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Key biodiversity areas as globally significant target sites for the conservation of marine biological diversity

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ABSTRACT

1. Recent approaches to the planning of marine protected area (MPA) networks for biodiversity conservation often stress the need for a representative coverage of habitat types while aiming to minimize impacts on resource users. As typified by planning for the Australian South-east Marine Region, this strategy can be manipulated by political processes, with consequent biased siting of MPAs. Networks thus created frequently possess relatively low value for biodiversity conservation, despite significant costs in establishment and maintenance.

2. Such biases can be minimized through application of the data-driven and species-based concept of key biodiversity areas (KBAs).

3. By mapping locations of threatened species and populations that are highly aggregated in time or space, the KBA process allows marine sites of global biodiversity significance to be systematically identified as priority

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conservation targets. Here, the value of KBAs for marine conservation planning is outlined, and guidelines and provisional criteria for their application provided.

KEY WORDS: endemism; marine protected area; MPA; IUCN Red List; systematic conservation planning; threatened species

INTRODUCTION

As threats to biodiversity increase, conservation managers and donor organisations require increasingly sophisticated tools for decision-making; above all, ways to prioritize conservation actions that are efficient, accountable and transparent. At the global scale, the 'biodiversity hotspots' approach (Myers *et al*., 2000) provides an example of the way that data-driven analyses can assist prioritization of conservation actions and significantly leverage new conservation dollars (Brooks *et al*., 2006). At regional and local scales, progress towards a data-driven approach to conservation is most evident in site planning, which is now central to much conservation action across the world (Margules and Pressey, 2000).

While species are the predominant units of biodiversity, time and resources are not available to conserve species one-by-one (Ehrlich, 1992). The conservation of important sites with associated habitats as protected areas or through other safeguard mechanisms is therefore generally seen as the best strategy to maintain biodiversity (Bruner *et al*., 2001; Brooks *et al*., 2004; Pressey, 2004). This is implicit, for example, in the call from the 5th IUCN World Parks Congress to: 'Maximize representation and persistence of biodiversity in comprehensive protected area networks, focusing especially on threatened and under-protected ecosystems and species globally threatened with extinction' (www.iucn.org/themes/wcpa/wpc2003/english/outputs/durban/cbdmessage.htm). Threat reduction and additional conservation actions are clearly also necessary (Cicin-Sain and Belfiore, 2005); nevertheless, even for species that are wide-ranging across landscapes and seascapes, protected areas often serve as 'anchors' for conservation strategies involving ecological networks or corridors.

Central to conservation planning is the question: 'How can the locations of protected areas be best identified?' This question is particularly pertinent within the field of marine planning given increasing recognition of the extent of deterioration in the marine environment and the potential importance of the role of marine protected areas (MPAs) (Roberts and Hawkins, 1999; Dulvy *et al*., 2003; Glover and Earle, 2004; Worm *et al*., 2006). The issue is also intertwined with social and political considerations, given that intense pressure is often placed on decision-makers to minimize perceived impacts of new protected areas on existing users (Davis, 1981; Lynch, 2006).

To provide the most effective outcomes, and to offset pressures of partisan stakeholders, efficient systems of identifying and prioritizing candidates for protected sites are required. Ideally, such systems should be quantitative and explicit, while also understandable to stakeholder groups and non-scientific decision-makers. Here, we briefly review the representative habitat approach typically used in planning networks of MPAs for biodiversity conservation, and highlight a serious shortcoming.

To help overcome current limitations in the development of MPA networks, we also outline the key biodiversity area (KBA) concept, where sites of global conservation significance that are, or can potentially be, managed for conservation are identified using the principles of vulnerability and irreplaceability (Eken *et al*., 2004; Langhammer *et al*., 2007). Vulnerable sites are those holding one or more globally threatened species while irreplaceable sites are those holding a significant proportion of the global population of a species.

The term 'site' in the KBA context refers to a landscape or seascape unit that (i) can be delimited on maps, (ii) encompasses the important habitat used by the species of conservation concern, and (iii) can actually or potentially be managed as a single unit for conservation. Following this definition, KBAs can vary greatly in size, each possessing a site boundary that corresponds to the most practical conservation unit, where contiguous habitat, local management units, and the potential for significant gene flow among populations are all considered.

In the marine context a KBA will often, but not always, be effectively safeguarded as an MPA. The choice of an appropriate conservation tactic will depend on the best mechanism to protect the target species within a given KBA. Managed fishery or tourism sites may in some instances provide better protection for a KBA than provided by MPA designation, particularly if the alternative strategy promotes local ownership and allows more effective control of illegal exploitation.

Within the conservation managers' toolbox, KBAs provide an important tool alongside strategies currently used to safeguard representative habitats, to protect individual species from idiosyncratic threats, and to reduce broad-scale pressures and threats affecting wide-ranging species. While further testing is necessary, the incorporation of a KBA approach into marine conservation planning should ensure that MPAs and other site conservation tactics are targeted towards the places where they are most necessary to prevent species' extinctions.

LIMITATIONS OF CURRENT MARINE CONSERVATION PLANNING METHODS

Despite recognition that site protection is fundamental to conservation, national systems of protected areas, including MPAs, are far from systematic in their coverage (Rodrigues *et al*., 2004; Mora *et al*., 2006). Gaps and redundancies in the distribution of protected sites result not only in inefficiency but failure to protect important biodiversity (Margules and Pressey, 2000).

Since pioneering work two decades ago (Kirkpatrick, 1983), a number of tools for identifying and prioritizing sites of conservation significance have been proposed or are currently under development (Pressey, 2004). These include reserve selection algorithms that use a complementarity approach to maximize the range of conservation targets such as habitats or species to be protected within a protected area network of given size (Beger *et al*., 2003; Leslie *et al*., 2003; Stewart *et al*., 2003).

While conservation planning tools have progressed most rapidly in terrestrial systems, data-driven procedures are also increasingly used when defining frameworks for marine conservation planning (Roff and Taylor, 2000; Roff and Evans, 2002; Roff *et al*., 2003), and to assist the development of systematic networks of MPAs (Ward *et al*., 1999; Sala *et al*., 2002; Beger *et al*., 2003; Roberts *et al*., 2003; Stewart *et al*., 2003; Campagna *et al*., 2007).

Planning of MPA networks to conserve biodiversity has relied heavily to date on the concept of representation — the need to encompass a wide range of different habitat types (e.g. ANZECC, 1999). Thus, MPA planning typically involves division of a seascape into mappable units such as bioregions and habitats, with stakeholder-driven processes then applied to select representative subsets of each of the mapped units for protection. A major limitation of this process is that it often, although not always (e.g. in the terrestrial realm, Cowling *et al*., 2003), overlooks vulnerability — recognition that some sites hold species that are at higher risk of extinction and require prioritized conservation action (Margules and Pressey, 2000).

A compounding problem of current MPA selection processes is that outcomes are affected by biases associated with stakeholder input. During negotiations over MPA establishment and boundaries, stakeholders who utilize local resources (e.g. fisheries, tourism, oil extraction) typically know where those resources are concentrated, and attempt to keep such areas out of protected area networks (Lynch, 2006). By contrast, the conservation sector rarely has accurate information on the most important sites for conservation, resulting in areas of low conservation value often ending up designated as 'representative' areas within finalized MPA systems.

MPA networks derived through such negotiations can possess reduced value for biodiversity conservation and involve large costs in terms of forgone opportunities to create more effective MPA networks for the same total cost. This situation parallels terrestrial conservation planning of the last century when alpine regions, deserts and other areas with little resource value were disproportionately designated national parks — the so-called 'worthless lands' scenario (Runte, 1977; Pressey, 1994), in the sense that lands allocated to conservation often possessed negligible economic value other than for tourism.

Biases in MPA location develop at the local level during face-to-face negotiations between stakeholders, and can also be formalized at regional levels within government-mandated MPA selection strategies. The Galapagos Marine Reserve (GMR) provides an example of a network of sanctuary zones developed using a bottom-up, stakeholder-driven process with substantial bias. In Galapagos, sanctuary zones were identified following a series of face-to-face meetings involving representatives of fishing, tourism, conservation, science and management sectors that extended over 12 months and culminated in an extended boat cruise to finalize zone boundaries (Heylings *et al*., 2002; Edgar *et al*., 2008). Negotiations followed guidelines outlined in the GMR Management plan, which called for the development of a network of conservation zones that included representative habitats, rather than species, within recognized Galapagos bioregions.

At the commencement of negotiations, the Galapagos artisanal fishing sector advocated that no areas be excluded from fishing, while the science and conservation sector proposed 36% of the coast as 'no-take' zones. Consensual agreement was eventually reached on a total of 14 'no-take' conservation zones (6% of the coast) and 62 small 'no-take' tourism zones that were also regarded as possessing high conservation value (additional 11% of coast). Because the fishing sector would not agree to sanctuary zones in important fishery areas, almost all conservation zones were located along coasts with little fishery resources or with limited commercial diver access. Sites with known concentrations of sharks were included within tourism zones.

The environmental outcome of these negotiations was quantified during archipelago-wide surveys of resources at the conclusion of negotiations (Edgar *et al*., 2004). Mean densities of major fishery species (sea cucumbers and spiny lobsters) were about three times higher in areas agreed to remain open for fishing compared to conservation zones, a consequence of fishers vetoing all conservation zones proposed for resource rich areas. Areas agreed as tourism zones possessed sharks — the major dive tourism resource — with mean densities five times higher than in conservation zones.

An example of bias that is formally embedded in MPA selection is the process used recently by the Australian Government to delineate MPAs in its South-east Marine Region, one of the largest MPA networks worldwide,

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encompassing 226 155 km^2 of seabed. The government agency responsible for this process initially consulted with stakeholders and then distributed a draft set of MPA zones for public comment. The specific criteria used to identify MPAs proposed in the initial draft were not clearly enunciated but appear to have been: (i) a wide range of habitat types should be included; and (ii) existing and prospective petroleum leases and fishing grounds should be avoided. Only one proposed MPA overlapped with a petroleum lease, while several had boundaries contiguous with petroleum leases (Figure 1(A); Buxton *et al*., 2006).

The fishing sector lobbied strongly during the public consultation phase that all sites with significant commercial fishing activity should not be included within MPAs because of social and economic costs, or, if this was not possible, then pre-existing fishing methods should be allowed to continue within protected zones (Buxton *et al*., 2006). Zoning amendments requested by fishers were largely accommodated (Figure 1(B)), allowing the Australian Minister for the Environment and Heritage to announce (5 May 2006: http://www.deh.gov.au/minister/env/2006/mr05may06.html) 'We have made more than 20 adjustments to boundaries and zoning that will reduce the impact on commercial fishing by more than 90 per cent. . . . The new MPA network will not prevent prospective oil and gas areas from being explored and developed.' Implicit in this statement is the assumption that a useful and comprehensive MPA network can be developed, and threats to biodiversity addressed, with negligible change to existing activity.

Although a total of 8% of the continental shelf in the South-east Marine Region is to be recognized as MPAs of one form or another, and 42% of the total MPA area comprises 'no-fishing' sanctuary zones, only one sanctuary zone is located on the continental shelf (*ca* 0.4% of regional shelf waters) within the MPA network (http://www.deh.gov.au/coasts/mpa/southeast/index.html). Longlining, charter fishing and aquaculture are permitted in almost all sections of MPAs on the continental shelf and slope, while sanctuary zones are located almost exclusively in abyssal areas >1500 m depth. Scallop dredging, the fishing activity within the region that is recognized to cause most environmental harm (Kaiser *et al*., 2006), is not affected by the new zoning scheme (Stump and Sansom, 2006). Sites of recognized conservation significance have not been included in fully protected zones, such as the highly productive Bonney Upwelling where Endangered blue whales (*Balaenoptera musculus*) congregate.

Thus, despite the impressive appearance and size of this MPA network on paper, human activity will continue virtually unchanged across the Australian seascape. Major conservation benefits arising from the new MPA network with respect to existing activities are negligible, despite clear evidence that marine biodiversity within the region has already declined substantially over the past century (Edgar and Samson, 2004; Edgar *et al*., 2005).

Rather than representing a failure of the representation approach to marine planning, the above example could also be regarded as an example of political opportunism producing poor conservation outcomes that are promoted as a great advance to a poorly-informed public; however, such manipulation is assisted by the latitude available to policy makers in defining which habitat types require representative coverage. Very little scientific information exists on which physical surrogates are most effective at delineating habitat or ecosystem types in conservation planning. Consequently, planners can credibly place emphasis on, for example, a division of the seascape into geomorphological units over a division based on primary productivity, water temperature, wave energy, current strength, or depth, whereas these latter factors, or others including history, may predominantly influence species' distributions.

This source of potential bias is likely to diminish with spatial scale (R.L. Pressey, pers. commun.). At broad regional scales, marine planners possess enormous flexibility in the siting of protected areas to achieve representation targets, hence lobbying by extractive interests can readily divert conservation attention away from areas and species most under threat. At finer scales, planners possess less spatial flexibility relative to the scale of exploited resources, and avoidance of such areas becomes more difficult.

KEY BIODIVERSITY AREAS AS PRIORITY SITE CONSERVATION TARGETS

To redress stakeholder-associated biases in MPA selection, conservation managers require an objective protocol to identify sites of highest significance for biodiversity conservation. The 'Key Biodiversity Area' (KBA) approach fills this need (Eken *et al*., 2004; Langhammer *et al*., 2007), providing a complement to existing methodologies that identify and select representative marine areas for protection within MPA networks.

KBA methods are based on the rationale that the extinction of any species represents a loss of global significance. Biodiversity clearly declines at the species and genetic levels following extinction. It can also decline at the ecosystem level, depending on the ecological role of lost species. KBAs are selected using standardized, globally-applicable criteria based on species vulnerability and site irreplaceability (Margules and Pressey, 2000). A KBA site meeting the vulnerability criterion comprises the confirmed locality of a Critically Endangered or Endangered species, or more than $\sim$30 individuals of a Vulnerable species (following the IUCN Red List categories

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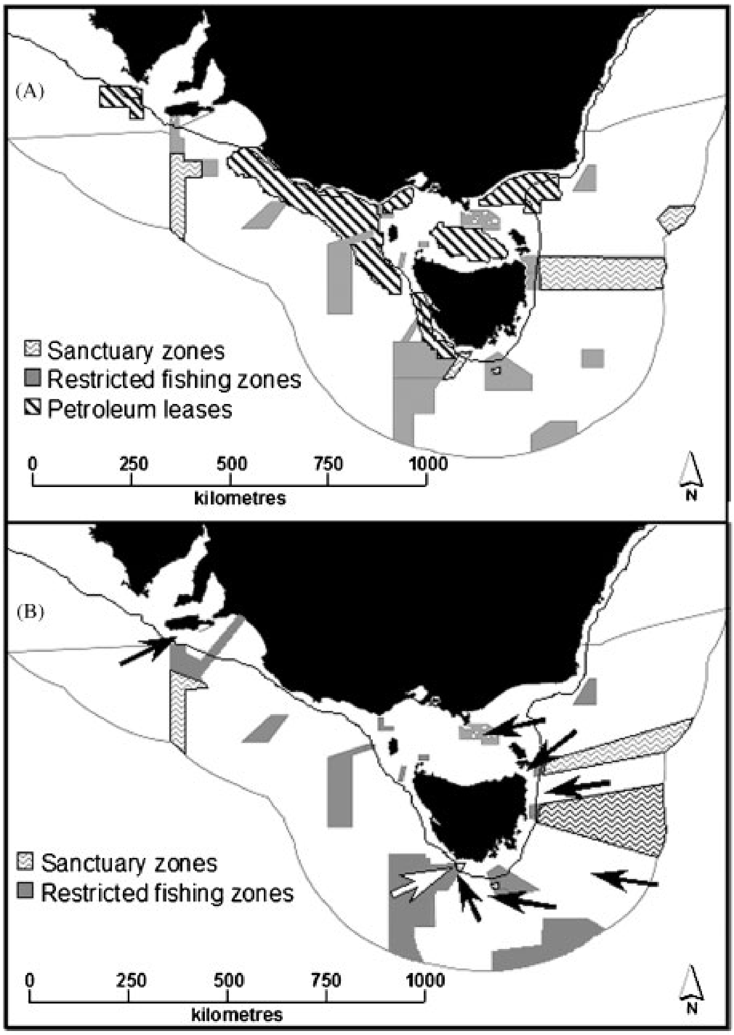

Figure 1. (A) Boundaries initially proposed for MPAs in the South-east Marine Region of Australia with 'no-take' sanctuary zones and with zones of limited permitted fishing. (B) Finalized MPA zone boundaries for the South-east Marine Region. Filled arrows indicate locations identified by the fishing industry as important for commercial fisheries after public dissemination of proposed zones depicted in (A). Open arrow indicates sole sanctuary zone present on continental shelf. Petroleum exploration leases (Figure 1(A)), the continental shelf break (black line) and Exclusive Economic Zone boundary (grey line) are also shown. Data largely from Buxton *et al.* (2006)

and criteria; IUCN, 2001), while an irreplaceable KBA comprises the location of a significant proportion of the global population of a species (Eken *et al*., 2004; Langhammer *et al*., 2007).

The KBA approach developed from quantitative criteria pioneered by the BirdLife International partnership for the designation of globally significant 'Important Bird Areas'. More than 10 000 Important Bird Areas, the avian subset of KBAs, have been identified in over 170 countries and territories (BirdLife International, 2004a).

KBAs are also widely used for the conservation of plants, mammals, amphibians and other vertebrate taxa. Sites encompassing the only known localities for highly-threatened species — the 'Alliance for Zero Extinction' (AZE) sites (Ricketts *et al*., 2005) — comprise a subset of KBAs. A total of 595 AZE sites based on mammals, birds, amphibians, tortoises, crocodiles, iguanas and conifers, with a median size of 120 km^2, have been identified globally to date (Ricketts *et al*., 2005).

Sites that uniquely contain particular threatened species should be regarded as the most urgent subset of sites required

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within networks of MPAs that follow 'CAR' principles of complementarity, adequacy and representativeness (ANZECC, 1998). For an MPA network to be totally representative and comprehensive it must include all sites with species that are confined to a single site. When single-site species face a high level of threat (i.e. are listed as CR or EN on the Red List), then sites that contain such species clearly represent the most urgent conservation priorities for inclusion within MPA networks. No alternatives in space exist to safeguard these threatened species within a network, and neither can management intervention be postponed for the future as the species may have become extinct by that time.

Other KBA sites need management intervention less urgently, but, if extinction risk is to be minimized, represent higher conservation priorities than sites with widely-distributed non-threatened taxa. A fully comprehensive and representative network of MPAs could nevertheless also include sites in this latter category, particularly in wilderness regions, in order to minimize the slide of species from non-threatened to threatened status.

A further likely benefit of the KBA process additional to its role in safeguarding well-researched species is to protect sites that include critical habitats for poorly-known threatened species. This benefit is outlined using the hypothetical example illustrated in Figure 2, where the two sides of the figure represent changes in a mosaic of different 'ecosystem units' distributed across the seascape from the recent past to the present. The term 'ecosystem unit' is used to refer to a spatial mapping unit that reflects habitats or assemblages, with species composition within each unit most similar to that in units of similar shading.

For the seascape represented by Figure 2, the primary target for safeguarding at present is the hatched ecosystem unit 'A', given that the total area of this ecosystem unit has declined by more than 90% over the period of analysis. If current threats

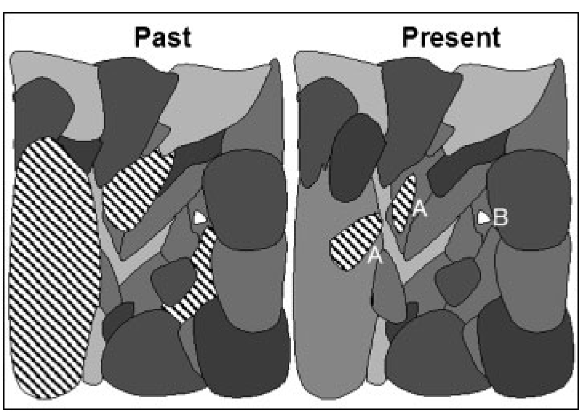

Figure 2. Changes in distribution of ecosystem units from the recent past to present

continue, then this unit could disappear from the seascape, with extinction of associated species. Hatched unit A may comprise, for example, seabed habitat threatened by trawling, shallow rocky reef ecosystem transformed by cascading fishing effects associated with the removal of large predators, upwelling cells affected by climate change, estuaries influenced by pollutants draining from catchments, or seagrass beds affected by eutrophication. While knowledge of the type of threat is necessary when attempting to ameliorate it, this information is not necessary when identifying conservation priority areas using such a mapping approach.

A secondary priority for protection in this example is the small white unit at right centre (marked B), which currently possesses a stable area but is unique within the seascape. Because of its small size and lack of replication, a localized threat acting stochastically, or negative external influences encroaching inside the limited habitat boundary, could cause extinction of species associated with this unit.

In practice, such a direct mapping approach can rarely be used to identify priority ecosystem units for several reasons. First, marine habitats are out of sight, hence mapping over large spatial scales relies on remote sensing, and ecosystem units often have poorly-resolved or fuzzy boundaries (Bruce *et al*., 1997). Second, habitat boundaries can be important biodiversity features in their own right, including frontal areas in offshore habitats (Malakoff, 2004; Campagna *et al*., 2007). Third, detailed 'before' data, such as shown at left in the hypothetical example, are rarely, if ever, available (Dayton *et al*., 1998). Fourth, 'ecosystem units' (or habitats, assemblages, etc.) are not homogeneous entities and are affected by history. The universe of environmental variables operating in the marine environment (e.g. sediment type, seabed structural heterogeneity, bedrock geology, wave exposure, currents, salinity, turbidity, depth, and concentrations of oxygen, nitrates, phosphates, silicates and iron) trend in different but interacting directions, and affect each species differently. Consequently, any habitat map should be regarded as a highly simplified representation, with the distribution of few, if any, species defined by sharp, mapped habitat boundaries (Brooks *et al*., 2004).

Given these limitations, the KBA approach arguably provides the best available mechanism to identify ecosystem units of highest conservation priority. While data relating to species distributions are patchy and also suffer many of the limitations outlined above for habitat data, 'before' information and complete spatial data sets are not necessary for KBA planning.

For the hypothetical example above, populations of species closely associated with hatched ecosystem A in Figure 2, and confined to the region mapped, will have declined greatly in recent years, triggering threatened species status for those species when criteria associated with the IUCN Red List of

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Threatened Species (hereafter Red List) are applied. By plotting the localities where threatened taxa are known to occur, it should be possible to identify the hatched areas because of the presence of one or more species with rapidly declining populations. Thus, the location of threatened species can provide a surrogate for threatened habitats, which in turn may provide a surrogate for other (unknown) threatened species. Moreover, by assessing the known localities for species with highly restricted ranges, it should also be possible to locate sites analogous to the small white ecosystem unit B.

A prediction of the hypothesis that remnant areas of relatively undisturbed habitat provide a refuge for multiple species with declining populations is that sedentary threatened marine species are not randomly distributed across the seascape, but will tend to co-exist in a restricted number of sites. This prediction is supported by the limited data available. A map of all localities where threatened endemic Tasmanian fish and sea stars have been recorded in the past 30 years, for example, indicates that threatened marine species are not randomly distributed (Figure 3). Large areas of coast lack threatened taxa, including the northern half of the island, whereas over 80% of known localities with threatened species are positioned within a 100 km span off the south-eastern coast. Three species occur in very close proximity at the encircled location.

MARINE KEY BIODIVERSITY AREAS

Criteria used to identify key biodiversity areas

KBAs are designated for species that regularly occur at a site and that will benefit from conservation and management actions undertaken at the site (Eken *et al*., 2004). Sites may be included in a KBA network where the species' occurrence is seasonal (e.g. for breeding or feeding) or episodic; however,

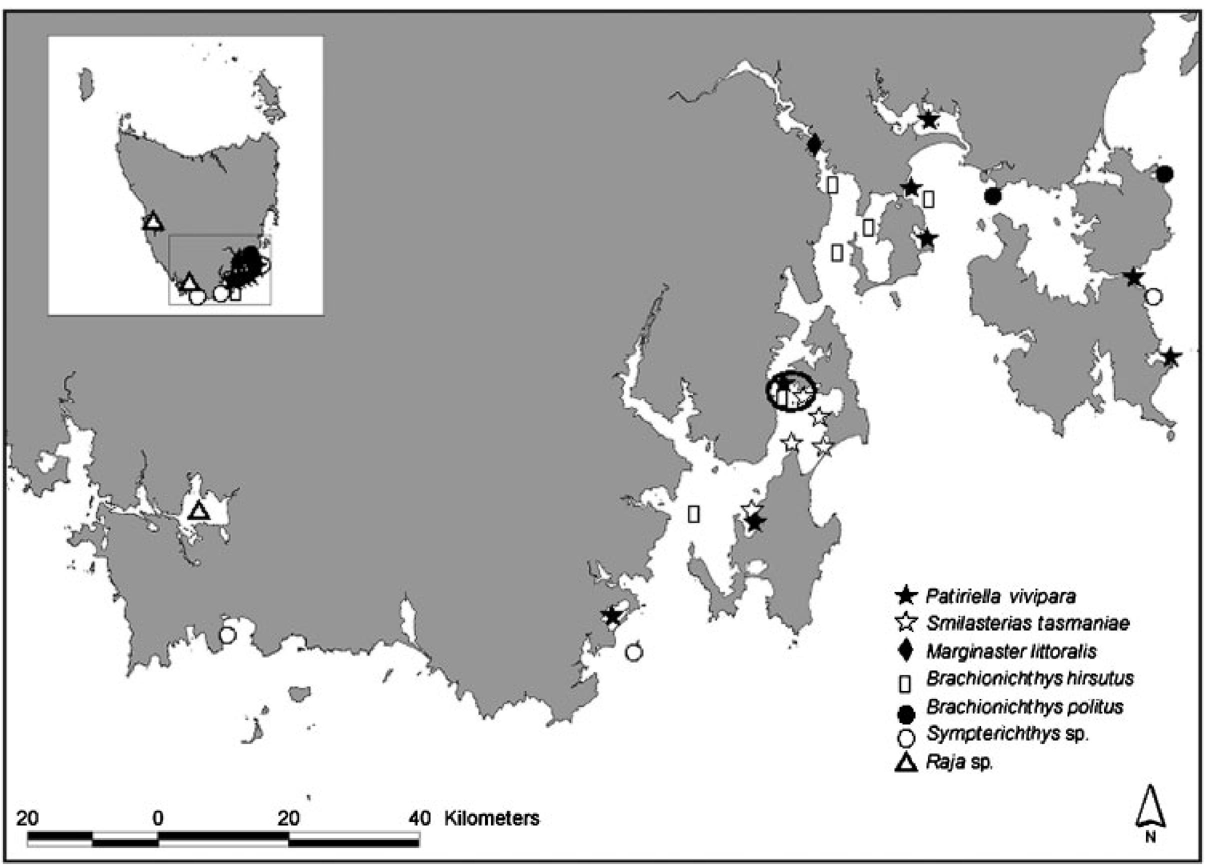

Figure 3. Localities at which endemic Tasmanian marine fish and invertebrate species listed as threatened under the Australian Conservation and Biodiversity Protection Act and Tasmanian Threatened Species Protection Act have been recorded during the past 30 years

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instances of vagrancy or marginal occurrence and historical records are excluded.

KBA criteria currently applied in terrestrial situations need some modification to work most effectively at marine sites because of differences between terrestrial and marine realms (Steele, 1985; Carr *et al.*, 2003), including greater connectivity and faster turnover rates of marine systems, and the three-dimensional nature of marine habitats. In addition, comparatively few data are available on the distribution of, and threats to, marine taxa.

As an initial step to address the perceived need for modified marine criteria, a 'Marine KBA Development Workshop' was held in Washington DC on 1–3 August 2005, involving authors of this paper and ~30 others closely involved in marine conservation planning and science. The goal of the workshop was to examine criteria used to identify terrestrial KBAs to determine whether they need adaptation for marine situations; Table 1 lists the marine KBA criteria that were consensually agreed by workshop participants. They are based on KBA criteria applied to date, mainly in terrestrial environments, with slight adaptation to facilitate marine application. While some initial testing has been undertaken in the Galapagos Marine Reserve (Edgar *et al.*, 2008), criteria and thresholds included in Table 1 are provisional, and are proposed within the evolving process for establishing agreed-upon thresholds for KBA criteria (Eken *et al.*, 2004). They need field-testing to ensure that the number of KBAs identified in practice is appropriate and reasonable, and that species for which site-scale conservation is important are not overlooked.

Vulnerability criterion associated with globally threatened species

The IUCN Red List provides an appropriate quantitative standard for measuring extinction risk among species (IUCN, 2001). The Red List recognizes three groups of threatened species with decreasing levels of vulnerability: Critically Endangered (CR), Endangered (EN) and Vulnerable (VU) species. Other categories are applied for species regarded as Extinct (EX), Extinct in the Wild (EW), Near Threatened (NT), Least Concern (LC) and Data Deficient (DD). Threatened species are categorized on the basis of standardized thresholds related to population size, population trends, distributional range, and persistence of threats. For example, a species with a population that has declined by >80% over the past 10 years in the face of persisting threats is categorized as CR.

One major shortcoming of the current Red List is that few marine species have been assessed, and these taxa are heavily biased towards large, charismatic, wide-ranging vertebrates (Rodrigues *et al.*, 2006). Only 19 benthic marine invertebrates (twelve molluscs, four crustaceans, one polychaete, two cnidarians) and one marine plant have been assessed as threatened and entered on the 2006 IUCN Red List, surely an insignificant proportion of the number that is actually threatened (Millar, 2003; Edgar *et al.*, 2005).

Some threatened marine fish, invertebrates and seaweeds are, however, recognized at national and state levels. Any species endemic to an assessment region that is evaluated using the required Red List process and meets threatened species criteria should be included in the application of the vulnerability criterion for KBAs. Within Australia, for example, the NSW Fisheries Management Act 1994 lists six marine fishes and two algal species as Vulnerable, Endangered or Extinct (http://www.fisheries.nsw.gov.au/threatened_species/threatened_species), while threatened species schedules of the Tasmanian *Threatened Species Protection Act* include four marine fishes, three seastars, one seaweed and one gastropod (http://www.dpiw.tas.gov.au/inter.nsf/WebPages/SJON-58K8WK?open). Nevertheless, such listings are incomplete even in comparatively well-studied regions such as Australia, and they are absent for most nations and states.

Table 1. Criteria and thresholds provisionally considered appropriate for the identification of marine KBAs

Criterion	Sub-criteria	Provisional thresholds for triggering KBA status
Vulnerability		Regular presence of a single individual for Critically
Regular occurrence of a globally threatened species (according to the IUCN Red List) at the site		Endangered (CR) and Endangered (EN) species; regular presence of 30 individuals or 10 pairs for Vulnerable species (VU)
Irreplaceability	(a) Restricted-range species	Species with a global range less than 100 000 km^2; 5% of global population at site
Site holds X% of a species′ global population at any stage of the species′ lifecycle	(b) Species with large but clumped distributions	5% of global population at site
	(c) Globally significant congregations	1% of global population seasonally present at site
	(d) Globally significant source populations	Site is responsible for maintaining 1% of global population

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The development of effective KBA networks clearly requires threat assessment of sedentary species in addition to the charismatic taxa already included on the IUCN Red List (Edgar *et al.*, in review). Species currently recognized as threatened may prove a somewhat unusual subset of all species because of the inclusion of many wide-ranging species such as whales and tuna that require broad-scale (national and global) action to address threats such as longlining, as well as management of sites where animals aggregate to feed, mate or spawn.

The low number of marine species with global threat assessments to date should be overcome through the acceleration of the IUCN Global Marine Species Assessment (GMSA; http://www.sci.odu.edu/gmsa/), which aims to systematically assess the Red List threat status of all species within major marine taxonomic groups such as sharks and rays, reef fish, corals, kelps and seagrasses. Both the GMSA and the Census of Marine Life (O'Dor, 2004) will also play an urgently needed role in the coordination, capture, and management of existing marine biodiversity data, further assisting the Red Listing process. Centralization of new and existing species data will better facilitate the assessment of a species' threatened status using standard criteria associated with extinction risk. To ensure this process remains rigorous and comprehensive, close collaboration is needed between taxonomic experts, regional data providers and assessors.

Irreplaceability criteria associated with species concentrations in space or time

KBAs can be identified using irreplaceability criteria for: (a) species with highly restricted global ranges ('range-restricted' or 'endemic' species); (b) species with highly clustered distributions ('clumped' species); (c) species that temporarily aggregate in particular sites ('congregatory' species); or (d) species with small sub-populations that are responsible for generating a significant proportion of recruitment ('source' species). Following conventions recognized for terrestrial KBA identification, provisional percentage thresholds for these categories are proposed (Table 1).

Until more detailed global analyses of species ranges are completed, the initial set of species with sites to be considered under the 'restricted-range' criterion are those with mapped extent of occurrence or EOO (*sensu* IUCN, 2001) of less than 100 000 km^2. A greater distributional area is applied here compared to the 50 000 km^2 used to define restricted-range species in terrestrial situations (Langhammer *et al.*, 2007) because of the greater mean range size for marine species. Approximately 3% and 4% of Indo-Pacific reef coral and reef fish species, respectively, are defined as range-restricted using the 50 000 km^2 EOO threshold, and 3% and 9%, using the 100 000 km^2 threshold (G. Allen, unpublished data; Hughes *et al.*, 2002; Allen, 2007). This compares with approximately 25% of all bird and mammal species, and 60% of amphibian species, that fall within the 50 000 km^2 EOO used for terrestrial taxa (Eken *et al.*, 2004). The occurrence in a site of, provisionally, 5% of the population (or range) of a 'restricted-range' species would be required to trigger the identification of a KBA under this criterion (Table 1).

Participants at the marine KBA workshop suggested that the area of continental shelf be used in calculations of EOO for species with mapped distributions that appear as long coastal strings. This technicality was considered necessary for coastal species with linear distributions that are difficult to quantify realistically in terms of area, most notably intertidal species distributed along continental margins. Larval dispersal of most coastal marine species extends seawards, with the shelf break used here to define the offshore distributional boundary for such species.

A second class of species that may trigger the irreplaceability criterion comprises those species that are widely distributed but have clumped distributions in parts of their range. In other words, large numbers of individuals may be concentrated in a single or few sites while the rest of the species is widely dispersed. Species with large extent of occurrence but small area of occupancy can trigger this criterion. A provisional threshold of 5% of the global population should trigger a KBA for such species (Table 1), paralleling the threshold for restricted-range species. An example is Guerney's sea pen *Ptilosarcus gurneyi*, which is distributed along the US west coast from the Gulf of Alaska to southern California, but with very high concentrations in Puget Sound (Birkeland, 1974). Species with such wide distributions should only be considered after other KBA criteria have been evaluated, given that most species with clumped distributions are concentrated in an area for only part of the year and should therefore also trigger the congregatory criterion.

KBAs for congregatory species include: (a) assembly sites where large numbers of individuals gather at the same time (e.g. feeding, breeding and spawning sites); and (b) bottleneck sites traversed by many individuals in the same season (e.g. migratory sites). To meet the KBA sub-criterion for congregations, a site must hold a significant proportion of the global population of a congregatory species on a regular basis. Based on the 1% thresholds in wide use under the Ramsar Convention (BirdLife International, 2002) and regional flyway initiatives (Asia-Pacific Migratory Waterbird Conservation Committee, 2001) a provisional threshold of 1% of the global population of a species is proposed. This threshold requires further testing, especially in comparison with a 5% threshold.

In conformity with Red List criteria, calculations of population size are based on the number of mature individuals, excluding individuals that will never produce

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new recruits to the global population (e.g. dioecious individuals reproductively isolated from other individuals, or individuals that produce larvae that drift offshore and are all lost). In contrast to sites with plants and animals that generate no new recruits, and hence are not considered as KBAs, some marine sites make a disproportionately high contribution to recruitment elsewhere. Such source sites should be designated as KBAs when they contribute $>1\%$ of recruits to the global population of the species, regardless of whether the total adult population is clumped or not. Recognition of source sites is necessary to safeguard sites such as the waters around particular Caribbean islands that generate the majority of juvenile spiny lobster recruitment to islands across the wider region (Stockhausen *et al.*, 2000).

An anomaly within the methodology developed for terrestrial KBA identification, and inherited by the provisional thresholds outlined here, is that a much smaller proportion of the total population is sufficient to trigger a KBA for species that aggregate seasonally (1%) compared to spatially (5%). This can be supported on the grounds that it has worked well so far in the terrestrial context. However, it could also be argued that these thresholds should be reversed in marine systems because restricted-range species face greater extinction risk from localized stochastic threats. Threats that are distributed at scales $<100\,000\,km^2$ may fully overlap the distribution of a restricted-range species and therefore threaten the total population with extinction if precautionary management measures are not enacted. By contrast, congregatory species with populations that are widely distributed but not currently threatened (i.e. potentially able to trigger KBA irreplaceability but not vulnerability criteria) are less likely to become extinct as a consequence of threats that are localized in time and space. The difference between thresholds for congregatory and restricted-range taxa clearly needs to be assessed in practice as a matter of urgency, then standardized if appropriate within ongoing processes to refine standard KBA methodology.

An additional irreplaceability criterion relating to 'Bioregionally-restricted assemblages' has been applied to terrestrial sites that hold 'a significant proportion of the group of species whose distributions are restricted to a biome or to a subdivision of it' (Eken *et al.*, 2004). This criterion has not been used as widely in identifying KBAs as the two criteria described above. Its usage evolved from Important Bird Area (IBA) and Important Plant Area (IPA) criteria (Eken *et al.*, 2004), although these in turn differ in some respects. For IBAs, this criterion has been defined as: 'a significant component of the group of species whose distributions are largely or wholly confined to one biome' (Fishpool and Evans, 2001). For IPAs, this criterion covers two situations, either: 'an exceptionally rich flora in a regional context in relation to its biogeographic zone', or 'an outstanding example of a habitat or vegetation type of global or regional plant conservation and botanical importance' (Plant Diversity Challenge, 2004).

Given that the bioregionally-restricted criterion has not been widely applied for a range of animal taxa in the terrestrial context and is still regarded as under development, we suggest that it be postponed from application to marine sites for the present. An informed decision on the application of this criterion and appropriate thresholds requires: (i) the exploration of methodologies that have been used to classify species assemblages and define biomes; (ii) analysis of different bioregional classifications applied in the marine context; and, importantly, (iii) the identification of important aspects of marine biodiversity that are not captured through application of the other criteria.

Regional testing of KBA criteria

To date, three main tests of KBA criteria in the marine environment have been conducted. First, application of KBA criteria described in this paper has been trialled within the Galapagos Marine Reserve, as described in the associated paper (Edgar *et al.*, 2008). A total of 38 KBAs were identified along the Galapagos coastline using the vulnerability criterion, comprising *ca* 10% of the total inshore area.

Second, although marine algae in the UK have received very little direct conservation attention, three IPA criteria have recently been applied to marine algae (Brodie *et al.*, in press): (A) significant populations of one or more species that are of global or European conservation concern; (B) an exceptionally rich flora in a European context in relation to its biogeographical zone; (C) an outstanding example of a habitat type of global or European plant conservation and botanical importance. Criterion A falls within KBA guidelines outlined here for species endemic to the Europe region, while Criteria B and C largely relate to the bioregionally-restricted assemblage criteria as applied to terrestrial KBAs.

Over 83 UK sites were suggested by members of the British Phycological Society as possible candidate IPAs, nine of which have been considered for possible European IPA designation. While the application of IPA criterion C of outstanding examples of habitat types was relatively uncomplicated, given that most relevant habitats, including maerl beds, chalk and eel grass beds at the UK level and reefs at the European level already had conservation legislation, the application of criteria A and B was less straightforward. This largely reflected the lack of verifiable data, hence a pragmatic approach was needed to apply criteria that were developed primarily with terrestrial organisms in mind. A novel approach was devised utilizing specimens from the algal herbarium and distribution maps. For criterion A, a list of species with limited distribution was made and then refined by specialists. This rare species list

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provided the basis for assessing species as potential Red List candidates.

Until recently, the inclusion of seabirds in BirdLife International's IBA programme has been largely confined to the identification and protection of terrestrial sites where more than threshold numbers are present on a regular basis, such as at nesting colonies, foraging grounds, and roosting locations. The programme is, however, undergoing extension into the marine realm, where four different 'types' of marine IBA are being explored. The initial focus is on delimiting seaward extensions to boundaries of existing IBAs designated for seabird breeding colonies, to include the colonies' adjacent foraging areas. In addition, it is likely that IBAs will be readily and increasingly identified for inshore concentrations of non-breeding seabirds, such as seaduck, whose distributions are largely circumscribed by water depth and access to sedentary benthic food resources. Straits, headlands and other places that act as migratory bottlenecks, through and around which large numbers of seabirds funnel seasonally, will also be identified. Finally, and most challenging of all, work is underway to examine ways of identifying and delineating IBAs for the key foraging areas of pelagic species. These studies utilize information provided by satellite tracking (BirdLife International, 2004b), combined with at-sea observations and other data sources, in order to develop and test criteria by which offshore IBAs may identified.

The process of identification of marine KBAs for a full range of taxa has recently commenced through projects undertaken by participants of the marine KBA workshop, and others, in the Eastern Tropical Pacific, Philippines, Indonesia, Madagascar, Australia, Brazil, Melanesia and Polynesia, thereby providing a vital opportunity for adaptively testing thresholds and criteria.

Outstanding issues associated with key biodiversity areas

Development of an optimal KBA methodology represents an ongoing challenge, both in order to reduce subjectivity associated with KBA identification, delineation and prioritization, and also to facilitate implementation of KBA and MPA networks with relevant stakeholders (Knight *et al*., 2007). While subjectivity is less than with methodologies based on the use of abstract habitat types as proxies, the KBA process involves several challenges, most significantly because:

(i) species distributional datasets are inevitably imperfect, hence KBA networks will also be imperfect, with a bias towards well-studied sites;
(ii) irreplaceability thresholds are often difficult to apply in practice because of the scarcity of good population data at the global and site levels, making it near impossible to accurately estimate the percentage of the global population present at many sites;
(iii) some threatened species, such as the napoleon wrasse *Cheilinus undulatus* (Donaldson and Sadovy, 2001) and green turtle *Chelonia mydas* (Edgar *et al*., 2008), occur widely, potentially creating a situation whereby the majority of the global coastline is designated within KBAs; and
(iv) boundaries of KBAs can potentially be manipulated by sectoral interests to achieve particular aims, with, in the extreme situation, the existence of some KBAs dependent on whether boundaries are drawn to encompass a sufficient target population to trigger thresholds.

With respect to the first of these challenges, KBA networks should be regarded as adaptive systems that will improve through time as new data become available. Protection of a site where a threatened species is known to occur, or which possesses a high concentration of the known global population, should be undertaken without delay regardless of the possible but uncertain existence of individuals in under-studied areas. If research later indicates that a species is more widely distributed than initially believed, then existing and potential KBAs involving that species should be re-evaluated, and conservation resources directed to KBAs reallocated if appropriate. An alternate strategy involving the protection of sites where the existence of threatened species is predicted but uncertain will, in many cases, result in overconfidence that species are adequately safeguarded.

In cases where population data are poor and the second challenge applies, proxies such as global percentage of suitable habitat or range polygons for all sites at which the species is known to occur can be used to generate proportionate population estimates. These estimates should subsequently be refined as better data become available.

The third challenge, that the KBA process will be debased if all localities with confirmed records of widely-distributed threatened species are recognized as KBAs, is partly alleviated through the exclusion of sites with vagrant occurrence of individuals, or less than 30 individuals of VU species. Nevertheless, if application of thresholds outlined in Table 1 in multiple regions is found to trigger an excessively high number of KBAs, then thresholds will need to be adjusted upwards to trigger fewer KBAs. Such a modification was suggested by Edgar *et al*. (2008), who recommended that marine KBAs not be recognised for wide-ranging EN and VU species that are well represented in existing KBAs, unless at least 1% of the global population is present at a site.

The fourth challenge relates to the variable potential size of KBAs, with dimensions depending on the scale of local management units and the extent of habitat considered necessary to safeguard populations of species that trigger the KBA. Boundaries of marine KBAs may follow habitat edges, depth contours, existing or potential MPA boundaries, exclusive economic zone or seabed-tenure borders, or other

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features, depending on the needs of the species that trigger the KBA. Although methods for working through various contrasting KBA delineation scenarios are outlined in Langhammer *et al.* (2007), the development of standardized globally-consistent methods for delineating KBAs would greatly reduce subjectivity in boundary delineation.

In many cases, appropriate KBA boundaries are self-evident, such as for species endemic to small islands, existing MPAs, or individual estuaries. In other cases, decisions may involve consideration of appropriate habitat or management unit boundaries, and whether to aggregate multiple localities with known species occurrence into a single KBA or to consider different localities as separate KBAs.

A long-term strategy for standardizing decision rules requires, first, identification of appropriate boundaries for numerous KBAs on a case-by-case basis. Information on the processes most often used to define individual KBA boundaries should then be integrated into generalized decision rules. This delineation process would be greatly assisted if facilitated by an international agency or a consortium with a mandate to maintain global KBA standards and to recommend useful changes to criteria and thresholds, in the same way that the Red List process is facilitated by the IUCN.

Standardized methods also need to be developed for ranking KBAs with respect to their priority for conservation intervention and investment. A non-prioritized network of KBAs has value in terms of providing a focal set of sites for conservation action, local ownership, and building institutional capacity; however, managers and conservation financiers additionally require information on such factors as cost of protection, level of local and government support, vulnerability of site to external threats, ecosystem services presently and potentially generated, and conservation value of each KBA. Decision rules to prioritize the need for conservation action among KBAs will greatly assist this process.

Alliance for Zero Extinction sites, which represent known places where extinctions are imminent unless immediate conservation action is taken (Ricketts *et al.*, 2005), comprise the highest priority set of KBAs with respect to conservation intervention. Other likely prioritization rules, in no set order and all else being equal, are: (i) KBAs with the greatest proportion of the global population of a threatened or aggregating species have highest priority, (ii) KBA sites identified for Critically-Endangered (CR) species have higher priority than sites identified for Endangered (EN) species, which have higher priority than sites for Vulnerable (VU) species; (iii) KBAs identified to safeguard a species not protected elsewhere have highest priority; (iv) KBAs safeguarding more than one species have higher priority than sites identified for a single species; (v) KBA sites that are highly vulnerable to known threats have higher priority than sites lacking apparent threats if a large percentage of the global population of threatened or aggregating species utilize that site (i.e. it is highly irreplaceable; Langhammer *et al.*, 2007); and (vi) KBAs identified to protect species with significant ecological roles, such as keystone or habitat-forming species, have high priority.

Algorithms such as MARXAN (Ball and Possingham, 2000) are available to maximize site complementarity (category iii above) and site representation (category iv) in MPA networks; however, additional decision rules are needed to incorporate species vulnerability (categories (i) and (ii)), site vulnerability (category (v)) and ecological interaction strength (category (vi)) into the prioritization framework. Decision rules based on interactions between the six categories above are particularly needed, such as: 'Is an MPA with 90% of the population of two EN species a higher conservation priority than an MPA with 50% of the population of a CR species?' Such questions are best answered by quantifying relationships between threat and extinction risk.

An issue related to the prioritization of KBAs is the identification of representation targets within MPA networks using reserve selection algorithms. KBAs represent an essential component of any representative MPA network because of their irreplaceability, hence should be identified as an initial step if networks are to be complementary, adequate and representative. Clearly, if a KBA is designated for a species not found elsewhere, then that site is a necessary component within any fully representative network.

Although a number of outstanding issues associated with development of KBA networks remain, as discussed above, the variety of benefits provided by KBAs should not be underestimated. The following strengths indicate that the role of KBAs in systematic MPA planning should be overwhelmingly positive:

(i) KBAs are founded on previous initiatives (e.g. Important Bird Areas, Alliance for Zero Extinction), hence existing *de facto* KBAs have already been identified in many countries.
(ii) KBAs consider all taxonomic groups for which data exist.
(iii) KBAs target all known biodiversity that would benefit from conservation activities undertaken at the scale of individual sites.
(iv) KBAs can be based on any species-level data, allowing the KBA process to begin immediately with iterative updating as more data become available.
(v) KBA identification relies on inexpensive and straightforward procedures that can typically be completed within a short timeframe.
(vi) Leadership and ownership of the KBA process occurs at local (or sometimes national or regional) levels, but

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follows global standards and criteria that allow comparability and consistency.

Local leadership is important because allocation of resources to protect KBAs will inevitably involve subjectivity in terms of tradeoffs between perceived needs, threats, benefits and costs, regardless of the availability of sophisticated analytical tools. Because these tradeoffs are best understood locally, and stakeholder ownership is critical to the success of protected areas, identification and implementation of the KBA process is best achieved through activities undertaken at local and national levels. Local activities should include monitoring, which is integral to any implementation strategy given that improved boundary delineation and more effective species protection depend heavily on an understanding of population trends for those species whose presence triggers a KBA.

Development of KBA networks involves feedback at many levels. On the one hand, KBA criteria will need to be modified if found deficient in the protection of threatened or aggregating species, while remaining simple and globally consistent. On the other, individual KBAs and KBA networks must adapt to changing environmental conditions and data availability. This is particularly important in the current era of global environmental change, when a major challenge is to foresee fragmentation of biota at existing locations through extirpation, emigration and immigration. Thus, marine KBA networks will inevitably evolve through time, as is also the case with terrestrial KBAs. Sites will be added to the network as new data on threats and species distribution become available, and, in cases of the delisting of threatened species, or local extirpation or populations decline below trigger values, sites will also occasionally lose KBA status.

CONCLUSIONS

The data-driven and species-based KBA concept allows systematic identification and conservation of sites of global biodiversity significance through application of simple criteria. This concept fills a critical need to incorporate species vulnerability into MPA planning, and should prove useful for overcoming deficiencies in current planning strategies. KBA criteria have been largely applied to date in terrestrial situations, and require some further modification to work effectively in marine situations because of the generally larger distributional ranges of marine taxa, the linear distributions of many coastal species when mapped at regional scales, and a paucity of observational data for marine plants and animals. Criteria and thresholds to identify marine KBAs provisionally outlined in this paper require testing as a matter of urgency, as do decision rules for delineating KBA boundaries, and for prioritizing conservation action amongst sites. Nevertheless, although the thresholds proposed here are provisional, marine sites have already been identified as KBAs that are globally significant and that represent clear targets for conservation action. Once an initial set of KBAs have been identified in a region, conservation activities should begin as soon as possible, rather than waiting for criteria to be finalized or a full network of sites identified.

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