VEGETATIONAL AND LEPIDOPTERAN CONSERVATION IN REHABILITATED ECOSYSTEMS
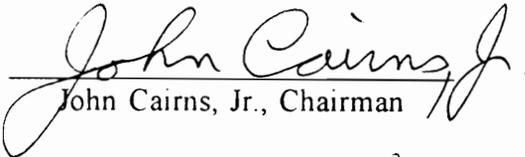
by

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Doctor of Philosophy

in

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APPROVED:

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ABSTRACT

Coal surface mining and associated reclamation practices have had an immense impact on the landscape of the Appalachian region of the United States. However, their effect on floral and faunal conservation has been poorly documented. Lepidopteran communities, vegetation, and nectar resources were studied on 19 mine sites reclaimed 0-30 years previously and five sites in the surrounding hardwoods in southwestern Virginia. The goals of this work were to characterize vegetational and lepidopteran communities of these sites; to understand the relationships between the two; and to assess the role of mine reclamation in regional conservation efforts.

Vegetational community composition of the reclaimed sites appeared to be approaching that of the hardwood sites as time since reclamation increased. However, it will take a number of years, if ever, before the vegetational community composition and structure approximate that of the hardwoods. Between-site variation in vegetational communities was greater in the hardwoods, than the reclaimed sites.

Recently reclaimed mined sites hosted a large number of both individuals and species of diurnal lepidoptera, comprising mostly widespread, generalist species. Multivariate analysis suggested that lepidopteran community composition of reclaimed sites was approaching that of the hardwoods and that lepidopteran communities of later

successional reclaimed sites were fairly similar to those of the surrounding hardwoods. Moth community composition closely reflected vegetational community composition and species richness, while butterflies were poor indicators of vegetational communities.

Reclaimed sites provided much more abundant and diverse nectar resources than hardwood forests. Results of nectar studies and butterfly behavioral observations suggested that adult butterfly community composition was strongly influenced by nectar abundance, but that nectar was not a limiting resource.

While reclaimed sites hosted a number of the more common plant and lepidopteran species, it remains questionable whether reclaimed areas will ever host the entire complement of the biota present prior to disturbance. In order to further conservation efforts, rehabilitation projects must be designed and monitored over larger spatial and longer temporal scales.

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GENERAL INTRODUCTION

Traditionally, ecologists have studied undisturbed ecosystems to better understand community ecology and have advocated protection of these ecosystems in order to conserve biodiversity. However, we now live in a world where human impact is ubiquitous. Vitousek et al. (1986) estimate that 40% of terrestrial primary production is used directly or indirectly by human beings. Cairns (1989) challenges us to find systems that have yet to be impacted by human endeavors. In fact, much ecological research is performed on sites that have been severely disturbed by human activities (Cairns 1988).

As humans continue to radically alter the natural environment, more frequent efforts are being made to restore ecosystems to their predisturbance conditions, or to reclaim severely disturbed land for an alternate use. While rehabilitation of disturbed ecosystems has been increasingly cited as a means of mitigating habitat destruction (e.g. Cairns 1988; Jordan et al. 1988), there has been little study of the long-term success of restoration and reclamation efforts. Because most rehabilitation efforts have been initiated in recent times, long term studies are not possible; coal surface mines, however, have been reclaimed for over thirty years. While reclamation practices have changed over the years and an even longer time period would be necessary to fully assess the successional trajectory on these sites, they do offer one of the best opportunities for study of the long-term effects of ecosystem rehabilitation.

Coal surface mining and mine reclamation have had a significant impact on the

landscape of the Appalachian region of the United States. According to the Office of Surface Mining, over 50,000 hectares in southwestern Virginia have been disturbed by mining since 1977, before which time detailed records do not exist. Zipper (1986) estimates that by 1986, 8.5% of this region had been directly impacted by coal surface mining. The majority of mining in the Appalachian region is contour mining, which results in many long, disturbed strips of land that wind through the mountains.

Despite the extensive environmental disturbance caused by surface mining there has been little study of the long-term development of ecological communities on these sites. The goals of this work were to characterize the succession of the vegetational and lepidopteran components of this ecosystem and to better understand the relationship between the two. This information is important in assessing the role of surface mine reclamation in the preservation of the biodiversity of this region.

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Chapter 1

VEGETATIONAL COMMUNITIES

Introduction

The Appalachian region of the United States was mined extensively for coal
beginning after World War II. However, until the mid-1960s, efforts were rarely made
to reclaim surface mined areas, and many such areas still remain sparsely vegetated
today (Stocum 1980). Until approximately 15 years ago mine reclamation practices
were haphazard and often resulted in exposed highwalls and unstable outslopes (Zipper
1990) (Figure 1.1, p. 21). During this time period, benches and outslopes were often
hydroseeded with a few grass species and black locust (*Robinia pseudoacacia*), a
nitrogen fixing tree species. White pine (*Pinus strobus*) was planted on the top of the
outslope, in what later proved to be an ineffective effort, to hide the highwalls.

The passage of the Surface Mining Control and Reclamation Act (SMCRA),
(Public Law 95-87 Federal Register 3 Aug. 1977, 445-532) in 1977 mandated and
standardized reclamation practices. This act requires that parties involved in surface
mining post a bond and detail a reclamation plan for a site after mining. Judgment of
revegetation success for bond release is almost exclusively determined by restoration
of original contour and achievement of a certain percentage ground cover, 90% in
Virginia, in order to stabilize the soil and control water quality (Virginia Department
of Mines, Minerals, and Energy 1979). Establishment of a minimum tree density per

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acre is required for the over 80% of surface mined land that is designated for forestry purposes ultimately. Commonly-planted vegetation is fairly similar to that used before enactment of legislation, although white pine is planted more extensively because of its rapid establishment and commercial value. Widely planted herbaceous species include such grasses as orchard grass (*Dactylis glomerata*), Kentucky-31 tall fescue (*Festuca arundinacea*), and Timothy (*Phleum pratense*), and such legumes as serecia lespedeza (*Lespedeza cuneata*), birdsfoot trefoil (*Lotus corniculatus*), and clover (*Trifolium* spp.). None of the commonly-used herbaceous species are native to the southeastern United States. Success of reclamation is judged after 5 years; if the bond is released at this time, the owner is not required to make further efforts towards ecosystem establishment.

The goals of this research were (1) to characterize the vegetation of reclaimed coal surface mined sites in southwestern Virginia; (2) to determine if the vegetation on reclaimed sites is following a successional trajectory towards that of the surrounding forest; and (3) to consider the implications of reclamation procedures for the conservation of the flora and fauna of this region.

Site Description

This research was performed in Wise County, Virginia. The majority of the sites are located on or near the Powell River Project research area (PRP), an approximately 700-ha area, about 125 ha of which have been mined. The PRP is

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owned by Penn Virginia, but is part of a research program administered by Virginia

Polytechnic Institute and State University. Only one sufficiently large area remains in

the vicinity of the PRP that was reclaimed during the 1960s and has not been remined;

consequently, four of the later successional reclaimed sites are located near the town

of Appalachia, at a distance of 10 to 15 km from the PRP.

The soil in this area is comprised of acidic and infertile sandstones, siltstones,

and shales (Daniels and Amos 1985). The premining vegetation of the reclaimed areas

was typical of the majority of the central Appalachian coal bearing-region, which is

covered by oak-hickory forests and Appalachian mixed-hardwood forest types with

species composition reflecting local microclimate and soil type (Burger and Torbert

1990). The unmined portions of the study area, like the vast majority of this region of

the country, have been logged for the past 200 years and are, therefore, not free from

human impact. The sites surveyed range in elevation from 700 to 925 m. All sites

are south facing with slopes ranging from 12.5° to 42.5° and aspects ranging from

140° to 225°.

Methods

Field methods

The vegetation was surveyed on 23 quarter-hectare sites that were rectangular

(62.5 x 40 m) with the longer side perpendicular to the slope. The sites were grouped

into five age classes for most analyses (Table 1.1). All sites of the same type were

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separated by a minimum of 0.5 km. Site ages were determined from mining permit maps and tree cores. Exact logging records were not available for the hardwood sites; the most recently disturbed site was selectively cut 9 years previously. Since the hardwood sites contained mixed-aged trees, the age used for analyses was determined by coring a tree in the oldest age class that was represented by large numbers of individuals.

Table 1.1 - Description of sites

Number of sites	Site type
3	Reclaimed <5 years ago
5	Reclaimed 5-12 years ago
5	Reclaimed 15-20 years ago
5	Reclaimed 25-30 years ago
5	Hardwoods

Nineteen sites were sampled during summer 1992, and the four remaining sites were sampled during summer 1993. Vegetation on the sites was sampled in three vegetational strata: herb (0-0.75 m), shrub (0.75-2 m), and tree (>2 m). The total percent cover and percent cover of individual herbaceous species were recorded in 16 1-m^2 quadrats. The cover of individual species was ranked using the Braun-Blanquet

cover-abundance scale (Mueller-Dombois and Ellenberg 1974). Quadrats were systematically distributed along four transects perpendicular to the slope, each separated by 10 m. Herbs were sampled in both late-May/early-June and in late-August to insure identification of all species. Total cover and individual species cover estimates from August were used for all species except bittercress (*Cardamine hirsuta*) and bedstraw (*Galium aparine*), which had wilted by that time.

Shrubs and trees were sampled in eight quadrats, 4 and 10 m^2, respectively, that were distributed systematically along the same transects as the herb quadrats. Total percent cover and cover of individual species were estimated in the shrub strata. Because of the difficulty in estimating cover in the canopy, the composition of this layer was quantified by measuring the diameter at breast height (DBH) of trees rooted in the quadrat. The DBH is generally a good predictor of tree crown cover (Spurr 1960). Shrubs and trees were sampled once in June/early-July.

Nomenclature follows Radford et al. (1968). All plants were identified to species except for a few whose determination to only the generic level was possible at the time of sampling. Gleason and Cronquist (1991) was used as a source for the original geographic ranges of species.

Numerical analyses

Importance values were calculated for the species in each vegetational stratum (Mueller-Dombois and Ellenberg 1974). The importance value was calculated as the

sum of the relative cover and relative frequency for herbs and shrubs. For trees, the importance value was calculated as the relative number of individuals and relative basal area. Since a measure of frequency and abundance was included for each layer, the sum of importance values in each layer totalled 200. The species-by-site matrices for each layer were combined for multivariate analyses, utilizing the largest importance value from the three layers. Because the shrub layer in the sites studied was characterized by only a few species, these species were combined with the tree strata for calculations of species richness. Vines were included with the strata in which they were most abundantly represented.

Because exact site ages were difficult to determine, ages were ranked and Spearman rank correlation coefficients between vegetational community variables and site age are reported here. The effect of slope, aspect, and elevation on the various vegetational species richness and structure variables was tested using dummy variable regression adjusting for type of site. The partial F-test for adding each of these variables separately, after the type and interaction (between type and the variable being tested) effects were included, was used as a measure of significance.

The vegetational dissimilarity between sites was quantified by calculating the percent dissimilarity between sites (Gauch 1982). As the distance between sites varied greatly, the relationship between geographic distance separating the sites and vegetational dissimilarity was compared using a Mantel test on the reciprocals of geographic distances and the percent dissimilarity matrix. Reciprocal distances were

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used to emphasize nearby sites, as beyond a certain distance relative separation of sites is unlikely to affect ecological similarity (Manly 1991). A one-sided test was used to determine how close the observed Z was to the Z based on 10,000 randomly permuted distances. $Z=\sum X_{ij}Y_{ij}$ where X_{ij} and Y_{ij} are the off-diagonal elements of the two distance matrices.

Trends in vegetational community composition were explored using detrended correspondence analysis (DCA) of the vegetational importance value-by-site matrix. Detrending was used because of the long species gradient, which resulted in a high correlation between the second axis and square of the first axis. Because DCA is an averaging procedure, sites containing only a few species are distinctive (Gauch 1982). This phenomenon was apparent for the most recently reclaimed site. Since the focus of this study was on long-term trends, this site was not included in the DCA analysis.

A ranking system was used to quantify less common species that tend to be downweighted in multivariate procedures. Species that were found in one site were given 5 points; those in two sites, 3 points, and those in three sites, 1 point. These rankings were summed for each site.

Results

In total, 156 plant species were found, including 112 herbaceous species and 44 shrub and tree species. Figures 1.2 and 1.3 (pp. 22 and 23) illustrate the species richness and structural trends of the sites surveyed. In Figure 1.2 cover is relativized

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to the maximum cover in each vegetational stratum, as the measurement units for the three strata differed. Table 1.2 shows the means, medians, and ranges for vegetational community variables. Table 1.3 lists the correlations between variables in Table 1.2 and ranked site age. Both species richness of trees and total species richness increased significantly with site age, although tree species richness was more highly correlated than total species richness with site age ($r=0.87$ vs. $r=0.74$). Herb species richness was not significantly correlated with site age. Average herb cover decreased significantly with site age, while tree cover increased and shrub cover was not correlated with site age. Tree species richness was correlated with tree cover; herb species richness was not correlated with herb cover, because the higher cover was largely the result of a few planted species.

Table 1.2 - Means, medians, and ranges of vegetational community variables for sites (n=23)

Variable	Mean	Median	Range
Species richness herbs (SRH)	20.6	20	10-35
Species richness trees (SRT)	11.4	9	2-24
Species richness total (SRV)	32.0	31	13-55
Average percent cover herbs (CH)	68.3	74	12-106
Average percent cover shrubs (CS)	14.8	10	<1-38
Average basal area trees ($m^2/10\ m^2$) (BAT)	0.1028	0.0650	0-0.3834

Table 1.3 - Spearman rank correlation coefficients between vegetational community variables and site age (n=23) Abbreviations are explained in Table 1.2.

Variable	SRH	SRT	SRV	CH	CS	BAT	AGE
SRH	1.0						
SRT	0.03	1.0					
SRV	0.60*	0.69**	1.0				
CH	-0.05	-0.79***	-0.63*	1.0			
CS	-0.29	0.50*	0.14	-0.41	1.0		
BAT	0.09	0.91***	0.68**	-0.66**	0.42*	1.0	
AGE	0.17	0.87***	0.74***	-0.70**	0.40	0.95***	1.0

*p<0.05, **p<0.001, ***p<0.0001

The small variations in slope, aspect, and elevation did not have significant effects on any of the vegetational species richness and structure variables, after adjusting for the site type and interaction effects (p>0.05 in all cases). In no case were the type-variable interaction effects significant. Using the Mantel test, the relationship between vegetational dissimilarity and geographic distance of all sites and of sites of a given type was not significant (p>0.05 in all cases, n=23 and 5).

Figure 1.4 (p. 24) shows the locations of the sites and the 60 most common species on the first two DCA axes. For a complete species list and the types of sites in which each species was found see Appendix 1 (p. 105). The first axis of DCA has an eigenvalue of 0.682 and explains 20% of the variation. This axis is strongly

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correlated with ranked site age (r=0.92, p<0.0001). The second and third dimensions only explain an additional 6 and 4% of the variation, respectively.

The most recently reclaimed sites were characterized by the planted species discussed previously. Most of the rapidly invading species were in the Asteraceae, including frost aster (*Aster pilosus*), horseweed (*Erigeron canadensis*), hawkweed (*Hierarcium pratense*), and a number of goldenrods *(Solidago* spp.). In mid-successional reclaimed sites, bittercress *(Cardamine hirsuta)*, virgin's bower *(Clematis virginiana)*, pokeweed (*Phytolacca americana*), and dwarf cinquefoil (*Potentilla simplex)*, as well as most species found in earlier sites, were common in the herb layer, while invading shrub and tree species included red maple *(Acer rubrum)*, sourwood (*Oxydendron arboreum*), wild cherry (*Prunus serotina*), and blackberry *(Rubus allegheniensis)*. In the sites reclaimed more than 25 years ago, the majority of the planted species were no longer present, and a number of species that were common in the surrounding hardwoods were found. Typical species included heart-leaved aster *(Aster divaricatus*), jewelweed *(Impatiens pallida)*, and violets (*Viola* spp.) in the herb strata and sweet birch *(Betula lenta)*, tulip poplar (*Liriodendron tulipifera)*, and grape *(Vitis aestivalis)* in the canopy. A number of species were common in the hardwood sites that were rarely found in reclaimed sites: beggar's tick *(Desmodium nudiflorum)*, wild yam *(Dioscorea villosa)*, red oak (*Quercus rubra)*, greenbriar *(Smilax rotundifolia)*, buffalo nut *(Pyruleria pubera)*, and rhododendron (*Rhododendron maximum)*.

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Figure 1.5 (p. 25) shows the ranking of the sites with respect to less common species. While there was a great deal of variability among different sites in the same age class, the hardwood sites hosted a greater number of species that were less common, particularly in the herb strata. Such species included a grass (*Brachyelytrum erechtum*), meadow parsnip (*Thaspium barbinode)*, trillium (*Trillium grandiflorum*), and bellwort (*Uvularia perfoliata*). Total less common species rankings were weakly correlated with site species richness (r=0.59, p=0.0031, n=23).

Discussion

Many obstacles exist to establishing vegetation on previously mined sites, such as low pH (Haufler et al. 1978; Stocum 1980), soil compaction (Daniels and Amos 1981; Davidson et al. 1984; Ashby 1990), elevated surface temperatures (Deely and Borden 1973; Stocum 1980; Bell and Ungar 1981), water stress (Brenner et al. 1984; Grunwald et al. 1988), and lack of nutrients such as nitrogen (Haufler et al. 1978) and phosphorus (Andrews 1992). Considering the stressful abiotic conditions and minimal management efforts, the degree of resilience of these systems is encouraging. A large portion of the flora common in the surrounding hardwoods has become established in the later successional reclaimed sites.

While the first DCA axis was strongly correlated with time, the vegetational communities of hardwood and older reclaimed sites still differed substantially. Similarly, the vegetational structure of the older reclaimed sites did not resemble that

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of the hardwoods. The data suggest that it will be a number of years, if ever, before even the oldest reclaimed sites approximate the vegetation of the surrounding area. The goal of mine reclamation is not necessarily to restore predisturbance condition. Regardless, these results raise the question of whether the success of reclamation can be judged after only 5 years, when succession in these disturbed systems takes much longer. A bond cannot be held for the time period required for development of a mature forest, but regulations should be designed to balance short- and long-term management needs.

A great deal of variation remains unexplained by site age in the multivariate analyses. There are a number of possible interpretations for this variation. First, the sites did not represent a true successional sequence because of changes in reclamation practices. Second, some variation may have resulted from differences in abiotic factors such as soil pH or nutrient levels, although past studies have suggested that within-site variation in soil variables on mined areas is often higher than between sites (Stocum 1980; Brenner et al. 1984). Third, a problem common to many efforts to rehabilitate disturbed lands is invading non-native species impeding succession; these species are often unable to outcompete natives in highly disturbed areas. While initial distributions of plant species are not always well-documented, at least 17 of the invading species found on these mine sites are not native to this area, whereas none of the species found in the surrounding hardwoods were introduced. Fourth, vegetational succession is likely affected by landscape-scale factors such as distance to a colonizing

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source. However, most of the sites were located within 50 m of hardwood forest. Interestingly, the relative geographic distances between sites did not explain a significant amount of this variation. The most likely explanation is that all sites were separated by at least 0.5 km. Also, comparisons by site type have little power due to small samples sizes.

While legislation has resulted in an improvement in environmental quality in this region, there are some potentially negative effects. First, the use of aggressive species to achieve 5-year herbaceous cover and tree establishment requirements mandated by legislation may inhibit the long-term development of the vegetational communities on these sites (Brenner et al. 1984; Andersen et al. 1989; Burger and Torbert 1990; Hughes et al. 1992). Second, legislation has encouraged the standardization of planted species, resulting in a homogeneous flora, a common problem with large-scale land rehabilitation efforts (Parmenter and MacMahon 1990).

While it is impossible to predict, results of this study raise the question of whether the full complement of species will ever reinvade the previously disturbed sites. Similarly, on a study of 35-year old reclaimed coal surface mines in Pennsylvania, Schuster and Hutnik (1987) noted the absence of a number of common hardwood species including hemlock *(Tsuga canadensis)*, rhododendron, and mountain laurel *(Kalmia latifolia)*. This problem is certainly due in large part to the stressful environmental conditions and the poor dispersal abilities of a number of native herbaceous species (e.g., Whigham 1974; Beattie and Culver 1981; Rust and Roth

1981). However, it is difficult for many native species to compete with the non-native species used for reclamation. These wide-scale reclamation practices impact not only the flora, but also the fauna of this region. For example, a number of studies have shown that animals are more commonly associated with volunteer than planted species on reclaimed sites (Brenner et al. 1984).

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Forested upland

Highwall

Coal seam

Bench

Outslope

Compacted mine spoils
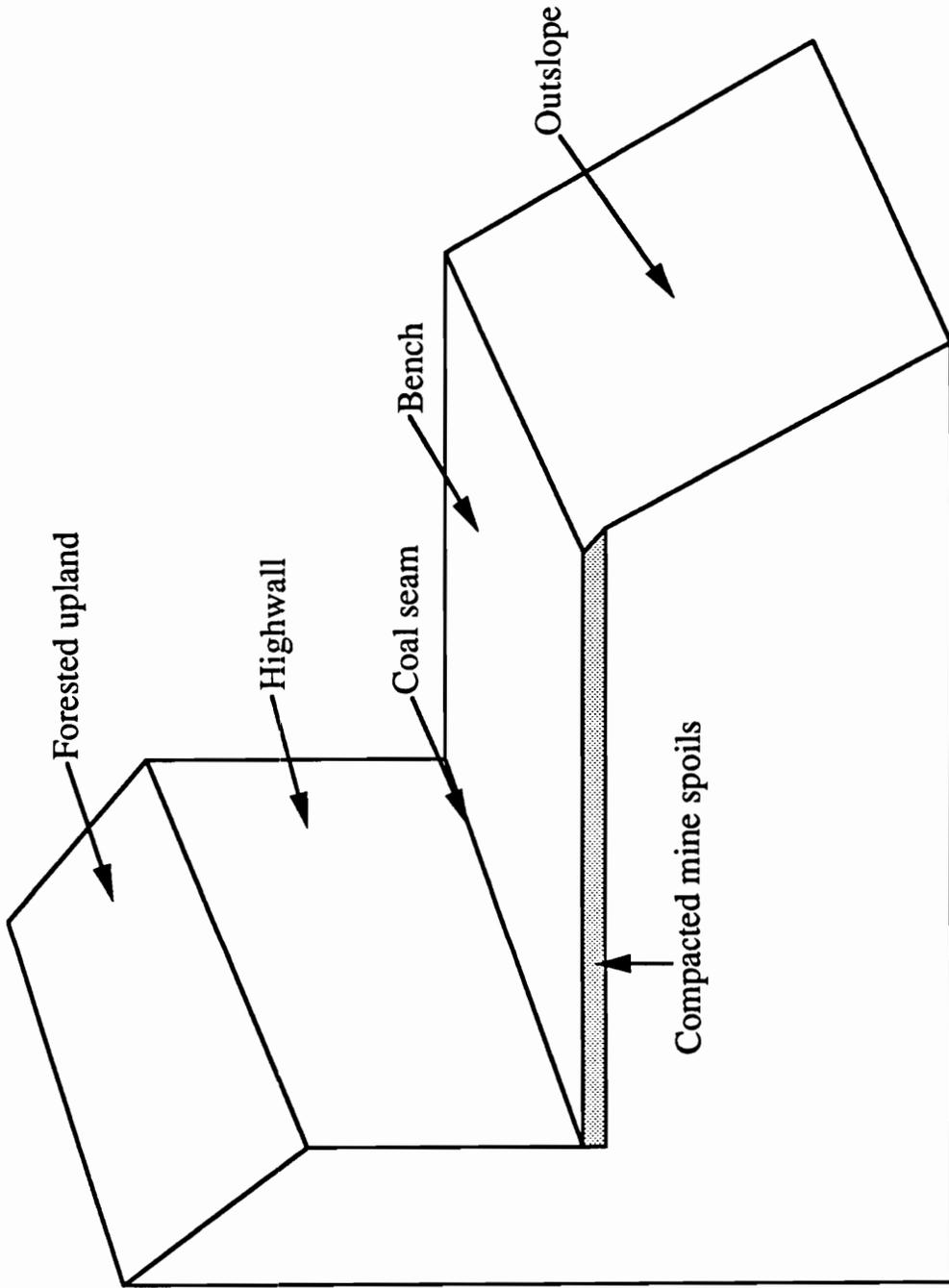
Figure 1.1 - Pre-SMCRA surface mined landscape.

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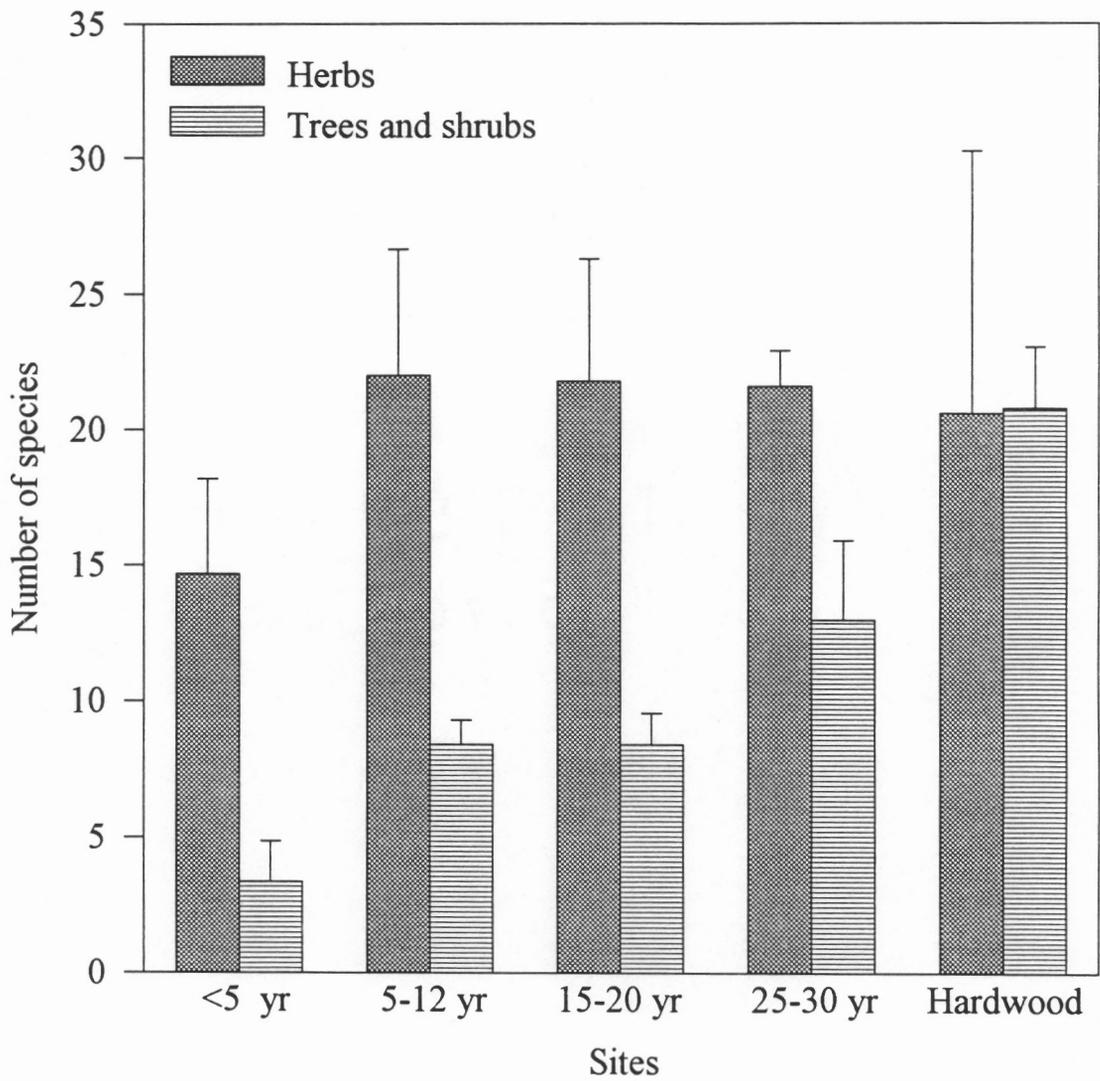
Figure 1.2 - Mean vegetational species richness by type of site. Error bars represent 1 SD.
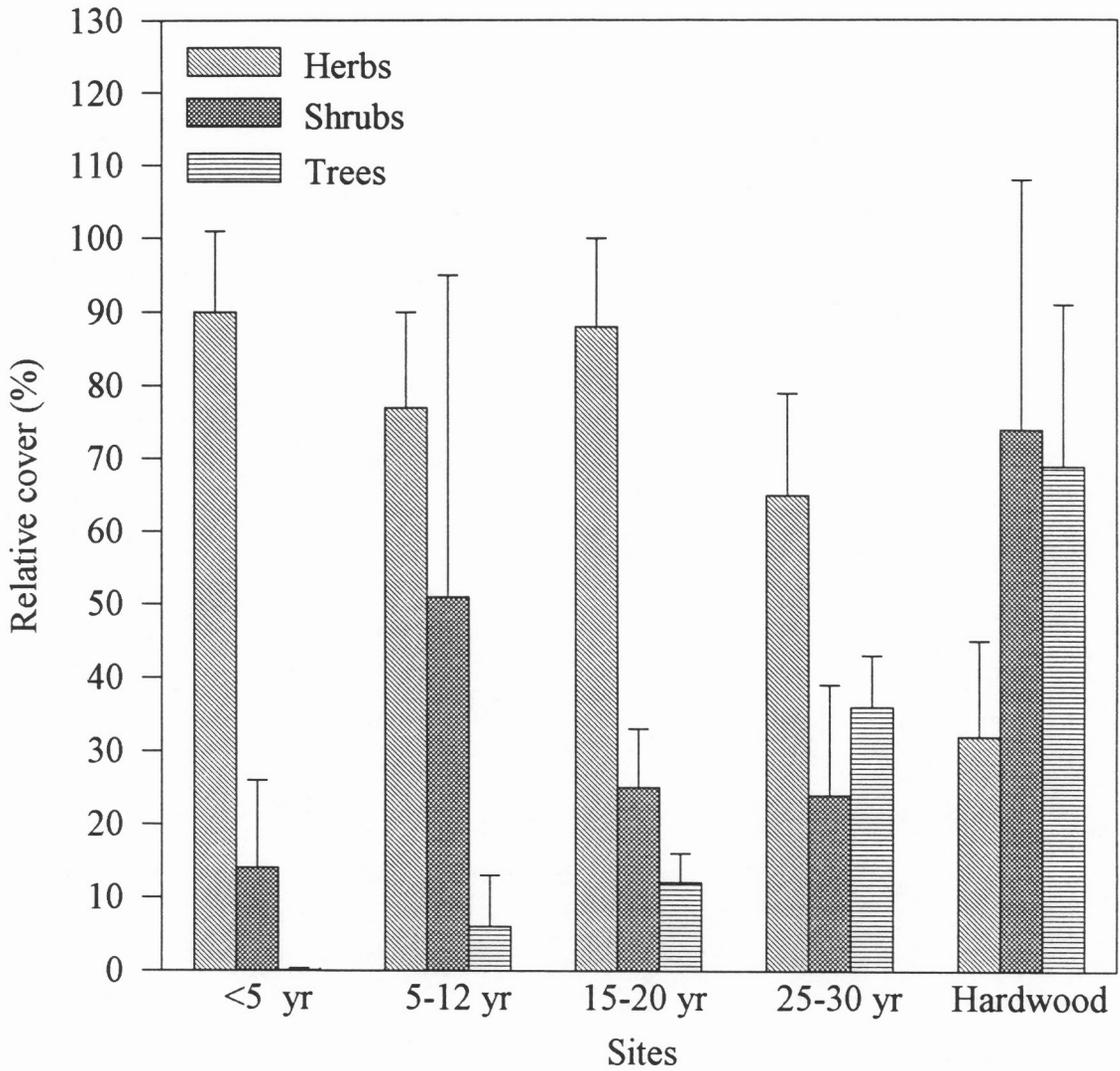
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Figure 1.4 - Vegetational site and species DCA scores. Sites are represented by symbols. Species are represented by the first three letters of the generic and specific epithets.
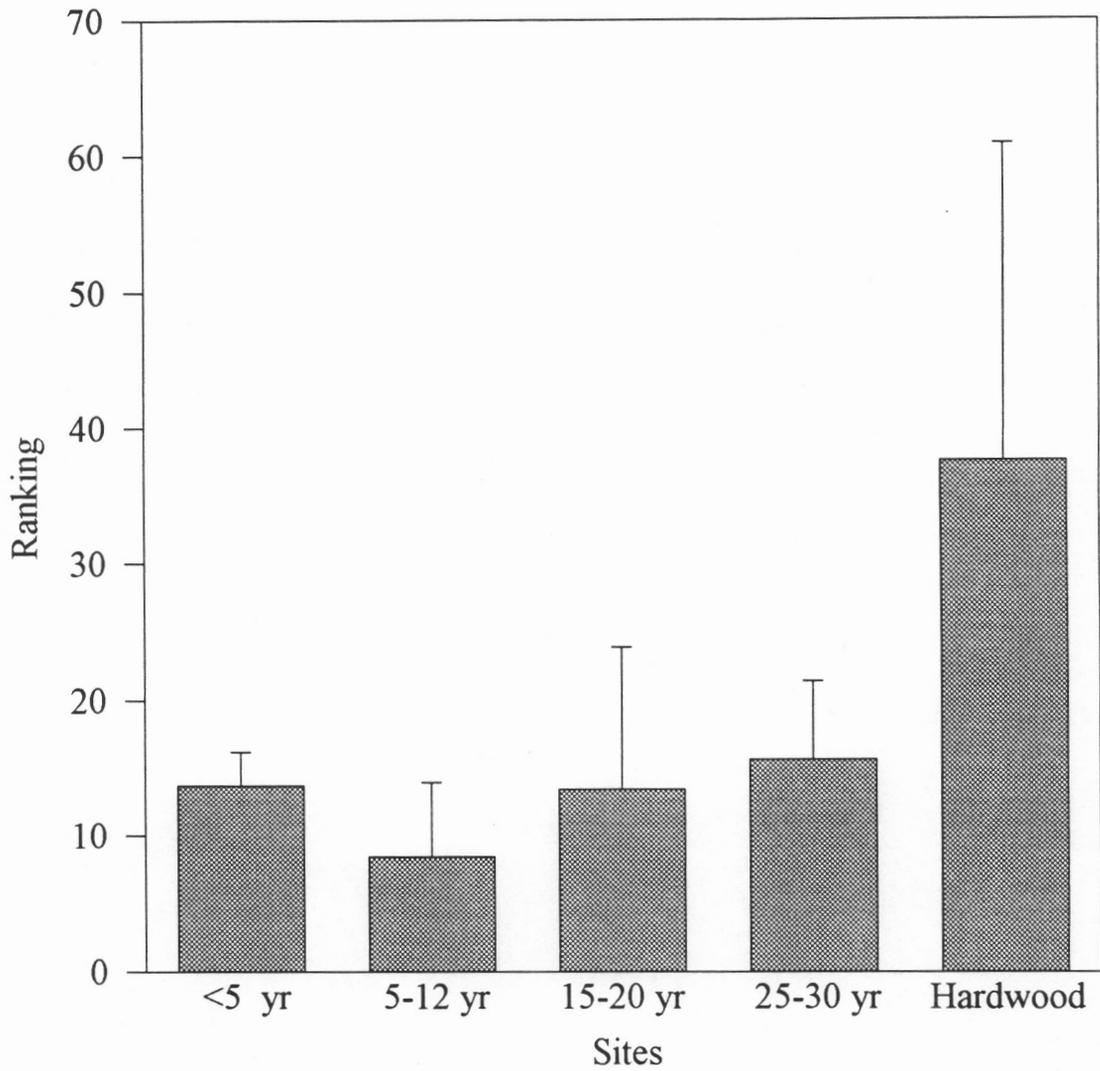
Figure 1.5 - Mean vegetational less common species ranking by type of site. Error bars represent 1 SD. Ranking system: species observed in 1 site = 5 points, 2 sites = 3 points, and 3 sites = 1 point.

Chapter 2

LEPIDOPTERAN COMMUNITIES

Introduction

While restored and reclaimed ecosystems are becoming an increasing part of

our landscape, most processes in these ecosystems are poorly understood. The

majority of land rehabilitation projects have focused on maintenance of certain abiotic

variables, such as erosion control and soil fertility, and the establishment of vegetation,

while the fauna has been almost universally overlooked. In a survey of articles on

restoration research, Majer (1991) found that only 6.3% of these articles cited the

fauna.

The lack of research on reestablishment of insect populations is particularly

alarming. Insects play critical roles in ecosystem functioning that are commonly

disrupted by human disturbance; these include pollination (e.g., Janzen 1974; Murphy

1984; Jennersten 1988), influencing plant succession (e.g., McBrien et al. 1983; Danell

and Ericson 1990), nutrient cycling (e.g., Springett 1978; Schowalter 1981), and spore

dispersal (Jennersten 1983). Studies of rehabilitated areas have only recently

addressed the recovery of these faunal communities (e.g., Hawkins and Cross 1982;

Parmenter and MacMahon 1987; Anderson 1993; Cullen and Wheater 1993; Williams

1993) despite the integral role they play in ecosystems processes. All of these studies

suggested that insect communities are fairly resilient to disturbance but concluded that

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more time was necessary to determine whether restoration had succeeded in reestablishing the predisturbance communities. These results are not surprising, as all these studies were performed on sites restored ≤8 years previously. The only research on lepidoptera in rehabilitated areas, the focus of this study, has been done on midwestern prairies. Panzer et al. (1987) and Selser and Schramm (1990) observed that restored prairies host a number of butterfly species, but often the less common species do not become reestablished.

Understanding of factors influencing lepidopteran community dynamics in minimally disturbed sites is limited. Knowledge of these issues in restored areas is nearly nonexistent. It is commonly recognized that lepidopteran communities are influenced by a number of factors including larval host plants, adult nectar resources, climate, distance to a colonizing source, and predators and parasites (Emmel and Emmel 1962; Gilbert and Singer 1975). However, the relative importance of these factors varies between systems and is widely debated. A better understanding of factors influencing lepidopteran communities is essential, as lepidoptera have been suggested repeatedly as indicator species (e.g., Weitzel 1982; Holloway 1984; Erhardt 1985; Wilcox et al. 1986; Kremen 1992).

This study investigated the diurnal lepidopteran communities on reclaimed coal surface mine sites and in the surrounding, minimally-disturbed hardwood forests. The goals of this work were: (1) to characterize the lepidopteran communities of these sites; (2) to determine if the reclaimed sites were hosting lepidopteran communities

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similar to those present prior to mining; (3) to investigate the degree to which the vegetation on these sites influences the lepidopteran communities; and (4) to assess the suitability of lepidoptera as indicators of ecosystem restoration.

For a number of reasons, areas reclaimed after coal surface mining provide one of the best opportunities for studying faunal population dynamics in rehabilitated systems. First, in contrast to most restoration projects which have been initiated in the past ten years, coal surface mines have been reclaimed for over 30 years, and reclamation has been mandated by law for over 15 years. Second, coal surface mining has caused widespread disturbance in both the southeastern and midwestern regions of the United States. Third, commonly utilized coal surface mine reclamation practices are typical of those of most large-scale restoration projects: a few aggressive plants species are used for revegetation in an effort to achieve ground cover requirements and maintain water quality. Little concern is given to the long-term development of the system, and the impact of reclamation practices on the fauna is rarely considered.

Methods

Field methods

The work described here was performed in Wise County, Virginia. For a detailed descriptions of the study sites, determination of the ages of the sites, and vegetation sampling methods see Chapter 1 (Field methods, p. 6). The diurnal lepidoptera were surveyed on 19 of the sites (excluding four of the sites reclaimed 25-

30 years previously) during the 1992 flight season and on all 23 sites during the 1993 flight season. Lepidopteran communities were sampled bimonthly, for a total of 12 times, between mid-April and mid-September. The line-transect method described by Douwes (1976) and modified by Erhardt (1985) was utilized (Figure 2.1, p. 49). Eight, permanently-marked, 62.5 m transects were walked, and all lepidoptera within 2.5 m to either side or in front of the observer were recorded. All transects were walked between 9:30 and 16:30 with transect time ranging between 15 and 45 minutes, depending on terrain and number of lepidoptera observed. The time of day a site was sampled was varied to avoid a bias against species whose activity is limited to a certain portion of the day. Minimum weather conditions used were those suggested by Pollard (1977); a minimum temperature of 13°C was required when clouds obscured the sun less than 40% of the transect time, and a minimum temperature of 17°C was necessary when conditions were cloudy more than 40% of the transect time. A number of researchers have suggested that, above a certain minimum, temperature does not affect the number of lepidoptera observed (Emmel and Emmel 1963; Douwes 1976; Pollard 1977; Erhardt 1985) and observations during this study confirm that idea. All sites were south-facing to minimize the effect of aspect.

To insure that sampling methods were equally thorough in different habitat types, in May and July of 1992 four consecutive transects were walked on two sites that were reclaimed 5-12 yr and 15-20 yr previously and on two hardwood sites. Percent similarities between lepidopteran communities observed on each of four

transect walks for a site on a sampling day were calculated.

All lepidoptera that could be reliably identified during transect walks were recorded, which excluded a number of smaller microlepidoptera. Larger species that could not be identified immediately were caught and preserved for later identification. If an individual could not be caught, it was included in the most common likely group (Pollard 1977). In a most cases, individuals were identified to species. However, the species in the genera *Agriphila, Colias, Erynnis, Herpetogramma, Poanes, Polygonia, Probole,* and *Thorybes* were combined for further analyses because of the difficulty of reliably determining the species of all individuals during transect walks.

Numerical analyses

The numbers of individuals seen on a transect each day were summed for the entire flight season. The logarithms of the abundances of each species were used for multivariate analyses, as there were a few species that were present in large numbers, while the majority of species were represented by <10 individuals. Since a few sites were sampled for only one flight season the data for each site from different years were analyzed separately.

Statistical analyses of trends in lepidopteran communities are identical to those described in Chapter 1 (Numerical analyses, p. 8). Spearman correlation coefficients between various lepidopteran and vegetation variables are reported here. Lepidopteran species richnesses and individual abundances from 1993 are given and were used for

correlations with environmental and vegetational variables. Determining actual

residence of a species in a site for such mobile organisms as lepidoptera is difficult

and somewhat arbitrary. Preliminary analyses of these data showed similar trends

regardless of whether all species or only more abundant species were considered. The

results presented here include all individuals found in a site.

Results

Table 2.1 shows the means by site type of the similarity of lepidopteran

communities observed during repeated transect walks in a site. The data suggest that

the sampling method was not biased towards a certain type of site.

*Table 2.1 - Average percent similarities of repeated
lepidopteran samples on different types of sites*

Site type	Mean similarity ± SD
Reclaimed 5-12 yr ago	77.2 ± 8.7
Reclaimed 15-20 yr ago	72.8 ± 10.1
Hardwood	75.0 ± 8.9

A total of 52 butterfly and 99 moth species were observed during transect

walks. Table 2.2 shows the means, medians, and ranges for lepidopteran community

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variables. Figures 2.2 and 2.3 (pp. 50 and 51) illustrate the species richness and individual abundance trends of the sites surveyed. Table 2.3 lists the correlations between variables in Table 2.2 and ranked site age.

Some lepidopteran species rapidly colonized reclaimed sites. While there was no trend in total lepidopteran species richness and abundance with site age, butterfly species richness and abundance decreased significantly with site age, and moth species richness and abundance increased significantly with site age. As would be expected, species richness and abundance were highly correlated for both butterflies and moths separately; total lepidopteran species richness and abundance were less strongly, though significantly, correlated. For the 19 sites surveyed in both 1992 and 1993, lepidopteran species richness was highly correlated between years ($r=0.73$, $p=0.0003$, $n=19$), while abundance of individuals was only weakly correlated ($r=0.52$, $p=0.0231$, $n=19$).

Table 2.2 - Means, medians, and ranges of lepidopteran community variables for sites in 1993 (n=23)

Variable	Mean	Median	Range
Species richness butterflies (SRB)	9.8	8	4-16
Species richness moths (SRM)	22.3	23	9-32
Species richness lepidoptera (SRL)	32.1	32	17-44
Abundance butterflies (ABB)	69.3	52	4-248
Abundance moths (ABM)	229.1	240	39-436
Abundance lepidoptera (ABL)	298.0	288	91-448

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Table 2.3 - Spearman rank correlation coefficients between lepidopteran community variables and site age (n=23) Abbreviations explained in Table 2.2.

Variable	SRB	SRM	SRL	ABB	ABM	ABL	AGE
SRB	1.0						
SRM	-0.27	1.0					
SRL	0.47*	0.72***	1.0				
ABB	0.75***	-0.43*	0.15	1.0			
ABM	-0.37	0.68**	0.36	-0.47*	1.0		
ABL	0.12	0.45*	0.50*	0.18	0.78***	1.0	
AGE	-0.57**	0.82***	0.33	-0.73***	0.66**	0.22	1.0

*p<0.05, **p<0.001, ***p<0.0001

The small variations in slope, aspect, and elevation did not have significant effects on any of the lepidopteran community variables, after removing the site type effect (p>0.05 in all cases). In no cases were the type-variable interaction effects significant. The lepidopteran dissimilarity and inverted geographic distance matrices for all sites were not significantly correlated using the Mantel test (p>0.05, n=23). The matrices for sites aged 15-20 yr were weakly negatively correlated (r=-0.48, p=0.009, n=5), while no other matrices were significantly correlated by type of site (p>0.05, n=5 in all cases).

Figure 2.4 (p. 52) shows the location of the lepidopteran communities of the sites during each year on the first two DCA axes. The first dimension of DCA of

lepidopteran communities has an eigenvalue of 0.532 and explains 24% of the lepidopteran community variation. This axis is strongly correlated with site age ($r=0.93$, $p<0.0001$, $n=40$). The second and third dimensions explain less than 5% of the variation. Lepidopteran communities of sites reclaimed ≤ 20 years previously fall close together along the first two DCA axes and far from the hardwood sites, while the lepidopteran communities of sites reclaimed 25-30 years previously are more similar to those of the hardwood sites. For the most part, lepidopteran communities on the same sites in different years are quite similar and are distinct from communities of other sites; however, there is some overlap in lepidopteran communities of earlier reclaimed sites, as these communities are more similar overall. The site scores on first DCA axes of the butterfly and moth communities analyzed separately are highly correlated with the first axis site scores for the entire lepidopteran communities ($r=0.91$ and $r=0.995$, $p<0.0001$ and $n=40$ in both cases), but the first axis of the moth analysis explains much more of the variation (25%, eigenvalue=0.546) than does the first axis of the butterfly analysis (15%, eigenvalue=0.292).

Figure 2.5 (p. 53) illustrates the locations of the 60 most common lepidopteran species on the first two DCA axes. For a full listing of all lepidopteran species and the types of sites in which they were observed see Appendix 2 (p. 110). The most rapidly invading species included clover looper moth (*Caenurgina crassiuscula*), sulphurs (*Colias* spp.), toothed somberwing (*Euclidea cuspidea*), eastern tailed blue (*Everes comyntas*), and pearl crescent (*Phyciodes tharos*). These species, as well as a

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number of others, such as cross-lined wave *(Calothysanis amaturaria)*, common wood nymph *(Cercyonis pegala)*, confused eusarca *(Eusarca confusaria)*, and grayish zanclognatha *(Zanclognatha pedipilalis)*, were commonly found in sites reclaimed 15-20 years previously. Later successional reclaimed sites were characterized by few butterfly species and a number of moth species common in the hardwoods: snowy geometer *(Eugonobapta nivosaria)*, sober renia *(Renia sobrialis)*, and white-striped black *(Trichodezia alvobittata)*. Common hardwood species included a microlepidopteran *(Blepharomastix ranalis)*, morbid owlet *(Chytolita morbidalis)*, decorated owlet *(Pangrapta decoralis)* and discolored renia *(Renia discoloralis)*. A few species, such as haploas *(Haploa* spp.), swallowtails *(Papilio* spp.), lacewing moth *(Scopula limboundata)*, and faint-spotted angle *(Semiothisa ocellinata)*, were found in all types of sites.

Table 2.4 illustrates the correlations between vegetational and lepidopteran community variables. Butterfly species richness and abundance were significantly positively correlated with herbaceous cover and were negatively correlated with tree species richness and basal area. Moth species richness and abundance were positively correlated with a number of vegetational variables, but were most strongly correlated with overall vegetational species richness and tree basal area respectively. Lepidopteran species richness was strongly correlated with herbaceous species richness and weakly correlated with overall vegetational species richness, but these results were primarily due to the high correlations between moth and vegetational species richness.

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Table 2.4 - Spearman rank correlation coefficients between vegetational and lepidopteran community variables (n=23) (SRH - species richness herbs, SRT - species richness trees, SRV - species richness vegetation, CH - cover herbs, CS - cover shrubs, BAT - basal area trees, SRB - species richness butterflies, SRM - species richness moths, SRL - species richness lepidoptera, ABB - abundance butterflies, ABM - abundance moths, ABL - abundance lepidoptera)

Variable	SRB	SRM	SRL	ABB	ABM	ABL
SRH	0.21	0.53*	0.64**	0.04	0.25	0.30
SRT	-0.48*	0.63*	0.23	-0.68	0.50*	0.08
SRV	-0.20	0.79***	0.58*	-0.44*	0.52*	0.26
CH	0.51*	-0.52*	-0.11	0.49*	-0.35	-0.04
CS	-0.25	0.14	-0.06	-0.27	0.23	0.07
BAT	-0.64**	0.60*	0.09	-0.69**	0.57*	0.14

*p<0.05, **p<0.001, ***p<0.0001

A number of results indicate that, as would be expected, vegetation has a strong influence on lepidopteran communities. The correlation between the first dimensions of vegetational and lepidopteran DCA was very high (r=0.91, p<0.0001, n=40). Both axes were strongly correlated with time. While the lepidopteran, butterfly, and moth dissimilarity matrices were all significantly correlated with the vegetational dissimilarity matrices using the Mantel test, the lepidopteran and moth matrices were more highly correlated than was the butterfly matrix (Table 2.5).

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Table 2.5 - Comparison of the lepidopteran, butterfly, and moth dissimilarity matrices with the vegetational dissimilarity matrix using the Mantel test (10,000 permutations) (n=23)

Dissimilarity matrix	r	ρ [random $Z \leq$ observed Z]
Lepidopteran	0.78	0.0001
Butterfly	0.49	0.0001
Moth	0.79	0.0001

Figure 2.6 (p. 54) illustrates the less common species rankings. The hardwood sites hosted a greater number of less common species than the reclaimed sites. As with the vegetation, the less common species were not evenly distributed among the hardwood sites. Few less common species were found in the oldest reclaimed sites. However, it is important to reiterate that four of these sites were only sampled for one field season. Lepidopteran species richness was weakly correlated with less common species ranking (r=0.46, p=0.0262, n=23). The less common species rankings of the vegetational and lepidopteran communities for each site were not significantly correlated (r=0.17, p=0.4374, n=23).

Discussion

Given the high degree of disturbance and the little effort made on long-term ecosystem management, the lepidopteran communities of reclaimed coal surface mines appear to be quite resilient, as do insect communities in other restored temperate systems (e.g. Selser and Schramm 1990; Cullen and Wheater 1993; Williams 1993). The DCA of the vegetation and lepidopteran communities suggests that the lepidopteran communities on later successional sites are more similar to the hardwoods than are the vegetational communities. This trend is not surprising, as the majority of temperate lepidopteran species are fairly mobile, though dispersal ability varies greatly among species. Therefore, lepidoptera have the ability to colonize rapidly and may be found in areas that do not possess all necessary habitat requirements.

A second interesting trend revealed by the DCA analyses is the separation between sites reclaimed ≤ 20 years and ≥ 25 years previously. It appears that while reclaimed sites are rapidly colonized initially, there may be a stalling in community development on these sites. Generally, it has been observed that succession does not proceed at a constant rate, but is a periodic process (e.g., Gleason 1927; Tansley 1935). In the case of reclaimed sites, commonly planted aggressive vegetational species may slow the rate of succession. These results, like those of the vegetation study, suggest that five years is insufficient to judge reclamation success.

As with the vegetation, there is a great deal of unexplained variation in the DCA of lepidopteran communities. This variation is certainly due in part to

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vegetational differences. It also reflects the mobility of the group being studied. Diurnal lepidoptera commonly pass through habitats in which they do not reside. As discussed earlier, determining degree of utilization of sites by lepidoptera is difficult. Diurnal lepidoptera may move large distances in search of nectar resources, to thermoregulate, to encounter mates, or to oviposit. Landscape variables may also explain some of the variation.

Reclaimed sites host a large number of both species and individuals of lepidoptera, particularly butterflies. Most of these species, however, are widespread geographically, as they are well adapted to other areas of human disturbance such as roadsides and vacant lots. These species tend to be the classic r-selected species with high mobility, the ability to utilize a wide variety of food resources, and multiple broods per flight season (discussed further in Chapter 4, p. 91); examples include sulfurs, eastern tailed blue, and silver spotted skipper *(Epargyreus clarus)*. The hardwood sites host more of the less common species, which are patchily distributed like the less common vegetational species.

In this study, differences in slope, aspect, and elevation had little influence on lepidopteran communities. These results were to be expected as sites were chosen that varied little with respect to these factors. Considering the degree of mobility of lepidoptera it is somewhat surprising that relative distances between sites were not more important. The geographic distance and lepidopteran dissimilarity matrices were significantly, though weakly (r=-0.48), correlated only for the sites reclaimed 15-20

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years previously. The lack of correlation between geographic distance and lepidopteran communities dissimilarities suggests that either (1) within-site factors were more important in determining lepidopteran community composition; (2) distances between sites were sufficiently large to preclude any contagion effects; or (3) sample sizes were insufficient to detect a relationship between the two.

Clearly, plant community composition plays a major role in determining habitat suitability for lepidoptera. Other studies have shown the number of diurnal lepidopteran (Erhardt 1985) and butterfly (Thomas and Mallorie 1985; Murphy and Wilcox 1986) species to be correlated with plant species richness. However, in this study, only moth species richness was strongly correlated with plant species richness. Multivariate statistical analyses suggested a strong correlation between lepidopteran and vegetational community composition. One indication was the strong correlation between the first DCA axes of both communities. However, the DCA analyses were strongly dominated by the first axes and little variation was explained by further axes. The Mantel test comparing lepidopteran and vegetational matrices indicated high correlation between the dissimilarities of these two communities, although the correlation coefficient was greater for the moth than butterfly dissimilarity matrix.

While appropriate host plants are a basic requirement for lepidoptera, more complex habitat requirements may also influence lepidopteran communities (Owen 1959; Gilbert and Singer 1975; Vane-Wright 1978). As will be discussed in the next chapter, nectar resources appear to be the primary factor influencing the adult butterfly

communities observed in this study. Another factor that has been demonstrated to influence the distribution of certain species is canopy cover (e.g., Warren 1985; Weiss and Murphy 1990; Greatorex-Davies et al. 1993); butterflies, in particular, may move to areas to thermoregulate (Clench 1966). In this study, butterfly species richness and abundance were highly negatively correlated with tree basal area (r=-0.64 and -0.69), which is a good indicator of canopy cover (Spurr 1960). Butterfly species richness was positively correlated with herbaceous cover (r=0.51), a trend observed in other studies (e.g., Thomas and Mallorie 1985). It is impossible without further study to determine if these correlations reflect an actual influence of plant structure on butterfly communities, or if they are the result of an association with nectar resources. As with any faunal group, a number of factors may influence any one species, making it difficult to determine the predominant factors influencing the entire community.

Given this challenge, the often suggested suitability of diurnal lepidoptera as indicator species remains in question. It is important when considering the use of an indicator species to specify what abiotic or biotic factors the species are being used to indicate (Landres et al. 1988). Erhardt (1985) and Murphy and Wilcox (1986) suggest lepidoptera as indicators of plant species richness, while both Holloway (1984) and Kremen (1992) disagree with this assertion. Holloway suggests moths as an indicator of plant community type, and Kremen's work shows butterflies to be distributed along a topographic/moisture gradient. Murphy and Wilcox (1986) found butterfly species richness to be correlated with bird and mammal species richness. These varied results

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may be due to differences in the spatial scales of the studies. However, they suggest that lepidoptera are not reliable indicators of any single habitat variable.

Results of this study suggest that diurnal moths are better indicators of plant species richness and community composition than are butterflies. In this study, butterflies were extremely poor indicators of plant species richness. Furthermore, a number of species including *Erynnis juvenalis, Pompeius verna,* and *Vanessa atalanta,* were found in sites where known host plants were not found, which demonstrates that the presence of adult butterflies does not necessarily indicate habitat suitability even for an individual butterfly species. Butterflies appeared to be more influenced by the presence of nectar resources, but were not very selective in the plant species they visited for nectar. However, butterflies have been more often suggested as indicators because they are relatively easy to identify, are recognized by the public, and have relatively well-known life histories. Moths, however, do not share these characteristics. Even in temperate systems such as the one studied, most moths have poorly-documented life histories and their identification can be challenging. A logical continuation of this work would be to sample nocturnal moths. However, it is even more difficult to determine residence of these species in a habitat than for diurnal lepidoptera, and sampling is difficult and time consuming. Another interesting extension of this work would be to examine whether butterflies could be used as indicators for the reestablishment of insect communities as a whole or other faunal guilds as has been suggested (e.g., Murphy and Wilcox 1986; Kremen et al. 1993).

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This seems unlikely as only a few butterfly species are commonly found in the hardwood areas in this region.

Species conservation would be easier if "rare species occur in, and therefore benefit from the conservation of, species rich habitats" (Prendergast et al. 1993, p. 335). In the current study, there was a significant, though relatively weak, correlation between lepidopteran species richness and less common species rankings. Thomas and Mallorie (1985) surveyed butterfly communities on 500 m transect walks in various habitat types in the Atlas Mountains of Morocco. They found a weak, though significant, correlation between rare butterflies and overall butterfly species richness. Prendergrast et al. (1993) compared total species richness and number of rare species, determined by geographic distribution, of butterflies in 10 km^2 areas in Great Britain. They found that protecting the 5% most species rich areas would include 11 of 14 rare or uncommon butterfly species. While conducted at different scales, these studies all suggest that focusing conservation efforts on species rich areas will protect many, but not all uncommon species. Conservation would also be facilitated if areas of high numbers of rare species in different groups coincided. Areas of high numbers of less common species of lepidoptera and plants were not correlated in the current study.

In drawing conclusions, it is important to recognize that this study is one of the first to address the influence of land rehabilitation practices on lepidopteran communities and to consider the use of lepidopteran species as indicators of rehabilitation success. The continued study of lepidopteran communities on reclaimed

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mine sites, and rehabilitated areas in general, is important for the conservation of this faunal group because of its important role in ecosystem development. However, lepidoptera appear to have limited use as indicators of revegetation success. The use of indicator species has generally been recommended to facilitate evaluation of the health of an area. Even sampling of butterflies on this terrain is challenging and must be repeated a number of times because of the high turnover of species during the flight season. It would seem to be more efficient to measure vegetation, using community composition in addition to current cover criteria, than to use lepidoptera as indicators of vegetational succession.

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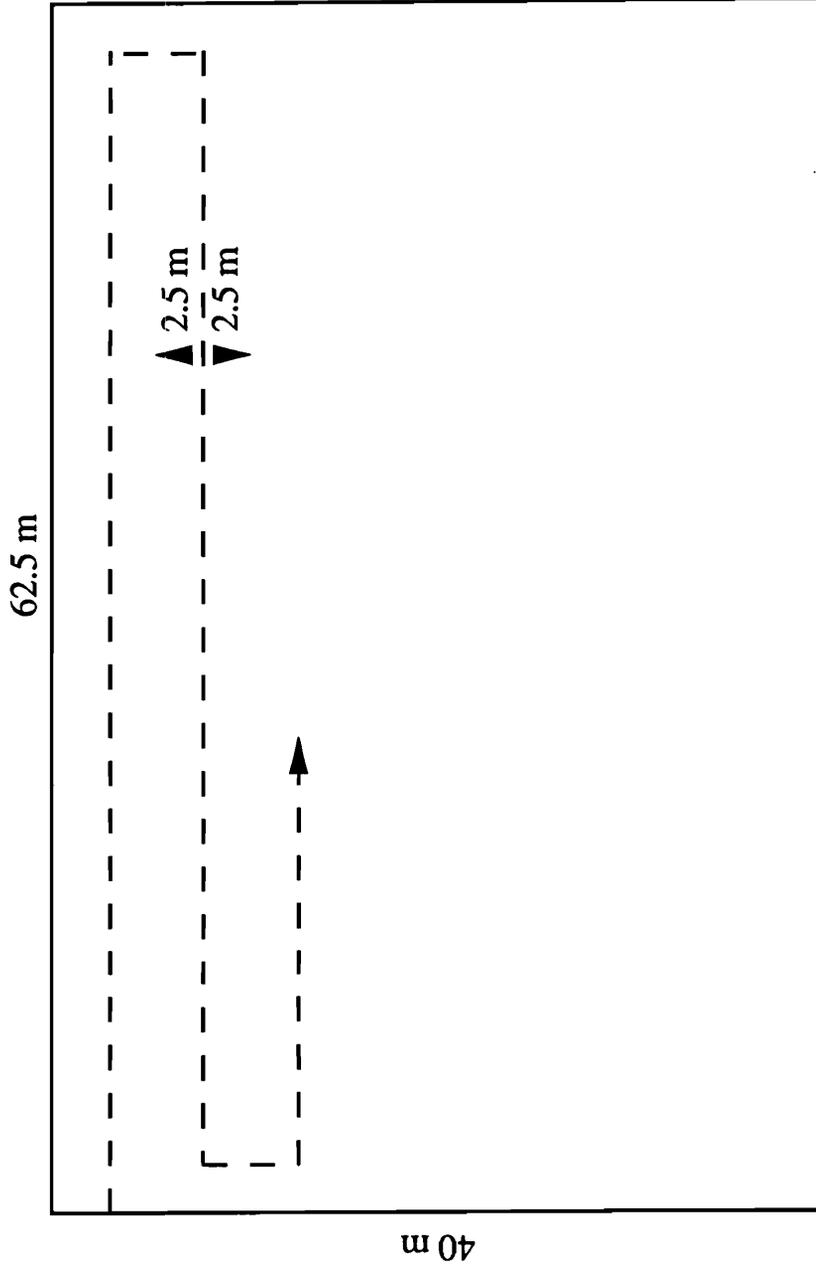
Figure 2.1 - Diagram of lepidopteran sampling method.
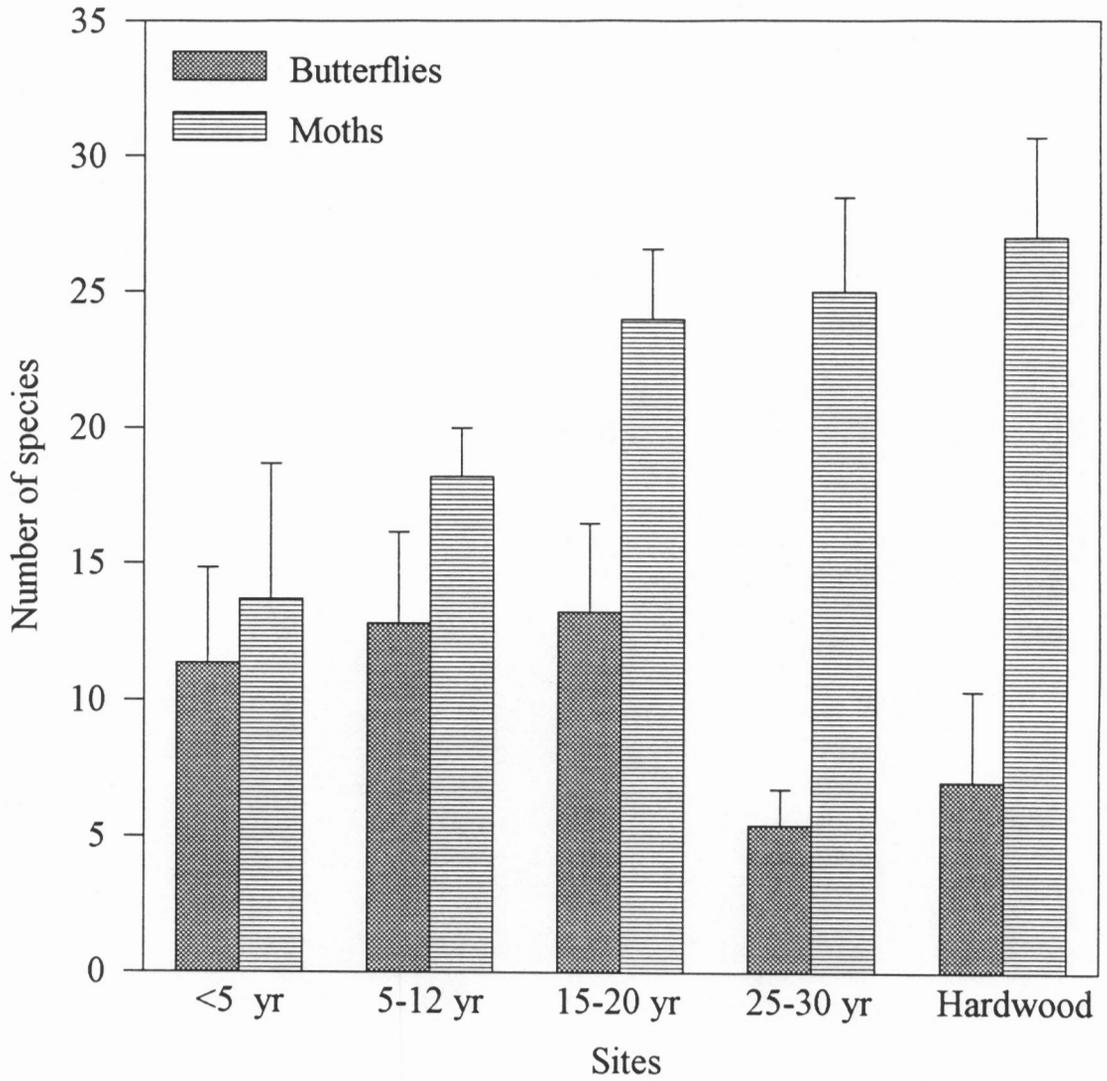
Figure 2.2 - Mean lepidopteran species richness by type of site.
Error bars represent 1 SD.
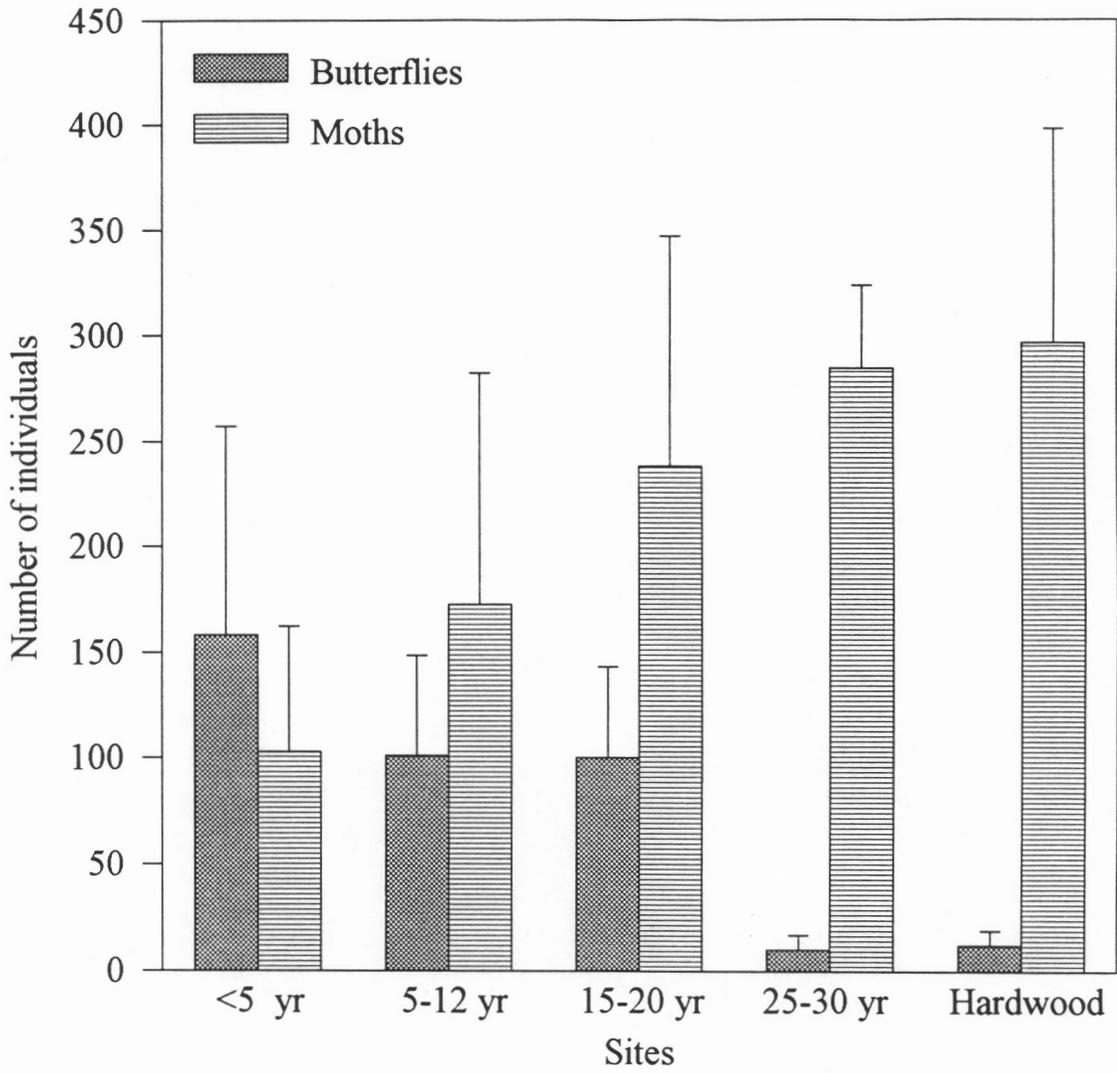
Figure 2.3 - Mean lepidopteran abundance by type of site. Error bars represent 1 SD.
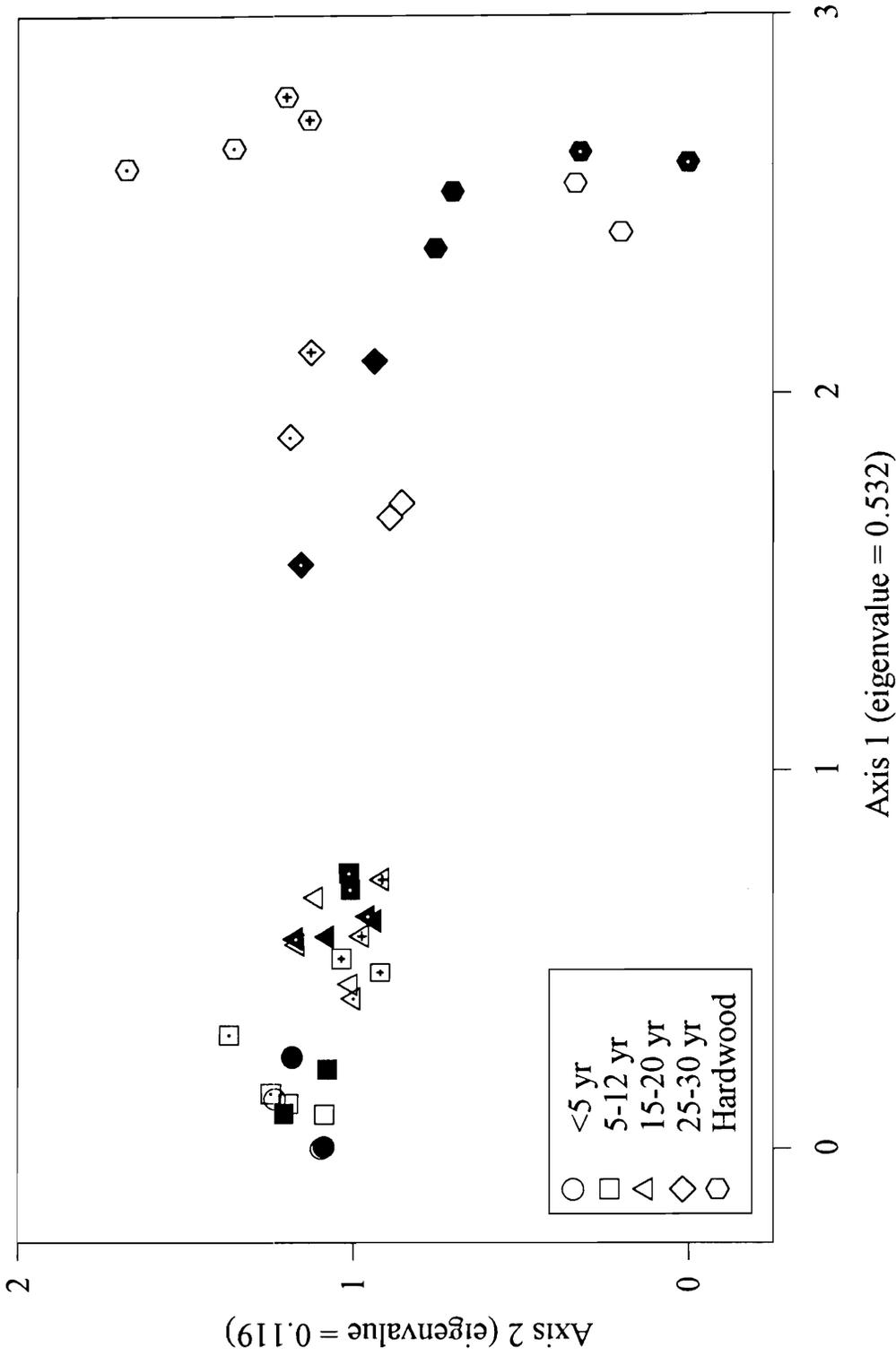
Figure 2.4 - Lepidopteran site DCA scores. Samples of the same site in different years are indicated by the same fill type.

Figure 2.5 - Lepidopteran species DCA scores. Species are represented by the first three letters of the generic and specific epithets. Sites are represented by crosshairs (refer to Figure 2.4).

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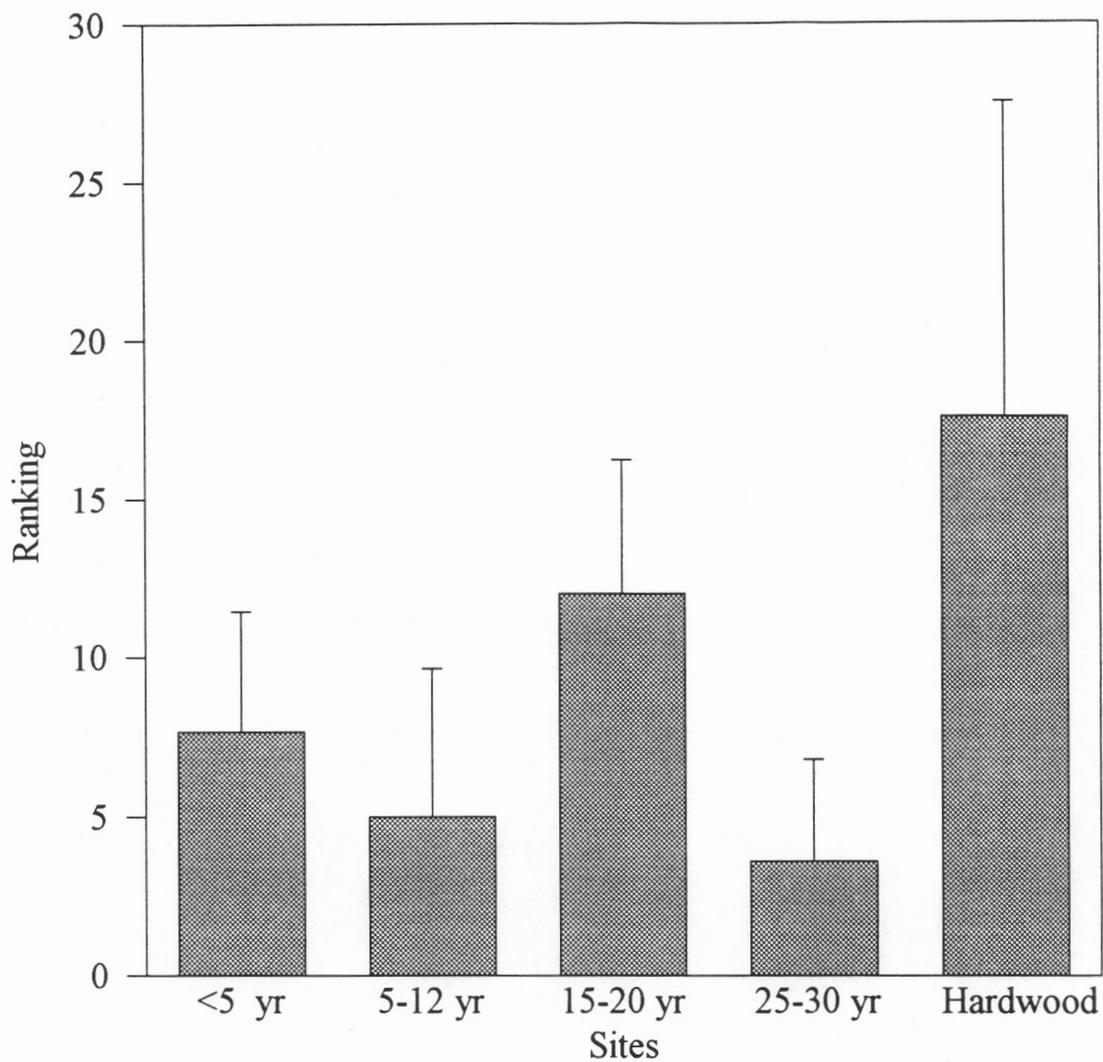
Figure 2.6 - Mean lepidopteran less common species ranking by type of site. Error bars represent 1 SD. Ranking system: species observed in 1 site = 5 points, 2 sites = 3 points, and 3 sites = 1 point.

Chapter 3

NECTAR RESOURCES AND THEIR INFLUENCE ON LEPIDOPTERAN

COMMUNITIES

Introduction

The ability of rehabilitated land to support any faunal community depends on a

number of habitat variables. While larval host plants are requisite for the survival of

lepidoptera, they are only one of a suite of factors, including nectar resources, that

may determine habitat suitability. Nectar resources are not only important for

lepidoptera, but also for many other insects including diptera, hymenoptera, and a

number of coleoptera (Hocking 1953).

Most adult butterflies obtain the majority of their energy and nutrients from

flower nectar, although tree sap, rotting fruit, animal dung, and carrion are important

sources for a number of species (Opler and Krizek 1984). Studies on individual

species have shown nectar intake by butterflies to be positively related to longevity

and fecundity (e.g., Murphy et al. 1983; Lederhouse et al. 1990; Boggs and Ross

1993). A number of studies suggest that nectar resources more strongly affect

butterfly abundance than does the presence of suitable host plants (e.g., Ehrlich and

Gilbert 1973; Gilbert and Singer 1973; Murphy 1983; Kremen 1992; Munguira and

Thomas 1992). Although some studies have demonstrated correlations between

butterfly species richness and either flowering plant species richness (Kremen 1992) or

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flowering plant number (Munguira and Thomas 1992), the influence of nectar resources on lepidopteran community composition is not universally supported by past studies (Sharp et al. 1975).

The purpose of this study was to quantify the nectar resources provided by reclaimed coal surface mined sites and to determine the effect of nectar resources on lepidopteran community composition on these sites. A further aim was to determine whether inflorescence number is a good indicator of the amount of sugar produced. Most previous studies have investigated correlations between butterfly abundance and species richness and the number and species richness of plants in flower without directly quantifying nectar abundance. However, nectar volume and concentration vary widely among plant species.

Methods

Nectar measurements

Most scientists studying the relationship between insects and nectar have quantified nectar abundance using a micropipette or capillary tube to extract and measure the volume of the nectar and a pocket refractometer to assay the sugar concentration (e.g., Hocking 1953; Watt et al. 1974; Sharp et al. 1975). This technique, however, cannot be used for extracting small amounts of nectar (<1 μl), which does not allow for the measurement of the nectar of plants with numerous small florets, such as the majority of species in the Asteraceae. Previous studies have

overlooked the nectar produced by these plants. Over three-quarters of the plant species commonly utilized as nectar sources in the present study produce insufficient amounts of nectar per floret to allow for accurate measurement by the technique described. The laboratory assay described by McKenna and Thomson (1988), using small pieces of filter paper for nectar extraction, is useful for measuring minute quantities of nectar. It also results in more accurate sampling of species that produce sufficient amounts of nectar for measurement by the traditional method, as nectar is more thoroughly extracted using filter paper than a capillary tube. The drawbacks of this technique are (1) it only allows for quantification of total sugar and not volume and concentration separately; and (2) it requires a great deal more time.

The number of flowers or inflorescences of 32 species were estimated during the bimonthly lepidopteran surveys of each site (see Chapter 2, Field Methods, p. 28). The smallest feasible unit of flowering was used for quantification (see Table 3.1, p. 62). Plants considered as nectar sources were determined by records of butterfly species' nectar utilization (Opler and Krizek 1984), reports of pollinator visits (Müller 1883; Beattie 1971; Rust 1977; Bertin and Sholes 1993), and personal observation. Species that were only represented by a few inflorescences were not sampled. A few plant species that are common in southwestern Virginia, and on which butterflies occasionally alight, do not produce nectar; these include *Clematis virginiana* (H. Baker pers. comm.), *Coronilla varia* (Al-Tikrity et al. 1970), *Lysimachia quadrifolia* (Simpson et al. 1983), *Rosa multiflora* (H. Baker, pers comm.), and *Solanum americanum* (H.

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Baker, pers. comm.).

Inflorescences were bagged with mosquito netting for 24 hours prior to sampling in order to prevent the majority of insects from extracting nectar; however, ants and a few other small insects were able to pass through the netting on occasion. The nectar was removed using small (approximately 2 mm x 10 mm) pieces of Whatman #1 filter paper, attached to insect pins to prevent contamination by secretions from the hands. Samples were taken between 12:00 and 17:00, as sampling earlier in the day could be confounded by extensive dew. Samples were allowed to dry on a pinning board overnight, and then were wrapped in plastic and stored in a freezer until being assayed in the laboratory. At the time of sampling, the number of florets per head and flowers per inflorescence, that were open and producing nectar, were estimated for each species.

In the laboratory, the sugar in 10 samples of each species was redissolved and measured using the anthrone method to determine the total carbohydrate content. Details of the assay are described in McKenna and Thomson (1988). As reports of nectar sugar composition for the majority of species assayed were not available, 1:1 fructose:glucose standards were used. McKenna and Thomson (1988) suggest that the method is relatively insensitive to the sugar composition of the standards.

For each species mean amount of sugar per sample was multiplied by the number of florets per head or flowers per inflorescence to calculate the nectar produced by the flowering unit. These numbers were multiplied by the total number

of inflorescences observed flowering on each sampling date and the products summed to give an estimate for total sugar produced during the entire flight season. Some long-lived flowers were counted more than once and some were not counted, as they completed flowering between lepidopteran sampling dates, but the same method was used for each site, allowing for comparison.

For the genera *Solidago* and *Erigeron* it was impossible to identify the species of all individuals flowering. For Solidago the average of the means for the two most common species, *S. gigantea* and *S. rugosum*, was used, and for *Erigeron* the value of the more common species, *E. philadelphicus*, was used. While it was possible to identify all species of *Viola*, some were represented by few flowers, and the average of the means of the most common species, *V. hastata* and *V. septemloba*, was used. Similarly, for *Melilotus* spp. the value for *M. officinalis* was used.

Butterfly behavior

The lepidopteran communities on 19 and 23 reclaimed and hardwood sites were sampled 12 times during the flight seasons of 1992 and 1993, respectively (see Chapter 2, Field methods, p. 28). The time, temperature, and percentage of transect in full sunlight were noted on each date. The behaviors of approximately 85% of observed butterflies were recorded and categorized as flying, basking, nectaring, mating, or chasing. If a butterfly was observed nectaring, the nectar source was recorded. Behaviors of the remaining 15% of butterflies were not recorded for one of

three reasons: (1) the behavior was not noted before the butterfly was disturbed by the observer; (2) numerous butterflies were sighted simultaneously, precluding recording of all behaviors; or (3) the tape recorder malfunctioned. Behaviors of moths were not recorded as they were nearly always basking unless disturbed by the observer. The number of butterflies recorded nectaring may have been slightly overestimated, as it was not always possible to verify that a butterfly had extended its proboscis into a flower due to disturbance by the observer and the large number of butterflies observed.

Numerical analyses

Spearman rank correlation coefficients between lepidopteran, vegetational, and nectar variables are reported here. The nectar values from 1993 were used for correlations between vegetational and nectar variables because four of the later successional sites were not sampled during 1992. Correlations between lepidopteran and nectar variables were calculated both for the entire flight season and by sampling date. Correlations between lepidopteran and nectar variables were calculated separately for each flight season. In only one case were the correlations significantly different between years, so values for the two years were combined and the one difference noted.

Results

<u>Nectar resources</u>

Table 3.1 lists the following values for each nectar source: (1) the means and standard deviations of the amount of sugar per sample; (2) the number of samples assayed; (3) the type of inflorescence used for abundance estimates during transect walks; (4) the number of florets per head or inflorescence; and (5) the total nectar per inflorescence. Table 3.2 lists previously reported values of nectar sugar content for a few species. It is difficult to assess the accuracy of the method utilized compared to other techniques because of the low number of species for which values have been reported and the large number of factors that can influence nectar production (reviewed by Hocking 1953). However, most of the values obtained in this study are similar to those previously reported.

Table 3.1 - Nectar production by various plant species (MS - mean amount of sugar produced in μg, SD - standard deviation, N - number of samples, INF - type of inflorescence, NF - number of florets or flowers per inflorescence, TN - amount of nectar per inflorescence in mg [MS*NF/1000])

Species	MS ± SD	N	INF	NF	TN
Achillea millefolium	45 ± 35	10	corymb[1]	200	9.00
Alliaria petiolata	90 ± 51	8	flower	1	0.09
Aster divaricatus	105 ± 59	10	head	6	0.63
Aster pilosus	78 ± 69	10	head	10	0.78
Barbarea vulgaris	158 ± 62	10	flower	1	0.16
Buddleya davidii	102 ± 34	10	thyrse	15	15.30
Chrysanthemum leucanthemum	65 ± 49	10	head	50	3.25
Daucus carota	24 ± 16	10	umbel	100	2.40
Erigeron philadelphicus	45 ± 33	10	head	50	2.25
Eupatorium maculatum	126 ± 77	10	corymb[2]	500	63.00
Eupatorium rugosum	113 ± 61	10	corymb[3]	500	56.50
Geranium maculatum	287 ± 126	10	flower	1	0.29
Helianthus microcephalus	144 ± 115	10	head	7	1.01
Hierarcium pratense	21 ± 11	10	head	10	0.21
Lespedeza bicolor	211 ± 85	10	flower	1	0.21
Lespedeza cuneata	142 ± 61	10	stem[4]	20	2.84
Lotus corniculatus	155 ± 85	10	umbel	5	0.78
Medicago sativa	62 ± 47	10	head	10	0.62
Melilotus officinalis	150 ± 86	10	raceme	10	1.50
Potentilla canadensis	224 ± 148	10	flower	1	0.22

[1]corymb of 50 heads with 4 florets. [2]corymb of 100 heads with 5 florets. [3]corymbs of 50 heads with 10 florets. [4]flowers are solitary but were estimated per stem.

Table 3.1 - (continued)

Species	MS ± SD	N	INF	NF	TN
Robinia hispida	2755 ± 945	10	flower	1	2.76
Rubus allegheniensis	894 ± 504	10	flower	1	0.89
Senecio aureus	14 ± 6	10	head	10	0.14
Solidago gigantea	62 ± 54	10	panicle[5]	200	12.40
Solidago rugosa	48 ± 13	10	panicle[5]	200	9.60
Solidago spp.	55	20	panicle[5]	200	11.00
Taraxacum officinale	75 ± 47	10	head	50	3.75
Trifolium pratense	38 ± 38	8	head	50	1.90
Trifolium repens	42 ± 32	10	head	40	1.68
Tussilago farfara	114 ± 54	10	head	30	3.42
Vicia villosa	101 ± 128	10	raceme	10	1.01
Viola hastata	108 ± 59	10	flower	1	0.11
Viola septemloba	185 ± 103	10	flower	1	0.19
Viola spp.	147	20	flower	1	0.15

[5]panicle of 100 heads with 2 florets.

Table 3.2 - Values for the amounts of sugar in this study compared to those previously reported. Abbreviations explained in Table 3.1

Species	INF	MS	Literature value	Source
Daucus carota	umbel[1]	2400	52.5-2670	Erickson and Peterson 1979
Lotus corniculatus	flower	155	45	Szabo and Naida 1985
Medicago sativa	floret	62	41	Szabo and Naida 1985
Trifolium pratense	floret	38	47	Szabo and Naida 1985
Trifolium repens	floret	42	42	Szabo and Naida 1985
Vicia villosa	floret	101	29.5	Weaver 1979

[1]The type of inflorescence listed here differs from that in Table 3.1 to correspond with the inflorescence type reported reported in the literature.

For the 19 sites sampled in both years the correlations between sugar abundance (r=0.86, p<0.0001), number of inflorescences (r=0.87, p<0.0001), and nectar plant species richness (r=0.92, p<0.0001) in the two years were quite high, as would be expected. The notable exception was the most recently reclaimed site, in which the nectar abundance in the second year was only about 20% of that in the first year. This result is explained by the fact that the majority of nectar in the first year was produced by *Lotus corniculatus*, a planted species that rapidly dies out when shaded. In most cases the number of inflorescences was slightly lower in 1993, which may have been the result of much drier conditions.

Both when analyzed by individual sampling date and summed over the entire flight season, sugar abundance was highly correlated with number of inflorescences

(r=0.94 and r=0.89, p<0.0001 in both cases, n=42 and n=504) and nectar plant species richness (r=0.87 and r=0.82, p<0.0001 in both cases, n=42 and n=504). As illustrated by Figure 3.1 (p. 83), the more recently reclaimed sites are outliers, resulting from the fact that the vegetation on these sites was predominantly comprised of a few planted species that have large numbers of inflorescences with little nectar.

Figure 3.2 (p. 84) shows mean sugar abundance and nectar plant species richness by site type. Both variables decreased with site age (r=-0.74 and r=-0.75, p<0.0001 and n=23 in both cases), although sugar abundance varied a great deal between sites of similar age. The more recently reclaimed sites provided approximately 300 times the amount of sugar produced in hardwood sites and had about five times as many nectar producing plant species.

Table 3.3 shows the correlations between sugar abundance, inflorescence number, and nectar plant species richness for the 1993 flight season and the vegetational community variables. Interestingly, nectar variables were not correlated with herbaceous species richness and were negatively correlated with tree and vegetational species richness. Amount of sugar, inflorescence number and nectar plant species richness were positively correlated with herbaceous cover and strongly negatively correlated with tree cover.

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Table 3.3 - Spearman rank correlation coefficients between nectar and vegetational community variables (n=23) (SAB - sugar abundance, IN - inflorescence number, NSR - nectar species richness, SRH - species richness herbs, SRT - species richness trees, SRV - vegetational species richness, CH - cover herbs, CS - cover shrubs, BAT - basal area trees)

Variable	SRH	SRT	SRV	CH	CS	BAT
SAB	0.06	-0.76***	-0.61*	0.67**	-0.17	-0.74***
IN	-0.05	-0.85***	-0.65**	0.63*	-0.28	-0.91***
NSR	0.13	-0.75***	-0.43*	0.52*	-0.29	-0.80***

*p<0.05, **p<0.001, ***p<0.0001

As with nectar resources there was a general decrease in number of individuals and species of butterflies with site age (r=-0.73 and r=-0.57, p<0.0001 and p=0.0041, n=23 in both cases), but there was a high degree of variability between sites of the similar successional stage (see Figures 2.2 and 2.3, pp. 50 and 51). Table 3.4 lists correlations between all nectar and lepidopteran variables summed over the entire flight season and Figures 3.3 and 3.4 (p. 85 and p. 86) show butterfly abundance and species richness plotted versus amount of sugar and nectar plant species richness.

When summed for the entire flight season, butterfly abundance and species richness were strongly correlated with nectar abundance and nectar plant species richness, while moth abundance and species richness were negatively correlated with sugar abundance and nectar plant species richness. The abundances of a few

individual butterfly species were weakly correlated with nectar abundance; these

included *Colias* spp. (r=0.42, p=0.0059), *Epargyreus clarus* (r=0.43, p=0.0048), *Everes*

comyntas (r=0.46, p=0.0019), *Phyciodes tharos* (r=0.38, p=0.0136), *Poanes* spp.

(r=0.35, p=0.0213), and *Strymon melinus* (r=0.41, p=0.0064).

Table 3.4 - Spearman rank correlation coefficients between nectar and lepidopteran community variables summed for the 1992 and 1993 flight seasons (n=42) (SAB - sugar abundance, IN - inflorescence number, NSR - nectar species richness, ABB- abundance butterflies, SRB - species richness butterflies, ABM - abundance moths, SRM - species richness moths, ABL - abundance lepidoptera, SRL - species richness lepidoptera)

Variable	ABB	SRB	ABM	SRM	ABL	SRL
SAB	0.72***	0.65***	-0.42*	-0.46*	-0.13	0.01
IN	0.84***	0.61***	-0.48*	-0.63***	-0.03	-0.15
NSR	0.76***	0.59***	-0.45*	-0.49**	-0.06	-0.05

*p<0.05, **p<0.001, ***p<0.0001

Table 3.5 lists correlations between lepidopteran and nectar variables by

sampling date. Butterfly abundance and species richness were correlated with nectar

abundance, but were equally correlated with site age when all sampling dates were

considered together. While site age in itself was probably not important, a number of

vegetational variables, including herbaceous cover and tree basal area, were correlated

with site age. In just a few cases were butterfly abundance and species richness

correlated with nectar abundance on different sampling dates in individual sites (Table

3.6). Only in very recently reclaimed sites were nectar and butterfly variables

correlated the majority of the time. When dummy variable regression was used to

assess the relative importance of the site and nectar abundance effects, sugar

abundance did not have a significant effect on butterfly species richness or abundance

after adjusting for type of site ($p > 0.05$ in both cases). Similarly, nectar plant species

richness explained a significant, but reduced, amount of the variation in butterfly

abundance ($p = 0.0384$) and species richness ($p = 0.0013$) once the effect of site type was

removed. Figure 3.5 (p. 87) illustrates a representative example of fluctuations in

butterfly and sugar abundance in a single reclaimed site during the 1993 flight season.

While both sugar and butterfly abundance peaked in late May/early June, later peaks

in the abundance of sugar and butterflies did not coincide. The abundance of nectar in

September was the result of a number of late-flowering species in the Asteraceae.

Table 3.5 - Spearman rank correlation coefficients between nectar and lepidopteran community variables by sampling date (n=504) Abbreviations explained in Table 3.4.

Variable	ABB	SRB	ABM	SRM	ABL	SRL
SAB	0.54***	0.56***	-0.09	-0.03	0.19***	0.13*
IN	0.54***	0.57***	-0.13*	-0.07	0.16**	0.10*
NSR	0.51***	0.53***	-0.16**	-0.11*	0.12*	0.04
AGE	-0.54***	-0.53***	0.20***	0.23***	0.02	-0.09*

*$p < 0.05$, **$p < 0.001$, ***$p < 0.0001$

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Table 3.6 - Spearman rank correlation coefficients between nectar and butterfly community variables within individual sites (number of sites with significant correlations in parentheses) Abbreviations explained in Table 3.4.

Site type	n	SAB/ABB	SAB/SRB	NSR/ABB	NSR/SRB
<5 yr	3	*(2) **(1)	*(2) **(1)	*(2)	*(1)
5-12 yr	5	*(2)	*(2)	*(1)	*(1)
15-20 yr	5	*(1)	**(1)	--	--
25-30 yr	5	--	--	--	--
Hardwood	5	*(1)	*(1)	*(1)	*(1)

*p<0.05, **p<0.001

Lepidopteran behavior

Butterflies were observed nectaring 13% of the time (Table 3.7). The percentage of butterflies observed basking, flying, and nectaring was fairly constant in all sites, although low numbers of observances in later successional reclaimed and hardwood sites resulted in higher variability. The number of butterflies observed nectaring in a site was correlated with sugar abundance and nectar plant species richness (r=0.66 and r=0.63, p<0.0001 and n=42 in both cases), as would be expected, since the overall number of butterflies in these sites was higher. However, the percentage of butterflies observed nectaring was not correlated with nectar abundance (r=0.16, p=0.2969, n=42) or nectar plant species richness (r=0.12, p=0.4521, n=42).

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Table 3.7 - Distribution of butterfly behaviors (n=2651)

Behavior	Percentage
flying	47%
basking	38%
nectaring	13%
chasing	2%
mating	0.5%

Table 3.8 lists the percentage of the time that more common species were observed nectaring. This percentage varied widely between species. Many of the wood satyrs, such as *Cercyonis pegala, Enodia anthedon,* and *Megisto cymela,* rarely nectar and obtain nutrients from sap flows, while certain mobile species, including *Epargyreus clarus, Papilio glaucus,* and *Vanessa cardui* spend a large percentage of time nectaring.

Table 3.8 - Percentage of time common lepidopteran species were observed nectaring (only species for which n>10 are included)

Percentage	Species
0	*Cercyonis pegala, Enodia anthedon*
1-5	*Megisto cymela, Papilio troilus*
5-10	*Battus philenor, Everes comyntas, Vanessa atalanta rubria*
10-20	*Ancyloxypha numitor, Celestrina ladon, Colias* spp., *Erynnis* spp., *Phyciodes tharos, Speyeria cybele*
21-30	*Epargyreus clarus, Papilio glaucus, Poanes* spp., *Strymon melinus, Thorybes* spp., *Vanessa cardui*

Table 3.9 lists the frequency with which the various plant species were utilized as nectar sources. Figure 3.6 (p. 88) shows the number of observed visits to each plant species plotted versus the total amount of sugar provided by those plants. A logarithmic scale for sugar abundance was used because of the large range of values. While the average amount of sugar produced by a plant species was significantly correlated with the frequency with which it was utilized by butterflies (r=0.54, p=0.0021, n=30), there were a few notable outliers. *Buddleya davidii* and *Rubus allegheniensis* were utilized by many more individuals than would have been expected from sugar abundance. Butterflies nectared less than would be predicted on *Lespedeza cuneata* and *Solidago* spp. This result was likely due to the fact that both

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Lespedeza and *Solidago* flower towards the end of the growing season when there are few adult butterflies present.

Table 3.9 - Number of observed visits to nectar sources

Frequency	Species
1-2	*Achillea millefolium, Alliaria petiolata, Aster pilosus, Barbarea vulgaris, Cercis canadensis, Chrysanthemum leucanthemum, Cirsium vulgare, Clematis virginiana, Hydrangea arborescens, Solanum americanum, Stelleria media, Viola* spp.
3-5	*Aster divaricatus, Coronilla varia, Melilotus* spp., *Robinia hispida, Rudbeckia hirta, Senecio aureus, Taraxacum officinale*
6-10	*Daucus carota, Eupatorium maculatum, Geranium maculatum, Helianthus microcephalus, Hieracium pratense, Medicago sativa, Vicia villosa*
11-20	*Erigeron* spp., *Eupatorium rugosum, Lespedeza bicolor, Lespedeza cuneata, Potentilla simplex, Solidago* spp., *Trifolium repens*
27	*Trifolium pratense*
45	*Buddleya davidii*
48	*Rubus allegheniensis*

Discussion

Clearly, reclaimed areas provide much more abundant and diverse nectar resources than the surrounding hardwoods. Few other studies have compared relative

availability of nectar in different ecosystem types. Hocking (1953) estimated that tundra ecosystems provided about 3.5 times the amount of nectar of boreal forests. In this study, the difference between areas with open and closed canopies was much higher.

Results of this study, both from correlations between nectar and butterfly variables and behavioral observations, suggest that nectar resources play an important role in determining adult butterfly community composition. Nectar resources appeared to have little influence on moth community composition since most diurnal moths do not nectar. Interestingly, butterfly abundance and species richness and sugar abundance and nectar plant species richness were correlated when summed over the entire flight season, but not when considered by sampling date. These results agree with previous research on the influence of nectar resources on the adult butterfly communities. Kremen (1992) found correlations between flowering plant species richness and butterfly species richness in forests in Madagascar. Munguira and Thomas (1992) reported correlations between number of flowering plants and both abundance and species richness of butterflies on roadside verges in England. In these two studies both nectar resources and butterfly community measurements from all sampling dates were summed. Sharp et al. (1975) found few correlations between nectar abundance and butterfly abundance in alpine meadows in Colorado. However, butterfly abundance and nectar resources were analyzed by sampling date. It appears that many adult butterfly species move in order to optimize nectar resources, but that

their life histories have not evolved to maximize available nectar resources temporally.

A number of possible explanations exist for the lack of correlation between butterfly and sugar abundance by sampling date and the low number of correlations between individual butterfly species abundances and nectar resources. The most obvious is that other factors constrain butterfly life histories. For a number of butterfly species flight period reflects the life history of a specific larval host plant (e.g., Ehrlich et al. 1975; Cappuccino and Kareiva 1985). However, most species commonly observed in this region can utilize widespread plant species, many of which have lengthy growing seasons. Second, the numerous species that overwinter as larvae and pupae are probably not able to utilize the late season peak of nectar because there would be insufficient time for life cycle completion. Third, sampling scale may also explain the lack of correlation between butterfly and sugar abundance by sampling date. It is difficult to determine the scale at which butterflies move to locate nectar resources and it varies in different species. The scale at which butterfly communities and nectar resources have been sampled in previous studies has varied greatly. Fourth, the observed results could be explained by plant structure rather than nectar resources. Plant structure may play an important role in thermoregulation (e.g., Warren 1985), oviposition (e.g., Sanjayan and Courtney 1992), and mate location (e.g., Daily et al. 1991) in certain species. Sugar abundance and nectar plant species richness were strongly negatively correlated with tree cover ($r=-0.74$ and $r=-0.80$) and positively correlated with herbaceous cover ($r=0.67$ and $r=0.52$). Correspondingly, butterfly

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abundance and species richness were strongly negatively correlated with tree cover (r=-0.69 and r=-0.64) and weakly positively correlated with herbaceous cover (r=0.49 and r=0.51). As vegetational structure changes much less during the flight season than nectar resources, vegetational structure would explain the apparent correlation between nectar resources and butterfly abundance summed over the entire flight season and the lack of correlation between these variables when considered by sampling date. It is impossible to separate the influences of vegetational structure and nectar resources without further manipulative experiments.

While recently reclaimed sites provide extensive nectar resources and host large numbers of adult butterflies, it is unclear whether these sites provide a resource that might otherwise limit butterfly populations in this region. The degree to which nectar resources influence butterfly population dynamics has been widely debated. Nectar may be a limiting resource for certain species (e.g., Gilbert and Singer 1973; Ehrlich et al. 1975; Gut et al. 1977). Clench (1967) argued that competition for adult food resources has resulted in temporal dissociation in a number of hesperiine butterfly species in Pennsylvania. However, the majority of temperate butterfly species are able to use a wider variety of plant species for nectar than as larval host plants (Opler and Krizek 1984; Erhardt and Thomas 1991). While butterflies may be limited in the species of plants from which they can obtain nectar by sugar concentration and corolla length, all the species observed in this study are able to utilize a number of nectar sources which are common in other disturbed areas such as roadsides, lawns, or logged

sites. In some ecosystems, plants may produce excess nectar in order to compete for pollinators, rather than pollinators being limited by nectar availability (Hocking 1968; Mosquin 1971). This appeared to be the case during the majority of the flight season in the sites studied; however, further studies in lepidopteran energetics would be necessary to make a conclusive statement.

As previous studies have generally used plant number as an indicator of nectar abundance (e.g., Sharp et al. 1975; Kremen 1992; Munguira and Thomas 1992), one goal of this study was to test this assumption. Results of this study showed a high correlation between nectar abundance and inflorescence number. However, it must be recognized that determining the type of inflorescence to quantify or how far an individual plant extends, as has been done in some previous studies, is largely subjective. Also, this assumption may not be valid if sites differ substantially with respect to common nectar producing plant species. In this study, most sites were dominated by similar nectar sources so there was a high correlation between nectar abundance and number of inflorescences. However, the factor relating these two variables was different in early successional sites in which other nectar producing species were commonly present.

For a number of reasons measuring sugar abundance is problematic. First, a considerable amount of time is necessary to measure the sugar content of nectar. Second, a number of factors have been demonstrated to affect nectar secretion including humidity, temperature, time of day, altitude, or position of floret or flower in

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an inflorescence (reviewed by Hocking 1953). Third, the amount of nectar secreted often varies greatly between different cultivars of commonly planted species (Erickson and Peterson 1979; Szabo and Naida 1985). Fourth, other characteristics of nectar besides total sugar content can be important to insects. Sugar composition (e.g., Watt et al. 1974; Baker and Baker 1981, 1983; Erhardt 1991) and sugar concentration (e.g., Watt et al. 1974) have been shown to affect butterfly nectar preferences. Presence of lipids and amino acids not only influence the quality of nectar for some butterfly species (Murphy et al. 1983; Erhardt 1991; May 1993), but may also affect the accuracy of nectar concentration measurements (Inouye et al. 1980). Differences in nectar composition could explain the preferences of butterflies for certain plant species such as *Buddleya* and *Rubus*, although little information is available about the nectar composition of these species.

Inflorescence number may actually influence butterfly abundance more than the amount of sugar. Insects often spend a great deal of time testing flowers before finding substantial nectar rewards (Southwick et al. 1981). In this study, butterflies were seen on occasion trying to nectar at plants that do not produce nectar, such as *Coronilla varia* and *Solanum americanum*, and flying towards colored flags marking the sites. These observations suggest the importance of visual cues rather than actual nectar rewards. However, other work has shown that butterflies can learn flower colors associated with nectar rewards reversibly (Weiss 1993), which implies that butterflies would soon learn to avoid plants that do not provide nectar of suitable

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quantity or composition.

Quantifying the amount of sugar available in nectar sources resulted in little additional understanding of the factors affecting butterfly community composition on the sites studied. Considering this fact, the extensive additional time necessary to assay sugar abundance does not seem warranted for studies of this type. However, the nectar assay utilized worked well for measuring small amounts of nectar and would serve as a valuable tool in studies specifically focused on insect energetics.

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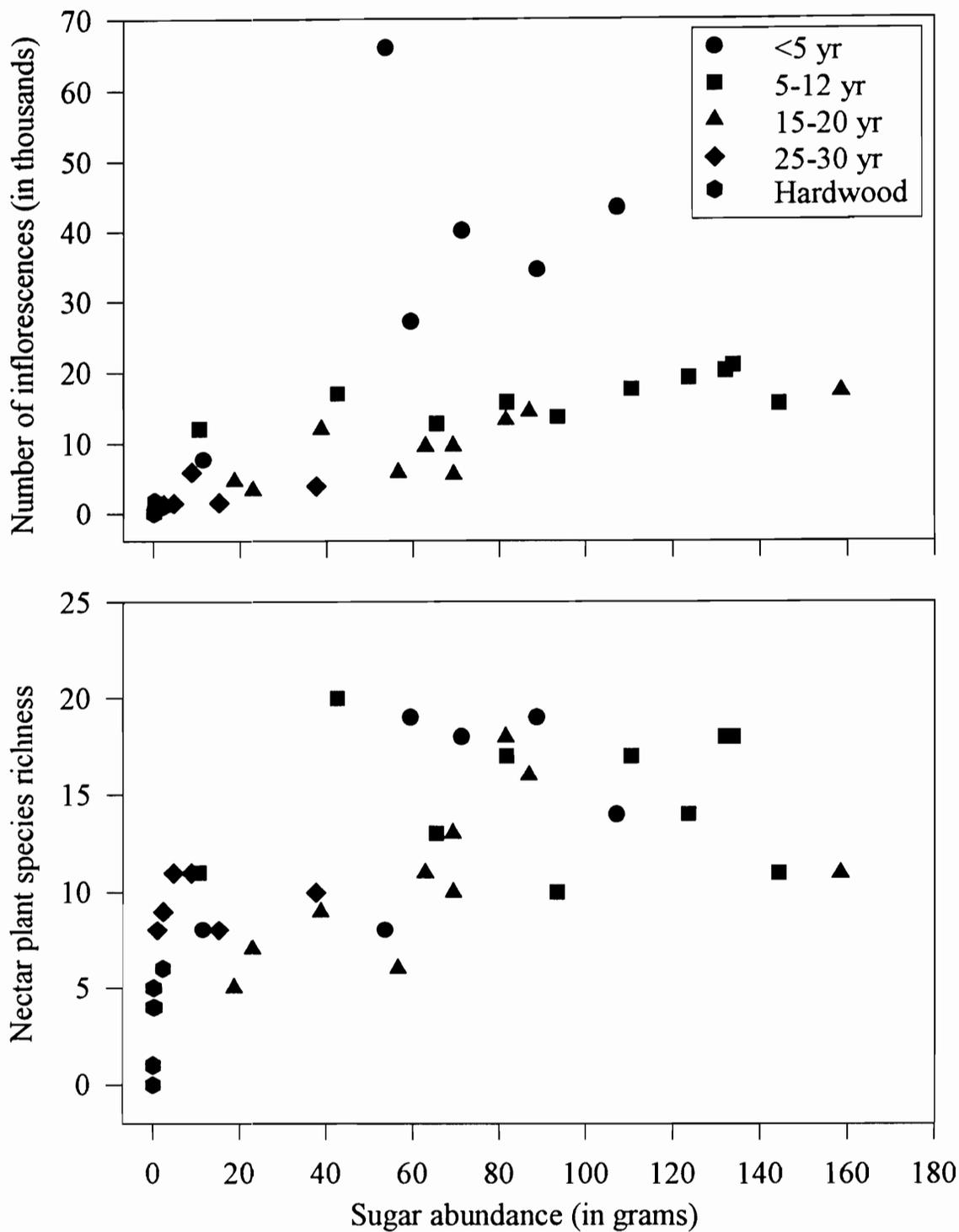
Figure 3.1 - Number of inflorescences and nectar plant species richness vs. sugar abundance. Some symbols for the hardwood sites overlap.

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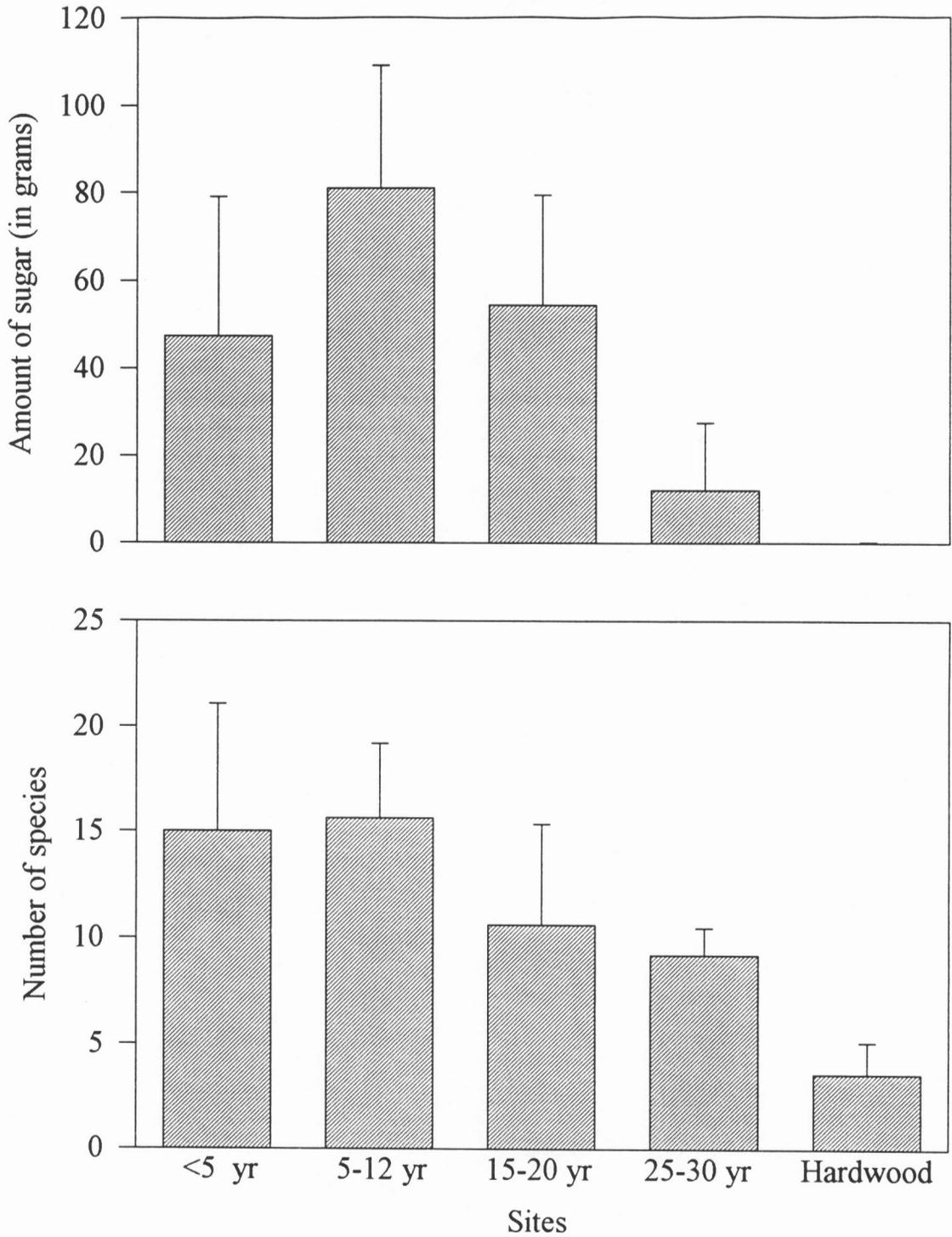
Figure 3.2 - Mean sugar abundance and nectar plant species richness by type of site for 1993. Error bars represent 1 SD.
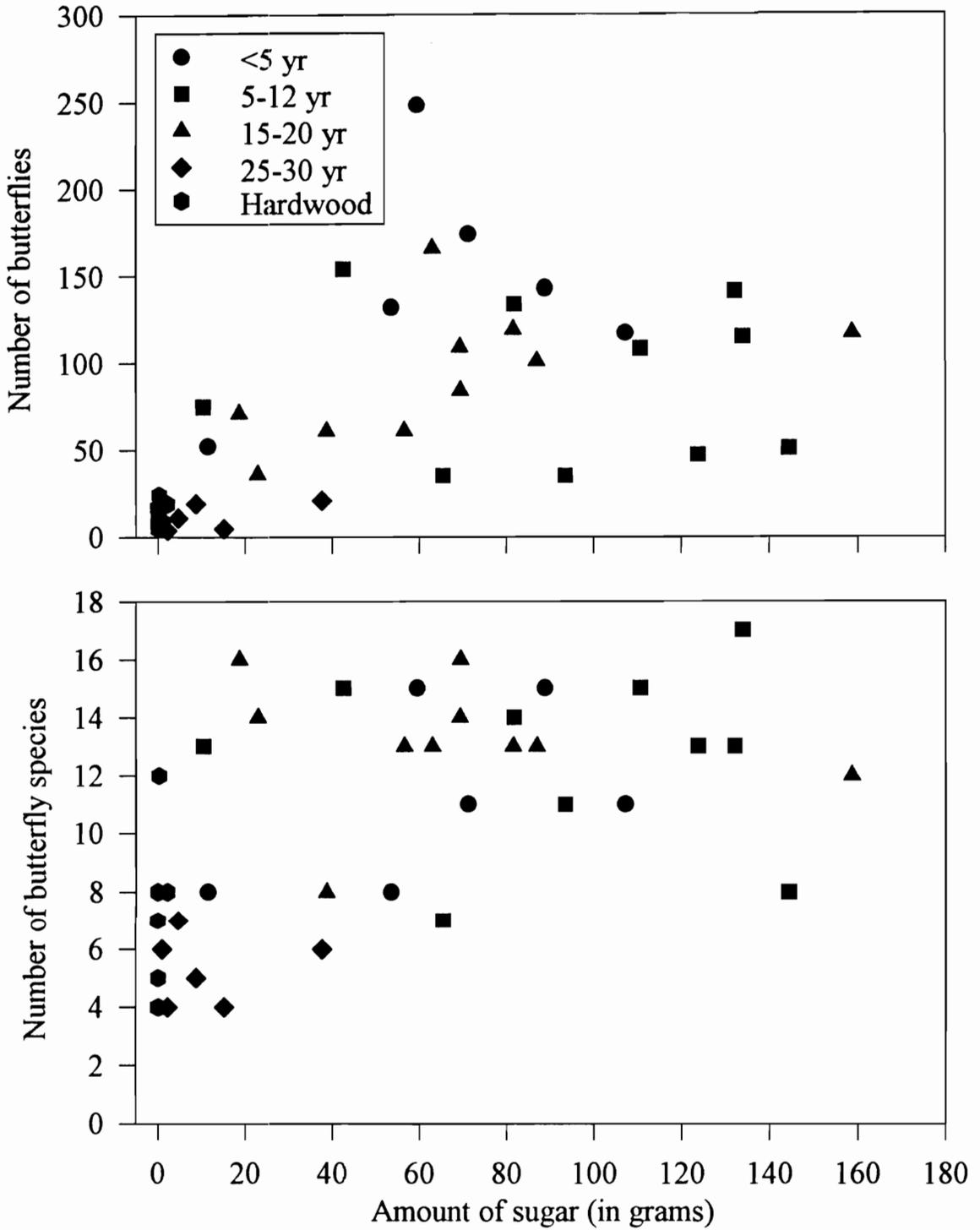
Figure 3.3 - Butterfly abundance and species richness vs. sugar
abundance. Some symbols for the hardwood sites overlap.
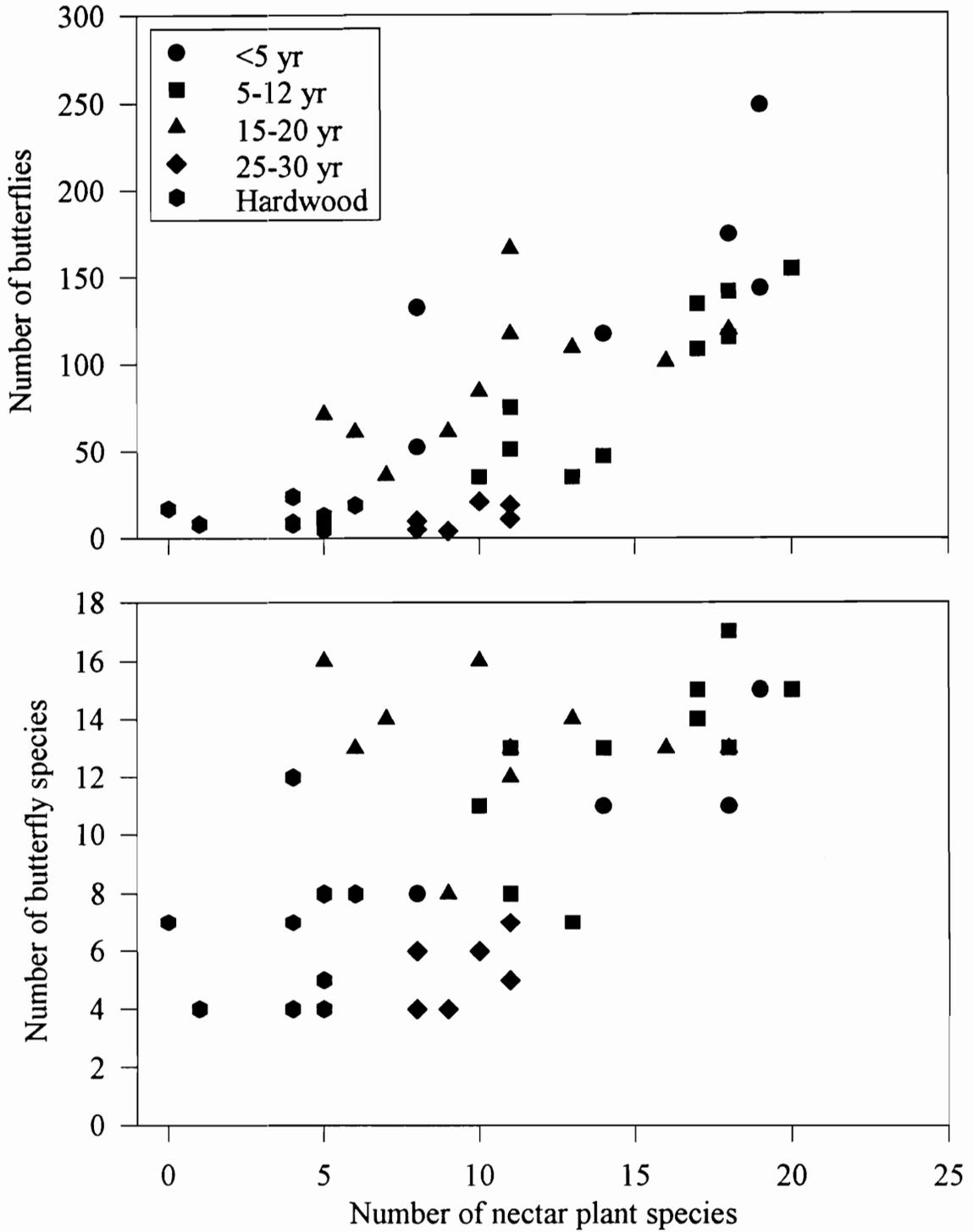
Figure 3.4 - Butterfly abundance and species richness vs. nectar plant species richness.
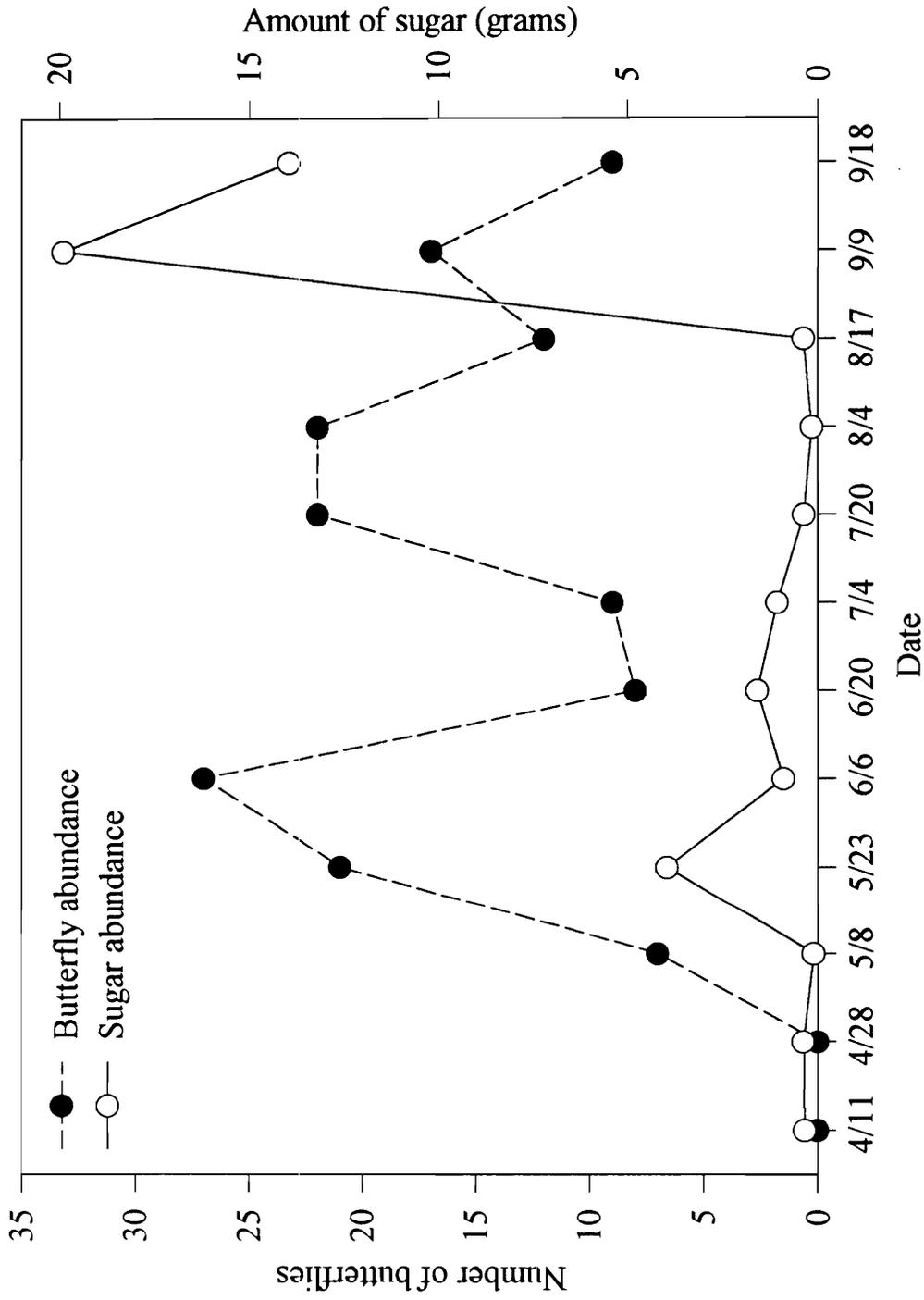
Figure 3.5 - Butterfly and sugar abundance in a 7 yr old reclaimed site during 1993.
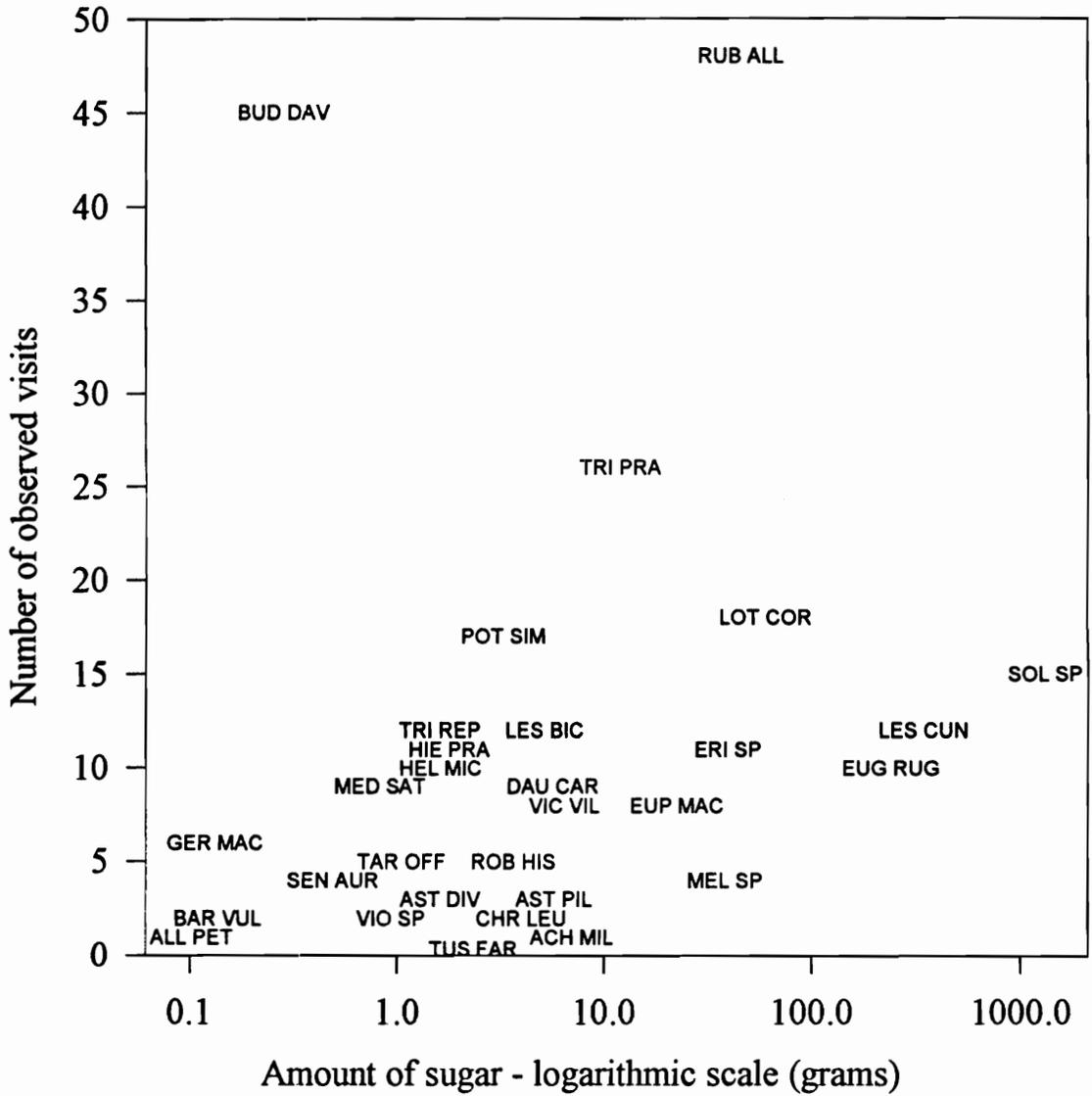
Figure 3.6 - Number of observed visits vs. sugar abundance for nectar sources. Plant names are located as close as possible to the actual location without overlapping.

Chapter 4

THE ROLE OF MINE RECLAMATION IN CONSERVATION

Introduction

Protecting extensive areas of undisturbed ecosystems is the ideal means of

conserving biodiversity and maintaining ecosystem services. However, the highly

disturbed state of our planet necessitates rehabilitating damaged ecosystems, in

addition, to achieve these ends. While an increasing amount of research is being done

to investigate the effects of land rehabilitation efforts on the biota, the vast majority of

this research has consisted of one-time sampling of one or a few sites. Research on

less disturbed systems indicates that maintaining high alpha (within-site) diversity does

not necessarily maximize beta (between-site) diversity (Noss 1983; Van Horne 1983;

Palmer 1992). Therefore, few studies provide the data necessary to evaluate the

effectiveness of restored and reclaimed areas in preserving native flora and fauna.

Determining criteria to evaluate the conservation value of an area is difficult;

the most commonly used criteria are diversity and rarity (Margules and Usher 1981),

although both measures have their limitations. Diversity alone means little if one does

not consider the composition of species in the community (Rodda 1993). Disturbed

habitats tend to favor r-selected, generalist species (Pianka 1978; Novotny 1991).

Normally rare species are given more conservation value. However rarity can be

defined in different ways including geographic range, habitat specificity, and local

population size (Drury 1974; Rabinowitz 1981).

Recognizing these caveats of lack of data and difficulty of choosing criteria, this chapter addresses the role of mine reclamation in conservation. In the first part of this chapter, the results of this study and the few others reported in the literature are discussed. Strategies for enhancing regional reclamation efforts are outlined in the second section. Conclusions from studies on mine reclamation provide valuable insight into other large-scale land rehabilitation efforts.

Regional conservation of flora and fauna

Plants and fungi

The few studies that have addressed the effect of mine reclamation on floral conservation suggest that reclaimed mine sites do not host all species present in the unmined areas. As discussed in Chapter 2 (p. 16), in this study reclaimed areas hosted both fewer total plant species and fewer less common plant species than hardwood forests. Schuster and Hutnik (1987) found that even after 35 years a number of common hardwood tree and shrub species had not invaded reclaimed mined sites in Pennsylvania. In a study of bryophytes in a reclaimed and undisturbed watershed in West Virginia, Engelmann and Weaks (1985) reported higher species richness and more "infrequent" and specialized species in the undisturbed area. Finally, Allen and MacMahon (1985) studied vesicular-arbuscular mycorrhizae (VAMs) on reclaimed mine sites in Wyoming. They reported that the number of VAM species on the mined

and unmined sites was similar. However, similarity between samples from unmined

sites was lower suggesting higher spatial heterogeneity in these sites.

Lepidoptera

A number of studies on lepidoptera suggest that disturbed areas favor

widespread species, while forests host more specialized species (e.g., Adams 1973;

Leps and Spitzer 1990; New and Thornton 1992). However, Erhardt and Thomas

(1991) and Thomas (1991) assert the importance of early successional grasslands in

the preservation of certain European butterflies. It is important to recognize that there

is a great deal of variability in the scale on which these studies base their conclusions.

Results of this study support the hypothesis that reclaimed areas favor

generalist species. First, as discussed in Chapter 2 (p. 39), while the total number of

species in the different types of sites was similar, a greater number of less common

species were found in hardwood than reclaimed sites. However, there are drawbacks

to this method of quantifying differences in community composition; the range of the

species and the reason for rarity are not considered. To further consider the effect of

mine reclamation on butterfly conservation an analysis of life history traits was done.

Such analyses with moths were not possible due to the poor documentation of their

life histories.

A total of 102 butterfly species have been recorded in southwestern Virginia

(Clark and Clark 1951; Opler and Krizek 1984; Opler, unpublished data), although 12

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of the species disperse or migrate to Virginia and are not known to breed in this area. Of the 90 resident species, 46 were observed on reclaimed sites during the summers of 1991-1993, while an additional 4 species were observed in upland forests and 3 species in a nearby riparian-wetland area. Table 4.1 lists the percentages of resident species observed in reclaimed sites with respect to certain life history traits. Reclaimed sites favor species with r-selected traits including oligophagy and polyphagy, multiple broods, good dispersal abilities, and a wide geographic range. Past studies have shown the Lycaenidae to exhibit the highest degree of endemism among butterfly families (e.g., Adams 1973), a trend supported by these data.

While only a few species in this region, most notably the regal fritillary *(Speyeria idalia)*, have experienced precipitous population declines in the past few decades (Covell 1990), the butterfly fauna of the eastern United States is increasingly dominated by the generalist species commonly found on reclaimed mine sites, such as eastern tailed blue *(Everes comyntas)*, silver spotted skipper *(Epargyreus clarus)*, and pearl crescent *(Phyciodes tharos)* (Opler and Krizek 1984). It is impossible to make a conclusive statement, but this trend likely reflects, in part, a homogenization of the flora. In this study, 19 (43.2%) of the butterfly species that were not observed typically reside in upland second growth or forest areas, but were lacking suitable host plants. The spread of non-native plant species has favored lepidopteran species that are able to utilize these species as host plants, such as yellow sulphur *(Colias philodice)* (Karowe 1990) and American copper *(Lycaena phleas)* (Opler and Krizek 1984).

Table 4.1 - Percentage of butterfly species resident in southwestern Virginia that were observed in reclaimed mine sites by life history traits and family (sources Opler and Krizek 1984; Scott 1986)

Host plant specificity	Monophagous (n=35)	Oligophagous (n=43)	Polyphageous (n=12)
Present	29.4	58.1	91.7
Absent	70.6	41.9	8.3
Number of broods	1 (n=35)	2 (n=31)	3≤ (n=24)
Present	37.1	61.3	58.3
Absent	62.9	38.7	41.7
Dispersal ability	Sedentary (n=18)	Semi-vagile (n=44)	Vagile (n=28)
Present	16.7	59.1	60.7
Absent	83.3	40.9	39.3
Geographic range	Southeastern U.S. (n=19)	Eastern N. America (n=37)	N. America≤ (n=35)
Present	15.8	52.8	68.6
Absent	84.2	47.2	31.4

Table 4.1 (continued)

Family	Hesperidae (n=31)	Lycaenidae (n=21)	Papilionidae (n=6)	Pieridae (n=7)	Nymphalidae (n=26)
Present	56.7	23.8	50.0	71.4	61.5
Absent	43.3	76.2	50.0	28.6	38.5

Birds

During June and August 1992 birds were censused on reclaimed benches and the surrounding hardwoods, in the general vicinity of the lepidopteran sampling sites, for a total of 24 observer hours (McCormick and Holl, unpublished data). As sampling efforts in different habitat types were not equal it is impossible to compare numbers of less common species, as was done with plants and lepidoptera. However, while more than twice the number of individuals was observed in reclaimed as compared to hardwood areas, similar numbers of species were observed in both areas (Table 4.2). A few species, such as indigo bunting *(Passerina cyanea)*, carolina wren *(Thryothorus bewickii)*, and song and field sparrow *(Melospiza melodia* and *Spizella pusilla)*, are very common on reclaimed sites. It is important to note that these data do not represent thorough sampling efforts. However, they suggest that reclaimed sites host high numbers of individuals, but this trend does not translate into more species. Crawford et al. (1978) recorded more species of birds on undisturbed than mined areas

in West Virginia, while Karr (1968) observed more species and individuals in 30 and 50 year old reclaimed mine sites than in an unmined forest. No studies were found that compared avian communities on multiple reclaimed sites to allow for between-site comparisons.

Table 4.2 - Percentage of individuals and species of birds observed in different habitat types. Species are included in the habitat type in which they were most commonly found. Edge indicates the border between hardwood and reclaimed sites.

Habitat type	Reclaimed	Edge	Hardwood	Wetland
Individuals (n=771)	48.5	8.5	21.6	21.4
Species (n=40)	37.5	7.5	35.0	20.0

Conclusions and Recommendations

Although few studies have considered the effect of mine reclamation on conservation and the scales of the studies have varied, the majority of studies suggest that reclaimed mine sites do not host the full complement of the flora and fauna of the unmined areas. While more time is necessary to make final conclusions on the role of mine reclamation in the conservation of biodiversity, the current rate of degradation of the planet necessitates considering strategies for improving land rehabilitation efforts. In order to enhance the role of land rehabilitation in conservation efforts larger spatial and longer temporal scales must be considered.

Increasingly, it is being recognized that ecological processes are not regulated at a single spatial scale, but may be constrained by processes at both larger and smaller levels (O' Neill et al. 1986; Risser 1987; Pulliam 1988). Surface mining and associated reclamation practices affect ecosystems far beyond the boundaries of the actual mined area (e.g., Vaughan et al. 1978; Matter and Ney 1981; Dick et al. 1986). Likewise, recovery of reclaimed areas is dependent on the composition of the surrounding landscape (e.g., Wolfe 1990; Nepstad et al. 1991; Anderson 1993). Therefore, it is essential that the surrounding landscape, and not just the mined area, be considered in the reclamation plans.

Currently, mining companies are responsible for maintaining certain water quality standards for the runoff into nearby streams. However, little consideration is given to sources of floral and faunal propagules. Most coal surface mines in the Appalachian region are long and narrow allowing for colonization from nearby hardwoods. However, increasingly, larger areas are being mined. As introduction of the entire complement of flora and fauna is not feasible, consideration must be given to how to facilitate natural colonization. One option is leaving remnant patches of forest as a source of propagules. Small clumps of planted trees (Janzen 1988) and artificial structures (Wolfe 1990) have been used successfully to encourage faunal seed dispersal in other large-scale restoration projects.

A second reason for increasing the spatial scale of reclamation projects is a need to increase landscape-scale habitat diversity. Patchiness is inherent to most

ecosystems (White 1987). However, the same few herb and tree species are planted on reclaimed areas throughout extensive regions. Increasing the conservation value of reclaimed areas within the current legislation is possible. For example, Fowler and Turner (1981) outline a reclamation plan designed to benefit wildlife that involves planting different species in small patches instead of using the same vegetation mixture for the entire area. Their results suggest that such a strategy is no more expensive than traditional reclamation practices. In addition to using different planted species, the benefits of small areas of wetlands to fauna has been well documented (e.g. Lacki et al. 1991; Lacki et al. 1992, Mulyani and DuBowy 1993). While using some innovative reclamation practices over large areas may be cost prohibitive, more variation in even small areas would benefit regional conservation efforts.

Reclamation projects must be planned and monitored for a longer time period than the five years currently considered. Current regulations call for establishing vegetation that "is capable of regeneration and plant succession" (Virginia Department of Mines, Minerals and Energy, p. II - 224). However, as discussed in Chapter 1 (p. 16), studies have shown that many of the non-native planted herbaceous species inhibit long-term vegetational development. Eastern white pine, which is commonly planted to achieve tree cover standards, may result in lower vegetational species richness over the long-term (Schuster and Hutnik 1987). Anderson (1993) reported that the shrub species planted in mines in Australia rapidly became established, but resulted in a stalling of the turnover of ant communities.

Alternative revegetation strategies exist that better balance the needs of minimizing short-term problems, such as erosion and acid mine drainage, and maximizing long-term ecosystem development. Studies have shown that a number of tree species, besides those currently utilized, have potential for reclamation (e.g., Brown et al. 1984; Torbert et al. 1985). Preliminary results suggest that some native herbaceous species can become established on these sites while permitting substantial invasion of volunteer species (Sabre and Holl, unpublished data). Farmer et al. (1982) and Wade (1989) found that forest topsoil amendments could be used to provide a diverse cover on mined areas quickly. Wade also reported higher biomass and equal cover at the end of the first growing season in the topsoiled area as compared to plot seeded with a standard reclamation mixture. Forest soils could be moved from areas that will soon be mined to areas currently being reclaimed (Garrison 1992). Also, applying sewage sludge to mine spoils may ameliorate soil conditions sufficiently to allow for a large number of volunteer species to become established (W. Daniels, pers. comm.).

It is questionable whether restoration to original condition is possible (Cairns 1989). However, it is clear that current reclamation practices could be improved. Changes in requirements are needed that encourage, rather than inhibit, innovation. For example, in order to pass the final inspection in Virginia 400 trees per acre must have survived 3 years. This requirement does little to insure erosion control, but precludes the use of a number of native tree species that do not rapidly become

established. Although it is challenging to legislate minimum reclamation requirements while allowing for the flexibility to use new procedures, feasible strategies exist. One legislative option to better balance short- and long-term management needs would be to use an incremental bonding system. In such a system, certain portions of the bond would be returned at the end of a number of time periods, such as 5, 15, and 25 years, instead of returning the entire bond if 5 year criteria were met. Also, Pielou (1986) outlines a rapid and easily-understandable method for quantifying the relative vegetational diversity of an area and comparing community composition of a disturbed and reference sites. Such criteria could be measured with little added time or expense.

While more research is needed to better understand ecosystem processes in restored systems, it is equally important that existing information be utilized in developing reclamation plans. Ecosystems are sufficiently complex to preclude our rebuilding them. Therefore, we must adopt a strategy of trying to facilitate natural successional processes. This goal can only be achieved when we view restoration efforts in the context of the surrounding landscape for a length appropriate to the resiliency of a given ecosystem.

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VEGETATIONAL SPECIES LIST

*average importance value <5
**average importance value 5-25
***average importance value >25
Pplanted species
Nspecies not originally native to the southeastern United States

	Reclaimed sites				
Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
HERBS					
Achillea millefolium L.	*		*		
PN*Agrostis alba* L.	**	*	*		
Agrimonia gryposepala Wallroth					*
N*Alliaria petiolata* Cavara & Grande				*	
Ambrosia artesimifolia L.			*		
Asclepias syriaca L.					*
Asplenium platyneuron Oakes			*	*	
Aster divaricatus L.		*		**	*
A. lateriflorus Britton			*	**	
A. pilosus Willd.	**	*	**	*	
Aureolaria laevigata Raf.					*
Brachyelytrum erechtum Beauvois					*
PN*Bromus japonicus* Thunbeg		*	*		
Campanula divaricata Michaux				*	*
N*Cardamine hirsuta* L.		*	*	**	
Carex purpurifera Mackenzie					*
C. virescens Muhl.				*	
N*Chrysanthemum leucanthemum* L.		*			
Circaea lutetiana Ascherson & Magnus					*
Clematis virginiana L.	*	**	**	**	*
N*Convolvulus arvensis* L.		*	**	*	
PN*Coronilla varia* L.	*				
PN*Dactylis glomerata* L.	**	*	*		

Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
[N]*Daucus carota* L.	*	*			
Dennstaedtia punctilobula Bernh.					*
Desmodium nudiflorum L.					**
[N]*Dianthus armeria* L.			*		
Dioscorea villosa L.		*			*
Disporum lanuginosum Salisbury					*
Epilobium coloratum Biehler	*		*		
Epigaea repens L.					*
Erechtites hieracifolia Raf.		*	*		
Erigeron annuus Persoon		*			
E. canadensis L.	*	*	*		
E. philadelphicus L.			*	*	
Eupatorium maculatum L.		*	*	*	
E. purpureum L.				*	*
E. rugosum Houttuyn		**	**	*	
[PN]*Festuca elatior* L	***	***	***	*	
F. obtusa Biehler				*	
Fragaria virginiana Duchesne		*		*	
Galax aphylla L.					*
Galium aparine L.			**	**	
G. circaezans Michaux					*
G. triflorum Michaux					*
Gaultheria procumbens L.					*
Geranium maculatum L.					*
Geum canadense Jacquin				*	*
Gnaphalium obtusifolium L.			*		
Helianthus microcephalus T. & G.		*	*	*	
Hieracium paniculatum L.					*
[N]*H. pratense* Tausch	*	*	*		
H. scabrum Michaux	*				
Impatiens capensis Meerb.			*		
I. pallida Nuttall			*	***	
Juncus tenuis Willd.		*			
Lactuca canadensis L.		*	*		
[N]*L. scariola* L.	*				
[PN]*Lespedeza cuneata* G.	***	**	**	*	
Lobelia spicata Lam.		*		*	
[PN]*Lotus corniculatus* L.	**				

Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
Lysimachia quadrifolia L.		*		*	*
[PN]*Medicago sativa* L.	*				
[PN]*Melilotus* spp. Miller	*				
Monarda clinopodia L.				*	
Muhlenbergia tenuiflora BSP.				*	*
Oenothera biennis L.	*	*	*		
Oxalis stricta L.		*			*
Panicum spp. L.		*	*	*	*
Parthenocissus quinquefolia Planchon		*	*	*	**
[PN]*Phleum pratense* L.	**				
Phytolacca americana L.		*	*	**	
Poa alsodes Gray				*	
[PN]*P. pratensis* L.	*				
[N]*Polygonum cespitosum* Stewart				*	
Polystichum acrostichoides Schott		*	*	**	*
Potentilla simplex Michaux.	*	**	*	*	*
Ranunculus spp. L.				*	*
[N]*Rumex acetosella* L.		*	*		
[N]*R. crispus* L.			*		
[N]*R. obtusifolius* L.				*	
Sanicula canadensis L.					*
Senecio aureus L.		*	*	*	
[N]*Setaria faberi* W. Herrmann			*		
Smilacina racemosa Desf.					*
Smilax rotundifolia L.			*	*	***
Solanum americanum Miller		*	*		
Solidago curtissii T. & G.				*	*
S. flexicaulis L.		*	*	**	*
S. gigantea Aiton	**	**	**	*	
S. nemoralis Aiton	*	**	*		
S. rugosa Miller		**	**		
Taenidia integerrima Drude					*
[N]*Taraxacum officinale* Wiggers	*	**	*		
Thaspium barbinode Nuttall					*
Thalictrum dioicum L.					*

Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
Thelypteris hexagonoptera Weatherby					*
Toxicodendron radicans L.			**		*
Trillium grandiflorum Salisbury					*
[PN]*Trifolium pratense* L.	**	*			
[PN]*T. repens* L.	**	*			
[N]*Tussilago farfara* L.	*	*			
Urtica gracilis Aiton			*		
Uvularia perfoliata L.					*
U. pudica Fernald					*
[N]*Veronica serpyllifolia* L.			*		
Viola blanda Willd.		*		*	
V. canadensis L.				*	
V. eriocarpa Schweinitz					*
V. hastata Michaux					*
V. septemloba House	*	*	*	*	**

SHRUBS AND TREES

Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
Acer negundo L.				*	
A. pennsylvanicum L.				*	*
A. rubrum L.		**	***	***	***
A. saccharum Marshall					*
Aesculus octandra Marshall					*
Amelanchier arborea Fernald					*
Betula lenta L.		**	*	***	*
[N]*Buddleja davidii* Franchet			*		
Carya spp. Nuttall			*	**	**
Castanea dentata Borkh					*
Cornus florida L.		*	*	*	**
[PN]*Elaeagnus umbellata* Thunberg	**	**	*		
Fagus grandifolia Ehrhart					*
Hydrangea arborescens L.				***	*
Juglans nigra L.					*
Kalmia latifolia L.		*			**
[PN]*Lespedeza bicolor* Turcz	*				
Liriodendron tulipifera L.		*	**	***	**
Magnolia acuminata L.			*	*	*
M. fraseri Walter				*	*

Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
Nyssa sylvatica Marshall			*	*	**
Oxydendron arboreum DC.		**	**	**	**
[P]*Pinus strobus* L.	***	***	*		
Prunus serotina Ehrhart		*	*	**	*
Pyrularia pubera Michaux					**
Quercus alba L.					*
Q. prinus L.					**
Q. rubra L.			*	*	**
Rhododendron catawbiense Michaux					*
R. maximum L.					**
Rhus glabra L.		*		*	
[P]*Robinia hispida* L.	*	**			
[P]*R. pseudoacacia* L.	**	***	***	***	*
[N]*Rosa multiflora* Thunbeg			*	**	
Rubus allegheniensis Porter	*	***	***	***	*
R. occidentalis L.			**	*	
Salix spp. L.				*	
Sassafras albidum Nees		**	*	**	**
Tilia americana L.					*
Tsuga canadensis Carr					*
Ulmus americana L.					*
Vaccinium arboreum Marshall					**
Viburnum acerifolium L.					*
Vitis aestivalis Michaux.			**	**	*

Appendix 2
LEPIDOPTERAN SPECIES LIST

*average number of individuals/site during a flight season <1
**average number of individuals/site during a flight season 1-10
***average number of individuals in a site during a flight season >10
+not observed during regular transect walks, but observed in this type of site
Wfound in wetland sites

Note - For species that were recorded at the generic level during transect walks, abundances are listed by genus followed by the specific epithet of identified species.

	Reclaimed sites				
Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
BUTTERFLIES					
Ancyloxypha numitor (Fabricius)	***	*	*		
Anthocaris midea (Hübner)		*		*	*
Asterocampa celtis (Boisduval & LeConte)					+
Atalopedes campestris (Boisduval)			+		
Battus philenor (Linnaeus)		*	*	*	*
W*Boloria bellona* (Fabricius)					
Calycopis cecrops (Fabricius)	*				
Celastrina ladon (Cramer)	**	*	*	**	**
Cercyonis pegala (Fabricius)	**	**	**		*
Chlosyne nycteis (Doubleday & Hewitson)			*		*
Colias spp. *eurytheme* Boisduval *philodice* Godart	***	**	*		
Cyllopsis gemma (Hübner)					*
Danaus plexippus (Linneaus)	+	+	+		
Enodia anthedon Clark			*	*	*
Epargyreus clarus (Cramer)	***	***	***	**	**

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Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
Erynnis spp.	**	**	**	**	**
baptisae (Forbes)					
brizo (Boisduval & LeConte)					
horatius (Scudder & Burgess)					
icelus (Scudder & Burgess)			.		
juvenalis (Fabricius)					
Euptoieta claudia (Cramer)	+	+			
^W*Euphydryas phaeton* (Drury)					
Everes comyntas (Godart)	***	***	***		
Glaucopsyche lygdamus (Doubleday)			*		
Hermeuptychia hermes sosybius (Fabricius)					+
Limentis archippus (Cramer)		*	*		
L. arthemis astenax (Drury)			*	*	
^W*Lycaena phleas* (Linnaeus)					
Megisto cymela (Cramer)	*	**	**		*
Nastra lherminier (Latreille)	*	*			
Nymphalis antiopa (Linnaeus)			*		*
Papilio glaucus Linnaeus	*	*	**	*	*
P. troilus Linnaeus	*	*	*		*
Phyciodes tharos (Drury)	***	***	***	*	*
Pholisora catullus (Fabricius)	+				
Pieris rapae (Linnaeus)	+				
P. virginiensis Edwards				*	*
Poanes spp.	*	**	**		*
homobok (Harris)					
zabulon (Boisduval & LeConte)					
Polites coras (Cramer)			*		
P. origenes (Fabricius)	*				
Polygonia spp.		*	*	*	*
comma (Harris)					
interrogationatis (Fabricius)					
Pompeius verna (Edwards)	*	*	**	*	
Satyrium calanus (Hübner)					*
Speyeria cybele (Fabricius)	*	*	**		*
S. diana (Cramer)			+		+
Strymon melinus (Hübner)	**	*	*		

Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
Thorybes spp.	*	*			*
bathyllus (J. E. Smith)					
pylades (Scudder)					
Vanessa atalanta rubria			*	*	*
(Fruhstarfer)					
V. cardui (Linnaeus)	*	*	*	*	
V. virginiensis (Drury)	+	+			
Wallengrenia egeremet (Scudder)	*	*			
MOTHS					
Agonopteryx thelmae (Clarke)				*	**
Agriphila spp.	*	**	***		
ruricollela (Zeller)					
vulgivagella (Clemens)					
Anagrapha falcifera (Kirby)		*			
Anania funebris (Strom)	*	**	*	**	
Anavitrinella pampinaria (Guenee)					*
Antepione thisoaria (Guenee)				*	
Anticarsia gemmatalis Hübner			*		
Blepharomastix ranalis (Guenee)		*	**	***	***
Bomolocha baltimoralis (Guenee)				*	*
B. deceptalis (Walker)					**
B. edictalis (Walker)				**	**
Cabera erythemaria Guenee					+
Caenurgia chloropha (Hübner)		*	*		
Caenurgina crassiuscula (Haworth)	***	**	**		
Calothysanis amaturaria (Walker)	*	**	***	**	
Campaea perlata (Guenee)					+
Catocala spp. Schrank		*	*		*
Celiptera frustulum Guenee			*		
Chytolita morbidalis (Guenee)			*	***	***
Crambus laqueatellus Clemins	***	***	**		
Desmia funeralis (Hübner)		*	*	*	
Diachrysia balluca Geyer					*
Dichorda iridaria (Guenee)			*		
Dyspteris abortivaria	**			*	*
(Herrich-Schaffer)					

Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
Ecliptopera atricolorata (Grote & Robinson)					*
Ectropis crepuscularis (Denis & Schiffermuller)					*
Ennomos subsignaria (Hübner)					+
Epimecis hortaria (Fabricius)				*	*
Eubaphe mendica (Walker)		*		*	**
Euchlaena obtusaria (Hübner)		*	*	*	
Euclidia cuspidea (Hübner)	**	**	**		
Eugonobapta nivosaria (Guenee)		*	**	**	**
Eulithis diversilineanis (Hübner)		*			
Eusarca confusaria Hübner	*	**	**	**	*
Feltia subgothica (Haworth)		*			
Gueneria similaria (Walker)			*	*	*
Hahncappsia marculenta Grote & Robinson		*			*
Haploa clymene (Brown)	*	**	**	*	**
H. lecontei (Guerin)	**	**	**	**	**
Heliomata cycladata Grote & Robinson	**	***	**	*	
H. infulata (Grote)		*	*		
Herpetogramma spp. *aeglealis* (Walker) *thestealis* (Walker)	**	***	***	***	***
Heterophleps refusaria (Walker)					*
H. triguttaria Herrich-Schiffer			*	**	*
Holomelina opella (Grote)					*
Hydrelia inornata (Hulst)				*	**
Hydria prunivorata (Ferguson)				*	**
Iridopsis larvaria (Guenee)					*
Itame pustularia (Guenee)				*	*
Lithacodea carneola (Guenee)		*	*		*
L. muscosula (Guenee)			*		*
Lomographa vestaliata (Guenee)		**	**	*	*
Macrochilo litophora (Grote)	**	***	***	***	
Melanophia canadaria (Guenee)			*	*	**
Mesoleuca ruficillata (Guenee)					**

Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
Metarranthis spp.		*	**	**	*
hypochraria (Herrich-Schaffer)					
indeclinata (Walker)					
Mocis texana (Morrison)	*	*	*		*
Nomophila nearctica Mun.		*	*	*	*
Oidaematophorus spp. Wallengren				+	
Orthonama centrostrigaria (Wollaston)			*	*	**
Pangrapta decoralis Hübner			**	**	***
Parallelia bistriaris Hübner				**	**
Phalaenophana pyramusalis (Walker)		**	**	**	***
Phoeberia atomaris Hübner					**
Plathypena scabra (Fabricius)	***	**	**	*	**
Probole spp.			*		**
alienaria (Herrich-Schaffer)					
nepiasaria (Walker)					
nyssaria (Guenee)					
Prochoerodes transversata (Drury)				*	
Pseudoplusia includens (Walker)			*		
Psychomorpha epimenis (Drury)			+		
Pyromorpha dimidiata Herrich-Schaffer				*	*
Renia discoloralis Guenee				**	**
R. sobrealis (Walker)	*	*	*	***	***
Rivula spp. Guenee		*	**	*	
Scopula inductata (Guenee)	*	*			
S. limboundata (Haworth)	*	**	***	**	**
Semiothisa aemulitaria (Walker)	*	**	*	*	*
S. bisignata (Walker)	*	**	***	**	***
S. ocellinata (Guenee)		*	*		*
S. quadrinotinaria (Herrich-Schaffer)				*	**
Spilosoma latipennis Stretch					+
S. virginica (Fabricius)	+				
Spodoptera frugiperda (J. E. Smith)			*		
Synedoidea grandirena (Haworth)					+
Tetracis cachexiata Guenee			*		*

Species name	<5 yr	5-12 yr	15-20 yr	25-30 yr	Hardwood
Thioptera nigofimbria (Guenee)	*		*		
Trichodezia alvobittata (Guenee)			*	***	**
Udea rubigalis (Guenee)					*
Urola nivalis (Drury)			*		
Xanthorrhoe lacustrata (Guenee)			*	**	*
Xanthotype urticaria Swett	**	*	**	*	
Zale undulinearis (Drury)			*		
Zanclognatha laevigata (Grote)				**	**
Z. lituralis (Hübner)					*
Z. pedipilalis (Geunee)	**	***	***	***	
Z. protumnusalis (Walker)					*

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CURRICULUM VITAE

KAREN DAVIS HOLL

Education

Ph.D. Biology, 1994, Virginia Polytechnic Institute and State University

B.S. Biology, with honors, 1989, Stanford University

Positions held

1994 (summer). Instructor, Rocky Mountain Biological Laboratory, Gothic, CO. Taught field course in restoration ecology.

1990-1994. National Science Foundation Predoctoral Fellow. Research projects have addressed: (1) the effect of surface mine reclamation on vegetational and lepidopteran conservation (2) the use of native plant species for landfill and surface mine restoration (3) knowledge of and attitudes towards the environment and population growth in Costa Rica.

1993 (fall). Instructor, Virginia Polytechnic Institute and State University. Responsible for designing curriculum for and teaching honors colloquium on restoration ecology for upper level undergraduates.

1991-1992. Graduate teaching assistant, Virginia Polytechnic Institute and State University. Taught general biology laboratory to non-science majors and plant taxonomy laboratory to upper level students.

1990 (Winter). Environmental advisor, León, Nicaragua. Wrote environmental impact statements in Spanish. Translated U.S. environmental legislation into Spanish. Worked on a river restoration project.

1989 (Fall). Environmental education intern, Hidden Villa, Los Altos Hills, CA. Taught environmental biology and organic farming to elementary school children.

1987-1989. Research assistant, Stanford University and Rocky Mountain Biological Laboratory, Gothic, CO. Assisted with experimental design and data collection on a number of projects studying behavior and population biology of birds and insects.

Publications

Holl, K. D., G. C. Daily, P. R. Ehrlich. 1990. Integrated Pest Management in Latin America. Environmental Conservation 17: 341-350.

Holl, K. D., G. C. Daily, and P. R. Ehrlich. 1993. The fertility plateau in Costa Rica: a review of causes and remedies. Environmental Conservation 20: 317-324.

Holl, K. D., and J. Cairns, Jr. 1994. Landscape indicators in ecotoxicology. Pages xxx-xxx in: D. J. Hoffman, B. A. Rattner, A. G. Burton, and J. Cairns, Jr. (eds.). Handbook of Ecotoxicology. Lewis Publishers: Chelsea, MI. (in press)

Holl, K. D., J. Cairns, Jr., and T. Rattray. 1994. Recycling by design. Speculations in Science and Technology 16:xxx-xxx. (in press)

Holl, K. D., and J. Cairns, Jr. In review. Vegetational community development on reclaimed coal surface mined sites. Submitted to Bulletin of the Torrey Botanical Club.

Holl, K. D. In review. The effect of coal surface mine reclamation on lepidopteran conservation. Submitted to Journal of Applied Ecology.

Holl, K. D. and John Cairns, Jr. In preparation. Nectar resources and their effect on butterfly population dynamics in a reclaimed area. For submission to Restoration Ecology.

Holl, K. D., G. C. Daily, and P. R. Ehrlich. In preparation. A survey of knowledge and attitudes in Costa Rica regarding population-environment-biodiversity issues. For submission to Conservation Biology.

Selected abstracts & conference proceedings

Holl, K. D. 1994. Restoration ecology: some new perspectives. Presented at the international workshop: Preservation of Natural Diversity in Transboundary Protected Areas. May 16-24, 1994. Bieszczady and Tatry Biosphere Reserves, Poland.

Holl, K. D., and J. Cairns, Jr. 1993. Diurnal lepidopteran communities of reclaimed coal surface mines. Pages 16-20 in Proceedings, Powell River Project Symposium Progress Reports. Clinch Valley College: Wise, VA.

Holl, K. D., and J. Cairns, Jr. 1993. The effect of coal surface mine reclamation on lepidopteran populations in Virginia. Bulletin of the Ecological Society of America 78(supplement): 279.

Holl, K. D., and J. Cairns, Jr. 1992. Diurnal lepidopteran communities of coal surface mined sites. Pages 79-82 in Proceedings, Powell River Project Symposium and Progress Reports. Clinch Valley College: Wise, VA.

Holl, K. D., and J. Cairns, Jr. 1992. The effect of coal surface mine reclamation on lepidopteran conservation in southwest Virginia. Page 139 in Program and Abstracts for the 6th Annual Meeting of the Society for Conservation Biology. June 27-July 1, 1992, Blacksburg, VA.

Holl, K. D., and J. Cairns, Jr. 1991. Reclaimed coal surface mined sites as diurnal lepidopteran habitat: a preliminary report. Pages 87-90 in Proceedings, Powell River Project Symposium and Progress Reports. Clinch Valley College: Wise, VA.

Grants

1994. U. S. Department of Energy Global Change Distinguished Postdoctoral Fellowship. Salary for 2 years

1992. Roanoke Regional Landfill Board. Restoration of the Roanoke Regional Landfill, $23,987

1992. Morrison Institute for Population and Resource Studies. Factors affecting the fertility rate in Costa Rica, $1500

1991. Virginia Academy of Sciences. Butterfly population dynamics on coal surface mined sites in southwestern Virginia, $746

1990. National Science Foundation. Predoctoral Fellowship. Salary and tuition for 3 years.

Public Service

1990-1994. Ecocycle (environmental group), Committee Chair

1992-1994. Unitarian Universalist Fellowship, Program Committee Chair

1993-1994. Unitarian Universalist Fellowship, Board Member

1993-1994. University Talloires Declaration Steering Comittee, Member

1991. Department of Biology Graduate Advisory Committee, Member

1991. Campus Earth Day Coordinator.

Awards

1994. Phi Sigma Award for Outstanding Doctoral Research, Virginia Polytechnic Institute and State University

1994. Paul Derring Graduate Student Service Award, YMCA, Virginia Polytechnic Institute and State University

1994. National Academy of Science Award recognizing promising young women in science

1992. Emily Stuart Volunteer Service Award, YMCA, Virginia Polytechnic Institute and State University

1991. Environmental Conservation Annual Foundation Paper Prize

1991. Elected into Omicron Delta Kappa honor society

Karen D. Holl