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1995
v.4

Gypsy Moth Management
in the United States:
a cooperative approach

Final
Environmental Impact Statement
Volume IV of V

Appendix G

Ecological Risk Assessment

November 1995

United States
Department
of Agriculture

Forest Service

Animal and Plant
Health Inspection
Service

The complete final environmental impact statement, Gypsy Moth Management in the United States: a cooperative approach, consists of five volumes:

Volume I.	Summary
Volume II.	Chapters 1-9 and Appendixes A-E
Volume III.	Appendix F, Human Health Risk Assessment
Volume IV.	Appendix G, Ecological Risk Assessment
Volume V.	Appendix H, Comments on the Draft Environmental Impact Statement, and Responses

The record of decision is a separate document published and available 30 days or longer after the notice of availability for the final environmental impact statement is published in the Federal Register (40 CFR Part 1506.10).

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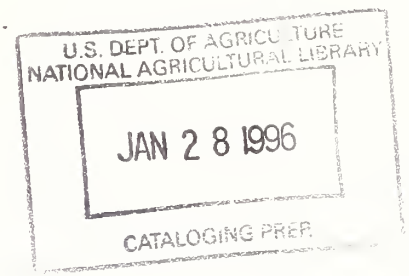
Ecological Risk Assessment
for the
National Gypsy Moth Management Program



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Published by:
U.S. Department of Agriculture
Forest Service
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Radnor, PA

2.8.3. Susceptible Stands (Greater than 60 Percent Defoliation) 2-49
2.8.4. Responses by Selected Habitat Generalists 2-53
2.9. Summary 2-54
2.10. References 2-55

3. Characterization of Active Management Strategies and Treatments 3-1

3.1. Description of Active Gypsy Moth Management Strategies 3-1
3.1.1. Suppression 3-2
3.1.2. Eradication 3-2
3.1.3. Slow-The-Spread 3-3
3.2. Description of Active Gypsy Moth Management Treatments . . . 3-4
3.3. References 3-6

4. Description of Program Areas

4.1. The Two Ecosystem Approach 4-1
4.2. The Development Continuum 4-2
4.3. Physical Characteristics 4-2
4.3.1. Climate 4-3
4.3.2. Soils 4-3
4.3.3. Hydrology 4-4
4.4. Biotic Characteristics 4-5
4.4.1. Flora 4-5
4.4.2. Terrestrial Fauna 4-6
4.4.3. Aquatic Fauna 4-8
4.5. Summary 4-8
4.6. References 4-10

5. Hazard Analysis

5.1. Toxicity Data 5-1
5.1.1. *Bacillus thuringiensis* var. *kurstaki* 5-1
5.1.2. Diflubenzuron 5-2
5.1.3. Nucleopolyhedrosis Virus 5-3
5.1.4. Dichlorvos 5-4
5.1.5. Disparlure 5-5
5.2. Terrestrial Field Studies 5-5
5.2.1. *Bacillus thuringiensis* var. *kurstaki* 5-5
5.2.2. Diflubenzuron 5-10
5.2.3. Gypsy Moth Nucleopolyhedrosis Virus 5-18
5.2.4. Dichlorvos 5-18
5.2.5. Disparlure 5-19

5.3. Aquatic Field Studies 5-19
5.3.1. *Bacillus thuringiensis* var. *kurstaki* 5-19
5.3.2. Diflubenzuron 5-20
5.3.3. NPV, Dichlorvos, and Disparlure 5-21
5.4. References 5-25

6. Endpoint Selection 6-1

6.1. Change in Forest Health 6-1
6.2. Change in Numbers of Nontarget Species Or Their Populations 6-2
6.3. Change in Water Quality 6-2
6.4. Change in Microclimate 6-3
6.5. Change in Soil Fertility, Productivity, Or Stability 6-3
6.6. Summary 6-3

7. The Fate and Transport of Insecticides in the Environment 7-1

7.1. Diflubenzuron 7-1
7.1.1. Fate in the Canopy 7-2
7.1.2. Fate on the Ground 7-3
7.1.3. Fate in the Water 7-5
7.1.4. Metabolites 7-9
7.1.5. Fate of Metabolites in the Canopy and in Soils 7-9
7.1.6. Fate of Metabolites in Water 7-11
7.1.7. Summary -- Undeveloped Forest 7-12
7.1.8. Summary -- Developed Forest 7-13
7.2. *Bacillus thuringiensis* var. *kurstaki* (*Btk*) 7-14
7.2.1. *B.t.k.* on Land 7-15
7.2.2. *B.t.k.* in Water 7-16
7.2.3. Summary -- Undeveloped Forest 7-17
7.2.4. Summary -- Developed Forest 7-17
7.3. Nucleopolyhedrosis virus 7-18
7.3.1. Fate and Location 7-18
7.3.2. Transport and Transmission 7-19
7.3.3. Summary - NPV 7-19
7.3.4. Summary - Undeveloped Forest 7-20
7.4. Disparlure 7-20
7.5. Dichlorvos 7-20
7.5.1. Dichlorvos in Air 7-21
7.5.2. Dichlorvos on the Ground 7-21
7.6. Environmental Fate And Transport Modeling 7-21
7.6.1. Methodology 7-22

7.6.2. Results and Discussion 7-32
7.7 References 7-34

8. Exposure Assessment 8-1

8.1. Diflubenzuron 8-1
8.1.1. Exposure Through Ingestion of Diflubenzuron 8-1
8.1.2. Exposure Through Dermal Contact of Diflubenzuron . . . 8-3
8.1.3. Exposure to Diflubenzuron in Aquatic Environments . . . 8-3
8.1.4. Summary
8.2. 4-Chloroaniline (Degradation Product of Diflubenzuron) 8-4
8.3. *Bacillus thuringiensis* var. *kurstaki* (*B.t.k.*) 8-4
8.3.1. Exposure through ingestion of *B.t.k.* 8-5
8.3.2. Exposure to *B.t.k.* in Aquatic Environments 8-5
8.3.3. Summary 8-5
8.4. Disparlure 8-6
8.5. Nucleopolyhedrosis Virus 8-6
8.6. Dichlorvos 8-6
8.7. References 8-8

9. Risk Analysis 9-1

9.1. Diflubenzuron 9-1
9.1.1. Methodology for Nontarget Species Endpoint 9-1
9.1.2. Results 9-4
9.2. *Bacillus thuringiensis* var. *kurstaki* 9-9
9.2.1. Methodology 9-9
9.2.2. Results 9-10
9.3. Disparlure 9-13
9.3.1. Nontarget Species 9-13
9.3.2. Forest Health, Water Quality, Microclimate, Soil Productivity and Fertility 9-13
9.4. Nucleopolyhedrosis Virus 9-13
9.4.1. Nontarget Species 9-13
9.4.2. Forest Health, Water Quality, Microclimate, Soil Productivity and Fertility 9-14
9.5. Dichlorvos 9-14
9.5.1. Nontarget Species 9-14
9.5.2. Forest Health, Water Quality, Microclimate, Soil Productivity and Fertility 9-14
9.6 References 9-15

10. Conclusions

10.1. Comparative Risks Between Treatments 10-1
10.2. Factors Affecting the Risk Associated with the Treatments 10-3
10.3. Comparative Risk Between Strategies 10-5
10.4. Comparative Risk Between Alternatives 10-5

Glossary Glossary-1

Subject Index . . . . . Subject Index-1

Species Index . . . . . Species Index-1

List of Tables

Table 2-1.	Comparison of European and Asian Strain of Gypsy Moths	2-81
Table 2-2.	European Gypsy Moth Host Susceptibility	2-82
Table 2-3.	Short-term (st) and long-term (lt) changes expected in various ecosystem components, assuming a moderate intensity outbreak (30-60 percent defoliation) that persists for 1, 2, or 3 successive years	2-83
Table 2-4.	Short-term (st) and long-term (lt) changes expected in various ecosystem components, assuming a high intensity outbreak (30-60 percent defoliation) that persists for 1, 2, or 3 successive years	2-85
Table 3-1.	Treatment Rates Used in Eradication, Suppression and Slow-the-Spread	3-8
Table 5-1.	Toxicity data of *Bacillus thuringiensis* var. *kurstaki*	5-50
Table 5-2.	Toxicity data of diflubenzuron	5-54
Table 5-3.	Toxicity data of nucleopolyhedrosis Virus	5-64
Table 5-4.	Toxicity data of dichlorvos	5-67
Table 5-5.	Toxicity data of disparlure	5-69
Table 5-6.	Effects of *B.t.k.* on nontarget Lepidoptera	5-70
Table 5-7.	Effects of *B.t.k.* on parasites of Lepidoptera	5-74
Table 5-8.	Effects of *B.t.k.* on birds	5-77
Table 5-9.	Effects of diflubenzuron on nontarget Lepidoptera	5-78
Table 5-10.	Effects of diflubenzuron on predators and parasites of insects other than gypsy moths	5-80
Table 5-11.	Effects of diflubenzuron on predators and parasites of gypsy moths	5-86
Table 5-12.	Effect of diflubenzuron on honey bees	5-88
Table 5-13.	Effect of diflubenzuron on other nontarget invertebrates	5-90
Table 5-14.	Effects of diflubenzuron on birds	5-94
Table 5-15.	Toxicity data of *Bacillus thuringiensis* var. *kurstaki* to aquatic organisms	5-96
Table 5-16.	Toxicity data of diflubenzuron to aquatic organisms	5-97
Table 6-1.	Ecological endpoints and their indicators	6-4
Table 7-1.	FSCBG model parameters	7-48
Table 7-2.	Summary of PRZM input parameters	7-49
Table 7-3.	Diflubenzuron estimated environmental residues	7-53
Table 7-4.	Multiple applications of diflubenzuron within the same year	7-55
Table 7-5.	Estimated diflubenzuron concentrations in various environmental components 1.0 oz/ac	7-56
Table 7-6.	Estimated diflubenzuron concentrations in various environmental components 0.5 oz/ac	7-57

Table 7-7. Estimated diflubenzuron concentrations in various environmental components 0.33 oz/ac 7-58
Table 7-8. Estimated diflubenzuron concentrations in various environmental components 0.25 oz/ac 7-59
Table 7-9. *Btk* estimated environmental residues 7-60
Table 8-1. Exposure to *Btk* in terrestrial diet items and aquatic resources 8-9
Table 9-1. Organisms at risk from diflubenzuron applications based on the screening index 9-25
Table 9-2. Variables used in Monte Carlo simulation 9-27
Table 9-3. Toxicological data used to estimate LC_{50} or EC_{50} distributions 9-29
Table 9-4. Percent of population affected by diflubenzuron application 9-35
Table 9-5. Risk of encountering a *Btk* drop 9-41
Table 10-1. Qualitative relative risk estimates by tactic 10-8

List of Figures

Figure 2-1. Generally infested, transition, and unifested areas as of November 1994 2-77

Figure 2-2. Predicted generally infested, transition, and unifested areas by the year 2010 2-78

Figure 2-3. The relation of larval population (N_L) in generation n+1 to larval population in generation n 2-79

Figure 2-4. Relationship between trend in egg density (I) and egg density per acre at the start of generation ($NE_{(n)}$); Glenville and Eastford data (1958-64 and 1965-68) 2-80

Figure 3-1. Decision-Tree for gypsy moth management 3-7

Figure 7-1. Fate and transport in the terrestrial environment 7-45

Figure 7-2. Fate and transport in the aquatic environment 7-46

Figure 7-3. *B.t.k.* growth cycle 7-47

Figure 9-1. Cumulative frequency distribution of toxicological studies conducted using diflubenzuron on terrestrial organisms . 9-22

Figure 9-2. Cumulative frequency distribution of toxicological studies conducted using diflubenzuron on aquatic organisms . . . 9-23

Figure 9-3. Relationship between environmental concentration of diflubenzuron and population level risk 9-24

Section 1
Introduction

1.1. Background and Purpose

The USDA Forest Service and the Animal and Plant Health Inspection Service are jointly preparing an environmental impact statement (EIS) for the USDA's gypsy moth program. Prior to preparing the draft EIS, comments were solicited from the general public and interested parties in order to help identify and define the major issues to be addressed in the EIS. Some of the issues identified during this scoping process were of an ecological nature, including potential adverse effects to nontarget species and effects on species diversity, water quality, microclimate, soil, and forest health. Based on the environmental concerns expressed by the public during the scoping process, the agencies decided to provide an ecological risk assessment.

The risk assessment process is designed to look at an activity or proposed action and carefully evaluate the potential impacts and the likelihood of those impacts occurring if that activity or action is carried out. Often the process identifies mitigating actions to decrease undesirable impacts or alternative actions that may improve the benefits derived from the activity.

The risk assessment process has been used most often to assess the likelihood of adverse impacts to human health. However, as people become more aware of potential negative impacts to various components of the environment, the process is being applied to environmental questions as well. The risk assessment process can be used to evaluate impacts to any environmental component that may be valued. Generally, ecological risk assessments analyze projects of limited geographical scope and examine impacts to a few nontarget species or other components of the environment that are of particular value to people, for example, a high value timber stand, a scenic vista or a prized trout fishery. However, as the interrelationships of the various components of the environment become clearer, the application of the risk assessment process naturally expands from consideration of a few components to consideration of many components and the ecological processes that connect them. This is an ongoing process that will continue to evolve.

This risk assessment is programmatic in nature because it was developed to support a programmatic EIS. It examines the risk on both large and small scales, but does not address specific sites or locations. It involves the analysis of three strategies (eradication, suppression, and slow-the-spread) which utilize treatment methods that include microbial and chemical insecticides. It also examines the risk posed by a strategy involving no active gypsy moth management (no treatment for gypsy moths).

1.2. Ecological Risk Assessment Process

Many risk assessments address the impact of chemical treatments (stressors) on some aspect or group of aspects (endpoints) of the ecosystem. This risk assessment is unique in that the stressors it addresses include biological agents, chemicals, and the pest species. Stressors evaluated include diflubenzuron, *Bacillus thuringiensis* variety *kurstaki* (*B.t.k.*), nucleopolyhedrosis virus, dichlorvos, Disparlure, and the gypsy moth.

The endpoints selected for evaluation include those that will alter the function or structure of the ecosystem if they are changed. This risk assessment also differs from most in that the endpoints evaluated are not limited to single species or populations of certain species. This risk assessment integrates ecosystem effects over several hierarchical scales of biological organization. These biological levels of organization include individual organisms, populations of organisms of the same species, groups of populations or individuals organized at the community level, and groups of communities organized at the ecosystem level.

The development of this risk assessment followed a 5-step approach: (1) identification of environmental components (endpoints) determined to be important in the context of the gypsy moth program based on the advice of individuals in the program and in the scientific community, as well as from the comments received and issues identified during the scoping process for the EIS; (2) identification of the potential hazard to the endpoints from each of the stressors (the gypsy moth and the treatment methods) based upon laboratory toxicity studies and field studies; (3) estimation of the potential exposure of each endpoint to each stressor, using mathematical models to evaluate the fate and transport of each treatment method, estimate environmental concentrations, and estimate nontarget exposures; (4) estimation of the risk posed by each treatment method to single species, population, community and ecosystem level endpoints, integrating the information gathered in the previous hazard and exposure analysis; (5) discussion of the ecological significance of the identified risks and a comparison of the risks posed by each treatment method. Because the treatment methods are combined in various strategies, a summary of risk was also provided for each of the strategies (eradication, suppression, slow-the-spread, and no active management).

Section 2
Exposure and Risk from Gypsy Moth

In this section the gypsy moths and their effects on the environment will be discussed, including: gypsy moths and the forest environment; gypsy moth biology; infestation history; outbreak duration; ecological effects of infestations; scenarios that include intensity, duration, and extent of outbreaks; exposure assessment; and an assessment of the risks from gypsy moths to forests, non-target species, water, microclimate, and soil.

2.1. The Forest Environment

Two forest types, oak-hickory and oak-pine, are both the most extensive forest types in the United States (Marquis and Johnson, 1989) and the types most likely to be host to outbreak populations of the gypsy moth. Among the three remaining forest types recognized by Marquis and Johnson (1989) only one, northern hardwoods, is generally immune to gypsy moth outbreaks. The other types, aspen-paper birch and the Allegheny and Appalachian mixed hardwoods, also host gypsy moth outbreaks.

Even though many people perceive ecosystems to be relatively stable and resisting change, and anything that alters the ecosystem is perceived to be destabilizing, there is a growing body of literature suggesting that forest ecosystems rarely achieve a steady state; they are instead interrupted by disturbances at irregular and frequent intervals (Twardus, 1994).

Among the forest-dwelling insects, the gypsy moth is one of the most thoroughly studied species and the target of the most intense efforts at containment, control, and eradication (McManus and McIntyre, 1981). Despite all such efforts, predictive abilities remain limited. In fact, many of the changes that are bound to occur in both gypsy moth populations and many other components of the hardwood forest (or any other) ecosystem will undoubtedly continue to contain surprises (Holling, 1986). A few examples follow:

Gypsy moth density: Gypsy moth egg masses can be very difficult to find, and density estimates based on egg mass counts may not always be representative of actual populations (Forbush and Fernald, 1896).

Density-defoliation: The population density-defoliation relationship is not always clear. For example, Liebhold and others (1993c) notes the uncertainty in the egg mass density-defoliation relationship and attributes it in part to measurement error, as well as the variable nature of the underlying relationship between "true" egg mass density and "true" defoliation. In another study, Williams and others (1991) compared their results with Gansner and others (1985), and suggested that those results may not be applicable in areas with different forest types or on the leading edge of the gypsy moth infestation. On a larger, region-wide scale, Twery (1991) noted that in areas with highly susceptible forest types, many stands experience few defoliations.

Favored hosts: North American plant species described as favored foods are usually similar to those listed long ago by Mosher (1915). Mauffette and others (1983), however, note that hornbeam is strongly preferred in Quebec, but in New England it is only an intermediate host, while gray birch and quaking aspen are both preferred in New England but classed as intermediate hosts in Quebec. Different gypsy moth strains also exhibit different food preferences. For example, Montgomery (1993) states that tests indicate the Asian strain of gypsy moth is more likely to be a serious problem in western forests than the European strain.

Tree mortality: Quimby (1993), Gansner and others (1993b), Tigner (1992), and others recognize differences in tree mortality between stands with comparable defoliation histories. This patchy tree mortality results in a mosaic of gaps interspersed within an otherwise continuous forest canopy (Hix and others, 1991).

2.2. Gypsy Moth Biology

2.2.1. Gypsy Moth Types

The types of gypsy moths considered in this risk assessment include European and Asian strain of the gypsy moth (AGM). Both gypsy moths are the same species (*Lymantria dispar L.*). As of 1994, the European gypsy moth is a permanent resident in all or part of 16 states and the District of Columbia. The Asian strain of the gypsy moth has been introduced into parts of the United States from the Russian far east and Germany.

2.2.2. Life Cycle

2.2.2.1. European Gypsy Moth

Eggs laid the previous year hatch into larvae from early April to late May. The newly hatched larvae often remains on the egg mass for several days before climbing toward foliage. After reaching the tree crown, most larvae spin silken threads and suspend themselves from leaves. Many of the threads break and wind may transport larvae several hundred yards in wooded areas and several miles in open areas. Gypsy moths in the larval stages are dark brown and hairy and may reach maximum lengths of 5.5 cm (Johnson and Lyon, 1988).

As the larvae mature, behavioral changes occur. While younger larvae remain on or near the foliage, older larvae seek resting locations farther down the tree in bark fissures, wounds, or flaps, or in the litter at the base of the tree. They descend to these protective resting locations at dawn and ascend at dark to feed in the canopy (Smith, 1985). During heavy infestations, larvae may continue to feed during the day.

The larval stage lasts from seven to eight weeks and normally consists of five instars (life stages) for males and six for females. After each instar, molting occurs in which the outer skin is shed to allow for growth during the next larval instar. Following the last instar, the mature larva finds a sheltered place and pupates in a brownish-black pupal case. Pupal cases are often found on the trunk of a host tree in clusters accompanied by molted skins of the last larval instar.

Moths begin to emerge about the middle of July, with males appearing several days earlier than females. As the gypsy moth expands into the more southern regions, this may occur as early as June. The European female cannot fly, and she emits a pheromone (sex attractant) that volatilizes and is carried in the air. Male moths can be attracted to such females for distances up to a mile. After fertilizing and depositing eggs the adult moths do not eat and soon die (Johnson and Lyon, 1988). Egg masses are deposited on tree trunks, rocks, and litter. Eggs overwinter, however egg mortality can be induced by below normal temperatures. Reduced defoliation following a winter of prolonged subzero temperatures was reported by Bess (1961) and was attributed to 90 percent overwintering egg mortality.

2.2.2.2. Asian strain of the gypsy moth

While similar in many respects, there are several significant differences between the European and Asian strain of the gypsy moth. Since the European gypsy moth was established in North America from a single introduction of closely related individuals,

genetic studies have shown little variation within or between populations (Wallner 1992). The Asian strain of the gypsy moth displays considerable variability within populations. This is expressed morphologically in the great variety of larval color forms, behaviorally in the female flight capability, and physiologically in the capacity of larvae to aggressively colonize a broad spectrum of hosts (Wallner, 1992).

While the female European gypsy moth is flightless, the Asian female is a strong flier capable of flights in excess of 18 miles (30 km) (Wallner, 1992). Since the female of the Asian strain of the gypsy moth is able to lay eggs far from the pupal site following flight, this characteristic alone may require the detection, delimitation, and control or eradication methods developed for the European gypsy moth be modified for the Asian strain of the gypsy moth (Wallner, 1992). A life stage comparison of the European and Asian strain of the gypsy moth is shown in Table 2-1.

2.2.3. Host Preferences

2.2.3.1. European Gypsy Moth

Larvae of the European gypsy moth eat foliage on a wide range of trees and shrubs, but they prefer hardwoods. The predominant species that suffer disproportionately high defoliation include oaks, apple, basswood, boxelder, hawthorne, poplars, and willow (Johnson and Lyon, 1988). Less preferred host species include elm, blackgum, hickory, maple, and sassafras. During heavy infestations, larvae may feed on otherwise non-preferred species. Included among these species are ash, butternut, cedar, dogwood, catalpa, fir, juniper, walnut, locust, and sycamore (Johnson and Lyon, 1988).

Despite a rather extensive number of potential host species, oaks historically suffer disproportionately more defoliation from gypsy moth. Most deciduous trees can survive one or two consecutive years of defoliation before severe decline or mortality occurs (Johnson and Lyon, 1988).

Although less susceptible than many broad-leafed species, some coniferous species such as white pine may be defoliated during heavy infestations. Since most coniferous species store carbohydrate resources necessary to refoliate in the leaves, they are usually unable to survive a single, complete defoliation (Johnson and Lyon, 1988).

Mosher (1915) found gypsy moth larvae favored certain tree and plant species over others, depending on the insect's stage of development (Gottschalk, 1988). The four classifications of common tree and shrub species shown in Table 2-2 are based on European gypsy moth food preference, with class I species being the most

preferred and susceptible species, and class IV being the least preferred and susceptible species.

2.2.3.2. Asian strain of the gypsy moth

The Asian strain of the gypsy moth feeds on numerous species of trees and shrubs. In the former Soviet Union, the Asian strain of the gypsy moth feeds on more than an estimated 600 tree or plant species. Comparison studies indicate the Asian strain of the gypsy moth feeds more voraciously and grows faster on white oak, larch, and paper birch than does the European gypsy moth. The Asian strain of the gypsy moth may not only thrive on the same tree species eaten by the European gypsy moth, but may do better on many species that the European gypsy moth does not favor, such as the Douglas-fir (Wallner, 1992). In the United States, studies show Asian strain of the gypsy moth grows better than European gypsy moth on 50 plant species, and the greatest differences in growth rates are on coniferous species (Wallner, 1994).

2.2.4. Population Dynamics

Gypsy moth populations are regulated by small mammal and avian predators, invertebrate predators, insect parasitoids, viral pathogens, availability of favored tree species, dispersal characteristics, competition for food, and weather (Elkinton and Liebhold, 1990). In general, predators play their most important role when gypsy moth population densities are low, while both viral disease and the effects of competition for food occur at high gypsy moth population densities and can cause outbreaks to collapse (Speight and Wainhouse, 1989).

Following an initial outbreak, populations generally decline and are usually maintained at innocuous densities. Subsequent outbreaks are usually less severe than the initial outbreak. Gypsy moth populations in North America are described as bimodal, existing either at innocuous densities or in an outbreak mode (Campbell, 1981).

Several causal factors have been described for gypsy moth outbreaks. A positive relation between the proportion of favored tree species and population densities has been widely reported (Baker and Cline, 1936; Behre, 1939; Behre and Reineke, 1943; Behre, 1939). Behre (1939) reported outbreaks to be unlikely in forest stands with less than 50 percent favored tree species composition.

Although dispersal of young larvae plays a role in gypsy moth outbreaks, it is thought to play a relatively minor role in outbreak initiation (Campbell, 1976). Larval dispersion may be the major cause of gypsy moth distribution enlargement and range

expansion at innocuous densities, but it does not cause outbreaks to spread. Spatial distribution of stand susceptibility is more likely to cause outbreaks and subsequent defoliation to spread (Liebhold and McManus, 1991). Outbreaks have been described as originating in small, discrete locations. These locations are referred to as "foci" and are usually characterized by stands growing on stressed sites such as ridgetops, upper slopes, and deep sands and frequently subjected to drought (Houston and Valentine, 1977). These areas are able to support moderate to high populations when gypsy moth is undetectable in surrounding areas (Liebhold and McManus, 1991). Protected resting locations that favor larval and pupal survival have been shown to support higher gypsy moth populations and lead to outbreaks (Houston and Valentine, 1977; Bess and others, 1947; Houston, 1975; Campbell and Sloan, 1977a). Other factors that may precipitate outbreaks include predator failure and specific climatic and meteorological conditions. Khanislamov and Girfanova (1964) have shown that weather variation may have more drastic effects on natural enemies of gypsy moth than on the pest itself. Population collapse at the end of an outbreak appears to result from the widespread occurrence of overpopulation phenomena, principally disease, reduced fecundity, and starvation (Campbell, 1981).

Forested areas subjected to repeated defoliation may eventually experience significant changes to forest stand composition. These changes may serve to reduce susceptible species as well as the capacity of forests to sustain further outbreaks.

2.2.5. Natural Regulators

2.2.5.1. Nucleopolyhedrosis virus

Nucleopolyhedrosis virus (NPV) is a natural component of the gypsy moth environment (Podgwaite, 1981). NPV is considered the primary natural regulator of dense gypsy moth populations in North American forests (Reiff, 1911; Glaser and Chapman, 1913; Doane, 1970). High density populations of gypsy moth will eventually collapse, for the most part due to pathogens, especially NPV (Elkinton and Liebhold, 1990).

2.2.5.2. Predators and Parasites

In the Northeast many species of wildlife eat gypsy moths and other forest-defoliating insects. Some predators eat only one lifestage of the gypsy moth while others consume two or more lifestages (Smith, 1985). Predation can regulate sparse, stable gypsy moth populations indefinitely. Once an outbreak starts, however, as well as during subsequent outbreak decline, predation has no significant effect on population densities (Smith and Lautenschlager, 1981).

In each life stage the gypsy moth becomes prey to several animals. Mice and shrews were shown by Bess and others (1947) to be important predators of gypsy moth, particularly during the pupal stage. Forbush and Fernald (1896) first identified birds as predators of gypsy moths, concentrating on instar IV-VI larvae. More recent studies have identified mammals as having a greater impact on gypsy moth populations (Smith and Lautenschlager, 1981; Elkinton and Liebhold, 1990).

Invertebrate predation of gypsy moth pupae was considered to be minor in comparison to vertebrate predation (Campbell and Sloan, 1977a). However, Smith and Lautenschlager (1981) suggested some of the mortality attributed to vertebrates was in fact caused by invertebrates, such as ground beetles (Elkinton and Liebhold, 1990). Both adult and immature stages of *Calosoma sycophanta*, a large ground beetle introduced from Europe, are known to feed on gypsy moth larvae and pupae (Elkinton and Liebhold, 1990).

Although gypsy moth populations respond to predator pressure, periods of low predatory pressure do not necessarily lead to an outbreak. Outbreaks become more likely when lack of predatory pressure coincides with other factors.

Entomophagous (feeding on insects) parasites of the gypsy moth have been widely studied for their application as biological control agents (Reardon, 1976), but most researchers believe that they do not play a major role in the population dynamics of gypsy moths in North America. Extensive efforts have been made to introduce European and Asian strain of the gypsy moth parasitoids to North America (parasitoids are insects, especially wasps, that complete their larval development inside the body of another insect). Ten species have become established (Elkinton and Liebhold, 1990). Gypsy moth mortality due to each type of parasite is specific to a given gypsy moth life stage.

2.2.5.3. Bacterial and Fungal Pathogens

The pathogenicity of the spore-forming bacterium *Bacillus thuringiensis kurstaki (B.t.k.)* against the gypsy moth was first reported in 1929 (Metalnikov and Chorine, 1929). During sporulation, *B.t.k.* produces a toxic delta-endotoxin protein crystal.

In addition to *Bacillus* spp., other bacterial pathogens include *Serratia marcescens, Serratia liquefaciens, Streptococcus,* and *Pseudomonas* spp. The collective mortality to gypsy moth associated with these microorganisms does not usually exceed 15 percent (Podgwaite, 1981).

Fungal pathogens account for insignificant levels of recorded gypsy moth mortality, and are of little importance in overall population regulation (Podgwaite, 1981). However, the fungal pathogen *Entomophaga maimaiga*, which plays an important

role in gypsy moth population dynamics on other continents has become widely established in North America.

2.2.5.4. Antiherbivore Compounds

Normal declines in gypsy moth populations may be due to changes in leaf biochemistry in the preferred host. The development of defensive mechanisms within the host may reduce susceptibility and prevent a second defoliation within the same year. Changes in the biochemistry of replacement leaves of red oak saplings that had been defoliated were analyzed by Schultz and Baldwin (1982). Compared with leaves of red oaks that had not been defoliated, the replacement leaves of defoliated oaks had increased levels of tannin and phenolic compounds as well as reduced water content. All these changes have been shown to retard gypsy moth larval growth. Gypsy moths reared on replacement leaves of black oaks that had been defoliated not only showed reduced larval growth and weight, but also experienced nondisease mortality of 80 percent. This was attributed to the increased total phenolic content and tannin levels (Schultz and Baldwin, 1982).

2.2.6. Factors Controlling the Rate of Spread

Spread of the gypsy moths occurs in two ways: windblown dispersal of newly hatched larvae and transport of egg masses due to human activity. The latter can include egg masses attached to vehicles, building material, or any other object that can be transported. Wind dispersal accounts for short-range movements, while transport due to human activity accounts for long-range movements, including remote, isolated outbreaks known as spot infestations (Talerico, 1981). Whether spread by either mechanism, gypsy moth populations still require susceptible trees and an acceptable climate to become established.

Between 1906 and 1920 the rate of spread in the New England states was measured at five miles (9.2 km) per year (McManus and McIntyre, 1981). Since 1965 the rate of spread in the Appalachians has been estimated at 13 miles (21 km) per year (Liebhold and others, 1992).

The capability for sustained long distance flight by the adult female is one characteristic that makes the establishment of the Asian strain of the gypsy moth in the United States such a threat. Should the Asian strain of the gypsy moth become established in the U.S., it would spread at an estimated rate of four times greater than the European variety (Wallner, 1994).

2.3. Gypsy Moth Infestation History

2.3.1. Brief History of Infestations

Originally introduced in eastern Massachusetts in 1869, subsequent spread of gypsy moth resulted in eradication efforts that were considered such a success they were abandoned in 1900. The gypsy moth continued to spread however, and eventually cooperative efforts to control gypsy moth were initiated between infested states and USDA. Various campaigns of eradication and control have continued since the early 1900s. With the exception of isolated outbreaks, the goal of eradication was largely abandoned in the 1960s. Current programs emphasize integrated pest management (IPM) techniques and the use of biological and chemical pesticides and treatment alternatives that pose minimal environmental impact.

2.3.2. Present Distribution

The generally infested area in the United States includes: all or portions of Maine, Massachusetts, New Hampshire, Vermont, Rhode Island, Connecticut, New York, New Jersey, Pennsylvania, Delaware, District of Columbia, Maryland, West Virginia, Virginia, Ohio, North Carolina and Michigan. The gypsy moth is presently extending its range into the southeast and midwestern states (Liebhold and McManus, 1991). Isolated infestations have occurred and been eradicated in at least 26 states since 1967. The area where gypsy moth is permanently established is shown in Figure 2-1.

2.3.3. Potentially Infested Area: Next 15 Years

Forest types classified as highly susceptible to infestation extend as far west as the eastern portions of Texas, Oklahoma, and Nebraska, as well as northern Minnesota (USDA, 1985). Given the currently accepted rate of spread, portions of North Carolina, and the remainder of Virginia, Ohio, Indiana, Illinois, and Wisconsin have the potential to become generally infested over the next 15 years. The projected area of gypsy moth spread is described in Figure 2-2.

2.3.4. Detecting and Evaluating Populations

Gypsy moth populations have proven to be difficult to sample. The most common method of population estimation is egg mass counts. Other methods include the use of pheromone traps, frass drop measurements, and timed-walk techniques. The

timed-walk method uses regression equations to estimate populations based on egg mass counts collected within a certain period of time, usually five minutes.

2.3.5. Population Size and Defoliation

Estimating or predicting the degree of defoliation from egg-mass counts alone ignores other important factors such as weather, parasites, predators, dispersal, and disease. Egg-mass data obtained by the fixed- variable-plot method were used by Wilson and Talerico (1981) to predict percent defoliation the following year in central Pennsylvania. A correlation coefficient of 0.73 was considered significant when considering other factors mentioned above (Wilson and Talerico, 1981). Although egg mass counts can be useful, they are an imperfect tool for forecasting defoliation.

2.4. Outbreak Duration

The longevity record for gypsy moth outbreaks in North America may belong to the town of Brewster, located on Cape Cod, Massachusetts. Bess and others (1947) reported the gypsy moths caused some defoliation in this town for 20 successive years (1925-44). On a much larger scale, a gypsy moth outbreak persisted for at least a decade (1911 to 21) in an approximately 50-mile wide strip of land that stretched along the Atlantic seaboard of New England, from Maine and New Hampshire to southern Massachusetts (Campbell, 1973). More recently, major multiyear gypsy moth outbreaks have occurred in parts of New Jersey, Pennsylvania, Delaware, Maryland, Virginia, West Virginia, and Michigan (USDA, Forest Service, 1993) almost entirely involving persistent initial outbreaks. In contrast to these extended outbreaks, most other gypsy moth outbreaks in North America flared-up and then dissipated in just one or two years. Such is the usual pattern in areas like New England, where the gypsy moth has been long established.

Unfortunately, the characterization of even the most important processes in the gypsy moth life system remains uncertain. Consequently, our ability to predict its population fluctuations is limited. Given these uncertainties and limitations, some tentative interpretations of the processes that underlay multiyear population fluctuation patterns during gypsy moth outbreaks are described below.

2.4.1. Key Factors

For many species, Morris (1957) proposed that population studies would ultimately reveal a few "key factors" mainly responsible for changes in population numbers. Understanding population dynamics depends in large part on the identification of these key factors (Price, 1984). In herbivorous species whose population numbers fluctuate

cyclicly, for example, this cyclic behavior often seems to be dominated by a process through which the herbivore alternately induces changes in host foliage (through defoliation) and then responds to this altered food quality (through population decline).

In a recent review of what is known about gypsy moth population dynamics, Elkinton and Liebhold (1990) summarized the results of several investigations. They concluded that important processes in this life system include the action of small mammal predators when populations are sparse and density-dependent parasitism by tachinids during incipient outbreaks. Once outbreaks are underway, they emphasize the roles of NPV and changes in foliage chemistry. They also suggest weather conditions influence the system in some manner, but the precise mechanism remains obscure. With additional research, a much different understanding of gypsy moth population dynamics may emerge (Elkinton and Liebhold, 1990).

2.4.2. Density-Dependence

Numerical changes in an herbivore population are often related to earlier herbivore density. Such density-related changes are said to exhibit density-dependence. Examples of responses to changes in herbivore numbers by density-related factors include herbivore-natural enemy relations (Hairston and others, 1960; Nicholson, 1958), herbivore-food plant relations (Baltensweiler and Fischlin, 1988), and density-related changes in herbivore behavior (Campbell, 1993; Wilson, 1975). In contrast to density-dependent factors, factors such as weather (Greenbank, 1956) and random influences (Cole, 1954) are said to be density-independent. Currently there appears to be general agreement that gypsy moth populations are regulated largely by density-dependent processes.

2.4.3. Population Equilibria

Morris (1963) proposed two very different extremes of population densities of the spruce budworm, *Choristoneura fumiferana* (Clem.) are both able to remain more or less constant (Figure 2-3). At the first of these equilibrium densities ((1) in Figure 2-3), the population is innocuous. If the population reaches point (R), however, it will probably continue to increase to a second stable equilibrium ((2) in Figure 2-3), and a sustained outbreak will ensue. More generally, patterns of population fluctuation characterized by multiple equilibria have been described for a variety of animal populations (Raffa and Berryman, 1986; Isaev and Khlebopros, 1973; Peterman and others, 1979). Similarly, Campbell and Sloan (1978a) proposed the gypsy moth also exhibits multiple equilibria (Figure 2-4). In one set of studies, for example, the gypsy moth population equilibrium was about 1,000 eggs per acre

("Eastford" in Figure 2-4). In another set, however, this equilibrium was about 100-fold higher--100,000 eggs per acre--("Glenville" in Figure 2-4).

With some reservations (Liebhold, 1992), there is general agreement that European gypsy moth populations in North America have two numerical equilibria (Elkinton and Liebhold, 1990; Liebhold and Elkinton, 1989). Based on published descriptions of their population fluctuations, Berryman (1983) classifies outbreaks of both the gypsy moth and the jack pine sawfly, *Neodiprion swainei*, as "sustained eruption."

2.4.4. Aggregating Uncertain Outcomes

A range of 0 to 100 percent defoliation can result even when egg mass densities range between 100 and 1,000 egg masses per acre, showing a weak correlation between egg mass density and defoliation (Williams and others, 1991; Liebhold and others, 1993a). Even including additional measurements, prediction of defoliation in this density range remains relatively imprecise. As these results suggest, it is not possible to predict the year when any given gypsy moth outbreak will end, and there exists only a limited ability to predict defoliation intensity during any given season. Proposals to mitigate this kind of uncertainty are described below.

2.4.4.1. Patchiness

During years when there is a wide range of gypsy moth subpopulation densities, outbreak conditions show a tendency to persist or even worsen. Spotty defoliation one to two years prior to a general outbreak are typical. Conversely, during years when subpopulation densities are roughly equal, outbreaks tend to decline. This implies that variability within some subpopulations may often be influenced more by conditions within adjacent subpopulations than by local, on-site conditions (Campbell and Sloan, 1978b). In describing gypsy moth population dynamics, Liebhold and Elkinton (1989) describe results which do not dispute this notion.

2.4.4.2. Focal Areas

Within any large, heterogeneous forested area, certain stands are much more likely than others to be the first to exhibit visible defoliation. Such observations gave rise to the notion that gypsy moth outbreaks start in such areas and expand outward in successive years (Houston and Valentine, 1977; Wallner, 1987). Recently, however, Liebhold (1988) emphasized the importance of the spatial pattern of stand susceptibility in the maintenance of outbreaks. In general, there is a gradation of stand susceptibility from the center of focal-areas out through the surrounding region (Houston and Valentine, 1985). This continuum of stand susceptibility may be

responsible for the observed pattern of defoliation. When some exogenous factor(s) such as weather triggers release, foci may reach defoliating densities rapidly. In less susceptible stands densities may increase to defoliating densities, but not as rapidly (Liebhold, 1988).

2.4.4.3. Using Categories

Williams and others (1991) developed predicted defoliation values based on population densities. In populations of intermediate density, use of their predicted defoliation value may prove inaccurate, resulting in either unnecessary action when defoliation is low or inaction when defoliation is high. To address this problem, these investigators followed the same procedures as Campbell (1966). Campbell partitioned data accumulated between 1911-31 by the now-defunct Melrose Highlands Gypsy Moth Laboratory into ten classes that included five density ranges and two population trends. Within each class, Campbell presented the frequency of defoliation of oak in the four quartile ranges of percentage defoliation. Williams and others (1991) reported results that compared remarkably well with Campbell's (1966) given the differences in time and space between the two studies.

2.4.4.4. Geostatistics and GIS

Geostatistics and Geographic Information Systems (GIS) are two relatively recent technologies that provide new avenues for analyzing spatial patterns of insect populations. GIS serves as a tool for analyzing interactions among and within spatially referenced data. Geostatistics are a family of statistics that describe correlations through space and time. Most of the published applications of GIS and geostatistics to insect problems are from forest and rangeland entomology since forests and rangelands are usually managed as units covering 100 to 10,000 heterogeneous ha (Liebhold and others, 1993b). Clearly, the life system of the gypsy moth is appropriate for exploring these new technologies. Already, several such exploratory studies have been published (Liebhold and Elkinton, 1989; Liebhold and others, 1990a, 1990b, 1991, 1994).

The use of tools such as GIS and geostatistics, the incorporation of spatial analysis into ecological theory, ecological models, and pest-management practices will not happen immediately because substantial developmental changes in both theory and practice must occur (Liebhold and others, 1993).

2.4.4.5. Homeostasis and Weather

Miller and others (1989) compared weather records in Massachusetts and Connecticut with the area in each state that was defoliated annually by the gypsy moth. In both states, drought conditions in October during the preceding generation followed by warmer than usual daily minimum temperatures around egg hatch (the first of May), were significant predictors for overall acreages defoliated. Most of the variation in both states in actual defoliation was associated with prior defoliation and indices of these weather conditions (in Massachusetts, model $R^2 = 0.819$; in Connecticut, model $R^2 = 0.834$). Michaels (1987) also developed a model that uses weather variables to project gypsy moth-induced defoliation. Michaels calculated that this model was able to correctly predict a major change in moth status (greater than 15 percent +/- change in defoliation from the previous year) 83 percent of the time in the "fit mode" and 79 percent of the time in the "test mode." Unfortunately, Michaels does not describe the weather variables that were found useful, indicating only that the most predictive model was one which uses only two principal components.

Once a large scale gypsy moth outbreak is underway, the above results suggest the ebb and flow of that outbreak is a function of outbreak size and weather. If these results prove to be generally applicable, their implication for the future of multiyear projections of gypsy moth outbreaks is enormous. Specifically, on a multiyear time horizon, weather events such as those specified in Miller and others (1989) are currently not themselves predictable with enough accuracy to make them operationally useful predictors of defoliation. Thus, our ability to project the course of areawide gypsy moth outbreaks may continue to be constrained to a single season.

2.5. Ecological Effects of Infestations

2.5.1. Terrestrial Vegetation

2.5.1.1. Trees and stands

Defoliation and secondary organisms

For at least half a century, the gypsy moth persisted at generally innocuous densities in the predominantly oak forests of northeastern Connecticut and adjacent Massachusetts (Bess and others, 1947; Brown and Sheals, 1944; Friend, 1945; Turner, 1963). During such intervals, gypsy moth larvae usually eat only a small proportion of the foliage of even their most favored host species; when defoliation, overall, is low, nearly all of it occurs on favored-food trees (Campbell and Sloan, 1977b). Once a large-scale outbreak is underway, however, the list of affected plants

of this pest expands to include some 300 species of broadleaved and coniferous trees and shrubs (Leonard, 1981). This list grows ever longer as the insect invades new areas (Liebhold and others, 1995). In the eastern United States alone, 76 percent of the hardwood forests have been classified as susceptible to the insect (USDA, 1990) and as many as 12.5 million acres have been defoliated in a single season (1981) (Williams, 1982).

Trees defoliated about 75 percent or more are likely to refoliate during the same season. The refoliated leaves are smaller and fewer, and repeated defoliations can cause additional reductions in leaf size (Wargo, 1981a). In addition, while many twigs and branches may be dying (Staley, 1965), there is also significant sprouting from adventitious and latent buds. According to Wargo (1981b), trees that refoliate are completely out of phase with the season. Visually, for example, the condition of trees in a mixed composite stand of oaks (red, black, scarlet, and white) in eastern New England showed rapid decline in the year after defoliation and continued to decline slightly during the next five years. Following a single heavy defoliation, about 10 years passed before these trees returned to their predefoliation condition (Campbell and Sloan, 1977b).

What happens when a tree is defoliated depends on five key factors: (1) severity (how much foliage is removed), (2) frequency (the number of successive years of defoliation), (3) timing (when in the growing season the tree is defoliated), (4) pathogens (the presence and number of secondary organisms), and (5) health and vigor (the physiological condition of the tree when it is defoliated) (Parker, 1981). Most trees are able to tolerate at least two years of defoliation before root starch content (useable energy) is depleted (Wargo, 1981a). Even when such reserves are depleted, mortality does not normally occur in the absence or scarceness of secondary organisms (Wargo, 1981b).

The principal secondary organisms involved in the further decline and possible death of previously defoliated eastern hardwood trees are the shoestring fungus, *Armillaria* spp. (Vahlex. Fr.) Kummer, and the twolined chestnut borer, *Agrilus bilineatus* Web. (Houston, 1981a; Wargo, 1981b). Ultimately, tree mortality is due to effects on photosynthesis, growth regulators and water, and nutrition relations. These effects are induced in the crown by the defoliator and the borer, in the main stem by the borer, and in the roots by the fungus (Wargo, 1977; Wargo, 1981b). As Gottschalk (1994) noted *Armillaria* and *Agrilus* play an important role in forest health, removing weak, sickly trees from the population.

Previous stand disturbance, which may allow partial colonization of root systems by *Armillaria*, increases rhizomorph abundance (Twery and others, 1990; Wargo, 1989). Even in the presence of abundant rhizomorphs, however, undefoliated and lightly defoliated trees may remain resilient (Twery and others, 1990). Hart (1989), for

example, reported mortality due to *Armillaria* root rot occurred only in aspen trees that experienced defoliation of 80 percent or more.

Stressed trees also provide an environment that favors the survival of the twolined chestnut borer (Côté, 1976; Twery, 1991; Wargo, 1977). This borer is attracted to volatile chemicals released by stressed oaks (Dunn and others, 1986a). Trees with low stored starch reserves were more likely to be attacked; and only those trees with extremely low winter root starch reserves were likely to die (Dunn and others, 1986b, 1987).

Tree mortality

The most useful variables for predicting tree mortality following gypsy moth defoliation appear to be species composition, defoliation duration, and defoliation intensity (Fosbroke and Hicks, 1989). Factors that contribute to interspecies differences in responses to defoliation include where in the tree reserve energy is stored, the amount of energy required to refoliate, and how much energy is needed to maintain growth during refoliation (Twery, 1991). Hemlock, for example, will not usually survive even one complete defoliation (Stephens, 1988), whereas some oaks on dry sites may survive repeated defoliations indefinitely (Houston and Valentine, 1977; Bess and others, 1947; Twery, 1991). Similarly, House (1963) reported that significant white pine mortality occurred only where white pines experienced 100 percent defoliation.

Heavy defoliation is much more common among the oaks than among trees that are not particularly favored as food by the gypsy moth (food class B). When heavy defoliation occurs, however, mortality rates among food class B trees may be much higher than rates found among comparable oaks. Following a single heavy defoliation in eastern New England, for example, 69 percent of the food class B trees that were rated "poor" died, compared to about 37 percent of the comparable oaks (Campbell and Sloan, 1977b).

In any given stand, heavy defoliation year after year tends to be a rare event. When such an event does occur, however, consequent tree mortality rates soar. In the area described by Campbell and Sloan (1977b), for example, only 7 percent of the mixed oaks rated "good" died following a single heavy defoliation. After two successive heavy defoliations, however, mortality rates in this category increased to 27 percent. Similar results have been reported by others (for example, Gansner and Herrick, 1984; Hicks and Fosbroke, 1987a, 1987b).

Gypsy moth infestations generally result in mortality losses of less than 15 percent of total basal area. However, losses of 15 to 35 percent are not uncommon, and occasionally losses occur that are greater than 50 percent (Gottschalk and others,

1987). Volume growth is reduced among surviving trees for about three years after a defoliation episode (Heichel and Turner, 1976; Picolo and Terradas, 1989; Twery, 1991). Specifically, Twery (1987) reported an average decrease of 20 percent in stem volume growth in oaks in any year a tree was defoliated compared to the previous, undefoliated year. Some of this reduction is a result of reduced leaf area in the recovering trees (Wargo, 1981a). Despite such losses, Quimby (1993) reported net growth in Pennsylvania presently exceeds that of removals (including inordinate tree mortality) by a factor of 2.2 to 1 (Powell and Considine, 1982). Similar results have been reported by Gansner and others (1993a). Further, depending on management objectives, some losses in tree volume caused by the gypsy moth could be advantageous, at least in some pine-oak stands (McGee and Bivens, 1984). Years ago, for example, Hall (1935) commented on the resistance of pitch pine, *Pinus rigida* Mill., to defoliation by the gypsy moth and its advantages as a forest tree on Cape Cod, Massachusetts. Subsequently, Campbell and Garlo (1982) showed that growth of pitch pine increased in New Jersey, at least temporarily, after the gypsy moth defoliated competing oaks.

Defoliation was heavy along the eastern seaboard of New England between 1911 and 1921. During this decade the oak component suffered about 60 percent mortality. About 30 percent of red maples and 33 percent of white pines also died (Campbell and Sloan, 1977b). During the next decade, both defoliation and the responses to it were significantly less (Baker, 1941; Campbell and Sloan, 1977b). In fact, a sustained outbreak of the magnitude that occurred from 1911 to 1921 has not been recorded since in New England (Houston, 1981a). In a very general way, the pattern of the most severe tree mortality occurring along and behind an advancing outbreak front has been repeated as the insect has invaded new areas (Gansner and Herrick, 1984; Herrick and Gansner, 1986; Twery and Gottschalk, 1988). Herrick and Gansner (1988), for example, found average damage that was "very similar" in case studies drawn from outbreaks in northeastern Pennsylvania (1971-1979) and central Pennsylvania (1978-1985). In short, results to date support the conclusion that most mortality occurs during and after the initial outbreak (Twery, 1991). In addition, Campbell and Sloan (1977b) found certain trees within any given species consistently suffered heavier defoliation than others and were also more likely to die. These authors suggested that such differential intraspecific mortality might also tend to reduce subsequent stand vulnerability. In this regard, Byington and others (1994a) reported that intraspecific genetic variability for tolerance and response to defoliation, and possibly for resource allocation, exists in red oak. This variability may allow natural populations of red oak to evolve in response to the presence of gypsy moth in the environment (Byington and others, 1994b).

Gottschalk (1994) reports that moderate to heavy overstory mortality in recent years has followed heavy defoliation on about 5 to 20 percent of defoliated Appalachian stands. Unfortunately, tree mortality rates in these stands have shown no clear signs of downturn in places where a second wave of equally heavy defoliation has

subsequently occurred. Possibly, the frequently-cited reductions in gypsy moth-induced effects in areas such as New England have come about primarily as a result of changes in forest composition.

Stand Structure

Consistently, subdominant trees suffer much higher mortality than the taller dominants after heavy defoliation (Brown and others, 1988; Campbell, 1979; Gansner and others, 1993b; Quimby, 1993). One reason for the higher, post-defoliation mortality rates among subdominant trees may be contained in the results of some greenhouse trials. In these trials, shaded plants experienced twice the level of defoliation-induced mortality of plants grown in full sunlight (McGraw and others, 1990). In any case, one typical result of heavy and repeated defoliation is a more one-storied stand. Although oak growing-stock volume in trees less than 10 inch diameter at breast height actually decreased between 1965 and 1989, these losses were offset by gains in larger trees (Gansner and others, 1993a).

From eastern New England to western Pennsylvania, the effect of defoliation on composite stands often appears to resemble the approximate, but naturally occurring equivalent of a "thinning from below" (Campbell and Sloan, 1977b; Gottschalk 1990a). Long-term consequences of such naturally occurring thinnings are only now being revealed, but intentional thinning from below in oak stands tends to have some beneficial effects. The production of high-quality logs is enhanced, mast production is increased, and the large trees improve the aesthetic qualities of the stand. However, growth per acre per year is reduced during the latter part of the rotation (Clark and Watt, 1971).

Mast Production

Even in healthy, undefoliated stands, heavy crops of hard mast (acorns) are only produced about every three or more years. During intervening years, mast crops may be poor or nearly absent (USDA, 1994). In addition, defoliation can virtually eliminate oak mast production, especially in the short-run. Defoliation can result in several consecutive years of complete mast failure, a difficult situation to which many wildlife species must adapt (Gottschalk, 1990b; Liscinsky, 1984; Palmer, 1988). During years of moderate and heavy defoliation, such effects on mast production can be attributed to three sources: (1) direct consumption of flowers, (2) abortion of immature acorns due to a low carbohydrate supply, and (3) lack of flower bud initiation. These effects are generally short-lived with residual effects lasting for only one or two years after defoliation ends. Available data suggests that abortion of immature acorns is the most significant of these three effects. As a result, up to five years of complete acorn failure is possible (Gottschalk, 1990a).

As previously noted, trees that do not die during a defoliation episode may take as long as 10 years to recover their full vigor. On the other hand, once trees have recovered their vigor, significant overstory mortality (more than 60 percent of the basal area) must occur before significant reductions in acorn production occur. This is due to mortality occurring for the most part in intermediate and suppressed trees that are not heavy mast producers. Production of residual trees may even be stimulated by this thinning (Gottschalk, 1990a). In the long term, loss of hard mast is partially compensated by an increase in soft mast (flower) production (Gottschalk, 1990b). Some effects of defoliation-induced losses of hard mast on wildlife are described later in this section.

2.5.1.2. Shrubs and Herbaceous Plants

When heavy defoliation is followed by massive overstory mortality, existing shrub and herb cover increase dramatically due to increases in available light, moisture, and nutrients. Coupled with increased nitrogen, increased light may also warm the soil enough to induce germination of long-buried seeds, such as those of raspberry.

In some situations, heavy defoliation and subsequent overstory mortality can result in dominance by shrubbery and herbaceous vegetation for several years. Gansner (1985) described an understory 10 years following defoliation as being dominated by blueberry (*Vaccinium* spp.), witch-hazel (*Hamamelis virginiana* L.), raspberries (*Rubus* spp.), and several species of ferns, along with some tree seedlings that had been heavily browsed by deer (Hix and others, 1991). Hix and others (1991) also noted that blueberries and raspberries were often the shrub species that increased the most following defoliation in the stands they studied. And among herbaceous plants, Brackley (1985) noted that gypsy moth-induced defoliation appeared to stimulate flowering in the Federally endangered orchid *Isotria meleoloides* in New Hampshire. In time, even such shrub-dominated stands almost certainly revert to growth representative of a maturing eastern hardwood forest.

Gypsy moth food preferences include a long list of species that normally occupy the forest understory. During outbreaks, differential defoliation of these species provides a short-term advantage to species that are rarely or never eaten. The following list of understory food preferences is from Twery (1991).

Susceptible. species readily eaten by gypsy moth larvae during all instars. Hawthorn, hazelnut, hophornbeam, hornbeam, serviceberry, witch-hazel.

Resistant. species eaten when preferred food is not available and/or only some instars. Blueberries, pin and choke cherry, pawpaw, persimmon, redbud, sourwood, sweetfern.

Immune. species that are rarely eaten. All Azalea species, dogwood, elderberry, grape, greenbrier, juniper, mountain, silver and striped maple, rhododendron, all Rubus species, sheep and mountain laurel, spicebush, sarsaparilla, all viburnum species.

2.5.1.3. Vegetative diversity.

Following defoliation, many investigators noted increases in red maple and other species, and corresponding decreases in oak in Appalachian forests (Gansner and others, 1994a, In press; Hix and others, 1991). Regeneration in the Allegheny Mountain region has been dominated by red maple, and red maple and birch in the Ridge and Valley regions. Oak reproduction has been sparse and seedlings small compared to red maple and non-commercial seedlings in the Allegheny Mountains. In one study, only 4 to 16 percent of the stems were northern red oak or white oak (Allen and Bowersox, 1989). A similar pattern emerged in Michigan (Witter and others, 1994). In addition, Hix and others (1991) described significant increases in total density due to increased light, nutrients, and moisture reaching the forest floor. Total density increase was from about 42,000 to 62,000 stems per acre.

Sparse oak reproduction is a major concern in many heavily defoliated stands. Because oaks die due to secondary agents after defoliation (Twery, 1991), the oak component in many future stands will depend to a great degree on the survival of both acorns and small oak seedlings (0-1 ft tall) (Galford and others, 1993; Hix and others, 1991). Unfortunately, there is limited knowledge about the ability of oak to successfully compete with birch and red maple (Allen and Bowersox, 1989).

Several insects, including acorn weevils, *Conothrachelus posticatus* (Boheman), nitidulids *Stelidota octomaculata* (Say), and acorn moths, *Valentinia glandulella* (Riley), may destroy great numbers of germinating acorns and new seedings (Galford and others 1993). Also, both white-tailed deer, *Odocoileus virginianus*, and smaller mammals routinely consume most of the acorns in a mature deciduous forest in Virginia (McShea and Schwede, 1993). Subsequently, white-tailed deer browse selectively on oak reproduction. Auchmoody and Walters (1992), for example, conclude that it is futile to plant northern red oak in northwestern Pennsylvania without protecting the seedlings from deer. In many areas, as Ellingwood and Caturano (1988) have stated, the deer population has exceeded the biological carrying capacity (number of deer the land can support in good physical condition) and cultural carrying capacity (maximum numbers of deer that can coexist compatibly with local human populations) (Witmer and deCalesta, 1992). In the face of such browsing pressure, even heroic measures to regenerate oak, such as the seed collection and direct seeding of oak described by Gottschalk (1993) as common practices in France and Germany, may fall short of expectations.

Commonly, moderately heavy defoliation accelerates forest succession toward more shade-tolerant (and less defoliation-prone) species (Campbell and Sloan, 1977b; Clement and Nisbet, 1972; Feicht and others, 1993; Houston, 1981b; Stephens and Hill, 1971). More specifically, Gansner and others (1994) used forest inventory plot data to estimate the likelihood of defoliation in each of south central Pennsylvania's 14 counties in 1978 and 1989. In every county, the latter rating was lower. Changes ranged from -1 to -18 percent and averaged -10.3 percent.

In contrast to widespread scarcity of oak regeneration in other infested areas, oaks often continue to dominate stands in frequently defoliated areas with excessively drained, sandy soils such as Cape Cod, Massachusetts, and the New Jersey coastal plain, or rocky, shallow, ridgetop soils such as those common near Medford, Massachusetts. Also, Oliver (1978) indicated that in central New England a small number of oaks in young stands may become dominant when the stands reach 50 years of age (Twery, 1991).

2.5.1.4. Susceptibility to fire.

Researchers generally agree that heavy defoliation from gypsy moth increases fire danger, although differences in fuels have not been measured nor has the increased fire hazard been calculated (Gottschalk, 1990b). Wildfires are more difficult to control in areas of extensive tree mortality. An abundance of heavy fuel, standing dead snags, dense understory vegetation and numerous fallen trees act in combination to promote spotting, impede line construction and extend mop-up (Tigner, 1992). Additionally, the numerous standing dead snags may act as lightning rods and further increase risk of fire starts by lightning. Fire caused by a lightning strike on one or more of these snags could smolder for several days before being detected (USDA, 1994). However, fires are infrequent during the growing season in eastern hardwood forests. Increased fire hazard in these regions would only be significant from long-term increases in woody fuels following mortality, and not from direct defoliation effects (Gottschalk, 1990b).

2.5.2. Terrestrial Animals and Aquatic Biota

2.5.2.1. Mammals

Among the 62 species of New England mammals listed by DeGraaf and others (1992), some species (black bear, white-tailed deer, and fisher, for example) that retreated as the forest was cleared, returned as the new forest has grown while a second group, such as meadow voles and woodchucks, increased on land cleared for agriculture, and then declined again, as the forest returned. A third group, such as

forest-dwelling moles, shrews, bats, mice, and voles tend to be habitat generalists (Gore 1986). Many of these species are seasonally herbivorous, insectivorous, or omnivorous. And a fourth group, the larger carnivores and omnivores, such as skunks, raccoons, foxes, and coyotes, forage opportunistically and seasonally in a wide range of habitats. In the long-run, the increasingly patchy forest mosaic that usually follows heavy and repeated defoliation should provide a basis for increased mammalian diversity.

Black Bears

A recent gypsy moth outbreak in the Shenandoah National Park caused widespread defoliation, hard mast (acorn) failures, and tree mortality. During this time, 54 radio-collared black bears (*Ursus americanus*) were monitored to determine the impact on the bear population. Cub production and survival and adult and subadult survival rates were not affected by defoliation. Bears exhibited different habitat preferences at all seasons but did not avoid defoliated habitat. Prior to infestation and mast failure, bears utilized acorns as their primary fall food. During defoliation, bears switched to fruits of grape (*Vitis* spp.), pokeweed (*Phytolacca americana*), and spicebush (*Lindera benzoin*) as their primary fall foods. Analysis of nutritional quality indicated there was no decline. Seventy-one percent of bear dens were in tree cavities, primarily in living oaks (mean diameter at breast height = 98 cm). Gypsy moth-induced mortality of den trees was high and by the end of the study 54 percent of living oaks used as dens were dead. The investigators (Vaughan and Kasbohm, 1993) concluded the long-term adverse impact of defoliation on this species may be a reduction in den sites, with natural replacement possibly requiring 50 years. In another study on the reproductive biology of female black bears in northeastern Pennsylvania, Alt (1990) found an average of 2.98 cubs per litter among 211 litters. These bears were studied from 1974 through 1989--a period that includes many years of heavy gypsy moth-induced defoliation in this same general area.

These results suggest that the changes initiated by gypsy moth-induced defoliation will have little effect on black bear populations.

Gray squirrels

In general, gray squirrels have three basic habitat requirements: (1) a diversity of hard mast producing trees,(2) den trees, and (3) a forest canopy closure between 40 and 75 percent (Uhlig, 1956; Allen, 1987). Defoliation has a major influence on hard mast production (Gottschalk, 1990a; McConnell, 1988), and mast production is directly related to squirrel productivity (Gorman and Roth, 1989). Nixon and others (1975) found an average of 4.1 placental scars for gray squirrels in abundant mast

years with low squirrel densities, while in mast-failure years average placental scars were 2.5.

The gray squirrel may be the game species most adversely affected by oak defoliation and mortality (USDA, 1994). The primary influence of the gypsy moth on squirrels may be the loss of acorns, or an increase in non-oak tree species as a result of high forest stand mortality. Thus the effect on habitat in forests with a major oak component may be persistent (Silvester, 1991).

Particularly in the short-term, grey squirrel populations can be expected to decline following heavy defoliation by the gypsy moth.

White-tailed Deer

In heavily forested portions of the Eastern Mixed Forest region, woody twigs probably make up less than 10 percent of the food consumed by deer. When available, foods such as acorns, other mast and fruits, leaves, herbs, clover, grasses, or cultivated crops are preferred foods (Barber, 1984). The loss of acorn production due to gypsy moth defoliation could lead to increased agricultural damage by whitetail populations (Silvester, 1991).

Dense brush or seedling stands are often used by deer for escape or resting cover. In many woodlands, this cover is virtually absent unless even-age management has been practiced recently. Such absence probably limits populations of species such as white-tailed deer (DeGraaf and others, 1992). Whitetails, together with ruffed grouse, wild turkey, and black bear also require both horizontal and vertical habitat diversity (DeGaaf and others, 1992).

Deer populations will definitely respond positively to the forest changes initiated by heavy defoliation.

Bats

Bats are among the habitat generalists. The principal effect of gypsy moth outbreaks on bat numbers will probably occur in places where defoliation and subsequent tree mortality provide more tree cavities. Five of the nine New England bat species use such cavities (DeGraaf and others, 1992).

South of New England the endangered Virginia big-eared bat, *Plecotus townsendii virginianus*, may be affected by gypsy moth since it feeds on insects within the forest. Additionally, the diet of this species consists almost exclusively of moths. Evidence described by Sample and Whitmore (1993) derived from moth wings dropped by the

bat in maternity caves suggests that as much as 70 percent of the moths consumed originated in the forest (Sample and others, 1994). Concern has been expressed that heavy defoliation might be more likely to deplete the bat's food than a chemical spray (Twery, 1990). In a two-year post treatment study following moderate defoliation, however, Sample and others (1994) found that both *B.t.* and defoliation produced fewer effects on the food of the bats than was observed from Dimilin.

Woodland mice

In most forested areas, species of *Peromyscus* are the most common mammals present (Baker, 1968). Once a home range is established, many individuals remain in the same area for the rest of their lives (Stickel, 1968). In general, their populations are relatively constant with few violent fluctuations (Termann, 1968). Recent studies, however, suggest that their densities in the spring in forests dominated by oaks are sometimes limited by the size of the acorn crop the previous autumn (McShea and Rappole, 1992; McShea and Schwede, 1993). Particularly at low acorn-crop densities, these authors suggest that the high proportion of the mast crop eaten by high densities of white-tailed deer may limit populations of these mice and other more mast-dependent species.

2.5.2.2. Birds

DeGraaf and others (1992) list 220 species of birds that spend at least part of each year in New England. Among these, 162 species (69 percent) make at least some use of forested habitats.

During outbreaks, gypsy moths may provoke aggregative responses by a few species of birds (Smith, 1985) and a distinct numerical response by cuckoos (*Coccyzus* spp.) (Campbell, 1975; Smith and Lautenschlager, 1981; Smith 1985). Other species may suffer increased nesting failures (see for example, Cooper and others, 1994; Crocoll, 1991). Generally, however, relations between the insect and birds appear to operate primarily through the effect of defoliation on vegetation.

Several reports describe the short-term effects of a sustained gypsy moth outbreak on both vegetation and birds (Cooper and others, 1987; Thurber, 1992, 1993). In one study area, for example, high canopy cover (greater than or equal to 12 m) declined sharply, while low canopy cover (3 to 5.9 m) increased. The live basal area decreased from 87 percent to 57 percent, and shrub cover averaged 40 percent higher after the outbreak, while shrub species richness increased 24 percent. Distribution of the effects of defoliation on forest vegetation and structure was uneven.

Overall bird density increased from 284 to 337 birds/40 ha. This increase was limited to moderate and low impact plots. Species richness increased from 19 to 23 species per plot, and declines were limited to tree nesters and flycatchers on high impact plots. Increases in low shrub and ground nesters, cavity nesters, low shrub and ground foragers, bark foragers, forest edge species, short-distance migrants, year-round residents, and woodpeckers were widespread, but most pronounced on moderate impact plots.

Dead wood provides critical habitat for many species, and this element is absent or scarce in most second-growth forests. Defoliation also leads to dense vegetation in lower forest strata, providing habitat that is scarce in closed-canopy forests. Limited defoliation would likely benefit non-game forest bird populations (Thurber, 1993).

DeGraaf and Holland (1978) reported similar results, finding significantly fewer numbers of only four out of 36 bird species examined in heavily defoliated areas (Cooper and others, 1987). Similar short term results were reported by DeGraaf (1987) for defoliated and nondefoliated stands in central Pennsylvania. Also, across a longer time horizon, Holmes and others (1986) showed that fluctuations in some bird species coincided with changes in tree species composition in a nondefoliated forest in New Hampshire.

In the long-term, non-game bird diversity will increase in areas that have been defoliated.

Ruffed Grouse

Although ruffed grouse are found in many cover types, their overall geographic range corresponds to that of certain species of poplar, primarily quaking aspen and balsam poplar. In the winter, the birds need cover that furnishes favorable shelter. In spring, they need breeding and nesting cover. In summer, they need feeding grounds for the broods. And in autumn, they need cover that supports fattening foods (Bump, 1947). Most of the woody plants that provide the bulk of the bird's diet are ecological pioneers, transition species in forest succession. Such species are normally present a few years after repeated heavy defoliation. Animal foods, mainly insects, which are abundant in juvenile stands, are eaten primarily by young chicks. Favored forests typically have the heavy shrub understory and fallen trees that are needed for the males' spring courtship and fall territorial displays (Johnsgard, 1989).

Heavy defoliation will create conditions that strongly favor the ruffed grouse.

Wild Turkeys

Wild turkey populations thrive across a broad range of climatic conditions. The most important factor limiting their distribution seems to be precipitation. Turkeys are limited by extremes: deep persistent snow cover at one end and insufficient rain to support the growth of trees at the other (Healy, 1992).

Several key requirements have been suggested regarding wild turkey habitat: (1) lateral cover and cover types with well-developed herbaceous or woody vegetation at 0 to 1.0 meter (0 to 3 feet) above the ground is key to nesting habitat; (2) moisture conditions at mesic sites provide an important microclimatic characteristic for nests; (3) close proximity to brood cover is an important criterion in selection of nest sites; (4) savannas are the best brood-rearing environment; (5) the wild turkey is an opportunistic breeder, adapted to widely fluctuating environmental conditions;(6) roost trees on northeast-facing slopes that allow turkeys to roost above cold-air drainage are important in regions of cold winter weather; and (7) this species is not restricted to wilderness environments, and even suburban environments provide suitable habitat (Porter, 1992).

For many areas of the central Appalachians, the limiting habitat factor for wild turkeys is believed to be the lack of quality brood range (USDA, 1994). Unfortunately, since savannas are the best brood-rearing environment (Porter, 1992), it seems very unlikely that gypsy moth-induced defoliation will often result in increases in such habitat. Also, the loss of hard mast that accompanies and follows heavy defoliation may have adverse effects on these birds. Conversely, the increase in forest diversity that often follows outbreaks may benefit the birds. Certainly, the current abundant wild turkey population in Pennsylvania does not suggest a long-term adverse effect of defoliation on this species.

Neotropical Migrants

Major declines have been observed in populations of many bird species that migrate to and from the tropics (Finch, 1991; Holling, 1988). In an eight-year study of possible effects on these migrants stemming from gypsy moth-induced defoliation, only four species were observed to decline: Acadian flycatcher, eastern wood peewee, blue-gray gnatcatcher, and black-throated green warbler. Of these, both the eastern wood-pewee and the black-throated green warbler also declined regionally according to the Breeding Bird Survey (BBS), a survey that covers a very large geographic area. Similarly, only five species (the black-and-white warbler, hooded warbler, ovenbird, rose-breasted grosbeak, and indigo bunting) exhibited increases. None of these species increased significantly using BBS data (Thurber, 1992). An increase in abundance of open-habitat or edge species, a decrease in abundance of deep-forest species, and an overall increase in bird species richness may be responses

of bird communities to gypsy moth (Cooper and others 1987). Productivity for some species may decrease temporarily, as habitats are made more favorable for nest predators and parasites. While some forest interior bird populations will respond negatively to changes in local conditions, those species will "recover" as the forest recovers. The maintenance of stable populations of neotropical migratory birds over long periods of time will not be damaged by the gypsy moth (Cooper and others, 1994).

Particularly in the long-run, these observations suggest that the chain of events triggered by gypsy moth-induced defoliation will not have an adverse impact on neotropical migratory birds.

2.5.2.3. Reptiles and Amphibians.

DeGraaf and others (1992) list 30 species of reptiles and 26 species of amphibians in New England. Among these, only one species (the five-lined skink) uses the forest exclusively. In addition, another 20 species of reptiles and 25 amphibian species make some use of the forest. Among these species, most (16 species of reptiles and two species of amphibians) make some use of all forest stages, while four species from each group are found only in stands of saplings and poles, or older. Naturally, many species in each of these groups use both marshes, swamps, and bogs (8 reptilian species and 6 species of amphibians), and ponds, lakes, streams, and rivers (13 species of reptiles and 17 species of amphibians) (DeGraaf and others, 1992).

Within the forest, habitat attributes used by these groups (DeGraaf and others, 1992) include the following:

Habitat Attribute	Number of species using attribute	
	Reptiles	Amphibians
Shrub layer	-	1
Wetland shrub layer	1	-
Ground vegetation	3	-
Wetland ground vegetation	2	10
Waterside logs	2	-
Dead and down material	6	11
Litter	2	3
Subterranean habitat	11	13
Slash piles	4	3

In the short term, defoliation-induced increases in insolation on dead and down material, litter, and the materials above subterranean habitats will definitely degrade some of the habitat attributes occupied by some of these species (Schweitzer, 1988). Longer term, species in both groups will benefit from increases in dead and down material.

Timber Rattlesnakes

The timber rattlesnake, *Crotalus horridus* L., is listed as a threatened species by the New York State Department of Environmental Conservation. Typical denning areas for these snakes are rock outcrops and talus near the crest of steep slopes. Such areas often support stands susceptible to gypsy moth outbreaks.

Recently, a field study was conducted on this species in south central New York (Peterson, 1990). Peterson concluded that gypsy moth-induced defoliation would have an adverse effect on rattlesnake denning areas, primarily through reductions in hard mast and consequent adverse effects on the small mammals that these snakes eat. To assure the diverse small mammal community that would provide a more stable and reliable food supply for the snakes, Peterson recommends that a variety of plant communities should be maintained near denning areas.

In the short term, adverse effects of heavy defoliation on rattlesnake numbers are likely. Long-term effects will depend primarily on the diversity and composition of the subsequent plant community.

Amphibians. Within this group, various species of frogs are major users of wetland ground vegetation, salamanders are the major users of dead and down material, and both salamanders and toads are most likely to use subterranean habitats (DeGraaf and others, 1992). These patterns suggest the following: (1) Short-term, adverse effects of defoliation will be most severe on salamander populations, intermediate on populations of toads, and least on frogs; (2) long-term, salamander populations should benefit the most from an increase in dead and down material.

2.5.2.4. Native Lepidopterans

A remarkably similar number of macrolepidopteran species, averaging about 400 species, have been recovered from sites in different places when light traps were used for sampling (Butler and Kondo, 1991; Grimble and others, 1994; Butler and others, 1995). Schweitzer (1994) suggests that these counts may represent roughly two-thirds of the macrolepidopteran fauna.

For at least two reasons, heavy, widespread and prolonged defoliation by the gypsy moth might be expected to reduce macrolepidopteran diversity, at least temporarily. First, a long-standing view holds that competition for food among species with very similar environmental needs will lead sooner or later to the local extinction of the species with inferior competitive ability (Gause and Witt, 1935; Hutchinson and Deevy, 1949). Many laboratory-based studies lend support to this view, but supporting evidence from the far more complex real world is virtually absent. The process of exclusion may be in the distant past and direct evidence for it comes only from a study of introduced species (Varley and others, 1973) such as the gypsy moth. Second, some of the principal parasitoids introduced to combat the gypsy moth have a broad host range and attack numerous species of native macrolepidopterous larvae (Butler, 1993). Under certain conditions, such parasitoids could reduce populations of some of these alternate hosts to very low levels.

In fact, the largely anecdotal information in Schweitzer (1988) strongly suggests that both the gypsy moth and many other lepidopteran species may increase together. For example, the gypsy moth defoliated a record number of acres in 1981. That same year produced the highest number of butterfly species ever for the New Haven, Connecticut area, and for many years stood as the record for eastern North America.

Long-term, some of the lepidopteran species that require oak-dominated forest canopies should decline. Other species may increase in response to the increased plant diversity that often follows a gypsy moth outbreak.

Spring-feeding Lepidopterans

During gypsy moth outbreaks, relatively high numbers of a few species of lepidopteran associates are not unusual, particularly eastern and forest tent caterpillars, *Malacosoma americana* and *M. disstria*, and various species of *Xylenini* (Campbell, 1993; Eggen, 1987; Schweitzer, 1988). According to Schweitzer (1988), very few spring-feeding lepidopterans hatch as late as the gypsy moth and grow as slowly. Thus, for the great majority of these species, many to virtually all larvae should finish feeding before gypsy moth-induced defoliation is severe enough to bring a risk of starvation.

Summer-feeding Lepidopterans

During outbreaks some lepidopterans that eat foods favored by the gypsy moth and emerge during midseason may simply starve. This possibility is ameliorated, however, by staggered oviposition by these species in time and space (Schweitzer, 1988). In addition, whatever defenses host plants of the gypsy moth have evolved against herbivory will probably adversely influence survival of many herbivorous

species (Myers, 1988). Thus, host defenses triggered by defoliation, such as those described by Schultz and Baldwin (1982), could reduce survival among such lepidopterans. Also, these species are among the group that may serve as alternate hosts or prey to several species of gypsy moth parasites and predators.

Short-term, at least, the abundance of this group of lepidopterans may decline somewhat during gypsy moth outbreaks.

2.5.2.5. Other Terrestrial Invertebrates

Sample and others (1995) conducted a study to assess the effects of *Bt* and the gypsy moth on native arthropods (invertebrates such as insects, spiders, and crustaceans). They concluded that gypsy moth defoliation had little effect on most non-lepidopteran taxa. No differences were observed among Coleoptera, Diptera and Trichoptera, except for a spray plot/gypsy moth plot interaction for elaterids and a greater number of Trichoptera at unsprayed plots in 1992. For Hymenoptera, no differences were observed at the order level, but ichneumonids were less abundant at sprayed plots and braconids more abundant at gypsy moth plots.

To examine patterns of arthropod establishment across a size range of forest disturbances, Shure and Phillips (1991) created five sizes of canopy openings in the southern Appalachian Mountains. All feeding guilds were well represented in small openings and herbivore biomass and load (mg of herbivores/g of foliage) were much higher than in larger patches. The results suggest that arthropod abundance and diversity in sprout-dominated forest openings are highly dependent on the extent of environmental differences between patch and surrounding forest.

In the short-run, gypsy moth defoliation may occasionally result in reduced abundance or diversity of terrestrial arthropods. In the long-run, a more diverse arthropod community can be expected.

Insect parasites and predators

Changes in the abundance of second order consumers are related to changes in the abundance of their hosts. For example, since braconid species parasitize gypsy moths (Doane and McManus, 1981), Sample and others (1995) suggested that the elevated abundance of braconids at their gypsy moth plots might indicate a greater number of gypsy moth hosts available to be exploited at these plots.

To persist, organisms in this group have developed an array of host-finding abilities. Consequently, concentrations of hosts or prey are often found and exploited by temporary concentrations of these natural enemies. In the short-term, it is possible

that some specialized species of parasitoids will become rare as gypsy moth-induced defoliation reduces (or even locally eliminates) populations of their hosts. In the long-term, the diversity of these organisms should increase, at least in places where heavy defoliation has led to more horizontal and vertical diversity in the plant community and, subsequently, the herbivores.

Pollinators

Organisms in this group range from a huge variety of arthropods to vertebrates such as hummingbirds (Stein, 1992) and bats (Erickson, 1989). In the main, however, wild bees are undoubtedly paramount in the role of pollinators. In one study, for example, Senft (1990) concluded that most species of endangered plants in an area in Utah seem to rely primarily on wild native bees for pollination.

Wild bees respond positively to a combination of arable fields, enclaves of meadows and pastures, shelterbelts and forests, and uncultivated roadsides (Banaszak, 1992). Within the forest, many kinds of bees nest in dry wood such as logs, branches and twigs. Some dig out their own nest holes, while others nest in holes previously made by insects such as woodboring cerambycids and buprestids. In addition, reductions in the overstory canopy encourage bee-pollinated shrubs such as blueberries; by increasing sunlit soil, such overstory reductions make nest sites for ground-nesting bees more attractive (Adams and Senft, 1994; Batra, 1984).

Wild bees should benefit from both an increased landscape diversity and an increase in sites that are suitable for nesting.

Spiders

Spiders evolved under conditions of limited prey availability, and as a result are extremely efficient prey-capture machines (Riechert and Luczak, 1982). Further, spiders as a group are numerous. In fact, except during outbreaks of herbivorous insects, spiders outnumber all other arthropods (except mites) on the foliage of both balsam fir in the eastern United States (Loughton and others, 1963; Renault and Miller, 1972) and Douglas-fir and the true firs in the Pacific Northwest (Mason, 1992).

As generalist predators, many species of spiders will eat hairy larvae. Fichter (1984), for example, concluded that all spiders will accept Douglas-fir tussock moth larvae in direct proportion to the numbers of tussock moth relative to total potential prey. However, in the gypsy moth life system, the role of these predators remains unclear (Smith and Lautenschlager, 1981).

Spiders are often cannibalistic. Not surprisingly, solitary spiders show a spatial pattern between individuals that is far more regular than random (Burgess and Uetz, 1982). Consequently, spiders showed little evidence of a numerical response to an increasingly dense population of the spruce budworm (Watt, 1963).

The results described in Watt (1963) suggest that the short-term effects of a gypsy moth outbreak on spider abundance should be minimal. Long-term effects are unknown.

Earthworms

Earthworm assemblages composed of native taxa are normally found on sites that have been disturbed only by logging and that are located greater than 50 m from severely disturbed areas (Kalisz and Dotson, 1989). Within specific stands, a significant relationship exists between active worm densities in summer and densities during the rest of the year. Worms are normally absent from stands with lower soil moisture, more acidic soil, shallower soil depths, and lesser amounts of high quality vegetation. High densities of worms are usually present on stands composed of apple, alder, ash, aspen, arrowwood, and fire cherry. Sites with low worm densities were composed of black cherry, beech, red maple, red oak, red pine, sugar maple, and white pine. Stands absent of earthworms also occur at old fields and at some stands composed of conifer species (Parris, 1986). Other studies (Cuendet, 1984; Nordstrom and Rundgren, 1974; Zicsi, 1983) also indicate that overstory vegetation is important in influencing earthworm populations.

These results suggest that changes wrought by a gypsy moth outbreak will tend to have a positive effect on earthworm density in places where heavy overstory mortality moves the plant community toward an earlier successional stage. Otherwise, defoliation should have little or no effect on these animals.

Other Litter and Soil Invertebrates

In a grassland ecosystem, moderate grazing of foliage often increases densities and biomass of belowground herbivores and detritivores. The positive numerical response of soil animals to foliage herbivory results from increased quality (nitrogen concentration) of roots and changes in consumer assimilation efficiencies. Root growth and senescence and acquisition of soil inorganic nitrogen by microbes colonizing senescent roots have been hypothesized as additional causal agents for the soil animal response (Seastedt and others, 1988).

Although the above summary information is both limited and drawn from another kind of ecosystem, it suggests that moderate gypsy moth-induced defoliation will perhaps be followed by increases in the biomass of litter and soil invertebrates.

2.5.2.6. Aquatic Biota

Riparian zones constitute the interfaces between terrestrial and aquatic systems. Changed conditions in the riparian zone will usually have a greater effect on adjacent water bodies with relatively smaller surface area. Consequently, the environmental effect of gypsy moth-induced defoliation can be expected to be greatest in 'upstream' areas where interface reactions are greatest (USDA, 1994).

Few studies have been conducted that might document linkages between defoliation and aquatic fauna. In the absence of such studies, much of the material used here is based on studies of responses by stream fauna to logging.

Fish diversity

In proximity to streams and rivers, gypsy moth outbreaks could result in significant increases in water temperature. In some trout waters, an increase of only one or two degrees could make the water temporarily uninhabitable for trout. Other species, such as sculpins, could also exhibit significant shifts (Moring, 1981).

Stream Salmonids

The removal of vegetation from the banks has immediate effects on the fish. Although trout require some cover, they are scarce in densely shaded water (Hynes, 1970). In the Pacific Northwest, coho salmon, *Oncorhynchus kisutch*, emerged up to six weeks earlier in streams that had been logged (Schrivener and Anderson, 1984). Consequently, with a longer growing season, fingerlings in a logged stream were significantly larger by the fall (Holtby, 1988).

Several investigators reported that logging debris led to increased salmonid densities (Schrivener and Anderson, 1984; Murphy and others, 1986; Tschaplinski and Hartman, 1983). Tschlaplinski and Hartman (1983) noted that log jams and debris act to provide shelter and reduce stream velocities.

In contrast to the increased temperatures and productivity found in streams in recently logged watersheds, Weatherley and Ormerod (1990) suggested that afforestation, by reducing summer temperatures, could lead to marked reductions in rates of development of some invertebrates and fish.

For several post-outbreak years, streams in watersheds defoliated by the gypsy moth may exhibit reduced Acid Neutralizing Capacity (ANC). Downey and others (1994) suggest a kill of stocked rainbow trout in an Appalachian stream resulted from a post-defoliation depression in ANC that exacerbated existing acid deposition effects.

In the short-term, defoliation may result in loss of trout populations in marginal trout streams. Large scale failure of trout populations is not likely (USDA, 1994).

Other Game Fish

Defoliation could conceivably provide increased nutrient supplies to small water bodies and such an event could have short-term adverse effects on the sport fishing in an affected lake.

Aquatic Insects

Macroinvertebrate samples were taken from four logged and four unlogged streams in the Great Smoky Mountains. The logged streams contained greater numbers of organisms and more taxa than the unlogged streams. These differences may be attributable to differences in quantity and quality of leaf litter inputs (Silsbee and Larson, 1983). In addition, Downey and others (1994) suggest that defoliation-induced reductions in ANC may shift the insect fauna in some headwater streams from predominantly acid intolerant species towards more acid tolerant ones.

Results similar to Silsbee and Larson's (1983) have been reported from several Pacific streams (Carlson and others, 1990; Murphy and Hall, 1981; Newbold and others, 1980). The greatest macroinvertebrate increases were found in small (first-order), high gradient (10 to 16 percent) streams (Murphy and Hall, 1981). The effect of buffer width was significant. The macroinvertebrate communities in streams with wide buffers (more than 30 m) could not be distinguished from those of controls (Newbold and others, 1980). In general, timber harvesting activities do not appear to damage aquatic insect habitat and pool abundance is not altered (Carlson and others, 1990).

In contrast to the above, a report on some effects of logging on environments in New Zealand, (Graynoth, 1979) found major modifications in aquatic insects. These included a reduction in the abundance of Plecoptera and certain Ephemeroptera larvae, and an increase in the abundance of oligochaetes, chironomids, and *Deleatium* (Ephemorata) larvae.

Although there are dangers in equating effects induced by logging to those induced by defoliation, the above results suggest that gypsy moth-induced defoliation will have a small and short-term positive effect on the diversity and abundance of aquatic insects.

Molluscs and Crustaceans

Heavy gypsy moth-induced defoliation could affect the abundance of stream-dwelling crayfish and snails. Representatives of these groups in vernal ponds may also be affected. In both streams and ponds, the direction of such effects will depend on local circumstances.

Algal Density

Algal density, particularly in small water bodies with an adequate nutrient supply, may be affected by increases in light penetration. For example, a spring-fed, headwater stream in central Rhode Island was examined from 1979 to 1982. In the first two summers, a dense riparian canopy reduced light penetration to 5 to 18 percent of incident radiation. The lotic macroalgal community was 1 to 4 species covering less than 1 to 35 percent of the stream bottom. In 1981, the surrounding leaf canopy was removed by a massive gypsy moth outbreak. Light penetration increased to 73 percent and the resulting rise in water temperature was 3.7° C. In early August, macroalgal cover increased to a peak of 80 percent of the stream bottom. A less severe gypsy moth defoliation in 1982 did not produce significant differences in macroalgal cover from 1979 and 1980 (Sheath and others, 1986).

In comparison, investigators on a stream in Shenandoah National Park observed no significant changes in periphyton abundance due to defoliation. These investigators speculated that many southern Appalachian streams are so low in nutrients that increased sunlight penetration alone is not enough to increase algal growth (USDA, 1994). Unfortunately, this latter report does not comment on the possible fertilizing effects of frass and leaf fragments on the stream biota.

Detrital Decomposition Rates

Particularly on small, first-order streams, defoliation of streamside vegetation may provide both increased sunlight at the water surface (Sheath and others, 1986) and an infusion of nutrients through frass and leaf fragments. Undoubtedly, such streams experience higher rates of microbial respiration (USDA, 1994). Large increases have been noted in fecal coliform and fecal streptococci densities in streams during periods of maximum defoliation (Corbett and Lynch, 1987).

During gypsy moth outbreaks, detrital decomposition rates should increase markedly in streams that are bordered by susceptible vegetation.

2.5.3. Abiotic Factors

Climatic and soil (edaphic) influences often dominate forest stand productivity. Of these abiotic factors, the quantity and quality of available water and nutrients during the growing season are perhaps the most critical. The lack of adequate soil water during the growing season often results in reduced tree growth and seedling mortality (Loftus and Fitzgerald, 1989).

Increased nutrient return as a result of defoliation (and other kinds of herbivory) has been viewed as a reaction that helps a forest ecosystem counteract losses in productivity in a maturing forest (Mattson and Addy, 1975).

With respect to defoliation caused by the gypsy moth, however, Grace (1986) disagrees. Grace studied defoliated and undefoliated plots in Pennsylvania and found the nutrient content of litter on defoliated plots was 94.6 kg/ha. On undefoliated plots it totaled 73.1 kg/ha. Defoliation was associated with statistically significant increases in N, P and K and a decrease in Ca. This nutrient shift was viewed as a further detriment to the health and vigor of host trees. In the long run, this relatively modest nutrient increase is not likely to compensate for lost productivity (Grace, 1986). An outline of what is known about within-tree nitrogen transfer provides support for Grace's position. Briefly, deciduous trees (such as red oak and larch) resorb most of their foliar nitrogen into stems before normal leaf-fall. In conifers, the resorption rate is much lower. Thus, the normal annual uptake need for this element is much higher in evergreens (Muzika and Gottschalk, 1994). Clearly, deciduous trees that lose their foliage through defoliation at midsummer are likely to lose most of this nitrogen.

2.5.3.1. Water

Temperature

When defoliation reduces the riparian canopy, water surfaces previously shaded from sunlight receive increased incident radiation. A likely consequence of reduced riparian canopy may be elevated stream temperatures during the warm season.

Increases in incident radiation penetration due to gypsy moth defoliation in central Rhode Island were described by Sheath (1986). Prior to defoliation, a dense riparian canopy reduced light penetration at the stream surface to a range of 5 to 18 percent of the incident radiation. Following a "massive" gypsy moth outbreak, light penetration

increased to 73 percent by early July. The resulting water temperature increase was 6.7 °F (3.7 °C).

The temperature regime within a stream is not determined by incident sunlight alone. Groundwater discharge and flow rate also influence stream water temperature. In general, reduced stream temperatures result from increased flow rates and increases in groundwater discharge.

Because stream water temperature at a particular point is influenced by stream flow volume, hydraulic gradient, and ground water discharge, as well as degree of shading (Barton and others, 1985) and upstream conditions, actual changes to water temperatures will vary greatly on a site specific basis and will depend in part upon the degree and duration of defoliation.

Dissolved Oxygen and Eutrophication

Increased organic loading to receiving streams and corresponding rise in biochemical oxygen demand (BOD) can lead to eutrophic conditions and reduced dissolved oxygen levels, particularly in benthic regions. The consequences of defoliation, such as increased nutrient content in forest litter, increased watershed yield and flow rates, could potentially cause increased nutrient transport to water bodies adjacent to severely defoliated areas.

Grace (1986) concluded that although nutrient levels (N, P, K, Mg) were higher for litter components falling during the growing season, increased quantities of nutrients that return to the forest floor as a result of defoliation are relatively modest. However, those nutrients that do reach the forest floor in increased amounts show little evidence of being lost from the site (Corbett and others, 1978). The majority of the N and P added to the forest floor during the growing season remained in the upper layers of the forest floor at the end of the summer, indicating that these nutrients are not rapidly remineralized. Gypsy moth defoliation will likely have little effect on soil fertility and the eutrophication of adjacent water bodies (Corbett and others, 1978; Grace, 1978).

pH Levels and Stream Acidification

Forest defoliation due to gypsy moth may contribute to alterations in stream water chemistry and a reduction in the acid-neutralization capacity (ANC) in streams associated with upland watersheds in the southern Appalachian region (USDA, 1994).

Downey (1991) noted that seasonal improvement in stream water chemistry (increased ANC) occurs when deciduous leaves are present. This ANC increase in the summer months may represent the result of hydronium ion exchange for base cations at the leaf surface. Defoliation due to gypsy moth would serve to temporarily produce conditions typical of the winter months, that is, reduced ANC and lower pH (Downey, 1991).

The geology and soil conditions typical of the southern Appalachian area create small headwater streams with low concentrations of bicarbonate, a natural acid buffering agent. The soils in this area tend to exhibit low water retention and streams are therefore primarily composed of rainwater or snowmelt that has spent relatively little time in the soil. Higher elevation geology is primarily made up of low solubility bedrock formations. These two factors result in low bicarbonate concentrations and reduced ANC. The reduced buffering capacity is probably adequate to buffer low levels of acidic compounds entering the stream. However, the effects of atmospheric deposition of nitrogen and sulfur over the long term must also be considered. Sulfate build-up in the soils may gradually replace bicarbonate until the buffering capacity of the stream becomes affected (Downey, 1991).

Gypsy moth disturbances are also suspected in causing increased nitrate mobility. Due to nitrogen demand as a limiting nutrient, nitrate concentrations in Appalachian upland streams are typically low (USDA, 1994). The tendency for forested watersheds to retain nitrogen was also reported by Grace (1986) and Eagle (1993). However, elevated stream nitrate concentrations have been associated with forest harvest (Vitousek and Melillo, 1979) and defoliation caused by insects including gypsy moth (Swank and others, 1981; USDA, 1994). Elevated nitrate levels were found in 30 streams in the southern Appalachian region over a four year period following initial gypsy moth defoliation. This increase in nitrate mobility has been associated with decreases in base-cation availability and reduced ANC. In none of these streams have nitrate levels fallen to pre-defoliation levels, which suggests that nitrogen retention in soils may be delayed. Based on observations of other defoliated regions, recurrences of defoliation can be expected. Thus the cumulative effects on nitrate mobility and stream acid-base chemistry may be significant (USDA, 1994).

Nutrient Concentration

Defoliation by gypsy moth can accelerate the transfer of nutrients from vegetation to the soil surface, but there is little evidence that these nutrients are lost from the site and enter adjacent water bodies to a significant degree (Grace 1986; Eagle 1993). Incidences of defoliation caused by insects other than gypsy moth provide some basis for which to evaluate gypsy moth defoliation impacts on stream water nutrient concentration. Swank and others, (1981) studied the impact of defoliation by the fall cankerworm on mixed hardwood forests in the southern Appalachians. Slight

elevations in nitrate-nitrogen were reported, but not changes in concentrations of ammonia-nitrogen, phosphorus, sulfate, or major cations such as calcium and magnesium (Corbett and Lynch, 1987; Eagle, 1993; Grace, 1986).

Flow Rate and Watershed Yield

Increases in stream discharge, runoff, and yield are principal factors governing nutrient and sediment transport from terrestrial to aquatic environments. Defoliation due to gypsy moth has been shown to increase water yield from a watershed (Corbett and Lynch, 1987). This is due in part to fewer leaves being available to transpire moisture from the soil (Twery, 1991). Reduced leaf area also results in reduced precipitation interception and transpiration from the leaf surface. Under normal conditions, interception has been shown to amount to 10 to 35 percent of annual precipitation (Waring and Schlesinger, 1984). Water yield measurements of a watershed that had undergone over 75 percent defoliation in New Jersey in 1971 showed a water production increase of over 146,000 gal/acre (1,365 m^3/ha) (Corbett and others, 1987). Stephens, and others (1972) found that soil moisture content (and also xylem pressure potential) were higher in defoliated stands. Eagle (1993) identified increased quantities of stream discharge as a potentially destabilizing effect of herbivorous insects.

Instances of other insect-induced tree mortality also provide evidence of similar watershed yield impacts. A mountain pine beetle epidemic that killed 35 percent of the total growing stock in a 122 km^2 watershed in Montana resulted in measured increases in water yield of 15 percent, a 10 percent increase in low flows, and little increase in peak runoff (Potts, 1984). Helvey and Tiedemann (1978), however, failed to detect changes in peak discharge following defoliations by the Douglas-fir tussock moth in three Oregon watersheds, although degree of defoliation was not reported (Corbett and Lynch, 1987).

Increases in watershed yields may also produce beneficial effects. Increases in water yields from forested watersheds may enhance ground water recharge to drinking water aquifers (Sharpe, 1982).

Sediment Load

Normal sediment loads from forested lands are low. However, increases in stream velocities due to increased yields could possibly lead to increased erosion, sedimentation, and stream turbidity. Also, reduced interception of precipitation by leaf surfaces can lead to increased erosion and weathering effects adding to erosion and sediment transport potential.

However, Corbett and Lynch (1987) reported that timber cutting by itself (excluding the disturbances caused by road building and log skidding) usually has little if any effect on stream turbidity and sedimentation. Thus, increased watershed erosion as a result of gypsy moth defoliation is unlikely.

Structural Habitat

The structural habitat of streams may be altered whenever gypsy moth defoliation results in significant tree mortality in riparian areas. These changes result from the deposition of large woody debris into affected streams. The formation of debris dams in headwater streams acts to trap leaves and other large particulate organic materials, lengthening the time available to be ingested by benthic macro-invertebrates and shredders, and allowing for more complete energy utilization. Also, large woody materials provide improved fisheries habitat by providing better cover (USDA, 1994).

2.5.3.2. Microclimate

Soil and Litter Temperature

Due to canopy loss and increased light penetration to the forest floor, soil and litter temperatures can be expected to increase during the warmer months; however, documented evidence of this occurring is sparse. During conditions of peak defoliation, forests in the Shenandoah National Park in Virginia suffered complete overstory cover loss resulting in increased maximum daily ambient air temperatures of 4.8 °C (Vaughan and Kasbohm, 1993).

Light Penetrating Canopy

Increases in light penetration following heavy defoliation provide increased available solar energy and, when combined with the increased nutrient content in the forest litter, may provide for increased forest productivity following defoliation (Eagle, 1993). Increased levels of soil moisture and decreased leaf litter depth also enhance conditions for the existing understory (Tomblin, 1994).

Relative Humidity Below Canopy

Increases in light penetration due to defoliation and the resulting increase in ambient temperatures act to reduce relative humidity levels. Decreased humidity serves to

reduce fire fuel moisture leading to increased fire hazard over the long term (White and Schneeberger, 1981).

2.5.3.3. Soil

Decomposition

Decomposition is the process by which organic matter is broken down to smaller particles and soluble forms of nutrients that are available for plant uptake. In forest ecosystems the primary organisms responsible for decomposition are fungi and bacteria in the upper soil layers. Decomposition is often referred to as mineralization, which is the transformation of organic matter to inorganic forms such as NH_4^+, CO_2, H_2O, and Ca^{2+}. Mineralization is a critical process determining site fertility and the development of forest soils (Waring and Schlesinger, 1985).

Decomposition rate is, for the most part, due to microbial activity largely influenced by moisture and temperature. Microbial activity increases exponentially with temperature. A generally accepted rule is a doubling of microbial activity with a 10-degree C rise in temperature. Increased soil temperature in localized areas due to defoliation may promote increased microbial activity and decomposition rate. Defoliation has been shown to increase maximum daily temperatures (Vaughan and Kasbohm, 1993). Soil decomposition may be accelerated, but the duration of these effects would be determined by the degree and frequency of defoliation. Moisture content is not usually a limiting factor in microbial activity (Waring and Schlesinger, 1985).

Decomposing bacteria and fungi have high nutrient requirements. Increased nutrient content in litter fall may thus enhance decomposition. This is the case in gypsy moth defoliation in the spring when leaf matter is consumed before nutrient reabsorption takes place (Grace, 1986). The effects of these increased nutrient levels and mineralization could have the effect of enhancing forest regeneration.

Production and Composition of Forest Litter

Defoliation has little effect on total litter production, but does impact the seasonal distribution and composition of forest litter (Grace, 1986). Gypsy moth defoliation has been shown to cause bimodal distribution of litter fall. The first litter fall period is in the spring due to heavy gypsy moth feeding and is composed of frass and small leaf particles as well as dead insects. The second period is the normal autumnal leaf-drop. For the most part, nondefoliated areas exhibit only one short period of whole leaf fall in the fall months.

Grace (1986) compared the distribution and composition of litter in defoliated and nondefoliated areas for one year. In defoliated areas, litter production (frass, leaf fragments, etc.) from May to August reached 1,940 kg/ha, while nondefoliated areas produced only 276 kg/ha. Defoliated areas produced autumnal litter production of 1,322 kg/ha, while nondefoliated areas produced 3,146 kg/ha.

The significance of changes in litter production is primarily due to changes in nutrient content of the litter. Before leaf abscission occurs in the fall, nutrients in the leaves are reabsorbed into perennial portions of the tree. Therefore, litter produced by normal autumnal leaf-drop does not contain high nutrient levels. Gypsy moth defoliation occurs in the spring, prior to nutrient reabsorption, and the frass and other litter components deposited to the forest floor during defoliation are high in nutrients (Eagle, 1993; Grace, 1986).

Amount of Organic Matter

Organic enrichment of the forest floor and litter is associated with changes in litter production and composition. The translocation of nutrients and organic material from the canopy to the forest floor occurs in the form of frass, dead insects, and uneaten plant parts (Eagle, 1993). Litter that is deposited during defoliation differs in composition compared to autumnal litter. Grace reported leaf fragments constituted 85 percent of litter deposited during defoliation, and contributed a greater return of nutrients to the forest floor than did insect frass. During insect outbreaks frass can account for up to 7.8 percent of above ground deposition of organic matter. Carbohydrate rapidly leached and metabolized by microbes in the soil can increase microbial activity and decomposition. These effects can be beneficial and enhance forest regeneration. However, phenolic compounds are also leached and can impede decomposition.

Soil pH

Chemically, pH is an expression of hydrogen ion (H^+) activity. Soil pH is an important factor governing nutrient availability to plants. Most plant nutrients are available within a specific pH range.

Soil pH is strongly correlated with precipitation, which is naturally acidic. Therefore, soils of arid and subhumid regions tend to be alkaline or neutral, and most soils of humid regions are acidic (Foth and Ellis, 1988).

Plant growth itself is an acidifying process since most plant nutrients are positively charged ions. To compensate for imbalances in charge potential, H^+ ions are released as part of nutrient uptake and plant growth. The process of nitrification, enhanced by

gypsy moth defoliation and soil nutrient increases, also generates H^+ ions contributing to acidic conditions.

Defoliation may raise soil pH levels. However, there is little published literature describing the direct effects of gypsy moth defoliation, or defoliation in general, on soil pH conditions.

Erosion Rate

Vegetative cover reduces the erosive potential of water for several reasons: (1) the soil is less prone to saturation, (2) frost-heave is reduced, (3) roots form reinforcing structures, and (4) associated dead organic matter and litter protects the soil surface from the mechanical effects of precipitation. Precipitation interception by leaves also reduces mechanical effects on exposed soil surfaces, but these effects are significantly more pronounced on sloped terrain (Bormann and others, 1974).

Most studies of disturbance-induced erosion have been concerned with the effects of fire where the understory and herbaceous plants have been removed. In these cases, increases in erosion can occur and are largely dependent on fire intensity. In the case of light fires, such as controlled burns, significant soil erosion does not usually occur (Kozlowski and others, 1991).

Gypsy moth defoliation does not normally remove herbaceous plants and shrub species from the understory. The ability of this forest vegetation to reduce erosive potential of water should not be significantly affected by gypsy moth defoliation. Tree removal during timber harvesting itself has been shown to have little effect on stream turbidity and sedimentation, and increased erosion due to gypsy moth defoliation is unlikely (Corbett and Lynch, 1987).

2.6. Scenarios

Dominant variables in determining the effects of a gypsy moth infestation on the ecosystem include defoliation intensity, the number of successive years of defoliation, the proportion of the forest trees that are favored as gypsy moth hosts, the diversity of landforms and cover types in the area, and the spatial scale of the outbreak. Consequently, an infinitely large number of infestation scenarios could be considered. To reduce this potential array of outbreak scenarios to a manageable number (four), the following background conditions will be assumed:

Background Number 1

The forest contains some stands that are immune to outbreaks (less than 20 percent of the stand basal area in preferred species); some stands that are resistant (20 to 50 percent of the stand basal area in preferred species); and some stands that are susceptible (greater than 50 percent of the stand basal area in preferred species). Streams that support trout run through the forest and the forest contains both wetlands and a lake that supports both cold water and warm water game fish.

Background Number 2

Three categories of average, standwide defoliation are considered. These are: normal background defoliation (less than 30 percent), which occurs in the immune stand; moderate defoliation (30 to 60 percent), which occurs in the resistant stand; and high (greater than 60 percent defoliation), which occurs in the susceptible stand.

Background Number 3

Particularly among the vertebrates, seasonal home ranges often shift among several optional cover types. Because defoliation is always patchy across a large, heterogenous area, the short-term consequences of defoliation are minimal for several of these species (for example, black bears and wild turkeys). In summer, for example, such animals may simply avoid a heavily defoliated area by moving to a more resistant cover type. Similarly, in the long term, such animals are able to take advantage of whatever favorable habitat conditions may follow the outbreak. For these species, a forest-wide scenario will be used. In this scenario, the animals range among a mixture of immune, resistant, and susceptible stands. Species and species groups that will be discussed under this scenario include bats, black bears, white-tailed deer, wild turkeys, and ruffed grouse.

Background Number 4

Except for normal background defoliation, each outbreak scenario is assumed to persist for one, two, or three years. During those years, the defoliation category does not change. After the outbreak, the gypsy moth population subsides. Recurring outbreaks are not considered.

2.7. Exposure Assessment

An assumption is that once an area is generally infested, virtually all acres will henceforth support at least a few gypsy moths every year, if those areas have host

plants acceptable to early stage larvae. As long as the gypsy moth population remains sparse, contacts with the insect have minor consequences for most biotic components of the ecosystem (except for the sustenance that they provide to a few specialized natural enemies), and no discernible effect on abiotic components. Rather, for a huge array of biotic and abiotic ecosystem components, the high defoliation that occurs during an outbreak is the event that triggers significant environmental consequences.

Clearly, both the intensity and duration of an outbreak and the interval between outbreaks are critical determinants of subsequent environmental consequences. Practically, then, the assessment of environmental exposure to this pest is largely a matter of assessing probable outbreak duration and the duration of interoutbreak interludes.

Unfortunately, attempts to predict multiyear trends in gypsy moth populations have thus far yielded operationally unsatisfactory results. Consequently, gypsy moth-related management decisions can only make use of projections of defoliation and population trend for the current year, together with information about prior outbreak duration and the current state of relevant ecosystem components. To conform with this limitation, and to avoid unnecessary duplication, further discussion of ecosystem exposure to a gypsy moth outbreak will be provided in the following section by describing how various ecosystem components might react to an outbreak that persists at each of three given intensities for one, two, or three successive years.

2.8. Risk Assessment

2.8.1. Immune Stands (Less than 30 Percent Defoliation)

2.8.1.1. Forest Health

In any forest, foliage-eating insects are always present. All trees are likely to lose some of their foliage every year, and leaves are clearly among the tissues that plants are most capable of replacing. Further, light to moderate defoliation has little, if any, detectable effect on wood production.
The arrival of the gypsy moth should not trigger any changes in forest health, in stands that are immune to outbreaks.

2.8.1.2. Nontarget Species

In the absence of outbreaks, the arrival of the gypsy moth will result in an increase in the density of certain natural enemies of the insect (NPV, entomophagous fungi, parasitoids). Changes in other faunal groups, if any, will be too subtle to measure.

2.8.1.3. Water, Microclimate, and Soil

No detectable changes are anticipated in any of these end points.

2.8.2. Resistant Stands (30 to 60 Percent Defoliation)

2.8.2.1. Forest Health

Short-term

After the first year of defoliation, a slight, but detectable decline will occur in the condition of susceptible overstory trees. With each additional outbreak year, this decline will become more obvious. After two years of moderate defoliation, accelerated mortality will begin to occur in this group, and will be particularly noticeable among subdominant trees. Wood production among susceptible trees will decline slightly. Growth rates will probably increase among many shrub and herbaceous species.

If the outbreak continues for a third year, the abundance of *Armillaria* and *Agrilus* will increase and site occupancy by shrubs and herbaceous cover may increase sharply. Acorn production will be reduced after two years of moderate defoliation, and may continue to be low for as long as five years after the end of defoliation. Soft mast production may increase.

Long-term

If the outbreak ends after one year, trees that declined in condition will return to their pre-defoliation configuration. Other changes in forest health, if any, will be too subtle to measure. After two outbreak years, the stand will begin to lose subdominant trees and become more one-storied. Species favored as food by the gypsy moth may decline in this overstory, and less favored species may increase slightly. Successional changes toward more shade-tolerant species may be accelerated and the ingrowth will probably contain few oak seedlings (relative to the composition of the pre-defoliation

overstory) and many red maples. Hard mast production among surviving dominant oaks will return to predefoliation amounts, or even higher. Susceptibility to fire may increase, and susceptibility to further gypsy moth outbreaks and vulnerability to further damage may be reduced, but only slightly.

2.8.2.2. Nontarget species

Short-term

Even if the outbreak lasts for only one season, it is likely to provoke an aggregative response by opportunistic gypsy moth parasites and (possibly) predators. Such short-term moderate outbreaks will not trigger clear positive or negative responses in other groups. If the outbreak continues for a second year, specialized predators (such as cuckoos and the carabid *Calosoma sycophanta*) will increase. Flocks of other bird species, such as red-winged blackbirds, grackles, and starlings, may also appear to take advantage of the gypsy moth as a food source. Both bird density and species richness may increase, but flycatcher numbers may decline. Declines may occur in amphibians that use dead and down material, litter, and subterranean habitats. In this overall group, these short-term declines may be most noticeable among juvenile red-spotted newts (efts).

Two or three years of moderate defoliation in oak dominated forests will probably trigger short-term reductions in gray squirrel productivity and abundance. If moderate defoliation continues for a third consecutive year, short-term increases may occur in the abundance of aquatic insects, algae, and detrital decay rates, and streams that are already too warm to be good trout habitat may lose their marginal trout populations.

Two or three years of moderate defoliation should trigger minor shifts in associated lepidopterans. Tent caterpillar populations may increase, while a few other lepidopteran species may decline. In response to such changes, corresponding shifts may occur in the diversity and abundance of insect parasites and predators. For these species (both the folivores and their insect natural enemies), most of the shifts that occur will be within the normal range of year-to-year fluctuations in diversity and abundance.

Short-term effects of moderate, sustained defoliation on spiders and earthworms should be positive, but may be too subtle to measure. The biomass of litter and soil invertebrates may increase.

Long-term

Except for the mobile habitat generalists, a moderate outbreak that lasts for only one season is unlikely to provoke any detectable long-term responses by nontarget species. After two or three outbreak years, however, subsequent gray squirrel and white-footed mouse productivity and abundance may either decline or increase, depending on the long-term survival rate and mast producing capability of dominant oaks. Non-game bird diversity may increase, but neotropical migrants may not be affected. Long-term, salamander populations should benefit from increases in dead and down material. Similarly, the diversity of pollinators and other terrestrial arthropods may increase slightly in response to a more diverse plant community.

Long-term detectable changes are not expected in spiders, earthworms, other litter and soil invertebrates, or any of the aquatic groups except stream salmonids.

2.8.2.3. Water

Short-term

In a moderate one-year outbreak, defoliation may not reduce the riparian canopy enough to affect seasonal stream temperature, and other effects should be inconsequential. If the outbreak persists for two or three seasons, however, progressive thinning of the tree crowns coupled with defoliation may result in increased water temperature at midsummer. In rare instances, small streams may also receive a sufficient increase in organic materials to trigger reductions in dissolved oxygen. Even if the outbreak does persist, however, sustained moderate defoliation should not greatly reduce stream acid-neutralization capacity (ANC), although slight elevations in nitrate-nitrogen may persist for several post-outbreak years.

Even moderate defoliation should result in short-term increases in watershed yield, but sediment loads should only increase slightly, or not at all.

Long-term

Sustained moderate outbreaks may result in a slight decade-long (or longer) seasonal increase in water temperature in small streams that are bordered by susceptible vegetation. Some additional woody debris will also be deposited in streams. No other long-term effects of moderate defoliation on water attributes are anticipated.

2.8.3.4. Microclimate and Soil

Moderate defoliation should increase seasonal soil and litter temperatures, and light penetration. Also, soil moisture should increase. Together, these changes will result in increased biological productivity among the plants and decomposers on the forest floor. Those changes are expected to be short-term only. There should be no long-term changes in microclimate or soil condition.

2.8.3. Susceptible Stands (Greater Than 60 Percent Defoliation)

2.8.3.1. Forest Health

Short-term.

After one season of heavy defoliation, the condition of overstory trees will be degraded, and mortality rates will increase slightly, particularly among subdominant trees. Temporary declines will occur in the production of both new wood and hard mast. Minor increases will occur in the growth rates of many shrubs and herbaceous plants.

If heavy defoliation continues for a second consecutive year, more than half the subdominant oaks will probably die within five years from the termination of defoliation. Heavy mortality is also likely among the dominant oaks and major mortality may occur among less-favored overstory species. The secondary pathogens *Armillaria* spp. and *Agrilus bilineatus* will become more abundant. Wood production among the surviving overstory trees will be drastically reduced for a few years and the production of hard mast will stop temporarily. Soft mast production may also be reduced for a few years. Growth rates will be reduced among some shrubs and will accelerate among others, depending largely on their suitability as food for gypsy moth larvae. Any such reductions will be short-lived, however, and the density and site occupancy by shrubs and herbaceous plants will increase greatly.

If heavy defoliation persists for a third season, heavy overstory mortality will ensue. Mortality will be very high among oaks and will extend to many less preferred species, such as red maple and white pine. Even some trees not usually accepted as food by the insect such as white ash and dogwood, may decline, and even die. *Armillaria* abundance will increase greatly, as will the abundance of the twolined chestnut borer. Wood production will be drastically curtailed, temporarily, among surviving susceptible trees; it will accelerate among any competing hard pines. Hard mast production will probably cease completely for about five years and soft mast production may be temporarily curtailed. Densities of susceptible shrubs, such as

hawthorn, hazelnut, and witch-hazel may decline temporarily, but site occupancy by shrubs and herbaceous plants will soon increase. In particular, long-buried seeds of *Rubus* spp. will germinate and may shortly form thickets.

Long-term.

If heavy defoliation stops after a single season, the stand will soon revert to its approximately predefoliation configuration. After secondary pathogens have removed an unusually high number of subdominant trees and a few additional dominants, this one-year outbreak will probably have no other long-term detectable effects on forest health.

If heavy defoliation is sustained for two years, some stands will become distinctly one-storied. Released from competition, surviving dominant trees will recover to their predefoliation condition and will generate accelerated new growth and abundant crops of mast. In such stands, shrub and herbaceous cover will increase. Except on very xeric sites, the abundance of understory red maple will increase. Other stands will revert to an earlier successional stage, as described below.

Many, and sometimes most, overstory trees will die in stands where heavy defoliation is sustained for three successive years. In such instances, the sites will revert to earlier successional stages. Some of these sites will be dominated initially by such plants as blueberries, sweetfern, raspberries or ferns. Ultimately, virtually all such communities will be replaced by young forests, but this process often takes decades. Other sites, where defoliation has not resulted in massive mortality of less-favored trees, may shortly be dominated by red maple or birches. Obviously, acorn (hard mast) production will be greatly reduced, but soft mast production will probably increase dramatically. Without salvage logging, fire hazard will increase greatly.

On excessively drained, sandy soils, oak-dominated stands subjected to a severe, sustained gypsy moth outbreak may continue to be occupied by susceptible, oak-dominated growth. On better sites, however, such stands may be succeeded by stands much less susceptible to further gypsy moth outbreaks.

2.8.3.2. Nontarget Species

Short-term.

With respect to nontarget species, the short and long-term consequences of one year of heavy defoliation, but not a complete defoliation, are probably about the same as the consequences of several consecutive years of moderate defoliation. These were described in a preceding section. Even one year of complete defoliation, however,

will have dramatic effects on Lepidoptera, who could suffer large scale starvation and increased mortality (Schweitzer, 1995).

If heavy defoliation persists for two or more years, opportunistic omnivorous mammals, such as skunks and raccoons, may visit infested sites and will incorporate both gypsy moth larvae and pupae and adults of several species of predaceous *Calosoma* beetles (particularly *C. sycophanta*) in their midsummer diets. Gray squirrel densities and productivity will decline.

Flycatchers will probably decline, temporarily, but other avian groups, such as forest edge species and woodpeckers, will probably increase. Temporarily, the abundance and diversity of neotropical migrants may decline.

Short-term, the small mammals that furnish food for timber rattlesnakes may decline, as progressive overstory decline and mortality leads to hard mast failures. Amphibians will also be adversely affected, as increased insolation heats and dries out habitats on and beneath the surface of the forest floor. These effects will be particularly difficult for several species of forest-dwelling salamanders. A variety of amphibians will also be adversely affected in places where increased sunlight dries up intermittent vernal ponds. The already high mortality rates among tadpoles, for example, will probably be even higher.

Trout populations may decline or disappear, temporarily, in many of the small streams that are bordered by susceptible vegetation. In some of these streams other fish species may increase. Similarly, stream-dwelling crayfish and snails in small streams may be adversely affected. In most larger bodies of water, however, no detectable consequences of defoliation are expected on warm-water game fish, molluscs, or crustaceans.

Both the diversity and the abundance of aquatic insects and algae will increase in streams bordered by susceptible riparian vegetation, and detrital decay rates will rise. In streams bordered by a nonsusceptible riparian zone, these changes, as well as changes in other stream-dwelling groups, will vary inversely with the width of that zone.

Locally and temporarily, some species of forest-dwelling lepidopterans will probably decline. Possibly, the abundance of both the oak-eating folivores, many of their insect parasites, and (possibly) some of their insect predators will decline drastically. In contrast, major increases will occur in populations of specialized gypsy moth natural enemies and possibly in a few species of leaf-eating lepidopteran associates. Densities of both spring- and summer-feeding lepidopterans that eat the foliage of species favored by the gypsy moth may decline slightly, in response to both reductions and biochemical changes in foliage that have been triggered by defoliation.

Short-term, both the abundance and the diversity of other terrestrial invertebrates may decline. Spider densities, however, may increase somewhat, and densities of both earthworms and other litter and soil invertebrates should increase.

Long-term.

Only a few species have been identified that will definitely exhibit long-term negative responses to a massive and sustained gypsy moth outbreak. These species include gray squirrels, which may lose both hard mast and much of their preferred oak overstory, and stream salmonids, which may lose some of their stream habitat for decades. Conversely, the diversity of many species groups will either increase or remain about the same, in response to the increased diversity of plant communities. Species and groups that can be expected to respond positively to more diverse forest vegetation include: any generalist mammals; nongame birds (including neotropical migrants) that do not require mature, closed canopy, multistoried forests; aquatic insects; insect parasites and predators; spiders; and other terrestrial invertebrates. Standing dead trees will be used by cavity-using mammals, cavity-nesting birds, bark foragers, and woodpeckers. Dead and down material will provide dens for mammals, courtship and display sites for nongame birds, and habitat for a variety of reptiles and amphibians. In streams, logs and debris dams will improve habitat for both fish and aquatic insects.

In the litter and soil earthworm densities will usually increase in places where the plant community has reverted to an earlier successional stage. Other litter and soil invertebrates may be unaffected.

2.8.3.3. Water

Short-term.

If the riparian canopy is composed of susceptible species, a one-year outbreak will result in elevated water temperatures, particularly in small streams. If the outbreak persists, this effect will continue. Small streams bordered by susceptible vegetation will receive an increase in organic materials. In response, a rise in biological oxygen demand may lead to reduced dissolved oxygen and increased eutrophication. In upland watersheds, a decrease may occur in the acid-neutralization capacity of streams. Elevated nitrate-nitrogen concentrations may persist for several years.

Watershed yields will increase, but sediment loads should only increase slightly.

Long-term.

In small streams that are bordered by susceptible vegetation, elevated water temperatures may persist for decades. Wood debris will increase in these and other streams, and will form debris dams and pools. Watershed yields will return to predefoliation quantities.

2.8.3.4. Microclimate and Soil

Increased light penetration will result in seasonal elevations in soil and litter temperature. Soil moisture content will increase, temporarily. In response, biological productivity will increase among the plants and decomposers on the forest floor. Increases in nutrient levels in litter fall will also promote microbial activity and increase the rate of soil decomposition and mineralization. As with the prior scenario, these changes are expected to be short-lived.

2.8.4. Responses by Selected Habitat Generalists

Probable short-term and long-term responses by some habitat generalists to a forestwide gypsy moth outbreak are described here. Species considered include black bears, white-tailed deer, wild turkeys, and ruffed grouse, together with one species group (bats).

As already noted, defoliation is always patchy across a large heterogeneous area. When seasonal defoliation episodes occur, these generalist species may remain in defoliated areas or may move on, depending on the extent to which the defoliated area meets their needs. Bears, for example, are highly insectivorous. Given an abundance of gypsy moth larvae and pupae, it seems likely that these mammals will join the list of gypsy moth predators. Similarly, ruffed grouse and wild turkeys are known to have very broad dietary ranges. Both these species probably sample gypsy moth larvae and pupae, on occasion. Also, several species of bats would definitely eat adult moths, although there is a temporal separation in their activity patterns. In any case, all of these species have large home ranges. For all but ruffed grouse, home range is in excess of 50 acres (DeGraaf and others, 1992). For the grouse, these authors specify a 10 to 50 acre home range. For all these species, the option to avoid a heavily defoliated area is usually viable. During nesting, however, any grouse or turkeys that have nested in heavily defoliated areas may experience an increase in nesting failures.

Long-term, black bears may have to cope with a reduction in den trees, and bears, white-tailed deer, and wild turkeys may have to adapt to a long-term reduction in hard mast. But for this group, most of the other long term affects should be positive. For

the omnivorous bears, the more diverse landscape will provide a wider range of foraging opportunities. For white-tailed deer, the abundance of young growth will provide both abundant forage and escape cover. For wild turkeys, this same young growth will provide nesting habitat. And for ruffed grouse, this same growth provides cover for drumming logs, a place for chicks to forage, and preferred autumn habitat.

As trees die, an increase in standing dead trees will provide den sites for bats. The increase in dead and down logs will provide an abundance of potential drumming and courtship sites for ruffed grouse.

2.9. Summary

Indices of 56 projected short-term and long-term ecosystem responses to various gypsy moth-induced defoliation regimes are shown in Tables 2-3 and 2-4. For each ecosystem component, the index can range from--(the greatest expected decrease), through 0 (no expected change), to ++ (the greatest expected increase). For example, a modest increase (+) is projected as the long-term change in soft mast production resulting from a high intensity outbreak that lasted one year (Table 2-6). A question mark indicates unusually high uncertainty about the category. For each category, a subjective estimate is provided of confidence in the projections.

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Figure 2-1. Generally infested, transition and uninfested areas as of November 1994

Figure 2-2. Predicted generally infested, transition and uninfested areas by the year 2010

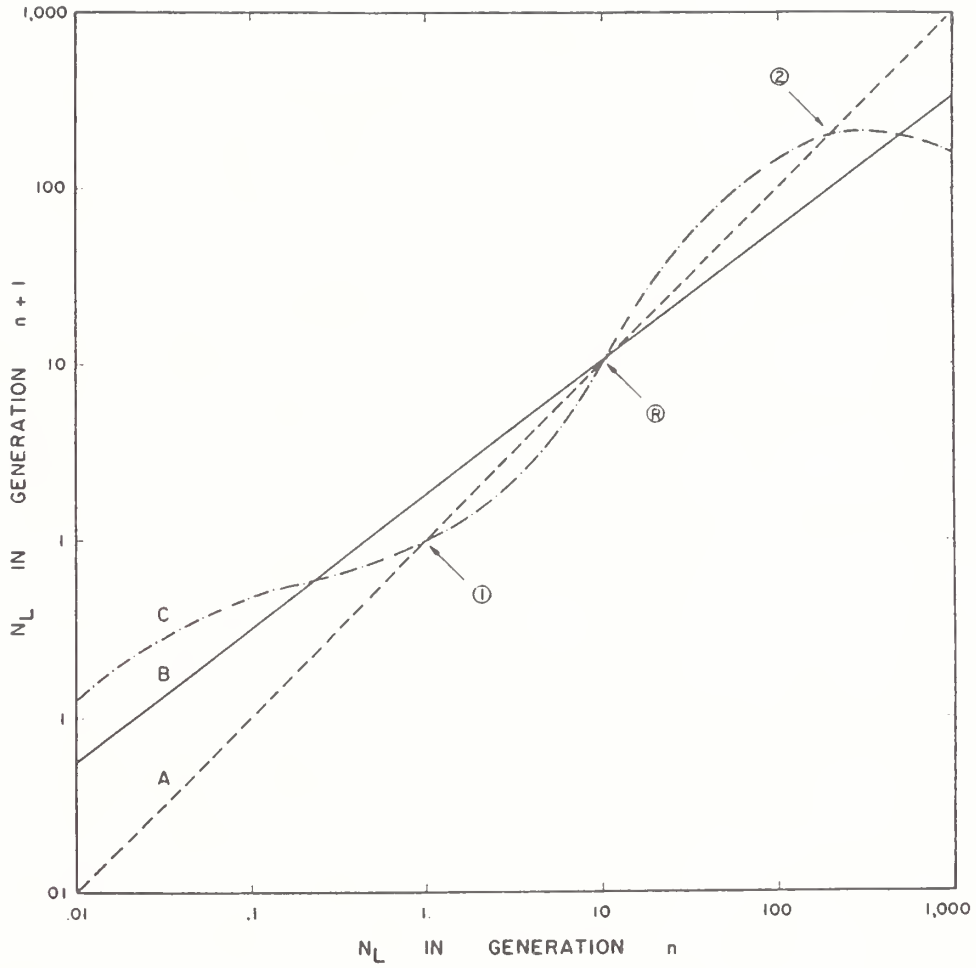

Figure 2-3. The relation of larval population (N_L) in generation n + 1 to larval population in generation n. A - hypothetical line with a slope of unity. B - actual regression line from key-factor analysis. C - hypothetical sinuous reproduction curve. (1) - low equilibrium point. (2) - high equilibrium point. (R) - population release point (Morris, 1963)

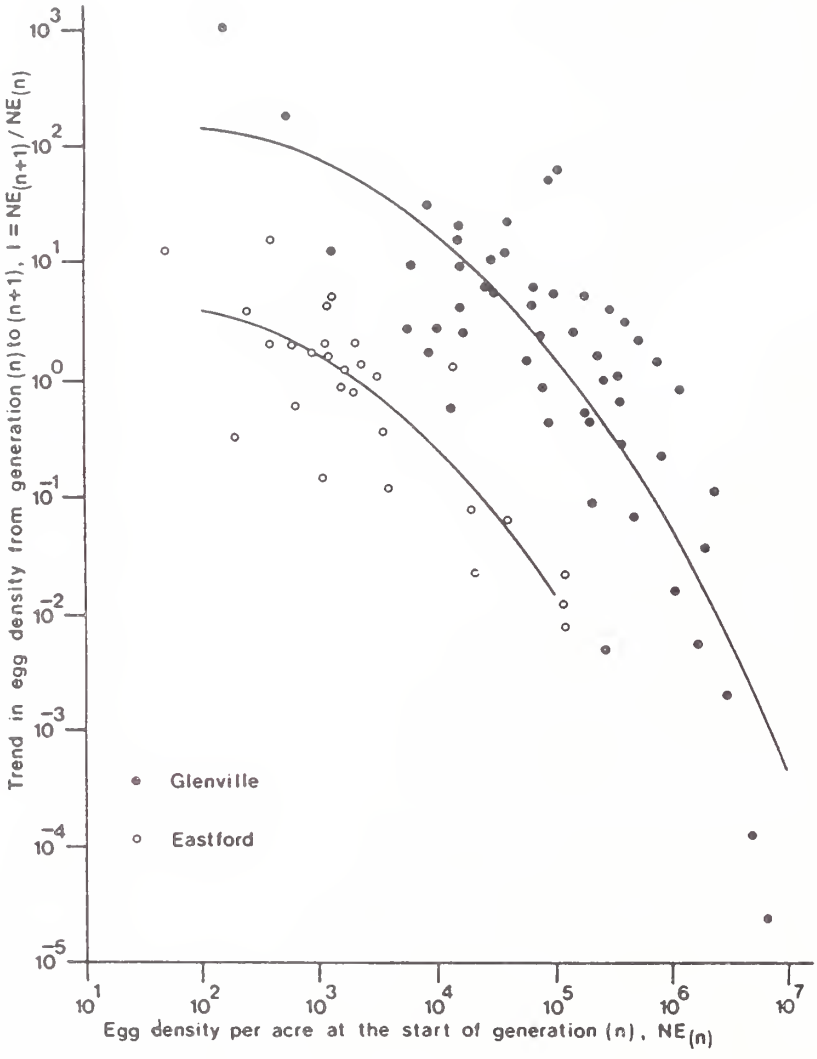

Fig. 2-4. Relationship between trend in egg density (I) and egg density per acre at the start of generation ($NE_{(n)}$); Glenville and Eastford data (1958-64 and 1965-68) (Campbell and Sloan, 1978a)

Table 2-1. Comparison of European and Asian strain of the gypsy moth (USDA, 1992)

	European	Asian
Adult - Male	Strong flier Attracted to pheromone	Strong flier Attracted to pheromone
Adult - Female	Flightless	Strong flier (>30 km) Attracted to light
Larvae	1st instars disperse Uniform color	1st and 2nd instars disperse Highly variable color
	Main hosts: oak, birch, poplar willow, alder	Main hosts: larch, birch, and willow as well as oak
	Larvae feed in the canopy at night and move to resting sites during the day	Larvae feed in the canopy at night and remain on the host during the day
	Late instars use artificial, or manmade objects for resting locations	Late instars use artificial, or manmade objects as resting locations
Pupae	Pupates in litter	Pupates on foliage
Egg Masses	On tree trunk, rocks, litter	On foliage, tree trunks, rocks, objects associated with light
Mortality	NPV, *Bt*, fungus, parasites and various predators	NPV, *Bt*, fungus, microsporidia, numerous parasites and predators

Table 2-2. European Gypsy Moth Host Susceptibility (Gottschalk 1988)

Class I: Species that are favored by gypsy moth during all larval stages.

Overstory species:

apple; basswood; bigtooth and quaking aspen; gray, paper, and river birch; boxelder; larch; American mountain-ash; lombardy poplar; all oak species; poplar; sweetgum; willow.

Understory species:

alder, hawthorne, hazelnut, eastern hophornbeam, serviceberry, all sumac species, witchhazel.

Class II: Species that are favored by gypsy moth following earlier larval stages.

Overstory Species:

chestnut; eastern hemlock; all pine species; all spruce species.

Class III: Nonpreferred species fed upon by later larval stages only when preferred foliage is unavailable.

Overstory species:

American beech; black and yellow birch; blackgum; Ohio and yellow buckeye; butternut; sweet and black cherry; eastern cottonwood; cucumbertree; American and slippery elm; hackberry; all hickory species; Norway, red, silver, and sugar maple; pear; silver poplar; sassafras; black walnut.

Understory species:

blueberries; pin cherry; chokecherry; American hornbeam; pawpaw; persimmon; redbud; sourwood; sweetfern.

Class IV: Unfavored species that are rarely fed upon.

Overstory species:

all other ash species; baldcypress; northern catalpa; eastern redcedar; balsam and fraser fir; American holly; horsechesnut; Kentucky coffee-tree; black and honey locust; mulberry; sycamore; yellow-poplar.

Understory species:

all azalea species; dogwood; elderberry; grape; greenbrier; juniper; mountain and striped maple; rhododendron, all rubus species; sheep and mountain laurel; spicebush; sarsaparilla; all viburnum species.

Table 2-3. Short-term (st) and long-term (lt) changes expected in various ecosystem components, assuming a moderate intensity outbreak (30-60 percent defoliation) that persists for 1, 2 or 3 successive years.

	Moderate defoliation (30-60%), persisting for:					
	1 yr		2 yrs		3 yrs	
Ecosystem component	st	lt	st	lt	st	lt
Trees						
Condition R[1], H[4]	-	0	-	0	-	0
Mortality R, I[4]	0	0	+	0	+	0
Diversity D[2], I	0	0	0	+	0	+
Tree pathogens A, H	0	0	+	0	+	0
Diameter growth R, H	0	0	-	0	-	0
Volume changes R, H	0	0	-	0	-	0
Shrubs A[3], H	0	0	+	0	+	+
Herbaceous cover A, I	0	0	+	0	+	+
Hard mast A, H	0	0	-	0	-	0
Soft mast A, L[6]	0	0	+	0	+	0
Fire hazard R, L	0	0	0	+	0	+
Gypsy moth natural enemies A, H	+	0	+	0	+	0
Mammals D, I	0	0	0	0	0	+
Bats A, L	0	0	0	0	0	0
Black bears A, I	0	0	0	0	0	0
White-tailed deer A, I	0	0	0	0	0	+
Gray squirrels A, H	0	0	-	?	-	?
Deer mice A, I	0	0	-	?	-	?
Birds D, H	0	0	0	+	0	+
Wild turkeys A, I	0	0	0	0	0	0
Ruffed grouse A, I	0	0	0	0	0	+
Neotropicals A, I	0	0	0	0	0	0
Reptiles and Amphibians D, L	0	0	-	0	-	0
Timber rattlesnakes A, I	0	0	0	0	-	0
Amphibians A, I	0	0	-	0	-	+
Fish D, L	0	0	0	0	0	0
Stream-using salmonids A, L	0	0	-	0	-	-
Other game fish A, L	0	0	0	0	0	0
Invertebrates D, I	0	0	0	0	0	+
Native Lepidoptera D, I	0	0	0	0	?	0
Spring Lepidoptera A, I	0	0	0	0	?	0
Summer Lepidoptera A, L	0	0	0	0	?	0

Table 2-3, *continued ...*

Ecosystem component	Moderate defoliation (30-60%), persisting for: 1 yr st	1 yr lt	2 yrs st	2 yrs lt	3 yrs st	3 yrs lt
Litter invertebrates A, L	+	0	+	0	+	0
Insect parasites and predators A, I	0	0	-	0	-	+
Spiders A, I	0	0	+	0	+	0
Pollinators A, I	0	0	0	+	0	+
Aquatic insects A, L	+	0	+	0	+	0
Earthworms A, I	0	0	+	0	+	0
Mollusks A, L	0	0	-	0	-	?
Crustaceans A, L	0	0	-	0	-	?
Detrital decay rate R, L	+	0	+	0	+	0
Algal density A, L	0	0	+	0	+	0
Water						
Temperature A, I	0	0	+	+	+	+
Dissolved oxygen A, L	0	0	-	0	-	0
Nutrients A, L	0	0	+	0	+	0
Flow rate R, I	+	0	+	0	+	0
Yield A, I	+	0	+	0	+	0
Sediment load A, L	0	0	0	0	+	0
Microclimate						
Soil-litter temp. A, H	+	0	+	0	+	0
Light below canopy A, H	+	0	+	0	+	0
R.H. below canopy A, H	-	0	-	0	-	0
Soil						
Decomposition rate R, I	+	0	+	0	+	0
Litter production A, I	0	0	0	0	0	0
Quantity organic matter A, I	+	0	+	0	+	0
Soil pH R, L	0	0	0	0	0	0
Erosion rate R, I	0	0	0	0	0	0

[1]R = rate or appearance
[2]D = diversity
[3]A = abundance or quantity
[4]H = relatively high confidence in these projections
[5]I = intermediate confidence in these projections
[6]L = relatively low confidence in these projections

Table 2-4. Short-term (st) and long-term (lt) changes expected in various ecosystem components, assuming a high intensity outbreak (>60 percent defoliation) that persists for 1, 2 or 3 successive years.

	High defoliation (>60%), persisting for:					
	1 yr		2 yrs		3 yrs	
Ecosystem component	st	lt	st	lt	st	lt
Trees						
Condition R[1], H[4]	-	0	--	0	--	0
Mortality R, I[5]	+	+	++	+	++	0
Diversity D[2], I	0	+	+	+	+	+
Tree pathogens A[3], H	+	+	+	+	++	++
Diameter growth R, H	-	0	--	+	--	+
Volume changes R, I	-	0	--	+	--	+
Shrubs A, I	+	+	+	+	+	++
Herbaceous cover A, I	+	+	+	+	+	++
Hard mast A, I	-	0	--	?	--	-
Soft mast A, I	0	+	0	++	0	++
Fire hazard R, I	0	+	0	++	+	++
Gypsy moth natural enemies A, H	+	0	++	0	++	0
Mammals D, I		0	0	0	+	0+
Bats A, I	0	0	0	0	0	+
Black bears A, I	0	0	0	0	0	0
White-tailed deer, A, H	0	+	0	+	0	++
Gray squirrels A, H	-	-	-	-	-	--
Deer mice A, H	-	0	-	-	-	-
Birds D, I	0	+	0	+	0	+
Wild turkeys A, I	0	0	0	0	0	0
Ruffed grouse A, H	0	+	0	+	0	++
Neotropicals A, I	0	0	-	0	-	0
Reptiles and Amphibians D, I	-	+	-	+	--	+
Timber rattlesnakes A, L	-	0	-	0	-	0
Amphibians A	-	+	-	+	--	+
Fish D, I	0	0	0	0	0	0
Stream-using salmonids A, H	-	-	-	-	-	-
Other game fish A, L[6]	0	0	0	0	0	0
Invertebrates D, L	0	+	0	+	0	+
Native Lepidoptera D, L	-	0	-	0	-	0
Spring Lepidoptera A, L	-	0	-	0	0	-
Summer Lepidoptera A, L	-	0	-	0	0	-

Table 2-4, *continued ...*

Ecosystem component	High defoliation (>60 %), persisting for: 1 yr st	1 yr lt	2 yrs st	2 yrs lt	3 yrs st	3 yrs lt
Litter invertebrates A, L	+	0	+	0	?	0
Insect parasites and predators A, H	-	+	-	+	-	+
Spiders A, L	+	0	+	0	+	0
Pollinators A, I	0	+	0	+	0	+
Aquatic insects A, L	+	+	+	+	+	+
Earthworms A, L	+	0	+	+	+	+
Mollusks A, L	-	?	-	?	-	?
Crustaceans A, L	-	?	-	?	-	?
Detrital decay rate R, I	+	+	+	+	+	+
Algal density A, I	+	0	+	+	+	+
Water						
Temperature A, H	+	+	+	+	+	+
Dissolved oxygen A, L	-	0	-	-	-	-
Nutrients A, I	+	+	+	+	+	+
Flow rate R, I	+	0	+	0	+	0
Yield A, I	+	0	+	0	+	0
Sediment load A, I	+	0	+	0	+	0
Microclimate						
Soil-litter temp. A, H	+	0	+	0	+	0
Light below canopy A, H	+	+	+	+	+	+
R.H. below canopy A, H	-	0	-	0	-	0
Soil						
Decomposition rate R, I	+	0	+	0	+	0
Litter production A, I	0	0	0	0	0	0
Quantity of organic matter A, I	+	0	+	0	+	0
Soil pH R, L	0	0	0	0	0	0
Erosion rate R, I	0	0	0	0	0	0

[1]R = rate or appearance
[2]D = diversity
[3]A = abundance or quantity
[4]H = relatively high confidence in these projections
[5]I = intermediate confidence in these projections
[6]L = relatively low confidence in these projections

Section 3
Characterization of Active Management Strategies and Treatments

3.1. Description Of Active Gypsy Moth Management Strategies

The Federal role within the Gypsy Moth Management Program is a function of the specific Federal agency's authority, the presence and numbers of gypsy moths, the geographical location of the infestation, and the control strategy required (Fig. 3-1). When gypsy moths are discovered in an area, a decision must be made regarding their management (Fig. 3-1). All gypsy moth management activities conducted by the Federal government follow an Integrated Pest Management (IPM) philosophy. IPM is defined as the selection, integration, and implementation of pest control actions on the basis of predicted economic, ecological, and sociological consequences (Council on Environmental Quality, 1972). Using IPM, treatments do not take place until damage has exceeded an acceptable threshold. This threshold is often purely economic, but other factors for which economic data are difficult to collect may also be considered. In the case of gypsy moths, aesthetic and nuisance factors are often considered. For instance, when gypsy moth populations increase, they become an increasing nuisance to residents. Eventually the gypsy moth population provides a sufficient nuisance to cause residents to consider active gypsy moth management. The economic threshold for gypsy moth is the maximum level of defoliation that a forest can sustain without a resulting decrease in overall forest productivity. This relates primarily to timber production, but may also be related to loss of revenue at certain locations from decreased tourist visits due to the lowered aesthetic appeal from the damage caused by gypsy moths to the forested area.

The acceptable threshold of defoliation due to gypsy moth feeding depends upon local conditions. A woodland area may tolerate defoliation one year with minimal loss of trees, but damage to trees may exceed the economic threshold in the first year of defoliation if the forest is already stressed due to other environmental factors. If the threshold for treatment has not been exceeded, the obvious management strategy is no active management of gypsy moths. In other words, moths are left to their own devices and no treatments are undertaken. This no active management strategy and its ecological risks are described in Section 2 of this risk assessment. If, however, the threshold for treatment is reached and active management of gypsy moths is required, the treatments will be applied according to one of three active management strategies: suppression, eradication, and slow-the-spread.

3.1.1. Suppression

Suppression programs are designed to limit the amount of damage to forested areas from infestations of gypsy moths. The goal of insecticide applications in a suppression strategy is to reduce gypsy moth populations to levels that result in damage below an acceptable threshold. Suppression is the active management strategy employed in the generally infested area. Effectiveness of a suppression program in limiting damage depends upon the density of gypsy moth populations and the size of the infested area relative to the area to be treated. In addition to limiting damage, this strategy also reduces the propensity of the moths to disperse by suppressing their populations to levels resulting in less impetus for movement to locate new food sources. A suppression strategy presumes established populations throughout the potential treatment area along with some continuing damage from gypsy moths at an acceptable level.

The Federal role in the gypsy moth suppression program, in part, varies depending upon the size and number of sites that require treatment. The goals of this strategy are achieved through IPM treatments using chemical and biological tools. Treatments in a suppression program may include applications of *Bacillus thuringiensis* var. *kurstaki* (*B.t.k.*), diflubenzuron, or gypsy moth nucleopolyhedrosis virus (NPV). *B.t.k.* is a bacterium that produces a crystal toxin pathogenic to gypsy moth another lepidopteran larvae. NPV is a naturally-occurring virus that causes a disease specifically in gypsy moth larvae. *B.t.k.* and NPV are biological insecticides. Diflubenzuron is a chemical insecticide that inhibits chitin synthesis (exoskeleton production) in molting insects. Formulations of *B.t.k.*, diflubenzuron, and NPV are applied in the early spring, soon after leaf emergence, to kill gypsy moth larvae (usually early instars). Because treatments in suppression programs are designed to prevent damage from exceeding a threshold, these programs are generally less intense than applications in eradication programs where the effort aims to eliminate gypsy moth populations in the treated area.

3.1.2. Eradication

Eradication strategies are designed to eliminate infestations of gypsy moth from localities outside the generally infested area. These infested areas have gypsy moth populations that are usually separated by more than 100 miles from other established populations of gypsy moths, have no previous history of gypsy moth population outbreaks, and previous trapping efforts have only occasionally yielded low numbers of male gypsy moths. The goal of insecticide applications in an eradication project is to reduce gypsy moth larvae to levels insufficient to maintain a viable population in the treated area. This strategy is based upon the contention that continuing survival of any gypsy moths within the treatment area poses too great a risk of damage to shade trees and forested areas within and adjacent to that area. In other words, the acceptable threshold gypsy moth population is zero. This presumes that the intense

effort to eliminate viable larvae is, in the long term, less costly than the effort to suppress, slow-the-spread, or take no action.

Eradication of isolated infestations has generally been more effective than eradication attempted at recently established sites near an established population that is dispersing. The decision to eradicate usually considers the cost of quarantines and the impact on commerce from the potential gypsy moth infestation in a given area. This strategy is usually applied to eliminate the gypsy moth when it spreads artificially to an area where it is not a permanent resident.

The Federal role in a gypsy moth eradication program, in part, varies depending upon the size and density of the gypsy moth population, the number and size of infested sites, and the potential for reinfestation of those sites due to reintroduction or close proximity to other infested areas. The goal of this strategy can be achieved through several treatments with chemical and/or biological tools (See Section 1. Suppression). These may include ground and/or aerial applications of formulations of *B.t.k.*, diflubenzuron, and NPV. Aerial applications of Disparlure, a male attractant, may be made to disrupt mating of the gypsy moth. Eradication strategies also make use of traps to detect, delimit (determine the extent), and monitor gypsy moth populations. The traps are baited with Disparlure to attract male moths. Either a sticky substance traps the male moths inside or an insecticide (dichlorvos) is used in the trap to kill the male moths. These traps also may be employed in a mass trapping effort to attract and kill adult male gypsy moths.

Treatments in eradication programs are generally more intense than in either suppression or slow-the-spread programs. This relates to the greater effort involved in elimination of the gypsy moth from a site rather than in suppressing populations or population spread.

3.1.3. Slow-The-Spread

The slow the spread strategy is designed to reduce the natural rate of dispersion of gypsy moths from the generally infested area to the uninfested area. The area where the slow-the-spread strategy is used is often referred to as the transition zone. The first outbreak of gypsy moths in a previously uninfested area often results in the most severe damage. The goal of insecticide applications in a slow-the-spread effort is to apply treatments so low-level populations of gypsy moths in previously uninfested areas are unable to increase to population levels that result in a propensity to disperse.

Dispersion reduction through slow-the-spread efforts has achieved mixed results. These efforts tend to be more effective in areas where the food plants of the uninfested areas are not the preferred hosts of gypsy moth and in years when the

weather conditions are less favorable to the moths. This strategy presumes the effort to restrict movement or decrease survival will slow the spread of dispersing gypsy moths into the uninfested area.

The Federal role in a program to slow the spread of gypsy moth dispersion will vary depending upon the size and density of the gypsy moth population, the number and size of gypsy moth detection sites, and the potential for dispersion from those sites to adjacent, uninfested areas. The goal of this strategy may be achieved through several IPM treatments. This includes applications of *B.t.k.*, diflubenzuron, Disparlure, and NPV (See Section 3.1.1. Suppression). Slow-the-spread may use gypsy moth trapping (utilizing the male attractant Disparlure and the insecticide dichlorvos) to detect, delimit, or monitor gypsy moth populations. Mass trapping of male moths may also limit dispersion.

3.2. Description Of Active Gypsy Moth Management Treatments

Figure 3-1 provides a general decision-tree for the National Gypsy Moth Management Program. If moths are detected, a decision must be made whether or not to initiate an active management program. This decision will depend upon whether or not the acceptable gypsy moth population thresholds are exceeded. If thresholds are exceeded, the type of management program must be determined. This is largely a question of geography: is the area to be treated within the generally infested area, clearly outside the generally infested area, or in the transition zone?

Once the strategy has been identified, the actual treatments to be used must be determined. There are seven basic treatments available to be used by themselves or in combination. Table 3-1 provides information on the rates for each treatment.

Diflubenzuron, a chemical treatment used in all three active management strategies, is generally applied aerially. It effectively reduces gypsy moth populations and protects foliage (Twardus and Machesky, 1992). It is considered the most effective treatment for controlling very high gypsy moth populations.

Bacillus thuringiensis var. *kurstaki* (*B.t.k.*) also is used in the three active management strategies, usually by aerial application. It has been found to be effective in preventing excessive defoliation. It is, however, somewhat less effective than diflubenzuron in protecting foliage and reducing gypsy moth populations (Twardus and Machesky, 1992).

The gypsy moth nucleopolyhedrosis virus (NPV) is very specific to gypsy moths in its toxicity. It has been found to provide effective reductions in dense gypsy moth populations (Reardon and Podgwaite, 1992). Because NPV is slower acting than diflubenzuron and *B.t.k.*, it does not provide as much protection from defoliation in

the year it is applied as do the other two insecticides. However, it does protect against defoliation in the following year (because it reduces gypsy moth populations). Unfortunately, NPV is in limited supply and therefore rather expensive to apply. Its high degree of specificity for gypsy moth, however, makes it particularly desirable for use in sensitive areas.

Mating disruption is a technique used in eradication and slow-the-spread strategies. It involves aerial or ground release of polymeric flakes, tapes, or beads impregnated with the gypsy moth sex pheromone. These releases confuse male gypsy moths who cue on the pheromone when looking for a mate, thus disrupting mating. This technique, however, is only effective at low gypsy moth population densities (generally less than 10 egg masses per acre).

Mass trapping, like mating disruption, is directed toward male gypsy moths. Two types of traps are used to attracted and capture male moths: (1) a standard delta trap baited with a pheromone impregnated strip to lure male moths which become stuck to the sticky inside walls of the trap; or (2) a milk carton trap, baited with pheromone and also containing a laminated plastic strip impregnated with dichlorvos (an insecticide) that kills the moths. The traps are laid out in an intensive grid pattern. The milk carton trap has a much larger capacity than the delta trap and is used most often in the transition area to monitor gypsy moth populations. Delta traps are used in eradication of isolated gypsy moth infestations. The use of mass trapping is highly labor intensive and is used only in small areas with low population densities (less than 10 egg masses per acre).

The sterile insect technique also has a potential in eradication and perhaps slow-the-spread strategies. Although more research is needed before this technique is fully operational. Because a female moth usually mates only once, if it mates with a sterile male (or a male that can pass on the sterile trait), then the offspring will not be viable. For this technique to be successful, large numbers of steriles must be released (a ratio of at least 34 steriles to 1 wild-type is desired). Steriles are generally released as sterile pupae. The logistical problems associated with producing sufficient numbers of sterile gypsy moths limit the use of this technique. It is practical only at very low gypsy moth densities (usually less than 2.5 egg masses per acre).

3.3 References

Council on Environmental Quality, 1972. Integrated pest management. November 1972.

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Twardus, Daniel B.; H.A. Machesky. 1992. Gypsy Moth Suppression in the Northeast: Summary of the Treatment Monitoring Database, 1989-1990. USDA Forest Service, Northeastern Area, State and Private Forestry, Forest Health Protection, Morgantown, WV, NA-TP-08-92. September 1992. 11 p. + app.

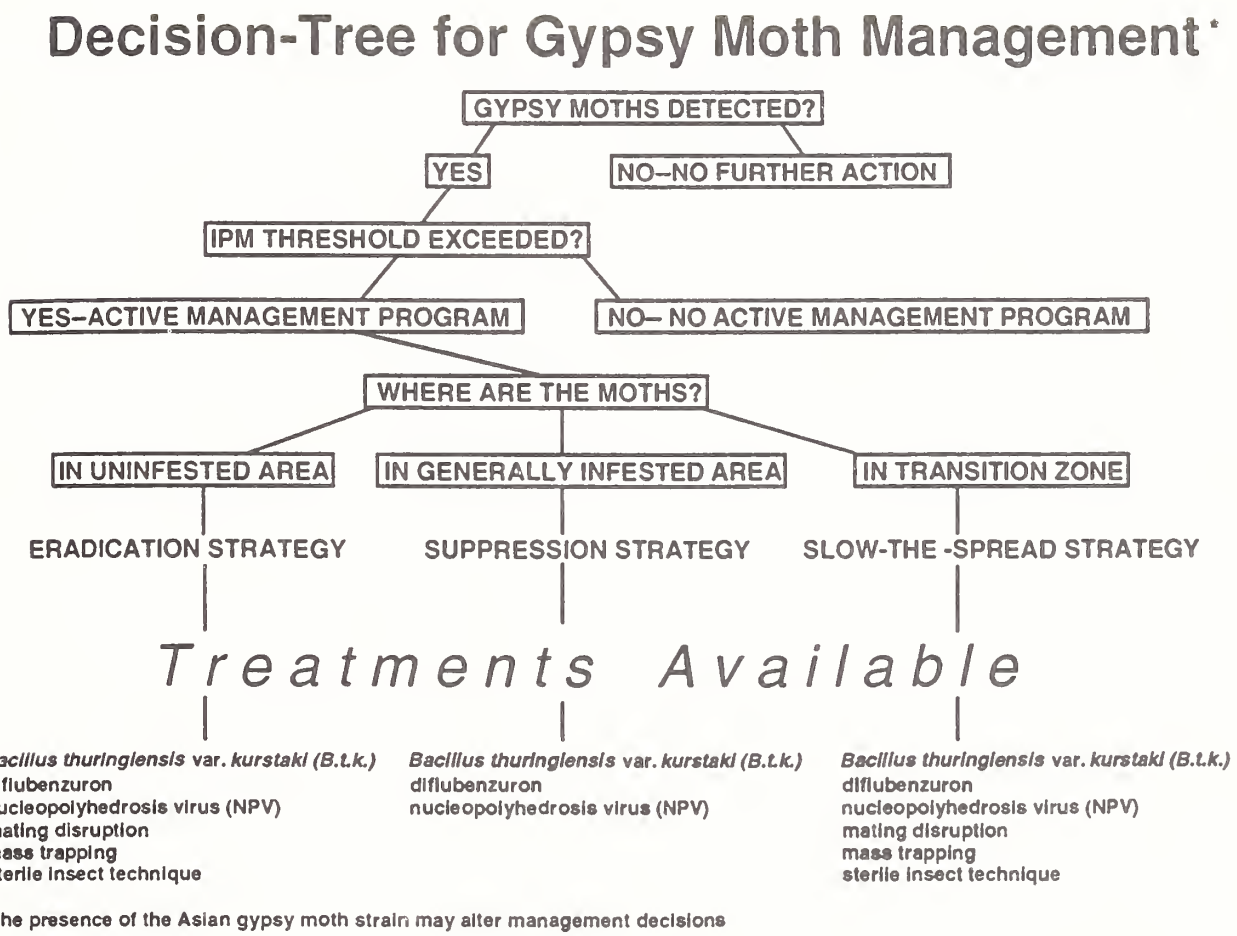

Figure 3-1. Decision tree for gypsy moth management.

Table 3-1. Treatment Rates used in Eradication, Suppression and Slow-the-spread

Tactics → / Strategies ↓	Diflubenzuron*		*B.t.k.*		NPV (Gypchek)		Disparlure		Traps	
	Rate[1]	Number of Times	Rate[2]	Number of Times	Rate[3]	Number of Times	Rate[4]	Number of Times	Mass Trapout (Traps/acre)	Delimiting (Traps/mi²)
Suppression										
Typical	0.50	1	24	1	$2 * 10^{11}$	2	Not Applicable			16
Maximum	1.00	1	40	2						36
Eradication										
Typical	0.50	2	24	2-3	$2 * 10^{11}$	2	30	1	9	16
Maximum	0.50	2	40	3			40			36
Slow-the-Spread										
Typical	0.25	1	24	1	$2 * 10^{11}$	2	30	1	9	16
Maximum	0.50	2	40	2			40			36

*Note - cumulative total diflubenzuron per season is not to exceed 1.00 oz. a.i./acre

[1] oz active ingredient/acre

[2] Billion international units (BIU)/acre

[3] Occlusion bodies (OB)/acre

[4] grams/acre

Section 4
Description of Program Areas

The environment analyzed in this risk assessment encompasses any area in the United States with trees susceptible to defoliation by gypsy moths. Susceptible species occur in tree-dominated plant communities throughout the U.S. Currently, the gypsy moth is largely confined to the northeast quadrant of the country (see maps in Section 2). This generally infested area is expanding westward and southward at a rate of 21 km/year (13 miles/year) (Liebhold and others, 1992). Additionally, isolated infestations have occurred in many areas well beyond the generally infested area and will probably continue to occur with increasing frequency as the generally infested area expands. The geographic scope of this analysis is nationwide to account for the continued expansion of the gypsy moth infestation.

4.1. The Two Ecosystem Approach

For the purposes of this risk assessment, the United States is separated into two ecosystems: the developed forest and the undeveloped forest ecosystems. About two-thirds of the gypsy moth management programs are conducted in the developed forest ecosystem. A review of factors that affect the fate and transport of insecticides (and thus exposure of various components of the ecosystem) used in the gypsy moth management program (see Section 7) suggests that the fate and transport of diflubenzuron and *B.t.k.*, the insecticides most commonly used for gypsy moth control, are influenced more by the density of human development than by the forest type. In other words, the overall difference in fate and transport of gypsy moth insecticides varies less among forest types than between forest types and developed areas, hence the two ecosystem approach. Developed areas are covered to a large extent by impervious (waterproof) surfaces such as concrete and asphalt, potentially leading to rapid runoff of insecticides used in gypsy moth control programs and subsequently to negative effects on aquatic systems in these areas. Runoff is neither as rapid nor as large in undeveloped forested watersheds.

Classifying these areas as ecosystems perhaps stretches the definition. Ecosystems are often thought of as relatively discrete ecological systems defined by the plants and animals in them and their interactions (Lawrence, 1989). In contrast, the two ecosystems in this risk assessment are defined in terms of the amount of impervious surfaces. In a general sense the developed forested ecosystems share many ecological features that cut across the larger, discrete ecological units in which they are found (such as ecoregions as defined by Bailey (1980)). To justify, in an ecological sense,

lumping all the remaining forested land into one ecosystem, the undeveloped forest, is more challenging. Many biotic characteristics of undeveloped forests would lead to their separation into different ecosystems (Bailey, 1980); however, none appears to be as important as the presence or absence of impervious surfaces in the fate and transport of the microbial and chemical insecticides used to control gypsy moths. The relative lack of impervious surfaces unites the various forests found throughout the United States into one ecosystem, the undeveloped forests. General biotic features of both ecosystems are described below.

4.2. The Development Continuum

The number of buildings is perhaps the most relevant measure of urbanization in any area, and thus of the impact humans have had on the immediate environment. Generally, the number of buildings is greatest in the business district of a metropolitan area and decreases with distance from the city center forming a continuum through city residences, suburbs, agricultural areas, finally reaching remote and relatively undisturbed habitats. Species diversity and ecosystem complexity tend to increase along this continuum, but greenways, parks, wildlife reserves, and agricultural land can interrupt or reverse the gradient (Peterson, 1982).

Within this continuum, the developed forest contains single- and multiple- family dwellings, interspersed with occasional parks, cemeteries, schools, churches, and small businesses; developed forests are mostly residential areas. Most gypsy moth management programs in developed forest areas are conducted only where the canopy coverage is at least 50 percent (Schneeberger, 1994). For consistency, the developed forest ecosystem also has at least 50 percent canopy coverage. Newer developments with younger or sparse trees might not meet this criterion and so might not qualify for gypsy moth management in most states, but older communities with trees that form a complete canopy (Detwyler, 1972) would qualify. Compared to the undeveloped areas, subcanopy trees and shrub-layer vegetation are more sparse in the developed forest, and the ground layer is sometimes very dense, for example where there are lawns. The developed forest is similar to the "suburbia" zone of VanDruff (1979) in his three-zone classification of wildlife habitats in developed areas. It is not an area of skyscrapers and parking garages, or the "metropolitan center" (VanDruff, 1979). In contrast, undeveloped forests lack paved roads, and have nearly complete canopy coverage with roughly three layers of subcanopy vegetation (subcanopy trees, shrub layer, and ground layer vegetation) in addition to a litter layer of dead leaves, stems, branches, and tree trunks all in various stages of decomposition.

4.3. Physical Characteristics

4.3.1. Climate

The difference in climate between developed areas and undeveloped areas is in proportion to the amount of "built-up" area that influences the amount of heat-absorbing surfaces in developed areas. These surfaces retain heat and warm the surrounding air throughout the day and far into the night. The "heat island" effect contributes to changes in wind and precipitation and is in proportion to the density of the population as well (Landsberg, 1976). In general, temperatures in the city are 1 to 3_°F (less than 1 to 2_°C) higher, relative humidity is about 6 percent less, average rainfall is greater by about 10 percent, and average wind speed is less by the same amount (Bryson and Ross, 1972).

The developed forest shields the immediate environment from heat island effects. However, weather patterns created over highly developed areas, for example, downtown business districts, are carried over adjacent residential areas (Hengeveld and De Vocht, 1982). These increases in temperature and rainfall in developed areas will affect the fate and transport of gypsy moth treatment relative to the undeveloped forests. While higher temperatures accelerate the degradation of diflubenzuron in aquatic and terrestrial habitats, greater rainfall increases the amount that reaches streams, ponds, or other bodies of water.

4.3.2. Soils

The traditional soil-layer scheme found in undisturbed habitats (greater organic content near the surface and increasingly mineralized soils below) is absent in many developed areas. Leaves and other organic debris from trees and shrubs, which create the organic layer of undisturbed soils, often are raked from the ground surface for composting or trash collection in developed areas. However, cultivated and planted soils sometimes have considerable organic additives. Soils in developed areas often are highly disturbed by mixing, or filling (Gilbert, 1989). They tend to be compacted, reducing aeration and drainage. Impervious surfaces prevent water and air from reaching the soil beneath. These characteristics result in reduced plant vigor, root penetration, and nitrogen fixation by legumes, and fewer micro- and macro-invertebrates which consume and recycle organic material (Gray, 1972).

Undeveloped areas typically have well-defined soil layers. Litter and organic layers overlay layers that become more mineralized and less organic with depth (Waring and Schlesinger, 1985). These soils tend to be less compacted, better aerated, and better drained than those in developed areas.

4.3.3. Hydrology

The high percentage of area covered by impervious materials in developed areas is the most significant ecological factor separating developed and undeveloped forested ecosystems from the standpoint of risk posed by gypsy moth management activities. Impervious surfaces such as roofs, parking lots, streets, sidewalks, and driveways shunt water directly into stormwater systems which feed local streams (Schaake, 1972; Walesh, 1989). Following a storm in developed areas, the flow rate, peak flow levels, and loadings of pollutants and debris in streams and rivers increase substantially compared to streams and rivers in undeveloped areas; these effects are directly related to the amount of impervious surface covering the land in the drainage system (Kim and others, 1978; Rimer and others, 1978; Waring and Schlesinger, 1985; Brown, 1988).

The percentage of an area covered by impervious surfaces in developed areas depends largely on the density of dwellings. More than 70 percent of some high-density developed areas (greater than eight dwelling units per acre) are covered by impervious surfaces (Fisher and Katz, 1984). In less densely inhabited areas as little as 10 percent of the area is covered by impervious surfaces (the "partly urban" category of Veenhius and Slade, 1990). In the developed forest ecosystem waterproof surfaces account for 20 to 50 percent of the ground surface (the "urban" classification of Veenhius and Slade, 1990); in this risk assessment 35 percent was the figure used in the model to estimate runoff in developed forest ecosystems (Section 7). In many developed areas soils are compacted, vegetation is less abundant, and litter is nonexistent, further reducing the ability of developed areas to retain precipitation (Hengeveld and De Vocht, 1982).

Impervious surfaces are essentially non-existent in undeveloped forests. (In modeling runoff in the undeveloped forest ecosystem, 5 percent was used as the impervious area input parameter (Section 7) to account for rock outcrops and some paved surfaces.) In addition, the canopy, understory, shrub layers, and ground cover all reduce the force of raindrops on the litter or bare earth, thereby reducing erosion (Gray, 1972); these elements also retain water during storms that evaporates later (Waring and Schlesinger, 1985). Water percolates through the soil after the litter and vegetation become saturated. Most forest soils have some natural channels that allow water to flow rapidly into streams (Waring and Schlesinger, 1985). Despite these direct inputs to streams in forested watersheds, streams in developed areas undergo larger and more rapid changes in flow in response to storm runoff (Veenhius and Slade, 1990). Overland runoff is rare in forests, but the norm in developed areas (Waring and Schlesinger, 1985; Hengeveld and De Vocht, 1982). One result of this fundamental difference between developed and undeveloped forested ecosystems is that pollutants enter aquatic systems directly and quickly in developed systems compared to undeveloped forests (Kim and others, 1978; Rimer and others, 1978; Waring and Schlesinger, 1985; Brown, 1988).

Diflubenzuron is relatively insoluble in water and adsorbs readily to organic material. It reaches water bodies either by direct application, or bound to soil particles or debris swept by overland flow. Exposed soil, common around developments, erodes easily during rain storms, increasing the runoff of sediments (Gray, 1972) which could increase diflubenzuron concentrations in creeks and ponds in sprayed areas. The force of raindrops directly hitting the earth, more common where the canopy and other vegetation layers are incomplete, increases soil erosion and results in further sedimentation of aquatic systems.

The behavior of diflubenzuron on asphalt has not been reported in the literature; however, diflubenzuron will likely adhere readily to asphalt because of the high organic content of asphalt. But, some runoff of unbound diflubenzuron might be expected during storm events shortly after spraying, in a fashion similar to the initial washoff recorded from plants (see Section 7).

In developed areas *B.t.k.* and NPV will probably be carried readily during storms into streams by overland runoff. In undeveloped forested ecosystems these products will wash from leaf surfaces, fall to the litter, and remain there until they degrade, germinate (in the case of *B.t.k.*), or are consumed.

4.4. Biotic Characteristics

4.4.1. Flora

Vegetation in developed areas is diverse. Older neighborhoods often have remnants of native forest trees lining streets and in yards. These areas often have nearly complete canopy coverage (Detwyler, 1972). Newer developments, sometimes built on former agricultural land, have younger trees with very little complete canopy coverage; the trees are susceptible to defoliation from gypsy moths nevertheless. Understory, shrub-layers, and ground cover either are lacking, or are composed of introduced ornamental species (Schmid, 1975). Lawns are the dominant ground layer in most developed areas, lending a park-like appearance to neighborhoods with mature trees, and a prairie-like appearance to neighborhoods with few tall trees (Schmid, 1975). Ornamental plantings increase the species diversity of the understory plant community in developed areas over conditions in many undeveloped areas (Detwyler, 1972).

The species composition of trees in developed areas is largely dependent on the age of the neighborhood and tree-planting history of the community. Older neighborhoods with mature "forests" are composed of native oaks, maples, sycamores, and other species. Developed forests in the West, especially California, are usually composed

of exotic trees, native to the Eastern United States that are often excellent gypsy moth hosts. Individual trees sometimes predate the neighborhood (Detwyler, 1972).

Hoehne (1981) inventoried the groundlayer vegetation in 31 forest fragments next to residential or agricultural areas near Milwaukee, Wisconsin. In a comparison among the 31 sites she found disturbance was the most influential factor regulating composition of the groundlayer vegetation. Disturbance from recreational use caused decreases in the presence and frequency of native herbaceous species. Diversity increased with disturbance because of increases in exotic and weedy species in a comparison among the sites; the exceptions were the most heavily used areas, such as near foot paths and campsites. Three stands in her study had also been sampled 23 years earlier. A comparison of the old and new surveys suggest that over time, soil compaction, disturbance, and a reduction in stand size has resulted in the loss of 25 to 60 percent of the species of groundlayer plants in these stands, despite increases in exotic and weedy species.

4.4.2. Terrestrial Fauna

Animal diversity is less in developed forest ecosystems than in undeveloped forest ecosystems, resulting in simplified competitive and predator-prey systems. The characteristics of animals that do well in developed settings include those that reproduce rapidly and have flexible behavioral patterns that enable them to exploit diverse food sources (Gill and Bonnet, 1973). Mammals requiring large home ranges, particularly large carnivores, and are sensitive to human disturbance are among the first to disappear from developed areas; for example, bear (*Ursus americanus*), wolf (*Canis lupus*), cougar (*Felis concolor*), and bobcat (*Lynx rufus*). In contrast, smaller omnivores such as opossum, raccoon, and skunk are quite successful in residential developments. Gray squirrels, fox squirrels, or both (*Sciurus carolinensis* and *S. niger*, respectively) can be found in abundance in most metropolitan areas. Fox (primarily red fox, *Vulpes fulva*) can maintain populations in some developed settings, as can coyotes (*Canis latrans*). Deer (*Odocoileus virginianus* or *O. hemionus*) are also found in developments where sufficient green space exists for cover and disturbance from dogs is not too great.

Matthiae and Stearns (1981) found fewer species of mammals in forest patches in urban and agricultural areas around Milwaukee compared to rural areas. The presence of dogs (which act as predators) and isolation from other forests were thought to be primarily responsible for this pattern.

The diversity of birds is lower in developed settings than in undeveloped forested ecosystems (Robinson and Bolen, 1984). Most species are either introduced, year-round residents, or short-distance migrants rather than neotropical migrants, which are

more common in the undeveloped forest ecosystem. Nest predation is higher in developed areas due to large populations of crows (*Corvus* spp.), jays (*Cyanocitta* and *Aphelocoma* spp.), and domestic cats (Whitcomb and others, 1981; Terborgh, 1989). In general, brood parasitism by cowbirds (*Molothrus ater*) is greater in fragmented habitats (Robinson and others, 1993); their effects on susceptible host species nesting in developed areas is likely to be great. Higher nest predation and parasitism in human-disturbed habitats are thought to be factors in the decline of neotropical migrants nation-wide, and are less severe or nonexistent in undisturbed, unfragmented forest tracts (Whitcomb and others, 1981; Terborgh, 1989).

Reptiles and amphibians are much reduced where development has eliminated native vegetation and disturbed breeding and cover sites. Travel barriers and pollution also contribute to the decline of reptiles and amphibians in developed areas (Campbell, 1974). Ponds in parks sometimes are home to a few species of frogs and turtles, but the native diversity of herpetofauna (reptiles and amphibians) is always reduced in these areas compared to undisturbed sites. Introduced species sometimes flourish in modified aquatic habitats and compete with or depredate populations of native species. Some species of snakes and lizards find refuge in hedgerows and undisturbed backyards, especially if close to reservoir populations in larger green spaces (Campbell, 1974). In general, urban and suburban populations of reptiles and amphibians must contend with high rates of predation and harassment from people and their pets (Campbell, 1974). Snakes, even nonvenomous species, are particularly subject to persecution by people (Bird, 1987).

Flying insects that are generalists are less affected by urbanization than other animals because of their capacity for dispersal. Insect diversity can be quite high in backyards because of the diversity of plants (Lutz, 1941; Owen, 1978). However, these are altered insect communities compared to what would be expected in native habitat in these areas. Species having specific requirements, for example, native species of plants to forage or reproduce on, tend to be less common in developed settings. Singer and Gilbert (1978) found some species of butterflies absent from developed areas because of a lack of specific plants required for feeding or laying. Other species were absent for unknown reasons. Taylor and others (1978) in Great Britain found that populations of moths in towns are not isolated; they were supplemented by surrounding undeveloped habitat. Ground dwelling arthropods (invertebrates such as insects) in developed areas are less diverse than their forest counterparts, especially if far removed from reservoir populations from which founders can disperse and repopulate recently developed areas (Gilbert, 1989). Studies on the differences between developed and undeveloped forests in litter- and soil-dwelling invertebrates were not found in the literature. The degree of disturbance to an area and the ability of the organisms to disperse is expected to be key to understanding any differences in species composition in these two ecosystems.

4.4.3. Aquatic Fauna

Increasing development around streams and rivers often has negative consequences for the invertebrates inhabiting them. Many aquatic insects and other invertebrates are sensitive to changes in flow rate, temperature, sedimentation, and pollutants (Hynes, 1963, 1970; Wiederholm, 1984). Measurement of nitrogen and phosphorous content, sediment loads, and coliform bacteria are high in runoff from developed areas ((Kim and others, 1978; Rimer and others, 1978). These factors promote algal growth which in turn reduces the amount of dissolved oxygen (Abel, 1989). Pollution of this sort alters species composition and reduces species diversity of aquatic insects, although at mild levels might increase overall abundance of some species benefitting from an increased food supply (Hynes, 1973; Wiederholm, 1984). Other pollutants such as heavy metals, oil, and pesticides further reduce aquatic invertebrate populations in developed areas. In cases where invertebrate biomass is reduced, negative effects will be noted among predatory fish populations as well (Moyle, 1976).

Ono and others (1983) listed habitat alteration and introduced fishes as the leading threats to threatened and endangered fish in the U.S. These same factors affect native fishes in developed streams; however, introduced fishes pose much less threat to eastern fishes than to native fishes in the west (presumably because the species-rich native eastern fishes are well-adapted for living in complex and shifting assemblages, in contrast to the relatively isolated and species-poor western fishes) (Moyle 1986). Streams in developed areas commonly channelized and receive inputs directly from storm drains which carry a variety of contaminants from gutters. Channelization, pollutants, and sediments alter the chemistry and flow rates of many urban and suburban streams (see Hydrology, above). These changes, in turn, have severe impacts on the ecology of streams (Hynes, 1970; Moyle, 1976; Moyle and Leidy, 1992). Channelized segments of streams flowing into San Francisco Bay, California, were either lacking fish or dominated by introduced fishes and a few hardy native species (Leidy and Fiedler, 1985). In another study in California, channelization reduced the biomass of invertebrate prey and of the fish inhabiting channelized regions compared to unchannelized regions (Moyle, 1976). Ponds are common in parks and sometimes have an introduced species of sunfish (*Lepomis* spp.), goldfish (*Carassius auratus*), bass (*Micropterus salmoides*), or carp (*Cyprinus carpio*).

4.5. Summary

In this risk assessment we divide the United States into two ecosystems, the developed and undeveloped forests, because of differences in the area covered by impervious surfaces. In developed areas 20 to 50 percent of the surface is made of waterproof or nearly waterproof materials (mostly asphalt and concrete). Roads, sidewalks, and roofs increase runoff of sediments and pollutants in local streams. Negative effects

from the use of insecticides in gypsy moth control could potentially be more severe on aquatic invertebrates and vertebrates in these areas than in undeveloped areas, because of increased runoff.

Despite regional variability the biological characteristics of wooded developed areas can be described generally in terms of their contrast to undisturbed forests. Disturbance from human activity affects the ecosystem at every level. Soils are compacted and turned relatively frequently in some areas. Litter and organic layers of soil are often removed or imported to create certain aesthetic effects. These changes alter the soil chemistry and reduce the diversity and abundance of animals in the soil arthropod community, which in turn affects rates of decomposition of organic materials. Natural communities of plants and animals are severely fragmented or completely absent. Native plants are often replaced by non-native species and all but the most resilient native mammals, birds, reptiles, and amphibians are displaced or are found only in adjacent greenways and parks. Aquatic systems in developed areas must contend with increased disturbance, siltation, and pollution, which result in different assemblages of species compared to aquatic flora and fauna in undeveloped areas.

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Section 5
Hazard Analysis

This section summarizes background information on the hazards of insecticides and the pheromone used in the National Gypsy Moth Management Program. Field applications of these agents in the program will result in direct and indirect exposures to some nontarget organisms. This potential exposure of nontarget organisms requires consideration of the toxicity and potential hazard to nontarget species in terrestrial and aquatic habitats from program field applications. Exposure to these agents may result in uptake through the cuticle or skin, gastrointestinal tract, or respiratory system. A broad literature review was conducted to identify pertinent data regarding the toxicity and effects of field applications on nontarget species.

5.1. Toxicity Data

Toxicity data are organized by insecticide or pheromone. Discussions include basic information about the toxic mode of action, the overall toxicity to specific classes of organisms.

5.1.1. *Bacillus thuringiensis* var. *kurstaki*

The pathogenicity of *Bacillus thuringiensis* var. *kurstaki* (*B.t.k.*) derives primarily from the presence of a crystal containing delta endotoxin produced by the bacteria. When crystals are ingested by an organism that has an alkaline gut and a receptor site specific for delta endotoxin, that organism is subject to the toxic effects of *B.t.k.*. This generally limits the susceptible organisms to certain invertebrates such as some Lepidopterans (butterflies and moths). Vertebrates are not susceptible to *B.t.k.* toxicity. The mode of action precludes any concern for dermal and inhalation routes. Toxicity data for *B.t.k.* are presented in Table 5-1. Phytotoxicity (the ability to poison plants) from *B.t.k.* has never been observed at field rates of application.

Among the terrestrial insects affected by exposure to *B.t.k.*, lepidopteran larvae are most likely to be adversely affected. Susceptible caterpillars include the following taxa: noctuid moths, geometer moths, pyralid moths, tortricid moths, tussock moths, notodontid moths, tiger moths, silk moths, tent caterpillars, bagworm moths, sphinx moths, swallowtail butterflies, and pierid butterflies (Faust and Bulla, 1982). *B.t.k.* is relatively nontoxic to honeybees (Atkins and others, 1981). Direct toxicity of *B.t.k.* to terrestrial insect predators and parasites has not been noted in any studies except

some low-level mortality in a laboratory study at doses higher than would occur at the highest recommended rates of application (9.4 L Dipel 4L/ha, or 79 BIU/ha) of *B.t.k.* (Haverty, 1982).

Most aquatic insects are not affected by *B.t.k.*. Immature and adult stages of mayflies, caddisflies, dragonflies, damselflies, beetles, midges, and dobsonflies are all unaffected by *B.t.k.*. Aquatic lepidopteran larvae, however, might be susceptible. Some species of black fly larvae, and two species of stoneflies in the families Leuctridae and Taeniopterygidae were found to be susceptible to *B.t.k.* at field application rates (30 BIU/ha or 2-6 IU/ml) for gypsy moth (Eidt, 1985; Lacey and others, 1978; Kreutzweiser and others, 1993; Kreutzweiser and others, 1992).

5.1.2. Diflubenzuron

The mode of toxic action of diflubenzuron is through interference in the formation of the insect's cuticle or shell. Chitin, a protein that is the building block of the insect exoskeleton, does not deposit properly and the hard exoskeleton does not form properly. A likely mechanism is through blockage of chitin synthetase, the ultimate enzyme in the biosynthesis pathway of chitin (Cohen, 1993). Exposure of invertebrates to diflubenzuron results in larvicidal and sometimes ovicidal effects. The larvae are unable to molt properly due to a lack of chitin in the new cuticle. Exposure of larvae may occur through dermal contact, but the primary route of intoxication is through ingestion. Ovicidal effects may occur through direct contact of eggs or through exposure of gravid females by ingestion or dermal routes. The larva develops fully in the egg, but is either unable to hatch or dies soon after hatching due to chitin deficiency in the cuticle. Diflubenzuron affects insects, other arthropods and some fungi. Chitinous algae (diatoms) are not adversely affected by diflubenzuron (Antia and others, 1985). Most other organisms, including vertebrates, are not affected by exposure to diflubenzuron.

Toxicity data for diflubenzuron are presented in Table 5-2. No information was located about toxicity to reptiles or amphibians, but it is likely that diflubenzuron is of low toxicity to these species based upon the selective nature of the toxic mode of action. Phytotoxicity has not been found to be of any concern to green plants when diflubenzuron is applied at recommended rates of application. Some fungi have exhibited inhibited growth at 50 ppm (lowest level tested), but most have not (Booth, 1978).

Toxicity of diflubenzuron to terrestrial arthropods varies and is very poorly quantified, but most show adverse effects at high exposures. Diflubenzuron often exhibits a greater toxic effect when ingested immediately prior to molting than if it is ingested at other times. Immature grasshoppers, beetle larvae, lepidopteran larvae, and chewing herbivorous and some juice-feeding insects are most susceptible (Elliott

and Iyer, 1982; Jepson and Yemane, 1991; Martinat and others, 1993; Büchi and Jossi, 1979; Mc Whorter and Shapard, 1971; Berry and others, 1993; Sinha and others, 1990; Butler, 1993; Sample and others, 1993c; Redfern and others, 1980; Martinat and others, 1988; Butler, 1995). Honeybees, parasitic wasps, predatory insects, and sucking insects exhibit greater tolerance to diflubenzuron exposure (Atkins and others, 1981; Brown and Respicio, 1981; Bull and Coleman, 1985; Webb and others, 1989; Deakle and Bradley, 1981; Keever and others, 1977; Turnipseed and others, 1974; Martinat and others, 1988). Diflubenzuron is moderately toxic to spiders and mites (Anderson and Elliott, 1982; Marshall, 1979; Everts, 1990).

Toxicity of diflubenzuron to aquatic organisms varies by taxa. Diflubenzuron is generally not toxic to fish, aquatic snails, and most bivalves. No information was located about toxicity to aquatic amphibians, but it is likely that diflubenzuron is of low toxicity to these species based upon the toxic mode of action. It is highly toxic to most aquatic insects, crustaceans, horseshoe crabs, and barnacles. The no-observed effect concentration for phytotoxicity in duckweed is 190 micrograms/liter (Thompson and Swigert, 1993d).

The principal metabolites and degradation products of diflubenzuron are 4-chloroaniline, 4-chlorophenylurea, 2,6-difluorobenzoic acid, 2,6-difluorobenzamide, and 4-chloroformanilide (Eisler, 1992). Both 4-chloroformanilide and 4-chlorophenylurea are further degraded to 4-chloroaniline (Metcalf and others, 1975).

Toxicity data regarding 4-chloroaniline is limited. 4-chloroaniline was shown to inactivate lactoperoxidase in cattle at concentrations well above those anticipated *in vivo* (Bumpus and others, 1993). When diflubenzuron and its metabolites were tested for toxicity to *Euglena gracilis Z.*, only 4-chloroaniline was found to cause inhibition of growth (Gattavecchia and others, 1981), but the researchers indicate that no effects from diflubenzuron exposure would be expected on growth and protein biosynthesis for this organism. The metabolite 4-chloroaniline was shown to be mutagenic (Prasad, 1970) and to have dose-related carcinogenic activity in male rats (NCI, 1979). Rapid metabolism and degradation make it highly unlikely that there would be sufficient exposure to cause any of the adverse toxicological effects noted in these studies.

5.1.3. Nucleopolyhedrosis virus

The active ingredient in gypsy moth nucleopolyhedrosis virus (NPV) is the polyhedra. Polyhedra are crystalline protein bodies that contain the infective virus particles (DNA) of this type of insect virus. This virus occurs naturally throughout the gypsy moth infested areas. NPV is highly pathogenic to early stage larvae of gypsy moth (Doane, 1967; Magnoler, 1970). The virus is often a major factor in bringing about the collapse of high gypsy moth populations (Doane, 1976a; Woods and Elkington,

1987; Murray and others, 1991). Gypsy moth NPV was shown to be non-infectious to other species of insects including the closely related tussock moth, *Orgyia pseudotsugata* (Maramorosch and others, 1976; Anonymous, 1976; Barber and others, 1993). The safety testing of NPV has shown that adverse effects are limited to the gypsy moth and do not affect other organisms. However, adjuvants (such as stickers and ultraviolet light protectants) may be added to NPV formulations. Data concerning the toxic properties of these adjuvants are lacking, thus adding a degree of uncertainty to the toxic effects of NPV formulations. Toxicity data for NPV are presented in Table 5-3.

5.1.4. Dichlorvos

Dichlorvos is an organophosphate insecticide. The toxicity of organophosphate insecticides occurs primarily through the inhibition of acetylcholinesterase (AChE) enzyme activity (Smith, 1987; Klaassen and others, 1986). The AChE enzyme is responsible for the breakdown (hydrolysis) of acetylcholine, a neurotransmitter that permits the transmission of nerve impulses across the nerve synapse. Inhibition of this enzyme results in an accumulation of acetylcholine at the nerve synapse and the continual transmission of nerve impulses. The extent of inhibition of AChE caused by a given dose of pesticide is usually expressed as a percentage of normal activity or a percentage reduction compared to normal activity.

Organophosphates exhibit an irreversible pesticide-enzyme binding reaction (phosphorylation) resulting in AChE inhibition for extended periods of time. This extended binding of AChE allows effects to accumulate, so that a sequence of low doses of an organophosphate can produce the same effect as a single higher dose if the frequency of exposures are fairly close in time.

Dichlorvos is readily absorbed by mammals through oral, dermal, and inhalation routes (Blair and others, 1975; Laws, 1966; U.S. EPA, OPTS, 1987). Dichlorvos is mildly irritating to skin and may cause dermatitis (U.S.EPA, OPP, 1986). Dichlorvos is also a mild eye irritant (U.S. EPA, OPTS, 1987). Carcinogenic effects were observed in laboratory studies on rodents. Dichlorvos induces gene mutations in bacteria with or without metabolic activation (Braun and others, 1982). Chromosomal aberrations and direct DNA damage have been shown in bacterial and mammalian test systems (Gupta and Singh, 1974; Dzwonkowska and Hubner, 1986; Shirasu and others, 1976).

Toxicity data for dichlorvos are presented in Table 5-4. Dichlorvos has been shown to be toxic to mammals, birds, aquatic and terrestrial invertebrates, and fish. It exhibits low phytotoxicity to most plants. No information was located about toxicity of dichlorvos to amphibians or reptiles, although an anecdotal reference to a reptile exposure indicated no apparent effect to a snake resulting from the vapors of a No-

Pest strip (impregnated with dichlorvos) that was placed on top of the cage (Lentz and Hoessle, 1971).

Bioconcentration or bioaccumulation is not expected for any animals because metabolism is rapid (Casida and others, 1962).

5.1.5. Disparlure

Disparlure is a chemical sex attractant that lures male gypsy moths. As shown in Table 5-5, toxicity data indicate that Disparlure is not toxic to mammals, birds, or fish. No information was found on toxicity of Disparlure to insects.

5.2. Terrestrial Field Studies

5.2.1. *Bacillus thuringiensis* var. *kurstaki*

B.t.k. has minimal effects on most terrestrial nontarget species, as the studies below indicate. For most of these groups, the effects of *B.t.k.* are indirect, if present at all. This review highlights the essential information currently available from field studies on *B.t.k.*-toxicity.

5.2.1.1. Lepidoptera

B.t.k. effectively reduces gypsy moths; in one study Dipel 4L (20 BIU/ha) reduced egg mass density by 85 to 95 percent (Andreadis and others, 1982). However, *B.t.k.* treatments designed to eradicate or suppress gypsy moths will also have negative consequences for nontarget moths and butterflies eating sprayed foliage, although the susceptibility to the toxic effects of *B.t.k.* varies widely with species (Peacock and Schweitzer, 1993). Only one study did not find effects from *B.t.k.* on nontarget Lepidoptera (Buckner and others, 1974). In this study the results were not analyzed statistically and sample sizes appeared too small to adequately evaluate the effects of the treatments. All other studies demonstrated some reductions among Lepidoptera (see Table 5-6).

Applications of *B.t.k.* significantly decrease the numbers of adult and larval Lepidoptera the year of spray (Miller, 1990a; Miller 1990b; Rodenhouse and Holmes, 1992; Crawford and others, 1993; Peacock and others, 1994; Sample and others, 1995). Both macrolepidoptera and microlepidoptera are affected (Peacock and others, 1994; Sample and others, 1995; Butler and others, 1995). Reductions of adult populations of Lepidoptera in general, and the larvae of some groups in particular (Noctuidae), are routinely noted the following year as well (Miller 1990a and b;

Peacock and others, 1994; Sample and others, 1995). The delayed effect of *B.t.k.* on adult populations is expected in species whose susceptible life-stage occurs the year previous to the appearance of the adults (Sample and others, 1995). Despite the nearly universal reductions in total lepidopteran biomass suggested by these studies, most investigators have not found reductions in overall species diversity (Miller, 1990a; Crawford and others, 1993; Peacock and others, 1994; Sample and others, 1995). However, some species appear to be particularly susceptible to *B.t.k.*, as evidenced by their total, or nearly total elimination from treated sites (Crawford and others, 1993; Peacock and others, 1994). Such results are expected given the variable susceptibility to *B.t.k.* noted among several species (some in the same genus) of Lepidoptera tested in laboratory experiments (Peacock and Schweitzer, 1993). A number of species Peacock and Schweitzer tested had little or no mortality. Even in controlled laboratory situations for most species some treated larvae produced normal adults. Miller (1990b) noted significant reductions in species richness of Lepidoptera larvae collected in sprayed areas the year of treatment and one year after treatment, but not two years after treatment.

5.2.1.2. Parasites

Field studies on the effects of *B.t.k.* on the parasites of gypsy moths, spruce budworm (*Choristoneura fumiferana*), and Lepidoptera in general (see Sample and others, 1995), have yielded somewhat variable results (Table 5-7). The direct toxicity of *B.t.k.* to parasitic Hymenoptera is low (Flexner and others, 1986). Laboratory tests on several species of wasp parasites of Lepidoptera have shown that only a few were susceptible to *B.t.k.* (Hamed, 1979; Muck and others, 1981). Doses were applied to food in these studies and were reported as spores/ml (for example, 10^7 to 10^9 spores/ml); as such they are difficult to compare to amounts used in the National Gypsy Moth Management Program, which are determined by a standard assay technique that determines potency of a formulation, a property loosely correlated with spore count. In addition to the two laboratory studies showing toxic effects of *B.t.k.* on beneficial wasps, one field study showed that two species of wasp parasitoids of the western spruce budworm (*Choristoneura occidentalis*) "not uncommonly" acquired lethal *B.t.k.* infections from their hosts. Parasite larvae appeared to develop normally and emerged from their hosts, which had been brought into the laboratory after exposure in the field, only to die before pupation (Thompson and others, 1977). The parasitism rate was the same between experimental and control plots. Most field studies, however, show no adverse toxic effects of *B.t.k.* on parasite populations, thus, the effects noted below are assumed to be indirect for this group, as well as for the parasitic flies.

Three field studies found no change in rates of parasitism (that is, the proportion of parasitized larvae per sample) by most species of wasp and fly parasites on gypsy moth or spruce budworm larvae due to the application of *B.t.k.* (Dunbar and others,

1973; Buckner and others, 1974; Abrahamson and others, 1979). A decrease in the rate of parasitism on treated plots was noted for some species of parasites in one study (Ticehurst and others, 1982). At least one parasitic fly (*Brachymeria intermedia*) is rare where gypsy moth populations are low because this species is attracted to the higher light levels in a defoliated forest (Reardon and others, 1979). One parasitic wasp (*Cotesia* (*Apanteles*) *melanoscelus*) increases its rate of parasitism of gypsy moths in plots sprayed with *B.t.k.* (Dunbar and others, 1973; Ticehurst and others, 1982; Andreadis and others, 1983; Webb and others, 1989). Moth larvae poisoned by *B.t.k.* suffer lethal or nonlethal gut paralysis and feed at lower rates than unaffected larvae, thus prolonging the larval period of the host and allowing greater time for the development of the parasite. Ovipositing females preferentially seek smaller or younger larvae, of which there are more on treated plots due to their slower rate of development. Thus, the rate of parasitism increases on treated plots because there are more suitable hosts (Weseloh, 1985). The same mechanism operates to increase the rate of parasitism on *B.t.k.*-poisoned gypsy moth larvae for an introduced parasitic wasp, *Rogas lymantriae* (Wallner and others, 1983).

Spray programs will also reduce the number of potential hosts for parasites, thus a reduction in trap captures of adult parasites in treated areas would be expected, as found by Reardon and others (1979). The same effect might account for the lower numbers of Ichneumonidae (of which many species parasitize lepidopteran larvae) caught in plots treated with *B.t.k.* in another study (Sample and others, 1995).

5.2.1.3. Other Arthropods

Field studies of insects other than Lepidoptera and their parasites and predators have found few other species or groups that are affected. Sample and others (1995) found no effects on sawfly larvae (Hymenoptera: Tenthredinidae and Pergidae), which are similar to caterpillars in appearance and diet. Sawflies were lumped with caterpillars in collections of larvae from foliage in another study (Rodenhouse and Holmes, 1992), so the effect of *B.t.k.* on this group cannot be distinguished from the general decline in caterpillar abundance. *B.t.k.* did not affect the overall abundance of other "clinging" arthropods, which included Coleoptera (beetles), Homoptera (sucking insects such as aphids, leaf-hoppers, cicadas), and Araneida (spiders) (Rodenhouse and Holmes, 1992).

The effect of *B.t.k.* treatments on predatory ground beetles (Carabidae) was studied in woodlands with heavy gypsy moth infestation in southwestern Pennsylvania (Cameron and Reeves, 1990). A weak trend suggested that beetles were more likely to feed on gypsy moth caterpillars in untreated plots than in treated plots, a trend perhaps easily explained by the greater availability of gypsy moth larvae on untreated plots. *B.t.k.*-sprayed plots did not affect the number of individuals per species trapped on treated and control plots (Cameron and Reeves, 1990). In another study, numbers of ground

beetles caught in pit-fall traps declined six days after treatment in treated plots (receiving 2 and 4 gal./ac of Dipel WP and Thuricide 16B), and continued to decline 30 days post-spray (Buckner and others, 1974). However, catches on control plots also were lower on day 30 which suggests that factors such as phenology (the relation of climate to biological phenomena), and factors other than *B.t.k.* treatments were responsible for the reduced trap catches. This same study noted no effects on the foraging behavior, pollen collection, or colony growth, of bees in hives placed in treated areas; and no negative effects were noted on other arthropods collected in pit-fall traps or from foliage (including insects in the orders Hemiptera, Homoptera, and Hymenoptera) (Buckner and others, 1974). Statistics were not used in the analysis of these data, and sample sizes were small. In another study, a predatory beetle (*Bembidion lampros*) suffered some mortality (10 to 15 percent) after exposure to soil sprayed with Dipel (Obadofin and Finlayson, 1977). However, this study also found no differences in the efficiency of this beetle as a predator of cabbage maggot (*Hylemya brassicae*) eggs that were placed on plants sprayed with Dipel (at about 44 BIU/ac) when compared with predation rates on unsprayed plants.

Several studies report no effects of *B.t.k.* spray on other beneficial predatory insects of crop pests (reviewed in Melin and Cozzi, 1990). Among the Hemiptera, or true bugs, predators such as spined stiltbug (*Jalysus spinosus*), Nabidae (damsel bugs, *Nabis* spp.), or Lygaeidae (big-eyed bug, *Geocoris* spp.), Anthocoridae (minute pirate bugs, *Orius* spp.), or Reduviidae (assassin bugs), and Pentatomidae (spined soldier bug, *Podisus maculiventris*). Among beetles, Coleoptera, several important predators in the family Coccinelidae (ladybird beetles) were unaffected by *B.t.k.* applications.

No effect was noted on ground-dwelling spiders in a spruce-fir forest in west-central Maine two weeks after Dipel 4L and Thuricide 16B were applied (at 9.35 l/ha (20 BIU/ha) and 5.8 l/ha (20 BIU/ha), respectively); fluctuations in trap catches before and after spray on treated sites were matched on control plots (Hilburn and Jennings, 1988).

5.2.1.4. Soil Organisms

A study of the effects of *B.t.k.* applications on the soil microflora found that application of *B.t.k.* to soils resulted in a moderate increase in numbers of soil bacteria, actinomycetes, fungi, and nematodes compared to controls (Petras and Casida, 1985). In a review of the effects of Bt on other soil organisms, *B.t.k.* applications were found to reduce populations of a species of predatory mite that is closely related to soil-dwelling species (Addison, 1993).

Some species of earthworms were unaffected by *B.t.k.* (Dipel WP®) applied to ash-maple forest soils at 6000 mg/m^2, 100 times recommended rates (Benz and Altweg,

1975), however, whether or not the earthworms were actually exposed to the *B.t.k.* was not clear in this study (Addison, 1993).

5.2.1.5. Birds

The following field studies examine the effects on insectivorous birds, including neotropical migrants, when food resources are reduced by the application of *B.t.k.* (Table 5-8). These studies show that the effects of reduced food levels on birds resulting from applications of *B.t.k.* are subtle.

In sprayed plots, Rodenhouse and Holmes (1992) found significant reductions in the number of nesting attempts per bird per year, and in the number of caterpillars in the diets of black-throated blue warblers (*Dendroica caerulescens*, a neotropical migrant). Nestling growth, nestling mortality from starvation, and fledgling success (measurable aspects of nesting birds that could be negatively affected by food reduction) did not differ between treated and control plots; the production of young per year (number of young produced per territory per season) was not significantly lower despite fewer nesting attempts by birds on treated plots.

Gaddis (1987) and Gaddis and Corkran (1986) studied the reproductive success and feeding activities of chestnut-backed and black-capped chickadees in control and *B.t.k.*-sprayed sites near Portland, Oregon. In two years of study, caterpillars constituted a significantly smaller proportion of the nestling diet in treated sites compared to controls, but the time between prey deliveries was not different between spray and control sites. In the first year they found no difference between treated and control sites in reproductive success or nestling growth measures. In the second year they found a significantly lower fledgling success at treatment sites. This was due to three nests that were abandoned in one of the treatment blocks near the third spray of the season. The relationship between the application of *B.t.k.* and the nest failures is uncertain. Prior to failure all of the nests contained nestlings that appeared healthy, and all failures occurred within 7 days. One nest in a control site was similarly abandoned the same week.

5.2.1.6. Mammals

As with birds, some bats might be affected indirectly by reductions in food abundance. Sample and others (1993b) found that *B.t.k.* applications to oak/hickory woodlands in West Virginia resulted in reduced abundance and biomass of moths the year following application. Although effects on bats were not measured, some effect is possible. This study also showed that defoliation by gypsy moths also reduced populations of adult native Lepidoptera and could affect bat foraging as well.

Belloq and others (1992) found increased emigration of adult male masked shrews out of treated areas, and diet shifts among females and young on treated areas. Diet of adult males was unchanged on treated plots. The abundance of shrews on treated plots was unaffected as adult males were replaced by juveniles. The masked shrew is a generalist predator, thus the effects of *B.t.k.* applications were not great.

5.2.1.7. *B.t.k.*/NPV Interactions

Two studies show that applications of *B.t.k.* to gypsy moth populations reduces the incidence of infection by the gypsy moth nucleopolyhedrosis virus (NPV) (Woods and others, 1988; Webb and others, 1989), and another study found no such effect (Abrahamson and others, 1979). In one of these studies the mortality from NPV was lower and NPV transmission in egg masses was lower in plots treated with *B.t.k.*; lower mortality from NPV in treated plots sometimes resulted in higher egg mass densities in plots treated with *B.t.k.* (Woods and others, 1988). Of several hypotheses presented that could explain this relationship, perhaps the most likely is related to the density-dependence of NPV transmission (Woods and others, 1988). The virus spreads more rapidly through dense larval populations than in sparse ones (Doane, 1976b), thus its transmission is hindered in larval populations reduced by *B.t.k.*.

5.2.2. Diflubenzuron

This section describes the results of field studies on the effects of aerial applications of diflubenzuron on nontarget organisms in terrestrial ecosystems. Note application rates (see tables 5-9 through 5-14) in nearly all these studies exceed the application rate currently used in the National Gypsy Moth Management Program (0.25-0.5 oz a.i./ac, or 17.5-35 g a.i./ha). Some interpretation is necessary to extrapolate from the results of these studies to the effects expected from current gypsy moth management efforts using diflubenzuron.

5.2.2.1. Lepidoptera

Lepidoptera (butterflies and moths) are the group of nontarget organisms perhaps most severely affected by applications of diflubenzuron. Larval butterflies and moths are herbivorous and succumb to diflubenzuron poisoning after ingesting contaminated vegetation just as do gypsy moth larvae. Recent field studies have documented measurable negative effects from diflubenzuron on populations of nontarget Lepidoptera in treated woodlands (Table 5-9). Effects on larval macrolepidoptera (large species of moths and butterflies) have been noted the year of application (Martinat and others, 1988; Butler, 1993; Butler and Kondo, 1993), with detrimental effects on population levels the next year despite recolonization potential from nearby

areas (Butler, 1993; Sample and others, 1993c). Reduced trap-catches of adults were found in the year of treatment and in the following year (Butler, 1993; Sample and others, 1993c). Recovery of populations of macrolepidopterous larvae had not occurred in the diflubenzuron treated watersheds 27 months post-treatment when the study concluded (Butler, 1995). Microlepidoptera (smaller species of moths) are less affected by diflubenzuron, perhaps because many feed in locations protected from diflubenzuron spray (Martinat and others, 1988; Sample and others, 1993c).

5.2.2.2. Insect Predators and Parasites

The effects of diflubenzuron on beneficial insects, such as the predators and parasites of gypsy moths and other insects, have shown mixed results in field studies (Tables 5-10 and 5-11). Two recent studies examined the effects of diflubenzuron spray on general insect parasites and predators in natural environments; far more studies have examined the impact of diflubenzuron on insects that depredate and parasitize crop pests. The studies on crop pests shed additional light on the impacts that gypsy moth management may have on nontarget insects that are predators and parasites in the undeveloped forest and developed forest ecosystems. A discussion of these studies is followed by a review of studies that examined the effects of diflubenzuron on predators and parasites of gypsy moths.

5.2.2.2.1. Predators and Parasites of Invertebrates Other Than Gypsy Moths

The effects of diflubenzuron on predators and parasites on invertebrates other than gypsy moths are presented in Table 5-10. Populations of parasitic wasps in the families Ichneumonidae and Braconidae were reduced for three weeks following a spray for grasshopper control in Africa; predacious wasps (Larrinae) whose prey were the target organism in this study were also reduced (Everts, 1990). The reductions could be explained by direct toxicity or by a reduction in prey. In this study diflubenzuron was applied in diesel oil. Some oils were found to be toxic to parasitoids of crop pests (discussed below). In contrast, the abundance and diversity of ichneumonids and braconids was unaffected in forests sprayed with diflubenzuron for gypsy moth management (Sample and others, 1993c).

Field studies on the control of cotton pests with diflubenzuron, applied at rates 4 to 8 times higher than rates used in gypsy moth management, found little to no effect on the abundance of major predators of bollworms, which include the big-eyed bug, lacewings, and ladybird beetles (Ables and others, 1977; Keever and others, 1977; Deakle and Bradley, 1982). However, laboratory studies and caged field studies suggest that nymphal lacewings suffer increased mortality, immature big-eyed bugs suffer higher mortality, and ladybird beetles have decreased fecundity after ingesting food treated with diflubenzuron (Ables and others, 1977; Keever and others, 1977).

In studies conducted in outdoor cages, no effects were found on the development of a wasp parasite inside bollworm eggs treated with diflubenzuron (Ables and others, 1977). In another study, diflubenzuron applied at 70 g a.i./ha (1 oz a.i./ac) affected parasitism by this same wasp only when applied in an oil carrier (House and others, 1980), an effect confirmed by others (Ables and others, 1980; Bull and Coleman, 1985). In the National Gypsy Moth Management Program diflubenzuron is applied in a water carrier only, and at rates 1/4 to 1/2 that described in this study.

Applications of diflubenzuron to soybeans display a dose-dependent effect, with no effects occurring at the lower application rate. Predators (nabids and geocorids) were significantly fewer on treated versus control fields in one study in which application rates of up to 562 g a.i./ha were used (Turnipseed and others, 1974), but were unaffected in the other study which used a lower application rate (250 g a.i./ha; Heinrichs and others, 1979). Application rates in these studies were from 7 to 32 times higher than currently used in gypsy moth management programs.

In Europe, application of diflubenzuron (rate not given) to apple orchards to control leafrollers and other caterpillar pests resulted in an increase in the wooly apple aphid. Although its wasp parasitoid was unaffected by the spray, one predator, the European earwig (*Forficula auricularia*), was missing on treated versus control fields. Earwigs are voracious predators of aphids and were severely affected by the spray (Ravensberg, 1981), an effect confirmed in the laboratory (Sauphanor and others, 1993).

Another example of an increase of a pest was noted when diflubenzuron was applied (at more than 10, 20, and 40 times gypsy moth management rates) to control codling moth in a pear orchard. Another pest, the pear psylla, increased at higher rates with diflubenzuron applications, and predators and parasites of the pear psylla were twice as numerous on the orchard with the lower rate of application (140 versus 280 and 560 g a.i./ha versus 4 and 8 oz a.i./ha) (Westigard, 1979). A study of pest and predatory species of mites in a pear orchard found no effect on the predators from diflubenzuron spray (sprayed until "runoff"), which controlled the pest species of mites equally well in treated and control parts of the orchard (Riedl and Hoying, 1980). In a citrus orchard, diflubenzuron sprayed (8 times at 350 g a.i./ha or 5 oz a.i./ac, at least 10 to 20 times current gypsy moth rates) to control the sugar cane rootstalk borer weevil (*Diaprepes abbreviatus*) resulted in significantly more Texas citrus mites on treated than control trees, while other species of mite and predacious fungi did not change in abundance (Schroeder and others, 1980). Parasitism of the oriental fruit moth by two wasp species was unaffected on trees in a peach orchard sprayed with diflubenzuron (at a very high rate of 1.1 kg a.i./ha; 15 oz a.i./ac) when compared to parasitism on control trees (Broadbent and Pree, 1984b).

When applied to beans to control Mexican bean beetle (applied at 1 to 8 times gypsy moth rates), diflubenzuron resulted in slightly higher success of a wasp parasite on

control fields compared to treated fields. The difference was not significant, but the field test suggested that this parasite has difficulty completing its development in treated bean beetle larvae, a result confirmed in the laboratory (Zungoli and others, 1983).

5.2.2.2.2. Predators and Parasites of Gypsy Moths

The effects of diflubenzuron on predators and parasites of gypsy moths are presented in Table 5-11. The effects of diflubenzuron on the wasp parasite of gypsy moth larvae (*Cotesia* (*Apanteles*) *melanoscelus*) have been studied in apple orchards and hardwoods. Several studies generally agree that this parasite is sensitive to diflubenzuron when applied to early gypsy moth instars, either as a result of direct toxicity of the insecticide to the parasite (Granett and Dunbar, 1975; Granett and others, 1976; Madrid and Stewart, 1981), or as a result of toxicity to the gypsy moth larva which succumbs before successful development of the wasp larva (Webb and others, 1989). Older stages of the wasp larvae are reported to be more hardy to the effects of diflubenzuron than younger stages (Granett and others, 1976). The toxic effects of diflubenzuron on developing wasp larvae are halted development inside treated gypsy moth larvae, or failure to spin proper cocoons upon emergence (Madrid and Stewart, 1981; Granett and Weseloh, 1977). Most field studies show no effect of diflubenzuron on adult wasps. one laboratory study found that female *Brachymeria intermedia* receiving topical treatments (2 and 4 μg technical grade diflubenzuron per adult) laid multiple eggs in a gypsy moth larvae rather than the usual 1 per larvae. Some developmental problems occurred with the generation produced in those larvae, as well; however, survivors produced viable young themselves (Khoo and others, 1985).

Eggs of gypsy moths were parasitized as heavily on diflubenzuron-treated sites as on control sites by the egg parasite (*Ooencyrtus kuvanae*), at application rates as high as 67 g a.i./ha (0.9 oz a.i./ac (2 to 4 times the rate used in current gypsy moth management programs). Laboratory studies confirm the apparent insensitivity of this parasite when developing inside eggs treated with diflubenzuron (Brown and Respicio, 1981).

Diflubenzuron (applied once at 30 g a.i./ha; 0.4 oz a.i./ac) was lethal to 100 percent of the larvae of a parasitic fly (Family Tachinidae) in a program in southern Quebec (Madrid and Stewart, 1981). The fly larvae may have been indirectly affected, as their hosts, the gypsy moth caterpillars, were killed before the fly larvae could emerge. A parasitic wasp (*Apanteles melanoscelus*) was also affected in this study; 66 percent of wasp cocoons failed to develop properly in the field, and about 80 percent failed to form cocoons or emerge from them in the lab.

5.2.2.2.3. Conclusions

Studies of the effects of diflubenzuron on parasites of crop pests suggest some parasitic wasps are quite tolerant of diflubenzuron-treated hosts, while studies on gypsy moth parasites suggest that wasps exposed in early stages of development are sensitive to diflubenzuron applications. In addition, an egg parasite of gypsy moths is quite resistant while a parasitic fly is very sensitive. Some parasitic organisms are more sensitive to the effects of diflubenzuron when the host species is in earlier, more susceptible, instars. Taken together, these studies suggest that diflubenzuron can be expected to have varying effects on larval or egg parasites of nontarget insects, depending upon the age of the host and the species of parasite.

Predators in immature stages (such as lacewings) eating contaminated prey suffered higher mortality; adult predators suffered reduced fecundity after eating contaminated prey (for example, ladybird beetles). Several studies showed no effects on populations of predators of crop pests in the field following application of diflubenzuron. In some cases immigration from untreated neighboring fields might have confounded trapping results. The controlling effect of predators on (normally inconsequential) pests was discovered in two studies after diflubenzuron reduced the predator but not the pest population.

5.2.2.3. Honey Bees

Negative effects of diflubenzuron on honey bees (*Apis mellifera*) are noted at high application levels and relatively long periods of exposure (Table 5-12). When fed for six or more weeks on supplementary food treated with 10 and 60 ppm diflubenzuron, honey bees produced less sealed brood (pupating larvae) and less wax comb than control hives and hives receiving lower doses (Barker and Waller, 1978; Stoner and Wilson, 1982). After 10 weeks, the number of individuals per colony was lower in hives receiving the 10 ppm the diet (Stoner and Wilson, 1982). Opportunities for feeding outside of the artificial food supply were limited in at least one of these studies, thus guaranteeing that the bees were exposed to diflubenzuron (Stoner and Wilson, 1982). Exposure was also guaranteed in another field study in which hives were caged with sprayed trees. In this study no effects on adult mortality or the production of larvae and sealed brood was evident after exposure to 110 to 400 g a.i./ha (1.5 to 5.7 oz a.i./ac) sprayed on the tree, rates that are 3 to 20 times higher than the rates used in the National Gypsy Moth Management Program (Emmett and Archer, 1980). In other studies, hives were located in fields or forests that were sprayed with diflubenzuron and no effects were found on adult survival, numbers of larvae, wax, comb, or honey production (Buckner and others, 1975; Matthenius, 1975; Robinson and Johansen, 1978; Robinson, 1979). A review of 20 field studies found one study in which brood was reduced for the first week after application of diflubenzuron at 20 g a.i./ha (in the range of rates used in the National Gypsy Moth

Management Program, 15 to 35 g a.i./ha, or 0.25 to 0.5 oz a.i./ac); this effect disappeared after the first week (Kuijpers, 1989).

A review of the fate and transport information shows that diflubenzuron is not absorbed or translocated in plants (see Section 7). Thus, nectar is not expected to contain diflubenzuron, and the results of the feeding studies is not particularly relevant to this program. In contrast, pollen exposed to direct spray could contain some residues, and honey bees could collect and use this pollen in the hive. However, studies of caged honey bees and free-foraging bees in sprayed areas found no effects on hives located in sprayed areas. In addition, the application rates used in these field studies were all greater than those used in the National Gypsy Moth Management Program.

5.2.2.4. Other Nontarget Invertebrates

The field studies on the effects of diflubenzuron show that immature stages of leaf-eating organisms are most susceptible (Table 5-13). Negative effects on nymphal stages of grasshoppers and other orthopterans were noted in several studies (Everts, 1990; FAO, 1992; Butler, 1993; Jech and others, 1993; Martinat and others, 1993). Negative effects on populations of other herbivorous insects such as sawflies, some beetles, and wood lice were noted (Martinat and others, 1988; Butler, 1993; Butler, 1995). Homopterans, hemipterans, and predacious arthropods were unaffected in one study (Martinat and others, 1988), but some homopterans were reduced in another (Butler, 1993). Many homopterans and hemipterans suck plant fluids, inserting mouthparts through the cuticle of the plant. Thus, they avoid ingesting residues of diflubenzuron on the surfaces of leaves. The abundance of insects representing several different families of beetles, wasps, ants, and flies were unaffected in another study (Sample and others, 1993c). However, many studies of nontarget invertebrates fail to determine whether living larvae observed in the field will remain viable until they emerge as adults and may underestimate actually mortality.

Spiders and mites were adversely affected by diflubenzuron spray in several studies (Marshall, 1979; Blumberg, 1986; Everts, 1990; Martinat and others, 1993; Perry and others, 1993). A four-week delay between spray application and decreased abundance was noted in two studies on spiders and could be explained by direct toxic effects or indirect effects through loss of prey (Everts, 1990; Martinat and others, 1993). In neither of these studies was species diversity affected, and Everts (1990) found that only 1 of 3 spider families was negatively affected. Studies of soil mites suggest that some species are susceptible while others are not (Marshall, 1979; Blumberg, 1986; Perry and others, 1993). In one study half of the taxa investigated showed significant decreases in abundance, but the overall number of soil mites was unaffected, as decreases in some groups were offset by increases in others. However, some species were apparently eliminated from a site receiving very high doses of

diflubenzuron (greater than 20 times rates proposed to be used in the National Gypsy Moth Management Program) and mites closer to the surface were more severely affected by diflubenzuron applications to the canopy than those below 3 cm (Marshall, 1979). Significant decreases in densities of soil mites and spiders, and in the abundance of 6 out of 24 common species, compared with pretreatment and control areas, were found in another study on the effects of diflubenzuron applications to soil arthropods (Perry and others, 1993); populations of reduced taxa rebounded the following year. The effects of the application appeared to be indirect for at least some of the taxa, perhaps acting through predator-prey relationships among them.

Several simultaneous investigations were conducted in West Virginia to evaluate the impacts of diflubenzuron applied to manage gypsy moths in a forest ecosystem in the Appalachian mountains. Terrestrial organisms sampled in these recent (mostly unpublished) studies included pollinating insects, canopy arthropods, soil microflora, soil arthropods, and salamanders (Reardon, 1993). Three years of pretreatment data (1989 to 1991) were compared with data from the same site the year of treatment, the following year, and from untreated sites. The results of some of these studies were cited [Butler, 1993 (canopy arthropods); Perry and others, 1993 (soil arthropods)]; the salamander study is discussed below under the subsection on vertebrates. The overall effects on pollinators was relatively slight; among 100 species of bees, 50 species of flower flies, 5 species of bee flies, beetles, yellowjackets, and the European hornet, significant decreases were found in the abundance of flower flies, yellowjackets, and hornets, on treated plots. The principal effect on canopy arthropods was on foliage eating groups, with major reductions noted in Lepidoptera (as expected). Cellular slime molds were unaffected the year of treatment in this study. Other organisms studied in this research effort were aquatic fungi, and aquatic macroinvertebrates.

5.2.2.5. Birds

Diflubenzuron is of little direct consequence to birds because of its low toxicity to vertebrates. The effects of this insecticide on insectivorous birds, including neotropical migrants, will be indirect, when food resources are reduced by applications of diflubenzuron. Indirect effects, such as enlarged territories on treated plots versus untreated plots, and reduced fat levels of birds collected on treated plots, have been correlated with significant reductions in populations of Lepidoptera which resulted from applications of diflubenzuron (Whitmore and others, 1993; Cooper and others, 1990; Sample and others, 1990) (Table 5-14). Whitmore and others (1993) found fat reductions in the neotropical migrants only. Fat levels of two resident species were not measurably affected. Sample and others (1993d) and de Reede (1982) found significant dietary shifts in some species as a result of treatments. In contrast, other studies in which more pesticide was applied (Buckner and others, 1975; Bart, 1975; Stribling and Smith, 1987; and Richmond and others, 1979)

reported no indirect effects on birds, as determined by measures of abundance and nesting success.

By examining territory size, feeding rates of adults, and fat levels of adults before and after treatment and in treated and control plots, the studies by Cooper and others (1990), Sample and others (1990), and Whitmore and others (1993) used very sensitive measures of response to chemical treatments. The results suggest that the adults must exert extra effort to feed nestlings in territories treated with diflubenzuron and replace caterpillars with foods of lower value. The negative effects from the extra energy expenditure may not affect immediate productivity (as in de Reede, 1982, and Richmond and others, 1979), but could have negative long-term effects, such as decreased future reproductive success, and reduced lifespan. Such responses suggest that breeding productivity could be compromised in certain species of birds in some forest ecosystems. The results of Whitmore and others (1993) indicate that at least some resident species (especially members of the chickadee and titmice family) are less affected by treatments than migrants. Possibly resident species are less affected because they often forage for prey less likely to be affected by treatments, such as leafrollers, and they may be more familiar with their territories. De Reede (1982) also studied resident species and found no negative effects on nesting success for these species.

5.2.2.6. Mammals

The results of two small, unreplicated field studies on small mammals suggest that diflubenzuron applications of 60 to 280 g a.i./ha (0.85 to 4 oz a.i./ac) have no deleterious effects on the abundance or reproductive activities of voles, field mice, and shrews (O'Connor and Moore, 1975; Henderson and others 1977). In one of these studies, small mammals increased in abundance on a plot receiving a high rate of application (280 g a.i./ha) compared with a control plot (Henderson and others 1977). The negative effects diflubenzuron might have on bot flies, a parasite of small mammals, was suggested as a possible explanation.

The effects on Lepidoptera in general could have adverse indirect effects on bats that eat moths. In a study of the effects of aerial applications of diflubenzuron on moths in an Appalachian forest, abundance and biomass of moths consumed by the endangered Virginia big-eared bat (*Plecotus townsendii virginianus*) were reduced (Sample, 1991).

5.2.2.7. Amphibians

The effect of diflubenzuron on amphibians in the field is represented by a single unpublished study conducted in the Appalachian mountains in West Virginia in

concert with several other investigations on other aspects of the ecosystem (Reardon, 1993). Diet, fat levels, and reproductive effects (egg volume and egg number) were noted in this study of one aquatic (*Desmognathus monticola*) and two terrestrial salamanders (*D. ochrophaeus* and *Plethodon cinereus*). Although the analysis is preliminary, there appears to be a dietary shift to include more ants and fewer winged hymenopterans for the aquatic species on treated sites compared to the same sites before treatment (Pauley, 1994, personal communication). No corresponding increase occurred in the proportion of ants in stomachs of this species collected on control sites during the post-treatment years, indicating a possible treatment effect, however there was a reduction in the proportion of winged Hymenoptera in their diet on "post-treatment" control sites. Egg volume was significantly reduced on treated and control sites during the post-treatment years for one of the terrestrial species (*D. ochrophaeus*), suggesting one or more environmental factors other than diflubenzuron were responsible.

5.2.3. Gypsy Moth Nucleopolyhedrosis Virus

The gypsy moth nucleopolyhedrosis virus (NPV, product name GypChek) is a naturally-occurring virus found wherever gypsy moths are established. NPVs belong to a group of viruses that have high host-specificity (Gröner, 1990). Few field studies were found demonstrating the nontarget effects of the gypsy moth NPV. However, in the laboratory, Barber and others (1993) demonstrated the host-specificity of the gypsy moth NPV in a test of 46 larval Lepidoptera native to North America (including 3 species in the same family as gypsy moths, Lymantriidae), one fly (Family Tachinidae), and a bee (Family Megachilidae). None of the species other than gypsy moths were infected by the virus. Gypsy moths suffered 92 percent infection. In addition, a beetle in the genus *Tenebrio* and some species of grasshoppers (Order Orthoptera) also were not affected by the virus (John Podgwaite, personal communication).

The number of parasitic wasps (*Cotesia melanoscelus*) found in plots sprayed with GypChek was significantly lower than in control plots (Webb and others, 1989). This parasitic wasp apparently avoids gypsy moth larvae infected with the virus (Versoi and Yendol, 1982). Larvae often die of the virus infection before the parasite completes its development, suggesting a good reason for wasps to discriminate between NPV-infected and uninfected hosts. Mammals and birds pass viable NPV through their gut unaffected (Lautenschlager and Podgwaite, 1979; Lautenschlager and others, 1977; Podgwaite and Galipeau, 1978).

5.2.4. Dichlorvos

No field studies on the effects of dichlorvos in gypsy moth traps on nontarget organisms were found. A general review of field studies on terrestrial organisms exposed to dichlorvos will not be attempted in this risk assessment because of the way the chemical is proposed to be used in the National Gypsy Moth Management Program. Dichlorvos is the insecticide contained in a Vaportape strip that is used inside gypsy moth traps. Very few organisms other than gypsy moths are exposed to this chemical.

5.2.5. Disparlure

Disparlure is a chemical which mimics a natural pheromone produced by female gypsy moths to attract males. It is used in the program to disrupt mating by leading males away from females or into gypsy moth traps. It is specific to gypsy moths and has few toxic effects on other organisms. One field study examined the effect of Disparlure applications on the degree of parasitism by the wasp *Ooencyrtus kuwanai*, which parasitizes gypsy moth eggs (Brown and Cameron, 1979). No effects of Disparlure applications were noted on parasitism rates in this study, where the lure also demonstrated no measurable success in disrupting mating among the moths.

5.3. Aquatic Field Studies

5.3.1. *Bacillus thuringiensis* var. *kurstaki*

5.3.1.1. Effects On Aquatic Plants

In order for *B.t.k.* to have toxic effects, *B.t.k.* must be ingested by an organism and exposed to appropriate digestive enzymes at a pH of 9.0 to 10.5 (Falcon, 1971). Therefore, aquatic plants are unaffected by *B.t.k.* because plants have no mechanism for ingesting the bacteria.

5.3.1.2. Effects on Aquatic Invertebrates

Otvos and Vanderveen (1993) have summarized information, including field studies (e.g., Eidt, 1985 and Kreutzweiser and others, 1992), on the effects of *B.t.k.* on aquatic invertebrates. That summary concluded that *B.t.k.* does not adversely affect the abundance and composition of benthic insects. Surgeoner and Farkas (1990) also reported that *B.t.k.* has no appreciable affect on aquatic invertebrates.

Although the effects on the overall invertebrate community may not be appreciable, certain groups such as black flies (Eidt, 1985) and stoneflies (Kreutzweiser and others, 1992) are susceptible to *B.t.k.* The effect of *B.t.k.* on aquatic lepidopterans has not been reported in field studies.

Invertebrates in marine and estuarine ecosystems also reportedly are not effected by *B.t.k.* Several studies are cited by Surgeoner and Farkas (1990) who report no effect of *B.t.k.* to oysters, mussels, shrimp, and periwinkles.

5.3.1.2. Effects on Aquatic Vertebrates

The lack of any documented fish kills, despite the use of *B.t.k.* in Canadian forestry and agricultural control programs for nearly 20 years, has been advanced as an argument that *B.t.k.* does not kill fish (Surgeoner and Farkas, 1990; Otvos and Vanderveen, 1993). Field studies by Buckner and others (1974) concluded that *B.t.k.*-contaminated water has no observable effects on resident fish behavior and reproduction (Table 5-15).

In addition to concerns regarding direct toxicity to fish, concerns have been expressed that fish will consume the cadavers of *B.t.k.*-infected insects that fall into aquatic systems (Forsberg and others, 1976). However, no evidence exists that consumption of *B.t.k.*-treated insects has adversely affected fish to any noticeable degree (Surgeoner and Farkas, 1990).

5.3.2. Diflubenzuron

5.3.2.1. Effects on Freshwater Lentic Systems

The biota found in freshwater lentic systems (for example, ponds, marshes, and lakes) vary considerably in terms of sensitivity to diflubenzuron (Fisher and Hall, 1992)(Table 5-16).

Freshwater aquatic plants such as those found in lentic habitats are generally unaffected by diflubenzuron (Eisler, 1992). Field studies have not been conducted to establish the effect of diflubenzuron on all types of primary producers in lentic habitats. However, laboratory toxicity studies on phytoplankton (floating plant life) have reported the following five-day EC_{50} values and no observed adverse effect concentration values respectively: greater than 300 and 300 μg/L (ppb) for

Selanastrum capricornutum; greater than 380 and 380 µg/L (ppb) for *Navicula pelliculosa*; and greater than 330 and 330 µg/L (ppb) for *Anabaena flos-aquae* (Thompson and Swigert, 1993a,b,c).

In littoral (nearshore) zones where periphytic algae and rooted aquatic macrophytes are the main primary producers, concentrations of diflubenzuron rapidly dissipate and do not accumulate (Booth and Ferrell, 1977). No effects were reported on a blue-green alga (*Plectonema*) growth rate at diflubenzuron concentrations of 100 µg/L (ppb) (Booth and Ferrell, 1977). The laboratory five-day EC-50 and no observed adverse effect concentration values for duckweed (*Lemna gibba*) exposed to diflubenzuron for 14 days were greater than 270 µg/L (ppb) and 270 µg/L (ppb), respectively (Thompson and Swigert, 1993d). In emergent macrophytes diflubenzuron degrades rapidly, within 7 to 10 days after application (Sundaram and others, 1991).

The invertebrate fauna of freshwater lentic habitats, especially crustaceans and insects, are subject to population reductions associated with diflubenzuron use (Lahr, 1990; Eisler, 1992). Reduced levels of crustacean zooplankton assemblages were also reported at diflubenzuron application rates ranging from approximately 30 to 280 g a.i./ha (0.4-4.0 oz a.i./ac) (Apperson and others, 1978; Ali and Mulla, 1978a and b; Lahr, 1990; and Sundaram and others, 1991). Additionally, in lentic habitats, populations of benthic (bottom dwelling) invertebrates are reported to be especially susceptible to diflubenzuron. These invertebrates include crustaceans such as amphipods (scuds) (Ali and Mulla, 1978a and b; Sundaram and others, 1991; Fisher and Hall, 1992), and insects such as dipterans (true flies) (Apperson and others, 1978; Sundaram and others, 1991), chironomids (midges) (Hansen and Garton, 1982a), mayflies (Sundaram and others, 1991), odonates (dragonflies and damsel flies) (Sundaram and others, 1991), and corixids (water boatmen) (Sundaram and others, 1991). Populations of other invertebrates such as gastropods (snails) and ostracods (seed shrimp) are not susceptible to diflubenzuron (Ali and Mulla, 1978a) or are affected only in the short term (Lahr, 1990) and quickly recover to pre-treatment levels.

Vertebrates in freshwater lentic habitats are highly resistant to toxic effects of diflubenzuron at concentrations less than or equal to 45 µg/L (ppb) (Eisler, 1992). For example, Colwell and Schaefer (1980) reported diflubenzuron residues in black crappies (*Pomoxis nigromaculatus*) and brown bullheads (*Ictalurus nebulosus*) were not detectable seven days after diflubenzuron treatments when residues in ponds had a mean concentration of 13.2 µg/L (ppb). No fish mortalities were reported in reservoirs intentionally treated with diflubenzuron at a rate of 40 g a.i./ha (0.6 oz a.i./ac) (Bannister, 1990).

Diflubenzuron can indirectly affect fish populations by reducing the invertebrate food base. Bluegills (*Lepomis macrochirus*) reportedly switched from feeding on

zooplankton to feeding more on midges and terrestrial insects presumably in response to zooplankton decreases after diflubenzuron treatments (Apperson and others, 1978). The diets of black crappies and brown bullheads were altered after diflubenzuron was applied to ponds resulting in a mean concentration of 13.2 μg/L (ppb) and causing a reduction in prey items (zooplankton) in the upper waters (Colwell and Schaefer, 1980). However, despite the alterations in prey selection, stomach fullness indices and condition factors of the fish in the treated ponds were not significantly reduced (Colwell and Schaefer, 1980).

5.3.2.2. Effects on Freshwater Lotic Systems

As was the case with the biota in lentic systems, the biota in freshwater lotic systems (for example, streams and rivers) vary considerably in terms of susceptibility to diflubenzuron (Table 5-16).

Few field studies have specifically addressed diflubenzuron effects on lotic vegetation. Periphytic algae, the main primary producers in many streams and smaller lotic systems, occur mostly on the upper, exposed surfaces of rocks and stones in running water and would likely be exposed to a toxicant. However, algae in general have were reported to be unaffected by exposure to diflubenzuron in laboratory studies (Booth and Ferrell, 1977, Hansen and Garton, 1982a). In artificial streams, algae experienced rapid and persistent increases in biomass and delayed reductions in diversity when exposed to diflubenzuron concentrations greater than 10 μg/L (ppb) for 5 months (Hansen and Garton, 1982a). Aquatic macrophytes generally degrade diflubenzuron rapidly (Sundaram and others, 1991).

Although some studies have reported no adverse effect on lotic invertebrate communities after diflubenzuron applications (Bocsor and Moore, 1975; Blumberg, 1986; Jones and Kochenderfer, 1987), many other studies have reported that certain invertebrates in lotic habitats experience reductions in population size after exposure to diflubenzuron. For example, application rates of 1,250 μg/L (ppb) directly applied to a river for pest control reduced dipterans (Satake and Yasuno, 1987).

Laboratory studies indicate that other invertebrate groups in streams and rivers would be adversely affected by diflubenzuron. These include midges (Ali and Lord, 1980; Julin and Sanders, 1978; Johnson and Finley, 1980), stoneflies (Mayer and Ellersieck, 1986), and mayflies (Harrahy and others, 1994). Insects, especially mayflies, stoneflies, and dipterans in laboratory stream communities were the segment of the biota most affected by diflubenzuron (Hansen and Garton, 1982). The effects reported by Hansen and Garton (1982) showed an apparent dose-dependent response with the greatest insect reductions at diflubenzuron concentrations of 10 and 50 μg/L (ppb) in the water.

Studies indicate that diflubenzuron exposure has little effect on fish in lotic ecosystems. No adverse effects were reported on dace (genus *Rhinichthys*) and minnow fry and adults (family Cyprinidae) in a river system after exposure to diflubenzuron levels of 1250 μg/L (ppb) (Satake and Yasuno, 1987). In laboratory studies trout and salmon were unaffected by exposure to diflubenzuron (Johnson and Finley, 1980; McKague and Pridmore, 1978).

5.3.2.3. Effects on Estuarine Systems

Little information has been published from field studies to assess the effects of diflubenzuron on aquatic plants in estuaries. However, laboratory studies indicate that chitin-producing and nonchitinaceous marine diatoms are unaffected by diflubenzuron (Anita and others, 1985). A five-day EC_{50} value of greater than 270 μg/L (ppb) and a no observed adverse effect concentration value of 270 μg/L (ppb) for the diatom *Skeletonema costatum* have been reported (Thompson and Swigert, 1993e) (Table 5-16).

Invertebrates in estuarine systems, especially crustaceans, are the most sensitive nontarget organisms to diflubenzuron (Eisler, 1992). Repeated applications of diflubenzuron (28 mg a.i./ha) over 18 months resulted in reduced populations of amphipods, dragonfly naiads, corixid nymphs and adult beetles (Farlow and others, 1978). Diflubenzuron applied at a rate of 45 g a.i./ha (0.6 oz a.i./ac) resulted in diflubenzuron concentrations of 0.69 to 3.6 μg/L (ppb) and caused a 46.5 percent mortality to juvenile blue crabs in a tidal pool (Hester and others, 1986). Numerous laboratory studies have also substantiated the effect of diflubenzuron on marine crustaceans. Summaries of these studies (Eisler, 1992; Fisher and Hall, 1992) indicate that organisms that molt, such as crustacean grass shrimp and crabs (Cunningham and Myers, 1987), are especially susceptible to diflubenzuron because of interference with chitin synthesis (Walker and Horst, 1992).

Fish abundance actually was reported to increase in a coastal marsh after diflubenzuron treatments, probably as a result of an abundance of prey (Farlow and others, 1978). In addition, laboratory studies indicate that saltwater fish are resistant to the effects of diflubenzuron (Fisher and Hall, 1992).

5.3.2.4. Summary of Diflubenzuron Effects

The effects on aquatic organisms summarized as follows (Eisler, 1992):

i. Rates as low as 2.5 to 16 μg/L (ppb) are highly effective against pestiferous dipterans (true flies), including many species of chaoborids (phantom midges), chironomids (midges), and culicids (mosquitoes). Presumably non-pestiferous

members of those groups would likewise be affected. These groups are primarily detritivores.

ii. Dosages of 2.5 to 16 μg/L (ppb) also suppress nontarget populations of cladocerans (water fleas), copepods, mayfly nymphs, corixids (water boatmen), and springtails.

iii. Moderately resistant to diflubenzuron are larvae of diving beetles, dragonfly adults and naiads, some ostracods (seed shrimp), backswimmers and water boatmen; organisms that are highly resistant include mosquitofish, frogs, toads, snails, and algae. These organisms are primarily predators.

iv. Most populations of species with short generation times affected by diflubenzuron begin to recover within days or weeks, and recovery is usually complete within 80 days after the last treatment. Recovery for species with longer generation periods would require more time unless recovery is associated with recolonization from upstream drift.

5.3.3. NPV, Dichlorvos, and Disparlure

No published reports were found from field studies that addressed the effects of either Disparlure or NPV treatments on aquatic ecosystems. Dichlorvos will be used solely in traps to capture male gypsy moths and is not anticipated to enter aquatic ecosystems. Therefore, field studies involving dichlorvos are not included in this section.

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Table 5-1. Toxicity data of *Bacillus thuringiensis* var. *kurstaki*

Species	Exposure/Dose	Effect	Reference
Mammals			
human	acute oral dose of 1 g/day for 3 consecutive days	no toxicity/ infectivity	U.S. EPA, OPTS, 1986
rat	acute oral dose	$LD_{50} \geq 4.7 \times 10^{11}$ spores/kg	U.S. EPA, OPTS, 1986
rat	acute oral dose	$LD_{50} \geq 2.67$ g/kg	U.S. EPA, OPTS, 1986
mice	acute oral dose of 10,000 mg/kg	no effect	Abbott Labs, 1992
rabbit	acute oral dose	$LD_{50} \geq 2.0 \times 10^{9}$ spores/animal	U.S. EPA, OPTS, 1986
dogs	acute oral dose of 10,000 mg/kg	no effect	Abbott Labs, 1992
rabbit	acute dermal dose (Dipel 6AF®)	$LD_{50} > 2{,}000$ mg/kg	Abbott Labs, 1992
rat	acute inoculation dose	$LD_{50} \geq 3.4 \times 10^{11}$ spores/kg	U.S. EPA, OPTS, 1986
rabbit	acute inoculation dose	$LD_{50} \geq 6.9 \times 10^{7}$ spores/kg	U.S. EPA, OPTS, 1986
rat	acute inhalation dose	$LD_{50} \geq 8 \times 10^{11}$ spores/animal	U.S. EPA, OPTS, 1986
sheep	oral diet of 500 mg/kg/day of Dipel D® or Thuricide-HP® (approx. 10^{12} spores per day) for 5 months	no toxicity or significant treatment-related effect (physical or clinical)	Hadley and others, 1987
Birds			
birds	acute oral dose	$LD_{50} = 178$ ppm, NOEL = 1 ppm	U.S. EPA, OPTS, 1988
bobwhite quail	acute oral dose	$LD_{50} > 10{,}000$ g/kg	Abbott Labs, 1992
mallard	acute oral dose	$LD_{50} > 2{,}000$ mg/kg	Abbott Labs, 1992
mallard	acute oral dose	$LD_{50} > 2{,}000$ mg/kg	Hudson and others, 1984
Fish			
rainbow trout	96 hour exposure	NOEL > 1,000 ppm	Abbott Labs, 1992
rainbow trout	96 hour exposure	$LC_{50} > 10$ mg/L	Mayer and Ellersieck, 1986
bluegill sunfish	96 hour exposure	$LC_{50} = 95$ mg/L	Mayer and Ellersieck, 1986

Table 5-1, *continued ...*

Species	Exposure/Dose	Effect	Reference
rainbow trout, bluegill sunfish, sheepshead minnow	30 days at 100 x maximum expected environmental concentration (MEEC) for label rates of Dipel	no adverse effects	Abbott Labs, 1992
eel	2,000 x MEEC for label rates of Dipel	no adverse effects	Abbott Labs, 1992
Terrestrial Invertebrates			
acarina	biweekly at field application rates for Dipel WP®	no toxic effects	Horsburgh and Cobb, 1981
honey bee	extensive lab and field studies with Dipel at labeled rates	no adverse toxic effects	Abbott Labs, 1992
honey bee	Dipel	relatively nontoxic	Atkins and others, 1981
Hymenoptera Diptera, Neuroptera, Orthoptera, Coleoptera, Hemiptera, Araneae	field and lab studies at labeled application rates for Dipel	no adverse toxic effects	Abbott Labs, 1992
damsel bug, bigeyed bug	label application rates of Dipel WP®	no adverse effects	Jensen, 1974
spined soldier bug (Pentatomid)	label application rates of Dipel WP®	no adverse effects	Wallner and Sturgeoner, 1974
gypsy moth	dietary Dipel HG® exposure	LC_{50} = 95.3 IU/ml diet	Ahmad and others, 1978
gypsy moth	HD-1 strain in diet	LC_{50} = 34 IU/ml diet	Frankenhuyzen and others, 1992
white-marked tussock moth	HD-1 strain in diet	LC_{50} = 12 IU/ml diet	Frankenhuyzen and others, 1992
hemlock looper	HD-1 strain in diet	LC_{50} = 162 IU/ml diet	Frankenhuyzen and others, 1992
jack pine budworm	HD-1 strain in diet	LC_{50} = 145 IU/ml diet	Frankenhuyzen and others, 1992
western spruce budworm	HD-1 strain in diet	LC_{50} = 11 IU/ml diet	Frankenhuyzen and others, 1992
spruce budworm	HD-1 strain in diet	LC_{50} = 63 IU/ml diet	Frankenhuyzen and others, 1992

Table 5-1, *continued ...*

Species	Exposure/Dose	Effect	Reference
tobacco budworm	HD-1 strain in diet	LC_{50} = 1.55 x 10^3 IU/ml diet	Bell and Romine, 1986
corn earworm	HD-1 strain in diet	LC_{50} = 6.06 x 10^3 IU/ml diet	Bell and Romine, 1986
spruce budworm	HD-1 strain in diet	LC_{50} = 163 IU/ml diet	Frankenhuyzen and Fast, 1989
western spruce budworm	HD-1 strain in diet	LC_{50} = 38 IU/ml diet	Frankenhuyzen and Fast, 1989
beet armyworm	dietary exposure of Dipel 2X®	LC_{50} = 196 μg/ml diet	Moar and Trumble, 1987
Trichoplusia ni	Dipel-HG® application	LC_{50} = 16 BIU/ha	James and others, 1993
cinnabar moth	Dipel-HG® application	LC_{50} = 19 BIU/ha	James and others, 1993
diamondback moth	topical Dipel 2X® application	direct dip LC_{50} > 100 mg/ml leaf dip LC_{50} = 0.014 mg/ml	Idris and Grafius, 1993
Diadegma insulare (ichneumonid parasite)	residual Dipel 2X® bioassay	LC_{50} > 100 mg/ml	Idris and Grafius, 1993
Helicoverpa armigera	dietary Dipel exposure	LC_{50} = 55 μg/ml diet	Teakle and others, 1992
Helicoverpa punctigera	dietary Dipel exposure	LC_{50} = 70 μg/ml diet	Teakle and others, 1992
green lacewing	18.7 L/ha of Dipel 4L®	5.3% increased mortality at 7 days	Haverty, 1982
convergent lady beetle	18.7 L/ha of Dipel 4L®	13.4% increased mortality at 7 days	Haverty, 1982
Aphytis melinus	18.7 L/ha of Dipel 4L®	no increased mortality	Haverty, 1982
Apanteles fumiferanae (braconidae)	8.4 BIU/L (Thuricide 48LV®) spray against spruce budworm	50-60% reduction in parasitoid populations with treatment of third instar, no reduction for fourth instar	Nealis and Frankenhuyzen, 1990
striped earwig	10 x label application rate of Dipel WP®	no mortality observed	Workman, 1977

Table 5-1, *continued ...*

Species	Exposure/Dose	Effect	Reference
Chinese praying mantis	consumption of cabbage looper larvae which had consumed *B.t.k.* for 15 hr in 150 μg/ml (18,000 IU/mg) diet	no effect on mortality or survival	Youston, 1973
Aquatic Invertebrates			
brine shrimp	acute static exposure	LC_{50} = 85 mg/L	U.S. EPA, OPTS, 1988
mussels	acute static exposure to 400 ppm	28 % mortality	U.S. EPA, OPTS, 1988
grass shrimp	100 x MEEC from Dipel applications in diet and water for 30 days	no adverse toxic effects	Abbott Labs, 1992
Chironomidae Trichoptera, Megaloptera, mayflies, Plecoptera, black flies	430 IU/ml (Thuricide 32LV®)	only black fly *Simulium vittatum* clearly affected, maybe black fly *Prosimulium fascum/mixtum*	Eidt, 1985
caddisflies, mayflies, stoneflies (12 taxa)	10 x label application rate for Dipel 64AF®	only the stonefly *Leuctra tenuis* was reduced at 4 days after treatment	Kreutzweiser and others, 1993
caddisflies, mayflies, stoneflies (12 species)	maximum concentration of 600 IU/ml in flow-through bioassay (Dipel 8AF®)	no mortality to any species except the stonefly *Taeniopteryx nivalis* which had 30% mortality	Kreutzweiser and others, 1992
stonefly *Pteronarcys californica*	24 hour exposure	LC_{50} = 80 mg/L	Mayer and Ellersieck, 1986
stonefly *Pteronarcys californica*	96 hour exposure	LC_{50} = 10 mg/L	Mayer and Ellersieck, 1986

Table 5-2. Toxicity data of diflubenzuron

Species	Exposure/Dose	Effect	Reference
Mammals			
rat	acute oral dose	LD_{50} = 4,640 mg/kg (technical) LD_{50} = 10,000 mg/kg (WP-25)	Willcox and Coffey, 1978
mouse	acute oral dose	LD_{50} = 4,640 mg/kg (technical) LD_{50} = 10,000 mg/kg (WP-25)	Willcox and Coffey, 1978
rabbit	acute dermal dose	LD_{50} = 4,640 mg/kg (WP-25)	Willcox and Coffey, 1978
sheep and swine	100 ppm diet	no adverse effects	Escobar, 1978
Birds			
mallard duck	acute oral dose 8-day dietary concentration	LD_{50} > 5,000 mg/kg LC_{50} > 4,640 mg/kg	Willcox and Coffey, 1978
bobwhite quail	acute oral dose 8-day dietary concentration	LD_{50} > 5,000 mg/kg LC_{50} > 4,640 mg/kg	Willcox and Coffey, 1978
red-winged black bird	8-day dietary concentration	LC_{50} = 3,763 mg/kg	Willcox and Coffey, 1978
Fish			
fish	40 g a.i./ha application	no adverse effects	Everts, 1990
rainbow trout	Dimilin 25-WP® - 96 hour exposure	EC_{50} = 240 mg/L	Julin and Sanders, 1978
channel catfish	Dimilin 25-WP® - 96 hour exposure	EC_{50} = 370 mg/L	Julin and Sanders, 1978
fathead minnow	Dimilin 25-WP® - 96 hour exposure	EC_{50} = 430 mg/L	Julin and Sanders, 1978
bluegill sunfish	Dimilin 25=WP® - 96 hour exposure	EC_{50} = 660 mg/L	Julin and Sanders, 1978
mummichog (*Fundulus heteroclitus*)	Dimilin 25-WP® - 96 hour exposure	LC_{50} = 32.99 mg/L NOEC = 29.86 mg/L	Lee and Scott, 1989

Table 5-2, *continued ...*

Species	Exposure/Dose	Effect	Reference
yellow perch	96 hour exposure	LC_{50} > 50 mg/L	Mayer and Ellersieck, 1986
brook trout	96 hour exposure	LC_{50} > 50 mg/L	Mayer and Ellersieck, 1986
cutthroat trout	Dimilin 25-WP® - 96 hour exposure	LC_{50} > 60 mg/L	Mayer and Ellersieck, 1986
Atlantic salmon	96 hour exposure	LC_{50} > 50 mg/L	Mayer and Ellersieck, 1986
smallmouth bass	96 hour exposure	LC_{50} = 10-100 mg/L LC_{50} = 250 mg/L (juvenile)	Willcox and Coffey, 1978
Terrestrial Invertebrates			
phytoseiid and stigmaeid mites	187 ppm spray to apple orchards	no adverse effects	Anderson and Elliott, 1982
European red and rust mites	187 ppm spray to apple orchards	no population increases following treatment	Anderson and Elliott, 1982
earthworm (*Eisenia fetida*)	soil exposure	NOEC = 1 g Dimilin WP-25 per kg dry soil	Berends and Thus, 1992
earthworm (*Eisenia fetida*)	soil exposure	NOEC = 780 mg diflubenzuron per kg dry soil	Berends and others, 1992
nematodes	10 day dietary exposure to Dimilin®	adults unaffected, but reproduction hindered and egg hatch prevented. population reductions of 5% for *Pelodera* sp., 47% for *Panagrellus redivivus*, and 94% for *Acrobeloides* sp. at 10 ppm.	Veech, 1978
honey bee	acute toxicity	diflubenzuron relatively nontoxic	Atkins and others, 1981

Table 5-2, *continued ...*

Species	Exposure/Dose	Effect	Reference
honey bee	Dimilin 25-WP® - exposure to 59 ppm in syrup and 100 ppm in water	elimination of brood production, less comb, and less new workers	Barker and Waller, 1978
honey bee	Dimilin® - topical exposure	LD_{50} = 52.9 mg/kg (3rd instar) LD_{50} = 45.51 mg/kg (4th instar) LD_{50} = 22.33 mg/kg (pupa)	Chandel and Gupta, 1992
bee *Apis cerana indica*	Dimilin® - topical exposure	LD_{50} = 56.15 mg/kg (3rd instar) LD_{50} = 49.13 mg/kg (4th instar) LD_{50} = 22.69 mg/kg (pupa)	Chandel and Gupta, 1992
honey bee	Dimilin 25-WP® - 0.4 kg a.i./ha to apple orchards	no adverse effects	Emmett and Archer, 1980
honey bee	Dimilin® - topical exposure	LD_{50} > 30 μg/bee LD_{50} > 200 μg Dimilin WP-25 per bee.	Gijswijt, 1978
	Dimilin® - dietary exposure	No adverse effects at 5.9 ppm	
honey bee	acute topical exposure	LD_{50} > 100 μg/bee (adult) LD_{50} >0.0125 μg/bee (larva)	Kuijpers, 1989
	acute oral exposure	LD_{50} > 100 μg/bee (adult) LD_{50} > 0.030 μg/bee (larva)	
honey bee	dietary pollen cakes with 10 ppm	50% reduction in stored sugar	Nation and others, 1986
honey bee	Dimilin® - dietary sugar cakes of 1 ppm	reduction in sealed brood	Stoner and Wilson, 1982
honey bee	Dimilin 25-WP® - dietary exposure	LC_{50} = 3.7 ppm	Wittmann, 1982
boll weevil parasite *Trichogramma* sp.	46.7 L/ha	unaffected in the absence of crop oil	Bull and Coleman, 1985

Table 5-2, *continued ...*

Species	Exposure/Dose	Effect	Reference
nontarget insects (lacewing *Chrysopa oculata*, braconid wasp *Macrocentrus ancylivorous*, assassin bug *Acholla multispinosa*)	Dimilin 25-WP® - topical exposure up to 300 ppm and contact with treated leaves. Dimilin 25-WP® - consumption of treated host larvae	considerable mortality and inhibition of molting to lacewing, but no effects to wasp or bug. reduced emergence of wasp, but no effect on lacewing or bug.	Broadbent and Pree, 1984
large milkweed bug *Oncopeltus fasciatus*	topical exposure to 1 microgram	inhibition of reproduction	Redfern and others, 1980
European earwig *Forficularia auricularia*	12.5 g a.i./ha	growth and mobility adversely affected	Sauphanor and others, 1993
migratory grasshopper *Melanoplus sanguinipes*	Dimilin 25-WP® - dietary exposure	LC_{50} = 0.08 ppm (lettuce diet) LC_{50} = 0.1 ppm (wheat seedling diet)	Elliott and Iyer, 1982
desert locust and grasshopper	application mortality	LD_{100} = 64 g a.i./ha LD_0 = 22.5 g a.i./ha	FAO, 1992
desert locust *Schistocerca gregaria*	dietary exposure	LD_{50} = 886.7 μg AI (2nd instar) LD_{50} = 207.4 μg AI (4th instar) LD_{50} = 325.2 μg AI (5th instar)	Jepson and Yemane, 1991
orthopteran *Oxya japonica*	Dimilin 25-WP® - topical exposure	LD_{50} = 0.06 μg per insect or 0.31 mg/kg	Lim and Lee, 1982
German cockroach *Blatella germanica*	Dimilin 25W® - contact with spray of treated cage plywood panels	population reduction of 67.3% at 30 mg/m^2, 93% at 60 mg/m^2, and 98.2% at 120 mg/m^2. egg hatch unaffected, but high first instar mortality.	Wadleigh and others, 1991
German cockroach	Dimilin 25-WP® - dietary exposure	no mortality at 4 mg/kg, 15% at 20 mg/kg, 88% at 100 mg/kg, and 100% at 500 mg/kg	Tsuji and Taneika, 1988
cockchafer *Melolontha*, leaf beetle *Gastroidea viridula*	beech or sorrel leaves treated with 0.1% Dimilin 25-WP®	repellant effects and 100% ovicidal effect to chafer. effective against larvae and eggs of beetle.	Büchi and Jossi, 1979

Table 5-2, *continued ...*

Species	Exposure/Dose	Effect	Reference
rove beetle *Aleochara bilineata*	consumption of cabbage maggot treated with Dimilin 25-WP®	no adverse effects	Gordon and Cornect, 1986
Mexican bean beetle	Dimilin 25-WP® - dietary exposure	LC_{50} = 3.4 ppm (3rd instar)	McWhorter and Shepard, 1977
Colorado potato beetle	foliage treatments	unaffected at 50 mg/L, slightly at 100 mg/L, and strongly at 300-500 mg/L	Tamaki and others, 1984
cotton leafworm *Spodoptera littoralis*	Dimilin 25-WP® at 0.5 kg/feddan	suppression of oviposition and high ovicidal activity	Abo-Elghar and others, 1980
gypsy moth *Lymantria dispar*	topical exposure	LD_{50} = 3.58 mg/kg (alder) LD_{50} = 8.96 mg/kg (douglas fir) LC_{50} = 0.06 ppm diet (alder) LC_{50} = 0.45 ppm diet (douglas fir)	Berry and others, 1993
	acute oral exposure		
gypsy moth	Dimilin 25-WP® - dietary exposure at 0.1 mg/kg	100% lethal to larvae	Martinat and others, 1988
cotton leafworm *Spodoptera littoralis*	dietary exposure	LC_{50} = 1 mg/kg LD_{50} 380 ng/larva	Neumann and Guyer, 1987
rice swarming caterpillar adult *Spodoptera mauritania*	Dimilin 25-WP® - dietary exposure	60-64% sterility at 10 ppm, 100% sterility at 100-1,000 ppm	Beevi and Dale, 1984
large white butterfly *Pieris brassicae*	topical exposure	LD_{50} = 2.5 μg/insect or 1.07 mg/kg	Sinha and others, 1990
hemlock looper *Lambdina fiscellaria*	Dimilin 25-WP® at 70 g a.i./ha	> 90% reduction in larval and adult populations	Retnakaran and others, 1988
forest tent caterpillar *Malacosoma disstria*	Dimilin 25-WP® at 70 g a.i./ha	total control	Retnakaran and others, 1979

Table 5-2, *continued ...*

Species	Exposure/Dose	Effect	Reference
house fly *Musca domestica* and parasitoid *Muscidifurax raptor*	Dimilin® - topical exposure	no effect to eggs or pupae at 10,000 ppm. > 90% mortality to intermediate to late stage larvae at 1.25 to 10 ppm. No effects to parasitoid.	Ables and others, 1975
house fly	Dimilin 25-WP® - soil treatment effects	LEL = 12.5 ppb. directly treated soil (75 g a.i./ha) results in 90-100% mortality	Barth, 1981
banana fruit fly *Zaprionus paravittiger*	dietary exposure	50% inhibitory dose for adult emergence = 0.16 ppm for 1st instar, 0.32 ppm for 2nd instar, and 0.95 ppm for 3rd instar	Chopra and Rup, 1985
horn fly *Haematobia irritans*	4 weeks after 0.5% spray treatment of Dimilin 25-WP® on range cattle	elimination of adult emergence	Kunz and others, 1977
tachinid parasite *Doryphorphaga doryphorae*	treatment of potato foliage for control of Colorado potato beetle	unaffected at 50 mg/L, but only 0-4% survival at 300-500 mg/L	Tamaki and others, 1984
Aquatic Invertebrates			
dragonfly nymphs *Orthemis* spp., *Pantala* sp.	TH 6040® (diflubenzuron) - 168 hour exposure	LC_{50} = 50 μg/L	Miura and Takahashi, 1974
mayfly nymphs *Callibaetis* sp.	TH 6040® (diflubenzuron) - 168 hour exposure	LC_{90} = 10 μg/L	Miura and Takahashi, 1974
Hydropsychidae (Trichoptera)	Dimilin 25-WP® 15 days after lab treatment	1.9% survival at 0.0025 ppm, 2.2% survival at 0.025 ppm, 0.59% survival at 0.25 ppm. No adult emergence from treated tanks and only 31.6% emergence from control tanks	Bradt and Williams, 1990

Table 5-2, *continued ...*

Species	Exposure/Dose	Effect	Reference
mayflies, stoneflies, Coleoptera, bacteria, oligochaetes, gastropods, algae, fungi	5 month flow-through stream exposure with Dimilin®	mayfly and stonefly affected at 1 μg/L, diptera as well as algae and fungi affected at 10 μg/L, but Coleoptera, bacteria, oligochaetes, and gastropods unaffected.	Hansen and Garton, 1982a
mayflies (heptageniid), stonefly nymph *Peltoperla arcuata*	Dimilin 25-WP® exposure in water and diet	mayflies sensitive to 0.6 ppb with partial molt inhibition and decreased activity, decreased survival of stonefly nymphs	Harrahy and others, 1994
perlodid stonefly *Skwala* sp.	Dimilin 25-WP® - 96 hour exposure	LC_{50} = 57 mg/L	Mayer and Ellersieck, 1986
Trichoptera, Plecoptera	Dimilin W-25® at 0.0625-0.25 lb a.i./acre	slight population reductions not attributed to applications	Rabeni and Gibbs, 1975
chironomid midge larvae, mayfly nymphs, caddisfly larvae	Dimilin W-25® in outdoor experimental stream exposures	high larval mortality to chironomid midges, mayfly nymphs and caddisfly larvae also affected at 1 and 10 mg/L	Yasuno and Satake, 1990
Chironomus plumosus, 4th instar larvae	Dimilin 25-WP® - 48 hour exposure	EC_{50} = 0.56 mg/L	Julin and Sanders, 1978
Tanytarsus dissimilis	96 hour exposure	LC_{50} = 1.02 μg/L	Hansen and Garton, 1982b
Aedes nigromaculatum	TH 6040® (diflubenzuron) - 48 hour exposure	LC_{50} = 0.5 μg/L	Miura and Takahashi, 1974
Culex quinquefascia-tus	Dimilin 25-WP® at 112 g/ha (0.1 lb/acre)	satisfactory control for 1-2 weeks in waste lagoon	Axtell and others, 1980
Culex quinquefascia-tus, *C. salinarius*, *C. restuans*	Dimilin 25-WP® at 0.125 ppm in waste lagoon	larval control for 14 to 35 days	Barker and Booram, 1979
water scavenger beetle larvae *Hydrophilus triangularis*	TH 6040® (diflubenzuron) - 48 hour exposure	LC_{50} = 100 μg/L	Miura and Takahashi, 1974

Table 5-2, *continued ...*

Species	Exposure/Dose	Effect	Reference
water scavenger beetle adults *Laccophilus* spp., *Thermonectus basillaris*, *Tropisternus lateralis*	TH 6040® (diflubenzuron) concentrations as high as 250 μg/L	no mortality	Miura and Takahashi, 1974
Daphnia magna (1st instar)	Dimilin 25-WP® - 48 hour exposure	EC_{50} = 0.015 mg/L (1st instar)	Julin and Sanders, 1978
Gammarus pseudolimnaeus (mature scuds)	Dimilin 25-WP® - 48 hour exposure	EC_{50} = 0.030 mg/L	Julin and Sanders, 1978
Daphnia magna	24 hour exposure	EC_{50} = 0.0680 mg/L	Kuijpers, 1988
	48 hour exposure	EC_{50} = 0.0071 mg/L NOEC = 0.00045 mg/L	
Daphnia magna	Dimilin 25-WP® - 48 hour exposure	LC_{50} = 0.00075 mg/L (neonate) LC_{50} = 0.02345 mg/L (adult)	Majori and others, 1984
harpacticoid copepod *Tigriopus californicus*	Dimilin® at 1-10 μg/L	development hindered	Antia and others, 1985
marine copepod *Acartia tonsa*	Dimilin® at 1000 ppb	survival or fecundity of adults unaffected.	Tester and Costlow, 1981
	Dimilin® - 10 ppb after 24 hours	less than 5% hatch of viable nauplii less than 50% hatch of viable nauplii	
	Dimilin® - 1 ppb after 12 hours		
brine shrimp *Artemisia salina*	Dimilin® at 2 ppb	reduced lifespan.	Cunningham, 1976
	Dimilin® at 1 ppb	no effect.	

Table 5-2, *continued ...*

Species	Exposure/Dose	Effect	Reference
grass shrimp *Palaemonetes pugio*	72 hour exposure	LC_{50} = 2.95 µg/L (technical) LC_{50} = 2.83 µg/L (WP-25)	Wilson and Costlow, 1986
		LC_{50} = 1.84 µg/L (technical) LC_{50} = 1.39 µg/L (WP-25)	
	96 hour exposure		
grass shrimp *Palaemonetes pugio* (pre-molt animals)	24 hour exposure	LC_{50} = 1.11 µg/L	Touart and Rao, 1987
mysid shrimp *Mysidopsis bahia*	Dimilin® - 96 hour exposure	LC_{50} = 1.97 µg/L MATC < 0.4 µg/L	Nimmo and others, 1981
Hyalella azteca	96 hour exposure	LC_{50} = 1.84 µg/L	Hansen and Garton, 1982b
crab larvae	Dimilin® at 1-10 ppb	inhibition of crab larval cuticle formation	Christiansen and others, 1978; Christiansen, 1987
fiddler crab *Uca pugilator*	Dimilin® acute exposure	NOEC = 20 µg/L (molting) NOEC = 2 µg/L (survival) NOEC = 0.2 µg/L (behavioral)	Cunningham and Myers, 1987
horseshoe crab *Limulus polyphemus*	Dimilin 25-WP® at 5 µg/L Dimilin 25-WP® at 50 µg/L	slight molt delay severe mortality	Weis and Ma, 1987
freshwater clam *Anodonta cygnea*	Dimilin 25-WP® at 200 mg/L	affects lamellar calcification	Machado and others, 1990
quahogs *Mercenaria mercenaria*	100 mg/L	adversely affected.	Surprenant, 1989
	320 µg/L	NOEC > 320 µg/L	
snail *Physa* sp.	acute exposure	LC_{50} > 125 ppm	Willcox and Coffey, 1978

Table 5-2, *continued ...*

Species	Exposure/Dose	Effect	Reference
barnacle *Balanus eburneus*	1000 ppb 50-500 ppb	lethal exposure; sublethal exposure; both exposures result in accelerated molting and decreased survival.	Gulka and others, 1980
Plants			
marine diatoms	Dimilin® - 5 mg/L exposure	unaffected	Antia and others, 1985
duckweed *Lemna gibba* G3	14 day exposure	NOEC at 190 μg/L	Thompson and Swigert, 1993d
terrestrial fungi	50 and 100 ppm exposure	NOEC at 100 ppm for *Rhizopus* sp., *Aspergillus* sp., *Fusarium* sp., *Trichoderma* sp., inhibition of growth at 50 and 100 ppm for *Pythium* sp.	Booth, 1978
terrestrial fungus *Rhizoctonia solani*	300 ppm exposure	inhibition of growth	Townshend and others, 1983

Table 5-3. Toxicity data of nucleopolyhedrosis virus

Species	Exposure/Dose	Effect	Reference
Mammals			
humans	fed 5.82 x 10^9 polyhedra/day for 5 days	no evidence of adverse effects	Heimpel and Buchanon, 1967 as cited in Ghassemi and others, 1983
white-footed mouse, short-tailed shrew, Virginia opossum	fed formulations per larvae with up to 6 x 10^8 OB/day	no adverse effects on physical condition or reproductive efficiency. no evidence of gross or microscopic lesions	Lautenschlager and others, 1977
rat	acute exposure by gastric intubation	no toxicity at 40 x 10^{19} polyhedra/rat	U.S. EPA, 1977
rat	2 year feeding study at 10^7 -10^8 polyhedra/rat/day	no mortality or tumorigenicity	U.S. EPA, 1977
guinea pig	acute dermal exposure at 40 x 10^9 polyhedra/pig	no acute toxicity	U.S. EPA, 1977
rabbit	primary skin irritation test at 40 x 10^9 polyhedra/ rabbit	no skin irritation or toxicity	U.S. EPA, 1977
rabbit	eye irritation test	no toxicity	U.S. EPA, 1977
Birds			
black-capped chickadee, house sparrow	fed larvae with 3.3 x 10^7 to 2.1 x 10^8 OB on alternate days	no short-term effects, no body weight changes, no histopathological changes noted	Podgwaite and Galipeau, 1978
mallard hens	five daily oral treatments of 18.34 x 10^6 polyhedra/mg (500 mg/kg/day)	no effects on survival	U.S. EPA, 1977
resident wild birds, quail (caged)	2.5 x 10^{12} OB/ha aerial gypsy moth application	no adverse effects on physical condition or populations. no evidence of gross or microscopic lesions	Lautenschlager and others, 1979

Table 5-3, *continued ...*

Species	Exposure/Dose	Effect	Reference
Fish			
brown trout, bluegill	96 hour exposure	$LC_{50} > 1.5 \times 10^9$ OB/g (equivalent to 5490 ppm)	U.S. EPA, 1977
longnose killifish, sheepshead minnow, spot	feeding studies	no sign of pathogenicity	U.S. EPA, 1977
Terrestrial Invertebrates			
gypsy moth (*Lymantria dispar*)	acute dietary exposure	first instar $LC_{50} = 0.93 \times 10^3$ OB/ml; second instar $LC_{50} = 1.75 \times 10^3$ OB/ml; third instar $LC_{50} = 2.84 \times 10^3$ OB/ml; fourth instar $LC_{50} = 21.73 \times 10^3$ OB/ml; fifth instar $LC_{50} = 103.62 \times 10^3$ OB/ml	Shapiro and others, 1986
gypsy moth	NPV epizootic effect	lowers female pupae to 25%, favors ichneumonid parasitism	Doane, 1976b
gypsy moth	spray for control of first instar	1-2.5 x 10^{12} polyhedra/ha gives population density below control threshold; 5 x 10^{11} polyhedra/ha gives only limited control	Cunningham and others, 1993
gypsy moth	1.25 x 10^{12} OB/ha	80-98% reduction in egg masses	Podgwaite and others, 1992
gypsy moth	2.5 x 10^{12} OB/ha	81% reduction in egg masses	Podgwaite and others, 1991
honey bee	fed 10 x 10^9 polyhedra/hive with sucrose	no adverse effects	U.S. EPA, 1977

Table 5-3, *continued ...*

Species	Exposure/Dose	Effect	Reference
Aquatic Invertebrates			
Daphnia magna	48 hour exposure	LC_{50} > 250 OB/ml (equiv. to application rate of 2.5 x 10^{12} OB/ha	U.S. EPA, 1977
juvenile oysters	feeding studies	no sign of pathogenicity	U.S. EPA, 1977

Table 5-4. Toxicity data of dichlorvos

Species	Exposure/Dose	Effect	Reference
Mammals			
mice, female	acute oral dose	LD_{50} = 133 mg/kg LD_{1} = 106 mg/kg	Haley and others, 1975
mice, male	acute oral dose	LD_{50} = 139 mg/kg LD_{1} = 81 mg/kg	Haley and others, 1975
mice	acute oral dose	LD_{50} = 135 mg/kg	Wagner and Johnson, 1970
rat, female	acute oral dose	LD_{50} = 135 mg/kg LD_{1} = 50 mg/kg	Wagner and Johnson, 1970
rat, male	acute oral dose	LD_{50} = 145 mg/kg LD_{1} = 30 mg/kg	Wagner and Johnson, 1970
rat	acute oral dose	LD_{50} = 56 mg/L	Gaines, 1969
pig	acute oral dose	LD_{50} = 157 mg/kg LD_{0} = 56 mg/kg	Stanton and others, 1979
dog	systemic effects	NOEL = 0.08 mg/kg/day LEL = 0.8 mg/kg/day	U.S. EPA, ORD, 1989
dog	AChE effects	NOEL = 0.08 mg/kg/day LEL = 0.8 mg/kg/day	U.S. EPA, ORD, 1989
cattle	AChE effects	NOEL = 0.18 mg/kg/day LEL = 0.91 mg/kg/day	U.S. EPA, ORD, 1989
rat	AChE effects	NOEL = 0.05 mg/kg/day, LEL = 0.5 mg/kg/day	U.S. EPA, OPP, 1989
mice	teratogenic effects	NOEL = 60 mg/kg/day	Schwetz and others, 1979
rabbit	teratogenic effects	NOEL = 5 mg/kg/day	Schwetz and others, 1979

Table 5-4, *continued ...*

Species	Exposure/Dose	Effect	Reference
Birds			
European starling	acute oral dose	LD_{50} = 12 mg/kg	Smith, 1987
Red-winged blackbird	acute oral dose	LD_{50} = 17 mg/kg	Smith, 1987
Terrestrial Invertebrates			
honey bee	topical dose	LD_{50} = 0.495 μg/bee	U.S. EPA, OPTS, 1987
honey bee	topical dose	LD_{50} = 0.501 μg/bee LD_1 = 0.2687 μg/bee	Atkins and others, 1981
carpenter ant *Camponotus novaboracensis*	acute exposure	LD_{50} = 4.1 μg/jar LD_1 = 0.4689 μg/jar	Gibson and Scott, 1989
carpenter ant (*Camponotus pennsylvanica*)	acute exposure	LD_{50} = 3.5 μg/jar LD_1 = 0.005042 μg/jar	Gibson and Scott, 1989
Aquatic Species			
rainbow trout	96 hour exposure	LC_{50} = 100 μg/L	Mayer and Ellersieck, 1986
fathead minnow	96 hour exposure	LC_{50} > 23,000 μg/L	Mayer and Ellersieck, 1986
stonefly nymphs	96 hour exposure	LC_{50} = 0.1 μg/L	Mayer and Ellersieck, 1986
hermit crabs	96 hour exposure	LC_{50} = 45 μg/L	HSDB, 1989

Table 5-5. Toxicity data of Disparlure

Species	Exposure/Dose	Effect	Reference
Mammals			
rat	acute oral dose	LD_{50} > 34,600 mg/kg	Industrial Bio-Test Laboratories, Inc., 1972
rabbit	acute dermal dose	LD_{50} > 2,025 mg/kg	Industrial Bio-Test Laboratories, Inc., 1972
rabbit	eye irritation test	not an eye irritant	Industrial Bio-Test Laboratories, Inc., 1972
rabbit	primary skin irritation test	not a skin irritant	Industrial Bio-Test Laboratories, Inc., 1972
Birds			
mallard ducklings	8-day dietary exposure	LC_{50} > 5,000 ppm (=NOEL)	USDI, Fish and Wildlife Service, 1975
bobwhite quail chicks	8-day dietary exposure	LC_{50} > 5,000 ppm (=NOEL)	USDI, Fish and Wildlife Service, 1975
Fish			
rainbow trout	96 hour exposure	LC_{50} > 100 mg/L	USDI, Fish and Wildlife Service, 1972
bluegill sunfish	96 hour exposure	LC_{50} > 100 mg/L	USDI, Fish and Wildlife Service, 1972

Table 5-6: Effects of *B.t.k.* on nontarget Lepidoptera

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Target = Spruce budworm (*Choristoneura fumiferana*)	Thuricide 16B Dipel WP, with and without chitinase. 2 & 4 lbs./acre	Algonquin Park, Ontario, & Spruce Woods, Manitoba. Spruce-Fir forests.	Numbers of hand-picked larvae from aspen, alder, and maple were not different on control and treated plots.	No	Yes	Buckner and others, 1974
32 species of Lepidoptera on tobacco brush, *Ceanothus velutinus*	20 BIU/ha	Estacada, Clackamas Co., OR Program to control spruce budworm (*Choristoneura occidentalis*)	Number of larvae on shrubs in treated site decreased 80% between pre- and posttreatment surveys compared to control site where number of larvae increase 6% in same time period, 2 weeks post-spray. 2 months post-spray there were no differences between spray and control sites. One year after spray and 1st post-spray sample, larval abundance was lower than pre-treatment the previous year and lower than control site. Late-season sample in 2nd year was again no different between control and treated sites. Species richness & diversity not statistically significantly different between control and treated site; *B.t.k.* tended to even-out the proportions of species.	No	Yes	Miller, 1990a

Table 5-6, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
35 spp. Lepidoptera in 10 families. All in Garry oak (*Quercus garryana*)	40 BIU/ha 3 times	Elmira, Lane Co., OR Program to manage gypsy moth	3 postspray samples. Significant differences in caterpillar density between treated and control plots for each postspray sample; remained significant by day 68, not by day 90. Species richness significantly lower on treated plots. Species richness and larval abundance significantly lower 1 year after spray, but not 2 years after spray. Results expected of univoltine species.	No	Yes	Miller, 1990b
Forest Lepidoptera on 30 ha plots	Thuricide 32LV 3.5 L/ha	White Mtns., NH. Northern hardwoods: maple, beech, birch.	Spraying in 1983 significantly reduced caterpillars relative to unsprayed plots. No differences in 1984 & 1985, because numbers were naturally low on control plots those years.	No	Yes	Rodenhouse and Holmes, 1992
Nontarget moths in Asian strain of the gypsy moth eradication program area	60 BIU/ha (24 BIU/acre)	Pierce and King Counties, WA	Full-spectrum lights. 49-97% lower catches at treated sites in 1993 v. same sites in 1992; stat. significant drop. Three species (*Orthosia hibisci, Protorthodes rufula, Perizoma curvilinea*) eliminated from site? Overall, moth diversity unaffected.	Yes	No	Crawford and others, 1993

Table 5-6, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Micro- and macrolepid-optera	89 BIU/ha (36 BIU/acre) Sprayed in 1992	Rockbridge Co., VA Oak woodland, 50 acre plots	Sampled in 1992 and 1993. Pre- and post (day 6 and 12) foliage samples from canopy, subcanopy and shrub-layer show reductions in the relative abundance of 16/19 most common taxa. 12/16 were microlepidopterans. In 1992 larval abundance reduced on 3/5 *B.t.k.* sites in canopy and subcanopy. Reduction in Microleps. in 4/5 sites in canopy and 3/5 sites in subcanopy. Uneven application accounted for variable effects. 2 plots consistently showed the greatest effects. Total numbers of lepidopterans on foliage were no different on treated and control sites in 1993. Microlepidoptera accounted for 95% of the individuals collected from foliage in 1992 and about 85% in 1993. 6/8 most common macrolepidopteran species trapped under burlap bands were reduced by treatment. 3 of these spp. were nearly absent in treated plots (*Satyrium calanus, Malacosoma disstria, Orthosia rubescens*). Other less common spp. appeared to be significantly less on treated plots. *Dasychira obliquata* was not affected apparently. Noctuidae also lower in 1993.	Yes	Yes	Peacock and others, 1994

Table 5-6, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Lepidoptera Sampled in 1990-1992	14.4 BIU/ha Sprayed in 1991	Grant and Pendleton counties, WV. 50 ha plots in Oak-hickory with pine; blueberry shrub layer.	4 treatments: control; bt-sprayed without gypsy moth (GM); bt with GM; GM alone (defoliated). Foliage and blacklight (b-l) samples. Total larval abundance reduced following *B.t.k.* application in 1991. No effects of *B.t.k.* and GM on several microlepidopterans noted. Total spp. richness (SR), SR of Noctuidae, and of Geometridae reduced in *B.t.k.* plots in 1991. Residual effects noted in 1992 on Noctuidae. B-l sampling found reductions in 1991 in abundance of total Lepidopterans, microleps., and Geometridae. Total lepidop., microleps. and noctuids were reduced in 1992. Few differences in adult Lepidoptera richness between sprayed and control plots.	Yes	Yes	Sample and others, 1995; Butler, 1995

1--The R column indicates whether the study design included replicates, and the C indicates controls.

Table 5-7. **Effects of *B.t.k.* on parasites of Lepidoptera**

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Gypsy moth parasites	Thuricide 10 & 20 BIU/ha (4 & 8 BIU/acre)	Cockaponset State Forest, CT Oak woodland	Numbers of cocoons of *Apanateles melanoscelus* on treated versus control plots no different. Rate of parasitism by *A. melanoscelus* significantly greater on treated plots. Other species of wasp parasites unaffected.	Yes	Yes	Dunbar and others, 1973
Spruce budworm and hymenopterous parasites	Thuricide 16B, & Dipel WP with & without chitinase. 2 & 4 lbs/acre	Algonquin Park, Ont.	No evidence of effects from *B.t.k.* treatments on rate of parasitism of spruce budworm (*Choristoneura fumiferana*) larvae on treated and control plots.	Yes	Yes	Buckner and others, 1974
6 species of wasp and 3 spp. of fly parasites of gypsy moth	Dipel 20 BIU/ha Applied twice	High Point State Park, NJ	Significantly fewer numbers of *Brachymeria intermedia* (wasp) in treated versus control plots in 1 year. Significantly fewer numbers of *Parasetigena silvestris & Compsilura concinnata* (flies) on treated versus control plots the next year. Significantly fewer wasp parasites in family Braconidae and fewer fly parasites in Sarcophagidae caught in the second year. Rate of parasitism not examined. Differences attributed to reduced host availability on treated sites rather than to direct toxicity of *B.t.k.* to parasites.	Yes	Yes	Reardon and others, 1979

Table 5-7, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Gypsy moth plus 2 wasp and 3 fly parasites.	Dipel 4L 19.8 BIU a.i./ha with sticker & water	Tuscarora State Forest Cumberland Co., PA mixed oak	Parasitism rate by *A. melanoscelus* (wasp) significantly greater on treated plots, but not for *Phobocampe unicincta* (fly). Significant reduction in rate of parasitism for the fly parasites *Blepharipa pratensis* and *C. concinnata*. *Parasetigena silvestris* unaffected. Enhancement of parasitism by wasp correlated with delayed development of *B.t.k.*-poisoned gypsy moth larvae. Parasitism rate reduced on all plots in the year following treatment. Cold winter & sparse snow cover could be the reason.	Yes	Yes	Ticehurst and others, 1982
Gypsy moth and wasp parasite *Cotesia* (=*Apanateles*) *melanoscelus*	Dipel 4L 30 & 40 BIU/ha	Lyme, CT Oak woodland	Parasitism 12.5% to 20% higher on treated plots. Parasitic flies were unaffected. Increase in wasp parasitism is probably due to delayed development of *B.t.k.*-poisoned gypsy moth larvae--gut paralysis and slower feeding rates of larvae increase time available for parasite development.	Yes	Yes	Andreadis and others, 1983
Gypsy moth and wasp parasite *Cotesia* (=*Apanateles*) *melanoscelus*	Dipel 8L 40 BIU/ha w/ 2% Bond sticker	Kent, Caroline, & Queen Anne counties, MD Mixed hardwood	Significantly greater number of parasites on replicate blocks treated with *B.t.k.* than on control blocks. Increased numbers were predicted based on increased development time of *B.t.k.*-poisoned hosts.	Yes	Yes	Webb and others, 1989

Table 5-7, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Hymenoptera in Gypsy moth defoliated and undefoliated plots Sampled in 1990-1992	14.4 BIU/ha Sprayed in 1991	Grant and Pendleton counties, WV. 50 ha plots in Oak-hickory with pine; blueberry shrub layer.	4 treatments: control; bt-sprayed without gypsy moth (GM); bt with GM; GM alone (defoliated). Foliage and blacklight (b-l) samples. No differences (at the order level of taxonomic categories) in numbers of Hymenoptera collected on treated and control plots. Wasps in the mostly parasitic family Ichneumonidae were less abundant at treated plots (with and without Gypsy moths), and braconid wasps more abundant on plots with Gypsy moths. Reduced abundance of lepidopteran hosts could explain the reduction among Ichneumonidae.	Yes	Yes	Sample and others, 1995

1--The R column indicates whether the study design included replicates, and the C indicates controls.

Table 5-8. Effects of *B.t.k.* on birds

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Black-throated blue warbler (*Dendroica caerulescens*)	Thuricide 32LV w/ Rhoplex sticker 3.5 l/ha	White Mtn. Natl Forest, NH. Sugar maple, American beech, yellow birch are dominant spp.	Caterpillar biomass significantly different between 1 sprayed and 2 control plots in 1983, and significantly fewer nesting attempts and significantly fewer caterpillars in the diets of young in those plots. No significant differences noted in production of young per pair; clutch size, hatching success, & no. of young fledgling per nest not different between treated and control sites. No differences noted between treated and control areas in caterpillar abundance in 2 years after treatment--natural dip in caterpillar abundance those years.	No	Yes	Rodenhouse and Holmes, 1992
Chestnut-backed and Black-capped chickadees (*Parus rufescens*, and *P. atricapillus*)	Not given. 60 BIU/ha (24 BIU/acre)?	Near Portland, OR and surrounding counties	No effects on growth rate or fledgling success in 1st year. Reduced fledgling success 2nd year due to unexplained nest abandonment on 3 treatment plots (also 1 nest on control plot). Significantly smaller proportion of caterpillars brought as food on treatment sites both years, but provisioning rate no different.	Yes	Yes	Gaddis, 1987 Gaddis and Corkran, 1986

1--The R column indicates whether the study design included replicates, and the C indicates controls.

Table 5-9. **Effects of diflubenzuron on nontarget Lepidoptera**

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Canopy arthropods	70 g a.i./ha (1 oz a.i./ac)	Sleepy Creek, Morgan Co., WV. Oak-pine and oak-hickory hardwood.	Some micro- and macrolepidoptera were significantly reduced on treated plots, as were other herbivorous invertebrate species, such as grasshoppers. Microlepidoptera were least affected.	Yes	Yes	Martinat and others, 1988
Canopy arthropods	70 g a.i./ha (2 oz a.i./ac)	Fernow Experimental Forest, Parsons, WV	Under tree band traps taxa richness of Lepidoptera (butterflies and moths) larvae was affected in 1992, treatment year. Foliage sampling found reduced abundance and richness Lepidoptera. Macrolepidoptera larval richness and abundance remained reduced in 1993, year following treatment.	Yes	Yes	Butler, 1993
Macro- and micro-lepidoptera	Dimilin 25W @ 56.7 g /ac. (0.5 oz a.i./ac) Applied in 1990	Coopers Rock State Forest, WV Oak woodland	Two-year study: year of treatment and post-treatment year. Study design limits data interpretation. Species richness and abundance of larvae on foliage lower on treated site after treatment. Species richness reduced the following year on treated site. No measurable effects from treatment on captures at 1 light trap on treated and 1 on control site. Tree bands yielded no differences in larvae trapped on treated and control sites. A general decline in species richness and abundance of nontarget Lepidoptera noted over 8 years could be partly due to applications of diflubenzuron.	No	Yes	Butler and Kondo, 1993

Table 5-9, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Insects greater than 3 mm long. Black-light trapped.	70.75 g a.i./ha (1 oz a.i./ac)	Grant Co., WV Oak, hickory, and pine.	Species richness of Lepidoptera was lower in sprayed plots. Significant reductions found in 3 of the 4 families of macrolepidopterans in both the year of spray and the following year. Significant reductions in the year following spray in Noctuidae. Microlepidopteran families not significantly different on sprayed and control plots. Largest effect found year after treatment--lag time expected.	No	Yes	Sample and others, 1993c

1--The R column indicates whether the study design included replicates, and the C indicates controls.

Table 5-10. Effects of diflubenzuron on predators and parasites of insects other than gypsy moths

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Nabids and geocorids	fields treated with 281 or 562 g a.i./ha (4 or 8 oz a.i./ac)	Soybeans in S.C.	Significantly fewer nabids and geocorids on treated v. control sites. Trapped predators on fields.	Yes	Yes	Turnipseed and others, 1974
Predators: lacewing (*Chrysopa carnea*), ladybird beetle (*Hippodamia convergens*), Wasp parasite *Trichograma pretiosum* of bollworm (*Heliothis*)	**Lab:** 10 mg on 9-cm filter paper (contact); and 5 ppm sugar-water fed to host. **Caged Field:** 280 g a.i./ha (4 oz a.i./ac) **Field:** 33, 66, & 140 g a.i./ha (0.5, 1, & 2 oz a.i./ac)	Cotton	**Lab:** Lab rearing of hosts on diflubenzuron diets and raising parasites on those eggs. And raised lacewings from topically treated eggs and adults. Negative effects on lacewing and ladybird beetle in lab; egg hatch of beetle returned to normal after 30-40 d. **Caged Field:** Caged lacewing suffered increased mortal. eating treated eggs. No effect on parasitic wasp through 2-3 generations; wasp developed in treated eggs & in eggs produced by treated adults, & direct exposure to adults was not toxic. **Field:** No evidence of negative effects on predators/parasites. Immigration from untreated fields could mask negative effects on beetles seen in lab.	Yes, but not in caged study	Yes	Ables and others, 1977

Table 5-10, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Predators were big-eyed bug (*Geocoris punctipes*), *Nabis* spp., ladybird beetle (*Hippodamia convergens*) *Coleomegillamaculata*, *Orius insidiosus*, lacewing (*Chrysopa*) spp.	2 oz. a.i./ac (140 g a.i./ha) in sun oil. Applied 9 times @ 4-7 day intervals.	Cotton in NC	Collected predators and found no difference between population levels of adults and immatures between control and diflubenzuron-treated fields except for big-eyed bug and lacewing. Significant difference in immature:adult ratios between treated and control fields for the bug.	Yes	Yes	Keever and others, 1977
Predators: 2 beetles, a Carabid *Calosoma argentatus* and a geocorid. A nabid, and fungus *Normuraea rileyi*. Host: velvetbean caterpillar *Anticarsia gemmatalis*.	250 g a.i./ha (3.6 oz a.i./ac). Applied 3x by mistblower.	Soybeans in Brazil.	Diflubenzuron had no effect on adult levels of predator *Calosoma*, nor on nabids or geocorids.	No	Yes	Heinrichs and others 1979

Table 5-10, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Host: Pear psylla, *Psylla pyricola*. Spray was to control codling moth (*Laspeyresia pomonella*)	560, 280, & 140 g a.i./ha (8, 4, & 2 oz a.i./ac). 2 & 3 treatments. Handgun and air-carrier sprayer.	Pear orchard in Oregon.	Nearly twice as many *Psylla* predators and parasites per season in the lower rate application. Higher rates resulted in higher populations of the pear psylla.	Yes	Yes	Westigard 1979
Host: *Heliothis* spp. Parasite: *Trichogramma pretiosum*.	**Lab:** diflubenzuron alone at 70 g a.i./ha (1 oz a.i./ac), and oil alone at 4.7 l/ha. **Field:** 70 g a.i./ha (1 oz a.i./ac) in 4.7 l/ha crop oil (savol) + H_2O, applied 6x @ 5 d. intervals	Cotton in Texas	**Lab:** Diflubenzuron alone did not affect parasitism of *Heliothis* by wasp, but the Savol oil affected it severely. **Field:** Treatments reduced parasitism by 44% after spray.	Yes	Yes	House and others, 1980
Phytophagous mites and entomophagous mites	Hand-held sprayer applied diflubenzuron until runoff.	Pear orchard in San Jose, CA	No effect on entomophagous mites. Phytophagous mites controlled as in control fields.	Yes	Yes	Riedl and Hoying, 1980

Table 5-10, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Host: sugar cane rootstalk borer weevil (*Diaprepes abbreviatus*)	350 g a.i./ha (5 oz a.i./ac). Sprayed 8 times.	Citrus grove, Florida	No effect on brood numbers in samples counted 1/month for 7 months. Texas citrus mite significantly more abundant on treated than control. Other mites and predacious fungus no different. 1 year later there were no differences between fields. Checked predators in treated and control fields.	No	Yes	Schroeder and others, 1980
Wasp parasitoid *Atractotomus mali* of wooly apple aphid. European Earwig (*Forficula auricularia*)	**Lab:** 0.01% ai fed to parasite (on wooly apple aphid) wasp *Atractotomus mali*. **Field:** Difluben. applied 3x (rate not given) to orchard.	Apple orchard sprayed to rid of leafrollers and other caterpillars. The Netherlands.	**Lab:** The parasite in lab was not affected. Young produced by dosed females and dosed host eggs were the same as controls. **Field:** Surveys of field predators found only one difference, the abundance of earwigs. Aphids increased. Earwigs are voracious aphid predators and were absent from DFB-sprayed fields. Causal relationship between increase of aphids and decrease in earwigs inferred.	Yes	Yes	Ravensberg 1981

Table 5-10, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Predators of bollworms (*Heliothis*): lacewings (*Chrysopa* spp.), ladybird beetle (*Hippodamia convergens*), *Coleomegilla maculata* big-eyed bug (*Geocoris punctipes*), *Nabis* spp., *Orius insidiosus*.	70 g a.i./ha (1 oz a.i./ac). Applied in paraffinic crop oil (Dimoil) and water.	Cotton in NC	Numbers of predators unaffected by 4 treatments 1 week apart. Sampling took place 1 week after each treatment. Fields 15 ha each. Did not look at parasite numbers. Crop oil could have affected that group.	Yes	Yes	Deakle and Bradley, 1982
Host: Mexican bean beetle (*Epilachna varivestis*). Parasite: wasp (*Pediobius foveolatus*).	**Lab:** 100, 1,000, and 10,000 ppm **Field:** 18, 53, & 88 g a.i./ha (0.25, 0.76, & 1.26 oz a.i./ac)	beans in MD	**Lab:** Topical application to adults did not affect survival or reproduction, nor that of their progeny. Emergence of parasite from larvae treated after parasitism and before was 0 or nearly 0. **Field:** Emergence higher in controls, but not significantly.	Yes	Yes	Zungoli and others, 1983
Wasps are *Macrocentrus ancylivorus* and *Glypta* spp.	1.1 kg a.i./ha (15 oz a.i./ac)	Peach orchard Ontario, Canada	Parasitism rate of wasp on oriental fruit moth (*Grapholitha molesta*) unaffected in sprayed vs. control fields.	Yes	Yes	Broadbent and Pree 1984

Table 5-10, *continued* ...

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Parasitic wasps, Ichneumonidae, Braconidae, Tiphiidae. Predatory wasp, Larrinae.	38 & 83 g a.i./ha Applied in diesel oil (0.54 & 1.19 oz a.i./ac)	Senegal Savannah habitat	Reduced populations of Ichneumonids and Braconids in sprayed plots for at least 3 weeks. Possibly due to effects on host species rather than direct toxicity. Tiphiids unaffected by treatments. Predatory wasp reduced in treated plot, possibly a response to prey reduction (grasshoppers).	No	Yes	Everts, 1990
Parasites >3 mm long. Black-light.	70.75 g a.i./ha (1 oz a.i./ac)	Grant Co., WV Oak, hickory, and pine.	Trap catches of 3 families of Hymenoptera were unaffected, including two parasitic families, Ichneumonidae and Braconidae.	No	Yes	Sample and others, 1993c

1--The R column indicates whether the study design included replicates, and the C indicates controls.

Table 5-11. Effects of diflubenzuron on predators and parasites of gypsy moths

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Parasitic wasp *Apanteles melanoscelus*	57 g a.i./10 gal water with spreader sticker. Applied with backpack sprayer.	Apple orchard in Union, CT	Parasitism rate on treated vs. control trees roughly equal before spray, but lower on treated trees 7 d. after spray (1.81% v. 0.67%). Some adult wasps developed successfully, perhaps those in later stages of development.	Yes	Yes	Granett and Dunbar, 1975
Parasitic wasp *Apanteles melanoscelus*	3.5 g a.i./10 gal. water with spreader sticker. Applied w/ backpack sprayer.	Apple orchard in Brooklyn, CT	1st application of spray decreased parasitism rate. 2nd and 3rd applications did not. Later applications after wasp sensitive period.	Yes	Yes	Granett and others, 1976
Wasp parasite on GM eggs (*Ooencyrtus kuvanae*) on gypsy moth.	67 g a.i./ha (0.96 oz a.i./ac)	Lewisburg, PA	Egg masses in treated plots were parasitized as heavily as egg masses in control plots. Lab data showed no effect on development and emergence from treated eggs or from eggs laid by treated adults.	Yes	Yes	Brown and Respicio 1981
Wasp parasite on GM larvae (*Apanteles melanoscelus*)Parasitic fly in family Tachinidae	30 g a.i./ha, in 4.78 l water (0.43 oz a.i./ac)	southern Quebec, Canada	Wasp mortality 80% in 2 weeks from field spray. Development halted in most cases, failed to spin cocoons upon emergence, etc. 100% mortality in tachinid parasite. Gypsy moths in 2nd, 3rd, and 4th instar. Results in line with Granett and Weseloh (1977) lab tests and with others on this wasp species.	Yes	Yes	Madrid and Stewart, 1981

Table 5-11, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Wasp parasite on GM larvae (*Cotesia melanoscelus*) Pathogen: gypsy moth nuclear polyhedrosis virus (NPV)	28 g a.i./ha (0.4 oz a.i./ac)	Hardwood plots, MD	Numbers of the wasp no different on Control v. treated plots. Incidence of NPV significantly lower in treated plots. Late instar spraying may preserve larvae long enough for parasitoid to complete development. Earlier spraying kills host too quickly, hence parasitoid as well. NPV lower in treated plots because fewer Gypsy moths to transmit virus.	Yes	Yes	Webb and others, 1989

1--The R column indicates whether the study design included replicates, and the C indicates controls.

Table 5-12. **Effect of diflubenzuron on honey bees**

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Honey bee, *Apis mellifera*	350 g a.i./ha (5 oz a.i./ac)	Manitoulin Island, Ontario	5 hives in treated and untreated sites. No effects on egg hatch, brood production, numbers of adults, and honey production.	Yes	Yes	Buckner and others, 1975
Honey bee, *Apis mellifera*	34 & 68 g a.i./ha (0.5 & 0.97 oz a.i./ac)	New Jersey	Hives placed in gypsy moth treatment blocks. No effects from applications on numbers of adults, larvae, or honey production.	Yes	Yes	Matthenius, 1975
Honey bee, *Apis mellifera*	60 ppm in water supplied to hive for 40 days	Arizona	Treated colonies produced significantly less brood and more eggs. They built less comb and used less water compared to control hives. Survival of adults was unaffected	Yes	Yes	Barker and Waller, 1978
Honey bee, *Apis mellifera*	2 & 4 oz a.i./ac	Washington	No effect on adult mortality or brood production.	Yes	Yes	Robinson and Johansen, 1978
Honey bee, *Apis mellifera*	0.5 & 2 oz a.i./ac, w/ crop oil, sprayed 8 times	Cotton field, Sanford, FL. Colonies inside field - received direct spray	No effects noted on adult mortality, rate of larval growth, brood production, or honey or wax production. No residues in wax or honey. Not caged study, so bees could have foraged outside of spray area. Hives were in spray, however.	Yes	Yes	Robinson, 1979

Table 5-12, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Honey bee, *Apis mellifera*	**Expt.1 & 2:** 110, 200, & 400 g a.i./ha (1.6, 2.9, & 5.7 oz a.i./ac) **Expt 3:** 220 g a.i./ha (3.1 oz a.i./ac)	southeastern England, apple trees in full flower. Caged with bees	**Expt. 1 & 2:** No effect from spray on trees on adults or larvae. **Expt. 3:** Direct spray on incoming adults. Neither adults nor larvae harmed by this treatment. No disappearance of unsealed larvae--a sign of diflub. effects in hives. Residues (0.11 & 0.3 μg/g) of diflub. in honey in hives within sprayed plots.	Yes	Yes	Emmett and Archer, 1980
Honey bee, *Apis mellifera*	0.1, 1, & 10 ppm in Sugar-cake for 12 wks. 0.01, 0.1, & 1.0 ppm in sucrose syrup next year for 10 weeks.	Laramie, WY Mountain meadow rangeland	In expt. using 10 ppm diflubenzuron in sugar-cake, significantly fewer sealed brood were produced, and colony size was reduced significantly compared to control and lower dosed colonies. No effects on brood production, colony size or adult bee mortality were seen the following year, when lower doses in a fluid solution was used. Degradation in sucrose solution might have reduced the potential for adverse effects.	Yes	Yes	Stoner and Wilson, 1982

1--The R column indicates whether the study design included replicates, and the C indicates controls.

Table 5-13. Effect of diflubenzuron on other nontarget invertebrates

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
56 taxa of soil mites	140 g a.i./ha (2 oz a.i./ac) at site 1; 280 g a.i./ha (4 oz a.i./ac) at site 2.	Kamloops, B.C.	Mites counted in the top 6 cm of soil. About half of the taxa showed significant decreases in abundance from diflubenzuron applications. Overall population unaffected by spraying; increases in some species compensated for decreases in others. Mites in upper 3 cm of soil more severely affected than mites below. Some predators decreased and some increased (trophic level not predictive of susceptibility). 4 species apparently eliminated from site 2, after a year; other species persisted at low levels a year after spray.	No	Yes	Marshall, 1979
8,000 indiv. soil invertebrates	140 g a.i./ha (2 oz a.i./ac). 4.05 ha in 41 ha woods.	NC, Stern Property. Wooded valley.	Some species of soil mites were adversely affected. Half the number in treated v. untreated samples. Consistent with Marshall, 1979.	No	Yes	Blumberg, 1986
Canopy arthropods	70 g a.i./ha (1 oz a.i./ac)	Sleepy Creek, Morgan Co., WV. Oak-pine and oak-hickory hardwood.	No significant treatment effects noted among herbivorous Hemiptera, Homoptera (which suck plant juices), predacious arthropods, and taxonomic richness of these groups.	Yes	Yes	Martinat and others, 1988
Grasshoppers, their insect predators, pollinators, ground spiders, soil organisms	38 & 83 g a.i./ha (0.54 & 1.18 oz a.i./ac) Applied in diesel oil	Senegal Savannah habitat	Nearly 90% reduction in grasshoppers (nymphs and adults) 7 d. after treatment at higher rate. Low rate had minimal effects on larval grasshoppers. At least one taxon of beetle showed reductions of 50% at highest dose. Possible reduction in trap catches of members of 1 of 3 families (the Gnaphosidae) of ground spiders, at highest dose, 4 weeks after treatment.	No	Yes	Everts, 1990

Table 5-13, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
7+ spp. of savannah inhabiting grasshoppers	variable: 22.5 g a.i./ha to 135 g a.i./ha (0.32 to 1.9 oz a.i./ac), depending on study	Africa	Treatments aimed at grasshopper control resulted in variable results. Lower rates of application appeared to have less effect (up to 40% mortality at 22.5 g a.i./ha). 0% mortality and deformed targets (presumably still consuming vegetation) were reported in two different applications at rates as high as 90 g a.i./ha. Mortality of 60 to 100% reported in 5 of 7 studies at rates >45 g a.i./ha.	Yes and No	?	FAO, 1992
Canopy arthropods	35 g a.i./ha (0.5 oz a.i./ac)	Fernow Exptl. Forest, Parsons, WV	Under tree bands, Carabidae (beetles), Gryllacrididae (grasshoppers), and two families of moths were significantly reduced in total taxa richness and abundance on treated sites. Foliage sampling found reduced abundance and richness in the following groups: Lepidoptera, Symphyta (sawflies, horntails), some herbivorous Coleoptera (beetles), Psocoptera (book lice, wood lice), predatory Thysanoptera (thrips), some Homoptera (leaf hoppers, aphids, cicadas), Diptera (flies), Orthoptera (grasshoppers), and Arachnida (spiders). Some affected by direct toxicity and others (predators) indirectly through prey reduction. Macrolepidopterous larval abundance had not recovered by the end of the study - 27 months post-treatment.	Yes	Yes	Butler, 1995

Table 5-13, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1] R	STUDY DESIGN[1] C	STUDY
30+ spp. of grasshoppers, counted on treated and control fields	About 11 and 22 g a.i./ha (0.75% & 1.0% a.i./kg. @ 1.1 & 2.2 kg/ha.) Treated bran bait.	South Dakota on rangelands	Total populations were reduced 28 days after treatment by 60 and 70% at highest rates of application (0.75 & 1.0% a.i./kg; 2.2 kg/ha). Populations reduced <20% at half that rate. Greater effects early instars.	Yes	Yes	Jech and others, 1993
120 species of spiders (Araneae) and orthopteroid (Orthoptera and Dictyoptera)	70 g a.i./ha (1 oz a.i./ac)	Sleepy Creek, Morgan Co., WV. Oak-pine and oak-hickory hardwood.	Significant effects from treatments noted on spider on 1 of 10 sampling dates, and on orthopteroid abundance on 2 of 10 sampling dates. Trend in expected direction on other dates. No change in diversity of these groups. Effect on spiders could be from loss of prey or direct toxicity. Orthropoids picking up from litter that they ingest.	Yes	Yes	Martinat and others, 1993
Soil microarthro-pods	70 g a.i./ha (1 oz a.i./ac)	Fernow Exptl. Forest, Parsons, WV	Significant decreases in densities of mites and spiders compared with control and pretreatment numbers. Significant decrease in 6/24 common species. Trophic categories (fungivores, herbivores, detritivores, carnivores) not significantly different after treatment. Effect appeared indirect on some groups (direct on others?)	Yes	Yes	Perry and others, 1993

Table 5-13, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
Insects greater than 3 mm long. Black-light trapped.	70.75 g a.i./ha (1 oz a.i./ac)	Grant Co., WV Oak, hickory, and pine.	No differences noted at order level among 9 orders of insects. Species richness of Lepidoptera was lower in sprayed plots. At family level of analysis, abundances of 2 of 9 families of beetles increased in treated plots, as did midges (chironomids) out of 6 families of flies examined. Trap catches of 3 families of Hymenoptera were unaffected, including two parasitic families, Ichneumonidae and Braconidae, and of ants, Formicidae. Largest effect found year after treatment. Many species not sampled might be affected, e.g. diurnal spp.--sawflies (see Martinat and others, 1988)	No	Yes	Sample and others, 1993c

1--The R column indicates whether the study design included replicates, and the C indicates controls.

Table 5-14. Effects of diflubenzuron on birds

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
5 species of forest birds	67 g a.i./ha (0.96 oz a.i./ac)	Robert B. Winter State Park, PA	Singing rates and abundances of singing birds did not change in diflubenzuron sprayed sites.	No	Yes	Bart, 1975
38 forest birds	350 g a.i./ha (5 oz a.i./ac)	Manitoulin Island, Lake Huron, Ontario.	Abundance, territory size: Censuses revealed no differences between abundance on treated and control plot. No territory expansion after treatment.	No	Yes	Buckner and others, 1975
Western forest birds	140-280 g a.i./ha (2-4 oz a.i./ac)	Oregon: Blue Mtns./Wallowa Mtns.	Nest success, abundance. No effects.	No	Yes	Richmond and others, 1979
Great and blue tits (*Parus major* and *P. caeruleus*), European tree sparrow (*Passer montanus*)	Applied to drip-off (300-3,300 g a.i./ha?)	Netherlands	Growth rate of young and nesting success: No effect. Great tit caterpillar use dropped 10-40%. Maximum daily dietary intake is far below LD50. Nesting success unaffected, though diet shifted in one species. Toxicity not an issue, even at highest application rates.	Yes	Yes	de Reede, R.H., 1982
20 spp. birds	140 g a.i./ha (2 oz a.i./ac)	Pennsylvania	No change in abundance or diversity.	No	Yes	Stribling and Smith, 1987
7 spp. birds	70.75 g a.i./ha (1 oz a.i./ac)	Oak/Hickory W. Virginia	Diet: Significantly fewer Lepidoptera in treated area, and foraging areas of red-eyed vireo (*Vireo olivaceus*) greater than 2 times larger there. No significant differences in bird abundance between treated and control plots. Enlarged territories of vireo suggests greater energy required to acquire food. Could jeopardize nesting success.	Yes	Yes	Cooper and others, 1990

Table 5-14, *continued ...*

SPECIES	APPLICATION RATE	SITE DESCRIPTION	RESULTS AND CONCLUSIONS	STUDY DESIGN[1]		STUDY
				R	C	
9 spp. birds, 2 residents and 7 neotropical migrants	70.75 g a.i./ha (1 oz a.i./ac)	West Virginia: Oak/hickory forest	Diet: Significant shift in diet for 5/9 spp. on treated vs. control plots. Fewer Lepidoptera and more beetles, flies, and other insects eaten on treated sites. Most birds switched prey, others ate less food. Migrants more affected than residents.	Yes	Yes	Sample and others, 1993d
9 spp. birds, 2 residents and 7 neotropical migrants	70.75 g a.i./ha (1 oz a.i./ac)	West Virginia: Oak/hickory forest	Fat levels: Significantly less fat in 7/9 spp. on treated vs. control plots, but insect biomass from intestines was the same. All 7 were neotropical migrants. Lowered fat levels result from lowered food resource from pesticide application. No change in insect biomass indicates a shift to less profitable prey. Possibly detrimental effects on longevity and nesting success.	Yes	Yes	Whitmore and others, 1993

1--The R column indicates whether the study design included replicates, and the C indicates controls.

Table 5-15. Toxicity data of *Bacillus thuringiensis var. kurstaki* to aquatic organisms

Species	Exposure/Dose	Effect	Reference
Fish			
fish	Dipel WP® spruce budworm program	no significant adverse effects	Buckner and others, 1974
Aquatic Invertebrates			
midges, mayflies, sponges, planarians, hydras, crayfish, clams, Trichoptera, Plecoptera, Odonata, Coleoptera,	Dipel WP® spruce budworm control program	insignificant knockdown of emerging adult midges and mayflies. No adverse effects to other aquatic species	Buckner and others, 1974
Chironomidae Trichoptera, Megaloptera, mayflies, Plecoptera, black flies	430 IU/ml (Thuricide 32LV®)	only black fly *Simulium vittatum* clearly affected, maybe black fly *Prosimulium fascum/mixtum*	Eidt, 1985
caddisflies, mayflies, stoneflies (12 taxa)	10 x label application rate for Dipel 64AF®	only the stonefly *Leuctra tenuis* was reduced at 4 days after treatment	Kreutzweiser and others, 1993
Ephemeroptera, Plecoptera, Trichoptera, Diptera	Foray 48B applied in one application: 96 oz./acre and a dosage of 36 BIU/acre	*Epeorus* significantly less abundant on sprayed than untreated streams; no other spray effects seen.	Oldham and others, 1994

Table 5-16. Toxicity data of diflubenzuron to aquatic organisms

Species	Exposure/Dose	Effect	Reference
Fish			
bluegill sunfish	Dimilin 25-WP® -5 ppb in lake	growth unaffected	Apperson and others, 1978
bullhead catfish, sunfish *Lepomis gibbosus*	Dimilin® - 125 ppm	no toxic effects	Buckner and others, 1975
fish	40 g AI/ha application	no adverse effects	Everts, 1990
mosquitofish *Gambusia affinis*	five monthly applications of Dimilin 25-WP® at 0.05 lb AI/acre	no adverse effects on populations	Takahashi and Miura, 1975
Aquatic Invertebrates			
Caenis sp., ceratopogonid, chironomidae, hydracarina, *Cyclops* spp., *Diaptomus* spp., *Alona* sp., *Bosmina* sp., *Ceriodaphnia* sp., *Diaphanosoma* sp., *Ilyocryptus* sp., *Macrothrix* sp., *Sida* sp., *Simocephalus* sp., Ostracoda, Oligochaeta, Nematoda, Rotifera	exposed pond in middle of citrus grove treated with Dimilin 25-WP® at 0.56 kg AI/ha	no adverse effects evident to any species present	Ali, 1987; Ali and others, 1988
Ephemeroptera, Chironomidae, Trichoptera, Plecoptera	TH 6040® at 0.06 lb/acre	no adverse effects	Bocsor and Moore, 1975
Ephemeroptera, Odonata, Diptera, Trichoptera, Coleoptera, Hemiptera, Decapoda, Branchiopoda, Copepoda	35.2 g AI/ha application	only branchiopoda and copepoda adversely affected	Everts, 1990
Chironomidae, Odonata, Ephemeroptera (*Baetis* sp.), Trichoptera (*Hydropsyche* sp.), black fly larvae (*Simulium* sp.)	Dimilin 25-WP® at 0.1 ppm	moderate toxicity to ephemeroptera, trichoptera, and black fly larvae, other species unaffected.	Mohsen and Mulla, 1982

Table 5-16, *continued ...*

Species	Exposure/Dose	Effect	Reference
Tricorixa louisianae nymphs, *Buenoa* spp. nymphs, *Coenagrionid* spp. naiads, coenagrionid spp. naiads, *Berosus infuscatus* adults, and *Hyalella azteca*	28 mg AI/ha	populations of listed taxa reduced considerably, all other taxa increased or unaffected	Farlow and others, 1978
Ephemeroptera, Plecoptera	Dimilin® at 0.06 lb AI/acre	no adverse effects	Jones and Kochenderfer, 1987
Trichoptera, Plecoptera	Dimilin W-25® at 0.0625-0.25 lb AI/acre	slight population reductions not attributed to applications	Rabeni and Gibbs, 1975
chironomid midges *Chironomus decorus* and *Chironomus paripes*, *Cyclops* spp., collembola, *Chaoborus* sp., *Baetis* sp.	Dimilin 25-WP® - lab exposure.	90% mortality of midges at 4-22 ppb.	Ali and Lord, 1980; Ali, 1980
	Dimilin 25-WP® field treatment at 28 g AI/ha.	3 week control of midges, reduced and suppressed populations of other species mentioned, recovery of nontarget species within a few weeks after treatment	
Chaoborus astictopus, zooplankton, cladocerans, rotifers, algae	Dimilin 25-WP® - applications to farm ponds to control gnats	44% reduction of gnats at 2.5 ppb, 88% at 5 ppb, and 98% at 10 ppb. 4-6 week emergence delay. zooplankton and cladocerans suppressed. rotifers and algae unaffected.	Apperson and others, 1978
Simulium venustum	Dimilin® at application rate of 0.7 ppm	drift of black fly larvae increased in 15 minutes	McKague and Pridmore, 1979
Culex quinquefascia-tus	Dimilin 25-WP® at 112 g/ha (0.1 lb/acre)	satisfactory control for 1-2 weeks in waste lagoon	Axtell and others, 1980
Culex quinquefascia-tus, *C. salinarius*, *C. restuans*	Dimilin 25-WP® at 0.125 ppm in waste lagoon	larval control for 14 to 35 days	Barker and Booram, 1979

Table 5-16, *continued ...*

Species	Exposure/Dose	Effect	Reference
water scavenger beetle adults *Laccophilus* spp., *Thermonectus basillaris*, *Tropisternus lateralis*	TH 6040® concentrations as high as 250 µg/L	no mortality	Miura and Takahashi, 1974
Daphnia pulex, *D. galeata*, *Hyalella azteca*, *Bosmina longirostris*, oligochaete worms, *Diaptomus* spp., *Cyprinotus* sp.	granular formulation at 0.11 and 0.22 kg AI/ha	cladoceran populations reduced at both rates, oligochaete worms and *Bosmina* unaffected, *Diaptomus* and *Cyprinotus* severely affected at higher rate	Ali and Mulla, 1978
amphipod, aquatic beetle larvae	Dimilin® at 350 g/ha	populations apparently reduced	Buckner and others, 1975
aquatic and terrestrial invertebrate faunas	Dimilin 25-WP® at 140 g AI/ha	no adverse effects	Blumberg, 1986

Section 6
Endpoint Selection

Endpoints are those components of the ecosystem that, when altered, cause changes of ecological or societal importance. Endpoints are ecological components that may be adversely affected by a stressor (treatment) used in gypsy moth management or by the gypsy moth itself. Relevant ecological endpoints were selected from issues identified by the EIS team through the scoping process. Available data were insufficient for the analysis of some of the endpoints and these data gaps were identified.

The five endpoints selected for analysis are: (1) change in forest health, (2) change in numbers of nontarget species or the size of their populations, (3) change in water quality, (4) change in microclimate, and (5) change in soil fertility, productivity, or stability. By their nature, endpoints are broadly defined and may not be directly measurable. Ecological indicators, or measurement endpoints, are measurable components of the ecosystem relevant to the state of an endpoint. A set of ecological indicators, for which sufficient data exist to complete an analysis, were selected for each endpoint. Data were not available for every combination of stressor and endpoint, however. For example, the effect of diflubenzuron on water temperature has not been studied. The collective response of these indicators to each of the stressors was used to assess the response of each endpoint.

Ecological endpoints utilize indicators from all levels of biological organization from the individual to the ecosystem level, including populations and communities. Special consideration was given to selecting indicators at the population level because effects at this level can have important ecological consequences, while the loss of a single individual may not be ecologically meaningful. Each endpoint and its associated indicators is discussed below and can be found in Table 6-1.

6.1. Change in Forest Health

Examination of changes in forest health provides a measure of the efficacy of the various treatment methods versus no action. The defoliation resulting from untreated gypsy moth infestations will have visible and aesthetic effects on forests. Healthy forests perform important functions in the ecosystem by producing oxygen while consuming carbon dioxide (carbon cycling), moderating the climate, and controlling soil erosion. Indicators of the health of the forest include changes in tree mortality rates, incidence of fungal or bacterial disease, degree of insect damage, and tree growth rates. Changes in species composition in the understory and in the canopy

trees are also indicative of forest health. Of special interest, particularly because of the effect of gypsy moth damage, is the proportion of oaks or other trees in the forest that are highly susceptible to gypsy moth damage. The productivity of the forest, or the increase of primary producer mass per area over time, is an indicator of forest health at the ecosystem level.

6.2. Change in Numbers of Nontarget Species or Their Populations

Species perform vital functions within the ecosystem, such as pollination, decomposition, or nutrient cycling. Wildlife species have societal importance to bird or nature watchers, hunters, and recreationists. Game fish species have societal importance to anglers and also serve as an indicator of the quality of aquatic resources. Native Lepidopteran species (moths and butterflies) are of interest due to the potential effects from the gypsy moth, diflubenzuron, and *B.t.k.* Lepidopteran species are of aesthetic and scientific interest to society and some may have ecological significance as pollinators. Changes in species richness of mammals, birds, reptiles, amphibians, and invertebrates may reflect changes in community structure and relative abundance of these organisms in the ecosystem. A general interest in the worldwide decrease in biodiversity makes local sensitivity to these issues imperative. Of particular interest are changes in amphibian populations due to a dramatic worldwide decline of these organisms. Changes in populations of crustaceans and aquatic insects are important indicators of water quality and could affect populations of game fish as these organisms make up the bulk of game fish diets. Changes in mollusk populations are especially important due to nationwide decline of these species, as well as the importance of the mollusk in maintaining water quality. Changes in bat populations are important due to the function of these organisms in reducing flying insect populations and because of decline in populations of several species.

6.3. Change in Water Quality

The presence or absence of water as well as its quality has important ecological consequences for plant and animal community distributions. Many sport fish have narrow requirements for water quality, and only live in cold, clear waters. High quality water resources are also important to society because they provide drinking water sources and opportunities for recreational use.

Temperature affects the amount of dissolved oxygen in water, the types of organisms living in the water, and temperature-dependent rate functions such as photosynthesis, nutrient uptake or decomposition. The dissolved oxygen concentration of water determines the types of organisms that will occur there. Nutrient concentration determines the types of plants, bacteria and fungi found in water resources. The flow rate of streams affects the dissolved oxygen concentration and determines the

occurrence of types of benthic invertebrates (those occurring at the bottom of water bodies) and planktonic organisms (those that float). The sediment load affects the amount of light penetrating the water for photosynthetic organisms and also affects the respiration of invertebrates and fish. High algal density is often an indicator of poor water quality with low dissolved oxygen concentration and high nutrient concentrations.

6.4. Change in Microclimate

Microclimate refers to climatic characteristics over a relatively small area within a similar type of habitat. Microclimate can refer to the climatic features observed within a forested habitat, or within specific portions of that habitat, for example, in the litter layer of the soil. Microclimate (for example, temperature and relative humidity) is affected by the amount of sunlight and heat reaching the habitat; increases in these factors are expected with defoliation. Changes in microclimate can have profound effects on terrestrial and aquatic systems.

6.5. Change in Soil Fertility, Productivity, or Stability

The nutrient concentration of the soil, absorbance capacity, and chemical characteristics of the soil influence the type of plant community that can be supported in an area. The amount of organic material in a soil also affects its ability to hold water. Decomposition of the organic material returns nutrients to soil. Litter invertebrates process the litter by shredding it into small pieces, facilitating further breakdown by bacterial or fungal activity. Soil pH (acidity or alkalinity) affects the type of microbial organisms and invertebrates found in the soil. Earthworms reduce soil compaction and increase the water retention capacity of the soil by increasing pore space. Erosion removes soil and nutrients from the ecosystem and can change the types of organisms present in the below-ground ecosystem by altering the physical and chemical composition of the soil.

6.6. Summary

The effects of the stressors on the ecological indicators were evaluated collectively to form an overall picture of their effects on the endpoints. Ecological indicators were assessed quantitatively whenever possible using data from monitoring studies and mathematical models. When quantitative analysis was not possible, the indicators were assessed qualitatively.

Table 6-1. Ecological Endpoints and their Indicators

Endpoints	Ecological Indicator
Change in forest health	• Forest productivity, tree growth rates, mast production • Successional state, stand age, species composition of tree and understory species • Susceptibility to fire • Incidence of disease, tree mortality rates, degree of insect damage
Change in nontarget species number or population densities	• Species richness of mammals, birds, reptiles, amphibians, fish, invertebrates • Population densities of groups of special concern: spring-feeding native Lepidopterans, summer-emerging native Lepidopterans, insect predators and parasites, pollinators, amphibians, mollusks • Population densities of organisms eaten by game fish: crustaceans, aquatic insects, small fish
Change in water quality	• Water temperature, dissolved oxygen concentration • Nutrient concentration, algal densities • Flow rate, water yield, and sediment load • Detrital decomposition rate
Change in microclimate	• Percent defoliation, amount of light penetrating canopy,soil and litter temperature, relative humidity below the tree canopy
Change in soil fertility, productivity, or stability	• Population sizes of organisms that alter soil composition or texture: litter invertebrates, earthworms, bacteria and fungi • Litter production, concentration of organic material, decomposition rate • Soil pH • Erosion rate

Section 7
The Fate and Transport of Insecticides in the Environment

The information presented about fate and transport of insecticides represents an extensive review of the literature for each of the microbial and chemical insecticides used in various treatment methods in the management of gypsy moths: diflubenzuron, *Bacillus thuringiensis* var. *kurstaki* (*B.t.k.*), nucleopolyhedrosis virus for gypsy moths (NPV), Disparlure, and dichlorvos. Information on the toxicology and mode of action of these insecticides and the attractant (Disparlure) can be found in Sections 3 and 5.

The information provided by the fate and transport section is used to determine where in the environment the pesticide is likely to be found and to estimate the amount of pesticide likely to accumulate. The estimated environmental concentrations are fundamental for estimating exposures and, therefore risks, to nontarget organisms and other environmental components (see Estimated Environmental Concentrations, this section).

7.1. Diflubenzuron

This discussion of diflubenzuron describes where it goes and why. Important chemical characteristics of diflubenzuron are described, followed by a review of its fate in the forest canopy, on ground, and in water. Next comes a description of the fate and transport of the major breakdown products (metabolites) of diflubenzuron. At the end is a summary of the fate and transport of diflubenzuron in undeveloped and developed forests throughout the United States.

Several chemical characteristics of diflubenzuron influence its behavior in the environment. Solubility in water is relatively low (1.0 mg/L @ 25 °C). The partition coefficient (K_{oc}=10,000) is very high (Dowd, and others 1992). These factors indicate that diflubenzuron, by itself, will not be readily transported in water and it will readily adsorb to soil particles, sediments, and organic material. Its low solubility and high adsorption characteristics make diflubenzuron relatively immobile in the environment and unlikely to enter ground water.

In addition, diflubenzuron is relatively photo-stable in normal light regimes (Schaefer and Dupras, 1976; Bull and Ivie, 1978). Some studies suggest that the chemical might be protected from photo-degradation when adsorbed to particulate matter suspended in the water column (Sundaram and others, 1991). In contrast, one study

found several degradation products formed when diflubenzuron (in methanol and aqueous dioxane) was subjected to light in the range of 254 nm (Metcalf and others, 1975). Similar results were obtained when diflubenzuron was subjected to light (peak energy at 300 nm) in a methanol solution (Ruzo and others, 1974). To what extent these solvents make up the inert ingredients of the product in which diflubenzuron is sold is unknown. However, the product soon dries after it lands. In addition, wavelengths shorter than 300 nm account for less than 2 percent of the spectrum of sunlight reaching the surface of the earth. Thus these studies are of limited use in defining the role of natural light in the degradation of diflubenzuron.

7.1.1. Fate in the Canopy

Diflubenzuron falling on leaves of the canopy or subcanopy is immediately subject to various climatic factors that affect its fate and transport in the environment. Rainfall and wind reduce diflubenzuron residues by 20 percent to 80 percent within two to three weeks of treatment; much of what remains adheres to the leaf surface until leaf-fall (Bull and Ivie, 1978; Nigg and others, 1986; Martinat and others, 1987; Wimmer and others, 1993). Martinat and others (1987) reported a sharp decline in leaf residues (roughly 70 percent reduction) after 3.45 cm rainfall within 10 days after application to an oak-hickory forest. Twenty-one days later the same amount remained despite an additional 3.51 cm rainfall. They concluded there was no consistent relationship between rainfall and the rate at which diflubenzuron disappeared from foliage. Their data are consistent with other studies that show a similar pattern of initial decline in diflubenzuron residues on leaf surfaces. Wimmer and others (1993) found, after a sharp decline initially, residues on leaves of several species of deciduous trees declined very slowly for the remainder of the summer (141 days) until leaf-drop in the fall. In another study all but about 20 percent of diflubenzuron residues were washed from cotton leaves after 7.6 cm (3 inches) of rain three weeks after application (Bull and Ivie, 1978). Roughly 50 percent of diflubenzuron residues on citrus leaves remained after two weeks in a four-week study (Nigg and others, 1986).

Loss of diflubenzuron from leaves due solely to wind effects has not been specifically tested. However, in two studies roughly 50 percent of the diflubenzuron residues declined on citrus leaves and hardwood leaves in the first 3 weeks after spray despite little rainfall (Nigg and others, 1986; Wimmer and others, 1993). Residue loss in the absence of rain suggests that wind is the likely cause (Eisler, 1992).

In coniferous forests, diflubenzuron and its metabolites declined on pine needles in 61 days to nondetectable levels in two of four samples and to 10 percent and 25 percent of original concentrations in the other two samples (Mutanen and others, 1988). Traces of either diflubenzuron or its various metabolites (residues were lumped in this analysis) persisted on needles for at least 319 days (Mutanen and others, 1988).

Diflubenzuron can be absorbed to the cuticle of plants, but does not move from the cuticle easily. Diflubenzuron is not readily translocated, or metabolized by plants. In three separate studies diflubenzuron applied to cotton in greenhouses and in the field resulted in less than 1 percent (Mansager and others, 1979), less than 5 percent (Verloop and Ferrell, 1977), and less than 7 percent (Bull and Ivie, 1978) of the chemical absorbing and translocating in the plant tissue in four weeks or more.

7.1.2. Fate on the Ground

In this risk assessment it is assumed that diflubenzuron weathering from leaf surfaces or filtering through the canopy and subcanopy will fall to the forest floor and be deposited on the litter or soil. This assumption is patterned after the results of Wimmer and others (1994; Wimmer, unpublished data) who measured diflubenzuron residues in the canopy and litter using a new, more sensitive method of residue analysis (Wimmer and others, 1991). Once diflubenzuron reaches the litter, it either filters to the soil, is ingested by terrestrial organisms, degraded by microorganisms in the litter, or carried (adhered to organic matter) to streams or other water bodies by runoff. In soils diflubenzuron is degraded by microbes or ingested by soil organisms. Diflubenzuron may also leave the terrestrial environment when soil particles are washed into aquatic systems. During leaf-drop in the fall, additional residues from the fallen leaves will add to those already present in the litter (Wimmer and others, 1993).

Studies show the degradation of diflubenzuron in some soils proceeds relatively rapidly in spring and summer. For example, low to non-detectable (less than 0.0038 ppm) residues were found in soil two weeks after four treatments (at 45 g a.i./ha) on California pasturelands in spring or summer (Schaefer and Dupras, 1977); in the laboratory, a half-life in sandy loam (incubated at 24 °C) was about 50 hours (Walstra and Joustra, 1990). The interrelated factors responsible for the degradation of diflubenzuron in soil and litter, that is, organic matter, microbes, temperature, and pH are discussed below under separate headings.

Organic Matter: Diflubenzuron has been shown to bind readily with organic matter in soils as well as with sediments. In two types of forest soils, 95 percent of the chemical remained in the top 2.5 cm (1 inch) following a simulated 50 cm (19.5 inches) rainfall (Sundaram and Nott, 1989). In soils of cotton fields diflubenzuron was measurable down to 10 cm (4 inches) (Bull and Ivie, 1978). These results reflect not only the affinity of diflubenzuron for organic matter in soil but also its low water solubility, and hence, low potential for leaching.

Once attached to organic matter diflubenzuron can remain biologically active for a long time, depending on temperature and the presence of microbes. Diflubenzuron applied to stored wheat (presumably a microbe-deficient environment) persisted

unaltered for six months (Aly and Dauterman, 1992). Bound residues were extracted from the wheat stored for six months and were added to media in which housefly larvae were kept. The larvae pupated, but adults failed to emerge; analysis of the pupae showed that they contained bound residues. In another experiment larval stoneflies (and other shredders) grew more slowly and suffered higher mortality after eating organic material treated with diflubenzuron than shredders eating diets free of diflubenzuron (Swift and others, 1988). Thus, diflubenzuron not only persists in detritus for long periods of time at low temperatures, but also remains biologically active and available to decomposers in both aquatic and terrestrial environments.

Not surprisingly, diflubenzuron remains available and potent to leaf eaters and shredders after adhering to organic matter. This is how gypsy moths encounter the chemical. One study showed that gypsy moths suffered 81 to 100 percent mortality within 10 to 13 days after exposure to oak seedlings treated with diflubenzuron (17.5 to 70 g/ha) and then subjected to 1 to 5 inches (2.54 to 12.7 cm) simulated rainfall (Uniroyal, 1989).

In an oak-hickory woodland in West Virginia, residues of diflubenzuron declined in leaf litter throughout the summer, from more than 1000 ppb (nanograms/gram of dry litter) after application of 15 to 200 ppb just before leaf-fall (1994; Wimmer, unpublished data). Residue levels rose to near 1000 ppb after leaf-fall with the addition of residues that remained on the recently dropped foliage. Residues remained near 1000 ppb over winter and dropped to 100 to 400 ppb by the end of the following summer, when uncontaminated leaves from the canopy diluted what remained in the litter (1994; Wimmer, unpublished data).

Microbial Breakdown: Studies conducted on sterile and natural soils have identified microbial activity as being a dominant factor regulating the fate of diflubenzuron in the soil. In one study, 98 percent of diflubenzuron applied to natural sandy loam degraded in four weeks, in contrast to only 6 percent degradation in sterile soils (Verloop and Ferrell, 1977). Diflubenzuron readily degraded in various agricultural soils and water-saturated soils (pH 5.1 to 7.4, 24 °C), with 50 percent being metabolized in two days or less (Nimmo and others, 1984). Small differences were observed in metabolic rates between soil types, and aerobic and anaerobic conditions. Sterile soil did not result in significant degradation over four weeks. The half-life of diflubenzuron under sprayed citrus trees is reported to be about 19 days (Nigg and others, 1986). During this time the soil presumably was receiving input from the leaves above, thus artificially increasing these half-life estimates. Several laboratory studies (conducted by scientists at Phillips-Duphar, the manufacturer of insecticides containing diflubenzuron as cited in Wilcox and Coffey, 1978) yielded half-lives of 0.5 to 1.0 week in natural soils, and negligible degradation in sterile soils in one year. Pintar and others (1975) identified over 111 soil bacteria which could use diflubenzuron as a sole carbon source.

Diflubenzuron was shown to be very stable over four weeks when applied in acetone to air-dried Drummer soil (Metcalf and others, 1975). These results are expected if Drummer soil is sterile, and if the acetone creates larger diflubenzuron particle sizes, thus reducing the efficiency of microbial degradation (Verloop and Ferrell, 1977; Maas and others, 1981; Nimmo and others, 1984). The Metcalf study also reported essentially no breakdown by *Pseudomonas putida* after incubation for six hours at 30 °C. This soil microorganism is effective in the degradation of a variety of organic compounds, but not diflubenzuron according to the conditions of this study.

The factors responsible for the 10-fold decline in litter residues (noted above) found in the litter of a forest in West Virginia are not yet clear (1994; Wimmer, personal communication). Degradation by microbes is the likely cause of the decline.

Temperature and pH: Since microbes are primarily responsible for the degradation of diflubenzuron, degradation rate is expected to be influenced by temperature. In one study, degradation of diflubenzuron residues in agricultural soils slowed during winter months, but accelerated during warmer spring and summer months (Bull and Ivie, 1978). Nigg and others (1986) reported the half-life to be significantly greater in cool, dry soil than hot, wet soil. And in another study, the half-life was two days at 14 °C and less than two days at 24 °C in dry sandy loam (Nimmo and others, 1984).

The role of soil pH in the degradation of diflubenzuron has not been studied in detail. As cited above, degradation of diflubenzuron in agricultural soils, pH 5.1 to 7.4, incubated at 24 °C, resulted in half-lives of about two days (Nimmo and others, 1984). Studies have also shown that diflubenzuron binds tightly to organic matter and rarely leaches deeper than the top 2.5 cm (1 inch) of soil (Sundaram and Nott, 1989). This layer of soil is generally organic rather than mineral in character, and therefore predominantly acidic (Froth and Ellis, 1988). The activity of detritus decomposers, such as bacteria and fungi, is hampered in low pH waters (Haines, 1981). Studies in water have shown that degradation is negligible at pH 4 at any temperature, probably as a result of the reduced microbial activity (Cunningham, 1986). The same response would be expected of decomposers in soils of very low pH.

7.1.3. Fate in Water

Diflubenzuron enters bodies of water by direct spray, drift, or by runoff carrying soil or litter to which diflubenzuron has adhered. Leaves with diflubenzuron residues fall or wash into streams and rivers in autumn, creating an additional pulse of pesticide that may persist until the following spring. Applications of diflubenzuron are not expected to travel to ground water because it rarely leaches deeper than the first inch or so of soil (see Fate on the Ground--Organic Matter, above).

Experiments conducted in ponds and lakes demonstrate that diflubenzuron breaks down relatively rapidly in water. The factors controlling its fate and transport in water are the same as in soil, that is, organic matter, microbes, temperature, and pH. Aeration, or the presence of oxygen, also apparently affects the rate of breakdown in water. In dechlorinated tap water, incubated at 20 °C, pH 7.2, the half-life of diflubenzuron (original concentration = 0.4 mg/l) was 0.5 days in aerated water and 1.0 days in water without aeration (Anton and others, 1993). The effect of oxygen on the formation of metabolites is discussed under Metabolites below. The rate of degradation increases with temperature and pH. Data from Schaefer and Dupras (1976) suggest a half-life of about two days for diflubenzuron in eutrophic ponds. The half-life in cold, acidic waters (mountain streams in the Appalachian mountains) is undoubtably longer.

Organic Matter: Diflubenzuron in water rapidly binds with organic material in sediments and is removed from the water column (Carringer and others, 1975). It remains as the parent compound in litter or other organic sediments in the water system until further breakdown by microbial or abiotic chemical means, just as in soils.

Schaefer and Dupras (1976) found diflubenzuron applied to ponds (pH 8 to 10, temperature not given) was not detectable within three days. Laboratory studies suggested a combination of adsorption to organic material and hydrolysis into its metabolites were principally responsible for its rapid disappearance from the water column.

Apperson and others (1978) found rapid dispersal and disappearance of diflubenzuron from the water when applied to ponds and a lake, varying from 80 percent disappearance to 98 percent after 14 days. No residues of diflubenzuron were found in lake sediments, suggesting that breakdown rather than adsorption to sediments was ultimately responsible for its disappearance. Cunningham and Myers (1986) found a ten-fold decline in diflubenzuron in the water column within four days after application to estuarine habitats, and hypothesized that this was due to diflubenzuron dispersing and adsorbing to sediments. The rapid decline in water column residues was followed by a leveling, with residues still measurable 15 days after application. Unlike studies of freshwater sediments, residues in estuarine sediments did not decline overall in a 15-day time period.

Sundaram and others (1991) did not detect diflubenzuron (less than 0.05 μg/L) in pond water 20 days after application (of 14 μg/L), and attributed this to adherence to particulate matter. However, diflubenzuron residues in pond sediments declined to the level of detection within three days, suggesting microbial breakdown was also a factor. Residues disappeared faster in streams than in ponds in this study. These results are similar to those of the field studies reported above (Schaefer and Dupras, 1976; Apperson, 1978; Cunningham and Myers, 1986).

Harrahy and others (1993) found diflubenzuron residues persisted on leaves of hardwoods soaked in streams in December, but degraded or were metabolized during the summer months (July and August). Warmer temperatures (see Temperature, below) presumably increased the activity of organisms consuming and metabolizing this chemical. Swift and others (1988) also found diflubenzuron very persistent on stream-soaked leaves in their winter-time study of the effects of this chemical on stream invertebrates.

Microbial Breakdown: As in drier soils, microbial degradation contributes to the degradation of diflubenzuron in aquatic sediments. The extent to which microbes contribute to the degradation of diflubenzuron in the water column is unclear (discussed under Temperature and pH, below). Chapman and others (1985) found rapid degradation of diflubenzuron in natural sediments compared to sterile sediments (both incubated at 28 °C), thus implicating soil microbes (fungi and bacteria in this experiment) in the degradation of diflubenzuron. As well, Nimmo and others (1984, 1986) found that anaerobic, water-saturated soils resulted in slower degradation of diflubenzuron than drier, aerobic soils, but was much faster than in sterilized hydrosoil. The soils tested were incubated at 24 °C and ranged in pH from 5.1 to 7.5. Booth and Ferrell (1977) reported that diflubenzuron was quickly degraded by algae, and was ingested but not degraded by *Pseudomonas* spp. in aquatic systems.

Schaefer and Dupras (1976) claimed that microorganisms had little or no effect on diflubenzuron degradation in water. However, they only tested degradation over a 24 hour period. This presumably is not long enough for the growth and development of diflubenzuron-eating microbes. Fischer and Hall (1992) cite two studies on this matter that conflict in their results: Pritchard and others (1979) found no significant differences in diflubenzuron half-life in estuarine and sterile water; but Schimmel and others (1983) found rapid breakdown (four-day half-life) of diflubenzuron in the same environments. The difference is probably due to the solvent added with the diflubenzuron, which likely increased the size of diflubenzuron particles (slowing microbial degradation) in one study but not in the other (Fischer and Hall, 1992).

Temperature and pH: The following studies find that the degradation of diflubenzuron in tap, deionized, and distilled water (presumably water that is relatively free of microorganisms) increases with increasing pH and temperature. In one study, the rate of degradation in tap water at low temperatures was nearly independent of pH (at 10 °C half-lives were 29 days at pH 7.7 and 32 days at pH 10). The half-lives at 24 °C were similar, but at higher temperatures differences in pH have greater impact on the rate of breakdown of diflubenzuron (at 38 °C half-lives were eight days at pH 7.7 and two days at pH 10) (Schaefer and Dupras, 1976). In another study conducted at 36 °C the half-life of diflubenzuron in deionized distilled water was two to six days at pH 10, seven to eight days at pH 6, and negligible at pH 4 (Ivie and others, 1980). The difference in the rate of degradation between high and low pH water at low temperatures was not pronounced in this

study, confirming the results of Schaefer and Dupras (1976). Thus, the rate of degradation is faster at higher water temperatures and in neutral and alkaline waters. Little or no degradation in water occurs below pH 4 in laboratory studies.

Not only are the effects of temperature and pH important to the chemical degradation of diflubenzuron, but also to the organisms that metabolize diflubenzuron in aquatic systems. For example, the rate of loss of diflubenzuron from leaf packs in streams decreased in December compared to July and August (Harrahy and others, 1993); reduced microbial activity in lower water temperatures was suggested as a possible cause. The activity of decomposers of detritus, such as bacteria and fungi, apparently is hampered in low pH waters (Haines, 1981). Acidification from pollution in the northeastern United States has reduced the pH of some lakes to between 4.5 and 5.0, with adverse effects on fish populations in particular and the aquatic ecosystem in general (Maybeck and others, 1990). The occurrence of water bodies with pH less than 4.5 is apparently rare. Thus, the rate of degradation of diflubenzuron would be expected to decline under low pH conditions due to effects on biological as well as chemical processes.

The pH of most rivers not affected by pollution is between 6 to 9 (Hem, 1985). Streams in the Fernow Experimental Forest, West Virginia, typical of the Appalachian mountains, have pH values of 5.9 or less and temperatures that range from a low of 4 °C in January to a high of 16 °C in August (Adams and others, 1994). It is unclear whether or not microbial activity is hampered by the acidity of these streams.

The rate of degradation of diflubenzuron in the water column of natural aquatic habitats might be more influenced by temperature and pH than by microorganisms. The above laboratory studies suggest that diflubenzuron degrades relatively rapidly in aerated waters of pH 7 and 20 °C or greater (Schaefer and Dupras, 1976; Ivie and others, 1980; Anton and others, 1993), with half-lives within the range found in ponds and lakes (Schaefer and Dupras, 1976; Apperson and others, 1978; Sundaram and others, 1991). The role of microbes in the degradation of diflubenzuron was clearly demonstrated in sediments, however (Nimmo and others, 1984, 1986; Chapman and others, 1985). Nearly all natural streams, lakes, or ponds have sediments of some kind. Thus, even though a definitive relationship between pH levels and diflubenzuron degradation in laboratory water can be demonstrated, the role of pH in environmental fate of diflubenzuron in most natural waters is probably minor in comparison to the sequestering effect of organic matter and sediment which provide the substrate for microbial degradation (Cunningham, 1986). This might not be true in cold, acidic mountain streams, where low pH and temperature could hinder microbial activity.

7.1.4. Metabolites

The initial products resulting from microbial degradation of diflubenzuron are 4-chlorophenyl urea (CPU) and 2,6-difluorobenzoic acid (DFBA). CPU further degrades into 4-chloroaniline (4-CA) and other products on its path to total breakdown. DFBA degrades rather quickly with carbon dioxide as the principle product. In laboratory studies, a small percentage of diflubenzuron degraded directly into 4-CA and 2,6-difluorobenzamide in agricultural sandy loam soil (2 percent, Verloop and Ferrell, 1977; less than 1 percent, Walstra and Joustra, 1990).

7.1.5. Fate of Metabolites in the Canopy and in Soils

A study of persistence in coniferous forests (see Fate in Canopy, above) found residues of diflubenzuron and its metabolites (CPU and 4-CA) on needles in the canopy for up to 319 days (Mutanen and others, 1988). This study demonstrates that diflubenzuron is undergoing some degradation into its various breakdown products while in the canopy, although the extent was not elucidated. Also in this study, no residues of diflubenzuron or its metabolites were found in the humus layer in this forest, but residues in the litter were found up to 319 days after application. These data suggest that the metabolites of diflubenzuron, as with the parent compound, bind to organic material and do not leach down through the soil. The relative contribution to the litter residues by diflubenzuron, CPU, and 4-CA was not resolved, although after 319 days it seems unlikely that diflubenzuron contributed significantly. Apparently, 4-CA is sensitive to ultraviolet light, degrading rapidly into other products in its presence (Ruppert and others, 1993). This factor could influence its persistence were it to occur on foliage, especially in the upper canopy.

In a laboratory study of persistence in sandy loam (pH 5.6, 1.8 percent organic content, 10.5 percent water content, incubated at 24 °C), diflubenzuron was applied at 976 g a.i./ha (about 40 times higher than the 23 g a.i./ha used in gypsy moth programs) and the half-life was found to be 50 hours (Walstra and Joustra, 1990). 4-CA accounted for less than 1 percent (0.4 to 1.5 percent) of the extractable residues. Unidentified soil-bound residues accumulated during the study, and after 21 days accounted for nearly 40 percent of the degradation of diflubenzuron. Degradation of DFBA was found to be "very rapid."

The laboratory studies by Nimmo and others (1986, 1990) are perhaps the most complete examination of the fate of the metabolites of diflubenzuron in soils. In dry sandy clay, CPU had a half-life of four weeks when incubated at 24 °C, and 10 weeks at 14 °C (Nimmo and others, 1986). Hydrosoils (nearly anaerobic) degraded CPU more slowly, with a half-life of 16 weeks at 24 °C. In sterile sandy clay incubated at 24 °C, 85 percent of originally applied doses of CPU were recovered after 39 weeks. When 4-CA was applied directly to soils, 45 percent was bound

within 30 minutes, and 80 percent was bound in 4 weeks. This research demonstrated that 4-CA is the principle breakdown product of CPU in dry sandy clay (accounting for slightly more than 50 percent of CPU, thus of diflubenzuron as well), and that 4-CA binds almost immediately to the soil making its identification difficult. Some studies, such as that by Walstra and Joustra (1990), find large amounts of unextractable soil residues from the degradation of diflubenzuron. The research of Nimmo and others (1986) suggests that these might be largely 4-CA. 4-CA was not produced as rapidly in hydrosoils, and accounted for considerably less of the degradation products in 48 weeks than was found in aerobic degradation in much shorter time periods. An alternative degradation pathway in anaerobic conditions might explain this phenomenon (Nimmo and others, 1986). In any case, it is apparent that CPU degradation is hindered in anaerobic conditions, just as anaerobic conditions hinder the degradation of diflubenzuron (Nimmo and others, 1986; Thus and others, 1991).

In water-covered silt loam (hydrosoil, pH 7.4, 24 °C, 5.2 percent organic content), the half-life of diflubenzuron was found to be 18.2 days during the first 56 days, and somewhat faster between days 56 and 90 (Thus and van Dijk, 1991). Diflubenzuron degraded to 2 percent of its original concentration in 90 days. These were nearly anaerobic conditions, and the hydrolysis products, CPU and DFBA, accumulated with no significant further breakdown during the course of this 90-day (13-week) study. This contrasts with the shorter half-lives and additional breakdown products produced in drier, aerobic soils (Nimmo and others, 1986; Walstra and Joustra, 1990).

In a study on the fate of 4-CA in soil, 49 percent of the originally applied diflubenzuron (5 ppm) bound to the soil in 24 hours (Bollag and others, 1978); 8 percent of the 4-CA was recovered as carbon dioxide from nonsterile soils after 6 weeks compared to none from sterile (autoclaved) soils. Soil extracts of radio-labeled bound residues showed degradation products not found in sterile soils, suggesting that microbes were able to degrade bound as well as unbound residues (Bollag and others, 1978). Three studies demonstrated that the degradation of 4-CA is facilitated by fungi (*Phanerochaete chysosporium*) and bacteria (*Alcaligenes* sp., *Pseudomonas acidivirans*) that use 4-CA as a carbon and nitrogen source (Surovtseva and others, 1992; Brunsbach and Reineke, 1993; Chang and Bumpus, 1993). In another study, 67.2 percent of the radio-actively labeled 4-CA applied to soils where barley, potatoes, and carrots were grown sequentially, was lost after 20 weeks, presumably as carbon dioxide from microbial degradation (Freitag and others, 1984). Residue losses in the following two years were negligible. More than 90% of the remaining product formed various soil-bound residues, and less than 1 percent of the applied radioactivity was found in the plants (Freitag and others, 1984). These studies on the fate of 4-CA in soil are consistent with the findings of Nimmo and others (1986).

The half-life of DFBA was found to be 9 and 12 days in humus sand and sandy clay, respectively, incubated at 24 °C (Nimmo and others, 1990). The products of DFBA

degradation are CO_2 and bound residues, produced in roughly equal proportions. These findings agree with those of Walstra and Joustra (1990), who also found "very rapid" degradation of DFBA, also producing CO_2 in the process.

7.1.6. Fate of Metabolites in Water

In an outdoor pond in California pastureland (pH 8 to 10, temperature not given) CPU increased in concentration as diflubenzuron decreased (Schaefer and Dupras, 1976). The rate of CPU formation suggested the half-life of diflubenzuron of about two days in the ponds, although diflubenzuron disappeared from the water column much faster as a combination of degradation and adsorption to organic matter. A final sample taken four days after application found CPU at a maximum concentration. The decrease in diflubenzuron followed by an increase in CPU mirrored the results of another study which lumped the two in an analysis of residues from sod-lined ponds. In this study, diflubenzuron was applied at 56 g ai/ha; an initial drop in residues (diflubenzuron degrading and adsorbing to sediments) was followed by a rise in residues (presumably an increase in CPU) which peaked four days after application (Madder and Lockhart, 1980). The residues (presumably CPU) then declined to about 30 percent of peak concentration in 10 days.

The fate of diflubenzuron and its metabolites was followed for four days in "pasture water" (pH 8.2, temperature 23 to 40 °C, 30 cm deep) after diflubenzuron was applied at 90 g ai/ha to a pasture (Schaefer and others, 1980). Diflubenzuron decreased to 1/10th original concentrations in the pond in 4 days (20.3 and 2.4 ppb 1 hour and 4 days after treatment). CPU increased from 5.6 to 7.2 ppb from 1 hour to 4 days post-spray, and 4-CA increased from 0.6 to 2.6 ppb in the same time. The water level was too low for additional samples. Thus, as the concentration of CPU increased, the concentration of its "parent" (diflubenzuron) decreased to very low levels, while CPU itself was being degraded to 4-CA. This dynamic process resulted in concentrations of 4-CA (2.6 ppb) which were 10 percent of the original concentration of diflubenzuron (20.3 ppb) in this particular aquatic situation.

In a water-filled pit created in a swamp (temperature and pH not given), diflubenzuron residues declined by 80 percent from days 1 to 6 after treatment, however residues at the level of detection were found 61 days after treatment (Mutanen and others, 1988). Residues at the level of detection were also found in water in a similar pit in mineralized soils (representing a ground water sample) up to 26 days after spray. In neither pit were CPU or 4-CA detected.

The fate of the metabolites of diflubenzuron is similar in many ways to that of diflubenzuron itself. They bind to organic matter and the principle route of degradation appears to be microbial. DFBA seems to be very unstable and is quickly degraded in the environment. CPU is somewhat more stable than diflubenzuron,

especially in oxygen-poor environments. 4-CA appears to be the most persistent of the degradation products, perhaps only because it binds so tenaciously to soils and therefore becomes somewhat unavailable to microbial degradation. 4-CA perhaps accounts for a majority of the unextractable or bound residues commonly found in studies of the fate of diflubenzuron in soils.

7.1.7. Summary--Undeveloped Forest

In the terrestrial environment diflubenzuron sprayed over forests settles on the canopy or falls directly to the forest floor. Immediately following application, diflubenzuron will begin to be transported, degraded, or ingested through various mechanisms and pathways (figures 1 and 2). Between 20 and 80 percent of the residue is washed or abraded from leaf surfaces during the first two to three weeks after spray (Wimmer and others, 1993). The abraded residues contribute to those already present in the ground litter from direct spray, and to those that overwintered from the previous year, and mostly degrade during the remainder of the spring and summer through the action of microorganisms in the litter and soil (Verloop and Ferrell, 1977; Nimmo and others, 1984; Nigg and others, 1986; 1994, Wimmer, unpublished data). Diflubenzuron in the canopy slowly diminishes throughout the summer so by autumn 5 to 50 percent of the diflubenzuron remains attached to leaves in the canopy (Wimmer and others, 1993). This amount, which is significant, falls to the forest floor with the leaves in autumn. It remains in the litter throughout the cool winter months until degradation (microbial action) accelerates with increasing temperatures the following summer (Bull and Ivie, 1978; 1994 Wimmer, unpublished data). Residues remaining on litter at the end of this second summer are diluted at leaf-fall by uncontaminated foliage, assuming the area was not resprayed in year two.

A relatively small amount of diflubenzuron will enter streams and rivers either by direct spray filtering through the canopy or by runoff from organic debris to which it has adhered (Figure 2). Diflubenzuron falling directly into streams or ponds is either diluted by stream flow or rapidly adheres to organic sediments where its fate parallels diflubenzuron in the soil (Carringer and others, 1975; Schaefer and Dupras, 1976; Booth and Ferrell, 1977; Chapman and others, 1985; Sundaram and others, 1991). In water itself, the stability of diflubenzuron depends on pH and temperature, being greater at pH values under 6 or at lower temperatures (Schaefer and Dupras, 1976; Ivie and others, 1980). During leaf-fall an additional pulse of diflubenzuron enters the stream with fallen leaves (Harrahy and others, 1993). As in the terrestrial environment, these residues persist on the submerged leaf material throughout the winter and degrade as temperatures increase the following spring and summer (Swift and others, 1988; Harrahy and others, 1993).

The persistence of diflubenzuron in the environment as described above might vary somewhat depending especially on the average summer and winter temperatures of a

region. Overwintering residues have been found in mountains in West Virginia and cotton fields in Texas, and might be expected to occur in most states, except perhaps in the extreme southwest and peninsular Florida where the growing season is nearly year round and soil temperatures in the fall and winter presumably are high enough to allow microbial activity to continue at moderately high rates. Conversely, residues might persist through a second winter in colder regions with short growing seasons.

7.1.8. Summary--Developed Forest

Virtually all the studies on the fate and transport of diflubenzuron have been conducted in woodlands away from substantial development. However, the same factors responsible for the degradation of diflubenzuron in wooded environments will also operate in forested residential areas. The main difference between these two environments that could affect the fate and transport of diflubenzuron is the percent of area covered by impervious (waterproof) surfaces (see Program Area Description). This characteristic of residential areas substantially changes the flow of water from that seen in natural ecosystems. Storm water and the debris it collects is deposited directly into local streams via runoff from streets, gutters, and storm sewers, rather than percolating through the soil with minimal overland flow. No studies were found on the effect of impervious surfaces (asphalt and concrete) on the fate and transport of diflubenzuron. This is considered a data gap in the risk assessment. The physical properties of diflubenzuron and its behavior in the field suggest that it will bind to asphalt (high in organic content) and not to concrete (a matrix of sand, gravel, clay, and limestone); thus, these surfaces ostensibly counterbalance each other in terms of their effects on fate and transport of this chemical. However, because of the lack of information on this point, the water runoff models in this risk assessment operate under the conservative assumption that runoff water washes all residues falling on these impervious surfaces (see Estimated Environmental Concentrations, this section).

Some diflubenzuron, assumedly, will wash from asphalt during a storm, as happens in the canopy, adding to washing from concrete surfaces. This results in greater inputs into streams in developed forests than those in undeveloped forests. The fate of diflubenzuron in many residential streams will likely be similar to that in streams in more undeveloped areas; it will bind quickly to organic debris and sediments and decline throughout the summer, assuming degrading microbes are present. Leaf packs formed by debris carried into waterways in the late fall and winter may harbor residues through the cold months and degrade when temperatures warm the following spring and summer. Many residential areas have leaf collection programs in the fall. Residues remaining on the leaves will be concentrated in these instances in landfills or compost heaps. Diflubenzuron is expected to degrade rapidly in compost heaps because microbial activity and temperatures are high in these environments (typically between 30 and 60 °C), even in autumn and winter (Fogarty and Tuovinen, 1991).

7.2. *Bacillus thuringiensis* var. *kurstaki* (*B.t.k.*)

Bacillus thuringiensis (*B.t.*) is an aerobic, gram-positive, endospore-forming bacterium. Over 20 varieties and hundreds of strains of *B.t.* have been isolated (Surgeoner and Farkas, 1992). Like other bacteria, *B.t.* has a two phase growth cycle composed of a vegetative stage when environmental conditions are favorable and the cells are young, and a spore-forming stage during adverse environmental conditions or when the cells approach old age. Exotoxins are released from the bacterium during vegetative growth. During the spore-forming stage, an endospore is formed within the vegetative cell during a process called sporulation. The endospore is referred to as a spore after the vegetative cell containing the endospore disintegrates.

A characteristic feature of *B.t.* during sporulation is the production within the vegetative cell of a bipyrimidal protein crystal, separate from the endospore (Surgeoner and Farkas, 1992). This crystal is released into the environment along with the spore when the bacterial cell wall degrades following sporulation (Figure 3) (Lambert and Peferoen, 1992). More than 25 proteins have been isolated from crystals produced by varieties and strains of *B.t.* (Lambert and Peferoen, 1992).

The variety of *B.t.* used in gypsy moth suppression or eradication programs is *Bacillus thuringiensis* var. *kurstaki* (*B.t.k.*). The protein crystal produced by this variety includes a delta endotoxin toxic to insects of the Order Lepidoptera (butterflies, moths, and skippers). *B.t.k.* does not produce a beta exotoxin, a protein which is toxic to some mammals, associated with other *B.t.* varieties (Faust and Bulla, 1982). *B.t.k.* does produce an alpha-exotoxin, however. Spores of *B.t.k.*, as well as protein crystals, are toxic to many Lepidoptera (Krieg and Lagenbruch, 1981; Peacock and Schweitzer, 1993).

B.t.k. is grown commercially for insecticidal formulations used against gypsy moths. A special growth medium is used to grow the vegetative cells. After a period of vigorous vegetative growth which depletes the essential nutrients in the medium, the bacteria forms endospores and protein crystals. The growth medium is then treated to remove most of the vegetative cells. The remaining spores and crystals are concentrated and dried to a fine powder. The amount of vegetative cells that remain and are incorporated into the powder varies from formulation to formulation. The powder is tested for potency based on its toxicity to cabbage looper caterpillars in a standard bioassay test. The potency of the powder is determined by comparison with the potency of a reference standard, *B.t.k.* strain HD-1-S-1980 (Surgeoner and Farkas, 1990). Potency is measured in International Units (IU)/mg. The potency of the final product is expressed as the number of Billion International Units (BIU)/ L.

7.2.1. *B.t.k.* on Land

Ultraviolet light, temperature, and moisture are important factors moderating the persistence of *B.t.* in the terrestrial environment. Ultraviolet light (a component of natural sunlight) is the most destructive environmental factor for *B.t.* regardless of the variety (Ignoffo, 1992). Ultraviolet light greatly reduces the viability of spores (Salama and others, 1984, Ignoffo, 1992). In one study the delta-endotoxin in the protein crystal was found to be slightly more resistant to ultraviolet light than the spore: exposure time to destroy half of the spores was 30 minutes while the protein crystals required 240 minutes (Ignoffo 1992). Twice as many spores remain viable after 24 hours on a cloudy day than on a sunny day (Frye and others 1973 as cited in Forsburg and others, 1976).

Temperature affects the persistence of both the spore and the delta-endotoxin. Spores and delta-endotoxin lose viability more quickly in higher temperatures (Ignoffo, 1992, Dulmage and Aizawa 1982). The viability of the spore is rapidly lost at temperatures over 75 °C (Salama and others, 1984) and with increasing moisture or humidity (Ignoffo 1992); however, temperatures this extreme do not reflect conditions found in nature. The combined effects of ultraviolet light, temperature and moisture may be more important than the effects of each factor alone (Leong and others, 1980).

Many studies indicate the insecticidal activity of *B.t.* is about a week in the environment, but a few reports suggest that the insecticidal effects of spraying *B.t.* persist for longer periods of time. *B.t.* falling on the forest canopy persists up to 30 days on some types of foliage (Smirnoff 1973 as cited in Forsburg and others 1976) and up to eight days on red oak (Sundarum and Sundarum, 1992). There is evidence that *B.t.k.* viability on leaves is related to tree species as the half-life for *B.t.k.* on oak is twice that of the half life on redbud (Pinnock and others, 1975 as cited in Forsburg and others, 1976). Although spore viability is quickly reduced on leaves, the insecticidal activity has been observed for up to 42 days following spraying (Morris and Hildebrand 1974 as cited in Dulmage and Aizawa 1982). It is unknown whether *B.t.* multiplies on the leaf surface (Lambert and Peferoen, 1992). One field study reported the insecticidal activity of *B.t.k.* persisted for more than 64 days (Miller and West, 1987) and another reported insecticidal activity after 90 days (Scribner, personal communication). It is unclear what is causing the prolonged insecticidal activity in these studies. Miller and West (1987), citing Yendol and others (1975), suggest that the presence of molasses in the *B.t.* mixture stimulates feeding by the caterpillars, which therefore ingest more *B.t.* than they otherwise would, thereby increasing their exposure.

Spores and crystals containing delta-endotoxin are ingested by animals along with vegetation. When ingested by certain types of insects, particularly lepidopterans, *B.t.* can multiply in the digestive tract (Lambert and Peferoen, 1992). Naturally-occurring *B.t.* almost never causes death in large numbers of individuals of susceptible species

(Dulmage and Aizawa 1982) since it is difficult to pass *B.t.* from dead carcasses of herbivorous insects to other individuals of their species (Dulmage and Aizawa 1982). *B.t.* spores are also ingested by vertebrates, but spores are incapable of multiplying in vertebrate digestive tracts because conditions, such as pH or moisture, are not favorable to growth. Intact *B.t.* spores have been isolated from vertebrate feces (Dunn 1960, Smirnoff and Macleod, 1961). These spores remain viable for up to 3 weeks in feces in natural environments.

B.t. has been isolated from soil in urban, forested, agricultural, and industrial locations, insect larvae and insect habitats such as rotting wood and stored product containers (Chilcott and Wigley 1993). *B.t.* varieties will not grow in acidic soils (Saleh et. al, 1970), but will multiply in neutral or alkaline soils. The presence of organic matter increases sporulation of *B.t.*, especially at lower pH values where *B.t.* does not multiply readily (Saleh et. al, 1970). *B.t.k.* can multiply in soil following application, but does not have secondary growth following sporulation (Akika and others 1977).

Spores can persist in the soil for several months; however, the insecticidal activity is greatly reduced (Pruett and others, 1980). In one report, after 63 days only 3% of the insecticidal activity remained, while 38 percent of the spores were still viable (Pruett and others, 1980). The delta-endotoxin appears to decay and lose potency much more quickly in soil than do the spores (Surgeoner and Farkas 1990, Dulmage and Aizawa 1982).

Microbial competition is an important factor determining the fate of *B.t.* in soil (Akika and others, 1977). When applied to sterilized soil, *B.t.* cell numbers did not decrease over time as did *B.t.* cells applied to unsterilized soil (Akika and others 1977 as cited in Dulmage and Aizawa 1982). Repeated applications of *B.t.* did not result in an increased concentration of *B.t.* in natural soils in the field. There appears to be a maximum level of *B.t.* that a soil will support (Dulmage and Aizawa 1982).

7.2.2. *B.t.k.* in Water

B.t. enters the aquatic environment through direct application to surface water, runoff from the terrestrial environment, or from feces of animals that have ingested the spores. A review notes several characteristics of the fate of *B.t.* spores in aquatic systems (Menon and de Mestral, 1985): viable spores persisted in the laboratory in dark, sterile water for 70 days; spores survive longer in nutrient-rich water than in tap or distilled water; and spores are degraded more quickly in seawater than in fresh water. Survival times under natural conditions are shorter than laboratory studies due to bacterial competition and predation. In two field studies, viable *B.t.k.* spores have been found in rivers for 13 days (Menon and de Mestral, 1985) and up to four weeks following spraying (Buckner and others, 1974).

There is conflicting evidence regarding the uptake of *B.t.* by mollusks. *B.t.k.* was observed in shellfish two days following spraying of a river, but was not observed within 30 days of the application (Buckner and others, 1974). No *B.t.k.* was found in shellfish following a similar spraying in the Moose River (Menon and de Mestral, 1985). *B.t.k.* was found not to affect aquatic microbial activity or decomposition (Kreutzweizer and others 1993).

7.2.3. Summary--Undeveloped Forest

B.t.k. is aerially applied in the early spring before the leaves of deciduous trees are fully expanded. The spray mixture containing spores, protein crystals, residual vegetative cells, spreaders, stickers, and sunscreens is released from an aircraft above the intended area. Spores and protein crystals are distributed on vegetation on and below the upper canopy layer, on litter, soil, and inadvertently, in some surface waters under the canopy or nearby due to drift. Some of the spores and crystals will wash off vegetation onto soil or litter, most will be degraded by sunlight, moisture, and temperature, and some will be ingested by animals. While most studies demonstrate that the insecticidal activity of *B.t.k.* on vegetation lasts about a week, a few reports suggest that it could be much longer in some situations (Miller and West, 1987; Scribner, personal communication). A small amount of *B.t.k.* may fall to the ground on leaves in autumn as observed by Pruett and others (1980). *B.t.* does not persist in the aquatic environment for more than a few weeks (Buckner and others, 1974, Menon and de Mestral, 1985), while in the terrestrial environment it can persist for several months (Dulmage and Aizawa 1982).

7.2.4. Summary--Developed Forest

The higher percentage of impervious surfaces in developed areas (see Section 4) leads to the assumption that *B.t.k.* might more easily be transported to aquatic systems in developed areas than in undeveloped areas. Higher concentrations of *B.t.k.* spores and crystals might occur in aquatic habitats in developed areas as a result, where they would degrade rather rapidly as in aquatic systems in undeveloped areas. However, no studies were located which examined the transport of *B.t.k.* over impervious surfaces. Thus, concentrations of *B.t.k.* in runoff were not estimated in this risk assessment (see Estimated Environmental Concentrations, this Section) because of this data gap and because the toxicological literature suggests that organisms in aquatic systems are not affected by *B.t.k.* (see Section 5).

7.3. Nucleopolyhedrosis virus

Nucleopolyhedrosis virus for gypsy moths (NPV) is a pathogen specific to gypsy moths now found naturally in the environment in most areas where gypsy moths are established. The National Gypsy Moth Management Program applies GypChek® (the product containing the virus) as an insecticide, augmenting naturally occurring populations or introducing NPV into new areas through aerial application. (A more detailed description of NPV is in Section 5.)

7.3.1. Fate and Location

The most significant environmental factor affecting the persistence of NPV in the environment is sunlight (Ignoffo, 1992). Several different NPV's specific to different species of pests (but not NPV for gypsy moths) had half-lives that ranged from less than 1 to 48 hours after exposure to natural and artificial sunlight (Ignoffo, 1992). In contrast, temperature and water have relatively little effect on the persistence of NPV in the natural environment. NPV specific to a species of *Heliothis* (a bollworm) were viable after 25 years stored at 5 °C, but lasted about 100 days at 50 °C. In general, NPV is stable at 10 °C to 30 °C (Ignoffo, 1992). NPV lasts longer dry than wet, but in general water does not greatly affect survival, especially for short-term exposures (less than 30 days) (Ignoffo, 1992).

NPV is now a natural component of the environment in the United States, although prior to the introduction of gypsy moths it was not. Presumably it was transported to this country on gypsy moths. It persists in any one location for at least one year after outbreaks of gypsy moths and subsequent epizootics (animal epidemics causing widespread mortality through infection by the virus) (Podgwaite and others, 1979). The persistence of NPV was tested on samples of leaf, bark, litter, and soil taken from woodland plots where natural outbreaks of the gypsy moth and virus had been observed, and on plots treated with NPV. Activity of NPV from spray deposits was measurable for 3 to 15 days (as determined by feeding studies using second-stage gypsy moth larvae). In contrast, NPV liberated from larval cadavers onto bark showed measurable activity for about one year. The difference is perhaps a result of where the NPV is deposited, being more protected from sunlight and rain in the crevices of bark where cadavers are found than where spray is deposited. In a study of the infectivity of different parts of the environment, it was found that soil under the litter and exuviae (sloughed skin) of larvae contained significantly more NPV than tree bark (Weseloh and Andreadis, 1986). However, another study showed that bark on trees where epizootics occurred the previous year was infective to larvae crawling over it, suggesting the importance of tree bark in transmitting NPV, particularly in the year following an epizootic (Woods and others, 1989). In a related study, it was found that egg masses laid on tree boles, had substantially higher levels of

contamination than those laid on rocks, ground debris, or understory brush (Woods and others, 1990).

The activity of NPV (or its potency as measured in bioassays using second-stage gypsy moth larvae), was found to be up to seven times greater in NPV harvested from cadavers than from live larvae (Shapiro and Bell, 1981). The difference in activity might be due to structural or biochemical changes in the infective parts of the NPV (the occlusion bodies, or OBs).

7.3.2. Transport and Transmission

NPV is probably transported in the environment in many ways: by wind, rain, parasites, predacious insects, and by dispersal of infected larvae (Lautenschlager and others, 1980). Field and laboratory studies have shown that viable gypsy moth NPV (as determined by bioassay using moth larvae) is found in the gut of several avian and mammalian species in the wild (Lautenschlager and others, 1980) and these animals pass viable NPV through their gut within a few days of ingestion (Lautenschlager and Podgwaite, 1979).

The transmission of NPV was found to be most effective when larvae crawled over exuviae from NPV-infected gypsy moth larvae, in contrast to crawling over contaminated soil or bark; larvae contaminated with NPV did not transmit NPV via contact with other larvae with which they were kept in the laboratory (Weseloh and Andreadis, 1986).

Other studies have shown that egg masses oviposited on contaminated bark become infected with NPV (Murray and Elkinton, 1990) and that the site of oviposition is more important than parental population (transovum transmission) in transmission of the virus (Murray and Elkinton, 1989). These results demonstrate the importance of oviposition site in the transmission of the virus in high-density populations or after epizootics and suggest that transmission via adults is not common. A laboratory study confirmed this last observation; NPV fed in sublethal doses to late-instar larvae is not transmitted via adults to the next generation (Murray and others, 1991). This study showed that pupae infected with NPV failed to eclose, whereas those that successfully emerged avoided infection and were incapable of transmitting the virus to the next generation (Murray and others, 1991).

7.3.3. Summary--NPV

NPV is a virus that infects gypsy moths only and is used as an insecticide in the USDA's gypsy moth program. It is now found in the environment in most places where gypsy moths are found. NPV is particularly susceptible to degradation by

sunlight. Applied aerially the insecticide persists for about two weeks, but when released from infected cadavers of caterpillars NPV can persist for up to a year or more. The virus is relatively stable in a normal range of temperatures and short-term exposures to water does not affect its persistence.

7.3.4. Summary--Undeveloped Forest

The higher percentage of impervious surfaces in developed areas (see Section 4) leads to the assumption that NPV might more easily be transported to aquatic systems in developed areas than in undeveloped areas. Higher concentrations of NPV might occur in aquatic habitats in developed areas as a result. However, given the specificity of NPV to gypsy moths this scenario is of no consequence to nontarget animals. Thus, concentrations of NPV in runoff in developed areas were not estimated in this risk assessment (see Estimated Environmental Concentrations, this Section).

7.4. Disparlure

Disparlure is a chemical simulating a natural pheromone emitted by female gypsy moths to attract males. The USDA's gypsy moth program applies Disparlure in vinyl flakes and beads to disrupt males seeking females. It is also used in traps to attract males.

The persistence of Disparlure concentrations in air emitted from flakes applied to a woodland was shown to decrease gradually over 34 days to between 1.5 percent and 15.5 percent of concentrations measured in the first 24 hours (Caro and others, 1981). At normal application rates, this level (0.04 ng/m^3) is predicted to be near the detection threshold of gypsy moths; it is within an order of magnitude of the threshold concentration for response of many species of insects to pheromones (Caro and others, 1981). Concentrations were highest between 1400 and 2200 hours each day, and at 0.3 m and 10 m height within the woodland (Caro and others, 1981).

Possibly, flakes impregnated with the pheromone could be carried into aquatic systems more easily in developed forests because of the higher percentage of area covered by impervious surfaces (see Section 4). The ecological significance of this is negligible.

7.5. Dichlorvos

Dichlorvos is the insecticide used in traps to kill gypsy moths. The chemical is added to a plastic strip, Vaportape II, which is placed inside the traps. Dichlorvos is

therefore unlikely to be released into the environment except through vaporization of the chemical from the tape, or by accident should a trap break and fall to the ground. The fate and transport of Dichlorvos on the Vaportape strip in developed and undeveloped forests is likely the same.

7.5.1. Dichlorvos in Air

Dichlorvos does not photodegrade, but does react with atmospheric gases such as ozone and various hydroxyl radicals. The estimated half-life of dichlorvos in air was determined to be about 16 hours (Hazardous Substances Database, 1989). Air samples taken at 0, 1, 2, 6, and 10 hours after turfgrass was sprayed with dichlorvos at 1.59 lbs a.i./acre were found to contain negligible levels of insecticide (Maddy and others, 1984).

7.5.2. Dichlorvos on the Ground

Dichlorvos is readily hydrolyzed in soil to dimethyl phosphate and dichloroacetaldehyde, then degraded to dichloroethanol (U.S. EPA, OPTS, 1987; Doyle, 1981). The half-life of dichlorvos is reported to be two hours in silt and eight hours in sand (U.S. EPA, OPTS, 1987). This rapid hydrolysis is both catalyzed by chemical bases and facilitated by activity of soil microorganisms (Lamoreaux and Newland, 1978). Various environmental factors can, however, result in a half-life as long as 17 days (Menzie, 1972).

Dichlorvos has an organic carbon partition coefficient of 28, which indicates a low tendency to adsorb to soils (Kenaga, 1980). This results in moderate to rapid mobility of dichlorvos in a range of soils (U.S. EPA, OPTS, 1987). This mobility increases in soils low in organic matter, but high in humic acid content (Khan and Khan, 1986). Mobility of dichlorvos is greater under acid conditions in sandy soils, and greater under alkaline conditions in silty soils (Sharma and others, 1985, 1986).

7.6. Environmental Fate and Transport Modeling

Characterization of risks to the ecosystem from insecticides used in gypsy moth management requires an estimate of residues levels. Residue levels are estimated for various parts of the environment. Residue levels for diflubenzuron and *B.t.k.* were estimated in the upper and lower canopy leaves, at the soil or litter surface, and in surface water bodies (streams and ponds). Diflubenzuron concentrations in runoff water were estimated for both the developed and undeveloped forest ecosystems. There is a great amount of uncertainty in the number of *B.t.k.* crystals or spores likely to be in runoff water. *B.t.k.* does not persist for long periods of time (less than

two weeks in aquatic systems) and does not present the same magnitude of hazard as diflubenzuron to aquatic organisms. For these reasons, the runoff of *B.t.k.* in the two ecosystems was not determined. NPV and Disparlure present negligible hazards to organisms other than gypsy moth; thus the maximum theoretical residue (flat surface) was estimated rather than residues in leaves, soil, and other components. Dichlorvos is not broadcast through the environment as are the other insecticides. It is used in insect traps where dichlorvos residues are expected to remain, embedded in a strip of Vaportape. Even if the trap is disturbed and the Vaportape strip is released into the environment, dichlorvos is expected to remain in the strip. Therefore, dichlorvos residues on other types of environmental media are not considered in this risk assessment.

Mathematical models were used to estimate post-application residues of diflubenzuron and *B.t.k.*. The aerial dispersion of diflubenzuron and *B.t.k.* was simulated using the Forest Service Cramer Barry Grim (FSCBG) model. Concentrations of diflubenzuron in runoff water were estimated with the Pesticide Root Zone Model (PRZM). A surface water model was developed to estimate the concentration of diflubenzuron in a typical stream and pond receiving storm runoff. *B.t.k.* concentrations resulting from directly spraying surface waters were determined to represent a reasonable worst-case scenario. Only *B.t.k.* concentrations in water resulting from direct application were determined. *B.t.k.* falling on plant leaves and exposed to ultraviolet radiation will rapidly degrade, leaving only the *B.t.k.* spores on soil or litter surface available for transport via storm runoff into aquatic habitats. Concentrations of *B.t.k.* in surface waters after storm runoff were not determined due to uncertainties about the transport of *B.t.k.* spores from soil or litter into water.

7.6.1. Methodology

7.6.1.1. Diflubenzuron and *B.t.k.* residues on vegetation and soil/litter surface

The Forest Service Cramer Berry Grim (FSCBG) model simulates aerial dispersion of insecticides, using the initial pesticide droplet size distribution, aircraft speed, aircraft type, and meteorological conditions to calculate the trajectory of a falling droplet of insecticide. The spatial area modeled includes all of the spray area (spray block) and a portion of the area adjacent to the spray block. The average mass of diflubenzuron or *B.t.k.* within the spray block was calculated, as well as the maximum, minimum and standard deviation. Based on typical application rates, four rates of application were used for estimating residues of diflubenzuron (0.25, 0.33, 0.5, 1.0 oz ai/ac, or 17.5, 23.3, 35.0, 70.0 g/ha respectively), and two rates of application were used for *B.t.k.* (24, 40 BIU/ac). Diflubenzuron residues were estimated by modeling a single application as well as a second application at the same rate one week later. This model does not apply to ground applications of either diflubenzuron or *B.t.k.*

The parameter values chosen, mostly from handbooks on aerial application equipment and from pesticide labels, represent the environmental conditions and equipment commonly encountered during a spray program (Table 7-1). However, many combinations of aircraft, spray equipment and meteorological conditions were not addressed. Residues predicted by FSCBG generally represent those expected from aerial spraying. Site-specific conditions may cause actual residues to deviate from those predicted using the parameter values given in Table 7-1.

The parameters that substantially affect model output include release height, wind speed, aircraft speed, aircraft type, and application rate (Teske and Curbishley, 1990). The model is not very sensitive to changes in temperature or humidity. The model is very sensitive to wind speed and release height (Teske and others, 1991). The wind was modeled as a 5 mph crosswind perpendicular to the flight lines. This wind speed was selected to result in the maximum deposition on vegetation and the soil surface. Greater wind speeds would cause more of the insecticide to drift away from the target area, while lower wind speeds would not allow the proper amount of turbulent mixing required for even coverage. A multiple story tree canopy 15.25 m (50 feet) high was used to simulate a mixed hardwood forest. The release height was about 12 m (39 feet) above the canopy. The model was run in the "near wake" mode of FSCBG to calculate the percentage of the insecticide on the soil surface, in the canopy, or suspended in the air.

The model has been validated in the field using data obtained from an aerial application of the pesticide Asana XL to a seed orchard (Teske and others, 1991). The validation results suggest that FSCBG adequately represented the spray system, although during the validation runs the model generally overpredicted the average insecticide mass within the spray block by 12.9 percent.

Several factors contribute to uncertainty in the results of the aerial dispersion model. Small differences in release heights resulted in large differences in the estimated concentration of diflubenzuron or *B.t.k.*. Since it is unlikely that a pilot would maintain a constant altitude during aerial application, the actual deposition may deviate from the model predictions. Meteorological conditions (wind, temperature, relative humidity) vary throughout a spray application and may also affect deposition, although this variation is not considered by FSCBG. FSCBG assumes, unrealistically, that the canopy is homogeneous throughout the spray block. Even when using the same configuration of aircraft and spray equipment, these factors combine to create more variability in observed residue levels than predicted by the model. In addition, different spray equipment, aircraft, and aircraft speeds can be expected to produce dissimilar distributions of residues. Despite the uncertainties associated with the model, it produces reasonable results when compared to monitoring results and can simulate residue levels following application rates for which no monitoring data are available.

7.6.1.2. Degradation of diflubenzuron on leaves over the growing season

Diflubenzuron residues decrease over time due to degradation of the pesticide on vegetation. Diflubenzuron residues were determined for leaves immediately after application and at the end of the growing season. Persistence was assumed to be similar to diflubenzuron on hardwood leaves in West Virginia (Wimmer and others, 1993); an average of 46 percent of the original residue remained on the upper canopy leaves at leaf-drop, while 62 percent remained on lower canopy leaves.

7.6.1.3. Diflubenzuron concentration on leaves

Diflubenzuron concentration on leaves was determined by the following equation:

$$C = \frac{DFB}{(Leaf\ Weight)}$$

where:

DFB = diflubenzuron residue ($\mu g/cm^2$)

$Leaf\ Weight$ = fresh weight (mg/cm^2)

Diflubenzuron concentrations on leaves were calculated at the time of application when leaves were not yet fully expanded and at the end of the growing season. The fresh weight of leaves was used rather than dry weight because nontarget organisms would be ingesting fresh leaves. Weights of leaves per unit area (specific weight in $\mu g/cm^2$) vary between species, however, fresh weights on a per area basis are not commonly reported. Due to lack of data in the literature, specific weights were measured for seventy leaves from three different species (dogwood, sourwood, oak). Fresh leaves were collected shortly after budburst (about four weeks) from a mixed hardwood stand in Prince William County, Virginia. The average specific weight for the seventy leaves was 13.20 mg/cm^2 (Rockwood, unpublished data).

Leaf weight and surface area increase as the leaves expand, however only one study was found that measured this change throughout a growing season. This study measured changes in specific weight ($\mu g/cm^2$) in fresh birch leaves in Finland in spring after budburst, in the middle of the growing season, and at senescence (Bogacheva, 1994). These data suggest that the specific weight of early spring leaves is 8.5% less than leaves at the end of the growing season. Considerable uncertainty is associated with using these data to estimate leaf weight because these data were collected close to the Arctic circle over a period that did not include budburst. Because diflubenzuron concentration is lower in leaves with higher specific weights, the conservative assumption used in this risk assessment was that specific leaf weights do not increase throughout the growing season.

7.6.1.4. Diflubenzuron concentration in litter

Diflubenzuron concentration in spring leaf litter was determined by dividing the residues at the soil or litter surface by the specific dry weight of a leaf. In the autumn, diflubenzuron concentration in leaf litter was estimated by dividing the residue remaining on the leaf at leaf drop by the dry weight per unit area of the leaf. The dry weight rather than the wet weight was used because fallen leaves are expected to desiccate when removed from the tree.

7.6.1.5. Diflubenzuron concentration in soil

Concentrations of diflubenzuron in the upper 1 cm of soil were estimated by the following equation:

$$c = \frac{DFB}{Soil\,weight}$$

where:

c = diflubenzuron concentration per 1 cm soil depth

DFB = diflubenzuron mass ($\mu g/cm^2$)

$Soil\ weight$ = wet weight of soil (mg/cm^3)

Assumedly, diflubenzuron will remain in the upper soil layer and will not leach or be incorporated to deeper depths based on fate and transport properties discussed earlier in this Section. The weight of a cubic centimeter of soil is dependent upon characteristics of soil that vary on a site-specific basis. As a conservative assumption, the weight of a cubic centimeter of a fine loam (1.2 g), a very light soil, was selected to maximize diflubenzuron concentration.

7.6.1.6. *B.t.k.* and diflubenzuron concentration in directly sprayed water bodies

Diflubenzuron is not applied directly to large bodies of water in gypsy moth programs. Inevitably, however, it filters through the forest canopy to small pools, springs, puddles, vernal pools, and streams. In order to provide a conservative (maximized) estimate of exposure, concentrations of diflubenzuron following direct application to a small stream are calculated. The concentrations of *B.t.k.* and diflubenzuron in directly-sprayed streams of 0.76 m and the concentrations of *B.t.k.* in 2 m ponds were calculated. Ponds were assumed to be cylindrical in shape. The total mass of *B.t.k.* falling on the surface area was determined by multiplying the average residue per area by the area of a circle. This mass was divided by the volume of water with a given depth. Mixing was assumed to be instantaneous. The stream was assumed to be triangular in cross section. The total mass of insecticide (*B.t.k.* or diflubenzuron) falling on the surface of a one meter long segment was

calculated. Concentration was determined by dividing this mass by the volume of water in a 1 m long stream segment.

7.6.1.7. Diflubenzuron concentration in aquatic sediments

Concentrations of diflubenzuron in the sediments were assumed to be 2 percent of the concentration in the water column based on a monitoring study by Kingsbury and others (1987).

7.6.1.8. 4-chloroaniline concentration in the water column

Concentrations of 4-chloroaniline in the water column were assumed to be 10 percent of the water column concentration of diflubenzuron based on monitoring following diflubenzuron treatment of a flooded pasture (Schaefer and others, 1980).

7.6.1.9. Diflubenzuron concentration in runoff water

The Pesticide Root Zone Model (PRZM) was used to estimate the diflubenzuron concentration in runoff water from a sprayed watershed. Model parameters were selected to simulate the highest concentrations of diflubenzuron that could reasonably be expected to provide a worst-case scenario for aquatic organisms (Gray and Beauman, 1994). The model is sensitive to the foliar extraction coefficient for pesticide washoff for diflubenzuron and the soil adsorption coefficient, K_d. Therefore, soils with small infiltration rates and low organic matter concentrations produce the highest concentrations of diflubenzuron in runoff water (Gray and Beauman, 1994). Soil types selected had high clay content, low infiltration rates, and low organic matter concentrations. Three locations with soils of this type were modeled: Arkansas/southern Missouri (Northern Arkansas forest), northern Georgia (Northern Georgia forest), and southern New York/Massachusetts/Connecticut (Northeastern forest).

Parameter estimates were obtained from soil surveys, agricultural chemical handbooks, and from PIRANHA, a set of programs including PRZM and a nation-wide physiographic and soil characteristics databases (Burns, 1990) (Table 7-2). Since PRZM is normally used to simulate crops, forests are not adequately described by the PIRANHA database. For the simulation, the canopy coverage was assumed to be 85 percent when fully extended, with a foliar washoff coefficient of 0.2. Diflubenzuron was applied to the partially extended tree canopy 10 days after budburst of gypsy moth-susceptible trees.

Annual application of diflubenzuron was simulated for the 20-year period from 1964-1983 for each site. This period was selected because adequate meteorological data describing rainfall frequency and intensity were available over this period for each site. The twenty simulations were independent of one another; residue levels of diflubenzuron from one simulation had no effect on residues generated by subsequent runs. The twenty years of meteorological data provided an estimate of variability in diflubenzuron concentration in runoff. The maximum concentration of diflubenzuron in runoff water was observed when a five cm rainstorm occurred shortly after application; this scenario was selected for the risk assessment.

Many parameter values used in the PRZM model (that is, slope, cover type, soil composition) are site-specific; therefore, pesticide concentration in runoff from particular sites may be different from values predicted by the model. The PRZM model results used in this analysis could be considered a worst-case scenario for sandy or silty loam soils. Therefore the concentration of diflubenzuron used in this analysis probably overestimates actual concentrations in many sites. This type of estimate is useful in the risk analysis because it sets an upper limit on the expected response of aquatic organisms to diflubenzuron applications in the field.

7.6.1.10. Diflubenzuron concentrations in waters receiving runoff

Diflubenzuron is not applied directly to large bodies of water, although small streams are often found in forested areas and may inadvertently receive a direct spray or drift. For the purposes of the risk assessment, the concentration of diflubenzuron in a small stream, directly sprayed, was determined as a worst case scenario. The concentrations of diflubenzuron in a small stream and shallow pond receiving runoff from the sprayed area were also determined for developed and undeveloped forest watersheds.

The surface water model used in this risk assessment to estimate diflubenzuron concentrations in streams and ponds was developed specifically to analyze the effects of nonpoint runoff in a watershed after aerial spraying. This model contains few site-specific parameters, and is used only to give an approximate estimate of diflubenzuron in streams or ponds receiving runoff. Concentrations of diflubenzuron at specific sites can reasonably be expected to vary from model predictions. The model predicts the concentration of diflubenzuron in a stream and a pond in a small watershed (9 mi^2). The entire watershed was assumed to be sprayed with diflubenzuron. The watershed consists of a 3,373 m (2.1 mile) long stream that drains 52.2 percent of the watershed before emptying into a 374-m diameter pond. The remaining 47.8 percent of the watershed drains directly into the pond via overland flow. The length of the stream was determined to be the average length of a second order stream draining a watershed of 4.7 mi^2 (van der Leeden and others, 1990). The surface area of the pond was determined by calculating the surface water

body size for a watershed of 9 mi^2 using the average basin to lake ratio of 212 to 1 reported by Reckhow and Chapra (1983). The simulated pond is two meters deep. Water enters the pond from overland runoff and from the stream. Water leaves the pond through a drainage outlet and from evaporative loss. Water loss due to evaporation is based on the evaporation rate (van der Leeden and others, 1990) and the available surface area for evaporation (surface area of the pond). The water level of the pond is assumed to be constant, and the outflow from the pond varies with stream inflow.

The model assumes that the stream has a base flow rate of 3.60 m/second and an initial depth of 0.76 m at base flow. These values were selected because similar values have been reported for a second order stream of that length (van der Leeden and others, 1990). The stream channel was modeled as a triangular area, the depth and width vary, depending on the volume of water in the stream. The stream is assumed to be twice as wide as it is deep, making the cross sectional area equivalent to depth squared.

The model simulates the change in diflubenzuron concentration, calculates the average concentration, and the maximum concentration over each 24 hour period within the first 96 hours following a rainstorm. Diflubenzuron concentration was calculated at each of the one second time steps until 96 hours had elapsed. The model may overestimate diflubenzuron concentrations due to the assumption that all diflubenzuron on impervious surfaces is carried into surface waters via runoff. Considerable uncertainty exists regarding the actual amount of diflubenzuron that is not bound to organic impervious surfaces, such as asphalt. Some of the diflubenzuron may not be available for transport via runoff water.

At the first time step, the stream depth at base flow was used to calculate the cross-sectional area of the stream. The volume of water in the stream was calculated using the following equation:

$$V = l \times xa$$

where:

V = volume of water in stream (m^3)
l = stream length (m)
xa = cross-sectional area of stream (m^2)

The volume of the stream is altered by the new volume of base flow entering the stream, the volume of runoff entering the stream, and the volume of water leaving the stream as discharge into the pond. With no runoff, the volume of base flow entering the stream is balanced by the volume of water discharged into the pond such that the stream volume does not change. When runoff occurs, the stream volume increases. Stream volume is calculated at each time step after the first, using the following equation, which also accounts for runoff entering, volume of base flow entering, and volume of discharge leaving the stream:

$$V_t = V_{t-1} + V_{RO} + V_{BF} - V_{SD}$$

where:

V_t = *volume* (m^3)

V_{t-1} = *volume at previous time step* (m^3)

V_{RO} = *runoff volume entering stream* (m^3)

V_{BF} = *base flow volume* (m^3)

V_{SD} = *stream discharge volume* (m^3)

The depth of the stream was calculated at each iteration after the first time step, using the following equation, assuming a stream channel twice as wide as it is deep:

$$d = \sqrt{\frac{V}{l}}$$

where:

d = *depth* (m)

V = *volume of stream* (m^3)

l = *stream length* (m)

Stream velocity and overland flow velocity were calculated with the following equation (Newberry, 1984):

$$v = \frac{\sqrt{m} \times \left(\frac{p}{xa}\right)^{\frac{2}{3}}}{n}$$

where:

v = *velocity* (m/s)

p = *wettable perimeter of stream* (m)

xa = *cross-sectional area of stream* (m^2)

n = *Manning's n*

m = *slope*

The maximum overland velocity in the model was determined by the highest overland flow velocity (greater than 0.61 m/second) reported for a land use type of residential dwellings and grass (USDA, 1983). When the velocity calculated by the model exceeded the maximum reported velocity, the simulated velocity was taken to be the maximum value.

Different modeling assumptions were made regarding overland flow of water from the terrestrial environment to the stream and pond, depending on whether surfaces were impervious (for example, pavement and roads) or pervious (that is, surfaces through which infiltration is possible). The model assumes that impervious areas account for 35 percent of the watershed in the developed forest ecosystem, and only 5 percent in the undeveloped forest ecosystem. The developed forest estimate is based on the average percentage of impervious surfaces in residential areas of single-family dwellings on quarter acre lots (Chapra and Reckhow, 1983). The single-family residence value was selected because gypsy moth programs are often conducted in this type of suburban residential setting. The volume of runoff produced on either pervious or impervious surfaces was determined by the following equations (USDA,1983):

$$S = \left(\frac{1000}{SCS}\right) - 10$$

where:

S = runoff parameter derived from SCS runoff curve number
SCS = the SCS runoff curve number for a particular cover class
98 = SCS Impervious areas (pavement)
61 = SCS Pervious areas (good grass)

$$Runoff = \frac{\sqrt{R - (0.2 \times S)}}{R + (0.8 \times S)} \times 0.0254$$

where:

Runoff = runoff produced (cm)
R = rainfall (inches)
S = runoff parameter derived SCS runoff curve

The model simulates the effect of diflubenzuron entering a pond and a directly-sprayed stream by means of overland runoff after a relatively large storm event six hours after application. The diflubenzuron concentration in the pond and directly sprayed stream six hours after application was used as the initial diflubenzuron concentration in the surface water model. When no runoff occurs within 6 hours of application, the concentration of diflubenzuron in both the stream and the pond was less than 1 μg/L. Diflubenzuron concentrations are expected to increase in the pond and stream as the overland flow enters them.

The surface water model was designed to provide a generalized representation of diflubenzuron transport in an aquatic system. This approach was selected over a more detailed site-specific model because of the difficulty in extrapolating from site-specific models to the geographically diverse program area. The predictions of the surface water model are useful for comparing the expected concentrations of diflubenzuron in the two ecosystems (developed and undeveloped forests); however, any spray may result in aquatic concentrations that differ from the model results because of site specific factors. Ponds less than two meters deep are likely to have higher diflubenzuron concentrations than the simulated pond, whereas larger, deeper ponds are likely to have lower diflubenzuron concentrations.

The model assumes homogeneous and instantaneous mixing, thus it simplifies the hydrological conditions of a stream. Consequently, there is uncertainty regarding the residence time of diflubenzuron in the stream. In reality, the concentration of diflubenzuron in a pond or stream is likely to vary spatially. Diflubenzuron residence time would vary if the model assumption of equal inflow and outflow of water in the pond were violated. Results of this model reflect the maximum diflubenzuron concentration observed rather than the average concentration (Table 7-5). The maximum concentration persists for very short time periods (less than 6 hours) and is similar between application rates. The average concentration is higher for higher application rates. The maximum concentration was used in the risk assessment due to uncertainty about the effect of varying diflubenzuron concentrations on the response of aquatic organisms. Given that there are no toxicological data to determine whether the response of an organism to varying diflubenzuron concentrations is more similar to that organism's response to the average concentration or the maximum concentration encountered, the maximum concentration was used in the risk assessment as a conservative estimate.

7.6.1.11. Disparlure

Disparlure residues were estimated by calculating the theoretical expected mass per area assuming a flat surface with no vegetation cover. This calculation overestimates the mass of Disparlure expected in a forest canopy and represents a worst-case scenario. Given an application rate of 30 g/acre this results in 0.713 $\mu g/cm^2$.

7.6.1.12. Nucleopolyhedrosis virus

NPV residues were also estimated by calculating the theoretical expected mass per area assuming a flat deposition surface. Using the maximum application rate of 5 x 10^{11} polyhedral inclusion bodies (OBs)/acre, resulting in 12355 OB/cm^2.

7.6.1.13. Dichlorvos

Dichlorvos is applied exclusively in traps and is not expected to come in contact with forest vegetation, soil, or water unless the traps are disturbed. The mass of Dichlorvos in the Vaportape II within the trap is 590 mg. If a trap is broken, the tape may come in contact with the soil surface or vegetation. Dichlorvos may leach out of the tape into soil. Traps are usually laid out on a grid and are not placed in clumps. If a trap is broken, and the tape released into the environment, it would represent a very small area within the ecosystem. Because so few organisms would be likely to encounter the broken trap, a comprehensive analysis of the concentration of dichlorvos in soil and on vegetation will not be conducted. Therefore, no

calculations of dichlorvos concentration were performed. The risk of an organism encountering a tape, and either eating it or coming in contact with it will be addressed.

7.6.2. Results and Discussion

7.6.2.1. Diflubenzuron

Discussion of diflubenzuron residues is important because these are the only data that are directly comparable to data collected in monitoring studies of field applications. Simulation of aerial application of diflubenzuron indicated that most of the residues fall on vegetation in the upper or lower canopy (Table 7-3). A smaller amount of diflubenzuron residues reach the soil or litter surface. Low concentrations of diflubenzuron occur in the directly sprayed stream, with even lower concentrations in waters receiving runoff. Waters in developed forests had higher concentrations of diflubenzuron than water bodies within undeveloped forests.

As expected, modeling results indicated the highest residues were associated with the highest of the four application rates examined. Higher residues were found in the upper canopy than the lower canopy. Applying diflubenzuron twice, rather than once, results in residue levels similar to those from a higher application rate used once (Table 7-4).

The most extensive monitoring study of diflubenzuron residues was conducted in a mixed hardwood forest in West Virginia following application with 0.5 oz a.i./ac (35.01 g a.i./ha) (Wimmer and others, 1993). Diflubenzuron residues are reported in nanograms per square centimeter, making them directly comparable to residue estimates predicted by the Forest Service Cramer Berry Grim model (FSCBG). The mean upper canopy residue (16.7 ng/cm^2), the mean lower canopy residue (9.02 ng/cm^2), and the standard deviations of upper (17.1 ng/cm^2) and lower canopy (6.59 ng/cm^2) residues from the hardwood forest compare well with the mean residues and standard deviations predicted by FSCBG. However, in this study two applications of diflubenzuron within 24 hours were made because a drenching thunderstorm occurred shortly after the first application. If most of the residues from the first application were not lost during the rainstorm, these data would overestimate the average residue level following a 0.5 oz a.i./ac (35.01 g/ha) application.

Upper canopy leaves in the spring had the highest diflubenzuron concentrations of any vegetation; grasses, lower canopy leaves and forbs had lower concentrations (Tables 7-5 through 7-8). A monitoring study (Martinat and others, 1987) following a 70.75 g a.i./ha (1.0 oz a.i./ac) application reports lower diflubenzuron concentrations in leaves (0.10 to 0.45 ppm) than estimated through modeling (0.74 to 1.13 ppm);

however, the leaf weight and species type are not reported, making direct comparison impossible.

Diflubenzuron concentrations in the litter in field studies (greater than 1 ppm spring, 1 ppm autumn, 1994; Wimmer, unpublished data) are similar to those estimated through modeling (1.4 ppm spring, 1.2 ppm autumn).

Surface water diflubenzuron concentrations were greater for streams and ponds in the developed forest ecosystem than those in the undeveloped forest. Concentrations in streams increase rapidly following runoff, but decline to less than 1 μg/L within 24 hours of runoff. Concentrations in ponds increase slowly and remain high (greater than 1μg/L) for a longer period of time than streams. For both ponds and streams the concentrations reported in Table 7-6 are the maximum concentrations that occurred within the first 24 hour period.

7.6.2.2. *B.t.k.*

B.t.k. residues were estimated on vegetation in the upper and lower canopy, and on the soil or litter surface. As with diflubenzuron, most (greater than 60 percent) of the *B.t.k.* residues were deposited on vegetation in either the upper or lower canopy (Table 7-9), with a lesser amount reaching the soil or litter surface. *B.t.k.* concentrations in directly sprayed streams and ponds were less than 1.5 mg/L.

7.6.2.3. Disparlure, Nucleopolyhedrosis Virus, and Dichlorvos

Based on theoretical deposition rates onto a flat surface, the expected residue of Disparlure is 0.741μg/cm^2, while the expected residue of nucleopolyhedrosis virus is 12355 OB/cm^2. Dichlorvos is expected to remain in the trap on the Vaportape at a concentration of 590 mg/tape or trap.

7.7. References

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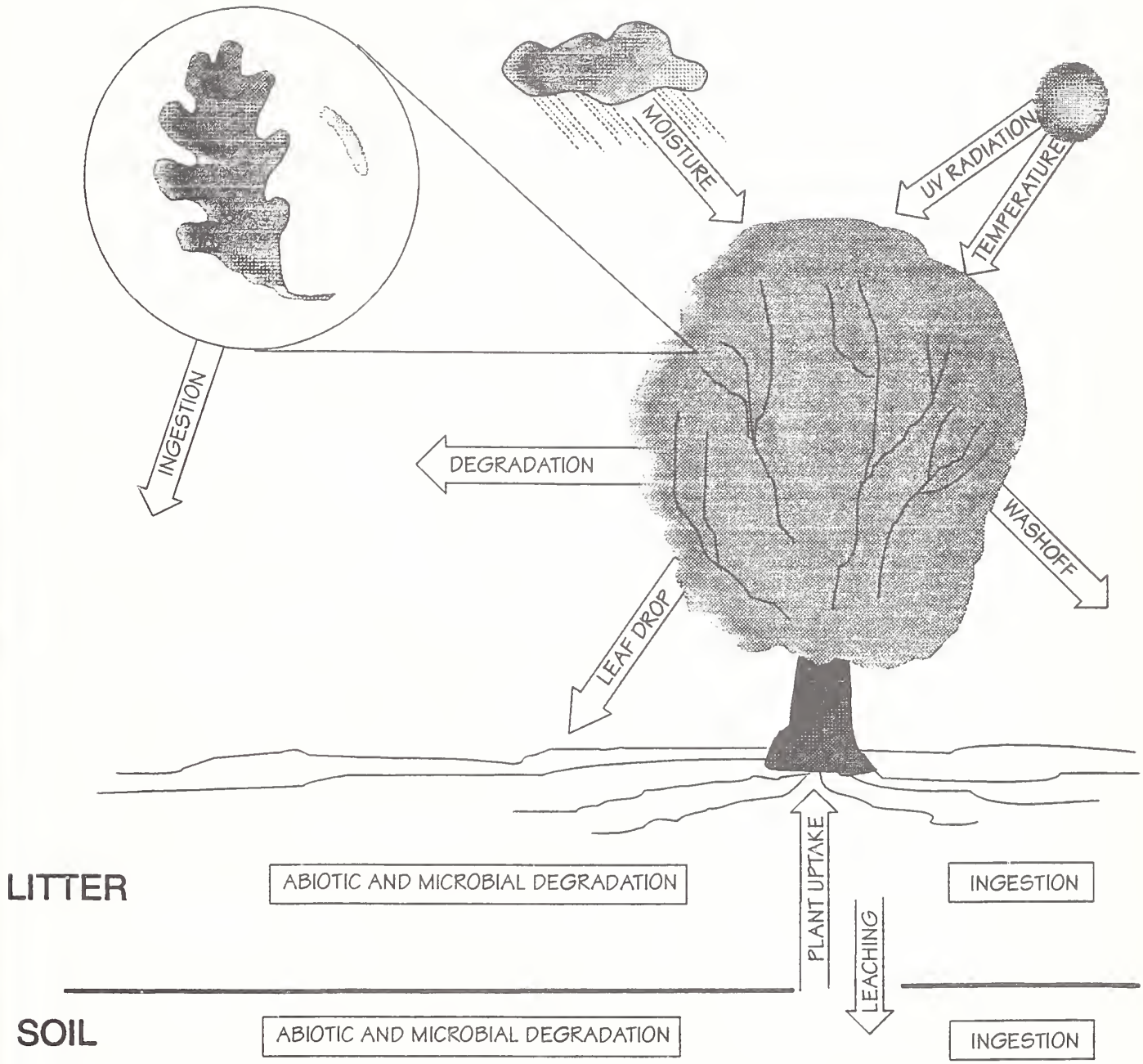

Figure 7-1. Fate and transport in the terrestrial environment.

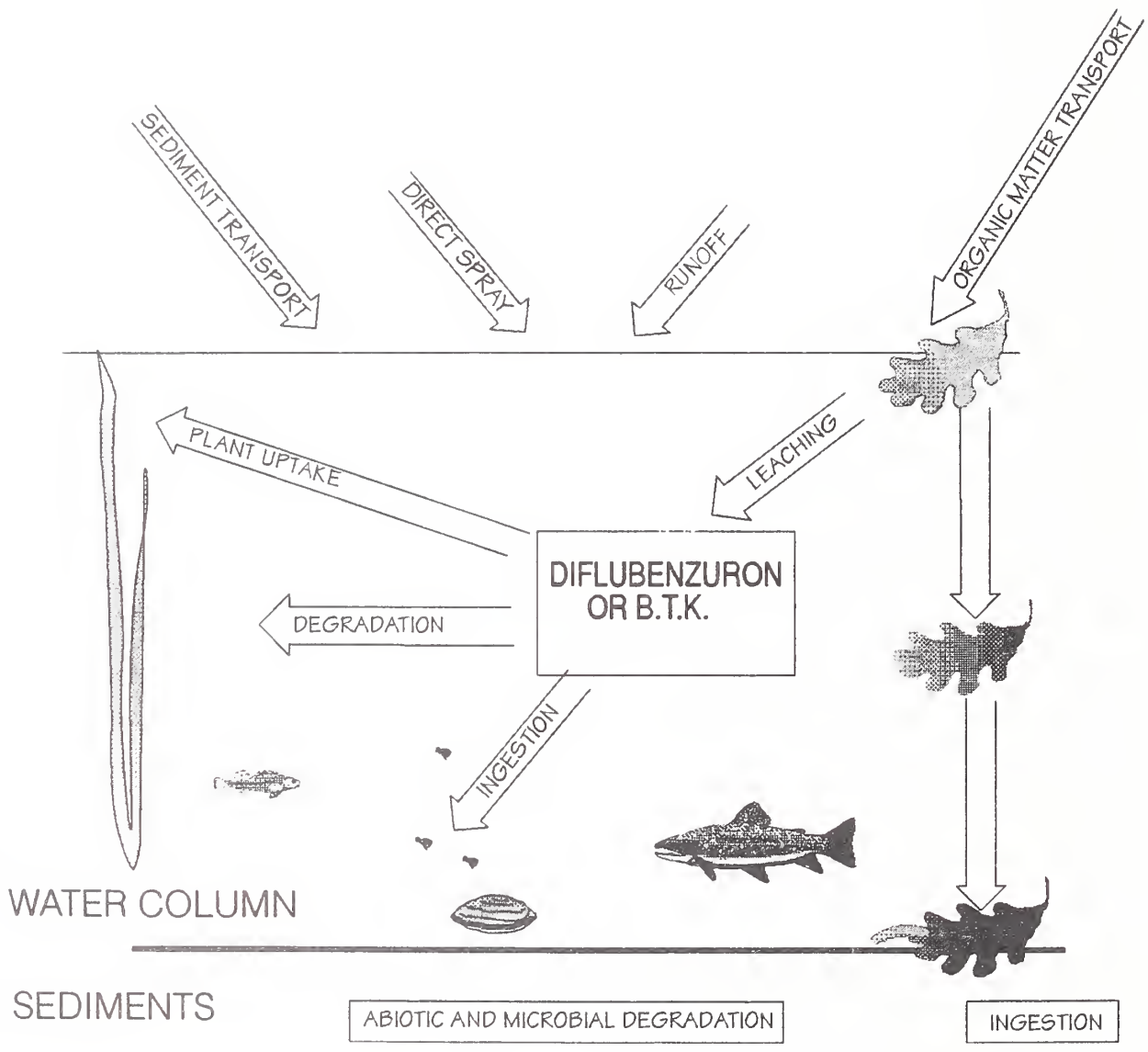

Figure 7-2. Fate and transport in the aquatic environment.

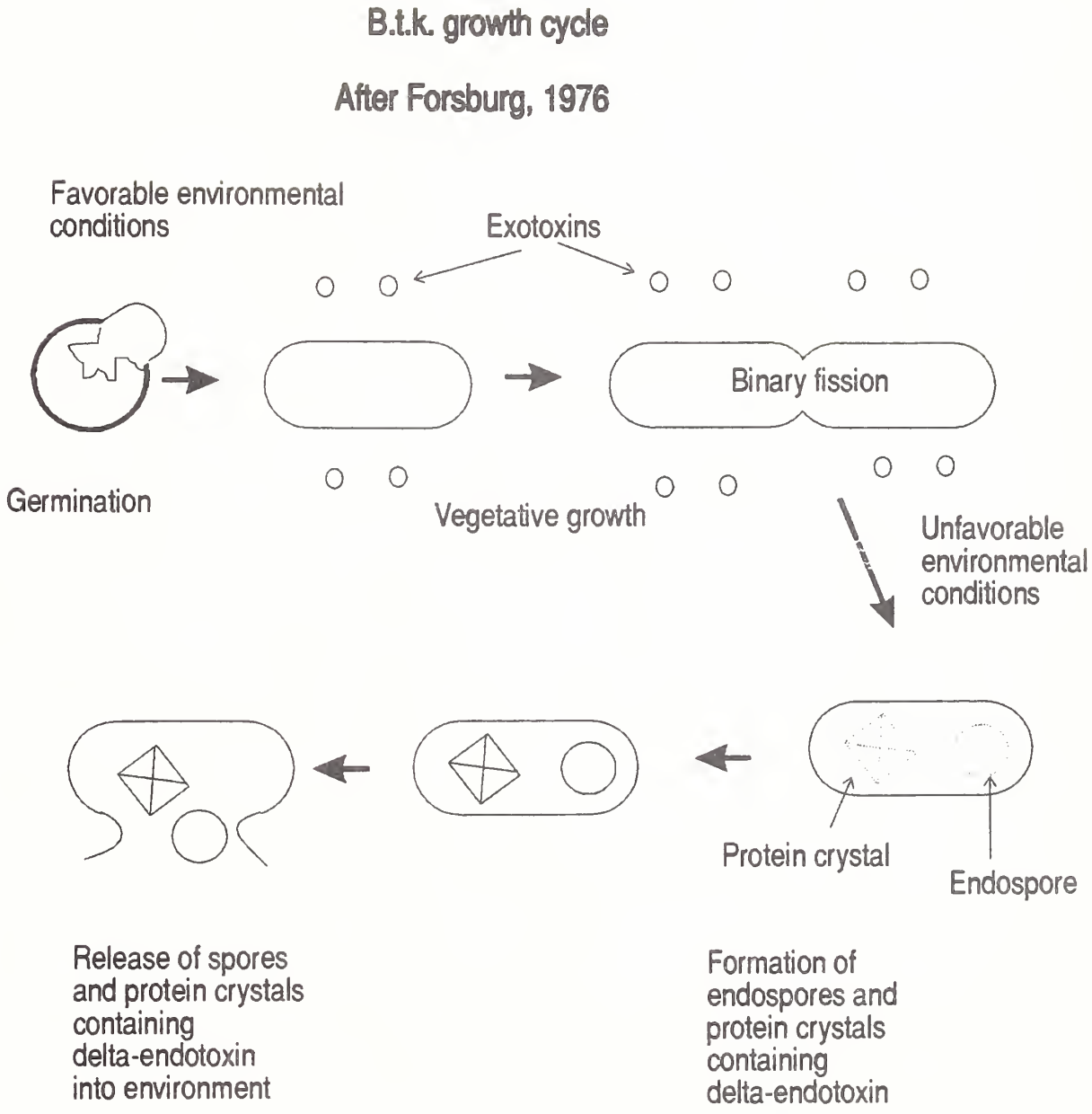

Figure 7-3. *B.t.k.* growth cycle.

Table 7-1. FSCBG Model Parameters

Parameters	Diflubenzuron	*B.t.*
Wind speed (mph)	5	5
Wind direction (°)	90	90
Temperature (°F)	66.7	66.7
Humidity (%)	20	20
Release height (m)	27.42	27.42
Emission rate (fl oz/acre)	2.4	2.4
Aircraft type	Bell 204	Bell 204
Swath width (feet)	200	200
Source diameter (m)	14.63	14.63
Aircraft speed (mph)	80.55	80.55
Density of carrier (g/cm^3)	1.23	1.23
Canopy type	15.23 m Story canopy	15.23 m Story canopy
Model type	Near Wake	Near Wake

Table 7-2. Summary of PRZM Input Parameters

Input Parameter	Northeastern Forest	Northern Georgia Forest	Northern Arkansas Forest	Source of Data or Remarks
Start date	1/1/64	1/1/64	1/1/64	
End date	12/31/83	12/31/83	12/31/83	
Meteorological data				
Pan evaporation factor	0.760	0.760	0.760	PIRANHA (PRZM companion program)
Snow melt factor	0.300	0.200	0.200	PIRANHA
Pan evaporation flag	0	0	0	PIRANHA
Min depth in which evaporation is extracted	15.00	17.00	16.00	PIRANHA
Duration of runoff hydrograph	5.800	5.90	5.70	PIRANHA
Crop data				
Initial crop number	1	1	1	
Surface condition before spraying	Residue	Residue	Residue	Leaf litter on ground
Area of field (ha)	1.0	1.0	1.0	
Number of crops	1	1	1	Only trees on plot
Maximum interception storage of crop (cm)	0.40	0.40	0.40	Dowd and others 1993
Maximum active root depth of crop (cm)	125.00	125.00	125.00	Roots extend below upper soil zones
Maximum aerial coverage at full canopy (%)	85	85	85	Assumed (Dowd and others 1993)
Surface condition directly after harvest (leaf fall)	Residue	Residue	Residue	Leaf litter on ground
Number of cropping periods	20	20	20	
Yearly date of crop emergence	4/22	3/13	3/13	
Yearly date of crop maturation	5/11	4/2	4/2	
Yearly date of crop harvest	10/15	11/15	11/15	
Foliar growth model	Linear	Linear	Linear	Leaves emerge linearly
Pesticide data				
Number of applications	20	20	20	One per year

Table 7-2, *continued ...*

Input Parameter	Northeastern Forest	Northern Georgia Forest	Northern Arkansas Forest	Source of Data or Remarks
Application date	5/1	3/23	3/23	
Depth of incorporation (cm)	0	0	0	No incorporation
Pesticide decay rate on foliage	0	0	0	No decay - worst case assumption
Foliar extraction coefficient for pesticide washoff	0.2	0.2	0.2	Assumed
Plant uptake	0	0	0	No uptake - worst case assumption
Soil data				
Soil name	Rockaway	Melvin	Taloka	
Soil texture	Sandy loam	Silty loam	Silty loam	
Hydrologic group	C	D	D	
Calculate erosion losses	Yes	Yes	Yes	
USLE soil erodibility K	0.24	0.42	0.42	PIRANHA
USLE topographic factor LS	1	1	1	PIRANHA
USLE supporting practice factor P	1	1	1	PIRANHA
Runoff curve number (CN) for antecedent soilwater condition II for fallow, crop, and residue	83 77 80	83 77 80	83 77 80	Agricultural Handbook 537
USLE cover management factor for fallow,crop, and residue	.01 .01 .01	.01 .01.01	.01 .01.01	Agricultural Handbook 537
Depth of soil core (cm)	100	100	100	Assumed
Soil hydraulics	Freely draining	Freely draining	Freely draining	Assumed
Number of soil horizons	3	2	2	PIRANHA
First horizon				
Thickness (cm)	10	18	72	PIRANHA
Bulk density of horizon	1.35	1.8	1.5	PIRANHA
Hydrodynamic dispersion(cm^2/day)	0	0	0	PIRANHA

Table 7-2, *continued ...*

Input Parameter	Northeastern Forest	Northern Georgia Forest	Northern Arkansas Forest	Source of Data or Remarks
Pesticide decay rate (day^{-1})	0.035	0.035	0.035	Estimated Wauchoppe and others 1991, Hartley and Kidd 1983)
Initial soil water content(cm^3/cm^3)	0.205	0.284	0.282	PIRANHA
Field capacity (cm^3/cm^3)	0.205	0.284	0.282	PIRANHA
Wilting point (cm^3/cm^3)	0.065	0.104	0.122	PIRANHA
Organic carbon %	0.58	0.29	0.29	PIRANHA
K_d	58.005	29.002	29.002	PIRANHA
Second horizon				
Thickness (cm)	86	82	28	PIRANHA
Bulk density of horizon	1.75	1.8	1.7	PIRANHA
Hydrodynamic dispersion(cm^2/day)	0	0	0	PIRANHA
Pesticide decay rate (day^{-1})	0.035	0.035	0.035	Estimated Wauchoppe and others 1991, Hartley and Kidd 1983)
Initial soil water content(cm^3/cm^3)	0.106	0.281	0.324	PIRANHA
Field capacity (cm^3/cm^3)	0.106	0.281	0.324	PIRANHA
Wilting point (cm^3/cm^3)	0.046	0.101	0.204	PIRANHA
Organic carbon %	0.116	0.174	0.174	PIRANHA
K_d	11.601	17.401	17.401	PIRANHA
Third horizon				
Thickness (cm)	4			PIRANHA
Bulk density of horizon	1.55			PIRANHA
Hydrodynamic dispersion(cm^2/day)	0			PIRANHA
Pesticide decay rate (day^{-1})	0.02			Estimated Wauchoppe and others 1991, Hartley and Kidd 1983)
Initial soil water content(cm^3/cm^3)	0.081			PIRANHA

Table 7-2, *continued ...*

Input Parameter	Northeastern Forest	Northern Georgia Forest	Northern Arkansas Forest	Source of Data or Remarks
Field capacity (cm^3/cm^3)	0.081			PIRANHA
Wilting point (cm^3/cm^3)	0.031			PIRANHA
Organic carbon %	0.058			PIRANHA
K_d	5.8			PIRANHA

Table 7-3. Diflubenzuron Estimated Environmental Residues

Application Rate 1.0 oz. a.i. /ac	
Upper canopy leaves mean residue ($\mu g/cm^2$)	0.0314
Upper canopy leaves standard deviation ($\mu g/cm^2$)	0.0226
Lower canopy leaves mean residue ($\mu g/cm^2$)	0.0206
Lower canopy leaves standard deviation ($\mu g/cm^2$)	0.0148
Soil/litter/pavement surface ($\mu g/cm^2$)	0.0160
Upper and lower canopy leaves mean residue in autumn ($\mu g/cm^2$)	0.0117
Litter in autumn; pre-spray the following spring	0.0149
Litter post-spray the following spring	0.0309
Application Rate 0.5 oz. a.i. /ac	
Upper canopy leaves mean residue ($\mu g/cm^2$)	0.0146
Upper canopy leaves standard deviation ($\mu g/cm^2$)	0.0116
Lower canopy leaves mean residue ($\mu g/cm^2$)	0.0096
Lower canopy leaves standard deviation ($\mu g/cm^2$)	0.0076
Soil/litter/pavement surface ($\mu g/cm^2$)	0.0075
Upper and lower canopy leaves mean residue in autumn ($\mu g/cm^2$)	0.0056
Litter in autumn; pre-spray the following spring	0.0071
Litter post-spray the following spring	0.0146
Application Rate 0.33 oz. a.i. /ac	
Upper canopy leaves mean residue ($\mu g/cm^2$)	0.0090
Upper canopy leaves standard deviation ($\mu g/cm^2$)	0.0076
Lower canopy leaves mean residue ($\mu g/cm^2$)	0.0059
Lower canopy leaves standard deviation ($\mu g/cm^2$)	0.0050
Soil/litter/pavement surface ($\mu g/cm^2$)	0.0046
Upper and lower canopy leaves mean residue in autumn ($\mu g/cm^2$)	0.0034
Litter in autumn; pre-spray the following spring	0.0043
Litter post-spray the following spring	0.0089

Table 7-3, *continued ...*

Application Rate 0.25 oz. a.i. /ac	
Upper canopy leaves mean residue ($\mu g/cm^2$)	0.0068
Upper canopy leaves standard deviation ($\mu g/cm^2$)	0.0058
Lower canopy leaves mean residue ($\mu g/cm^2$)	0.0045
Lower canopy leaves standard deviation ($\mu g/cm^2$)	0.0038
Soil/litter/pavement surface ($\mu g/cm^2$)	0.0035
Upper and lower canopy leaves mean residue in autumn ($\mu g/cm^2$)	0.0026
Litter in autumn; pre-spray the following spring	0.0033
Litter post-spray the following spring	0.0068

Table 7-4. Multiple Applications of Diflubenzuron within the Same Year

0.5 oz ai/ac		
Upper canopy residues ($\mu g/cm^2$)	0.0146	0.0249
Lower canopy residues ($\mu g/cm^2$)	0.0096	0.0164
Soil/litter surface residues ($\mu g/cm^2$)	0.0075	0.0113
0.33 oz ai/ac		
Upper canopy residues ($\mu g/cm^2$)	0.0090	0.0154
Lower canopy residues ($\mu g/cm^2$)	0.0059	0.0101
Soil/litter surface residues ($\mu g/cm^2$)	0.0046	0.0069
0.25 oz ai/ac		
Upper canopy residues ($\mu g/cm^2$)	0.0068	0.0117
Lower canopy residues ($\mu g/cm^2$)	0.0045	0.0077
Soil/litter surface residues ($\mu g/cm^2$)	0.0035	0.0053

Table 7-5. Estimated Diflubenzuron Concentrations in Various Environmental Components

1.0 oz/ac (70.02 g/ha) application rate

	Spring Concentrations in Diet	Autumn Concentrations in Diet
Terrestrial Concentrations (ppm)		
Upper canopy leaves	1.903	0.784
Lower canopy leaves	1.370	0.754
Understory Forb leaves	1.333	0.823
Understory Grass leaves	1.778	1.097
Plant juices (sap)	0[1]	0[1]
Pollen/nectar	0[1]	0[1]
Wood	0[1]	0[1]
Litter - fallen leaves	3.037	2.700
Soil	0.013	0[2]
Aquatic Concentrations (μg/L)		
Pond after runoff in residential forest		
Open water column - maximum estimate	6.21	0[2]
Sediments	0.124	0[2]
4-chloroaniline	0.621	0[2]
Pond after runoff in natural forest		
Open water column - maximum estimate	1.22	0[2]
Sediments	0.0244	0[2]
4-chloroaniline	0.122	0[2]
Streams		
Directly sprayed stream	16.01	0[2]
Stream after runoff in residential forest	13.14	0[2]
Stream after runoff in natural forest	2.76	0[2]
Leaf packs (ppm)	-[3]	2.70

[1] Not expected to contain diflubenzuron residues
[2] Diflubenzuron or 4-chloroaniline will have degraded or been transported out of this habitat by autumn
[3] Leaf packs containing diflubenzuron coated leaves are not expected to form until autumn

Table 7-6. Estimated Diflubenzuron Concentrations in Various Environmental Components

0.5 oz/ac (35.01 g/ha) application rate

	Spring Concentration in Diet	Autumn Concentration in Diet
Terrestrial Concentrations (ppm)		
Upper canopy leaves	1.125	0.426
Lower canopy leaves	0.739	0.418
Forb leaves	0.790	0.487
Grass leaves	1.05	0.650
Plant juices (sap)	0[1]	0[1]
Nectar	0[1]	0[1]
Wood	0[1]	0[1]
Litter - fallen leaves	1.798	1.528
Soil	0.008	0[2]
Aquatic Concentrations (μg/L)		
Pond after runoff in residential forest		
Open water column - maximum estimate	5.97	0[2]
Sediments	0.119	0[2]
4-chloroaniline	0.597	0[2]
Pond after runoff in natural forest		
Open water column - maximum estimate	0.90	0[2]
Sediments	0.018	0[2]
4-chloroaniline	0.090	0[2]
Streams		
Directly sprayed stream	8.01	0[2]
Stream after runoff in residential forest	12.50	0[2]
Stream after runoff in natural forest	2.02	0[2]
Leaf packs (ppm)	-[3]	1.528

[1] Not expected to contain diflubenzuron residues
[2] Diflubenzuron or 4-chloroaniline will have degraded or been transported out of this habitat by autumn
[3] Leaf packs containing diflubenzuron coated leaves are not expected to form until autumn

Table 7-7. Estimated Diflubenzuron Concentrations in Various Environmental Components

0.33 oz/ac (g/ha) application rate

	Spring Concentration in Diet	Autumn Concentration in Diet
Terrestrial Concentrations (ppm)		
Upper canopy leaves	0.545	0.225
Lower canopy leaves	0.358	0.197
Forb leaves	0.383	0.237
Grass leaves	0.511	0.315
Plant juices (sap)	0[1]	0[1]
Nectar	0[1]	0[1]
Wood	0[1]	0[1]
Litter - fallen leaves	0.873	0.740
Soil	0.004	0[2]
Aquatic concentrations (μg/L)		
Pond after runoff in residential forest		
Open water column - maximum estimate	5.97	0[2]
Sediments	0.119	0[2]
4-chloroaniline	0.597	0[2]
Pond after runoff in natural forest		
Open water column - maximum estimate	0.90	0[2]
Sediments	0.018	0[2]
4-chloroaniline	0.090	0[2]
Streams		
Directly sprayed stream	5.38	0[2]
Stream after runoff in residential forest	12.50	0[2]
Stream after runoff in natural forest	2.02	0[2]
Leaf packs (ppm)	-[3]	0.740

[1] Not expected to contain diflubenzuron residues
[2] Diflubenzuron or 4-chloroaniline will have degraded or been transported out of this habitat by autumn
[3] Leaf packs containing diflubenzuron coated leaves are not expected to form until autumn

Table 7-8. Estimated Diflubenzuron Concentrations in Various Environmental Components

0.25 oz/ac (17.50 g/ha) application rate

	Spring Concentration in Diet	Autumn Concentration in Diet
Terrestrial Concentrations (ppm)		
Upper canopy leaves	0.412	0.170
Lower canopy leaves	0.273	0.150
Forb leaves	0.292	0.180
Grass leaves	0.389	0.240
Plant juices (sap)	0[1]	0[1]
Nectar	0[1]	0[1]
Wood	0[1]	0[1]
Litter - fallen leaves	0.664	0.562
Soil	0.003	0[2]
Aquatic concentrations (μg/L)		
Pond after runoff in residential forest		
Open water column - maximum estimate	5.87	0[2]
Sediments	0.117	0[2]
4-chloroaniline	0.587	0[2]
Pond after runoff in natural forest		
Open water column - maximum estimate	0.792	0[2]
Sediments	0.016	0[2]
4-chloroaniline	0.070	0[2]
Streams		
Directly sprayed stream	4.00	0[2]
Stream after runoff in residential forest	12.18	0[2]
Stream after runoff in natural forest	1.77	0[2]
Leaf packs (ppm)	-[3]	0.562

[1] Not expected to contain diflubenzuron residues
[2] Diflubenzuron or 4-chloroaniline will have degraded or been transported out of this habitat by autumn
[3] Leaf packs containing diflubenzuron coated leaves are not expected to form until autumn

Table 7-9. *B.t.k.* Estimated Environmental Residues

	drops/cm^2	mg/cm^2	IU/cm^2
24 BIU/ac			
Upper Canopy Leaves Mean	2.17	0.00873	92.58
Upper Canopy Standard Deviation	1.03	0.00260	27.59
Lower Canopy Leaves Mean	1.10	0.00518	54.89
Lower Canopy Standard Deviation	0.61	0.00162	17.17
Soil/litter surface	0.98	0.00434	46.00
Soil/Litter surface Standard Deviation	0.56	0.00138	14.61
Directly sprayed stream (2 ft)	-[1]	500.14[2]	5301[3]
Directly sprayed pond (6 ft)	-[1]	138.08[2]	1463[3]
40 BIU/ac			
Upper Canopy Leaves Mean	3.59	0.01447	153.38
Upper Canopy Standard Deviation	1.70	0.00431	45.71
Lower Canopy Leaves Mean	1.83	0.00858	90.91
Lower Canopy Standard Deviation	1.01	0.00265	28.13
Soil/litter surface Mean	1.63	0.00919	97.40
Soil/Litter surface Standard Deviation	0.92	0.00228	24.20
Directly sprayed stream (2 ft)	-[1]	828.56[2]	8778[3]
Directly sprayed pond (6 ft)	-[1]	228.74[2]	2674[3]

[1] *B.t.k.* drops are no longer discrete entities when mixed with the water column and have no meaning in the aquatic environment
[2] μg/L
[3] IU/L

Section 8
Exposure Assessment

An exposure assessment is an estimate of the amount of insecticide an organism is likely to encounter, including an estimate of the period of time the insecticide will be available in the organism's environment. This section presents exposure estimates for each insecticide used in the National Gypsy Moth Management Program. Environmental concentrations of the insecticides calculated in Section 7 are used to estimate exposure to diflubenzuron and *B.t.k.* through ingestion, and exposure to dichlorvos through ingestion, dermal absorption, and inhalation.

8.1. Diflubenzuron

Diflubenzuron is aerially applied to forested areas at various rates: 1 a.i. oz/ac, 0.5 a.i. oz/ac, 0.33 a.i. oz/ac, 0.25 a.i. oz/ac (70.02 g/ha, 35.01 g/ha, 23.34 g/ha, 17.51 g/ha). Most of the diflubenzuron will be deposited on the tree canopy (greater than 70 percent) with a lesser amount deposited on the soil surface (see Section 7).

Organisms will be exposed dermally to diflubenzuron through direct spray and through contact with residues on vegetation and soil. They will also be exposed through ingestion of residues with their food items. Dietary exposure has been calculated by determining the concentration of diflubenzuron in food items (see Section 7). This approach was taken rather than explicitly calculating the mass of diflubenzuron ingested because of uncertainty about the amount of food ingested per day for most nontarget organisms and because the greater body of literature reports toxicological test data as dietary concentrations rather than as whole body burdens (mg diflubenzuron/kg body weight). Concentrations of diflubenzuron were determined for tree, forb, and grass leaves, soil, leaf litter, and insects.

8.1.1. Exposure Through Ingestion of Diflubenzuron

Organisms eating leaves, leaf litter, or soil in the treated area will ingest diflubenzuron. Diflubenzuron binds readily with organic material and persists throughout the growing season on leaves. Exposure to diflubenzuron in litter continues throughout two growing seasons. In the developed forest ecosystem this would include the leaves of turf grasses on the ground surface. Although diflubenzuron residues on canopy leaves have been measured throughout the growing season (Wimmer and others, 1993, Martinet and others, 1987, Mutanen and others,

1988), similar studies have not been conducted on turf grasses, thus creating a data gap for diflubenzuron degradation rates on turf grasses. As a conservative estimate, diflubenzuron persistence on turf grasses is assumed to be similar to its persistence on canopy leaves. This is conservative because grasses grow from their base continuously, and in most residential lawns, grass (with its residue) would be cut, raked, and removed, or cut grass would be left on the lawn or soil surface. Due to cutting, exposure to diflubenzuron on turf grasses in the developed forest ecosystem will be much shorter (one to several weeks) than exposure to diflubenzuron on subcanopy vegetation in the undeveloped forest ecosystem.

Organisms exposed to diflubenzuron through ingestion of leaves would be restricted to those organisms consuming the entire leaf or the leaf's outer surface. This includes a wide variety of insects, as well as some vertebrates such as mammals (deer, groundhogs, mice), birds, and reptiles (box turtles). Small omnivorous finches and sparrows feed upon the leaves, buds, and flowers of forbs, shrubs, and trees at certain times of the year, commonly in the spring when the vegetation first appears (Bent, 1968). Although green foliage does not contribute significantly to the annual intake of most of these species, fresh vegetable matter has been found to constitute up to 50 percent of individual stomach samples of some species (e.g. white-crowned sparrow) (Bent, 1968).

In the developed forest ecosystem, grazing animals such as cows, horses, sheep, or goats would be exposed if they consumed turf grass within the sprayed area. Organisms eating only the outer surface of the leaf will have higher concentrations of diflubenzuron in their diet than organisms consuming the entire leaf. Concentrations of diflubenzuron used in this risk assessment and reported in Section 7 are based on consumption of the upper and lower leaf surfaces, including the leaf material between these surfaces (see Table 7-5). Over the growing season the concentration of diflubenzuron will decrease as the leaves increase in size and weight and diflubenzuron degrades.

Sucking insects (gall-forming insects, phytophagous mites, aphids, coccids, aleyrodids, psyllids), leafminers, or stemborers feed on plant juices (phloem) and are not expected to be exposed to diflubenzuron in their diet. Due to the persistence of diflubenzuron throughout the growing season, insects that change their method of feeding as they mature from sucking, mining or boring to consumption of the leaf's outer surface, will be exposed during part of their life cycle. Pollen and nectar-feeding organisms (bees and adults of some butterflies and moths) and wood-feeding organisms (termites and specialized forms of beetles, ants, and caterpillars) are also not expected to be exposed to diflubenzuron in their diets.

Terrestrial soil arthropods consume dead or decomposing organic matter, fallen leaves or branches of trees, and the microorganisms living on these items. Soil arthropods that specialize in eating fallen leaves, including many lepidopteran species, would be

exposed to two distinct pulses of diflubenzuron during the year: immediately following application and later when leaves fall during autumn. Exposure to the first pulse lasts until leaf drop when the second pulse occurs, although concentrations of diflubenzuron in litter decline throughout the summer (from 1 ppm to 0.2 ppm, Wimmer, 1994). Exposure to diflubenzuron fallen leaves in autumn continues at about the same level throughout the winter, as diflubenzuron does not degrade rapidly at low temperature. Lepidopteran species such as *Abagratis*, *Rhychagrotis*, *Anomogynen*, *Protoboarmia*, and *Euchlaena* will consume diflubenzuron in leaf litter throughout the winter (Schweitzer, 1994 personal communication). Exposure to the second pulse continues until the following autumn when newly fallen diflubenzuron-free leaves dilute the diflubenzuron concentration in litter (Wimmer, 1994). Organisms that consume are assumed to be exposed only if they consumed soil from the upper (1 cm) layer, as diflubenzuron is not predicted to leach or become incorporated into the soil (See Section 7). Preliminary results from the Wimmer laboratory at West Virginia University suggest that diflubenzuron may not move significantly into the underlying soil when levels drop off in the ground litter; however, only a small percentage of samples had been analyzed at this writing (1994; Wimmer, unpublished data).

8.1.2. Exposure Through Dermal Contact of Diflubenzuron

Organisms that move through vegetation and come into contact with the soil or litter surface, leaves, or turf grass will be exposed to diflubenzuron. Due to the strong affinity of diflubenzuron for organic material, residues will probably not be easily dislodged from leaves, grass, or organic matter in the litter or soil. More data need to be collected to fully support this assumption, however. Diflubenzuron generally is assumed to be more toxic to invertebrates when ingested than when applied dermally (Grosscurt, 1978, Retnakaran and others, 1979), with a few exceptions, most notably dipterans and a lepidopteran, *Spodoptera littoralis*, the Egyptian cotton leafworm. Because diflubenzuron is primarily a stomach poison, a quantitative assessment of dermal exposure is not given.

8.1.3. Exposure to Diflubenzuron in Aquatic Environments

In the aquatic environment organisms are simultaneously exposed to diflubenzuron through ingestion, dermal contact, and inhalation. Diflubenzuron concentrations vary within a water body due to differences in microhabitat. Therefore, exposure to diflubenzuron in various microhabitats within the aquatic environment was examined.

Planktonic organisms and most fish are exposed to concentrations of diflubenzuron in the water column. Benthic invertebrates are exposed to diflubenzuron adsorbed to sediment particles or dissolved in sediment pore spaces. Benthic stream invertebrates

sediment particles or dissolved in sediment pore spaces. Benthic stream invertebrates are exposed either to the diflubenzuron concentration in the water column if they are located on upper surfaces of rocks, or to the smaller concentration of diflubenzuron in sediments.

Aquatic organisms that consume detritus are exposed primarily during autumn when large quantities of leaves are transported to streams. The concentration of diflubenzuron in leaf packs is assumed to be similar to the concentration in leaf litter.

8.1.4. Summary

The highest diflubenzuron residues occur on tree canopy leaves, with much lower residues being deposited on the ground surface (Section 7). Organisms consuming leaves are exposed to greater diflubenzuron concentrations than ground-dwelling organisms (see Table 7-5). Diflubenzuron exposure to ground-dwelling organisms differs between the two ecosystems (undeveloped forest and developed forest) due to increased amounts of impervious surfaces and turf grass cultivation in the developed forest ecosystem and the generalization that the ground surface of the undeveloped forest ecosystem is covered with leaf litter except for occasional small patches of bare soil. Because diflubenzuron residues would be removed with cut turf grass, ground-dwelling organisms in the developed forest ecosystem will be exposed to diflubenzuron for a shorter time period than ground-dwelling organisms in the undeveloped forest. As application rates increase, so does exposure to diflubenzuron in terrestrial and aquatic environments.

8.2. 4-Chloroaniline (Degradation Product of Diflubenzuron)

Organisms exposed to ingestion of diflubenzuron will also be exposed to 4-chloroaniline; however, 4-chloroaniline concentrations are much lower than diflubenzuron concentrations. Exposure to 4-chloroaniline also occurs in the soil and litter, affecting the same set of soil arthropods as with ingestion of diflubenzuron. If 4-chloroaniline behaves in the water column as does diflubenzuron (Kingsbury and others, 1987), then only organisms in the open water column would be exposed to concentrations of 4-chloroaniline exceeding 0.5 μg/L. Exposure to 4-chloroaniline is greater in developed forest ecosystems than undeveloped forest ecosystems due to increased runoff of diflubenzuron from impervious surfaces.

8.3. *Bacillus thuringiensis* var. *kurstaki*

Aerial application of *B.t.k.* results in residues on both the tree or shrub canopy and the soil or litter surface. Ingestion is the only route of exposure to *B.t.k.* considered in this risk assessment. Spores and crystals cannot enter the gut through either the

dermal or inhalation routes of exposure. Without exposure in an alkaline gut, *B.t.k.* exhibits no toxic properties to nontarget organisms.

8.3.1. Exposure Through Ingestion of *B.t.k.*

As with diflubenzuron, terrestrial organisms consuming leaves, litter, or soil will be exposed through their diet. The same groups of organisms will be exposed to *B.t.k.* as discussed for diflubenzuron: organisms eating portions of both upper and lower leaf surfaces, organisms eating litter or the uppermost layer of the soil, and organisms eating dead or dying caterpillars which had consumed spores or crystals. Dietary exposure to *B.t.k.* was calculated as exposure to either *B.t.k.* drops rather than to *B.t.k.* concentration in the diet (see Table 8-1). The toxicological data available for *B.t.k.* were primarily reported in IU/ml of diet; however, neither the exact concentration of *B.t.k.* in the diet nor the exact surface residues were reported. Only one study reported *B.t.k.* toxicity in units of *B.t.k.* mass per body weight, and no studies reported dietary concentrations. Due to the lack of toxicological data, exposure to drops of *B.t.k.* was calculated. For calculation of risk, the IU/cm^2 are also given. Exposure to *B.t.k.* among ground-dwelling organisms would be of shorter duration in the developed forest ecosystem where grass containing *B.t.k.* is cut and removed than in the undeveloped forest ecosystem. However, the difference in length of exposure between these two ecosystems would be small due to short half-life of *B.t.k.*. Exposure to *B.t.k.* in the terrestrial environment (or vegetation, litter, or soil) is of relatively short duration (less than 300 hours to reduce *B.t.k.* to 1/1000 of its original concentration given a half-life of 30 hours).

8.3.2. Exposure to *B.t.k.* in Aquatic Environments

Organisms living in the open water column are primarily exposed to *B.t.k.* when it is sprayed directly on water bodies. There is uncertainty about the duration of exposure to *B.t.k.* due to the small number of studies conducted under natural conditions. Monitoring studies suggest that exposure would not exceed two weeks in duration (Menon and de Mestral, 1985).

8.3.3. Summary

The highest *B.t.k.* residues occur in the upper tree canopy leaves, with lower residues on subcanopy vegetation and the ground surface (see Section 7). Organisms consuming upper canopy leaves are exposed to more *B.t.k.* than ground-dwelling organisms consuming subcanopy vegetation. The more upper canopy leaves consumed per body weight, the more an organism will be exposed. Ground-dwelling organisms consuming subcanopy vegetation could be exposed to *B.t.k.* for shorter

duration in the developed forest ecosystem than the undeveloped forest ecosystem if grass is cut and clippings removed shortly after spraying. As application rates increase, so does exposure.

8.4. Disparlure

Aerial application of disparlure results in a distribution in the environment similar to diflubenzuron and *B.t.k.*. Disparlure is found in the canopy, subcanopy, and on the forest floor. Organisms eating leaves, litter, or soil will be exposed to Disparlure in their diets. Organisms coming in contact with Disparlure will also be exposed dermally. The maximum exposure to Disparlure in any of these environments would be the expected residue of 0.741 $\mu g/cm^2$ on a flat surface. Aquatic organisms would be exposed to Disparlure concentrations of 7.41 $\mu g/L$ in a 1 m water body. Due to the low toxicity of Disparlure for vertebrates and its unknown effects on invertebrates, a quantitative exposure analysis was not performed.

8.5. Nucleopolyhedrosis Virus

Nucleopolyhedrosis virus (NPV) when aerially applied is distributed on the canopy, with some penetrating to the soil or litter surface. The same suite of organisms exposed to ingestion of diflubenzuron or *B.t.k.* will also be exposed to NPV. In addition, organisms in contact with vegetation or the soil or litter surface will be dermally exposed to NPV. The maximum exposure to NPV would be the expected residue of 12355 PIB/cm^2 on a flat surface. A quantitative exposure analysis of NPV was not performed due to its host specificity for gypsy moth and its lack of effect on other nontarget species.

8.6. Dichlorvos

Exposure to dichlorvos is limited to those organisms entering a trap containing the dichlorvos-Vaportape II, or encountering the Vaportape II on the forest floor from a broken trap. Dichlorvos can be inhaled, ingested, or absorbed through dermal contact.

Ingestion of dichlorvos is unlikely to occur by the organisms caught within the trap, mainly flying insects. Because the dichlorvos is impregnated into the Vaportape, it will not easily be ingested unless the tape is also ingested. Most insects within the trap will succumb to inhaling dichlorvos before they can ingest it. Ingestion is primarily a concern for vertebrates discovering a Vaportape from a broken trap. The tape is not palatable and it will probably not be eaten by most organisms. Black bears

have damaged traps during past gypsy moth management programs. The risk assessment considers consumption of the Vaportape by bears that may disturb a trap.

Dermal contact with the Vaportape will result in exposure to dichlorvos for organisms entering the trap. The majority of these organisms will be flying insects. Organisms encountering the Vaportape from a broken trap will also be exposed. These would include soil or litter arthropods, small vertebrates, and black bears. Inhalation of dichlorvos would primarily be restricted to organisms caught within the trap, mainly flying insects. Organisms encountering a Vaportape from a broken trap would be exposed if they stood directly over the Vaportape or picked it up in their mouths.

8.7. References

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Table 8-1. Exposure to *B.t.k.* in terrestrial diet items and aquatic resources

40 BIU/ac application rate	Residues in diet items drops/cm	Residues in diet items IU/cm
Upper canopy leaves	3.59	153.38
Lower canopy leaves	1.83	90.91
Forb leaves	1.63	97.40
Grass leaves	1.63	97.40
Plant juices (sap)	0[1]	0[1]
Nectar	0[1]	0[1]
Wood	0[1]	0[1]
Litter - fallen leaves	1.63	97.40
Soil	1.63	97.40
24 BIU/ac application rate	**Residues in diet items drops/cm**	**Residues in diet items IU/cm**
Upper canopy leaves	2.17	92.58
Lower canopy leaves	1.03	27.59
Forb leaves	0.98	46.00
Grass leaves	0.98	46.00
Plant juices (sap)	0[1]	0[1]
Nectar	0[1]	0[1]
Wood	0[1]	0[1]
Litter - fallen leaves	0.98	46.00
Soil	0.98	46.00
***B.t.k.* concentration in water (IU/L)**	**24 BIU/ac**	**40 BIU/ac**
Directly sprayed stream (0.76 m deep)	5301	8778
Directly sprayed pond (2 m deep)	1463	2674

[1] *B.t.k.* is not expected in these substances

Section 9
Risk Analysis

In this section the effect of each of the insecticides and the pheromone on the endpoints is discussed. Endpoints are components of the ecosystem (see Sections 1 and 6). Within the discussion of each treatment, the nontarget species will be addressed first, because many of the endpoints are influenced by the actions of nontarget species. The effects of the treatments and the methods used to analyze effects are discussed.

9.1. Diflubenzuron

9.1.1. Methodology for Nontarget Species Endpoint

Comparison of the dietary LC_{50} values of various organisms with estimated environmental concentrations of diflubenzuron in diet items suggests that species at risk are primarily invertebrates (Figures 9-1, 9-2). Species at risk were identified using a screening index shown below:

$$SI = \frac{DFB}{Toxicological\ Benchmark * SF}$$

SI = Screening Index
DFB = Diflubenzuron Concentration
Toxicological Benchmark = LC_{50} , EC_{50}
SF = Safety factor

Safety factors were 1/5 for terrestrial organisms and 1/10 for aquatic organisms. These are based on values used by the EPA in screening indices evaluating pesticides during the registration process (Urban and Cook, 1986). As a conservative assumption, all of the organism's diet was assumed to be contaminated. Screening indices were developed for a variety of different types of organisms, including all major groups identified as potentially exposed to ingesting diflubenzuron (Section 8 and Table 9-1). Organisms were considered to be potentially at risk if the environmental concentrations of diflubenzuron exceeded those necessary to cause the benchmark response given the safety factor (for example, screening index exceeds one).

The screening index assumes there is no variability in either the diflubenzuron concentration in the environment or the toxicological response to diflubenzuron.

However, residues of diflubenzuron in leaves, soil, water, and other environmental media vary spatially within the spray area due to numerous site-specific factors and imprecisions inherent with aerial application; variability in specific leaf and soil weight contribute to the variability of diflubenzuron concentrations in leaves, litter, and soil. Toxicological responses (LC_{50}, EC_{50}) also vary depending upon the age of the organism, whether the organism is about to molt, and genetic variability. To account for these sources of variability, a Monte Carlo simulation of the screening index was conducted using randomly selected values from frequency distributions for the toxicological benchmarks and for the parameters used to calculate diflubenzuron concentration in leaves, litter, soil, and water. Monte Carlo simulation is an iterative process in which random variates are selected from probability distributions to provide parameter values used in the model at each iteration, in this case, the screening index. This simulation considers variability in the diflubenzuron deposition, leaf, forb, grass, soil and litter weights, and LC_{50} value. The type of distribution and its statistics of dispersion are given in Table 9-2, for each of the parameters simulated using Monte Carlo techniques.

Spatial distribution of diflubenzuron residues was simulated using the mean and standard deviation of residues determined by the Forest Service Cramer Berry Grim model in Section 7. Residues were assumed to be log normally distributed based on monitoring data for diflubenzuron and other aerially applied insecticides (Bryant and others, 1987; Mierzejewski and others, 1993). The proportion of the original diflubenzuron residue remaining on leaves from spring to autumn was based on diflubenzuron persistence in a West Virginia forest (Wimmer and others, 1993), as was the proportion of the residue remaining on the litter over the winter months (Wimmer, 1994). Diflubenzuron concentrations in water were assumed to range from one-half to one and one-half the value estimated by the surface water model (Section 7). Both diflubenzuron concentrations in sediments and 4-chloroaniline concentrations are calculated based on diflubenzuron concentrations in water and therefore are distributed similarly to diflubenzuron concentrations in water.

The specific leaf weight is an important factor determining the diflubenzuron concentration of leaves. The distribution of specific leaf weights was assumed to be similar to spring leaves collected in Prince William County, Virginia. (Rockwood, 1994). Forbs and grasses were assumed to have the same distribution of specific leaf weights with forbs having a mean two-thirds that of trees, and grasses one-half that of trees.

Toxicological data from taxonomically similar species were pooled to obtain an estimate of variability between species and studies. Although many studies report LC_{50} values in parts per million diflubenzuron in the diet, not all of the studies controlled the concentration of diflubenzuron ingested by the test organisms. Studies where diflubenzuron was sprayed on diet items without measuring the diflubenzuron concentration of the diet item are difficult to interpret because the results cannot be

related to a known dose. Comparison of these diet spray studies with studies explicitly measuring dietary diflubenzuron concentrations shows higher LC_{50} values for the diet sprays, often more than twice as high as studies measuring the actual concentrations. Therefore, diet spray studies were not used to estimate toxicological benchmarks unless there were no other data. When the LC_{50} or EC_{50} values from diet spray studies were used, they were reduced by half as a conservative assumption. This assumption was based on observed differences between diet spray studies and studies measuring dietary diflubenzuron concentrations for lepidopterans, the only group in which a large number of both types of studies were performed. The studies used to estimate distributions of either LC_{50} or EC_{50} data used in the Monte Carlo simulation are given in Table 9-3.

Toxicological data are generally lognormally distributed because within a population there will be more younger, sensitive individuals than older, more resistant ones resulting in a distribution with few higher LC_{50} values and many smaller ones. However, for groups other than lepidopterans, not enough data were available to describe the distribution statistically. For those groups having enough toxicological studies to estimate the likeliest value and a range, the triangular distribution was used. For groups which have fewer toxicological studies, the uniform distribution was used.

The Monte Carlo simulation was run for 4000 iterations using a Latin Hypercube sampling technique to select random numbers from the probability distributions. The mean square error was less than 0.001 for all of the model parameters generated using Monte Carlo techniques. The probability that the screening index exceeds one was taken to be equivalent to the proportion of Monte Carlo iterations producing screening index values over one. Species with a greater than 1 percent risk of exceeding the screening index were considered potentially at risk from diflubenzuron and are shown in Table 9-1.

The percentage of the population at risk from diflubenzuron was determined using the relationship between diflubenzuron concentrations and population response defined by probit analysis. In probit analysis the log of the diflubenzuron concentration is linearly related to a probit value which corresponds to some level of population response. Knowing the slope of the probit line and the LC_{50}, the diflubenzuron concentration can be related to a probit value by using the equation for a straight line. The same sources of variability exist for this calculation as for the screening index previously described. In addition, the slope of the probit line can also vary. To address these sources of variability at the population level, a Monte Carlo simulation of the probit estimation was performed for every group at or exceeding the screening index. The LC_{50} , probit slope, and diflubenzuron concentration were randomly selected from the distributions in Table 9-2, resulting in a distribution of probit values (Figure 9-3). The mean population response and the risk of 50 percent, 75 percent or 90 percent of the population responding are given in Table 9-3. Risk was equivalent

to the proportion of iterations in which the probit values for 50 percent (5.0), 75 percent (5.52) or 90 percent (6.28) were exceeded.

9.1.2. Results

9.1.2.1. Nontarget species

The screening index indicates that lepidopterans (moths and butterflies), orthopterans (grasshoppers), parasitic wasps, benthic crustaceans, aquatic insects, and immature planktonic crustaceans are directly at risk from diflubenzuron. For these species, increasing the application rate causes a larger proportion of the population to be affected (Table 9-4). At higher application rates there is a high risk of affecting at least 50 percent of these populations (Table 9-4). Higher application rates also increase the number of species groups affected; more aquatic organisms are at risk at the highest application rate than the lowest.

9.1.2.1.1. Terrestrial

Although there were differences in diflubenzuron concentrations between the upper and lower canopy leaves, species were either at risk in both habitats, or not at risk in either. Differences in diflubenzuron concentrations between spring and fall likewise did not change the groups at risk for those analyzed in both seasons (Lepidoptera and Orthoptera). The mean population reduction was greater, however, for species feeding in the upper canopy than the lower canopy and for species feeding in the spring than the fall (Table 9-4).

Population reductions based on lepidopteran LC_{50} values probably substantially underestimate risk because these toxicological studies only addressed mortality of test organisms over a relatively short period. The EC_{50} data are probably more indicative of natural mortalities because these data are based on failure to molt, which would result in mortality had the toxicological test been conducted longer.

Results from field studies on the effects of diflubenzuron on nontarget terrestrial organisms largely corroborate the results of the risk modeling. Field studies show significant reductions in populations of butterflies and moths (Lepidoptera) and grasshoppers (Orthoptera) in blocks sprayed with diflubenzuron (see Section 5) as do predictions from this risk assessment (Tables 9-1 and 9-4). Field studies show small butterflies and moths (microlepidopterans) are less affected than large (macrolepidopterans), perhaps as a result of life history and behavior rather than susceptibility to diflubenzuron. This difference was not explored in the risk models.

Most field studies did not infer the percent reduction in populations from their data, but rather looked for statistical differences between population levels on treated and control plots. Often, relatively soon after applications, many to possibly all caterpillars found alive in treated areas undoubtedly died at subsequent molts. Thus, a comparison between the field studies and the modeling results in this regard is not possible in most cases. Nevertheless, two field studies on grasshoppers yield some comparable results (Everts, 1990; Jech and others, 1993), however the modeling results predict somewhat higher probabilities of mortality than found in the field studies. Everts (1990) found nearly a 90 percent reduction in grasshoppers after treatment at 2.4 fl. oz./ac, and no noticeable reduction at 1.1 fl. oz./ac, whereas the model predicts a 31 percent to 88 percent risk of a 90 percent reduction in population at a rate of 1 fl. oz./ac. The results of Jech and others (1993) were more similar to the modeling results (see Tables 5-10 and 9-4).

Parasitic wasps of gypsy moths are at risk according to field studies and risk modeling; however, the mechanism by which they suffer is unclear. Toxicity studies suggest that early larval stages of the parasites are directly affected by the diflubenzuron in their hosts, older larval parasites might be more resistant, and some mortality could result from death of the host succumbing to diflubenzuron rather than from direct toxicity to the parasite. Field studies suggest that populations of parasites of other orders of insects besides lepidopterans will be mostly unaffected by diflubenzuron treatments (Section 5). Modeling results and field studies on the effects of diflubenzuron on predators such as lacewings, ladybird beetles, big-eyed bugs, and others suggest that they will be largely unaffected by diflubenzuron treatments at rates used in the USDA's gypsy moth program. However, these organisms may be affected by a reduction in food as their prey are killed by the diflubenzuron. Species feeding on terrestrial lepidopterans or orthopterans will experience a reduction in these food items within a spray area. Predatory species will have to increase their foraging areas or leave the spray area if they are dependent upon affected prey species.

Ground spiders are a group of terrestrial, above-ground organisms studied in the field whose risk was not modeled in this assessment because of a lack of toxicological data. Ground spiders could be indirectly affected through prey reduction or directly affected by diflubenzuron applications; the effects noted in the field studies were not universal among the groups analyzed, some taxa being more affected than others, and overall species diversity remaining unchanged (Everts, 1990; Martinet and others, 1993; Perry and others, 1993).

Vertebrates, some beetles, and earthworms are not directly at risk from diflubenzuron. The majority of species not exposed to diflubenzuron through ingestion are also not at great risk. These include sucking insects, nectar feeding insects such as bees, leaf miners, insects living inside wood, many types of ants, and insects feeding entirely on forbs which emerged after diflubenzuron application. This

conclusion is suggested by field studies (Section 5), considerations of the natural history of these groups, and the fate and transport of diflubenzuron.

Although birds were not identified as at risk from population reduction due to diflubenzuron applications in the modeling results, field studies suggest that subtle effects might occur for some insectivorous species. Thus far studies have not found reductions in overall reproductive success for breeding birds in treated plots; however, there are clear indications of negative effects resulting from a reduction in food resources (diet switching, reduced fat loads of adults, and enlarged foraging territories have been noted after applications). A mitigating factor for birds is that caterpillar population reduction is not immediate, but occurs gradually over a few weeks, especially if some caterpillars are not killed until the second molt after diflubenzuron application. Similar indirect effects could have negative consequences for bats that forage on Lepidoptera.

There are no differences in risk between the two ecosystems (developed and undeveloped forests) when considering the period immediately following spraying. However, the duration of exposure to diflubenzuron is shorter for ground-dwelling organisms in the developed forest than in the undeveloped forest ecosystem. This makes the risk over the growing season less for ground-dwelling organisms in the developed forest ecosystem.

9.1.2.1.2. Aquatic

Susceptible aquatic organisms are at higher risk in the developed forest ecosystem than in the undeveloped forest ecosystem due to higher concentrations of diflubenzuron in runoff water. Although the estimated environmental concentrations differ for aquatic habitats in undeveloped and developed forests (Section 7), there were few differences in groups of aquatic organisms at risk from diflubenzuron applications between the two ecosystems. Benthic insects were at risk in all but the undeveloped forest pond (the habitat with the lowest diflubenzuron concentration) at the lowest application rate. Planktonic crustaceans were at lower risk in aquatic habitats in undeveloped forests than in residential areas. The risk to some types of benthic insects may be overestimated because the toxicological data used in this analysis were LC_{50} values from mayflies and caddisflies. At least one species of stonefly had an EC_{50} value higher than the LC_{50} data used. Mollusks were not at risk according to the modeling results.

The results from modeling aquatic systems reflect the findings of field studies. Populations of benthic and planktonic crustaceans and insects in ponds were found to suffer mortality from diflubenzuron applications in several field studies (Section 5).

Fish are not at risk of mortality from diflubenzuron in every habitat, however, the concentration of diflubenzuron exceed the no observed effect concentration (NOEC) in the directly sprayed stream and the developed forest stream. Fish which feed on aquatic invertebrates may be affected indirectly by diflubenzuron through a reduction in their prey items. Fish are not at risk from 4-chloroaniline in any of the habitats.

9.1.2.1.3. Multiple Applications and Other Considerations

Two applications, seven to fourteen days apart, would present the same risk to most organisms as one application at a higher rate. This is based on the similarity between the estimated diflubenzuron residues from two lower application rates which equal one higher application rate (Section 7). Annual application of diflubenzuron would result in higher risk to litter invertebrates than an application made for only one year due to the persistence of diflubenzuron residues over the winter and into the following summer (Section 7).

Determining the species at risk of becoming locally extinct within the spray area is more difficult. For many invertebrates there are no data on invasion rates into new areas. Natural population sizes and variability around those sizes are also not known for most species. Terrestrial invertebrates may use one area as larvae, and disperse into other areas to lay eggs as adults; elimination of the larval stage of lepidopterans does not imply that no eggs from dispersing adults will be laid in a treated area. Some generalizations can be made, however. Susceptible species in rare or patchily distributed specialized habitats are at greater risk than species with large geographic distributions. For organisms inhabiting specialized habitats, there may be extremely low or no dispersal from other such habitats. Susceptible organisms will not successfully reinvade the sprayed area as long as diflubenzuron concentrations remain high enough to cause substantial population mortality.

Species that have more than one generation per season will be at more risk than species with a single annual brood. Susceptible species with multiple generations will suffer high mortality with each generation produced, thus magnifying the population reduction over that of a single annual generation species. Organisms with high dispersal rates will be able to reinvade treated areas if the treatment is not repeated. Organisms whose populations were severely reduced by diflubenzuron and have slow population growth rates and low dispersal rates will be affected for the longest time. Clearly, large block size, low dispersal capabilities, and frequent retreatment do not favor recolonization. The rapidity with which man-made lakes and newly created streams are invaded is evidence of the dispersal ability of many aquatic invertebrates (Sheldon, 1984). Terrestrial species should also recolonize sprayed areas, however the species composition may be different from that prior to the spray.

9.1.2.2. Forest Health

Diflubenzuron is not phytotoxic and has no direct effect on plants. Indirectly, diflubenzuron affects the health of trees by reducing the density of leaf-eating insects and therefore, reducing the amount of insect damage to trees and tree mortality. Tree mortality from non-outbreaking insects is minimal, isolated, and considered natural. Because diflubenzuron does not affect invertebrates until they molt, some defoliation will occur despite diflubenzuron treatment.

9.1.2.3. Water Quality

Water quality is degraded when pesticides enter water resources. The half-life of diflubenzuron in water is short and water quality degradation due to increased concentrations of diflubenzuron in the water column will be brief. However, diflubenzuron transported into water resources on fallen leaves has a much longer half-life and is available to organisms consuming these leaves. Although diflubenzuron affects stream invertebrates that process detritus, field studies have not shown any decline in detrital decomposition rates (Swift and others, 1988). Populations of many invertebrates that feed on algae are reduced by diflubenzuron. An increase in algal biomass could occur following the loss of algal herbivores; however, this is not likely and has not been demonstrated in field studies.

9.1.2.4. Microclimate

Diflubenzuron does not directly affect microclimate in any way other than decreasing the amount of defoliation in forests infested with gypsy moths. Indirectly through reducing defoliation, diflubenzuron also reduces the effects of gypsy moths on microclimate (that is, soil temperature, light penetration through the canopy). However, because leaf-eating insects do not die immediately, some defoliation does occur. Any changes in microclimate due to defoliation following diflubenzuron treatment should be of brief duration as most trees will refoliate following gypsy moth damage.

9.1.2.5. Soil Productivity and Fertility

Although earthworms are not at risk from diflubenzuron, other types of litter invertebrates appear to be at risk based on field studies (Perry and others, 1993). Significant reductions in density in mites and ground-dwelling spiders were observed following application of 35 g a.i./ha (0.5 oz/ac) (Perry and others, 1993). Reductions in mites populations were found in other studies as well (Section 5). Leaf-eating or litter-eating ants also appear to be at risk based on the Monte Carlo simulations

litter-eating ants also appear to be at risk based on the Monte Carlo simulations discussed earlier in this section. Although these soil or litter-dwelling species appear to be at risk, at least one field study suggests that decomposition rates are not affected by diflubenzuron (Rockwood, 1995). Further toxicological testing of litter invertebrates and field studies measuring decomposition, organic matter content, bacterial and fungal populations and litter invertebrate populations need to be conducted to evaluate the effect of diflubenzuron on soil productivity and fertility.

9.2. *Bacillus thuringiensis* var. *kurstaki*

9.2.1. Methodology

Due to the short half-life of *B.t.k.* spores and crystals, exposure is limited to the time period immediately following application (see Section 5). Lepidopteran species are the primary group at risk from *B.t.k.*. Most lepidopteran species that consume the outer surface of the leaves, entire leaves, or leaf litter will be exposed. Not all of the leaf-eating caterpillars will be exposed, because a caterpillar may spend some of its time molting, an activity which reduces exposure to *B.t.k.* because caterpillars do not feed when molting (Slansky, 1993). Aquatic lepidopteran species may be affected.

Most toxicological studies do not estimate the concentration of *B.t.k.* in the diet of test organisms, reporting instead IU (International Units) potency sprayed on cubes of artificial diet (IU/ml). There is no indication of either the concentration of *B.t.k.* in the diet as it is not homogeneously distributed, or how much diet or *B.t.k.* an organism consumes, making it difficult to relate these studies to field conditions. No studies have provided estimates of IU per mg of organism or IU per mg of diet. Only one study calculated the LD_{50} in terms of IU per organism (Ratcliffe and Yendol, 1993). Toxicity of *B.t.k.* to Lepidoptera varies greatly between species and in some cases among instars.

In this assessment, the risk of encountering a *B.t.k.* droplet is calculated rather than the risk of mortality once the droplet is ingested, due to the difficulty in relating toxicological studies to concentration of *B.t.k.* ingested. The number of *B.t.k.* drops encountered was estimated by the following equation:

Drops encountered = Density *B.t.k.* drops/cm^2 x Area consumed by caterpillar(cm^2)/day

Caterpillars were considered at risk if the number of *B.t.k.* drops encountered exceeded one. In susceptible lepidopteran species, a single *B.t.k.* crystal can cause death; the first symptoms develop within minutes of ingestion. Thus for some susceptible species, the risk of encountering a drop may be equivalent to the risk of

encounter. An organism's diet and instar stage may moderate the effect of *B.t.k.* sometimes drastically.

B.t.k. deposition varies spatially within the spray area, as does the area of foliage consumed by an individual caterpillar per day. To address these sources of variability, a Monte Carlo simulation estimated the risk of exposure to *B.t.k.* by randomly selecting *B.t.k.* drop environmental concentrations, and the square centimeters of foliage consumed by a caterpillar per day.

The foliage area consumed by a caterpillar per day was estimated by converting the mass consumed into area using the specific leaf weight of spring leaves in Prince William County (Virginia). Relative consumption rates estimate the amount of food ingested per body weight making it possible to calculate feeding rates for different sizes of caterpillars. Relative consumption rates are usually reported as fresh or dry weight ingested per dry weight of caterpillar (Slansky, 1993). Rates based on fresh weight are higher than those based on dry weights with rates as high as 18 mg fresh weight per mg dry weight of caterpillar. Relative consumption rates vary with the nutrient and water content of leaves and plant species. Relative consumption rates using fresh weights of leaves vary widely (4 to 18 mg per mg dry weight of caterpillar) with higher rates reported in the spring than in the autumn probably because leaves contain more water in the spring. To convert caterpillar dry weight to fresh weight an assumption was that a minimum of 50 percent of their fresh body weight is water.

Relative consumption rate was represented with a triangular distribution ranging from 3 to 9 mg with the likeliest value at 6 mg fresh weight leaves per mg fresh weight caterpillar. The spring leaf weight distribution used for diflubenzuron was used with the relative consumption rate to calculate the number of square centimeters of leaf a caterpillar ingested. The *B.t.k.* drop density was assumed to be lognormally distributed (Mierzejewski, 1993) with the mean and standard deviation for the two application rates given in Table 7-9. The model was run for 4000 iterations using a Latin Hypercube sampling technique. The risk of encountering a droplet was equivalent to the proportion of iterations in which the number of drops encountered exceeded one.

9.2.2. Results

9.2.2.1. Nontarget species

Butterflies and moths (Lepidoptera) are clearly at risk of exposure to *B.t.k.*. For some species, however, the risk of mortality or sublethal effects is high while for others it is essentially zero. For some species which suffer some mortality, the

surviving larvae recover and produce viable adults (Peacock and Schweitzer, 1993). Larger caterpillars consume a larger area of vegetation than smaller caterpillars and are more likely to encounter a *B.t.k.* drop (Table 9-5). For some species the smaller, earlier instars are more susceptible to *B.t.k.*, while for other species the larger, later instars are more susceptible. The risk of encountering a drop increases with application rate and height above the ground in the canopy. Only spring-feeding caterpillars are at risk due to the short half-life of *B.t.k.*. Most field studies corroborate the results of the modeling (Section 5). Reductions in the total number of caterpillars and reductions in species richness have been noted (Section 5). Recovery to pre-spray numbers of caterpillars has taken one to two years in field studies in Oregon and West Virginia (Miller, 1990; Sample and others 1993a). The effects of multiple sprays of *B.t.k.* on butterflies and moths is largely unknown, however, new research suggests that for resistent species, multiple applications will not cause additional mortality over that observed in the first spray (Carter and others, 1995). It must be stressed, however, that just as the initial effects of *B.t.k.* vary from species to species, the effects of multiple sprays will also vary and these results can not be generalized to all species of Lepidoptera.

Field studies indicate some soil invertebrates (nematodes, ground beetles) may also be at risk (Addison, 1993). All strains of Bt are toxic to the eggs of the nematode, *Trichostronjulas colubriformus*; however, populations of other nematode species increased following field applications of *B.t.k.*. Population declines of ground beetles following *B.t.k.* applications have also been noted (Addison, 1993). *B.t.k.* has caused mortality in predatory mites (Addison, 1993). A quantitative risk assessment for these groups was not performed due to a data gap concerning their toxicological response.

Although some laboratory studies suggest that some species of parasitic wasps are directly susceptible to *B.t.k.*, doses used in these feeding studies are difficult to relate to field applications. Field studies suggest the predominate effect of *B.t.k.* on parasites is indirect, through effects on its host species. These studies show either no effects on rates of parasitism or increased rates, as *B.t.k.* affected larvae slow their own development and are therefore preferentially sought by ovipositing wasps because of their smaller size (Section 5).

Terrestrial vertebrates are not directly affected by *B.t.k.*, although some irritation or allergy-like symptoms may occur. Vertebrates that feed on lepidopteran species that are caterpillars in the spring will have a reduced number of prey items to eat. The severity of such reductions should vary depending on the composition of dominant lepidopteran species. There is some evidence that reductions in caterpillars from *B.t.k.* applications forces birds and mammals to switch diets; in one study on birds the number of nesting attempts per year was reduced, although not the overall production of fledglings per breeding territory in the year of application or subsequent years (Section 5). Bats feeding nearly exclusively on Lepidoptera could also be

indirectly affected through a reduction in prey base, as strongly suggested by a study of the Virginia big eared bat in West Virginia (Sample and others, 1993b).

B.t.k. has not been observed to cause fish kills following field application, however, it has been suggested that some types of fish, such as carp or koi, that have alkaline digestive secretions may be affected. This data gap could be filled by conducting laboratory toxicological tests on fish with alkaline digestive secretions.

9.2.2.2. Forest Health

B.t.k. reduces the damage to trees from leaf-eating lepidopterans soon after ingestion of the crystal. Reduction in the amount of defoliation allows more products of photosynthesis to be used in tree growth and reproduction rather than forming a new set of leaves.

9.2.2.3. Water Quality

B.t.k. has been observed to increase microbial respiration and also to decrease decomposition rates. Initially following *B.t.k.* application, stream insects such as mayflies, may increase their drift rate, however, this effect is not long-lasting. Indirectly, through the reduction of defoliation, *B.t.k.* reduces changes in water quality associated with gypsy moth infestations.

9.2.2.4. Microclimate

B.t.k. tends to promote the full establishment of the tree canopy by inhibiting leaf-eating lepidopterans from consuming it. There should be no change in microclimate from that of a healthy uninfested forest due to *B.t.k.* application.

9.2.2.5. Soil productivity and fertility

Soil invertebrates (nematodes, ground beetles) may be affected by *B.t.k.* (Addison, 1993). More research needs to be conducted to determine the effects of *B.t.k.* on decomposition rates.

9.3. Disparlure

9.3.1. Nontarget species

Quantitative calculations of risk were not performed for Disparlure due to the lack of necessary toxicological data for invertebrate groups. Toxicological data for fish, bobwhite quail chicks, and mallard ducklings indicate that Disparlure is not a mortality risk to any of these groups at the application rates used. Disparlure would have to be applied at well over 1000 times its present application rate to affect the most sensitive of the vertebrate species, fish. Due the mode of action of Disparlure, it is not likely that invertebrates will be killed; however, this should be interpreted with caution due to the data gap for invertebrate toxicological studies. The risk from Disparlure is not substantially different between the two ecosystems (developed and undeveloped forests).

9.3.2. Forest Health, Water Quality, Microclimate, Soil Productivity and Fertility

Disparlure does not directly affect these endpoints. Disparlure does not protect foliage from gypsy moth damage; therefore forest health, water quality, microclimate and soil productivity will continue to be affected by defoliation. Disparlure will reduce risk that gypsy moth populations will alter these endpoints in the subsequent year as gypsy moth egg masses are reduced by mating disruption.

9.4. Nucleopolyhedrosis Virus (NPV)

9.4.1. Nontarget Species

Quantitative calculations of risk were not performed for NPV due to lack of toxicological data showing effects from NPV. Toxicological and field tests show no effects for terrestrial vertebrates at concentrations greater than the application rate used to control gypsy moth. Concentrations of NPV will be below those causing effects from fish or *Daphnia*, the only aquatic groups for which toxicity data exist. Based on these data, NPV is not expected to put any group at risk of mortality, other than gypsy moths, due to its application. Again caution should be used in interpreting these results due to the lack of toxicological data for many invertebrate groups. The risk from NPV to nontarget species is not different between the two ecosystems.

The effect of NPV applications to a parasitic wasp was evaluated in one field study, and demonstrated a lower rate of parasitism due to parasite avoidance of NPV-infected hosts (Section 5).

9.4.2. Forest Health, Water Quality, Microclimate, Soil Productivity and Fertility

NPV itself poses no risk to altering these endpoints due to its host specificity. However, if such application prevents subsequent defoliation(s), it will indirectly have a positive effect on forest health.

9.5 Dichlorvos

9.5.1. Nontarget Species

The method by which dichlorvos is applied greatly reduces the number of organisms that may be exposed to it. Most of these will be small invertebrates that become caught within the insect trap. A very small number of invertebrates in the soil or litter are at risk from contacting tapes from broken traps. For these organisms there is a high risk of mortality; however, due to the small numbers of invertebrates caught in traps relative to natural population sizes, these species will not be at risk of populations reductions from dichlorvos. Of more concern is the risk to vertebrates which disturb the trap and remove the Vaportape II tape strip. Using the Human Health Risk Assessment to derive risk calculations, large vertebrates, such as bears, will not be at risk from dichlorvos. Each strip contains 590 mg of dichlorvos, making the dose for a small bear (30 kg) about 20 mg/kg/day. This dose is lower than the LD_1 reported for rats (Wagner and Johnson, 1970), although it is higher than the NOEL for acetyl cholinesterase inhibition and systemic effects in dogs (0.08 mg/kg/day) (USEPA 1989). Vertebrates consuming the tape strips may experience some sublethal effects from dichlorvos. Few vertebrates relative to natural population sizes will be exposed due to the low density of traps used. The risk from dichlorvos is not different between the two ecosystems.

9.5.2. Forest Health, Water Quality, Microclimate, Soil Productivity and Fertility

Dichlorvos will not directly affect these endpoints.

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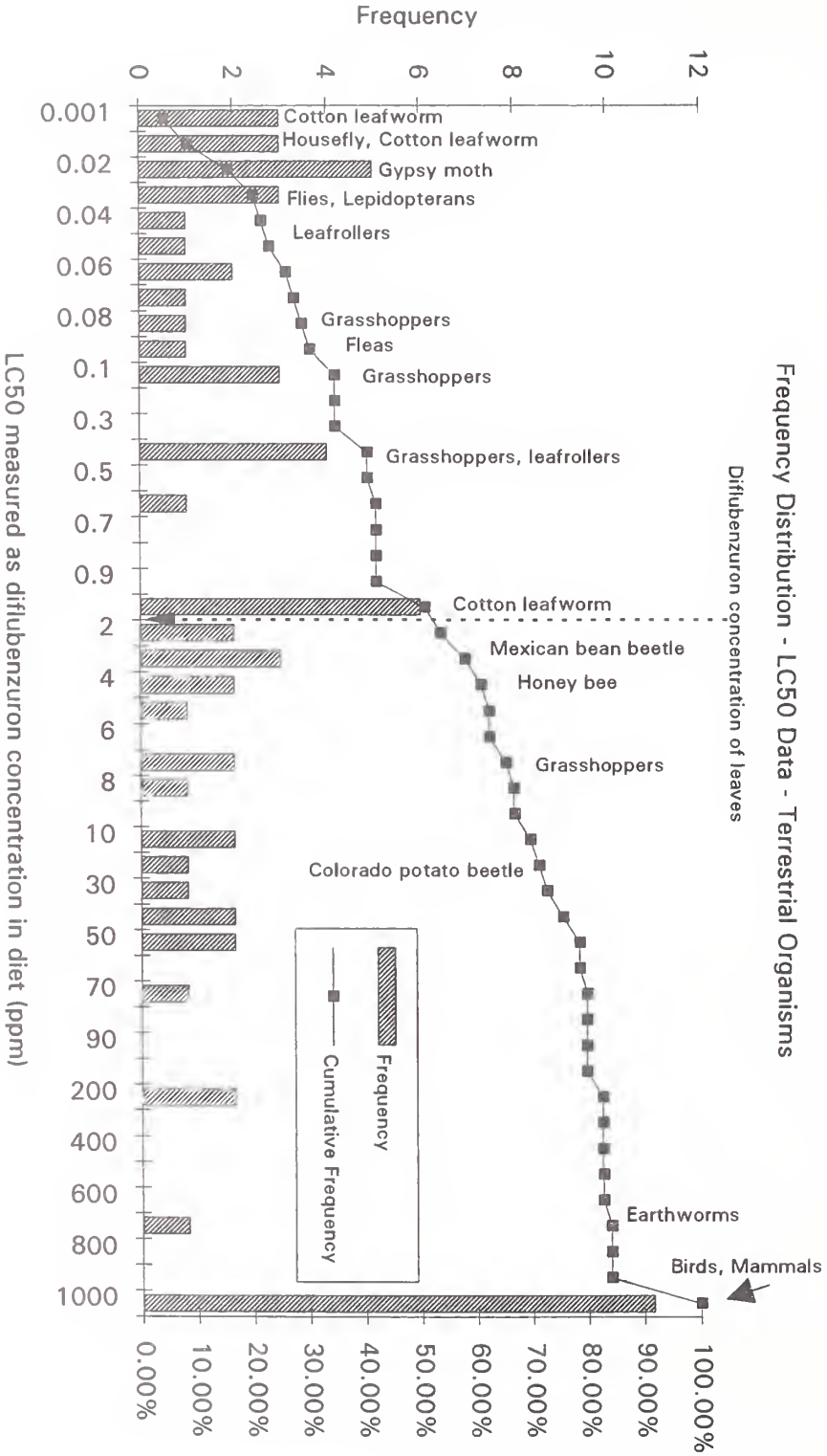

Figure 9-1. Cumulative frequency distribution of toxicological studies conducted using diflubenzuron on terrestrial organisms.

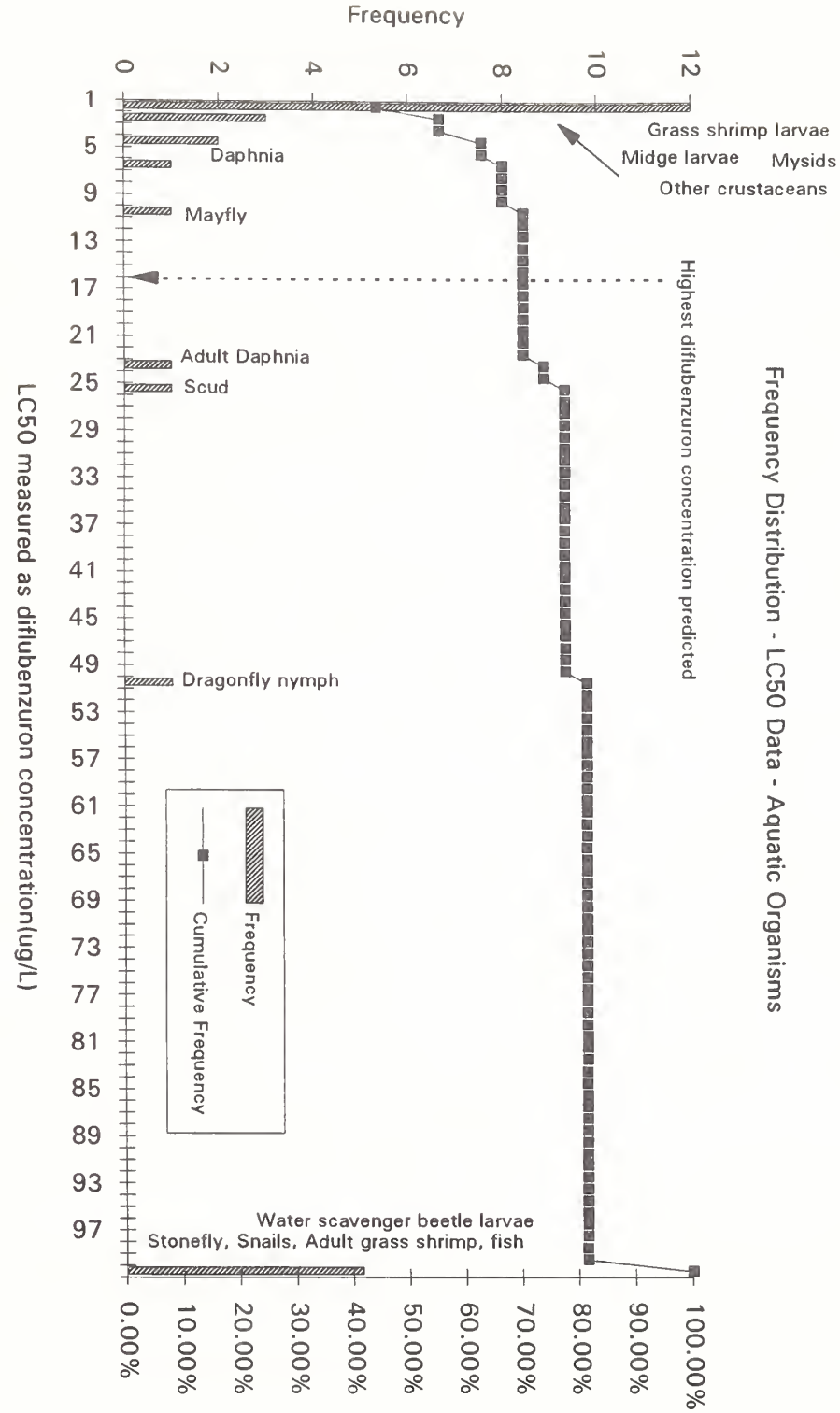

Figure 9-2. Cumulative frequency distribution of toxicological studies conducted using diflubenzuron on aquatic organisms.

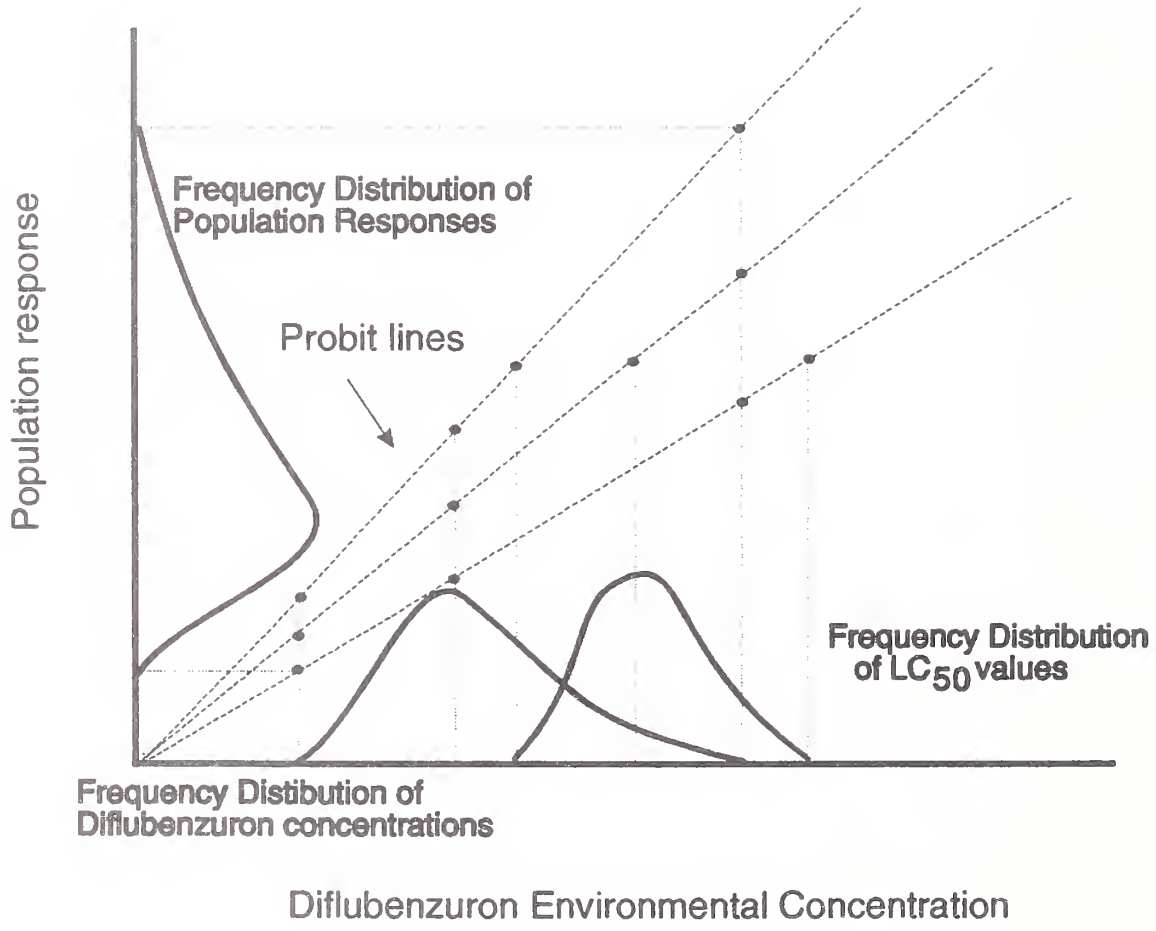

Figure 9-3. Relationship between environmental concentration of diflubenzuron and population level risk

Table 9-1. Organisms at risk from diflubenzuron applications based on the screening index

Terrestrial Organisms at risk at ALL application rates (1.0, 0.5, 0.33, 0.25 ai oz/ac)
Spring larval lepidopterans upper and lower canopy (based on LC_{50} and EC_{50})
Fall larval lepidopterans upper and lower canopy (based on LC_{50} EC_{50})
Spring orthopterans (grasshoppers) upper canopy and lower canopy
Fall orthopterans upper and lower canopy
Parasitic wasps of gypsy moth upper canopy
Aquatic Organisms - Direct spray At risk at all application rates (1.0, 0.5, 0.33, 0.25 ai oz/ac)
Benthic midges
Benthic crustaceans
Benthic insects (mayflies caddisflies)
Planktonic insects
Planktonic crustaceans nymphs
Planktonic crustaceans adults
Fish (NOEC [1]) ONLY at 1.0
Aquatic Organisms - Developed forest pond At risk at ALL application rates (1.0, 0.5, 0.33, 0.25 ai oz/ac)
Benthic crustaceans
Benthic insects (mayflies caddisflies)
Planktonic insects
Planktonic crustaceans nymphs
Planktonic crustaceans adults

Table 9-1, *continued ...*

Aquatic Organisms - Developed forest stream At risk at ALL application rates (1.0, 0.5, 0.33, 0.25 ai oz/ac)
Benthic midges ONLY at risk in the 1.0 and 0.5 ai oz/ac application rate
Benthic crustaceans
Benthic insects (mayflies caddisflies)
Planktonic insects
Planktonic crustaceans nymphs
Planktonic crustacean adults
Fish (NOEC [1]) ONLY at 1.0, 0.5, 0.33
Aquatic Organisms - Forest pond At risk at ALL application rates (1.0, 0.5, 0.33, 0.25 ai oz/ac)
Benthic crustaceans
Planktonic insects
Planktonic crustaceans nymphs
Aquatic Organisms - Forest stream At risk at ALL application rates (1.0, 0.5, 0.33, 0.25 ai oz/ac)
Benthic crustaceans
Benthic insects (mayflies caddisflies)
Planktonic insects
Planktonic crustaceans nymphs
Planktonic crustaceans adults ONLY at risk at 1.0 ai oz/ac

[1] NOEC - No Observed Effects Concentration

Table 9-2. Variables used in Monte Carlo simulation

For Normal or LogNormal Distributions:	Distribution	Mean	Standard Deviation	
DFB [1] Upper canopy spring	LogNormal	Table 7-3	Table 7-3	
DFB Lower canopy spring	LogNormal	Table 7-3	Table 7-3	
DFB at soil surface spring	LogNormal	Table 7-3	Table 7-3	
DFB water concentration - direct spray (μg/L)	LogNormal	Table 7-3	Table 7-3	
Lepidopteran LC_{50} (ppm diet)	LogNormal	1.153	1.447	
Lepidopteran EC_{50} (ppm diet)	LogNormal	0.115	0.198	
For Triangular or Uniform Distributions:	Distribution	Likeliest	Lowest	Highest
Percent DFB remaining upper canopy	Triangular	0.3	0	1.0
Percent DFB remaining lower canopy	Triangular	0.3	0.4	1.0
Leaf Weight Spring (mg/cm ^{2})	Triangular	0.0165	0.008	0.0240
Leaf Weight Autumn (mg/cm ^{2})	Triangular	0.0185	0.010	0.0254
Forb leaf weight (mg/cm ^{2})	Triangular	0.0122	0.066	0.0168
Grass leaf weight (mg/cm ^{2})	Triangular	0.0092	0.005	0.0127
DFB in undeveloped forest stream (μg/L)	Triangular	Table 7-3	Likeliest - ½ of Likeliest	Likeliest + ½ of Likeliest
DFB in developed forest stream	Triangular	Table 7-3	Likeliest - ½ of Likeliest	Likeliest + ½ of Likeliest
DFB in undeveloped forest pond	Triangular	Table 7-3	Likeliest - ½ of Likeliest	Likeliest + ½ of Likeliest
DFB in developed forest pond	Triangular	Table 7-3	Likeliest - ½ of Likeliest	Likeliest + ½ of Likeliest
DFB in sediments	2% of water column value			

Table 9-2, *continued* ...

For Triangular or Uniform Distributions:	Distribution	Likeliest	Lowest	Highest
4CA[2] in water column	10% of water column value			
Orthopteran LC_{50} (ppm diet)	Triangular	0.12	0.08	0.41
Coleopteran LC_{50} (ppm diet)	Triangular	14.9	1.7	25
Ant LC_{50} (ppm diet)	Uniform		0.07	0.51
Parasitoid EC_{50} (ppm diet)	Uniform		0.0053	0.0064
Thysanopteran LC_{50} (ppm diet)	Triangular	70	30	140
Earthworm NOEC (ppm soil)	Uniform		780	1000
Benthic midges LC_{50} (μg/L)	Triangular	1.8	1	1.9
Benthic crustaceans LC_{50} (μg/L)	Triangular	1.8	0.15	2
Benthic insects LC_{50} (μg/L)	Triangular	10	.75	100
Mollusks LC_{50} (μg/L)	Uniform		112.5	137.5
Planktonic insects LC_{50} (μg/L)	Uniform		0.45	0.55
Planktonic crustaceans immatures LC_{50} (μg/L)	Triangular	1.5	0.062	6.89
Planktonic crustaceans adults LC_{50} (μg/L)	Uniform		21.11	25.80
Fish LC_{50} (μg/L)	Triangular	50	30	130
Benthic insects -DFB sensitive EC_{50} (μg/L)	Triangular	4.9	0.1	4.9
Benthic insects -DFB tolerant EC_{50} (μg/L)	Uniform		57.5	100
Benthic crustaceans EC_{50} (μg/L)	Triangular	25	2	71
Planktonic crustaceans NOEC	Triangular	0.4	0.45	2
Mollusks EC_{50} (μg/L)	Uniform		4.1	5.3
Crustaceans 4CA LC_{50} (μg/L)	Uniform		90	100
Fish 4CA LC_{50} (mg/L)	Uniform		31	33

[1] DFB = Diflubenzuron
[2] 4CA = 4-chloroaniline

Table 9-3. Toxicological data used to estimate LC_{50} or EC_{50} distributions

Mortality	*LD_{50} mg/kg*	*LD_{50} µg/insect*	*LC_{50} ppm*	*Reference*
Lepidopterans				
Diflubenzuron in diet				
Brassia tada last instar			0.02	
Mamestra brassicae (armyworm)			0.02	Tada and others 1986
Laspeyresia pomonella			0.03	Van Eck 1981
Laspeyresia pomonella hatch-pupation			0.03	as cited Van Eck 1981
Laspeyresia pomonella egg - pupae			0.04	Van Eck 1981
Gypsy Moth - fed alder leaves			0.06	Berry and others 1993
Laspeyresia pomonella fifth instar			0.4	Van Eck 1981
Gypsy Moth - fed fir needles			0.45	Berry and others 1993
Spodoptera littoralis Cotton leafworm 5th instar			1	Ishaaya, 1990
Spodoptera exigua (beet armyworm) 1st instar			1.1	Granett and Hejazi 1983
Spodoptera exigua 4th instar			1.3	Granett and others 1983
Spodoptera exigua 1- 2 instar			1.4	Granett and others 1983
Spodoptera exigua 4th instar			1.5	Granett and others 1983
Adoxophyes orana (leafroller) hatching-pupation observed mortality			2.5	Van Eck 1981
Adoxophyes orana (leafroller) hatching - 5th instar estimated mortality			3.6	Van Eck 1981
Spodoptera littoralis 2nd instar			5	Osman and others 1986

Lepidopteran Diet Spray Studies	*LD_{50} mg/kg*	*LD_{50} µg/insect*	*LC_{50} ppm*	
Spodoptera exempta 3rd instar 96 hr after exp.			0.00001	Degheele and others, 1993
Orgyiz psuedotsugata (tussock moth) 2nd instar			0.1	Rappaport and Robertson, 1981
Spodoptera littoralis			0.43	Grosscurt and Wixley 1991

Table 9-3, *continued ...*

Spodoptera exigua			7.23	Grosscurt and Wixley 1991
Spodoptera littoralis 3rd instar		0.38	12.3	El Saidy and others, 1988
Plutella xylostella			51.1	Grosscurt and Wixley 1991
Spodoptera littoralis (Ahmed)DFB resistant strain			71.43	Ahmed and others, 1987
Schoenobius incertulas Walker		250		Zaidi and others 1988
Pseudoplusia includens 48 hr		43.9		Reed and Bass, 1980
Spodoptera littoralis	1	0.38		Neumann and Guyer 1987

Lepidopteran EC_{50} studies	*ED_{50} mg/kg*	*ED_{50} µg/insect*	*EC_{50} ppm*	*Reference*
Gypsy moth 3rd instar contin. feeding pupation effect - failure to pupate			0.006	Abdel-Monem and Mumma 1981
Gypsy moth 2 instar diet effect - failure to molt the second time in this instar			0.0075	Granett and Weseloh, 1975
Gypsy moth 3rd instar effect - failure to molt the second time in this instar			0.009	Abdel-Monem and Mumma 1981
Gypsy moth 3rd instar effect - failure to feed			0.013	Granett and Dunbar 1974
Gypsy moth 5th instar continuous feeding pupation effect - failure to pupate			0.009	Abdel-Monem and Mumma 1981
Gypsy moth 3rd instar effect - partial molt			0.021	Granett and Dunbar 1974
Gypsy moth 3rd instar effect - partial molt			0.031	Granett and Dunbar 1974
Gypsy moth 3rd instar effect - failure of first molt of this instar			0.052	Abdel-Monem and Mumma 1981
Gypsy moth 5th instar effect - failure of first molt of this instar			0.122	Abdel-Monem and Mumma 1981
Laspeyresia pomonella 28 hr effect - failure to molt			0.4	Van Eck 1981
Adoxophyes orana 28 h effect - failure to molt			0.6	Van Eck 1981

Table 9-3, *continued ...*

Orthopterans (Mortality)	LD_{50} *mg/kg*	LD_{50} *µg/insect*	LC_{50} *ppm*	*Reference*
Migrator grasshopper diet 20 day exposure.			0.08	Elliott and Iyer 1982
Migratory grasshopper 20 day exposure diet spray			0.1	Elliott and Iyer 1982
Migratory grasshopper 1 day exposure diet spray			4.5	Elliott and Iyer 1982
Migratory grasshopper 8 day exposure			0.41	Elliott and Iyer 1982
Migratory grasshopper 12 day exposure			0.12	Elliott and Iyer 1982
Beetles (Mortality)	LD_{50} *mg/kg*	LD_{50} *µg/insect*	LC_{50} *ppm*	*Reference*
Mexican bean beetle 3rd instar diet spray			3.4	McWhorter and Shepard, 1977
Leptinotarsa decemlineata diet spray			36	Grosscurt and Wixley 1991
Colorado potato beetle 1st instar top			50	Hegazy and others 1989
Beetles (No observed effect concentration)	*NOEC mg/kg*	*NOEC µg/insect*	*NOEC ppm*	*Reference*
Hippodamia convergens (convergent lady beetle) top adult			10000	Keever and others, 1990
Geocoris punctipes (Big eyed bug) top nymph			10000	Keever and others, 1990
Erythroneura variablis (Homoptera)			>1000	Grosscurt et al., 1988
Fleas, Lice, and Mites Mortality	LD_{50} *mg/kg*	LD_{50} *µg/insect*	LC_{50} *ppm*	*Reference*
Bovicula limbatus coated diet day 5			4.1	Hopkins and Chamberlain, 1978
Bovicula limbatus coated diet day 4			14.4	Hopkins and Chamberlain, 1978
Bovicula limbatus coated diet day 1			57.6	Hopkins and Chamberlain, 1978
Cat flea diet 1.5-3 day old			0.09	El-Gazzar and others 1988
Bovicula limbatus 3rd instar coated diet day 6			1.8	Hopkins and Chamberlain, 1978
bulb mite			>100 ppm	Knowles and others 1988

Table 9-3, *continued ...*

Ants (Mortality)	LD_{50} *mg/kg*	LD_{50} *µg/insect*	LC_{50} *ppm*	*Reference*
Capriguara goncalves (ant) 4 day diet		102.5		Busoli and others 1992
Capriguara goncalves (ant) 28 day diet		14		Busoli and others 1992
Parasitoid Hymenopterans (Mortality)	EC_{50} *mg/kg*	EC_{50} *µg/insect*	EC_{50} *ppm*	*Reference*
Apanteles melanoscelus DFB in host diet emergence			0.0059	Granett and Weseloh, 1977
Apanteles melanoscelus DFB in host diet molting			0.079	Granett and Weseloh, 1977
Apanteles melanoscelus DFB in host diet pupation			0.02	Granett and Weseloh, 1977
Earthworms and litter invertebrates (Mortality)	*NOEC mg/kg soil*	*NOEC µg/insect*	*NOEC ppm*	*Reference*
Eisenia fetida	1000			Berends and Thus 1992a
Eisenia fetida	780			Berends and Thus 1992b
Thysanoptera Thripidae adults LC_{50} diet spray			140	Kubota, 1989

Aquatic Invertebrates

Mortality	LC_{50} *ppb*	*Reference*
Aedes aeygyti	0.3	Grosscurt and Wixley 1991
Aedes nigromaculis 48 h	0.5	Miura and Takahashi 1974
Chironomus decorus 4th instar midge (LC_{90}- 6.0 ppm)	1.9	Ali and Lord 1980
Clam shrimp *Eulimnadia* 48 h	0.15	Miura and Takahashi, 1974
Cricotopus sp. 4th instar to pupae	1.79	Hansen and Garton 1982
Daphnids 48 h	1.5	Miura and Takahashi, 1974
Daphnia magna 48 h adult	23.45	Majori and others 1984
Daphnia magna - 48 h neonate	0.75	Majori and others 1984
Daphnia magna 48 h - same study as full life cycle	4.42	Hansen and Garton 1982
Daphnia magna 48 h	6.89	Hansen and Garton 1982

Table 9-3, *continued ...*

Daphnia magna 48 h	4.55	Hansen and Garton 1982
Daphnia magna full life cycle	0.062	Hansen and Garton 1982
Dragonfly nymph *Orthemis, Pantala* 168 h	50	Miura and Takahashi, 1974
Eurytemora affinis copepod 48 h	2.2	Savitz, 1991
Gammarus psuedolinmaeus 96 h	25	Julin and Sanders 1978
Glyptotenidipes paripes 4th instar midge- (LC_{90} 4.1)	1.8	Ali and Lord 1980
Grass shrimp - 24 h	1.11	Touart and Rao 1987
Grass shrimp - 72 h	2.83	
Grass shrimp - 96 h	1.39	Wilson and Costlow, 1986
Grass shrimp larvae - 96 h	1.44	Wilson and Costlow 1987
Grass shrimp ovigerous females- 96 h	6985	Wilson and Costlow 1987
Grass shrimp post larval - 96 h	1.62	Wilson and Costlow 1987
Hyalella azteca - 96 h	1.84	Hansen and Garton 1982
Hyalella azteca - 96 h	2	Nebeker and others, 1983
Hydrophilus triangularis 48 h	100	Miura and Takahashi, 1974
Juga plicifera (FW mollusk)	no effect	Nebeker and others, 1983
Callibaetis Mayfly nymph - 168 h LC_{90}	10	Miura and Takahashi, 1974
Mysid shrimp - 96 h	1.97	Nimmo and others, 1981
Mysidopis bahia mysid 21 day	1.2	Nimmo and others, 1979
Perlodid stonefly - 96 h	57000	Mayer and Ellersieck, 1986
Snail *Physa* spp (>125 h)	125	Nebeker and others, 1983
Stone crab *Menippe mercenaria*	0.5	Costlow 1979
Tanytarsus dissimilis - 96 h 2-3 instar	1.02	Hansen and Garton 1982
Triops longicaudatus 24 h	0.75	
Grass shrimp 72 h technical[1]	2.95	Wilson and Costlow, 1986
Grass shrimp 96 h technical[1]	2.08	Wilson and Costlow, 1986
Grass shrimp 72 h Dimilin WP20	3.27	Wilson and Costlow, 1986
Grass shrimp 96 h Dimilin WP20	1.54	Wilson and Costlow, 1986
Eurytemora affinis 48 h nauplii	2.2	Miura and Takahashi, 1974

Table 9-3, *continued ...*

Mesocyclops thermocyclopoides 48 h copidodites	1000		Rao and Paul 1988
Water scavenger beetle larvae - 48 h	100		Miura and Takahashi, 1974
Aquatic invertebrates *Sublethal effects*	*EC_{50}* *ppb*	*NOEC* *ppb*	*Reference*
Brine shrimp (NOEC life span)		2	Cunningham, 1976
Chironomus plumoosus, 4th instar - 48 h larvae immoblilized	560		Julin and Sanders, 1978
Clam affected laminellar calcification	200000		Machado and others, 1990
Clistoronia magnifica caddisfly emergence inhibited	0.1		Nebeker and others, 1983
Crayfish *Orconectes virilis*		125000	Nebeker and others, 1983
Cricotopus spp. 7 day static test emergence stopped, molting affected	4.9		Nebeker and others, 1983
Daphnia magna - 24 h	68		Kuijpers, 1988
Daphnia magna - 48 h	71		Kuijpers, 1988
Daphnia magna - 48 h NOEC		0.45	Kuijpers, 1988
Daphnia magna 1st instar - 48 h immobilization	15		Julin and Sanders, 1978
Fiddler crab - behavior	0.2		Cunningham and Myers, 1987
Fiddler crab - molting	20		Cunningham and Myers, 1987
Fiddler crab - survival	2		Cunningham and Myers, 1987
Gammarus pseudolimnaeus - 48 h	30		Julin and Sanders, 1978
Gammarus pseudolimnaeus - 96 h	25		Julin and Sanders, 1978
Grass shrimp *Palaemontes pugio*	640		Fischer and Hall, 1992
Juga 3 wks		4.9	Nebeker and others, 1983
Mysid	2.1		Nimmo and others, 1981
Odonata 96 h	>1000		USEPA, 1989
Oyster *Crassostrea virginica* 96 h	>250000		USEPA, 1989
Physa Snail 3 wks		4.9	Willcox and Coffey, 1978
Quahogs 24 h		320	Surprenant, 1989
Stonefly *Skwala* spp. 96 h	57.5		Mayer and Ellersieck, 1986
Stonefly *Skwala* spp. 96 h	>100		

Table 9-3, *continued ...*

Tanytarsus dissimilis adult 7 day emergence stopped	1.6		
Tanytarsus dissimilis larva 5 day molting affected	4.9		Nebeker and others, 1983
Tanytarsus dissimilis 48 h	1.02		Hansen and Garton, 1982
Goeldichironomus holoprasinus 168 h LC_{90}	10		Miura and Takahashi, 1974
Copepods MATC		0.75	Savitz, 1991
Palaemonetes pugio MATC	0.3		Wilson and others, 1987
Aquatic Plants *Sublethal effects*	*EC_{50}* *ppb*	*NOEC* *ppb*	*Reference*
Marine diatoms		5000	
duckweed *Lemna gibba* 14 day	190		
4 chloroaniline *(Mortality)*	*LC_{50}* *ppm*		*Reference*
Brachydanio rerio 96 h	31		Lee and others 1993
Sublethal effects	*EC_{50}* *ppb*	*NOEC* *ppb*	*Reference*
Midge larvae 4th instar 48 h	43000		
Daphnia magna 48 h	200		Lee and others 1993

[1] Technical grade diflubenzuron

Table 9-4. Percent of population affected by diflubenzuron application

Application Rate	1.0 oz ai/ac				0.5 oz ai/ac			
Organism type	Mean Pop. reduct.	Risk of population reduction as large as: 50%	75%	90%	Mean Pop. reduct.	Risk of population reduction as large as: 50%	75%	90%
Spring Lepidopterans upper canopy LC_{50}	65	75	41	7	50	49	19	2
Spring Lepidopterans lower canopy LC_{50}	57	62	27	4	41	37	11	<1
Fall Lepidopterans upper canopy LC_{50}	44	42	15	2	29	21	5	<1
Fall Lepidopterans lower canopy LC_{50}	42	38	12	1	27	17	4	<1
Spring Lepidopterans upper canopy EC_{50}	95	99	95	69	91	97	86	47
Spring Lepidopterans lower canopy EC_{50}	93	98	91	58	85	93	77	37
Fall Lepidopterans upper canopy EC_{50}	87	93	78	42	76	83	61	23
Fall Lepidopterans lower canopy EC_{50}	86	94	78	37	75	82	58	19
Spring Orthopterans (grasshoppers) upper canopy	97	100	100	88	91	100	94	52
Spring Orthopterans lower canopy	95	100	98	72	85	98	84	31
Fall Orthopterans upper canopy	85	96	83	42	72	81	56	12
Fall Orthopterans lower canopy	84	93	85	31	69	81	49	8
Predatory insects	17	9	2	0	<1	1	0	0
Parasitic wasps of gypsy moth upper or lower canopy	> 99	100	100	100	> 99	100	100	100
Direct spray								
Benthic midges	20	10	6	1	5	2	1	0
Benthic crustaceans	98	100	100	60	92	100	100	60

Table 9-4, *continued ...*

Application Rate	1.0 oz ai/ac				0.5 oz ai/ac			
Organism type	Mean Pop. reduct.	Risk of population reduction as large as: 50%	75%	90%	Mean Pop. reduct.	Risk of population reduction as large as: 50%	75%	90%
Benthic insects (mayflies caddisflies)	60	90	2	0	58	87	1	0
Planktonic insects	>99	100	100	100	98	100	100	100
Planktonic crustaceans nymphs	98	100	100	62	81	98	74	22
Planktonic crustaceans adults	11	2	1	0	5	1	<1	<1
Residential forest pond								
Benthic crustaceans	89	100	99	38	89	100	99	36
Planktonic insects	98	100	100	100	98	100	100	100
Planktonic crustaceans nymphs	94	95	62	15	76	94	60	13
Residential forest stream								
Benthic midges	19	9	2	1	17	10	1	0
Benthic crustaceans	97	100	100	97	97	100	100	95
Benthic insects (mayflies caddisflies)	58	86	1	0	57	82	1	0
Planktonic insects	> 99	100	100	100	>99	100	100	100
Planktonic crustaceans nymphs	91	100	97	49	90	100	96	46
Forest pond								
Forest stream								
Benthic crustaceans	90	100	100	40	61	88	27	2

Table 9-4, *continued ...*

Application Rate	1.0 oz ai/ac				0.5 oz ai/ac			
Organism type	Mean Pop. reduct.	Risk of population reduction as large as: 50%	75%	90%	Mean Pop. reduct.	Risk of population reduction as large as: 50%	75%	90%
Planktonic insects	98	100	100	100	97	100	100	99
Planktonic crustaceans nymphs	76	95	61	15	44	34	10	0

Application Rate	0.33 oz ai/ac				0.25 oz ai/ac			
Organism type	Mean	Risk of population reduction as large as: 50%	75%	90%	Mean	Risk of population reduction as large as: 50%	75%	90%
Spring Lepidopterans upper canopy LC_{50}	39	34	9	<1	34	25	6	<1
Spring Lepidopterans lower canopy LC_{50}	32	22	5	<1	27	17	3	<1
Fall Lepidopterans upper canopy LC_{50}	21	12	2	<1	17	8	1	<1
Fall Lepidopterans lower canopy LC_{50}	19	9	2	<1	16	6	1	<1
Spring Lepidopterans upper canopy EC_{50}	84	92	75	34	80	89	68	26
Spring Lepidopterans lower canopy EC_{50}	78	87	64	27	74	82	56	19
Fall Lepidopterans upper canopy EC_{50}	68	73	46	14	63	67	40	10
Fall Lepidopterans lower canopy EC_{50}	66	71	43	11	61	64	37	8
Spring Orthopterans upper canopy	83	97	79	27	78	92	67	16
Spring Orthopterans lower canopy	75	89	60	13	68	82	47	7
Fall Orthopterans upper canopy	59	65	33	5	51	53	23	2
Fall Orthopterans lower canopy	56	59	25	2	48	48	16	1

Table 9-4, *continued ...*

Application Rate	0.33 oz ai/ac				0.25 oz ai/ac			
	Mean	Risk of population reduction as large as:			Mean	Risk of population reduction as large as:		
Organism type		50%	75%	90%		50%	75%	90%
Predatory insects	< 1	0	0	0	< 1	0	0	0
Parasitic wasps of gypsy moth upper or lower canopy	> 99	100	100	100	> 99	100	100	100
Direct spray								
Benthic midges	< 1	1	0	0	< 1	1	0	0
Benthic crustaceans	86	99	92	28	79	96	71	16
Benthic insects (mayflies caddisflies)	56	86	1	0	55	85	1	0
Planktonic insects	96	100	100	94	94	100	100	71
Planktonic crustaceans nymphs	71	83	48	11	61	65	34	7
Residential forest pond								
Benthic crustaceans	88	100	99	35	59	100	99	36
Planktonic insects	98	100	100	100	98	100	100	100
Planktonic crustaceans nymphs	76	94	59	12	76	94	59	14
Residential forest stream								
Benthic crustaceans	97	100	100	95	97	100	100	95
Benthic insects (mayflies caddisflies)	56	80	0	0	56	81	1	0
Planktonic insects	> 99	100	100	100	> 99	100	100	100
Planktonic crustaceans nymphs	90	100	95	44	90	100	95	45

Table 9-4, *continued ...*

Application Rate	0.33 oz ai/ac				0.25 oz ai/ac			
	Mean	Risk of population reduction as large as:			Mean	Risk of population reduction as large as:		
Organism type		50%	75%	90%		50%	75%	90%
Forest pond								
Planktonic insects								
Forest stream								
Benthic crustaceans	61	79	20	0	60	77	20	0
Planktonic insects	84	100	99	8	84	100	99	5
Planktonic crustaceans nymphs	40	29	8	0	40	28	8	0

Table 9-5. Risk of encountering a *B.t.k.* drop

24 BIU/ac application rate			
Caterpillar Size (mg)	Upper Canopy % Probability	Lower Canopy % Probability	Ground surface % Probability
0.75	12	2	2
2.5	90	40	32
5	99	82	76
10	100	98	97
50	100	100	100
100	100	100	100
40 BIU/ac application rate			
Caterpillar Size (mg)	Upper Canopy % Probability	Lower Canopy % Probability	Ground surface % Probability
0.75	42	8	6
2.5	99	76	69
5	100	96	94
10	100	100	100
50	100	100	100
100	100	100	100

Section 10
Conclusions

This section contains a comparison of the relative risks to the endpoints from each of the treatments (diflubenzuron, *B.t.k.*, Disparlure, NPV, and Dichlorvos). Risks are also compared between gypsy moth management strategies (suppression, eradication, and slow-the-spread) and between alternatives proposed in the environmental impact statement.

10.1. Comparative Risks Between Treatments

10.1.1. Forest Health

None of the treatments directly affect tree health, as none are phytotoxic or plant growth initiators (Table 10-1). Diflubenzuron, *B.t.k.* and NPV reduce defoliation though the reduction in number of leaf-eating stages of pest species, thus indirectly affecting forest health by limiting defoliation related damage. Diflubenzuron, *B.t.k.*, and NPV all act to reduce defoliation in the year following the spray by reducing the current gypsy moth population and, therefore, also reducing the number of egg masses produced. Disparlure and Dichlorvos used in mass trapping may also reduce the number of egg masses produced, thereby reducing the amount of defoliation the subsequent year.

By reducing the densities of defoliating insects, all the treatment act to reduce tree mortality and reduce the incidence of disease from secondary pests such as shoe-string fungus. Treatments that effectively reduce gypsy moth population densities present the least risk of gypsy moth mediated changes in forest health indicators.

10.1.2. Nontarget Species

Nontarget species are at risk from diflubenzuron, *B.t.k.*, and to a far less degree, the dichlorvos used in traps. Diflubenzuron affects far more species and is available to nontarget species for a longer time than *B.t.k.*.

Diflubenzuron primarily affects arthropods whose immatures feed on leaves. Vertebrates are generally not affected. In the aquatic environment, diflubenzuron affects planktonic and benthic crustaceans (*Daphnia*, scuds) and many aquatic insects

(especially mayflies, caddisflies, and Diptera). However, for both the terrestrial and aquatic organisms, only the juvenile life stages which undergo molting are at risk of mortality. Adults are not killed; however, fecundity may be affected.

B.t.k. puts fewer types of organisms at risk than diflubenzuron. Vertebrates are generally not affected. It primarily affects lepidopteran species that have larval stages in the spring. However, even many of these are only moderately sensitive or insensitive to dosages used. Due to lack of research, there is uncertainty about the effect of *B.t.k.* on some other types of invertebrate organisms, primarily the litter invertebrates. Dichlorvos primarily affects insects that enter gypsy moth traps, unless the traps are disturbed and the Vaportape strips are removed. Vertebrates (bears) encountering the strip are not at risk of mortality, although cholinesterase inhibition may occur.

Predators and parasites of invertebrates are indirectly affected by both *B.t.k.* and diflubenzuron. Bats, birds, and small mammals dependent upon invertebrates are at higher risk of a reduction in their prey base from diflubenzuron than *B.t.k.* because diflubenzuron affects more invertebrates for a longer period of time than *B.t.k.*. Species specializing in feeding on lepidopterans will have less risk of food reduction in *B.t.k.* sprayed sites than in sites treated with diflubenzuron because *B.t.k.* affects only some of those Lepidopteran species that are caterpillars in the spring.

Many of the species affected by diflubenzuron, such as aquatic crustaceans and insects, and some of the terrestrial insects, experience large natural fluctuations in population densities. These species are likely to recover from even the large population reductions that accompany the use of diflubenzuron. Species likely to suffer long-term reductions in population density due to diflubenzuron are those with low growth rates and low rates of dispersal, those restricted to spatially rare habitats, or those whose larvae feed in canopy gaps where ground cover plants may receive very high doses (for example, many butterflies). Recolonization of rare isolated habitats may be impossible. Species experiencing long-term effects due to *B.t.k.* spraying will include only certain lepidopteran species or predators or parasites of lepidopterans.

The specificity of NPV and Disparlure for gypsy moths reduces the risk that these treatments will directly affect nontarget species. Indirectly, species that rely on gypsy moths for a substantial portion of their diet will be forced to find new prey items; however, few species rely exclusively upon gypsy moths.

10.1.3. Water Quality

Many of the ecological indicators for water quality are adversely affected by increased defoliation; therefore, treatments that limit defoliation will reduce the risk of altering

these indicators. *B.t.k.* provides the greatest degree of foliage protection and would offer the least risk of defoliation-mediated changes in water quality. Varying degrees of defoliation are associated with the other treatments, all of which pose greater risks that defoliation will alter water quality than *B.t.k.*. Diflubenzuron is the only treatment with the potential to affect water quality by actions other than reducing defoliation. Diflubenzuron may allow algal densities to increase by reducing populations of algal grazers (crustaceans, insects). These increased algal densities should be quickly reduced by grazing from the remaining algal herbivores and their progeny. Any risk of altering water quality from NPV, Disparlure or dichlorvos will be associated with reductions in gypsy moth egg masses hatching in the year following spraying.

10.1.4. Microclimate

Changes in microclimate are related to defoliation and therefore treatments that reduce defoliation will also pose the least risk of defoliation-mediated changes in microclimate. *B.t.k.*, diflubenzuron and NPV have less risk of defoliation capable of altering microclimate than Disparlure and dichlorvos. Disparlure and the dichlorvos used in mass trapping will have no effect on the ecological indicators for microclimate during the year of application. Any reduction in risk from Disparlure or dichlorvos will occur in the subsequent year due to the effect of these treatments in reducing gypsy moth densities.

10.1.5. Soil Productivity and Fertility

Soil productivity and fertility are affected by a reduction in leaf litter caused by defoliation. Diflubenzuron, *B.t.k.*, and NPV provide the least risk of decreased leaf litter production. Disparlure and dichlorvos only affect defoliation in the year after spraying and therefore have the same risk of reduced litter production as gypsy moth infested areas.

Diflubenzuron does not pose a great risk to most litter invertebrates, bacteria, or fungi, with the possible exception of ground-dwelling spiders. *B.t.k.* may have an effect on litter invertebrates such as mites, carabids, and a nematode (Addison, 1994). Disparlure, dichlorvos and NPV do not increase risk of mortality to litter invertebrates or earthworms because of limited use or lack of toxicity. Other ecological indicators of soil productivity and fertility (erosion rate and soil pH) are not affected by any of the tactics.

10.2. Factors Affecting the Risk Associated with the Treatments

10.2.1. Size of Treated Area

The larger the area treated, the more difficult it will be for species to reinvade areas from which they were removed by the specific treatments (for example, invertebrates removed by diflubenzuron or lepidopterans removed by *B.t.k.*). Small patches, if not retreated, can be reinvaded more easily than large areas due to the proximity of untreated areas and their populations of species unaffected by the treatment.

10.2.2. Frequency of Treatment Within a Growing Season

More frequent applications of diflubenzuron will present a greater risk to organisms susceptible to diflubenzuron, but living in habitats where diflubenzuron is not persistent (streams, ponds, soil). More frequent applications in habitats where diflubenzuron is persistent throughout the growing season (litter, on vegetation) will present the same degree of risk to susceptible organisms in these habitats as would a single increased rate of diflubenzuron.

More frequent applications of *B.t.k.* would put more lepidopteran species at risk than a single application due to the rapid half-life of *B.t.k.*. *B.t.k.*, used only once, would affect only some of the leaf-eating lepidopterans present within a week of the spray. Multiple applications within the same year may put different species at risk as new caterpillars hatch throughout the spray period. There is a data gap as to the effect of multiple applications of *B.t.k.* on caterpillars unaffected by the initial spray.

More frequent applications of Disparlure, NPV, and dichlorvos should not increase the risk posed by those tactics to nontarget species.

10.2.3. Number of Years of Treatment

Treatment of the same area over multiple years would prolong the effects noted for each tactic for a single year. Invertebrate populations would be reduced for several years rather than one if diflubenzuron were for consecutive years. Diflubenzuron concentrations would gradually increase in the litter as all the diflubenzuron from the first year's application will not degrade before the second year's application. There is a data gap as to the effect of applying *B.t.k.* one year and diflubenzuron the next. It is not known if there are cumulative effects from this type of application pattern. There is also a data gap concerning the effect of applying *B.t.k.* for multiple years.

10.2.4. Life History Strategies of Susceptible Organisms

Organisms which have multiple generations per growing season would be at greater risk from diflubenzuron than those organisms with a single generation. The toxic effects of diflubenzuron would affect each generation causing a greater cumulative population reduction than for organisms producing a single generation.

Organisms susceptible to diflubenzuron or *B.t.k.* which occupy specialized habitats that are widely separated from other such habitats may be removed from an area sprayed with these insecticides. This is especially true of aquatic organisms living in sinks or caves which may receive water containing diflubenzuron, or terrestrial organisms dependent upon patchily distributed plant species.

Species feeding on plants found in canopy gaps will be exposed to higher concentrations of sprayed insecticides (diflubenzuron or *B.t.k.*) and will experience higher mortality rates than those species in the canopy. Examples of species of this type would include butterfly larvae dependent upon grasses or certain herbaceous plants which are normally not able to grow in the forest due to the low light levels under the canopy.

10.3. Comparative Risk Between Strategies

Treatments are not uniquely associated with only one of the strategies--eradication, suppression, and slow-the-spread. Many strategies use essentially the same treatments and the same application rates of the insecticides. The largest difference between strategies lies not in the treatments, but in the geographic areas treated. Slow-the-spread is used only along the leading edge of the gypsy moth front and is currently applied to the smallest amount of acreage of any strategy. Eradication is applied to more acreage than slow-the-spread, but less than suppression, which is applied to the largest amount of acreage as it can be used throughout the generally infested area. Individual eradication and suppression projects can vary in size. In some states, eradication projects can be quite small, and can be smaller than suppression projects in other states. On a very broad scale, the strategy applied to the smallest amount of acreage also poses the least risk of affecting those nontarget species at risk from the various treatments. However, in areas where gypsy moth is already established, the strategy applied to the smallest amount of acreage is also the strategy with the greatest risk, on a very broad scale, of altering forest health, microclimate, water quality, and soil productivity and fertility by the gypsy moth.

The effect of the strategy on the risks of gypsy moth mediated change in forest health, water quality, microclimate, or soil productivity and fertility also differs between strategies. In eradication and suppression, gypsy moth populations are higher than in the slow-the-spread strategy, and thus are capable of potentially changing some of the

endpoints if not controlled. In slow-the-spread, populations of gypsy moths are low in the treatment areas. The risk to forest health posed by these low densities is much lower than risk to forest health from the higher densities of gypsy moths treated by eradication or suppression. Treatment of gypsy moth populations at low densities may prolong the period during which the forest experiences low risk of changing forest health due to gypsy moth infestations. Treatment of gypsy moths at low densities may not reduce the risks associated with a high density gypsy moth infestation in the future.

10.4. Comparative Risk Between Alternatives

10.4.1. Forest Health

The risk of altering the ecological indicators for forest health is related to the degree of gypsy moth damage, the number of years the infestation causes heavy defoliation, and the original species composition of the forest. Forests composed primarily of susceptible species have a greater risk of changing the ecological indicators of forest health than forests with few susceptible species. Likewise, there is greater risk to forests defoliated for multiple years rather than once. Risk to forest health from gypsy moth is therefore a range of risk based on forest characteristics throughout the program area, rather than a single level of risk for all forests.

For all of the ecological indicators of forest health, the risk of changing the indicators is greater for no active management than for any of the alternatives that include control strategies (suppression, eradication or slow-the-spread). The difference in risk between no active management and the other alternatives will depend upon characteristics of the forest and application rates of insecticides. In the most susceptible forests (greater than 60 percent defoliation) experiencing high rates of insect damage, risk of increased tree mortality, increased incidence of secondary disease, decreased diameter growth, and change in species composition is great. In forests experiencing defoliation of 30 to 60 percent or less for a single year, these risks are very much lower.

10.4.2. Nontarget Species

Each alternative poses some risk to some of the ecological indicators for changes in nontarget species numbers or population densities. Reduction of food for organisms consuming oak mast (acorns), and reduction in nutritional quality and number of leaves available for summer or fall leaf-eating, phloem-feeding, leaf-mining, or sucking organisms will alter abundance of grey squirrels, deermice, deer, and probably some phytophagous invertebrates in the no active management strategy.

Increased numbers and growth of understory plants following tree mortality or defoliation will cause changes in the small mammal, deer, and bird species using the area. Defoliation allows a shrubby undergrowth to develop which causes the bird species diversity to change (increase). However, some bird species require forest interiors and are sensitive to gaps and edges such as ones created by defoliation. Predation and nest parasitism could increase whenever defoliation is heavy. In severe defoliation, aquatic species may be affected as water temperatures rise and nutrient concentrations increase.

The other alternatives, if diflubenzuron is used, present a high risk for phytophagous invertebrates which consume entire leaves or the leaf outer surface. This risk is reduced or eliminated for all except a few leaf-eating Lepidopteran species and possibly some litter invertebrates when *B.t.k.* is used. Risk to diflubenzuron susceptible species is increased in eradication programs using higher application rates or multiple applications, as is risk to lepidopterans if higher rates and multiple applications of *B.t.k.* are used. There is a data gap concerning the effect of multiple treatments on lepidopterans. It is uncertain whether species which survive one treatment will also survive a second or third treatment within the same season. Although recent research indicates that for species which are moderately resistent to the effects of *B.t.k.*, multiple exposures may not cause additional mortality over that suffered in the first exposure (Section 9.2.2.1) Species depending upon invertebrates as food items will either change food preferences, spend more time foraging, or leave the area. Fewer types of species are affected when Disparlure and NPV are used exclusively due to the specificity of these insecticides.

10.4.3. Water Quality

Effects of gypsy moth infestations on water quality are related to gypsy moth densities and the amount of defoliation. Heavy infestations in forests composed primarily of susceptible species will have greater risk of changing water temperature than lower density infestations. The risk of altering the indicators of water quality is greater in the no active management alternative than the other alternatives due to effects of gypsy moths. Gypsy moth infestations can increase water temperature though defoliation, increase nutrient concentrations by converting leaves into frass, and for infestations which cause tree mortality can alter water flow rate and water yield.

10.4.4. Microclimate

Changes in microclimate are primarily caused by defoliation. The risk of changing the ecological indicators for microclimate is greater for the no active management alternative than for the other alternatives. As with other endpoints affected by gypsy

moth infestations, the higher the population densities and greater percentage of susceptible trees in the forest, the more risk there will be to changing microclimate.

10.4.5. Soil Productivity and Fertility

Risk to the ecological indicators of soil productivity and fertility differ between the no active management alternative and the other alternatives. No active management increases the risk of changing the nutrient content of litter and litter invertebrate populations. Short-term studies suggest decomposition rates are not changed by tactics used in the other alternatives; however, no long term studies have been conducted.

Table 10-1. Qualitative relative risk estimates by tactic

Endpoint	Ecological Indicator	Diflubenzuron	*B.t.k.*	NPV	Disparlure	Dichlorvos
Change in Forest Health	Forest productivity, tree growth rates, mast production	Minimal	Minimal	Minimal	Minimal	Minimal
	Successional state, stand age, species composition of tree and understory species	Minimal	Minimal	Minimal	Minimal	Minimal
	Susceptibility to fire	Minimal	Minimal	Minimal	Minimal	Minimal
	Incidence of disease, tree mortality rates, degree of insect damage	Minimal	Minimal	Minimal	Minimal	Minimal
Change in nontarget species numbers or population densities	Species richness of mammals, birds, reptiles, amphibians, fish, invertebrates	High (Decrease in numbers of invertebrate species)	High (Decrease in number of lepidopteran species)	Minimal	Minimal	Minimal
	Population densities of groups of special concern: spring-feeding native lepidopterans, summer-feeding native lepidopterans, insect predators and parasites, pollinators, amphibians, mollusks	High (Decrease in lepidopteran populations throughout the growing season, possible reduction parasitoid species)	High (Decrease in lepidopteran populations feeding on leaves during the spring)	Minimal	Minimal	Minimal
	Population densities of game species: wild turkeys, deer, black bears, trout. salmon	Low (Decrease in fish populations, especially in the forested residential ecosystem)	Minimal	Minimal	Minimal	Minimal
	Population densities of organisms eaten by game fish: crustaceans, aquatic insects, small fish	High (Large decreases in populations of aquatic invertebrates; small population decreases in fish)	Minimal	Minimal	Minimal	Minimal
Change in water quality	Water temperature, dissolved oxygen concentration	Minimal	Minimal	Minimal	Minimal	Minimal

Table 10-1, *continued ...*

Endpoint	Ecological Indicator	Diflubenzuron	*B.t.k.*	NPV	Disparlure	Dichlorvos
	Nutrient concentration, algal densities	Minimal - Slight (Increases in algal populations)	Minimal	Minimal	Minimal	Minimal
	Flow rate, water yield and sediment load	Minimal	Minimal	Minimal	Minimal	Minimal
	Detrital decomposition rate	Minimal	Minimal	Minimal	Minimal	Minimal
Change in microclimate	Percent defoliation, amount of light penetrating canopy, soil and litter temperature, relative humidity below the tree canopy	Minimal	Minimal	Minimal	Minimal	Minimal
Change in soil fertility, productivity or stability	Population sizes of organisms that alter soil composition or texture: litter invertebrates, earthworms, bacteria and fungi	Moderate (Decreases in ground spiders and mite populations	Moderate (Change in nematode diversity: decreases in ground beetles)	Minimal	Minimal	Minimal
	Litter production, concentration of organic material, decomposition rate	Minimal	Minimal	Minimal	Minimal	Minimal
	Soil pH	Minimal	Minimal	Minimal	Minimal	Minimal
	Erosion rate	Minimal	Minimal	Minimal	Minimal	Minimal

GLOSSARY

ac: acre

Acute Toxicity: The potential of a substance to cause injury or illness when given in a single dose or in multiple doses over a period of 24 hours or less.

a.i.: active ingredient

Arthropod: Invertebrates characterized by a hard, chitinous exoskeleton and jointed appendages. Includes crustaceans, centipedes, millipedes, spiders and mites, and insects.

Benchmarks: Results of toxicological tests, such as LC_{50} or EC_{50} values.

Beneficial Organism: Any organism that eats, parasitizes, or regulates in some way populations of other organisms that are pests.

Benthic: Of or pertaining to the bottom of the sea, lake, or stream. The habitat in which many aquatic organisms live.

Biodiversity: The variety of life, and its processes. It includes the variety of living organisms, the genetic differences among them, and the communities and ecosystems in which they occur.

Biomass: Total weight, volume, or energy equivalent of organisms in a given area.

Biota: Plants and animals.

BIU: Billion International Units

Carcinogenicity: The ability of a substance to produce cancer.

Chitin: A chain of a molecules, N-acetylglucosamine, produced by arthropods to form the hard exoskeleton. Also produced in the cell walls of fungi.

Chronic Toxicity: An adverse biologic response, such as mortality or an effect on growth or reproductive success, resulting from repeated or long-term (equal to or greater than 3 months) doses (exposures) of a compound usually at low concentrations.

Community: An association of potentially interacting plants and/or animals, more or less distinguishable from other such associations, usually defined by the nature of

their interaction or the place in which they live.

Conspecific: Belonging to the same species.

Cover Type: The vegetation, described in terms of its general form or dominant species, comprising the plant community in a given area.

Crustacean: Organisms such as crabs, lobsters, shrimp, crayfish, wood lice, pill bugs, and water fleas. Crustaceans have hard exoskeletons made of chitin, as do other members of the Phylum Arthropoda (Arthropods).

Data gap: A lack of information on a particular subject.

Degradation: Breakdown of a compound by physicochemical or biochemical processes into basic components with properties different from those of the original compound.

Dermal: Refers to the body surface of an animal.

Detritus: Freshly dead or partially decomposed organic matter.

Developed Forest: Privately-owned, forested, residential areas.

Dipteran: An insect belonging to Diptera. A large order of insect including flies and mosquitoes.

Direct Effect: The reaction of an organism after exposure to a control or eradication method, chemical or nonchemical, not mediated through another organism. For example, caterpillars that eat leaves with diflubenzuron on them fail to molt and die as result of their direct exposure to this pesticide. As well, the direct effect of an unchecked gypsy moth infestation could be the change in species composition of trees in the woodland.

Dose: A quantity of material that is taken into the body; dosage is usually expressed in amount of substance per unit of animal body weight often in milligrams of substance per kilogram (mg/kg) of animal body weight, or other appropriate units; in radiology, the quantity of energy or radiation absorbed.

EC_{50}: The concentration of a substance that results in some effect exhibited by 50% of the test organisms. Also called the Median Effective Concentration.

Ecoregion: Large geographical areas usually containing many different ecosystems which share general characteristics of climate, vegetation, and topography, and that are distinct from other such regions.

Ecosystem: A relatively self-contained ecological system defined by the types of organisms found in it and their interactions, e.g. forest, grassland, soil, etc.

Endpoint: Structural or functional property of the ecosystem of ecological or societal importance.

Ephemerata: An order of aquatic insects including mayflies.

Eradication: Elimination of gypsy moth from an area infested as a result of artificial movement of gypsy moth life stages from the generally-infested area.

Exposure: Contact, through skin contact, inhalation, or ingestion, with a substance that may have a harmful effect.

Exuviae: Cast off skins or outer coverings of insects and other animal which shed skin.

FIFRA: The Federal Insecticide, Fungicide, and Rodenticide Act. This act establishes procedures for the registration, classification, and regulation of pesticides.

Food Chain: A feeding sequence used to describe the flow of energy and materials through the system.

Food Web: The interconnected food chains in the ecosystem, representing the various paths of energy flow through populations in the community.

Forest: Land at least 10 percent occupied by forest trees or formerly having had such tree cover and not currently developed for non-forest use. Lands developed for non-forest use include areas for crops, improved pasture, residential, or administrative areas, improved roads of any width, and adjoining road clearing and power line clearing of any width.

Formulation: The way in which a pesticide is prepared for practical use; includes preparation as wettable powder, granular, or emulsifiable concentrate; a pesticide preparation supplied by a manufacturer for practical use; a pesticide product ready for application; also, refers to the process of manufacturing or mixing a pesticide product in accordance with the EPA-approved formula.

Fumigant: pesticide applied a liquid or powder which volatilizes to gas; usually applied beneath a tarp, sheet, or other enclosure.

Fumigation: The process of using a fumigant to destroy pests, usually applied under a cover or shelter.

FWS: The Fish and Wildlife Service, an agency of the U.S. Department of the Interior.

g: gram

Gene: The basic unit of inheritance, by which hereditary characteristics are transmitted from parent to offspring. Genes consist of short lengths of DNA (or RNA in some viruses) that direct the synthesis of protein. These in turn influence the form and function of the organism.

Generally infested area: The area where gypsy moth lives permanently.

Genotoxicity: A specific adverse effect on the genome (the complement of genes contained in the **haploid** set of **chromosomes**) of living cells, that upon the duplication of the affected cells can be expressed as a **mutagenic** or a **carcinogenic** event because of specific alteration of the molecular structure of the genome.

Geocorid: A Big-eyed bug

Growth regulator: A chemical that controls the rate of growth, or interferes with successful growth in an animal. Diflubenzuron is a growth regulator for insects and other chitinous animals.

Guild: A group of species that makes a living in a similar way.

ha: hectare

Half-life: The time required for the concentration of a chemical to decrease by 50%; a measure of the persistence of a chemical in a specific set of circumstances (the greater the half-life, the more persistent a chemical is likely to be).

Hazard: The potential adverse effect of a pesticide; the intrinsic ability of a stressor to cause adverse effects under a particular set of circumstances.

Hazard Assessment: A component of risk assessment that consists of the review and evaluation of toxicological data to identify the nature of the hazards associated with a chemical, and to quantify the relationship between dose and response.

Hemipterans: An insect belonging to the Order Hemiptera including the true bugs.

Homopterans: An insect in the order Homoptera, an order which includes aphids, scale insects and cicadas.

Host: Any plant or animal that supports in some way, often as a food source, part or all of the life history of another plant or animal.

In vitro: In glass; a test-tube culture; any laboratory test using living cells taken from an organism.

In vivo: In the living organism; *in vivo* tests are those laboratory experiments carried out on whole animals or human volunteers.

Indirect Effect: The reaction of an organism to a change in the environment that is a direct result of some control or eradication method. For example, wasps that prey on caterpillars that eat leaves with diflubenzuron on them could obtain diflubenzuron through their caterpillar prey, thus being exposed indirectly to the chemical. As well, the indirect effect of an unchecked gypsy moth infestation could be the change in the bird community that results from the change in woodland structure, a direct effect of the gypsy moths themselves.

Insecticide: A pesticide that kills, debilitates, or controls the growth of insects.

Instar: A larval stage of an insect

Integrated Pest Management: Use of many different types of control methods, such as chemical, biological, and cultural controls, to manage pest damage below economically damaging levels.

IPM: See Integrated Pest Management

IU: International Unit

Land Use: The type of activity occurring on the land surface,
e.g. forestland, farmland, pastureland, etc.

Landscape: The physical features of an area (e.g. slope, aspect, drainage) that affect the characteristics of the plant and animal communities in the ecosystem.

Latin Hypercube: A stratified sampling technique designed to sample from all portions of a distribution.

LC_{50}: See Median Lethal Concentration.

LD_{50}: See Median Lethal Dose.

Leaf Expansion: The percentage of leaf growth from 0 to 100.

Lentic: Water bodies which do not flow (e.g. lakes, ponds)

Lepidopteran: An insect belonging to the order Lepidoptera including moths, butterflies, and skippers.

Lethal Concentration: A concentration of a substance in water or air that is lethal to a test organism.

Lethal Dose: A dose of a substance that is lethal to a test organism.

Lotic: Water bodies which flow and have running waters (e.g. streams, rivers)

Lognormally: Distributed as in a log normal distribution such as a logarithmic function with a normal distribution.

Median Lethal Concentration: The concentration of a toxicant necessary to kill 50% of the organisms in a population being tested; usually expressed in part per million (ppm), milligrams per liter (mg/L), or milligrams per cubic meter (mg/m^3).

Median Lethal Dose: The dose necessary to kill 50% of the test organisms; usually expressed in milligrams of chemical per kilogram of body weight (mg/kg).

mg/cm^2: milligrams per square centimeter

mg/kg: milligram per kilogram

mg/m^3: milligram per cubic meter

Microorganism: An organism so small that a microscope is necessary to see it.

Monte Carlo simulation: A technique used to simulate systems with probabilistic elements. One or more variable in a Monte Carlo simulation is determined by drawing a random number from a probability distribution (such as the normal or uniform distribution) which describes the natural variation in that variable.

Mutagenicity: The ability of a substance to produce a mutation, an alteration of the chemical linkages in the DNA.

Nabid: A damselbug belonging to Order Hemiptera of Class Insecta.

ng: nanogram, one billionth of a gram

nm: nanometer, One billionth of a meter

NOEL: No Observed Effects Level. The dose of a chemical at which no treatment related effects were observed in a laboratory toxicological test.

normal distribution: A theoretical frequency distribution of variable data generally shaped in a bell-shaped curve.

Nontarget Organism: A plant or animal that is not the object of a control or eradication effort.

Oral: Relating to the mouth.

Oral Toxicity: The toxicity of a compound when given or taken by mouth, usually expressed as milligrams of chemical per kilogram of body weight of animal.

Orthopterans: An insect in the Order Orthoptera, including grasshoppers, crickets, locusts, and cockroaches.

Ovicide: A chemical toxic to the eggs of the target animal.

Persistence: The characteristic of an insecticide or a compound to remain in the environment as an effective residue; persistence is related to volatility, chemical stability, and biodegradation.

Pesticide: A substance or mixture of substances that kill insects, rodents, fungi, weeds, or other forms of plant or animal life that are considered to be pests.

Planktonic: Suspended in the water column of seas, lakes, rivers, or other water bodies.

Plecoptera: An order of insects including the stoneflies.

Phytoplankton: Small algal cells suspended in the water column of water bodies.

Phytotoxic: Toxic or harmful to plants

Population: A group of individuals of one species in an area, though the size and nature of the area is defined, often arbitrarily, for the purposes of study being undertaken.

Potentiation: The action of two or more substances from which one or more (the potentiator) enhances the toxicity of another.

ppb: Parts per billion; the number of parts of chemical substance per billion parts of the substrate in question.

ppm: Parts per million; the number of parts of chemical substance per million parts of the substrate in question.

probit analysis: An analysis technique that relates doses to measures of standard deviation away from the 50% response level using the cumulative normal distribution.

Recreational Forest: Publicly-owned forest used predominantly for hiking, hunting, camping, day-use, and sight-seeing.

Residue: Quantity of pesticide and its metabolites remaining on and in a crop, soil, or water.

Resistance: The ability of a population or ecosystem to absorb an impact without significant change from normal fluctuations; for plants and animals, the ability to withstand adverse environmental conditions and/or exposure to toxic chemicals or disease.

Risk: The probability that a particular harmful event will occur.

Safety factor: An factor used to give a margin of error to the screening index. Safety factors are selected based on the amount of error likely in the estimation of the toxicological benchmark (LC_{50} or EC_{50}) or estimating concentrations of the toxicant in the environment.

Screening index: A unitless index used to determine whether a species exposed to a toxicant is potentially at risk or not. The screening index is a conservative estimate of those species at risk. The screening index is less likely to miss species that are at risk and more likely to indicate a species at risk when it may not actually be at risk.

Species Composition: The assemblage of species inhabiting a defined area.

Species Richness: The number of species in a local area, region, or community.

Species Diversity: An ecological concept that incorporates both the number of species in a given area and the number of individuals per species.

Stressors: Entities which cause stress to the ecosystem. These can be chemical or biological insecticides such as diflubenzuron or B.t.k., or they can be organisms such as the gypsy moth.

Subchronic Toxicity: Adverse biologic response of an organism, such as mortality or an effect on growth or reproductive success, resulting from repeated or short-term (3 month) doses (exposures) of a compound, usually at low concentrations.

Succession: Replacement of populations in a habitat through a regular progression to a stable state.

Suppression: Reduction of gypsy moth populations in heavily infested areas.

Synergism: The action of two or more substances to achieve an effect of which each is individually incapable; synergistic effects may be greater or less than the sum of effects of the substances in question.

Systemic: Entering and then distributing throughout the body of an organism.

Teratogenicity: The ability of a substance to produce birth defects, evident at birth or shortly thereafter.

Toxic: Poisonous to organisms.

Toxicant: A poisonous substance such as the active ingredient in pesticide formulations that can injure or kill plants, animals, or microorganisms.

Toxicity: The capacity of a substance to cause adverse effects, a specific quantity of a substance which may be expected, under specific conditions, to do damage to a specific living organism.

triangular distribution: A theoretical frequency distribution shaped like a triangle and described by a minimum, maximum and likeliest value.

Trophic Levels: Feeding level - for example, primary producer, herbivore, and first-level carnivore.

Uniform distribution: A theoretical frequency distribution described by a minimum and a maximum value. All values in the uniform distribution have an equal probability of occurrence.

Volatility: The tendency of a substance to evaporate at normal temperatures and pressures.

Subject Index

Asian gypsy moth, 2-2, 2-3 to 2-4, 2-5, 2-7, 2-8, 2-80, 5-31, 5-71
***Bacillus thuringiensis* var. *kurstaki* (*B.t.k.*)**
application rates, 3-8, 7-22
by strategy, 3-8
used in models, 7-22
B.t.k. and NPV interactions, 5-10
characteristics of
crystals, 2-7, 3-2, 5-1, 7-14, 7-15, 7-17, 7-21, 7-46, 8-4, 8-5, 9-9, 9-12
delta-endotoxin, 2-7, 5-1, 7-14, 7-15, 7-16, 7-46
description of, 5-1, 7-14
growth cycle, 7-14, 7-46
half-life, 7-15, 8-5
persistence, 7-15, 7-16
spores, 5-6, 5-50, 7-14, 7-15, 7-16, 7-17, 7-21, 7-22, 7-46, 8-4, 8-5, 9-9
ecological effects (see B.t.k. risk)
effects on
aquatic invertebrates, 5-2, 5-19, 5-53, 5-95,
beneficial predatory insects, 5-8
birds, 5-9, 5-50, 5-77, 9-11
fish, 5-20, 5-50, 5-95
Lepidoptera, 5-1, 5-5 to 5-6, 5-19, 5-51 to 5-52, 5-70 to 5-73, 7-14, 7-15, 9-9, 9-10, 9-11, 9-12, 9-41, 10-9
mammals, 5-9 to 5-10, 5-50, 7-14, 9-11
parasites of Lepidoptera, 5-6, 5-74 to 5-76, 9-11
parasitic wasps, 5-6, 5-52, 9-11
soil invertebrates, 5-8, 9-11, 9-12, 10-9, 10-10
terrestrial invertebrates, 5-7, 9-11, 9-12, 9-41, 10-9, 10-10
exposure
through ingestion, 8-5
in aquatic environments, 8-5
fate 7-14 to 7-17
aquatic environments, 7-16, 7-45
in the developed forest, 7-17
in soil, 7-16
in water, 7-16
in the undeveloped forest, 7-17
persistence, 7-15, 7-16
terrestrial environments, 7-44
effect of moisture, 7-15
effect of temperature, 7-15
effect of ultraviolet radiation, 7-15
fate and transport modeling
methodology, 7-22 to 7-23, 7-25
estimated environmental concentration
in water resources, 7-33, 7-59
on leaves, 7-33, 7-59
on litter, 7-33, 7-59
on soil, 7-33, 7-59
risk
birds, 9-11
forest health, 9-12, 10-9
Lepidoptera, 9-10 to 9-11, 9-41, 10-9
methodology, 9-9 to 9-10
microclimate, 9-12, 10-10
parasitic wasps, 9-11
soil invertebrates, 9-11, 9-12, 10-9, 10-10
soil productivity and fertility, 9-12, 10-10
vertebrates, 9-11
water quality, 9-12, 10-9
toxicity of (see B.t.k. effects on), 5-1 to 5-2
aquatic invertebrates, 5-53
beneficial predatory insects
birds, 5-9, 5-50
fish, 5-50
Lepidoptera, 5-51, 7-14
mammals, 5-50
terrestrial invertebrates, 5-51 to 5-53
***B.t.k.* (see *Bacillus thuringiensis* var. *kurstaki*)**
DDVP (see dichlorvos)
Developed forest ecosystem
description of
climate, 4-3
fauna,
aquatic, 4-8
terrestrial, 4-6
flora, 4-5
hydrology, 4-4
soils, 4-3
B.t.k. residues, 7-17, 8-5, 8-6
diflubenzuron residues
aquatic, 7-13, 7-30 to 7-31, 7-32, 7-33, 7-55 to 7-58, 8-4, 9-6, 9-7, 9-25, 9-27
terrestrial, 7-13, 8-1, 8-2, 8-4
Dichlorvos
application
description of, 7-31
rates, 3-8
chemical characteristics
half-life on air, 7-21

Dichlorvos
chemical characteristics (continued)
half-life in soil, 7-21
ecological effects (see dichlorvos, risk)
effects on
amphibians, 5-4, 5-18
aquatic invertebrates, 5-4, 5-18, 9-13
beneficial predatory insects, 5-4, 5-18
birds, 5-4, 5-18, 5-67
fish, 5-4, 5-18, 5-67
mammals, 5-4, 5-18, 5-67, 9-13
reptiles, 5-4, 5-5, 5-18
terrestrial invertebrates, 5-4, 5-18, 5-67, 9-13
exposure
dermal, 8-6
ingestion, 8-6
fate , 7-33
risk, 9-13 to 9-14
bears, 9-14
invertebrates, 9-13
toxicity
amphibians, 5-4
aquatic invertebrates, 5-4, 5-18
beneficial predatory insects, 5-4, 5-18
birds, 5-4, 5-18, 5-67
fish, 5-4, 5-18, 5-67
mammals, 5-4, 5-18, 5-67
reptiles, 5-4, 5-5, 5-18
terrestrial invertebrates, 5-4, 5-18, 5-67
Diflubenzuron
application rates
by strategy, 3-8
used in models, 7-22
characteristics of
degradation products
4-chloroaniline, 5-3, 5-29, 7-9, 7-26, 7-35, 7-36, 7-42, 7-55, 7-56, 7-57, 7-58, 8-4, 9-2, 9-7, 9-28
4-chlorophenylurea, 5-3
description of, 5-2 to 5-3, 7-2 to 7-3
half-life on vegetation, 7-5
half-life in soil, 7-5
half-life in water, 7-7
persistence, 7-2, 7-3, 7-7
ecological effects (see diflubenzuron, risk,)
effects on
amphibians, 5-2, 5-17 to 5-18
aquatic invertebrates, 5-3, 5-20 to 5-24, 5-59 to 5-63, 5-96 to 5-98, 9-32 to 9-35
beneficial predatory insects, 5-3, 5-11 to 5-12, 5-57, 5-80 to 5-86,
birds, 5-2, 5-16 to 5-17, 5-54, 5-93 to 5-94
fish, 5-3, 5-54 to 5-55, 5-21, 5-22, 5-23, 5-96, 9-35
Lepidoptera, 5-10, 5-58, 5-78 to 5-79, 9-29 to 9-30
mammals, 5-2, 5-17, 5-54
parasites of Lepidoptera, 5-12, 5-85 to 5-86
reptiles, 5-2
soil invertebrates, 5-55
terrestrial invertebrates, 5-2 to 5-3, 5-14 to 5-16, 5-55 to 5-59, 5-78 to 5-92, 9-31 to 9-32
exposure
4-chloroaniline, 8-4
dermal, 8-3
ingestion, 8-1 to 8-3
in aquatic environments, 8-3 to 8-4
fate
in the canopy, 7-2 to 7-3,
degradation, 7-24
persistence, 7-2, 7-9
washoff from canopy leaves, 7-2
wind abrasion of diflubenzuron, 7-2
in the developed forest, 7-13
on the ground, 7-3 to 7-5
effect of microbial degradation on, 7-4
effect of temperature on, 7-5
persistence, 7-3
of metabolites, 7-9 to 7-12
in the water, 7-5 to 7-8
effect of microbial degradation on, 7-7
effect of pH on, 7-7
effect of organic matter on, 7-6
effect of temperature on, 7-7 to 7-8
persistence, 7-6, 7-7
in undeveloped forest, 7-12 to 7-13
fate and transport modeling, 7-22 to 7-31,
methodology, 7-22 to 7-31, 7-48 to 7-52
estimated environmental concentration, 7-53 to 7-59
aquatic sediments, 7-55 to 7-58
leaves, 7-55 to 7-58
litter, 7-55 to 7-58
metabolites, 7-55 to 7-58
runoff water, 7-55 to 7-58
water resources, 7-55 to 7-58
risk
aquatic insects, 9-6 to 9-7
birds, 9-6
crustaceans, 9-6 to 9-7
earthworms, 9-5
fish, 9-7

Diflubenzuron
risk (continued)
forest health, 9-8, 10-1
from multiple applications
methodology, 9-1 to 9-4
microclimate, 9-8, 10-3
parasitic wasps, 9-5, 9-36, 9-39
percent population reduction
aquatic organisms, 9-36 to 9-38, 9-39 to 9-40
Lepidoptera, 9-4, 9-36, 9-38
invertebrate predators, 9-36, 9-39
Orthoptera, 9-36, 9-38
parasitic wasps of gypsy moth, 9-36, 9-39
screening index, 9-1, 9-2, 9-3, 9-4, 9-25
spiders, 9-5
soil productivity and fertility, 9-8 to 9-9, 10-3
terrestrial invertebrates, 9-4 to 9-6
vertebrates, 9-5 to 9-6
water quality, 9-8, 10-2 to 10-3
toxicity
amphibians, 5-2
aquatic invertebrates, 5-3, 5-59 to 5-63, 9-32 to 9-35
beneficial predatory insects, 5-3, 5-57
birds, 5-2, 5-54
fish, 5-3, 5-54 to 5-55
Lepidoptera, 5-58, 9-29 to 9-30
mammals, 5-2, 5-54
reptiles, 5-2
soil invertebrates, 5-55
terrestrial invertebrates, 5-2 to 5-3, 5-55 to 5-59, 9-31 to 9-32
Dimilin (*see also* diflubenzuron)
Dimilin 4L, 5-2, 5-5, 5-8, 5-52, 5-75
Dimilin 25W, 5-57, 5-78
Dipel (*see also Bacillus thuringiensis kurstaki*), 5-2, 5-5, 5-8, 5-25, 5-35, 5-36, 5-40, 5-45, 5-48, 5-50, 5-51, 5-52, 5-53, 5-70, 5-74, 5-75, 5-95
Disparlure
application rate
by strategy, 3-8
characteristics
description of, 5-5, 5-19
effects on
birds, 5-69, 9-12
fish, 5-79, 9-12
mammals, 5-79
ecological effects (see disparlure, risk)
exposure, 8-6
fate , 7-20, 7-33
risk
forest health, 9-13
microclimate, 9-13
nontarget species, 9-12
soil productivity and fertility, 9-13
water quality, 9-13
toxicity , 5-5, 5-69
birds, 5-69
fish, 5-69
mammals, 5-69
Endpoints
indicators of, 6-1 to 6-3, 6-4
selection of, 6-1 to 6-3
Eradication
effects of, 5-71, 7-14, 10-1, 10-5, 10-6, 10-7
methodology, 3-1, 3-2 to 3-3, 3-5, 3-8
treatments used in, 3-2 to 3-3, 3-5, 3-8
Foray (*see also Bacillus thuringiensis kurstaki*), 5-41, 7-39
Forests (*see also* developed and undeveloped forest ecosystems, and gypsy moth, ecological effects of)
FSCBG (Forest Service Cramer Barry Grim) aerial dispersion model, 7-22, 7-23, 7-32, 7-42
Gypchek (*see also* nucleopolyhedrosis virus), 3-8, 5-18, 5-41, 5-48, 7-18
Gypsy moth (see also Asian gypsy moth)
biology
distribution, 2-9,
host preference, 2-4 to 2-5
infestation history, 2-9 to 2-10
life cycle, 2-3 to 2-4
outbreak duration, 2-10
parasites, 2-6 to 2-7
pathogens, 2-7 to 2-8
population dynamics, 2-5 to 2-6, 2-10, 2-11
predators, 2-6 to 2-7
rate of spread, 2-8
detection techniques, 2-9
ecological effects of
forest health,
fire susceptibility, 2-21
mast production, 2-18 to 2-19
shrubs and herbaceous plants, 2-19 to 2-20
stand structure, 2-18
tree mortality, 2-16 to 2-18
understory plants, 2-19 to 2-10
vegetative diversity, 2-20 to 2-21
microclimate
light penetration, 2-40
relative humidity, 2-40

Gypsy moth
ecological effects of
microclimate (continued)
soil and litter temperature, 2-40
nontarget populations
soil fertility and productivity
decomposition, 2-41
earthworms, 2-32, 2-40
erosion rate, 2-39, 2-43, 2-83, 2-85
litter production, 2-41
organic matter content, 2-42
soil pH, 2-42
water quality
acidification, 2-37 to 2-38
detrital decomposition rates, 2-35
dissolved oxygen concentration, 2-37, 2-48, 2-52, 2-83, 2-85
eutrophication, 2-37, 2-38
flow rate, 2-39
sediment load, 2-39 to 2-40
structural habitat, 2-40
effects of defoliation
secondary organisms, 2-14 to 2-16
tree mortality, 2-16 to 2-18
effects on
amphibians, 2-27, 2-28, 2-47, 2-51, 2-52, 2-82, 2-84
aquatic insects, 2-34, 2-51, 2-52, 2-83, 2-85
birds, 2-25 to 2-27
neotropical migrants, 2-26 to 2-27, 2-26, 2-47, 2-51, 2-52, 2-82, 2-84
ruffed grouse, 2-23, 2-25, 2-44, 2-53, 2-54, 2-56, 2-64, 2-82, 2-84
wild turkey, 2-23, 2-25 to 2-26, 2-62, 2-69, 2-82, 2-84
crustaceans, 2-30, 2-35, 2-51, 2-83, 2-85
detrital decomposition rates, 2-35
dissolved oxygen concentration, 2-37, 2-48, 2-52, 2-83, 2-85
earthworms, 2-32, 2-47, 2-48, 2-51, 2-52, 2-58, 2-64, 2-68, 2-83, 2-85
eutrophication, 2-37, 2-52
game fish, 2-33 to 2-34, 2-44, 2-51, 2-82, 2-84
gray squirrels, 2-22, 2-47, 2-50, 2-52, 2-55, 2-61, 2-71, 2-73, 2-82, 2-84
insect parasites, 2-30, 2-83, 2-85
insect predators, 2-30, 2-83, 2-85
Lepidopterans, 2-28 to 2-29, 2-30, 2-47, 2-51, 2-56, 2-62, 2-65, 2-74, 2-82, 2-84
spring-emerging, 2-29, 2-51, 2-82, 2-84
summer-emerging, 2-29, 2-51, 2-82, 2-84
litter invertebrates, 2-32, 2-40, 2-41, 2-47, 2-48, 2-51, 2-52, 2-83, 2-85
mammals, 2-21 to 2-24, 2-82, 2-84
bats, 2-23 to 2-24, 2-31, 2-44, 2-53, 2-54, 2-82, 2-84
black bears, 2-22, 2-44, 2-53, 2-55, 2-73, 2-82, 2-84
white-tailed deer, 2-20, 2-21, 2-23, 2-24, 2-44, 2-53, 2-55, 2-67, 2-82, 2-84
woodland mice, 2-24, 2-47, 2-82, 2-84
mollusks, 2-35, 2-51, 2-83, 2-85
pH, 2-37, 2-42, 2-43, 2-58, 2-59, 2-60, 2-61, 2-68, 2-72, 2-83, 2-85
pollinators, 2-31, 2-48, 2-55, 2-83, 2-85
reptiles, 2-27 to 2-28, 2-52, 2-82, 2-84
timber rattlesnakes, 2-28
spiders, 2-30, 2-31, 2-32, 2-47, 2-48, 2-51, 2-52, 2-56, 2-66, 2-69, 2-83, 2-85
exposure assessment, 2-44 to 2-45
immune stands (see risk assessment)
management strategies, 3-1 to 3-8
resistant stands (see risk assessment)
risk assessment
immune stands, 2-44, 2-45
resistant stands, 2-44, 2-46
susceptible stands, 2-13, 2-44, 2-49
susceptible stands (see risk assessment)
Gypsy moth virus (see nucleopolyhedrosis virus)
Mass trapping, 3-3, 3-4, 3-5
Mating disruption, 3-5, 9-13
Monte Carlo simulation, 9-2, 9-3, 9-9, 9-10, 9-27
NPV (see nucleopolyhedrosis virus)
Nucleopolyhedrosis virus (NPV)
application rates
by strategy, 3-8
characteristics
description of, 5-3 to 5-4, 7-18
ecological effects (see nucleopolyhedrosis virus, risk)
effects on
aquatic invertebrates, 5-66
birds, 5-18, 5-64
fish, 5-65
mammals, 5-18, 5-64
parasitic wasps, 5-18
terrestrial invertebrates, 5-18, 5-65
exposure, 8-6
fate , 7-18 to 7-20
fate and transport modeling
risk
forest health, 9-13

Nucleopolyhedrosis virus
risk (continued)
microclimate, 9-13
nontarget species, 9-13
soil productivity and fertility, 9-13
water quality, 9-13
toxicity , 5-4
aquatic invertebrates, 5-66
birds, 5-64
fish, 5-65
mammals, 5-64
terrestrial invertebrates, 5-65
transmission of nucleopolyhedrosis virus, 7-19
Probit analysis, 9-3, 9-4
Program areas, 3-1 to 3-8, 4-1, 5-71, 7-13, 7-30, 10-46
PRZM (Pesticide Root Zone Model), 7-22, 7-26, 7-27, 7-35, 7-48
Slow-the-spread 1-1, 1-2
effects of, 10-1, 10-5, 10-6
methodology, 3-1, 3-3, 3-4, 3-5, 3-8
treatments used in, 3-1, 3-3, 3-4, 3-5, 3-8
Sterile insect technique, 3-5
Suburban areas (see developed forest ecosystem)
Suppression, 1-1, 1-2, 2-73 4-12
effects of, 5-25, 5-30, 5-58, 7-14, 10-1, 10-5, 10-6
methodology, 3-1, 3-2, 3-3, 3-4, 3-6, 3-8
treatments used in, 3-1, 3-2, 3-3, 3-4, 3-6, 3-8
Traps, pheromone, 2-9, 2-28, 3-3, 3-5, 3-8, 5-7, 5-8, 5-18, 5-19, 5-24, 5-78, 7-20, 7-22, 7-31, 8-6, 9-13, 9-14,
delta traps, 3-5
insecticide used in traps (*see also* dichlorvos or disparlure), 3-5
milk carton traps, 3-5
Undeveloped forest ecosystem
description of
climate, 4-3
fauna
aquatic, 4-8
terrestrial, 4-6
flora, 4-5
hydroiogy, 4-4
soils, 4-3
B.t.k. residues, 7-17, 8-5, 8-6
diflubenzuron residues
aquatic, 7-12, 732 to 7-33, 7-55 to 7-58
terrestrial, 7-12, 8-1, 8-2, 8-4
nucleopolyhedrosis virus, 7-20
Urban areas (see developed forest ecosystem)
Vaportape II strip, 5-18, 7-20, 7-31, 7-33, 8-6, 9-13

Species Index

Abagratis, 8-3
Acarina, 5-51
Acholla multispinosa, 5-57
Actinomycetes, 5-8
Adoxophyes orana, 9-21, 9-29, 9-30
Aedes nigromaculis, 9-32
Agrilus bilineatus, 2-15, 2-49
Agrilus, 2-15, 2-46, 2-49
Alcaligenes, 7-10, 7-42
Aleochara bilineata, 5-58
Algae, 2-47, 2-51, 5-2, 5-20, 5-22, 5-24, 5-60, 5-97, 7-7, 9-8
Algal, 2-35, 2-83, 2-85, 4-8, 6-2, 6-4, 9-8
Alona, 5-96
Amphibians (*see also* frogs, tadpoles, toads), 2-27, 2-28, 2-47, 2-51, 2-52, 2-82, 2-84, 4-7, 4-9, 4-10, 5-2, 5-3, 5-4, 5-17, 6-2, 6-4
Amphipods (scuds), 5-21, 5-23, 5-98, 10-1
Anabaena flos-aquae, 5-20
Anodonta cygnea, 5-62, 9-18
Anomogynen, 8-3
Anticarsia gemmatalis, 5-36, 5-81
Ants (*see also* Formicidae), 2-71, 5-15, 5-17, 5-68, 5-92, 8-2, 9-5, 9-9, 9-28, 9-32
Apanteles melanoscelus (*see also Cotesia melanoscelus*), 5-13, 5-85, 9-16, 9-32
Apanteles (*see also Cotesia*), 5-13, 5-52, 5-85, 9-16, 9-32
Apanteles fumiferanae, 5-40, 5-52
Aphelocoma, 4-6
Aphids, 5-7, 5-12, 5-42, 5-83, 5-90, 8-2
Aphytis melinus, 5-52
Apis mellifera, 5-14, 5-87, 5-88
Aquatic invertebrates, 4-9, 5-19, 5-53, 5-59, 5-66, 5-95, 5-96, 9-7, 9-32, 9-34, 10-9
Arachnida, 5-90
Araneae, 2-66, 5-51, 5-91
Araneida, 5-7
Armillaria, 2-15, 2-46, 2-49
Asian gypsy moth,2-2, 2-3, 2-4, 2-5, 2-7, 2-8, 2-80, 5-71
Aspergillus, 5-63
Avian (*see also* birds), 2-5, 2-51, 7-19, 7-37
Bacillus thuringiensis kurstaki, 2-7, 8-4, 9-9
Backswimmers, 5-23
Bacteria, 2-41, 4-8, 5-1, 5-4, 5-8, 5-19, 5-60, 6-2, 6-4, 7-4, 7-5, 7-7, 7-8, 7-10, 7-14, 7-36, 7-39, 10-3
Baetis, 5-96, 5-97
Bagworm moths, 5-1
Balanus, 5-63
Barnacles (*see also Balanus*), 5-3
Bass, 4-8, 5-55, 9-19, 9-30
Bats, 2-21, 2-23, 2-31, 2-44, 2-53, 2-54, 2-82, 2-84, 5-9, 5-17, 6-2, 9-6, 9-11, 10-2
Bears, 2-21, 2-22, 2-23, 2-44, 2-53, 2-55, 2-73, 2-82, 2-84, 4-6, 8-6, 8-7, 9-13, 9-14, 10-2, 10-9
Bees (*see also* honeybees), 2-31, 5-8, 5-14, 5-16, 5-87, 5-88, 8-2, 9-5
Beet armyworm (*see also Spodoptera exigua*), 5-52, 9-29
Beetles, 2-7, 2-39, 2-50, 5-2, 5-7, 5-8, 5-11, 5-12, 5-14, 5-15, 5-16, 5-18, 5-23, 5-52, 5-80, 5-81, 5-83, 5-84, 5-89, 5-90, 5-92, 5-94, 5-98, 8-2, 9-5, 9-11, 9-12, 9-17, 9-18, 9-31, 9-34, 10-10
Beneficial organisms, 2-18, 2-39, 2-42, 5-6, 5-8, 5-11, 9-17
Berosus infuscatus, 5-97
Big-eyed bugs (*see also Geocoris*), 5-8, 5-11, 5-51, 5-81, 5-83, 9-5
Birches, 2-1, 2-2, 2-5, 2-20, 2-50, 2-80, 2-81, 5-71, 5-77, 7-24
Birds, 2-7, 2-24, 2-25, 2-26, 2-27, 2-47, 2-52, 2-58, 2-59, 2-60, 2-63, 2-64, 2-72, 2-82, 2-84, 4-6, 4-7, 4-9, 4-10, 4-12, 4-13, 5-4, 5-5, 5-9, 5-16, 5-17, 5-18, 5-50, 5-54, 5-64, 5-68, 5-69, 5-77, 5-93, 5-94, 6-1, 6-2, 6-4, 7-37, 7-41, 8-2, 9-6, 9-11,10-6, 10-7
Bivalves (*see also* clams, oysters, or quahogs), 5-3
Black-and-white warbler, 2-26
Black-throated green warbler, 2-26
Black-capped chickadee, 5-64
Black flies (*see also Simulium* or *Prosimulium*), 5-2, 5-19, 5-53, 5-95, 5-96, 5-97
Black crappies, 5-21
Blepharipa pratensis, 5-75
Blue-gray gnatcatcher, 2-26
Blueberries, 2-19, 2-31, 2-50, 2-81, 5-73, 5-76
Bluegill, 5-21, 5-50, 5-51, 5-54, 5-65, 5-69, 5-96
Bobwhite quail, 5-50, 5-54, 5-69, 9-12
Bollworms (*see also Spodoptera*), 5-11, 5-80, 5-83, 7-18
Bosmina longirostris, 5-98
Bosmina, 5-96, 5-98

Bovicula limbatus, 9-31
Brachydanio rerio, 9-35
Brachymeria intermedia, 5-7, 5-13, 5-74
Braconidae, 5-11, 5-52, 5-74, 5-84, 5-92
Branchiopoda, 5-96
Brine shrimp, 5-53, 5-61, 9-34
Brook trout, 5-55
Brown trout, 5-65
Brown bullheads, 5-21
Buenoa, 5-97
Butterflies, 2-29, 4-7, 4-13, 5-1, 5-5, 5-10, 5-58, 5-78, 6-1, 7-14, 8-2, 9-4, 9-10, 10-2, 10-5
Cabbage maggot, 5-8, 5-58
Caddisflies, 5-2, 5-53, 5-60, 5-95, 9-6, 9-25, 9-26, 9-34, 9-37, 9-39
Caenis, 5-96
Callibaetis, 5-59, 9-33
Calosoma sycophanta, 2-7, 2-47
Canis latrans, 4-6
Carabid, 2-47, 5-40, 5-81
Carabidae, 5-7, 5-90
Carassius auratus, 4-8
Carp, 4-8
Carpenter ants, 5-68
Cat flea, 9-16, 9-31
Cats, 4-6
Cattle, 5-3, 5-59, 5-67
Cavity-nesting birds, 2-52
Ceratopogonid, 5-96
Ceriodaphnia, 5-96
Channel catfish, 5-54
Chaoborids, 5-23, 5-97, 7-34
Chaoborus astictopus, 5-97, 7-34
Chaoborus, 5-97, 7-34
Chestnut-backed, 5-9, 5-77
Chironomid, 2-34, 5-21, 5-23, 5-60, 5-92, 5-97
Chironomus plumoosus, 9-34
Choristoneura occidentalis, 5-6, 5-70
Choristoneura fumiferana, 2-11, 5-6, 5-70, 5-74
Chrysopa oculata, 5-57
Chrysopa carnea, 5-80
Cicadas, 5-7, 5-90
Cinnabar moth, 5-52
Cladocera, 9-18
Cladocerans, 5-23, 5-97
Clam shrimp, 9-32
Clams, 5-62, 5-95, 9-32, 9-34
Clistoronia magnifica, 9-34
Coccinelidae, 5-8
Codling moth (*see also Laspeyresia pomonella*), 5-12, 5-82
Coenagrionid, 5-97
Coleomegilla maculata, 5-83
Coleoptera, 2-30, 5-7, 5-8, 5-51, 5-60, 5-90, 5-95, 5-96, 9-28
Collembola, 5-97
Colorado potato beetle, 5-58, 5-59, 9-31
Compsilura concinnata, 5-74
Conifers, 2-36
Conothrachelus posticatus, 2-20
Copepods, 5-23, 5-61, 5-96, 9-20, 9-33, 9-35
Corixids, 5-21, 5-23
Corn earworm (*see also Heliothus*), 5-52
Corvus, 4-6
Cotesia melanoscelus (*see also Apanteles*), 5-18, 5-86
Cotesia (*see also Apanteles*), 5-7, 5-13, 5-18, 5-75, 5-86
Cougar, 4-6
Cowbirds, 4-7
Coyotes, 2-21, 4-6
Crabs, 5-3, 5-23, 5-62, 5-68, 9-33, 9-34
Crassostrea virginica, 9-34
Crayfish, 2-35, 2-51, 5-95, 9-34
Cricotopus, 9-32, 9-34
Crotalus horridus, 2-28, 2-69
Crows, 4-6
Crustaceans, 2-30, 2-35, 2-51, 2-83, 2-85, 5-3, 5-21, 5-23, 6-2, 6-4, 7-35, 9-4, 9-6, 9-25, 9-26, 9-28, 9-36, 9-37, 9-38, 9-39, 9-40, 10-1
Cuckoos, 2-24, 2-47
Culex quinquefascia-tus, 5-60, 5-97
Culex, 5-60, 5-97
Culex salinarius, 5-60, 5-97
Culex restuans, 5-60, 5-97
Culicids, 5-23
Cutthroat trout, 5-55
Cyanocitta, 4-6
Cyclops, 5-96, 5-97
Cyprinidae, 5-22
Cyprinotus, 5-98
Cyprinus carpio, 4-8
Dace, 5-22
Damsel bugs, 5-8
Damselflies, 5-2
Daphnia galeata, 5-98
Daphnia magna, 5-61, 5-66, 9-32, 9-34
Daphnia pulex, 5-98
Daphnia, 5-61, 5-66, 5-98, 9-13, 9-32, 9-34, 10-1
Decapoda (*see also* crabs or shrimp), 5-96
Deer (*see also* white-tailed deer), 2-19, 2-20, 2-21, 2-23, 2-24, 2-44, 2-53, 2-82, 2-84, 4-6, 8-2, 10-6, 10-9

Deleatium, 2-34
Dendroica caerulescens, 5-9, 5-77
Desmognathus, 5-17
Desmognathus ochrophaeus, 5-17, 5-18
Diadegma insulare, 5-52
Diamondback moth, 5-52
Diaphanosoma, 5-96
Diaprepes abbreviatus, 5-12, 5-82
Diaptomus, 5-96, 5-98
Diatom, 5-23
Dictyoptera, 5-91
Dipterans, 5-21, 5-22, 5-23, 8-3
Dobsonflies, 5-2
Dogs, 4-6, 5-50, 9-14
Doryphorphaga doryphorae, 5-59
Dragonflies (*see also* Odonata), 5-2, 5-21, 5-23, 5-59, 9-33
Duckweed, 5-3, 5-20, 5-63, 9-35
Earthworms, 2-32, 2-47, 2-48, 2-51, 2-52, 2-83, 2-85, 5-8, 5-55, 6-2, 6-4, 9-5, 9-8, 9-28, 9-32, 10-3, 10-10
Eisenia fetida, 5-55, 9-32
Entomophaga maimaiga, 2-7
Ephemeroptera, 2-34, 5-96, 5-97
Epilachna varivestis, 5-84
Erythroneura variablis, 9-31
Euchlaena, 8-3
Euglena gracilis, 5-3, 5-33
Eulimnadia, 9-32
Eurytemora affinis, 9-20, 9-33
Fathead minnow, 5-54, 5-68
Felis concolor, 4-6
Ferns, 2-19, 2-50
Fiddler crab, 5-62, 9-34
Fish, 2-33, 2-34, 2-44, 2-51, 2-52, 2-82, 2-84, 4-8, 4-11, 4-12, 5-3, 5-4, 5-5, 5-20, 5-21, 5-22, 5-23, 5-50, 5-54, 5-65, 5-69, 5-95, 5-96, 6-1, 6-2, 6-4, 7-8, 7-36, 7-41, 8-3, 9-7, 9-12, 9-13, 9-25, 9-26, 9-28
Five-lined skink, 2-27
Fleas, 5-23, 9-31
Flies, 5-6, 5-15, 5-16, 5-17, 5-19, 5-21, 5-23, 5-53, 5-74, 5-75, 5-90, 5-92, 5-94, 5-95
Flycatcher, 2-26, 2-47
Forficularia auricularia, 5-12, 5-57, 5-83
Formicidae, 5-92, 9-15
Fox, 4-6
Fox squirrels, 4-6
Frogs, 2-28, 4-7, 5-24
Fungi, 2-41, 2-45, 5-2, 5-8, 5-12, 5-16, 5-60, 5-63, 6-2, 6-4, 7-5, 7-7, 7-8, 7-10, 10-3, 10-10
Fusarium, 5-63
Game fish, 2-34, 2-44, 2-51, 2-82, 2-84, 6-1, 6-2, 6-4, 10-9
Gastropods, 5-21, 5-60
Geocorids, 5-12, 5-80, 5-81
Geocoris punctipes, 5-81, 5-83, 9-31
Geocoris, 5-8, 5-81, 5-83, 9-31
Geometer moths, 5-1
Geometridae, 5-73
Glypta, 5-84
Glyptotenidipes paripes, 9-33
Goeldichironomus holoprasinus, 9-35
Goldfish, 4-8
Grackles, 2-47
Grass shrimp (*see also Palaemontes pugio*), 5-23, 5-53, 5-62, 9-33, 9-34
Grasshoppers, 5-2, 5-11, 5-15, 5-18, 5-57, 5-78, 5-84, 5-89, 5-90, 5-91, 9-4, 9-5, 9-25, 9-31, 9-36
Gray squirrels, 2-22, 2-47, 2-50, 2-52, 2-82, 2-84, 4-6
Great tit, 5-93
Gryllacrididae, 5-90
Guinea pig, 5-64
Haematobia irritans, 5-59
Hamamelis virginiana, 2-19
Helicoverpa punctigera, 5-52
Helicoverpa armigera, 5-52
Heliothis, 5-80, 5-82, 5-83, 7-18
Hemiptera, 5-8, 5-51, 5-89, 5-96
Hemipterans, 5-15
Hemlock looper, 5-51, 5-58
Hermit crabs, 5-68
Hippodamia convergens, 5-80, 5-81, 5-83, 9-31
Homoptera, 5-7, 5-8, 5-89, 5-90, 9-31
Honey bee (*see also Apis mellifera*), 5-1, 5-3, 5-51, 5-55, 5-56, 5-65, 5-68, 5-87, 5-88
Hooded warbler, 2-26
Horn fly, 5-59
Hornets, 5-16
Horseshoe crabs, 5-3, 5-62
House sparrow, 5-64
House fly, 5-59
Hyalella azteca, 5-62, 5-97, 5-98, 9-33
Hydracarina, 5-96
Hydras, 5-95
Hydrophilus triangularis, 5-60, 9-33
Hydropsyche, 5-96
Hydropsychidae, 5-59
Hylemya brassicae, 5-8
Hymenoptera, 2-30, 5-6, 5-7, 5-8, 5-17, 5-51, 5-76,

Hympenoptera (continued), 5-84, 5-92, 9-15
Hymenopterans, 5-17, 9-32
Ichneumonidae, 5-7, 5-11, 5-76, 5-84, 5-92
Ichneumonids, 2-30, 5-11, 5-52, 5-65, 5-84
Ictalurus nebulosus, 5-21
Ilyocryptus, 5-96
Indigo bunting, 2-26
Invertebrates, 2-5, 2-7, 2-30, 2-32, 2-33, 2-40, 2-47, 2-48, 2-51, 2-52, 2-82, 2-83, 2-84, 2-85, 4-3, 4-7, 4-8, 4-9, 4-11, 5-1, 5-2, 5-4, 5-11, 5-15, 5-19, 5-21, 5-22, 5-23, 5-51, 5-53, 5-55, 5-59, 5-65, 5-66, 5-68, 5-78, 5-89, 5-95, 5-96, 5-98, 6-2, 6-4, 7-7, 7-35, 7-36, 7-37, 7-39, 7-40, 7-41, 7-43, 8-3, 8-4, 8-6, 9-1, 9-7, 9-8, 9-9, 9-11, 9-12, 9-13, 9-32, 9-34
Isotria meleoloides, 2-19
Jalysus spinosus, 5-8
Jays, 4-6
Juga plicifera, 9-33
Juga, 9-33, 9-34
Laccophilus, 5-61, 5-98
Lacewings (*see also Chrysopa*), 5-11, 5-14, 5-52, 5-57, 5-80, 5-81, 5-83, 9-5
Lady beetle, 5-52, 9-31
Ladybird beetle (*see also Hippodamia convergens*), 5-80, 5-81, 5-83
Lambdina fiscellaria, 5-58
Large white butterfly, 5-58
Larrinae, 5-11, 5-84
Laspeyresia pomonella, 5-82, 9-29, 9-30
Leaf-hoppers, 5-7
Leaf miners, 9-5
Leafrollers, 5-12, 5-17, 5-83
Lemna gibba, 5-20, 5-63, 9-35
Lepidoptera, 2-56, 2-82, 2-84, 5-5, 5-6, 5-7, 5-9, 5-10, 5-16, 5-17, 5-18, 5-70, 5-71, 5-73, 5-74, 5-78, 5-79, 5-90, 5-92, 5-93, 5-94, 7-14, 7-37, 7-39, 7-43, 8-8, 9-4, 9-6, 9-9, 9-10, 9-11
Lepidopterans, 2-28, 2-29, 2-30, 2-47, 2-51, 3-2, 5-1, 5-2, 5-6, 5-7, 5-19, 5-72, 5-73, 5-76, 6-1, 6-4, 7-15, 8-2, 8-3, 9-3, 9-4, 9-5, 9-7, 9-9, 9-11, 9-12, 9-25, 9-27, 9-29, 9-30, 9-36, 9-38
Spring-emerging, 2-29, 2-51, 6-4, 10-9
Summer-emerging, 2-29, 2-51, 6-4, 10-9
Lepomis macrochirus, 5-21
Lepomis, 4-8, 5-21, 5-96
Leptinotarsa decemlineata, 9-17, 9-31
Leuctra tenuis, 5-53, 5-95
Leuctridae, 5-2
Limulus polyphemus, 5-62
Lindera benzoin, 2-22
Litter invertebrates, 2-83, 2-85, 6-2, 6-4, 9-7, 9-8, 9-9, 9-32
Lizards, 4-7
Longnose killifish, 5-65
Lymantria dispar (*see also* Asian gypsy moth), 2-2, 5-58, 5-65, 9-15
Macrocentrus ancylivorous, 5-57, 5-84
Macroinvertebrates, 2-34, 5-16, 5-28
Macrolepidopterans, 2-28, 5-5, 5-10, 5-72, 5-78, 5-79, 9-4
Macrothrix, 5-96
Malacosoma disstria, 2-29, 5-58, 5-72
Malacosoma americana, 2-29
Mallard, 5-50, 5-54, 5-64, 5-69, 9-12
Mamestra brassicae, 9-20, 9-29
Mammalian, 2-22, 5-4, 7-19, 7-37
Mammals, 2-5, 2-7, 2-11, 2-20, 2-21, 2-24, 2-28, 2-50, 2-51, 2-52, 2-53, 2-82, 2-84, 4-6, 4-9, 4-11, 5-4, 5-5, 5-9, 5-17, 5-18, 5-50, 5-54, 5-64, 5-67, 5-69, 6-2, 6-4, 7-14, 7-37, 8-2, 9-11, 10-6
Maples, 2-4, 2-17, 2-19, 2-20, 2-32, 2-46, 2-49, 2-50, 2-73, 2-81, 4-5, 5-8, 5-70, 5-71, 5-77
Masked shrews, 5-9
Mayflies, 5-2, 5-21, 5-22, 5-23, 5-59, 5-60, 5-95, 9-6, 9-25, 9-26, 9-33, 9-37, 9-39, 10-2
Megaloptera, 5-53, 5-95
Menippe mercenaria, 9-33
Mercenaria mercenaria, 5-62, 9-20
Mesocyclops thermocyclopoides, 9-19, 9-34
Mexican bean beetle, 5-12, 5-58, 5-84, 9-31
Mice, 2-7, 2-21, 2-24, 2-82, 2-84, 5-17, 5-50, 5-67, 8-2, 9-21
Microlepidopterans, 5-5, 5-11, 5-72, 5-73, 5-78, 5-79, 9-4
Micropterus salmoides, 4-8
Midges, 5-2, 5-21, 5-22, 5-23, 5-60, 5-92, 5-95, 5-97, 9-25, 9-26, 9-28, 9-32, 9-33, 9-35, 9-36, 9-37, 9-39
Minnow, 5-22, 5-51, 5-54, 5-65, 5-68
Minute pirate bugs, 5-8
Mites, 2-31, 5-3, 5-8, 5-12, 5-15, 5-55, 5-82, 5-89, 5-91, 8-2, 9-8, 9-11, 9-31, 10-3, 10-10
Mollusks, 2-35, 2-51, 2-83, 2-85, 6-2, 6-4, 7-17, 9-6, 9-28, 9-33, 10-9
Molothrus ater, 4-7
Mosquitoes (*see also Culex*), 5-23
Mosquitofish, 5-24, 5-96

Mouse (*see also* mice), 2-47, 5-54, 5-64
Mummichog, 5-38, 5-54
Muscidifurax raptor, 5-59
Mussels, 5-19, 5-53
Mysidopsis bahia, 5-62
Nabidae, 5-8
Nabids, 5-12, 5-80, 5-81
Nabis, 5-8, 5-81, 5-83
Nematoda, 5-96
Nematodes, 5-8, 5-55, 9-11, 9-12, 10-3, 10-10
Neotropical migrants, 2-26, 2-47, 2-51, 2-52, 4-6, 4-7, 4-12, 5-9, 5-16, 5-94
Noctuidae, 5-5, 5-72, 5-73, 5-79
Noctuids, 5-73
Normuraea rileyi, 5-81
Oaks, 2-1, 2-4, 2-5, 2-8, 2-13, 2-14, 2-15, 2-16, 2-17, 2-18, 2-20, 2-21, 2-22, 2-23, 2-24, 2-29, 2-32, 2-36, 2-46, 2-47, 2-49, 2-50, 2-51, 2-52, 2-80, 2-81, 4-5, 5-9, 5-71, 5-72, 5-73, 5-74, 5-75, 5-76, 5-78, 5-79, 5-84, 5-89, 5-91, 5-92, 5-93, 5-94, 6-1, 7-2, 7-4, 7-15, 7-24, 10-6
Odonata, 5-95, 5-96, 9-34
Odonates, 5-21
Oligochaeta, 5-96
Oligochaetes, 2-34, 5-60, 5-98
Oncopeltus fasciatus, 5-57
Oncorhynchus kisutch, 2-33, 2-63, 2-70, 2-72
Ooencyrtus kuwanai, 5-19
Ooencyrtus kuvanae, 5-13, 5-85
Orgyia psuedotsugata, 5-4, 9-29
Oriental fruit moth, 5-12, 5-84
Orius insidiosus, 5-81, 5-83
Orius, 5-8, 5-81, 5-83
Orthemis, 5-59, 9-33
Orthoptera, 5-18, 5-51, 5-90, 5-91, 9-4
Orthopterans, 5-15, 5-57, 9-4, 9-5, 9-25, 9-28, 9-31, 9-36, 9-38
Orthosia hibisci, 5-71
Orthosia rubescens, 5-72
Ostracods, 5-21, 5-23
Oysters, 5-19, 5-66, 9-34
Palaemontes pugio, 9-34, 9-35
Pantala, 5-59, 9-33
Parasetigena silvestris, 5-74, 5-75
Parasites, 2-6, 2-7, 2-10, 2-26, 2-29, 2-30, 2-47, 2-51, 2-52, 2-80, 2-83, 2-85, 5-1, 5-6, 5-7, 5-11, 5-12, 5-13, 5-14, 5-17, 5-18, 5-52, 5-56, 5-59, 5-74, 5-75, 5-80, 5-82, 5-83, 5-84, 5-85, 6-4, 7-19, 9-5, 9-11, 9-13
Parasitic wasps, 5-3, 5-7, 5-11, 5-13, 5-18, 5-80, 5-84, 5-85, 9-4, 9-5, 9-11, 9-13, 9-25, 9-36, 9-39
Parasitic flies, 5-6, 5-7, 5-13, 5-75, 5-85
Parasitoids, 2-5, 2-7, 2-29, 2-30, 2-45, 5-6, 5-11
Parus atricapillus, 5-77
Parus caeruleus, 5-93
Parus rufescens, 5-77
Parus major, 5-93
Pathogens, 2-5, 2-6, 2-7, 2-15, 2-49, 2-82, 2-84, 5-86, 7-18
Pediobius foveolatus, 5-84
Perizoma curvilinea, 5-71
Perlodid, 5-60, 9-33
Peromyscus, 2-24, 2-55, 2-71
Phanerochaete chysosporium, 7-10
Phobocampe unicincta, 5-75
Physa, 5-62, 9-33, 9-34
Phytolacca americana, 2-22
Pierid butterflies, 5-1
Pieris brassicae, 5-58
Pigs (*see also* swine), 5-64, 5-67
Pinus rigida, 2-17
Planarians, 5-95
Plecoptera, 2-34, 5-53, 5-60, 5-95, 5-96, 5-97
Plecotus townsendii virginianus, 2-23, 5-17
Plectonema, 5-20
Plethodon cinereus, 5-17
Plutella xylostella, 9-30
Podisus maculiventris, 5-8
Pollinators,2-31, 2-48, 2-83, 2-85, 5-16, 5-89, 6-2, 6-4, 10-50
Praying mantis, 5-53
Predators, 2-5, 2-6, 2-7, 2-10, 2-11, 2-26, 2-29, 2-30, 2-31, 2-47, 2-51, 2-52, 2-53, 2-80, 2-83, 2-85, 4-6, 5-1, 5-7, 5-8, 5-10, 5-11, 5-12, 5-13, 5-14, 5-15, 5-24, 5-80, 5-81, 5-82, 5-83, 5-85, 5-89, 5-90, 6-4, 9-5
Predatory insects, 5-3, 5-8, 9-36, 9-39
Prosimulium fascum/mixtum, 5-53, 5-95
Protoboarmia, 8-3
Protorthodes rufula, 5-71
Pseudoplusia includens, 9-30
Psocoptera, 5-90
Psylla, 5-12, 5-82
Psylla pyricola, 5-82
Pteronarcys californica, 5-53
Pyralid moths, 5-1
Pythium, 5-63
Quahogs, 5-62, 9-20, 9-34
Quail, 5-50, 5-54, 5-64, 5-69, 9-12
Rabbit, 5-50, 5-54, 5-64, 5-67, 5-69

Rainbow trout, 2-33, 5-50, 5-51, 5-54, 5-68, 5-69
Raspberries, 2-19, 2-50
Rats, 5-3, 5-50, 5-54, 5-64, 5-67, 5-69, 5-38, 9-14
Rattlesnakes, 2-28, 2-51, 2-82, 2-84
Red-winged blackbirds, 2-47, 5-68
Red maple, 2-20, 2-32, 2-49, 2-50
Red-spotted newts, 2-47
Reduviidae, 5-8
Reptiles, 2-27, 2-52, 2-82, 2-84, 4-7, 4-9, 4-10, 5-2, 5-4, 6-2, 6-4, 8-2, 10-9
Rhinichthys, 5-22
Rhizoctonia solani, 5-63
Rhychagrotis, 8-3
Rice swarming caterpillar, 5-58
Rose-breasted grosbeak, 2-26
Rotifera, 5-96
Rove beetle, 5-58
Rubus, 2-19, 2-49, 2-81
Ruffed grouse, 2-23, 2-25, 2-44, 2-53, 2-54, 2-82, 2-84
Salamanders, 2-28, 2-51, 5-16, 5-17
Salmon, 2-33, 2-63, 2-70, 2-72, 5-22, 5-55, 10-50
Salmonids, 2-33, 2-48, 2-52, 2-68, 2-82, 2-84
Sarcophagidae, 5-74
Satyrium calanus, 5-72
Sawflies, 2-12, 5-7, 5-15, 5-90, 5-92
Schoenobius incertulas, 9-30
Sciurus carolinensis, 2-61, 4-6
Sculpins, 2-33, 2-67
Secondary organisms, 2-14, 2-15
Selanastrum capricornutum, 5-20
Serratia liquefaciens, 2-7
Serratia marcescens, 2-7
Sheep, 2-19, 2-81, 5-50, 5-54, 8-2
Sheepshead minnow, 5-51, 5-65
Short-tailed shrew, 5-64
Shrimp, 5-19, 5-21, 5-23, 5-53, 5-61, 5-62, 9-32, 9-33, 9-34
Shrubs, 2-4, 2-5, 2-15, 2-19, 2-31, 2-46, 2-49, 2-82, 2-84, 4-3, 5-70, 8-2
Sida, 5-96
Simocephalus, 5-96
Simulium venustum, 5-97
Simulium vittatum, 5-53, 5-95
Skeletonema costatum, 5-23
Skwala, 5-60, 9-34
Smallmouth bass, 5-55
Snails, 2-35, 2-51, 5-3, 5-21, 5-24, 5-62, 9-33, 9-34
Snakes, 2-28, 4-7, 5-4
Soil invertebrates, 2-32, 2-47, 2-48, 2-51, 2-52, 5-89, 9-11, 9-12
Species richness, 2-24, 2-26, 2-47, 5-6, 5-70, 5-71, 5-78, 5-79, 5-92, 6-2, 6-4, 9-11, 10-50
Species diversity, 1-1, 4-2, 4-5, 4-8, 4-11, 5-6, 5-15, 9-5, 10-47
Sphinx moths, 5-1
Spiders, 2-30, 2-31, 2-32, 2-47, 2-48, 2-51, 2-52, 2-83, 2-85, 5-3, 5-7, 5-8, 5-15, 5-89, 5-90, 5-91, 9-5, 9-8, 10-3, 10-10
Spined stiltbug, 5-8
Spined soldier bug, 5-8, 5-51
Spodoptera exempta, 9-29
Spodoptera mauritania, 5-58
Spodoptera littoralis, 5-58, 8-3, 9-29, 9-30
Spodoptera exigua, 9-29, 9-30
Spodoptera (*see also* bollworms), 5-58, 8-3, 9-29, 9-30
Sponges, 5-95
Spot, 2-8, 2-61, 5-65
Springtails, 5-23
Spruce budworm (*see also Choristoneura fumiferana*), 2-11, 2-31, 5-6, 5-51, 5-52, 5-70, 5-74, 5-95, 7-34
Starlings, 2-47, 5-68
Stoneflies, 5-2, 5-19, 5-22, 5-53, 5-60, 5-68, 5-95, 7-4, 9-6, 9-33, 9-34
Streptococcus, 2-7
Striped earwig, 5-52
Sucking insects, 5-3, 5-7, 8-2, 9-5
Sugar cane rootstalk borer weevil, 5-12, 5-82
Sunfish, 4-8, 5-50, 5-51, 5-54, 5-69, 5-96
Swallowtail butterflies, 5-1
Sweetfern, 2-19, 2-50, 2-81
Swine (*see also* pigs), 5-27, 5-32, 5-54
Sycamores, 4-5
Symphyta, 5-90
Tachinidae, 5-13, 5-18, 5-85
Tadpoles, 2-51
Taeniopterygidae, 5-2
Tanytarsus dissimilis, 5-60, 9-33, 9-35
Tenebrio, 5-18
Tent caterpillar (*see also Malacosoma disstra*), 2-47, 5-58, 8-8
Terrestrial invertebrates, 2-30, 2-51, 2-52, 5-4, 5-51, 5-55, 5-65, 5-68, 9-7
Thermonectus basillaris, 5-61, 5-98
Thysanoptera, 5-90, 9-17, 9-32
Tiger moths, 5-1
Toads, 2-28, 5-24
Trichoderma, 5-63
Trichograma pretiosum, 5-80
Trichogramma, 5-29, 5-56, 5-82

Trichoplusia ni, 5-52
Trichostronjulas colubriformus, 9-11
Tricorixa louisianae, 5-97
Triops longicaudatus, 9-33
Tropisternus lateralis, 5-61, 5-98
Trout, 1-1, 2-33, 2-34, 2-44, 2-47, 2-51, 5-22, 5-50, 5-51, 5-54, 5-55, 5-65, 5-68, 5-69, 10-9
Turkeys, 2-23, 2-25, 2-26, 2-44, 2-53, 2-82, 2-84
Tussock moth (*see also Orygyia psuedotsugata*), 2-31, 2-39, 5-1, 5-4, 5-51, 9-19, 9-29
Uca pugilator, 5-62, 9-16
Ursus americanus, 2-22, 4-6
Vaccinium, 2-19
Valentinia glandulella, 2-20
Vertebrates, 2-7, 2-31, 2-44, 4-9, 5-1, 5-2, 5-16, 5-20, 5-21, 7-16, 8-2, 8-6, 8-7, 9-5, 9-11, 9-13, 9-14
Vireo olivaceus, 5-93
Virginia big-eared bat, 2-23, 5-17
Vitis, 2-22
Vulpes fulva, 4-6
Wasps, 2-7, 5-3, 5-6, 5-7, 5-11, 5-12, 5-13, 5-15, 5-18, 5-19, 5-57, 5-74, 5-75, 5-76, 5-80, 5-82, 5-83, 5-84, 5-85, 5-86, 9-4, 9-5, 9-11, 9-13, 9-25, 9-36, 9-39
Water boatmen, 5-21, 5-23
White-footed mouse, 2-47
White-tailed deer, 2-20, 2-21, 2-23, 2-24, 2-44, 2-53, 2-82, 2-84
Wolf, 4-6
Wood lice, 5-15, 5-90
Wood peewee, 2-26
Woodpeckers, 2-24, 2-51, 2-52
Worms, 2-32, 5-98
Xylenini, 2-29
Yellow perch, 5-55
Yellowjackets, 5-16
Zaprionus paravittiger, 5-59
Zooplankton, 5-21, 5-97

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