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The earth is experiencing a wave of extinctions possibly unprecedented in recent evolutionary history. The ecological advantage conferred upon humans by the evolution of reason and language has enabled us to obtain competitive dominance over all other species and expand our realized niche to include components of almost all habitats on the planet. We have simplified complex and diverse natural systems and processes for the production of food and shelter, leading to massive extinctions of species (Taylor 2004). The previous six mass extinctions caused by cataclysmic events decimated life on the planet and the current wave shows signs of approaching this level. These previous waves of extinction occurred over millennia; brief in geological time but long in ecological time. Life forms had the opportunity to adapt and evolve. Many species became extinct, but many others prospered and new species evolved. The current wave is not yet 300 years old; a brief aberration even in ecological time. Though humans have undoubtedly changed ecological processes and caused extinctions for longer than that, it was not at the current scale. Perhaps what is most troubling about the current wave of extinctions is that, unlike previous events, it is taking place because of a type of irreversible transformation of the land and even the oceans that is likely to make them unsuitable as habitat for many forms of life into the distant future.

The task for conservation biology is to halt the current extinction wave and chart a course for a future which includes biological diversity not only for its direct contribution to human welfare as a resource, but also because it appeals to important human values (Norton 1987; Sarkar 2005). In that future, nature conservation must exist alongside nature exploitation, not instead of it. Better still, the protection of biodiversity should become integrated into natural resource management that has sustainable livelihoods as an equally important goal. The protection of biodiversity will not win out in direct competition with other needs and aspirations of people (Rosenzweig 2003). Indeed, the research frontiers

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in conservation biology clearly must link a biophysical understanding of the world around us with a socio-economic understanding of what drives decisions on natural resource use, as well as an understanding of natural resource governance mechanisms and the institutional arrangements that societies use to regulate the exploitation of natural resources. This task of planning for the future is made even more complex by global environmental changes, especially climate change, the detailed consequences of which we can only dimly foresee, and even that only for a very few species. We will return to this issue of linking socio-economic goals with biodiversity conservation in Chapters 5 through 9, especially Chapters 7, 8 and 9. The focus in Chapters 2, 3 and 4 is on methods for deriving biodiversity data sets suitable for use in conservation planning.

1.1 Conservation area networks

Central to the conservation of biological diversity is the establishment of networks of conservation areas which are managed to minimize the risk of extinction. These are often called protected areas, but we use the term conservation areas to include all areas that perform a conservation function, whether they are strictly protected or not (Sarkar 2003). These are the priority areas for the allocation of scarce biodiversity management resources. *Systematic conservation planning* consists of the use of specific protocols to identify such priority areas and separate them from processes which threaten their persistence. Identifying and securing biodiversity priority areas will not in itself protect biodiversity, but networks of priority areas in each region of the world should form the framework upon which other conservation actions build.

The key concept underpinning systematic conservation planning is complementarity. Complementarity is a measure of the contribution an area in a planning region makes to the full complement of biodiversity features: species, assemblages, ecological processes, etc. As discussed below and in later chapters, we can never know the full complement of biodiversity features; consequently we must use partial measures or surrogates and we must set goals for the representation of these surrogates. Complementarity can then be measured by the contribution an area makes to the conservation goal. For example, pretend that the conservation goal is to represent at least one population of all of the vascular plant species that occur in a planning region in a minimal set of biodiversity priority areas. The first area we choose might be the one with the most species. The next area we choose will be the one with the most

species that were not represented in the first area and the next area after that will be the one with the most remaining unrepresented species, and so on until all species are represented. There are two important points to note about this process. The first is that as areas are added to the set, the contribution that remaining areas make to the goal changes. This is because some of the species in those areas may have already been contributed by other areas previously selected for the set. The second is that areas with the highest complementarity will not necessarily be those with the most species. In this example, complementarity is measured at each step as the number of unrepresented species. If an area has few species, but they do not occur widely in the planning region, it may have higher complementarity than an area with many species that are widespread throughout the planning region (depending on which species occur in areas that have already been selected). Thus, species richness cannot be used to measure complementarity. There are many variations on the simple area selection process outlined above and many of these are described later in the book. As it turns out, for example, it is often more economical to begin the area-selection process by selecting that area with the rarest species on it and then the one with the highest number of rarest unrepresented species, and so on until all species are represented (Sarkar *et al.* 2002). Vane-Wright *et al.* (1991) coined the term “complementarity,” but it appears to have been independently discovered at least four times (Justus & Sarkar 2002), twice in Australia (Kirkpatrick, 1983; Margules & Nicholls 1987; Margules *et al.* 1988), in the United Kingdom (Ackery & Vane-Wright 1984) and in South Africa (Rebelo & Siegfried 1990). Complementarity is described in detail in Chapter 5 (Section 5.4).

Systematic conservation planning is usually implemented with software tools using digital georeferenced data sets and area selection algorithms. Algorithmic approaches ensure that planning exercises are standardized and repeatable – this is the most important sense in which conservation planning is systematic. However, this numerical computer-based approach has led to some criticisms, in particular that large data sets are needed to run the software (Redford *et al.* 1997; Prendergast *et al.* 1999), that local expertise is not consulted and incorporated (Redford *et al.* 1997) and that they are expensive and money could better be spent acquiring reserves (Prendergast *et al.* 1999). Any reading of the conservation planning literature will show that these criticisms are unjustified. There is no substitute for local expertise, which should always be consulted. In fact, systematic conservation planning works best when local experts are the ones doing the planning. The methods are

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inexpensive, can be used without computers and with minimal data. It is just that computers speed up the process and enable large data sets to be used efficiently when they are available. Also, the recent growth in biological and environmental databases, in part due to rapid improvements in remote sensing and modeling, means that there is now almost no terrestrial region on Earth with so little data that systematic planning tools cannot improve policy formulation.

Where population density is high and land is in private ownership, conservation agencies have, in the past, found it necessary to take opportunities whenever and wherever they occurred believing that when land is privately owned there is no opportunity for a more strategic systematic approach. However, if conservation NGOs (non-government organizations) such as The Nature Conservancy in the United States and Bush Heritage in Australia are purchasing land, as they currently are, they need to know what the genuine biodiversity priorities are or they will not make best use of their funds. If they purchase the cheapest land or just whatever becomes available they will not make the contributions to biodiversity conservation that they could make, and which are sorely needed. As several past studies have shown, ad hoc conservation-area selection has been remarkably cost-ineffective (Pressey *et al.* 1996; Pressey & Cowling 2001). This is because conservation areas all over the world contain a biased sample of biodiversity from ecosystems and habitats that were selected because they are remote and inaccessible, or they are unfit for alternative uses such as agriculture, which means they are cheap. The identification of biodiversity priority areas should be based on good science and sound common sense and not on such ad hoc grounds. Some priority areas will inevitably be expensive. But even if they prove too expensive to buy, we should know about them. They might become subject to policy change and/or management agreements in future. Planning tools support that process. They do not replace it. Conservation planning is a dynamic iterative process and these planning tools are designed for decision support to help local experts identify good policy options, not formulate policy on their behalf.

In the United States, Texas provides a striking example of the potential continued inefficient use of resources when conservation policy is not determined by systematic planning (Alford 2005). In 2003 tax exemptions given to landowners for maintaining or restoring their land as wildlife habitat were estimated to be over \$1 billion. Land management options included habitat control (such as clearing juniper overgrowth), erosion control, control of predators (such as feral cats or dogs), providing water

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resources, providing supplemental food for wild animals, growing nutritional plants, providing animal shelters (such as leaving dead trees for bird nests) and conducting animal censuses. There is no coordination among efforts of individual landowners. Although those who receive the tax break have to formulate explicit wildlife management plans, these do not have to be filed with any state agency, but only be appraised by county officials who need not have any wildlife expertise. There is no monitoring to determine whether plans are successful in promoting wildlife. In some cases anecdotal evidence suggests no change: the deer population growth rate remained the same before and after the policy was introduced in 1995. In other cases, there may be some success: while the decline of wild turkey populations was 54 % in the five years preceding 1995, it was only 9 % in the subsequent five years. Given the lack of coordination and monitoring it is impossible to evaluate what is being conserved or restored and how effectively. Perhaps more importantly, there is no prioritization of land on the basis of its potential contribution to a biodiversity goal, given appropriate management, and therefore no prioritization of the landowners who should receive these tax breaks. This is the type of problem that systematic conservation planning aims to mitigate.

In some places where land is privately owned and there are few opportunities for biodiversity protection using traditional approaches, market-based instruments are being tested and evaluated. In a recent pilot study in the highly fragmented landscape of the Western Australian wheatbelt called “Auction for Landscape Recovery” (Gole *et al.* 2005), landowners were invited to nominate portions of their holdings and propose management actions for these portions and then bid for funds to implement those actions. Bids were judged on two criteria. One was the contribution the nominated areas made to a regional biodiversity goal, i.e., complementarity, and the other was the likely success of the proposed management actions. This is an example of how the systematic approach is beginning to find wide application in the development and implementation of conservation policies and practices.

1.2 What do we mean by biodiversity?

Biodiversity consists of the biological variety bequeathed to us by evolutionary processes over millennia. It is what we have to conserve if we do not want to squander this inheritance. The biological realm is characterized by variability and complexity at every level of structural, taxonomic and functional organization. The term “biodiversity” was introduced in

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the mid 1980s as a contraction of “biological diversity” to refer to the totality of this variability (Takacs 1996). From a biological perspective, all such diversity is important because it provides the raw material for evolution. Ideally, all of it should be conserved.

Ecosystem processes such as nutrient cycling, the movement of water and energy and the dispersal of propagules are necessary for the persistence of biodiversity, but in this book we will restrict the definition of “biodiversity” to biological patterns such as the distribution patterns of biological entities and dynamic behavioral phenomena such as migrations. Two hierarchical schemes are used for the classification of biological entities (Sarkar 1998). One is a spatial (or ecological) hierarchy starting with molecules and macromolecules, then cell organelles, cells, individuals, populations and metapopulations, communities, ecosystems and ultimately the biosphere. The second is a taxonomic hierarchy from alleles to loci, linkage groups, genotypes, subspecies, species, genera, families, orders, classes, phyla and kingdoms. Both hierarchies reflect evolutionary history and are constrained by evolutionary mechanisms. Since the future for biodiversity is dependent on evolutionary processes, understanding the relationships between these two hierarchies and phylogeny is also necessary for the formulation of successful conservation strategies. In practice we never know these relationships fully but must proceed anyway, accepting that conservation priorities and strategies will change as knowledge accumulates.

There are two points to note about both hierarchies. The first is that they are not clean and tidy in the sense that biological entities fall neatly into well-defined classes without exception. The second is that classes at all levels in each hierarchy are heterogeneous: there is variety within each class at every level. The variety of viable biological configurations at all hierarchical levels is extremely large, currently unknown, and probably unmeasurable. This second point is almost trivial, but emphasizes the complexity of biodiversity. Almost any two populations, even of the same subspecies, differ in their genetic profiles. Except for some clonal organisms, almost any two individuals of the same species have different genotypes. There are virtually no two identical ecological communities, and so on. The first point is equally important when we consider how to describe biodiversity. While some entities such as organelles and cells are reasonably well defined, examples such as fungi, symbionts and clonal organisms show that even biological “individuals” are not always precisely defined. Asexual species are notoriously difficult to define and even sexual species, usually defined by the ability to interbreed and produce

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offspring (Mayr 1957), cause problems, for instance in the case of so-called “ring species.” In such species, “rings” are composed of circular geographically successive populations that evolved from a single ancestral population. Each pair of neighboring populations can interbreed except for the two terminal ones which, by definition, should now be regarded as different species. Examples include the greenish warbler, *Phylloscopus trochiloides*, populations surrounding Tibet and the salamander, *Ensatina eschscholtzii*, surrounding the central valley of California (Wake 2001).

As we will discuss in detail in Chapter 2, biodiversity, described in such a complex way, is impossible to estimate or quantify in the field; it is not an operational concept (Sarkar 2002; Sarkar & Margules 2002). In practice, the concept of biodiversity has often been simplified to refer to diversity at three levels of organization: genes (alleles), species and ecosystems (e.g., Meffe & Carroll 1994; Vermeulen & Koziell 2002 and see Lindenmayer & Burgman 2005 for a detailed discussion of biodiversity at different organizational and functional levels). Any such definition is necessarily partly conventional. However, that does not mean that it must be arbitrary. Such a definition is conventional because we know that it does not include all of what we mean by diversity. For instance, the definition of biodiversity as diversity of alleles, species and ecosystems excludes interspecific hybrids. More importantly, it excludes biological phenomena such as the annual migrations of north American monarch butterflies (*Danaus plexippus*), which may be endangered and deserving protection (Brower & Malcolm 1991; see Chapter 2). Nevertheless, the definition is not arbitrary because focusing conservation efforts on genes, species and ecosystems will protect much of the diversity within species, taxonomic diversity at levels higher than species and many communities (see Chapter 2).

As Austin and Margules (1986) first explicitly pointed out, the concept of biodiversity must be operationalized through the use of “surrogates,” features of the landscape such as the presence of species or other taxa, habitat type, etc., that can *in principle* be quantified and assessed in the field. Surrogates that are used to represent biodiversity in its full generality we call “true” surrogates. For example, “biodiversity in general” is often thought to be adequately represented by species diversity. The identification and adoption of true surrogates operationalizes the concept of biodiversity for systematic conservation planning. But, as we noted earlier, any choice of a true surrogate set is partly conventional. Moreover, since general biodiversity cannot be quantified, the success of a true surrogate set in representing general biodiversity also cannot be quantitatively

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assessed. What we must do is give good reasons for what we choose to be the true surrogate set in a given planning context. Quite often, we are not in a position even to assess the full distribution of a true surrogate set. For instance, if we take all species within even well-documented taxa such as mammals and birds, there are many areas of the world for which we will not have all the distributional data we need (see Chapter 3). In many situations we may not even be able to model such distributions completely (see, however, Chapter 4). We then have to use “estimator surrogates” for biodiversity; those features which we can *in practice* quantify and assess in the field. The question whether an estimator surrogate set adequately represents a true surrogate set is amenable to quantitative assessment, as we discuss in detail in Chapter 2.

1.3 Systematic conservation planning

Systematic conservation planning is a structured step-wise approach to mapping conservation area networks, with feedback, revision and re-iteration, where needed, at any stage. Though prioritizing new areas for conservation is central to systematic conservation planning, this process does not ignore or throw away, literally or metaphorically, existing conservation areas or networks. In almost all regions of the world, there is a heritage of conservation areas, which more than likely have been accumulated opportunistically and are therefore unrepresentative of regional biodiversity. Systematic planning normally accepts these imperfect networks and maps onto, or builds on, what already exists, with the objective of transforming them into better networks. In addition, analyses of the extent to which existing conservation areas contribute to regional biodiversity goals might provide options for future rationalization. For example, it may be possible to trade existing conservation areas making low contributions to biodiversity representation for new areas that would make higher contributions.

Margules and Pressey (2000), Groves *et al.* (2002), Cowling and Pressey (2003) and Sarkar (2004) identified several distinguishing characteristics of systematic conservation planning. In the first place, it requires the identification and engagement of stakeholders. The people who rely on or influence the use of natural resources in the planning region must be party to the planning process or it will fail. This point deserves emphasis: far too often conservation policies have failed because people who have a stake in the land or water that is the subject of those policies have not been consulted (Sarkar 1999; Justus & Sarkar 2002). Sometimes,

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when the planners have come from international organizations or distant national governments, this has led to accusations of paternalism, colonialism and other ideological remnants of the Northern colonial era (Guha 1989). Recent reports have documented the creation of “conservation refugees,” people involuntarily displaced by conservation policies such as the creation of reserves and the abrogation of traditional resource-use rights such as hunting (Dowie 2005). Throughout this book we emphasize the point that systematic conservation planning must be viewed as part of social policy which explicitly recognizes and addresses these issues.

Systematic conservation planning also requires that clear choices are made about how biodiversity is to be measured and mapped. This is partly the surrogacy issue raised in Section 1.2 above and discussed in detail in Chapter 2. But it also includes collecting data, building and managing data sets and databases, and carrying out data treatments to derive the chosen surrogates. These issues are discussed in Chapters 3 and 4. Similarly, clear goals have to be set and preferably then translated into explicit measurable conservation targets. Next, the extent to which conservation goals have been met in existing reserves or priority areas must be recognized so that the simple explicit methods described in Chapter 5 can be used to identify new priority areas to complement existing conservation areas in achieving the set goals. Finally, explicit criteria for implementing planning choices on the ground have to be formulated and applied, objectives for individual conservation areas have to be set, the achievement of those objectives has to be monitored and appropriate management actions have to be taken to ensure continued contribution of individual conservation areas to overall conservation goals. This book addresses all of these issues, though Chapter 6 is the only one to focus on the last, while also addressing issues of threat and vulnerability at all stages in the process.

Table 1.1, modified from Margules and Pressey (2000) and Sarkar (2004), describes this overall approach in eleven more-detailed stages. The first stage is stakeholder engagement. Stakeholders will often be local residents, farmers or pastoralists, but can also include government agencies responsible for managing natural resources such as water and forests, non-government organizations (NGOs), including conservation NGOs, both local and global, and industries, e.g., mining and agri-businesses. Stakeholders include all those people who have decision-making powers over a region, all those who will be affected by the conservation plans that are formulated, those with scientific or other expertise about the region and those who may commit resources for conservation planning and implementation. For a conservation plan to be successful, the involvement of

