RESIDUAL EFFECTS OF THINNING AND HIGH WHITE-TAILED DEER DENSITIES ON NORTHERN REDBACK SALAMANDERS IN SOUTHERN NEW ENGLAND OAK FORESTS

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Abstract: Research has demonstrated that even-aged regeneration harvests, especially clearcutting, can have a major and long-lasting detrimental effect on forest amphibians, but the effects of less intensive silvicultural treatments have not been well documented. Additionally, the chronic overabundance of white-tailed deer (*Odocoileus virginianus*) has become a problem in many parts of North America, with associated effects on vegetation composition and structure and on other wildlife. I assessed the effects of crown thinning and deer overabundance on the relative abundance of forest-floor salamanders in a southern New England mixed oak–hardwood forest. I surveyed salamanders by using cover boards in 16 forest stands with thinned or unthinned treatments and with histories of low (3–6 deer/km^2) or high (10–17 deer/km^2) deer densities. Surveys were conducted 5 times a year for 3 years. Northern redback salamanders (*Plethodon cinereus*) were the dominate species in all surveys and in all treatment classes. Redbacks were most abundant in spring and fall surveys and in the second and third year of the study. Neither thinning nor white-tailed deer density had a significant effect on the number of redback observations: stands with high numbers of redbacks occurred in all treatment classes. At the stand level, numbers of redback observations were positively correlated with the number of pieces and area of coarse woody debris and with the density of tall ($\geq$1 m) woody stems. The study suggests that a stand disturbance, where a large percentage of the canopy is retained and that results in an increase in cover of understory vegetation, would result in no long-term effect on forest-floor salamanders.

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Although the policies and procedures of ecosystem management of forests are evolving, there is consensus that management should have a long-term neutral effect on the structural and functional integrity of the ecosystem (Healy 1994). Within forest ecosystems, the role of amphibians and the relation between forest management and amphibian ecology has been thoroughly reviewed by deMaynadier and Hunter (1995). Salamanders can account for a significant component of the biomass of forest fauna in the eastern United States (Burton and Likens 1975*a*, Hairston 1987), and thus can also play an important role in detritus processing as predators of forest-floor invertebrates (Burton and Likens 1975*b*, Wyman 1998).

Although forestry practices can have a negative effect on some amphibians, maintaining sustainable populations can be compatible with timber harvesting if precautions are taken (deMaynadier and Hunter 1995, Waldick 1997, Harpole and Haas 1999). Forest clearcutting can result in a temporary decline in amphibian numbers (Pough et al. 1987, Petranka et al. 1993), but this effect is not certain (Chazal and Niewiarowski 1998). In New England, this decline is seen principally in numbers of the terrestrial-breeding northern redback salamander (DeGraaf and Yamasaki 1992) and the pond-breeding wood frog (*Rana sylvatica*) and mole salamanders (*Ambystoma* spp.; deMaynadier and Hunter 1998). The effects of intermediate levels of forest harvesting on amphibians (i.e., thinning or selection-based silviculture) are less well known (Harpole and Haas 1999), but small-scale modifications (e.g., firewood cutting) appear to have little detrimental effects on populations of some northern salamanders (Pough et al. 1987).

White-tailed deer overabundance also affects forested ecosystems in many parts of North America (Warren 1997). With sustained high deer densities, for example, browsing can negatively affect habitat composition and structure for other wildlife species (Waller and Alverson 1997), but the effects of chronic high densities of white-tailed deer on amphibians are unknown.

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Over the last 10 or more years, studies in the oak forests of the Quabbin Reservation in central Massachusetts have documented changes in vegetation, birds, gray squirrels (*Sciurus carolinensis*), and small mammals following intermediate timber harvests in areas of high- and low-density white-tailed deer populations (Healy et al. 1987, Brooks and Healy 1988, DeGraaf et al. 1991, Healy 1997). I surveyed the composition and relative abundance of terrestrial amphibians of these forests to assess the lasting, long-term effects of intermediate timber harvests and chronic high levels of deer browsing. Harvested and control stands were located both in areas that supported low-density deer populations and in areas that had supported high-density populations for many years.

STUDY AREA

The study was conducted within the watershed of the Quabbin Reservoir in Franklin, Hampshire, and Worcester counties in central Massachusetts (approximate centroid of study sites at 42°25′N, 72°22′W). The watershed contains a 9,713-ha reservoir and 22,663 ha of surrounding uplands (O'Connor et al. 1995). Approximately 19,000 ha of the uplands are managed forest. The area immediately adjacent to the reservoir was set aside as a wildlife sanctuary, and public access has been controlled since 1938. The property is managed by the Metropolitan District Commission (MDC) to provide municipal water, forest products, wildlife, and recreation.

Study stands were identified from MDC forest inventory data. All stands met the following criteria: (1) white pine (*Pinus strobus*)–northern red oak (*Quercus rubra*)–red maple (*Acer rubrum*) forest type (Type 20; Eyre 1980:27–28); (2) even-aged, 60–100 years old; (3) sawtimber-sized with dominant trees having a diameter at breast height (dbh) >30 cm; and (4) not on extremely steep or rocky sites (Healy et al. 1987, Healy 1997). This forest type and other compositionally similar types occur from central Maine, across southern New England, south through the Middle Atlantic States, west down the Ohio River and central Mississippi River watersheds, and in southern portions of the Great Lakes States (Eyre 1980). The stands generally originated in the first 2 decades of this century, following removal of the pine component. All stands were surrounded by extensive forest land, predominantly sawtimber size and of the same type. Slope and aspect varied among the stands and within individual stands. Soils across the study area were loamy and sandy glacial till, shallow to deep, and well drained (Soil Conservation Service 1967, 1989).

I selected 4 stands in each of 4 treatment categories of a 2 × 2 factorial design: thinned and unthinned control in low deer density and high deer density areas (Healy 1997). Densities of white-tailed deer in the low deer density, hunted stands were estimated from harvest records to be about 3–6 deer/km^2 (Healy 1997). Deer densities in the high deer density, sanctuary stands were estimated by distance-based, line transect sampling at 10–17 deer/km between 1983 and 1992 (Healy 1997). A controlled hunt for white-tailed deer was begun on the high deer density portion of the study site in 1991, 4 years prior to the first year of this study, and has continued every year thereafter and is drastically reducing deer numbers.

Thinnings in the silviculturally treated stands occurred between 12 and 21 years prior to the study (Healy 1997). Thinning was done by removing trees of undesirable species, form, or condition from the main canopy to promote growth on residual trees in accordance with the silvicultural guidelines for upland oaks, and left a residual stocking of 50–60%. In 1983, following this silvicultural treatment, thinned stands were found to have significantly fewer trees ≥2.5 cm in diameter, with significantly less basal area, than unthinned stands (Healy et al. 1987, Healy 1997). Stands with high densities of deer had fewer trees ≥2.5 cm in diameter than stands with low deer densities because of a lack of recruitment into the smaller diameter classes. Average stand diameter was unaffected by thinning but was larger in high deer density stands because of reduced numbers of small-diameter trees.

Thinning resulted in an increase in seedlings and saplings <2.5 cm in diameter, especially for individuals ≥30 cm in height (Healy et al. 1987). In stands with chronic high deer densities, numbers of these stems were greatly reduced. Thinnings had no consistent effect on other understory herbaceous and shrub vegetation cover, but high deer numbers resulted in less forb and greater graminoid cover. In thinned stands with high deer numbers, fern cover increased dramatically (Healy et al. 1987).

The most recent survey of understory vegetation composition and structure in the study stands occurred in 1994, >10 years since the most recent thinning and 3 years after deer numbers were reduced (W. H. Healy, U.S. Forest Service, unpublished data). Thinned stands had a greater density of stems <2.5 cm in diameter and a greater cover of herbaceous and shrub cover than did unthinned stands. Significant differences existed for woody stems 30–99 cm in height (unthinned stand average = 3,500 stems/ha; thinned stand average = 8,600 stems/ha; $F_{1,12} = 8.85$, $P = 0.025$), stems ≥100 cm in height (unthinned stand average = 1,400 stems/ha; thinned stand average = 6,600 stems/ha; $F_{1,12} = 7.02$, $P = 0.038$), and for forb cover (unthinned stand average = 12.2%; thinned stand average = 30.5%; $F_{1,12} = 9.95$, $P = 0.017$). Stands with a history of high densities of deer had nonsignificantly lower densities of woody stems ≥30 cm in height, greater fern cover, and lower shrub cover than stands that had always supported low densities of deer.

METHODS

I surveyed the relative abundance of terrestrial salamanders by using cover boards (DeGraaf and Yamasaki 1992, Fellers and Drost 1994, Davis 1997). The use of cover boards, a type of artificial cover object (ACO), for surveying terrestrial salamanders has not been thoroughly validated (North American Amphibian Monitoring Program [NAAMP]. 1997. The terrestrial salamander monitoring program. Bias: the disguiser of population trends. URL: http://www.im.nbs.gov/sally3.html). A small number of studies have found that the numbers of salamanders under boards were correlated with independent indices of salamander abundance (DeGraaf and Yamasaki 1992; NAAMP. 1997. The terrestrial salamander monitoring program: recommended protocol for running cover object arrays. URL: http://www.im.nbs.gov/sally4.html). The use of boards or other types of ACOs is being recommended for monitoring terrestrial salamanders (Fellers and Drost 1994, Davis 1997, NAAMP. 1997. The terrestrial salamander monitoring program: recommended protocol for running cover object arrays. URL: http://www.im.nbs.gov/sally4.html). I chose to use cover boards based on these recommendations, the nondestructive nature of the methodology, and the ability to implement a highly standardized sampling design.

In early December 1994, I placed 2 rough-cut hemlock (*Tsuga canadensis*) boards, 1 m × 25 cm × 4 cm, in contact with the humus, 2.5 m apart at each of 30 vegetation survey plots in each stand. The plots were systematically located ≥20 m apart along transects that paralleled the long axis of the stands. The boards had been air-dried prior to purchase and installation.

Individual salamanders and other vertebrates under each board were counted 5 times each year between the months of May and October, 1995 through 1997. Animals were handled only if there was a question as to species identification; otherwise animals remained where observed when the board was carefully lowered. Northern redback salamanders (hereafter, redback salamanders) were visually identified as unsexed or young-of-the-year (<30 mm snout–vent length), immature or 1 year old (30–40 mm), or mature or ≥2 years old (>40 mm; Sayler 1966). The interval between successive surveys was never <2 weeks and generally was 4 weeks. The surveys were not selected to follow precipitation events. The exact dates of a survey were scheduled to avoid having a precipitation event occur during a survey. It took between 1–4 days to turn boards in all stands, but usually between 2–3 days.

Coarse woody debris (CWD), including both logs and stumps, was measured on 33.3-m^2 circular plots centered on a point midway between the 2 boards. To be surveyed, CWD had to be in contact with the forest floor. The minimum diameter for CWD was 10 cm; no minimum length was set (Harmon and Sexton 1996). The CWD diameters were measured with calipers, length with a tape, and each piece was assigned to 1 of 4 decay classes (U.S. Forest Service 1995). Decay classes ranged between sound (Class 1) to fully rotted (Class 4) and were assigned based on structural integrity, texture of rotten portions, and bark conditions. Depths of organic soil horizons (litter, fragmentation, humus) were measured in a shallow profile excavated about 30 cm from the outer edge of each cover board on each CWD survey plot. Because of time constraints, CWD was surveyed only on even- or odd-numbered plots ($n = 15$), with the decision determined by the toss of a coin in each stand.

The relative abundance of redback salamanders and forest floor habitat attributes were analyzed for the effects of deer density and thinning by analysis of variance. Redback salamanders observations were summed across plots to the stand because no salamanders were seen at

Table 1. Number of observed fauna under cover boards[a] by year and species, Quabbin Reservation, central Massachusetts, 1995–97.

	1995	1996	1997	Total
Red-spotted newt (*Notophthalmus viridescens*[b])	15	36	33	84
Spotted salamander (*Ambystoma maculatum*)	1	3	6	10
Northern dusky salamander (*Desmognathus fuscus*)	2	0	4	6
Northern two-lined salamander (*Eurycea bislineata*)	1	3	2	6
Four-toed salamander (*Hemidactylium scutatum*)	1	0	0	1
Northern redback salamander (*Plethodon cinereus*)	388	985	907	2,280
Northern spring peeper (*Hyla crucifer crucifer*)	0	1	0	1
Pickerel frog (*Rana palustris*)	0	1	0	1
Northern ringneck snake (*Diadophis punctatus*)	0	3	4	7
Eastern garter snake (*Thamnophis sirtalis sirtalis*)	3	0	3	6
Northern short-tailed shrew (*Blarina brevicauda*)	0	0	1	1
White-footed mouse (*Peromyscus leucopus*)	17	1	1	19
Southern redback vole (*Clethrionomys gapperi*)	13	0	0	13

[a] n = 4,800/year (2 boards/plot × 30 plots/stand × 16 stands × 5 surveys/year).
[b] Nomenclature follows Collins (1997) for amphibians and reptiles, and Banks et al. (1987) for mammals.

many plots. These values were transformed by the square root of the number of observations per stand per survey occasion to improve normality of distribution (Mateu 1997), and then analyzed via a repeated-measures design, where both year and survey occasion were individually evaluated (Damon and Harvey 1987). The distribution of redback salamander total numbers by size class and treatment class was analyzed via the $r \times c$ contingency table (Conover 1971: 149–154). All analyses were performed with SYSTAT (Wilkinson et al. 1992). Significance level was $\alpha = 0.05$.

RESULTS

Redback salamanders were the most abundant vertebrates observed under cover boards (Table 1). Of the 2,280 redback salamanders observed during the study, 1,950 occurred as single specimens under a board. There were 148 observations where 2 redback salamanders occurred under a single board, 10 triples, and 1 observation with 4 salamanders. Redback salamanders were observed under 2,111 of 14,400 boards turned during the 15 surveys. The maximum number of boards in a stand (n = 60) with redbacks was 43; no redbacks were observed in >1 stand in several surveys. During the study, there was a redbacks salamander observed under an average of 3.8 boards in a stand. More than 75% of the redback salamanders were sexually mature, 13% were immature, and 12% were young-of-the year. No difference was found in the distribution of redback salamanders by size class across the 4 treatment classes ($\chi^2_6 = 8.1$, $P = 0.231$).

Seven additional amphibian species were observed (Table 1). Red-spotted newts (*Notophthalmus viridescens*) were most often observed as terrestrial efts and occurred in 4 stands located near permanent ponds. Spotted salamander (*Ambystoma maculatum*) observations were of immatures, presumedly dispersing from natal ponds. Northern dusky (*Desmognathus fuscus*) and northern two-lined salamanders (*Eurycea bislineata*) breed in or near streams (DeGraaf and Rudis 1983) and were observed on plots near streams.

Neither thinning nor deer density had a significant effect on the number of redback salamander observations ($F_{1,12(\mathrm{deer})} = 0.998$, $P = 0.338$; $F_{1,12(\mathrm{thin})} = 2.606$, $P = 0.132$). Thinning and the interaction of thinning and deer density were greater sources of treatment variance ($\mathrm{MSE_{thin}} = 5.62$; $\mathrm{MSE_{deer \times thin}} = 5.79$) than deer density alone ($\mathrm{MSE_{deer}} = 2.15$). Redback salamander numbers were frequently lowest in the unthinned, low deer density treatment and generally greatest in the thinned, low deer density treatment, but no treatment class consistently had the greatest number of redback salamander numbers every year (Table 2).

Numbers of redback salamander observations varied significantly by year ($F_{2,24} = 44.8$, $P < 0.001$). Numbers of redback salamanders increased from 388 the first year of study to 985 the second year, and then declined slightly the third year. Numbers of redback salamander observations by month (survey occasion) varied considerably, between a low of 16 salamanders in July 1997 to a high of 443 in October 1996. The significant effect of survey occasion ($F_{4,48}$

Table 2. Northern redback salamanders observed under cover boards by treatment, year, and month, Quabbin Reservation, central Massachusetts, 1995–97.

Year Month	Low deer density No thin $\bar{x}$	Low deer density No thin 95% CI	Low deer density Thin $\bar{x}$	Low deer density Thin 95% CI	High deer density No thin $\bar{x}$	High deer density No thin 95% CI	High deer density Thin $\bar{x}$	High deer density Thin 95% CI
1995								
May	5.6	0.7–15.1	4.7	3.0–6.7	4.2	2.7–6.0	5.6	3.1–8.8
Jun	3.6	0.9–8.2	6.1	3.6–9.3	6.5	3.1–11.2	4.0	1.6–7.6
Jul	2.6	1.0–4.9	4.5	1.9–8.3	4.9	1.8–9.7	6.0	2.6–10.8
Aug	4.1	0.3–12.5	6.5	2.2–13.2	9.0	2.6–19.2	5.4	2.1–10.3
Sep	1.1	<0.1–5.4	1.1	<0.1–4.3	1.2	<0.1–5.2	2.5	0.3–7.2
1996								
May	14.0	8.7–20.6	22.9	17.4–29.1	17.1	12.8–22.0	16.2	6.6–30.0
Jun	5.1	1.7–10.5	9.1	6.0–12.9	9.0	4.6–14.8	9.1	4.0–16.3
Jul	4.2	0.1–19.0	2.1	0.8–4.1	8.3	3.1–15.8	4.6	0.2–14.8
Aug	1.4	<0.1–5.9	2.2	1.0–3.8	0.1	<0.1–1.8	2.0	<0.1–7.9
Sep	17.1	12.9–22.0	31.6	24.6–39.5	32.1	14.5–56.7	27.1	12.0–48.3
1997								
May	20.7	17.9–23.7	24.1	18.7–30.2	22.7	13.1–35.0	22.9	11.0–39.2
Jun	7.2	0.7–20.5	12.4	5.3–22.6	9.1	3.7–16.7	6.9	3.0–12.5
Jul	0		0.7	<0.1–2.9	0.9	<0.1–3.7	1.2	<0.1–5.2
Aug	4.6	0.7–11.9	12.2	5.1–22.2	7.9	5.0–11.3	6.5	2.2–13.2
Oct	7.6	0.1–27.5	18.2	11.1–27.0	12.9	9.6–16.8	18.5	10.0–29.6

= 75.9, $P < 0.001$) was the single largest source of variation in redback salamander observations. The Pearson correlation between total redback salamander numbers and precipitation for the 3 days prior to the survey was 0.651 (Bonferroni-adjusted probability = 0.051). Generally, numbers of redback salamanders were highest in the spring and fall surveys and lowest in the summer surveys.

Organic soil horizon depths did not differ among treatments. Litter depths averaged 24–33 mm by treatment class (Table 3). Fragmentation depths averaged 32–37 mm, and humus depths averaged 14–46 mm among treatment classes. The greatest fragmentation and humus depths were recorded in 2 stands in the thinned, high deer density treatment that contained small wetlands.

Density and area of CWD were greatest in the thinned, low deer density treatment but not

Table 3. Organic soil horizon depths and coarse woody debris amounts by treatment, Quabbin Reservation, central Massachusetts, 1997.

Category Attribute	Low deer density No thin $\bar{x}$	Low deer density No thin SD	Low deer density Thin $\bar{x}$	Low deer density Thin SD	High deer density No thin $\bar{x}$	High deer density No thin SD	High deer density Thin $\bar{x}$	High deer density Thin SD
Organic soil horizon depths (mm)								
Litter	23.8	3.4	27.0	5.7	32.7	2.5	32.2	8.9
Fragmentation	35.9	6.6	36.1	5.2	32.2	2.8	37.3	7.1
Humus	24.3	12.3	15.4	9.6	14.3	8.2	46.1	40.4
Coarse woody debris counts (no./100 m^2) by decay class[a]								
1	0		0		0.55	0.971	0.25	0.5
2	0.05	0.1	0.35	0.342	0.3	0.258	0.1	0.115
3	3.05	0.943	3.35	1.29	3.55	1.159	2.2	0.365
4	3.25	0.998	5.15	1.05	3.95	0.79	4.55	1.482
All classes	6.4	1.681	8.85	1.32	7.95	1.886	7.2	2.085
Coarse woody debris area (m^2/100 m^2) by decay class[a]								
1	0		0		0.022	0.044	0.03	0.06
2	0.017	0.035	0.061	0.061	0.055	0.049	0.027	0.048
3	0.4	0.323	0.52	0.316	0.676	0.156	0.457	0.342
4	0.434	0.29	0.826	0.291	0.589	0.242	0.61	0.27
All classes	0.854	0.588	1.406	0.52	1.347	0.304	1.145	0.356

[a] See METHODS for definition of coarse woody debris decay classes.

statistically different from other treatment classes (Table 3). There was very little sound CWD (Classes 1 and 2), and generally equivalent amounts of CWD in the more decayed Classes 3 and 4.

The number of redback salamander observations differed among the 16 forest stands. Stands with high numbers of redback salamander observations occurred in all treatment classes, whereas 3 of the 4 stands with the lowest number of redback salamander observations occurred in the low deer density, unthinned treatment. Stands 11, 13, and 16 regularly had few redback salamander observations, while stands 1, 12, and 18 had high numbers of observations. At the stand level (n = 16), total redback salamander observations were positively correlated (Spearman's *rho*) with the average area of CWD ($r_S = 0.568$, $P = 0.01$–0.025), average count of CWD ($r_S = 0.479$, $P = 0.025$–0.05), and tall woody stem ($\geq$1 m tall) density ($r_S = 0.381$, $P = 0.05$–0.1).

DISCUSSION

The predominance of redback salamanders observed in this study was typical of terrestrial salamander surveys in the northeastern United States (Burton and Likens 1975*a*, Wyman 1988, Gibbs 1998), especially when artificial cover objects were used for the surveys (DeGraaf and Yamasaki 1992, Bonin and Bachand 1997). The terrestrial herpetofauna of New England is not very diverse, especially compared to southeastern regions of the United States (DeGraaf and Rudis 1983, Wyman 1998), so the dominance of a single species was expected.

The silvicultural treatment evaluated in this study was an intermediate treatment, not a regeneration harvest where a larger portion of the canopy would be removed. The effects of thinning on forest-floor fauna associated with closed-canopy forests would therefore be expected to be less severe than even-aged regeneration harvests and especially clearcutting, a practice that has been shown to have negative effects on some forest amphibians (Pough et al. 1987, Petranka et al. 1993, deMaynadier and Hunter 1995, Harpole and Haas 1999). Stands thinned to silvicultural guides (Hibbs and Bentley 1983, Johnson 1994) should have sufficient residual canopy trees to create shaded refugia for resident salamanders throughout the stand. In 1987, 4–12 years after thinning, basal area in the thinned stands of this study was an average of 66% of the basal area in the control, unthinned stands (Healy et al. 1987).

Reproduction from these survivors could replace any salamanders lost during or subsequent to the thinning (Pough et al. 1987). Alternatively, resident salamanders may have been able to survive within the patches where trees were removed by moving into deeper soil horizons or by tolerating the changed temperature and moisture conditions of the forest floor in gaps created by thinning canopy trees (Messere and Ducey 1998). It is unlikely that thinned stands were functioning as sink habitats (Griffis and Jaeger 1998) populated by emigrating mature salamanders from adjacent unthinned stands, as the distribution of redback salamanders by size class in the thinned stands was the same as in the control stands. Size class is a good surrogate measure of age class and sexual maturity (Sayler 1966). The equivalent abundance of immatures in the treated stands indicates that redback salamander reproduction was similar to that in the control stands.

The thinnings that were evaluated in this study are typical of many timber harvests (Smith 1986) and are a recommended intermediate treatment for oak in New England (Hibbs and Bentley 1983) and elsewhere throughout the range of northern red oak (Johnson 1994). The residual stands following the thinnings of this study were similar in structure to the reverse-J diameter distribution expected from uneven-aged or selection regeneration systems (Smith 1986). This structure suggests that selection harvests, while not recommended for stands dominated by northern red oak (Johnson 1994), would have a nonsignificant effect on redback salamanders (Pough et al. 1987, Messere and Ducey 1998), as observed following thinning in this study.

The lack of significant differences in the numbers of redback salamanders between thinned and control treatments found in this study, and the results of 2 other wildlife studies conducted in these stands, suggests this common silvicultural practice is compatible with the maintenance of faunal diversity. The effects of thinnings on vegetation structure and composition in southern New England oak stands were sufficiently minor as to have no significant effect on small mammal composition or the relative abundance of dominant small mammal species (Brooks and Healy 1988). Thinning increased the richness of the breeding-bird community by increasing the numbers of species active in understory vegetation (DeGraaf et al. 1991).

My failure to identify a lasting effect of chronic high numbers of white-tailed deer on redback salamander numbers differed from other studies, which found significantly lower numbers of red-backed voles (*Clethrionomys gapperi*) and short-tailed shrews (*Blarina brevicauda*) and canopy-gleaning birds in the high deer density stands (Brooks and Healy 1988, DeGraaf et al. 1991). Decreased bird populations have been observed in other locations with chronic high densities of white-tailed deer, which resulted in structural and compositional changes in mid- and understory vegetation due to overbrowsing (deCalesta 1997, McShea and Rappole 1997).

Residual effects of thinnings on understory vegetation structure and composition continued to be observed in 1994, >12 years after the stands were thinned, with greater densities of seedlings and saplings and greater forb cover in thinned stands. Likewise, residual effects of chronic high deer numbers were observed in lower numbers of seedlings and saplings, greater fern cover, and lesser shrub cover 3 years after deer densities were reduced by hunting. These residual effects in understory composition and structure had no apparent effect on redback salamander numbers.

No significant residual effects of thinning were observed on CWD density or area. If differences had occurred immediately following thinnings, natural recruitment of CWD has since removed any differences that may have existed between thinned and unthinned stands. Likewise, no differences were found in the thicknesses of soil organic horizons between treatment classes. Pough et al. (1987) demonstrated that depth of leaf litter is an important predictor of aboveground salamander activity. I do not know how litter depths changed immediately following the thinnings (Ash 1995), but if reduced, they had recovered by the time of this study.

The largest sources of variation in numbers of redback salamander observations were temporal effects. The dramatic increase in numbers between the first and second year of study may have been a result of aging of the boards, the length of time of board placement, or an increase in carrying capacity of the habitat caused by the addition of cover objects (Bonin and Bachand 1997). Additionally, precipitation during the spring months of 1995 was 3.9 (Mar), 3.8 (Apr), and 4.6 (May) cm below the 30-year normal for the area (National Oceanic and Atmospheric Administration 1995). Drought conditions could have resulted in redback salamanders retreating to lower soil depths and avoiding the drier forest floor (Heatwole 1962) and, consequently, the cover boards, resulting in the reduced numbers observed in the May and June surveys in 1995.

Differences in numbers of redback salamander observations also differed significantly among survey occasions. Numbers of redback salamander observations were generally higher in spring and fall surveys. Soil moisture in New England is typically high in the spring and fall and low in the summer (Hornbeck and Leak 1992), reflecting seasonal precipitation patterns, and could account for seasonal patterns in redback salamander numbers. Optimum temperatures for redback salamander surface activity also occur in the spring and fall (Burton and Likens 1975*a*). Some variation in redback salamander numbers among individual survey occasions was also due to precipitation during the week prior to the survey. Precipitation can temporarily increase moisture levels of the forest floor, allowing salamanders to be active on the ground surface and take refuge under cover boards (Jaeger 1980).

MANAGEMENT IMPLICATIONS

Differences in the composition and structure of residual vegetation, in the forest floor cover, and in the observations of redback salamanders leave the impression that a stand disturbance, where a large percentage of the canopy is retained and that results in an increase in cover of understory vegetation, would result in no long-term effect on redback salamanders. This effect was observed following thinning and following chronic overbrowsing by white-tailed deer. While not measured, it is likely that increased low herbaceous cover increased humidity and reduced temperatures at the ground level, which improved microhabitat quality for redback salamanders (Heatwole 1962, Pough et al. 1987). These findings demonstrate that the maintenance of sustainable redback salamander populations are compatible with timber harvesting if precautions are taken to minimize disturbance to the overstory canopy and to forest floor microhabitats.

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