THESIS

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FIRST-YEAR RESPONSES OF WILDLIFE AND WILD-
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IN A NORTHERN HARDWOODS FOREST

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Anne Husted Thomas

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Jonathan B. Haufler
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FIRST-YEAR RESPONSES OF WILDLIFE AND WILDLIFE HABITAT TO SEWAGE SLUDGE APPLICATION IN A NORTHERN HARDWOODS FOREST

by

Anne Husted Thomas

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ABSTRACT

FIRST-YEAR RESPONSES OF WILDLIFE AND WILDLIFE HABITAT TO SEWAGE SLUDGE APPLICATION IN A NORTHERN HARDWOODS FOREST

By

Anne Husted Thomas

A single application of sewage sludge was applied to a 50-year old northern hardwoods forest in Montmorency County, Michigan, in July 1982. The application technique required construction of 5-m wide trails at 20 m intervals throughout the forest. Vegetative community composition and structure were analyzed, small mammal communities monitored via live-trapping, and red-backed salamanders censused on 9 study plots (3 with trails, 3 with sludge application and trails, and 3 controls).

First year results indicated that tree seedling densities were lowered in trails by trail construction and use, and in forest interiors by sludge damage. Vertical vegetative cover was reduced in tall strata by tree removal for trail construction, and in low strata by sludge damage. Trails appeared to provide suboptimal salamander above-ground foraging habitat; but did not affect small mammal habitat use. *Peromyscus* mice increased in number on sludge-treated plots, compared to trail-only and control plots, presumably due either to differential survival or to a behavioral response to some change in their environment.

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TABLE OF CONTENTS

LIST OF TABLES

Number		Page

LIST OF FIGURES

INTRODUCTION

Sewage waste disposal poses a growing problem for many municipalities. Technological advances can only partially solve these problems by modifying processes to reduce the amount of waste generated and to make it less harmful to humans and to the environment. Advanced secondary sewage treatment uses filtration and aerobic or anaerobic bacterial digestion to break down organic material to its inorganic constituents. The process destroys or immobilizes most toxins, pathogens and parasites (Sopper and Kardos 1979). Much water can be reclaimed from sewage, but the remaining solids must be dealt with. Final sewage solids, called sludge, can be disposed of by: (1) discharge into rivers, lakes, or oceans, (2) incineration, (3) landfill, or (4) land application. Federal clean water criteria dictate against sludge disposal in any manner that might affect the inland or coastal navigable waters of the United States (Schmid et al. 1975). Offshore ocean dumping is undesirable, since sludge constituents reduce seabed oxygen and in other ways affect benthic organisms (National Academy of Sciences 1978). Incineration leaves noncombustible material which must be disposed of, wastes raw materials, and creates air pollution (Turk et al. 1972, National Academy of Sciences 1978). Landfill, if properly done, can be a safe sludge disposal method (National Academy of Sciences 1978). As with incineration, however, landfill wastes raw materials. In addition, safe landfill sites, with no threat of groundwater contamination, are scarce. Burial necessitates removing existing vegetation and stockpiling an amount of excavated material equal to the volume of sludge to be buried. Stockpile sites also will be temporarily devoid of vegetation (Schmid et al. 1975). Thus, available sites meeting all environmental and economic criteria for landfill are quite limited.

Sludge disposal by land application is the remaining alternative. Terrestrial ecosystems can act as living filters for the compounds and elements in sludge (Sopper 1975), while sludge-borne nutrients can promote plant productivity. Sludge or sewage effluent has been applied to agricultural lands, often resulting in increased yield and higher nutrient content of crop plants (Sopper and Kardos 1979). However, for human health and aesthetic reasons, and because they are less effective in retaining the mineral elements applied (Woodwell 1977), agricultural systems are not always the ideal candidates for sewage sludge disposal.

Forests, in contrast, are generally considered stable, nutrient-conservative ecosystems with greater potential to act as living filters of mineral enriched discharges (Sopper 1975, Woodwell 1977). Forests are attractive as potential sewage disposal sites because of their ecosystem longevity, stability, and their tendency to be nitrogen-limited (Wollum and Davey 1975, Bormann and Likens 1979). Sewage effluent or sludge applied to aggrading northern hardwood forests has been shown to achieve significant nutrient removal, primarily by the ecosystem's plant communities (Woodwell 1977, Sopper and Kardos 1979). Sludge fertilization can result in increased forest productivity, particularly by trees (Saffort 1973, Weetman and Hill 1973, Woodwell 1977, Sopper and Kardos 1979). However, fertilizer effects on forest plant and animal community composition in terms of numbers of individuals, dominance, and structure can alter the integrity and stability of the ecosystem (Weetman and Hill 1973), thus pointing to the need for ecosystem study and monitoring.

Past studies of forest fertilization effects have concentrated on tree production, and usually have dealt with tree plantations and coniferous forests. In the deciduous forests studied, responses of tree growth have ranged from negligible growth increases in 50-70 year old stands (Koterba et al. 1979, Sopper and Kardos 1979, Stone et al. 1982) to threefold growth increases compared to controls in a younger stand (Sopper and Kardos 1979).

There is some evidence that sludge or effluent fertilization initiates plant community composition and structure changes, with a general acceleration in canopy closure, an increase in leaf area index and in herbaceous cover, a decrease in midstory cover, increased dominance by shade tolerant understory species, and a decline in plant species diversity (Anthony and Wood 1979, Sopper and Kardos 1979).

The sewage sludge application process for a mature forest may involve clearing narrow (5m wide) application trails at intervals throughout the forest in order to allow access by a spray vehicle. This procedure would produce a unique forest thinning effect. The influence of this particular thinning pattern on forest communities has not been investigated, either alone or in conjunction with sludge fertilization. One would expect, however, the invasion of post-disturbance plant species, (e.g. brambles, cherry) a flourish of germination and growth by species intermediate in shade tolerance, and stump sprouting by species that characteristically use that form of regeneration.

Very little work has been done on animal community responses to forest sludge fertilization. Bierei et al. (1975) found that Peromyscus leucopus population densities on effluent and sludge-injected effluent treated areas were significantly greater than control area densities in the fall, but not in spring. The authors attributed this to increased herbaceous growth following fertilization, which improved summer food and cover resources that were unavailable in winter. Wood and Simpson (1973) conducted pilot studies on sewage effluent forest irrigation effects on wildlife and wildlife habitat, and their data suggested no difference in P. leucopus populations between treatment and control sites. They noted greatly enhanced herbaceous vegetation growth due to sewage treatment, theoretically improving habitat for herbivores, which they did not monitor. Sludge treatment in a 40-year-old Douglas fir forest, according to early reports, resulted in lower herbivore numbers on treated plots, apparently due to reduction of the animals' required food and cover plant species by sludge treatment (West 1981). Woodyard (1982) monitored small mammal populations for 2 years following sludge fertilization of a 4-year-old jack pine clearcut. He found increases in small mammal species diversity and foliage height diversity on treated plots, with 3 small mammal species showing increased colonization on treatment plots.

Although research has not examined mammal responses to the forest thinning pattern created by cutting application trails, more extensive thinning or clearcutting in eastern deciduous forests has prompted herbivore increase and immigration and changes in numbers within granivore-omnivore species (Krull 1970, Kirkland 1977). A study of mammal use of 15-m-wide undisturbed winter roads and surrounding forest in Canada revealed increased Microtus captures in the predominantly grass and sedge dominated roads, and fewer Clethrionomys gapperi captures on roads than in forest (Douglass 1977).

The effects of forest fertilization and/or thinning on reptile and amphibian population size and distribution have not been documented.

Thus, the research to date leaves many questions to be answered concerning sludge fertilization effects on forest ecosystems. Evidence does suggest that such a manipulation can potentially alter forest plant and animal community composition, structure, and dominance and hence the ecosystem's integrity and stability. Depending on the goals of forest and wildlife managers, the effects may be positive or negative, and in either case society would benefit greatly by increasing its understanding of and ability to predict the short and long range environmental soundness of this form of sludge disposal.

OBJECTIVES

This study was undertaken with the primary objective of determining the first year impact of sewage sludge fertilization of a northern hardwoods ecosystem, on various plant, small mammal, and terrestrial salamander communities. Specifically, the responses of plant community structure and composition, plant current annual production, small mammal community size and composition, and terrestrial salamander populations to sludge application were investigated.

SITE DESCRIPTION

The area studied was part of the Thunder Bay River State Forest in Montmorency County, northern lower Michigan (Fig. 1). The site is an approximately 25 ha, 50-year old northern hardwoods forest located 16 km north-east of Atlanta - roughly halfway between Gaylord and Alpena. It occupies the SW ¼ of the SW ¼ of section 19, and the NW ¼ of the NW ¼ of section 30, T. 32 N., R. 2 E., Montmorency County. The land is flat to gently sloping. Soils were of the Mancelona, Melita and Menominee series, which are deep, well-drained soils in sandy material (MSU Forestry Dept., unpubl. data. Elevation is approximately 300 m above sea level.

Sugar maple (*Acer saccharum*) and red maple (*Acer rubrum*) were the dominant overstory tree species, with beech (*Fagus grandifolia*), basswood (*Tilia americana*), birch (*Betula lutea* and *Betula papyrifera*), hemlock (*Tsuga canadensis*), red oak (*Quercus rubra*) and white ash (*Fraxinus americana*) as subdominants (MSU Forestry Dept. unpubl. data).

The study area was located just north of the Polar-Equator Trail, which marks the 45th parallel. Atlanta, Michigan (44° 59' N, 84° 10' W) and has a climate typical

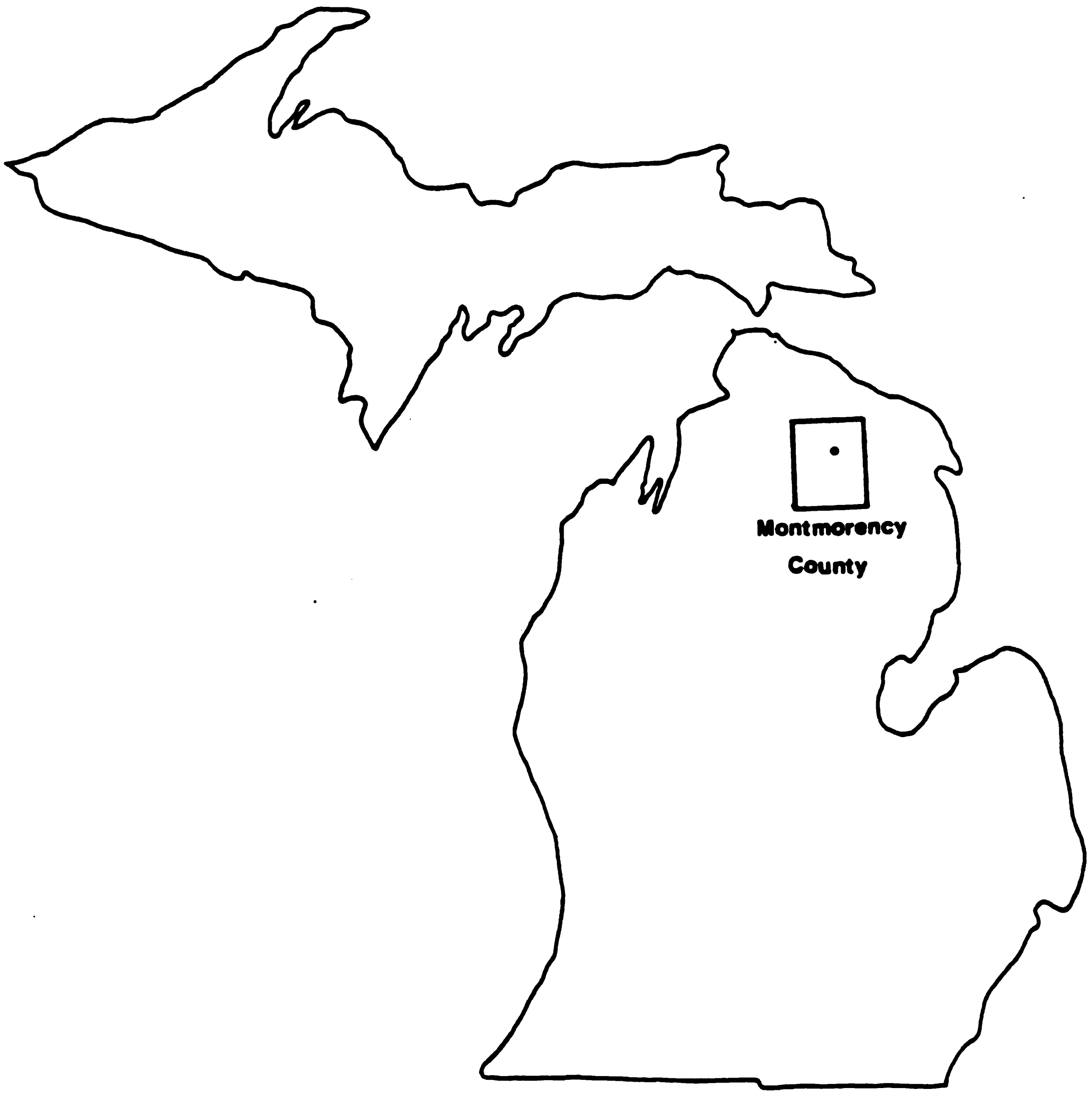

Figure 1. Map showing location of study site in Montmorency County, MI.

of northern lower Michigan, with long, severe winters, short, cool summers and an abbreviated growing season. The average annual precipitation (including melted snow) is 76.66 cm. The mean annual temperature is 5.83° C. Average temperature extremes range from -7.4° C in January to 19.6° C in July (NOAA 1981). Average precipitation gradually declines from a monthly high of 8 cm in June to a February low of 3.4 cm. During the study period (August, 1981 through August, 1982), temperatures closely followed the average except during the winter, which was unusually cold. Precipitation from fall to January tended to equal or exceed the normal, but winter and spring levels were abnormally low (Fig. 2) (NOAA 1982).

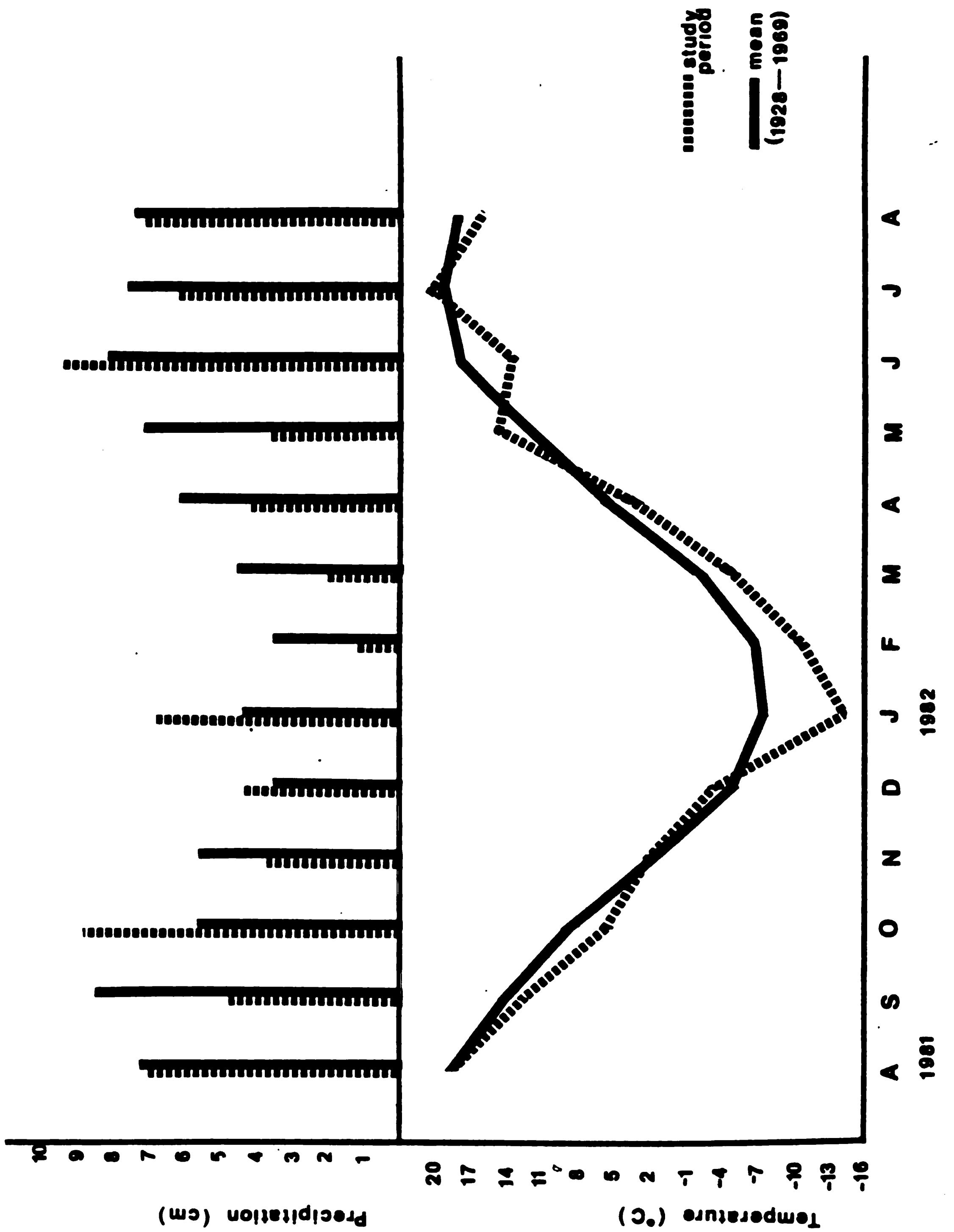

Figure 2. Average precipitation and temperature, by month for normal period and study period, 1981-1982, Atlanta, MI.

METHODS AND MATERIALS

Experimental Design

The study was organized as a completely randomized design. The study area was divided into 9 study plots, each 1.5 ha in size and separated from each other by at least 20 m buffer zones. In order for the sludge application vehicle to gain access to the interiors of the study plots, 5-m wide application trails were needed, running the length of each plot and situated at 20 m intervals. Because these trails were expected to have a treatment effect independent of the sludge effect, the study was designed with 2 treatments and a control in order to be able to separate trail effects from response to sludge application. The 9 study plots were randomly divided into 3 plots with trails-only treatment, 3 plots with trails and sludge, and 3 control plots which received no manipulation at all (Fig. 3).

Trail and Sludge Treatment

In September 1981, application trails were created on the 6 treatment plots, along with an east-west access trail. All trees and shrubs greater than 2 m in height

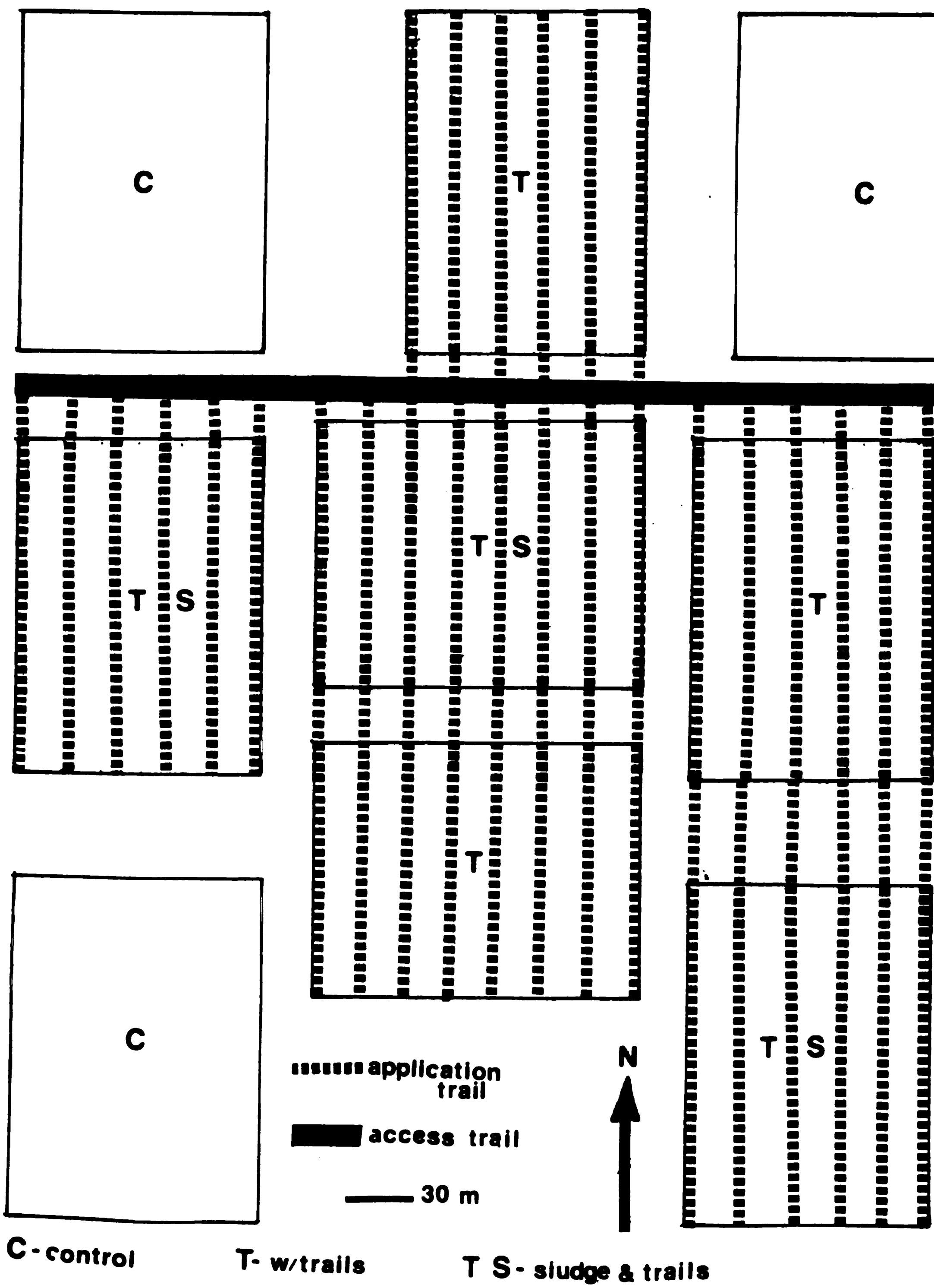

Figure 3. Study plot arrangement, northern hardwoods study site, Montmorency County, MI.

removed from the trails by felling with chain saws and skidding cut material to a site outside the study area. Sludge application, scheduled for fall 1981, was postponed until summer 1982. In late June and early July 1982, each sludge treatment plot received approximately 14 metric tons (224,601 liters) of anaerobically digested, municipal sewage sludge from Rogers City, Michigan. Three tanker trucks, making repeated trips, transported the sludge to a field near the study area. Sludge was transferred from these trucks to a smaller tank pulled by a tractor originally designed for logging operations. The tank sprayed sludge onto the adjacent forest strips, designated as "interiors," as it moved slowly along the application trails. In order to achieve the desired nitrogen loading level on the forest floor, the application vehicle had to make several passes around each interior. Application trails themselves did not receive sludge. Application was accomplished in 2½ weeks.

The U.S. Forest Service - MSU Cooperative Analytical Laboratory analyzed sludge samples to determine element content and loading levels, which are listed in Table 1. Concentrations reflect the fact that this was "clean" sludge, from a nonindustrial source, with very low levels of trace elements - well below the maximum metal limits allowable for sludges used on food crops (Chaney 1973). Heavy and trace metal loading levels were low, while

Table 1. Mean element concentrations in wet sludge from Rogers City, MI, and mean loading levels applied to the soil on the northern hardwood study site, Montmorency County, MI in July 1982.

Element	Chemical concentration ($\overline{X}$ for 3 plots)	Loading levels ($\overline{X}$ kg/ha for 3 plots)
Solids (%)	5.04	9210
Nitrogen (%)	0.427	783.1
Phosphorus (%)	0.21	383.7
Zn (ppm)	47.63	8.60
Cd (ppm)	0.42	0.08
Mn (ppm)	9.18	1.66
B (ppm)	1.50	0.27
Fe (ppm)	2568	465.9
Al (ppm)	440.7	79.80
Mg (ppm)	275	49.84
Cu (ppm)	59.7	10.82
K (ppm)	65.4	11.89
Ca (ppm)	2781	503.0
Ni (ppm)	1.17	0.21
Cr (ppm)	3.22	0.58
Na (ppm)	102.4	18.57

the application rates of nitrogen, phosphorus, calcium and potassium were high and are comparable to high conventional fertilization treatments (MSU Forestry Department unpubl. data).

Vegetative Community Composition

Vegetation sampling used a stratified random sampling design since plant populations were expected to vary in a predictable pattern among the application trails and the interior forest strips between trails. Trails were sampled as 1 stratum and interiors as another, in order to keep variation within strata small and avoid inflating the sampling error of the estimated population mean (Steel and Torrie 1960). In addition, separate stratum means could be estimated and compared to each other (e.g. sludged interiors vs. unsludged interiors vs. controls).

Plant community composition was characterized using nested quadrats, placed according to the above-mentioned stratified random sampling design. Woody plants were counted and recorded, by species in each of 5 size classes: 0-30 cm tall, 30 cm - 1 m tall, 1 m - 2 m tall, > 2 m tall but <10 cm dbh, and >2 m tall and >10 cm dbh.

The height classes chosen conformed to the natural life forms which occurred on the site, and to correspond with the vertical cover heights also measured - heights considered to be of particular importance to the small mammals occupying the habitat (M'Closkey and Lajoie 1975). Quadrats (plots) used to sample the smallest size class (0-30 cm) were 1 m X 10 m, class II (30 cm - 1 m) were 2 m X 20 m in interiors and controls and 1 m X 30 m in the trails, classes III (1 m - 2 m) and IV (> 2 m high, <10 cm dbh) were also 2 m X 20 m, and class V (>2 m high, < 10 cm dbh) were 4 m X 20 m. Long, narrow rectangular plots were found to be most effective, since much of the vegetation occurred in a clumped distribution. Quadrats were randomly located in trails, interiors, and controls. Trail quadrats were placed lengthwise along the trails, while interior quadrats angled across the interiors in order to fit 20 m plots into a 15 m-wide interior. Frequency of herbaceous vegetation was recorded, by species, using 2 m X 5 m quadrats.

Vertical Vegetative Cover

The line intercept method (Gysel and Lyon 1980) was used to estimate vertical cover and foliage height diversity. All vegetative cover 1 cm or greater intercepting an imaginary vertical plane rising from 1 edge of a 20 m-long tape was recorded. Gaps of less than 10 cm were ignored. Cover was measured for each of 4 height strata known to be important to small mammals: 0-10 cm, 10-30 cm, 30 cm - 2 m, and >2 m (M'Closkey and Lajoie 1975). Cover lines were randomly located in control and treatment plots, and in treatment plots they were placed so that they ran across a trail and an adjacent interior. Cover across the 15 m wide interiors was recorded in 5 m segments, so that an edge profile could be constructed. Segments adjacent to a trail were expected to show an edge effect. Vegetation measures were taken during the last week in July, 1983, after full leaf-out but before senescence had led to significant leaf loss.

Annual Plant Production

To estimate primary production below 2 m in height, samples of above-ground vegetative growth to up to 2 m were collected. Quadrats ½ m x 15 m were randomly located in control plots. In treatment plots, ½ m x 20 m plots were randomly located across plots so that they spanned an entire interior (15 m) and an adjacent trail (5 m). Within a quadrat all living herbaceous vegetation was clipped, as was all current annual growth from living woody plants, from ground level to a height of 2 m. Tissue taken from interior strips was kept separate from that collected in trails. Seven categories of vegetation were chosen, based on consistent abundance throughout all study plots: Hop-hornbeam (*Ostrya virginiana*), beech, white ash, sugar maple, and bracken fern were segregated into individual groups. All other woody species were combined (e.g. red oak, red maple, basswood) into a single group, and all herbaceous species were combined together. Plant production above 2 m in height was not collected, since it was assumed that such material would be beyond the reach of foraging herbivorous mammals expected to inhabit the area. Vegetation was collected during the last week in August, 1982. Samples were placed in paper bags, oven-dried at 60°C until they no longer lost weight (@ 24 hr), at which time dry weights were recorded.

Small Mammal Populations

Small mammal populations were monitored by conducting several periods of live-trapping, each lasting for 5 consecutive days and occurring once per month.

In mid-August 1981, a single 5-day trapping session was conducted in an attempt to gather pre-treatment, baseline data on the study area's small mammal community. A 5 x 5 grid of trap stations was centered within each study plot. Stations were 15 m apart and had a single Sherman live-trap (H. B. Sherman Co., Tallahassee, FL) (13 x 13 x 38 cm). Bait consisted of rolled oats, raisins, and anise extract. Traps were left open throughout the 5-day period, and were checked and re-set each morning. All newly captured animals were marked with ear tags or toe clipping; and for each captured individual ID, species, sex, relative age and trap station were recorded.

Trapping resumed in May, 1982, after trail cutting but before sludge application. In order to accommodate the high mammal populations indicated by the previous August's results, and also to standardize methods with concurrent trapping on 3 other sites as part of a larger project, trapping methods were revised. In 1982, 6 x 7 trapping grids were used, with trap stations 10 m apart and 2 Sherman traps per station. One half of the traps were placed in trails and ½ located in interiors, in order to check for differential use of the habitat. Each trap was baited with whole oats, animal fat and anise extract and equipped with cotton nest material. The June 1982 trapping period was abandoned due to sludge application, but trapping resumed in July, 2 days after sludging was completed, and a final period was conducted in mid-August.

Terrestrial salamanders were censused along belt transects using the method described by Burton and Likens (1975). On rainy summer nights after litter and understory plants are well soaked, terrestrial salamanders such as the red-backed salamander (Plethodon cinereus) can be found foraging above ground on litter, plant leaves or stems up to 2.8 m above ground level. Randomly located belt transects, 2 m x 95 m, each spanning a plot's interiors and trails, were marked out in advance with string. On appropriate rainy nights, beginning 1 hr after sunset, teams of observers using bright flashlights slowly walked transects recording each salamander observed in the transect. Each team of observers worked a pair of transects: 1 transect in a trail-only plot and 1 in a sludge and trails plot. An equal number of transect pairs began with observation of a sludge plot and a trail-only plot, so as to minimize any effect from time or observer fatigue.

Data Analysis

Since the study was designed around 3 types of treatment plots (the 2 manipulations: trails-only, and trails and sludge application, and the controls), 1-way analysis of variance was used to compare vegetation data and identify significant differences among treatments (Steel and Torrie 1960). In the seedling size classes, variances often differed enough to be classified as heterogeneous when examined with Bartlett's test for homogeneity of variance. Heterogeneous data were subjected to a log transformation, which often resulted in homoscedasticity, thus making analysis of variance the appropriate test for significance. In the few instances when transformation did not correct heterogeneity, pairs of treatments were compared using the t-test for unequal variances (Steel and Torrie 1960).

Percent cover data, not expected to follow a normal distribution, were subjected to the arcsin $\sqrt{X+1}$ transformation (Steel and Torrie 1960).

The stratified random sampling procedure for vegetation yielded estimates of means for entire plots, in which data from the 2 strata (trails and interiors) were combined. In addition, estimates from interiors or trails alone were available, and were compared to each other and to controls, again using 1-way analysis of variance. Finally, trail data were compared to interior data, within treatments, using t-tests.

Required sample sizes for all vegetation measures were calculated using Snedecor's (1956) formula:

$$n = \frac{s^2 t^2}{d^2}$$

in which n = required number of plots or lines

s^2 = sample variance

t = normal deviate at confidence limit level ($\alpha = 0.10$) and appropriate degrees of freedom

d = margin of error (sample $\bar{X}$ times designated accuracy of 20%)

The number of individual mammals captured in August, 1981 was relatively high, while populations were greatly reduced in 1982. Since 1982 numbers were too low to make the use of conventional capture-recapture population estimators feasible, enumeration was chosen to estimate all mammal populations. Beginning at time t on a plot, the number of animals caught at time t was summed with the numbers of previously marked animals caught after time t, but not at time t (Krebs 1966). Since treatment plot sizes were small, the study was concerned with comparisons of relative numbers of animals rather than with population densities, and density estimates were unnecessary. One-way analysis of variance compared mammal numbers among treatments on a month-by-month basis. T-tests compared the number of trail captures to interior captures.

Salamander populations were estimated by direct count, and the data for pairs of transects counted on the same night were evaluated using a paired-comparison t-test. A chi-squared test compared numbers seen on trails to numbers seen in interiors.

Foliage height diversity and mammal species diversity were each estimated with the Shannon-Wiener equation: $H^1 = -\Sigma pi \ell npi$ (Brower and Zar 1977), where pi is the proportion of the total (vertical cover or mammal species) which the ith category contributes. One-way analysis of variance evaluated treatment differences.

Linear correlations were used to test for associations between mammal species diversity and foliage height diversity, and associations between small mammal numbers and various cover estimates and FHD. In conjunction with analyses of variance, specific treatment differences were isolated using Duncan's new multiple range test (Chew 1976). Differences were considered significant at the $\alpha = 0.10$ level, for all comparisons.

RESULTS

Vegetative Community Composition

When trail and interior data were combined to represent an entire plot, neither sludge and trail nor trail-only treatment had a discernible effect on total living woody stem densities in 4 of the 5 size classes. In the large tree size class, >2 m in height and >10 cm dbh, however, control plots had stem densities significantly greater than plots with trails or plots with both sludge and trail treatment ($P < 0.05$) (Table 2).

The density data were also broken down into interior-only and trail-only comparisons, by species, between the 2 treatments and the controls. These comparisons revealed several treatment responses which varied according to species and size class. The general trend was for sludged interiors and sludge plot trails to have lower stem densities relative to those in unsludged interiors, controls, or trails in trail-only plots.

The small seedling class (0-30 cm) followed this trend. When interior strips were compared between treatments

Table 2. Density (stems/ha) of all woody plants in entire plots (interiors and trails) per size class: for controls, plots with trails, and plots with trails and sludge application in Montmorency County, MI, July, 1982.

Size class	Control plots[1]	w/Trails plots	Sludge and trails plots
0-30 cm	53,276 ± 11,300	56,649 ± 4,106	42,016 ± 9,847
30 cm - 1 m	2,853 ± 374	3,961 ± 510	2,795 ± 240
1 m - 2 m	1,856 ± 550	1,738 ± 133	1,522 ± 158
>2 m, <10 cm dbh	3,222 ± 652	2,306 ± 137	2,267 ± 428
>2 m, >10 cm dbh	888[a][2] ± 50	676[b] ± 31	645[b] ± 43

[1] $\overline{X}$ value with standard error, from 3 plots per treatment.

[2] numbers in the same horizontal row with different letter superscripts are significantly different ($P < 0.05$).

(Table 3), sludge-treated interiors had fewer total seedlings, and fewer "combined maple"*, red oak, and "combined other species" seedlings than did controls or interiors in trail-only plots. However, the sludge plot interior densities of total seedlings and combined maple seedlings were not significantly lower than control densities, nor were sludged interior numbers significantly lower than trail-only plot densities of "combined other" species. White ash seedling densities proved the consistent exception to the trend, and in this smallest size class, interiors in both types of treatment plots had significantly more ash seedlings than were found in controls.

Sludge was not applied to the trails themselves. In trail density comparisons between treatments, in the 0-30 cm size class (Table 4) while ash had significantly higher densities on trails in both treatment types than on control plots ($P < 0.01$). As in the interiors, red oak had fewer seedlings on sludge-plot trails than on other trails or controls, although the difference was significant only between control plots and sludge plot trails.

Among larger seedlings (30 cm – 1 m) in interior strips, only white ash and total seedling numbers differed

*Sugar and red maple data were combined in this size class, due to difficulty in their identification.

Table 3. Density (stems/ha) of each species in the 0-30 cm size class in interiors: for controls, plots with trails, and plots with trails and sludge application, in Montmorency County, MI, July 1982.

Species	Control plots[1]	w/Trails plots	Sludge and trails plots
Hophornbeam (*Ostrya virginiana*)	3,202 ± 834	2,633 ± 896	1,870 ± 193
Combined maples (*Acer rubrum* & *A. saccharum*)	$36,408^{ab}$[2] ± 7,856	$61,111^{a}$ ± 9,420	$27,664^{b}$ ± 7,739
American beech (*Fagus grandifolia*)	2,456 ± 1,347	1,733 ± 788	756 ± 87
White Ash (*Fraxinus americana*)	$2,584^{a}$ ± 544	$17,222^{ab**}$ ± 5,646	$9,267^{b}$ ± 1,059
American basswood (*Tilia americana*)	1,812 ± 265	2,745 ± 770	4,215 ± 2,778
Red oak (*Quercus rubra*)	$1,856^{a}$ ± 411	$2,833^{a}$ ± 356	565^{b} ± 77
Combined other species	$4,309^{a}$ ± 1,732	$2,311^{ab}$ ± 387	688^{b} ± 29
Total	$53,276^{ab}$ ± 11,300	$90,400^{a*}$ ± 13,572	$44,640^{b}$ ± 10,623

[1] $\bar{X}$ value with standard error, from 3 plots per treatment.

[2] numbers in the same horizontal row with different letter superscripts are significantly different at $P < 0.05$ unless otherwise specified.

(*significantly different at $P < 0.10$)
**heterogeneous variance uncorrected by transformation, n.s.d. when tested with t-test for unequal variances.

Table 4. Density (stems/ha) of each species in the 0-30 cm size class in trails: for controls, plots with trails, and plots with trails and sludge application, in Montmorency County, MI, July 1982.

Species	Control plots[1]	w/Trails plots	Sludge and trails plots
Hophornbeam (*Ostrya virginiana*)	3,202 ± 834	1,483 ± 192	1,943 ± 306
Combined maples (*Acer rubrum* & *A. saccharum*)	36,410 ± 7,853	23,779 ± 3,540	17,700 ± 5,632
American beech (*Fagus grandifolia*)	2,546 ± 1,347	1,303 ± 221	1,559 ± 372
White ash* (*Fraxinus americana*)	2,594[a][2] ± 544	13,320[b] ± 3,160	8,330[b] ± 938
American basswood (*Tilia americana*)	1,812 ± 265	1,609 ± 281	3,116 ± 1,547
Red oak (*Quercus rubra*)	1,856[a] ± 411	1,286[ab] ± 431	463 ± 124
Combined other species	4,309 ± 1,732	2,839 ± 1,459	1,279 ± 728
Total	53,276 ± 11,300	45,399 ± 2,250	34,143 ± 7,823

[1] $\overline{X}$ value with standard error, from 3 plots per treatment

[2] numbers in the same horizontal row with different letter superscripts are significantly different at $P < 0.01$ unless otherwise specified.

*significantly different, $P < 0.10$

Table 5. Density (stems/ha) of each species in the 30 cm - 1 m size class in interiors: for controls, plots with trails, and plots with trails and sludge application, in Montmorency County, MI, July 1982.

Species	Control plots[1]	w/Trails plots	Sludge and trails plots
Hophornbeam (*Ostrya virginiana*)	756 ± 152	717 ± 279	642 ± 96
Red maple (*Acer rubrum*)	154 ± 97	29 ± 9	98 ± 46
Sugar maple (*Acer saccharum*)	786 ± 87	1,135 ± 302	1,056 ± 210
American beech (*Fagus grandifolia*)	411 ± 58	320 ± 12	339 ± 22
White ash (*Fraxinus americana*)	342^{a}[2] ± 104	$1,636^{b}$ ± 385	883^{ab} ± 229
American basswood (*Tilia americana*)	51 ± 12	106 ± 68	60 ± 13
Red oak (*Quercus rubra*)	74 ± 59	19 ± 9	19 ± 7
Combined other species	242 ± 19	459 ± 180	117 ± 49
Total	$2,853^{a*}$ ± 374	$4,440^{b}$ ± 576	$3,200^{ab}$ ± 227

[1] $\bar{X}$ value with standard error, from 3 plots per treatment.

[2] numbers in the same horizontal row with different letter superscripts are significantly different at $P < 0.05$, unless otherwise specified

*significantly different, $P < 0.10$

significantly among treatments (Table 5). In both cases, densities were significantly higher on interiors in trail-only plots than on controls, with sludges interior densities at nonsignificant intermediate levels.

Trail densities had nearly the reverse pattern in large seedlings (Table 6). Control plots had more hophornbeam, beech, and "combined-other-species" seedlings than did trails in either type of plot with trails. However, in the case of hophornbeam and "combined other species," control densities were not significantly greater than trail-only plot densities.

The final treatment influences on stem densities appeared in the interior strips in the 1 m - 2 m size class (Table 7). White ash and American basswood both had significantly higher ($P<0.05$) densities in trails-only plot interiors than in either sludged interiors or controls.

Trails did not contain trees or shrubs greater than 1 m tall. Pole size sapling (> 2 m tall, > 10 cm dbh) and large tree (> 2 m tall, >10 cm dbh) densities did not differ significantly between control plots and treatment plot interiors (Tables 8 and 9).

When trail densities were compared to interior densities within the same treatment, a consistent pattern emerged with interiors containing similar or higher densities than found in trails. In the 0-30 cm size class,

Table 6. Density (stems/ha) of each species in the 30 cm - 1 m size class in trails: for controls, plots with trails, and plots with trails and sludge application, in Montmorency County, MI, July 1982.

Species	Control plots[1]	w/Trails plots	Sludge and trails plots
Hophornbeam (*Ostrya virginiana*)	$756^{a[2]} \pm 152$	$307^{ab} \pm 191$	$207^{b} \pm 49$
Red maple (*Acer rubrum*)	154 ± 97	73 ± 28	52 ± 10
Sugar maple (*Acer saccharum*)	786 ± 87	946 ± 282	665 ± 152
American beech (*Fagus grandifolia*)	$411^{a*} \pm 58$	$116^{b} \pm 46$	$173^{b} \pm 52$
White ash (*Fraxinus americana*)	342 ± 104	784 ± 259	349 ± 176
American basswood (*Tilia americana*)	51 ± 12	46 ± 5	73 ± 25
Red oak (*Quercus rubra*)	74 ± 59	19 ± 10	43 ± 17
Combined other species	$242^{a} \pm 19$	$185^{ab} \pm 115$	$25^{b} \pm 25$
Total	$2,853 \pm 374$	$2,527 \pm 550$	$1,578 \pm 429$

[1] $\bar{X}$ value with standard error, from 3 plots per treatment

[2] numbers in a horizontal row with different letter superscripts are significantly different at $P < 0.10$ unless otherwise specified.

* significantly different, $P < 0.05$

Table 7. Density (stems/ha) of each species in the 1 m - 2 m size class in interiors: for controls, plots with trails, and plots with trails and sludge application, in Montmorency County, MI, July 1982.

Species	Control plots[1]	w/Trails plots	Sludge and trails plots
Hophornbeam (*Ostrya virginiana*)	697 ± 291	633 ± 123	564 ± 59
Red maple (*Acer rubrum*)	56 ± 36	22 ± 3	0 ± 0
Sugar maple (*Acer saccharum*)	403 ± 116	506 ± 81	686 ± 116
American beech (*Fagus grandifolia*)	589 ± 180	520 ± 6	628 ± 122
White ash (*Fraxinus americana*)	64^{a}[2] ± 23	433^{b} ± 154	122^{a} ± 35
American basswood (*Tilia americana*)	3^{a} ± 3	14^{b} ± 3	3^{a} ± 3
Red oak (*Quercus rubra*)	0 ± 0	6 ± 3	0 ± 0
Combined other species	53 ± 18	183 ± 85	31 ± 10
Total	1,856 ± 550	2,317 ± 177	2,031 ± 211

[1] $\overline{X}$ value with standard error, from 3 plots per treatment

[2] numbers in a horizontal row with different letter superscripts are significantly different ($P < 0.05$).

Table 8. Density (stems/ha) of each species in the >2 m tall, <10 cm dbh size class in interiors: for controls, plots with trails and plots with trails and sludge application, in Montmorency County, MI, July 1982.

Species	Control plots[1]	w/Trails plots	Sludge and trails plots
Hophornbeam (*Ostrya virginiana*)	272 ± 173	461 ± 129	203 ± 66
Red maple[2] (*Acer rubrum*)	208 ± 83	25 ± 1	8 ± 5
Sugar maple (*Acer saccharum*)	1,936 ± 223	1,950 ± 113	2,297 ± 422
American beech (*Fagus grandifolia*)	672 ± 267	353 ± 61	456 ± 143
White ash (*Fraxinus americana*)	6 ± 6	106 ± 74	42 ± 26
American basswood (*Tilia americana*)	25 ± 8	8 ± 5	11 ± 3
Red oak (*Quercus rubra*)	0 ± 0	0 ± 0	3 ± 3
Combined other species	75 ± 59	167 ± 97	8 ± 8
Total	3,222 ± 652	3,075 ± 183	3,022 ± 570

[1] $\overline{X}$ value with standard error, from 3 plots per treatment.

[2] Heterogeneous variances, uncorrected by transformation; no significant difference when tested with t-test for unequal variances.

Table 9. Density (stems/ha) of each species in the >2 m tall, >10 cm dbh size class in interiors: for controls, plots with trails, and plots with trails and sludge application, in Montmorency County, MI, July 1982.

Species	Control plots[1]	w/Trails plots	Sludge and trails plots
Hophornbeam (*Ostrya virginiana*)	4 ± 2	21 ± 2	22 ± 2
Red maple (*Acer rubrum*)	94 ± 49	19 ± 6	1 ± 1
Sugar maple (*Acer saccharum*)	508 ± 41	557 ± 22	572 ± 75
American beech (*Fagus grandifolia*)	94 ± 51	61 ± 17	64 ± 23
White ash (*Fraxinus americana*)	10 ± 4	14 ± 1	18 ± 14
American basswood (*Tilia americana*)	110 ± 28	107 ± 20	115 ± 37
Red oak (*Quercus rubra*)	35 ± 6	21 ± 9	25 ± 23
Combined other species	40 ± 23	101 ± 46	42 ± 30
Total	888 ± 50	901 ± 41	860 ± 58

[1] $\bar{X}$ value with standard error, from 3 plots per treatment.

red oak, combined species of maple, and total seedling densities were significantly higher in trail-only plot interiors than in trails in the same plots ($P < 0.10$); no other significant differences occurred (Tables 10 and 11). In the large seedling size class (30 cm - 1 m) the pattern was not so well defined, although interiors still had similar to higher densities. Here, hophornbeam and total seedling densities in sludged interiors exceeded those in trails in the same plots (Table 11). Trail-only plot interiors had significantly more beech, basswood and total seedlings than were found in the trails (Table 10).

Herbaceous Species Frequency

Herbaceous understory species tended to occur in a varied distribution from plot to plot, making it difficult to characterize the forest floor's herbaceous composition in other than broad terms. However, some patterns emerged (Fig. 4).

There were 7 plants common to nearly all 9 of the study plots: starflower (Trentalis borealis), Canada mayflower (Maianthemum canadense), sedge (Carex spp.), bedstraw (Galium aparine), violet (Viola spp.), heart-leaved aster (Aster cordifolius), and brambles (Rubus spp.) Their distributions closely resembled each other in control plots and trail-only plot interiors (Fig. 4). These same herbaceous species had a somewhat different occurrence profile in the trails. Trails had similar profiles in

Table 10. Density (stems/ha) of each species in the 0-30 cm and 30 cm-1 m size classes, in interiors and trails, for plots with trails-only, Montmorency County, MI, July 1982.

Species	0-30 cm		30 cm - 1 m	
	Interiors[1]	Trails	Interiors	Trails
Hophornbeam (*Ostrya virginiana*)	2,633±896	1,483±192	717±279	307±191
Red maple (*Acer rubrum*)	—	—	29±9	73±28
Sugar maple (*Acer saccharum*)	—	—	1,135±302	946±282
Combined maples	61,111**±9,420	23,779±3,540	—	—
American beech (*Fagus grandifolia*)	1,733±788	1,303±221	320**±12	166±46
White ash (*Fraxinus americana*)	17,222±5,646	13,320±3,160	1,636±385	784±259
American basswood (*Tilia americana*)	2,745±770	1,609±281	106**±68	46±5
Red oak (*Quercus rubra*)	2,833*±356	1,286±431	19±9	19±10
Combined other species	2,311±387	2,839±1,459	459±180	185±115
Total	90,400**±13,572	45,399±2,250	4,440*±576	2,527±550

[1] $\overline{X}$ value with standard error, from the 3 treatment plots

*significantly greater than trail density ($P < 0.10$)

**significantly greater than trail density ($P < 0.05$)

Table 11. Density (stems/ha) of each species in the 0-30 cm and 30 cm - 1 m size classes, in interiors and trails, for plots with trails and sludge application, in Montmorency County, MI, July 1982.

Species	0-30 cm		30 cm - 1 m	
	Interiors[1]	Trails	Interiors	Trails
Hophornbeam (*Ostrya virginiana*)	1,870±193	1,943±306	642**±96	206±49
Red maple (*Acer rubrum*)	—	—	98±46	52±10
Sugar maple (*Acer saccharum*)	—	—	1,056±210	665±152
Combined maples	27,663±7,739	17,700±5,632	—	—
American beech (*Fagus grandifolia*)	756±87	1,559±372	339*±22	173±52
White ash (*Fraxinus americana*)	9,267±1,059	8,330±938	883±229	349±176
American basswood (*Tilia americana*)	4,215±2,778	3,116±1,547	60±13	73±25
Red oak (*Quercus rubra*)	565±77	463±124	19±7	43±17
Combined other species	688±29	1,279±728	117±49	25±25
Total	44,640±10,623	34,143±7,823	3,200*±227	1,578±429

[1] $\bar{X}$ value with standard error, from the 3 treatment plots

* significantly greater than trail density ($P < 0.10$)

**significantly greater than trail density ($P < 0.05$)

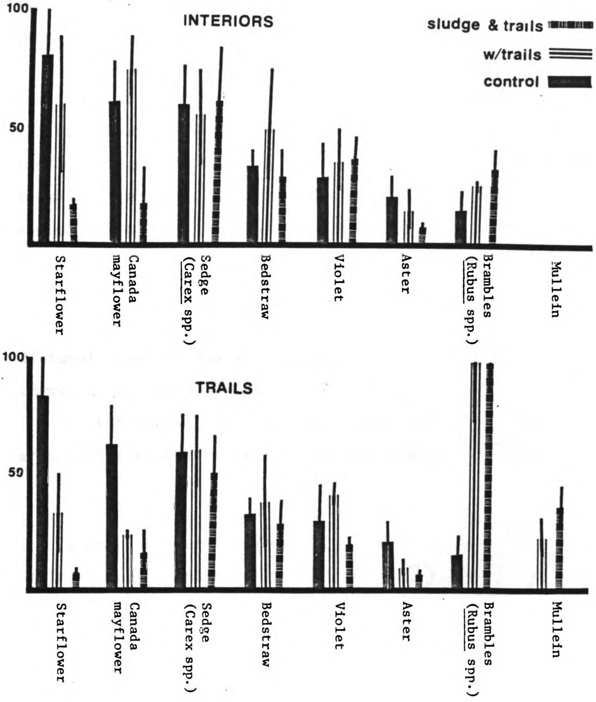

Figure 4. Frequency of commonly occurring herbs and shrubs (% of quadrats in which the plant occurred) in control plots, plots with trails, and plots with trails and sludge application, Montmorency County, MI, July 1982 ($\bar{X}$ with SE).

both types of plots with trails. A third, intermediate pattern emerged from herbaceous plant occurrence in sludged interiors.

As the profiles indicate, Carex, violet, bedstraw and heart-leaved aster occurred with similar frequency in all situations. Starflower and Canada mayflower were the most common plants in controls and trail-only plot interiors, and were among the least frequent in the trails and in sludged interiors. Rubus had a relatively low occurrence in controls and trail-only plot interiors, was relatively more common in sludged interiors, and became the most frequently found herbaceous plant in the trails. An additional species, mullein (Verbascum spp.), appeared commonly in the trails and did not occur elsewhere.

A list of all plant species identified on the study area indicates the tremendous variety present (Appendix 1).

Vertical Vegetative Cover

When plots were considered in their entirety by combining interior and trail percent cover data, percent vertical cover did not differ significantly between controls and/or treatments in stratum 1, the 0-10 cm height class (Fig. 5). In the 10-30 cm stratum (stratum 2), sludged plots had significantly less vertical cover than did trail-only plots. Sludged plots and trail-only plots both had less cover than did control plots in the 30 cm - 2 m and the >2 m height classes (strata 3 and 4). This reduction

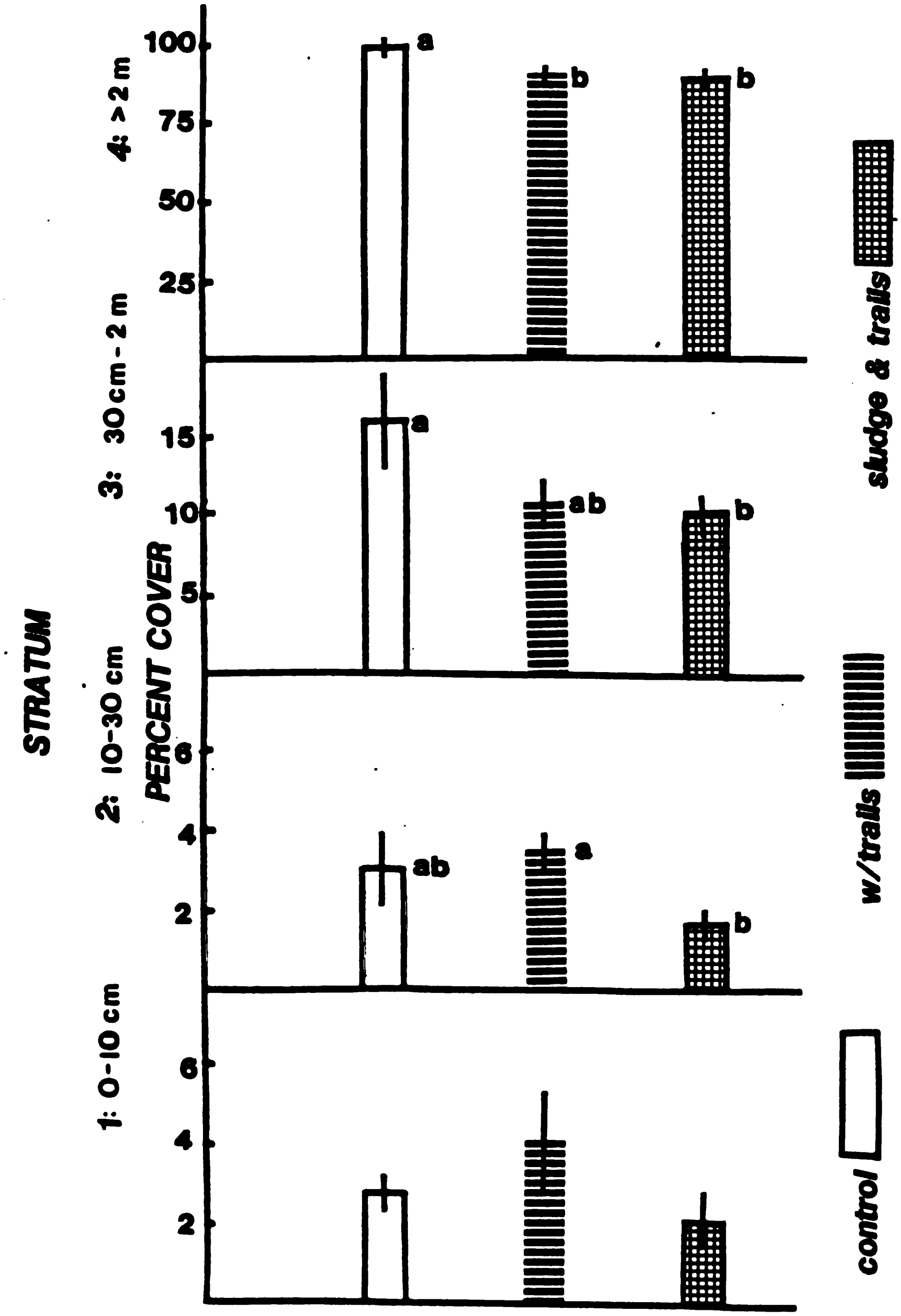

Figure 5. Percent vertical vegetative cover in 4 height strata in entire plots: for control plots, plots with trails, and plots with trails and sludge application, Montmorency County, MI, July 1982 ($\bar{X}$ with SE).

[1]Bars in the same stratum with different letter superscripts are significantly different ($P < 0.05$).

*Significantly different, $P < 0.10$.

in cover was not statistically different from control plot cover, however, in trail-only plots for stratum 3 (Fig. 5).

Considering the data for interior and trail cover individually revealed a breakdown of the treatment effects. When only interior cover was compared (Fig. 6), sludged interiors had significantly less vertical cover ($P < 0.05$) in the 10-30 cm stratum than did trail-only plot interiors. This parallels the lower seedling densities in the same height range. In stratum 4, control plot and sludged interiors had significantly more cover than trail-only plot interiors (Fig. 6). However, when interior cover was separated into 2, 5-m edge segments and a 5-m inner-interior segment, the cover differed significantly between treatment interiors and controls only in the edge segments, adjacent to the trails.

Trail cover comparisons (Fig. 7) indicated the most pronounced treatment differences in cover. As with entire plot and interior-only data, trail cover in stratum 1 (0-10 cm) was highly variable and no significant differences between treatments emerged. In stratum 2 (10-30 cm), sludge-plot trails had significantly less cover than did control plots. Strata 3 and 4, together representing all cover above 30 cm in height, had significantly less ($P < 0.01$) vertical cover in trails of both treatment types than in control plots.

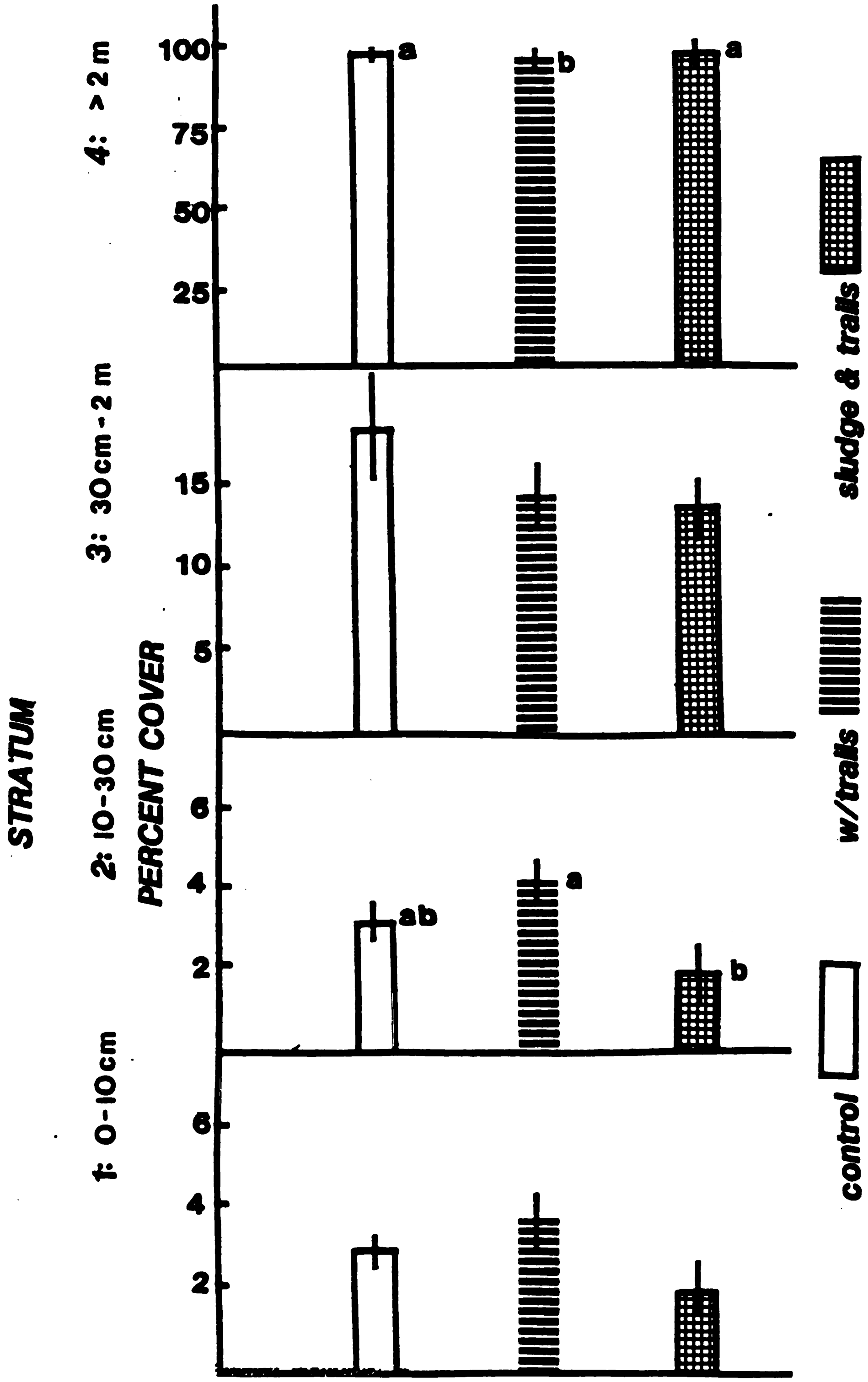

Figure 6. Percent vertical vegetative cover in 4 height strata in forest interiors: for control plots, plots with trails, and plots with trails and sludge application, Montmorency County, MI, July 1982 ($\overline{X}$ with SE)

[1]Bars in the same stratum with different letter superscripts are significantly different ($P < 0.05$).

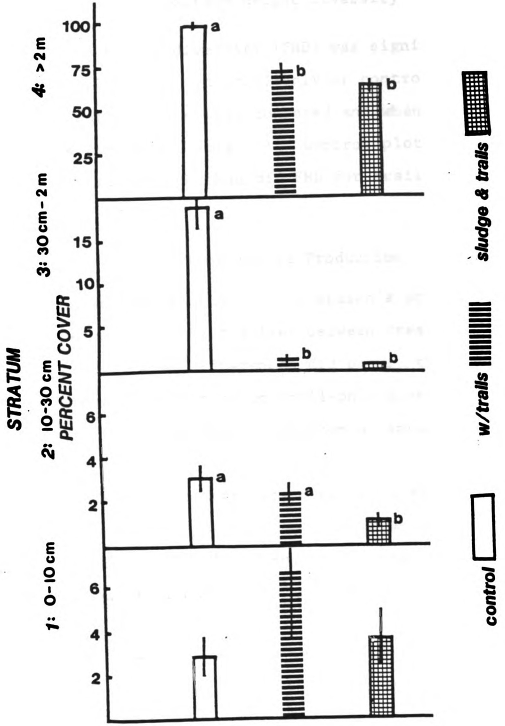

Figure 7. Percent vertical vegetative cover in 4 height strata in application trails: for control plots, plots with trails, and plots with trails and sludge application, Montmorency County, MI, July 1982 ($\overline{X}$ with SE).

[1]Bars in the same stratum with different letter superscripts are significantly different at P < 0.01 unless otherwise specified.

*Significantly different P < 0.10.

Foliage Height Diversity

Foliage height diversity (FHD) was significantly lower in sludged plots than in trail-only or control plots, both when entire plots were compared and when interiors alone were compared (Table 12). Control plots had significantly greater FHD than did FHD for trails in sludged plots.

Plant Annual Production

For most species the current season's primary production below 2 m did not differ between treatment and/or control plots. When differences did occur, the trend was for greater production on trail-only plots than on controls, with sludge plot production at intermediate levels.

When entire plot data were compared (Table 13), trail-only plots had nearly 3 times as much sugar maple current annual production as did control plots. Total annual production on trail-only plots exceeded that on control plots by 261%.

Interior-only comparisons yielded only 1 treatment difference: sugar maple annual production was twice as great on trail-only interiors as on controls (Table 14). Trails in trail-only plots had 4 times the amount of sugar maple production and 2.7 times as much total annual vegetative production as did control plots

Table 12. Foliage height diversity (Shannon Wiener Index values) for control plots, plots with trails, and plots with trails and sludge application, in Montmorency County, MI, July 1982.

Comparisons	Control plots[1]	w/Trails plots	Sludge and trails plots
Entire plots	0.5275^{a}[2] $\pm$ 0.0198	$0.4941^{a} \pm 0.0275$	$0.3844^{b} \pm 0.0137$
Interiors only	$0.5275^{a} \pm 0.0198$	$0.5230^{a} \pm 0.0247$	$0.4297^{b} \pm 0.0029$
Trails only	$0.5275^{a*} \pm 0.0198$	$0.4072^{ab} \pm 0.1164$	$0.2484^{b} \pm 0.0540$

[1] $\bar{X}$ value with standard error, from 3 plots per treatment

[2] numbers in a horizontal row with different letter superscripts are significantly different P 0.05 unless otherwise specified.

*significantly different, $P < 0.10$

Table 13. Above ground net annual production (kg/ha) <2 m: for controls, plots with trails, and plots with trails and sludge application in Montmorency County, MI, August 1982.

Species	Control plots[1]	w/Trails plots	Trails and sludge plots
Herbaceous species	2.28 ± 0.72	17.03 ± 9.22	6.36 ± 2.68
Hophornbeam (_Ostrya virginiana_)	3.86 ± 1.48	5.56 ± 1.71	6.05 ± 0.36
American beech (_Fagus grandifolia_)	6.38 ± 0.44	6.17 ± 2.18	7.28 ± 1.73
White ash (_Fraxinus americana_)	0.86 ± 0.09	8.32 ± 5.54	3.19 ± 1.05
Sugar maple (_Acer saccharum_)	$7.48^{a} \pm 0.87$	$20.78^{b} \pm 3.48$	$15.08^{ab} \pm 2.46$
Other woody species	3.44 ± 0.99	6.05 ± 1.54	4.03 ± 2.33
Bracken fern (_Pteridium aquilinum_)	1.73 ± 1.50	4.11 ± 2.10	0.50 ± 0.50
Total	$26.00^{a} \pm 1.39$	$68.00^{b} \pm 17.73$	$42.20^{ab} \pm 5.14$

[1] $\overline{X}$ value with standard error, from 3 plots per treatment.

[2] numbers in a horizontal row with different letter superscripts are significantly different ($P < 0.05$).

Table 14. Above ground net annual production (kg/ha) <2 m, from interiors: for controls, plots with trails, and plots with trails and sludge application in Montmorency County, MI, August 1982.

Species	Control Plots[1]	w/Trails plots	Sludge and trails plots
Herbaceous spp.	2.43 ± 1.32	14.89 ± 8.24	4.47 ± 1.42
Hophornbeam (*Ostrya virginiana*)	4.12 ± 1.58	6.77 ± 1.93	6.60 ± 0.62
American beech (*Fagus grandifolia*)	6.80 ± 0.46	7.03 ± 2.79	9.56 ± 1.74
White ash (*Fraxinus americana*)	1.01 ± 0.18	7.35 ± 5.22	3.23 ± 0.92
Sugar maple (*Acer saccharum*)	7.99[a][2] ± 0.92	16.79[b] ± 3.83	13.53[ab] ± 0.95
Other woody species	3.67 ± 1.05	6.37 ± 1.76	4.12 ± 1.54
Bracken fern (*Pteridium aquilinum*)	1.85 ± 1.60	5.48 ± 2.80	0.63 ± 0.63
Total	27.73 ± 1.46	64.67 ± 17.32	41.60 ± 3.39

[1] $\bar{X}$ value with standard error, from 3 plots per treatment.

[2] numbers in a horizontal row with different letter superscripts are significantly different ($P < 0.10$).

(Table 15). In contrast, beech annual production was significantly greater on control plots than on trails in either type of plot with trails.

Small Mammal Community

In August, 1981, a total of 4 small mammal species was captured on the study area. Eastern chipmunks (Tamias striatus), white-footed mice (Peromyscus leucopus) and woodland deer mice (Peromyscus maniculatus gracilis) occurred on all plots, and a single boreal red-backed vole (Clethrionomys gapperi) was captured (Table 16). This pre-treatment trapping period yielded significantly higher Peromyscus ($P < 0.01$) and total ($P < 0.05$) average per plot numbers of animals known to be alive on controls than were on either type of plot scheduled for treatment (Table 17).

In 1982, an additional species was captured on all 3 types of plots: the woodland jumping mouse (Napaeozapus insignis) (Table 18). There were incidental captures of a masked shrew (Sorex cinereus) and a short-tailed shrew (Blarina brevicauda), but no red-back voles were captured.

In contrast to the 1981 capture data, the 1982 average per-plot numbers of individual animals known to be alive did not differ significantly between treatments and/or controls for any month, either pre-sludge (May) or post-sludge application (July and August) (Table 19). When the major species were compared separately, however, there

Table 15. Above ground net annual production (kg/ha) <2 m, from trails: for controls, plots with trails, and plots with trails and sludge application in Montmorency County, MI, August 1982.

Species	Control plots[1]	w/Trail plots	Sludge and trails plots
Herbaceous species	2.43 ± 0.76	23.44 ± 12.24	12.06 ± 6.63
Hophornbeam (*Ostrya virginiana*)	4.12 ± 1.58	1.92 ± 1.07	5.6 ± 1.71
American beech (*Fagus grandifolia*)	6.80^{a}[2] ± 0.46	3.58^{b} ± 1.34	1.07^{b} ± 0.54
White ash (*Fraxinus americana*)	0.92 ± 0.09	11.28 ± 6.54	3.08 ± 1.55
Sugar maple (*Acer saccharum*)	8.00^{a} ± 0.92	32.76^{b} ± 3.19	19.73^{ab} ± 7.99
Other woody species	3.66 ± 1.05	5.06 ± 1.82	3.77 ± 0.68
Bracken fern (*Pteridium aquilinum*)	1.85 ± 1.60	0.00 ± 0.00	0.11 ± 0.11
Total	27.76^{a} ± 1.48	78.04^{b} ± 19.95	44.24^{ab} ± 11.69

[1] $\overline{X}$ value with standard error, from 3 plots per treatment

[2] numbers in a horizontal row with different letter superscripts are significantly different ($P < 0.10$)

Table 16. Total small mammals known to be alive in August 1981, in controls, plots designated to have trails, and plots designated for trails and sludge application in Montmorency County, MI.

Species	Control plots	w/Trails plots	Sludge and trails plots
Eastern chipmunk (_Tamias striatus_)	8	8	11
Peromyscus spp.	51	55	75
Boreal redback vole (_Clethrionomys gapperi_)	0	1	0
Total	59	61	86

Table 17. Average number of small mammals known to be alive in August, 1981, major Species only, in controls, plots designated to have trails, and plots designated for trails and sludge application in Montmorency, County, MI.

Species	Control plots[1]	w/Trails plots	Sludge and trails plots
Eastern chipmunk (_Tamias striatus_)	2.67 ± 1.20	1.67 ± 0.33	3.67 ± 0.88
Peromyscus spp.	25.00^{a}[2] ± 1.00	$18.67^{b} \pm 1.20$	$17.00^{b} \pm 1.00$
Total	$27.67^{a*} \pm 0.88$	$20.67^{b} \pm 1.67$	$20.67^{b} \pm 1.20$

[1] $\bar{X}$ number of individuals captured per plot, with standard error, from 3 plots per treatment.

[2] numbers in a horizontal row with different letter superscripts are significantly different ($P < 0.01$) unless otherwise specified.

*significantly different ($P < 0.05$)

Table 18. Total small mammals known to be alive in summer, 1982, by species, in controls, plots with trails, and plots with trails and sludge application in Montmorency County, MI.

Species	Control plots			w/Trails plots			Sludge and trails plots		
	5/12	7/12	8/14	5/12	7/12	8/14	5/12	7/12	8/14
Eastern chipmunk (Tamias striatus)	0	13	19	1	10	9	1	6	15
Woodland deer mouse (Peromyscus maniculatus gracilis)	0	1	3	1	6	9	1	3	19
White-footed mouse (Peromyscus leucopus)	0	2	8	3	5	9	2	6	21
Woodland jumping mouse (Napaeozapus insignis)	0	4	12	2	0	3	1	1	3
Masked shrew (Sorex cinereus)	0	0	1	0	0	0	0	0	0
Short-tailed shrew (Blarina brevicauda)	0	0	0	0	0	0	0	0	1
Total	0	20	43	7	21	30	5	16	59

Table 19. Average number of small mammals known to be alive in summer, 1982, all species combined, in controls, plots with trails, and plots with trails and sludge application in Montmorency County, MI.

Time of trapping; mammals mammal species diversity (Shannon-Wiener Index)	Control plots[1,2]	w/Trails plots	Sludge and trails plots
May 1982			
Small mammals	0 ± 0	2.33 ± 1.45	2.00 ± 1.15
Diversity value	0 ± 0	0.58 ± 0.31	0.58 ± 0.31
July 1982			
Small mammals	6.67 ± 0.88	7.00 ± 1.53	5.33 ± 1.45
Diversity value	0.30 ± 0.20	0.56 ± 0.28	0.66 ± 0.28
August 1982			
Small mammals	14.33 ± 5.55	10.00 ± 1.53	19.67 ± 3.38
Diversity value	0.79 ± 0.40	1.07 ± 0.20	1.09 ± 0.13

[1] $\overline{X}$ number of individuals captured per plot, with standard error, from 3 plots per treatment.

[2] $\overline{X}$ diversity index value with standard error, from 3 plots per treatment.

were significantly higher numbers of Peromyscus captured on sludged plots than on either control or trail-only plots (Table 20).

No significant differences were found among comparisons of captures in traps placed in trails with traps located in interiors.

Mammal species diversity did not correlate well with FHD ($r = 0.14$). Peromyscus numbers showed a strong negative correlation ($r = -0.835$, $P < 0.01$) with FHD; and also correlated negatively with percent vertical cover in the 10-30 cm stratum ($r = -0.63$, $P < 0.05$) and percent cover in the 2 m stratum ($r = -0.59$, $P < 0.10$).

Salamander Populations

The only salamander species found on the study area was the red-backed salamander, Plethodon cinereus. This terrestrial species did not demonstrate a detectable population response to sludge application when its numbers were compared between sludged plots and trail-only plots. There were an estimated 1484 salamanders per ha in trail-only plots, and 1447 per ha in sludged plots. A chi-squared analysis comparing numbers of salamanders observed in trails to numbers seen in interiors showed a significantly greater ($P < 0.01$) number of salamanders in interiors compared to very few animals in trails.

Table 20. Average number of small mammals known to be alive in summer, 1982, major species only, in controls, plots with trails, and plots with trails and sludge application in Montmorency County, MI.

Time of trapping species	Control plots[1]	w/Trails plots	Sludge and trails plots
May 1982			
Eastern chipmunk (*Tamias striatus*)	0 ± 0	0.33 ± 0.33	0.33 ± 0.33
Peromyscus spp.	0 ± 0	1.3 ± 0.88	1.0 ± 0.58
July 1982			
Eastern chipmunk (*Tamias striatus*)	4.33 ± 1.33	3.33 ± 0.33	2.00 ± 2.00
Peromyscus spp.	1.0 ± 0.58	3.7 ± 1.45	3.00 ± 1.00
August 1982			
Eastern chipmunk (*Tamias striatus*)	6.33 ± 4.1	3.00 ± 0.58	5.00 ± 3.51
Peromyscus spp.	6.67^{a}[2] ± 1.45	$4.00^{a} \pm 1$	$13.33^{b} \pm 1.86$

[1] $\bar{X}$ number of individuals captured per plot, with standard error, from 3 plots per treatment

[2] numbers in a horizontal row with different letter superscripts are significantly different ($P < 0.05$).

DISCUSSION

Vegetative Community Response

Both the forest thinning to create application trails and the sewage sludge application to forest interiors prompted notable first-year changes in the hardwood forest plant community. Composition was altered in tree seedling size classes by increased density of some species, such as white ash, and decreased density of others (e.g. red oak, beech, and the maples). Some understory species, such as Canada mayflower, were suppressed, while post-disturbance plants such as brambles and mullein were favored. Structural changes included loss of cover from middle and upper strata in trails and from lower strata in sludged interiors, which led to reduced foliage height diversity.

Because the 2 largest stem density size classes comprised trees that were sapling-size and larger, they would not be expected to respond to either treatment by increased density in the first year following treatment. Therefore density differences in these size classes would logically be attributable to actual tree removal or to inherent differences in plant community composition. Forest interior and control densities did not differ, in these large size classes, either by species or in total, indicating that the tree community composition was homogeneous among all plots. Not surprisingly, total large tree stem densities in entire plots (interior and trail data combined) were reduced in both types of plots subjected to trail cutting (i.e. tree removal). Sapling size trees (> 2 m tall, < 10 cm dbh), even though also eliminated from trails, showed a nonsignificant trend of greater control plot total densities. Since the variances were particularly high among treatment and control plots in this size class, there appeared to be inherently high variation in sapling densities throughout the study area. Large variances coupled with the low replication of 3 plots per treatment, affected the power of tests for treatment differences.

In order to differentiate plant community changes resulting from the 2 treatments, vegetation measures in trails and interior areas were also considered separately. In the smallest seedling size class (0-30 cm), the reduced numbers of red oak, combined maple, combined-other species and total seedlings on sludged interiors when compared to controls and/or to trail-only plot interiors suggest a negative response to the sludge itself. Shortly after treatment many withered seedlings and herbaceous plants were observed, heavily coated with sludge, indicating sludge induced leaf and plant mortality. This mortality may have been a function of applying sludge during the growing season, since a related experiment involving sludge application the previous fall in a mixed-oak forest produced no forest interior seedling density differences (Haufler et al. unpubl. data).

In contrast, the dramatic increases in white ash seedling numbers on both types of plots with trails, in both the interiors and in the trails, suggest a not unexpected, positive response to forest thinning. White ash is known to regenerate heavily (via seedlings) in recently cut northern hardwood forests (Marquis 1965, Borman et al. 1970, Leak and Solomon 1975). White ash regeneration appeared to have occurred in this study, either by increased germination or survival, or both. Other ash seedlings presumably present before either treatment may have benefited from release by forest thinning (i.e. trail construction), thereby gaining sufficient increased growth and/or survival to account for the observed greater densities of tall seedlings (30 cm - 2 m) in the forest interiors of both types of plots with trails. Bicknell (1982) monitored seedling growth following the clearcut of a northern hardwood forest, and her data for first-year post-cut growth by white ash seedlings substantiate this possibility. In reality, forest thinning such as that caused by trail construction may have provided the optimal release conditions for both ash and basswood, which also had greater tall seedling density in trail-only plots. White ash and basswood are considered "gap phase" species, which normally persist in climax forests by colonizing gaps in the canopy.

Such species tend to be able to make more extension growth in a season than, for instance, sugar maple or beech, and are best adapted to competing in conditions intermediate between large disturbed areas and a closed canopy (Marks 1975).

The trails themselves did not receive sludge treatment. However, they were subjected to considerable disturbance both by tree removal and later by the sludge applicator. Therefore, differences in stem densities should reflect either the effects of trail construction (e.g. trauma during logging, loss of soil or leaf litter, microclimate changes, etc.) and/or, in the case of sludged plots, possible physical trauma from the sludge vehicle's tires. The data exhibit vegetation changes in response both to characteristics of the trails or their construction and to trauma during sludging. Small (0-30 cm) red oak seedlings, and large (30 cm - 1 m) hophornbeam and combined-other-species seedlings had significantly reduced densities only in sludge plot trails, and appear to have suffered damage by the vehicle. In addition, red oak is another gap phase species and would seem unlikely to react negatively to light forest thinning such as this study's trail construction, unless the procedure itself was damaging. On the other hand, trails in both types of treatment plots had fewer large beech seedlings, and this species appears to have been more vulnerable to trail construction.

In density comparisons of trails to interiors within treatments, the trend for interiors to have similar or greater seedling densities than trails also suggests losses due to trail construction or changes in microenvironment. In trail-only plots, interior densities of red oak, beech, basswood and total seedlings did not exceed those on controls, but were greater than those in trails. A similar situation occurred in sludged plots, with sludged interior densities of hophornbeam, beech, and total large seedlings exceeding trail densities in the same plots, but not in controls. Theoretically, the interior vs. trail density differences could be due to gains in interiors, to losses in trails, or to a combination of both (e.g. slight gains, perhaps in response to release or fertilization, when juxtaposed against moderate losses would together constitute a significant difference). Since in the 2 cases mentioned above the interior densities were not significantly greater than in controls, the data favor either of the latter choices. Particularly in the sludged plots, however, there is evidence of sludge damage, and the short time span (2-3 weeks) between sludge application and vegetation measures would not seem long enough for fertilizer-induced growth to have exerted much measurable effect in the interiors. This argues against widespread density gains in sludged interiors. Some apsects of the trails, whether construction, microenvironment, or use seem to have reduced stem densities. In the oak forest which received sludge in the fall of 1981 as a parallel to this study, 1982 trail seedling densities (0-1 m) were over 2 times greater in plots that did not receive sludge (Haufler et al., unpubl. data), which further supports the hypothesis that sludge applicator destruction was responsible for some of the reduced trail densities. This effect should be short-term and trail densities should recover, although not without initial competition from typically post-disturbance species (such as brambles, pin cherry, _Prunus pensylvanica_, and aspen, _Populus tremuloides_) (Auchmoody 1979, Bicknell 1982).

Final composition changes were produced, primarily by trail construction and to a lesser degree by sludge damage, in the herb-shrub segment of the plant community. Trails and sludged interiors possessed relatively sparser distributions of the typically deep woods herbs starflower and Canada mayflower, presumably due to microenviroment changes in the trails (perhaps increased light or temperature, or litter loss) and to sludge-induced damage or enviornmental changes in the interiors. The several-fold increase in occurrence of brambles in the trails was expected, since _Rubus_ spp. characteristically invade after recent disturbances. _Rubus_ seeds are usually present in the forest floor and are quick to take hold in an area whenever conditions become favorable (Stearns 1951). Common mullein, an early successional inhabitant of old fields, roadsides and waste places (U.S. Dept. of Agriculture 1971), was an anticipated invading species.

Plant damage by sludge application, both from the sludge itself and from vehicle use of the trails, along with mechanical removal of larger trees, also produced structural changes in the forest plant community. When data were combined to represent entire plots, large tree removal from the trails resulted in reduced cover above 2 m in both types of plots with trails, although these plots retained more than 90% of total available cover. Tree removal reduced trail canopy cover itself to about 65-75%, but trails made up only 25% of each plot's area. The 16-fold decrease in trail cover in the 30 cm - 2 m level can also be attributed to trail construction, presumably by physical damage, since it occurred in all trails regardless of treatment type. This trail effect accounts for the observed entire plot cover decrease in sludge plots. In the 10-30 cm cover stratum, which had low (less than 5%) cover throughout the forest, only sludge plot trails and interiors lost vertical cover. This reduced forest interior low level cover parallels the decreased woody stem density in the corresponding size class, and reflects sludge-induced leaf and plant mortality in the herb-shrub vegetation. Ultimately, cover reduction in sludge plots from the 3 strata above 10 cm was great enough in combination to drop foliage height diversity indices for sludge plot trails, interiors, and plots considered as a whole.

Small Mammal Response

Small mammal populations, as indicated by the August 1981 trapping data, were moderately high on the study site. The 2 species of Peromyscus and the eastern chipmunk were apparently the site's only common small mammal inhabitants, although the bait used may not have been attractive to shrews. The final day of trapping produced from 20% to 50% newly captured individuals, indicating that 1 trap per station was inadequate for capturing a majority of the site's inhabitants. The fact that Peromyscus numbers were 1/3 higher on controls plots than on either set of plots designated for treatment suggested inherent differences in pre-treatment mouse populations. In 1981, however, no habitat data were available for the area, so it was impossible to explain the mouse capture differences by any habitat discrepancies. No age or sex differences were found between mouse populations.

Trails were cut through both types of treatment plots in September, 1981. When mammal trapping resumed in May, 1982, numbers had equalized between the 3 types of plots but were exremely low. The drastically reduced 1982 populations followed an unusually severe winter, with record-breaking low temperatures coupled with heavier than normal snows in December and January (NOAA 1982). The extent to which this weather was responsible in reducing small mammal populations is unknown, but it is expected to have played a major part. The responsible factors appear to have been widespread, as southern Michigan Peromyscus populations also suffered a precipitous decline between 1981 and 1982 (Haigh, MSU personal comm.).

The July trapping period followed sludge application by less than 1 week, and much of the litter was still wet, with frequent pooling of sludge. Nevertheless, the treatment had no discernible effect on small mammal population numbers or their use of the area. Mammal numbers rose equally from their May figures on all 3 types of plots. There was no differential preference for trail or interior use of the habitat, as indicated by equal captures in both trap locations. In mid-August, however, sludged plots yielded significantly greater Peromyscus numbers (twice as large) than found in control or trail-only plots. There was insufficient time between sludge application and the August census (4-5 weeks) for this increase to reflect augmented mammal production in response to the treatment. The gestation period for both Peromyscus species is 21 days, but new mice do not usually leave their nests for the first 20 days of life and therefore would not be in the trapped population for at least 40 days following conception. Neither sex nor age ratios differed significantly between the 3 types of plots, although there were so few adults that such comparisons are suspect. Thus the increased mouse captures on sludged plots were presumably due to differential survival, or to a behavioral response by the animals to some change in their environment. An increase in survival might be attributed, for instance, to increased food availability. *Peromyscus* spp. are fairly strongly insectivorous (Van Horne 1982), and forest fertilization may result in increased litter invertebrate populations (Weetman and Hill 1973).

Neither species of *Peromyscus* demonstrated a greater population on a given type of plot than did the other, nor did either show a greater increase between trapping periods. Juvenile and subadult fractions of the trapped populations were slightly higher on sludged plots, but not significantly so. This argues against attributing the increase to differential habitat use patterns by young mice, as a result of niche displacement, which Van Horne (1982) described in *P. maniculatus* in the northwest. Trapped populations of all animals in 1982 comprised over 90% recaptures for all plots by the final day of trapping, indicating that the estimated populations were reliable for comparative purposes.

Most studies have failed to correlated *P. leucopus* and *P. maniculatus* spring and summer breeding with any identifiable, exclusively food-associated parameters, such as plant species diversity (Verts 1957, M'Colskey and Lajoie 1975). Jameson (1955) asserted that good mast years supported continued reproduction in *Peromyscus* through fall and into winter, which led to high populations in the following year. This would not, however, account for a differential increase in midsummer on some parts of the study site. Specifically food-associated variables other than net above-ground vegetative production, woody species density and herb-shrub frequency were not examined in this study. Density, frequency, and annual production each failed to demonstrate a positive response (e.g. increased production) confined exclusively to sludged plots. For the primarily granivorous-insectivorous *Peromyscus* species occupying this habitat, the measured first year vegetative changes occurring on the area, (sludged or otherwise) such as increased ash seedling density, would not be expected to provide significant additionaly food resources (the invading *Rubus* did not produce fruit in the 1982 season).

Several studies have looked for associations between *Peromyscus* numbers and habitat structural diversity or available vegetative cover at various heights; results provide conflicting evidence. *Peromyscus leucopus* numbers have correlated positively with vegetation density below 7.6 cm but not above it (M'Closkey and Lajoie 1975), indicating that the herb-shrub profile may provide an important component in habitat utilization. Drickamer (Unpubl. data) found a positive association between structural complexity canopy 1 m or more above traps, and *P. leucopus* numbers, while *P. maniculatus* numbers did not show any correlation with those variables. Verts (1957) found no association between P. maniculatus populations and cover or lack of it, while Lobue and Durnell (1959) and Miller and Gretz (1977) reported negative P. maniculatus population responses to herb cover in their habitat. Van Horne (1982) separated cover into several categories (e.g. tree cover less than 1.5 m, seedling cover less than 1.5 m, shrub cover less than 25 cm), and found positive correlations between adult P. maniculatus densities and all forms of low cover except seedling cover, which had a negative association with mouse densities. Juvenile densities correlated negatively with the cover variables measured. There is, then some evidence linking higher P. maniculatus numbers, in particular, to a paucity of low-level vegetative cover. In this study, both species of Peromyscus appeared to increase equally in numbers on sludged plots, and their combined numbers showed negative correlations with foliage height diversity and percent vertical cover in the 10-30 cm and greater than 2 m strata. This suggests that Peromyscus numbers may have been determined largely by the structure of their vegetative cover profile or a correlate of it. The strength of such an argument is somewhat weakened, however, by the 1981 mouse population figures. Any differences in habitat variables that might account for higher 1981 Peromyscus numbers on control plots than on plots designated to receive treatment were undocumented. All indications from 1982 vegetation measures suggest that for the vegetation variables measured, the study site was homogeneous prior to treatment. Dice (1931) and Fitch (1979) maintain that Peromyscus distribution is influenced by a behavioral habitat selection in reaction to visual and other stimuli associated with forests and prairies, and that a suite of interacting factors are probably responsible. Van Horne's (1982) and Drickamer's (1979; unpubl. data) data support this. The data from this study are not comprehensive enough over time or habitat measures to implicate any causal relationships between Peromyscus numbers and habitat characteristics. The results do indicate, however, that neither trail nor sludge treatment has a negative impact, in the sense of lowering a population even immediately post-treatment, on small mammal population numbers or distribution. This was consistent with the findings of other forest fertilization studies (Wood and Simpson 1973, Bierei et al. 1975, West et al. 1981, Woodyard 1982).

The appearance of jumping mice among captures in 1982 was not limited to either of the treatments or the controls, and therefore cannot be attributed to any known isolated habitat change. Studies of this animal's food habits (Whitaker 1963, Vickery 1978) indicate that they are similar to those of woodland deer mice, although jumping mice consume relatively fewer arthropods and more fungus. Therefore, there was no reason to suspect that the bait used in 1981 would not have attracted jumping mice, had they been present. The study site was altered in general character, however, by the creation of application trails and the wide, east-west access road between the northern study plots. Jumping mice are known to colonize recently cut, open woodlands (Krull 1970, Kirkland 1977), suggesting that this species may have immigrated onto the study area after it was opened up by trail construction.

Salamander Response

The red-backed salamender is an entirely terrestrial salamander, and as such was the only species expected to occur in workably high densities throughout the study area. In a similar forest in New Hampshire, it comprised 93.5% of total salamander biomass, while remaining species occurred only along streams (Burton and Likens 1975). In the present study, red-backed salamander populations demonstrated no response to sludge treatment. Post-sludge censusing began 3 weeks after sludge treatment, after much of the sludge had dried sufficiently to produce a surface crust, but large pools still remain. Unfortunately, census procedures were too labor-intensive to allow for censusing control plot populations in order to investigate trail treatment effects on salamander populations. The disproportionately high number of salamanders seen in forest interiors compared to those seen in trails suggested that application trails do affect above-ground, rainy night foraging distribution. Jaeger (1978) found that surface population densities of red-backed salamanders (on or above soil level) remain in a steady state, even when the litter is dry, and are not correlated with temperature, soil depth or litter depth. However, the percent of the surface population in the litter (rather than under rocks and logs) is positively and significantly correlated with rainfall. Studies have found that foraging success was high in wet periods, and that during rainfall the salamanders shifted their microhabitat use upward from the litter-soil interface and foraged instead on top of the litter and on vegetation (Burton and Likens 1975, Burton 1976, Jaeger 1978). Although the present study did not attempt to analyze salamander habitat components, it was noted that many of the salamanders observed in interiors were crawling on the lower (below 1 meter) portions of tree trunks. Since trails lacked any vegetation above 1 m, and all tree trunks were cut as close to ground level as possible, it may be that vegetation and tree removal, in particular, rendered application trails suboptimal above-ground foraging habitat for red-backed salamanders.

SUMMARY AND RECOMMENDATIONS

Sewage sludge disposal on forest lands offers an attractive solution to a serious waste management problem. It allows significant nutrient reclamation while potentially stimulating increased production of both forest and wildlife resources. The response of a forest ecosystem to sludge amendment appears to depend heavily on the age and type of forest, the method of application, and the timing of application.

This study examined the first-year response of a mature northern hardwoods forest to a single application of sludge, at commercial fertilizer loading levels, using a heavy equipment agricultural application technique. The technique required construction of a series of application trails throughout the forest, and this aspect of the treatment had significant effects on the forest that were independent of sludge effects. In the first growing season following trail construction, the trails appeared to alter plant community composition. They stimulated tree seedling and stump sprout regeneration, favored some post-disturbance understory species, and suppressed other typically deep forest understory species. Application trails, by their nature, altered forest structure by reducing canopy cover. Trails did not appear to affect small mammal population size or use of the area, but did appear to provide suboptimal foraging habitat for terrestrial salamanders.

Sludge applied during the growing season reduced vegetative vertical cover in lower strata and decreased seedling densities, through leaf and plant mortality. This appeared to be a short term response. Red-backed salamander populations showed no effect from sludge application. *Peromyscus* mice increased in number on sludged plots a month after sludge application, apparently due to a behavioral response; other small mammal populations showed no effects from the sludge treatment. Sludge and trail construction together reduced vegetative structural diversity, but neither treatment affected small mammal species diversity.

Other studies (Campa 1982, Woodyard 1982) have reported that sludge application to a young jack pine (*Pinus banksiana*) clearcut improved nutritive quality, digestibility, and productivity of wildlife forages, increased (small) mammal diversity, and may have accelerated ecosystem succession. Thus sludge amendment of forest soils can help achieve a variety of common wildlife management objectives.

The present study identified some immediate effects of sludge and its application using a specific technique, in a mature northern hardwoods forest. The application technique required cutting trails through the forest, which set back succession and favored plant species uncommon to woodlands in the narrow, localized strips. In addition, it opened the forest and apparently stimulated strong regenerative efforts by some dominant climax tree species while others suffered seedling density losses. Certain of the favored species, such as brambles and sugar maple stump sprouts, are highly preferred foods for browsing herbivores.

The immediate effects of sludge on vegetation were primarily destructive, causing understory stem density and cover losses, and combining with trail cutting to reduce vegetative structural complexity. Such damage to existing plants by the sludge itself may be avoided by applying sludge when plants are dormant, such as autumn or early spring. Over time, forest fertilization may lead to increased net primary production, encouraged in addition by forest thinning such as that achieved by the sludge application trails. In a mature forest such as this one, fertilization and/or thinning may not change timber production appreciably, (Ellis 1979, Koterba et al. 1979, Sopper and Kardos 1979, Stone et al. 1982), but fruit and seed production may increase (Weetman and Hill 1973, Daniel et al. 1979) and some understory plants may flourish (Anthony and Wood 1979, Auchmoody 1979, Koterba et al. 1979). Such changes may alter animal community composition, and/or support its increased production.

Any potential wildlife and forest resource benefits from these changes must be evaluated in light of the costs associated with the application technique, which is costly both in terms of equipment and site preparation. Future research should compare the relative costs and benefits of this application procedure with a more economical, less radical one, such as one using heavy irrigation spray equipment.

The potential bioaccumulation and toxicity and food chain transfer of sludge constituents in forest flora and fauna, including litter-dwelling invertebrates and herpetofauna, are virtually unknown. These topics need study before the environmental soundness of widespread sludge application to forests can be evaluated.

Forest land application of sewage sludge is a feasible habitat management and waste disposal technique. However, mature forests present technical challenges to application which need further experimentation and assessment. The benefits derived from sludge fertilization and forest thinning may not outweigh the costs of the procedure investigated in this study, for most wildlife management objectives. Sludge is, however, a relatively inexpensive, readily available fertilizer which presents a potentially useful habitat management tool. It may be used, in various situations, to increase forage production and nutritive quality, to alter ecosystem succession, and to change plant and animal community composition and structure.

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APPENDIX

APPENDIX

List of vascular plants on the northern hardwoods study site, Montmorency Co., MI, summer 1982.

Common Name	Scientific Name
Indian cucumber-root	Medeola virginiana
Wild sarsaparilla	Aralia nudicaulis
Spikenard	Aralia racemosa
Bellwort	Uvularia grandiflora
Hooked crowfoot	Ranunculus recurvatus
Sweet cicely	Osmorhiza claytonii
Trillium	Trillium grandiflorum
Starflower	Trientalis borealis
Bedstraw	Galium aparine
Woodland strawberry	Fragaria vesca
White baneberry	Actaea pachypoda
Fringed polygala	Polygala paucifolia
Violet	Viola spp.
Brambles	Rubus spp.
Columbine	Aquilegia canadensis
Small Solomon's seal	Polygonatum pubescens
False Solomon's seal	Smilacina racemosa
Canada mayflower	Maianthemum canadense
Waterleaf	Hydrophyllum virginianum
Common mullein	Verbascum spp.
Rattlesnake plantains	Goodyera spp.
Indian pipe	Monotropa uniflora
Pyrola	Pyrola spp.
Pipsissewa	Climaphila umbellata
Wintergreen	Gaultheria procumbens
Partridgeberry	Mitchella repens
Enchanter's nightshade	Circaea quadrisulcata
White lettuce	Prenanthes alba
Heart-leaved aster	Aster cordifolius
Dogbane	Apocynum spp.
Willow-herb	Epilobium spp.
Hop clover	Trifolium agrarium
Troup lily	Eurythronium americanum
Sedge	Carex spp.
Leather wood	Dirca palustris

APPENDIX (cont'd.)

Common Name	Scientific Name
Eastern hophornbeam	Ostrya virginiana
American beech	Fagus grandifolia
American basswood	Tilia americana
Sugar maple	Acer saccharum
Red maple	Acer rubrum
Striped maple	Acer pensylvanicum
White ash	Fraxinus americana
Eastern hemlock	Tsuga canadensis
Red oak	Quercus rubra
Paper birch	Betula papyrifera
Yellow birch	Betula lutea
Viburnum	Viburnum spp.
Juneberry	Amelanchier spp.
Pin cherry	Prunus pensylvanica
Aspen	Populus spp.
Balsam fir	Abies balsamea
Dogwood	Cornus spp.