The Biodiversity Integrity Index: An Illustration Using Ants in Western Australia

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Abstract: *Although Western Australia is a relatively unpopulated region, considerable areas of native vegetation have been modified by agricultural clearing, rangeland grazing, urbanization, road construction, and mining. Ant diversity is reduced and community composition changed by each of these land uses. Road construction has the greatest long-term effect on the alpha diversity of ants, followed by agricultural clearing, mining, urbanization, and rangeland grazing. We present data on the extent of these various land uses in each major Western Australian vegetation association. Then, examples of ant diversity and community composition for each land use are coupled with geographic information system data on the extent of each land use in the various vegetation associations to calculate indices of "biodiversity integrity." The extent of biodiversity integrity in each region concurs with a subjective opinion of the condition of each unit. Agricultural clearing, followed by rangeland grazing, were found responsible for the greatest loss of ant biodiversity integrity. The findings relate to Australia in general and may serve as a framework for estimating losses of biodiversity integrity in other regions of the world in taxa other than ants.*

El Indice de la Biodiversidad Integral: Una ilustración utilizando hormigas en Australia occidental

Resumen: *Si bien Australia occidental es una región relativamente deshabitada, áreas considerables de vegetación nativa han sido modificadas por clareos agrícolas, pastoreo de ganado, desarrollo urbano, construcción de rutas y explotación de minas. Cada uno de estos usos de la tierra reduce la diversidad especifica de las hormigas y cambia la composición de la comunidad. La construcción de rutas tuvo el efecto a largo plazo más grande sobre la diversidad alfa de las hormigas, seguido por el clareo agrícola, la minería, el desarrollo urbano y el pastoreo de ganado. Presentamos los datos sobre la extensión de los distintos usos de la tierra para cada una de las principales asociaciones vegetales de Australia occidental. Luego, relacionamos los ejemplos de diversidad de hormigas y la composición de la comunidad para cada uso de la tierra con los datos del sistem a de información gráfica sobre la extensión de cada uso de la tierra en las distintas asociaciones vegetales, para calcular indices "de la biodiversidad integral." El grado de la biodiversidad integral en cada región coincide con una opinión subjetiva del estado de cada unidad. Se encontró que el clareo agrícola, seguido por el pastoreo de ganado fueron los reponsables de la mayor pérdida de la biodiversidad integral de las hormigas. Estos resultados se refieren a Australia en general y pueden servir como marco para estimar las pérdidas de la biodiversidad integral en otras regiones del mundo, en términos de otros taxones distintos de las hormigas.*

Introduction

The concept of biological diversity means different things to different groups of people. Assessment of biological diversity has tended to focus on species lists for the areas of interest (alpha diversity surveys), and considerable importance is attached to the magnitude of species richness at the site and/or to the existence of threatened species (Noss 1990). Consequently, a site or an ecosystem may become flagged as being of particular significance if it contains exceptionally high species richness

Paper submitted September 19, 1994; revised manuscript accepted March 8, 1995.

or if one or more rare, vulnerable, or endangered species exist there (World Conservation Union 1980).

These are undeniably important aspects of biological diversity, but we believe that an important feature is missed when the conservation of biological diversity is reduced to a consideration of these basic statistics. What is missing is a consideration of the integrity, or departure from the pristine state, of biological diversity across the entire landscape. Consider an intact expanse of jarrah (*Eucalyptus marginata*) forest in the southwest of Australia or of mulga (*Acacia anura*) shrubland in the north of Australia. The quality, or degree of pristineness, of these environments is reduced if the forest is punctuated with roads, orchards, mine sites, and powerline easements, or if the mulga is heavily grazed by livestock. In both cases the full complement of plant and animal species may still be present at least somewhere in these ecosystems, so conventional measures of diversity may fail to portray the obvious changes in the quality of the environment.

Although there is clearly a need to assess biological diversity in selected areas (Noss 1990), its value would be enhanced if this was combined with a consideration of landscape condition or quality. We propose a measure, referred to as biodiversity integrity (BI), that provides a statement about the degree of intactness of the original biological diversity over a particular land unit. We used ants in Western Australia as a demonstration of the use of this index because their distribution is well known and their response in terms of biological diversity to major land uses has been the subject of a range of investigations. There is no reason why this procedure should not be applied to other regions of the world and to many other plant and animal taxa. The index is the product of a measure of diversity for a particular landscape unit and the area that unit occupies. The diversity measure could be almost any one of the conventionally used diversity measures, such as species richness, Shannon's diversity index, or even the number of threatened species within a particular area. We used two complementary measures: species richness and the quotient of similarity between disturbed and control sites. The former provides an indication of the biological variety contained within the habitat, and the latter indicates the degree of change in species composition that has accompanied the alteration of the habitat.

Consider a landscape that contains four major habitat units. The units are first standardized to the same area, because the index attributes equal importance to each habitat, whatever the area it occupies. Unit 1 is in a pristine state, unit 2 contains a mine that occupies 10% of the area, and unit 3 contains a mine on 10% of the area and farmland covering 45% of the area. The remaining unit has been totally cleared for agricultural purposes. Diversity is set at 1 in the pristine state. No weighting is given to habitat units that support exceptionally low or high diversity because equal importance is assigned to the biological diversity of each unit when it is in its pristine state.

The diversity of a taxonomic group is then assessed within each habitat unit and in each type of disturbance. In the example given, diversity drops to half its original value in the post-mining situation and to a third in the farmland. The BI in unit 1 is 100×1.0, or 100, which is the maximum attainable figure. In unit 2 it is 90×1.0 (pristine) plus 10×0.5 (mine), which gives a value of 95. In unit 3 the BI in the disturbed areas is obtained by summing the indices for the pristine area (45×1.0) and the two types of disturbance (10×0.5 [mine] + 45×0.33 [farm]), which gives a value of 65. In unit 4 there are no pristine habitats, so the BI is 100×0.33, or 33. In instances where diversity increases above pristine levels due to the presence of species favored by disturbance, diversity is still considered to be 1. The rationale behind this is that those species favored by disturbance are already present somewhere in the habitat unit and also that such species tend to be of lower conservation interest than species that occur under pristine conditions. The use of the complementary measure of species similarity should provide a better reflection of how much BI has been affected in these circumstances.

The resulting BI may be used in a number of ways. It can provide numerical data that can contribute to an audit of the state of biological diversity across the entire landscape. Thus, in the hypothetical region mentioned above the BI has dropped from a maximum of 400 down to 293. It is also possible to report on the habitat units where biological diversity is most adversely affected. Thus, the BI is lowest in unit 4 (33), followed by unit 3 (65). The BI in unit 2 is only slightly lower (95) than that in the pristine unit 1. An additional use of the procedure is to provide summaries of the impact of different land uses on biological diversity across the entire landscape, or subsets thereof. This information is obtained by subtracting the BI value for a particular land use from the value that would have been obtained had species diversity or similarity been 1. Thus mining, which occupies 10% of each of two units, has caused a cumulative loss of 10 BI units ([10×1.0] − [10×0.5] for each unit). Farming, on the other hand, occupies 45 and 100% of two units and has caused a cumulative loss of 30 + 67, or 97 BI units.

Methods

The Study Area

The vegetation of Western Australia has been mapped by Beard (1990) and, in a coarse-grain sense, can be reduced to three provinces and 21 districts (Fig. 1). These phytogeographic regions, and the area they occupy, are listed in Table 1. The Darling district in the south-west

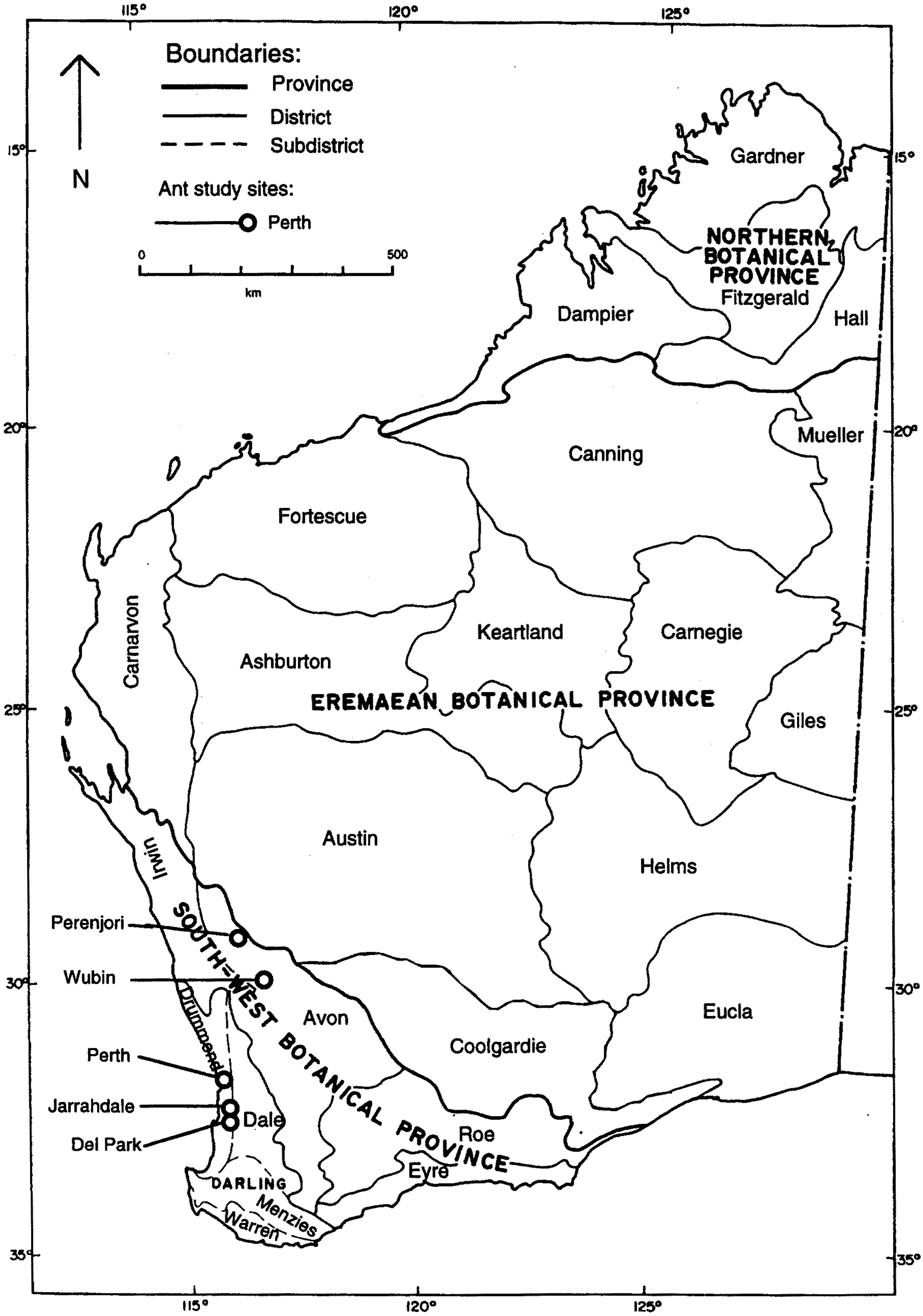

Figure 1. Phytogeographic regions of Western Australia (adapted from Beard 1990).

has been divided into four subdistricts because it is here that the greatest heterogeneity of vegetation and land use occurs. The state occupies a third of the Australian continent and supports a population of about 1.6 million; its revenue is derived mainly from farming and mining. The major land uses can broadly be described as mining (generally with subsequent rehabilitation), agriculture, grazing, urbanization, and roadways.

Information on the extent of uncleared vegetation, agricultural clearing, and rangeland grazing was obtained by superimposing digital data sets of vegetation boundaries over land use using the Intergraph® Microstation Geographic Information System Environment. The uncleared land consisted of nature reserves, national parks, and vacant government and private land. The digitized phytogeographic regions of the southwest of Western Australia were those of Beard (1990), and those for the remainder of the state were the groupings of the Western Australian Rangeland Monitoring System (WARMS; P. Thomas, personal communication). The data from the latter part of the analysis were assigned to the Beard phytogeographic region to which they most closely corresponded.

The extent of urbanization was determined from population figures from the 1991 national census (Castles 1993). We obtained statistics for the population living in each phytogeographic region. Then we obtained from maps the area occupied by a range of Western Australian townsites. The populations of these townsites were regressed against area in order to elucidate the area requirement per capita of population. The resulting area, namely 0.15 ha, was then multiplied by the population in each phytogeographic region to estimate the area that has been urbanized.

The roadway area was obtained from data on the length of roadways in each local government area (Western Australian Municipal Association 1993). These areas were assigned to the phytogeographic region in which they occurred, and the total length of road was multiplied by the mean width of road plus drainway (10 m) to give an estimate of road area. The road verges were not included because these can still support a considerable diversity of plants and animals (Keals & Majer 1991).

Mining area was obtained by counting the number of operating and closed mine sites (Department of Minerals and Energy 1993) in each phytogeographic region. It has been determined that a total of 1500 km^2 of land surface has been disturbed by mining and associated infrastructure (Chamber of Mines and Energy of Western Australia 1990), so the area disturbed in each district was obtained by multiplying this figure by the proportion of the total number of mines in each district. This provided only an approximation of the true figure because mines differ widely in size. But the effect of size differences is

Table 1. Areas of major vegetation associations in Western Australia, showing proportions of each that have been modified by five broad categories of land use.

Beard's (1990) phytogeographic region				*Percentage of total area modified*					
Province	*District or subdistrict*	*Natural region*	*Area occupied (km^2)*	*Uncleared*	*Urban*	*Roads*	*Mining*	*Agricultural clearing*	*Rangeland grazing*
Northern	Dampier	Dampierland	84,400	20.98	0.02	0.02	0.03	0.00	78.96
	Hall	East Kimberley	50,510	4.65	0.00	0.02	0.05	0.00	95.28
	Fitzgerald	Central Kimberley	83,330	45.51	0.05	0.02	0.02	0.00	54.40
	Gardner	North Kimberley	99,100	45.51	0.01	0.01	0.01	0.00	54.46
Eremaean	Eucla	Nullabor Plain	148,764	55.40	0.00	0.02	0.00	0.00	44.58
	Helms	Great Victoria Desert	209,206	76.78	0.00	0.02	0.00	0.00	23.20
	Austin	Murchison	316,239	21.51	0.01	0.04	0.14	0.00	78.31
	Carnarvon	Carnarvon	91,046	44.09	0.02	0.03	0.02	0.00	55.84
	Ashburton	Gascoyne	181,453	21.51	0.00	0.02	0.02	0.00	78.45
	Keartland	Little Sandy Desert	110,314	76.78	0.01	0.01	0.01	0.00	23.20
	Carnegie	Gibson Desert	149,784	76.78	0.00	0.01	0.00	0.00	23.21
	Giles	Central Ranges	60,788	76.78	0.00	0.01	0.01	0.00	23.21
	Fortesque	Pilbara	178,017	57.96	0.04	0.02	0.08	0.00	41.90
	Canning	Great Sandy Desert	262,275	76.78	0.00	0.01	0.01	0.00	23.20
	Mueller	Tanami Desert	63,934	76.78	0.00	0.02	0.00	0.00	23.20
	Coolgardie	South-western Interzone	126,500	72.45	0.05	0.03	0.26	0.00	27.21
South-western	Drummond	Swan Coast Plain	14,637	43.00	11.66	1.04	1.10	43.20	0.00
	Dale	Northern Jarrah Forest	19,473	50.00	0.90	0.40	0.16	48.54	0.00
	Menzies	Southern Jarrah Forest	26,572	55.00	0.33	0.29	0.07	44.31	0.00
	Warren	Karri Forest	8323	72.00	0.16	0.31	0.19	27.34	0.00
	Irwin	Northern Sandplains	39,656	30.00	0.16	0.14	0.13	69.57	0.00
	Avon	Wheatbelt	93,520	12.00	0.08	0.35	0.09	87.49	0.00
	Roe	Mallee	78,957	23.00	0.01	0.12	0.05	76.82	0.00
	Eyre	Esperance Plains	28,702	47.00	0.06	0.15	0.06	52.73	0.00

likely to be averaged out by the large number of mines in most of the phytogeographic regions.

Impacts of Land Uses

Surveys of the impact of land uses on the ant community have been carried out using reasonably similar sampling techniques at a range of sites throughout Western Australia. We selected a series of studies, representing each of the major land uses, to illustrate the application of the biodiversity integrity concept. In view of the fact that the reactions of ant communities to disturbance can vary in relation to factors such as rainfall (Majer 1990), we selected studies in which the sites were likely to represent the mid-range of responses. In the selected studies sampling was carried out using transects or clusters of pitfall traps run for seven-day periods in disturbed areas and, in order to provide a benchmark for comparison, in relatively pristine controls. In some of the studies referred to, the pitfall traps were augmented by hand collecting, vegetation sweeping, and tree beating using the sampling protocol described in Allen (1989) and Majer (1993).

Data on the impact of urbanization were obtained from Majer and Brown (1986), who surveyed the ant fauna of 33 gardens situated on the coastal plain of Perth (31°57′, 115°51′). This study did not report on ants of native vegetation, so we used the four data points obtained by Rossbach and Majer (1983) for the original vegetation of these gardens, namely tuart-jarrah-marri (*Eucalyptus gomphocephala*, *E. marginata*, *E. calophylla*) open forest and jarrah-banksia (*Banksia* spp.) tall woodland.

The mine data were obtained from Majer et al. (1984), who surveyed the ant fauna of 30 rehabilitated bauxite mines and three jarrah-marri open-forest controls situated at Jarrahdale (32°20′, 116°07′) and Del Park (32°43′, 116°04′), approximately 45 and 90 km south-southeast of Perth, respectively. A full range of rehabilitation options and ages were considered, so the data can be regarded as representing the range of responses of the biota to mining and subsequent rehabilitation.

Information on the ant fauna in areas cleared for agricultural purposes were obtained from Keals and Majer (1991), who surveyed the ants in cleared paddocks and in remnants of open or closed scrub heath situated between Wubin (36°06′, 116°38′) and Perenjori (36°26′, 116°17′), 200 km north of Perth. Three areas of native vegetation and three paddocks of lupin stubble were sampled.

Although a number of studies have been carried out in Western Australia on the responses of invertebrates to rangeland grazing, none have resulted in published lists of ants in grazed and pristine areas. Also, it is extremely difficult to find pristine areas because most of the relevant areas have been grazed by domestic stock at some time or other. Greenslade and Mott (1979) have described the species richness of ants in grazed and ungrazed paddocks of *Eucalyptus* woodland over perennial grasses at Manbulloo (14°31′, 132°12′) in the Northern Territory, and Greenslade and Smith (1994) have described ants at the generic level from lightly and heavily grazed areas of tall, open mulga (*Acacia anura*) shrubland at Lake Mere (30°11′, 145°00′) in New South Wales. These vegetation associations are similar to those that occur in Western Australia and may be considered representative of that state. In order to provide data that are numerically comparable with the other disturbances, we have resorted and identified Greenslade and Smith's material to the species level. The data represent replicated samples from both the lightly grazed (the closest we can get to a pristine area) and heavily grazed areas.

We did not sample the ant fauna of roadways because we considered it eliminated by construction of the road pavement and associated drainways.

Data Analysis

The ant data sets were first converted to presence/absence matrices, and the total number of species per site was summed. The mean value for species richness in both pristine controls (lightly grazed site in the case of the Lake Mere data) and the disturbed site was then calculated, and the disturbed-site mean was converted to a proportion of the control-site value.

The similarity between all sites within a data set was then calculated by the Systat® computer package using Sokal and Sneath's (1963) quotient of similarity ($S3$). This is calculated by the following formula: $S3 = c/(a + b)$, where a is the number of species in site A, b is the number in site B, and c is the number of species in common. The sites were then hierarchically clustered using the complete linkage procedure. In most cases the control sites formed a separate grouping from the disturbed sites. The distance along the $S3$ axis of the dendrogram was taken as a measure of the degree of similarity between the disturbed and control sites. A problem we encountered was that when large numbers of disturbed sites were compared with the three or four control sites used in these studies, the effects of disturbance were accentuated. Consequently, in the case of the mine site and urban data sets, we randomly selected mine and garden sites so that equal numbers of control and disturbed sites were used in each analysis. This was not necessary for the other two data sets, which were already balanced.

A second problem we encountered was that the degree of similarity of the ant fauna between control sites differed for the four studies. This may be a feature of the ant communities in the regions where the studies were carried out, although in part it could also be an artifact of the sampling procedures. To allow for this effect we measured the similarity level that separated the cluster of disturbed sites from that of the control sites and expressed this as a proportion of the similarity level at which all the control sites were joined. This procedure provided a ratio that could be related to a maximum value of 1 (disturbed and control sites exhibit no less similarity than the natural variation between control sites).

Results

The percentage of area in each phytogeographic region occupied by each land use is shown in Table 1. The amount of land devoted to rangeland grazing is high in the northern province, intermediate in most of the Eremaean province, and zero in the south-western province. By contrast, the area cleared for agriculture is high in the south-western province, particularly in the Avon subdistrict, and no land is cleared in the other two provinces. The amount of land under urbanization or roads is proportionately very low and tends to reflect the distribution of the state's relatively small population. Urbanization and road values are highest in the Drummond subdistrict, which contains Perth, the capital city of Western Australia. The other districts or subdistricts of the south-west province have experienced some degree of land alienation by these two activities and, by comparison, levels in the other two provinces are minute. The percentage area of districts disturbed by mining is generally small, although higher levels are evident in the Austin and Coolgardie districts (gold mines), the Fortescue district (iron ore mines), the Drummond subdistrict (mineral sand and limestone mines), the Dale subdistrict (bauxite mines), and elsewhere in the south-west province (mineral sand and limestone mines).

The mean values of ant-species richness, expressed as a proportion of the mean value in the control area, were reduced the most in the areas cleared for roads, followed by agricultural areas, the rehabilitated mines, and then the urban areas (Table 2). By contrast, ant-species richness in the heavily grazed rangeland was actually higher than in the lightly grazed area, a phenomenon also noted by Greenslade and Mott (1979) at Manbulloo. We therefore record the proportion of the original species richness as 1.

The dendrograms derived from the four studies all produced separate clusters of disturbed and control sites. Consequently, we were able to record the similarity level at which the disturbed sites were separated from the control site and to express this as a ratio of this value and the corresponding value that combined all control sites (Table 2). The higher the ratio (maximum = 1) the lower the degree of modification of the ant community by disturbance. Thus, despite the fact that rangeland grazing does not result in a reduction of ant-species richness, there is a change in the ant species present. This land use resulted in the lowest degree of modification of the ant community, followed by mining, agricultural clearing, urbanization, and finally road construction.

Table 2. Mean ant-species richness in disturbed areas within the study areas, and change in ant species similarity between disturbed and control areas.

Land use	*Proportional species richness[a]*	*Change in ant species[b]*
Urban	0.51	0.20
Roads	0.00	0.00
Mining	0.43	0.38
Agricultural clearing	0.41	0.32
Rangeland grazing	1.00	0.71

[a] Species richness is expressed as a proportion of species richness in the corresponding control areas.
[b] Change in similarity between disturbed and control areas expressed as a proportion of similarity between control areas.

Interestingly, the rankings of the five land uses in terms of the two measures are not the same, even though rangeland grazing caused the least modification of the fauna in terms of either measure. This shows that the two community measures respond in different ways to modification of the habitat.

When based on species richness, the BI values range from 47.95 in the Avon district to 99.99 in the Carnegie and Giles districts (Table 3). Although BI values vary considerably among the south-western provinces and generally reflect the known degree of habitat change in these areas, the differentiation among the districts in the Eremaean and northern provinces is small (all values greater than 99.80).

The BI values that are based on species similarity range from 40.04 in the Avon district to 93.26 in the Carnegie and Giles districts. Although the variance among BI values in the south-western province is similar to that obtained by the species richness BI measure, the species similarity approach produced a greater differentiation between districts in the Eremaean and northern provinces.

Although there was a significant linear fit between the two BI measures for the south-western province ($y = 13.497 + 0.86819x$, $r^2 = 0.995$; Fig. 2), there was no relationship in the other two provinces. Thus, although the two measures may be used interchangeably in the south, the BI derived from species richness is not useful in the north because of the insensitivity of ant-species richness to grazing.

Discussion

The failure of the BI based on species richness to clearly differentiate between regions in the northern and Eremaean provinces results from ant-species richness being maintained at high levels in rangeland grazed areas, even though species composition undergoes some change.

Table 3. Biodiversity integrity (BI) values associated with each of the major land uses within each phytogeographic region.

Ant species richness

District or subdistrict	Biodiversity integrity values						
	Uncleared	*Urban*	*Roads*	*Mining*	*Agriculture*	*Rangeland*	*Total*
Dampier	20.98	0.01	0.00	0.01	0.00	78.96	99.96
Hall	4.65	0.00	0.00	0.02	0.00	95.28	99.95
Fitzgerald	45.51	0.03	0.00	0.01	0.00	54.40	99.94
Gardner	45.51	0.00	0.00	0.00	0.00	54.46	99.98
Eucla	55.40	0.00	0.00	0.00	0.00	44.58	99.98
Helms	76.78	0.00	0.00	0.00	0.00	23.20	99.98
Austin	21.51	0.00	0.00	0.06	0.00	78.31	99.88
Carnarvon	44.09	0.01	0.00	0.01	0.00	55.84	99.95
Ashburton	21.51	0.00	0.00	0.01	0.00	78.45	99.97
Keartland	76.78	0.00	0.00	0.00	0.00	23.20	99.98
Carnegie	76.78	0.00	0.00	0.00	0.00	23.21	99.99
Giles	76.78	0.00	0.00	0.00	0.00	23.21	99.99
Fortesque	57.96	0.02	0.00	0.04	0.00	41.90	99.92
Canning	76.78	0.00	0.00	0.00	0.00	23.20	99.98
Mueller	76.78	0.00	0.00	0.00	0.00	23.20	99.98
Coolgardie	72.45	0.02	0.00	0.11	0.00	27.21	99.80
Drummond	43.00	5.95	0.00	0.47	17.71	0.00	67.13
Dale	50.00	0.46	0.00	0.07	19.90	0.00	70.43
Menzies	55.00	0.17	0.00	0.03	18.17	0.00	73.37
Warren	72.00	0.08	0.00	0.08	11.21	0.00	83.37
Irwin	30.00	0.08	0.00	0.06	28.52	0.00	58.66
Avon	12.00	0.04	0.00	0.04	35.87	0.00	47.95
Roe	23.00	0.01	0.00	0.02	31.50	0.00	54.52
Eyre	47.00	0.03	0.00	0.02	21.62	0.00	68.67

Ant species similarity

District or subdistrict	Biodiversity integrity values						
	Uncleared	*Urban*	*Roads*	*Mining*	*Agriculture*	*Rangeland*	*Total*
Dampier	20.98	0.00	0.00	0.01	0.00	56.06	77.06
Hall	4.65	0.00	0.00	0.02	0.00	67.65	72.32
Fitzgerald	45.51	0.01	0.00	0.01	0.00	38.62	84.15
Gardner	45.51	0.00	0.00	0.00	0.00	38.67	84.18
Eucla	55.40	0.00	0.00	0.00	0.00	31.65	87.05
Helms	76.78	0.00	0.00	0.00	0.00	16.47	93.25
Austin	21.51	0.00	0.00	0.05	0.00	55.60	77.16
Carnarvon	44.09	0.00	0.00	0.01	0.00	39.65	83.75
Ashburton	21.51	0.00	0.00	0.01	0.00	55.70	77.22
Keartland	76.78	0.00	0.00	0.00	0.00	16.47	93.25
Carnegie	76.78	0.00	0.00	0.00	0.00	16.48	93.26
Giles	76.78	0.00	0.00	0.00	0.00	16.48	93.26
Fortesque	57.96	0.01	0.00	0.03	0.00	29.75	87.75
Canning	76.78	0.00	0.00	0.00	0.00	16.47	93.25
Mueller	76.78	0.00	0.00	0.00	0.00	16.47	93.25
Coolgardie	72.45	0.01	0.00	0.10	0.00	19.32	91.88
Drummond	43.00	2.33	0.00	0.42	13.82	0.00	59.57
Dale	50.00	0.18	0.00	0.06	15.53	0.00	65.77
Menzies	55.00	0.07	0.00	0.03	14.18	0.00	69.27
Warren	72.00	0.03	0.00	0.07	8.75	0.00	80.85
Irwin	30.00	0.03	0.00	0.05	22.26	0.00	52.34
Avon	12.00	0.02	0.00	0.03	28.00	0.00	40.04
Roe	23.00	0.00	0.00	0.02	24.58	0.00	47.60
Eyre	47.00	0.01	0.00	0.02	16.87	0.00	63.91

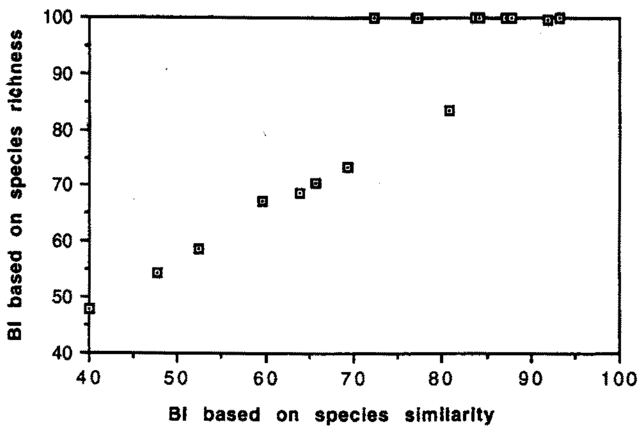

Figure 2. Relationship between biodiversity integrity (BI) values based on species richness and species similarity in the 24 phytogeographic regions. The regression line has been fitted to the data from the southwest province only; the 16 points from the northern and Eremaean provinces occur on the top right of the graph.

Thus, although this measure produces a good differentiation in the south-western province, where rangeland grazing does not occur, it is probably unsuitable for use in the grazed areas. By contrast, the BI based on species similarity was effective at separating regions in all three provinces.

The BI totals for the phytogeographic regions, especially those derived using the index of similarity, appear to separate the various regions in terms of the extent of land alienated for the various land uses. In the Western Australian State of the Environment Report (Environmental Protection Authority 1992) a subjective assessment, based on an overall consideration of many facets of the natural and human-made environment, was made of the general condition of the major vegetation districts. The ranking of regions in the south-western province and in the other two provinces derived from the present study generally agrees with these subjective rankings.

The procedure takes a coarse-grain approach to summarizing the extent of modification of ant diversity in the phytogeographic regions. It has not included the impact of some of the other, possibly more subtle, perturbations such as fire, diseases such as *Phytophthora cinnamomi*, or tourist activity. There is the potential to include these factors if the relevant data on their impact on the ant fauna are available, although an impediment to their inclusion might be that the spatial extent of these perturbations is not known.

A major objective of this paper is to demonstrate the potential for using the procedure with taxa other than ants, or in other parts of the world. For the procedure to work, however, certain criteria need to be met. First, the taxon used should be reasonably species-rich in the areas under consideration. In the case of Australian ants a large number of species are found in each sampling unit within control areas. There is therefore considerable information content within the collections obtained from each area. The same would be true for many other taxa, such as springtails, mites, beetles, birds, or flowering plants. The procedure is unlikely to be useful with less speciose groups because there would be less potential to extract statistical information.

A second criterion is that the taxon used for deriving the BI should be ubiquitous across the study area. This criterion is met for Australian ants, although the procedure may be less robust in the island state of Tasmania, where ant diversity is relatively low. Similarly, although ants would be suitable taxa to use in Africa, South America, and tropical Asia, they would have less utility in Great Britain, where only 45 species occur.

The third criterion is that a good data base should exist on the distribution of the indicator taxon and on the response of this taxon to disturbance. Alternatively, the taxon should be amenable to surveying so that such information can be readily obtained. Taxa for which such information may already exist in Western Australia include birds (Blakers et al. 1984), mammals (Kitchener & Vicker 1981), and many groups of flowering plants (Bureau of Flora and Fauna 1981).

One limitation of the coarse-grain approach is that it fails to detect variations in intensity of a land use, such as grazing, and therefore masks important changes in diversity. This could be rectified by quantifying the impact of varying intensities of a particular land use on the indicator taxon and applying the information to fine-scale geographic information system data. Such an approach would be quite feasible, although it would be better suited to studies of the variation in biological diversity over a more restricted area.

The BI procedure is probably more suitable for environmental auditing at the larger scale. The resulting information could be used in map, graphical, or tabular form to convey information on the quality of the environment in an easily interpretable form. It may also be used to rank sites in terms of BI. This could be particularly useful if decisions are to be made on which regions might be selected for environmental mitigation.

A further spinoff of this procedure is that it provides a statement on the relative impact of different land uses on the BI. This may be considered within a particular district or group of districts, or across the whole country or state. The state-wide summary for Western Australia (Table 4) indicates that agricultural clearing has had by far the greatest impact on ant BI, closely followed by rangeland grazing (if the loss of BI based on species similarity is used). The lower losses of BI as a result of urbanization, roads, and mining puts into context the lower impact of these relatively restricted land uses.

Table 4. Loss of biodiversity integrity (BI) values in the 24 phytogeographic regions (maximum loss = 2400) as a result of five broad land uses.*

Biodiversity integrity based on	*Land use*				
	Urban	*Roads*	*Mining*	*Agricultural clearing*	*Rangeland grazing*
Species richness	6.64	3.10	1.42	265.50	0.00
Similarity index	10.84	3.10	1.54	306.00	217.10

**Loss is calculated by subtracting the BI values shown in Table 3 from the values that would have been produced if ant-species richness or similarity values were 1.00 in each of the land uses.*

Although this procedure would be of value in environmental auditing, it must be appreciated that a coarse-grain approach will not detect change at the smaller scale. A major highway may provide an obstacle to the free movement of and genetic flow within animal populations (Mader 1984), and a mining operation may threaten a local ecosystem. It is therefore important that the procedure be used in the appropriate context and not to produce statistical information that glosses over important local change. In addition, it should not be seen as a substitute for other important measures of biological diversity, such as species turnover, the existence of rare species, the degree of endemism, or representativeness of communities. Specifically, the BI procedure is an adjunct to these measures that can synthesize changes in the integrity of biological diversity of a geographical range. If used in the right context, this procedure could contribute to national or more localized reporting on the current status of biological diversity.

Acknowledgments

We wish to thank Penelope Greenslade for making the Lake Mere ant material available to us. Natalie Keals sorted this material and assisted with data preparation. Neville Wallis kindly prepared the data on road lengths, and Alan Andersen, Harry Recher, Ian Briggs, Richard McKellar, and an anonymous referee commented on an earlier draft of the manuscript.

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