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LETTER

An Evaluation of Marine Important Bird and Biodiversity Areas in the Context of Spatial Conservation Prioritization

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Abstract

Important Bird and Biodiversity Areas (IBAs) are sites identified as globally important for bird species conservation. Marine IBAs are one of the few comprehensive multi-species datasets available for the marine environment, and their use in conservation planning will likely increase as countries race to protect 10% of their territorial waters by 2020. We tested 15 planning scenarios for Australia's Exclusive Economic Zone to guide best practice on integrating marine IBAs into spatial conservation prioritization. We found prioritizations based solely on habitat protection failed to protect IBAs, and prioritizations based solely on IBAs similarly failed to meet basic levels of habitat representation. Further, treating all marine IBAs as irreplaceable sites produced the most inefficient plans in terms of ecological representativeness and protection equality. Our analyses suggest that marine spatial planners who wish to use IBAs treat them like any other conservation feature by assigning them a specific protection target.

Introduction

Spatial conservation prioritization is the process of identifying priority sites for conservation actions in space and time (Moilanen *et al.* 2009). When designing marine protected area (MPA) networks, priority areas are selected based on several core principles: Representation—ensuring all aspects of biodiversity receive protection (e.g., setting targets for species distributions, abundances, ecological processes, habitats, and/or cultural sites); Adequacy—ensuring what is protected is sufficient to enable biodiversity to persist through time; and Cost-efficiency—ensuring the feasibility of the conservation action has been accounted for and the socio-economic impacts minimized (Ban & Klein 2009; Brown *et al.* 2015). Two additional concepts aid spatial prioritization: (1) Complementarity - selecting suites of sites that collectively ensure all conservation features receive protec-

tion (Moilanen *et al.* 2009); and (2) Irreplaceability—the contribution of a site to meet a pre-established set of biodiversity conservation targets (Ferrier *et al.* 2000). Spatial prioritization is typically performed with software, such as Marxan and Zonation, which operationalize the principles described above (Moilanen *et al.* 2009).

Given the paucity of data available at relevant planning scales for the oceans, practitioners routinely rely on publicly available spatial datasets of habitats and species ranges. Seabird distributions assist in identifying priority areas for marine conservation (Nur *et al.* 2011; Lascelles *et al.* 2012; Ronconi *et al.* 2012; McGowan *et al.* 2013; Bax *et al.* 2016) as seabirds are believed to be important indicators of marine ecosystem function (Furness and Camphuysen 1997; Zacharias and Roff 2001). The seabird conservation community strives to make comprehensive global seabird data available. One such dataset comes from BirdLife International's

Important Bird and Biodiversity Areas (IBA) program which uses a threshold-based approach to identify priority sites based on fulfilling one or more species related criteria (see Appendix A; BirdLife International 2010a).

IBAs are intended to delineate sites that are essential for the survival of birds (O'Dea *et al.* 2006) and the biodiversity they represent (BirdLife International 2010b). More than 12,000 IBAs have been identified, and an additional 2,000 candidate sites have been proposed for the global oceans (BirdLife International 2010b; Lascelles *et al.* 2016). Candidate marine IBAs consist of seaward extensions of seabird breeding colonies, non-breeding coastal congregations, migration bottlenecks and pelagic distributions. The IBA dataset is one of the most comprehensive species-specific datasets available for the oceans.

Marine IBAs will play a significant role in achieving 2020 country goals to conserve 10% of their Exclusive Economic Zones (EEZs; CBD; 2011). Hence there is an urgent need to understand how best to use these sites in future MPA planning. To date, there are no specific guidelines on how best to use marine IBAs for spatial conservation prioritization. Here, we examine 15 different planning approaches for Australia's EEZ (6 million km²) based on using habitat data with three different treatments of marine IBAs. We evaluate the resulting spatial plans with respect to their efficiency and how equally they distribute protection across biodiversity features. We ask the following questions: (1) Are marine IBAs (including candidate sites) effective surrogates for benthic and pelagic marine habitats and to what extent does selecting sites for those habitats also represent IBAs? (2) How does treating marine IBAs as irreplaceable sites influence spatial planning outcomes? (3) What is the best way to integrate IBAs with other biodiversity features when identifying MPA networks? Our analysis aims to identify the best ways to include marine IBAs in spatial prioritization, rather than identify where new MPAs in Australia should be located.

Methods

Spatial data

Marine ecoregions provide a spatial planning framework that captures unique biogeographic assemblages, including biophysical and oceanographic processes (Spalding *et al.* 2007). We used marine bioregions of Australia (Commonwealth of Australia 2006) to stratify seafloor geomorphic features (Harris *et al.* 2014) across seven depth classes and created a dataset of over 1,600 habitat features (Appendix B). We also included the Australian marine IBA inventory (see <https://maps.birdlife.org/marineIBAs/default.html>). This inventory consists of

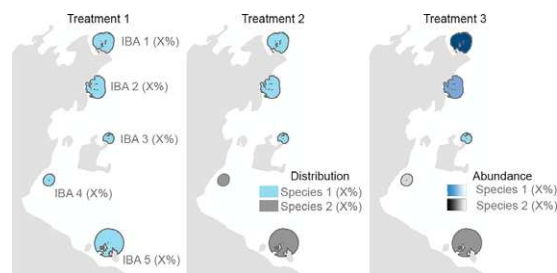


Figure 1 A visualization of how IBAs and targets are considered under our three treatments: (1) as individual features; (2) as the core habitats for the associated trigger species; and (3) as the abundances of associated trigger species where the distribution is weighted by the local population estimates. Note: for clarity in this example each IBA is only triggered by one species but many IBAs are triggered by more than one species.

69 marine IBAs (mean size = 34 km²), triggered by 27 seabird species, most of which are bird congregation sites that fulfill Criterion A4 (Appendix C, Figure 1A). In addition, there are 67 candidate IBAs (mean size = 13,000 km²) triggered by 25 seabird species that fulfill Criterion A1 and A4 (Appendix C, Figure 1B). While provisional, we included candidate IBAs in our analyses because they are published online and distributed for use in conservation planning.

Marine IBA treatments

We treated marine IBA polygons in three ways that are typical of how species data are used in spatial prioritizations (Figure 1). First, we considered each individual IBA as a unique conservation feature (Treatment 1: IBAs as features) noting that there can be many IBAs identified for the same species, and an individual IBA can be designated because of more than one species. Second, we assumed that IBAs associated with individual species represent the most important parts of their distribution throughout the EEZ, and treated the species as the conservation feature for which we set a target (Treatment 2; IBAs as core habitats). Third, we used the maximum population size for each species recorded for each IBA location and assumed this population to be evenly distributed in the planning units found within each IBA (Treatment 3: IBAs as abundances). In this case the conservation feature is also the species, but its value is weighted by its local abundance in each planning unit. For more details on the spatial prioritization problem (see Appendix D).

Conservation targets

Targets for conservation features are often set based on characteristics such as endemism, rarity, risk of

Table 1 Scenario matrix to derive 15 spatial planning approaches for habitats and IBAs

Habitats (features N=1659)		Treatment 1: IBAs as features (N=136)			
		100%	Sliding scale	20%	0%
		20%	1b	1d	1c
0%	0	1f	1e	-	
Treatment 2: IBAs as core habitats (N=33)					
20%	-	2a	2c	-	-
0%	-	2b	2d	-	-
Treatment 3: IBAs as abundances (N=33)					
20%	-	3a	3c	-	-
0%	-	3b	3d	-	-

extinction or threat, or political drivers such as policy or planning precedent (Carwardine *et al.* 2007). We constructed planning scenarios by varying the targets for habitats and marine IBAs (Table 1). We used a constant 20% target for habitat features following the zoning plan for Australia's Great Barrier Reef (Fernandes *et al.* 2005). We varied the targets for marine IBAs across treatments in three ways. First, we considered them as irreplaceable sites and set a target of 100% for each IBA (Scenario 0 and 1b). Second, we used a "sliding scale" approach (Figure 1) based on the IUCN Red List status of the species used to identify the IBAs (the "trigger" species). Following Australian terrestrial reserve policies (Commonwealth of Australia 1997), we set targets as: Least Concern = 10%; Near Threatened = 20%, Vulnerable = 60%, Endangered = 90%. When more than one trigger species was used to identify an IBA, we set the target based on the species with the highest threat category. Third, we set targets for IBAs equal to the other conservation features at 20%. We note that while setting different targets based on the criteria for which an IBA is established may seem logical, a defensible, systematic and appropriate way to justify this approach does not exist yet. We assessed a suite of 15 typical approaches for setting marine conservation priorities based on three treatments of the IBA data (Table 1).

Evaluation metrics

We defined surrogacy as the incidental coverage of features captured in a spatial plan when no specific targets were set for those features (Grantham *et al.* 2010). Sutcliffe *et al.* (2015) introduced the percentage gap metric as an evaluation of surrogate effectiveness in spatial prioritization. This metric measures the average target shortfall for a given scenario when surrogates drive the prioritization. For example, if all features miss their target by 20%, or if a third miss their target by 60%, then the percentage gap is 20%. If all targets are met the percentage gap is 0. We used this metric to evaluate the surrogate effectiveness of plans based on marine IBAs (Scenario 0)

and habitat features (Scenario 1a) to meet 20% targets for each other, as well as the geographic ranges of 58 seabird species found within Australia's EEZ (BirdLife International and the Handbook of the Birds of the World 2016).

We also used the proportional protection equality metric (*PE*) to evaluate how equally the network of protected areas represents features in each scenario (see Appendix E; Chauvenet *et al.* 2017). We evaluated scenarios in terms of the trade-off between representation (measured in *PE*) and cost-effectiveness with respect to two kinds of conservation features—habitats (measured in proportions of bioregions protected) and species (measured in proportions of total abundances protected derived from IBA Treatment 3) and the average of these two values. We defined the upper and lower bounds of this trade-off as efficiency and inefficiency frontiers, respectively.

Results

Surrogate performance to achieve representation targets

The effectiveness of individual marine IBAs and habitats (including bioregions) to act as surrogates for each other was poor. A conservation prioritization based solely on IBAs (Scenario 0) adequately met targets for only 519 of the 1,659 habitat features, and offered no protection to 828 of the smaller range features with a percentage gap of 62%. Similarly, a conservation prioritization based solely on habitats (Scenario 1a) met targets for only 55 of 136 marine IBAs, with a percentage gap of 49% (percentage gap graphs for these scenarios can be found in Appendix F).

Interestingly, spatial prioritizations driven solely by IBAs (Scenario 0) performed poorly as surrogates for trigger species' ranges when we set 20% targets (Figure 2A). IBAs biased the protection of seabird ranges away from nontrigger species (seven of which are listed as Vulnerable or higher on the IUCN Red List), with large discrepancies in the amount of species-level protection provided (percentage gap = 19%; Figure 2B). In contrast habitats were effective surrogates for representing seabird ranges at the 20% level, meeting targets for all IBA trigger species (Figure 2C) and with only a negligible percentage gap (0.02%) for nontrigger species (Figure 2D).

PE, cost-efficiency and IBA treatments

Given IBAs and habitats were poor surrogates for one another, we focus the following results only on those scenarios where both habitats and IBAs were included in the analysis (for all scenarios see Appendix G-H). Setting

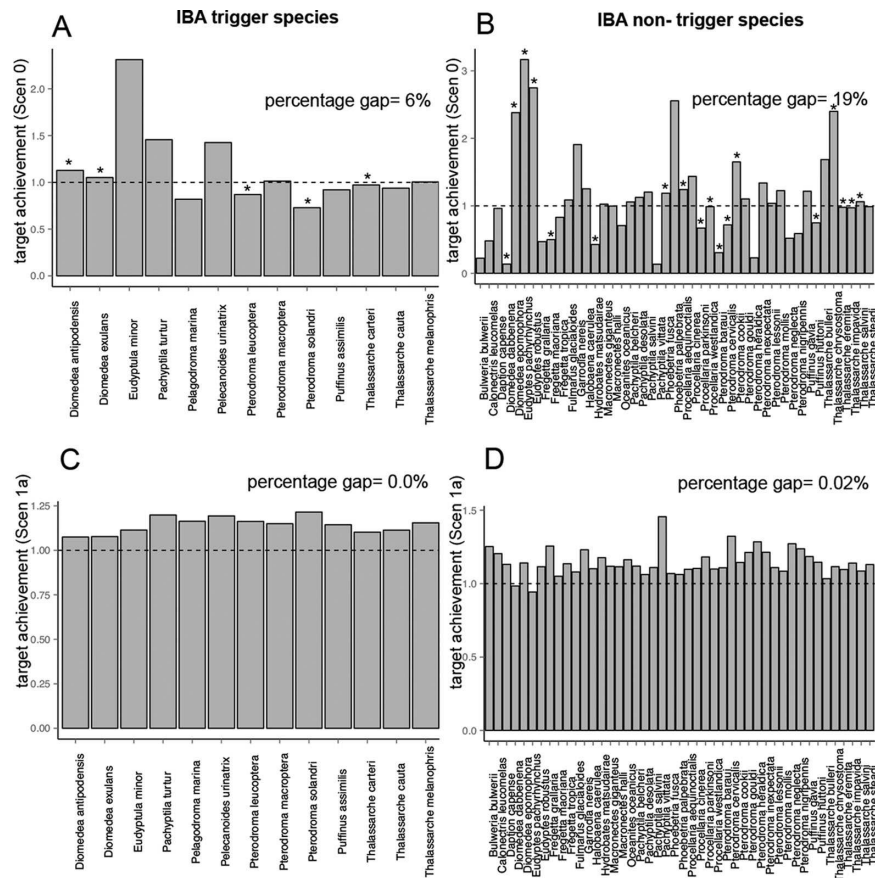


Figure 2 The surrogate effectiveness of Australia’s marine IBAs (A–B), and habitats/bioregions (C–D) to achieve 20% conservation targets for the pelagic distributions of Australian seabird species. Results are reported as the percentage gap metric of each scenario’s performance for species that do and not trigger an IBA. Asterisks note species listed as Vulnerable or higher according to the IUCN Red List.

targets for individual IBAs (Treatment 1) at 100% (e.g., treating them as irreplaceable sites) and habitats targets of 20% (Scenario 1b) was the worst scenario in terms of *PE* (Figure 3A–C). When we instead assigned a 20% target to both IBAs and habitats (Scenario 1c), the bioregional *PE* improved by 28% and the cost-efficiency of the prioritization improved by more than 50% (Figure 3A; Table 2). While this treatment did not perform as well as other treatments for *PE* across species (Figure 3B), it formed the upper bound of the efficiency frontier for the average *PE* values (Figure 3C).

Treating IBAs as species core habitats (Treatment 2) produced mixed results. Setting sliding scale targets for the species and 20% targets for habitats (Scenario 2a) returned relatively inefficient results compared to setting a flat 20% target for each species distribution (Scenario 2c). While Scenario 2c provided the most equal protection for bioregions of all scenarios (*PE* = 0.94, Table 1), the network performed worse than average for species *PE* (*PE* = 0.62; Figure 3B).

Weighting IBAs by species abundances (Treatment 3) and setting sliding scale targets performed roughly the same for average *PE* (Scenario 3a: *PE* = 0.84) as setting 20% targets for both IBAs and habitats (Scenario 3c: *PE* = 0.83). However, the latter was accomplished at nearly half the cost (Figure 3C; Table 2) and is a scenario on the upper bound of the efficiency frontier (Figure 3).

Discussion

Marine IBAs provide species-level data to inform spatial planning for expanding the global MPA estate. However, in the absence of best practice guidelines on how to use the marine IBA dataset, spatial planners are left with several options: not use the IBAs; treat them as irreplaceable sites (i.e., protect 100% of every IBA); or treat them like any other conservation feature and assign them a protection target. Our analysis explores these options by first assessing IBAs as surrogates for benthic and pelagic biodiversity, followed by an evaluation of different IBA

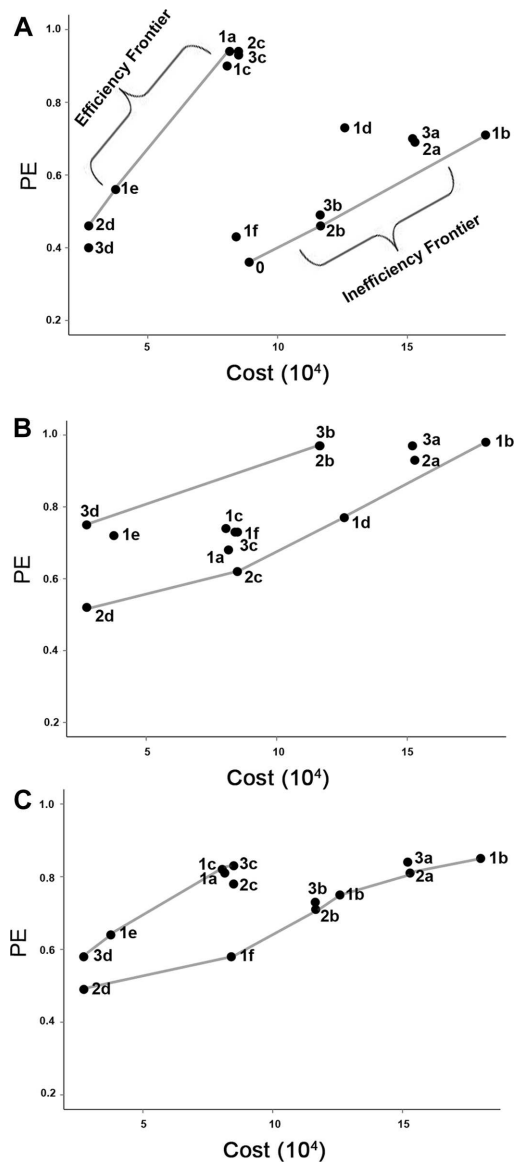


Figure 3 The network costs and protection equality (*PE*) for each scenario's best solution (see Table 2). *PE* is calculated for (A) Australia's bioregions; (B) IBA trigger species abundances (derived from Treatment 3); and (C) the average of the two. To assist with evaluating the trade-off between costs and *PE*, we consider those scenarios on the upper and lower bounds of the graphs as efficiency and inefficiency frontiers. Point labels correspond to the scenarios described in Table 1.

target-setting strategies and the influence they hold in providing cost-effective and equitable MPA networks.

Ignoring IBAs overlooks unique sites for seabirds

Neither marine IBAs nor habitat features were effective surrogates for each other. IBAs were also poor surrogates

for the pelagic ranges of Australian seabirds, but much of the range of these species consists of low-quality habitat that is not likely essential for species persistence. This suggests that IBAs help address the principle of adequacy by identifying areas that support these critical processes rather than representing patterns of biodiversity. However, we cannot attest to the merits of marine IBAs to capture biodiversity in other regions of the world. Having an IBA dataset based on a comprehensive evaluation of Australia's entire EEZ would likely increase the surrogate performance of IBAs for other biodiversity. Similarly, targeting a higher percentage of habitats would likely improve their performance as surrogates for IBAs. Importantly, our analysis suggests that it is prudent to include IBAs as an additional data layer despite them not representing broader marine biodiversity because IBAs contain unique spatial information on the processes that drive seabird abundances and distributions.

Treating marine IBAs as irreplaceable is very inefficient

BirdLife International's IBA program considers all IBAs to be equally important for marine biodiversity (BirdLife International 2010b; Butchart *et al.* 2012). The strictest translation of these guidelines suggests that every IBA should therefore be conserved in its entirety and treated as irreplaceable. When we treated marine IBAs as irreplaceable sites (e.g., 100% of every IBA is conserved), we arrived at the worst performing scenario for habitat-based *PE* (Figure 3A; Scenario 0). Treating IBAs as irreplaceable sites and setting additional targets for biodiversity features resulted in the most inefficient network across all 15 scenarios (Figure 3A–C; Scenario 1b). When only considering habitat features, we observed far better performance for cost-efficiency (~50% cost reduction) for a small reduction in the averaged *PE* of species abundances and habitats (Figure 3C; Scenario 1a). These results demonstrate that treating marine IBAs as irreplaceable by conserving the full extent of every marine IBA, particularly large IBAs, is not a practical way to build MPA networks that are ecologically representative and cost-efficient.

Recommendations for integrating IBAs into spatial prioritization as conservation features

Spatial conservation prioritization delivers spatial plans that cost-effectively represent biodiversity within a complementary network of sites. IBAs can be an important source of such biodiversity data and our results suggest marine IBAs should be treated as conservation features for which a target is set. When we set 20% targets for both IBAs and habitats, we produced plans that are

Table 2 Results of the best solutions for each scenario and associated *PE*. The best performing scenarios are based on the trade-off between *PE* and cost-effectiveness (in bold) derived from Figure 3. Scenario 0 has NAs because *PE* = 1 by default due to the inclusion of 100% of each IBA

Scenario	Cost of network	No. planning units	<i>PE</i> of best solution (bioregions)	<i>PE</i> of best solution (species abundance)	Average <i>PE</i>
Surrogacy scenarios					
0	89,043	10,095	0.36	NA ⁺	NA
1a	81,510	16,189	0.94	0.68	0.81
Treatment 1: IBAs as features					
1b	180,087	27,439	0.71	0.98	0.85
1c	80,549	15,864	0.90	0.74	0.82
1d	125,904	21,553	0.73	0.77	0.75
1e	37,605	8,017	0.56	0.72	0.64
1f	84,047	15,178	0.43	0.73	0.58
Treatment 2: IBAs as core habitats					
2a	152,920	23,346	0.69	0.93	0.81
2b	116,572	16,266	0.46	0.97	0.71
2c	84,951	16,416	0.94	0.62	0.78
2d	27,235	6,713	0.46	0.52	0.49
Treatment 3: IBAs as abundances					
3a	152,008	23,667	0.70	0.97	0.84
3b	116,415	16,601	0.49	0.97	0.73
3c	84,988	16,326	0.93	0.73	0.83
3d	27,182	6977	0.40	0.75	0.58

reasonably efficient and representative—defining the upper bounds of the efficiency frontier for average *PE* and cost (Figure 3C; Scenarios 1c, 2c, 3c). Notably, these targets were somewhat arbitrary and proportional representation does not guarantee the long-term viability of species.

Large inefficiencies emerged when we set targets for IBAs based on species threat status. This was principally due to the size of the candidate IBAs driven by Endangered and highly mobile species. For example, the Endangered Black-browed albatross defines six candidate IBAs covering approximately 248,000 km² of Australia's Southern Ocean and for which we set a 90% target according to Australian policy (see Appendix G). This exposes the challenge of defining pelagic IBAs where no clear biogeographic or habitat boundary exists and which are identified using seabird tracking datasets (Lascelles *et al.* 2016). While we encourage improving the conservation return on investment from global seabird tracking (McGowan *et al.* 2017), protecting large pelagic IBAs is probably not politically or socially feasible. Such IBAs may be best used to inform broad policy not MPAs.

While planners may consider setting differential targets based on species attributes, such as range size (Rodrigues *et al.* 2004), the characteristics of individual IBAs could also influence target setting. For example, planners could set higher conservation targets for IBAs capturing seaward extensions of nesting colonies or mi-

gration bottlenecks for declining populations, than for large pelagic IBAs. Appropriate targets will depend on the objectives of the spatial plan, the size of the planning region, the number of marine IBAs identified for the region and what other species-level data are available.

The future of IBAs in marine spatial planning

Using criteria to identify sites that are globally significant for biodiversity is not equivalent to identifying priority conservation areas for action (Knight *et al.* 2007; Di Marco *et al.* 2016; IUCN 2016). Criteria-based delineations often do not explicitly state what actions should be taken to ensure species persist within those sites, nor do they routinely account for the financial, social and political constraints associated with implementing conservation actions. Further, these sites do not account for complementarity in site identification (Brown *et al.* 2015). Di Marco *et al.* (2016) stressed the importance of complementing the threshold-based identification of terrestrial IBAs with the systematic identification of irreplaceable sites. Our results support the same claim for the marine realm. While we found little support for treating marine IBAs as universally irreplaceable, our recommendations do not preclude setting 100% targets for particular IBAs when appropriate. Following the terrestrial analysis of Di Marco *et al.* (2016), evaluating how different IBA criteria reflect the irreplaceability values of marine IBAs would be a valuable next step towards establishing a

systematic method for setting targets based on underpinning criteria.

The integration of marine IBAs into spatial conservation prioritization demands planners be equipped with more specialized knowledge of how and why individual IBAs exist. Attributes of the IBA trigger species and criteria, as well as the method of establishment (e.g., whether through telemetry tracking (Lascelles *et al.* 2016), at-sea surveys (Smith *et al.* 2014), expert opinion, or identifying foraging hotspots (Arcos *et al.* 2012)) should be provided. While some IBAs may be too large for strict protection they could inform specific non-spatial actions such as bycatch reduction policies and defining places where new marine activities must go through more rigorous impact assessment. Releasing prescriptive actions associated with these spatial data will further advance the utility of IBAs in spatial conservation prioritization. We believe these findings are relevant to other threshold-based approaches, such as Key Biodiversity Areas (IUCN 2016) and Ecologically or Biologically Significant Areas (Bax *et al.* 2016), where sites are delineated with the intent of influencing global conservation priority setting.

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

Appendix A. Criteria for Marine Important Bird and Biodiversity Areas.

Appendix B. List of habitat features.

Appendix C. Hierarchical visualization of confirmed and candidate Marine IBAs.

Appendix D. Explanation of how Lorenz curves derive proportional protection equality (PE) values.

Appendix E. Results of the surrogacy effectiveness analysis derived from the Percentage gap metric.

Appendix F. Best solutions for all 15 planning scenarios.

Appendix G. Lorenz curves to derive protection equality (PEP) for (a) bioregions (b) species abundances (See Appendix D for explanations).

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