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EVALUATING THE EFFECTS OF ECOSYSTEM MANAGEMENT: A CASE STUDY IN A MISSOURI OZARK FOREST

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Abstract. Many federal and state management agencies have shifted from commodity-based management systems to multiple resource-based management systems that emphasize sustainable ecosystem management. Long-term sustainability of ecosystem functions and processes is at the core of ecosystem management, but a blueprint for assessing sustainability under different management strategies does not exist. Using the Missouri Ozark Forest Ecosystem Project (MOFEP) as a case study, we present one approach to evaluating the landscape-scale, short-term (one and two years posttreatment) consequences of even-aged and uneven-aged forest management treatments on community-level biological diversity. We chose changes in density of ecological species groups, representing groups of species with similar resource requirements, as our response variable. Changes in density are detectable before species completely disappear from an area, and these changes may be an early indicator of significant alterations to community structure and ecosystem function. Meta-analysis was used to statistically combine changes in densities across multiple species groups and assess the overall impacts of management treatments on the animal community. We also separately examined changes in density for each ecological species group. Our findings demonstrated that, in the short-term, even-aged and uneven-aged forest management treatments caused changes in animal community density in Missouri Ozark forests. Even-aged management sites showed greater changes than uneven-aged management sites after harvesting, and changes in species' densities were larger two years posttreatment (1998) than one year posttreatment (1997). Evaluation of treatment effects on individual ecological groups revealed that toads, forest interior birds, and edge/early successional birds were significantly affected by management treatments. We did not expect most species groups to exhibit treatment effects because relatively little forest biomass was removed per experimental site (only 10%), forest cover at the regional landscape level did not change and was generally high during the study, and the time scale was relatively short. The challenges facing ecosystem management evaluation parallel the challenges of ecological science in general: identifying appropriate variables, spatial and temporal scales, and experimental/management treatments. The integrative approach demonstrated in this paper is a first step toward the analysis of the effects of management treatments on multiple organisms within an ecosystem.

Key words: animal communities; community-level diversity; ecosystem management; even-aged vs. uneven-aged forest; forest management; meta-analysis; Missouri Ozark Forest Ecosystem Project; Missouri Ozarks; species groups.

INTRODUCTION

Maintenance of biological diversity and ecosystem productivity are primary concerns of both conservation biologists and resource managers in this period of "new forestry" and "ecosystem management" (Johnson 1997). Many federal and state management agencies have shifted from commodity-based management systems to multiple resource-based management systems that emphasize sustainable ecosystem management (Salwasser 1990, Swanson and Franklin 1992). In practice, however, many agencies do not have the necessary tools to implement the goals of sustainable ecosystem management or evaluate the extent to which goals are achieved (Christensen et al. 1996). Long-term sustainability of ecosystem functions and processes is at the core of ecosystem management (Grumbine 1994, Christensen et al. 1996, Franklin 1997), but a blueprint for assessing sustainability under different management strategies does not exist. Ecosystem sustainability

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is defined in terms of explicit goals that state "desired future trajectories or behaviors" for a specific ecosystem (Christensen et al. 1996). Evaluation is based on short-term and long-term monitoring to assess whether these goals are being reached.

An important aspect of ecosystem sustainability is maintaining viable populations of associated organisms (e.g., Noss 1990, Poiani et al. 2000). The overall complexity of an ecosystem is critical to its sustainability (e.g., Elton 1958, McNaughton 1993, Tilman 1996, 1999, Doak et al. 1998, Tilman et al. 1998), and the maintenance of biological diversity is an integral part of ecosystem complexity (Christensen et al. 1996). Biological diversity can be characterized at multiple levels of biological organization (e.g., gene, population, or community), and at multiple spatial and temporal scales; different levels of resolution are appropriate for different questions (e.g., Noss 1990, Hunter 1999, Poiani et al. 2000). For example, many management agencies monitor the stand-level effects of forest management treatments on population-level diversity by evaluating changes in abundance, distribution, or reproductive success of focal species at different time intervals (e.g., Yahner 1992, Petranka et al. 1994, Annand and Thompson 1997, Herbeck and Larsen 1999). The biological, spatial, and temporal resolution of these studies generates information about the direct effects of habitat alteration on individual species (e.g., Herbeck and Larson [1999] found that recently clear-cut forest stands supported few if any Plethodontid salamanders). However, these stand-level studies do not address questions about species' persistence across a landscape, the regional reproductive consequences of a highly fragmented landscape, or the indirect effects of changes in population density on species interactions. Thus, biological diversity consists of many components, and evaluation of each component depends on analyzing data of appropriate biological, spatial, and temporal resolutions.

A robust approach to evaluating the effects of ecosystem management protocols on biological diversity is to test experimentally the responses of known perturbations at spatial and temporal scales relevant to ecosystem processes (Carpenter 1998). The Missouri Department of Conservation is currently in the process of experimentally testing the landscape-scale effects of even-aged and uneven-aged forest management on a wide range of associated organisms. In this paper, we present one approach to evaluating the landscape-scale, short-term consequences of forest management treatments on community-level biological diversity, using the Missouri Ozark Forest Ecosystem Project (MOFEP) as a case study.

The primary goal of this paper is to determine if community-level animal diversity changed in MOFEP, relative to pretreatment levels in the years immediately following the first treatment application. We do not include the plant communities in these analyses because they were deliberately impacted in the management treatments. Coarse measures and indices of diversity (i.e., species diversity, evenness, species richness) were not informative and masked important differences among the nine study sites. Instead, we chose changes in density or relative abundance of ecological species groups, representing groups of species with similar resource requirements, as our response variable. Changes in density, particularly declines beyond the range of normal population variability (Poiani et al. 2000), are detectable before species completely disappear from an area, and these changes may be an early indication of approaching alterations to community structure and ecosystem function (e.g., Noss 1990, Christensen et al. 1996). Meta-analysis, a statistical approach that facilitates synthesis of results across a set of studies (Cooper and Hedges 1993, Gurevitch and Hedges 1993), is used as a quantitative method for statistically combining changes in densities across multiple species groups. To explore the treatment effects at a finer biological resolution, we also examine changes in density for each ecological species group separately. Specifically, we address three questions to evaluate treatment effects on the animal community as a whole, as well as for individual species groups in MOFEP: Was there a short-term effect of even-aged and uneven-aged management on animal communities? Did even-aged and uneven-aged management techniques exert different short-term effects on animal communities? Were even-aged and uneven-aged management effects different one year vs. two years after harvest? Finally, we discuss the challenges and limitations associated with analyzing data from multiple studies within an ecosystem project.

METHODS

Missouri Ozark Forest Ecosystem Project (MOFEP)

The Missouri Ozark Forest Ecosystem Project (MOFEP), administered by the Missouri Department of Conservation, is a multi-investigator landscape-level project encompassing nearly 20 independent studies of biotic and abiotic ecosystem components (Brookshire and Shifley 1997) in southeastern Missouri. The MOFEP study area includes experimental sites in the Current River and Peck Ranch Conservation Areas (Reynolds, Shannon, and Carter counties in southeastern Missouri, USA). Pre-1880, these forests were dominated by continuous *Pinus echinata* communities, but intensive harvesting (1880–1920) followed by repeated burning and grazing altered the landscape to produce mature upland oak–hickory and oak–pine communities (Cunningham and Hauser 1989). In the Ozarks, *Quercus alba* shares the canopy with other species of oaks, including *Q. stellata, Q. velutina, Q. coccinea,* and with *P. echinata,* and *Carya tomentosa* (Kurzejeski et al. 1993).

The primary goal of MOFEP is to experimentally

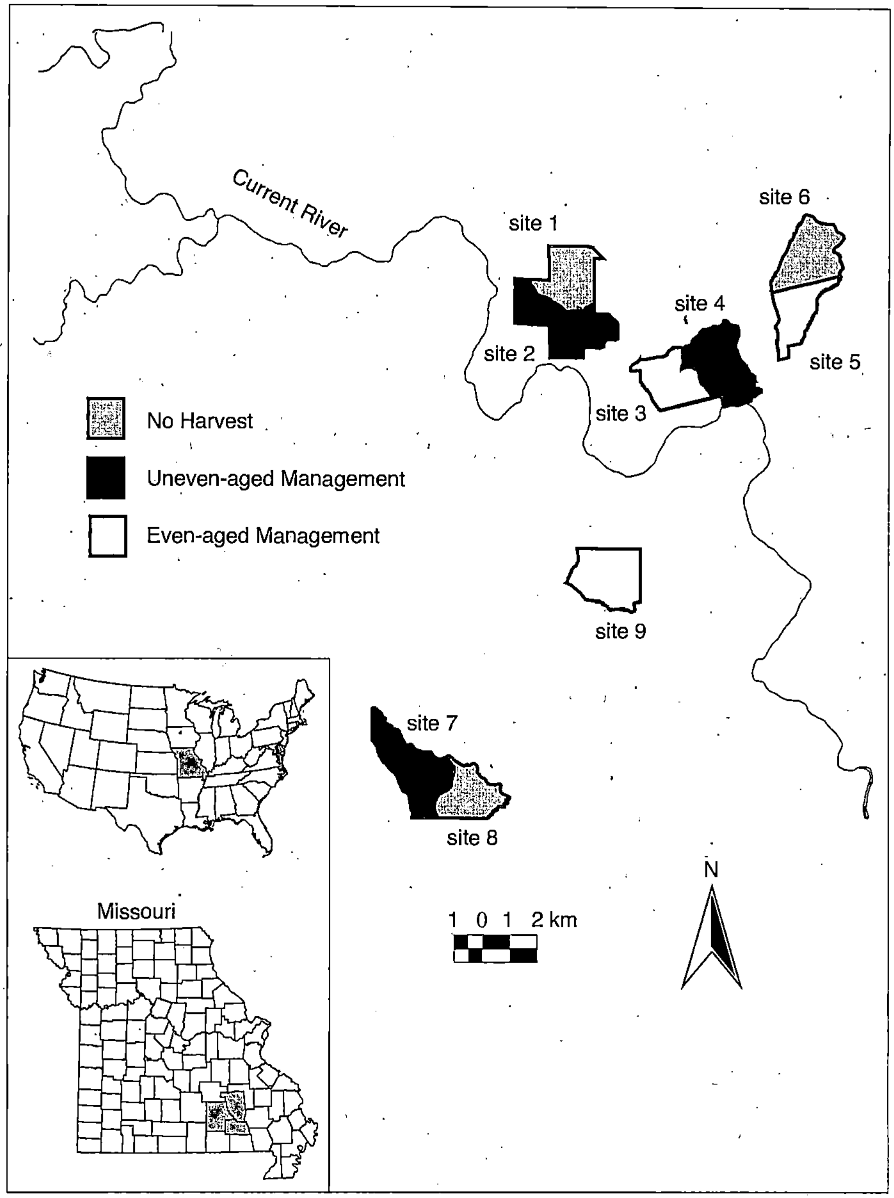

FIG. 1. Map of Missouri Ozark Forest Ecosystem Project (MOFEP) experimental sites 1–9 and location of the study area in southeastern Missouri, USA.

evaluate the effects of forest management on native animal and plant communities. The overall project was designed as a 100-yr experiment, with treatment/cutting intervals of 10 yr. The study area includes nine experimental sites, with sizes ranging 266–527 ha, located in a region that is 84% forested (Brookshire and Hauser 1993, Xu et al. 1997). Each experimental site was further divided into forestry stands that averaged five hectares in size. Experimental sites were assigned to three blocks based on subjectively determined physical similarity and then randomly assigned even-aged management treatment, uneven-aged management treatment, or no-harvest (control) treatment (Fig. 1). The result was a randomized complete block design with three sites per treatment and a total sample size of nine experimental sites (Sheriff and He 1997). Block 1 included sites 1–3, block 2 included sites 4–6, and block 3 included sites 7–9.

Treatments were designed to mimic realistic timber harvest practices administered by the Missouri Department of Conservation. The general management goal for treatment sites was to remove ~10% of forest biomass from even-aged and uneven-aged management experimental sites. No timber was removed from the no-harvest control sites. In even-aged and uneven-aged management sites, a contiguous track of forest representing 10% of the total area of each site was designated as old growth forest where no timber was removed. Treatments were applied to experimental sites by cutting selected forestry stands within each experimental site to achieve the desired amount of tree removal (i.e., 10% biomass). All animal density data, however, were collected at the experimental-site level, resulting in a landscape-level approach.

Even-aged sites were managed under a regime typical of Missouri Department of Conservation Forest Land Management Guidelines (1986). Ten to 12% of the remaining 90% of the forest was treated with clear-cutting and intermediate cutting (e.g, removal or girdling of single trees) in 1996 (Brookshire et al. 1997). Under Missouri Department of Conservation guidelines, clear-cuts were 3–12 ha in size, resulting in six to nine clear-cut stands per even-aged treatment site (Brookshire et al. 1997). A total of 78–110 ha were harvested on even-aged sites to achieve a 10% treatment level. The goal of even-aged management was to create a specific tree size class distribution in experimental sites: 10% in regeneration, 20% in small trees (trees 6–14 cm diameter at breast height [dbh]), 30% in poletimber (14–29 cm dbh), and 40% in sawtimber (≥29 cm dbh).

The uneven-aged management regime was a combination of small group openings and single-tree selection harvests. Small group openings ranged 21–43 m in diameter, depending upon aspect, and were designed to promote oak–hickory–short-leaf pine tree regeneration. Five percent of the study site was treated through small group openings (153–267 small group openings per uneven-aged site) during the 1996 harvest. These openings were scattered throughout the remaining 90% (i.e., excluding old-growth areas) of an uneven-aged management site. The harvest was completed with single-tree selection to obtain a balance of tree size classes (Law and Lorimer 1989) equal to the goals for even-aged management. A total of 203–348 ha were harvested on uneven-aged sites to achieve a 10% treatment level. Thus, uneven-aged and even-aged treatments resulted in the removal of similar amounts of biomass using different spatial configurations.

We collected pretreatment data during 1991–1995, the management treatments were implemented in 1996, and we then collected posttreatment data in 1997 and 1998. The data consisted of densities or relative abundances for amphibian and reptilian, bird, small mammal, and leaf-chewing insect species for each experimental site. Animal populations were sampled at the experimental-site level, not the forestry stand level. Thus, sampling plots for the animal studies were located throughout each experimental site, including forestry stands that were not harvested, and these data were analyzed at the experimental-site scale, resulting in a total sample size of nine for each animal group.

Amphibians and reptiles were sampled throughout each experimental site using 12 randomly placed trap arrays modified from Jones (1981), with six arrays on north- and east-facing slopes and six arrays on south- and west-facing slopes. These arrays included nine funnel traps and one central pitfall trap arranged along aluminum drift fences placed 120° apart. Arrays were open for sampling during August–October during 1992–1995 and 1997–1998. We checked arrays every three days, and all animals were marked and released after recording individual data (for details, see Renken [1997]). Relative abundance per study site was expressed as mean abundance per 100 trap days per year (1 trap day = 1 array open for 1 day).

Bird species' densities were determined using spot mapping (Robbins 1970). Each experimental site was divided into seven 45-ha spot-mapping plots that were each sampled 10 times (twice weekly) from mid-May through the end of June in 1991–1995 and 1997–1998. All birds detected (i.e., seen and heard) during each visit were recorded by location on topographic maps of the plot. We created composite maps per species and sampling plot each year to determine the total number of territories per site, which was then divided by total area sampled to determine a species' density per site per year (for details, see Clawson et al. [1997]).

Small mammals were sampled using Sherman small mammal live traps arranged in a 12 × 12 station grid, with 25 m between traps within the grid. Two grids were randomly placed on north- and east-facing slopes in each experimental site. Small mammals were sampled for six consecutive nights on each site during April or May of 1994, 1995, and 1998 (for details, see Fantz and Renken [1997]). We calculated relative abundance because captures and recaptures were too low to use population modeling density estimates. Relative abundance was defined as the number of individuals captured per site per 100 trap nights per year (1 trap night = 1 trap open for one night). Mammals were sampled less frequently than the other taxa, but because standard deviation is a component of our analyses, this factor does not negatively impact the analyses. At most, it is more difficult to find a significant change in relative abundance for mammals than for other groups with a smaller standard deviation in pretreatment abundance.

Leaf-chewing insects (e.g., Lepidoptera caterpillars) were sampled by searching the top and bottom of leaves, branches, and trunks near ground level (0.5 to 2.5 m) of approximately five trees of each of two species (*Q. alba* and *Q. velutina*) per sample plot. A minimum of 3000 *Q. alba* leaves and 1200 *Q. velutina* leaves were censused per plot. Six sample plots, stratified between north- and east-facing slopes and south- and west-facing slopes, were randomly selected in each experimental site and sampled in May of 1993–1995 and 1997–1998 (for details, see Marquis and Le Corff [1997]). We calculated mean insect density (number of insects per leaf area per tree) per site per year for use in these analyses.

Data analysis

We compiled results from the species under investigation and created ecological groups of similar spe-

TABLE 1. Species and characteristics of ecological animal groups analyzed in this study.

Species and characteristics of ecological group
Birds
Forest interior birds: neotropical migrant species that breed in forest interior habitat
Acadian Flycatcher (*Empidonax virescens*)
Kentucky Warbler (*Oporornis formosus*)
Ovenbird (*Seiurus aurocapillus*)
Wood Thrush (*Hylocichla mustelina*)
Worm-eating Warbler (*Helmitheros vermivorus*)
Edge/early-successional birds: resident and migrant species that breed in second-growth or forest edge habitats
Blue-winged Warbler (*Vermivora pinus*)
Eastern Bluebird (*Sialia sialis*)
Hooded Warbler (*Wilsonia citrina*)
Indigo Bunting (*Passerina cyanea*)
Mourning Dove (*Zenaida macroura*)
Northern Cardinal (*Cardinalis cardinalis*)
Prairie Warbler (*Dendroica discolor*)
White-eyed Vireo (*Vireo griseus*)
Yellow-breasted Chat (*Icteria virens*)
Mammals
Peromyscus species: mouse species that feed on seeds, fruit, and insects
Deer mouse (*Peromyscus maniculatus*)
White-footed mouse (*P. leucopus*)
Herpetofauna
Ambystoma salamanders: salamanders that breed in ponds
Marbled salamander (*Ambystoma opacum*)
Spotted salamander (*A. maculatum*)
Plethodon salamanders: salamanders that breed in forests
Slimy salamander (*Plethodon glutinosus g.*)
Southern redback salamander (*P. serratus*)
Toads: nocturnal toads that feed on insects and breed in shallow water
Eastern American (*Bufo americanus*)
Woodhouse's toad (*B. woodhousii*)
Skinks: medium-sized lizards that feed primarily on small invertebrates
Broadhead skink (*Eumeces laticeps*)
Five-lined skink (*E. fasciatus*)
Southern coal skink (*E. anthracinus pluvialis*)
Small snakes: small snakes that feed on invertebrates
Northern redbelly snake (*Storeria occipitomaculata o.*)
Prairie ringneck snake (*Diadophis punctatus arnyi*)
Western earth snake (*Virginia valeriae elegans*)
Black and white oak caterpillars (See the Appendix for species list)
Leaf-rolling caterpillars: leaf-chewing insects that feed within rolled leaves during April–May
Free-feeding caterpillars: externally feeding leaf-chewing insects that occur during April–May

cies (Table 1). Ecological groups were defined as species with similar resource requirements, such as habitat use or food acquisition (see Table 1). For species without known ecological characteristics, we relied on taxonomic classifications (e.g., salamander groups defined by breeding substrate) and observed habitat use (e.g., free-feeding caterpillars vs. leaf-rolling caterpillars). Our analyses were limited to ecological groups of species that were detected on all experimental sites during at least one phase of the experiment. We chose changes in density or relative abundance of ecological groups as the unit of study because we presumed that species within each group would respond similarly to the management treatments (Verner 1984, Szaro 1986, Block et al. 1995).

To evaluate the overall short-term effects of even-aged and uneven-aged management on multiple MOFEP animal communities, we conducted a meta-analysis (Hedges and Olkin 1985, Gurevitch and Hedges 1993). We calculated effect size by finding the standardized difference in mean density for each ecological group between control sites and either even-aged or uneven-aged treatment sites. Mean density/abundance difference was defined as the difference between mean pretreatment density/abundance (1991–1995) and posttreatment density/abundance in 1997 and 1998 separately. For example, we calculated the abundance of toads in each study site for each year. The pretreatment abundance was the mean abundance per site for 1992–1995. Repeated-measures analyses of density/abundance data revealed that all groups except oak caterpillars did not have significant year effects during the pretreatment phase of the experiment (Brookshire and Shifley 1997), justifying use of the pretreatment mean for further analyses. To find the difference in abundance between pretreatment and the first posttreatment year, we subtracted the 1997 toad abundance from mean pretreatment toad abundance for each site. Thus, yearly variation due to factors other than the management treatments was removed by using the pre- to posttreatment differences on control sites as the "zero" or "no effect" standard.

For each ecological group, we calculated an effect size of even-aged management and uneven-aged management, independently, for one year after treatment (1997) and two years after treatment (1998). Effect size (d_j) was defined as

$$d_j = \frac{M_T - M_C}{\mathrm{SD}_{TC}}$$

where M_C is the difference between mean density pretreatment and mean density in 1997 or 1998, for control groups; M_T is the difference between the mean density pretreatment and mean density in 1997 or 1998, for treatment groups; SD_{TC} is the pooled standard deviation of density differences for control and treatment groups; and j indexes the ecological group.

Effect size for a group had a variance (v) of

$$v = \frac{N_T + N_C}{N_T N_C} + \frac{d_j^2}{2(N_T + N_C)}$$

where N_T and N_C are the total number of sites in the treatment and control groups, respectively.

The cumulative effect size across any combination of groups was a weighted average of the group effects sizes and could be positive or negative, depending on whether M_C or M_T was larger. We calculated cumulative effect sizes with effect size direction and without direction (absolute value of d) to demonstrate the overall treatment effects when magnitude and direction of change were combined and when only magnitude of change was considered. Because we were interested in detecting any kind of change, and we expected the directionality of the density responses to management treatments to vary among ecological groups, cumulative effect sizes without direction are most relevant to this study.

Cumulative effect size (d_+) is defined as

$$d_+ = \frac{\sum_{j=1}^{k} w_j d_j}{\sum_{j=1}^{k} w_j}$$

where k is the number of groups in the treatment class, and $w_j = 1/v$. Cumulative effect size has a variance of

$$v_+ = \frac{1}{\sum_{j=1}^{k} w_j}.$$

We report effect size with direction and variance for all groups. We also show cumulative effect size, both with and without direction, and 95% confidence intervals for even-aged, uneven-aged, and both treatments combined one year after treatment and two years after treatment. Confidence intervals that do not overlap zero were considered significant at $P < 0.05$. We used the homogeneity statistic Q to evaluate the assumption of this fixed-effect model that all true effect sizes within a class were equal, as well as to partition the total heterogeneity (Q_T) into total within-class heterogeneity (Q_W), and between-class heterogeneity (Q_B) (Gurevitch and Hedges 1993). Thus, significant within-class heterogeneity would indicate that an assumption of the fixed-effect model that we used had not been met, and significant between-class heterogeneity would indicate that the responses of the treatment classes (i.e., even-aged and uneven-aged management) were different.

To test for effects of management treatments on each ecological group, we compared the pre- to posttreatment density/abundance difference on control sites to density/abundance difference on treatment sites using a randomized complete block analysis of variance (ANOVA) model. As in the meta-analysis, difference in pre- to posttreatment density/abundance per study site ($n = 9$) was the measurement unit. The main effects in the ANOVA model were block and treatment, with the block × treatment interaction used as the error term. All ANOVAs were performed with SAS version 6.12.

Results

Meta-analysis revealed significant cumulative effect sizes for even-aged and uneven-aged treatments in 1998 and for all treatments combined in 1997 and 1998 when effect size direction was not considered; cumulative effect sizes were not significant when direction of effect size was considered (Table 2). Effect sizes in 1998 were generally larger than effect sizes in 1997, particularly for the even-aged treatment. We did not find evidence for heterogeneity of effect sizes within the even-aged class in 1997 ($Q_{W(even)} = 4.50$, df = 10, $P > 0.90$) or 1998 ($Q_{W(even)} = 8.59$, df = 11, $P > 0.50$), or within the uneven-aged class in 1997 ($Q_{W(uneven)} = 3.48$, df = 10, $P > 0.95$) or 1998 ($Q_{W(uneven)} = 7.10$, df = 11, $P > 0.75$). Likewise, total heterogeneity (Q_T), total within-class heterogeneity (Q_W), and between-class heterogeneity (Q_B) were not significant in 1997 ($Q_T = 7.95$, df = 21, $P > 0.99$; $Q_W = 7.70$, df = 20, $P > 0.99$; $Q_B = 0.26$, df = 1, $P > 0.50$) or 1998 ($Q_T = 16.18$, df = 23, $P > 0.75$; $Q_W = 15.69$, df = 22, $P > 0.75$; $Q_B = 0.49$, df = 1, $P > 0.40$). The heterogeneity results were the same regardless of whether effect size direction was included or not included. Thus, effect sizes did not vary more than would be expected with random sampling, and the responses of even-aged and uneven-aged treatments were not significantly different.

We found significant treatment effects for edge/early successional birds in 1997 and 1998, and for toads and forest interior birds in 1998 (Table 3). All groups experienced changes in density or abundance after treatment, but for many groups (e.g., *Plethodon* salamanders, *Peromyscus* species, all caterpillar groups) populations in control sites (i.e., no harvest) increased or decreased in conjunction with treatment populations, resulting in no treatment effect (Fig. 2). For edge/early successional bird species, even-aged and uneven-aged treatments were different than controls; only even-aged treatments were different than controls for toads and forest interior birds (Dunnett's t test, $P < 0.05$).

Discussion

General treatment effects

In Missouri Ozark forests, animal community diversity, as represented by ecological species' group densities, showed an overall, short-term change in response to even-aged and uneven-aged forest management treatments. We found a trend towards greater changes in animal densities on even-aged management sites than uneven-aged management sites one year and two years after timber harvest, indicating that clearcuts may have affected the ecosystem in the short term more than smaller, scattered clearings. Our finding that changes in species' group densities were larger two years posttreatment (1998) than changes in density one year posttreatment (1997) suggests that treatment effects were not simply an immediate response to harvesting that quickly disappeared. In terms of sustainable ecosystem management, functionality or integrity of an area may best be judged by the extent to which species composition and population structure remain within their natural ranges of variability (Poiani et al. 2000). The short-term changes in animal densities that we observed do not indicate that ecosystem components or processes are changing at this time.

Ecological species groups responded variably to the management treatments (i.e., some groups increased in density, while other groups decreased in density after harvest treatments). Thus, when effect size direction (i.e., positive or negative) was factored into the meta-analysis, we did not find significant effects of even-aged or uneven-aged management on animal groups. The negative effect sizes statistically decreased the cumulative effect size, which was an average of ecological group effect sizes, and shifted the confidence intervals to the negative side of zero. Because the ecological group effect sizes are based on the difference between difference variables, negative and positive effect sizes do not necessarily indicate decreases and increases, respectively, in densities. Rather, effect size direction is dependent on both the direction of change from pre- to posttreatment on control sites relative to the change on treatment sites and the relative magnitudes of these changes. Thus, in this synthesis, we consider the magnitude of change more relevant than direction and suggest that the meta-analysis results without effect size direction are most appropriate.

Few landscape-level ecosystem studies simultaneously consider multiple species groups in their analyses of disturbance effects. Some studies have employed a landscape-modeling approach to predict the effects of proposed management actions on habitat availability and then, by linking species to habitats, predicted the effects of management on species (e.g., Hansen et al. 1993, Raphael et al. 1998). Based on species–habitat associations, Raphael et al. (1998) counted the number of terrestrial species that would likely be negatively impacted by proposed management strategies in the Interior Columbia Basin Ecosystem Management Project (Haynes et al. 1996). This approach was very useful for predicting the long-term effects of management plans that could result in different habitat configurations. When data from multiple species were available, most management research summaries described the effects of treatments on focal species or groups of species (e.g., Lautenschlager et al. 1998), generated lists of species that were restricted to particular habitats or managed areas (e.g., Carey 1989), or analyzed biological diversity properties such as species diversity indices or species richness (e.g., Scott et al. 1987, Margules et al. 1988, Hunter 1990, Hansen et al. 1991, Noss and Cooperrider 1994, Pickett et al. 1997). In the Missouri case study, species diversity and richness were not variable among control and treatment sites at any taxonomic scale. Coarse community measures, such as species diversity and richness, may not be sensitive enough to detect potential impacts of disturbance over a short time period or at the landscape-level, particularly when relatively little area is disturbed. Moreover, species diversity measures can be misleading because they do not indicate whether endemic species are replaced with introduced species. Thus, the population changes in preexisting species across a range of taxa provide valuable information about the impact of landscape-level management.

Treatment effects on separate ecological species groups

Evaluation of treatment effects on individual ecological groups provided useful insight into the meta-analysis results and the effects of management treatments from a finer biological resolution (i.e., species groups vs. animal community). Many ecological groups of species independently exhibited density/abundance changes on control and harvest treatment sites after treatment, but only toads, forest interior birds, and edge/early successional birds demonstrated a significant treatment effect. Toads declined less on even-aged sites than on control sites two years after timber harvest (Table 2, Fig. 2). A potential increase in food resources for toads following cutting in treatment sites may be responsible for inhibiting the natural decline seen in no-harvest compartments. Toads con-

TABLE 2. Differences pre- to posttreatment for control sites and treatment sites, standardized effect size, and variance of effect size for even-aged management and uneven-aged management treatments, when compared to controls, for ecological groups of species in 1997 (one year after harvest treatment) and 1998 (two years after harvest treatment).

	1997 (one year after treatment)			
Ecological group	Control pre–post	Treatment pre–post	Effect size	Variance or [95% CI]
Even-aged treatment				
Ambystoma salamanders	−1.623	−0.624	−0.769	0.716
Plethodon salamanders	−4.994	−6.355	0.217	0.671
Toads	5.245	3.317	0.583	0.695
Skinks	−0.197	−0.744	0.804	0.720
Small snakes	−1.269	−2.721	0.597	0.696
Peromyscus species				
Forest birds	21.968	31.184	−0.286	0.673
Edge birds	−0.359	−9.856	1.089	0.766
Free-feeding caterpillars, black oak	1.086	1.081	0.004	0.667
Leaf-rolling caterpillars, black oak	7.661	7.382	0.038	0.667
Free-feeding caterpillars, white oak	0.491	0.476	0.018	0.667
Leaf-rolling caterpillars, white oak	2.001	2.366	−0.139	0.668
Uneven-aged treatment				
Ambystoma salamanders	−1.623	−1.380	−0.143	0.668
Plethodon salamanders	−4.995	−6.814	0.275	0.673
Toads	5.245	5.258	−0.003	0.667
Skinks	−0.197	−1.301	0.949	0.742
Small snakes	−1.269	−2.941	0.637	0.700
Peromyscus species				
Forest birds	21.968	26.394	−0.154	0.669
Edge birds	−0.359	−7.299	1.074	0.763
Free-feeding caterpillars, black oak	1.086	0.810	0.300	0.674
Leaf-rolling caterpillars, black oak	7.661	6.175	0.232	0.671
Free-feeding caterpillars, white oak	0.491	0.487	0.004	0.667
Leaf-rolling caterpillars, white oak	2.001	2.250	−0.098	0.667
Summary without effect size direction (±)†				
Total even-aged			0.399	[−0.092–0.890]
Total uneven-aged			0.337	[−0.153–0.826]
Grand total			**0.368**	**[0.021–0.714]**
Summary with effect size direction (±)†				
Total even-aged			0.185	[−0.306–0.676]
Total uneven-aged			0.262	[−0.227–0.752]
Grand total			0.223	[−0.123–0.570]

† Summaries for management classes and years represent cumulative effect sizes, which are weighted averages of individual effect sizes, both with effect size direction (±) and without direction (i.e., absolute value of effect size). Effect sizes for summaries with confidence intervals (CI) that do not overlap zero are significant at $P < 0.05$ and noted in bold print.

sume many insects (Johnson 1997), and Harper and Guynn (1999) have noted that invertebrate density increased in clear-cuts following timber harvest. Forest interior birds declined further on treatment sites than on control sites two years after harvest. This result suggests that the loss of habitat discouraged individuals from establishing territories in the vicinity of forest openings, which decreased the density of these species in harvested sites. Conversely, edge/early successional birds increased both one and two years after timber harvest on both even-aged and uneven-aged sites. These species typically invade recently cleared areas. Stand-level studies have shown that densities of these species will peak in harvested stands two to four years after harvest and then start to decline as the forest canopy closes (e.g., Crawford et al. 1981, Thompson et al. 1992, Annand and Thompson 1997, Robinson and Robinson 1999).

Many groups of species would not be expected to exhibit treatment effects because relatively little forest biomass was removed per experimental site (only 10%), the regional landscape (i.e., counties in southeastern Missouri that encompass the Missouri Ozark Forest Ecosystem Project [MOFEP] study area) remained mostly forested, and the time scale was relatively short. With minimal habitat loss, individuals in undesirable areas may move to nearby areas that remain forested. Studies of population persistence after habitat loss suggest that amount of edge habitat and fragmentation in the region were important determinants of animal population responses (Thompson et al. 1992, Robinson et al. 1995, Donovan et al. 1997, Gibbs 1998, Tewksbury et al. 1998). For example, nest predation and cowbird parasitism, the leading causes of population decline in many forest-breeding bird species, increased with forest fragmentation at the landscape level

TABLE 2. Extended.

1998 (two years after treatment)			
Control pre–post	Treatment pre–post	Effect size	Variance or [95% CI]
−0.3401	0.137	−0.916	0.737
−3.543	−3.880	0.067	0.667
6.065	3.627	0.617	0.699
0.081	−0.330	1.085	0.765
−0.241	0.599	−0.870	0.730
2.420	0.550	1.604	0.881
29.295	39.467	−0.261	0.672
0.110	−20.081	1.134	0.774
0.916	0.806	0.127	0.668
6.176	7.049	−0.122	0.668
0.377	0.170	0.256	0.672
1.177	1.514	−0.154	0.668
−0.340	0.139	−0.918	0.737
−3.543	−0.233	−0.896	0.734
6.065	5.561	0.093	0.667
0.081	−0.791	1.179	0.782
−0.241	0.558	−0.844	0.726
2.420	1.331	0.688	0.706
29.295	31.877	−0.078	0.667
0.110	−15.043	1.134	0.773
0.916	0.771	0.172	0.669
6.176	5.142	0.180	0.669
0.377	0.185	0.237	0.671
1.177	1.436	−0.121	0.668
		0.563	**[0.086–1.041]**
		0.521	**[0.047–0.996]**
		0.542	**[0.206–0.879]**
		0.179	[−0.298–0.657]
		0.061	[−0.414–0.535]
		0.120	[−0.217–0.456]

(Robinson et al. 1995). Moreover, Donovan et al. (1997) demonstrated that edge effects in Midwestern forests depend on landscape context, with increased landscape-scale forest cover leading to less severe edge effects. In the Missouri Ozarks, the region is >80% forested, and edge habitats consist mostly of naturally regenerating forest bordering mature forest, suggesting that edge effects on MOFEP sites are probably not severe. The relatively short time scale of these results also favors a lack of treatment effects because population densities may not have responded to habitat changes.

Potential limitations

A puzzling complication in our experiment is the general decline or increase in densities across all species groups and treatments after timber harvest in 1996. Variability among pretreatment years was minimal compared to the changes observed posttreatment (see Brookshire and Shifley 1997). The results of this study clearly demonstrate the importance of sampling control sites over the same time period as treatment sites. If we had simply compared pretreatment densities to posttreatment densities for each species group, we would have concluded that both even-aged and uneven-aged management exerted large treatment effects (Fig. 2). The control sites, however, showed similar changes in densities after 1996. This finding suggests that some of the changes observed on treatment sites may not have been the result of the management treatments, but rather a regional environmental event. In 1997, the MOFEP region experienced El Niño weather patterns and a late frost in May, and, in 1998, the simultaneous emergence of two species of periodical cicadas. With data from control sites, we eliminated the effects of these potentially confounding factors from the statistical analyses, which explains why we found very small effect sizes despite visibly large changes in densities of most species groups on MOFEP sites. Ecologically,

TABLE 3. *F* values and probabilities for treatment main effects tested with analysis of variance (ANOVA) for ecological groups of species in 1997 (one year after harvest treatment) and 1988 (two years after harvest treatment).

Ecological group	1997 (one year after treatment)		1998 (two years after treatment)	
	$F_{(2,4)}$	P	$F_{(2,4)}$	P
Ambystoma salamanders	0.58	0.603	1.94	0.268
Plethodon salamanders	0.09	0.917	0.54	0.620
Toads	4.83	0.086	12.97	0.018
Skinks	1.30	0.368	1.04	0.434
Small snakes	0.26	0.786	0.50	0.642
Peromyscus species	…	…	4.15	0.106
Forest interior birds	0.89	0.480	30.40	0.004
Edge/early successional birds	21.33	0.007	33.79	0.003
Free-feeding caterpillars, black oak	0.44	0.673	0.12	0.894
Leaf-rolling caterpillars, black oak	0.47	0.654	0.63	0.577
Free-feeding caterpillars, white oak	0.00	1.000	0.27	0.776
Leaf-rolling caterpillars, white oak	0.12	0.889	0.08	0.924

Note: Analyses were performed on pre- to posttreatment difference variables (see *Methods*).

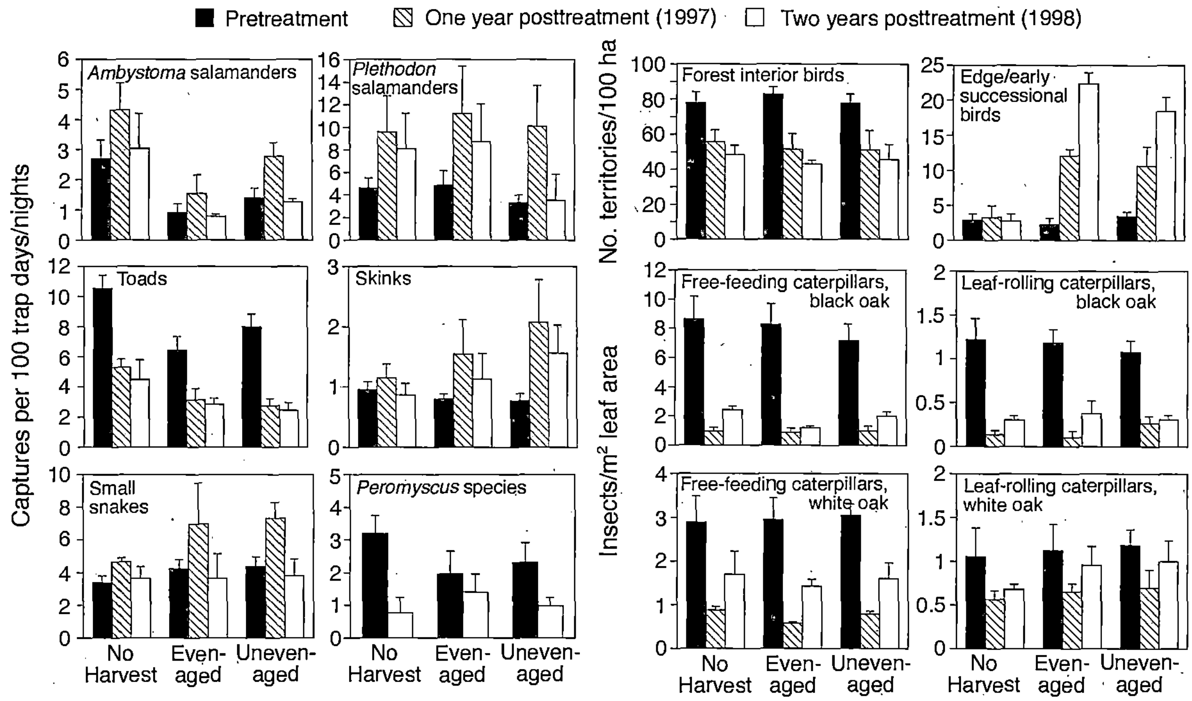

FIG. 2. Densities or relative abundance (± 1 SE) for ecological species groups on no-harvest, even-aged, and uneven-aged management sites pretreatment (1992–1995), one year after treatment (1997), and two years after treatment (1998) in the MOFEP experimental sites.

however, we acknowledge that this event complicates interpretation of the results. The meta-analysis results reveal a general trend towards landscape effects of the management treatments on animal densities, but the absolute magnitude of these changes is unclear given the regional changes.

Another possible explanation for the widespread density changes after harvest is that the treatments impacted an area larger than the target experimental sites. If so, treatments may have affected no-harvest control sites that were located adjacent to treatment sites. We find this interpretation unlikely because many MOFEP studies, including those focused on organisms with little mobility, found dramatic differences between their pretreatment and posttreatment data thousands of meters from any timber harvest in no-harvest control sites (MOFEP investigators, *personal communication*). While landscape-level studies need to be concerned that the scale of the treatments does not exceed the scale of the measurements, results from MOFEP studies do not indicate that this problem is of major concern for our study.

Based on the changes in animal densities observed in our study and pretreatment density patterns already reported (Brookshire and Shifley 1997), we conclude that most changes were within the range of normal population variability and that the management options examined in this study probably have only localized effects. Thus, the short-term response does not suggest that these treatments threaten ecosystem functionality. If the magnitude of change moves beyond the range of pretreatment variability, or if the management practices cause changes at a much larger scale than anticipated (e.g., across the entire MOFEP landscape instead of within study sites), then we would conclude that the treatments substantially affect resident animal populations. Ongoing research will provide further insights to these larger questions.

Finally, we comment on the use of meta-analysis as a quantitative method for evaluating overall treatment effects. Meta-analysis is an attractive method for synthesizing the effects of management on multiple groups of organisms, which is necessary as managers evaluate the overall effects of disturbance on an ecosystem. An assumption of meta-analysis is independence of individual studies (Hedges and Olkin 1985, Gurevitch and Hedges 1993, 1999). In the MOFEP case study, the species groups are not independent because they are spatially correlated. We assumed that the species groups were responding independently at this point in the experiment, given that we were only analyzing short-term data. The potential lack of independence implies that the confidence intervals may be larger than reported in the results, but the general conclusions remain the same. At this time, meta-analysis seems to be a practical option for detecting overall trends across multiple species groups. Other approaches that can synthesize data across multiple scales and levels of organization need to be developed. For example, we encountered problems when working with a variety of organisms that probably view the "landscape" very differently (e.g., a bird species' territory may cover two hectares of forest, whereas a leaf-chewing insect may remain on the same branch of a single tree for most of its life). We chose to evaluate treatment effects at one specific spatial scale, but other approaches may define landscape-scale from the point of view of the focal organisms. Furthermore, evaluating management effects at one spatial scale (e.g., experimental site in MOFEP) may hide changes at a different spatial scale (e.g., habitat types within experimental sites). R. B. Renken, W. K. Gram, S. C. Richter, D. K. Fantz, K. Ricke, and T. Miller (*unpublished manuscript*) found that some amphibians were impacted differently on north- and east-facing slopes than on south- and west-facing slopes in MOFEP, and R. Marquis (*unpublished manuscript*) found that leaf-chewing insects in tree canopies responded differently to management treatments than insects near ground level in other MOFEP studies. The same issues arise when considering the most appropriate temporal scale to evaluate management treatments. Ultimately, effective evaluation will include examining patterns at a variety of spatial and temporal scales using a variety of approaches.

Conclusions

We conclude that even-aged and uneven-aged management influence animal community densities in the Missouri Ozark landscape, but we are uncertain about the absolute magnitude of change and how long these impacts will persist. Clearly, individual species groups are responding differentially to the management treatments and continued monitoring at the population level is critical to determining the ecological processes that cause community-level changes. We suspect that short-term impacts will produce further changes in the animal communities and other parts of the ecosystem through indirect effects and species interactions. Time and further monitoring will tell us how these changes manifest themselves in terms of overall ecosystem sustainability.

The challenges facing ecosystem management evaluation parallel the challenges of ecological science in general (Christensen et al. 1996): identifying appropriate variables, spatial and temporal scales, and experimental/management treatments. We present one approach for integrating the effects of management treatments on multiple organisms within an ecosystem; many more alternatives are needed. As ecosystem management experiments are independently implemented in different ecosystems and regions, meta-analysis will become more useful and robust. Our current knowledge about ecosystem function and sustainability is limited, and we are only beginning to evaluate management from an ecosystem perspective. Acknowledging this uncertainty and integrating what we learn into future management efforts (i.e., adaptive management: Holling 1978, Walters 1986, Walters and Holling 1990) will ensure progress in the practice and science of ecosystem management.

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APPENDIX

The species listing for caterpillars is available in ESA's Electronic Data Archive: *Ecological Archives* A011-020.