CHAPTER 11

Conservation planning and priorities

Thomas Brooks

Maybe the first law of conservation science should be that human population—which of course drives both threats to biodiversity and its conservation—is distributed unevenly around the world (Cincotta *et al.* 2000). This parallels a better-known first law of biodiversity science, that biodiversity itself is also distributed unevenly (Gaston 2000; Chapter 2). Were it not for these two patterns, conservation would not need to be planned or prioritized. A conservation investment in one place would have the same effects as that in another. As it is, though, the contribution of a given conservation investment towards reducing biodiversity loss varies enormously over space. This recognition has led to the emergence of the sub-discipline of systematic conservation planning within conservation biology.

Systematic conservation planning now dates back a quarter-century to its earliest contributions (Kirkpatrick 1983). A seminal review by Margules and Pressey (2000) established a firm conceptual framework for the sub-discipline, parameterized along axes derived from the two aforementioned laws. Variation in threats to biodiversity (and responses to these) can be measured as vulnerability (Pressey and Taffs 2001), or, put another way, the breadth of options available over time to conserve a given biodiversity feature before it is lost. Meanwhile, the uneven distribution of biodiversity can be measured as irreplaceability (Pressey *et al.* 1994), the extent of spatial options available for the conservation of a given biodiversity feature. An alternative measure of irreplaceability is complementarity—the degree to which the biodiversity value of a given area adds to the value of an overall network of areas.

This chapter charts the history, state, and prospects of conservation planning and prioritization, framed through the lens of vulnerability and irreplaceability. It does not attempt to be comprehensive, but rather focuses on the boundary between theory and practice, where successful conservation implementation has been explicitly planned from the discipline's conceptual framework of vulnerability and irreplaceability. In other words, the work covered here has successfully bridged the "research–implementation gap" (Knight *et al.* 2008). The chapter is structured by scale. Its first half addresses global scale planning, which has attracted a disproportionate share of the literature since Myers' (1988) pioneering "hotspots" treatise. The remainder of the chapter tackles conservation planning and prioritization on the ground (and in the water). This in turn is organized according to three levels of increasing ecological and geographic organization: from species, through sites, to seascapes and landscapes.

11.1 Global biodiversity conservation planning and priorities

Most conservation is parochial—many people care most about what is in their own backyard (Hunter and Hutchinson 1994). As a result, maybe 90% of the ~US$6 billion global conservation budget originates in, and is spent in, economically wealthy countries (James *et al.* 1999). Fortunately, this still leaves hundreds of millions of dollars of globally flexible conservation investment that can theoretically be channeled to wherever would deliver the greatest benefit. The bulk of these resources are invested through multilateral agencies [in particular, the Global Environment Facility (GEF) (www.gefweb.org)], bilateral donors, and non-governmental organizations. Where should they be targeted?

11.1.1 History and state of the field

Over the last two decades, nine major templates of global terrestrial conservation priorities have been developed by conservation organizations, to guide their own efforts and attract further attention (Figure 11.1 and Plate 9; Brooks *et al.* 2006). Brooks *et al.* (2006) showed that all nine templates fit into the vulnerability/irreplaceability framework, although in a variety of ways (Figure 11.2 and Plate 10). Specifically, two of the templates prioritize regions of high vulnerability, as "reactive approaches", while three prioritized regions of low vulnerability, as "proactive approaches". The remaining four are silent regarding vulnerability. Meanwhile, six of the templates prioritize regions of high irreplaceability; the remaining three do not incorporate irreplaceability. To understand these global priority-setting approaches, it is important to examine the metrics of vulnerability and irreplaceability that they use, and the spatial units among which they prioritize.

Wilson *et al.*'s (2005) classification recognizes four types of vulnerability measures: environmental and spatial variables, land tenure, threatened species, and expert opinion. All five of the global prioritization templates that incorporated vulnerability did so using the first of these measures, specifically habitat extent. Four of these utilized proportionate habitat loss, which is useful as a measure of vulnerability because of the consistent relationship between the number of species in an area and the size of that area (Brooks *et al.* 2002). However, it is an imperfect metric, because it is difficult to assess in xeric and aquatic systems, it ignores threats such as invasive species and hunting, and it is retrospective rather than predictive (Wilson *et al.* 2005). The "frontier forests" approach (Bryant *et al.* 1997) uses absolute forest cover as a measure, although this is only dubiously reflective of vulnerability (Innes and Er 2002). Beyond habitat loss, one template also incorporates land tenure, as protected area coverage (Hoekstra *et al.* 2005), and two incorporate human population

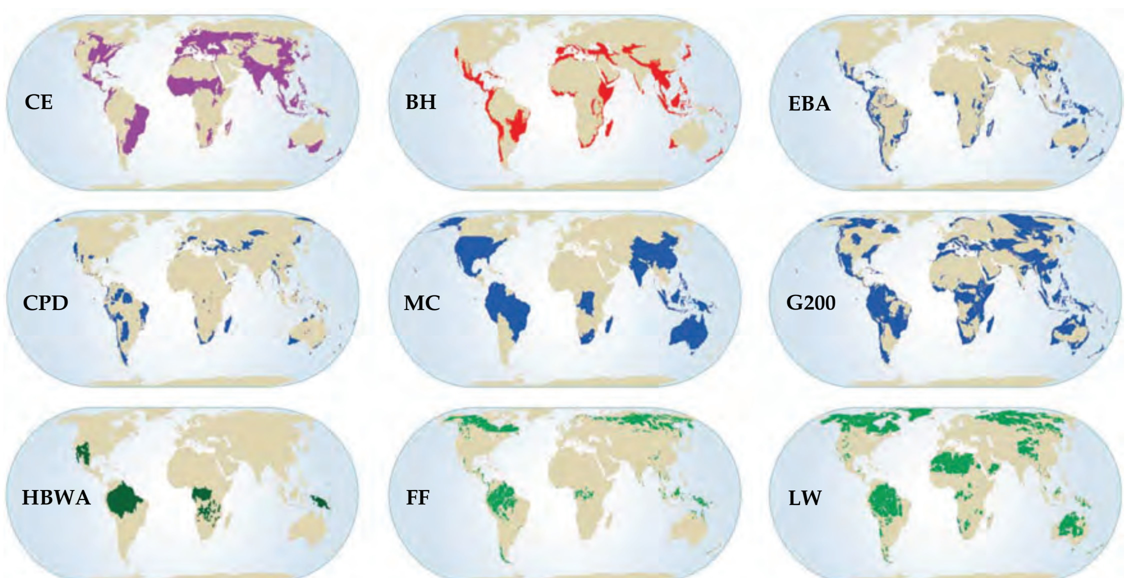


Figure 11.1 Maps of the nine global biodiversity conservation priority templates (reprinted from Brooks *et al.* 2006): CE, crisis ecoregions (Hoekstra *et al.* 2005); BH, biodiversity hotspots (Mittermeier *et al.* 2004); EBA, endemic bird areas (Stattersfield *et al.* 1998); CPD, centers of plant diversity (WWF and IUCN 1994–7); MC, megadiversity countries (Mittermeier *et al.* 1997); G200, global 200 ecoregions (Olson and Dinerstein 1998); HBWA, high-biodiversity wilderness areas (Mittermeier *et al.* 2003); FF, frontier forests (Bryant *et al.* 1997); and LW, last of the wild (Sanderson *et al.* 2002a). With permission from AAAS (American Association for the Advancement of Science).

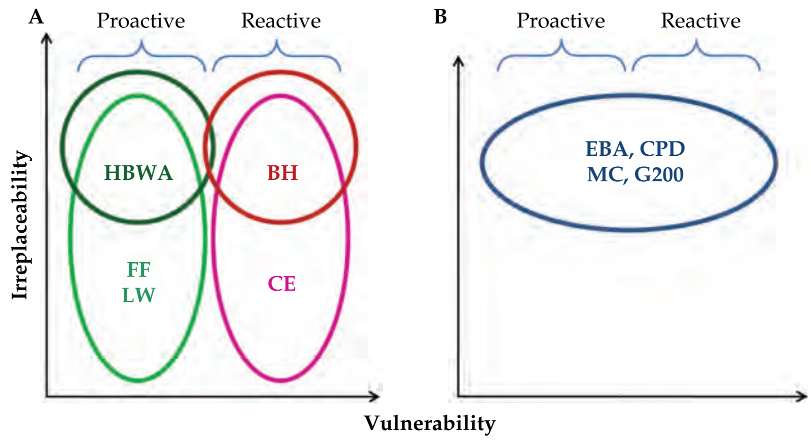


Figure 11.2 Global biodiversity conservation priority templates placed within the conceptual framework of irreplaceability and vulnerability (reprinted from Brooks *et al.* 2006). Template names follow the Figure 11.1 legend. (A) Purely reactive (prioritizing high vulnerability) and purely proactive (prioritizing low vulnerability) approaches. (B) Approaches that do not incorporate vulnerability as a criterion (all prioritize high irreplaceability). With permission from AAAS (American Association for the Advancement of Science).

density (Mittermeier *et al.* 2003; Sanderson *et al.* 2002a).

The most common measure of irreplaceability is plant endemism, used by four of the templates, with a fifth (Stattersfield *et al.* 1998) using bird endemism. The logic behind this is that the more endemic species in a region, the more biodiversity lost if the region's habitat is lost (although, strictly, any location with even one endemic species is irreplaceable). Data limitations have restricted the plant endemism metrics to specialist opinion estimates, and while this precludes replication or formal calculation of irreplaceability (Brummitt and Lughadha 2003), subsequent tests have found these estimates accurate (Krupnick and Kress 2003). Olson and Dinerstein (1998) added taxonomic uniqueness, unusual phenomena, and global rarity of major habitat types as measures of irreplaceability, although with little quantification. Although species richness is popularly but erroneously assumed to be important in prioritization (Orme *et al.* 2005), none of the approaches relies on this. This is because species richness is driven by common, widespread species, and so misses exactly those species most in need of conservation (Jetz and Rahbek 2002).

One of the priority templates uses countries as its spatial unit (Mittermeier *et al.* 1997). The remaining eight utilize spatial units based on biogeography, one using regions defined *a posteriori* from the distributions of restricted-range bird species (Stattersfield *et al.* 1998), and the other seven using units like "ecoregions", defined *a priori* (Olson *et al.* 2001). This latter approach brings ecological relevance, but also raises problems because ecoregions vary in size, and because they themselves have no repeatable basis (Jepson and Whittaker 2002). The use of equal area grid cells would circumvent these problems, but limitations on biodiversity data compilation so far have prevented their general use. Encouragingly, some initial studies (Figure 11.3) for terrestrial vertebrates (Rodrigues *et al.* 2004b) and, regionally, for plants (Küper *et al.* 2004) show considerable correspondence with many of the templates (da Fonseca *et al.* 2000).

What have been the costs and benefits of global priority-setting? The costs can be estimated to lie in the low millions of dollars, mainly in the form of staff time. The benefits are hard to measure, but large. The most tractable metric, publication impact, reveals that Myers *et al.* (2000), the benchmark paper on hotspots, was the single most cited paper in the ISI Essential Science Indicators category "Environment/Ecology" for the decade preceding 2005. Much more important is the impact that these prioritization templates have had on resource allocation. Myers (2003) estimated that over the preceding 15 years, the hotspots concept had focused

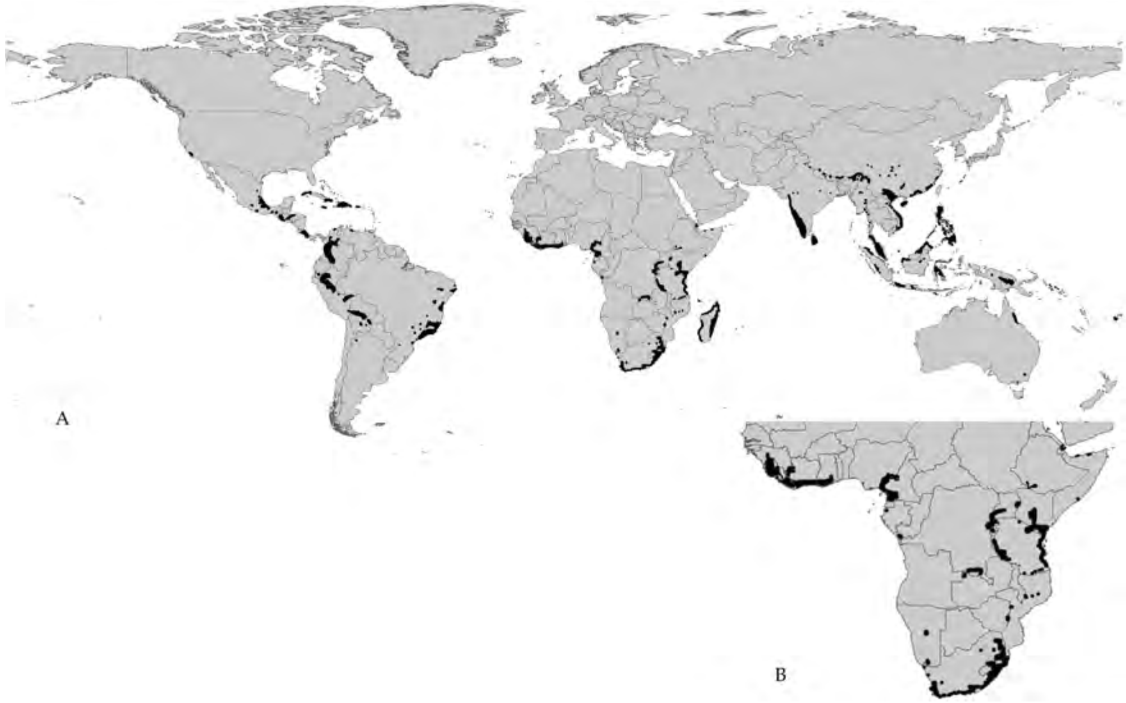


Figure 11.3 Incorporating primary biodiversity data in global conservation priority-setting (reprinted from Brooks *et al*. 2006). Global conservation prioritization templates have been based almost exclusively on bioregional classification and specialist opinion, rather than primary biodiversity data. Such primary datasets have recently started to become available under the umbrella of the IUCN Species Survival Commission (IUCN 2007), and they allow progressive testing and refinement of templates. (A) Global gap analysis of coverage of 11 633 mammal, bird, turtle, and amphibian species (∼40% of terrestrial vertebrates) in protected areas (Rodrigues *et al*. 2004a). It shows unprotected half-degree grid cells characterized simultaneously by irreplaceability values of at least 0.9 on a scale of 0–1, and of the top 5% of values of an extinction risk indicator based on the presence of globally threatened species (Rodrigues *et al*. 2004b). (B) Priorities for the conservation of 6269 African plant species (∼2% of vascular plants) across a 1-degree grid (Küper *et al*. 2004). These are the 125 grid cells with the highest product of range-size rarity (a surrogate for irreplaceability) of plant species distributions and mean human footprint (Sanderson *et al*. 2002a). Comparison of these two maps, and between them and Fig. 11.2, reveals a striking similarity among conservation priorities for vertebrates and those for plants, in Africa.

US$750 million of globally flexible conservation resources. Entire funding mechanisms have been established to reflect global prioritization, such as the US$150 million Critical Ecosystem Partnership Fund (www.cepf.net) and the US$100 million Global Conservation Fund (web.conservation.org/xp/gef); and the ideas have been incorporated into the Resource Allocation Framework of the Global Environment Facility, the largest conservation donor.

11.1.2 Current challenges and future directions

Six major research fronts can be identified for the assessment of global biodiversity conservation priorities (Mace *et al.* 2000; Brooks *et al.* 2006). First, it remains unclear the degree to which priorities set using data for one taxon reflect priorities for others, and, by extension, whether priorities for well-known taxa like vertebrates and plants reflect those for the poorly-known, megadiverse invertebrates, which comprise the bulk of life on earth. Lamoreux *et al.* (2006), for example, found high congruence between conservation priorities for terrestrial vertebrate species. In contrast, Grenyer *et al.* (2006) reported low congruence between conservation priorities for mammals, birds, and amphibians. However, this result was due to exclusion of unoccupied cells; when this systematic bias is corrected, the same data actually show remarkably high congruence

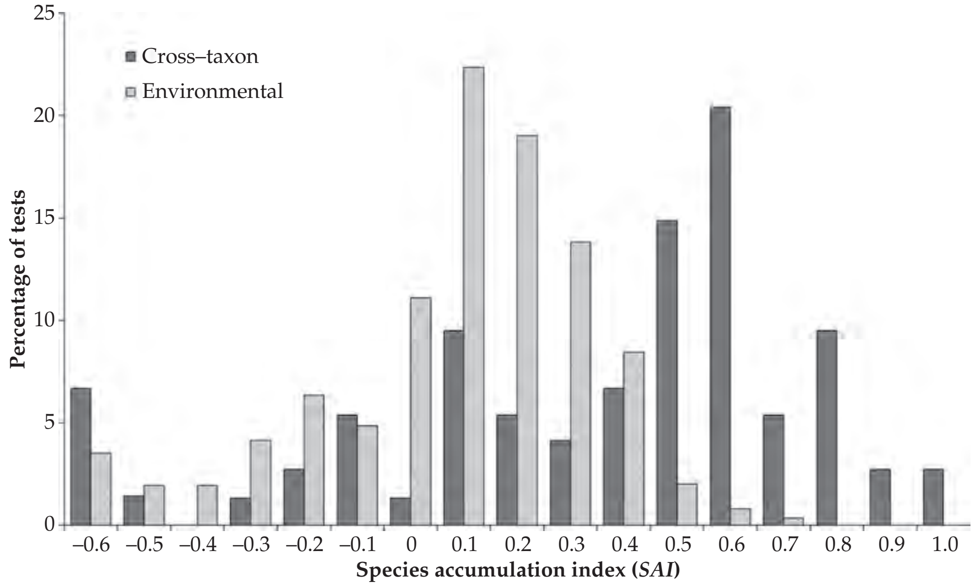


Figure 11.4 Frequency distribution of values of a species accumulation index (SAI) of surrogate effectiveness for comparison between tests on terrestrial cross-taxon and on environmental surrogates; the SAI has a maximum value of 1 (perfect surrogacy), and indicates random surrogacy when it has a value of 0, and surrogacy worse than random when it is negative (reprinted from Rodrigues and Brooks 2007).

(Rodrigues 2007). More generally, a recent review found that positive (although rarely perfect) surrogacy is the norm for conservation priorities between different taxa; in contrast, environmental surrogates rarely function better than random (Figure 11.4 and Plate X; Rodrigues and Brooks 2007).

While surrogacy may be positive within biomes, none of the conservation prioritization templates to date have considered freshwater or marine biodiversity, and at face value one might expect that conservation priorities in aquatic systems would be very different from those on land (Reid 1998). Remarkably, two major studies from the marine environment suggest that there may in fact be some congruence between conservation priorities on land and those at sea. Roberts *et al.* (2002), found that 80% of their coral hotspots, although restricted to shallow tropical reef systems, were adjacent to Myers *et al.* (2000) terrestrial hotspots. More recently, Halpern *et al.* (2008) measured and mapped the intensity of pressures on the ocean (regardless of marine biodiversity); the pressure peaks on their combined map are surprisingly close to biodiversity conservation priorities on land (the main exception being the North Sea) (see also Box 4.3). While much work remains in marine conservation prioritization, and that for freshwater biodiversity has barely even begun, these early signs suggest that there may be some geographic similarity in conservation priorities even between biomes.

Another open question is the extent to which conservation priorities represent not just current diversity but also evolutionary history. For primates and carnivores globally, Sechrest *et al.* (2002) showed that biodiversity hotspots hold a disproportionate concentration of phylogenetic diversity, with the ancient lineages of Madagascar a key driver of this result (Spathelf and Waite 2007). By contrast, Forest *et al.* (2007) claimed to find that incorporating botanical evolutionary history for the plants of the Cape Floristic hotspot substantially altered the locations of conservation priorities. Using simulations, Rodrigues *et al.* (2005) argued that phylogeny will only make a difference to conservation prioritization under specific conditions: where very deep lineages endemics are endemic to species-poor regions. Addressing the question globally across entire classes remains an important research priority.

Even if existing conservation priorities capture evolutionary history well, this does not necessarily mean that they capture evolutionary process. Indeed, a heterodox view proposes that the young, rapidly speciating terminal twigs of phylogenetic trees should be the highest conservation priorities (Erwin 1991)—although some work suggests that existing conservation priority regions are actually priorities for both ancient and young lineages (Fjeldså and Lovett 1997). Others argue that much speciation is driven from ecotonal environments (Smith *et al.* 1997) and that these are poorly represented in conservation prioritization templates (Smith *et al.* 2001). The verdict is still out.

The remaining research priorities for global conservation prioritization concern intersection with human values. Since the groundbreaking assessment of Costanza *et al.* (1997), much work has been devoted to the measurement of ecosystem service value—although surprisingly little to prioritizing its conservation (but see Ceballos and Ehrlich 2002). Kareiva and Marvier (2003) suggested that existing global biodiversity conservation priorities are less important than other regions for ecosystem service provision. Turner *et al.* (2007), by contrast, showed considerable congruence between biodiversity conservation priority and potential ecosystem service value, at least for the terrestrial realm. Moreover, that there is correspondence of both conservation priorities and ecosystem service value with human population (Balmford *et al.* 2001) and poverty (Balmford *et al.* 2003) suggests that biodiversity conservation may be delivering ecosystem services where people need them most.

Maybe the final frontier of global priority-setting is the incorporation of cost of conservation. This is important, because conservation costs per unit area vary over seven orders of magnitude, but elusive, because they are hard to measure (Polasky 2008). Efforts over the last decade, however, have begun to develop methods for estimating conservation cost (James *et al.* 1999, 2001; Bruner *et al.* 2004). These have in turn allowed assessment of the impact of incorporating costs into conservation prioritization—with initial indications suggesting that this makes a substantial difference within regions (Ando *et al.* 1998; Wilson *et al.* 2006), across countries (Balmford *et al.* 2000), and globally (Carwardine *et al.* 2008). Further, and encouragingly, it appears that incorporation of costs may actually decrease the variation in conservation priorities caused by consideration of different biodiversity datasets, at least at the global scale (Bode *et al.* 2008). The development of a fine-scale, spatial, global estimation of conservation costs is therefore an important priority for global conservation prioritization.

11.2 Conservation planning and priorities on the ground

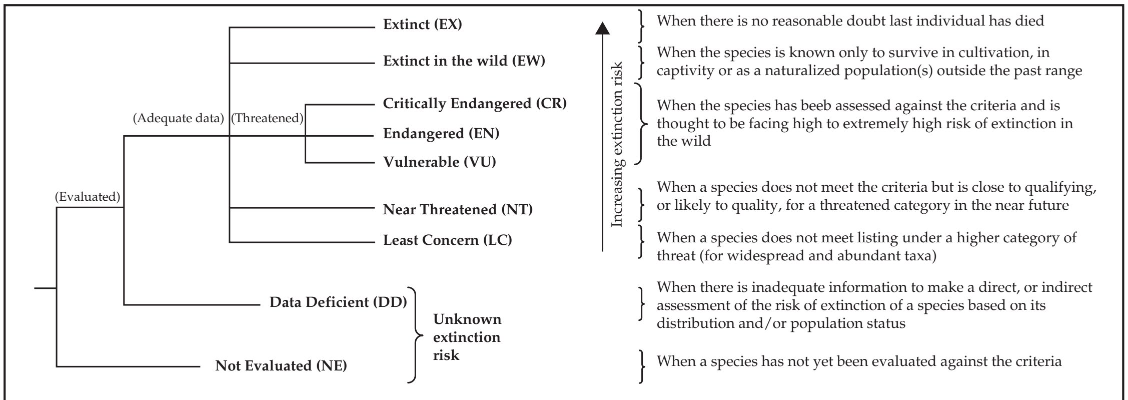
For all of the progress of global biodiversity conservation priority-setting, planning at much finer scales is necessary to allow implementation on the ground or in the water (Mace *et al.* 2000; Whittaker *et al.* 2005; Brooks *et al.* 2006). Madagascar can and should attract globally flexible conservation resources because it is a biodiversity hotspot, for example, but this does not inform the question of where within the island these resources should be invested (see Box 12.1). Addressing this question requires consideration of three levels of ecological organization—species, sites, and sea/landscapes—addressed in turn here.

11.2.1 Species level conservation planning and priorities

Many consider species the fundamental unit of biodiversity (Wilson 1992). Conversely, avoiding species extinction can be seen as the fundamental goal of biodiversity conservation, because while all of humanity's other impacts on the Earth can be repaired, species extinction, Jurassic Park fantasies notwithstanding, is irreversible. It is fitting, then, that maybe the oldest, best-known, and most widely used tool in the conservationist's toolbox informs conservation planning at the species level. This is the IUCN Red List of Threatened Species (www.iucnredlist.org).

History and state of the field

The IUCN Red List now dates back nearly 50 years, with its first volumes published in the



Criterion	Critically Endangered	Endangered	Vulnerable	Qualifiers and notes
A1: reduction in population size	≥90%	≥70%	≥50%	Over ten years/three generations[a] in the past, where causes of the reduction are clearly reversible AND understood AND ceased
A2-4: reduction in population size	≥80%	≥50%	≥30%	Over ten years/three generations[a] in past, futute or combination
B1: small range (extent of occurance)	<100km^2	<5000km^2	<20 000km^2	Plus two of (a) severe fragmentation and/or few locations (1, ≤5, ≤10); (b) continuing decline; (c) extreme fluctuation
B2: small range (area of occupancy)	<10km^2	<500km^2	<2000km^2	Plus two of (a) severe fragmentation and/or few locations (1, ≤5, ≤10); (b) continuing decline; (c) extreme fluctuation
C: small and declining population	<250	<2500	<10 000	Mature individuals. Continuing decline either: (1) over specified rates and time periods; or (2) with (a) specified population structure or (b) extreme fluctuation
D1: very small population	<50	<250	<1000	Mature individuals
D2: very restricted population	N/A	N/A	<20km^2 area of occupancy or ≤ five locations	Capable of becoming Critically Endangered or even Extinct within a very short time frame
E: quantitative analysis	≥50% in ten years/three generations[a]	≥20% in 20 years/five generations[a]	≥10% in 100 years	Estimated extinction risk using quantitative models (e.g. population viability analyses)

[a]Whichever is longer.

Figure 11.5 The IUCN Red List categories and criteria (reprinted from Rodrigues *et al.* 2006 © Elsevier). For more details see Rodrigues *et al.* (2006).

1960s (Fitter and Fitter 1987). Over the last two decades it has undergone dramatic changes, moving from being a simple list of qualitative threat assessments for hand-picked species to its current form of quantitative assessments across entire taxa, supported by comprehensive ancillary documentation (Rodrigues *et al.* 2006). The heart of the IUCN Red List lies in assessment of vulnerability at the species level, specifically in estimation of extinction risk (Figure 11.5). Because the requirements for formal population viability analysis (Brook *et al.* 2000) are too severe to allow application for most species, the IUCN Red List is structured through assessment of species status against threshold values for five quantitative criteria (IUCN 2001). These place species into broad categories of threat which retrospective analyses have shown to be broadly equivalent between criteria (Brooke *et al.* 2008), and which are robust to the incorporation of uncertainty (Akçakaya *et al.* 2000).

As of 2007, 41 415 species had been assessed against the IUCN Red List categories and criteria, yielding the result that 16 306 of these are globally threatened with a high risk of extinction in the medium-term future (IUCN 2007). This includes comprehensive assessments of all mammals (Schipper *et al.* 2008), birds (BirdLife International 2004) and amphibians (Stuart *et al.* 2004), as well as partially complete datasets for many other taxa (Baillie *et al.* 2004). Global assessments are underway for reptiles, freshwater species (fish, mollusks, odonata, decapod crustaceans), marine species (fish, corals), and plants.

It is worth a short digression here concerning irreplaceability at the species level, where phylogenetic, rather than geographic, space provides the dimension over which irreplaceability can be measured. A recent study by Isaac *et al.* (2007) has pioneered the consideration of this concept of "phylogenetic irreplaceability" alongside the IUCN Red List to derive species-by-species conservation priorities. A particularly useful application of this approach may prove to be in prioritizing efforts in *ex situ* conservation.

The benefits of the IUCN Red List are numerous (Rodrigues *et al.* 2006), informing site conservation planning (Hoffmann *et al.* 2008), environmental impact assessment (Meynell 2005), national policy (De Grammont and Cuarón 2006), and intergovernmental conventions (Brooks and Kennedy 2004), as well as strengthening the conservation constituency through the workshops process. Data from the assessments for mammals, birds, amphibians, and freshwater species to date suggest that aggregate costs for the IUCN Red List process average around US$200 per species, including staff time, data management, and, in particular, travel and workshops. This cost is expected to decrease as the process moves into assessments of plant and invertebrate species, because these taxa have many fewer specialists per species than do vertebrates (Gaston and May 1992). However, it is expected that the benefits of the process will also decrease for invertebrate taxa, because the proportion of data deficiency will likely rise compared to the current levels for vertebrate groups (e.g. ~23% for amphibians: Stuart *et al.* 2004). However, a sampled Red List approach is being developed to allow inexpensive insight into the conservation status of even the megadiverse invertebrate taxa (Baillie *et al.* 2008).

Current challenges and future directions

The main challenge facing the IUCN Red List is one of scientific process: how to expand the Red List's coverage in the face of constraints of taxonomic uncertainty, data deficiency, lack of capacity, and demand for training (Rodrigues *et al.* 2006). Some of the answer to this must lie in coordination of the IUCN Red List with national red listing processes, which have generated data on thousands of species not yet assessed globally (Rodriguez *et al.* 2000). To this end, IUCN have developed guidelines for sub-global application of the Red List criteria (Gärdenfors *et al.* 2001), but much work is still needed to facilitate the data flow between national and global levels.

One specific scientific challenge worth highlighting here is the assessment of threats driven by climate change. Climate change is now widely recognized as a serious threat to biodiversity (Thomas *et al.* 2004). However, it hard to apply the Red List criteria against climate change threats, especially for species with short generation times (Akçakaya *et al.* 2006), because climate change is rather slow-acting (relative to the time scale of the Red List criteria). Research is underway to address this limitation.

11.2.2 Site level conservation planning and priorities

With 16 306 species known to be threatened with extinction, threat rates increasing by the year (Butchart *et al.* 2004), and undoubtedly many thousands of threatened plants and invertebrates yet to be assessed, the task of biodiversity conservation seems impossibly daunting. Fortunately, it is not necessary to conserve these thousands of species one at a time. Examination of those threatened species entries on the Red List for which threats are classified reveals that habitat destruction is the overwhelming driver, threatening 90% of threatened species (Baillie *et al.* 2004). The logical implication of this is that the cornerstone of conservation action must be conserving the habitats in which these species live—establishing protected areas (Bruner *et al.* 2001). This imperative for protecting areas is not new, of course—it dates back to the roots of conservation itself—but analyses of the World Database on Protected Areas show that there are now 104 791 protected areas worldwide covering 12% of the world's land area (Chape *et al.* 2005). Despite this, however, much biodiversity is still wholly unrepresented within protected areas (Rodrigues *et al.* 2004a). The Programme of Work on Protected Areas of the Convention on Biological Diversity (www.cbd.int/protected) therefore calls for gap analysis to allow planning of "comprehensive, effectively managed, and ecologically representative" protected area systems. How can such planning best take place?

History and state of the field

Broadly, approaches to planning protected area systems can be classified into four groups. The oldest is *ad hoc* establishment, which often increases protected area coverage with minimal value for biodiversity (Pressey and Tully 1994). The 1990s saw the advent of the rather more successful consensus workshop approach, which allowed for data sharing and stake-holder buy-in, and certainly represented a considerable advance over *ad hoc* approaches (Hannah *et al.* 1998). However, the lack of transparent data and criteria still limited the reliability of workshop-based site conservation planning. Meanwhile, developments in theory (Margules and Pressey 2000) and advances in supporting software (e.g. Marxan; www.uq.edu.au/marxan), led to large scale applications of wholly data-driven conservation planning, most notably in South Africa (Cowling *et al.* 2003). However, the black-box nature of these applications led to limited uptake in conservation practice, which some have called the "research–implementation gap" (Knight *et al.* 2008).

To overcome these limitations, the trend in conservation planning for implementation on the ground is now towards combining data-driven with stakeholder-driven techniques (Knight *et al.* 2007; Bennun *et al.* 2007). This approach actually has a long history in bird conservation, with the first application of "important bird areas" dating back to the work of Osieck and Mörzer Bruyns (1981). This "site-specific synthesis" (Collar 1993–4) of bird conservation data has gained momentum to the point where important bird area identification is now close to being complete worldwide (BirdLife International 2004). Over the last decade, the approach has been extended to numerous other taxa (e.g. plants: Plantlife International 2004), and thence generalized into the "key biodiversity areas" approach (Eken *et al.* 2004). Several dozen countries have now completed key biodiversity area identification as part of their commitment towards national gap analysis under the Convention on Biological Diversity's Programme of Work on Protected Areas (e.g. Madagascar: Figure 11.6; Turkey: Box 11.1), and a comprehensive guidance manual published to support this work (Langhammer *et al.* 2007). Furthermore, all of the world's international conservation organizations, and many national ones, have come together as the Alliance for Zero Extinction (AZE), to identify and implement action for the very highest priorities for site-level conservation (Ricketts *et al.* 2005, Figure 11.7 and Plate 12).

The key biodiversity areas approach, in alignment with the conceptual framework for conservation planning (Margules and Pressey 2000), is based on metrics of vulnerability and irreplaceability (Langhammer *et al.* 2007). Their vulnerability criterion is derived directly from the IUCN Red List, through the identification of sites regularly holding threshold populations of one or more threatened species. The irreplaceability

Figure 11.6 Location and protection status of the Key Biodiversity Areas (KBAs) of Madagascar (reprinted from Langhammer *et al.* 2007).

criterion is based on regular occurrence at a site of a significant proportion of the global population of a species. This is divided into sub-criteria to recognize the various situations under which this could occur, namely for restricted range species, species with clumped distributions, congregatory populations (species that concentrate during a portion of their life cycle), source populations, and biome-restricted assemblages. The reliance on occurrence data undoubtedly causes omission errors (where species occur in unknown sites) and hence the approach overestimates irreplaceability. These omission errors could in theory be reduced by use of modeling or extrapolation techniques, but these instead yield dangerous commission errors, which could lead to extinction through a species wrongly considered to be safely represented (Rondinini *et al.* 2006). Where such techniques are of proven benefit is in identifying research priorities (as opposed to conservation priorities) for targeted field surveys (Raxworthy *et al.* 2003).

To facilitate implementation and gap analysis, key biodiversity areas are delineated based on existing land management units, such as protected areas, indigenous or community lands, private concessions or ranches, and military or other public holdings (Langhammer *et al.* 2007). Importantly, this contrasts with subdivision of the entire landscape into, for example, grid cells, habitat types, or watersheds. While grid cells have the advantage of analytical rigor, and habitats and watersheds deliver ecological coherence, these spatial units are of minimal relevance to the stakeholders on whom conservation on the ground fundamentally depends. Indeed, the entire key biodiversity areas process is designed to build the constituency for local conservation, while following global standards and criteria (Bennun *et al.* 2007). The costs and benefits of site conservation planning approaches have yet to be fully evaluated, but some early simulation work suggests that the benefits of incorporation of primary biodiversity data are large (Balmford and Gaston 1999).

Current challenges and future directions

Three important challenges can be discerned as facing site level conservation planning. The first stems from the fact that most applications of these approaches to date come from fragmented habitats—it often proves difficult to identify sites of global biodiversity conservation significance in regions that retain a wilderness character, for instance, in the Amazon (Mittermeier *et al.* 2003). Under such circumstances, the omission errors attendant on use of occurrence data (because of very low sampling density) combine with difficulty in delineating sites (because of overlapping or non-existent land tenure). These problems can, and indeed must, be overcome by delineating very large key biodiversity areas, which is still a possibility in such environments (e.g. Peres 2005).

The second challenge facing site level conservation planning is its extension to aquatic environments. Human threats to both freshwater and marine biodiversity are intense, but species assessments in these biomes are in their infancy (see above), seriously hampering conservation planning. Difficulties of low sampling density and delineation are also challenging for conservation planning below the water, as in wilderness regions on land. Nevertheless, initial scoping suggests that the application of the key biodiversity areas approach will be desirable in both freshwater (Darwall and Vié 2005) and marine (Edgar *et al.* 2008a) environments, and proof-of-concept from the Eastern Tropical Pacific shows that it is feasible (Edgar *et al.* 2008b).

The third research front for the key biodiversity areas approach is prioritization (Langhammer *et al.* 2007)—once sites have been identified and delineated as having global biodiversity conservation significance, which should be assigned the most urgent conservation action? This requires the measurement not just of irreplaceability and species vulnerability, but also of site vulnerability (Bennun *et al.* 2005). This is because site vulnerability interacts with irreplaceability: where irreplaceability is high (e.g., in AZE sites), the most threatened sites are priorities, while where irreplaceability is lower, the least vulnerable sites should be prioritized. This is particularly important in considering resilience (i.e. low vulnerability) of sites in the face of climate change. As with global prioritization (see above), it is also important to strive towards incorporating

Box 11.1 Conservation planning for Key Biodiversity Areas in Turkey

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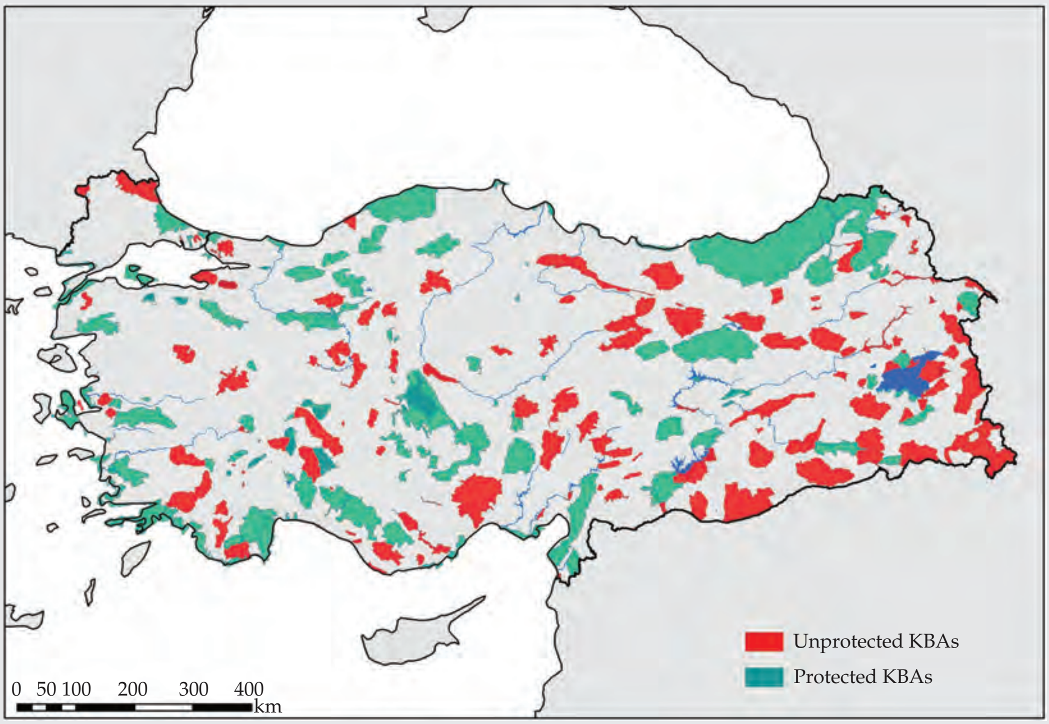
An impressive set of projects has already been carried out to map priority areas for conservation in Turkey. These include three inventories of Important Bird Areas (Ertan *et al.* 1989; Magnin and Yarar 1997; Kılıç and Eken 2004), a marine turtle areas inventory (Yerli and Demirayak 1996), and an Important Plant Areas inventory (Özhatay *et al.* 2003). These projects, collectively, facilitated on-the-ground site conservation in Turkey and drew attention to gaps in the present protected areas network.

We used the results of these projects as inputs to identify the Key Biodiversity Areas (KBAs) of Turkey, using standard KBA criteria across eight taxonomic groups: plants, dragonflies, butterflies, freshwater fish, amphibians, reptiles, birds, and mammals. As a result of this study, an inventory of two volumes (1112 pages) was published in Turkish fully documenting the country's KBAs (Eken *et al.* 2006).

We used the framework KBA criteria developed by Eken *et al.* (2004) and assessed 10 214 species occurring in Turkey against these criteria. Two thousand two hundred and forty six species triggered one or more KBA criteria. These include 2036 plant species (out of 8897 in Turkey; 23%), 71 freshwater fish (of 200; 36%), 36 bird (of 364; 10%), 32 reptile (of 120; 27%), 28 mammal (of 160; 18%), 25 butterfly (of 345; 7%), 11 amphibian (of 30; 37%), and 7 dragonfly (of 98; 7%) species. Then, we assessed all available population data against each KBA criterion and its threshold to select KBAs.

We identified 294 KBAs qualifying on one or more criteria at the global scale (Box 11.1 Figure and Plate 11). Two KBAs met the criteria for seven taxon groups, while 11 sites met them for six and 18 for five taxon groups. The greatest number of sites, 94, met the KBA criteria for two taxon groups, while 86 sites (29%) triggered the criteria for one taxon group only.



Box 11.1 Figure The 294 KBAs (Key Biodiversity Areas) of global importance identified in Turkey. While 146 incorporate protected areas (light), this protection still covers <5% of Turkey's land area. The remaining 148 sites (dark) are wholly unprotected.

continues

Box 11.1 (Continued)

The greatest number of sites, 223, was selected based on the criteria for plants, followed by reptiles and birds with 108 and 106 sites selected respectively. For other groups, smaller numbers of sites triggered the KBA criteria at the global scale: 95 KBAs were selected for mammals, 66 for butterflies, 61 for freshwater fish, and 29 each for amphibians and dragonflies. The number of sites selected for plants is actually rather low, given the high number of plant species in Turkey which trigger the KBA criteria. This can be explained by the overlapping distributions of restricted-range and threatened plants. The other taxon groups have relatively greater numbers of sites. For instance, the seven dragonfly species triggered the KBA criteria for 29 sites. One exception is the freshwater fish, which, like plants, have highly overlapping ranges.

Large scale surface irrigation, drainage, and dam projects form the most significant threats to Turkey's nature. Irrigation and drainage projects affect 74% of the KBAs and dams have an effect on at least 49%. Inefficient use of water, especially in agriculture, is the root cause of these threats. A total of 40 billion m^3 of water is channeled annually to agriculture (75%), industry (10%), and domestic use (15%), but 50–90% of water used for agriculture is lost during the transportation from dams to arable land. As a result of these threats, wetlands and associated grasslands are Turkey's most threatened habitat types. At least five wetland KBAs (Eşmekaya Marshes, Hotamış Marshes, Sultan Marshes, Ereğli Plain, and Seyfe Lake) have been lost entirely over the last decade, and other sites have lost at least 75% of their area during the same period.

Less than 5% of the surface area of Turkey's KBAs is legally protected, and so this should be expanded rapidly and strategically. Steppe habitats, river valleys, and Mediterranean scrublands are particularly poorly covered by the current network of protected areas. Wildlife Development Reserves, Ramsar Sites and, in the future, Natura 2000 Sites, would likely be appropriate protected area categories for this expansion.

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cost of conservation. Given these complexities, considerable promise may lie in adapting conservation planning software to the purpose of prioritizing among conservation actions across key biodiversity areas.

11.2.3 Sea/landscape level conservation planning and priorities

The conservation community has more than 40 years experience with conservation planning at the species level, and more than 20 at the site level. However, the recent growth of the field of landscape ecology (Turner 2005) sounds a warning that while species and site planning are essential for effective biodiversity conservation, they are not sufficient. Why not, and how, then, can conservation plan beyond representation, for persistence?

History and state of the field

The first signs that conserving biodiversity in isolated protected areas might not ensure

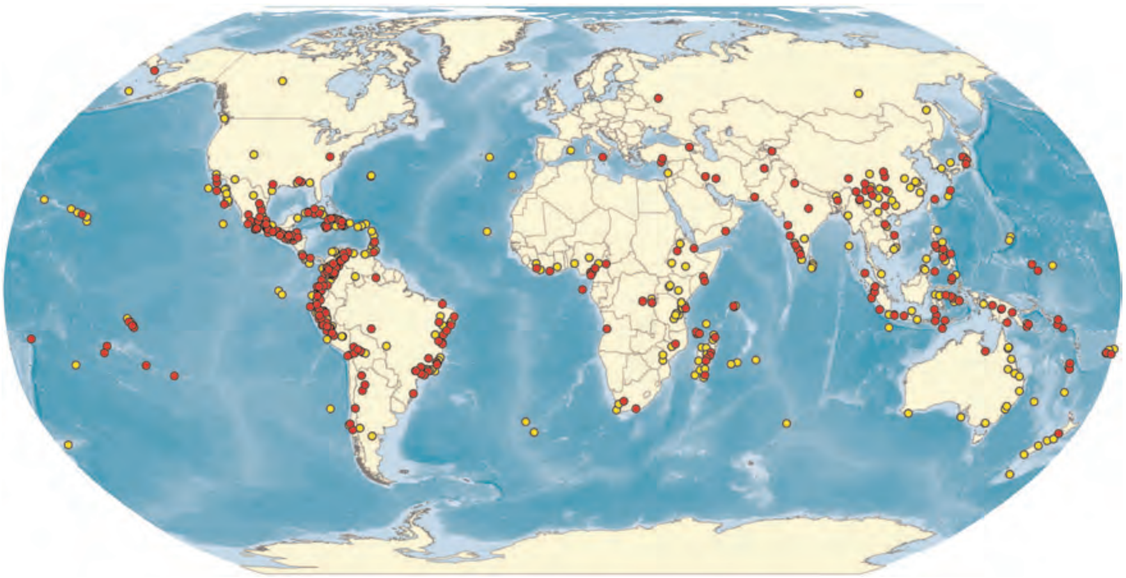

Figure 11.7 Map of 595 sites of imminent species extinction (reprinted from Ricketts *et al.* 2005). Yellow sites are either fully protected or partially contained within declared protected areas (n = 203 and 87, respectively), and red sites are completely unprotected or have unknown protection status (n = 257 and 48, respectively). In areas of overlap, unprotected (red) sites are mapped above protected (yellow) sites to highlight the more urgent conservation priorities.

persistence came from evidence of long-term extinctions of mammal species from North American national parks (Newmark 1987). Over the following decade, similar patterns were uncovered across many taxa, unfolding over the time-scale of decades-to-centuries, for megadiverse tropical ecosystems in Latin America (e.g. Robinson 1999), Africa (e.g. Brooks *et al.* 1999), and Asia (e.g. Brook *et al.* 2003). Large-scale experiments, most notably the Manaus Biological Dynamics of Forest Fragments project, provide increasingly refined evidence (Bierregaard *et al.* 2001). The mechanisms determining persistence—or extinction—in individual sites spans the full spectrum from the genetic scale (Saccheri *et al.* 1998; see Chapters 2 and 16) through populations (Lens *et al.* 2002) and communities (Terborgh *et al.* 2001), to the level of ecosystem processes across entire landscapes (Saunders *et al.* 1991; see Chapter 5).

The first recommendations of how conservation planning might address persistence at landscape scales were generic design criteria for the connectivity of protected areas (Diamond 1975). Conservation agencies were quick to pick up the concept, and over the last twenty years a number of large scale conservation corridors have been designed (Crooks and Sanjayan 2006), for example, the "Yellowstone to Yukon" (Raimer and Ford 2005) and Mesoamerican Biological Corridor (Kaiser 2001). There is no doubt that the implementation of corridors benefits biodiversity (Tewksbury *et al.* 2002). However, the establishment of generic corridors has also been criticized, in that they divert conservation resources from higher priorities in protected area establishment, and, even worse, have the potential to increase threats, such as facilitating the spread of disease, invasive, or commensal species (Simberloff *et al.* 1992).

Given these concerns, there has been a shift towards specification of the particular objectives for any given corridor (Hobbs 1992). A promising avenue of enquiry here has been to examine the needs of "landscape species" which require broad scale conservation (Sanderson *et al.* 2002b). Boyd *et al.* (2008) have generalized this approach, reviewing the scales of conservation required for all threatened terrestrial vertebrate species (Figure 11.8 and Plate 13). They found that 20% (793) of these threatened species required urgent broad scale conservation action, with this result varying significantly among taxa (Figure 11.9). They also asked why each of these species required broad scale conservation. This yielded the surprising finding that while only 43% of these 793 species were "area-demanding" and so required corridors for movement, no less than 72% were dependent on broad scale ecological processes acting across the landscape (15% require both). In this light, recent work in South Africa to pioneer techniques for incorporating ecosystem processes into conservation planning is likely to be particularly important (Rouget *et al.* 2003, 2006).

Current challenges and future directions

As at the species and site levels, the incorporation of broad scale targets into conservation planning in aquatic systems lags behind the terrestrial environment. Given the regimes of flows and currents inherent in rivers and oceans, the expectation is that broad scale conservation will be even more important in freshwater (Bunn and Arthington 2002) and in the sea (Roberts 1997) than it is on land. Boyd *et al.*'s (2008) results are consistent with this, with 74% of threatened marine tetrapods requiring broad scale conservation, and 38% in freshwater, and only 8% on land (Figure 11.9). This said, some recent work suggests that marine larval dispersal occurs over much narrower scales than previously assumed (Jones *et al.* 1999) and so there is no doubt that site level conservation will remain of great importance in the water as well as on land (Cowen *et al.* 2006).

A second research front for sea/landscape conservation planning concerns dynamic threats. Recent work has demonstrated that changes in the nature and intensity of threats over time have important consequences for the prioritization of conservation actions among sites (Turner and Wilcove 2006). Such dynamism introduces particular complications when considered at the landscape scale, the implications of which are only just beginning to be addressed (Pressey *et al.* 2007). Climate change is one such threat that will very likely require extensive landscape scale response (Hannah *et al.* 2002), and may be even

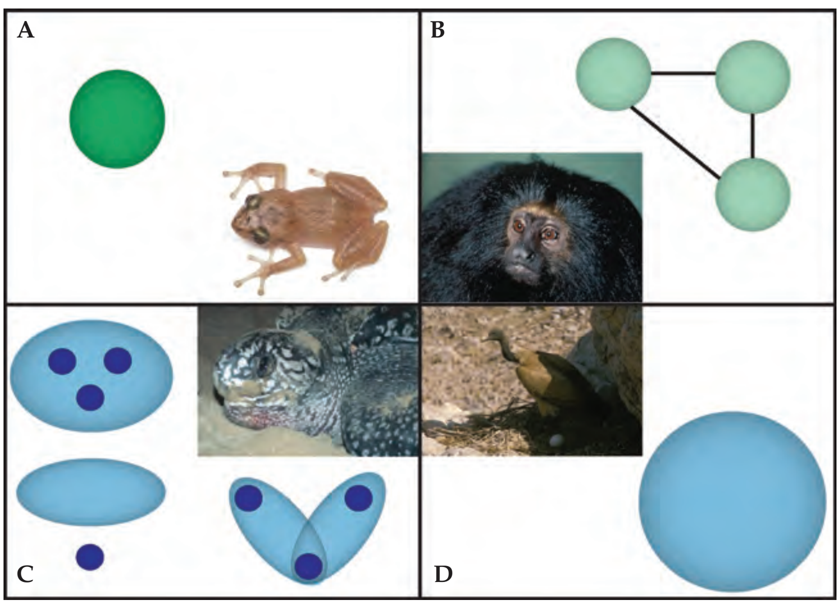

Figure 11.8 Scale requirements for the conservation of globally threatened species in the short- to medium term (reprinted from Boyd *et al.* 2008). (A, dark green) Species best conserved at a single site (e.g. *Eleutherodactylus corona*); (B, pale green) Species best conserved at a network of sites (e.g. black lion-tamarin *Leontopithecus chrysopygus*); (C, dark blue) Species best conserved at a network of sites complemented by broad-scale conservation action (e.g. leatherback turtle *Dermochelys cariacea*); (D, pale blue) Species best conserved through broad-scale conservation action (e.g. Indian vulture *Gyps indicus*). Photographs by S. B. Hedges (A), R.A. Mittermeier (B), O. Langrand (C), and A. Rahmani (D).

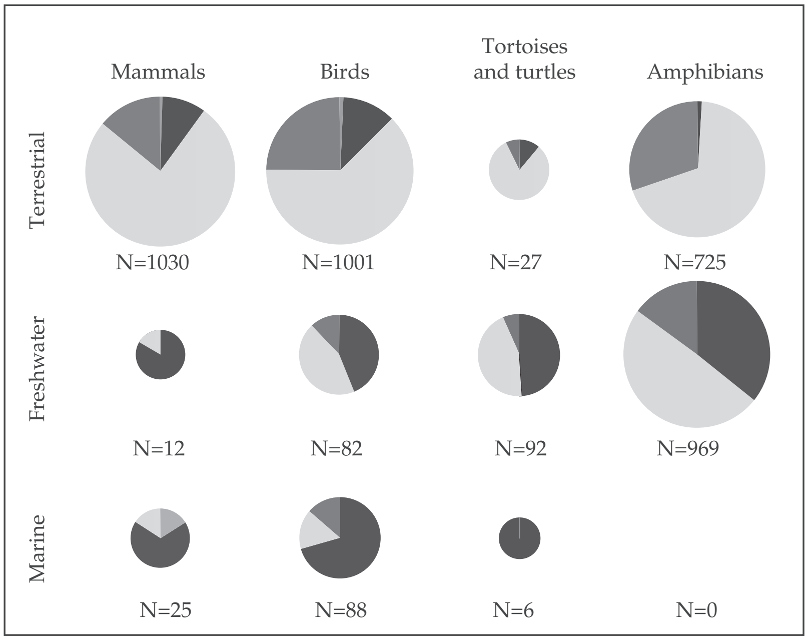

Figure 11.9 Percentages of globally threatened species requiring different scales of conservation in the short- to medium term (reprinted from Boyd *et al.* 2008). Dark green = species best conserved at a single site; pale green = species best conserved at a network of sites; dark blue = species best conserved at a network of sites complemented by broad-scale conservation action; pale blue = species best conserved through broad-scale conservation action. The totals exclude species insufficiently known to assess the appropriate scale required. Relative size of pies corresponds to the number of species in each taxon/biome combination.

more serious in freshwater (Roessig *et al.* 2004) and the ocean (Xenopoulos *et al.* 2005) (see Chapter 8).

Maybe the largest open research challenge for sea/landscape conservation planning is to move from maintaining current biodiversity towards restoring biodiversity that has already been lost (Hobbs and Norton 1996). Natural processes of succession provide models of how this can proceed most effectively (Dobson *et al.* 1997). However, restoration is much more expensive and much less likely to succeed than is preservation of biodiversity before impacts occur, and so explicit planning towards the specific biodiversity targeted to be restored is essential (Miller and Hobbs 2007). Given these costs and challenges, most efforts to date target very tightly constrained ecosystems that, as restoration proceeds, are then managed at site scales—wetlands are the best example (Zedler 2000). A few ambitious plans for landscape level restoration have already been developed (Stokstad 2008). Moreover, the current explosive growth in markets for carbon as mechanisms for climate change mitigation will likely make the restoration of forest landscapes increasingly viable in the near future (Laurance 2008). Ultimately, planning should move from simple restoration to designing landscapes that allow the sustainability of both biodiversity and human land uses, envisioned as "countryside biogeography" (Daily *et al.* 2001) or "reconciliation ecology" (Rosenzweig 2003).

11.3 Coda: the completion of conservation planning

The research frontiers outlined in this chapter are formidable, but conservation planning is nevertheless a discipline with its completion in sight. It is not too far of a stretch to imagine a day where top-down global prioritization and bottom-up conservation planning come together. Such a vision would encompass:

- The completion and continuous updating of IUCN Red List assessments of all vertebrate and plant species, plus selected invertebrate groups.
- Iterative identification of key biodiversity areas, based on these data, representing the full set of sites of global biodiversity conservation significance.
- Measurement and mapping of the continuous global surface of seascape and landscape scale ecological processes necessary to retain these species and sites into the future.
- Continuous measurement and mapping of the threats to these species, sites, and sea/landscapes, and of the costs and benefits of conserving them.
- Free, electronic, continuously updated access to these datasets, and to tools for their interpretation, planning, and prioritization.

A particularly important characteristic of such a vision is its iterative nature. As knowledge of biodiversity increases, threats and costs change, and conservation is implemented successfully (or not) it is crucial that mechanisms exist to capture these changing data, because changes to any one of these parameters will likely impinge on conservation planning across the board.

Under such a vision, it would be possible, at any given point in time, to maximize the overall benefits of a conservation investment at any scale, from *ex situ* management of a particular species, through gap analysis by a national protected areas agency, to investment of globally flexible resources by institutions like the GEF. Given the pace of advance in conservation planning over the last 20 years, it is possible that such a vision is achievable within the coming few decades. Its realization will provide great hope for maintaining as much of the life with which we share our planet as possible.

Summary

- Conservation planning and prioritization are essential, because both biodiversity and human population (and hence threats to biodiversity and costs and benefits of conservation) are distributed highly unevenly.
- Great attention has been invested into global biodiversity conservation prioritization on land over the last two decades, producing a broad consensus that reactive priority regions are concentrated in the tropical mountains and islands, and proactive priorities in the lowland tropical forests.
- Major remaining research fronts for global biodiversity conservation prioritization include the examination of cross-taxon surrogacy, aquatic priorities, phylogenetic history, evolutionary process, ecosystem services, and costs of conservation.
- Maybe the most important tool for guiding conservation on the ground is the IUCN Red List of Threatened Species, which assesses the extinction risk of 41 415 species against quantitative categories and criteria, and provides data on their distributions, habitats, threats, and conservation responses.
- The predominant threat to biodiversity is the destruction of habitats (Chapter 4), and so the primary conservation response must be to protect these places through safeguarding key biodiversity areas.
- While protecting sites is essential for biodiversity conservation, persistence in the long term also requires the conservation of those landscape and seascape level ecological processes that maintain biodiversity.

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Relevant websites

- BirdLife International Datazone: http://www.birdlife.org/datazone.
- IUCN Red List of Threatened Species: http://www.iucnredlist.org.

- World Database on Protected Areas: http://www.wdpa.org.
- Alliance for Zero Extinction: http://www.zeroextinction.org.

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