Wildl. Soc. Bull. 20:260–264, 1992

A NONDESTRUCTIVE TECHNIQUE TO MONITOR THE RELATIVE ABUNDANCE OF TERRESTRIAL SALAMANDERS

RICHARD M. DEGRAAF, *U.S. Forest Service, Northeastern Forest Experiment Station, Holdsworth Hall, University of Massachusetts, Amherst, MA 01003*

MARIKO YAMASAKI, *U.S. Forest Service, Northeastern Forest Experiment Station, P.O. Box 640, Durham, NH 03824-9799*

Salamanders are abundant vertebrates in many forest ecosystems, and their annual biomass production can be important in forest food webs (Pough et al. 1987). Population densities of eastern redback salamanders (*Plethodon cinereus*) can exceed 2 individuals/m^2 in deciduous forests of the United States (Heatwole 1962, Jaeger 1980). Eastern redback salamanders were the most abundant terrestrial vertebrate in northern hardwood forests on the Hubbard Brook Experimental Forest in New Hampshire (Burton and Likens 1975*a*). Salamander biomass may equal that of small mammals and be twice that of birds (Burton and Likens 1975*b*). Annual production of eastern redback salamander biomass exceeds that of either small mammals or birds (Burton and Likens 1975*b*). In the southern Appalachians, the congeners *P. jordani* and *P. glutinosus* are the most abundant terrestrial salamanders (Hairston 1981). Terrestrial salamanders are usually inconspicuous, but they are important components of forest-floor detritus-based food webs, where they are consumers of small invertebrate prey (Pough 1983). Salamanders are prey to certain predators (e.g., snakes), but their toxic skin secretions may cause avoidance by others (e.g., small mammals) (Brodie et al. 1979).

Salamanders are rarely considered in forest planning, though they are abundant and important (Bury et al. 1980, Pough et al. 1987). Forest vegetation and structural features that affect the temperature, moisture, and acidity of the forest floor influence the microclimates and microhabitats available to salamanders (Heatwole 1962, Wyman and Hawksley-Lescault 1987, Wyman 1988).

In northern New England, eastern redback salamanders are more abundant in red maple (*Acer rubrum*) and northern hardwood than in balsam fir (*Abies balsamea*) stands (DeGraaf and Rudis 1990). We evaluated the use of boards (that may simulate logs) to monitor eastern redback salamanders in even-aged managed stands of New England northern hardwood forests.

STUDY AREA

We sampled salamanders on the Kilkenny Wildlife Management Area in the northern White Mountain National Forest (WMNF), New Hampshire (44°30'N, 71°20'W). The region is mostly forested; soils are typically stony, sandy, and acidic (Hoyle 1973). The present forest was regenerated following extensive logging and fires around the turn of the twentieth century (Belcher 1980:36–40). Northern hardwoods constituted most of the forest and cover about half the area. We used fully stocked hardwood stands containing American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), yellow birch (*Betula alleghaniensis*), red maple, paper birch (*B. papyrifera*), and an occasional red spruce (*Picea rubens*). Shrubs present were primarily striped maple (*A. pensylvanicum*), mountain maple (*A. spicatum*), and hobblebush viburnum (*Viburnum alnifolium*). Sapling and seedling stands contained the above species and pin cherry (*Prunus pensylvanica*). We selected stands for this study that were even-aged, at least 16 ha in size, and at elevations from 400–775 m. Sawtimber stands (at least half the stocking in trees >31 cm dbh) were 80–100 years old, poletimber stands (at least half the stocking in trees 10–30 cm dbh) were 40–50 years old, sapling (2.5–10 cm dbh) stands were 14–20 years old, and seedling (<2.5 cm in diameter) stands were 3–5 years old, as determined from WMNF compartment records.

METHODS

We surveyed salamanders along 3 replicate transects across each of 3 edge types (sawtimber-pole, sawtimber-sapling, and sawtimber-seedling). Transects were 270 m long and extended 135 m into the stand on each side of each edge type. We used 18 stands (3 replicates × 3 edge types × 2 stands/edge). At each of 7 points along each transect (135 m, 65 m, and 35 m from the edge of the stand to each side and at the interface between stands), we placed a cluster of 3 pine boards in contact with the litter in July 1988 and left them undisturbed for 12 months to allow the boards to weather (R. B. Bury, U.S. Fish and Wildl. Serv., Ft. Collins, Colo., pers. commun., 1989). Boards were 1 m long × 20 cm wide × 2.5 cm thick and were placed to simulate sections of fallen trunks and of limbs of trees. Our boards covered a total surface area of 37.8 m^2.

We lifted boards 8 times (all boards lifted the same day) during June–August 1989 and counted the exposed salamanders. Boards were carefully replaced after counting. Periodic counts of salamanders at sample points ($n = 63$) were tested for serial randomness using a mean square successive difference procedure (Zar 1974:305–306) on each set of square-root-transformed station observations. The goodness of fit of the set of observed data ($n = 504$ observations, 63 board clusters × 8 observations) to the Poisson distribution was tested by chi-square (Zar 1974:302–305). We also counted salamanders 3 times during August 1991 beneath 3 randomly chosen logs in each stand that had at least as much area as a board cluster and that could be turned without breaking.

We hypothesized that the number of salamanders at sample points along transects within edge type would be similar. Salamander counts were analyzed by chi-square procedures; expected values were obtained by apportioning the total counts along the 3 replicate transects within edge type equally among the 7 sample points.

The depth of undecomposed and decomposing litter (Oe, Oa, and Oi layers) (U.S. Dep. Agric. 1981) was measured at 2 randomly chosen points within 2 m of each cluster of 3 boards. We hypothesized that litter depths at sample points along transects within edge type would be similar. Litter depths were analyzed by ANOVA procedures; litter depth data met the assumptions of homogeneity of variances with Bartlett's test, and an analysis of the residuals confirmed an assumption of normality without data transformations.

RESULTS AND DISCUSSION

We counted 109 eastern redback salamanders and 1 northern dusky salamander (*Desmognathus f. fuscus*) under boards along transects. We counted an average of 0.21 (SE = 0.02) salamanders/board cluster. Counts ranged from 0–3 salamanders/cluster/observation (Table 1). The goodness of fit of the Poisson distribution to the set of observed data showed the observations to be distributed randomly ($\chi^2 = 1.004$, 2 df, $0.5 < P < 0.25$). Periodic counts of 44 of 63 board clusters (70%) were determined to be random observations. Sixteen of 63 clusters (25%) had 0 observations throughout the sampling period, and 3 of 63 clusters (5%) did not yield random observations at $P = 0.01$. It would seem that these observations can be considered independent for this type of relative abundance sampling as long as the sampling periods are sufficiently separated in time and all stations are sampled within a 24-hour period to minimize weather-related salamander movements.

Table 1. Frequency of occurrence of eastern redback salamanders under board clusters in Kilkenny Wildlife Management Area, White Mountain National Forest, New Hampshire, June–August 1989.[a]

Salamanders/ cluster (X_i)	Frequency	$P(X_i)$[b]	F_i[c]
0	395	0.816	412.001
1	88	0.168	84.996
2	9	0.017	8.968
3	1	0.001	0.644

[a] $n = 504$ cluster observations (7 stations × 9 replications × 8 counts).

[b] $P(X_i)$ = the probability of X_i salamanders/cluster if the salamanders are distributed randomly.

[c] $F_i = (P(X_i))(n)$; F_i is the expected frequency of X_i salamanders/board cluster.

No salamanders were found under boards in seedling stands, 11 were found in sapling stands, and 20 were found in poletimber stands; most were in sawtimber stands or edges of sawtimber (Table 2). Redback salamanders occurred in regenerated clearcuts that were 14–20 years old but lower numbers were detected ($P < 0.05$) than in adjacent sawtimber stands (interface values excluded). Numbers of salamanders observed in poletimber stands were not different from those in adjacent sawtimber stands (Table 2). Mean litter depths were not significantly different ($P < 0.14$) along transects between age classes (Table 2).

We deployed boards to sample terrestrial salamanders because we were unable to ob-

Table 2. Eastern redback salamanders in each edge type ($n = 3$) and mean litter depths (cm) by stand age and distance from edge ($n = 6$) in northern hardwoods, Kilkenny Wildlife Management Area, White Mountain National Forest, New Hampshire, June–August 1989.

	Distance from edge (m)							
	Sawtimber stand			Younger stand[a]				
Edge type	135	65	35	0	35	65	135	Total
Sawtimber-seedling	8	6	5	1	0	0	0	20
Litter depth	10.9	15.6	17.2	11.5	10.1	8.6	13.7	
Sawtimber-sapling	5	12	5	3	2	5	4	36[b]
Litter depth	5.9	12.9	11.5	17.1	11.7	9.4	8.6	
Sawtimber-poletimber	7	13	6	7	6	9	5	53
Litter depth	13.9	10.7	12.4	8.8	10.1	11.3	8.4	
Total salamanders	20	31	16	11	8	14	9	109[c]

[a] The younger stand (seedling, sapling, or poletimber) abutting sawtimber.

[b] Distribution is different ($P < 0.05$) than expected; $\chi^2 = 13.71$, 6 df.

[c] Distribution is different ($P < 0.01$) than expected; $\chi^2 = 24.51$, 6 df.

serve any salamanders active aboveground during 6 nights during or just after rain in July–August 1988 when all transects were walked using the method of Pough et al. (1987). Use of boards that simulate cover (e.g., fallen trees) proved to be an effective method to document the presence of terrestrial salamanders. Advantages are that counts can be made during daylight hours and that useful results can be obtained without the laborious installation of pitfall traps. Also, the use of boards does not degrade salamander habitat by turning or breaking existing logs or disrupting forest litter as in other techniques. Plethodontids did not use board clusters in very recent clearcuts. Dead and downed logs were present along all transects in all stands. Mean numbers of salamanders under logs (interface values excluded, all distances from edge values combined, and replications pooled among 3 seedling, 3 sapling, 3 poletimber, and 9 sawtimber stands) were correlated with mean numbers under boards by stand type ($n = 18$, $r = 0.788$, $P = 0.028$).

This technique may indicate the relative abundance of eastern redback salamanders in stands of various ages, but may not indicate the occurrence of other salamander species, especially those that breed in aquatic habitats. Pitfall traps and drift-fences in upland stands in the WMNF produced captures of 2-lined (*Eurycea bislineata*), spotted (*Ambystoma maculatum*), and spring salamanders (*Gyrinophilus porphyriticus*), and red efts of the eastern newt (*Notophthalmus viridescens*), although captures of these species were relatively few (DeGraaf and Rudis 1990). These species do not normally occur under surface-cover objects.

Our observations suggest that clearcutting in New England northern hardwoods, which contain ubiquitous and abundant eastern redback salamanders (Burton and Likens 1975*b*), may dramatically reduce their numbers. In New York stands with different disturbance histories, Pough et al. (1987) found few eastern redback salamanders and red efts in 7-year-old clearcuts, but populations in a 60-year-old second-growth forest were similar to those of adjacent old growth. In Virginia (Blymer and McGinnes 1977) and California (Bury 1983), salamander populations were low in recent clearcuts (<10 years).

The depth of the leaf litter was the best predictor of aboveground salamander activity in New York stands (Pough et al. 1987). The recovery of the litter layer to precutting levels apparently took 60 years in New York, and has been estimated to take 65 years in New Hampshire (Likens et al. 1978). Recovery of organic matter on the forest floor after clearcutting is likely influenced by tree species composition and soil moisture or site wetness. Recovery of eastern redback salamanders in New York oak (*Quercus* spp.) woods, which typically occupy dry sites, took 60 years (Pough et al. 1987), whereas in mesic northern hardwoods, recovery had occurred by 50 years. However, we found that litter depths were highly variable between edge types and station locations (Table 2). Differences between mean litter depths in sawtimber and treated stands (interface values excluded) were almost significant ($P < 0.058$).

Litter depth measurements can vary with slope (Leak 1974), microtopography, time of year, and site, as well as time of last major disturbance. Humus depths tend to reach a maximum depth in old-growth hardwoods (Leak 1974) but, at times, are less than humus depths in cutover and partially cut stands on similarly well-drained soils in New Hampshire (Hart 1961, Leak 1974). The variability in litter depths in our study was within the range of variability in old stands.

We did not encounter red efts in this study. Although red efts can tolerate drier conditions than eastern redback salamanders (Pough 1974), they were likely not present because required breeding ponds were not present in the vicinity of our sample stands.

Lungless salamanders (plethodontids) are the most abundant forest salamanders (Burton and Likens 1975*b*), and their surface foraging activities are regulated by forest moisture levels (Jaeger 1978, Feder 1983). Eastern redback salamanders and red efts can accommodate small-scale changes in habitat (e.g., firewood cutting, single-tree selection harvesting) with little change in populations or activity (Pough et al. 1987). Clearcutting, however, seems to reduce site moisture and interrupts the accumulation of organic litter, which are factors that limit the abilities of salamanders to forage successfully.

SUMMARY

Our study indicated that the use of boards to simulate fallen timber was a useful technique to evaluate changes in the abundance or surface activity of terrestrial salamanders that accompany forest management. Counts were made during daylight hours, and useful results were obtained without the laborious installation of pitfall traps. Use of boards did not degrade salamander habitat by turning or breaking existing logs or disrupting forest litter as in other techniques.

Acknowledgments.—We thank S. Staats and S. Plambeck for counting salamanders; F. H. Pough, W. M. Healy, R. B. Bury, and W. B. Leak for their reviews; 2 anonymous reviewers for their comments and suggestions; and M. A. Sheremeta for typing the manuscript.

LITERATURE CITED

BELCHER, C. F. 1980. Logging railroads of the White Mountains. Appalachian Mt. Club, Boston, Mass. 242pp.

BLYMER, M. J., AND B. S. MCGINNES. 1977. Observations on possible detrimental effects of clearcutting on terrestrial amphibians. Bull. Md. Herpetol. Soc. 13:79–83.

BRODIE, E. E., JR., R. T. NOWAK, AND W. R. HARVEY. 1979. The effectiveness of anti-predator secretions and behavior of selected salamanders against shrews. Copeia 1979:270–274.

BURTON, T. M., AND G. E. LIKENS. 1975*a*. Energy flow and nutrient cycling in salamander populations in the Hubbard Brook Experimental Forest, New Hampshire. Ecology 56:1,068–1,080.

———, AND ———. 1975*b*. Salamander populations and biomass in the Hubbard Brook Experimental Forest, New Hampshire. Copeia 1975:541–546.

BURY, R. B. 1983. Differences in amphibian populations in logged and old-growth redwood forests. Northwest Sci. 57:167–178.

———, H. W. CAMPBELL, AND N. J. SCOTT. 1980. Role and importance of nongame wildlife. Trans. North Am. Wildl. and Nat. Resour. Conf. 45:197–207.

DEGRAAF, R. M., AND D. D. RUDIS. 1990. Herpetofaunal species composition and relative abundance among three New England forest types. For. Ecol. and Manage. 32:155–165.

FEDER, M. E. 1983. Integrating the ecology and physiology of plethodontid salamanders. Herpetologica 39:291–310.

HAIRSTON, N. G. 1981. An experimental test of a guild: salamander competition. Ecology 62:65–72.

HART, G. 1961. Humus depths under cut and uncut northern hardwood forests. U.S. For. Serv. Res. Note NE-113. 4pp.

HEATWOLE, H. 1962. Environmental factors influencing local distribution and abundance of the salamander *Plethodon cinereus*. Ecology 43:460–472.

HOYLE, M. C. 1973. Nature and properties of some forest soils in the White Mountains of New Hampshire. U.S. For. Serv. Res. Pap. NE-260. 18pp.

JAEGER, R. G. 1978. Plant climbing by salamanders: periodic availability of plant-dwelling prey. Copeia 1978:686–691.

———. 1980. Microhabitats of a terrestrial forest salamander. Copeia 1980:265–268.

LEAK, W. B. 1974. Some effects of forest preservation. U.S. For. Serv. Res. Note NE-186. 4pp.

LIKENS, G. E., F. H. BORMANN, R. S. PIERCE, AND W. H. REINERS. 1978. Recovery of a deforested ecosystem. Science 199:492–496.

POUGH, F. H. 1974. Natural daily temperature acclimation of eastern red efts, *Notophthalmus v. viridescens* (Raffinesque) (Amphibia: Caudata). Comp. Biochem. Physiol. 47A:71–78.

———. 1983. Amphibians and reptiles as low-energy systems. Pages 141–188 *in* W. P. Aspey and S. I. Lustick, eds. Behavioral energetics: the cost of survival in vertebrates. Ohio State Univ. Press, Columbus.

———, E. M. SMITH, D. H. RHODES, AND A. COLLAZO. 1987. The abundance of salamanders in forest stands with different histories of disturbance. For. Ecol. and Manage. 20:1–9.

U.S. DEPARTMENT OF AGRICULTURE. 1981. Soil survey manual: 430-V-SSM. Soil Conserv. Serv., Washington, D.C. Unnumbered.

WYMAN, R. L. 1988. Soil acidity and moisture and the distribution of amphibians in five forests of south central New York. Copeia 1988:394–399.

———, AND D. S. HAWKSLEY-LESCAULT. 1987. Soil acidity affects distribution, behavior, and physiology of the salamander *Plethodon cinereus*. Ecology 68:1,819–1,827.

ZAR, J. H. 1974. Biostatistical analysis. Prentice-Hall, Englewood Cliffs, N.J. 620pp.

Received 19 December 1990.
Accepted 31 January 1992.
Associate Editor: Holler.

***Wildl. Soc. Bull.* 20:264–273, 1992**

WILDLIFE SPECIES RICHNESS IN SHELTERBELTS: TEST OF A HABITAT MODEL

RICHARD L. SCHROEDER, ***National Ecology Research Center, U.S. Fish and Wildlife Service, 4512 McMurry Avenue, Fort Collins, CO 80525-3400***

TED T. CABLE, ***Department of Forestry, Call Hall, Kansas State University, Manhattan, KS 66506***

SANDRA L. HAIRE, ***National Ecology Research Center, U.S. Fish and Wildlife Service, 4512 McMurry Avenue, Fort Collins, CO 80525-3400***

Shelterbelts are human-made habitats consisting of rows of shrubs and trees planted either in fields or on the windward side of farmstead dwellings. Shelterbelts provide wooded habitat for a large variety of birds and other wildlife. A model to predict wildlife species richness in shelterbelts (Schroeder 1986) was published as part of the U.S. Fish and Wildlife Service Habitat Suitability Index (HSI) model series (Schamberger et al. 1982). HSI models have been used extensively by wildlife managers and land use planners to assess habitat quality. Several HSI models have become the focus of a test program that includes analysis of field data for corroboration, refutation, or modification of model hypotheses. Previous tests of HSI models focused either on single species (e.g., Cook and Irwin 1985, Morton et al. 1989, Schroeder 1990) or examined portions of HSI models, such as the relationship between cavity abundance and tree diameter (Allen and Corn 1990). The shelterbelt model, however, assesses habitat value at the community level. The effects of habitat characteristics, area, and perimeter on diversity and abundance of bird and mammal species in