Lehigh University
Lehigh Preserve

Theses and Dissertations

1-1-1984

Community structure variations in an eastern deciduous forest contaminated with zinc and cadmium.

Marianne Burke

Follow this and additional works at: http://preserve.lehigh.edu/etd

 Part of the Biology Commons

Recommended Citation
Burke, Marianne, "Community structure variations in an eastern deciduous forest contaminated with zinc and cadmium." (1984). *Theses and Dissertations.* Paper 2225.

This Thesis is brought to you for free and open access by Lehigh Preserve. It has been accepted for inclusion in Theses and Dissertations by an authorized administrator of Lehigh Preserve. For more information, please contact preserve@lehigh.edu.

COMMUNITY STRUCTURE VARIATIONS IN AN EASTERN DECIDUOUS FOREST CONTAMINATED WITH ZINC AND CADMIUM

by

Marianne Burke

A Thesis

Presented to the Graduate Committee

of Lehigh University

in Candidacy for the Degree of

Master of Science

in

Biology

Lehigh University

1984

ProQuest Number: EP76501

All rights reserved

INFORMATION TO ALL USERS
The quality of this reproduction is dependent upon the quality of the copy submitted.

In the unlikely event that the author did not send a complete manuscript
and there are missing pages, these will be noted. Also, if material had to be removed,
a note will indicate the deletion.

ProQuest EP76501

Published by ProQuest LLC (2015). Copyright of the Dissertation is held by the Author.

All rights reserved.
This work is protected against unauthorized copying under Title 17, United States Code
Microform Edition © ProQuest LLC.

ProQuest LLC.
789 East Eisenhower Parkway
P.O. Box 1346
Ann Arbor, MI 48106 - 1346

This thesis is accepted and approved in partial fulfillment of the requirements for the degree of Master of Science.

12/17/84
date of approval

Professor in charge

Chairman

Committee

ACKNOWLEDGEMENTS

Thanks are due to the landowners who generously allowed the use of their land for this study, and to John Washington from the Pennsylvania State University for allowing a preview of his unpublished data and zinc deposition model, and for his helpful discussions.

I am grateful to my advisor Jon I. Parker, and committee members David L. Cundall, Steven Krawiec, Richard G. Malsberger and Craig E. Williamson for their assistance and guidance during the study.

I am also grateful to Robert R. Kendi and Monica A. Newman for their computer assistance which went above and beyond the call of duty, to Howard K. Reinert for his helpful discussions on statistical analysis, to Lehigh University's Energy Research Center for summer fellowships, and to Lehigh University's Biology Department and Center for Marine and Environmental Studies for supplies.

A special thanks is extended to H. Joan Burke and Alexander M. Burke for their major contributions of gracious and unwavering support throughout the study.

TABLE OF CONTENTS

List of tables

List of figures

ABSTRACT

Variation in the community structure and soil chemistry of an eastern deciduous forest was examined and related to contamination from zinc smelters at Palmerton, Pa. Vegetation measurements and soil samples were collected on a nearby ridge at fifteen sites with similar geology, climate, exposure, aspect, elevation and history of disturbances, but at varying distances from the smelters. A species diversity index, an importance index and univariate and multivariate statistics were used to identify vegetation patterns. Abnormally high pH and percent base saturation, and abnormally low cation exchange capacity in the A1 soil horizons were associated with the areas of greatest contamination by zinc and cadmium. Patterns in species diversity were not conclusive, and were most dependent on sample size. Although total basal area did not vary significantly between any sites, stem density was significantly higher near the source of contamination. Variation in community structure is most closely associated with contamination. Rather than a clear exclusion of intolerant species, there appears to be an increase in density or basal area of chestnut oak, red maple and black gum, all of which may be tolerant to conditions caused by the contamination.

INTRODUCTION

Air pollutants are suspected causes of reduced stability and production in forest ecosystems. Forest decline adjacent to point emmission sources has been well documented (Gordon and Gorham 1963, Freedman and Hutchinson 1980, Rosenberg et al. 1979), and a growing body of evidence now suggests that forest decline in remote areas is related to deposition of atmospheric pollutants (Knabe 1976, Johnson et al. 1981, Ember 1982, Puckett 1982, Tomlinson and Silversides 1982, Johnson and Siccama 1983, Tomlinson 1983).

Atmospheric pollutants which are suspected of causing forest decline include acids, ozone and trace metals. Anthropogenic emmissions of zinc, cadmium and other heavy metals began in the 1880's (which correlates with the time of increased fossil-fuel consumption) and have increased to the present (Moore and Ramamoorthy 1984). Reported zinc concentrations in bulk deposition range from 0.378 ppm (in pre-1900 dated samples from a Greenland glacier) to 280 ppm for New York City in 1970 (Nriagu and Davidson 1980). Concentration patterns indicate that many pollutants may be transported considerable distances (Reiners et al. 1975, Hanssen et al. 1980) and can accumulate in soils (Hanson et al. 1982, Reiners et al. 1975). Zinc concentrations in for-

est floor samples from Virginia through Massachusetts were found to vary from 15.5 to 2279 ug/g (Andresen et al. 1980).

Air pollution may cause specific physiological, physical and chemical changes in trees. These include reduced rate of growth, browning and loss of foliage (Raynal et al. 1980, 1982, Johnson and Siccama 1983), reduced radial growth rate (Nash et al. 1975, Strand 1980, Johnson et al. 1981, 1982, Puckett 1982), change in percent earlywood and latewood (Lawhon and Woods 1976, Strand 1980), reduced cone size and weight (Smith 1981), reduced rooting depth, increased susceptibility to drought, decreased ability to use subsoil nutrients, induced mineral deficiencies (Foy et al. 1978), and increased susceptibility to bacterial infection (Raynal et al. 1982).

Forest communities in severely contaminated areas may change in species composition, species morphology, diversity, and production. Approaching major contamination sources in previously forested areas, the community changes from apparently unaffected forest, to areas of abundant trees with foliar damage, to areas of trees with grass and scrub, to areas of grass and scrub, to denudation in the highly contaminated area (Guderian 1977, Woodwell 1970, Nash 1975, Freedman and Hutchinson 1980). Other forest community

changes related to high levels of contamination are reduced species diversity (Freedman and Hutchinson 1980, but see Rosenberg et al. 1979), reduced annual growth increment (Nash et al. 1975, Baes and McLauphlin 1984, Stone and Skelly 1974), and a reduction in tree density (Freedman and Hutchinson 1980, Buchauer 1971, 1973, Jordan 1975).

This study examines forest community responses to contamination from zinc and cadmium particulates emitted by a smelting complex in Palmerton, Pennsylvania. The smelters are owned and operated by New Jersey Zinc Company and are located just north of the Lehigh River Gap in the southernmost ridge of the Appalachians (Figure 1).

The smelters (which have been in operation since 1898) are a major pollution source in the area (Buchauer 1971, 1973, Jordan 1975, Jordan and Lechvalier 1975, Nash 1975, Strojan 1978a,b). Large amounts of particulates rich in zinc and cadmium are dispersed in a predominantly southeastern direction. Concentrations of zinc and cadmium (respectively) in Palmerton were estimated at 135,000 ppm and 1750 ppm, in the organic horizons (Buchauer 1973), and 320 ppm and 12.4 ppm in the mineral soil (Washington, pers. comm.). These values compare with backround levels of 25 ppm for Zn and 6 ppm for Cd (Buchauer 1973). Elevated metal levels

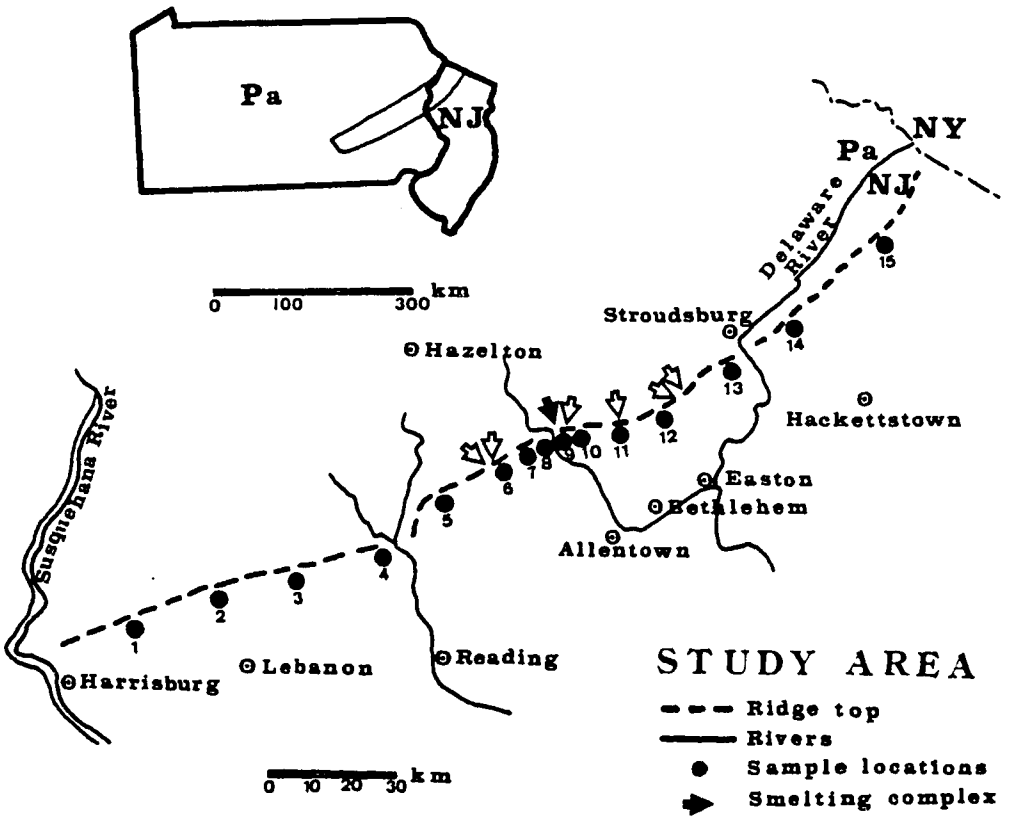

Figure 1. Study area and sample locations. Open arrows indicate ridgetop sites of vegetation sampling by Buchauer (1971) and Jordan (1975).

were detected 25 km east and 16 km west from the smelter for organic horizon (Buchauer 1973) and 28 km east and 7 km west from the smelter for the mineral horizon (Washington, pers. comm.).

Sulfation boundaries around the smelter were described by Nash (1975). Sulfation rates between 9 and 21 ug SO4/cm2/day occurred along an 8 km region of the north slope of the ridge near Palmerton. Consistent clean air values (<2.0 ug SO4/cm2/day) were measured beyond a 17 km length of the north slope of the ridge, and beyond 2 km south of the smelters, which coincides approximately with the top of the ridge. Low sulfation rates near the smelter are attributed to an efficient acid recovery plant at the smelting complex, and to the topography which prevents the spread of emitted sulfates (Nash 1975).

Additional sources of air pollution near the ridge include (from west to east) the urban-industrial areas at Harrisburg, Lebanon, Reading, Allentown, Bethlehem and Easton, and coal-fired electric power plants south of the Delaware Water Gap (Figure 1).

This region of the southernmost ridge of the Appalachians (Blue Mountain and Kittatiny Ridge) is ideal for forest community studies because it contains uncontaminated areas as

well as areas which are highly contaminated with zinc and cadmium. Further, it is possible to find numerous sites along the ridge that have similar slope, topography, aspect, geology, and climate.

The present structure of the forest community is presumed to be a dynamic state representing long-term response to fire (Buchauer 1971, Jordan 1975), logging (Lappin 1973), deposition of atmospheric pollutants, chestnut blight (Endothia parisitica) (Braun 1950), and regional severe gypsy moth (Lymantria dispar) infestation.

Previously documented ecological responses to the contamination near the Lehigh Gap include changes in forest species abundances (Buchauer 1971, Jordan 1975) and lichen communities (Nash 1975), as well as reductions in soil microflora (Jordan and Lechvalier 1975), rates of litter decomposition (Strojan 1978a), and litter arthropod numbers (Strojan 1978b).

Rather than examining communities along the ridgetop, as was done by Buchauer (1971) and Jordan (1975) (Figure 1), this study examines the influence of the smelter complex on the communities of the southfacing mid-slope. The combined effects of proximity to a contaminated source and deposition patterns on community structure are examined in this study.

A number of contaminated as well as uncontaminated sites were analyzed, and forest diversities, species composition and production are related to both metal and nutrient concentrations in the soil.

The hypotheses tested are that species diversity, species abundances, species composition, production and the elemental nutritional content of the soil are influenced by heavy metal contamination.

METHODS

Study area

The study was conducted at fifteen sites (Figure 1) on the southernmost ridge of the Appalachian Ridge and Valley province in Pennsylvania and New Jersey. The ridge, (called Blue Mountain in Pennsylvania and Kittatiny ridge in New Jersey) lies on an almost straight compass bearing of 40 degrees north of east and provides many sites with similar aspect, slope, geology and climatic conditions.

Geologically, the sites are characterized by sandstone and conglomerate with thin shale interbeds, except at the westernmost site where shale with limestone is also found (Gray and Shepps 1960). The ridge was covered by the first (Kansan) glacier. Presently, shallow channery soils predominate on most of the ridge and colluvial material increases in abundance toward the base of the ridge (Buchauer 1971).

Procedure

The forest was studied to determine if variation in community structure could be associated with pollution sources. The forest was sampled along a 200 kilometer area of the ridge, and fifteen sites of 0.15 ha each were established on the south slope of the mountain. Site locations 1 to 7 lay

west of the smelters, sites 8 to 11 lay within the zone known to receive heavy metal contamination from the smelters (Washington, pers. comm.) and sites 12 to 15 lay east and northeast of the heavily contaminated area.

All site locations were verified with 7.5 minute topographic maps and had an elevation of 300 meters, a south southeast aspect, a 12% slope (clinometer measured) and no visible signs of recent logging or fire damage.

All woody stems were tallied and identified to species. If they exceeded 1.2 meters in height, their diameter (DBH) was measured. Basal area was calculated from this measurement and densities were calculated from the tallies. All sampling was completed during June and July of 1983.

Soil pH, cation exchange capacity and percent base saturation were measured at the Pennsylvania State University Soil and Environmental Chemistry Laboratory from two samples of both the A1 and A2 horizons taken at random locations along each of the 15 transects. Measurements are based on the methods of Baker and Amacher (1981). Heavy metal concentrations were also measured at the lab for one A1 horizon sample from sites 1, 8, 9, 10, and 15. The procedure for metal analysis is that described in Hinish et al. (1967).

Species area curves (Cain 1938) were used to determine

the quadrat at which minimal adequate sampling occurred. The modified method proposed by Oosting (1956) for intensive sampling (5% relationship) was used. Relative density, dominance, frequency and importance values (Curtis and McIntosh 1951) were calculated for all sites in the uncontaminated areas, and for all sites in the area contaminated with metals.

Species diversity was measured in three ways. The number of species per quadrat (0.01 ha) was termed "alpha diversity". The number of species per transect (0.15 ha) was termed "species richness", and the Shannon-Wiener index (Shannon and Weaver 1949) was calculated for each transect. Mean alpha diversity and mean values for the Shannon-Wiener index values were calculated for the eleven uncontaminated and four contaminated sites.

Pearson's product-moment correlation (Sokal and Rohlf 1981) was performed on transect means for all possible pairs of soil and vegetation variables. Means were used because sample sizes for vegetation and soil variables were not equal.

Oneway analysis of variance was used to test for significant differences between transects for vegetation features. Analysis of variance was not conducted on the soil variables because the two samples from each site were not considered

adequate. In this and all subsequent analyses all data were transformed by adding 0.1 to all values. This prevented samples with zero values for a variable from being deleted from multivariate analyses, thus permitting maximum utilization of the available data.

Cochran's C test was used to test for departures from homogeneity of variance, although F tests are known to be robust with respect to departures from homogeneity of variance (Winer 1971). Transect means with significant F values were subjected to LSD (Least Significant Difference) multiple range tests to determine which transects were significantly different.

Stepwise discriminant function analysis (Rao's V) was used to identify linear combinations of vegetation variables which are most effective in discriminating between transects (Kleka 1975, Pimental 1979). In performing this analysis, cases were individual quadrats at each site and groups corresponded to sites. Group centroids are used to illustrate separation in discriminant space. Because many of the uncontaminated sites were predicted to have similar features, attempts to define linear combinations of variables that discriminate between all sites would not necessarily select variables associated with contamination, unless those variables represented a major component of total variance in

the data.

Because the total number of variables far exceeded the number of cases in each group, entry criteria for selection of variables were increased to reduce the total number of variables ultimately used in calculating the discriminant scores. As in other stepwise procedures, this does not necessarily result in the selection of those variables contributing to the greatest possible discrimination between groups because all possible subsets of variables are not examined. However, the approach was necessary to prevent the group covariance matrices from being singular. The correlation between individual variables and each discriminant function was used to evaluate the contribution of each variable to separation of transects along each discriminant axis. The probability of correct group assignments within individual cases (quadrats) was also used to evaluate the discrimination value of canonical variate for the first three discriminant functions.

Principle component analysis was used to gain a measure of the distribution of sites based on the total variance represented by the included variables. Because transformed linear and interval data were entered in the analysis, factors and factor scores were derived from the correlation matrix. Biological characteristics of the first three prin-

ciple components were based on correlations between the original variable and each principle component.

Because neither the Discriminant Function Analysis nor the Principle Component Analysis provide statistics to test for differences between groups, means of variables for transects within the contaminated area (as defined by Washington, pers. comm.) were compared with uncontaminated transects. To maintain the same sample sizes, the four transects in the contaminated area (8, 9, 10 and 11) were compared with the four westernmost transects (1, 2, 3, and 4) and four easternmost transects (12, 13, 14 and 15). As in the oneway analysis of variance by site, Cochran's C was calculated to test for homogeneity of variance. LSD was used to determine which of the three categories were significantly different.

Statistical analyses were performed on a Cyber 720 using SPSS (Statistical Package for the Social Sciences, Nie et al. 1975, Hull and Nie 1981), and BMDP (Biomedical Computer Programs, Dixon and Brown 1977).

RESULTS

Trace metal analyses of the soil samples showed high levels of zinc and cadmium at sites 8, 9, and 11, and low to normal concentrations at sites 1 and 15. Concentrations of other metals were low at all five sites, with generally lower concentrations of aluminum, iron and manganese at sites most heavily contaminated by zinc and cadmium (Table 1).

Soil pH at sites contaminated with trace metals showed abnormal profile patterns. Normally, pH is higher at the subsurface (A2) than at the surface (A1) horizons where production of acids occurs during decomposition. The normal pattern was shown for all sites which were not contaminated by metals except site 1. Mean pH values for the uncontaminated sites are 4.09 in the A1 horizon and 4.79 in the A2 horizon. Metal contaminated sites show a distinct reversal of the normal profile pattern with mean pH values of 5.4 in the A1 horizon and 5.06 in the A2 horizon (Figure 2).

Cation exchange capacity at sites contaminated with trace metals also showed abnormal profile patterns. Cation exchange capacity is usually greater in the A1 (due to the abundance of organic matter) than the A2 horizon. All uncontaminated sites showed this pattern, with mean values

Table 1. Metal concentrations (ppm) for A1 horizon soils at five sample locations.

	Sites				
metal	1	8	9	11	15
Manganese	84.0	6.9	1.7	19.4	42.0
Iron	72.0	77.0	2.4	120.0	162.0
Copper	2.1	0.8	0.3	0.9	0.9
Zinc	31.0	191.0	440.0	77.0	3.2
Sodium	26.4	27.2	26.7	27.6	26.9
Aluminum	26.8	2.5	0.6	14.0	14.9
Lead	2.5	1.0	0.2	8.1	0.8
Nickel	2.1	0.2	0.7	0.2	0.1
Cadmium	0.4	1.2	2.9	1.5	0.1

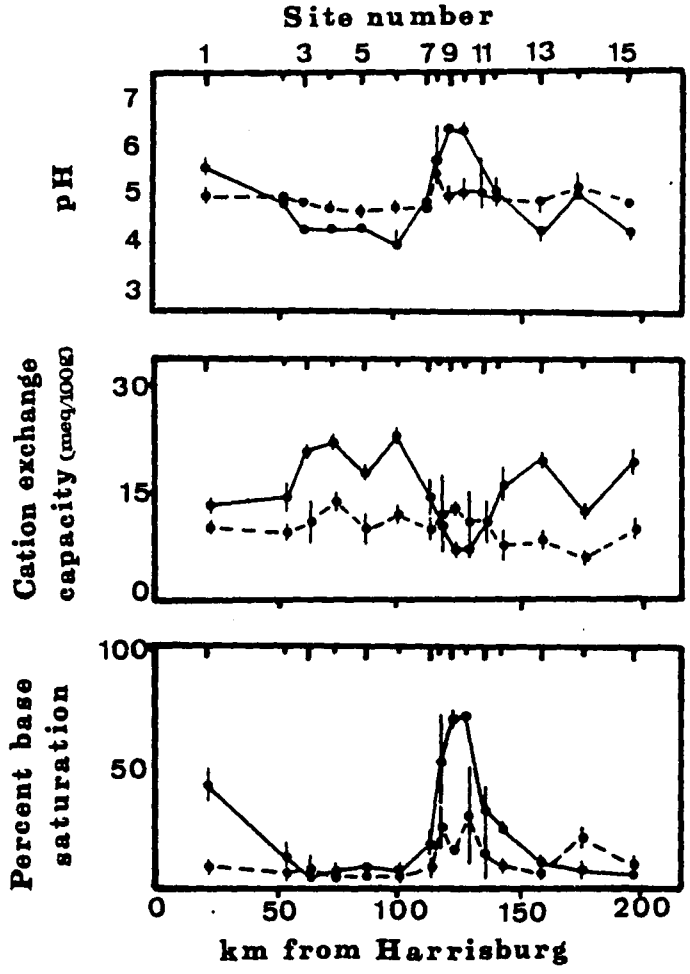

Figure 2. Mean soil pH, cation exchange capacity and percent base saturation in the A1 (———) and A2 (– – –) horizons at the fifteen sample locations. Vertical lines represent standard error.

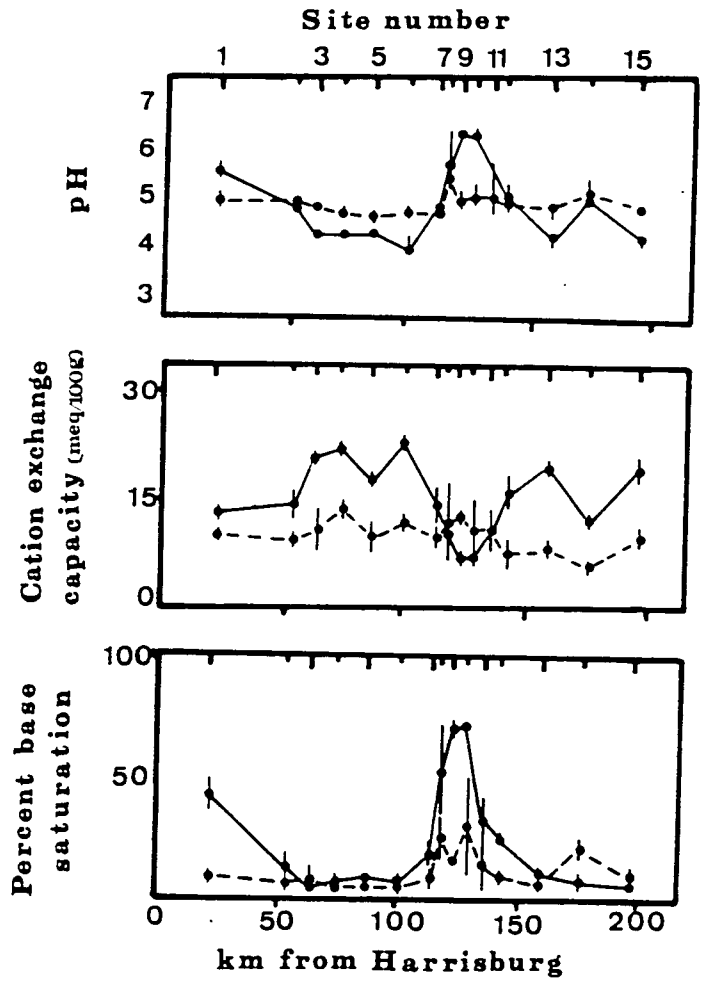

Figure 2. Mean soil pH, cation exchange capacity and percent base saturation in the A1 (———) and A2 (– – –) horizons at the fifteen sample locations. Vertical lines represent standard error.

of 17.62 meq per 100g in the A1 horizon and 10.02 meq per 100g in the A2 horizon. Metal contaminated sites show a distinct reversal with values of 9.39 meq per 100g in the A1 horizon and 11.65 meq per 100g in the A2 horizon. A highly significant ($p < 0.01$) negative correlation exists between pH and cation exchange capacity in the A1 horizon ($r = -0.802$) (Figure 2).

Percent base saturation is greater in the A1 than in the A2 horizon at the uncontaminated sites, but the reverse is true in the contaminated sites. At the uncontaminated sites, mean percent base saturation is 12.66 in the A1 horizon and 8.81 in the A2 horizon. Percent base saturation is highly correlated with pH in the A1 horizon in the contaminated ($r = 0.99$, $p < 0.01$) and uncontaminated sites ($r = 0.80$, $p < 0.01$) and with pH in the A2 horizon at the uncontaminated sites ($r = 0.56$, $p < 0.05$). Percent base saturation is not correlated with pH in the A2 horizon of the contaminated sites ($r = 0.50$) (Figure 2).

Importance values (Tables 2 and 3) show the uncontaminated area is a red maple - mixed oak - sweet birch forest, and the contaminated area is a black gum - red maple - chestnut oak forest. This difference in community structure is due to greater relative density and dominance of chestnut

Table 2. Relative density, dominance, frequency and importance values for tree species in the uncontaminated area.

Species	Density	Dominance	Frequency	Importance
Red maple	26.4	17.3	100.0	143.8
Red oak	8.0	28.4	100.0	136.4
Sweet birch	17.0	15.7	100.0	132.7
Chestnut oak	7.6	16.6	100.0	124.2
Witch hazel	13.7	3.0	81.8	98.5
Black gum	10.6	1.8	81.8	94.2
Hickories	3.0	4.7	81.8	89.5
Sassafras	3.4	2.6	81.8	87.8
American chestnut	2.7	0.2	81.8	84.7
Dogwood	1.0	0.1	81.9	82.9
White oak	0.6	1.4	63.6	65.5
Yellow poplar	1.5	4.7	45.4	51.5
White pine	0.3	9.8	36.4	46.5
Choke cherry	0.4	0.1	45.4	45.9
American beech	4.4	0.6	36.4	41.3
White ash	0.9	0.4	36.4	37.7
Buckthorn	1.0	0.1	36.4	37.5
Striped maple	1.1	0.4	27.3	28.8
Big-tooth aspen	0.2	0.5	27.3	28.0
Sugar maple	0.2	4.3	9.1	13.6
Hornbeam	0.3	0.01	9.1	9.4
Butternut	0.03	0.1	9.1	9.2

Table 3. Relative density, dominance, frequency and importance value for tree species in contaminated sites.

Species	Density	Dominance	Frequency	Importance
Black gum	27.2	1.8	100.0	147.8
Red maple	26.4	17.3	100.0	141.2
Chestnut oak	10.6	30.5	100.0	141.1
Red oak	5.9	16.3	100.0	122.2
Sweet birch	10.2	10.5	100.0	120.7
Sassafras	3.6	3.7	100.0	107.4
Witch hazel	4.8	3.4	75.0	83.4
American chestnut	1.1	0.2	75.0	76.3
Dogwood	5.6	1.4	50.0	57.0
White pine	0.7	4.0	50.0	54.6
Choke cherry	0.4	0.03	50.0	52.5
White oak	0.5	1.2	50.0	51.7
Striped maple	0.6	0.1	50.0	50.7
Big-tooth aspen	0.1	1.4	25 0	26.5
White ash	0.1	0.2	25.0	25.3
Paper birch	0.2	0.02	25.0	25.2
Pin cherry	0.1	0.04	25.0	25.1
Butternut	0.1	0.02	25.0	25.1
Buckthorn	0.1	0.01	25.0	25.1
Hickories	0.1	0.01	25.0	25.1

oak and black gum, as well as greater relative frequency of black gum in the contaminated compared to the uncontaminated areas. Sweet birch and red oak have lower relative density and dominance in the contaminated area compared to the uncontaminated area.

Species area curves showed sampling for species was adequate at each site (Table 4). Although no clear trend for species richness can be related to contamination (Table 5), the site with the highest recorded soil zinc and cadmium concentrations (9) has the lowest values for species richness, and the mean value for the contaminated area is lower than the mean value for the uncontaminated area. However, alpha diversity is greatest at the most contaminated sites. The Shannon-Wiener index does not show trends related to contamination.

Although basal area was not significantly different between the easternmost, westernmost and contaminated categories, or between sites, total density was significantly ($p < 0.001$) greater at the contaminated sites and variance was homogeneous. Analysis of variance and the multiple range test by site showed significantly greater ($p < 0.001$) numbers of stems at sites 8, 9 and 10. However, variance was significantly ($p < 0.05$) heterogeneous. Analysis of

Table 4. Results of modified method of using species area curves to determine the minimum number of 10 x 10 meter quadrats needed for adequate species sampling.

Site number	Quadrat number where adequate sapling occurred
1	9
2	12
3	6
4	6
5	6
6	4
7	8
8	7
9	6
10	9
11	6
12	6
13	8
14	8
15	7

Table 5. Diversity measurements for the fifteen sites in the study area.

Site	Species Richness	Mean Alpha Diversity	Shannon-Wiener Index
1	12	3.9	2.06
2	15	6.0	1.90
3	15	5.5	1.77
4	13	5.4	1.58
5	12	6.8	1.88
6	14	6.5	2.11
7	14	6.3	1.68
8	11	6.0	1.58
9	9	5.7	1.72
10	15	6.9	2.00
11	11	6.8	1.87
12	13	6.4	1.84
13	17	9.1	2.60
14	15	3.7	2.50
15	14	6.1	1.78
means			
Contaminated	11.5	6.4	1.79
Uncontaminated	14.0	5.95	1.97

variance of species variables showed significant differences between sites, but no pattern associated with soil was identified.

Discriminant Function Analysis demonstrates that no linear combination of variables accounted for more than 32.14% of total variance. This suggests that no uniform trend in change of community structure exists at the sites sampled. The limits of the first discriminant function are defined by the transect means (group centroids) of sites 1 and 10 (Figure 3-A). The majority of the uncontaminated sites lie closer to site 1 than site 10, and the two sites closest to the smelter (8 and 9) are placed between site 10 and the edge of the cluster of uncontaminated transect means. Sites marginal to the highly contaminated sites (7 and 11) also have transect means closest to the contaminated site means along the first discriminant function. Discriminant scores for the fifteen quadrats of each transect show a large amount of dispersion around each mean, although only means are illustrated in figure 3-A.

The first discriminant function is most highly correlated with the densities of dogwood, red maple and chestnut oak, and the basal area of black gum. The second discriminant function is most highly correlated with the density of dog-

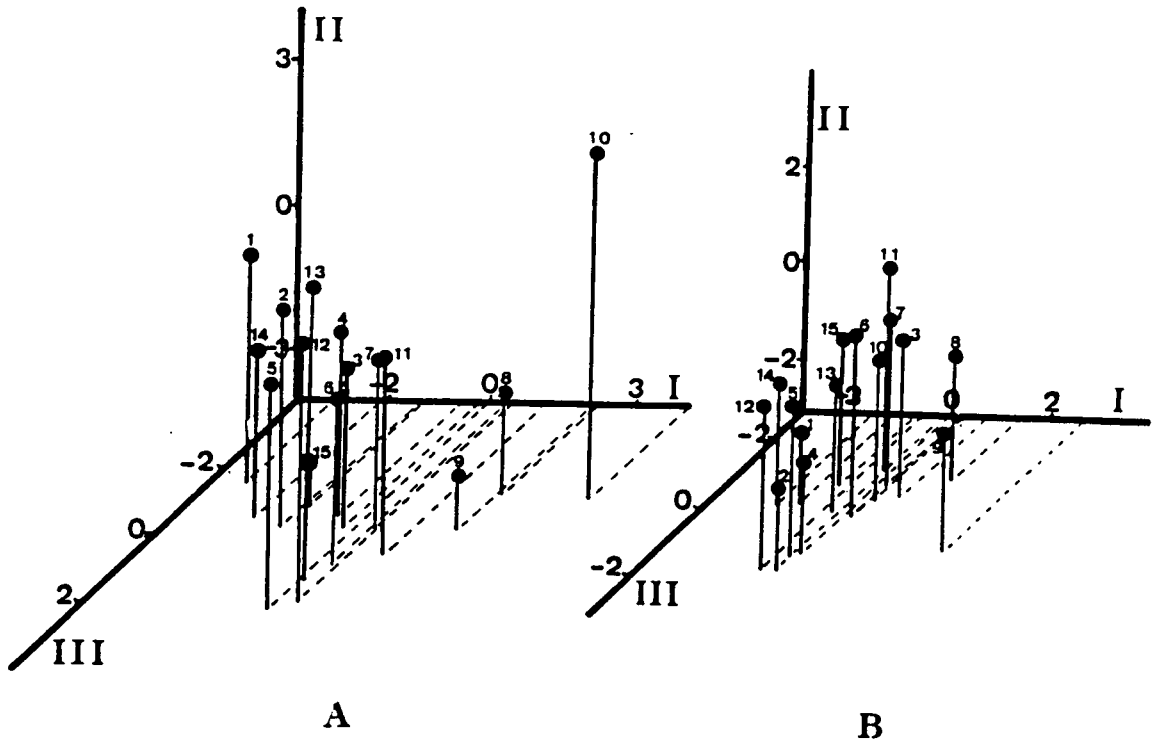

Figure 3. Three-dimensional graph of centroids for the first three Discriminant Functions (A) and factor scores for the first three Principal Components (B) for the fifteen sample locations.

wood and is negatively correlated with density of chestnut oak. The third discriminant function is most highly correlated with the density of witch hazel and sweet birch, and with the basal area of chestnut oak (Table 6). Table 7 presents the results of an analysis of variance and least significant difference test on the fourteen variables identified as most discriminating.

Principal component analysis demonstrates that no linear combination of the fourteen most discriminating variables account for more than 25.41% of the total variance for those variables. This further suggests there is no uniform trend in the change of community structure, but rather a complex continuum exists.

The limits of the first principal component are defined by sites 14 and 9 (Figure 3-B). Site 8 lies between site 9 and the cluster of uncontaminated site means. Site 10, which was extreme on the first discriminant axis, is not separated from the group of uncontaminated sites in the principal component analysis. The first principal component is most highly correlated with the densities of chestnut oak and red maple, basal area of chesnut oak, and with alpha diversity. The second principal component is most highly correlated with the basal area of black gum. The third

Table 6. Correlations between canonical discriminant functions and discriminating variables for the first six functions.

Variable	Functions					
	1	2	3	4	5	6
Red maple density	.3314	-.1474	.1451	-.0545	.1563	-.1888
Alpha diversity	.2558	.0476	.2380	.0605	-.0990	.1288
Dogwood density	.4582	.7314	-.0852	-.2620	.2168	-.2082
Witch hazel density	.0419	.1540	.3958	.3676	-.4835	-.0851
Dead basal area	-.0009	.0075	.1478	-.4230	.3550	.2700
Dead density	-.1185	.1948	-.3826	.2530	.3220	.0308
Sassafras density	.1095	-.1088	.0001	.3350	.1321	-.2000
Sweet birch density	.0948	-.1829	.4483	.2280	.4835	.0272
Sweet birch basal area	-.0428	.0783	.2025	.3954	.2166	-.0670
Chestnut oak basal area	.0826	-.0106	.1477	.0609	-.3168	-.1707
Red maple basal area	.0674	-.1714	.2198	.0694	.2574	.2170
Black gum basal area	.3059	-.1488	-.3458	.0477	-.4182	.7062
Chesnut oak density	.2800	-.2519	-.1141	.1337	-.1991	-.4190
White ash density	-.0849	.0408	-.0941	-.1335	-.0899	-.1868
Eigenvalues	3.15	2.01	1.20	1.00	0.73	0.61
% variance accounted for	32.14	20.55	12.20	10.26	7.42	6.23

Table 7. Results of LSD multiple range test on variables selected by discriminant function analysis. Means and standard errors by sample locations for 14 most discriminating variables, arranged in order of increasing mean values. Sample locations which are not significantly different are connected by horizontal lines.

Red maple density														
1	14	13	12	2	5	6	8	4	7	11	15	10	3	9
0.43	0.43	2.70	4.23	4.43	4.57	4.70	5.37	5.77	5.97	6.77	7.70	8.17	8.17	9.83
0.16	0.16	0.76	0.58	0.68	0.57	0.64	1.05	0.75	0.89	1.07	1.15	1.48	1.42	1.33
Alpha diversity														
14	1	4	3	13	9	8	2	15	7	5	12	6	11	10
3.83	4.03	5.57	5.63	5.70	5.83	6.10	6.10	6.17	6.38	6.43	6.50	6.63	6.90	6.96
0.33	0.25	0.34	0.44	0.50	0.25	0.31	0.46	0.40	0.27	0.61	0.30	0.26	0.29	0.24
Dogwood density														
9	8	15	1	13	2	3	5	6	14	7	4	11	12	10
0.10	0.10	0.10	0.10	0.17	0.17	0.17	0.17	0.23	0.23	0.23	0.23	0.63	1.03	5.57
				0.07	0.07	0.07	0.07	0.13	0.09	0.09	0.09	0.19	0.34	0.90
Witch hazel density														
9	14	1	15	2	4	13	8	10	6	7	11	3	5	12
0.10	0.10	0.10	0.17	0.17	0.23	0.37	1.03	1.43	2.63	2.83	2.97	3.03	6.37	8.37
			0.07	0.09	0.15	0.45	0.45	0.47	1.08	0.75	0.82	1.19	1.04	2.13
Dead basal area														
14	5	11	9	6	3	8	12	13	4	2	7	1	10	15
286.62	300.86	316.07	339.48	340.79	352.58	389.77	406.21	490.60	606.68	654.02	662.72	663.80	710.46	1361.20
282.34	74.60	126.05	48.44	40.60	56.12	76.75	159.09	99.97	102.12	76.28	83.67	109.49	134.46	246.09
Dead density														
15	14	11	6	5	9	8	7	13	12	4	10	3	2	1
2.83	5.10	5.17	5.70	6.03	7.30	7.43	9.03	9.23	10.03	10.43	15.23	8.70	15.23	15.37
0.44	0.82	1.02	0.65	0.85	0.70	1.36	0.74	1.19	0.78	1.06	1.75	1.26	1.64	2.06

Sassafras density

14	15	7	1	11	6	13	10	2	3	12	8	4	5	9
0.10	0.10	0.30	0.37	0.43	0.50	0.50	0.63	0.77	0.77	0.97	1.03	1.17	1.30	1.97
		0.11	0.15	0.27	0.16	0.16	0.29	0.27	0.23	0.26	0.33	0.40	0.35	0.40

Sweet birch density

1	7	3	14	13	11	10	8	5	2	12	6	15	9	4
0.23	0.50	0.70	0.70	0.83	1.37	1.50	2.17	3.57	3.77	5.10	5.23	5.63	5.83	6.23
0.09	0.19	0.27	0.24	0.27	0.36	0.49	0.94	1.06	0.61	0.63	0.94	1.42	1.12	0.85

Sweet birch basal area

1	7	3	15	8	13	10	11	6	9	4	14	5	12	2
8.22	17.50	60.69	92.76	94.88	168.68	174.63	199.06	218.28	408.95	452.81	459.38	514.31	747.32	780.38
5.55	16.33	39.76	23.39	35.68	84.99	84.45	85.78	86.13	125.08	83.47	226.33	149.51	171.55	128.38

Chestnut oak basal area

2	4	1	14	13	15	8	7	10	9	6	5	3	12	11
105.61	137.88	234.22	251.53	354.40	419.98	447.82	490.34	495.48	615.20	700.81	783.63	846.58	864.85	1086.43
59.98	106.29	130.20	157.76	201.24	168.10	131.09	120.71	118.45	107.29	168.63	315.94	225.75	247.05	272.95

Red maple basal area

13	14	1	8	9	12	11	10	5	2	15	7	3	6	4
136.67	142.01	142.09	195.56	242.73	264.39	302.78	375.88	399.67	447.29	504.18	522.13	542.74	623.80	892.67
35.88	103.84	118.52	48.21	37.00	51.90	50.05	93.78	76.82	110.52	104.43	82.05	89.24	166.62	107.30

Black gum basal area

14	1	12	13	15	3	5	2	4	9	6	10	11	7	8
0.10	0.10	3.29	6.04	11.57	67.73	59.51	67.73	114.55	121.56	142.89	262.21	402.80	526.52	996.83
		1.50	2.91	11.47	33.28	34.41	33.28	42.20	84.92	55.41	108.38	107.91	92.82	187.82

Chestnut oak density

4	14	1	2	15	13	5	12	10	7	11	5	8	3	9
0.37	0.37	0.37	0.57	0.90	1.03	1.37	1.63	2.03	2.10	2.17	2.83	3.17	3.77	5.03
0.12	0.15	0.15	0.24	0.24	0.36	0.39	0.44	0.34	0.38	0.81	0.56	0.74	0.69	0.56

White ash density

9	8	11	15	1	2	3	4	10	14	7	5	6	12	13
0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.10	0.17	0.17	0.17	0.17	0.17	0.23	1.43
								0.07	0.07	0.07	0.07	0.07	0.09	0.57

principal component is most highly correlated with basal area of dead trees (Table 8).

Product moment correlations between site means of soil variables and both discriminant scores and factor scores show a correlation for percent base saturation in the A1 horizon ($r = 0.628$, $p < 0.01$ and $r = 0.555$, $p < 0.05$, respectively). Other soil variables were not significantly correlated with the discriminant or factor scores.

Significant correlations include red maple basal area with: 1) cation exchange capacity of the A1 horizon ($r = 0.525$, $p < 0.05$) and, 2) pH of the A2 horizon ($r = -0.684$, $p < 0.01$); red maple density with cation exchange capacity of the A1 horizon ($r = 0.627$, $p < 0.01$) and; black gum basal area with pH in the A2 horizon ($r = 0.518$, $p < 0.05$).

Results of the analysis of variance, Cochran's C tests and Least Significant Difference tests by category are presented in table 9. In the contaminated category there was significanlty greater percent base saturation in the A1 and A2 soil horizons, ($p < 0.01$ and $p < 0.05$, respectively), greater basal area and density of black gum ($p < 0.05$), greater density of chestnut oak ($p < 0.10$), and higher total density ($p < 0.05$) than in the uncontaminated categories. Cation exchange capacity in the A1 soil horizon was signifi-

Table 8. Variable loadings for the first six components with the fourteen vegetation variables identified as most discriminating between sites by stepwise discriminant analysis.

Variable	Components					
	1	2	3	4	5	6
Black gum basal area	.282	-.547	.167	.344	.014	-.214
Chestnut oak basal area	.745	-.331	-.306	-.294	-.178	-.106
Chestnut oak density	.729	-.327	-.025	.480	-.042	.154
Dead basal area	-.283	.149	.802	-.273	.018	.091
Dead density	-.389	.179	-.125	.308	.657	-.074
Dogwood density	.153	-.241	.234	-.294	.756	-.074
Red maple basal area	.313	.431	.550	-.130	-.083	-.193
Red maple density	.777	-.049	.494	.138	.125	.162
Sassafras density	.583	.430	-.261	.495	.226	.210
Sweet birch basal area	.132	.749	-.455	-.110	.195	-.015
Sweet birch density	.365	.779	.227	.052	-.210	.185
White ash density	-.329	-.244	-.253	-.247	.076	.793
Witch hazel density	.539	.086	-.508	-.510	.005	-.268
alpha diversity	.739	-.094	.144	-.413	.236	.205
eigenvalues	3.56	2.25	2.05	1.48	1.25	1.00
% variance accounted for	25.41	16.04	14.66	10.57	8.96	7.18

Table 9. Means and standard errors and results of LSD for soil and vegetation variables which are significantly different by category. Horizontal lines indicate categories which are not significantly different. Categories are westernmost (W), easternmost (E), and contaminated (C).

Variable			
Cation exchange capacity A1	C	E	W
	9.35	16.82	17.75
	0.92	1.76	2.24
Cation exchange capacity A2	E	W	C
	8.22	11.12	11.65
	0.89	0.78	0.47
Percent base saturation A1	E	W	C
	11.85	17.05	57.48
	4.20	8.78	9.62
Percent base saturation A2	W	E	C
	7.79	11.04	21.76
	1.00	3.85	3.64
Black gum basal area	E	W	C
	5.25	52.41	448.85
	4.86	25.01	192.11
Black gum density	E	W	C
	0.28	1.36	7.34
	0.09	0.50	5.22
Chestnut oak density	E	W	C
	0.98	1.27	3.10
	0.26	0.83	0.69
Dead density	E	C	W
	6.46	7.48	12.56
	1.50	0.99	1.66
Hickories density	C	W	E
	0.12	0.67	0.80
	0.02	0.24	0.27
Total density	E	W	C
	20.15	26.20	34.45
	4.81	2.35	1.84

cantly lower ($p < 0.01$) in the contaminated than in the uncontaminated categories. The density of dead trees is significantly greater ($p < 0.05$) in the western category than the contaminated and eastern categories, and hickories are significanlty ($p < 0.01$) more abundant in the eastern than the western and contaminated sites (although they do not form a large portion of total basal area in any category).

DISCUSSION

A comparison of soil metal concentrations between this study and a study by Buchauer (1973) shows zinc and cadmium concentrations are lower on the southfacing slope than on the ridgetop. This difference is probably a result of the closer proximity to the smelter of sites in this study, and the protection from high levels of contamination afforded by the ridge. Data from both studies support a zinc deposition model (Washington, pers. comm.) which predicts a decline in soil concentrations of zinc with increasing distance from the smelters. Although metal concentrations in the A1 horizon at sites closest to the smelters are in the range classified 'very high' by Baker (pers. comm.), concentrations of these metals are much higher at the surface organic layer and only a small amount of downward movement occurs (Buchauer 1973).

High levels of zinc and cadmium correspond with very low levels of aluminum, manganese and iron (Table 1). Solubilities of the latter three elements are low at circumneutral pH levels (Drever 1982), and the abnormally high soil pH in the contaminated areas may restrict the dissolution of these elements from parent materials. This suggests aluminum toxicity is not a factor, but iron and manganese deficiencies may be factors influencing vegetation in the most contami-

nated areas.

Abnormal soil profile patterns for pH and cation exchange capacity associated with high metal levels suggest an interruption of natural soil processes. Elevated pH of the surface soil has been previously documented and was attributed to the amphoteric character of deposited zinc oxide (Buchauer 1971, 1973). If decomposition is reduced at the contaminated sites, as is suggested by Strojan (1978a,b), reduced production of acids may also contribute to high pH. Reduced cation exchange capacity at sites nearest to the smelters may be the result of lower mineralization rates which accompany reduced decomposition, or due to the exchange of metal cations for nutrient cations (Ruhling and Tyler 1973). Damage to nutrient cycling processes has been described in other polluted areas (Watson et al. 1976, Dudzik et al. 1976, Tyler 1974), and may precede forest community changes (Grodzinsky and Yorks 1981).

Communities in the uncontaminated areas are assumed to be representative of the normal forest community on the ridge. The description of uncontaminated communities in this study is similar to previous descriptions of community structure near site 6 (Reinert 1984), as well as descriptions for a chestnut oak forest in the Great Smoky Mountains (Whittaker 1965), and for a mixed oak forest in Pennsylvania (Rosenberg

et al. 1979).

Comparison of importance values for contaminated and uncontaminated sites suggests forest responses to contamination include a change from red maple-red oak and sweet birch forest to a black gum-chestnut oak-red maple forest. Increases in relative dominance for both black gum and chestnut oak and a decrease in relative density and dominance of red oak and sweet birch suggest the former species are relatively tolerant, and the latter species are relatively intolerant of the changed conditions. Red maple is important in both the contaminated and uncontaminated areas and, from this method of analysis, seems to be neutral to conditions related to contamination.

Previous studies of the ridge forests near the Palmerton smelter (Buchauer 1971, Jordan 1975) and near other smelters (Gordon and Gorham 1963, Freedman and Hutchinson 1980) have documented an increase in density and basal area with distance from the point source. Although there were no significant ($p \not\leq 0.14$) differences between sites or categories for basal area in this study, there was a significantly greater density of trees in the contaminated area. Differences between apparent community structure responses to contamination in this study and the studies by Jordan and Buchauer may be due to the greater level of contamination on the rid-

getop than on the south slope.

A trend in this study toward trees of smaller stature has also been documented in other contaminated areas (Woodwell 1970), and may be due to younger age or reduced radial growth. A preliminary comparison of radial growth between contaminated and uncontaminated areas investigated the nature of size differences. Chestnut oak in the contaminated area appears to have reduced radial growth increment compared to trees in uncontaminated areas, but there is no evident difference for red maple, red oak and sassafras. Although these are preliminary findings, this suggests reduced radial growth is one of the factors contributing to the small stature of trees in the contaminated area.

Although reduced species diversity has previously been documented for other contaminated forests (Rosenberg et al. 1979, Freedman and Hutchinson 1980), the diversity index did not show trends related to contamination in this study. An interesting opposite trend occurred, however, when the number of species for two different size samples were compared by site. The number of species per 0.15 ha sample was lowest at the most contaminated sites, but the number of species per 0.01 ha was greatest at these sites. This seeming paradox may be related to density differences and may be an indication of an inadequately sized vegetation sample.

Results of the multivariate analyses and product moment correlations suggest the greatest source of community variation on the ridge is related to contamination. Other factors which could affect community structure have not been measured, cannot be discounted, but seem unlikely as the cause of variation. No evidence of recent logging exists at any site, and gypsy moth or other herbivore damage is an unlikely selective influences on vegetation in the contaminated area. Fires are not uncommon in the vicinity of Lehigh Gap, but records are not complete (Jordan 1975). The lack of evidence of fire, thick litter layer at all sites, and great density and basal area of fire intolerant black gum (Fowells 1965) in the contaminated area, suggests fire is not a recent influence.

Results of the analysis of variance by category showed soil chemistry and some species abundances and dominances are significantly different between contaminated and uncontaminated areas. Chestnut oak and black gum seem to be more closely related to zinc and cadmium contamination than other species on the ridge.

Similar relationships between vegetation and contamination were documented in other areas. Rosenberg et al. (1979) also found red maple to be persistent and chestnut oak, black gum and dogwood intermediate in persistence near

a coal-fired power plant in Pennsylvania. Red maple was also persistent near metal smelters in Canada (Gordon and Gorham 1963), near a radiation source in New York (Woodwell 1970), and was previously noted as persistent in contaminated areas on the ridge near Lehigh Gap (Jordan 1975). Nash (1975) noted a greater number of black gum, sassafras and red oak near the Lehigh Gap than at the Delaware Water Gap, but reports an absence of red maples near the Lehigh Gap. Jordan's finding of fewer chestnut oaks near the smelters contrasts with this study, possibly due to previously mentioned proximity and deposition differences.

Because the metal contaminated soil is known to inhibit germination and seedling establishment (Jordan 1975), the success of black gum, chestnut oak and red maple may be due to their ability to reproduce vegetatively. Community difference may therefore be a trend toward a community composed of species which are able to avoid the sensitive seedling stage through vegetative reproduction. If the nutrient cycle is damaged, as has been suggested by this and previous studies (Strojan 1978a,b), a shift away from nutrient demanding species and toward species able to survive on nutrient poor soil is expected. The change appears to be less dramatic on the southfacing slope than on the ridgetop.

A more rigorous presentation of the data is not possible

due to limits on interpretation. This study exemplifies the difficulty in accurately describing a large vegetated area from a relatively small area sampled. Although species area curves suggest the number of species were adequately sampled, an underlying assumption of homogeneity of variance was not met when quadrats were treated as samples. Quadrats of 0.01 ha are, therefore, inadequate in size for this type of analysis. Samples sized at 0.15 ha (the sum of 15 quadrats at each sample location) exhibited homogeneous variance for all significantly different variables. An adequate sample for this type of vegetation analysis can, therefore, be considered to be 0.15 ha. Adequate sampling for soil pH, cation exchange capacity and percent base saturation was suggested by homogeneity of variance when mean values were grouped to categories. Four soil samples, rather than two, seem adequate for statistical analysis of pH, cation exchange capacity and percent base saturation in this study. Although larger quadrats and a greater number of samples at each site would have provided better estimates of actual conditions, underlying assumptions for univariate tests were met by grouping of samples.

This study did not address soil-plant interaction, but rather showed a relationship between soil chemistry and forest community structure. Variation in structure is correlated with contamination, and suggests the forest community

on the southernmost ridge of the Appalachians in Pennsylvania has responded to the deposition of metal contaminated particles from the zinc smelter. Rather than a clear exclusion of species in the contaminated area, only a subtle change in community structure has been detected.

LITERATURE CITED

Andresen, A. M., A. H. Johnson and T. G. Siccama. 1980. Levels of lead, copper and zinc in the forest floor in the northeast United States. J. Environ. Qual. 9:293-296.

Baes, C. F. III and S. B. McLaughlin. 1984. Trace elements in tree rings: evidence of recent and historical air pollution. Science 224:494-496.

Baker, D. E. and M. C. Amancher. 1981. The development and interpretation of a diagnostic soil-testing program. Bulletin 826. Agricultural Experiment Station, The Pennsylvania State University.

Braun, E. L. 1950. Deciduous forests of eastern North America. Hafner Publishing, New York. pp. 596.

Buchauer, M. J. 1971. Effects of zinc and cadmium pollution on vegetation and soils. Ph.D. thesis, Rutgers-The State University. New Brunswick, New Jersey.

Buchauer, M. J. 1973. Contamination of soil and vegetation near a zinc smelter by zinc, cadmium, copper and lead. Environ. Sci. Technol. 7:131-135.

Cain, S. A. 1938. The species area curve. Am. Midl. Nat. 19:573-581.

Curtis, J. T. and R. P. McIntosh 1951. An upland forest continuum in the prarie-forest border region of Wisconsin. Ecology, 32(3):476-496.

Dixon, W. J. and M. B. Brown. 1977. BMDP-77: Biomedical computer programs P-weries. University of California Press, pp. 880.

Drever, J. I. 1982. The geochemistry of natural waters. Prentice-Hall, Inc. Englewood Cliffs, New Jersey. pp. 388.

Dudzik, M., J. Harte, D. Levy, J. Sandusky. 1975. Stability indicators for nutrient cycles in ecosystems. Lawrence Berkeley Laboratory. Report LBL-3264. University of California.

Fowells, H. H. 1965. Silvics of forest trees of the United States. Agric. Handb. No. 271, U.S. Gov. Print. Off. Wash. D.C. pp.762.

Foy, C. D., R. L. Chaney and M. C. White. 1978. The physiology of metal toxicity in plants. Annu. Rev. Plant Physiol. 29:511-566.

Freedman, B. and Hutchinson, T. C. 1980. Pollutant inputs from the atmosphere and accumulation in soils and vegetation near a nickel-copper smelter at Sudbury, Ontario, Canada. Can. J. Bot. 58(1):108-132.

Gordon, A. G. and E. Gorham. 1963. Ecological aspects of air pollution from an iron-sintering plant at Wawa, Ontario. Can. J. Bot. 41:1063-1078.

Gray, C. and W. S. Shepps. 1960. Geologic map of Pennsylvania, Pennsylvania Geological Survey. Fourth series.

Grodzinski, W., J. P. Yorks. 1981. Species and ecosystem level bioindicators of airborne pollution: and analysis of two major studies. Water Air Soil Pollut. 16:33-53.

Guderian, R. 1977. Air pollution. Phytotoxicity of acidic gasses and its significance in air pollution control. (Translated by C. J. Brandt). Ecological Studies Vol.22, Springer Verlag, Berlin/Heidelberg, 127 pp.

Hanson, D. W., S. A. Norton, J. S. Williams. 1982. Modern and paleolimnological evidence for accelerated leaching and metal accumulation in soils in New England, caused by atmospheric deposition. Water Air Soil Pollut. 18:227-239.

Hanssen, J. E., J. P. Ranbaek, A. Semb, and E. Steinnes. 1980. Atmospheric deposition of trace elements in Norway. Proc. Int. Conf. Ecol. Impact Acid Precip. Norway 1980. SNSF project.

Hinish, W. W., R. R. Heddleson, and J. H. Eakin, Jr., 1967. Soil testing handbook, Cooperative Extension Service, The Pennsylvania State University, University Park, Pennsylvania.

Hull, C. H. and N. H. Nie. 1981. SPSS update 7-9. McGraw-Hill Book Company. pp. 401.

Johnson, A. H. and T. G. Siccama. 1983. Acid deposition and forest decline. Environ. Sci. Technol. 17(7):294-305.

Johnson, A. H., T. G. Siccama, D. Wang, R. S. Turner, T. H. Barringer. 1981. Recent changes in patterns of tree growth rate in the New Jersey Pinelands: a possible effect of acid rain. J. Environ. Qual. 10(4):427-430.

Johnson, A. H., R. S. Turner, D. G. Lord, T. G. Siccama. 1982. Assessing the possibility of a link between acid precipitation and decreased growth rates of pitch-pine (Pinus rigida), shortleaf pine (Pinus echinata) and red spruce (Picea rubens). Abs. of Pap. Amer. Chem. Soc. 183(Mar):93.

Jordan, M. J. 1975. Effects of zinc smelter emissions and fire on a chestnut-oak woodland. Ecology 56:78-91.

Jordan, M. J. and M. P. Lechvalier. 1975. Effects of zinc-smelter emissions on forest soil microflora. Can. J. Microbiol. 21:1855-1865.

Kleka, W. R. 1975. Discriminant analysis. In Statistical package for the social sciences. N. H. Nie (ed.). McGraw-Hill, Inc. pp.434-467.

Knabe, W. 1976. Effects of sulfer dioxide on terrestrial vegetation. Ambio 5:213-218.

Lappin, J. 1973. An estimate of the composition of the forests in early 1900's which covered the northwest slopes of Blue Mountain near Palmerton, Pennsylvania. MS Thesis, Rutgers, The State University, New Brunswick, New Jersey.

Lawhon, W. T. and F. W. Woods. 1976. Radial growth and wood density of white pine in relation to fossil-fired power plant operations.

Proc. of the First Int. Symp. on Acid Precip. and the Forest Ecosystem. USDA Forest Service general technical report NE-23.

Moore, J. W. and S. Ramamoorthy. 1984. Heavy metals in natural waters. Springer-verlag New York Inc., New York. 268 pp.

Nash, T. H. III. 1975. Influence of effluents from a zinc factory on lichens. Ecol. Monogr. 45:183-198.

Nash, T. H. III., H. C. Fritts, M. A. Stokes. 1975. A technique for examining non-climatic variation in widths of annual tree rings with special reference to air pollution. Tree-ring Bulletin. 35:15-24.

Nie, N. H., C. H. Hull, J. G. Jenkins, K. Steinbrenner, D. H. Bent. 1975. Statistical package for the social sciences. McGraw-Hill Book Company, pp. 675.

Nriagu, J. O., and C. I. Davidson. 1980. Zinc in the atmosphere. In zinc in the environment. Part I.: Ecological cycling. J. O. Nriagu (ed). P. 113-159. John Wiley and Sons, New York.

Oosting, H. S. 1956. The study of plant communities. W. H. Freeman Co., San Fransisco, Ca. 440 pp.

Pimentel, R. A. 1979. Morphometrics: the multivariate analysis of biological data. Kendall/Hunt, Dubuque, Iowa. pp.276.

Puckett, L. J. 1982. Acid rain, air pollution and tree growth in southeastern New York. J. Environ. Qual. 11(3):376-381.

Raynal, D. J., A. L. Leaf, P. D. Manion and C. J. Wang. 1980. Actual and potential effects of acid precipitation on a forest ecosystem in the Adirondack Mountains. New York State Energy and Development Authority, Research Report 80-28.

Raynal, D. J., J. R. Roman, W. M. Eichenlaub. 1982. Response of tree seedlings to acid precipitation II. Effect of simulated acidified canopy throughfall on sugar maple seedling growth. Environ. and Experimental Bot. 22(3): 385-392.

Reiners, W. A., R. H. Marks, P. M. Vitousek. 1975. Heavy metals in subalpine and alpine soils of New Hampshire. Oikos 26:264-275.

Reinert, H. K. 1984. Habitat separation between sympatric snake populations. Ecology 65(2): 478-486.

Rosenberg, C. R., R. J. Hutnick, D. D. Davis. 1979. Forest composition at varying distances from a coal-burning power plant. Environ. Pollut. 18:307-317.

Ruhling, A., and G. Tyler. 1973. Heavy metal pollution and decomposition of spruce needle litter. Oikos 24:402-416.

Shannon, C. E. and W. Weaver. 1949. The mathematical theory of communication. Univ. Illinois Press, Urbana. pp. 117.

Smith, W. H. 1981. Air pollution and forests. Springer-Verlag, New York 379 pp.

Sokal, R. R. and F. J. Rohlf. 1981. Biometry. W. H. Freeman and Company, USA.

Stone, L. L. and J. M. Skelly. 1974. The growth of two forest tree species adjacent to a periodic source of air pollution. Phyto-pathology 64:773-778.

Strand, L. 1980. The effect of acid precipitation on tree growth. Proc. Int. Conf. Ecol. Impact Acid Precip., Norway SNSF project. 1980:64-67.

Strojan, C. L. 1978a. Forest leaf litter decomposition in the vicinity of a zinc smelter. Oecologia 32:203-212.

Strojan, C. L. 1978b. The impact of zinc smelter emmissions on forest litter arthropods. Oikos 31:41-46.

Tomlinson, G. H. 1983. Air pollutants and forest decline. Env. Sci. and Technology. 17(6): 246-256.

Tyler, G. 1974. Heavy metal pollution and soil enzymatic activity. Plant and Soil 41: 303-311.

Watson, A. P., R. I. Van Hook, D. R. Jackson, D. E. Reichle. 1976. Impact of a lead mining-smelting complex on the forest floor litter arthropod fauna in the Lead Belt region of southwest Missouri. ORNL/NSF/EATC-30. Oak Ridge National Laboratory.

Winer, B. J. 1971. Statistical principles in experimental design. McGraw-Hill Book Company. 907 pp.

Woodwell, G. M. 1970. Effect of pollution on the structure and physiology of ecosystems. Science 168:429-433.

PERSONAL VITA

Address: 735 Fyler Rd
Kirkville, NY
13082

Office (315) 470-6793
Home 656-7078

Date of Birth: September 11, 1953

Place of Birth: Allentown, Pa

Parents: Alexander M. Burke
H. Joan Burke

Present Position: Graduate student, research assistant
Department of Environmental and Forest Biology
College of Environmental Science and Forestry
State University of New York
Syracuse, NY 13210

Education: University of Tennessee
Knoxville, TN
Forestry Department, 1976
B.Sc.

Affiliation: Sigma Xi

Previous Positions: 1979-1981 Wildlife Biologist
US Forest Service
Fall River Mills, CA

1976-1978 Environmental Education Officer
Fiji Department of Forestry
Suva, Fiji
(US Peace Corps/
Smithsonian Institution)

Awards: Summer Fellowships, 1983, 1984
Lehigh University Energy Research Center

Teaching Assistantships, 1983,1984
Biology Department, Lehigh University