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Terrestrial invertebrate surveys and rapid biodiversity assessment in New Zealand: lessons from Australia

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Abstract: Although invertebrates play a key role in the environment, their conservation and use in environmental monitoring is often considered “too difficult” and consequently ignored. One of the main problems in dealing with invertebrates is that even limited sampling can yield large numbers of specimens and an enormous diversity of species. Other problems include the taxonomic impediment (i.e. high proportions of invertebrate taxa are undescribed and there are few specialists available to identify specimens), the lack of knowledge on species distribution, diversity and ecological roles, and the fact that invertebrates are undervalued by the general public. A number of rapid biodiversity assessment (RBA) approaches have been suggested to overcome these problems. RBA approaches generally fall into four categories: (1) restricted sampling in place of intensive sampling (sampling surrogacy); (2) the use of higher taxonomic levels than species (species surrogacy); (3) the use of recognisable taxonomic units (RTUs) identified by non-specialists (taxonomic surrogacy); and (4) the use of surrogate taxa in place of all taxa (taxon-focusing). Australia has a long history of using invertebrates in terrestrial ecological studies, and in developing and using RBA approaches. Therefore, New Zealand could benefit from the experienced gained in Australia. Potentially one of the most useful RBA approaches to take in New Zealand involves focusing resources and attention on a limited range of taxa. However, this requires substantial communication, discussion, and agreement over which taxa should be selected for conservation priorities and environmental monitoring in terrestrial ecosystems.

Keywords: broad-based surveys; indicator species; invertebrates; monitoring; RTU; rapid biodiversity assessment; taxon-focusing.

Introduction

One of the principal directions of conservation biology over the last decade has been a shift from species-centered conservation towards the conservation and management of communities, landscapes and ecosystems (Grumbine, 1994; Hobbs, 1994; Hutcheson, 1994; Park, 2000). The traditional single-species approach to conservation has drawn increasing criticism for a number of reasons: it consumes a disproportionate amount of funding [for example, Given (1994) estimated that approximately 10% of critically endangered species consume 90% of conservation funding]; it cannot be conducted at a rate sufficient to deal with environmental threats; and does little to protect the major component of biodiversity, the invertebrates (Hobbs, 1994; Lambeck, 1997). The protection of large vertebrates does not necessarily extrapolate to conservation gains for lower organisms, communities or entire ecosystems. For example, Mac Nally *et al*. (2002) demonstrated that correlations of diversity for vertebrates and invertebrates in central Victoria, Australia, were generally weak [but see Martikainen *et al*. (1998), who showed that protecting the endangered white-backed woodpecker would also protect the habitats of a number of threatened beetle species].

Invertebrates are now recognised as important components of biodiversity (Kim, 1993; Kremen *et al*., 1993; Oliver and Beattie, 1996; Yen and Butcher, 1997). They are important in all ecosystems in terms of species numbers and biomass, and play vital roles in processes such as pollination, soil formation and fertility, plant productivity, organic decomposition, and the regulation of populations of other organisms through predation and parasitism (Daily *et al*., 1997; Yen and Butcher, 1997). Invertebrates are also the principal food-source of many vertebrates.

Furthermore, invertebrates are increasingly being recognised as important indicators of environmental changes. Kremen *et al*. (1993) suggested that terrestrial arthropods could be used for virtually any monitoring challenge. Conservation and biodiversity assessments that use invertebrates allow patterns of diversity and environmental quality to be measured at scales that are often more meaningful than those measured using plants and vertebrates (Yen and Butcher, 1997). The majority of invertebrates are also more sensitive to environmental perturbations than plants and vertebrates due to their rapid breeding rates and relatively short generation times (Kremen *et al*., 1993; Hilty and Merenlender, 2000). In addition, invertebrates exhibit a wide range of body sizes, growth rates, life history strategies and ecological preferences, which can be linked with specific variables to provide a greater understanding of invertebrate responses to environmental conditions and to generate predictive models for ecosystem biodiversity (Clarke, 1993; Niemelä *et al*., 2000).

The wider acceptance of invertebrates as indispensable components of biodiversity has led to a rapid increase in broad-based surveys (i.e. a survey incorporating a wide range of invertebrate taxa) and greater pressure to provide information and guidelines for invertebrate conservation and monitoring. The latter are necessary to ensure that surveys produce comparable information (e.g. presence or absence of species, their geographical and ecological distribution) that can then be used both to establish measurements of rarity and diversity, data for baseline monitoring, and to provide an objective base for supporting priority-setting decisions in conservation (Yen and Butcher, 1997).

Nevertheless, the inclusion of invertebrates in environmental monitoring programmes is still too often ignored; deemed too difficult, or not cost-effective for a number of reasons. Even limited sampling of invertebrates can yield an enormous number of specimens and an immense array of species. This precludes the use of the traditional single-species approach to conservation for the vast majority of invertebrate species. In addition, and this is particularly true in countries such as New Zealand and Australia, high proportions of invertebrate taxa are undescribed or unknown, and the number of specialists able to process samples and identify specimens is limited (New, 1999a). This situation is termed the 'taxonomic impediment.' Additional difficulties arise mostly from the limited knowledge of ecological roles, patterns of species diversity and distribution, little specific knowledge about invertebrate responses to environment changes, and the lack of clearly documented, easy-to-understand, standardised sampling protocols. There is also limited sympathy for invertebrates from the public, politicians, administrators and funding agencies who do not necessarily perceive invertebrates as charismatic animals warranting additional attention and funding. The rationale to conserve invertebrates needs to change (New, 1994).

The rise of Rapid Biodiversity Assessment (RBA)

Rapid Biodiversity Assessment (RBA) approaches have arisen mainly to help overcome many of the difficulties associated with large-scale invertebrate surveys. The two main objectives of RBA are to reduce the effort and cost of sampling, and to summarise complex ecological details so they can be understood by non-specialists (New, 1998a). Although RBA approaches have been subject to criticism (Brower, 1995; Goldstein, 1997; Trueman and Cranston, 1997), there is a need for invertebrate survey methodologies that can evaluate large numbers of species, increase ecological understanding, and that can be undertaken at a reasonable financial cost (Majer, 1983; Greenslade and New, 1991; Kremen *et al*., 1993; Sparrow *et al*., 1994; Oliver and Beattie, 1996).

In Australia, RBA approach have been strongly advocated, developed, and widely used in invertebrate surveys (Majer, 1990; Andersen, 1995; Oliver and Beattie, 1996; Yen and Butcher, 1997; New, 1998a; 1999a; Oliver *et al*., 1999). Possible reasons for this include a mega-rich continental invertebrate fauna (conservative estimate of 125 000 insect species), a small scientific community, and a relatively short scientific history (Greenslade and New, 1991). In comparison there have been relatively few such surveys in New Zealand. For example, Hutcheson and Kimberley (1999) showed that Malaise-trapped beetles were characteristic of different vegetation types, and the use of Malaise traps over the summer months provided an excellent assessment of beetle diversity. Harris and Burns (2000) examined the beetle assemblages of kahikatea forest fragments surrounded by pasture-dominated landscapes and concluded that forest fragments serve as important refuges for indigenous beetle fauna. Watts and Gibbs (2000) also examined the indigenous beetle fauna from revegetated habitats of different ages on Matiu–Somes Island and showed that the proportion of indigenous beetle species was positively correlated with that of indigenous plant species richness. The increased interest in invertebrate biodiversity has also been recently documented in a number of more generalised treatments: Hutcheson *et al*. (1999) (the value of indicator species), Department of Conservation/Ministry for the Environment (2000) (The New Zealand Biodiversity Strategy), and McGuinness (2001) (Department of Conservation threatened invertebrate species).

The reasons for adopting RBA approaches in New Zealand are similar to those observed in Australia even though the fauna may not be as rich (but it is largely endemic and undescribed), and the specialised workforce is even smaller than in Australia. The difficulties associated with broad-based invertebrate surveys and the significant gaps of scientific knowledge on invertebrate biology outlined at the beginning of this paper, apply equally well to New Zealand. Therefore, it is desirable to develop and apply RBA approaches to invertebrate surveys here. Because relatively few broad-based surveys of invertebrates have been completed so far in New Zealand, it is possible to benefit from the experience and research gained in Australia.

Oliver and Beattie (1996) outlined four general categories of RBA approaches: (1) sampling surrogacy (restricted sampling in place of intensive sampling); (2) species surrogacy (use of taxonomic levels higher than species, i.e. families, orders); (3) taxonomic surrogacy (use of recognisable taxonomic units [RTUs] identified by non-specialists); and (4) taxon-focusing (use of surrogate taxa in place of all taxa). The aim of this paper is to outline, and provide examples of, the four different approaches to rapid biodiversity assessment, particularly from an Australian perspective, with recommendations for New Zealand.

Sampling surrogacy

This approach to rapid biodiversity assessment involves reduced sampling. This may include shorter sampling duration, a reduced number of sampling methods employed, the use of less-intensive sampling methods than usual, as well as sub-sampling existing material. Statistical extrapolation methods can serve to estimate species richness using reduced sampling. Niemelä *et al*. (1990) suggested sampling ground beetles for periods of 20 days was representative of a season and could thus be used to reduce sampling effort when studying individual species or groups of locally abundant species. Sparrow *et al*. (1994) showed that sampling frequency and intensity were important factors in determining sampling effort for monitoring the species richness of neotropical butterflies. Colwell and Coddington (1994) provided a range of statistical methods (accumulation curves, non-parametric techniques) to estimate local species richness. Hammond (1994) used ratios and a hierarchical step-by-step approach to estimate species richness from groups and sub-groups of taxa. Samu and Lövei (1995) showed how the combination of field sampling and species accumulation curves can provide robust and accurate estimates of species richness for spider communities.

The number of sampling methods and choices of sampling protocols available for broad-based surveys are large, and the design of a survey must take into account a number of considerations. First, care must be taken to minimise the reduction in data quality with a reduction in sampling effort: surveys should have sufficient replication for statistical analyses. Second, sampling methods do not collect all species equally (New, 1998a; Kitching *et al*., 2001; Ward *et al*., 2002). For example, pitfall traps are used extensively to catch ants in Australia, but, when used alone, this method undersamples ant species richness, and provides a skewed representation of functional groups because ants in deep litter, soil or on vegetation are not sampled (Majer, 1997). Similar biases occur with ground beetles (Carabidae) in the Northern Hemisphere (Spence and Niemelä 1999; Niemelä *et al*., 2000). The issues of sampling surrogacy and sampling design for invertebrates surveys are extremely important and this paper cannot give them full justice, however, see Oliver and Beattie (1996), Oliver *et al*. (1999) and New (1998a) for details.

Species surrogacy

The use of taxonomic levels other than the species (e.g. genus, family, order) has received substantial attention in RBA. Higher taxonomic levels are frequently used in marine and freshwater environments where they respond to environmental gradients in a predictable manner, and related species have similar ecological requirements (Warwick, 1993; Marchant *et al.,* 1995). In forested areas of Australia the richness of ant species can be predicted by the richness of ant genera as the relationship between ant genera and species is close to 1:1 (i.e. monospecific) (Pik *et al*., 1999; Neville and New, 1999). However, in arid areas the relationship between generic and species richness does not hold because of species-rich genera in arid areas (Andersen 1995). Thus, genus richness (as a measure of species richness) is reliable only under limited circumstances and can be confounded by habitat, biogeography and sampling effort (Andersen, 1995).

The advantage of using higher taxonomic levels in surveys is that costs could be substantially reduced as the time-consuming task of identifying specimens to species level becomes unnecessary. However, New (1996) notes that it remains unproven whether the use of higher taxonomic levels is applicable more widely, particularly to terrestrial ecosystems where species richness and ecological heterogeneity are greater than for freshwater and marine ecosystems. In addition, higher taxonomic levels often contain species with a variety of feeding types and trophic levels (Beattie *et al*., 1993), and as a result individual species' responses can be masked by analysis at higher levels. This "cancelling-out effect" has been observed for ground-dwelling invertebrates at the family level (Neville and Black, 1997).

Taxonomic surrogacy

The use of recognisable taxonomic units (RTUs) [also known as morphospecies (MSP) or operational taxonomic units (OTUs)] has been advocated, where RTUs act as a surrogate for species-level identification (Beattie and Oliver, 1994). This approach has received recent attention in RBA and has been particularly controversial as opponents argue against the reduction of taxonomic "accuracy" in specimen identification (Brower, 1995; Goldstein, 1997). However, the use of "parataxonomists" (non-specialist taxonomists) to sort mass samples of invertebrates into RTUs before specialist identification could increase cost-effectiveness (Janzen 1991; Cranston and Hillman, 1992).

Oliver and Beattie (1995) tested the relationship between RTU-level identification by a non-specialist and species identification by a specialist taxonomist from pitfall trap and litter samples of ants, beetles and spiders from four Australian forest types. Estimates of species richness of ants and spiders varied little between RTU and actual species inventories. Richness estimates for beetles were influenced by non-specialist identification errors in two speciose families, Curculionidae and Staphylinidae. However, species turnover (β diversity), ordination patterns of species assemblages, and rankings of species richness by forest type were similar for RTUs and species inventories (Oliver and Beattie, 1995).

Several interrelated problems arise with the RTU level approach. First, descriptive taxonomy and formal names facilitate information retrieval and communicate information about biodiversity (New 1999b). Dall (1997) showed that species selected for conservation are almost always those that can be recognised and have scientific names. Second, the two roles of practical specimen identification are (1) consistent and unambiguous recognition of a particular species/taxon, and (2) the identification of members of assemblages for use in ecological measurements (Yen and Butcher, 1997). Identifications must be consistent across sites and sampling occasions, and ideally across different ecological surveys and projects. Surveys of the ant and beetle faunas in Victoria, Australia, over the last two decades resulted in many "RTU#1s". However, the delineation of "RTU#1" is not consistent across surveys and, as such, includes a number of different species, which severely limits the information gained about species distributions. Third, use of RTUs introduces a concern about data interpretation without knowledge of species assemblages. For example, Greenslade and Majer (1993) examined Collembola from forests and rehabilitated mines in Western Australia. Although both habitats contained similar numbers of species, the forest habitat largely contained native species, while cosmopolitan species predominated in the mined areas. Such differences can be overlooked without species-level identification (New, 1996).

Thus, RTUs should only be used as a stepping-stone before formal species-level identification. This approach is not always advocated in Australia where RTU identification is often the end-point. Under such conditions, the deposition of voucher specimens in appropriate collections is very important.

Taxon focusing

Taxon focusing includes a range of approaches that aim to identify a species, or a group of species, that act as a surrogate for a wider range of taxa. These approaches are based on the assumption that the selection, and protection, of a restricted number of "focal taxa" will also help the protection of other taxa. The use of focal taxa (such as keystone species, umbrella species and indicators) has received considerable attention. Unfortunately, there are few guidelines for the selection of specific focal taxa, although a number of authors have suggested objective and standardised scientific criteria. These typically involve taxonomic, biogeographic, biological and logistic parameters (Kremen *et al*., 1993; New, 1993; Pearson, 1994; Yen and Butcher, 1997; McGeoch, 1998; Andersen, 1999). However, the selection of focal taxa is usually based on practical grounds rather than assessed scientific value. This practicality emerges from a combination of taxonomic difficulties, availability of co-operation, personal interest, and prior selection in relation to particular aims or hypotheses (New, 1998a; 1999b).

The "taxon focusing" approach has been criticised because there is little evidence that patterns demonstrated by a few species can appropriately reflect biodiversity patterns of all elements of the biota (Prendergast *et al*., 1993; Duelli and Obrist, 1998; Lindenmayer *et al*., 2002). For example, Prendergast *et al*. (1993) showed that species richness hotspots can be in different places for different groups, and rare species often occur outside areas of high species richness. Few studies have extended their research to determine whether the indicator species or group they advocate is representative of other taxa. As a consequence of criticism, there has been considerable refinement of the taxon-focusing approach. Hammond (1994) suggested the idea of a "shopping basket" containing a number of taxa, each representing the response of a wider set of taxa (also see Niemelä and Baur, 1998). Lawton *et al*. (1998) used a range of taxa including birds, beetles, ants, termites, butterflies and nematodes to investigate the effect on biodiversity of logging practices in the tropics. Strong differences in the response to logging were found between taxonomic groups, indicating that a broad "shopping basket" approach is needed to gain a holistic picture of the environment. Mac Nally *et al*. (2002) went further, and defined "biodiversity management units" (BMUs) to protect regional biodiversity in central Victoria, Australia. These units were based on a combination of biological (vegetation communities), climatic (temperature, rainfall) and landform characteristics (topography), rather than solely on taxon-specific criteria. The protection of a representative collection of BMUs in a region may act as an umbrella, protecting a broad range of communities and ecosystems.

An optimal course to follow?

In spite of the difficulties associated with the selection of focal taxa (Simberloff, 1998; Lindenmayer *et al.*, 2002), the taxon-focusing approach remains one of the more practical rapid biodiversity approaches to invertebrate conservation. In New's (1996) opinion, the detailed study of a limited number of carefully chosen taxonomic groups will be more productive and realistic than attempting to evaluate a larger number of groups superficially. We are not suggesting that other RBA approaches be ignored, but rather that they are used in combination with taxon-focusing. We advocate the taxon-focusing approach and believe it will provide the greatest benefit for invertebrate conservation and biodiversity monitoring, we note, however, there is also a need for research on the representativeness of selected taxa in New Zealand ecosystems.

Taxonomic knowledge of invertebrate taxa is highly uneven, and, as such, different invertebrate taxa generally fall into the following three major categories (following New, 1999a). First, "well-known" groups, which, in Australia, include ants (Andersen, 1990), while in Europe, include the ground beetles (Stork, 1990). Second, "catch-up" groups, which consist of a large variety of invertebrate groups for which reasonable knowledge is available but the groups are not "well-known", for example, ground beetles (New, 1998b) and springtails (Greenslade, 1997) in Australia. These groups could be found to have value for conservation if research on them is increased. They could then be transferred to the "well-known" category. Third, "black-hole" groups, which are generally difficult to study and recognise, for example, nematodes. Incorporation of these groups in practical conservation or environmental monitoring would require an enormous effort (New, 1999b).

Which groups?

New (1996) has suggested that the most useful way to proceed is to focus on "well-known groups", but also to simultaneously concentrate on a number of "catch-up groups" so that they could become "well-known groups". This would seem to be a logical course of action to follow in New Zealand. Consequently, the three categories of groups defined above could serve as general selection criteria for focal-taxa. For example, in New Zealand, tenebrionid beetles (Coleoptera: Tenebrionidae) could fall into the "well-known" groups category (Watt 1992). Ground-beetles (Coleoptera: Carabidae) could be considered a "catch-up" group verging on the "well-known", depending on the tribes being considered (Larochelle and Larivière 2001). Most true bugs (Hemiptera) could fall into the "catch-up" groups category, with only some families verging on the "well-known" e.g. leafhoppers (Cicadellidae), seed bugs (Lygaeoidea), plant bugs (Miridae) and shield bugs (Acanthosomatida, Pentatomidae). These are only a few examples of groups familiar to the authors. What would be desirable is a thorough inventory and classification of invertebrate taxa occurring in New Zealand according to the above mentioned criteria. Such a compilation is a substantial task that cannot be fully addressed in this paper, but it seems useful to elaborate on the concept of taxon focusing to stimulate further discussion.

Aspects other than taxonomic criteria would also need to be considered when selecting focal taxa. Clearly, biological, biogeographical and logistical parameters will need to be assessed. The availability of taxonomic expertise is also important, as some "well-known" groups may still be very difficult to identify by non-specialists, and thus may not be good candidates for focal taxa. Ideally, a suite of taxa would also represent all major functional guilds and would convey as much information as possible on the sampled sites or habitats (New, 1996; 1998a). However, it is likely that the various suites of invertebrate taxa chosen will change in the different habitats and climatic regions (Yen and Butcher, 1997). For example, while ants are well studied in Australia because of their diversity (Andersen, 1990; 1997), there are very few native species in New Zealand and the fauna is not widely distributed, thus limiting their use as focal taxa. Beetles, however, have received recent attention (Hutcheson and Kimberley, 1999; Harris and Burns, 2000; Watts and Gibbs, 2000) in New Zealand and are likely to remain a key group.

Future directions for New Zealand

Invertebrate surveys have historically been designed and completed with little consideration for subsequent ecological analysis and comparison, and the use of broad-based surveys has often been overlooked because of cost (New, 1998a). However, extensive surveys (i.e. broad-based, over a large area, to determine the species present and their distributions) are becoming increasingly important for invertebrate conservation and environmental monitoring (Oliver and Beattie, 1996; Yen and Butcher, 1997). New (1994) outlined a number of important steps to follow: (1) discussion on the values of taxon-focusing over a traditional single-species approach; (2) categorising invertebrate taxa into the three categories of "well-known", "catch-up" or "black-hole"; (3) gaining a broad consensus of invertebrate taxa to study [see Brown, (1997) for an illustration in the Neotropics]; and (4) formulating standardised protocols for sampling those taxa [see the recent notable example set by Stark *et al*. (2001) for freshwater sampling protocols in New Zealand].

New (2000) noted that a good start for providing relevant information is to compile a bibliography of taxonomic information, such as that already completed by Ramsay and Crosby (1992) on the New Zealand fauna. In New Zealand it should also be easier to implement a coordinated national taxon-focusing approach without the bureaucracy associated with several levels in the Australian government. New Zealand also has only one major collection repository, the New Zealand Arthropod Collection (hosted at Landcare Research, Auckland, New Zealand). This repository, comprising the most comprehensive national holding of invertebrate taxa, together with its associated databases and systematics programme, can be relied on for checking identification as well as for depositing voucher specimens and data. In addition, recent advances in information systems such as electronic bar-coding of specimens and samples, computerised data management, and virtual collections are being increasingly explored to help in managing large datasets [Oliver *et al*. (2000): BioTrack™; Larivière and Rhode (2001): KOIORA-BIOASSIST™]. These computerised systems will allow for greater standardisation of RTUs and formal identifications across many projects from a variety of spatial and temporal scales, if RTU images and protocols become widely available and are used.

The suggested taxon-focusing will hopefully achieve greater inclusion of invertebrates in conservation assessment and environmental monitoring. A consensus of invertebrate taxa to focus on would represent a major advance in invertebrate conservation (New, 1996; Yen and Butcher, 1997). With careful planning and selection, a suite of nominated invertebrate taxa that encompasses a wide range of ecological roles and habitats might also serve as a comprehensive conservation umbrella for other taxa (New, 1996; Yen and Butcher, 1997). We are not suggesting that other RBA approaches be ignored, rather that they be used in combination with taxon-focusing. Single species should not be used in management plans to protect habitat and other species. Simberloff's (1998) extensive review of focal taxa showed the inadequacy of such an approach. Ideally, a number of invertebrate groups should be used to achieve a realistic measure of the environment (see Lawton *et al*., 1998). In essence, we are taking a "middle position" along a continuum, from using only a single species at one end (extreme RBA) to measuring every species at the opposite end (single-species conservation *v.* total inventory).

A definitive goal for invertebrate conservation would be the wide use of a set of relatively standardised sampling methods, to collect and study a relatively limited number of taxa (i.e. focal taxa) from many different habitats and ecosystems, with electronic access to a large amount of taxonomic and ecological data. Standardised and comparable data would be collected on species presence/absence, distribution patterns, habitat associations, diversity, rarity and abundance. These data are needed for effective conservation and monitoring of invertebrate species and their habitats. The above scenario is not unreasonable because there are already a number of groups that have the expertise to undertake large-scale invertebrate surveys in New Zealand. The technology to provide people with integrative databases of taxonomic and ecological data also exists. What we need is guidance on the value and direction of rapid biodiversity approaches and broad-based surveys for invertebrate conservation. The alternative is to continue working with non-standardised environmental monitoring regimes, and a fragmented approach to the daunting task of invertebrate conservation.

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