Effect of Diflubenzuron on Nontarget Canopy Arthropods in Closed, Deciduous Watersheds in a Central Appalachian Forest

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ABSTRACT In a 6-yr study (1989–1994), we evaluated the impact of diflubenzuron on the diversity and abundance of arthropods in the Fernow Experimental Forest in West Virginia. Diflubenzuron is commonly used in gypsy moth, *Lymantria dispar* (L.), suppression programs in eastern forests. For the evaluation, foliage samples were taken with pole pruners from the forest canopy on 4 small deciduous watersheds; burlap bands were used on tree trunks on all watersheds. Pretreatment sampling was conducted mid-May through mid-August 1989 through 1991. Diflubenzuron was applied by helicopter to 2 watersheds; the 2 remaining watersheds served as control plots. Analysis of variance was used to compare treatment means. Gypsy moth larvae were reduced on the treated watersheds, particularly during the treatment and posttreatment year. Possible nontarget arthropod effects were researched for 27 mo after treatment. A significant reduction in the diversity of arthropod families was observed beneath burlap bands in treated plots. However, no reduction was observed for arthropod abundance. The diversity and abundance of macrolepidoptera larvae also were reduced by diflubenzuron during the treatment year. On foliage, overall arthropod family diversity and abundance, and numbers of macrolepidoptera and beetles were reduced significantly in treated watersheds. No significant reduction was seen for macrolepidoptera larval diversity on foliage. At 27 mo after treatment, total arthropod abundance and macrolepidoptera abundance on foliage remained significantly reduced. Declines were seen on treated watersheds for Carabidae, Gryllacrididae, Psocoptera, Phlaeothripidae, and some sapfeeders but were nonsignificant.

KEY WORDS gypsy moth, diflubenzuron, nontarget effects, microlepidopteran diversity, microlepidopteran abundance, canopy arthropods

THE NONTARGET IMPACT of insecticides applied for forest pest suppression has received attention in recent years. The microbial insecticide *Bacillus thuringiensis* Berliner variety *kurstaki* produced significant reduction of nontarget canopy lepidopteran larvae when applied against spruce budworm (Miller 1990a) and gypsy moth (Miller 1990b, Sample et al. 1996). Dimilin (Duphar, Graveland, The Netherlands) or diflubenzuron [1-(4 chlorophenyl)-3-(2, 6 diflubenzoyl) urea] has been used to suppress gypsy moth, *Lymantria dispar* (L.), for 20 yr (Eisler 1992) and is considered highly effective for that purpose (Twardus and Machesky 1990). Diflubenzuron is persistent on foliage; Wimmer et al. (1993) reported that 13 of 20 study trees treated with diflubenzuron retained >20% of the originally applied insecticide at leaf fall in October. It was applied to 499,124 ha in 7 eastern states from 1990 to 1993 as part of the federal and state cooperative gypsy moth suppression program (USDA 1994).

Diflubenzuron inhibits chitin synthesis (Grosscourt and Jongsma 1987) and acts primarily against immature insects at molting. It may serve also as an ovicide (Maas et al. 1981). Diflubenzuron is toxic to early-instar gypsy moth larvae in small doses (≤0.10 ppm) (Grannett and Dunbar 1975), but it is not specific to insect pests (Eisler 1992).

The impact of diflubenzuron on nontarget canopy arthropods has been the subject of several research studies. Martinat et al. (1988) reported significant reductions in abundance of canopy macrolepidopteran and nonlepidopteran mandibulate herbivores. Butler and Kondo (1993) found the abundance of nontarget macrolepidopteran larvae to be reduced following diflubenzuron application. Spider and orthopteroid litter communities were reduced in a central Appalachian forest after a diflubenzuron treatment (Martinat et al. 1993). Sample et al. (1993b) reported that captures of moths in blacklight traps were reduced in abundance and diversity following a diflubenzuron application. Barrows et al. (1994) inferred that diflubenzuron significantly reduced numbers of yellowjacket (Hymenoptera: Vespidae) workers in

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the treatment year possibly because of a reduction in caterpillars that served as prey.

Previous studies of diflubenzuron impact were conducted with Dimilin 25 W (wettable, Uniroyal, Bethany, CT). From 1989 through 1994, potential impacts of Dimilin 4 L (liquid) were evaluated on selected nontarget organisms in broadleaf forested watersheds typical of the mountainous regions of the eastern United States (Reardon 1995). Our contribution to this study was an evaluation of potential impact to canopy arthropods. In earlier articles, we listed all macrolepidopterous larval species and arthropod families under bands and on foliage during the 6-yr study (Butler et al. 1995a, b). In this article, we present results of the 6-yr evaluation of arthropod diversity and abundance on the study watersheds 3 yr before and 3 yr after aerial application of diflubenzuron. We tested the null hypothesis that diversity and abundance of arthropods were not different between treated and untreated plots.

Earlier diflubenzuron studies were conducted over a 1- to 3-yr period with a maximum of 1-yr pretreatment and 16 mo posttreatment evaluation of nontarget arthropods. Only a few taxa were sampled, taxa were dealt with as operational taxonomic units or only a few sampling periods were used. The unique aspects of our study were the length of the evaluations, the intensity of sampling, the depth of the taxonomic study, and the use of entire small watersheds as treatment and control plots.

Materials and Methods

The study was conducted on the Fernow Experimental Forest in northcentral West Virginia. This forest is located on the unglaciated Allegheny Plateau (latitude 39° 95′ N, longitude 79° 41′ W) at elevations ranging from 533 to 1,112 m. The topography is rugged with slopes up to 10–60%. The experimental forest encompasses 1,902 ha and has been divided into a series of watersheds for research purposes (Northeastern Forest Experiment Station 1987). This study was conducted on watersheds 1, 4, 7, and 13. Watersheds 1, 7, and 13 are contiguous; watershed 4 is 200 m from watershed 7 (Fig. 1).

Before establishment of the Fernow Experimental Forest in 1934, the forest was heavily logged between 1905 and 1910. Additional thinnings for research occurred on several of the 4 watersheds from the early 1950s through the late 1970s. Common tree species now on the better quality sites include yellow poplar, *Liriodendron tulipifera* L., sugar maple, *Acer saccharum* Marsh., black cherry, *Prunus serotina* Ehrh., white ash, *Fraxinus americana* L., basswood, *Tilia americana* L., and northern red oak, *Quercus rubra* L. Dominant tree species on the poorer quality sites include various species of oak, *Quercus spp.*, hickory, *Carya spp.*, sourwood, *Oxydendrum arboreum* (L.) DC, and sassafras, *Sassafras albidum* (Nutt.) Nees (Adams et al. 1993).

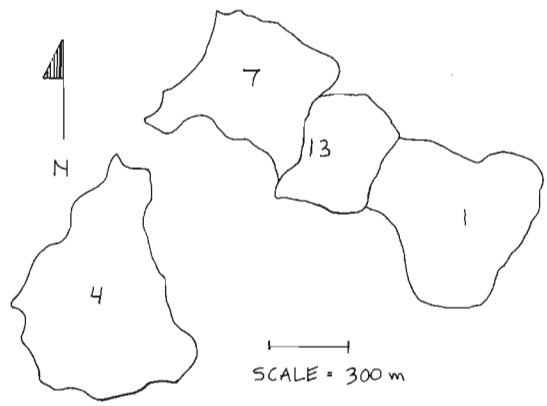

Fig. 1. Arrangement of study watersheds: control, 4, 7; diflubenzuron-treated, 1, 13.

Watershed 1 is 30.1 ha with a NE aspect (exposure). It has not been harvested since 1958. Predominant tree species are sugar maple, basswood, beech, *Fagus grandifolia* Ehrh., black locust, *Robinia pseudoacacia* L., cucumber magnolia, *Magnolia acuminata* L., and red oak. Watershed 4 is 38.7 ha with an ESE aspect; it is a relatively mature forest not having been logged since 1910. The most abundant tree species are beech, sugar maple, striped maple, *Acer pensylvanicum* L., yellow poplar, basswood, and red oak. Watershed 7 is 24.2 ha with an E aspect and it is the youngest of the forest stands, having been last cut and treated with herbicides in 1969. The most abundant species are black birch, *Betula lenta* L., sugar maple, red oak, and yellow poplar. Watershed 13 is 14.2 ha with a NNE aspect. Light-selective cutting of this watershed occurred until 1960 but it retains the characteristics of a relatively mature stand. Predominant species are sugar maple, red oak, yellow poplar, black cherry, and black birch.

We began collecting arthropods under burlap bands and from pole pruned foliage on all 4 watersheds in 1989. Trees that were sampled included black cherry, mixed birch (black and yellow), mixed maple (red and sugar), and mixed oak (red and white). Burlap bands mimic naturally occurring bark flaps and are useful for sampling arthropods that move up and down the tree trunks. Bands 24 cm wide were stapled at breast height (1.32 m) to trees. Ten black birch and black cherry trees were banded in late April 1989 on all 4 watersheds. Ten mixed maples also were banded on each watershed in early July 1989. In early May 1990, 10 mixed oaks per watershed were added giving a total of 160 banded trees for the study. Each year, bands were installed in late April and sampling was begun on the following dates: 4 May 1989, 14 May 1990, 6 May 1991, 11 May 1992, 10 May 1993, and 9 May 1994. All bands were checked weekly and all arthropods removed and

returned to the laboratory for identification to family. Macrolepidopteran larvae were identified to species directly or were reared to the adult for confirmation (Butler 1992). Bands remained in place until early to mid-August each year and then removed after the last sampling date: 15 August 1989, 13 August 1990, 12 August 1991, 10 August 1992, 9 August 1993, and 8 August 1994.

The 1st foliage samples were taken 17 May 1989 and consisted of two 25-branch-tip samples of mixed birch and 2 of black cherry on each of the 4 watersheds, providing a total of 16 samples per sampling date. On 27 June 1989, mixed maples were sampled using this procedure resulting in a total of 24 samples per week for the 1989 season. In 1990, sampling of mixed oak (primarily red oak) was initiated providing a total of 32 samples of foliage on each sampling date.

Foliage was collected from low- to midcanopy with pole pruners equipped with large plastic bags to catch the clipped foliage. Foliage sampling was begun soon after leaf expansion each spring: 17 May 1989, 15 May 1990, 7 May 1991, 12 May 1992, 11 May 1993, and 8 May 1994, and was continued weekly until mid-August for a total of 15 sampling dates each season.

The 32 foliage samples collected from mixed birch, black cherry, mixed maple, and mixed oak were returned to the laboratory where all arthropods were removed and identified to family. Macrolepidopteran larvae on foliage were identified to species. After arthropods were removed from foliage, leaves were removed from twigs in each sample, placed in paper bags, oven dried, and weighed. Arthropod abundance was based on foliage samples standardized to 50g dry leaf weight.

On 16 May 1992, Dimilin 4 L was applied by helicopter to watersheds 1 and 13 at a dose of 35.1 g (AI)/ha. The Dimilin-treated watersheds totaled 44.3 ha, and the untreated control watersheds 4 and 7, totaled 62.9 ha. Comparisons were made between treated and control plots for diversity and abundance of arthropods (without macrolepidopteran larvae) on foliage and under bands; species diversity and abundance of macrolepidopterous larvae on foliage and under bands; abundance of Psocoptera, Phlaeothripidae, selected mandibulate herbivores (Chrysomelidae and Curculionidae), and haustellate herbivores (Cicadellidae, Membracidae, Aphididae and Miridae) herbivores on foliage and Gryllacrididae, Carabidae, and Agelenidae under bands (Table 1). These taxa were chosen for analysis because they were the more abundant or best represented groups that had potential to show sensitivity to diflubenzuron. For analysis, data from each plot was pooled for all 4 tree species groups, by sampling date and by year. Analysis of variance (ANOVA) was used to test for possible differences between means of the 2 treatment and 2 control watersheds for each of the 6 yr of the study. We did not conduct between-year comparisons because of high between-year variation in both diversity and abundance parameters. We tested the null hypothesis that diversity and abundance of taxa did not differ between treated and control watersheds. The analysis was performed using the GLM Procedure (SAS Institute 1990). An alpha level of $P = 0.05$ was used to determine significance in all statistical analyses. Voucher specimens from the study are deposited in the West Virginia University Arthropod Collection.

Table 1. Dependent variables for taxa under burlap bands and on foliage used in tests comparing the difference between diflubenzuron-treated and untreated control watersheds

Variable	Description
Bands[a]	
Famdivers	No. arthropod taxa, primarily families (without macrolepidopterans)
Arthabund	Total count of all arthropods (without macrolepidopterans)
Macrolepdivers	No. species of macrolepidopterous larvae
Macrolepabund	Total count of macrolepidopterous larvae
Gryllabund	Abundance of Gryllacrididae
Carababund	Abundance of Carabidae
Agelenabund	Abundance of Agelenidae
Foliage[a,b]	
Famdivers	No. arthropod taxa (primarily families) (without macrolepidopterans)
Arthabund	Total count of all arthropods (without macrolepidopterans)
Macrolepdivers	No. species of macrolepidopterous larvae
Macrolepabund	Total count of macrolepidopterous larvae
Psocopabund	Abundance of Psocoptera
Phlaeoabund	Abundance of Phlaeothripidae
Haustabund	Abundance of pooled Aphididae, Cicadellidae, Miridae, and Membracidae
Coleopabund	Abundance of pooled Curculionidae and Chrysomelidae

[a] Data pooled from the 4 tree species groups.
[b] Counts expressed as treatment means standardized to 50 g dry weight foliage samples.

Results

Burlap Bands Samples. Total family diversity of arthropods under burlap bands during the 6-yr study, regardless of treatment, was 188 (several taxa were classes, orders, or superfamilies) (Butler et al. 1995a). Mean arthropod family diversity under bands was similar between treated and control watersheds in all years of the study except 1992, the Dimilin application year (Table 2), when diversity on the treated watersheds was significantly lower ($F = 22.15$; df = 1, 2; $Pr > F$ 0.0423) than on the control watersheds. No Dimilin impact on arthropod abundance under bands was noted (Table 2); abundance was higher on treated watersheds for 5 of the 6 yr of the study, including the treatment and posttreatment years. Phalangida was the most abundant taxon under bands for all years (Table 3). No treatment effect was seen for Phalangida or several other abundant taxa under bands (Table 3) including Formicidae, Agelenidae, Diplopoda, Miridae, Theridiidae, or Chilopoda.

Table 2. ANOVA of a diflubenzuron treatment on burlap band arthropod family diversity and abundance (without macrolepidopterans) and band macrolepidopteran diversity and abundance (without gypsy moth)

Dependent variable	Year	Treatment	Mean ± SE	*F*	*Pr* > *F*
Famdivers	1989	T	26.00 ± 1.00		
		C	24.00 ± 1.00	2.00	0.2929
	1990	T	63.5 ± 4.26		
Pretreatment		C	66.0 ± 4.26	0.17	0.1783
	1991	T	71.00 ± 4.14		
		C	76.50 ± 4.14	0.88	0.4465
	1992	T	65.50 ± 1.80		
		C	77.50 ± 1.80	22.15	0.0423*
	1993	T	57.50 ± 2.69		
Posttreatment		C	60.50 ± 2.69	0.62	0.5133
	1994	T	70.00 ± 9.96		
		C	72.50 ± 9.96	0.03	0.8754
Arthabund	1989	T	146.00 ± 9.62		
		C	107.00 ± 9.62	8.22	0.1032
	1990	T	1,349.00 ± 277.63		
Pretreatment		C	1,354.50 ± 277.63	0.00	0.9901
	1991	T	3,930.50 ± 1,088.18		
		C	3,492.50 ± 1,088.18	0.08	0.8027
	1992	T	2,427.00 ± 516.33		
		C	2,109.00 ± 516.33	0.19	0.7057
	1993	T	1,581.50 ± 366.69		
Posttreatment		C	1,446.00 ± 366.69	0.07	0.8183
	1994	T	2,546.50 ± 507.08		
		C	2,234.50 ± 507.08	0.19	0.7060
Macrolepdivers	1989	T	11.00 ± 2.83		
		C	9.00 ± 2.83	0.25	0.6667
	1990	T	17.00 ± 2.26		
Pretreatment		C	15.50 ± 2.26	0.25	0.6855
	1991	T	26.00 ± 3.16		
		C	37.00 ± 3.16	6.05	0.1331
	1992	T	6.00 ± 0.35		
		C	16.50 ± 0.35	441.00	0.0023*
	1993	T	23.50 ± 1.12		
Posttreatment		C	18.50 ± 1.12	10.00	0.0871
	1994	T	24.00 ± 4.53		
		C	25.00 ± 4.53	0.02	0.8902
Macrolepabund	1989	T	28.50 ± 15.35		
		C	33.00 ± 15.35	0.04	0.8550
	1990	T	86.50 ± 10.64		
Pretreatment		C	80.00 ± 10.64	0.19	0.7078
	1991	T	201.00 ± 14.75		
		C	239.50 ± 14.75	3.41	0.2063
	1992	T	16.00 ± 5.10		
		C	63.00 ± 5.10	42.48	0.0227*
	1993	T	138.00 ± 11.32		
Posttreatment		C	133.50 ± 11.32	0.08	0.8050
	1994	T	260.00 ± 72.76		
		C	182.50 ± 72.76	0.57	0.5299

T, treatment plots; C, untreated control plots; *, $P < 0.05$.

Trends for Carabidae and Gryllacrididae showed greater abundance in treated watersheds in pretreatment and treatment years and reduced abundance on treated watersheds in the last 2 posttreatment years (Tables 3 and 4).

Total diversity of macrolepidoptera larvae under bands, regardless of treatment, was 86 species (Butler et al. 1995b). Macrolepidopterous larval diversity (F = 441.0; df = 1, 2; $Pr > F$ 0.0023) and abundance (F = 42.48; df = 1, 2; $Pr > F$ 0.0227) (without gypsy moth) were significantly reduced under bands on treated watersheds in 1992 (Table 2) but not in the subsequent years. Of the 5 most abundant macrolepidopterous larval species, only *Abagrotis alternata* (Grote) (Noctuidae) appeared unaffected, at least during 1992 (Table 3). No gypsy moth larvae were found under bands in treated watersheds during the treatment year.

Foliage Samples. Total diversity of arthropods on foliage during the 6-yr study, regardless of treatment, was 225 families (Butler et al. 1995b). Family diversity on foliage (without macrolepidoptera) was significantly reduced (F = 19.88; df = 1, 2; $Pr > F$ 0.0468) on treated watersheds in 1993 (Table 5). Arthropod abundance was significantly reduced in 1992 (F = 19.06; df = 1, 2; $Pr > F$ 0.0487) and 1994 (F = 18.84; df = 1, 2; $Pr > F$ 0.0492). Nonsignificant treatment effects were observed for di-

Table 3. Mean abundance and standard deviation of the 10 most abundant arthropod taxa (exclusive of macrolepidoptera families) and the 5 most abundant macrolepidopterous species under burlap bands on treated and control watersheds for the years 1989–1994

Taxon	1989		1990		1991		1992		1993		1994		Total
	T	C	T	C	T	C	T	C	T	C	T	C	
Phalangida	22 ± 2	19 ± 5	454 ± 182	564 ± 170	2,207 ± 798	2,084 ± 974	618 ± 201	481 ± 120	418 ± 81	399 ± 145	705 ± 66	636 ± 261	8,604
Carabidae	21 ± 12	10 ± 4	222 ± 51	146 ± 76	197 ± 47	164 ± 20	380 ± 77	231 ± 76	189 ± 67	209 ± 87	93 ± 30	106 ± 23	1,966
Formicidae	4 ± 3	1 ± 1	32 ± 1	41 ± 30	105 ± 1	145 ± 14	465 ± 118	280 ± 124	244 ± 78	72 ± 7	247 ± 163	48 ± 2	1,683
Gryllacrididae	15 ± 1	12 ± 8	145 ± 10	115 ± 24	267 ± 13	147 ± 58	153 ± 5	129 ± 3	106 ± 16	164 ± 55	198 ± 61	203 ± 75	1,652
Agelenidae	20 ± 5	7 ± 3	88 ± 30	58 ± 42	228 ± 36	123 ± 82	231 ± 17	102 ± 59	241 ± 25	135 ± 75	195 ± 6	122 ± 62	1,546
Diplopoda	1 ± 0	1 ± 0	119 ± 13	142 ± 84	44 ± 3	19 ± 3	90 ± 60	66 ± 41	5 ± 2	5 ± 2	40 ± 10	59 ± 35	588
Miridae	0	0	2 ± 2	3 ± 1	15 ± 9	26 ± 6	84 ± 63	81 ± 58	113 ± 88	133 ± 111	68 ± 15	44 ± 15	567
Entomobryidae	3 ± 2	1 ± 1	6 ± 1	3 ± 1	235 ± 94	29 ± 2	15 ± 2	47 ± 33	6 ± 3	19 ± 13	85 ± 81	9 ± 4	455
Theridiidae	2 ± 1	1 ± 1	8 ± 1	9 ± 2	67 ± 15	65 ± 21	35 ± 11	32 ± 7	33 ± 7	27 ± 14	23 ± 4	30 ± 16	329
Chilopoda	1 ± 1	1 ± 0	8 ± 3	14 ± 9	11 ± 1	18 ± 5	51 ± 14	43 ± 15	8 ± 7	13 ± 4	3 ± 2	7 ± 2	174
Lymantria dispar (L.)	3 ± 1	1 ± 0	34 ± 13	25 ± 15	80 ± 24	113 ± 87	0 ± 0	205 ± 156	163 ± 12	729 ± 485	367 ± 44	495 ± 24	2,215
Polia latex (Guenée)	15 ± 5	18 ± 13	37 ± 1	33 ± 5	85 ± 16	82 ± 21	0	14 ± 5	34 ± 6	28 ± 10	8 ± 2	18 ± 6	369
Orthosia rubescens (Walker)	1 ± 1	1 ± 1	1 ± 0	2 ± 1	9 ± 4	5 ± 1	0	5 ± 1	30 ± 9	25 ± 7	122 ± 48	44 ± 19	243
Abagrotis alternata (Grote)	2 ± 1	2 ± 1	14 ± 3	18 ± 6	14 ± 2	22 ± 13	11 ± 6	14 ± 5	4 ± 2	4 ± 1	3 ± 1	4 ± 4	110
Lithophane hemina Grote	1 ± 1	1 ± 1	9 ± 0	9 ± 3	8 ± 3	13 ± 9	0	6 ± 4	28 ± 9	27 ± 5	1 ± 1	3 ± 2	103

T, treatment plots; C, control plots.

versity in 1992 and abundance in 1993 (Table 5). Among some of the more abundant taxa on foliage, significant treatment effects were not noted, but trends indicated reduction of several haustellate sapfeeders (Aphididae, Miridae, Membracidae, and Eriosomatidae), predatory thrips (Phlaeothripidae), and bark lice (Psocidae and Polypsocidae) at least in the treatment year (Tables 6 and 7). Analysis of the pooled coleopterous herbivores Curculionidae and Chrysomelidae showed significant reduction during 1992 ($F = 58.68$; df = 1, 2; $Pr > F$ 0.0166) and nonsignificant trends of reduction during the post-treatment years. We note that these beetles were more abundant on control watersheds in 1991 as well, but not significantly so (Tables 6 and 7).

During the study, 111 total species of macrolepidopterous larvae were recorded on foliage regardless of treatment (Butler et al. 1995a). Although macrolepidopteran larval diversity was reduced in 1992, the reduction was not significant. Caterpillar abundance on foliage was reduced from 1992 through 1994, the reduction being significant during 1993 ($F = 89.43$; df = 1, 2; $Pr > F$ 0.0110) and 1994 ($F = 27.47$; df = 1, 2; $Pr > F$ 0.0345) (Table 5). The 10 most abundant macrolepidopterous larvae on foliage (Table 6) tended to have similar abundance on treated and control watersheds before the diflubenzuron treatment, and reduced abundance on treated watersheds after the application.

Discussion

These data for diversity and abundance of forest arthropods reported here compare favorably with those reported at other locations in West Virginia (Butler 1992; Butler and Kondo 1993; Butler et al. 1995 a–c). We suggest that the results of our current study are therefore applicable to other eastern deciduous forests.

Significant diflubenzuron treatment effects were seen for arthropod family diversity under bands and on foliage, arthropod abundance on foliage, macrolepidopterous larval diversity under bands, macrolepidopterous larval abundance under bands and on foliage, and herbivorous coleopteran abundance on foliage. No significant effects were observed for other taxa including Phalangida, Formicidae, Araneae, Diplopoda, or Chilopoda. These results agree with those of Martinat et al. (1988) who noted diflubenzuron effects on macrolepidopterous and microlepidopterous larvae and nonlepidopterous mandibulate herbivores. In our study, numbers of our most abundant microlepidopterous larvae, Gelechiidae and Tortricidae, were too low for analysis, but treatment effects did not appear to occur. Of all mandibulate herbivores, Martinat et al. (1988) found the microlepidoptera to be the least affected by diflubenzuron. Earlier studies (Grannett and Retnakaran 1977) indicated that certain forest microlepidoptera such as spruce

Table 4. ANOVA of a diflubenzuron treatment on abundance of arthropods found under burlap bands

Dependent variable	Year	Treatment	Mean ± SE	*F*	*Pr* > *F*
Gryllabund	1989	T	15.00 ± 5.35		
		C	11.50 ± 5.35	0.21	0.6891
	1990	T	144.50 ± 18.25		
Pretreatment		C	115.00 ± 18.25	1.31	0.3715
	1991	T	267.00 ± 42.03		
		C	147.00 ± 42.03	4.08	0.1810
	1992	T	153.00 ± 4.12		
		C	129.00 ± 4.12	16.94	0.0543
	1993	T	106.00 ± 40.16		
Posttreatment		C	163.50 ± 40.16	1.02	0.4179
	1994	T	197.50 ± 68.14		
		C	203.00 ± 68.14	0.00	0.9597
	1989	T	20.50 ± 8.50		
Carababund		C	9.50 ± 8.50	0.84	0.4567
	1990	T	222.00 ± 64.72		
Pretreatment		C	146.00 ± 46.72	0.69	0.4937
	1991	T	197.00 ± 36.12		
		C	164.00 ± 36.12	0.42	0.5845
	1992	T	379.50 ± 76.00		
		C	230.50 ± 76.00	1.92	0.3000
	1993	T	189.00 ± 77.65		
Posttreatment		C	209.00 ± 77.56	0.03	0.8723
	1994	T	92.50 ± 26.45		
		C	106.00 ± 26.45	0.13	0.7527
Agelenabund	1989	T	19.50 ± 3.64		
		C	6.50 ± 3.64	6.38	0.1275
	1990	T	87.50 ± 36.00		
Pretreatment		C	57.50 ± 36.00	0.35	0.6154
	1991	T	228.00 ± 63.00		
		C	122.50 ± 63.00	1.40	0.3580
	1992	T	230.50 ± 42.98		
		C	101.50 ± 42.98	4.50	0.1678
	1993	T	240.50 ± 55.79		
Posttreatment		C	135.00 ± 55.79	1.79	0.3130
	1994	T	194.50 ± 44.01		
		C	122.00 ± 44.01	1.36	0.3642

T, treatment plots; C, control plots.

budworm, *Choristoneura fumiferana* (Clemens) (Lepidoptera: Tortricidae), may not be highly sensitive to diflubenzuron.

Of the macrolepidopterous larvae under bands, *A. alternata* was the only species that appeared unaffected by the diflubenzuron treatment. In 1992, it was the only species represented by more than a single individual during the season. This caterpillar overwinters in leaf litter as a late instar and activates in the spring to complete its development feeding on such understory plants as *Vaccinium*. By the time the diflubenzuron application was made, these larvae were mature and appeared unaffected.

Martinat et al. (1993) conducted pitfall trap sampling in a diflubenzuron treated forest and found a treatment effect for spider and orthopteroid abundance. In our study, numbers of spiders and orthopteroids on the foliage were too few for analysis. We did note a trend of reduced abundance of spiders in the families Theridiidae and Dictynidae in 1993 and 1992, respectively. Only spiders in the family Agelenidae were abundant enough under bands for analysis, and we saw no treatment effect.

The most abundant orthopteroids under bands in our study were the Gryllacrididae. Although no significant treatment effect was found, a reduction in 1993 was noted. Because diflubenzuron is known to persist on treated foliage throughout the treatment season (Wimmer et al. 1993) and move into the litter layer at leaf fall (Wimmer 1995), we expected diflubenzuron availability to Gryllacrididae to increase after leaf fall. Approximately 80% of the orthopteroids in the Martinat et al. (1993) study were Gryllacrididae; this short-term study found orthopteroid treatment effects within the treatment year.

Although diflubenzuron did not show a significant treatment effect on predatory Carabidae or haustellate taxa under bands or on predatory Phlaeothripidae or lichen feeding Psocoptera (Psocidae and Polypsocidae) on foliage, reduction of these groups was noted on treated plots during 1992 or 1993. Previous studies in the laboratory have shown the effect of diflubenzuron on beneficial predatory insects (Ables et al. 1977, Broadbent and Pree 1984).

Diflubenzuron has its greatest impact on immature insects and may show no effect on adults

Table 5. ANOVA of a diflubenzuron treatment on foliage arthropod family diversity and abundance (without macrolepidopterans) and foliage macrolepidopteran species diversity and abundance

Dependent variable	Year	Treatment	Mean ± SE	*F*	*Pr* > *F*
Famdivers	1989	T	82.00 ± 2.85		
		C	83.50 ± 2.85	0.14	0.7455
	1990	T	79.00 ± 5.26		
Pretreatment		C	79.00 ± 5.26	0.37	0.6065
	1991	T	105.00 ± 2.24		
		C	100.00 ± 2.24	2.50	0.2546
	1992	T	94.00 ± 5.26		
		C	102.50 ± 5.26	1.31	0.3712
	1993	T	77.50 ± 2.06		
Posttreatment		C	90.50 ± 2.06	19.88	0.0468*
	1994	T	77.00 ± 5.70		
		C	81.00 ± 5.70	0.25	0.6690
Arthabund	1989	T	6,038.13 ± 722.03		
		C	5,223.36 ± 722.03	0.64	0.5086
	1990	T	1,017.30 ± 153.71		
Pretreatment		C	1,043.51 ± 153.71	0.01	0.9150
	1991	T	6,648.57 ± 1114.63		
		C	7,627.81 ± 1114.63	0.39	0.5978
	1992	T	5,669.71 ± 386.70		
		C	8,057.14 ± 386.70	19.06	0.0487*
	1993	T	2,768.77 ± 547.89		
Posttreatment		C	3,598.07 ± 547.89	1.15	0.3965
	1994	T	2,695.31 ± 189.67		
		C	3,859.52 ± 189.67	18.84	0.0492*
Macrolepdivers	1989	T	24.50 ± 2.57		
		C	19.00 ± 2.57	2.28	0.2699
	1990	T	36.50 ± 2.69		
Pretreatment		C	34.50 ± 2.69	0.28	0.6518
	1991	T	34.50 ± 3.95		
		C	42.00 ± 3.95	1.80	0.3118
	1992	T	13.00 ± 3.02		
		C	25.50 ± 3.02	8.56	0.096
	1993	T	28.50 ± 3.34		
Posttreatment		C	39.00 ± 3.34	4.96	0.1559
	1994	T	34.00 ± 1.46		
		C	37.50 ± 1.46	2.88	0.2317
Macrolepabund	1989	T	181.47 ± 17.20		
		C	183.70 ± 17.20	0.01	0.9352
	1990	T	223.09 ± 59.59		
Pretreatment		C	248.93 ± 59.59	0.09	0.7882
	1991	T	500.85 ± 102.46		
		C	553.22 ± 102.46	0.13	0.7524
	1992	T	127.53 ± 46.40		
		C	316.83 ± 46.40	8.32	0.1221
	1993	T	166.00 ± 10.82		
Posttreatment		C	310.72 ± 10.82	89.43	0.0110*
	1994	T	225.41 ± 15.98		
		C	343.84 ± 15.98	27.47	0.0345*

T, treatment plots; C, control plots. *, $P < 0.05$.

(Maas et al. 1981). However, diflubenzuron also may act as an ovicide either by direct contact with the egg or through the female insect (Ables et al. 1980, Maas et al. 1981). Diflubenzuron may affect arthropods directly through impact on larvae at molts or through effects on egg development. The effects also may be indirect such as reducing availability of prey to predators or parasitoids.

Our findings on diflubenzuron effects on mandibulate herbivores in the forest canopy were expected based on the results of Martinat et al. (1988). However, we did not expect to find a significant reduction in arthropod total abundance (without macrolepidoptera) and macrolepidopteran abundance in the forest canopy that persisted through the 3rd posttreatment season.

Recovery from perturbation occurs through immigration and natality (MacArthur and Wilson 1967). Immigration should be most influenced by mobility of arthropods and by size of treatment blocks. Miller (1990b) suggested that small treatment blocks in a forest may not show long-term effects of pesticide treatment because of the dynamics of colonization similar to the concepts associated with island biogeography. We consider our treatment blocks in this study to have been small (30 and 14 ha), yet recovery in certain diversity and abundance parameters had not occurred for 27 mo

Table 6. Mean abundance and standard error of the 10 most abundant arthropod families (exclusive of macrolepidoptera families) and macrolepidopterous species on foliage on treated and control watersheds for the years 1989–1994

Taxon	1989		1990		1991		1992		1993		1994		Total
	T	C	T	C	T	C	T	C	T	C	T	C	
Aphididae	2,378 ± 32	1,147 ± 428	115 ± 25	99 ± 39	1,863 ± 271	1,792 ± 129	1,980 ± 45	2,178 ± 329	186 ± 35	276 ± 53	555 ± 49	824 ± 336	13,694
Cicadellidae	349 ± 22	269 ± 7	47 ± 25	31 ± 15	874 ± 82	1,497 ± 212	688 ± 150	978 ± 129	152 ± 23	166 ± 0	184 ± 32	181 ± 32	5,415
Miridae	230 ± 84	189 ± 81	37 ± 3	35 ± 17	380 ± 21	310 ± 90	633 ± 130	843 ± 273	635 ± 152	710 ± 331	432 ± 28	397 ± 61	4,832
Phlaeothripidae	354 ± 5	364 ± 63	28 ± 26	62 ± 16	68 ± 39	156 ± 123	144 ± 96	500 ± 294	331 ± 175	446 ± 137	258 ± 24	452 ± 217	3,162
Curculionidae	134 ± 4	151 ± 3	38 ± 10	32 ± 10	115 ± 30	294 ± 69	69 ± 20	184 ± 38	82 ± 12	317 ± 149	108 ± 4	438 ± 151	1,961
Psocidae	10 ± 3	11 ± 2	2 ± 2	2 ± 2	257 ± 156	222 ± 39	60 ± 46	424 ± 204	229 ± 87	106 ± 32	95 ± 11	168 ± 52	1,586
Polypsocidae	342 ± 99	236 ± 46	56 ± 9	32 ± 16	368 ± 161	406 ± 163	17 ± 1	110 ± 58	10 ± 6	5 ± 2	2 ± 1	1 ± 0	1,584
Theridiidae	58 ± 4	35 ± 16	39 ± 19	25 ± 4	296 ± 38	340 ± 135	203 ± 122	132 ± 64	136 ± 1	207 ± 122	0	1 ± 1	1,471
Membracidae	0	3 ± 0	7 ± 1	4 ± 0	8 ± 7	20 ± 7	126 ± 54	218 ± 31	138 ± 63	183 ± 31	192 ± 28	263 ± 10	1,161
Eriosomatidae	47 ± 45	164 ± 126	0	0	23 ± 22	2 ± 2	210 ± 68	579 ± 295	0	18 ± 18	0	0	1,044
Lomographa glomeraria (Grote)	40 ± 1	48 ± 12	13 ± 6	12 ± 1	26 ± 16	28 ± 1	37 ± 19	92 ± 6	11 ± 2	53 ± 14	23 ± 4	24 ± 1	407
Erannis tiliara (Harris)	11 ± 11	4 ± 2	31 ± 13	56 ± 37	123 ± 82	105 ± 28	11 ± 2	35 ± 6	0	4 ± 2	8 ± 5	6 ± 2	395
L. vestaliata (Gueneé)	19 ± 3	23 ± 5	30 ± 7	10 ± 6	41 ± 9	39 ± 2	9 ± 9	45 ± 6	36 ± 8	39 ± 8	15 ± 8	36 ± 11	342
Melanolophia canadaria (Gueneé)	8 ± 5	13 ± 5	21 ± 7	24 ± 2	60 ± 18	56 ± 3	21 ± 2	44 ± 4	9 ± 4	36 ± 9	6 ± 5	23 ± 2	320
Alsophila glometaria (Harris)	1 ± 1	3 ± 3	6 ± 4	3 ± 3	12 ± 7	15 ± 3	9 ± 7	14 ± 2	20 ± 14	42 ± 4	65 ± 33	74 ± 19	264
Orthosia hibisci (Gueneé)	20 ± 9	23 ± 12	12 ± 1	30 ± 4	30 ± 11	57 ± 16	10 ± 10	6 ± 2	4 ± 4	8 ± 1	7 ± 6	21 ± 0	227
Polia latex (Gueneé)	13 ± 7	15 ± 7	14 ± 8	11 ± 1	26 ± 10	40 ± 2	2 ± 2	10 ± 1	5 ± 0	18 ± 1	5 ± 0	9 ± 1	168
Itame pustularia (Gueneé)	4 ± 2	3 ± 1	4 ± 1	13 ± 9	4 ± 1	8 ± 2	1 ± 1	6 ± 4	8 ± 2	12 ± 3	9 ± 3	74 ± 11	147
Lymantria dispar (L.)	0	0	4 ± 1	3 ± 1	5 ± 1	3 ± 1	6 ± 3	8 ± 6	7 ± 4	11 ± 1	36 ± 1	62 ± 8	145
Morrisonia confusa (Hubner)	18 ± 6	16 ± 4	4 ± 3	4 ± 2	9 ± 0	7 ± 0	1 ± 1	7 ± 3	8 ± 2	4 ± 1	1 ± 0	1 ± 0	80

T, treatment plots; C, control plots.

Table 7. ANOVA of a diflubenzuron treatment on abundance of arthropods on foliage

Dependent variable	Year	Treatment	Mean ± SE	*F*	*Pr* > *F*
Phlaeoabund	1989	T	353.75 ± 44.73		
		C	364.17 ± 44.73	0.03	0.8843
	1990	T	28.19 ± 21.50		
Pretreatment		C	62.37 ± 21.50	1.26	0.3778
	1991	T	67.55 ± 91.44		
		C	155.88 ± 91.44	0.47	0.5651
	1992	T	143.65 ± 218.33		
		C	499.89 ± 218.33	1.33	0.3679
	1993	T	330.72 ± 156.78		
		C	445.77 ± 156.78	0.27	0.6555
Posttreatment	1994	T	258.25 ± 154.39		
		C	451.99 ± 154.39	0.79	0.4685
Psocopabund	1989	T	352.31 ± 79.60		
		C	246.44 ± 79.60	0.88	0.4463
	1990	T	58.23 ± 11.40		
Pretreatment		C	33.33 ± 11.40	2.39	0.2624
	1991	T	625.24 ± 265.60		
		C	627.64 ± 265.60	0.00	0.9955
	1992	T	76.72 ± 188.12		
		C	533.44 ± 188.12	2.95	0.2282
Posttreatment	1993	T	239.09 ± 61.39		
		C	111.17 ± 61.39	2.17	0.2785
	1994	T	96.75 ± 37.68		
		C	169.80 ± 37.68	1.88	0.3039
Coleopabund	1989	T	210.27 ± 77.81		
		C	316.37 ± 77.81	0.93	0.4367
	1990	T	63.64 ± 26.43		
Pretreatment		C	82.13 ± 26.43	0.24	0.6669
	1991	T	154.63 ± 40.67		
		C	393.58 ± 40.67	17.26	0.0534
	1992	T	91.38 ± 17.97		
		C	286.05 ± 17.97	58.68	0.0166*
	1993	T	98.74 ± 97.76		
Posttreatment		C	360.82 ± 97.76	3.59	0.1985
	1994	T	104.04 ± 107.85		
		C	526.87 ± 107.85	6.43	0.1266
Haustabund	1989	T	2,947.42 ± 371.31		
		C	1,908.05 ± 371.31	3.99	0.1837
	1990	T	205.62 ± 49.66		
Pretreatment		C	168.88 ± 49.66	0.27	0.6531
	1991	T	3,124.79 ± 403.86		
		C	3.618.17 ± 403.86	0.75	0.4787
	1992	T	3,428.68 ± 254.87		
		C	4,216.99 ± 254.87	4.78	0.1603
	1993	T	1,110.58 ± 237.31		
Posttreatment		C	1,334.27 ± 237.31	0.04	0.5737
	1994	T	1,363.01 ± 187.95		
		C	1,665.55 ± 187.95	1.30	0.3730

T, treatment plots; C, control plots. *, $P < 0.05$.

after treatment. These blocks represent entire small watersheds, however, the sides are formed by steep slopes of up to10–60%, topography typical of hilly or mountainous areas of eastern United States. Nonetheless, the top and bottom of each watershed opened into untreated woodlands that would be ready sources of colonizing arthropods.

Degree and mode of mobility of the most abundant arthropods in our study vary. Aphididae, Cicadellidae, Miridae, and Phlaeothripidae, for example, are winged in at least some adult forms and also are wind assisted in flight. Among the macrolepidopterans, adult females of *Erannis tiliaria* (Harris), *Alsophila pometaria* (Harris), and *L. dispar* are flightless (Barbosa et al. 1989), and dispersion is a function of ballooning early instar larvae. The remaining more abundant macrolepidoptera have females with relatively strong powers of flight.

Natality may be strongly influenced by voltinism. Miller (1990b) noted that *B. thuringiensis* most strongly affected lepidopteran species that were univoltine and unable to respond as quickly as multivoltine species to local extinction. In our study, among the macrolepidopterans under bands, the 5 most abundant species are univoltine; on foliage, 7 of the 10 most abundant species are univoltine. *Lomographa vestaliata* (Gueneé), *Melanolophia canadaria* (Gueneé), and *Itame pustularia* (Gueneé) may be bivoltine in the study area (Butler 1992, Butler et al. 1995a, Forbes 1948); these

3 species did not recover any more rapidly than the univoltine species (Table 6).

During our 6-yr study, we recorded considerable year to year variation, particularly in abundance of total arthropods and macrolepidopteran larvae. Population fluctuation of forest arthropods has been addressed by numerous researchers (Berryman et al. 1987, Myers 1988, Wolda 1978). Natural fluctuations are controlled by combinations of biotic and abiotic factors (Strong et al. 1984). We suggest that dynamics of arthropods in forest ecosystems are examples of ecological communities in which environmental perturbations are continuous and equilibrium densities are never achieved (i.e., stochastic systems [Ives 1995]). Application of diflubenzuron for forest pest control superimposes an additional perturbation on an already unstable system.

Martinat et al. (1988) concluded that diflubenzuron significantly reduced diversity and abundance of certain nontarget forest arthropods that may affect birds and small mammals by reducing their food supply. However, these authors contend that because insects are highly unpredictable as a food source for vertebrates, even without diflubenzuron perturbation, the impact of the insecticide on the vertebrate community is probably minimal. We suggest that because insect populations fluctuate so dramatically, an additional perturbation may result in impacts more severe than minimal. Indeed, short-term studies on the impact of diflubenzuron on forest songbirds showed that in treated plots, birds increased forage time, expanded their food search area and consumed fewer Lepidoptera (Cooper et al. 1990). Songbirds also shifted diets to less diflubenzuron-susceptible prey (Sample et al. 1993a), and possessed reduced fat reserves (Whitmore et al. 1993). In diflubenzuron treatment plots, abundance of moths available as food for the endangered Virginia big-eared bat was reduced (Sample 1991); and white-footed mice showed dietary shifts and altered juvenile-adult ratios (Seidel and Whitmore 1995). Spring populations of migrant warblers are known to feed almost exclusively on forest lepidopteran larvae on host trees; nonlepidopterans are considered to be insufficient to support migrant populations (Graber and Graber 1983). Diflubenzuron has its greatest impact on lepidopteran larvae.

During our study, gypsy moth was suppressed on the Dimilin-treated watersheds in the treatment and posttreatment years. Although few studies have been conducted on nontarget effects of gypsy moth defoliation on arthropods (Sample et al. 1996), we assume that heavy defoliation would affect competing nontarget arthropods.

Martinat et al. (1988) noted that the chance of detecting pesticide treatment effects on forest arthropods is low when foliage arthropod density or sampling intensity are low, so that as arthropod density declines, the chance of detecting treatment effects also diminishes. We agree and add that the chances of detecting statistically significant differences with poorly replicated studies also is very low. In our Fernow study, only 4 watersheds were made available for the number of years required in this study. Even so, significant treatment effects were found over multiple years indicating a major disruption in some components of the forest arthropod community.

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