# Response of Herpetofauna to Silvicultural Prescriptions in the Daniel Boone National Forest, Kentucky

**Jennifer P. Adams,** *Department of Forestry, University of Kentucky, Lexington, KY 40546-0073*

**Michael J. Lacki,** *Department of Forestry, University of Kentucky, Lexington, KY 40546-0073*

**Michael D. Baker,** *Department of Forestry, University of Kentucky, Lexington, KY 40546-0073*

*Abstract:* We compared the response of herpetofaunal communities in 16 hardwood stands treated with a high-leave harvest (7 $m^2$/ha residual basal area), low-leave harvest (3.5 $m^2$/ha residual basal area), clearcut harvest or no-harvest prescription in the Daniel Boone National Forest, Kentucky, from 1992 to 1996. Animals were captured with straight-line drift fences and pitfall traps. We sampled 800 trap nights and captured 24 species of amphibians ($N$ = 1,363) and 12 species of reptiles ($N$ = 163). Diversity of amphibians was lower in low-leave harvest stands after removal of timber than in no-harvest stands ($P < 0.05$). Numerical abundance and species richness of reptiles were higher after timber removal in high-leave, low-leave, and clearcut harvest stands than in no-harvest stands ($P < 0.05$), and diversity of reptiles was higher after removal of timber in low-leave harvest stands than in no-harvest stands ($P < 0.05$). These data indicate a comparable response by herpetofaunal communities in harvested stands in the Daniel Boone National Forest, regardless of the amount of basal area harvested.

Proc. Annu. Conf. Southeast. Assoc. Fish and Wildl. Agencies 50:312-320

Deforestation and habitat fragmentation by humans are among the environmental influences suspected of altering the diversity of herpetofauna at local, landscape, and global scales (Hairston and Wiley 1993, Blaustein et al. 1994). More specifically, effects of timber harvesting can change forest habitats, creating a variety of stand conditions either favorable or unfavorable for amphibians and reptiles (Bennett et al. 1980, Pough et al. 1987). Timber harvesting in forests of eastern North America is believed to benefit (Christman et al. 1979, Campbell 1980), have no effect (Greenberg et al. 1994), or adversely impact populations of amphibians (Stiven and Bruce 1988, Petranka et al. 1993, Phelps and Lancia 1995), and to enhance (Christman et al. 1979, Enge and Marion 1986, Phelps and Lancia 1995) or modify species composition of reptiles (Greenberg et al. 1994). Data on the effects of timber harvesting on herpetofauna in southern Appalachian forests remain limited (Stiven and Bruce 1988, Petranka et al. 1993).

The United States Forest Service is now using 2-age harvests in place of clearcuts in eastern Kentucky. This method begins with a deferment cut, where some predetermined amount of basal area remains standing. The stand then grows for a complete rotation, around 80 to 100 years in Kentucky, where it is again treated with a deferment cut (Smith et al. 1989). With this approach a 2-age rather than an even-age stand develops. The implications of this type of harvest for herpetofaunal communities remain unclear because existing data on responses to timber harvesting in the southern Appalachians are from stands that were clearcut or treated with salvage logging (Ash 1988, Stiven and Bruce 1988, Petranka et al. 1993).

Clearcutting alters forest-floor microhabitats by reducing shade, leaf litter, and soil-surface moisture, and increasing soil-surface temperatures (Pough et al. 1987, Ash 1988, Welsh 1990). Given that a 2-age harvest leaves some trees standing, changes in forest floor microhabitats with a 2-age harvest may be reduced from that of a clearcut harvest resulting in less change in the local herpetofaunal community. In this study we compare the response of herpetofaunal communities among stands treated with 2-age harvests, clearcut harvests, or no-harvest prescriptions. We test the null hypothesis that timber harvesting does not alter herpetofaunal communities in Appalachian hardwood forests in the Cumberland Plateau, eastern Kentucky.

Funding for this study was provided by the Daniel Boone National Forest (DBNF) and the Department of Forestry, University of Kentucky. This investigation (No. 96-09-133) is connected with a project of the Kentucky Agricultural Experiment Station and is published with the approval of the director. Methods for use of animals were conducted under approval of the Animal Care and Use Committee, University of Kentucky, protocol No. 92-0014A.

## Methods

The study was conducted in the Morehead Ranger District, DBNF, in southeastern Bath County, Kentucky. The Morehead Ranger District is the northernmost district of the DBNF and is situated on the Pottsville Escarpment at the northwest edge of the Cumberland Plateau (Smalley 1986). The landscape of the region is rough topography with narrow ridgetops, steep slopes, and narrow, deep valleys. Vegetation of the area is characterized as transitional between mixed mesophytic forest to the east and oak-hickory forest to the west (Hinkle et al. 1993). Prior to harvest, the overstory of stands consisted of white oak (*Quercus alba*), chestnut oak (*Q. prinus*), hickories (*Carya* spp.), northern red oak (*Q. rubra*), yellow-poplar (*Liriodendron tulipifera*), scarlet oak (*Q. coccinea*), and black oak (*Q. velutina*).

Sixteen stands were divided evenly among 4 silvicultural prescriptions, including no-harvest, high-leave harvest (7 $m^2$/ha residual basal area), low-leave harvest (3.5 $m^2$/ha residual basal area), and clearcut harvest. Overstory plant species composition was comparable among stands before harvest and ranged in age from 73 to 113 years. Stands were 4.4 to 16.2 ha in size, and were assigned to silvicultural prescriptions evenly throughout the study area to avoid bias in landscape position. Stands were harvested between late June 1993 and early May 1994.

Amphibians and reptiles were captured with straight-line drift fences of aluminum flashing connected to 15-liter plastic containers. One array of 10 pitfall traps was placed in each stand. Traps were arranged in the shape of a "y," with a single container in the center connected to 5-m long arms of component fences; remaining containers were placed between and at the end of the sections of each fence. Containers were filled part-way with water and sealed with lids when not in use. Traps were opened from 15 to 24 September 1992 (harvested stands) and from 17 to 26 September 1993 (no-harvest stands) to compare herpetofaunal communities among stands prior to harvest. After harvest, traps were opened from 14 to 23 September 1994, 18 to 27 April 1995, 30 September to 9 October 1995, and 28 March to 6 April 1996. We collected animals daily by scooping traps with dipnets. Animals were placed in plastic bags, iced down, and returned to the laboratory for identification. Animals kept as a reference collection are preserved following protocol in Martof (1956).

We measured 8 habitat variables at each trap array between 8 and 19 May 1995. At the center container of each array, we visually estimated canopy closure (%), and measured the minimum distance to standing water (m), forest edge (m), and stream drainage (m). Maximum height of understory vegetation (cm) and maximum litter depth (cm) were measured on meter-squared plots located 7.5 m from the center container, on lines bisecting the angle between each arm of the trap arrays. We measured the maximum height of the shrub layer (m) and coverage of woody debris (%) within a 10 $m^2$-circular plot centered on these same plots.

Data for amphibians and reptiles were analyzed separately. The community-level indices evaluated were numerical abundance (number of animals), species richness (number of species), and species diversity (Shannon index, H′). Analysis of variance was performed on community indices to test for effect of silvicultural prescription. Data prior to harvest (1992 and 1993) and data after harvest (1994, 1995, and 1996) were tested separately. Analysis of variance was also performed on habitat variables, and on data of no-harvest stands from autumn sampling periods (1993, 1994, and 1995) to test for any temporal effect. Data for canopy closure and coverage of woody debris were arcsine transformed prior to analysis. Tukey's honestly significant difference, multiple comparison procedure was performed when ANOVAs were significant to identify the source of the difference. With all statistical tests on data of amphibians or reptiles, we assumed that capture rate of individuals within a species did not vary among habitats, permitting relative comparison of abundance of species among silvicultural prescriptions (Corn 1994).

## Results

We sampled 800 trap nights (i.e., trap = 1 pitfall array) and captured 24 species of amphibians and 12 species of reptiles, totaling 1,363 amphibians and 163 reptiles.

Overall capture success was 1.9 animals/trap/night. We captured 17 species of salamanders, 7 species of frogs and toads, 1 species of turtle, 4 species of lizards, and 7 species of snakes. The American toad (*Bufo americanus*) was the most frequently captured species, comprising 45.6% ($N$ = 622) of the amphibians captured, followed by wood frogs (*Rana sylvatica*) at 9.2% ($N$ = 125), and ravine salamanders (*Plethodon richmondi*) at 7.4% ($N$ = 101). All 3 of these species were captured in stands representative of all harvest prescriptions after timber removal. The longtail salamander (*Eurycea longicauda*, $N$ = 2) was the sole amphibian captured prior to harvest that was not recorded after harvest. The mud salamander (*Pseudotriton montanus*, $N$ = 1) and the southern leopard frog (*R. utricularia*, $N$ = 2) were the only amphibians not captured in stands in an uncut condition.

The fence lizard (*Sceloporus undulatus*) was the most frequently captured reptile, comprising 36.2% ($N$ = 59) of the reptiles captured, with ringneck snakes (*Diadophis punctatus*) second in capture frequency among reptiles at 14.7% ($N$ = 24). No species of reptile was captured solely in stands in an uncut condition; however, 6 species of reptiles were not captured until after harvest and were never recorded in no-harvest stands. These include five-lined skinks (*Eumeces fasciatus*, $N$ = 13), fence lizards, redbelly snakes (*Storeria occipitomaculata*, $N$ = 2), garter snakes (*Thamnophis sirtalis*, $N$ = 3), earth snakes (*Virginia valeriae*, $N$ = 3), and a milk snake (*Lampropeltis triangulum*).

No difference was observed prior to harvesting (1992 and 1993) among stands assigned to the 4 silvicultural prescriptions in numerical abundance, species richness, or species diversity of amphibians or reptiles ($P > 0.05$), indicating that stands were comparable in community indices prior to timber harvesting. No-harvest stands demonstrated temporal stability across years for all community measures except species richness of amphibians ($F = 4.3$, $P = 0.0488$), with the difference occurring between 1995 ($\bar{x}$ = 8.75 species/stand) and 1994 ($\bar{x}$ = 3.5 species/stand), 2 of the years post-harvest. Thus, for community measures of herpetofauna, the no-harvest stands provide a valid comparison for evaluating effects of timber harvesting.

After harvest, no difference was observed among silvicultural prescriptions in numerical abundance or species richness of amphibians ($P > 0.05$), but diversity of amphibians varied among silvicultural prescriptions ($F = 3.11$, $P = 0.0333$; Fig. 1); diversity of amphibians was lower in low-leave harvest stands than in no-harvest stands ($P < 0.05$). Differences among silvicultural prescriptions after harvest were observed for numerical abundance ($F = 5.03$, $P = 0.0037$), species richness ($F = 8.28$, $P = 0.001$) and species diversity ($F = 5.17$, $P = 0.0032$) of reptiles. Numerical abundance and species richness of reptiles were higher in high-leave, low-leave, and clearcut harvest stands than in no-harvest stands ($P < 0.05$), and diversity of reptiles was higher in low-leave harvest stands than in no-harvest stands ($P < 0.05$).

No difference was observed among silvicultural prescriptions for the following habitat variables measured after harvest, minimum distance to standing water, minimum distance to forest edge, minimum distance to stream drainage, maximum height of understory vegetation, maximum litter depth, and maximum shrub height ($P >$ 0.05; Table 1). As expected, percent canopy closure ($F = 31.9$, $P = 0.0001$) differed

**Table 1.** Habitat variables measured at trap arrays in forest stands after timber harvest in the Daniel Boone National Forest, Kentucky, 1995. Based on $N = 4$ stands/harvest prescription.

| | No-harvest | | High-leave | | Low-leave | | Clearcut | |
|---|---|---|---|---|---|---|---|---|
| Habitat variable | $\bar{x}$ | SE | $\bar{x}$ | SE | $\bar{x}$ | SE | $\bar{x}$ | SE |
| Canopy closure (%) | 86.2[a] | 2.4 | 41.2[b] | 5.9 | 12.5[b] | 2.5 | 19.2[b] | 11.7 |
| Distance to standing water (m) | 70.0 | 33.4 | 81.2 | 39.7 | 61.2 | 30.4 | 70.0 | 33.4 |
| Distance to edge (m) | 53.8 | 17.5 | 25.0 | 8.7 | 17.5 | 3.2 | 13.8 | 4.7 |
| Distance to drainage (m) | 5.2 | 4.9 | 15.0 | 6.1 | 13.8 | 6.2 | 10.0 | 5.8 |
| Height of understory (cm) | 28.4 | 6.9 | 32.1 | 3.6 | 51.6 | 14.7 | 45.8 | 8.7 |
| Litter depth (cm) | 5.8 | 0.7 | 8.8 | 4.0 | 4.8 | 1.0 | 10.8 | 3.9 |
| Height of shrub layer (m) | 3.4 | 1.3 | 1.6 | 0.3 | 2.2 | 0.5 | 2.0 | 0.3 |
| Coverage of woody debris (%) | 6.2[a] | 0.8 | 29.5[ab] | 9.0 | 38.2[b] | 5.9 | 37.5[b] | 7.8 |

[a,b]Means within rows without common letters are different ($P < 0.05$).

among silvicultural prescriptions, with closure higher in no-harvest stands than in high-leave, low-leave, and clearcut harvest stands. Coverage of woody debris was also different among silvicultural prescriptions ($F = 4.74$, $P = 0.021$), with coverage higher in low-leave and clearcut harvest stands than in no-harvest stands.

## Discussion

Timber harvesting is often advocated as a sound management practice on the basis of maximizing species diversity (Bury et al. 1980), but as Probst and Crow (1991) suggest, such practices may maximize species richness at local scales and benefit generalist species at the expense of habitat specialists. Petranka et al. (1993) strongly implicate timber harvesting as a causal mechanism for loss of salamander populations in the southern Appalachians because timber harvesting eliminates salamander populations within the harvested stand and salamanders require a long recovery period before they can reoccupy a harvested stand. With the exception of Ash (1988), however, no published study documents the immediate response to timber harvesting by amphibians or reptiles in the southern Appalachians. Our study, by sampling prior to and following timber harvesting, documents the immediate response by herpetofauna to various harvesting methods in a southern Appalachian hardwood forest, and has established a baseline for long-term investigation of the effects of timber harvesting on amphibians and reptiles.

Although all community indices of amphibians appear to show declines with timber harvesting in our study (Fig. 1), only the test of species diversity was significant, and then only between low-leave harvest stands and no-harvest stands. Further, we observed no disappearance of any species of amphibian with timber harvesting, except for the longtail salamander, a species captured too infrequently to draw any inference on possible response. Some authors suggest that amphibians in the southeastern United States are sufficiently rich in species and abundant in numbers to permit recovery from local disturbances, such as timber harvesting (Blaustein and

Wake 1990, Blaustein et al. 1994). Regardless, we urge caution in the application of any management practice resulting in habitat fragmentation. Blaustein et al. (1994) suggest that many species of amphibians exhibit nonequilibrium population fluctuations, rendering interpretation of numerical responses by amphibians over short time frames somewhat speculative.

A significant response of reptiles to timber harvesting in our study was evident in all community indices (Fig. 1), indicating that timber harvesting on the DBNF may enhance species richness and numerical abundance of reptiles. Increases in all community indices occurred regardless of the amount of basal area removed, except for species diversity which was higher only in low-leave harvest stands relative to no-harvest stands. Further, 6 species of reptiles were only recorded in stands after timber was harvested. These data are consistent with the response of reptiles recorded in other forest ecosystems impacted by timber harvesting (Christman et al. 1979, Enge and Marion 1986, Phelps and Lancia 1995), and constitute the first data available on response of reptiles to timber harvesting in southern Appalachian hardwood forests. Reptiles are more mobile than amphibians covering greater distances in migration and daily movements (Blaustein et al. 1994) and may benefit from disturbance-associated changes in habitat and microclimate (Greenberg et al. 1994). We suggest that timber harvesting on the DBNF serves to some extent as a substitute for natural disturbance events, such as fire and windthrow, now reduced in importance on this actively managed forest, that historically opened forest canopy and provided conditions favorable to many species of reptiles. Prescribed fire, however, remains a frequently used management practice on the southern half of the DBNF.

Canopy closure was reduced in all harvested stands relative to no-harvest stands (Table 1), reflecting measured declines in average basal area to 10.2 $m^2$/ha (high-leave), 4.4 $m^2$/ha (low-leave), and 0 $m^2$/ha (clearcut) for harvested stands, compared to an average basal area before harvest among all 16 stands of 30.4 $m^2$/ha (Raulerson 1996). We recorded higher coverage of woody debris in low-leave and clearcut harvest stands relative to no-harvest stands (Table 1). The presence of "slash" (e.g., woody debris) in clearcut harvest strands is hypothesized to enhance cover for reptiles and their prey in South Carolina swamp forests (Phelps and Lancia 1995) and may confer a similar advantage to species inhabiting harvested stands in the DBNF.

How the switch from clearcutting to 2-age harvest prescriptions on the DBNF will influence herpetofaunal populations in the long term remains unclear. Our data suggest that for herpetofaunal communities the immediate response is comparable among high-leave, low-leave, and clearcut harvest stands, with only subtle differences measured relative to no-harvest stands. Based on the premise that timber harvesting produced undesirable impacts, a few authors have suggested recovery periods of 50 to 70 years for populations of amphibians affected by timber harvesting in the eastern United States (Pough et al. 1987, Petranka et al. 1993). For at least the stands we examined, the DBNF allowed 83 to 113 years of recovery before reharvesting (Raulerson 1996), a length of time in excess of that recommended. We recommend that, until more data are available on herpetofaunal responses to timber harvesting, the DBNF take a conservative approach to selecting stands for harvest and ensure that rotation

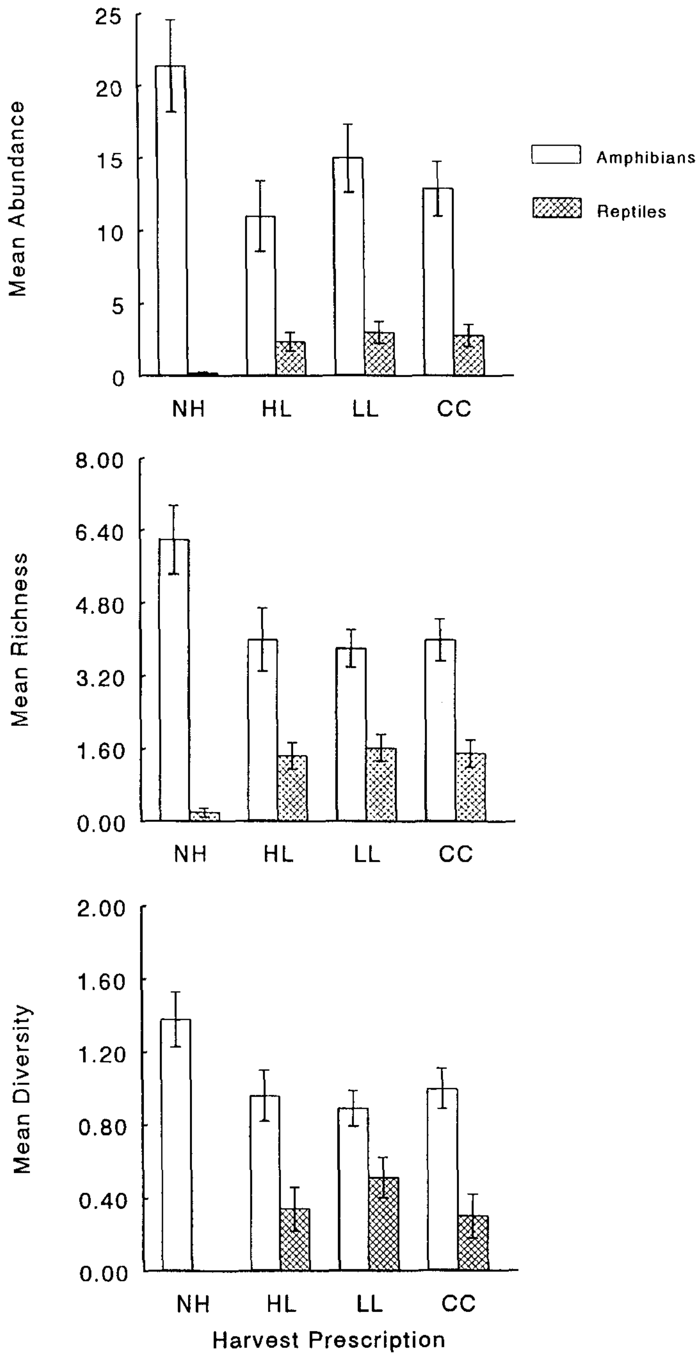


**Figure 1.** Mean abundance, species richness, and species diversity of amphibians and reptiles in no-harvest (NH), high-leave harvest (HL), low-leave harvest (LL), and clearcut harvest (CC) stands in the Daniel Boone National Forest, Kentucky. Bars represent 1 SE about the mean. Data are from post-harvest years, 1994 to 1996.

lengths (i.e., recovery periods) remain comparable to or longer than those of stands examined in our study.

## Literature Cited

Ash, A. 1988. Disappearance of salamanders from clearcut plots. J. Elisha Mitchell Sci. Soc. 104:116–122.

Bennett, S. H., J. W. Gibbons, and J. Glanville. 1980. Terrestrial activity, abundance, and diversity of amphibians in differently managed forest types. Amer. Midl. Nat. 103:412–416.

Blaustein, A. R. and D. B. Wake. 1990. Declining amphibian populations: a global phenomenon? Trends Ecol. and Evol. 5:203–204.

———, ———, and W. P. Sousa. 1994. Amphibian declines: judging stability, persistence, and susceptibility of populations to local and global extinctions. Conserv. Biol. 8:60–71.

Bury, R. B., H. W. Campbell, and N. J. Scott. 1980. Role and importance of nongame wildlife. Trans. North Am. Wildl. Nat. Resour. Conf. 45:197–207.

Campbell, H. W. 1980. St. Marks National Wildlife Refuge: forestry management and nongame wildlife. U.S. Dept. Int., U.S. Fish and Wildl. Serv., Gainesville, Fla. 163pp.

Christman, S. P., H. I. Kochman, H. W. Campbell, C. R. Smith, and W. S. Lippincott, Jr. 1979. Successional changes in community structures: amphibians and reptiles in Florida sand pine scrub. U.S. Dept. Int., U.S. Fish and Wildl. Serv., Gainesville, Fla. 21pp.

Corn, P. S. 1994. Straight-line drift fences and pitfall traps. Pages 109–117 *in* W. R. Heyer, M. A. Donnelly, R. W. McDiarmid, L. C.Hayek, and M. S. Foster, eds. Measuring and monitoring biological diversity: standard methods for amphibians. Smithsonian Inst. Press, Washington, D.C.

Enge, K. M. and W. Marion. 1986. Effects of clearcutting and site preparation on herpetofauna of a north Florida flatwoods. For. Ecol. and Manage. 14:177–192.

Greenberg, C. H., D. G. Neary, and L. D. Harris. 1994. Effect of high-intensity wildfire and silvicultural treatments on reptile communities in sand-pine scrub. Conserv. Biol. 8:1047–1057.

Hairston, N. G. and R. H. Wiley. 1993. No decline in salamander (Amphibia: Caudata) populations: a twenty year study in the southern Appalachians. Brimleyana 18:59–64.

Hinkle, C. R., W. C. McComb, J. M. Safley, and P. A. Schmalzer. 1993. Mixed mesophytic forests. Pages 203–253 *in* W. H. Martin, S. G. Boyce, and A. C. Echternacht, eds. Biodiversity of the southeastern United States: upland terrestrial communities. John Wiley and Sons, Inc., New York.

Martof, B. 1956. The amphibians and reptiles of Georgia. Univ. Ga. Press, Athens. 94pp.

Petranka, J. W., M. E. Eldridge, and K. E. Haley. 1993. Effects of timber harvesting on southern Appalachian salamanders. Conserv. Biol. 7:363–370.

Phelps, J. P. and R. A. Lancia. 1995. Effects of a clearcut on the herpetofauna of a South Carolina bottomland swamp. Brimleyana 22:31–45.

Pough, H. F., 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.

Probst, J. R. and T. R. Crow. 1991. Integrating biological diversity and resource management. J. For. 89:12–17.

Raulerson, G. E. 1996. Effects of silvicultural prescriptions on the quality and quantity of wildlife forage in the Daniel Boone National Forest, Kentucky. M.S. Thesis, Univ. Ky., Lexington. 103pp.

Smalley, G. W. 1986. Classification and evaluation of forest sites on the northern Cumberland Plateau. U.S. Dep. Agric., For. Serv. Gen. Tech. Rep. SO-60, New Orleans, La. 74pp.

Smith, H. C., N. I. Lamson, and G. W. Miller. 1989. An aesthetic alternative to clearcutting? J. For. 87:14–18.

Stiven, A. E., and R. C. Bruce. 1988. Ecological genetics of the salamander *Desmognathus quadramaculatus* from disturbed watersheds in the southern Appalachian Biosphere Reserve Cluster. Conserv. Biol. 2:194–205.

Welsh, H. H., Jr. 1990. Relictual amphibians and old-growth forests. Conserv. Biol. 4:309–319.