CHAPTER 6

SYSTEMATIC CONSERVATION PLANNING: PAST, PRESENT AND FUTURE

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6.1 INTRODUCTION

In general, the best farming land is the first to be cleared. In long-settled regions of the world, this has meant that by the time biodiversity conservation became a social priority, a very much non-random subset of the 'original' habitat types has been available for conservation management. Historical decisions on where protected areas were located were rarely based solely (if at all) on scientific assessment of biodiversity value or biogeographical representativeness. Rather, these decisions were based on other factors, such as the suitability of alternative land uses, availability of an area for conservation management, scenic beauty, and recreational values (Chapters 2 and 5; Pressey *et al.*, 1993; Margules *et al.*, 2002; Gaston *et al.*, 2008).

This has resulted in a legacy of protected areas that are biased towards habitats that are generally not threatened, such as dry, infertile or steep habitats (Pressey *et al.*, 1993, 2002; Soulé & Terborgh, 1999). For example, five per cent of the Earth's entire terrestrial protected area (972,000 km^2) is the Greenland National Park, which contributes little to biodiversity conservation as it contains mostly ice (Chape *et al.*, 2003; WDPA Consortium, 2006).

This form of bias can be demonstrated quantitatively, as shown in a regional-scale analysis by Pressey *et al.* (2002). In their paper, a part of their analysis was an assessment of protected area coverage as a function of slope and fertility in the northern eastern region of New South Wales, Australia (Figure 6.1). This analysis highlights the bias often found in reserve systems, with the steepest slopes and the soils of lowest fertility being far more represented in the reserve system than the converse. There are examples like this found on all inhabited continents on Earth (Brooks *et al.*, 2004; Rodrigues *et al.*, 2004a; Joppa & Pfaff, 2009).

The threat to biodiversity as a result of habitat loss and change over the second half of the 20th century led to an increased interest in enhancing the coverage and representativeness of the protected areas network (McNeely, 1994). The efforts taken towards these goals at a global and regional scale gained impetus from the development of the IUCN biogeographical regions (Dasmann–Udvardy) framework discussed in Chapter 5. This coarse-scale analysis did not, however, offer guidance on designing networks within regions at the scale of landscapes.

The first efforts to take a more scientific approach to designing protected area networks were based on the theory of island biogeography (e.g. Chapter 8; MacArthur & Wilson, 1967; Diamond, 1975a). The rationale followed was that nature reserves and other protected areas can be considered forms of

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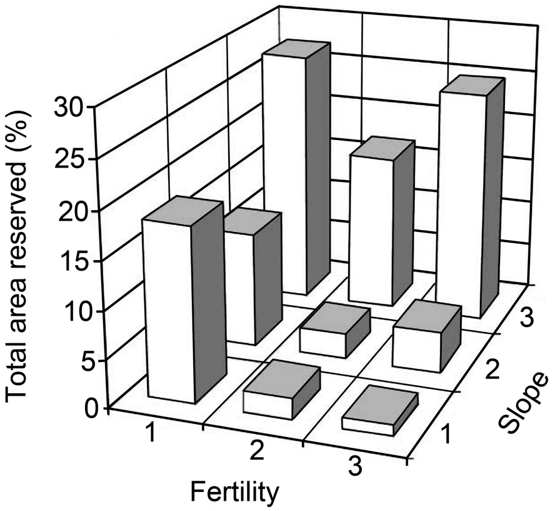

Figure 6.1 Assessment of reserve coverage as a function of slope and fertility in the northern eastern region of New South Wales, Australia. The vertical axis represents the percentage of the total area of each broad environmental unit captured in reserves in the region. The other two axes are measures of slope and soil fertility, with the lower numbers (i.e. 1) indicating flatter slope and lower fertility and the higher numbers (i.e. 3) indicating steep slopes and high fertility, respectively. From Pressey *et al.* (2002).

habitat islands, isolated from other reserves by anthropogenically transformed habitats (sometimes named the 'matrix') that are generally unsuitable for the species of conservation concern. These early efforts were guided by basic ecological principles, such as that bigger protected areas are better than smaller ones because they are likely to contain more species (Diamond, 1975a).

Initial approaches to systematic conservation planning were developed based on simple scoring systems, using criteria such as species richness or number of endemic species, to provide an indication of how new areas might contribute to protected area networks if they were chosen (Margules & Usher, 1981; Smith & Theberge, 1986). The integration of these basic principles into conservation planning was a useful first step, but both conservation scientists and practitioners have since criticized their simplicity (e.g. Simberloff & Abele, 1982). For example, it has been argued that the number of species contained in any single area alone should not determine its priority. More important is how any one area complements the existing protected area network, along with a suite of wider landscape conservation issues (Moilanen, 2008). Similarly, simply relying on the idea that 'a big reserve is better' is not useful for making planning decisions in landscapes also required for other human uses (e.g. agriculture, mining).

Systematic conservation planning has evolved as a discipline to enhance the efficiency of protected area network design and, through creating alternative proposed networks, to allow scientists and stakeholders to better engage with the complexity of multi-sectoral spatial planning across landscapes and within regions. Therefore, in general, the tools discussed in this chapter are typically employed at finer scales of analysis than the global/regional approaches discussed in Chapter 5 (but see Venter *et al.*, 2009; K.A. Wilson *et al.*, 2009).

The 1980s saw the first attempts to use detailed biogeographical information and selection algorithms in the design of protected area networks (Kirkpatrick, 1983). The field of systematic conservation planning has grown significantly since. It has influenced conservation planning by some of the major environmental organizations such as The Nature Conservancy (Groves *et al.*, 2002) and Conservation International (Myers *et al.*, 2000), and it has shaped policy legislation and conservation in both terrestrial (Knight *et al.*, 2006; Kremen *et al.*, 2008) and marine (Davis, 2005; Fernandes *et al.*, 2005) environments. It has featured in hundreds of peer-reviewed papers (Pressey *et al.*, 2007) and in recent books (e.g. Margules & Sarkar, 2007; Moilanen *et al.*, 2009).

In this chapter we review the key principles of systematic conservation planning and some of the current decision support tools available to assist conservation planners in making decisions. Decision support tools are information systems intended to help decision-makers compile and analyse data to help solve conservation problems. The increasing power and ease of use of such computer-based systems in the last two decades has opened up exciting possibilities for applications to conservation planning. We illustrate some of these applications from contemporary case studies, providing examples of the use of different techniques and tools. The field of conservation planning is rapidly changing, and we discuss advances (and future challenges) in systematic conservation planning at the end of the chapter.

6.2 WHAT IS SYSTEMATIC CONSERVATION PLANNING AND WHY USE IT?

The science of systematic conservation planning is concerned with the optimal application of spatially-explicit conservation management actions to promote the persistence of biodiversity and other natural features *in situ* (Margules & Pressey, 2000; Margules & Sarkar, 2007). It involves a transparent process of setting clear goals and objectives, and of planning conservation actions that meet them (Bottrill & Pressey, 2009).

A fundamental characteristic of systematic conservation planning is the principle of complementarity (Kirkpatrick, 1983). Since the first publications in this research field, systematic methods have identified systems of conservation areas that are complementary to one another in terms of collectively achieving objectives. Areas identified in this way will each contain, for example, different species or complementary portions of the required areas of different habitat types. As will be discussed further in this chapter, this represents a major improvement on the additive scoring procedures that were used extensively before the application of complementarity methods. Additive scoring approaches are incapable of dealing with the fundamental notion of building a system of protected area where the value of the whole system is not the same as summing the values of the separate protected areas.

Systematic conservation planning has traditionally been applied to design strict protected area networks (those areas that are managed for conservation values only, e.g. IUCN management categories I–IV; see Table 2.2). More recently, however, it has been expanded to include planning other types of conservation actions, such as stewardship payments or other land management, in space (Carwardine *et al.*, 2008) and time (Wilson *et al.*, 2007). Here we use the term 'protected area' loosely, in reference to any place where an action is applied for conservation purposes. We acknowledge that much of what is written in this chapter is focused on the literature behind planning protected area networks, but at the end of the chapter we provide examples of other forms of systematic conservation planning.

It should be noted that systematic conservation planning involves designing protected area networks based on clear objectives, as well as an understanding of constraints on where and how implementation can occur (Smith *et al.*, 2006). Constraints include factors such as the cost of acting in a particular area or the willingness of landholders to participate in a conservation initiative (Knight *et al.*, 2009).

Good systematic planning processes, as we will see, include input, information and values from a wide variety of stakeholders, incorporated within a transparent and inclusive process (Knight *et al.*, 2006; Bottrill & Pressey, 2009) in order to reduce conflicts between opposed interests. However, as in any such field, the approaches discussed herein have also spawned many analyses that are of largely heuristic rather than immediate practical value. This allows analysts to explore 'what-if' scenarios concerning future landscapes and climate surfaces or to undertake 'tests' of the effectiveness of existing protected area networks or schemes (e.g. Araújo *et al.*, 2004a,b, 2008; see Chapter 7).

6.3 CONCEPTS AND PRINCIPLES

6.3.1 Representativeness

An overarching goal of conservation is to ensure that there is no loss of biodiversity. As discussed in earlier chapters, representativeness is a fundamental principle in systematic conservation planning and refers to how well protected area networks contain representative samples of every feature of biodiversity that we aim to protect. Biodiversity features normally reflect some combination of genetic, species and community diversity. However, it is also important to consider the structure of habitats, e.g. the availability of coarse woody debris in temperate woodland, and ecological processes, such as fire dynamics in Mediterranean ecosystems.

It is often difficult for protected areas to achieve complete representation for two reasons:

1 in regions with high species compositional turnover over small distances, such as Mediterranean ecosystems (Judd *et al.*, 2008), a large proportion of the region will be required to represent all of the unique biodiversity features; and

2 even for the best studied regions, systematic data are lacking for some aspects of biodiversity.

This second problem has two elements, termed 'Linnean' and 'Wallacean' shortfalls (Chapter 4; Whittaker *et al.*, 2005). The Linnean shortfall refers to our lack of knowledge of how many, and what kind, of species there are. Almost two million species have had formal scientific names given to them, but this is still only a fraction of the total of all species (Groombridge & Jenkins, 2002). Estimates have been made that if the collection and description of new species were to continue at the current rate, it would take several thousand years to catalogue the world's biodiversity (Soulé, 1990). The Wallacean shortfall refers to our inadequate knowledge of the global, regional, and local distributions of the species that we know. Even for the best known taxa such as birds and mammals, and in the best studied regions, there are still huge gaps in our knowledge of distributions (Chapter 4).

6.3.2 Persistence (adequacy)

Having a representative protected area network does not ensure that biodiversity within the network will persist into the future. This is because although protected areas might contain a particular species or habitat type, the area might not alone be sufficient to ensure their persistence. Therefore, protected areas should ideally also be designed to maximize persistence. This can involve an analysis of viability requirements (Lande *et al.*, 2003); the configuration of protected areas, including dealing with issues such as connectivity and the permeability of the matrix (McIntyre & Hobbs, 1999; Lindenmayer & Franklin, 2002); and predicting what ecological processes are needed to sustain biodiversity (Soulé *et al.*, 2004).

While persistence is considered one of the most fundamental concepts of systematic conservation planning, exactly what constitutes adequate conservation is not well defined (Woinarski *et al.*, 2007; Watson *et al.*, 2008; Carwardine *et al.*, 2009). For example, is a conservation plan that gives every species a 75 per cent chance of persisting for 1,000 years adequate?

6.3.3 Efficiency

A simple way to ensure representativeness and persistence is to conserve everything. This is obviously impossible, and so some degree of compromise is necessary. If the impact of conservation actions on the rest of society is minimized, there is a better chance that the plan will succeed politically and socially and thus provide a platform from which to expand further actions.

The principle of efficiency is based on the idea that conservation planners should try to achieve biodiversity objectives for the least possible cost. 'Cost' here may reflect the financial cost of implementing and managing protected areas or the costs of lost opportunities for economic development (Naidoo *et al.*, 2006). It can also include other socio-economic considerations, such as the willingness of people to assist with conservation management, with the expectation that it is more cost-effective to do conservation where people are willing to act.

For example, take the matrix on the distribution of four species at five sites shown in Table 6.1. If you were to select the minimum number of sites to represent each species, the optimal combination would be sites 1 and 2 (at a cost of $25). However, when we take cost into account, the combination of sites that represents all species with the least cost is the set comprising sites 1, 4 and 5 ($11). By such consideration of cost, conservation planners are able to maximize the conservation 'return on investment' and hence make an efficient plan.

There is an increasing number of studies that provide evidence that incorporating financial constraints into conservation planning increases the likely biodiversity benefits for a given amount of money (Ando *et al.*, 1998; Naidoo *et al.*, 2006; Carwardine *et al.*, 2008). Other benefits from biodiversity conservation can be factored into such analyses, including ecosystem services – the benefits that humans derive from natural systems, such as clean air and water. By dealing with multiple measures of benefit, conservation planners may provide a more comprehensive evaluation of the returns from conservation investments.

Table 6.1 Matrix showing the distribution of four species at five sites.

	Site 1	Site 2	Site 3	Site 4	Site 5
Species 1	1	1	1	0	0
Species 2	0	1	0	0	1
Species 3	0	1	1	1	0
Species 4	1	0	0	0	0
Cost	$5	$20	$5	$4	$2

It is also possible to modify the cost of conservation by incorporating into the analysis the benefits obtained from the delivery of ecosystem services, such as the amount of carbon sequestrated or the reduced cost to filter water (Venter *et al.*, 2009). Such payment for ecosystem services has the potential to increase the support and resources available for conservation (Costanza *et al.*, 1997; Daily *et al.*, 2009).

6.3.4 Flexibility

Objectives can often be achieved in a number of alternative places, particularly when the distribution of biodiversity features is widespread. Moreover, proposed new conservation areas or networks must be accepted and implemented by planning bodies, which brings economic, political and social considerations to bear upon decisions. Therefore, a key principle of systematic conservation planning is flexibility.

A flexible conservation plan provides alternative solutions and assists planners to take account of opportunities (Knight & Cowling, 2007). This is because socio-economic constraints may not be fully understood and, in any event, may constantly be changing. For example, a piece of land with high conservation value might not initially be available for conservation management, but may later become available for sale, lease or other management intervention (McDonald-Madden *et al.*, 2008). Adopting a flexible plan also gives scope for sensible resolutions of resource/use conflicts.

6.4 DEVELOPING A SYSTEMATIC CONSERVATION PLAN

In this section we provide examples of how objectives can be set against the key principles outlined in section 6.3. The process of defining measurable objectives is one of the principal components of systematic conservation planning (Nicholson & Possingham, 2006). Defining objectives gives the planning approach transparency and a benchmark by which to evaluate progress towards goals. We discuss how all stakeholders (and not just planners sitting in academic or government institutions) need to be involved in the process of developing these objectives to ensure the plan is successfully implemented. We also provide two real-world case studies to help describe how each of these principles has been successfully integrated into an applied systematic conservation plan through the careful choice of conservation objectives.

6.4.1 Achieving representation

As discussed earlier, 'Linnean' and 'Wallacean' shortfalls in biogeographical data are highly problematic for any plan trying to achieve representation. Given such a deficit of knowledge and data on biodiversity, a partial measure of biodiversity is almost always used as a surrogate for the rest of biodiversity. To develop biodiversity surrogates, conservation planners must gather all existing data sets and determine which are fit for purpose. Decisions on which data sets to use will often be based on the likely effectiveness of the particular data set and biodiversity metric as a surrogate for other components of biodiversity for which we have no data or poor data.

However, the mere existence of a data set does not necessarily guarantee fitness for purpose (see examples in Chapter 4). For instance, where the underlying survey regime is too geographically biased, it could skew the selection of protected areas towards places that have been well-surveyed but which are not particularly biodiverse. In data-poor areas, one alternative is to use environmental surrogates (e.g. vegetation types) as a 'coarse filter', with the aim of capturing biodiversity attributes that are likely to correlate with the chosen data layers (e.g. Faith & Walker, 1996). A limitation of such approaches is that unless very fine-scale environmental data are available, 'fine filter' features indicative of resource hotspots, such as salt-licks, are likely to be missed, as may be the factors controlling the distributions of the subset of threatened and rare species (see: Araújo *et al.*, 2001, 2003, 2004a; Faith *et al.*, 2004a; Noss, 2004). Below are some examples of different theoretical approaches in developing surrogates for conservation planning purposes.

Species-based surrogates

A variety of criteria have traditionally been used to select species-based surrogates in systematic conservation plans. These surrogates have often been called 'indicator' species and there are a number of different types that have been used in past planning techniques (see Box 6.1).

Box 6.1 Some examples of species-based surrogates that have been used in systematic conservation planning approaches

Keystone species have a disproportionate effect on the ecosystem relative to their abundance (Mills *et al*., 1993; Paine, 1995). As such, they affect the types and abundance of many other species in a community. The identification and management of these species can be important in conservation planning (Fleishman *et al*., 2000). The keystone concept, although intuitive, has received criticism because it is not always clear whether ecological communities *have* keystone species – and even if they do, this may be hard to demonstrate quantitatively because of the complexity of community structure and environmental dynamics of many ecosystems (e.g. Power *et al*., 1996; Andelman & Fagan, 2000).

Figure B6.1a The African elephant (*Loxodonta africana*) plays a significant role in altering the vegetation structure and type throughout its range, and as such is considered a keystone species. Photograph courtesy of Peter Baxter.

Focal species are, in the present context, species that are most endangered by the threatening processes within a system (Lambeck, 1997; Watson *et al*., 2001). The logic of using a focal species is that if a conservation plan meets their minimum needs, they should capture the needs of all the other species in that system in relation to that particular threat. This approach has, however, been criticized by Lindenmayer *et al*. (2002), who have argued: (i) that it may be too difficult to identify species most affected by each threatening process because of a lack of data on all taxa, and (ii) that the approach is over-reliant on the untested assumption that protecting the most threatened species will inevitably protect those that that are less threatened.

Umbrella species are those species that are used as surrogates to represent the 'health' of an ecosystem or the distribution patterns of other species; or they are species that require such extensive resources for their conservation that many other species will be protected by default. Top predators are often used as umbrella species. There has been mixed support for the umbrella species concept in conservation planning. Andelman and Fagan (2000), in a study of umbrella species of the southern Californian sage-shrub community, found that selecting areas using umbrella surrogates performed barely better than a randomly selected set of species. However, Fleishman *et al*. (2001) have reported more positive results.

Figure B6.1b The hooded robin (*Melanodryas cucullata)* has been identified as a focal species in the woodland ecosystems of south-eastern Australia as it is highly threatened by habitat fragmentation and requires large woodlands remnants that are close together to persist (Watson *et al.*, 2001). Photograph courtesy of Mat Gilfedder.

Figure B6.1c The tiger (*Panthera tigris*) is often used as an umbrella species for conservation planning in countries such as India. Photograph courtesy of Liana Joseph.

Threatened taxa. It has been argued that conservation planning should concentrate on the needs of species currently endangered or threatened with extinction (Sarakinos *et al*., 2001; Conservation International, 2004). It is often less controversial to use these species as they should be of special concern for biodiversity conservation and, in some cases (and in particular regions), they may be relatively well known (and their locations mapped) (Gaston *et al*., 2002; Bottrill *et al*., 2009).

Figure B6.1d The marine iguana (*Amblyrhynchus cristatus*) is found only on the Galapagos Islands. Uniquely among modern lizards, this animal lives and forages in the sea. It is threatened by predation by exotic species. Photograph: James Watson.

Phylogenetic difference. Some ecologists argue that species that are more phylogenetically distinct contribute more to the total genetic and morphological diversity and so should be given priority for protection (Weitzman, 1993; Faith *et al*., 2004b; Faith, 2009; and see Box 7.1). It has been suggested that a good way to generate a plan using this criterion is to use higher taxa (i.e. genus, family) instead of species in the planning process (Mooers, 2007).

Figure B6.1e The little known Guianan cock-of-the-rock (*Rupicola rupicola)* is a spectacular, phylogenetically distinct member of a two-species family inhabiting northern South America. Photograph: James Watson.

Associated with the use of species surrogates (in particular using umbrella species, keystone species and focal species) to achieve representation is the concept of achieving functional or ecological redundancy, which refers to the situation where there are multiple species within an ecosystem that play similar ecological roles. Achieving functional redundancy is seen as important objective, because the consequences of losing all of the species that perform a particular ecological function within an ecosystem (e.g. losing all of the algae feeders from a coral reef) could result in a dramatic shift to a lower biodiversity system. Thus, the amount of functional redundancy in a system is of considerable importance in terms of retaining ecosystem integrity (Walker, 1992).

This approach requires that species with similar ecological roles (termed functional groups) are identified, along with the key processes that maintain ecological integrity. Conservation efforts can then be aimed at maintaining a full suite of functional groups, and functional groups with little or no redundancy should be prioritized for conservation action (Walker, 1992).

The functional redundancy concept has been enthusiastically applied to the problem of conserving coral reef ecosystems (Steneck & Dethier, 1994; Bellwood *et al*., 2004). Coral reefs are particularly prone to collapsing from a high diversity system to a low diversity system dominated by algae and a few species of fish (Scheffer *et al*., 2001) although, until recently, the causes of such phase shifts were poorly understood.

Recent comparative analysis of functional groups in coral reefs from around the world strongly suggests that high species diversity provides the potential for functional redundancy (Bellwood *et al*., 2004). Hence, Caribbean reefs that are lacking several critical functional groups, or have groups represented by a small number of species, have been particularly prone to phase shifts to low diversity systems (Scheffer *et al*., 2001). However, it should be noted that even in high diversity coral reef systems, such as the Great Barrier Reef in Australia, there are still some functional groups with low redundancy (e.g. that are represented by a small number of species) (Bellwood *et al*., 2003).

Despite the support of concepts such as functional redundancy by some systematic conservation planners, the overall level of support for species-based surrogates has been variable (Beger *et al*., 2003, 2007; Faith *et al*., 2004a). Since it is unlikely that it will ever be possible to measure the true variation of biodiversity within or between regions, or the overall functional role played by all species in a region, the true effectiveness of a species-based surrogate is indeterminable. Moreover, the underlying assumption that the needs of a particular surrogate group of indicator species will ensure the long-term persistence of all of biodiversity may never be true as all individual species, have, by definition, evolved to have their own specialized needs (see discussion on individualism in Chapter 3) and these needs will never be captured by a surrogate.

Because of this, many recent conservation planning exercises have used sets of species covering entire taxa (i.e. all birds, all mammals, etc.), or assemblages of species in a given area (e.g. combining plant, vertebrate and invertebrate data), as a surrogate for biodiversity in developing a conservation plan (Chapter 5, and see, for example, Williams *et al*., 1996; Sarakinos *et al*., 2001). In the case study outlined in Box 6.2, 53 species were identified that, when taken together, were considered representative of the system in Maputaland. These data were then combined with other data layers in a systematic conservation planning exercise.

Environmental surrogates

In the last decade, systematic conservation planning studies have predominantly used environmental surrogates as general surrogates for biodiversity representation (e.g. Carwardine *et al*., 2008; Klein *et al*., 2008). 'Environmental surrogate' is a generic term covering land or ecological classifications based primarily on physical and climatic variables, which can incorporate some biotic variables, such as vegetation type (Margules & Sarkar, 2007). These variables are assumed to correlate with the patterns of species distribution, and have been argued by some to be more useful than species-based surrogates (compare: Araújo *et al*., 2001, 2003, 2004a; Ferrier, 2002; Lombard *et al*., 2003; Faith *et al*., 2004a).

Environmental surrogates are often used because these data are usually more readily available compared to more detailed biological data. In the Californian marine case study outlined in Box 6.3, a number of key habitats and a range of different depth classes were considered good environmental surrogates. In the Maputuland case study outlined in Box 6.2, it was argued that capturing the 44 land-cover types, as well as the 53 species, was the most effective way to get a representative system.

Box 6.2 Conducting systematic conservation planning in the terrestrial environment: a Maputaland case study

Written by Robert J. Smith, Durrell Institute of Conservation and Ecology, University of Kent, drawing on Smith *et al.* (2008).

The Maputaland Centre of Endemism is a region of high conservation value that falls within the countries of Mozambique, South Africa and Swaziland (Figure B6.2a). Its climate and soils have played a large role in maintaining high levels of species richness and endemism (Steenkamp *et al*., 2004), but they have also influenced the conservation of this biodiversity, because much of the land has little agricultural value and so has not been cleared by commercial farmers. Instead, most people rely on small-scale farming and harvesting natural resources for their livelihoods. This, together with an increasing human population and a history of political marginalization, has led to widespread poverty.

The governments of the region are keen to reduce these poverty levels and have recognized that ecotourism and game ranching are the most profitable forms of land use. Consequently, they have developed the Lubombo Transfrontier Conservation Area (TFCA) initiative, which seeks to conserve the region's biodiversity and reconnect important large mammal populations while creating jobs by developing new conservation areas, both privately and communally managed.

The TFCA initiative is guided by the Maputaland conservation planning system, which is based on systematic conservation planning principles (Margules & Pressey, 2000). This approach involves producing a list of important conservation features, setting targets for each feature and then identifying priority areas for meeting these targets.

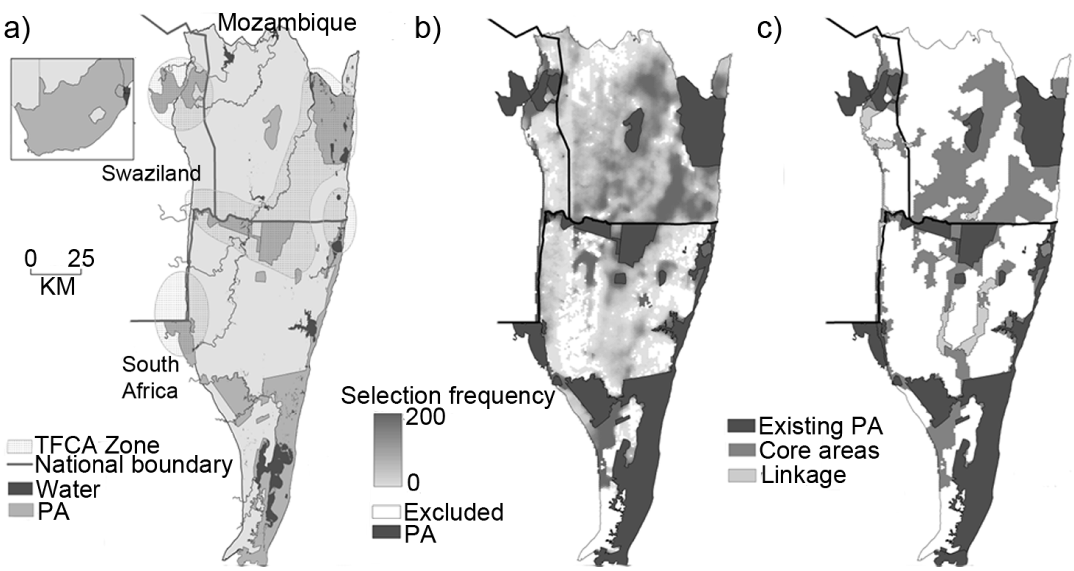

Figure B6.2
(a) Protected areas (PA) and TFCA (Transfrontier Conservation Area) zones in the Maputaland Centre of Endemism.
(b) Priority areas for conservation outside the existing protected areas.
(c) Proposed conservation landscape.
From Smith *et al.*, 2008, with permission from Elsevier.

The Maputaland system involved identifying 44 land-cover types, 53 species and 14 ecological processes as important conservation features, and mapping their distributions using satellite imagery and expert opinion (Smith *et al*., 2008). It also involved using data on the predicted spread of subsistence agriculture as a measure of both threat and opportunity cost, together with data on potential revenue from game ranching, which has key relevance for implementing the results (Knight *et al*., 2006).

The first conservation assessment used the *Marxan* conservation planning software, which uses a simulated annealing approach (Ball & Possingham, 2000). This involved:

1 dividing the region into a number of planning units;

2 assigning a cost to each planning unit based on its modelled risk of being cleared for agriculture;

3 using *Marxan* to identify near-optimal portfolios of these units for meeting the targets, maintaining connectivity and minimizing impacts on subsistence agriculture (Figure B6.2b).

These initial outputs were then used to develop a conservation landscape plan that could boost economic development through nature-based tourism and game ranching. The analysis identified 4,291 km^2 of new core protected areas and 480 km^2 of land that would function as ecological linkages (Figure B6.2c). The game ranching data were then used to estimate potential revenue from this proposed expansion of the protected area system.

The results showed that these new areas could provide US$18.8 million per annum, thereby helping to create jobs and reduce poverty. These results have already been used to guide South Africa's National Protected Area Expansion Strategy and the Critical Ecosystem Partnership Fund initiative in the Maputaland–Pondoland–Albany hotspot, although more work is needed to ensure that the system becomes part of day-to-day land use planning in all three countries.

6.4.2 Achieving persistence

Identifying how to secure the long-term persistence of species, ecosystems and the ecological and evolutionary processes that maintain them is difficult. For most systematic conservation plans, persistence objectives are formed as targets. These targets should be informed by ecological theory and empirical knowledge of species autoecology and biogeography (Carwardine *et al*., 2009).

The research that went into designing a conservation plan for the critically endangered Leadbeater's possum (*Gymnobelidius leadbeateri*) is a good example of how an objective for persistence can be calculated using a species minimum viable area. This possum, considered an umbrella species (see Box 6.1), inhabits the tall forests of southern Victoria, Australia, but its habitat has receded due to industrial logging and changed fire regimes.

Lindenmayer and Possingham (1995) showed that the species needed several patches, each of at least 100 ha in size, in each forest catchment in which they were present, to ensure their persistence in the long term. It was argued that all remaining patches of habitat containing this species must be protected and, if possible, enlarged by restoration activities to hit this minimum viable patch size, which has been the basis of conservation plans in the region.

In a similar example, Carroll *et al*. (2003) developed a conservation plan based on the needs of mammalian carnivores in the Rocky Mountains region of North America, using a spatially explicit population model that informed the design of the protected area network.

Persistence targets can also be set for environmental surrogates, especially when planning at coarser spatial scales. These are often based on achieving representational targets for biodiversity features while implicitly accounting for consequences for other stakeholders (e.g. agriculturalists or the forestry sector). For example, in a series of Regional Forestry Agreements developed in Australia, it was agreed by all stakeholders, including conservation biologists, that each distinct forest type was adequately protected if at least 15 per cent of its area was within a protected area (Pressey, 1998).

In the Californian marine case study outlined in Box 6.3, different persistence targets based on

Box 6.3 Conducting systematic conservation planning in the marine environment: a case study from the central coast of California

Written by Carissa J. Klein (The University of Queensland, Australia).

California's Marine Life Protection Act mandates the design and management of a network of marine protected areas to protect marine life, habitats, ecosystems and natural heritage, and to improve recreational, educational and study opportunities provided by marine ecosystems (State of California 1999). As part of the initiative to implement the Marine Life Protection Act, California's central coast (from Pigeon Point to Point Conception, covering an area of 2,978 km^2) was the first of five regions to undergo a stakeholder-driven process to design a network of marine protected areas.

To help inform the design of marine protected areas consistent with the Act's goals, a representative group of stakeholders from California's central coast developed a very broad set of *Regional Goals and Objectives* with the help of administrators, managers, and scientists in the period 2005–2006. A scientific advisory team was then tasked to provide guidelines that quantified the science-related Regional Goals and Objectives. These guidelines were as follows:

1 The diversity of species, habitats, and human uses prevents a single optimum network design.
2 Every 'key' marine habitat should be represented in the network.
3 Protected areas should extend from the intertidal zone to deep waters.
4 Protected areas should have an alongshore span of 5–20 km.
5 Protected areas should be placed within 50–100 km of each other.
6 Each 'key' habitat should be replicated at least 3–5 times.
7 Placement should take into account local resource use and stakeholder activities.
8 Placement should take into account the adjacent terrestrial environment and associated human activities.
9 Network design should account for the need to evaluate and monitor biological changes within the protected areas.

Systematic conservation planners were asked to produce a network of marine protected areas consistent with the scientific guidelines. These planners decided that it could be accomplished using the systematic conservation planning decision support tool *Marxan* (see Klein *et al*., 2008a,b and www.uq.edu.au/marxan/). *Marxan* identifies possible locations for protection that achieve a set of conservation targets (e.g. protect 20 per cent of each habitat type, 50 per cent of threatened species' distributions) for a minimal 'cost' (e.g. cost of closing conservation areas to fishermen; see Box 6.4 for more information on what a minimal-set problem is).

The nine guidelines outlined above, with the exception of **8** and **9**, were able to be factored into the *Marxan* analysis as follows:

• Guideline 1, which is related to the systematic conservation planning principle of flexibility, was accounted for by using *Marxan* to produce a number of different reserve networks which achieved a similar objective. *Marxan* produces multiple different solutions for the location of protected areas, all of which achieve the same set of conservation goals.
• Guideline 2, which is related to the principle of representation, was addressed by representing each key habitat identified in the different reserve networks.
• Guideline 3, which is also related to the principle of representation, was addressed by targeting each feature in five different depth zones.
• Guidelines 4, 5, and 6, which are related to the principle of persistence, were addressed by employing user-defined parameters to ensure reserves were of an adequate size, spacing, and replication.
• Guideline 7, which is related to the principle of efficiency, was addressed by minimizing the impact on commercial and recreational fisheries.

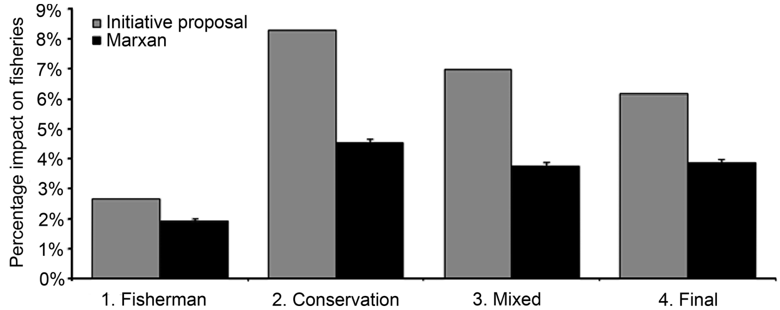

Figure B6.3a Comparison of the impact on commercial and recreational fisheries of marine reserve networks designed by stakeholders, based on expert judgment, with that designed using *Marxan*. The fishing impact (defined as loss of overall fishing yield) of both solutions is displayed, and the *Marxan* analysis is defined as an average ± standard deviation of 100 different solutions that achieved the planning objectives. Network 1 was developed by commercial and recreational fishers, network 2 by conservationists, network 3 by a mixed interest group, and network 4 was the solution considered for implementation by The California Fish and Game Commission. Adapted from Klein *et al.* (2008b).

Biodiversity data used in the analysis were environmental surrogates which included rocky reef, soft bottom, kelp forests, submarine canyons, eelgrass, surfgrass, and estuaries. Each of these habitats was targeted for inclusion in a network of marine protected areas under a number of different scenarios (e.g. 10 per cent representation, 20 per cent representation and 50 per cent representation in the reserve system).

The socio-economic data included information on the number of recreational fishing trips and an expert-derived assessment of the relative importance of an area for commercial fishing. The expert-driven assessment involved 109 commercial fishermen being interviewed to determine accurate spatial data on fishing effort and to map their fishing grounds. From this, an index of relative fishing effort was used to calculate the impact of fisheries in the reserve design (i.e. those marine waters that would be closed to fishing). The two types of fishing data were combined to deliver a relative index of fishing effort, which was used as a 'cost' to minimize in the *Marxan* software.

Marine reserves were chosen that would meet the different biodiversity targets and minimize the impact on fishermen in terms of lost fishing effort due to reservation. Explicitly considering commercial and recreational fisheries in the analysis allowed the impact to the fisheries to be reduced by up to 21 per cent, depending on the scenario selected (Klein *et al*., 2008a).

In a separate analysis, Klein *et al*. (2008b) were able to compare the marine reserve network designed without using a systematic planning tool by the three stakeholder groups (commercial and recreational fishermen, conservationists and a mixed interest group) against those designed using *Marxan*. They found that the *Marxan* analysis represented an equal or greater amount of habitat, yet for a lower cost in terms of the impact on commercial and recreational fisheries (Figure B6.3a). Interestingly, of all stakeholder groups, the proposal developed by stakeholders from the fishing industry was the most proficient at representing biodiversity and minimizing the impact to the fishing industry.

These results indicate the important role stakeholders have in systematic conservation planning and that conservation planning decision support tools should be used to support stakeholder-driven planning processes, not replace them.

environmental surrogates were used. The research team formed a scientific advisory team that gave them advice on what would be a good target for reservation for each habitat and water depth class. They were advised that the key habitats and different depth classes had to be captured and replicated at least three to five times to achieve an adequate outcome. See Box 6.3 for the results of this exercise.

Despite their continued use, there has been a large amount of criticism over the use of simple percentage targets in systematic conservation plans (Soulé & Terborgh, 1999; Recher, 2004; Watson *et al*., 2008). The main criticism is that fixed percentages do not account for landscape context. The habitat fragmentation literature (Chapter 8; and see Lindenmayer & Fischer, 2006) reveals that the size and isolation of the protected area, its 'shape' in terms of edge to core ratio, and also the similarity (or 'hostility') of the matrix habitat surrounding the protected area, can each affect the chances of persistence for many species. Fixed percentage targets do not take these patch- and landscape-scale effects into account.

There have been a number of recent analyses in the systematic conservation literature to address this problem. Specific design criteria based on the characteristics of environmental surrogates (e.g. a specific habitat type) have been incorporated into the persistence objective in some systematic conservation plans. For example, Leroux *et al*. (2007) introduced a framework for determining a minimum reserve size required to incorporate natural disturbance and maintain ecological processes by identifying criteria for estimating the size, location, and efficacy of a *minimum dynamic reserve*. The size and location of such a reserve is determined by the estimated maximum extent of the largest disturbance event, and by the extent and distribution of communities of species that are differentially affected by disturbance.

They illustrated their approach using a study of the Mackenzie Valley region of Canada, where forest fire is the major natural disturbance that influences vegetation community dynamics and dependent fauna. In this research, Leroux *et al*. (2007) designed and evaluated a candidate minimum dynamic reserve using a spatially explicit dynamic simulation model that incorporates locally calibrated fire and the vegetation dynamics (i.e. the minimum area they need for persistence and recolonization following a fire event) of five broad vegetation types (closed spruce, open spruce, mixed-wood, tall shrub, small shrub).

Using simulations, they showed that minimum extent of vegetation types ranged from 10 km^2 of tall shrub to 1,948 km^2 of open spruce, while the mean extent of the five communities available to burn in the study area varied from 118 km^2 of mixed-wood to 3,407 km^2 of open spruce. Using these thresholds, they showed that their minimum dynamic reserve maintained its recolonization sources through time, suggesting that minimum dynamic reserves may provide an operational framework for determining reserve size in dynamic landscapes under the influence of large natural disturbances such as fire. Of course, it is very difficult to validate such an estimation – hence the use of simulations.

Another way in which systematic conservation planners have attempted to achieve persistence is to develop a form of redundancy within the plan, i.e. to set multiple representation targets. Here, the idea is that reserve planning algorithms are set with the goal of selecting a network of areas that ensures, for example, that each species occurs in a minimum of five separate sites. Building in this degree of redundancy may be desirable to provide the protected area networks with a degree of resilience to ensure that a species (or other desired biodiversity attribute) survives in the face of natural catastrophes, disease epidemics, the chronic ecological and genetic effects of small population size, or the loss of a reserve to legal or illegal human intervention.

It should be noted that this use of the term 'redundancy' has somewhat negative connotations in conservation planning, as it was used as a key theme in criticisms of formerly widely used scoring procedures that disregarded complementarity and yielded systems of protected areas that had high redundancy and were inefficient (i.e. they were expensive and achieved few targets – see Pressey & Nichols, 1989; Pressey, 1994). Thus, the term redundancy is rarely used and 'multiple representation' is the favoured expression. This is considered more appropriate because multiple representations are not a by-product of the selection process but, rather, they are actively pursued.

Rodrigues *et al*. (2000) provide a useful demonstration of the potential advantages of multiple representations. They used presence/absence data from the Common Birds Census (CBC) in the UK to test the effectiveness of three families of selection models:

i Single and multiple representations. Single representations calculated the minimum area such that each species was represented in at least one site. The multiple representations method selected the minimum area needed to ensure that each species was represented in at least *n* sites (or the maximum number of sites, if this was less than *n*).

ii Percentage of range. This method was used to select the minimum area of sites so that each species was represented in at least *p* per cent of its range within the study area.

iii Permanence rate. A permanence rate was calculated for each species in each site, being the frequency with which a species was recorded in relation to the number of visits to a site within a specified time period. The minimum area was selected so that each species was represented in the site, or one of the sites, where it has the highest permanence rate.

The results of the Rodrigues *et al.* (2000) study clearly demonstrated that a single representation strategy (a minimum of one site containing each species) leads to very high efficiency but low long-term effectiveness. A multiple representation strategy appeared to be safer than a strategy based on percentage of area. This is explained by the prioritization of rare species that is an inevitable by-product of the multiple representation approach. For example, if a rare species only occurs in three sites and the multiple representation criterion (*n*) is set to three sites or more, then all the sites containing the species necessarily will be included in the selection.

The drawback of a simple multiple representation approach is that it assumes that all sites where the species occur have a similar potential for sustaining a population over a period of time. Strategies that target sites where species are most likely to persist give the greatest probability of long-term effectiveness (Williams, 1998). Unsurprisingly then, the Rodrigues *et al.* (2000) study found that choosing the best site based on permanence rate was a better strategy than investing in multiple, but blind, redundancy. Unfortunately, estimating persistence rate requires a lengthy and accurate time series, and other methods of choosing the 'best' site such as using abundance data are also expensive and time-consuming.

Ultimately, the decision about whether built-in redundancy is a good way to select a reserve network depends on data and resource availability (e.g. what area/pattern of reserves can be maintained).

An additional approach beyond planning for multiple representation is to plan ways to maximize the biophysical connections among protected areas. This is considered important for a number of reasons:

• First, as natural landscapes become more fragmented, an increasing number of species will need to disperse through an increasingly 'hostile' landscape matrix if they are to maintain their genetic variability in viable metapopulations. It is probable that connected landscapes improve the chances of this happening (Mackey *et al.*, 2008).

• Second, it is increasingly recognized that a large number of species need a very large area to survive – far larger than a protected area network will provide. For example, the European goshawk (*Accipter principalis*) has a home range of 30–50 km^2, and male mountain lions (*Felis concolor*) in the western United States have home ranges in excess of 400 km^2 (Wilcove *et al.*, 1986). Moreover, many species have evolved to be highly dispersive and regularly migrate vast distances to find suitable conditions. These species clearly require more space than could reasonably occur in a small number of isolated protected areas, as the resources they require for existence vary both spatially and temporally (Gilmore *et al.* 2007). The survival of these species will depend on their ability to move between protected areas, and also the hostility of the matrix habitat between protected areas.

• Third, habitat connectivity is likely to play an even larger role with the onset of anthropogenic climate change. Studies have estimated that by the middle of the 21st century, range shifts due to climate change will commonly span tens of kilometres (Kappelle *et al.*, 1999). There will be a clear need to have some form of connectivity to find suitable locations to which species can migrate or take refuge (Peters & Darling, 1985; Mackey *et al.*, 2008).

Planning for 'connectivity' has recently moved beyond simply creating corridors or stepping-stones between protected habitat patches. The concept of connectivity conservation is now encompassed within the concept of maintaining the ecological and evolutionary processes that generate and sustain biodiversity at various spatial and temporal scales (Soulé *et al.*, 2004, Pressey *et al.*, 2007; Watson *et al.*, 2009). Incorporating information on connectivity within a systematic conservation planning framework enables networks of priority areas to be designed with the goal of maintaining genetic and demographical flows, which may thus ensure the resilience of populations to the effects of landscape conversion and climate change.

To date, few studies have incorporated ecological and evolutionary processes into conservation planning

(Rouget *et al.*, 2003; Possingham *et al.*, 2005; Pressey *et al.*, 2007). However, in a national scale analysis in Australia, Klein *et al.* (2009) accommodated ecological and evolutionary processes in four ways:

1 using sub-catchments as planning units rather than arbitrarily delineated grids;
2 targeting refugia from drought;
3 targeting evolutionary refugia;
4 preferentially selecting planning units along connected waterways.

The researchers identified drought refugia as areas with relatively high and regular herbage production, while evolutionary refugia were identified as areas thought to be important for maintaining and generating biota during long-term climatic changes. They identified priority areas for conservation in Australia that met biodiversity and ecological process targets while minimizing acquisition cost.

Other examples of incorporating ecological processes in conservation planning include the comprehensive analyses undertaken in South Africa, where spatial surrogates for processes, such as edaphic interfaces, animal movement corridors, and macroclimatic and environmental gradients were targeted (Cowling *et al.*, 1999, 2003; Rouget *et al.*, 2003, 2006).

Clearly, the dynamic nature of ecological processes makes them difficult to quantify (Possingham *et al.*, 2005), but they are now recognized as an important consideration when persistence objectives are being defined. See further discussion in Chapter 7.

6.4.3 Achieving efficiency

As discussed in section 6.2, a key concept in identifying areas to achieve representation efficiently is complementarity. The basic idea behind complementarity is that conservation areas should complement one another in terms of the 'features' they contain, the species, communities, habitats, ecological processes, etc. Each conservation area should be as different from the others as possible until all the 'differences' (e.g. different species, communities, etc.) are adequately represented.

Complementarity can be defined in a number of ways. The most commonly used implementation is that a proposed new conservation area is assigned a higher complementarity value than another if it has more surrogates that have not already met their assigned target of representation in a conservation area network. For example, two proposed new areas, both with high species richness, may have different numbers of surrogates that can be captured in the reserve network. The efficient choice would be selecting the area that adds the most complementarity. Complementarity is, therefore, related to the concept of beta diversity (Whittaker, 1972), but whereas beta diversity is the difference between two areas, complementarity is a measure of the dissimilarity between the species complements of sets of selected areas.

It is important to note that the principle of efficiency is not simply about achieving complementarity. As we discussed earlier, achieving an efficient network is also a matter of achieving objectives for the least possible cost, where cost may reflect the financial cost of implementing and managing protected areas or the costs of lost opportunities for economic development (Naidoo *et al.*, 2006).

There is an increasing number of examples of where cost data have been implemented into systematic conservation analyses. For example, in the Californian marine case study outline in Box 6.3, the authors conducted and interviewed 109 commercial fishermen to find spatial data on fishing effort and to map their fishing grounds. From this, an index of relative fishing effort was used to calculate the impact of fisheries in the reserve design (i.e. those marine waters that would be closed to fishing). Using these stakeholder data, Klein *et al.* (2008a,b) were able to produce a systematic conservation plan that was efficient in that it maximized biodiversity conservation and minimized cost to livelihoods for fisherman.

As outlined in Section 6.3.3, it is also possible to factor in the returns from ecosystem service protection into conservation planning analyses. There may, however, be trade-offs between the achievement of objectives (Mertz *et al.*, 2007; K.A. Wilson *et al.*, 2009), depending on the spatial congruence between ecosystem services and between ecosystem services and biodiversity features. Some analyses have found high levels of congruence (Turner *et al.*, 2007; Venter *et al.*, 2009), but in other areas overlap has been more limited (Chan *et al.*, 2006; Naidoo *et al.*, 2008).

There are several ways to integrate ecosystem services into conservation planning analyses (Egoh *et al.*, 2007). Ecosystem services can be included as a feature for which a target can be set (Chan *et al.*, 2006) and the set of planning units that meet these and other targets for the lowest cost can be identified. Alternatively, it is possible to modify the relative weighting for conserving ecosystem services versus the conservation of other biodiversity features (K.A. Wilson *et al.*, 2009) and then seek to maximize the overall benefit that is derived.

6.4.4 Achieving flexibility

As we have discussed throughout this chapter, the selection and creation of new protected areas in a network is not a simplistic, one-off process. Protected area networks have to be accepted socially and politically, and it is therefore of critical importance that there should be several alternatives available when a systematic conservation plan is developed. These alternatives mean that the plan is flexible (Pressey *et al.*, 1993). It must be clear, however, why areas are selected and why some areas are not, and hence transparency is a clear part of flexibility (Nicholls & Margules, 1993).

Measuring the 'irreplaceability' of sites is arguably the commonest way to show flexibility in a systematic conservation plan. The irreplaceability of a site reflects the importance of including that site in the protected area network if all conservation objectives are to be achieved (Pressey *et al.*, 1994; Ferrier *et al.*, 2000). Irreplaceability can be viewed in two contexts: the likelihood that an area is necessary to achieve conservation objectives for the features it contains; or the extent to which the options for achieving conservation objectives are reduced if the area is unavailable for conservation.

In systematic conservation planning, a completely irreplaceable area is essential for a plan to meet its conservation objectives, whereas an area with a very low irreplaceability can be substituted by other sites. For example, when planning a reserve system in a landscape, you may find that some areas are completely unique or have been altered to such as extent that the last remaining sites are highly irreplaceable. If there is a risk of these areas being lost to threatening processes, then it might be a large loss for biodiversity conservation in that region. Consequently, irreplaceability can be used as a measurement of conservation value.

It is important to note that although irreplaceability can help determine which areas are priorities for conservation, other constraints and considerations may mean that areas with lower irreplaceability are more suitable for conservation. For example, some combination of vulnerability, ecological condition, and financial cost of an area might influence its priority for protection. When this occurs, it is important to acknowledge the conservation cost of not including these sites within the overall plan. Moreover, the irreplaceability rank of an area will change as individual areas are designated as part of the conservation area network. Therefore, the process of identifying irreplaceable sites must be reiterated after each stage, when new areas are included in a network and others are removed. Such a process was involved in the Maputaland study highlighted in Box 6.2.

6.5 DECISION SUPPORT TOOLS TO IDENTIFY AND PRIORITIZE NEW PROTECTED AREAS

As discussed in the introduction to this chapter, the development and use of systematic planning tools for designing protected areas is only a recent phenomenon. The early approaches to designing systematic conservation plans using simple scoring systems (e.g. Margules & Usher, 1981; Smith & Theberge, 1986) were perceived to be a great improvement on previous approaches due to their transparency and repeatability. However, due in large part to technical limitations of data processing up until the end of the 1980s, these early systematic conservation planning approaches did not take into consideration complementarity, nor did they have the ability to set spatial objectives like connectivity and spatial compactness (Margules *et al.*, 1991; Pressey, 1997).

Over the past decade, decision support tools have been increasingly used to help inform conservation planning decisions. Decision support tools are often computer-based information systems intended to help decision-makers compile and analyse data to help solve conservation problems. A range of mathematical techniques have been developed that are incorporated into these tools (see Box 6.4 and Moilanen *et al.*, 2009). It is important to note that the use of any decision support tool, simple or complex, requires properly defined conservation problems.

A common framework for defining conservation priorities is through the use of decision theory. This framework centres on achieving explicitly stated objectives while acknowledging constraints on conservation actions and the levels of uncertainty involved within the decision process. In Table 6.2, we outline a

Box 6.4 Three broad classes of mathematical problems used in systematic conservation planning: the minimum set problem, the maximal coverage conservation prioritization problem and the conservation resource allocation problem

Conservation planning began without a well-posed mathematical problem, which is not uncommon in conservation science (Possingham *et al.*, 2001). Cocks and Baird's (1989) seminal paper provided the first formal statement of a conservation planning problem – the minimum set problem. In the *minimum set problem* the goal is to conserve a variety of conservation features to an adequate level for minimum total cost where cost can be the cost of acquisition and management or the estimated foregone opportunity cost (Naidoo & Ricketts, 2006). The simplest variant of this problem is:

$$\min \sum_{i=1}^{N_S} c_i x_i$$

given that

$$\sum_{i=1}^{N_s} x_i r_{ij} \geq T_j, \qquad \text{for all features } j,$$

where r_{ij} is the occurrence level of feature j in site i, c_i is cost of site i, N_s is the total number of sites and T_j is the target level for feature j. The control variable x_i has value 1 for selected sites and value 0 for sites not selected (Moilanen *et al.*, 2009). This became the foundational problem of systematic conservation planning.

Since then, various authors have produced alternatives, but arguably the *maximal coverage conservation prioritization problem* is the most dominant. This problem is used when resources are insufficient for satisfying all targets and the objective is to find the solution that satisfies the largest number of conservation targets, given a budget constraint. The maximal coverage problem is related to the minimum set coverage problem, in that minimum set coverage can be achieved by solving the maximal coverage problem at different budget levels and finding the minimum budget level that satisfies all targets. A simple version of the maximal coverage problem can be written as:

$$\max \sum_j I_j \left(\sum_i x_i r_{ij} \geq T_j\right),$$

given that

$$\sum_i x_i c_i \leq B,$$

where B is the conservation budget (money, trained personnel, time, etc.), and $I(z)$ is an indicator function, with $I_j(z) = 1$ when condition z is true, i.e. the target for feature j is met when

$$\left(\sum_i x_i r_{ij} \geq T_j\right),$$

and $I_j(z) = 0$ otherwise (Moilanen *et al.*, 2009).

Both the minimum set and maximum coverage problems are limited to specific problems. However, it is possible to define a fairly general *conservation resource allocation problem* that includes most, if not all, previous problem definitions. In general, all of conservation involves taking actions in a place and at a time in an attempt to achieve a variety of outcomes.

Our general task is therefore to decide how much to spend on each kind of action (e.g. invasive species control, changed logging practices, or reduced grazing) in each place, at a particular time (which we will refer to henceforth as the year). These actions will alter the dynamics of a variety of state variables, $y_{ijt\cdot}$, such as the size of the population of a species in a site, or the amount of an ecosystem service in a site. Mathematically this means that our control, or decision, variable is a_{jkt}, the amount of money we spend on action k in place j in year t. A fairly general formulation of the conservation resource allocation problem is to:

$$\max \sum_{t=1}^{T} f(y_{ijt,1})$$

subject to a budgetary constraint each year

$$\sum_{j=1}^{N} \sum_{k=1}^{P} a_{jkt} \leq b_t \text{ for all } t,$$

and contingent on the dynamics of the state variables, that is, how key system states move from year to year in response to actions or forces we do not control:

$$y_{ijt.1} \cdot g(a_{..t}\ y_{..t}, x_{..t},) \text{ for all } i, j \text{ and } t,$$

where f is a function that turns our state variables into a reward function that we are trying to maximize (this could be highly non-linear), g is a function that determines how the state variables, *yijt,* evolve in space and time as a consequence of actions and forces we do not control, *xijt.* In this formulation, N is the number of different places and P is the number of different sorts of actions.

This mathematical formulation of a problem that considers expenditure of money on different conservation actions in space and time is a fairly general formulation of all resource allocation problems. It is called a resource allocation problem because there is a fixed annual budget. Evaluating actions based on their cost-effectiveness (Joseph *et al*., 2009) provides one algorithm that can often provide rough solutions to this very complex optimization problem.

seven-step decision theory framework which has been articulated by a number of authors for systematic conservation planning (Table 6.2; Possingham *et al*., 2001; K.A. Wilson *et al*., 2009).

There is now a large amount of literature on optimal protected area design based on this decision theory framework (summarized in Moilanen *et al*., 2009). The problems generated using this framework can be expressed mathematically and then solved by one of a number of methods. There are two classic problem definitions commonly used in conservation planning, the minimum set and maximal coverage conservation prioritization problem (Box 6.4).

The minimum set problem minimizes the resources expended while meeting the conservation objectives. For this problem, the objective is to minimize cost and the constraint is the conservation objectives.

The maximal coverage conservation prioritization problem maximizes the objectives (e.g. target level achievement) given a fixed amount of resources. Here, the problem is reversed: the constraint is the budget and the objective is to maximize conservation objectives.

Methods for solving systematic conservation planning problems fall into several classes: local heuristic algorithms, which select sites in a stepwise manner (Pressey *et al*., 1993; 1994); global heuristic algorithms, which select sites in sets (e.g. simulated annealing, Ball & Possingham 2000); and optimization algorithms (Cocks & Baird, 1989). These methods are dealing with increasingly large and more complex problems (see section 6.7), which includes having multiple and conflicting objectives and multiple types of management actions.

It must be noted that decision problems can be quite complex, and there are now several software packages that can support systematic conservation planning (e.g. *Marxan*, *C-Plan*, *Zonation*, *ConsNet*; see Moilanen *et al*. (2009) for a thorough review of each platform). However, as Bottrill & Pressey (2009) point out, these

Table 6.2 The application of a seven-step systematic conservation planning decision theory framework (Possingham *et al.*, 2001; K.A. Wilson *et al.*, 2009) to a hypothetical example based on the problem of acquiring new land to add to a protected area network to protect threatened species.

Step	Details	Example: Acquiring new land to add to a protected area network with the aim of protecting threatened species.
1 Statement of objective(s)	This is a statement of what is hoped to be achieved and is measurable.	To maximize the representation of threatened species in protected areas.
2 List of management actions	This can range from one action to a number of actions.	Purchasing new areas to add to the protected area network. The available option is either to acquire each parcel of land or not.
3 State variables	This is the knowledge about the system, including both biodiversity and human variables.	Where the threatened species are located and how much each parcel of land costs.
4 State dynamics	This step requires knowledge about how the state variables may change (which may be dependent or independent of the management action).	Fluctuation of property prices for parcels of land. These may vary independently or may increase with the implementation of the extended reserve network (Armsworth *et al.*, 2006).
5 Constraints	The constraints are what limit the application of any management action.	Size of budget, willingness of landholders to sell their properties, etc.
6 Uncertainty	Most data will contain a degree of uncertainty.	Inaccuracies in species data regarding presence and absence due to surveying methods and species detectability variation.
7 Solution methods	A range of mathematical approaches are used to solve problems (Box 6.4).	Algorithm to maximize representation and minimize cost.

software systems are not designed to replace people by making decisions for them; they operate interactively to facilitate decisions by people.

6.6 CONSULTATION AND IMPLEMENTATION OF SYSTEMATIC CONSERVATION PLANS

Much of the systematic conservation planning literature to date has focused on advancing the 'tools' of the systematic conservation planning trade. Far less attention has been dedicated to implementing conservation plans in the 'real world' (Salafsky *et al.*, 2002; Knight & Cowling, 2003). Indeed, some experts have argued that the discipline of systematic conservation planning is mired in an 'implementation crisis' (Knight & Cowling, 2003), because '... few academic conservation planners regularly climb down from their ivory towers to get their shoes muddy in the messy, political trenches, where conservation actually takes place' (Knight *et al.*, 2006, p. 410). There has been some critical discussion around this quite stark assertion (see, for example, Pressey & Bottrill, 2009), and a number of operational case studies show that development of a systematic conservation plan for a particular area by academics can integrate the diverse disciplines and activities needed for successful conservation action into a single, comprehensive process (Boxes 6.2 and 6.3 are good examples). Nonetheless, this debate highlights the point that while the tools of systematic conservation planning are important, they do not in themselves deliver conservation action.

To be successfully implemented, all systematic conservation plans must be complemented with social, political, and institutional tools and processes (Knight *et al.*, 2009). There are several operational models established that outline key considerations that can help to guide a transparent planning and implementation process (Pressey & Cowling, 2001; Knight *et al.*, 2006).

Those who have participated in such processes stress the importance of providing participants with clear and transparent explanations of the stages of a conservation planning process and what things need to be done to achieve them. This includes, for example, an assessment of conservation issues; identifying opportunities for and constraints on conservation actions; developing an implementation strategy; and products such as maps that help guide implementation. The flexibility of an overriding conservation model is also important, as it has been increasingly shown that implementing successful conservation actions is an ongoing process with feedbacks between planning and implementation.

Bottrill & Pressey (2009), for example, have produced a detailed 11-step process for the successful implementation of a conservation plan. The steps are outlined in Table 6.3. They argue that all 11 stages must be completed for a conservation plan to be conducted successfully.

An alternative operational framework is provided by Knight *et al.* (2006), who identified the key components for 'doing' pragmatic conservation planning (Figure 6.2). In their schematic, the thematic components are grouped into three interlinked foundations:

1 empower individuals and institutions;
2 undertake the systematic conservation assessment;
3 secure effective action.

Each foundation is essential for an effective conservation planning process.

The reality is that the implementation of any conservation plan is difficult. Wherever systematic plans are actually implemented, it quickly becomes apparent that human society is not an entity with a single value system (see Chapter 2). Whereas a conservationist or amateur naturalist may value a particular site because it contains habitat for an endangered species, a timber company may value that site because of the potential revenue that might be generated from harvesting trees, or a group of mountain bike enthusiasts may value the site for its recreational values.

Table 6.3 Steps in the process of developing and implementing a conservation plan, as outlined by Bottrill & Pressey (2009).

Steps	Processes
Stage 1	Scoping and costing the planning process
Stage 2	Identifying and involving stakeholders
Stage 3	Identifying the context for conservation areas
Stage 4	Identifying conservation goals
Stage 5	Collecting socio-economic and threat data
Stage 6	Collecting data on biodiversity and other natural features
Stage 7	Setting conservation objectives
Stage 8	Reviewing objective achievement in existing conservation areas
Stage 9	Selecting additional conservation areas
Stage 10	Applying conservation actions to selected areas
Stage 11	Maintaining and monitoring established conservation areas

However, when a plan is integrated with expert knowledge (Pressey & Cowling, 2001) and coupled with an implementation strategy that takes into context the needs for stakeholder collaboration (Driver *et al.*, 2003), the planning process itself can provide a foundation for effective conservation action. This is a major, often forgotten, value of systematic conservation planning – it not only identifies the priority conservation areas, but also provides a mechanism for stakeholder collaboration.

6.7 WHAT DOES THE FUTURE OF SYSTEMATIC CONSERVATION PLANNING HOLD?

In this chapter, we have delved into the fundamental principles of systematic conservation planning, while also providing some contemporary case studies demonstrating the use of different techniques and tools. In

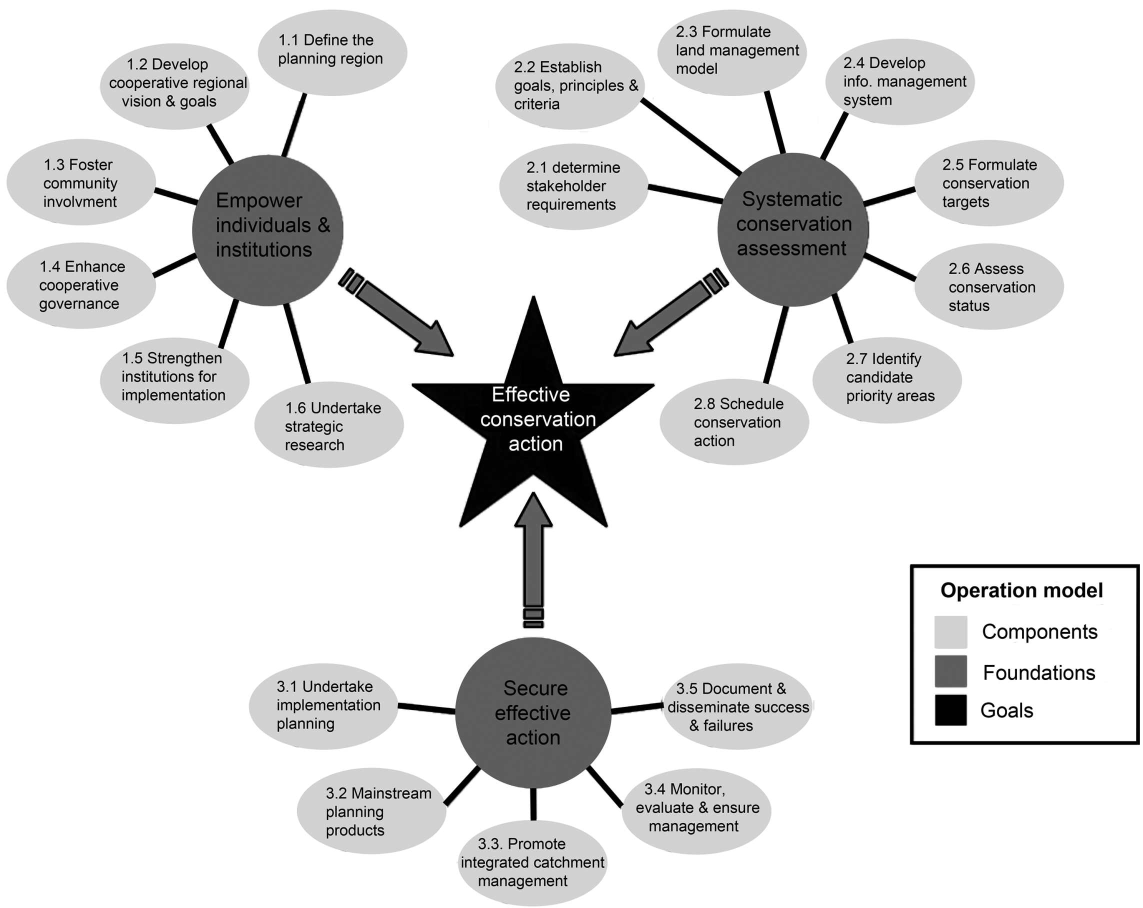

Figure 6.2 An operational model for pragmatic conservation planning. From Knight *et al.*, 2006.

this final section, we discuss future challenges in the field of systematic conservation planning and some recent advances.

The first two decades of systematic conservation planning primarily focused on a restricted suite of problems. These problems have generally assumed that systematic conservation planning:

1 is a static problem that determines a once-off reserve system;
2 can ignore the dynamic nature (including evolution) of biodiversity assets (e.g. species, habitats);
3 assumes a binary world where sites are either protected or not;
4 can use the area or number of sites as a surrogate for cost;
5 can ignore uncertainty;
6 can ignore risk and threat, and
7 can rely on simple targets for biodiversity assets so that once achieved, we are content that persistence is achieved.

All of these issues are challenges that need to be overcome if the discipline is to be taken seriously by those responsible for implementing conservation action. Here we briefly discuss some recent work that has contributed to this furthering this research agenda.

6.7.1 Conservation planning is a dynamic problem

Possingham *et al.* (1993) provided one of the first analyses that formulated the dynamic site selection problem. In each year they assumed one site could be bought (due to a constrained budget), sites had a fixed probability of becoming available and sites that were unreserved had a fixed probability of being destroyed. At the time, these authors found that taking a static approach was suboptimal compared to solving the dynamic problem using stochastic dynamic programming.

Various authors have subsequently considered and solved larger and more realistic versions of this original problem (e.g. Costello & Polasky, 2004; Meir *et al.*, 2004; Drechsler, 2005; Strange *et al.*, 2006). Such advances enable systematic conservation planners to include complexities like dynamic budgets and feedback between acquisition actions and the cost of reservation. In principle, any sort of dynamic complexity can be included in the site selection problem; however, the optimal solution of stochastic dynamic problems can only be found exactly using stochastic dynamic programming, which is computationally intractable for any but the smallest problem. There is therefore a need to derive simple heuristics that sequentially choose actions through time, such as choosing the actions that maximize the short term gain in biodiversity or minimize the short term loss of biodiversity.

6.7.2 Conservation assets change through time

The biodiversity assets that we would like to conserve are continually changing: local populations become extinct; species' distributions change; species evolve; and vegetation types change through succession (as discussed in Chapter 3). This adds further complexity and uncertainty to the dynamic conservation planning problem described above, and in principle it can be dealt with within the same approach.

However, there are some short cuts possible. Sites with evolutionary potential can be preferred in planning (Cowling *et al.*, 2003), present and future predicted distributions can be accommodated in the plans (Hannah & Hansen, 2005) and successional changes can be predicted and allowed for in target setting (Drechsler *et al.*, 2009).

6.7.3 A mix of conservation actions could occur at any site

As discussed briefly in the introduction, formal protection of habitat is just one of many conservation actions. In many cases, especially where there are multiple players in land ownership issues plus complex social and cultural constraints, reservation is an unlikely option for conservation. What we need is tools to help us determine which package of actions to activate at any site.

This sort of idea is effectively zoning – a common practice in fisheries, forestry and conservation where there are multiple interests (Watts *et al.*, 2009). These zoning tools are useful to guide broad policy decisions, and other methods have been developed to systematically select among specific conservation actions. For example, the Project Prioritization Protocol is a cost-effectiveness analysis that has been demonstrated to be useful for selecting among specific management projects for threatened species in New Zealand (e.g. Joseph *et al.*, 2009).

6.7.4 Better economics and socio-economics

Ando *et al.* (1998) were arguably the first to highlight in the peer-reviewed literature the naivety of building conservation plans that ignored realism in respect of financial costs. While the inclusion of the estimated cost of conservation is now more common in conservation planning (see section 6.3.3) it is still a challenge for most conservation researchers who are more familiar with the nuances of biological data (Bode *et al.*, 2008). To this end, there is a need for more real collaboration between economists and conservation biologists.

However, it is also being recognized that using simple cost-layer data (i.e. the price of land), without considering socio-economic factors such as a landholder's willingness to conduct a conservation action, regardless of cost, may lead to some erroneous results.

6.7.5 Dealing with uncertainty

There is some level of uncertainty in every aspect of conservation planning (Regan *et al.*, 2009). For example, semantic uncertainty underpins the actual definition of the conservation problem, while parametric uncertainty is rife in all the data that are used to develop conservation plans (Whittaker *et al.*, 2005). While Regan *et al.* (2005, 2009) argue that we can generally deal with parameter uncertainty quite well using sensitivity analysis, uncertainty about problem formulation or issues like species viability represent serious challenges at the interface of social science, philosophy, economics, mathematics and ecology. So far there are too few papers that deal credibly with uncertainty in conservation planning (but see Moilanen & Wintle, 2006).

6.7.6 Properly accounting for threats

There are several ways of dealing with threats in conservation planning. One is to rate sites in terms of the likelihood that they will be destroyed relative to their irreplaceability, with preference given to sites that are under most threat (Araújo *et al.*, 2002a; Pressey *et al.*, 2007). In practice, some planners use the likelihood of a site being converted to other uses as a surrogate of conservation cost and hence, by reference to the principle of efficiency, they avoid sites with a high probability of conversion.

Ironically, this will give us the reverse outcome to the first approach. Indeed, some of the confusion about how we deal with threats arises because some threats are mitigated by conservation action, while others are not. Ideally, threats are dealt with properly in a full dynamic framework (Wilson *et al.*, 2006; Game *et al.*, 2008) within which the consequences of taking action at a site, or not, are explicitly modelled.

6.7.7 Persistence – attainable goal or impractical utopia?

Persistence (also known as adequacy) is the bugbear of systematic conservation planning science because the question it asks – how much is enough? – is probably unanswerable. Governments and non-government organizations would often like to know that a suite of conservation actions in time and space is sufficient. However, in reality, more is always better, although that 'more' comes at an additional cost.

Probably the best way forward for conservation planners is to explicitly acknowledge and derive trade-offs, recognizing that no single answer is best but offering a range of good options that reflect different societal aspirations (Whittaker *et al.*, 2005; Polasky *et al.*, 2008). An alternative might be to represent different levels of risk (e.g. 75 per cent, 80 per cent or 95 per cent probability of persistence for 100 years) or varying levels of persistence (80 per cent probability of persistence for 10, 100 or 1000 years) based upon available knowledge.

Further discussion of the challenges of planning for persistence in a changing world is provided in the following chapter.

6.7.8 How much should we invest in improving a conservation plan?

As we have discussed throughout this chapter, there are usually many assumptions about what the most appropriate conservation actions in any given area may be and whether the data are truly fit for purpose. Recent research has shown that if learning processes and data collection strategies are intentionally included into the conservation planning process, it is likely that future conservation decisions will become more effective (Grantham *et al.*, 2009).

There is a complex and not very well understood trade-off between acting and learning when developing and implementing a systematic conservation plan. It is important to recognize that any given planned conservation action has been traded off with all other actions and also against the cost of delaying a conservation action.

FOR DISCUSSION

1 Describe and give examples for each of the key principles of systematic conservation planning. Describe some ways of achieving each of these principles when developing a hypothetical systematic conservation plan in both the marine and terrestrial environments.

2 How should scientists assess the fitness for purpose of data for use in systematic conservation planning?

3 What are the limitations and strengths of using targets for achieving persistence in a conservation plan?

4 What are the respective strengths and weaknesses of using species-based surrogates versus environmental-based surrogates when developing a systematic conservation plan?

5 What is the difference between conducting a minimum set problem as opposed to a maximum coverage problem when undertaking a systematic planning process? Give some examples of when each type of problem should be applied.

6 Are the key principles of systematic conservation relevant to both marine and terrestrial environments? What differences are there between how they are interpreted and the data used to achieve them in each of these environments?

7 Why is it important to ensure that all stakeholders participate in the planning process? How can planners ensure that stakeholders participate in the conduct of the systematic conservation plan?

SUGGESTED READING

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