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REVIEW

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Synergies between the key biodiversity area and systematic conservation planning approaches

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Abstract

Systematic conservation planning and Key Biodiversity Areas (KBAs) are the two most widely used approaches for identifying important sites for biodiversity. However, there is limited advice for conservation policy makers and practitioners on when and how they should be combined. Here we provide such guidance, using insights from the recently developed Global Standard for the Identification of KBAs and the language of decision science to review and clarify their similarities and differences. We argue the two approaches are broadly similar, with both setting transparent environmental objectives and specifying actions. There is however greater contrast in the data used and actions involved, as the KBA approach uses biodiversity data alone and identifies sites for monitoring and vigilance actions at a minimum, whereas systematic conservation planning combines biodiversity and implementation-relevant data to guide management actions. This difference means there is much scope for combining approaches, so conservation planners should use KBA data in their analyses, setting context-specific targets for each KBA type, and planners and donors should use systematic conservation planning techniques when prioritizing between KBAs for management action. In doing so, they will benefit conservation policy, practice and research by building on the collaborations formed through the KBA Standard's development.

KEYWORDS

decision science, irreplaceability, Key Biodiversity Areas, spatial prioritization, systematic conservation planning, targets

1 | INTRODUCTION

Site-based conservation is vital for stemming biodiversity loss, but many important species, ecosystems and ecological processes are missing from current conservation area systems (Butchart et al., 2015; Klein et al., 2015). In response, signatories to the Convention on Biological Diversity have committed through Aichi Target 11 of the Strategic Plan for Biodiversity to better conserve "areas of particular importance for biodiversity" (CBD, 2010). Two approaches widely used to inform the implementation of Aichi Target 11 are based on the systematic conservation planning and Key Biodiversity Area (KBA) approaches (IUCN, 2016; Margules & Pressey, 2000). Systematic conservation planning is an operational model for identifying and implementing the conservation of priority areas (Knight, Cowling, & Campbell, 2006; Pressey & Bottrill, 2009). It has been used in dozens of countries to help prioritize the expansion or management of protected area networks and design conservation landscapes and marine spatial plans (Álvarez-Romero et al., 2018; Sinclair et al., 2018). The KBA approach identifies sites that "contribute significantly to the global persistence of biodiversity" based on a set of globally standardized criteria and quantitative thresholds (IUCN, 2016). It builds on a methodology originally developed for identifying Important Bird and Biodiversity Areas (IBAs) (BirdLife International, 2014; Donald et al., in press) that was then adapted to identify sites of importance for a range of different taxa (Edgar et al., 2008; Eken et al., 2004; Holland, Darwall, & Smith, 2012; Langhammer et al., 2007), with over 15,000 sites identified to date (BirdLife International, 2017).

Traditionally, the two approaches were developed and applied by different communities, but the two groups began working closely together in 2012 as part of the process to develop a new global KBA Standard (IUCN, 2016). This collaboration involved comparing the systematic conservation planning and KBA approaches, addressing previous critiques of the KBA methodology (Knight et al., 2007) and investigating the scope for unifying the theory and practice that underpins them. Initial appearances suggest that options are limited, as the two approaches have seemingly different aims and methodologies. For example, systematic conservation planning uses complementarity-based algorithms to identify sets of priority conservation areas (Moilanen, Wilson, & Possingham, 2009), whereas the KBA approach is applied on a case-by-case basis, identifying sites that meet the KBA criteria at threshold levels for one or more biodiversity elements (Text Box). In addition, although systematic conservation planning ideally includes implementation-relevant data such as threats and costs (Moilanen et al., 2009), KBAs are identified using biodiversity data alone and so are not necessarily priorities for formal protection or any other particular form of conservation management (Maxwell et al., 2018).

Despite these differences, recent research from a global study using bird distribution data found that many terrestrial IBAs were also identified as important by the complementarity-based algorithms used in systematic conservation planning (Di Marco et al., 2016). This finding suggests the two approaches could be used together but there is limited guidance for conservation policy makers and practitioners on how this is best achieved. Here we provide such guidance, for the first time using the language of decision science to review and clarify the similarities and differences. We show that both approaches fit well within the decision science paradigm of specifying conservation features and setting environmental objectives (Pressey & Bottrill, 2009), and discuss why the value system underpinning KBAs justifies them being identified using biodiversity data alone to guide monitoring and vigilance actions. We then describe how the two approaches can complement and strengthen each other, and argue they should be used in tandem in future to identify, prioritize, and delineate new areas for conservation actions.

2 | USING DECISION SCIENCE TO FRAME THE TWO APPROACHES

Decision science "aims to help people make the best decision in pursuit of a stated objective, particularly in situations that are highly complex or uncertain" (Game, Kareiva, & Possingham, 2013). It is an increasingly important component of conservation science (Kareiva & Marvier, 2012) that represents an extensive body of theory, methods and tools. A fundamental part of decision science is framing the problem, which involves: (a) defining the context; (b) identifying the set of potential actions for implementation (Adams et al., in press); and (c) specifying the environmental objectives to be achieved through these actions (Groves & Game, 2015). In this section we frame systematic conservation planning and the KBA approach in terms of these three aspects and then use this to clarify their similarities and differences.

2.1 | Systematic conservation planning

Systematic conservation planning has been used around the world to identify priority conservation areas in a range of different contexts (Álvarez-Romero et al., 2018; Sinclair et al., 2018). This means the process of defining the context varies case-by-case and is the first step in any prioritization process (Knight et al., 2006; Margules & Pressey, 2000). Identifying the set of potential actions for implementation is also context-dependent, although the importance of explicitly defining these actions has only been recognized relatively recently (Game et al., 2013). Both these problem-framing steps involve a series of value judgments but so too does the third step of developing the environmental objectives. For example, planners must choose which conservation features should be used to represent biodiversity. These are generally species and ecosystem types but can also include populations, biological and ecological processes, and ecosystem services (Beger et al., 2010; Margules & Pressey, 2000). Just as importantly, they must decide on how much of these conservation features should be conserved. One common approach requires planners to specify how much of each conservation feature should be included in the priority area networks by setting quantitative targets. These quantitative targets should be set high enough to ensure the long-term persistence of each feature, based on the best available knowledge (Carwardine, Klein, Wilson, Pressey, & Possingham, 2009), but also reflect the broader context and value systems that underpin them. Thus, for example, targets for a rewilding project will be different from targets for establishing a network of intensively managed protected areas.

This is followed by undertaking a spatial conservation prioritization, and is based on dividing the planning region into a number of planning units or sites, calculating the amount of each conservation feature in each site, and identifying sets

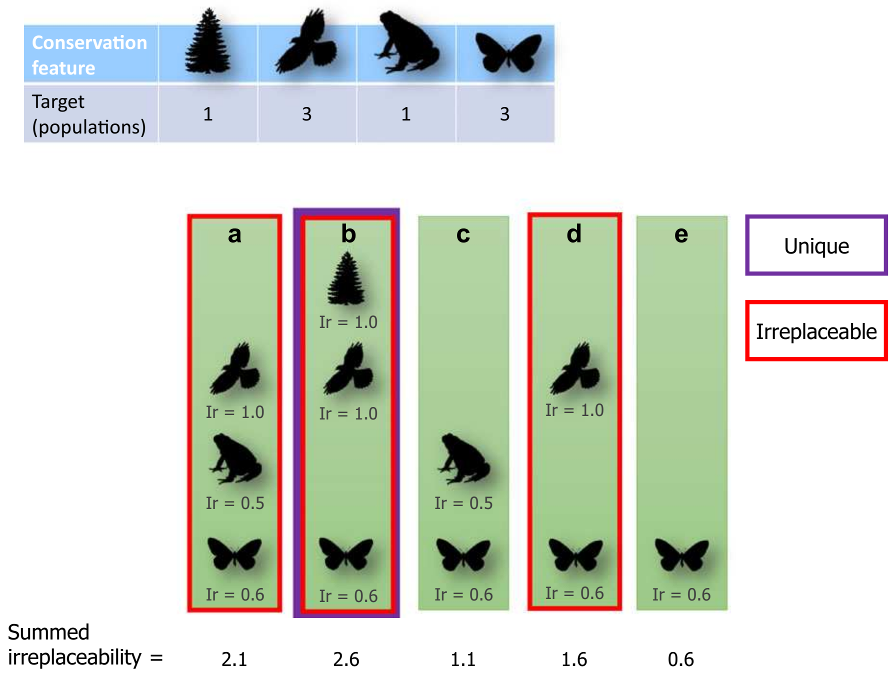

FIGURE 1 Schematic based on five planning units (a–e) to illustrate the concepts of irreplaceability, irreplaceability scores (Ir), summed irreplaceability and uniqueness, and their relationship to conservation targets. Planning units A, B and D are all irreplaceable because they have to be selected to meet the targets for the tree and bird species. Only planning unit B is unique because it contains the only tree population. The irreplaceability score for a specific conservation feature in a specific planning unit is calculated as: the number of planning unit combinations that meet the feature target containing the planning unit, divided by the total number of planning unit combinations that meet the feature target (excluding combinations containing superfluous planning units). The summed irreplaceability score for the planning unit is the sum of the irreplaceability scores for each of the conservation features found within it

of sites that meet all the targets (Margules & Pressey, 2000). This typically involves using prioritization software, but their complementarity-based algorithms often identify different sets of sites that meet the same targets with similar efficiency (Kukkala & Moilanen, 2013), although some sites appear in the alternative solutions more than others. The concept of irreplaceability was developed to distinguish between these sites (Pressey, Johnson, & Wilson, 1994), so that irreplaceability scores can be calculated for each conservation feature in each planning unit and range between 0 and 1 (Ferrier, Pressey, & Barrett, 2000). If a planning unit has an irreplaceability score of 1 for one or more conservation features then it is "irreplaceable," as it is always needed to meet the targets (Figure 1). However, it should be noted that irreplaceable sites are not equally important, as their irreplaceability can derive from meeting targets for one, several or many conservation features (Figure 1). Conversely, for sites with irreplaceability scores closer to zero there is flexibility because targets can be achieved by swapping them with many other similar sites. Either way, it is important to recognize that irreplaceability values are emergent properties and depend on the targets set by the planners overseeing the process.

As well as determining the irreplaceability of different sites, setting targets lets planners incorporate implementation-relevant considerations into the spatial prioritization, so that they influence the spatial location of the priority areas without undermining the conservation goals. Including these additional factors has no influence on the location of irreplaceable sites, as these have to be selected to meet targets. But when choosing between similar sites with lower irreplaceability scores there is flexibility, so the prioritization software selects sites that minimize costs and threats whenever possible (Ball, Possingham, & Watts, 2009). Planners should choose cost metrics that best reflects the conservation context, for example, land acquisition and management cost when buying land to create new protected areas, opportunity costs when seeking to minimize impacts on other sectors like farming or fishing, or stakeholder support metrics when seeking to avoid sites where support for conservation is lowest (Knight et al., 2011; Naidoo et al., 2006; Venter et al., 2014).

2.2 | The KBA approach

Framing the KBA approach as a decision science problem involves going through the same three steps of defining the context, identifying the set of potential actions for implementation, and specifying transparent environmental objectives (Groves & Game, 2015). The first step is straightforward because the KBA Standard (2016) describes the conservation context, stating the approach aims to inform multiple issues, such as: the expansion of protected area networks; private sector safeguard policies, environmental standards and certification schemes; supporting conservation planning; and providing local communities with opportunities for employment, recognition, economic investment, societal mobilisation and civic pride (Dudley et al., 2014). Because of this context, the KBA approach is designed for consistent application by different people and institutions in different places using the locally available data needed to apply the criteria and thresholds. This means the KBA methodology needs to be sophisticated enough to identify sites of true global significance but also reasonably quick and practical to implement, given the rapid rate of global biodiversity loss. It also accounts for the fact that KBAs are identified in a site-by-site manner and that data availability and relevant capacity varies widely between regions and taxonomic groups (Brooks et al., 2015; Meyer et al., 2015).

The second step is specifying the actions that should take place as a result of KBA identification. Given the situation described above it is expected that a corresponding range of context-specific activities will occur in different KBAs once they have been identified, but the common thread is based on monitoring and vigilance. This is because the status of a KBA depends on checking for the continued presence of the biodiversity elements that triggered its recognition, with formal reassessment at least once every 8–12 years (IUCN, 2016). Such monitoring and vigilance actions should therefore involve: identifying and monitoring threats, including those that originate outside the site; monitoring the biodiversity elements for which the site has been identified as important; and ensuring that relevant constituencies are informed when any of these elements are threatened.

The third framing step is to define the environmental objectives. These are to identify sites that "contribute significantly to the global persistence of biodiversity" (IUCN, 2016). Given this, it has to be possible for KBAs to be identified based on estimates of the global distribution of a biodiversity element and the proportion of that element at the site, without requiring data on the biodiversity values of other sites. This is why the revised KBA Standard and its predecessors primarily use a threshold-based system (Langhammer et al., 2007), such that any site meeting at least one threshold qualifies as a KBA. The new KBA methodology (IUCN, 2016) uses 10 threshold-based criteria grouped within 4 higher-level categories to define KBAs (Text Box 1): (A) threatened biodiversity, (B) geographically restricted biodiversity, (C) ecological integrity, and (D) biological processes. Thus, in these cases the KBA element (which is equivalent to the conservation feature in systematic conservation planning) is defined as the biodiversity feature that meets a specific criterion. For example, one KBA element could be an Eastern gorilla (*Gorilla beringei*) population that meets Criterion A1 for threatened species, whereas another element could be a Grauer's cuckooshrike (*Ceblepyris graueri*) population that meets Criterion B1 for geographically restricted biodiversity (Text Box). In addition, the KBA Standard includes a Criterion E—irreplaceability through quantitative analysis, which uses the systematic conservation planning approach to define the environmental objectives and identify sets of KBAs that complement existing sites.

The KBA methodology provides explicit guidance on how to define the elements for which a site is important. In contrast the process of setting targets, which is the other part of defining environmental objectives, is implicit. It derives from the requirements that every KBA has to be monitored as part of the KBA approach and every KBA is important for the global persistence of biodiversity, so destroying any one of them would have negative outcomes that should not go unnoticed. Thus, every KBA must be the focus of the monitoring and vigilance actions stated above, which when translated into the language of decision science means the KBA approach is based on setting 100% targets for every KBA element for these actions (where each KBA element is an assemblage, community, ecosystem type patch or population that meets the relevant criterion threshold—Text Box).

2.3 | Similarities and differences with the two approaches

The systematic conservation planning and KBA approaches can both be framed in the language of decision science, showing they are broadly similar in terms of context and defining environmental objectives. There are also important differences that stem from the context under which the approaches developed (Table 1). The most significant difference is that systematic conservation planning seeks to guide conservation interventions, whereas the KBA approach seeks to identify sites that are globally important for biodiversity. This is reflected in the actions linked to the two approaches: systematic conservation planning identifies where to focus a range of context-specific conservation management, whereas KBA actions are based on monitoring and vigilance at a minimum, making it clear that KBAs are not automatically priority areas for specific types of conservation action (we discuss how KBAs could be used in conservation planning in the next section). It should also be noted that the groups responsible for monitoring and vigilance actions within KBAs may be different from those responsible for implementing the results of any conservation planning exercise.

Other important differences relate to how the environmental objectives are defined. The conservation features in systematic conservation planning are generally biological entities, such as valued species or ecosystems, whereas the KBA

Box 1. Summary of IUCN Key Biodiversity Area Standard criteria

KBAs are based on the following criteria, grouped into five higher-level categories. Each criterion threshold is based on the site containing a specified percentage of the global population size or extent of a species or ecosystem (Criteria A–D) or having a high irreplaceability score (Criterion E). Definitions of the different terms and threshold details are given in the KBA Standard (IUCN, 2016).

A. Threatened biodiversity

Sites trigger identification because they hold a significant proportion of the global population/extent of a species at risk of global extinction or an ecosystem type facing risk of global collapse.

A1. Threatened species

A2. Threatened ecosystem types

B. Geographically restricted biodiversity

Sites trigger identification because they hold a significant proportion of the global population/extent of one or more geographically restricted species, assemblages of species, or ecosystem type.

B1: Individual geographically restricted species

B2: Cooccurring geographically restricted species

B3: Geographically restricted assemblages

B4: Geographically restricted ecosystem types

C. Ecological integrity

Sites trigger identification because they hold the most outstanding global examples of intact ecological communities with supporting large-scale ecological processes.

D. Biological processes

Sites trigger identification because they hold a significant proportion of the global population of a species during one or more life history stage or process, or during periods of environmental stress, or because they produce a very high proportion of the global adult population of a species.

D1: Demographic aggregations

D2: Ecological refugia

D3: Recruitment sources

E. Irreplaceability through quantitative analysis

Sites trigger identification because they have very high irreplaceability for the global persistence of biodiversity as identified through a comprehensively quantitative analysis of irreplaceability.

features are identified (for Criteria A–D) based on the presence of threshold levels of biodiversity elements. A more stark difference comes from the target setting process, which in systematic conservation planning is context-specific. For the KBA approach the implicit targets are 100%, which would be highly unusual in systematic conservation planning, but there are two reasons why they make sense within the KBA context. First, these 100% targets give the KBA methodology a solid theoretical basis within the framework of decision science and, specifically, systematic conservation planning: it means that a site containing any KBA element automatically becomes irreplaceable for monitoring and vigilance. Therefore, it does not matter that KBAs are identified without considering data on costs, threats, or the extent to which the biodiversity in one KBA complements other KBAs or conservation areas. Second, the required monitoring and vigilance actions can be low cost if resources are limited, as illustrated by current KBA monitoring efforts that incorporate remote sensing and/or citizen scientists (Bennun, Matiku, Mulwa, Mwangi, & Buckley, 2005; Kiragu, Butchart, Bennun, & Munyekenye, 2010; Tracewski et al., 2016). In addition, this process is supported by the organizations that form the KBA Partnership (www.keybiodiversityareas.org), as well as many individuals and national institutions involved in KBA identification and monitoring.

This means the KBA Standard (IUCN, 2016) helps address the issues previously identified by members of the systematic conservation planning community (Knight et al., 2007), partly

TABLE 1 Summary of the differences between the systematic conservation planning and KBA approaches

Systematic conservation planning	The Key Biodiversity Area approach
Important sites for conservation.	Important sites for biodiversity.
Accounts for complementarity (i.e., considers the characteristics of other sites).	Does not account for complementarity (only considers the characteristics of the site).
Informs where context-specific conservation management actions should take place.	Informs where monitoring and vigilance actions should take place.
Conservation features include valued biological entities (e.g., species, ecosystems, ecological processes).	KBAs are identified (for Criteria A–D) based on the presence of threshold levels of biodiversity elements.
Conservation features can include ecosystem services.	Does not select sites based on the presence of ecosystem services (but many KBAs are important for ecosystem service provision).
Based on context-specific targets for conservation management actions.	Based on targets of 100% for monitoring and vigilance actions.
Includes cost data (in the simplest cases the cost metric is area selected).	No cost data included.

through the process of updating the thresholds and criteria and partly by the new emphasis on KBAs being selected for their biodiversity importance, rather than as priority sites for any specific type of conservation action. This does not mean that the KBA methodology is perfect, and further testing and possible future revisions may be needed once the criteria and thresholds have been implemented for a wide range of taxa and ecosystems (IUCN, 2016). However, in the meantime, and in the face of rapid global change (Aukema, Pricope, Husak, & Lopez-Carr, 2017; Harfoot et al., 2018), it is reasonable to assert that we should "keep an eye" on every KBA and report to the world whenever one is threatened or loses its status.

3 | POTENTIAL LINKS BETWEEN SYSTEMATIC CONSERVATION PLANNING AND KBAS

Most systematic conservation planning projects that guide conservation management action on the ground aim to achieve local, national and global goals, and should account for corresponding implementation-relevant considerations. Thus, planners will have to make context-specific decisions on whether and how they could incorporate KBAs into their analyses. However, we would argue that every conservation plan would benefit from using KBA elements as conservation features. This is because the identification of every KBA has to go through a stakeholder consultation, review and endorsement process, which involves data validation and boundary delineation using a standard methodology that considers the local context (IUCN, 2016). In addition, some KBA elements represent aspects of biodiversity that are less commonly included in systematic conservation planning because of a lack of data, such as important migration stop-over and aggregation sites, and sites with high ecological integrity. Just as importantly, it provides information for planners who might only know the local importance of particular KBA elements, and support for planners to make the case for conserving globally important biodiversity in the face of competing interests. In both cases, data on KBAs can encourage them to set higher targets for species and ecosystems that are locally common but globally rare. Thus, the World Database of KBAs (www.keybiodiversityareas.org) is an important source of biodiversity data for systematic conservation planning, especially outside of data-rich regions and for poorly studied taxa, ecosystems and biological processes.

When using the KBA data, conservation planners will need to consider two issues. First, they must decide how to set management action targets for each KBA element found within their planning region. These will be different to the implicit 100% monitoring and vigilance targets used in the KBA approach, as although the KBA methodology identifies sites that are globally important for biodiversity, it makes no judgment on how they should be managed or how available conservation resources should be assigned (Dudley et al., 2014). Thus, planners might decide that some types of KBA element should not be priorities for active conservation management given the absence of current threat, limited funding, or other constraints (Figure 2). They might also set management targets for KBA elements based on their relative abundance in their planning region or their importance for achieving other conservation goals (McGowan, Smith, Di Marco, Clarke, & Possingham, 2018). For example, they might decide to set 100% management targets for every KBA element in a region with only a few KBAs (Figure 2a), whereas in a biodiversity-rich region (Figure 2b) targets could range from relatively low for KBA elements found throughout the region to 100% for elements that are highly threatened or rare.

The second issue arises from most KBAs being larger than the spatial resolution of planning units commonly used in national or regional spatial conservation prioritizations. This is because KBAs are delineated to both include the boundaries

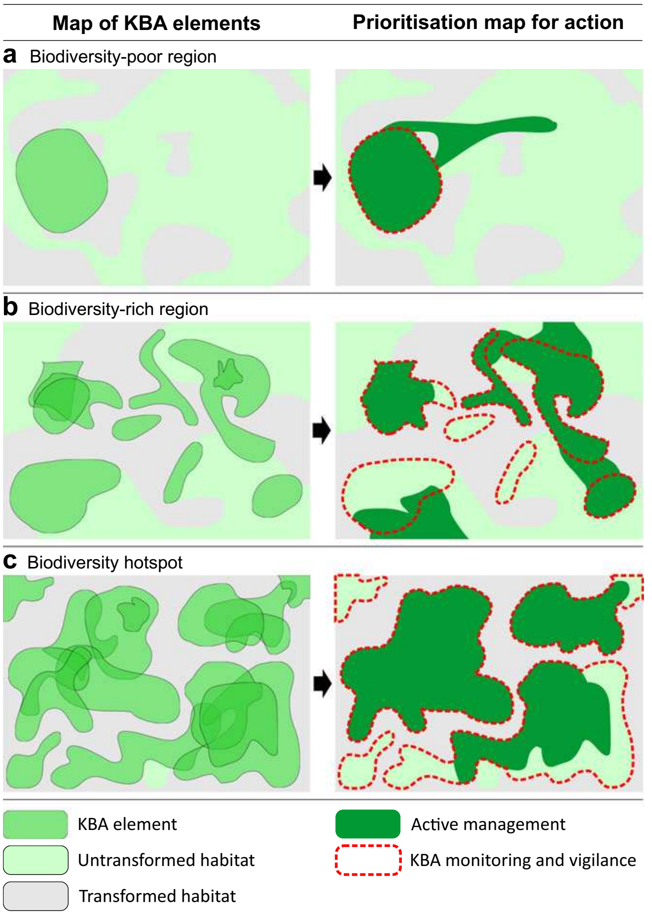

FIGURE 2 Scenarios of how KBA data could inform spatial conservation prioritization. In (a) biodiversity-poor regions there will be few KBA elements and so every KBA is likely to be identified as a priority for active management. In (b) biodiversity-rich regions only some KBAs and KBA sections will be identified as conservation priorities, with a focus on sites containing more than one KBA element. In (c) biodiversity hotspots most untransformed habitat will have KBA status and so prioritization should be informed by KBA element irreplaceability. In all cases: prioritization should be informed by data on threats, opportunities, and resource availability; additional areas will be needed to meet other targets and maintain connectivity; and monitoring and vigilance actions should take place in all KBAs, and ideally throughout the region

of any overlapping KBA elements and to be manageable as a single unit, so often adopt existing or potential conservation area boundaries (IUCN, 2016). If planners decide that their prioritization analysis should select/not select whole KBAs then they should represent them as single planning units, adopting the approach that is commonly used to represent protected areas (Margules & Pressey, 2000). However, if the conservation context is that there are insufficient resources to conserve whole KBAs for some KBA types then they should be represented in the same way as other conservation features (McGowan et al., 2018). This process is supported by the KBA Programme, which aims to encourage KBA proposers to provide maps of where each KBA element occurs within a KBA, providing important data for any conservation planning analysis. Thus planners can calculate the amount of each KBA element (or KBA if that is the only data available) in each planning unit. In most cases this will involve calculating the area of occupancy in each planning unit, but for some species it should be possible to produce actual estimates of population size (McGowan et al., 2018). Planners can then set management action targets for each of these KBAs and KBA elements based on the guidance outlined above.

If planners set a 100% management target for a particular KBA element then it will always be identified in their spatial conservation prioritizations as irreplaceable, so they can begin managing these sites for conservation immediately, even before running the prioritization process. This has obvious advantages, as although rapid systematic conservation planning analyses can provide important information (Smith, Goodman, & Matthews, 2006), collecting the necessary data for a full spatial conservation prioritization takes considerable time and resources (Bottrill & Pressey, 2012). However, a spatial prioritization will still be needed to identify priority areas for meeting the other environmental objectives, using complementarity-based algorithms to select the KBA elements and other features that best meet conservation management targets for developing functional landscapes and seascapes (Iwamura, Waroux, & Mascia, 2018; Knight et al., 2007; Sayer, Carr, & Darwall, in press). Donors, policy makers and management agencies with a KBA conservation remit should also use spatial conservation prioritization to decide which KBAs should be priorities for investment, based on factors such as the number and type of conservation features they contain, the relative importance of the site for those features, threats to the site, and management costs and opportunities. This will be particularly important in biodiversity hotspots (Mittermeier et al., 2004), which typically contain many KBAs because the little remaining habitat contains species with highly restricted ranges (Figure 2c).

Systematic conservation planning also forms the basis of KBA Criterion E, which identifies sites that are irreplaceable because of their combination of biodiversity features. Thus, application of Criterion E in regions where other KBA criteria have been applied will be important for identifying additional sites with high irreplaceability that should qualify as KBAs (Figure 1), despite not meeting thresholds for Criteria A–D (Di Marco et al., 2016). In addition, where a spatial conservation prioritization has already taken place, existing priority areas that meet the threshold-based KBA criteria can be listed, helping give them a higher global profile (Smith, Veríssimo, Leader-Williams, Cowling, & Knight, 2009). Finally, the KBA methodology for delineating important sites could be used when defining the boundaries of priority areas identified through spatial conservation prioritization (IUCN, 2016). Until recently, such delineation was a neglected topic in the scientific literature (Pressey, Mills, Weeks, & Day, 2013), but the developers of the original and revised KBA methodology have covered this issue in depth and systematic conservation planning would benefit from incorporating these methods. Thus, after nearly forty years of parallel development, it is time for planners to integrate the systematic conservation planning and KBA approaches to enrich future conservation policy, practice and research, building on the links and collaborations formed through the process of developing and finalizing the KBA Standard.

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