Conservation Planning and Biodiversity: Assembling the Best Data for the Job

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Readers of the note by Brooks et al. (2004) will, I hope, be persuaded of the urgent need for more comprehensive data on species. It would be unfortunate, though, if readers formed the impression that biodiversity processes cannot be seriously considered by conservation planners because “Techniques for mapping and measuring ecological and evolutionary processes are in their infancy.” More unfortunate would be the impression that “broad-scale biodiversity attributes” (Brooks et al. 2004), hereafter “land types” (such as vegetation units or land systems) are alternatives to data on species. Most conservation planners would agree with Brooks et al. that different biodiversity currencies have different advantages and problems. From that starting point, divergent courses can be plotted.

Brooks et al. recommend species data as a more promising option than relying on land types and call for an end to “armchair environmental classification.” Others have taken a different course. Since the early 1990s, many regional conservation plans have been shaped by a mixture of biodiversity surrogates, selected from a list of potential types (Table 1) and assembled into composite data sets (Noss 1993; Davis et al. 1999; Groves et al. 2000; Cowling et al. 2003). The rationale is straightforward: all surrogates are limited idiosyncratically in their depictions of biodiversity pattern and process, so a more comprehensive array of surrogates gives a better picture.

There is a broader context within which to consider the valid concerns of Brooks et al. about grossly incomplete species data. Conservation planners need better data on multiple biodiversity surrogates (Table 1), better methods for testing their effectiveness, and a deeper understanding of their relative advantages and limitations. The importance of multiple surrogates and the complementarity between them can be illustrated with a brief review of the respective limitations and advantages of land types and one form of species data, point locality records.

Locality records generally look best from a distance. Across continents decisions about the appropriate grid size for display and analysis of records strike a balance between gaps in observations at fine scales and loss of resolution at coarse scales (Fjeldså et al. 1999; Brooks et al. 2001). Closer inspection reveals spatial biases related to access. At scales relevant to conservation decisions on the ground, most locality records map roads, rivers, airfields, field stations, and towns (Freitag et al. 1996; Maddock & du Plessis 1999; Richardson & Funk 1999; Brooks et al. 2001). Conservation areas based on these data can be caricatured as circles around field stations. More intensive surveys are only a partial solution because they reduce spatial biases but do not eliminate them. Even in 1995, although the region was one of the most thoroughly surveyed in Australia, most of the 5,500 flora survey sites in northeastern New South Wales still marked out the network of minor roads (Brown et al. 2000). Biases aside, maps of locality records also contain many false negatives (unsurveyed areas where species might occur [Freitag et al. 1996]). A generous estimate of the total extent of the 1995 flora survey sites in northeastern New South Wales is about 500 ha or 0.01% of the region’s 4 million ha of native vegetation (from data in Brown et al. 2000). Predicting distributions, although important in addressing these spatial limitations (Nicholls 1989), is not a panacea. Accuracy can be affected by biases in locality records (Kadmon et al. 2004) and, even in well-surveyed regions, varies enough that some species cannot be reliably modeled because of insufficient records, inadequate data on explanatory variables, or both (Pearce et al. 2001).

The well-known taxonomic bias of locality records toward vascular plants and vertebrates is another limitation. The tiny proportion of the landscape within survey sites is generally searched only for a small proportion of the species likely to be present. Reasons include taxonomic limitations (with most species not described [May 1999]), limited resources, and the taxonomic training and
### Table 1. Biodiversity surrogates for conservation planning

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<sup>a</sup>Surrogates comprised the two composite data sets used in 1998 for negotiation and design of 323,000 ha of new strict reserves in the upper and lower subregions of northeastern New South Wales. Methods and details are at http://www.affa.gov.au (Industry Development, Regional Forest Agreements, Regions, New South Wales: North East [Upper and Lower], Environment Report). Numbers of species and forest ecosystems given here have been reduced to account for those occurring in both subregions.

<sup>b</sup>See Gaston (1994) for review and Stattersfield et al. (1998) for example.

<sup>c</sup>See Donald and Fuller (1998) for review.

<sup>d</sup>See Margules and Austin (1994) and Freitag et al. (1996) for discussion of limitations.

<sup>e</sup>See Robertson et al. (1995) and Boshoff et al. (2001) for two approaches to deriving density data.

<sup>f</sup>See Boshoff and Kerley (2001) for an example.

<sup>g</sup>See Nicholls (1989) for rationale and example and Guisan and Zimmermann (2000) for review.

<sup>h</sup>See Rouxe and Sheard (1981) for discussion and Couling and Heijns (2001) for example.

<sup>i</sup>See Mackey et al. (1988) and Lugo et al. (1999) for examples and supporting arguments.

<sup>j</sup>See Belbin (1993) and Faith and Walker (1996) for two different approaches.

<sup>k</sup>Processes including population dynamics, interspecific interactions, patch dynamics, migration, adjustment to climate change, and ongoing diversification of lineages.

<sup>l</sup>Biophysical features associated with processes of interest. Examples are soil interfaces (Rouget et al. 2003), areas of climatic stability (Fjeldsà et al. 1999), drought refugia, migration staging areas, and dispersal barriers.

<sup>m</sup>Criteria include size and connectivity of reserves (Noss et al. 1997). When applied qualitatively, these are expressed as preferences (e.g., bigger is better).

<sup>n</sup>Quantitative parameters for criteria. For species persistence, these can include spatial variation in habitat suitability, estimates of population density and permeability of landscapes to movement, and targets for effective population sizes. See Carroll et al. (2003) and Akcakaya et al. (2004) for examples. Other processes such as patch dynamics can be handled quantitatively if the key parameters of their spatial requirements can be identified.

**prefences of field biologists. The ability of adequately recorded taxa to serve as surrogates for others is uncertain. The voluminous literature on taxonomic surrogacy has produced mixed results, largely because of the variety of intertaxon comparisons, study regions, geographic scales, and testing methods (Margules & Pressey 2000).**

Even with these limitations, locality records play a vital role in conservation planning by improving information on the natural history of species, providing direct information on their distributions, and contributing the raw material for spatial predictions beyond survey sites. Raising the general standards of locality records to or beyond those in northeastern New South Wales will take plenty of money and time, even for vertebrates and vascular plants. More money and time will be needed to extend the taxonomic breadth of locality data. Meanwhile, conservation decisions have to be made.

Land types are geographic entities defined in many ways for many reasons (Rowe & Sheard 1981), including interpretation of biodiversity patterns. Conservation planners sometimes use land types not developed specifically for biodiversity. Even so, most land types reflect variation in abiotic factors and/or characteristics of vegetation known to influence the distribution of species and used in predictive models (Nicholls 1989; Guisan & Zimmermann 2000). A key difference is that models are used to select and weight explanatory variables for individual species in the minority that are described taxonomically and can be predicted with acceptable accuracy. Classification of land types, in attempting to define geographic
subdivisions that are relatively homogeneous in characteristics of interest (Pressey & Bedward 1991), identifies gradients and discontinuities that are seldom tuned to particular species. Planners therefore use land types as generalizations about environmental variation assumed to be relevant for at least a proportion of the species lacking descriptions, adequate point records, or reliable models. A similar rationale motivates the use of regional boundaries as a stratification for conservation planning (Brunckhorst et al. 1998; Olson & Dinerstein 1998).

In using land types to compensate for the limitations of data on species, conservation planners do not assume that land types alone are adequate biodiversity surrogates. Studies showing that rarer species are missed by reservation of land types (Kirkpatrick & Brown 1994; Lombard et al. 2003) do not, therefore, ring alarm bells; instead, they confirm the expectations of earlier work that emphasized the complementarity of data on land types and species (Noss 1987). The use of land types in conservation planning does assume that they are better than random at guiding reservation to represent biodiversity that is undescribed, unsurveyed, or unpredicted. How good is this assumption? It depends on the taxa and land types analyzed, the geographic scale, the study region, and the choice of test. Methods for testing environmental surrogates (Margules & Pressey 2000), like those for taxonomic surrogates, are diverse and still developing, but there are many fewer case studies from which to generalize. Brooks et al. cite only one (negative) study (Araújo et al. 2001). Others are more encouraging (Wessels et al. 1999; Pharo & Beattie 2001; Mac Nally et al. 2002; Oliver et al. 2004).

How good should species data be before conservation planners forget about land types? When biodiversity experts in northeastern New South Wales prepared for negotiations on further reservation in 1998, they combined improved mapping of land types with augmented species surveys and extensive modeling to produce composite data sets (Table 1). Not many regions on the planet had denser and more comprehensive species data at that time, and most still do not. More generally, when planners are confident that locality records and predictive models adequately represent the distributions of all currently known and undescribed species, they could safely dispense with land types. Some of the resources needed to improve biological data to that stage could be used to improve other biodiversity surrogates, and the balance between those objectives remains unresolved. Until then, land types provide consistent frameworks for conservation planning that are likely to be complementary to other surrogates and that are relatively easy and inexpensive to derive and revise. Information on spatial turnover of species has the potential to improve the delineation of land types by identifying previously overlooked environmental and geographical gradients that can be interpreted remotely (Muster 2002; Tuomisto et al. 2003).

The frequent arbitrary nature of percentage targets (Brooks et al.) is a serious concern, but is not a reason to abandon land types. If it were, the frequent use of simplistic single-occurrence targets (reviewed in Rodrigues & Gaston 2002) would argue against the use of species data. Recent targets for land types involve defensible relative percentages, although the “right” absolute percentages remain elusive (Pressey et al. 2003). Questions about which specific parts of land types should contribute to percentage targets (Brooks et al. 2004) are often solved in practice by the spatial constraints imposed by other types of surrogates in composite data sets. Whether or not this happens, land types provide an environmental and geographic stratification for reservation that is likely to improve the representation of biodiversity.

I have focused on locality records and land types, but neither of these surrogates tells planners much about promoting the persistence of biodiversity processes. Planning for biodiversity pattern might favor some processes incidentally but will tend to fail those that operate over large areas and particular spatial configurations (Pressey et al. 2003). This is serious because species are generated and sustained by dynamics operating across spatial and temporal scales. Systematic conservation planning has been dominated by considerations of pattern (Cowling et al. 1999), but is now addressing process more decisively than acknowledged by Brooks et al. The emerging focus on process combines quantitative planning methods with previously disparate lines of work, including population dynamics, metapopulation modeling, landscape ecology, and evolution. There are at least three practical, complementary approaches to planning for biodiversity process (Table 1). Approaches to quantitative design, involving targets for configuration or effective population sizes in which configuration is implicit, promise to become more influential in conservation decisions.

Systematic conservation planning is more about explicitness than objectivity. Uncertainty and subjectivity pervade the process (Margules & Pressey 2000). Testing and comparison of surrogates are sufficiently undeveloped that decisions about data involve not just dispassionate appraisal but also the histories and preferences of conservation planners. Many experienced planners believe that composite data sets are the best approach. As one of them, I share the hopes of Brooks et al. about a resurgence of interest and investment in taxonomy and natural history. This will be an important part of a larger strategy for better understanding and protecting biodiversity.

**Literature Cited**


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