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The Redevelopment of Plant Community Diversity on a Surface Coal Mine in Southwestern Wyoming

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THE REDEVELOPMENT OF PLANT COMMUNITY DIVERSITY
ON A SURFACE COAL MINE IN
SOUTHWESTERN WYOMING

by

Thomas Joseph Hatton

A dissertation submitted in partial fulfillment
of the requirements of the degree

of

DOCTOR OF PHILOSOPHY

in

Range Science

Approved:

Major Professor	Committee Member
Committee Member	Committee Member
Committee Member	Dean of Graduate Studies

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Thomas Joseph Hatton

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ABSTRACT

The Redevelopment of Plant Community Diversity
On a Surface Coal Mine in
Southwestern Wyoming

by

Thomas Joseph Hatton, Doctor of Philosophy

Utah State University, 1986

Major Professor. Dr. Neil E. West

Department. Range Science

A study was initiated in 1981 to investigate the redevelopment of plant communities on a recontoured, surface-mined site in southwestern Wyoming. The landscape pattern of vegetation was compared with topographic position, the initial topsoil pattern, and cultural practices including shrub planting density, shrub planting pattern and topsoil treatments for the years 1982-1985. In addition, differences in vascular plant species diversity within these various factors and treatments were determined, along with the trend in intracommunity (alpha), intercommunity (beta) and landscape (gamma) diversity for the entire study area.

The influence of site factors and tested cultural practices on the differentiation of the landscape vegetation pattern generally declined over time. There were no persistent differences in diversity among any cultural treatments or topographic positions, with the exception of planting pattern. The vegetation classified by cultural treatment, topographic position, or the initial plant communities identified in 1982 showed strong successional convergence by 1985. This is reflected in a decline in beta and gamma diversity over the period 1984-1985. Alpha diversity remained stable over this same period, though species richness increased, indicating that a subset of species is becoming more dominant. The landscape vegetation pattern apparently converged to a rather uniform composition, increasingly dominated by perennial grasses, forbs and shrubs.

(120 pages)

INTRODUCTION

Statement of the problem

The redevelopment of plant community diversity on drastically disturbed lands is a subject of both legal and ecological importance, but also one that has received little attention. If the goal of land rehabilitation is to reconstruct a stable ecosystem similar to the pre-disturbance condition, then a knowledge of the system's inherent capacity to diversify at both the community and landscape level is necessary to interpret and direct plant succession in a manner which optimizes diversity and stability.

The Surface Mining Control and Reclamation Act of 1977 (Public Law 95-87) requires the establishment of a:

> diverse, effective, and permanent vegetative cover...capable of self-regeneration and plant succession at least equal in extent of cover to the natural vegetation of the area...

on all disturbed post-mining lands. Of the sundry requirements for revegetation success, the re-establishment of diversity may be the most difficult to satisfy (Sindelar, 1980; Narten et al., 1982; Depuit, 1984).

But beyond the legal question, the redevelopment of plant community diversity on drastically disturbed lands is also of ecological interest. Along which environmental gradients do plant communities rediversify? Are there cultural practices that can redirect succession and therefore increase community and landscape diversity? Will early landscape diversity disappear if communities converge successionally? These questions have direct implications for succession and disturbance theory.

Objectives

This study examined plant succession and plant community redevelopment on a recontoured surface-mined site, in relation to soil patterns, topographic position and cultural practices. The specific objectives and associated null hypotheses were as follows:

1. For each year of the study, to determine which factor(s) affect redeveloping plant communities.

H1.1: Plant community diversity is unrelated to cultural practices, topographic position or the initial soil landscape pattern.

H1.2: Clumped planting pattern of shrubs produces similar plant communities to regularly spaced shrub plantings.

H1.3: High density planting of shrubs produces similar plant communities to low density shrub plantings.

H1.4: Plant community diversity is not related to topographic position.

H1.5: Plant community diversity is not related to the initial soil landscape pattern.

2. To determine the alpha, beta and gamma diversity of the redeveloping vegetation.

H2: Diversity does not change over time at the alpha, beta or gamma level of resolution.

3. To determine if cultural practices or topographic positions produce varying degrees of species diversity.

H3.1: All topographic positions are equally diverse.

H3.2: Low and high density shrub plantings are equally diverse.

H3.3: Nonstored and stored topsoil treatments are equally diverse.

H3.4: Clumped and regular shrub planting treatments are equally diverse.

H3.5: Medium density, regular shrub planting treatments on stored and nonstored topsoil are equally diverse.

4. To determine if the redeveloping vegetation classified by either topographic position or cultural treatment has similar successional trajectories.

H4.1: Vegetation classes on different topographic positions have similar successional trajectories.

H4.2: Vegetation classes on treatments with varying shrub planting densities have similar successional trajectories.

H4.3: Vegetation classes on treatments with clumped shrub planting patterns has a trajectory similar to vegetation on treatments with a regular planting pattern.

H4.4: Vegetation classes on unplanted, nonstored topsoil have successional trajectories similar to vegetation on unplanted, stored topsoil.

5. To determine if the plant communities identified with cluster analysis in 1982 have similar successional trajectories.

H5.1: Plant communities identified with cluster analysis in 1982 have similar successional trajectories.

REVIEW OF LITERATURE

Diversity defined

Because this study is concerned principally with diversity, a discussion of this concept is necessary due to the complex and varied treatment diversity has received in the ecological literature. Indeed, due to the great ambiguity in terms and interpretations, Hurlbert (1971) suggested abandoning the term in favor of better-defined concepts. Peet (1974), in his review of the subject, found no generally-accepted definition of diversity in the literature. If diversity is to be considered, therefore, a precise definition of terms is necessary.

This study employed the dual-concept of diversity first proposed by Simpson (1949), wherein diversity is a function not only of the number of species in an area (richness), but also how the dominance among those species is apportioned (eveness). This approach was advocated by Lloyd & Ghelardi (1964), Margalef (1969) and Pielou (1969), and is synonymous with Peet's (1974) term "heterogeneity."

Ecological implications of diversity

The legal requirement for diversity is based on strong ecological considerations; one of the hallowed tenets of modern community ecology has been that diversity correlates with stability (Krebs, 1985). The cause-and-effect relationship between these two concepts is no longer entirely clear, however (Odum, 1971). Part of the problem is that there is no concensus on the definition of stability in the literature; stability has been variously defined as (a) relative constancy of numbers over time (Margalef, 1968; Goodman, 1975); (b) the resistance to outside perturbation (Pielou, 1975), and (c) the ability of a system to reorganize after ecological displacement, or elasticity (Margalef, 1969; Holling, 1973).

Whittaker (1969) concluded that there are broad trends of decreasing diversity of vascular plant communities toward environments less stable, less favorable, and more extreme, but that the details of these trends are strongly modified by more complex relationships involving species interactions, particularly with dominant species. Ulrich (1976) stated that ecosystems with good biological soil conditions and high species richness are less sensitive to partial or complete destruction of vegetation. Whittaker (1975) stated that the instability of desert ecosystems has become an aspect of environment to which plants respond with niche differentiation and consequent diversity. If Whittaker's (1975) statement is true for arid lands in the western United States, then the contra-positive must also be true; that is, a reduction in species diversity may be equated with a lessened ability of these ecosystems to adjust to environmental fluctuation.

May (1973) attacked the argument that increased community complexity necessarily leads to increased stability, and cautioned community ecologists that if diversity seems linked to stability in the real world, it is not a consequence of species interactions; evolution may have merely produced nonrandom assemblages of species in which diversity and stability are related. Alternative models developed by Jeffries (1974) and DeAngelis (1975) suggested conditions under which stability is facilitated by diversity. Mellinger and McNaughton (1975) concluded that resolution of the relationship between species diversity and functional stability of ecosystems must be empirical. In one such empirical study, McNaughton (1977) confirmed the hypothesis that plant community diversity stabilizes ecosystem functional properties of the community (at least at the producer level) against environmental perturbations, whether experimentally induced or intrinsic to the ecosystem. McNaughton (1977) stated that the traditional model (as stated above) is considerably more robust in application than more recent mathematical models to the contrary. McNaughton (1978) concluded that more diverse ecosystems may be more stable because connectance decreases to accomodate greater diversity. Therefore, species are organized as relatively small guilds; the interaction strength among species declines as diversity increases. In this context, however, the causal relationship between higher diversity and increased stability depends on how the community is structured.

Another ecological reason for the reestablishment of plant diversity is its relationship to animal diversity. MacArthur (1965) showed a strong correlation between species diversity in bird communities and the corresponding complexity of their respective plant communities. More pertinent to the theme of this study, habitat (or community) diversity bears a strong relationship to the abundance and diversity of animal populations. Hansson (1977) suggested that landscape diversity may be positively correlated with the abundance of animal species requiring different habitats during the day or year, species exposed to predation, and generalist species capable of living and reproducing in a variety of habitats. Romme & Knight (1982) suggested that changes in landscape pattern may influence a variety of natural features, including wildlife, water and nutrient flow, and the probability of different kinds of natural disturbances. Several papers have identified the need to create or maintain diverse plant communities on mined lands (Karr, 1968; Allaire, 1979; Tyus & Lockhart, 1979; Steele & Grant, 1982; Emrich, 1983).

Allen (1984), however, pointed out the need to consider functional diversity as well as plant species diversity in the evaluation of ecosystem reconstruction. Allen (1984) suggested that the importance of specific functions such as persistence, palatability, or structural diversity of the vegetation may be lost in any similarity or diversity index which gives equal emphasis to all species. Specifically, she suggested that the unique functions attributed to shrubs are not expressed by indices which do not retain species identities.

The concern over shrub reestablishment is based on a large body of literature relating vegetation structure with animal habitat selection (Willson, 1974; M'Closkey, 1975; Atkeson & Johnson, 1979; Andersen et al., 1980; Sullivan & Sullivan, 1982). Wiens & Rottenberry (1981), however, found that while birds respond to some elements of habitat structure, their within-habitat responses may be more strongly correlated with details of habitat floristics. These authors suggested that a complete understanding of ecological patterns in avian community habitat relationships requires a knowledge of vegetational floristics as well as physiognomy. Rice et al. (1984) questioned previous studies correlating avian habitat selection with vegetation profiles, and concluded that floristics plays a more important role in avian habitat selection than previously reported. MacMahon (1976) found that vertical and horizontal foliage complexity did not correlate well with small mammal diversity.

Shrubs clearly have a unique function in semiarid ecosystem nutrient cycling; organic matter and nutrients accumulate under shrub canopies, and therefore these soil properties spatially covary with shrub location. This horizontal, anisotropic distribution was demonstrated for sodium and potassium (Rickard & Keough, 1968), organic carbon (Charley & West, 1975), phosphorus (Charley & West, 1975; West et al., 1984), total sulfur (Tiedemann & Klemmedson, 1973), and nitrogen (Garcia-Moya & McKell, 1970; Rixon, 1971; Tiedemann & Klemmedson, 1973; Charley & West, 1975; Wikeem & Pitt, 1982). Because of the suggested and demonstrated importance of shrubs in semiarid ecosystem function, it is desirable to give them unique consideration in ecosystem evaluation apart from any synthetic index.

Measures of diversity

A variety of quantitative indices of diversity has been proposed, and the choice of any particular index may greatly influence the biological interpretation of the data. Pielou (1975) stated that an index of diversity should have the following properties:

1. For a given number of species (s), an index value (H') should have its greatest value when the proportion of the community belonging to the _i_th species (p_i) is

equal to 1/s for all _i_, which is equivalent to complete eveness.

2. Given two completely even communities, one with s species and the other with s+1, the latter should have the greater H'.
3. The index should lend itself to an unbiased evaluation of diversity at more than one hierarchical level.

Khinchin (1957) and Pielou (1969) claimed that the only known function of p_i having these properties is the Shannon (Shannon-Weiner) index, first proposed by Shannon & Weaver (1949).

Dickman (1968) found that the Shannon index failed to reflect significant changes in plankton community structure. Sager & Hasler (1969) criticized the Shannon index response to changes in species importance, claiming that the Shannon index is insensitive to rare species. Although Fager (1972) claimed that the Shannon index is most sensitive to species with an importance value of 1/e, Peet (1974) demonstrated that while the maximum contribution that a species can make to index response is 1/e, the index responds most strongly to changes in importance of the rarest species.

Hurlbert (1971) concluded that the Shannon index is useful only if one is interested in the number of information bits per individual, yet the relevance of information theory to biology has never been demonstrated. Goodman (1975) could not identify any ecological process that corresponds in any obvious way to the information theory upon which the Shannon index is based; he concluded that the Shannon index is "dubious," and that whatever the index does measure seems to have no biological interpretation. Washington (1984) claimed that despite the extensive use of the Shannon index, none of the above criticisms have been answered by proponents of information theory, and that the continued use of this index seems to be because of its entrenched nature rather than any belief in its biological relevance. Washington (1984) concluded that the biological relevance of Simpson's (1949) index, as modified by Hurlbert (1971), is more apparent than that of other indices.

Routledge (1984) developed hierarchical estimates of diversity (sensu Whittaker, 1972) employing both the Shannon index and the Simpson index as modified by Hurlbert. Routledge (1984) observed that the extreme sensitivity of the Shannon index to the abundances of rarer species foreshadows potential difficulties in constructing estimators of hierarchical diversity, and that these estimators based on the Shannon index had substantial bias, while the estimators based on the Simpson index had negligible bias.

In a review of techniques for evaluating diversity on reclaimed surface mines, Chambers (1983) criticized the use of diversity indices such as the Shannon index primarily on the basis of a lack of statistical comparability between values obtained from two different communities. In addition, Chambers (1983) concluded that diversity indices do not reveal changes in apportionment of species among communities, and for this reason recommended the use of similarity indices for the evaluation of revegetated sites against a reference area. Similarity to a reference area may be inappropriate; the drastic disturbance to soil and microtopography associated with surface mining creates an ecosystem which may resemble an undisturbed site only in terms of macroclimate.

Chambers' (1983) recommendation, however, confounds the concepts of diversity and similarity. It is quite possible to envision a situation where a revegetated area consists of communities with the same species richness and eveness as the reference area, but supporting a somewhat different list of native species. Under these conditions, there would be the same degree of diversity but a similarity value less than unity. Another case in which similarity indices seem inadequate for evaluating diversity is when a revegetated area consists of all the species found in the reference area plus a number of others not found there, and with an overall greater eveness. In this case the revegetated site is obviously more diverse, but the similarity index is again less than one. Finally, Chambers (1983) was dealing with only alpha diversity, and similarity indices do not lend themselves to evaluating hierarchical diversity. In short, similarity indices are most appropriate for evaluating similarity, not diversity.

The problem of establishing variances for diversity indices is not restricted to the alpha level alone. Pielou (1975) stated that a method for estimating the sample variance of beta diversity does not exist. Zahl (1977), however, applied a jackknifing procedure (a randomization test) to the estimation of the Shannon index which produced an approximately normal distribution, even when based on a nonrandom sample, variable quadrat sizes, lack of independence among samples, and a rather small sample size. Zahl (1977) concluded that the only test necessary was for normality of the distribution of the jackknife estimates, and that normality can be produced most quickly by random, independent samples. Tukey (1958) conjectured that the values derived from a jackknife procedure are approximately independently and normally distributed. Heltshe & Forrester (1983) presented an exact expression for the jackknife estimate of the number of species using quadrat samples and examined the procedure's bias-reducing properties. Smith & van Belle (1984) showed jackknifing to reduce bias in species richness, and to give better estimates than bootstrapping (an alternative randomization procedure) when the number of quadrats is small. Heltshe & Forrester (1985) demostrated that the jackknife estimate of Simpson's index is unbiased when using quadrat sampling techniques, although the confidence intervals may be somewhat conservative at large sample sizes.

Routledge (1984) presented estimators of alpha, beta and gamma diversity along with estimators of variance at each of these levels through use of a jackknife technique with both the Simpson and Shannon indices. Wilson & Shmida (1984) evaluated several measures of beta diversity, and concluded that the estimator developed by Routledge (1977) is independent of the number of community changes, alpha diversity, and excessive sampling even when based upon presence-absence data. Despite the assertion by Khinchin (1957) and Pielou (1969) that only the Shannon index satisfies the criteria for diversity indices developed by Pielou (1975), the development by Routledge (1984) of hierarchical diversity estimators with the Simpson index demonstrates that this index also satisfies these criteria.

Most diversity indices, including the Simpson, take into account the frequencies or counts of individuals of each species for an estimate of species importance (eveness). However, other expressions of importance are possible as well. Whittaker (1972) suggested that productivity was the best expression of a species' command of the resources within a given area, but recognized that productivities are not easily measured, and that often species must be compared by other parameters such as coverage, density or biomass. Lyons (1981) concluded that the importance of a species should be based on the relative amount of a continuous quantity, such as coverage or biomass.

Landscape diversity and environment

Landscape diversity may be related to a number of factors. Under the individualistic hypothesis first proposed by Ramensky (1924) and Gleason (1926), species respond to environmental gradients according to their own unique manner of relating to both the physical environment and interactions with other species. This condition may result in communities intergrading continuously along gradients. Van der Maarel (1978) stated that gradients are of great importance for niche differentiation of plant species, leading to local concentrations of different species, and that preservation or restoration of gradients will effectively contribute to the fulfillment of information functions.

Competition along these environmental gradients does not generally result in sharp community boundaries (Whittaker, 1975). However, two additional factors may produce a landscape of communities with sharp boundaries: environmental discontinuities along environmental gradients (Whittaker, 1975) or patterns resulting from portions of a landscape being in different stages of recovery following disturbance (Romme, 1982). Where sharp community boundaries exist, for whatever reason, a more organismic (Tansley, 1935) view of plant communities is appropriate.

Plant succession on mined lands

Research on vegetational succession on mined lands has focused primarily on within-community changes. Although plant succession on seeded mine spoil is influenced by the choice and success of species seeded, cultural practices and weather, general patterns of community development are discernable (Sindelar, 1979). On mined lands in the western Unites States with replaced topsoil, Parton & Woodmansee (1980) stated that succession typically proceeds through an annual pioneer stage consisting of species of _Salsola_, _Kochia_ and annual grasses, which increase in biomass to a peak at about two or three years after recontouring. Allen & Knight (1984) attributed the success of introduced annuals during the earliest successional stages on mined lands to effective dispersal mechanisms, deep rooting, and temporal resource partitioning, where the introduced species maximize growth either earlier or later in the season than most of the native species. As these annual weeds decline in importance, they are replaced by short-lived perennials and biennuals. Allen (1984) stated that weedy species, which either do not form or have little dependence upon mycorrhizae are often the initial colonizers of mined lands, and that the reestablishment of mycorrhizae in the soil may improve the ability of desirable species to compete with weeds and increase the rate of succession from a weed stage to a diverse and stable community consisting of perennial forbs, grasses and shrubs. Wali (1980) identified a final seral condition where the community consists solely of those species found in the undisturbed native condition.

Within-community species diversity on seeded mine spoils may initially be high, perhaps even within the range of undisturbed native areas, but these levels may not be indicative of the level of diversity in subsequent stages of plant community development (Depuit et al., 1980). This may be a response to progressively greater dominance by a subset of the seeded species coupled with a loss of pioneer species, generally following Margalef's (1968) model of diversity and succession. Harthill & McKell (1979) and Wali (1980), however, employed a model developed by Connell (1978) to describe changes in diversity with succession on surface mined lands wherein diversity is initially quite low, increases with time to some maximum, and then decreases somewhat to a stable level. This model is akin to Odum's (1969) view that species richness generally increases as a community matures.

The development of landscape diversity in vegetation on surface-mined lands in the arid west has received little attention. Wali (1980) observed the effects of aspect and topographic position with uniform soil parent material on the distributions of plant species on undisturbed and mined sites in western North Dakota. He also described five native communities separable by these site factors, though he failed to discern plant associations on the mined site.

Because the differentiation of the landscape into plant communities depends in part upon environmental gradients, several studies have investigated the redevelopment of gradients in soil chemical and physical properties on mine spoils and have attempted to relate this to the pattern of plant reestablishment. Wali & Freeman (1973) compared pairs of unmined areas with spoil banks for several soil series. The authors then described changes in a number of soil properties and the type of vegetational changes. Unfortunately, their design was confounded by highly variable site histories, treatments, and geographical locations, and they drew no conclusions regarding site heterogeneity over time.

Schafer & Neilsen (1979) and Sindelar (1979) related differences in successional patterns of vegetation on suface-mined lands to the spatial distribution of soil texture; both studies concluded that perennial grasses came to dominate the fine-textured soils at the expense of shrubs. Schafer & Nielsen (1979) found that coarser soil textures favored annual grasses, while Redente & Cook (1981) found that these textures favored the growth of forbs and shrubs.

Jonescu (1979) concluded that plant communities redeveloping on ridges were different from those redeveloping on interridge areas of stripmines. The latter had a greater plant species diversity and were successionally accelerated.

Depuit (1980) concluded that after as short a time as three growing seasons , the effect of cultural practices such as seeding mixture, rate, and technique on diversity declined substantially. Doerr & Redente (1983) concluded that the effects of cultural practices do not persist long, and that environmental factors dominate the patterns of plant community development thereafter. Allen (1983) concluded that the early abundance of introduced annuals did not have a significant effect on species richness nor species eveness after seven years of succession following discing in eastern Wyoming sagebrush grassland, although species composition was affected.

Fisher & Deutsch (1983) noted that in the undisturbed condition, the uplands of the Wasatch and Fort Union geologic formations are characterized by pronounced soil diversity; highly contrasting soils can occur at intervals of less than twenty meters. They concluded that such diversity is lost as a result of mining and reclamation, which produces a relatively homogeneous soil with a commensurate loss of biological diversity. Schafer (1979) compared the variability of soils on a recently mined site with the variability of soils in a natural, undisturbed soil landscape in southeastern Montana. He concluded that large variations in soil existed on mined land, but primarily on a local scale (0 to 10 meters). Natural soils were more variable on a landscape scale (>500 meters), and this variability was highly correlated with landscape features. Variability in minesoils was influenced primarily by mining techniques. Schafer (1979) recognized fourteen soil families on the 300 hectares of undisturbed landscape, but only two families occurred on a 400 hectare mined site.

Though it is the horizontal differentiation of soil properties that is most directly related to plant community differentiation on initially homogeneous mine spoil, the subsequent processes of soil genesis on mine spoil may ultimately contribute to the redevelopment of landscape patterns, at least indirectly (Huggett, 1975). Wali (1980) noted two soil processes characteristic of the genetic development of recontoured spoil in the arid west. The first of these is the leaching of soluble salts from the upper soil zone into a lower zone over time. The second of these processes is the increase of organic matter and the concentration of major plant nutrients in the upper soil zone. This increase in organic matter leads in part to an increase in soil aggregate structure (Wilson, 1957). Schafer et al. (1976) observed that sheet erosion on spoil material is likely to be very significant until sufficient plant cover is established.

METHODS AND PROCEDURES

Study area

The research area was located on the Elkol-Sorenson Mine of the Pittsburgh and Midway Division of Chevron USA, Incorporated, near Kemmerer in southwestern Wyoming. Annual precipitation at Kemmerer averages 22.6 centimeters with the majority coming in spring and early summer. Average monthly temperatures range from -8 degrees Celsius in January to 17 degrees Celsius in July at Kemmerer (Parmenter & MacMahon, 1983). Climatic diagrams for the general area and the study site appear in Appendix 1. Precipitation was markedly above average for the years 1982-1984, and was relatively normal in 1985. The dominant native vegetation type was classified by West (1983) as temperate semi-desert sagebrush steppe. Current land use other than mining consists principally of livestock grazing and wildlife habitat (May et al., 1971). These are the identified post-mining land uses as well.

The experimental plots were laid out on a 50 hectare site, known as 12UC, which was mined in the late 1970's and recontoured in 1979. Topsoil was replaced according to design specifications in the fall of 1981. The site has a generally southern exposure with two west-facing and two east-facing slopes.

Experimental design

The experimental design was a large, multifactorial, replicated series of treatments, composed of three main portions: (1) natural revegetation, (2) management, and (3) successional augmentation.

Every treatment is represented by three replicated plots, either of 0.5 or 0.13 acres in extent. Plot location was limited to the two east-facing slopes, and assignment was random save for the lack of interspersion of the management replicates.

Revegetation plots were initiated in the fall of 1980. Three different treatments were involved. On some plots, topsoil stored since 1976 was applied. On other plots no topsoil was applied; the plant growth medium was spoil material. Finally, a third treatment involved the placement of fresh topsoil, directly applied after removal from a sagebrush-grass site a few tens of meters south of 12UC. No planting was done on these natural revegetation plots.

Management plots, initiated in 1980, include replicates which were managed by the mining company according to the State of Wyoming Department of Environmental Quality prescription. This management included an initial cover crop of domestic barley in the fall of 1980 (*Hordeum vulgare*) and subsequent seeding with several species of *Agropyron* in the spring of 1981. There is some indication that *Artemisia tridentata* and *Atriplex canescens* were included in the seed mixture, though these species did not successfully establish.

The major design effort was subsumed under the successional augmentation experiments, initiated in the spring of 1981. An objective of this study was to investigate the roles of shrub planting density and planting dispersion in ecosystem rehabilitation. The treatments involved two dispersion patterns (clumped and regular) and four density levels. The latter were defined with respect to the normal shrub density of the surrounding native vegetation, approximately 16,000 shrubs per hectare. Density treatment levels were based on proportions of this normal level (0.2, 0.6, 1.0, and 1.5).

Three shrub species (*Artemisia tridentata*, *Chrysothamnus viscidiflorus* and *Atriplex gardneri*) were planted in equilateral triads, with 40 centimeters between plants and with each apex consisting of one of the shrub species. Clumped experiments involved plant clumps containing three triads and an additional tenth plant so that all plants had the same inter-plant distance as in the triads. The tenth plant was alternated among the three species. Independent of shrub plantings, a number of points equal to the number of triads, but randomly allocated, were seeded with *Agropyron smithii* (90 seeds per point) and *Hedysarum boreale* (10 seeds per point).

Finally, a number of control plots were established. All controls were planted at the normal shrub tubeling density. Sets of replicate controls represent both dispersion patterns. Additionally, some control plots were planted with a single species of shrub at all of the apices of the triads to assess the effects of shrub species composition.

Vegetation sampling

Since slope position may play an important role in the pattern of vegetation dynamics, all experimental plots were divided into four equal quarters, with the first quarter located closest to the base of the slope and the fourth quarter closest to the ridge. Vegetation sampling was divided equally among the four quarters, resulting in a simple, stratified design.

The area within each quarter-slope replicate plot was subdivided into three types: (1) area planted with shrubs, (2) area planted with grass and forb, and (3) unplanted (interspace) area. The relative contribution each type made to the whole was dependent upon the treatment; high density plots have the greatest possible planted area, and unplanted plots have none.

Interspace areas were of two types: random interspace and interspace adjacent to a location planted with shrubs or grass (associated interspace). On every slope quarter, five random interspace locations were monitored within a 0.25 m^2 circular quadrat. Random interspace quadrats were located randomly except that the location could not contain a planted shrub, grass or forb individual. Associated interspace quadrats were located at a permanent compass direction and distance (0.28 meters) from the planted shrub or grass location; the direction with respect to the shrub or grass quadrat was random except that the quadrat could not contain a planted shrub, grass, or forb individual. These quadrats were monitored within the same kind of quadrat as the random interspaces. All quadrat centers were permanently located with steel concrete reinforcement rods (1/4 inch diameter) and the same locations were sampled each year.

Greig-Smith (1983) stated that successive changes detected with permanent plots might be correlated, though the likelihood of this occurring in native vegetation is uncertain. Austin (1981), however, strongly recommended the use of permanent plots based on the need to discriminate between temporal and spatial variability.

The methods used to obtain vegetation data were constrained by the requirement of nondestructive sampling. Sampling is timed to correspond to the average peak phytomass for the species on the reclaimed site, which usually occurs in the latter half of July. Cover by species was determined by ocular estimate (Hatton et al., 1986), based on a vertical projection of the quadrat boundary, and included the above-ground coverage of vascular plants or portions of plants within this volume.

Greig-Smith (1983) suggested that in large-scale investigations involving multiple observers (such as this one), personal bias is best included in the sampling error and not in between-treatment differences. Thus a given observer should not estimate all of the replicates of a treatment. In any year of this study, no observer estimated all three replicates of any treatment. In addition, of the approximately twenty observers employed over the period of this study, only two estimated vegetation in more than one year. Thus, temporal changes should be relatively free from personal bias.

Vegetation data analyses

All data analyses were based on information collected with the 0.25 m^2 interspace rings. By using this information, the direct effect of the presence of planted individuals is eliminated, and the analyses are restricted to the evaluation of plant community dynamics as affected by the initial planting pattern, planting density, topsoiling, or management treatments.

There were three types of interspace quadrats of this size sampled: random interspace, grass- and shrub-associated. Inspection of the species-area curves developed within a slope quarter of a given treatment replicate plot suggested that about ten such quadrats were necessary to adequately characterize species richness, based on the criterion of the curve approaching some assymptote assumed to represent the parametric value of richness. However, there were at most only five quadrats of any type sampled within each slope quarter. Therefore, it was desirable to use information from all three sources collectively to characterize a slope quarter of a treatment replicate plot.

However, because the grass- and shrub-associated quadrats were located near planted individuals, there was the potential that they might be biased with respect to the random interspace quadrats. To inspect for this proximity effect, the Bray-Curtis similarity coefficient was used to construct a resemblance matrix for the quadrats from the three different sources for the lowest slope quarter of a nonstored topsoil treatment and for the uppermost slope quarter of a low density clumped treatment in 1985. The selection of these two treatment quarters was to test for this proximity effect at the expected extremes in species richness and plant density, respectively.

Differences in similarities among the three sources of quadrat data were tested for with the Wilcoxon rank sum test (Sokal & Rohlf, 1981). Lack of a significant difference at the 90 percent level of confidence among the three sources of quadrat data was used as justification for their combination.

Given the above results, each slope quarter of each treatment replicate was characterized by the average percent cover, by species, based on the random interspace, grass- and shrub-associated quadrats. These became the basic units for all of the subsequent analyses on 12UC.

The use of a sample by species cover matrix can pose some problems, however. Though cover is a continuous variable, it is bounded (Smartt et al., 1976). Because many of the species present were unique to a given slope quarter, the data matrix for any given year consisted of a large number of zero values. Gauch (1982) pointed out the bias created with a matrix dominated by zero values when attributes are rescaled, and suggested the deletion of species occurring in less than five percent of the samples. However, Lambert & Dale (1964) stated that in the case of a matrix of continuous quantities with many zero values, dichotomizing at the zero/nonzero boundary loses little information, and demonstrated that the loss of information is less than might be intuitively expected. Clifford & Stephenson (1975) stated that indices which ignore joint absences are appropriate for ecological studies where a few species are common to many plots, and many species are restricted to a few plots. This is the situation which developed on 12UC. These authors demonstrated that similarity indices which ignore joint absences include the Jaccard, Czekanowski, and Bray-Curtis coefficients; the latter finds its most frequent use in ecology (Romesburg, 1984). Beals (1973, 1985) argued for the use of the Bray-Curtis similarity coefficient on the basis that all species contribute to the measure in proportion to their relative differences in the two samples.

Identification of plant communities

To address any question concerning plant community development, it was first necessary to identify plant communities within the study area. Gauch (1982) stated that communities sampled should be homogeneous in structure and composition if the research purpose is to represent communities by samples. The traditional approach among plant ecologists has been that the existence of homogeneous communities is discernible to the naked eye, and normally homogeneity is either assumed or subjectively determined (Curtis & McIntosh, 1951; McIntosh, 1967). On reclaimed mined lands, however, the redeveloping plant communities are usually so subtly different or spatially interspersed that subjective identification is impossible.

Cluster analysis provided an alternative means for a more objective identification of plant communities, which allows the synthesis of information (attributes) from multiple samples (objects). This sequential comparison of samples required the choice of a similarity index. The cluster analyses of vegetation data in this study employed the Bray-Curtis similarity coefficient. The unweighted pair-group method using arithmetic averages (UPGMA) was used as the clustering algorithm. Romesburg (1984) pointed out that this is the most common algorithm, chiefly because it judges similarity in a manner less extreme than either single or complete linkage clustering. Equal emphasis was given to all species in the data matrix (no standardization or transformation), following Gauch's (1982) recommendation. CLUSTAR (Romesburg & Marshall, 1984) was selected as the computer program by which the analyses were performed. The resultant dendrogram was reduced to a nonhierarchical classification principally according to the criterion of Ruben (1967), which states that the similarity within a group of objects should exceed the mean similarity of all the objects in the dataset. This rule was modified at times on the basis of an *a priori* standard of six to ten clusters, with at least five elements each.

Community patterns

The Type I errors associated with the null hypotheses were estimated for contingency tables relating object (quarter slope replicates) membership to cluster analysis classes. Expected class membership was derived from random assignments of objects based on row and column frequencies. CHITEST (Romesburg & Marshall, 1985), a program that uses Monte Carlo methods to test the null hypothesis that the row and column factors are independent of one another, was used to estimate p-values associated with the hypotheses concerning factors potentially related to the distribution of plant communities. Ten thousand random tables and their chi-square statistics were generated by means of this program, and the p-value was calculated as the fraction of these which exceeded the test statistic for the observed table.

When the p-value of such tests is significant, one can say with some certainty that the row and column factors are not independent. Nevertheless, the significance alone tells almost nothing about the strength of the association. It is desirable to estimate the predictive strength of the relation as well. Therefore, the symmetric index of predictive association, lambda (Goodman & Kruskal, 1954), was calculated for each contingency table.

Diversity estimates

Based on the identification of plant communities (cluster analysis classes), diversity at the alpha, beta and gamma levels was calculated for 12UC along with estimates of variance for these values according to methods described by Routledge (1984), based on Routledge's (1980) estimator of Simpson's index of concentration. The distribution of the jackknife pseudovalues converges to normal for these diversity estimators with increasing sample size, but because the number of communities in a dataset may be small, the variance estimators may be biased. The distributions of all three components of diversity were therefore tested for normality with the one-sample Kolmogorov-Smirnov goodness of fit statistic for the intrinsic hypothesis (Sokal & Rohlf, 1981) prior to any parametric comparisons.

Although beta is defined as the quotient of gamma over alpha, and so might not appear independent of gamma and alpha, Routledge (1984) proved that alpha, beta and gamma are distributed asymptotically normal. However, for small sample sizes, the asymtotic joint distribution may not be achieved. As an alternative, beta was also calculated as the quotient of the jackknife gamma over the jackknife alpha, and its variance approximated as a function of the variances and covariance of the jacknife gamma and alpha diversities (Mood et al., 1974).

Jackknife estimates of diversity were also determined for the following subsets of treatments on 12UC: (1) successional augmentation plots; (2) low planting density plots; (3) medium planting density plots; (4) normal planting density plots; (5) high planting density plots; (6) regular planting pattern plots; (7) clumped planting pattern plots; (8) stored topsoil plots; (9) nonstored topsoil plots; and (10) management plots. Because of the minimal number of replicates involved in the overall experimental design, the interaction of planting pattern and density was not tested. *A priori*, planned comparisons within each year between (1) high and low planting density treatments; (2) regular and clumped planting pattern treatments; (3) stored and nonstored topsoil treatments; and (4) medium density, regular shrub planting treatments on stored and nonstored topsoil treatments were performed by means of t-tests of equality of means when variances are assumed to be unequal (Sokal & Rohlf, 1981).

Because diversity as used in this study is a dual-concept, the components were also estimated separately for 12UC. Species richness was taken as the total number of species recorded. Eveness was calculated for each year as the quotient of Simpson's index over the number of species.

Successional trajectories

Because this study is limited to a relatively short period of recovery following disturbance, it is necessary to determine whether communities persist over time or change, and whether different sites or treatments show similar behavior over time. It is quite possible that the communities existing at the end of this study will converge during subsequent succession and reduce beta diversity. Austin (1977) suggested a technique that used multivariate ordination (reciprocal averaging) of communities over successive years to track the trajectories of succession for those communities over time. This technique allows an estimate of the degree of convergence or divergence among communities as they redevelop following disturbance.

To provide an estimate of the successional trajectories for the vegetation data organized by topographic position, cultural treatments, and the initial plant communities identified with cluster analysis in 1982, reciprocal averaging with the program ORDIFLEX (Gauch, 1977) was performed, and the centroid of each community, topographic position or cultural treatment was plotted for the years 1982-1985 in the species-space of the first two ordinal axes. The axes for species-space diagrams are arbitrary (Austin, 1977).

Species which differentiated composition among the four years of the study were determined by means of the computer program TWINSPAN (Hill, 1979) to provide a descriptive characterization of the overall pattern of plant succession on 12UC.

Because the identity of any individual species is lost in the mathematics of calculating a diversity index, and because the fates of the planted species are of unique interest, the establishment of Artermisia tridentata, Chrysothamnus viscidiflorus and Agropyron smithii within the unplanted interspaces was monitored independently. The low success of Hedysarum boreale and Atriplex gardneri precluded any statistically relevant treatment. The mean and variance of percent cover was calculated for each species following a Box-Cox transformation to maximize the normality of the data (Sokal & Rohlf, 1981).

Soil sampling and analysis

The replaced and recontoured topsoil was sampled in 1981 on a coordinate system to estimate the landscape pattern in plant growth media properties. The upper eight centimeters was analyzed for total nitrogen (Sher, 1955), bicarbonate phosphorous (Murphey & Riley, 1962), and sulfate (Rasnick & Nakayama, 1973). Electrical conductivity (EC) was determined on a saturation extract using a soil moisture salinity bridge and was corrected to 25 degrees Celsius. The soil pH was determined on a saturation extract by means of a KCl gel-filled combination electrode. Potassium and sodium contents were determined by flame emission. Calcium was determined by adsorption, in a solution containing one percent Lanthanum and five percent HCl to suppress interference. All soil sampling and analyses were performed by staff and with facilities of the Department of Soil Science and Biometeorology, Utah State University.

These eight soil variables, along with the percent silt, clay, and sand in the less than two millimeter soil fraction (Day, 1965) were used as the attributes to characterize each sample in a cluster analysis. Data were standardized such that each attribute had a mean of zero and a standard deviation of one. Euclidean distance was selected as the similarity measure, and the UPGMA algorithm was employed. The resultant dendrogram was reduced to a nonhierarchical classification with the same criteria as were used for the vegetation cluster analyses.

RESULTS

Similarity of sampling points

The mean similarities within the three types of 0.25 meter square plots (random interspace, grass- and shrub-associated interspace) did not exceed the similarities among these three types for the uppermost slope quarter of the second replicate of the low density clumped treatment and the lowest slope quarter of the second replicate of the medium density, nonstored topsoil treatment for the years 1984 and 1985 ($p > 0.1$). This was used as justification for combining the data from these three types of plots for subsequent analyses. Mean percent cover by species by treatment for the years 1982-1985 appears in Appendix 3.

Patterns in plant community redevelopment

Table 1 presents the degree of predictive association (lambda) and significance between the classes resulting from the cluster analysis of the vegetation for the years 1982-1985 and topographic position, planting treatments (including management treatments), planting treatments (excluding management treatments), planting density, planting pattern, and the 1981 topsoil pattern. The contingency tables developed for these tests appear in Appendix 2.

The association between the vegetation classes and topographic position (H1.4) was significant ($p < 0.05$) for the last three years of the study (1983-1985). The degree of predictive association was weak, though relatively stable over the study period.

Table 1. Predictive association (lambda) and significance between vegetation classes and topographic position, planting treatments and the 1981 topsoil classes for the years 1982-1985.

	1982 lambda p	1983 lambda p	1984 lambda p	1985 lambda p
Topographic position	0.11 0.39	0.12 0.007	0.12 0.021	0.13 0.04
All planting treatments	0.33 0.00001	0.22 0.00001	0.24 0.00001	0.22 0.00001
Planting treatments (no management)	0.25 0.00001	0.18 0.00001	0.21 0.00001	0.19 0.004
Planting density	0.26 0.00001	0.14 0.001	0.20 0.001	0.22 0.007
Planting pattern	0.10 0.06	0.31 0.001	0.16 0.09	0.14 0.28
1981 topsoil pattern	0.10 0.01	0.20 0.00001	0.09 0.056	0.14 0.074

The association between the vegetation classes and the planting treatments (including the management treatment) was highly significant ($p = 0.00001$) for all years. The strength of the association exhibited a general decline between 1982 (lambda = 0.33) and 1985 (lambda = 0.22). Because the management plots represent a radically different treatment, this same analysis was repeated excluding management replicates. Again, there was a significant ($p < 0.05$) association between planting treatments and vegetation classes for all years, and the strength of the association again declined between 1982 (lambda = 0.25) and 1985 (lambda = 0.19). In each year, the strength of the association was less than if management plots had been included.

There was a significant association ($p < 0.05$) between the vegetation classes and planting density for all years (H1.3). The strength of this association was highest (lambda = 0.26) in the first year of the study. Only in 1983 was there a significant association (lambda = 0.31, $p = 0.001$) between planting pattern and the vegetation classes (H1.2).

The association between the 1981 topsoil classes and the vegetation classes (H1.5) was significant in 1982 ($p = 0.01$) and 1983 ($p = 0.00001$), and not significant ($p > 0.05$) in the following two years.

Patterns in plant community diversity

Table 2 presents the jackknife estimates of alpha, beta and gamma diversity for the entire study area, and beta diversity alternatively

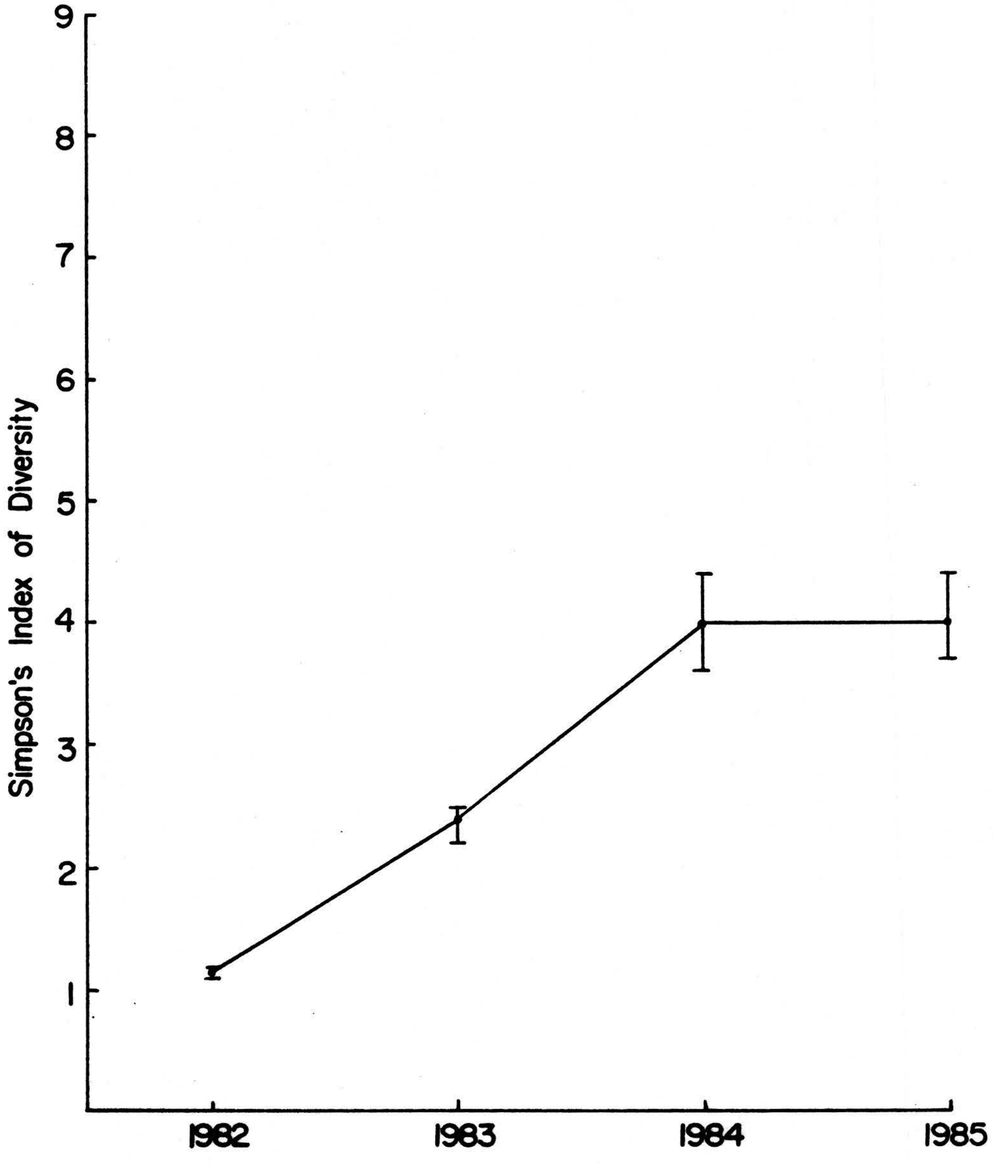
Table 2. Jackknife alpha, beta and gamma diversity, with beta also calculated as a quotient, on 12UC for the years 1982-1985.

	Mean	95 percent confidence interval	Kolmogorov-Smirnov statistic
Alpha			
1982	1.14	1.09 - 1.20	0.194
1983	2.38	2.24 - 2.54	0.161
1984	3.98	3.64 - 4.38	0.150
1985	4.03	3.74 - 4.37	0.136
Beta (jackknife)			
1982	1.05	1.01 - 1.08	0.387
1983	1.32	1.29 - 1.37	0.170
1984	2.11	1.94 - 2.27	0.174
1985	1.46	1.36 - 1.56	0.168
Beta (quotient)			
1982	1.05	1.01 - 1.08	-
1983	1.33	1.29 - 1.37	-
1984	2.11	1.93 - 2.27	-
1985	1.46	1.36 - 1.58	-
Gamma			
1982	1.19	1.13 - 1.26	0.204
1983	3.17	3.00 - 3.35	0.049
1984	8.38	8.01 - 8.78	0.155
1985	5.87	5.57 - 6.21	0.065

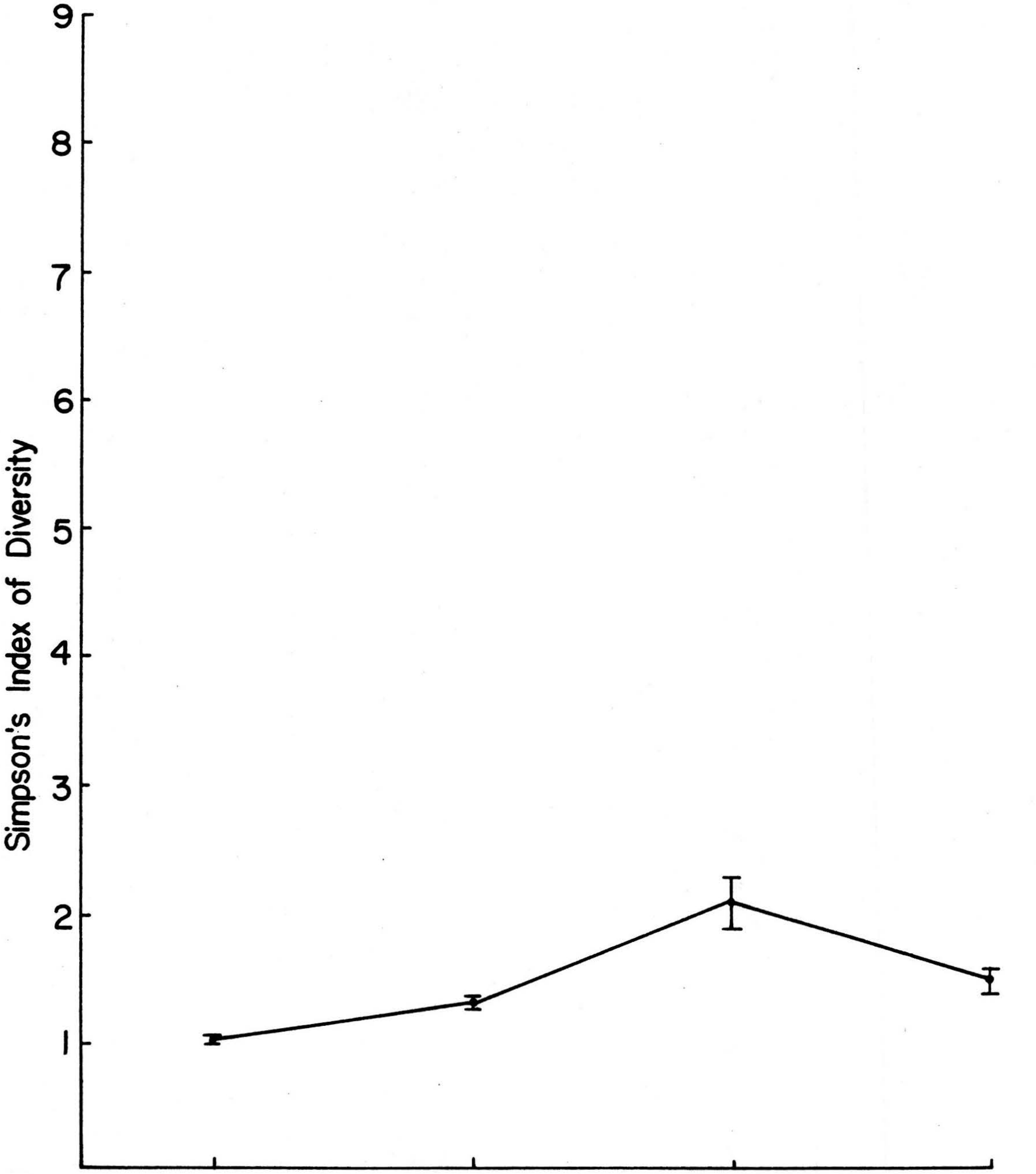
calculated as the quotient of the jackknife gamma over the jackknife alpha. Figure 1 presents the trend of alpha diversity over time. Alpha diversity increased significantly ($p < 0.001$) over the period 1982-1984, and then stabilized in 1984-1985. Figure 2 presents the trend of beta diversity over time; beta diversity also increased significantly ($p < 0.001$) over the period 1982-1984, and then significantly decreased ($p < 0.001$) in 1984-1985. The beta diversity estimates based on the quotient of the jackknife gamma over alpha, and the accompanying confidence intervals based on the variances and covariance of these same quantities were quite similar to their jackknife counterparts, suggesting that the latter were indeed distributed independently of alpha and gamma.

Gamma diversity, the total landscape diversity of 12UC, increased significantly ($p < 0.001$) over the years 1982-1984, and then significantly declined ($p < 0.001$) in 1984-1985 (Figure 3). The pattern of species richness, however, increased steadily over the period of the study, with 27, 40, 51 and 53 species on 12UC for 1982, 1983, 1984 and 1985, respectively. Eveness of the vegetation on 12UC increased over the first three years of the study (0.04, 0.08, and 0.16, respectively), but decreased in 1985 (0.11).

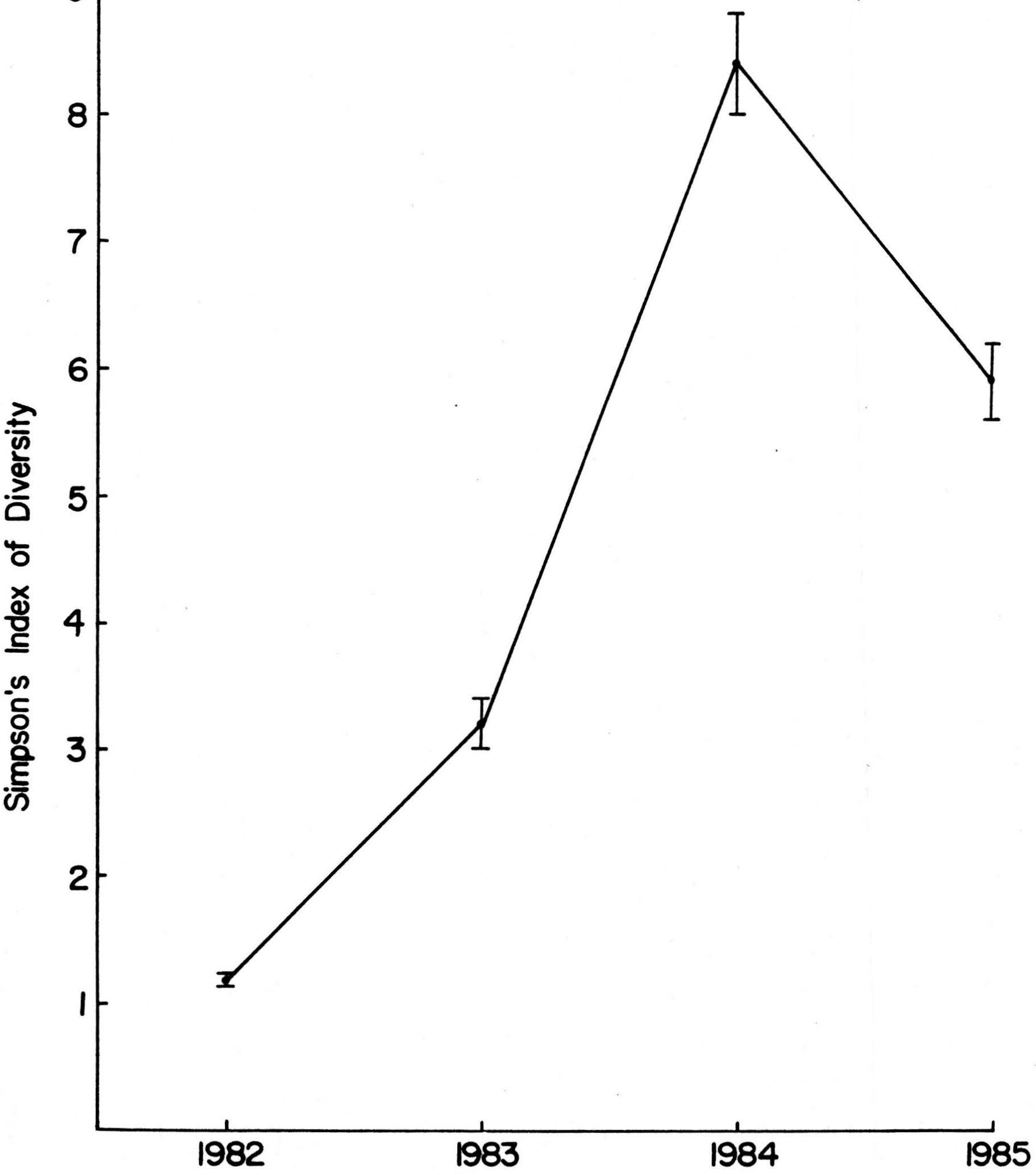
The jackknife diversity estimates of the planting density treatments appear in Table 3. The pattern in diversity over time for the low and high density treatments is presented in Figure 4. In no year was there a significant difference ($p > 0.1$) in diversity between these two extreme planting density treatments (H3.2).



1. Jackknife estimate of alpha diversity (Simpson's index) of 12UC over the period 1982-1985. Bars represent 95 percent confidence intervals.



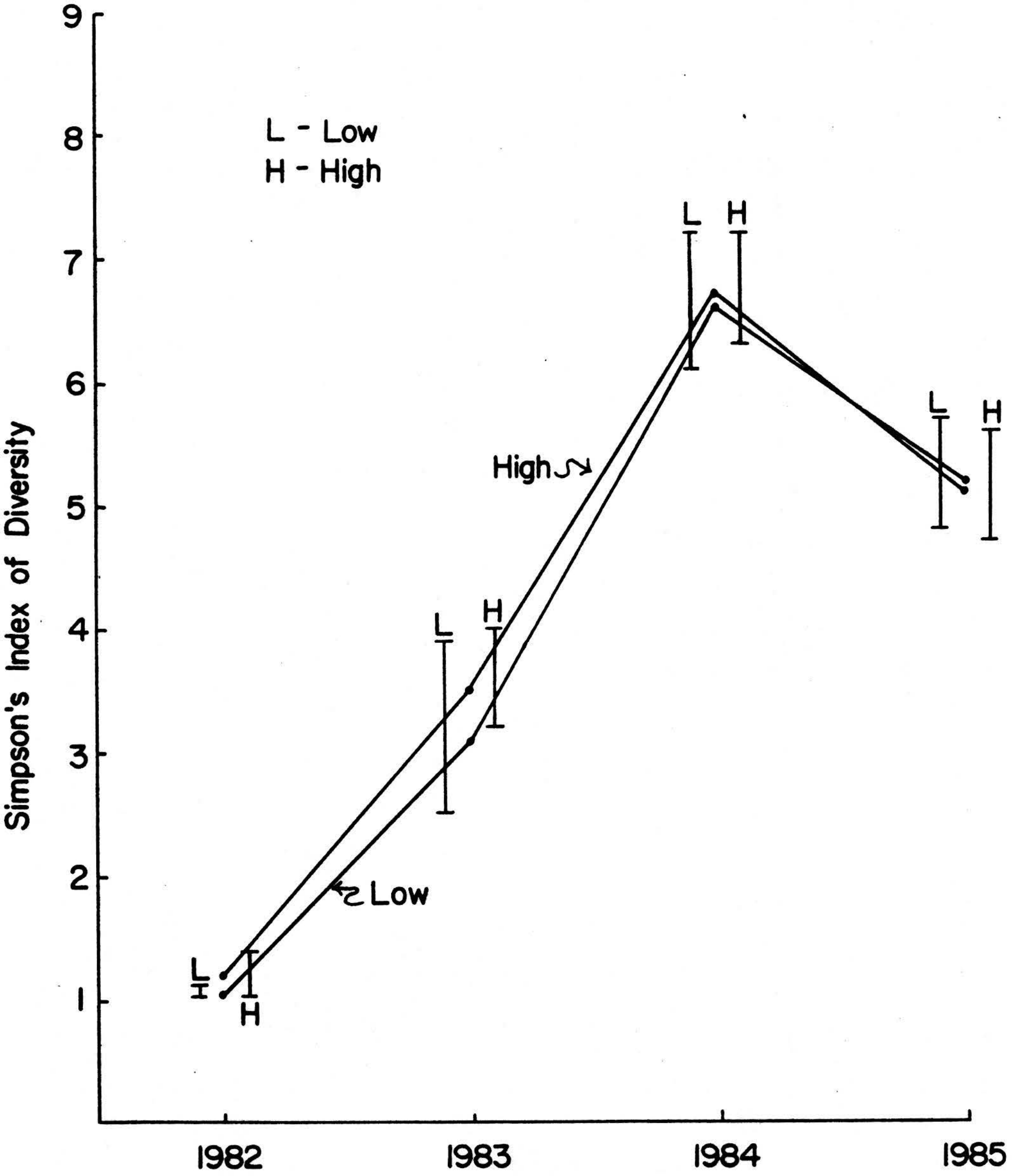
2. Jackknife estimate of beta diversity (Simpson's index) of 12UC over the period 1982-1985. Bars represent 95 percent confidence intervals.



3. Jackknife estimate of gamma diversity (Simpson's index) of 12UC over the period 1982-1985. Bars represent 95 percent confidence intervals.

Table 3. Diversity of shrub planting density treatments on 12UC for the years 1982-1985.

	Mean	95 percent confidence interval	Kolmogorov-Smirnov statistic
1982			
Low density	1.08	1.05 - 1.12	0.234
Normal density	1.11	1.05 - 1.18	0.209
Medium density	1.37	1.22 - 1.54	0.219
High density	1.20	1.04 - 1.40	0.294
1983			
Low density	3.07	2.54 - 3.86	0.158
Normal density	2.44	2.03 - 3.07	0.101
Medium density	2.86	2.65 - 3.11	0.164
High density	3.51	3.16 - 3.95	0.142
1984			
Low density	6.61	6.11 - 7.19	0.104
Normal density	6.64	6.31 - 7.01	0.178
Medium density	4.82	4.03 - 6.00	0.192
High density	6.69	6.25 - 7.20	0.103
1985			
Low density	5.19	4.80 - 5.66	0.108
Normal density	4.92	4.00 - 6.39	0.181
Medium density	6.52	5.75 - 7.54	0.154
High density	5.10	4.69 - 5.58	0.154



4. Jackknife estimates of Simpson's index of diversity for low and high density planting treatments over the period 1982-1985. Bars represent 95 percent confidence intervals.

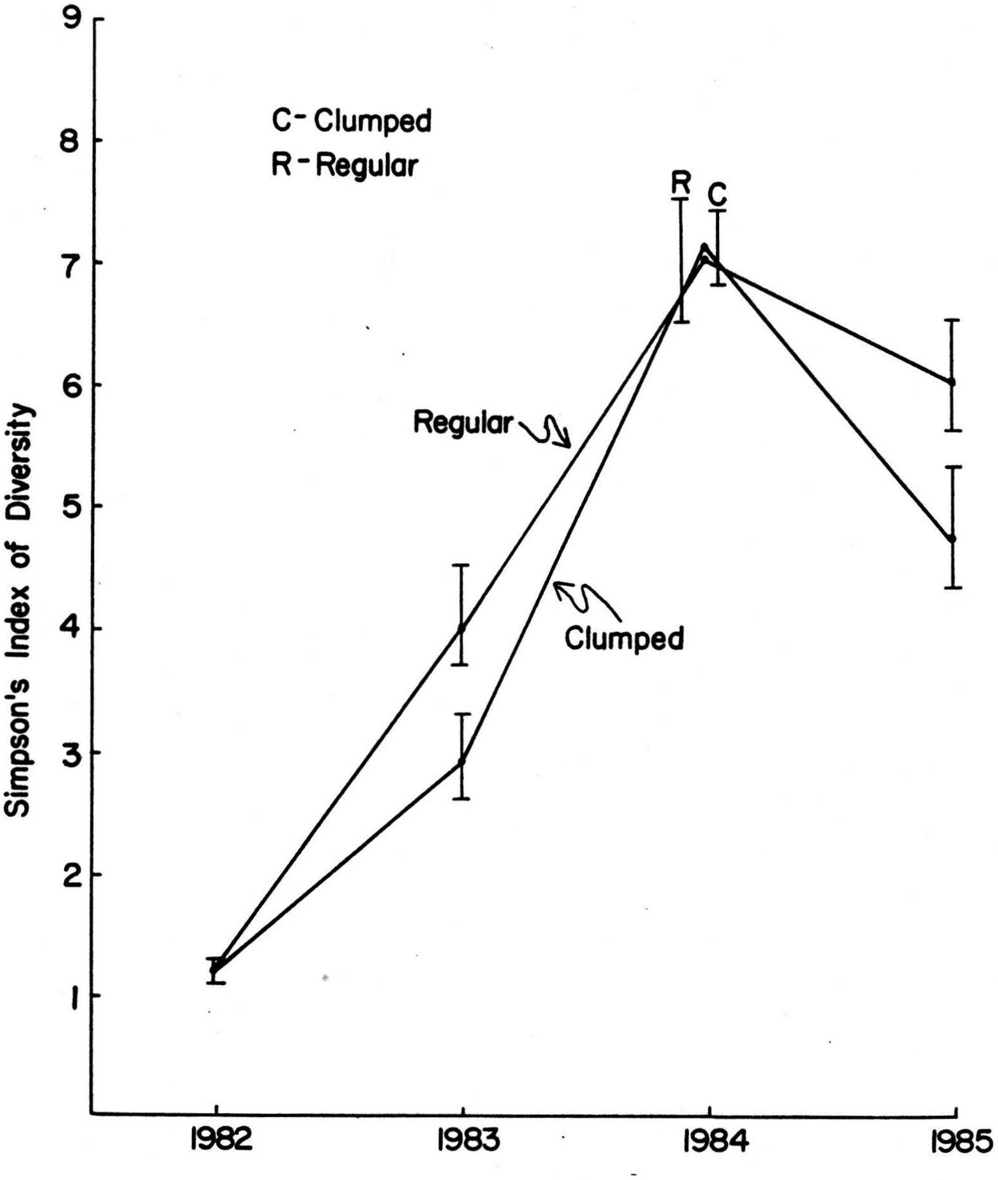
The jackknife diversity estimates for the two planting pattern treatments are presented in Table 4, and are plotted over time in Figure 5. Initially (1982), there was no significant difference ($p > 0.5$) in diversity, though in 1983 and 1985 the regular planting pattern was significantly more diverse ($p < 0.005$) than the clumped pattern (H3.4). The variances in 1984 were too great to show any statistically significant difference.

The jackknife diversity estimates for the planted and unplanted topsoiling treatments appear in Table 5. The diversity of the unplanted stored versus nonstored topsoil treatments appears in Figure 6. There was no initial difference in diversity ($p > 0.2$), though in 1983 and 1984 the nonstored topsoil treatment was more diverse ($p < 0.001$, H3.3). In 1985, however, there was no longer any significant difference ($p > 0.05$) between these treatments. The planted stored versus nonstored toposoil treatments is plotted over time in Figure 7. For the first two years of the study, the planted nonstored topsoil treatment was more diverse ($p < 0.05$), but by 1984 and 1985 there was no longer any significant difference ($p > 0.1$) between the two treatments (H3.5).

The jackknife diversity estimates for the four topographic positions for the years 1982-1985 appear in Table 6. There were no significant differences ($p > 0.05$) between topographic positions in any year (H3.1).

Table 4. Diversity of shrub planting pattern treatments on 12UC for the years 1982-1985.

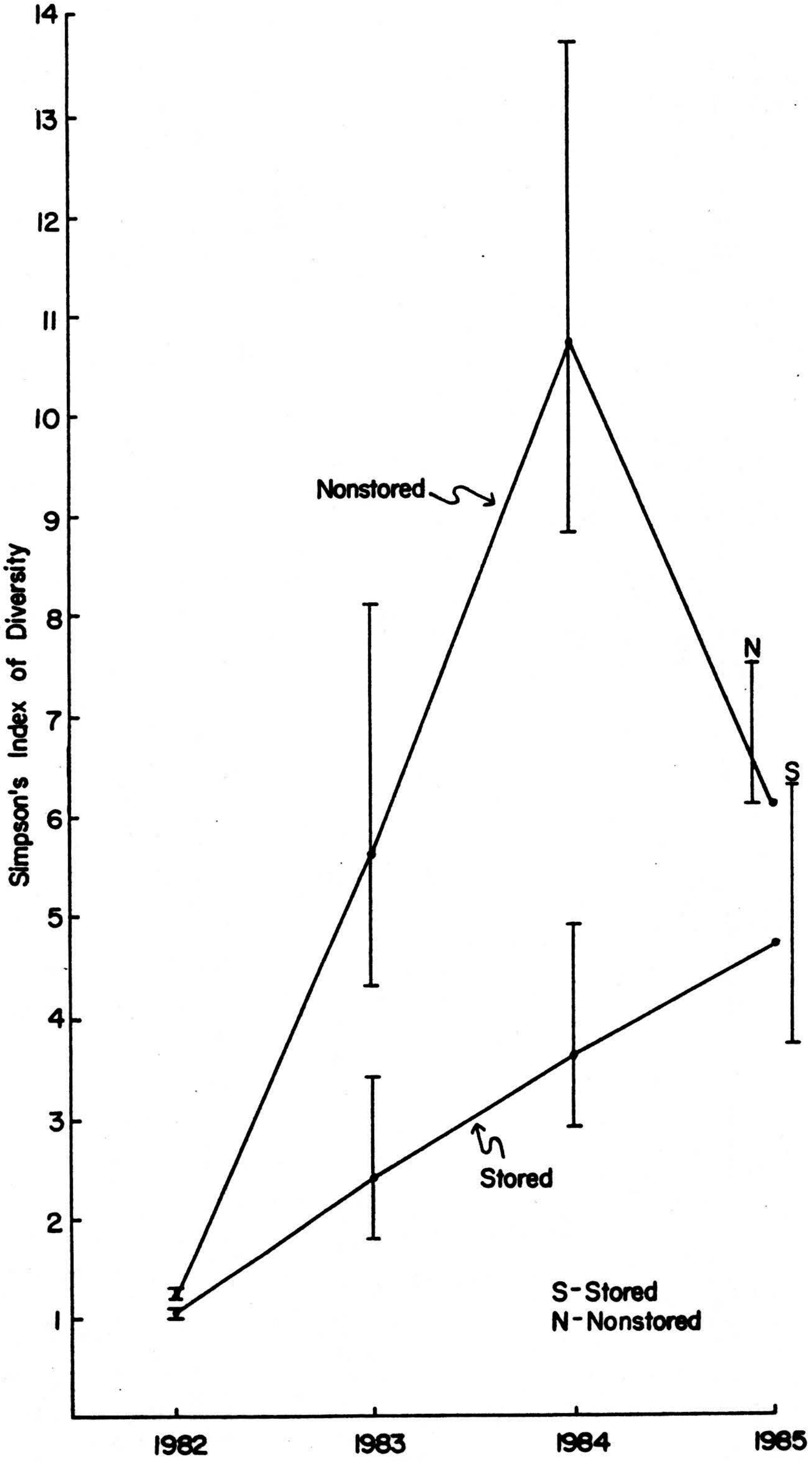
	Mean	95 percent confidence interval	Kolmogorov-Smirnov statistic
1982			
Clumped	1.16	1.08 - 1.25	0.275
Regular	1.19	1.13 - 1.25	0.135
1983			
Clumped	2.89	2.59 - 3.28	0.090
Regular	4.04	3.66 - 4.50	0.188
1984			
Clumped	7.06	6.77 - 7.38	0.146
Regular	6.97	6.53 - 7.47	0.145
1985			
Clumped	4.72	4.30 - 5.25	0.116
Regular	6.03	5.59 - 6.54	0.143



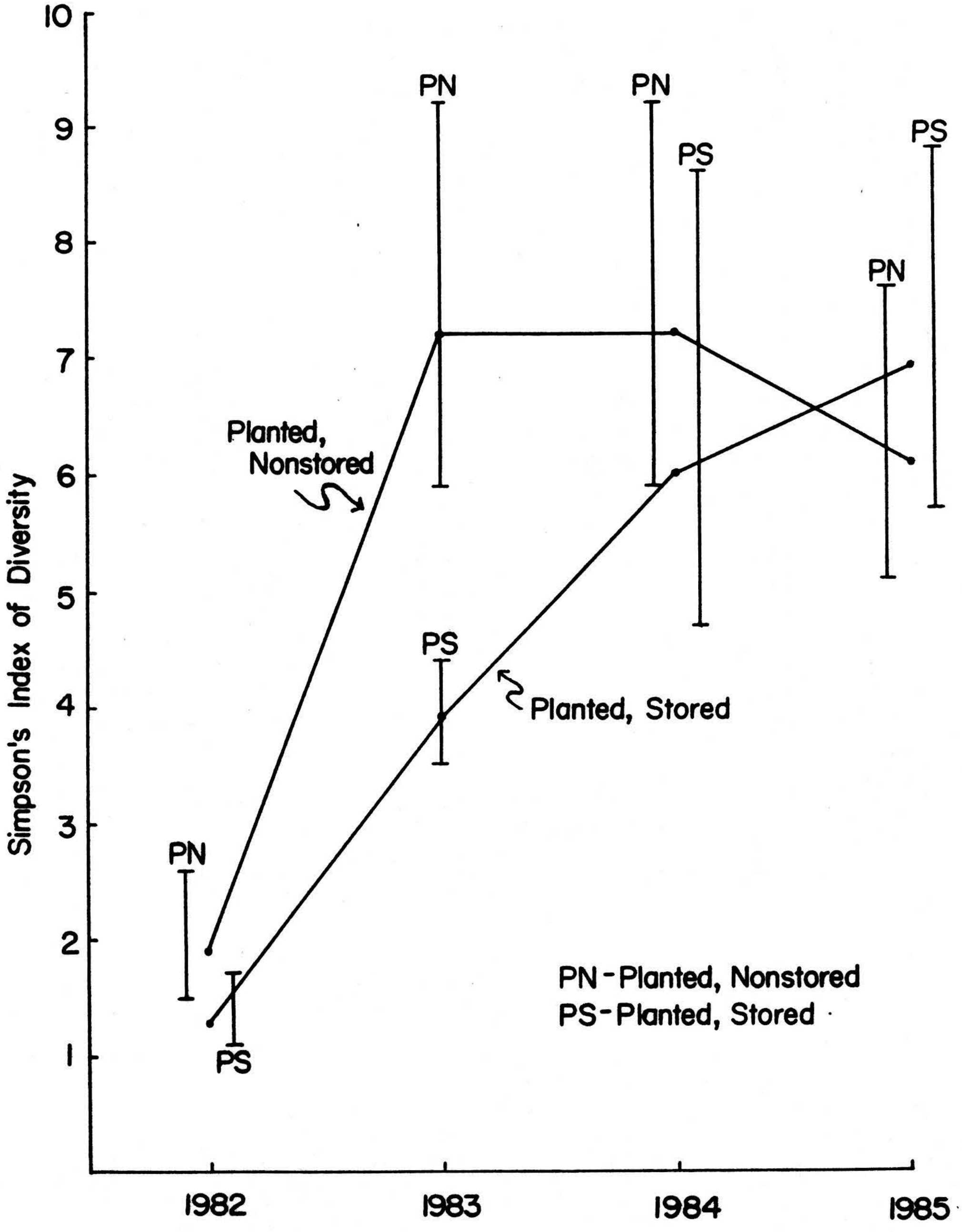
5. Jackknife estimates of Simpson's index of diversity for clumped and regular planting treatments over the period 1982-1985. Bars represent 95 percent confidence intervals.

Table 5. Diversity of topsoil and planted topsoil treatments on 12UC for the years 1982-1985.

	Mean	95 percent confidence interval	Kolmogorov-Smirnov statistic
Stored topsoil			
1982	1.10	0.98 - 1.24	0.366
1983	2.39	1.83 - 3.43	0.178
1984	3.59	2.85 - 4.85	0.307
1985	4.68	3.71 - 6.33	0.174
Nonstored topsoil			
1982	1.19	1.07 - 1.35	0.221
1983	5.59	4.27 - 8.09	0.195
1984	10.71	8.80 - 13.69	0.226
1985	6.09	5.11 - 7.56	0.198
Medium regular stored topsoil			
1982	1.35	1.12 - 1.69	0.210
1983	3.88	3.45 - 4.42	0.161
1984	6.04	4.66 - 8.58	0.200
1985	6.91	5.71 - 8.76	0.186
Medium regular nonstored topsoil			
1982	1.92	1.52 - 2.62	0.263
1983	7.19	5.88 - 9.24	0.177
1984	7.19	5.88 - 9.25	0.176
1985	6.11	5.11 - 7.56	0.198



6. Jackknife estimates of Simpson's index of diversity for unplanted stored and nonstored topsoil treatments over the period 1982-1985. Bars represent 95 percent confidence intervals.



7. Jackknife estimates of Simpson's index of diversity for planted stored and nonstored topsoil treatments over the period 1982-1985. Bars represent 95 percent confidence intervals.

Table 6. Diversity of four topographic positions on 12UC for the years 1982-1985.

	Mean	95 percent confidence interval	Kolmogorov-Smirnov statistic
Position 1 (ridge)			
1982	1.22	1.12 - 1.33	0.147
1983	3.98	3.40 - 4.79	0.102
1984	7.46	6.83 - 8.22	0.169
1985	5.24	4.47 - 6.32	0.098
Position 2			
1982	1.23	1.11 - 1.38	0.214
1983	3.54	3.08 - 4.16	0.086
1984	8.37	7.46 - 9.52	0.136
1985	5.89	5.27 - 6.66	0.052
Position 3			
1982	1.29	1.14 - 1.48	0.161
1983	2.85	2.38 - 3.57	0.085
1984	9.18	8.30 - 10.3	0.152
1985	6.84	5.97 - 8.00	0.107
Position 4 (toe)			
1982	1.15	1.07 - 1.23	0.249
1983	2.34	1.93 - 2.98	0.115
1984	7.49	6.04 - 9.86	0.094
1985	6.90	5.96 - 8.18	0.120

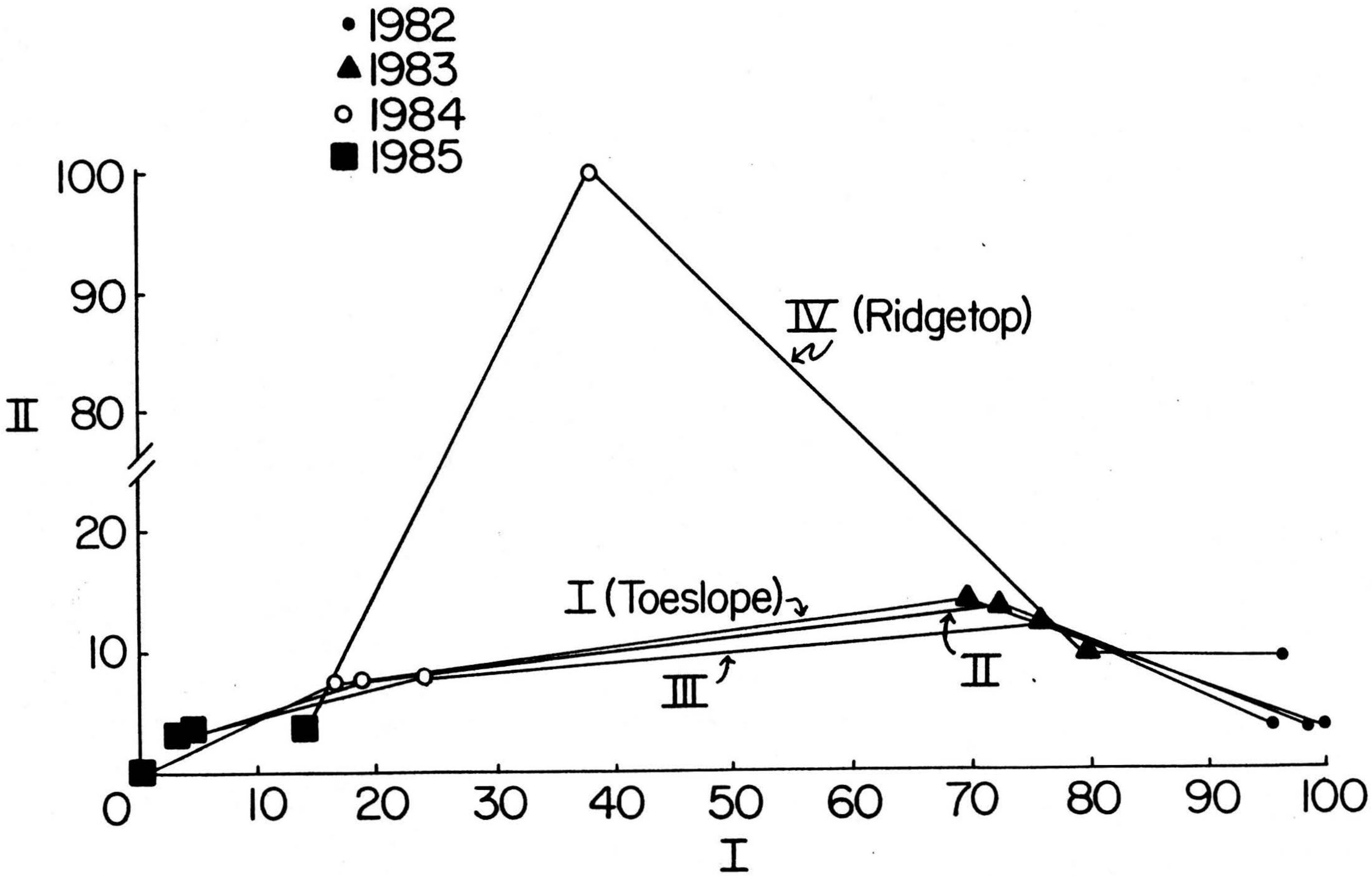
Successional trajectories

The reciprocal averaging ordination of vegetation samples classified by topographic position appears in Figure 8. The first two eigenvectors (axes I and II) explained 50.0 and 22.5 percent of the variation in the data, respectively. With the exception of the ridgetop slope quarter in 1984, all four topographic positions track quite similarly, and appear to have parallel, even convergent, trajectories (H4.1).

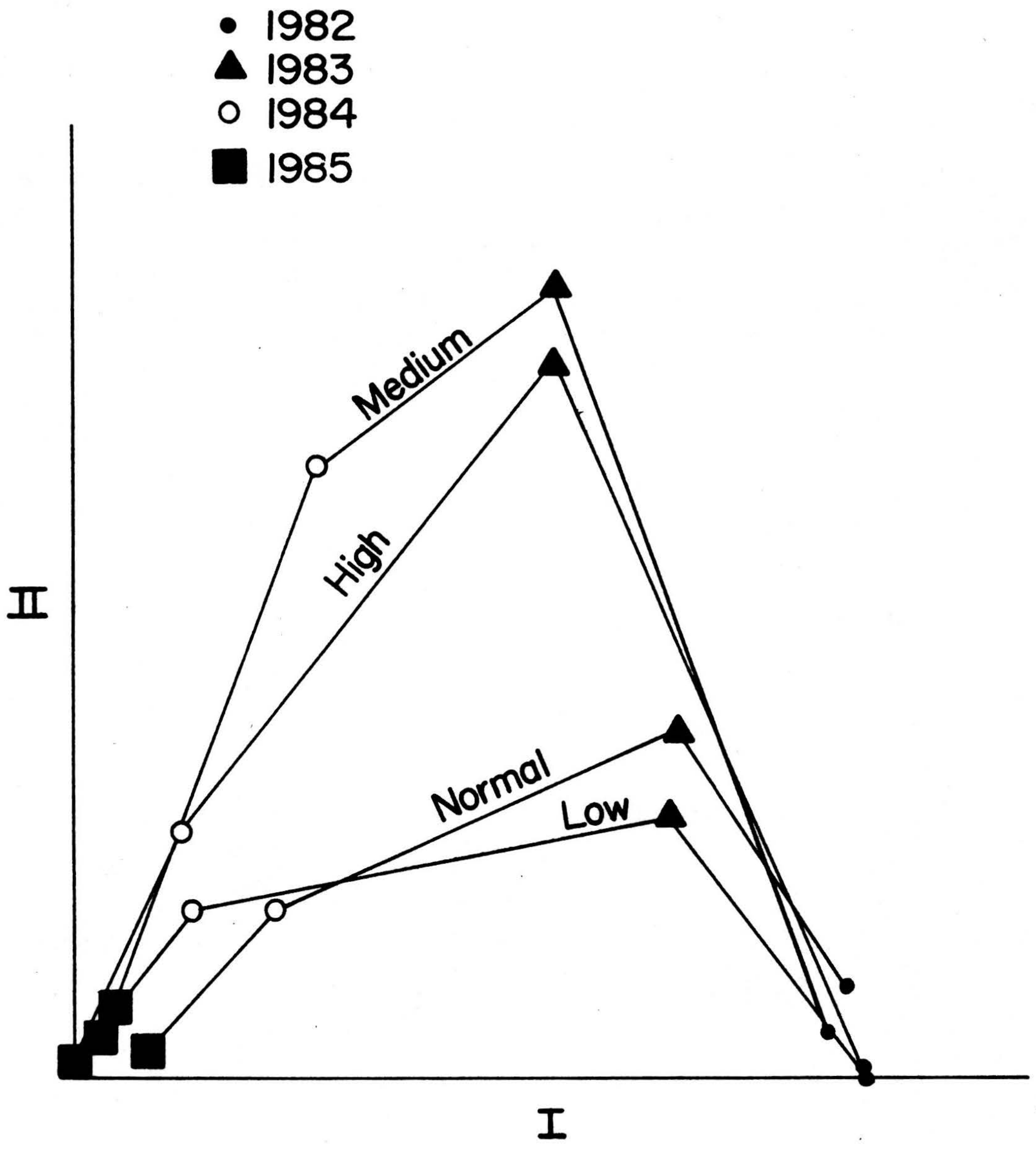
The ordination of samples classified by planting densities appears in Figure 9. All four treatments began in a very small region of the species-space, and while there was some divergence in 1983 and 1984, by 1985 all four treatments converged to essentially the same location (H4.2). The two principal eigenvectors accounted for 55.3 and 21.4 percent of the variation in the data, respectively.

The successional trajectories of the clumped and regular planting patterns appear in Figure 10. The first two eigenvectors accounted for 66.1 and 19.3 percent of the variation in the data. Again, the treatments ordinated quite closely in 1982, began to diverge in 1983, and then converged to very similar compositions by 1985 (H4.3).

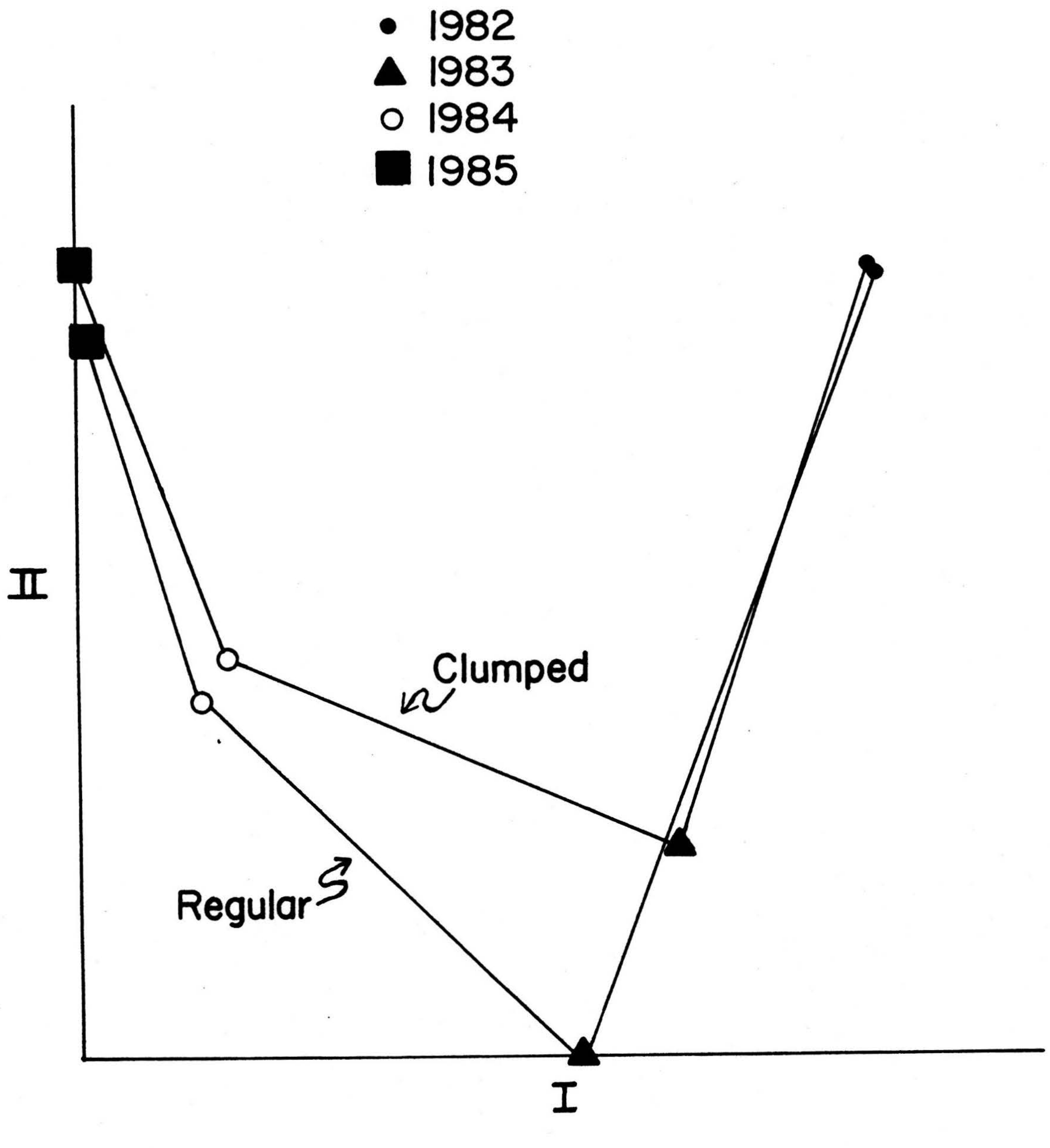
The ordination of the topsoiling treatments shows the greatest degree of divergence; while all four combinations of planted versus unplanted, stored versus nonstored topsoil treatments ordinated quite closely in 1982, by 1984 there was a considerable spread among the locations of the four treatments. In 1984, the stored topsoil treatments were located more closely to one another than they were to the nonstored topsoil treatments, and vice-versa, though the planted



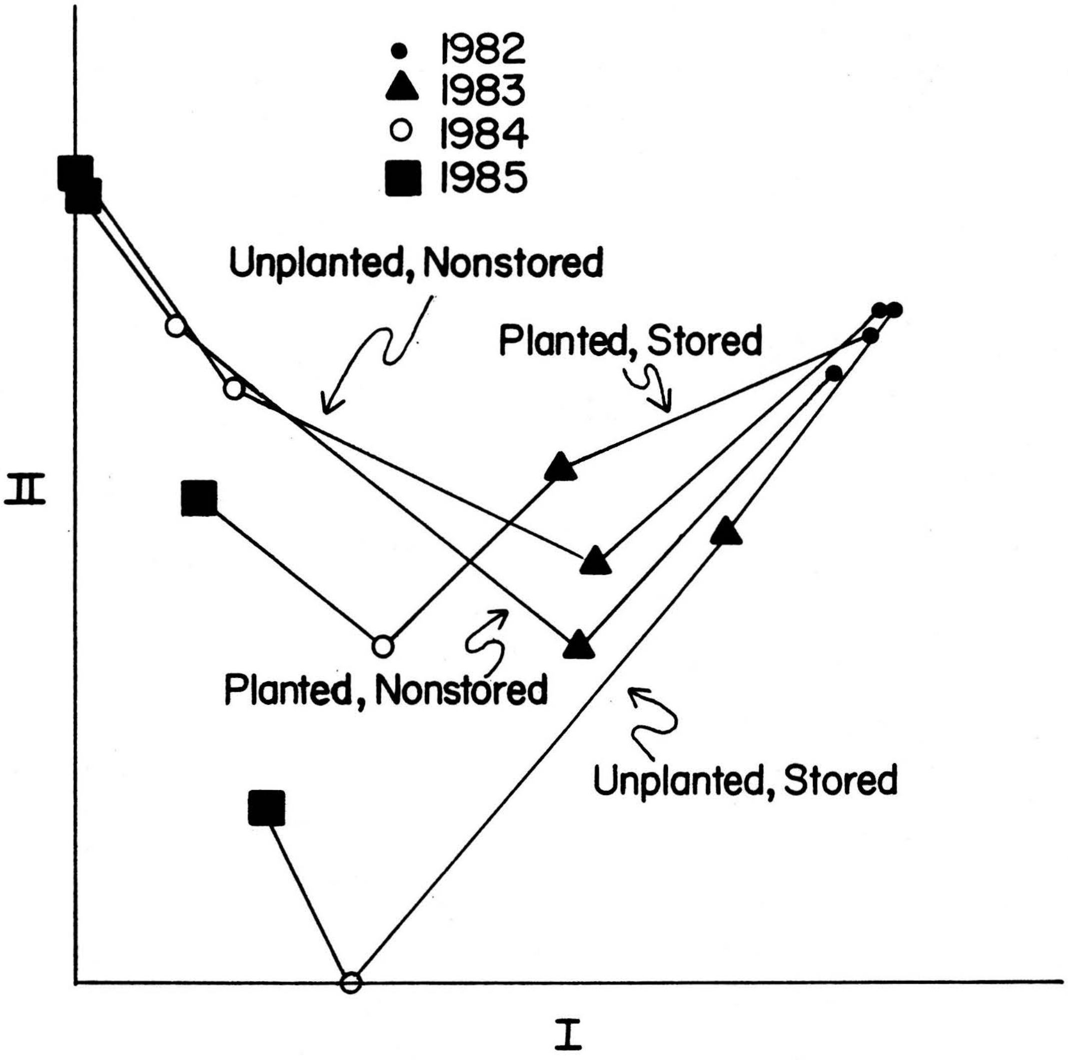
8. Successional trajectories of vegetation classified by four topographic positions over the period 1982-1985. Axes are first two eigenvectors of reciprocal averaging ordination.



9. Successional trajectories of the vegetation of planting density treatments over the period 1982-1985. Axes are first two eigenvectors of reciprocal averaging ordination.



10. Successional trajectories of the vegetation of planting pattern treatments over the period 1982-1985. Axes are first two eigenvectors of reciprocal averaging ordination.

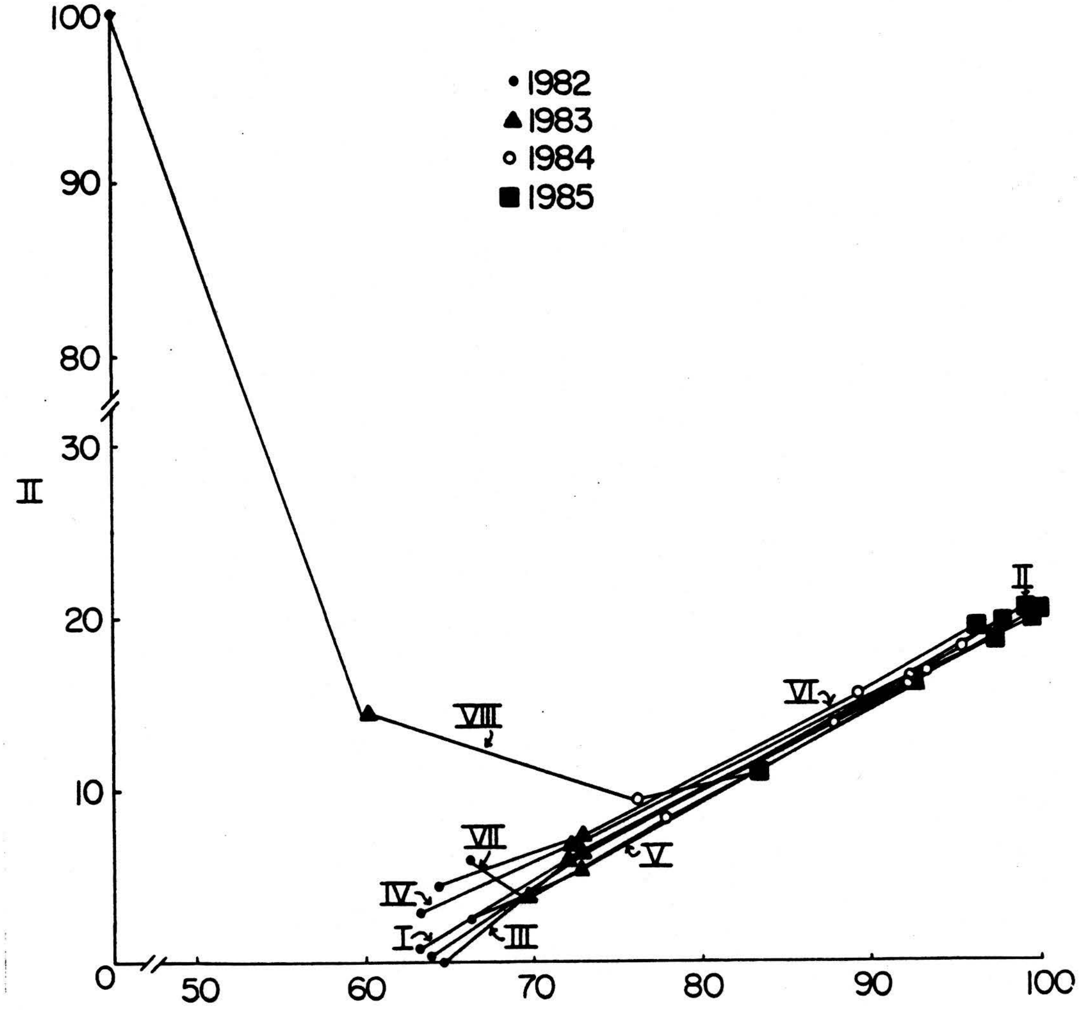


11. Successional trajectories of the vegetation of planted and unplanted, stored and nonstored topsoil treatments over the period 1982-1985. Axes are first two eigenvectors of reciprocal averaging ordination.

stored topsoil treatment was ordinated more closely to the nonstored treatments than was the unplanted stored topsoil treatment. By 1985, however, all four treatments appeared to be converging, with the nonstored topsoil treatments ordinated closely, and the stored topsoil treatments tracking toward this same point, though somewhat retarded in time (H4.4). The first two eigenvectors accounted for 46.5 and 18.9 percent of the variation in the data, respectively.

The successional trajectories of the communities identified in 1982 appear in Figure 12. Seven of the eight classes ordinate quite closely in all years, and apparently produced similar plant communities at a rather uniform rate. Class VIII however, ordinated in an extremely different position in 1982, but thereafter quickly converged toward the other classes, such that by 1985 it occupied approximately the same position in species-space as did the other seven classes. Class VIII was almost exclusively composed of management treatment plots.

Species composition for 12UC in 1982 was differentiated from other years by dominance of the annuals *Salsola iberica*, *Chenopodium album*, *Arenaria congesta* and *Monolepis nuttaliana*. The vegetation of 12UC in 1983 was differentiated by the annuals *Halogeton glomeratus*, *Lepidium perfoliatum*, *Lappula redowskii*, *Polygonum aviculare* and *Salsola iberica*, and the perennials *Agropyron dasystachum* and *Phlox multiflora*. The vegetation of 12UC was differentiated in 1984 by the annuals *Tragopogon dubius* and *Epilobium paniculatum*, and the perennials *Poa scabrella*, *P. fendleriana*, *Sitanion hystrix*, *Aster glaucoides*, *Machaeranthera canescens* and *Sphaeralcea coccinea*. The



12. Successional trajectories of the vegetation classes indentified in 1982 over the period 1982-1985. Axes are first two eigenvectors of reciprocal averaging ordination.

vegetation in 1985 was differentiated by the perennials *Hordeum jubatum*, *Bromus inermis*, *Oryzopsis hymenoides*, *Poa sandbergii*, *P. fendleriana*, *P. scabrella*, *Sitanion hystrix*, *Agropyron trachycaulum*, *A. dasystachium*, *Chrysothamnus viscidiflorus*, *Hedysarum boreale*, *Oxytropis sericea*, *Sidalcea oregana*, *Aster glaucoides* and *Machaeranthera canescens*.

The mean percent cover of *Artemisia tridentata* in the unplanted interspace (volunteer cover) over time appears in Table 7. There were no significant differences in cover among planting density or pattern treatments ($p > 0.05$). Management and unplanted nonstored topsoil treatments lacked any volunteer sagebrush cover, though the unplanted stored topsoil treatment recruited some volunteer plants.

The mean percent cover of volunteer *Chrysothamnus viscidiflorus* over time appears in Table 8. There were no significant differences in cover among planting density or pattern treatments ($p > 0.05$). All unplanted treatments had no volunteer shrubs in interpace areas.

The mean percent cover of volunteer *Agropyron smithii* over time is presented in Table 9. There were no significant differences in cover among planting density or pattern treatments ($p > 0.05$). All treatments, including unplanted ones, recruited *Agropyron smithii* in the interspace areas.

Table 7. Mean percent cover of <u>Artemisia tridentata</u> on interspace plots.

	1982 mean 95% C.I.	1983 mean 95% C.I.	1984 mean 95% C.I.	1985 mean 95% C.I.
Low density	0 -	0.09 0.02-0.16	0.28 0.003-0.56	0.50 0.19-0.81
Normal density	0 -	0.17 0.06-0.28	0.23 0.06-0.39	0.48 0.23-0.72
Medium density	0 -	0.05 0.004-0.10	0.14 0.02-0.29	0.52 0.22-0.81
High density	0 -	0.28 0.07-0.43	0.49 0.22-0.78	0.61 0.29-0.93
Clumped pattern	0 -	0.08 0.04-0.42	0.25 0.14-0.35	0.47 0.28-0.65
Regular planting	0 -	0.20 0.11-0.12	0.32 0.14-0.50	0.59 0.38-0.79
Stored topsoil	0 -	0 -	0.02 0.01-0.05	0.17 0.01-0.53
Nonstored topsoil	0 -	0 -	0 -	0 -
Planted, nonstored topsoil	0 -	0.21 0.02-0.40	0.57 0.03-1.10	0.64 0.14-1.15
Management	0 -	0 -	0 -	0 -

Table 8. Mean percent cover of Chrysothamnus viscidiflorus on interspace plots.

	1982 mean 95% C.I.	1983 mean 95% C.I.	1984 mean 95% C.I.	1985 mean 95% C.I.
Low density	0	0	0.004	0.29
	-	-	0.0004-0.01	0.009-0.07
Normal density	0	0.13	0.03	0.48
	-	0.002-0.03	0.0006-0.05	0.23-0.72
Medium density	0	0.02	0.08	0.50
	-	0.003-0.04	0.02-0.13	0.20-0.79
High density	0	0.06	0.09	0.33
	-	0.008-0.43	0.03-0.15	0.05-0.63
Clumped planting	0	0.01	0	0.36
	-	0.0008-0.02	-	0.19-0.53
Regular planting	0	0.04	0.06	0.31
	-	0.003-0.07	0.03-0.10	0.14-0.47
Stored topsoil	0	0	0	0
	-	-	-	-
Nonstored topsoil	0	0	0	0
	-	-	-	-
Management	0	0	0	0
	-	-	-	-
Planted, nonstored topsoil	0	0.05	0.18	0.16
	-	0.07-0.09	0.03-0.34	0.03-0.29

Table 9. Mean percent cover of Agropyron smithii on interspace plots.

	1982 mean 95% C.I.	1983 mean 95% C.I.	1984 mean 95% C.I.	1985 mean 95% C.I.
Low density	0.03	0.64	2.84	7.72
	0.003-0.05	0.26-1.02	1.57-4.11	5.19-10.3
Normal density	0.01	0.58	2.77	7.58
	0.002-0.03	0.34-0.81	1.49-4.05	4.39-10.7
Medium density	0.03	0.86	1.55	4.21
	0.03-0.09	0.22-1.51	0.94-2.17	2.52-5.90
High density	0	0.49	2.57	5.58
	-	0.29-0.68	1.41-3.74	3.90-7.26
Clumped planting	0	0.57	2.61	7.29
	-	0.36-0.78	1.87-3.35	5.50-9.09
Regular planting	0	0.71	2.26	5.25
	-	0.38-1.03	1.49-3.03	3.87-6.62
Stored topsoil	0.03	0.23	1.48	3.73
	0.02-0.08	0.05-0.40	0.01-3.47	1.18-6.29
Nonstored topsoil	0.01	0.38	2.15	3.42
	0.05-0.25	0.01-0.92	0.53-3.77	0.16-6.68
Management	0	0.37	0.73	2.33
	-	0.22-0.51	0.04-1.41	0.02-4.65
Planted, nonstored topsoil	0.09	0.33	3.96	5.54
	0.01-0.19	0.22-0.45	2.55-5.37	2.66-8.42

DISCUSSION

The investigations into the associations between the vegetation pattern and the pattern of site factors and cultural practices demonstrate that any one of these factors, at best, only weakly differentiated the landscape pattern of vegetation. In general, the ability of the various factors to differentially direct plant community succession declined over time.

The initial topsoil pattern was significantly associated (though weakly) with the landscape vegetation pattern only in the first two years of the study. Thereafter, the vegetation pattern was distributed randomly with respect to the initial topsoil pattern. There are two alternative, though not mutually exclusive, hypotheses to account for this observation: (1) the spatial variability at the scale of investigation, and the range of the soil factors used to discriminate the initial soil pattern, did not represent gradients steep enough to express differences in species' tolerances, or (2) the soil landscape pattern changed significantly over time. The homogeneity of mined land soils is well documented (Schafer, 1979; Fisher & Deutsch, 1983). It is likely that the topsoil on 12UC was too uniform, and the levels of plant growth factors too favorable, to differentiate the landscape vegetation pattern (Allen & MacMahon, 1985). There is, unfortunately, no information available on the changes in the soil landscape pattern to test the second hypothesis.

Topsoiling treatments showed a tendency to converge successionally; the ordination of these four treatments demonstrates this convergence. By 1984, there was no longer a significant difference in diversity between the planted stored and nonstored topsoil treatments, and by 1985 there was no difference in diversity between unplanted stored and nonstored topsoil treatments. This lack of a persistent difference in diversity and successional divergence contradicts the widely-held dictum that directly applied (nonstored) topsoil is a better plant growth medium, and capable of producing greater native plant diversity, than is stockpiled (stored) topsoil (Hargis & Redente, 1984). Greater diversity, and accelerated succession, is proposed to be a result of the greater presence of viable propagules of later successional species, which often decline as a result of soil stockpiling (Howard & Samuel, 1979; Hinchman & Taylor 1981). However, Allen & MacMahon (1985) demonstrated that the stored, respread topsoil on 12UC was chemically and physically similar to the soil on an undisturbed site; the only exceptions were bicarbonate-extractable phosphorous, which was actually higher in the overburden underlaying the respread topsoil than in the undisturbed site, and total nitrogen, which remained at levels 80 percent those of the undisturbed site. Further, Johnson (1984) demonstrated that the number of viable seeds and species of seeds in a stored topsoil pile at the Kemmerer mine showed no decrease with increasing length of storage time. Thus, the two principal mechanisms proposed to accelerate succession and create species diversity on nonstored versus stored topsoil apparently did not exist between the topsoiling treatments on 12UC.

Allen & MacMahon (1985) concluded that the major differences between the undisturbed soil and the stored, respread topsoil on the

Kemmerer mine were biological in nature, including lower vesicular-arbuscular (VA) mycorrhizal hyphal length on the stored, respread topsoil. However, the soil levels of VA mycorrhizae, of demonstrated importance in disturbed land succession (Miller, 1979; Reeves et al., 1979; Allen & Allen, 1980), declined sharply in 1983-1984 (Allen et al., in prep.), and the levels of microorganisms in the nonstored topsoil were reduced to the level in the stored topsoil treatments. Thus, the only significantly different parameter between the topsoiling treatments was largely equalized, and therefore successional convergence is not too surprising.

The association between the landscape vegetation pattern and all cultural treatments was significant in all years, but the strength of this association declined over time. Exclusion of the management treatment plots from consideration did not change this general pattern, though it lowered the strength of association with the vegetation pattern. The amount by which the strength of association was lowered by removal of the management treatment plots also declined over time, suggesting that the vegetation on the management treatment plots was more different from other planting treatments in 1982 than in 1985. Management treatment plots were initiated with quite a different species composition, including a cover crop of the annual Hordeum vulgare. Nevertheless, these plots successionally converged with other treatments. This convergence is dramatically demonstrated in Figure 12; class VIII, consisting almost exclusively of management treatment samples, initially ordinates in an extreme position relative to the other vegetation, but by 1985 has compositionally converged with the rest of the vegetation on 12UC. This would seem to be in agreement with Allen (1983), who noted that the early abundance of annuals does not have a significant effect on the patterns of mined land succession.

If the remaining planting treatments are classified by either planting density or planting pattern, it is clear that the planting treatment effect described above is more attributable to planting density than pattern. The former is significantly associated with the landscape vegetation pattern in all four years; planting pattern is only significant in 1983. Both planting density and pattern are less significant and less strongly associated with the landscape vegetation pattern in 1985 than in 1982. This is consistent with the observed successional convergence among density and pattern treatments. These results support the observations by Depuit (1980) and Doerr & Redente (1983) that the effects of cultural practices do not persist, and agree closely with the three year length of time estimated as the rate of convergence for differing cultural practices by Depuit (1980).

In no year did high planting density produce a more diverse community than the low density planting treatment. In 1983 and 1985, the regular planting pattern was more diverse than the clumped pattern. This is contrary to Turner's (1983) suggestion that clumping may facilitate the accumulation of propagules. Definitive conclusions concerning the effects of planting pattern should be avoided, however, for treatments were placed on the lee side of the two ridges which form the study area. Neuber (1984) demonstrated that the effect of the ridges on the distribution of snow on the lee sides of 12UC greatly overshadows any possible local effects produced by shrub clumps.

Hatton et al. (1986) demonstrated that the coefficient of variation associated with the ocular estimation of percent cover rises geometrically at the lower limit of actual cover. Because the cover values for volunteer planted species were usually approaching this lower limit, observer error may be quite significant. This error, however, was not accounted for in the cover estimates presented in this paper. Thus, these results are most reliably interpreted on a presence-absence basis.

The degree of recruitment of planted species in the interspaces, as expressed by percent cover, was not significantly affected by planting density or pattern. However, unplanted treatments generally did not recruit the planted shrub species. These results lead to two conclusions: (1) a very local seed source, on the scale of only a few meters, seems necessary and sufficient for significant recruitment of *Artemisia tridentata* and *Chrysothamnus viscidiflorus* in the first four years following disturbance, and (2) planting these shrubs at densities as low as one-fifth that of undisturbed levels develops volunteer shrub cover as well as planting at 1.5 times the undisturbed density.

Agropyron smithii, however, was a much more aggressive colonizer of interspace areas, though the degree to which it spread was again independent of planting density or pattern. This species was apparently much more vagile than the two shrub species; by 1985, unplanted treatments had similar volunteer cover as the planted treatments. It is not known whether this colonization was principally due to vegetative or sexual reproduction.

There is, of course, some risk in making conclusions concerning the effects of varying shrub planting density and pattern treatments on succession and diversity after only four years of observation. However, the strong convergence, coupled with the widespread abundance of volunteer shrubs across 12UC, suggests not only that there is no persistent treatment effect on succession, but also that the planting treatments themselves do not persist.

The landscape pattern was significantly associated with topographic position in 1983 to 1985; this is the only factor tested which showed a stable degree of association with the landscape vegetation pattern over time. This agrees with Jonescu's (1979) observation that plant communities developing on ridges were different from those developing on interridge positions, though Jonescu (1979) found significantly higher diversity on the ridge positions, and this study found no significant differences among topographic position diversity in any year. The continuing importance of topography in differentiating the landscape pattern in vegetation generally follows the model proposed by Doerr & Redente (1983), in which environmental factors dominate the pattern of community development after the effects of cultural treatments wane.

The ordination of vegetation classified by topographic position, however, shows a strong degree of convergence, with the exception of the behavior of the ridgetop position in 1984. The extreme behavior of the ridge position in 1984 may be due to the fact that this slope position was largely spared the negative effects of snow mold and the crash of VA mycorrhizal populations associated with the wet winter of 1983-1984 (Allen et al. , in prep.). Nevertheless, by 1985 all topographic positions had successionally converged, though the ridgetop position remained least closely ordinated with the other three positions.

The pattern of the beta and gamma diversity estimates for the vegetation on 12UC reflect the general convergence noted above. Though the within-community diversity, alpha, was apparently stable over the last two years of the study, beta diversity, and consequently gamma diversity, declined. This suggests that the plant communities initially developing on the site are becoming compositionally less distinct. This pattern follows the models of Harthill & McKell (1979) and Wali (1980). Because species richness increased over the length of the study, this decline in diversity can only be interpreted as a response to progressively greater dominance by a subset of the species present on 12UC (less eveness), and Odum's (1969) observation that species richness generally increases as a community matures cannot be rejected.

The changes in the vascular plant composition of 12UC with time reflect the general pattern of replacement of annuals by perennial grasses, forbs and shrubs identified by Parton & Woodmansee (1980), Wali (1980) and Allen & Knight (1984), and most closely follows Connell & Slatyer's (1977) second model of succession.

CONCLUSIONS

There is a strong pattern of successional convergence among vegetation communities identified by cluster analysis or classified by membership to the initial topsoil pattern, topographic position, or various cultural treatments. This is reflected in a decrease in the intercommunity (beta) and landscape (gamma) diversity over the last year of the study, and in the lack of any significant differences in diversity among topographic positions, planting density extremes, or topsoiling treatments in 1985. The only difference in diversity observed by 1985 was the higher diversity on the regular planting treatments than on the clumped planting treatments. The lack of a persistent treatment effect on plant community differentiation due to planting pattern or topsoiling treatments may be explained in part by the absence of the mechanisms supposed to produce differences in diversity and successional trajectory.

This successional convergence is further reflected in the declining strength of predictive association, and significance, between the landscape vegetation pattern and the initial topsoil pattern and cultural practices over the study period. The only factor tested which appeared to be either increasing or stable in strength of association and significance with respect to the landscape vegetation pattern was topographic position.

Though the alpha diversity of the study area increased over the first three years of the study, it remained relatively stable in the final year despite an increase in species richness, indicating that the eveness component of diversity is decreasing.

Therefore, despite topographic differences and the spatial variability of the initial topsoil treatment, and despite rather extreme differences in cultural treatments, after four years of succession the vegetation on 12UC converged to a relatively homogeneous species composition principally characterized by perennial vegetation.

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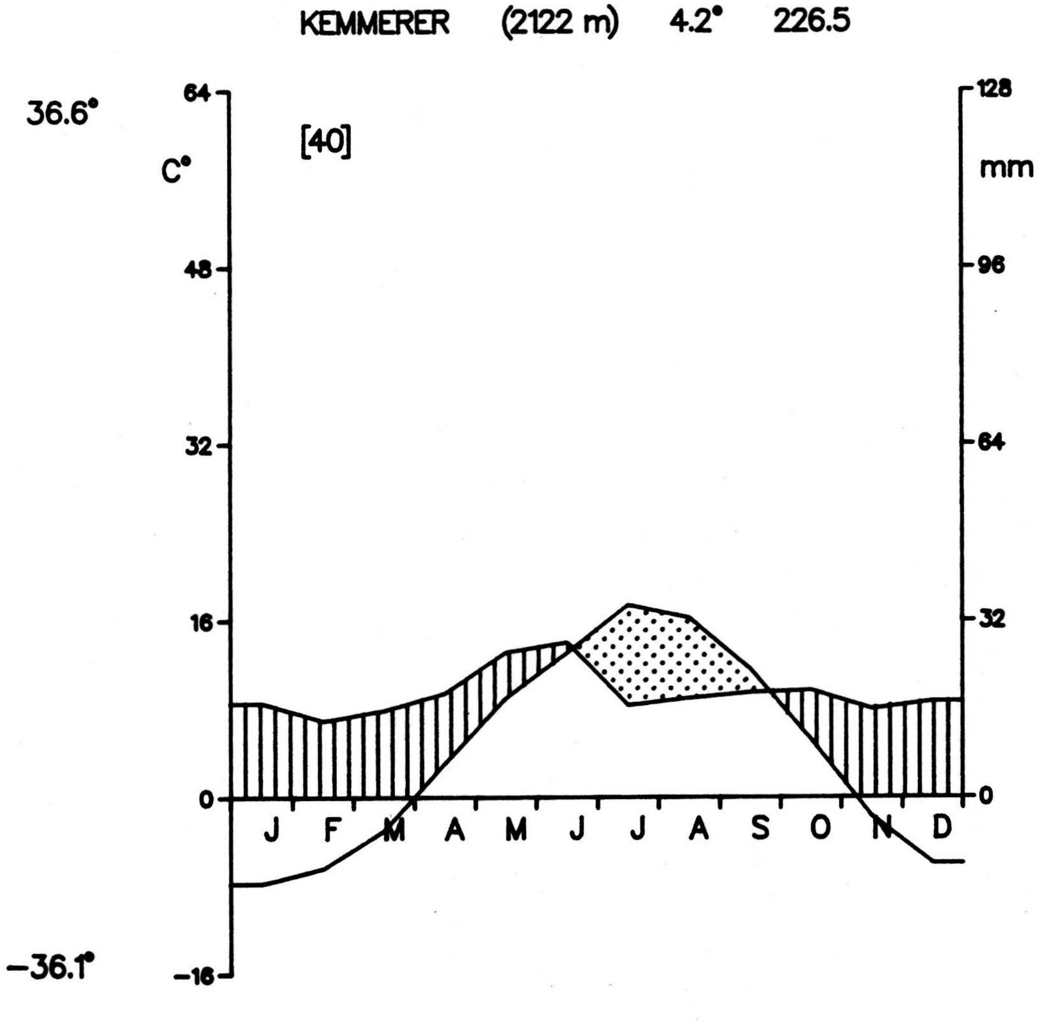
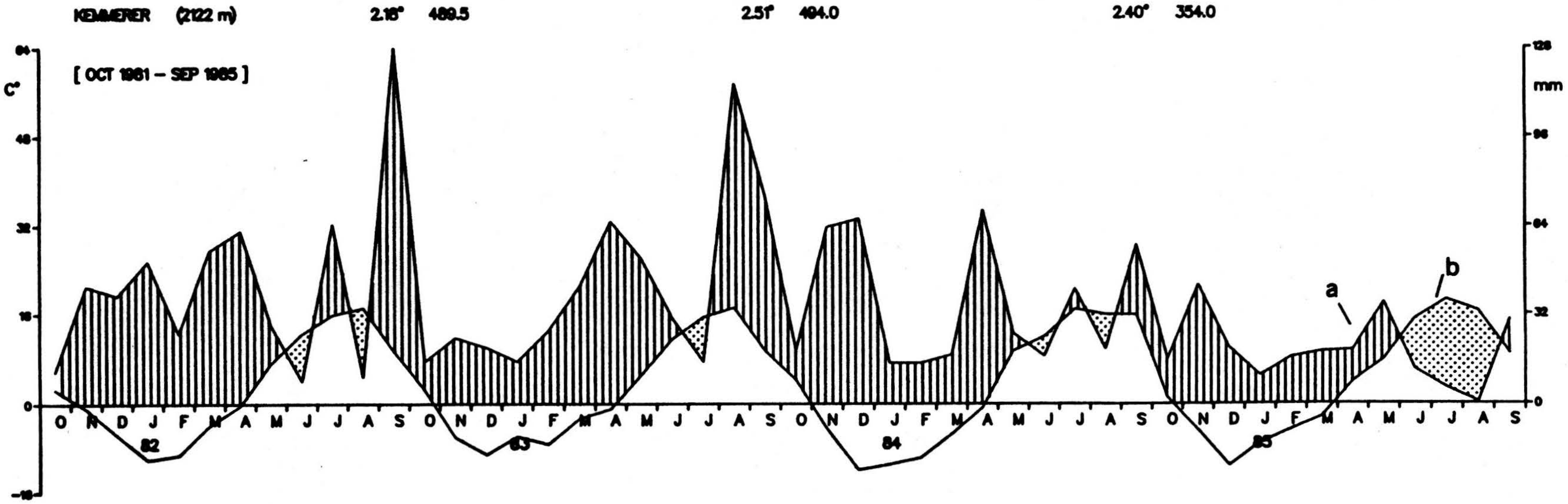
APPENDICES

Appendix 1. Climatic diagrams



13. Mean monthly temperature and precipitation at the National Weather Service Station, Kemmerer, Wyoming for the years 1951-1980.



14. Monthly temperature and precipitation for the study area, 12UC, for October, 1982 through September, 1985.

Appendix 2. Contingency tables

Table 10. Contingency table of 1982 vegetation classes including all treatments on 12UC versus topographic position.

	Topographic position				
Class	1	2	3	4	Row Sum
1	4	6	6	7	23
2	17	15	6	5	43
3	5	4	6	5	20
4	7	7	5	6	25
5	3	6	7	6	22
6	10	9	12	12	43
7	2	0	5	6	13
8	2	3	3	3	11
Column sum	50	50	50	50	200

Table 11. Contingency table of 1982 vegetation classes of successional augmentation and management treatments versus treatment membership.

	Classes					
Treatments	1	2	3	4	5	Row sum
ART	5	1	2	0	0	8
CHR	3	5	4	0	0	12
ATR	0	4	0	0	0	4
LDR	8	0	4	0	0	12
LDC	7	4	1	0	0	12
NDR	2	6	4	0	0	12
NDC	6	2	4	0	0	12
HDR	0	12	0	0	0	12
HDC	5	7	0	0	0	12
MDR	3	6	0	3	0	12
MDC	0	5	2	5	0	12
MGT	0	0	0	2	10	12
Column sum	39	40	21	22	10	132

Table 12. Contingency table of 1982 vegetation classes for successional augmentation treatments only versus treatment membership.

	Classes				
Treatments	1	2	3	4	Row sum
ART	5	1	2	0	8
CHR	3	5	4	0	12
ATR	0	4	0	0	4
LDR	8	0	4	0	12
LDC	7	4	1	0	12
NDR	2	6	4	0	12
NDC	5	3	4	0	12
HDR	0	0	0	12	12
HDC	6	6	0	0	12
MDR	3	6	0	3	12
MDC	0	5	2	5	12
Column sum	39	40	21	20	120

Table 13. Contingency table of 1982 vegetation classes versus shrub planting density treatments.

	Classes				
Density	1	2	3	4	Row sum
Low	15	4	5	0	24
Medium	3	11	2	8	24
Normal	8	8	8	0	24
High	5	7	0	12	24
Column sum	31	30	15	20	96

Table 14. Contingency table of 1982 vegetation classes versus shrub planting pattern treatment.

	Classes				
Pattern	1	2	3	4	Row sum
Regular	13	12	8	15	48
Clumped	18	18	7	5	48
Column sum	31	30	15	20	96

Table 15. Contingency table of 1982 vegetation classes for 12UC versus 1981 soil classes.

	Soil Classes							
Vegetation classes	1	2	3	4	5	6	7	Row sum
1	1	8	5	0	1	0	1	16
2	2	9	8	6	3	1	2	31
3	2	5	9	1	1	1	11	30
4	1	2	4	2	0	4	1	14
5	2	2	2	0	0	1	2	9
6	2	3	2	0	0	3	0	10
7	1	4	3	0	1	0	0	9
8	0	5	3	0	0	0	0	8
Column sum	11	38	36	9	6	10	17	127

Table 16. Contingency table of 1983 vegetation classes including all treatments on 12UC versus topographic position.

	Topographic position				
Class	1	2	3	4	Row Sum
1	6	7	9	8	30
2	5	8	10	9	32
3	5	2	4	13	24
4	5	3	0	1	9
5	4	7	1	1	13
6	16	9	7	8	40
7	2	3	6	2	13
8	4	3	3	0	10
9	0	2	4	2	8
10	3	6	6	6	21
Column sum	50	50	50	50	200

Table 17. Contingency table of 1983 vegetation classes of successional augmentation and management treatments versus treatment membership.

	Classes										
Treatments	1	2	3	4	5	6	7	8	9	10	Row sum
ART	3	4	1	0	0	0	0	0	0	0	8
CHR	1	4	4	0	3	0	0	0	0	0	12
ATR	3	0	0	1	0	0	0	0	0	0	4
LDR	0	0	1	0	1	0	5	0	2	3	12
LDC	4	6	2	0	0	0	0	0	0	0	12
NDR	2	3	0	0	2	0	3	1	1	0	12
NDC	3	2	2	0	2	0	1	0	0	2	12
HDR	0	1	0	0	0	3	3	2	1	2	12
HDC	0	3	0	0	0	3	2	2	0	2	12
MDR	0	1	0	0	2	1	5	0	3	0	12
MDC	0	1	0	0	3	3	1	4	0	0	12
MGT	0	1	2	6	0	1	1	0	0	1	12
Column sum	16	26	12	7	13	11	21	9	7	10	132

Table 18. Contingency table of 1983 vegetation classes for successional augmentation treatments only versus treatment membership.

	Classes										
Treatments	1	2	3	4	5	6	7	8	9	10	Row sum
ART	3	4	1	0	0	0	0	0	0	0	8
CHR	1	4	4	0	3	0	0	0	0	0	12
ATR	3	0	0	1	0	0	0	0	0	0	4
LDR	0	0	1	0	1	0	5	0	2	3	12
LDC	4	6	2	0	0	0	0	0	0	0	12
NDR	2	3	0	0	2	0	3	1	1	0	12
NDC	3	2	2	0	2	0	1	0	0	2	12
HDR	0	1	0	0	0	3	3	2	1	2	12
HDC	0	3	0	0	0	3	2	2	0	2	12
MDR	0	1	0	0	2	1	5	0	3	0	12
MDC	0	1	0	0	3	3	1	4	0	0	12
Column sum	16	25	10	1	13	10	20	9	7	9	120

Table 19. Contingency table of 1983 vegetation classes versus shrub planting density treatments.

	Classes									
Density	1	2	3	4	5	6	7	8	9	Row sum
Low	4	6	3	1	0	5	0	2	3	24
Medium	0	2	0	5	4	6	4	3	0	24
Normal	5	5	2	4	0	4	1	2	2	24
High	0	4	0	0	6	5	4	1	4	24
Column sum	9	17	5	10	10	20	9	7	9	96

Table 20. Contingency table of 1983 vegetation classes versus shrub planting pattern treatments.

	Classes									
Pattern	1	2	3	4	5	6	7	8	9	Row sum
Regular	2	4	1	5	4	16	3	7	6	48
Clumped	7	12	4	5	6	4	6	0	4	48
Column sum	9	16	5	10	10	20	9	7	10	96

Table 21. Contingency table of 1983 vegetation classes for 12UC versus 1981 soil classes.

	Soil classes							
Vegetation classes	1	2	3	4	5	6	7	Row sum
1	1	4	10	3	1	0	2	23
2	0	10	1	2	3	0	0	16
3	0	6	1	2	2	2	0	13
4	1	0	1	0	0	0	0	2
5	2	3	0	0	0	0	0	5
6	0	4	3	1	0	1	4	13
7	0	2	0	0	0	2	1	5
8	4	1	1	0	0	0	0	6
9	0	1	0	2	0	1	0	4
10	0	2	0	0	0	3	1	6
Column sum	10	33	17	10	6	9	8	93

Table 22. Contingency table of 1984 vegetation classes including all treatments on 12UC versus topographic position.

	Topographic position				
Classes	1	2	3	4	Row Sum
1	1	1	7	12	21
2	3	3	1	1	8
3	3	3	4	2	12
4	6	4	5	3	18
5	4	6	5	9	24
6	5	3	3	3	14
7	15	14	8	7	44
8	3	3	2	3	11
9	1	8	11	2	22
10	6	2	1	5	14
Column sum	47	47	47	47	188

Table 23. Contingency table of 1984 vegetation classes of successional augmentation and management treatments versus treatment membership.

	Classes								
Treatments	1	2	3	4	5	6	7	8	Row sum
ART	3	0	0	0	0	4	1	0	8
CHR	4	0	4	1	0	1	2	0	12
ATR	1	0	0	1	0	0	2	0	4
LDR	0	2	0	2	3	0	0	5	12
LDC	4	0	0	0	2	5	1	0	12
NDR	1	1	4	2	1	1	1	1	12
NDC	4	0	0	2	2	2	1	1	12
HDR	1	0	5	2	2	2	0	0	12
HDC	0	2	0	4	0	5	0	1	12
MDR	2	5	2	1	0	1	0	1	12
MDC	0	4	7	0	0	0	0	1	12
MGT	1	8	2	1	0	0	0	0	12
Column sum	21	22	24	16	10	21	8	10	132

Table 24. Contingency table of 1984 vegetation classes for successional augmenation treatments only versus treatment membership.

	Classes								
Treatments	1	2	3	4	5	6	7	8	Row sum
ART	3	0	0	0	0	4	1	0	8
CHR	4	0	4	1	0	1	2	0	12
ATR	1	0	0	1	0	0	2	0	4
LDR	0	2	0	2	3	0	0	5	12
LDC	4	0	0	0	2	5	1	0	12
NDR	1	1	4	2	1	1	1	1	12
NDC	4	0	0	2	2	2	1	1	12
HDR	1	0	5	2	2	2	0	0	12
HDC	0	2	0	4	0	5	0	1	12
MDR	2	5	2	1	0	1	0	1	12
MDC	0	4	7	0	0	0	0	1	12
Column sum	20	14	22	15	10	21	8	10	132

Table 25. Contingency table of 1984 vegetation classes versus shrub planting density treatments.

	Classes								
Density	1	2	3	4	5	6	7	8	Row sum
Low	4	2	0	2	5	5	1	5	24
Medium	2	9	9	1	0	1	0	2	24
Normal	5	1	4	4	3	3	2	2	24
High	1	2	5	6	2	7	0	1	24
Column sum	12	14	18	13	10	16	3	11	96

Table 26. Contingency table of 1984 vegetation classes versus shrub planting pattern treatments.

	Classes								
Pattern	1	2	3	4	5	6	7	8	Row sum
Regular	2	10	11	7	6	4	1	7	48
Clumped	8	6	7	6	4	12	2	3	48
Column sum	10	16	18	13	10	16	3	10	96

Table 27. Contingency table of 1984 vegetation classes for 12UC versus 1981 soil classes.

	Soil Classes							
Vegetation classes	1	2	3	4	5	6	7	Row sum
1	0	2	6	1	1	1	2	13
2	0	4	0	0	1	0	0	5
3	0	1	0	2	0	0	0	3
4	0	2	2	0	0	1	0	5
5	1	5	2	0	1	3	2	14
6	2	4	2	0	1	0	1	10
7	8	10	6	4	2	0	0	30
8	1	1	4	2	0	0	0	8
9	0	4	2	0	0	1	0	7
10	0	4	0	1	0	2	2	9
Column sum	11	37	24	10	6	8	7	104

Table 28. Contingency table of 1985 vegetation classes including all treatments on 12UC versus topographic position.

	Topographic position				
Classes	1	2	3	4	Row Sum
1	3	4	3	3	13
2	2	1	2	1	6
3	10	8	4	4	26
4	9	8	10	6	33
5	5	5	4	4	18
6	8	7	4	2	21
7	0	1	6	16	23
8	6	8	8	6	28
9	2	3	3	2	10
10	2	2	3	3	10
Column sum	47	47	47	47	188

Table 29. Contingency table of 1985 vegetation classes of successional augmentation and management treatments versus treatment membership.

	Classes								
Treatments	1	2	3	4	5	6	7	8	Row sum
ART	1	2	0	1	0	3	1	0	8
CHR	1	2	2	1	0	4	2	0	12
ATR	2	0	0	0	2	0	0	0	4
LDR	1	1	0	2	4	0	4	0	12
LDC	2	4	1	4	0	1	0	0	12
NDR	2	1	1	1	2	2	3	0	12
NDC	0	2	2	2	3	3	0	0	12
HDR	0	1	2	5	0	1	3	0	12
HDC	1	3	0	7	0	0	1	0	12
MDR	0	1	1	3	1	1	5	0	12
MDC	0	2	2	0	0	1	7	0	12
MGT	0	0	0	0	0	3	0	9	12
Column sum	10	19	11	26	12	19	26	9	132

Table 30. Contingency table of 1985 vegetation classes for successional augmentation treatments only versus treatment membership.

	Classes							
Treatments	1	2	3	4	5	6	7	Row sum
ART	1	2	0	1	0	3	1	8
CHR	1	2	2	1	0	4	2	12
ATR	2	0	0	0	2	0	0	4
LDR	1	1	0	2	4	0	4	12
LDC	2	4	1	4	0	1	0	12
NDR	2	1	1	1	2	2	3	12
NDC	0	2	2	2	3	3	0	12
HDR	0	1	2	5	0	1	3	12
HDC	1	3	0	7	0	0	1	12
MDR	0	1	1	3	1	1	5	12
MDC	0	2	2	0	0	1	7	12
Column sum	10	19	11	26	12	16	26	120

Table 31. Contingency table of 1985 vegetation classes versus shrub planting density treatments.

	Classes							
Density	1	2	3	4	5	6	7	Row sum
Low	3	5	1	6	4	1	4	24
Medium	0	3	3	3	1	2	12	24
Normal	2	3	3	3	5	5	3	24
High	1	4	2	12	0	1	4	24
Column sum	6	15	9	24	10	9	23	96

Table 32. Contingency table of 1985 vegetation classes versus shrub planting pattern treatments.

	Classes							
Pattern	1	2	3	4	5	6	7	Row sum
Regular	3	4	4	11	7	4	15	48
Clumped	3	11	5	13	3	5	8	48
Column sum	6	15	9	24	10	9	23	96

Table 33. Contingency table of 1985 vegetation classes for 12UC versus 1981 soil classes.

	Soil Classes							
Vegetation classes	1	2	3	4	5	6	7	Row sum
1	0	6	1	1	0	1	0	9
2	3	1	2	0	0	0	0	6
3	1	8	6	2	1	0	1	18
4	0	6	4	1	1	1	4	17
5	5	1	0	1	1	2	2	12
6	1	2	3	0	1	0	0	7
7	2	1	2	1	1	2	1	10
8	0	3	3	2	0	3	0	11
9	0	1	0	0	0	0	0	1
10	0	4	0	0	0	0	0	4
Column sum	11	33	21	8	5	9	7	95

Appendix 3. Species cover by treatment, 1982-1985

Table 34. Species codes.

Code	Species	Code	Species
1	Salsola iberica	2	Atriplex rosea
3	Polygonum aviculare	4	Bromus tectorum
5	Agropyron spicatum	6	Monolepis nuttaliana
7	Hordeum jubatum	8	Halogetum glomeratus
9	Agropyron smithii	10	Hordeum vulgare
11	Cirsium arvense	12	Chenopodium album
13	Lappula redowskii	14	Arenaria congesta
15	Collinsia sp.	16	Brachyactis sp.
17	Descurania sophia	18	Atriplex gardneri
19	Oryzopsis hymenoides	20	Lactuca serriola
21	Artemisia tridentata	22	Thlaspi arvense
24	Lithospermum ruderale	25	Chrysothamnus viscidiflorus
26	Polygonum douglasii	27	Epilobium paniculatum
28	Allium sp.	29	Poa sandbergia
30	Orthocarpus tolmei	31	Agropyron dasystachum
32	Agropyron trachycaulum	35	Hedysarum boreale
36	Plagiobothrys scouleri	38	Sisymbrium linifolium
40	Achillea miilifolium	42	Sphaeralcea munroana
43	Sysimbrium altissimum	44	Lepidium perfoliatum
45	Bromus inermis	46	Bromus japonicus
47	Hordeum brachyantherum	48	Collomia linearis
52	Sitanion hystrix	53	Taraxacum officinale
57	Poa scabrella	59	Phlox multiflora
60	Lappula myosotis	63	Lappula redowskii
64	Poa glaucifolia	69	Tragopogon dubius
71	Machaeranthera canescens	72	Sphaeralcea coccinea
75	Arenaria hookeri	76	Bromus anomalus
78	Carduus nutans	84	Melilotus officinalis
85	Arabis drummondii	89	Aster glaucoides
93	Poa fendleriana	97	Oxytropis sericea
98	Chaenactis douglasii	102	Erigeron sp.
105	Senecio sp.	116	Silene sp.
113	Rumex salcifolius	117	Sidalcea oregana
118	Poa pratensis		

Table 35. Percent cover by species by treatment for 1982 (L=low, M = medium, N = normal, H = high, C = clumped, R = regular, UN = unplanted nonstored topsoil, US = unplanted stored topsoil, PN = planted nonstored topsoil, PS = planted stored topsoil, MGT = management; T = < 0.1 percent cover).

	Density				Pattern		Topsoiling				
Code	L	M	N	H	C	R	UN	US	PN	PS	MGT
1	31.8	20.3	26.4	28.6	28.6	24.9	23.5	23.0	16.3	22.1	1.7
2	0.9	1.9	1.3	T	0.8	1.3	0.7	0.5	2.4	1.9	0.9
3	T	1.4	0.2	0.8	0.7	0.6	0.8	T	3.6	1.4	T
4	T		T	T		T		T	T	T	
5		T	T			T					
6			T			T			T		
7	T		T	T	T	T			T		
8	T	T	T		T	T		0.3	T	T	
9	T	T	T	T	T	T	T	T	T	T	
10				0.2	T	T	T				7.3
11				T	T						
12	T	T		T	T	T		T	T	T	
13	T			T		T	T		0.4		
14	T			T		T	T				
15	T					T	T		T		
17	0.2	0.2				T		T		0.2	
18				T	T				T		
20	T		T	1.4	0.7	T			T		
21				T	T						
22	T			T		T					
24									T		
25				T							
26						T					
27						T					
28						T					
29				T	T	T	T		T		
30							T				

Table 36. Percent cover by species by treatment for 1983 (L=low, M = medium, N = normal, H = high, C = clumped, R = regular, UN = unplanted nonstored topsoil, US = unplanted stored topsoil, PN = planted nonstored topsoil, PS = planted stored topsoil, MGT = management; T = < 0.1 percent cover).

	Density				Pattern		Topsoiling				
Code	L	M	N	H	C	R	UN	US	PN	PS	MGT
1	19.9	6.8	17.3	9.0	17.0	10.3	9.0	15.5	6.3	7.9	13.0
2	2.6	0.3	0.6	0.3	1.5	0.5	0.5	0.4	1.1	0.5	8.0
3	4.3	11.5	5.8	12.5	8.6	7.6	6.8	7.7	9.3	9.1	7.0
4	2.0	1.9	0.8	1.9	1.0	2.0	0.8	1.2	T	3.1	
5	T	T	T		T	T	T	T			T
6			T	T		T	T		T		
7	0.4	0.2	0.5	0.3	0.5	0.2	1.0	T	0.2	0.3	T
8			T			T					T
9	0.6	0.9	0.6	0.5	0.6	0.7	0.4	0.2	0.3	1.4	0.4
10	T					T		T	T		3.0
12	1.3	T	T	T	T	0.8	T	T	T	T	T
13	0.2	T	T	T	T	0.2	T	T	1.4	T	T
15							0.7		T		
16		T	T	0.3	T	0.2			T	T	
17	2.4	0.7	0.8	0.9	1.3	1.1	0.3	1.1	0.2	1.0	T
18	T	T		T	T	T			T	T	
19		T		T	T				T	T	
20	2.4	0.6	2.2	3.5	2.6	2.1	2.7	0.8	1.3	1.1	0.4
21	T	T	0.2	0.3	T	0.2			0.2	T	
22	1.8	T	T	0.2	0.2	1.1	1.5	T	T	T	T
25		T	0.1	T	T	T			T	T	
26							0.4				
27		T	T	T	T	T	1.7	0.3	0.4	T	
29		T					T		T	T	
30							T		T		
31	T	T	T		T	T		T	T	0.2	
32	T	T	T	T	T	T	0.9		2.6	0.1	
35	T			T		T			T		
38	T				T						
43		T	T			T				T	
44								T			
45			T								T
46				T	T	T	T				T
52		T				T	T		0.4	T	
57							T		T		
59						T			T		
60									T		T
63					T						
64			T								
72								T			

Table 37. Percent cover by species by treatment for 1984 (L=low, M = medium, N = normal, H = high, C = clumped, R = regular, UN = unplanted nonstored topsoil, US = unplanted stored topsoil, PN = planted nonstored topsoil, PS = planted stored topsoil, MGT = management; T = < 0.1 percent cover).

	Density				Pattern		Topsoiling				
Code	L	M	N	H	C	R	UN	US	PN	PS	MGT
1	2.0	1.3	T	0.9	2.2	1.5	2.3	2.4	1.1	1.8	4.0
2	T	T	2.2	T	T	T		T	T	T	T
3	1.3	5.5	0.2	2.0	2.3	2.7	2.1	2.2	3.4	5.2	13.5
4	5.3	2.8	T	3.7	3.1	4.4	3.0	9.4	2.4	3.2	T
5	0.2	0.3		0.3	T	0.4	0.8		0.2	0.4	4.1
7	5.6	1.2	T	3.9	3.6	3.5	5.8	1.2	6.4	2.0	0.5
9	2.8	1.5	2.8	2.6	2.6	2.3	2.2	1.5	4.0	1.4	0.7
10	T			T	T	T	T		T		0.3
11							0.5		T		
12	T	T		T	T	T				T	
13	T	T	T	T	T	T	1.1		T	T	T
14			0.2								
16				T		T					
17	0.8	0.2		T	0.4	0.3	0.4	3.2	0.2	0.2	1.3
18		T			T	T				T	
19	T	T			T	T	T	T	T	T	
20	3.9	0.4			2.2	3.1	3.4	1.1	1.4	0.8	1.0
21	0.3	0.1	0.2	0.5	0.3	0.3		T	0.6	T	
22	1.0	T			0.3	0.3	T	0.9	T	T	0.4
24			T	T							
25	T	T	T	T		T			0.2	T	
26					T						
27	T	T			T	T	0.4		T	T	
28			T	T							
29		T			T	T	T				
30									7.2		
31	T	T			T	T		T		T	0.4
32	0.5	0.4			0.2	0.3	4.0	0.2		0.5	0.4
35		T	T	T	T	T					
39				T	T						
40									T		
42	T					T		T	T		
43		T									
44							T				
45											T
46						T	0.2				T
52		T				T	1.3		0.9	0.3	
53	T	T		T	T	T	T		T	T	T

Table 37 (continued).

	Density				Pattern		Topsoiling				
Code	L	M	N	H	C	R	UN	US	PN	PS	MGT
57		T				T	0.2		0.3		
60	T	T				T				T	
69	T	T			T	T	T				
71		T				T					
75				T							
76	0.2					T					
84	T					T					
89	T	T				T				T	
93						T	0.9		0.5		T
94									T		
96									T		
97		T									
102								T			

Table 38. Percent cover by species by treatment for 1985 (L=low, M = medium, N = normal, H = high, C = clumped, R = regular, UN = unplanted nonstored topsoil, US = unplanted stored topsoil, PN = planted nonstored topsoil, PS = planted stored topsoil, MGT = management; T = < 0.1 percent cover).

	Density				Pattern		Topsoiling				
Code	L	M	N	H	C	R	UN	US	PN	PS	MGT
1	0.6	T	T	0.2	0.9	0.8	0.4	2.3	T	1.1	4.9
2	T	1.3	2.5	T	T	T					0.5
3	T	0.5	0.4	0.3	0.3	0.3	T	T	T	0.4	T
4	5.4	T	T	1.6	2.3	3.9	1.7	7.6	1.4	2.2	T
7	5.2	0.2	T	5.0	4.3	3.5	7.0	4.1	4.9	4.2	1.0
8	T				T	T		T			
9	7.7	4.2	7.6	5.6	7.3	5.3	3.4	3.7	5.5	3.8	2.3
10				T	T	T				T	
11	T	T	T		T		0.5		T		
12		T		T	T	T				T	
13	T	0.4	0.4	T	T	T	T	T	0.2	T	T
14		0.9	0.4								
15								T	T		
17	0.6	0.1		T	0.5	T	0.3	3.4	T	0.2	0.3
18				T	T	T				T	
19	T			T	T	T		T	T	T	T
20	3.5			2.5	2.3	2.6	0.3	1.5	0.7	1.9	0.5
21	0.5	0.5	0.5	0.6	0.5	0.6		0.2	0.6	0.2	
22	0.4			T	T	T	T	0.3			T
24		T	T								
25	0.3	0.5	0.5	0.3	0.4	0.3			0.2	0.2	
27	T	T	T	T	T	T	T	T	T	T	0.2
28		T	T								
29				T	T	T	T		T	T	
30						T				T	
31	0.2			0.2	0.4	0.3	T	T	0.2	0.6	3.2
32	1.4			0.8	0.8	0.9	7.2	0.5	7.8	1.2	0.9
35	T	T	T	T	T	T			T	T	
40							T		T		
42	T		T		T	T		T	T	T	T
44	T		T		T						
45				T	T	T				T	T
46	T			T	T	T	T	T			T
47									T		
52	T			T	T	T	3.1	T	1.7	T	
53	T			T	T	T	T		T	T	T
57				T	T	T	0.6		0.4		
60	T					T			T		

Table 38 (continued).

	Density				Pattern		Topsoiling				
Code	L	M	N	H	C	R	UN	US	PN	PS	MGT
69	T			T		T			T		
71	T	T		T	T			T			
72				T		T					
75	T										
76	T				T	T					
88	T										
93				T	T	T	11.0		0.5		
94									T		
97				T		T					
98				T		T					
105									T		
113						T				T	
116				T		T					
117				T	T						
118											T

Appendix 4. Soil attribute values for 1981 topsoil classes

Table 39. Mean values (with standard errors) for soil attributes of 1981 topsoil, 12UC, by cluster analysis class.

	Class						
	1	2	3	4	5	6	7
Total nitrogen (% by weight)	0.16	0.12	0.13	0.14	0.09	0.09	0.10
	(0.005)	(0.005)	(0.006)	(0.008)	(0.022)	(0.006)	(0.01)
Nitrate-N (mg N/kg soil)	18.3	22.7	38.2	57.8	33.8	17.6	9.7
	(2.39)	(1.51)	(1.75)	(3.58)	(5.36)	(3.37)	(1.51)
Ammonium-N (mg N/kg soil)	15.6	12.2	14.8	16.9	14.9	11.9	11.8
	(0.42)	(0.04)	(0.07)	(0.11)	(0.31)	(0.11)	(0.04)
Bicarbonate-P (mg P/kg soil)	18.1	22.1	18.2	22.3	24.5	18.8	22.9
	(0.69)	(0.75)	(0.66)	(0.86)	(2.68)	(1.32)	(4.11)
Electrical conductivity (dS/m)	2.04	2.86	3.65	4.13	3.61	1.96	1.58
	(0.24)	(0.16)	(0.16)	(0.21)	(0.37)	(0.31)	(0.23)
pH	7.3	7.6	7.4	7.5	7.7	7.3	7.9
	(0.01)	(0.05)	(0.07)	(0.01)	(0.02)	(0.01)	(0.02)
Sand content (%)	55.1	57.5	54.3	58.3	54.1	72.7	63.3
	(0.68)	(0.85)	(0.48)	(0.76)	(1.79)	(1.40)	(1.79)
Silt content (%)	18.4	18.0	20.3	18.7	19.2	12.0	17.2
	(0.47)	(0.71)	(0.32)	(0.33)	(0.63)	(0.62)	(0.78)
Clay content (%)	26.2	22.8	26.0	23.0	28.1	15.2	19.5
	(0.77)	(0.48)	(0.30)	(0.69)	(0.24)	(0.76)	(1.12)

Appendix 5. Estimating components of plant community diversity on mined lands

ESTIMATING COMPONENTS OF PLANT COMMUNITY DIVERSITY ON MINED LANDS[1]

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Abstract.--Techniques suggested to date for estimating plant community diversity on mined lands have focused on intracommunity (alpha) diversity, largely ignoring intercommunity (beta) and landscape (gamma) diversity. These techniques have traditionally lacked any formal procedure for expressing sampling variability in the resulting diversity estimates. Jackknife methods now exist, however, which not only estimate alpha, beta and gamma diversity, but provide confidence intervals for these estimates as well. These methods are dependent upon an a priori identification of plant communities, which poses a potential problem on mined lands where the redeveloping plant communities may be so spatially interspersed as to preclude immediate identification. A method which combines non-hierarchical clustering of quadrat data with the jackknife estimates of diversity is presented as a technique for the evaluation of the three components of plant community diversity on mined lands. The feasibility of the approach is demonstrated with a sampling of native reference vegetation adjacent to a southwestern Wyoming coal strip mine.

INTRODUCTION

The evaluation of diversity is perhaps the most difficult challenge in the assessment of revegetation success on surface-mined lands. Techniques suggested to date for estimating plant community diversity have focused on intracommunity (alpha) diversity, largely ignoring intercommunity (beta) and landscape (gamma) diversity, even though many papers have identified the need to create or maintain diverse habitats on mined lands (Karr 1968, Allaire 1979, Tyus and Lockhart 1979, Steele and Grant 1982, Emrich 1983). In addition, these techniques typically lack any formal procedure for expressing sampling variability in the resulting diversity estimates, though bond release criteria should ideally be defined on a statistical basis (Chambers 1983).

A robust bias reduction technique introduced by Quenouille (1956) and termed "jackknifing" by J.W. Tukey provides a means of estimating the central tendency and variance of a parameter independently of traditional parametric theory (Efron 1982). The technique reduces bias and allows parametric-like inference even in cases where the distributional properties of the variate are unknown (Gregoire 1984).

Zahl (1977) applied jackknifing methods to obtain estimates of diversity index sampling error. Routledge (1980) examined the bias-reducing properties of the jackknife method of estimating diversity with quadrat samples, and included a procedure for the calculation of confidence intervals for the diversity estimates. Routledge (1980) demonstrated that his jackknife estimator of diversity satisfied Jones' (1974) specification that the estimator must be a function of all the sample stratum means. Routledge's (1984) equations are derived from Jones' (1974) equations for first-order estimations, but incorporate the assumption that the number of quadrats sampled in a community is very much smaller than the total number of quadrats potentially sampled.

Routledge (1984) combined the jackknifing approach with Whittaker's (1972) hierarchical concept of diversity, providing a method for obtaining confidence interval estimates of alpha, beta and gamma diversity. In addition, Routledge (1984) demonstrated that the Shannon index's extreme sensitivity to rare species abundances

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leads to substantial bias in the jackknifed estimates, while Simpson's index has negligible bias. Heltshe and Forrester (1985) also determined the jackknife estimate of Simpson's index to be unbiased using quadrat samples.

The application of Routledge's (1984) technique to mined land assessment requires some modifications, however. The technique retains Whittaker's original idea that species turnover (beta diversity) occurs along elevational gradients. While Whittaker demonstrated that this was the case for an eastern hardwood community, it is unlikely that this assumption will hold for surface-mined lands. Therefore, some other a priori identification of the pattern of species distribution is necessary to employ Routledge's methods.

If we assume that the vegetation itself is the most appropriate expression of habitat (Daubenmire 1968, Gauch 1982), then we may use quadrat information to provide estimates of the plant communities present in a reclaimed area for input into the jackknifing calculations. Redeveloping plant communities may be identified subjectively, which poses a potential problem on mined lands where communities may be so subtle or spatially interspersed as to preclude immediate identification. Cluster analysis, however, provides a more objective alternative for the identification of plant communities from quadrat data, and can provide the non-hierarchical classification of quadrats necessary for the jackknifing approach. However, in order to jackknife estimates of diversity and variance, we must assume that the number of quadrats allotted to a cluster by the classification algorithm is proportional to the total number of quadrats in the community defined by the cluster. This is not an unrealistic assumption if quadrats are random samples in the landscape.

MATERIALS AND METHODS

To demonstrate the technique, data from a native reference area near a mine in southwestern Wyoming were obtained. The data consisted of weight estimates for all plant species within 121 one meter square permanent quadrats within a fenced, undisturbed area one hectare in size. The analysis was performed on three consecutive years' data (1982-1984).

As in all attempts to estimate plant community parameters from quadrat samples, it is necessary to establish the relationship between the size and distribution of plant communities within the study area and the sampling design and intensity necessary to adequately characterize the parameter in question. In the case of diversity, quadrat size must be smaller in magnitude than the smallest community considered, otherwise characteristics of the patch in question will be washed out by information from the surrounding matrix. Further, quadrat number must be demonstrated to adequately characterize diversity within the communities potentially present in the study area, at least those communities identifiable at the scale of the investigation. The choice of this scale is necessarily a subjective one.

For example, in the hectare of native vegetation used to demonstrate this technique, vascular plant communities were apparently distributed on a scale of tens of meters. On this basis, meter square quadrats were chosen for sampling. The minimal number of these quadrats necessary to characterize a community at this scale was determined by inspection of species-area and diversity-area curves.

Data for each year were first subjected to cluster analysis to identify the pattern of plant communities. CLUSTAR (Romesburg and Marshall 1984) was selected as the clustering program. Because the input data matrix of quadrats by species contained a large number of zero values, the Bray-Curtis coefficient was used to determine similarity on the basis that this index ignores joint absences (Clifford and Stephenson 1975).

Jackknife estimators of the three components of diversity were calculated based on the estimator of Simpson's index of concentration as it appears in Routledge (1980) and additional formulae in Routledge (1984). Variance estimators of alpha, beta, and gamma diversity were based on Routledge (1984). The distribution of jackknife pseudovalues converges to normal for these diversity estimators with increasing sample size; but because the number of clusters in a data set may be small, the variance estimators may be biased. The distributions of all three components of diversity were tested for normality with the one-sample Kolmogorov-Smirnov goodness of fit statistic for the intrinsic hypothesis (Sokal and Rohlf 1981).

Although beta is defined as the quotient of gamma over alpha and so might not be independent of alpha and gamma, Routledge (1984) proved that alpha, beta, and gamma are distributed asymptotically multivariate normal. However, for small sample sizes like ours, the asymptotic joint distribution may not be achieved. As an alternative, beta was also calculated as the quotient of the jackknife gamma over the jackknife alpha, and its variance approximated as a function of the variances and covariance of the jackknife gamma and alpha diversities (Mood et al. 1974).

For the purposes of comparison, the three components of diversity were calculated according to Whittaker's (1972) original method.

RESULTS

Jackknife statistics for the three components of diversity for the three years of data appear in table 1 along with the values for these same parameters calculated according to Whittaker's methods.

Table 1--Components of species diversity for a native reference area as calculated by (A) Whittaker's original methods and (B) jackknife method, with 95% confidence intervals.

Year		A (Whittaker)	B (jackknife)	95% C.I. (jackknife)
1982				
	alpha	2.16	2.26	2.00, 2.60
	beta	1.68	1.77	1.59, 1.95
	gamma	3.63	4.10	3.46, 5.03
1983				
	alpha	2.41	2.42	2.16, 2.74
	beta	1.75	1.74	1.50, 1.99
	gamma	4.22	4.26	3.80, 4.85
1984				
	alpha	3.27	3.14	2.31, 4.91
	beta	1.42	1.57	1.30, 1.85
	gamma	4.65	4.96	3.65, 7.74

Only in 1982 did we fail to reject ($p > 0.05$) the hypothesis of normal distribution of cluster pseudovalues for all three diversity components. In 1983, normal distributions of gamma and beta were rejected, as were the alpha and gamma pseudovalues for 1984. Thus, we might suspect that some of the variance estimates are biased. Heltshe and Forrester (1985) showed that the estimates of standard error for Simpson's index for sample sizes of similar magnitude to that of this study were either unbiased or slightly underbiased. These obsevations imply that slight deviations from normality may not significantly affect the estimation of diversity sampling variance.

The jackknife estimates for beta diversity variance were in all cases smaller than the beta estimates based on the variances of alpha and gamma diversities, though the latter were quite similar and in all years fell within the jackknife 95 percent confidence intervals.

For illustration, we assumed that no variance estimate was overly biased and tested the hypothesis of equality of diversity components with Student's t at a 0.05 significance level. Each yearly alpha estimate was significantly different from every other year; alpha diversity steadily increased from 1982 to 1984. No yearly gamma estimate was shown to be significantly different from any other year. Beta diversity in 1982 was not significantly different from beta diversity in 1983, but both were significantly greater than beta diversity in 1984.

DISCUSSION

Regardless of the calculation method, the components of diversity are estimates from sample data, not the true values. As such, there is sampling variability associated with the estimates. The jackknife estimates for diversity are all quite close to the estimates derived from Whittaker's approach, the latter in all cases falling within the jackknife 95% confidence intervals. However, the latter approach does not allow between-site or between-year comparisons which incorporate sampling variability information. The value of the jackknife technique lies in its ability to quantify the variance of the diversity index estimates and so to permit construction of confidence intervals and testing of hypotheses.

It should be noted that the landscape diversity will increase with the size of the study area. To employ any estimate of diversity for the purposes of comparison, the areas sampled should be equivalent.

The jackknifing method presented in this paper may also be used to evaluate diversity of life form classes as well as species. The method is also appropriate for data based on dominance parameters other than aboveground phytomass, e.g. cover or density.

Successful reclamation restores ecosystem function as well as structure and, ideally, reclamation criteria should consider both of these aspects. Any procedure based on an index of structure, such as species diversity (or similarity), does not necessarily evaluate function. Further, any structural index will fail to adequately characterize the contribution of a given species to ecosystem function if the magnitude of that function is not proportional to the dominance parameter for that species used to evaluate structure. For example, a nitrogen-fixing forb critical to the nutrient cycling of an ecosystem may have a very low relative importance when expressed as phytomass or cover, and thus its small contribution to a diversity or similarity index may underestimate that species' functional importance to the system.

Unfortunately, we have yet to define the roles that a given species plays in ecosystem function. Function may be approximated by reorganizing the data to life forms, but most such schema are based on structural attributes, which may or may not be correlated with function. Until functional relationships on mined lands are better defined and quantified, we are limited to the consideration of structural attributes and synthetic concepts such as species (or life form) diversity. The method for the evaluation of diversity presented in this paper was formulated for the purpose of estimating on an ecologically appropriate and statistically sound basis a single, synthetic, structural criterion of revegetation success.

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