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Use of terrestrial invertebrates for biodiversity monitoring in Australian rangelands, with particular reference to ants

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Abstract Taken literally, the aim of biodiversity monitoring is to track changes in the biological integrity of ecosystems. Given the overwhelmingly dominant contribution of invertebrates to biodiversity, no biodiversity monitoring programme can be considered credible if invertebrates are not addressed effectively. Here we review the use of terrestrial invertebrates, with a particular focus on ants, as bioindicators in Australia in the context of monitoring biodiversity in Australia's rangelands. Ant monitoring systems in Australia were initially developed for assessing restoration success following mining, and have since been applied to a wide range of other land-use situations, including grazing impacts in rangelands. The use of ants as bioindicators in Australia is supported by an extensive portfolio of studies of the responses of ant communities to disturbance, as well as by a global model of ant community dynamics based on functional groups in relation to environmental stress and disturbance. Available data from mining studies suggest that ants reflect changes in other invertebrate groups, but this remains largely undocumented in rangelands. The feasibility of using ants as indicators in land management remains a key issue, given the large numbers of taxonomically challenging specimens in samples, and a lack of invertebrate expertise within most land-management agencies. However, recent work has shown that major efficiencies can be achieved by simplifying the ant sorting process, and such efficiencies can actually enhance rather than compromise indicator performance.

Key words: ant communities, bioindicators, biological integrity, disturbance, functional groups, sampling efficiency.

INTRODUCTION

The term biodiversity monitoring can mean different things to different people. In the present study, we take it literally to mean monitoring the variety of life, and assume that its aim is to track changes in the biological integrity of ecosystems. This is a different issue from monitoring particular components of biodiversity in isolation, for their own particular values. The most commonly used operational units for measuring biodiversity are multicellular species (Purvis & Hector 2000), and the vast majority of these are invertebrates, especially insects and other arthropods (Wilson 1988). Given their overwhelming dominance, no biodiversity monitoring programme can be considered credible without invertebrates being addressed effectively (Taylor & Doran 2001).

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The distribution of terrestrial invertebrates is far more finely patterned than is the case for either vertebrates or vascular plants (Oliver *et al.* 1997; Ferrier *et al.* 1999; French 1999; Pik *et al.* 2002a), and vegetation has repeatedly been shown to be a poor surrogate for patterns of invertebrate biodiversity (Crisp *et al.* 1998; Jonsson & Jonsell 1999; Eyre & Luff 2002). In contrast to vertebrates, where species distributions can be closely correlated with habitat complexity (Coops & Catling 1997), it seems futile to seek attributes of habitat structure that meaningfully act as surrogates for invertebrate biodiversity at anything other than a very coarse level (Abensperg-Traun *et al.* 1996; Newell 1997; York 1999). Specific information on invertebrates requires invertebrates to be monitored directly.

In the present study, we review the use of terrestrial invertebrates as bioindicators in Australia in the context of monitoring biodiversity in Australia's rangelands. We focus on ants because they are the dominant terrestrial invertebrate group in the Australian environment, and are by far the most commonly used invertebrate indicators in Australian land management.

INVERTEBRATES AS BIOINDICATORS

Invertebrates are widely regarded as powerful monitoring tools in environmental management because of their great abundance, diversity and functional importance, their sensitivity to perturbation, and the ease with which they can be sampled (Rosenberg *et al.* 1986; Brown 1997; McGeoch 1998). In contrast, vertebrates tend to be too mobile, generalized or uncommon to be effective indicator taxa at local scales (Read 1998; Hilty & Merenlender 2000).

Invertebrates provide the cornerstone of biological monitoring in aquatic systems, where there are well-developed procedures for using them to assess biological integrity (Norris & Norris 1995; Harig & Bain 1998; Hawkins *et al.* 2000). The use of invertebrates as bioindicators in terrestrial ecosystems, in contrast, has been far less enthusiastically embraced. This reflects the lower prominence of invertebrates within terrestrial ecology more broadly when compared with limnology, especially in Australia. It can be attributed to the far greater prominence of vegetation and charismatic vertebrates in terrestrial compared with aquatic systems, which means that few researchers in land management have an appreciation of, and familiarity with, invertebrates.

The most direct way of monitoring invertebrate biodiversity is to sample entire invertebrate assemblages. This inevitably involves vast numbers and a great variety of specimens. Although the development of sophisticated computer-aided processing systems has greatly assisted the management of such samples (Oliver *et al.* 2000; Pik *et al.* 2002b), the efficiency of an entire-assemblage approach is open to question (Hilty & Merenlender 2000). A far more common approach is to focus on one or more indicator groups that reflect broader patterns of invertebrate biological integrity.

In the northern hemisphere, the most widely used invertebrate indicators are beetles, especially Carabidae (Stork 1990; Eyre & Luff 2002). This has led to the establishment of GLOBENET, a global initiative for assessing landscape change using carabid beetles (Niemelä *et al.* 2000). However, the use of carabids as bioindicators in Australia has been extremely limited, due to their generally low abundance in invertebrate samples and lack of ecological and taxonomic understanding of the Australian fauna (New 1998). Such a lack of understanding can severely compromise the interpretation of results from monitoring, given the need to separate environmental impacts from background variability in the face of the low statistical power that is typical of impact studies (Andersen 1999). Other invertebrate groups, such as spiders (Churchill 1997), grasshoppers (Andersen *et al.* 2001) and moths (McQuillan 1999; Kitching *et al.* 2000), have also been proposed as potentially useful indicators in Australia, but they likewise suffer from our poor knowledge of them (New 1999). In contrast, the community ecology of ants is particularly well known in Australia.

ANTS AS BIOINDICATORS IN AUSTRALIA

Ant monitoring systems in Australia were initially developed for assessing restoration success following mining (Majer 1983), and represent some of the earliest uses of insects as bioindicators in land management anywhere in the world. Ant monitoring is now widely adopted in the Australian mining industry as part of best-practice environmental management (Andersen 1997a; Majer & Nichols 1998). Ant monitoring has also been applied to a wide range of other land-use situations (Andersen 1990), including off-site mining impacts (Read 1996; Madden & Fox 1997; Read & Pickering 1999; Hoffmann *et al.* 2000), forest management (Neumann 1992; York 1994, 2000; Vanderwoude *et al.* 1997, 2000), conservation assessment (Yeatman & Greenslade 1980; Burbidge *et al.* 1992; Clay & Schneider 2000) and grazing impacts in rangelands (Landsberg *et al.* 1999; Hoffmann 2000; Read & Andersen 2000; Woinarski *et al.* 2002).

All relevant Australian studies have shown rangeland ant communities to be sensitive to disturbance, and sometimes particularly so (Landsberg *et al.* 1999; Woinarski *et al.* 2002). This contrasts with the findings of a North American study that concluded that rangeland ants were insensitive to land-use impacts (Whitford *et al.* 1999). The conclusion was based on an ordination of 44 sites representing a range of habitats and land uses in Arizona and New Mexico, which showed that intrinsic soil and vegetation variables have a far greater effect on ant communities than land use does. However, one would expect such a result for any animal group, and the interpretation that this makes ants insensitive to disturbance seems flawed. An earlier study in the same region, which did not confound the effects of disturbance with those of intrinsic habitat variation, found ants to be highly sensitive to grazing impacts, with the authors concluding that ants 'provided interpretable data for developing an indicator of exposure to ecosystem stress' (Nash *et al.* 1998).

More generally, the use of ants as bioindicators in Australia is supported by an extensive portfolio of studies of the responses of ant communities to disturbance (Hoffmann & Andersen 2003). Responses of individual species will inevitably vary with disturbance type and intensity, and with habitat, but groups of species can be identified throughout the country as relatively consistent 'increasers' or 'decreasers' in relation to disturbance. For example, species of the *metallica* group of *Rhytidoponera* from southern Australia typically increase in abundance following habitat disturbance, as do species of the *denticulatus* group of *Camponotus* from central and northern Australia (Hoffmann & Andersen 2003).

A predictive understanding of the responses of Australian ant communities to disturbance has been formalized in a global model of ant community dynamics based on functional groups in relation to environmental stress and disturbance (Andersen 1995). These functional groups have been developed as a framework for analysing ant communities at biogeographical scales (Andersen 1997b), but can also provide a useful basis for assessing the responses of local ant communities to land use. This is especially true when land management involves major habitat modification, as is the case in habitat restoration (Andersen 1997a; King *et al.* 1998; Bisevac & Majer 1999a), plantation forestry (Pik *et al.* 1999) and frequent burning (Andersen 1991; Vanderwoude *et al.* 1997). The system is less useful for analysing disturbance in open habitats of inland Australia, where habitat structural change is less marked than in well-forested areas (Hoffmann & Andersen 2003). Nevertheless, the functional group scheme can still be a useful tool for analysing grazing impacts in rangelands. For example, preliminary results from 25 sites in the Cobar region of western New South Wales show that functional groups perform as well as species in discriminating land condition in relation to grazing (A. Andersen, A. Fisher & R. Richards, unpubl. data, 2002).

RELIABILITY

Ultimately, the reliability of any indicator group is determined by the extent to which it reflects the broader entity it purports to represent, and not by its sensitivity per se to environmental change (Andersen 1999). The extent to which responses of ants to land use reflect that of invertebrate biodiversity more generally has been poorly documented, and is virtually unknown in rangelands. As far as we are aware, relevant data are available only from mining studies. For example, a study of rehabilitated bauxite minesites in south-western Australia showed that ant species richness was correlated with the richness of a range of other key invertebrate groups (Majer 1983). Similarly, ant species composition positively correlated with that of other key invertebrate groups, as well as to overall assemblage composition, at a wide range of natural and disturbed sites in and around the Ranger uranium mine in the Northern Territory (Andersen 1997a).

FEASIBILITY

Despite increasing recognition that ants provide a useful indication of change in biological integrity associated with land use, their feasibility as indicators remains a contentious issue. For example, Landsberg *et al.* (1999) compared the feasibility of sampling a range of indicator taxa and questioned the cost-effectiveness of ants compared with more familiar groups such as vascular plants and birds. However, the study did not include anyone with expertise in invertebrate surveying, and quite a different conclusion was reached in another study that did (Bisevac & Majer 1999b). Moreover, it is becoming increasingly clear that simplified approaches to invertebrate sampling and processing can provide effective results. For example, in a study of off-site mining impacts at Mount Isa in north-western Queensland, a greatly simplified sampling protocol was shown to reproduce the key findings from a comprehensive ant survey (Andersen *et al.* 2002). An assessment of large ants only, collected as bycatch from vertebrate pitfall traps, was able to detect emission impacts up to 35 km from the emission source (Fig. 1). The ant bycatch from vertebrate pitfall traps was similarly effective in documenting ant community responses to landscape position, grazing and military use in north-eastern Queensland (Woinarski *et al.* 2002).

The ongoing Cobar study mentioned previously has a major focus on improving the efficiency of ant monitoring. Preliminary results indicate that simplifying the ant sorting process can actually improve rather than diminish performance. For example, small selected subsets of ant genera performed better at discriminating land condition than the total species did (A. Andersen, A. Fisher & R. Richards, unpubl. data, 2002).

Table 1. Contribution of different land uses to percentage biodiversity loss in two contrasting bioregions of Western Australia according to the biodiversity integrity index from the study by Majer and Beeston (1996)

Land use	Fortescue†	Avon‡
Rangeland grazing	12.15	0.00
Agricultural clearing	0.00	59.49
Mining	0.05	0.06
Urbanization	0.03	0.06
Roads	0.02	0.35
Total biodiversity loss	12.25	59.96

†Fortescue is in the remote Pilbara region, where rangeland grazing is the most extensive land use; ‡Avon has extensive cropping lands.

SPATIAL SCALE

Invertebrate biodiversity is too finely patterned to be effectively sampled at anything other than the plot scale, and, as previously mentioned, it appears unrealistic to think that invertebrate biodiversity can be usefully characterized by surrogates measured at broader spatial scales. However, plot-scale measurements of invertebrates can readily be scaled-up to provide information on biodiversity at broader spatial scales.

Ants have been used to derive a biodiversity integrity index as a framework for estimating biodiversity change at regional scales (Majer & Beeston 1996; Beeston & Majer 2000). The process involves the following steps. First, the major land uses in the region are identified, and their spatial extent recorded. Second, the impacts of each land-use type on ant biodiversity are determined. The most robust way of doing this is to use compositional change. In their analysis for Western Australia, Majer and Beeston (1996) assigned rangelands a score of 71%, intensive agriculture 32%, mining 38%, urbanization 20% and roads 0% based on the compositional similarities of their associated ant communities compared with nearby reference sites (100%). Next, the spatial extent of each land use is multiplied by its relative impact on ant biodiversity. Finally, the resultant values are summed to provide the biodiversity integrity index for the region. Conversely, loss of biodiversity integrity is calculated by assigning each land use a loss value of (100 – relative impact)%, and multiplying this figure by spatial extent (Table 1). Majer and Beeston (1996) used the indices from all bioregions to calculate that the land use causing most biodiversity loss across the State was intensive agriculture (306 units of biodiversity loss), followed by rangeland grazing (217 units of biodiversity loss). Mining, in contrast, had a negligible impact (1 unit of biodiversity loss).

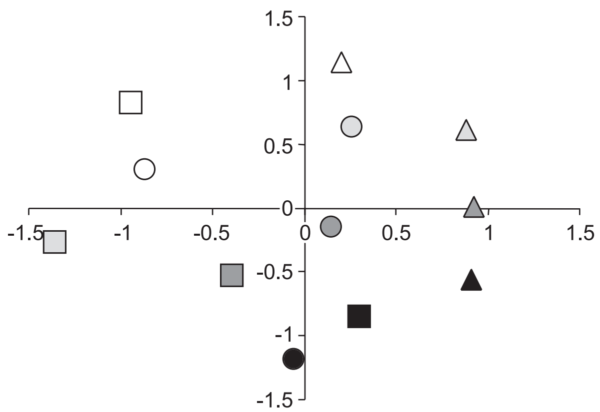

Fig. 1. Multivariate ordination based on results of a simplified ant-sampling protocol at Mount Isa in north-western Queensland. The protocol involved collecting large (>4 mm) ant species from the bycatch of vertebrate pitfall traps, and recording species presence/absence only. The protocol was able to differentiate between different habitat types (rocky ridges, triangles; rocky plains, circles; alluvial plains, squares) and different levels of SO_2 (background, open symbols; low, lightly shaded symbols; medium, heavily shaded symbols; high, closed symbols). Modified from Andersen *et al.* (2002).

CONCLUSION

Over the past decade there has been increasing recognition of the value of ants as bioindicators in the Australian terrestrial environment. However, ants or any other invertebrate group will not be routinely included in biodiversity monitoring programmes until two key impediments are overcome. The first is a traditional wildlife culture that sees agencies responsible for biodiversity management having little, if any, familiarity with terrestrial invertebrates. This problem is exacerbated by a general unavailability of species-level identification tools for non-specialists; in the absence of such tools, for any invertebrate group there are only a handful of people in Australia with the experience required to sort and identify species with confidence. The second impediment is the need to improve monitoring efficiency, and, in particular, to simplify the process of dealing with very large numbers of specimens from numerous, taxonomically challenging species. Our work has shown that invertebrate monitoring does not require comprehensive surveys, and that processing can be greatly simplified by using presence/absence or frequency data, and by considering only subsets of species, without compromising reliability.

These impediments highlight two research and development priorities in relation to using ants as bioindicators in rangeland monitoring. The first is the identification of subsets of species that track biological integrity most efficiently in different rangeland situations and the way data about these subsets are treated. Given the general uniformity at the genus level of ant faunas throughout much of inland Australia (Andersen 2003), it is likely that the use of particular subsets of taxa will have wide applicability. The second priority is the development of effective species-level identification tools for the non-specialist. The number of undescribed ant species in Australia is so large, and the number of practising taxonomists so small, that formal taxonomic treatment in published papers will never meet this need in the short- to medium-term. However, there would be enormous value in establishing regional 'virtual' museums of species-coded reference specimens using web-based imaging and identification technologies, which could be readily used by workers with limited taxonomic experience.

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