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Cover description: Young mountain gorilla photographed in the Volcanoes National Park in Rwanda. Photo credit: Liana Joseph.

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Conservation Letters is a scientific journal publishing empirical and theoretical research with significant implications for the conservation of biological diversity. The journal welcomes submissions across the biological and social sciences - especially interdisciplinary submissions – that advance pragmatic conservation goals as well as scientific understanding. Manuscripts will be published on a rapid communications schedule and therefore should be current and topical. Research articles should clearly articulate the significance of their findings for conservation policy and practice.

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VIEWPOINT

Global Biodiversity Targets Require Both Sufficiency and Efficiency

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With the adoption of the 2011–2020 Strategic Plan of the Convention on Biological Diversity (CBD), 196 nations agreed to achieve ambitious biodiversity-related targets. These targets encompass conservation inputs, such as increasing the amount of financial resources invested in biodiversity conservation (Target 20), conservation outputs, such as protecting areas of particular importance for biodiversity and ecosystem services (Target 11), and conservation outcomes, such as preventing the extinction of threatened species (Target 12). The evidence to date reveals limited progresses in achieving these targets, especially those related to conservation outcomes, and an alarming disparity between the rate of biodiversity decline and the rate at which conservation actions take place (Tittensor *et al.* 2014).

International biodiversity targets are essential for coordinating global conservation efforts, and we believe that the conservation community should improve upon existing CBD targets to have a better chance of achieving the overall vision of ending the ongoing biodiversity crisis. We argue that it is now time that targets clearly outline what is "sufficient" in conservation terms, and that nations identify "efficient" ways to achieve these targets.

Defining sufficient biodiversity targets

"How much is enough?" is a core question that should guide the definition of sufficient biodiversity targets, that is, adequate levels of conservation inputs, outputs, and outcomes necessary for the protection of biodiversity. However, this question does not seem to guide current CBD targets, which, despite more than two decades of development and monitoring, still suffer from ambiguity, unquantifiability, complexity, and redundancy (Butchart *et al.* 2016). For example, Target 11 calls for the conservation of at least 17% of terrestrial and 10% of marine areas—"especially areas of particular importance for biodiversity and ecosystem services"—through "effectively and equitably managed, ecologically representative, and well-connected systems of protected areas." This target includes seven different elements (Butchart *et al.* 2016), most of which are not quantified and none of which reflect what is sufficient from a biodiversity perspective. Many have argued that even if the static areal element of this target was globally achieved, it would not be enough to protect marine and terrestrial biodiversity (Venter *et al.* 2014; Butchart *et al.* 2015; O'Leary *et al.* 2016).

Specifically Butchart *et al.* (2015) found that protection of 26% of terrestrial land is required to adequately represent known threatened species and their habitats (28% if also considering nonthreatened species). This finding is likely to have correspondence in the marine realm, where scientists called for at least 30% protection of the oceans (O'Leary *et al.* 2016). We recognize that value judgements are involved here, for example, in determining an "adequate" representation for species. However, this does not reduce the need for pursuing sufficiency in biodiversity targets setting, based on the best available scientific knowledge.

As different elements vary in scale and purpose within the protected area target (e.g., protecting areas important for biodiversity, achieving a representative sample of ecosystems, achieving connectivity), and within all the other targets, there is a need for clear science to derive measures of sufficiency to help define the targets. This is doable. In the case of the above-discussed Target 11, a sufficient protection can be sought in relation to the areal extent required to ensure coverage for all known threatened species and habitats, for example, 30% coverage for the currently unprotected Clarke's Gazelle (if scaling the target according to species' range size; Venter *et al.* 2014; Butchart *et al.* 2015). In the case of Target 15, which calls for the restoration of at least 15% of degraded ecosystems globally, a possible sufficient formulation could be set around restoring the average abundance of native species to 90% or more of their value in natural habitats (Newbold *et al.* 2016).

Defining efficient conservation strategies

The achievement of biodiversity targets is often hindered by the inefficient allocation of conservation resources, for example by not locating protected areas in the most cost-efficient places for protecting threatened species (Venter *et al.* 2014). One solution to overcome this inefficiency is for countries to adopt explicit formulations of the resources allocation problem (Wilson *et al.* 2006), in which investments are allocated in space and time toward specific actions for achieving multiple biodiversity targets, such as protected area expansion and extinction risk reduction. Empirical evidence demonstrates that, if implemented, this strategic approach can produce a much more efficient allocation of conservation resources, with small changes in budget (Venter *et al.* 2014; Polak *et al.* 2016). An example of where improvement could be easily made is the derivation of national conservation strategies which explicitly prioritize protection in areas where underrepresented ecosystems are subject to the greatest threat levels (Watson *et al.* 2016).

An important part of an efficient global plan for biodiversity conservation is the establishment of an efficient framework for monitoring progress toward targets. However, the set of indicators used for target monitoring is sometimes inadequate, hindering the ability to accurately monitor some of the targets (Shepherd *et al.* 2016). More alarmingly, there is evidence that different indicators can lead to contrasting assessments. For example, species richness can remain stable in an area for a long period of time even when species abundance declines drastically (Hill *et al.* 2016). Identifying a comprehensive set of indicators, which are able to represent the changing state of a study system (e.g., the threatened species of a country), is an important step to be taken every time new targets are being defined. For each indicator, it is important to clarify whether it refers to conservation outputs (e.g., more protected areas) or outcomes (e.g., higher species abundances), what is the availability of baseline data, and what is the cost of collecting and maintaining new data. There are now new metrics that are readily available for target monitoring, such as "protection equality", which can be used for measuring the ecological representation of national protected area systems (Kuempel *et al.* 2016).

The role of conservation scientists in pursuing sufficiency and efficiency

Many studies have shown that global biodiversity targets do not set out what is sufficient to prevent ongoing biodiversity decline, and that national strategies to achieve these targets have been inefficient in their allocation of limited resources. We believe it is timely to constructively build on these findings, and that more scientists become proactively engaged with parties involved in targets setting. Scientists should provide policy makers with direct evidence of how alternative formulations of targets, and strategies to achieve them, can lead to improved biodiversity outcomes. An opportunity for this increased engagement will be the definition of post-2020 targets. These future targets are likely to play a fundamental role in supporting the UN's Agenda for Sustainable Development, through which the world's governments have agreed to achieve ambitious social, economic, and environmental goals by 2030. We believe that incorporating elements of sufficiency and efficiency into future global biodiversity targets is key to support their role in guiding global conservation efforts.

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LETTER

Effective Coverage Targets for Ocean Protection

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Abstract

The UN's globally adopted Convention on Biological Diversity coverage target for marine protected areas (MPAs) is ≥10% by 2020. In 2014, the World Parks Congress recommended increasing this to ≥30%. We reviewed 144 studies to assess whether the UN target is adequate to achieve, maximize, or optimize six environmental and/or socioeconomic objectives. Results consistently indicate that protecting several tens-of-percent of the sea is required to meet goals (average 37%, median 35%, modal group 21–30%), greatly exceeding the 2.18% currently protected and the 10% target. The objectives we examined were met in 3% of studies with ≤10% MPA coverage, 44% with ≤30% coverage, and 81% with more than half the sea protected. The UN's 10% target appears insufficient to protect biodiversity, preserve ecosystem services, and achieve socioeconomic priorities. As MPA coverages generated from theoretical studies inherently depend on scenario(s) considered, our findings do not represent explicit recommendations but rather provide perspective on policy goals.

Introduction

Global concern regarding environmental degradation and anthropogenic impacts on marine ecosystems has led to urgent calls to increase the global coverage of marine protected areas (MPAs), the aim being to preserve and recover what remains of ecosystems, and prevent further declines. The Convention on Biological Diversity (CBD) target currently commits signatory governments to conserving ≥10% of marine environments by 2020 through "ecologically representative" protected area networks (Convention on Biological Diversity 2010). The 2014 World Parks Congress called for at least 30% of each marine habitat to be included within highly protected MPAs, increasing a previous recommendation for 20–30% coverage made in 2003 (World Parks Congress 2014).

MPAs are one of the principal tools advocated to preserve and maintain biodiversity and ecosystem services, and to mitigate negative effects of anthropogenic activities (e.g., Lubchenco *et al.* 2003; Angulo-Valdés & Hatcher 2010; Halpern *et al.* 2010; Roberts 2012). They are areas where human activities have been restricted to varying degrees with the aim of protecting living and nonliving resources and, while most commonly established for conservation purposes, they are also recognized as a tool for commercial fish stock management and recovery (FAO 2011; Vandeperre *et al.* 2011; Rice *et al.* 2012; Roberts & Hawkins 2012).

Although protected area coverage targets have been controversial (Carwardine *et al.* 2009), they have driven international and national policy and collective action to increase conservation both on land and for the sea (Jenkins & Joppa 2009; Gleason *et al.* 2013; Botsford *et al.* 2014; Watson *et al.* 2014). While undoubtedly political, such targets should be based on robust scientific evidence if they are to meet their environmental objectives. Given the recent adoption by the UN of a Sustainable Development Goal for the oceans, with the 10% MPA goal embedded within it (Goal 14: Conserve and sustainably use the oceans, seas, and marine resources), and of the upcoming CBD Conference of Parties in 2016, it is timely to evaluate the evidence base for effective MPA coverage.

Previous reviews in 2003 (N = 40 studies, Gell & Roberts 2003) and 2010 (N = 33 studies, Gaines *et al.* 2010) suggested that 20–40% coverage is warranted. In view of the large disconnect between the UN 10% MPA target and the results of these studies, a broader synthesis of current research is required. We investigate six objectives of MPAs that together encompass the results of all studies examined: (1) protect biodiversity; (2) ensure population connectivity among MPAs; (3) minimize the risk of fisheries/population collapse and ensure population persistence; (4) mitigate the adverse evolutionary effects of fishing; (5) maximize or optimize fisheries value or yield; and (6) satisfy multiple stakeholders (i.e., studies contain analyses designed to identify the required percentage coverage to minimize trade-offs between stakeholders and maximize value [e.g., Boncoeur *et al.* 2002]). These objectives were chosen following an initial scoping study and represent objectives orientated toward conservation goals (objectives 1, 2, and 4), socioeconomic priorities (objective 5), or elements of both (objectives 3 and 6).

Here we conduct an assessment of scientific literature to determine whether existing targets for ocean protection are adequate to achieve, maximize, or optimize the various objectives expected from MPAs, as appropriate to the goals considered.

Methods

Selection of articles

An intensive search of peer-reviewed scientific literature was undertaken in Web of Science and Scopus. In addition, we conducted a bibliographic search of all relevant review articles identified in our searches to ensure all relevant articles were identified. Initial searches were undertaken in December 2014 in Web of Science and subsequently updated in March and October 2015 in Web of Science and Scopus. Subsequent updates restricted searches in Web of Science to articles published during or after 2014 or 2015, respectively, and undertook new searches in Scopus without date restriction. Search terms were identified by reference to articles cited in relevant reviews (Gell & Roberts 2003; Gaines *et al.* 2010), consultation with subject experts within the review group and simplified trial searches. Table S1 (Supporting Information) details the combinations of the search terms used. Only English language articles were assessed.

Study inclusion criteria

We established an a priori protocol for the search strategy and criteria for inclusion and exclusion of studies in our review. Included studies were required to contain the following elements: *Population*: Any marine environment. Studies considering protected areas in estuarine, freshwater, or terrestrial environments were excluded; *Intervention*: Included studies should consider the proportion of the sea that should be protected within MPAs to achieve, maximize, or optimize the objectives they investigated. Studies that used an inadequate sample size to enable investigation of appropriate coverage were excluded (i.e., scenarios should assess a minimum of 4% coverage values across a range and results should clearly indicate where objective(s) were achieved/maximized/optimized); *Time and Place*: Studies produced at any time and using any location as a case study were included, as were those using theoretical mathematical modeling approaches with numerical illustration; and *Outcomes*: Included studies must contain results that indicate a percentage, or range of percentages, of MPA coverage to achieve, maximize, or optimize the objective(s) investigated within each study. Objectives may be related to environmental or socioeconomic impacts, including but not restricted to: ecosystem functioning [biodiversity, abundance, connectivity] and human health and well-being [income, employment, fisheries yield]. Note that this percentage may be zero and that overall coverage can be calculated from appropriate size and spacing recommendations, e.g., an MPA size of 20 km width with spacing recommendations of 40 km would give a coverage of 33%. Studies that consider the design (size, spacing, shape, etc.) of MPAs but not overall coverage and where overall coverage cannot be calculated were excluded.

Article screening

The first 100 hits (based on sorting of relevance of results) from each search in Web of Science were screened. All articles identified in Scopus were screened. The results from each search were combined in a single Endnote library file and duplicates removed. All articles retrieved were assessed for inclusion in our review based on a hierarchical assessment of relevance by screening article titles, then abstracts of articles with relevant titles, followed by the full text of potentially relevant articles. Studies were considered relevant based on the inclusion criteria. If the relevance of articles was unclear at title and abstract stages they were included and assessed at full text. The aim of this process was to systematically remove articles that did not contain relevant information to our study. A schematic showing the processes involved in this review and numbers of articles and studies moving between stages is shown in Figure S1.

Data handling and analyses

Data extracted from each relevant article included the full reference and the percentage(s) of area or stock protected which achieved, optimized, or maximized the investigated objective(s).

Percent coverages were recorded according to each objective as either a range or single value depending on results reported within each study. Where multiple individual percent coverages were reported, we recorded these as a range (i.e., the minimum and maximum values reported were recorded) to encompass the full spread of results. Analyses were undertaken based on the median of the range, or the single value reported by each assessed article. Equal weighting was assigned to each study contained within this synthesis as included studies are essentially theoretical and therefore no a priori reason exists for weighting one study more highly than another. Each study was assigned to one or more objectives as appropriate. As results for each objective inevitably consisted of different sample sizes, numerically dominant groups will therefore be overrepresented in combined results. Coverages for different objectives were statistically compared using the Kruskal-Wallis H test to ensure overall results were representative of all objectives. To further test for possible bias resulting from uneven sample sizes, equal weighting was assigned to each objective by calculating the proportion of studies within each MPA coverage class (0–10%, 11–20%, etc.), i.e., each objective totals to one, and then averaging results across objectives. For these data, the mean, median, and modal group were estimated to enable comparison with our unweighted results. We used Mann-Whitney U to test for differences between required coverages in temperate and tropical ecosystems.

Results

We identified 126 relevant articles published between 1995 and 2015 (Figure S2); 96.8% of which used modeling approaches (including numerical simulations, decision-support tools, species-area relationships, and GIS modeling) with the remainder using literature review techniques ($N = 2$) or expert/stakeholder-driven processes ($N = 2$). These articles collectively contributed 144 studies (i.e., data points) to our analysis, given some papers addressed multiple objectives. Included studies are detailed in Table S2. Considerable variability in required MPA coverage among studies was found however mean and median results were highly consistent across a diverse range of objectives, converging between 30% and 40% (Figure 1 and Table S3). There was no significant difference in required coverage among the five different goals with sufficient sample sizes to offer adequate statistical power (protect biodiversity, ensure connectivity, avoid collapse, fisheries value, and multiple stakeholder satisfaction: Kruskal-Wallis H = 2.59, 4 df, $P = 0.63$).

On average, the required coverage for protection to achieve, maximize, or optimize the objective(s) investigated was 37% of the sea (median 35%, modal group 21–30%). Over half of all studies (56%) indicate that >30% of the sea should be protected to meet the goal they investigated (Figure 2). Eighty-one percent of goals were met with >50% coverage, but only 3% of goals were met with ≤10% coverage (Figure 2). When equal weighting was applied to each MPA objective, it had minimal effect on the results (equal weighting of objectives: mean 35% coverage, median 32%, modal group 21–30%; Figure S3). Table S3 provides summary statistics for each of the six objectives investigated and overall results. We also found no significant difference in required MPA coverage between studies undertaken with specific regard to either tropical (mean 34%, $N = 33$ studies) or temperate (mean 38%, $N = 47$ studies) ecosystems (Mann-Whitney U = 726, $Z = 0.48$, $N_1 = 47$, $N_2 = 33$, 1 df $P = 0.63$).

Discussion

MPAs are a critical part of the toolkit for biodiversity conservation and fisheries management (Roberts *et al.* 2005). However, while observational evidence detailing their many potential benefits exists (e.g., Lester *et al.* 2009; Fenberg *et al.* 2012; Baskett & Barnett 2015; Caselle *et al.* 2015; Huijbers *et al.* 2015) it is not practical to experimentally answer how much of the sea requires protection to safeguard biodiversity, preserve ecosystem services, and ensure socioeconomic priorities. Consequently, syntheses of theoretical research examining aspects of this question are required.

Previous reviews (Gell & Roberts 2003; Gaines *et al.* 2010) have suggested between 20% and 40% of the sea should be protected to achieve MPA goals. We update, expand and increase the rigor of these analyses, identifying an additional 93 articles previously not considered and discounting a further 41 articles previously included on the basis of those articles not being sufficiently thorough to meet our inclusion criteria. Our findings suggest that the objectives we examined are rarely secure with MPA coverage in single percentage figures—the status quo—and the picture was little improved with ≤10%

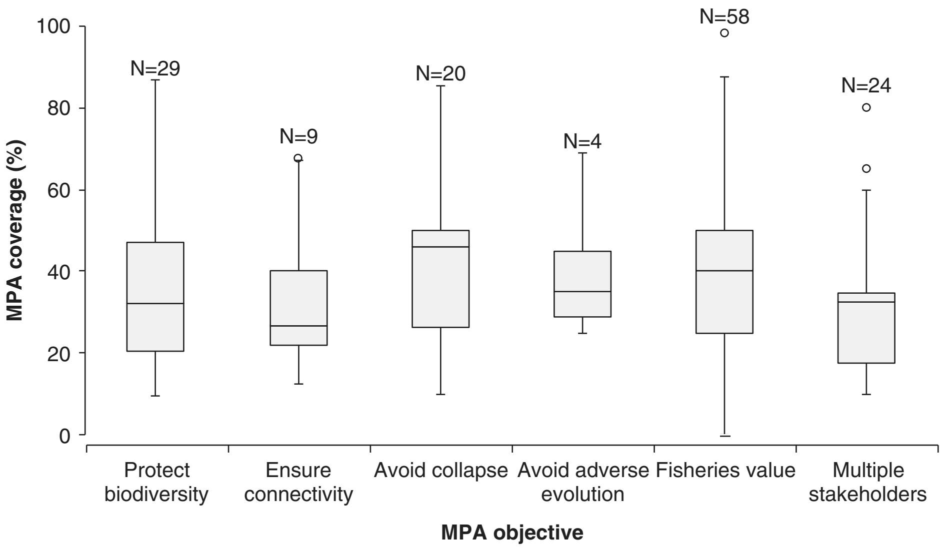

Figure 1 Tukey boxplot showing the range of required coverage for each MPA objective: (1) protect biodiversity ($N = 29$, median 32%, range 9–80%); (2) ensure population connectivity ($N = 9$, median 27%, range 13–68%); (3) minimize the risk of fisheries/population collapse and ensure population persistence ($N = 20$, median 46%, range 10–76%); (4) mitigate the evolutionary effects of selective fishing ($N = 4$, median 35%, range 25–59%); (5) maximize or optimize fisheries value or yield ($N = 58$, median 40%, range 0–98%); and (6) satisfy multiple stakeholders ($N = 24$, median 33%, range 10–80%). Outliers shown by open circles.

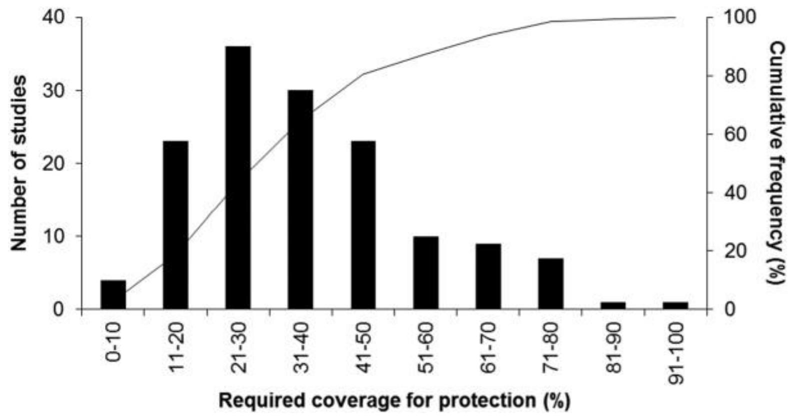

Figure 2 Frequency distribution of the required coverage for protection to meet MPA objectives based on 144 studies. Cumulative frequency (solid line) showing the percentage of studies that consider MPA goals will be met at each coverage level.

coverage. While achieving 10% coverage by 2020 is extremely ambitious politically, our research strongly indicates that 10% is only a waypoint toward effective ocean protection and governance, not the endpoint. Even the more ambitious target of ≥30% protection called for by the 2014 World Parks Congress (World Parks Congress 2014) may not be enough to meet all of the multiple objectives expected of MPA networks (e.g., Angulo-Valdés & Hatcher 2010), particularly if surrounding areas are not subject to good management (e.g., Micheli *et al.* 2004; Rodwell & Roberts 2004; White *et al.* 2010). However, improving management outside protected areas should ease the performance burden for MPAs and lower the eventual target coverage to be attained (e.g., Rodwell & Roberts 2004; White *et al.* 2010).

MPA coverages generated from any theoretical study inherently depend on the scenario(s) considered (e.g., species' life history characteristics, conservation objectives, MPA design characteristics, management outside the MPA(s), etc.) and most studies identify a range of coverages for protection rather than a specific fraction. In addition, it should be noted that none of the studies included within this analysis explicitly set out to address the question of how much of the sea should be protected globally but rather considered the implementation of MPAs within the scenario and/or case study area examined. Our findings do not therefore represent explicit recommendations for what global targets should be but rather offer perspective on political targets.

The evidence we examined showed that MPA coverages required to achieve objective(s) within individual studies varied substantially, largely due to the different scenarios considered (e.g., protection of rare species vs. optimizing a fishery for a highly mobile species). Indeed, the range of values reported by studies illustrates the theoretical potential for optimal management to either exclude MPAs entirely (e.g., Holland & Stokes 2006) or to restrict human activities to very small areas of the sea and protect the remainder (e.g., Tanner 2001). The former example focused on the application of an MPA in a well-managed (fished at or below maximum sustainable yield) and previously unexploited fishery, while the latter identified maximum harvest in a prawn trawl fishery with the majority of the fishing ground protected and a very small area heavily exploited. The level of MPA coverage required can vary considerably from one place, habitat type, or species to the next depending on their characteristics and the specific goals (e.g., representation of particular species or habitat or rebuilding overexploited fisheries, etc.) and management outside the MPA. Extreme values like 0% or 98% MPA coverage generally arose in studies considering single, narrow objectives. Given that MPA networks are always designed to achieve multiple objectives, with significant trade-offs between them, mean and median values of coverage will be more representative of those needed in practice than the extremes. Nonetheless, while there is strong consensus in the findings justifying global targets of the order of tens-of-percent MPA coverage, one should always consider specific circumstances at local scales.

The CBD target does not stipulate how much protection MPAs should have. Countries could therefore meet this target with MPAs that offer little protection from extractive or damaging activities. Estimates vary, but according to the authoritative MPAtlas (Marine Conservation Institute 2016) only just over 1% of the sea out of 2.18% in MPAs can be considered as highly protected.[1] While partially protected areas have been shown to provide some benefits to species' density and biomass (Sciberras *et al.* 2013), highly protected MPAs, also known as "marine reserves" or "no-take zones", have much greater benefits for habitats and species of conservation concern (Sciberras *et al.* 2013). Some MPA benefits may be achievable only with near complete protection (e.g., conservation of fragile habitats, or of highly vulnerable species) while others would likely require a greater coverage of partially protected MPAs to achieve the same outcomes. Highly protected MPAs also offer important contributions to fishery management goals (Vandeperre *et al.* 2011) and, if cooperatively designed and managed, may act to reduce conflict among stakeholders in multiple use areas (e.g., Mazor *et al.* 2014; Ruiz-Frau *et al.* 2015).

Some critics have argued that different design principles and MPA coverage are necessary for different environmental settings, or to meet different objectives such as biodiversity conservation versus fisheries enhancement (Hilborn *et al.* 2004). We found required MPA coverages of several tens of percent to be highly consistent across a diverse range of objectives (Figure 1) and in temperate versus tropical settings (Table S3). This convergence of results reveals the considerable opportunities for strategic designs to achieve many objectives simultaneously.

Percentage based targets have been criticized for several reasons. Some people consider them to be based on little scientific evidence or ecological knowledge or to imply guaranteed protection if targets are met regardless of enforcement and additional management measures for the matrix surrounding MPAs (Carwardine *et al.* 2009). In addition, the significant areas indicated by this study may be considered politically unachievable (Carwardine *et al.* 2009). However, while the 10% target is simpler politically, the evidence suggests it is highly unlikely to generate the benefits aspired to by the CBD. In the rush to fulfill targets, there is also concern that MPAs will be designated in areas with low biodiversity value or few human activities to increase social and political acceptability. Likewise there is the risk of creating networks of paper parks where management and enforcement is negligible if it exists at all. Both these outcomes would limit effectiveness. Having said that, establishing MPAs to protect intact environments in areas of limited human activity to prevent degradation before it occurs, such as seen in the recent creation of many very large, remote MPAs (Wilhelm *et al.* 2014), will make a highly valuable contribution to a global MPA network and is comparable to the wave of designation of large and intact terrestrial protected areas that occurred decades ago (Naughton-Treves *et al.* 2005; Cantú-Salazar & Gaston 2010).

All management strategies have drawbacks. However, establishing a global MPA target has many advantages which we, and others, believe outweigh such shortcomings: they are simple to convey, politically tractable, and explicitly incorporate the ecosystem approach (Carwardine *et al.* 2009); they help mobilize support for conservation and generate political will (Wood 2011); and, if designed appropriately, they provide measurable objectives and a clear purpose (Wood 2011). Based on our review, we conclude the UN 10% target is too low and that the 2014 World Parks Congress call for ≥30% of the sea in highly protected MPAs is strongly supported by existing evidence.

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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:

Figure S1. Schematic of review stages from searches.

Figure S2. Publication year of articles included in the review.

Figure S3. Average proportion of studies identifying the required coverage for protection to meet MPA objectives based on 144 studies.

Table S1. Search terms used to identify relevant articles, where ∗ denotes a wildcard to search for alternate endings.

Table S2. Included studies in review (median values reported to two significant figures [s.f.]).

Table S3. Summary statistics of the review (reported as integers).

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Endnote

1. Including currently proposed MPAs would result in this coverage increasing to 2.4% no-take MPAs out of 5.7% MPAs (Marine Conservation Institute 2016).

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LETTER

Reconciling Biodiversity Indicators to Guide Understanding and Action

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Abstract

Many metrics can be used to capture trends in biodiversity and, in turn, these metrics inform biodiversity indicators. Sampling biases, genuine differences between metrics, or both, can often cause indicators to appear to be in conflict. This lack of congruence confuses policy makers and the general public, hindering effective responses to the biodiversity crisis. We show how different and seemingly inconsistent metrics of biodiversity can, in fact, emerge from the same scenario of biodiversity change. We develop a simple, evidence-based narrative of biodiversity change and implement it in a simulation model. The model demonstrates how, for example, species richness can remain stable in a given landscape, whereas other measures (e.g. compositional similarity) can be in sharp decline. We suggest that linking biodiversity metrics in a simple model will support more robust indicator development, enable stronger predictions of biodiversity change, and provide policy-relevant advice at a range of scales.

Introduction

Concerns over the global loss of biodiversity and the degradation of ecosystem goods and services have led to international commitments aimed at preventing further declines. For example, the parties to the Convention on Biological Diversity (CBD) committed to the Strategic Plan for Biodiversity 2011–2020, supported by 20 Aichi Biodiversity Targets to be met by 2020, which calls for effective and urgent action during this decade to tackle biodiversity loss. These targets are echoed in the United Nations' newly approved Sustainable Development Goals (SDGs) and, in particular, their goals 14 and 15 concerning the conservation of seas and terrestrial ecosystems respectively. It is critical both to be able to measure progress against these targets and to identify the most effective policies and interventions for achieving them. However, there are a number of difficulties associated with both these needs. We highlight five of the most pressing.

First, biodiversity is a complex concept and no single indicator can effectively summarize its status or trend. Many different metrics of biodiversity are used for reporting trends although most are based upon the number of species or individuals present. Some aspects of biodiversity, such as phylogenetic and functional diversity, are rarely assessed despite their potential relevance (Diaz *et al.* 2013; Mace *et al.* 2014; Steffen *et al.* 2015). A framework for essential biodiversity variables (EBVs) (Pereira *et al.* 2013) is now gaining support and providing a basis for collaboration (see http://geobon.org/essential-biodiversity-variables/ebv-classes-2/) but is still far from being streamlined, with six classes of metrics and 22 categories of measurement (Pereira *et al.* 2013). Furthermore, being based on ecological principles, the EBVs may not easily link to decisions or policies designed to achieve the CBD targets (Jones *et al.* 2011; Nicholson *et al.* 2012) or even to the targets themselves (Tittensor *et al.* 2014). Even when considering a single target, several different metrics may be in use. For example, Aichi Target 12 calls for the prevention of extinctions and progress is currently assessed using three indicators: the Red List Index that measures change in the number of threatened species since the previous assessment, the Living Planet Index that assesses changes in abundance within populations of vertebrates since 1970, and the Wildlife Picture Index that uses modeled changes in species occupancy of birds and mammals in 16 sites since 2007. In addition to the differing metrics of change used by such indicators, different indicators may give different results because they sample different places or taxa, or because they calculate change from different baselines.

Second, most indicators of global biodiversity are extrapolated or modeled from local observations at a particular time and place. However, the processes of biodiversity change (e.g., migration and local extinction) interact and vary across scales of space and time, so that global trends are not a simple function of local or regional trends (Sax & Gaines 2003; Thomas 2013). This complicates the description of global trends (McGill *et al.* 2015) and confounds efforts to extrapolate and forecast future changes.

Third, there are substantial gaps in data and observations due to the accessibility, popularity, measurability, and even fundamental knowledge of different components of biodiversity. Observations sourced for the most widely used indicators are inevitably biased; generally toward recent decades, large-bodied and charismatic species, in terrestrial, temperate, economically-developed, and easily-accessible environments (Boakes *et al.* 2010; Hudson *et al.* 2014; Pimm *et al.* 2014; Geijzendorffer *et al.* 2015; Meyer *et al.* 2015; Newbold *et al.* 2015; Gonzalez *et al.* 2016). Certain areas of significant biodiversity, such as soils and oceans, especially involving invertebrate and microscopic organisms, are extremely poorly known and weakly sampled (Mora *et al.* 2011).

Fourth, the system within which biodiversity loss is observed is not well understood. Often, including for the Aichi targets, the drivers–pressure–state–impact–response (DPSIR) framework is used (Han *et al.* 2014; Marques *et al.* 2014). However, the framework linkages are assumed rather than evidence-based, and the metrics of biodiversity are rather weak proxies for global metrics, being based on available data but without evidence of causal associations or knowledge of the dynamic relationships involved. Developing linked indicator sets, based on established cause-effect and feedback relationships, has been recognized as important (Sparks *et al.* 2011), especially considering the different Aichi targets that are heterogeneous in intent and unlikely all to be achievable simultaneously (Perrings *et al.* 2011; Joppa *et al.* 2013; Di Marco *et al.* 2016).

Lastly, it has proven difficult to link biodiversity change into models of socioeconomic and environmental change, with the result that biodiversity is at best weakly involved in integrated assessment models (IAMs), and often only as a response metric (van Vuuren *et al.* 2006), rather than in the system dynamics (Harfoot *et al.* 2014a).

As a consequence of all these factors it is difficult to present a summary of biodiversity loss that is comprehensive and consistent. Recent studies indicating that there is no recent loss of local species richness or diversity (Vellend *et al.* 2013; Dornelas *et al.* 2014) have been challenged due to systematic biases in the data (Gonzalez *et al.* 2016). But other recent reports state the following suite of conclusions: a sixth global mass extinction is already underway (Ceballos *et al.* 2015), global species survival measured by the Red List Index could fall by about 0.2 by 2020 (Tittensor *et al.* 2014), species extinction rates are about 100 times background rates (Pimm *et al.* 2014), land-use pressures have reduced average local terrestrial species richness by about 14% (Newbold *et al.* 2015), vertebrate populations have declined by 52% (WWF 2014), terrestrial vertebrate populations have declined by about 25%, and invertebrate populations by about 45% (Dirzo *et al.* 2014). Can all these be true, and if so what explains the differences?

Here, we develop a simple narrative of global biodiversity change drawing upon current knowledge as well as experts' understandings of the system. We implement this in a stylized spatially-explicit simulation of a hypothetical region and show how commonly used biodiversity metrics might be expected to respond to anthropogenic impacts in human-modified landscapes over different spatial scales. We measure biodiversity indicators in the modeled system. We suggest that this approach, as well as being a useful heuristic device, has practical and applied value for refining global biodiversity metrics in order to 1) measure the most influential changes, 2) identify key points for intervention within the system, and 3) reconcile apparent conflicts between biodiversity indicators.

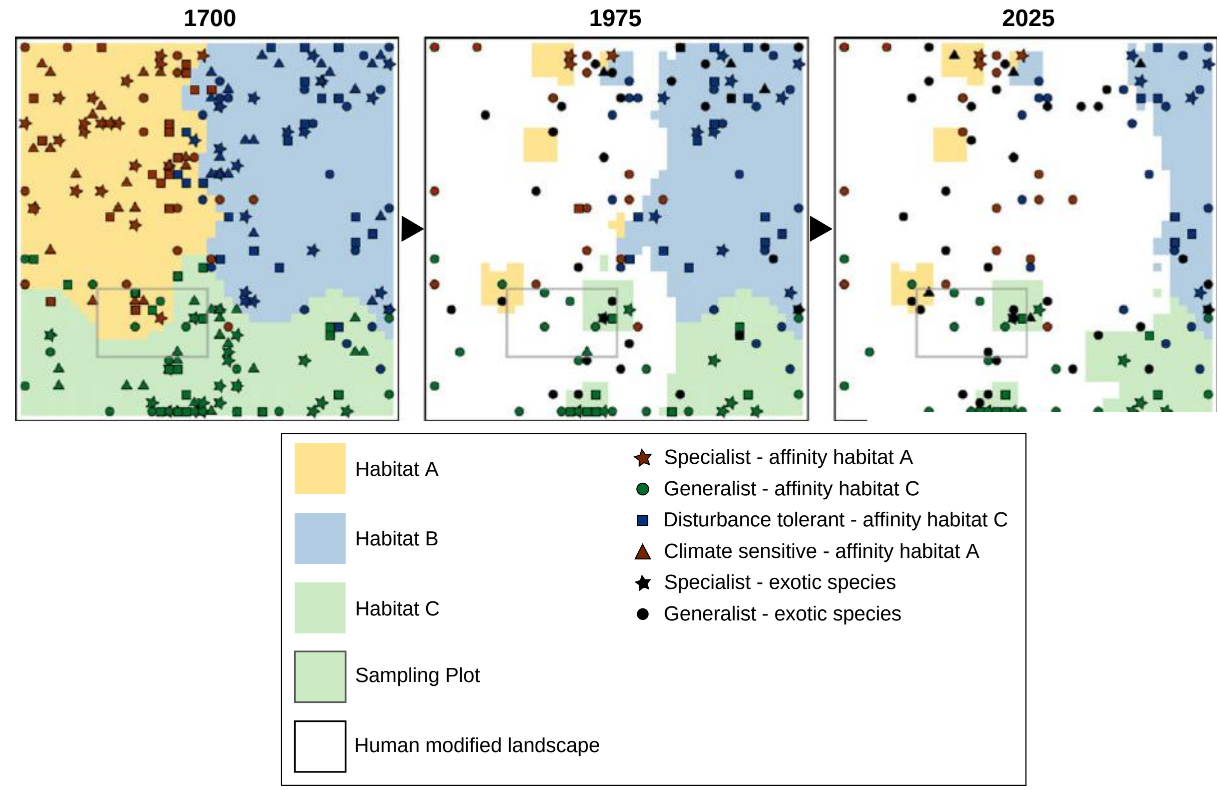

Figure 1 Regional landscape and community composition at three time points along a trajectory of human disturbance, starting from a pristine state in 1700 and running forward to 2100, as simulated by the idealized biodiversity response model. Pristine areas are indicated by coloured regions and human-dominated areas by white regions. A set of species exist in this landscape, each with an affinity for a particular habitat type (indicated by the color of the symbol) and with traits indicated by the symbol shape. In the first 275 years, 55% of the pristine habitat in the landscape was converted to human-modified matrix, while the ambient temperature increased by 2.75 °C. In the last 125 years, a further 25% of the pristine habitat is converted, while temperature increases by 1.25 °C.

Methods

Constructing the narrative

Our first step was to develop a picture of the current state of understanding of global biodiversity change and the most well-established causes and effects. We invited 26 biodiversity specialists, representing five countries, employed within academia, nongovernmental organizations, and private companies, to state what they considered to be the most important changes taking place to biodiversity (causes, states, and foci) based on their expert knowledge (see Section S1 for details). To assess the evidence base for these changes, we asked them to state also their level of confidence in each statement by indicating its comprehensiveness (taxonomic, geographic and across disciplines, and the extent of scientific consensus among experts). This information was used to construct the narrative that formed the basis for the simulation. The narrative guided our selection of anthropogenic impacts and provided the baseline of current state with which to test the simulation's results.

Exploring and visualizing the narrative through a simulation

The information gathered in the "Constructing the narrative" section was sufficiently complete to create a representation of a hypothetical terrestrial landscape. A stylized, spatially-explicit agent-based model of ecosystem change in response to anthropogenic impacts was constructed (Reconciling Biodiversity Indicators—http://reconciling-biodiversity-indicators.unep-wcmc.org) using the graphic language *Processing* (see Fry, B. & Reas, C. Processing URL https://processing.org/). The model simulates the time dynamics of individuals belonging to a set of 14 hypothetical species (see Table S1), living in a randomly generated landscape initially comprising three contiguous habitat types and a set of protected areas (Figure 1). The region is initialized in a pseudorandom pristine state. The extent and configuration of each habitat is generated randomly from defined spatial ranges, while the location of protected areas is drawn randomly and the configuration determined randomly within a defined range. The species agents are subsequently distributed randomly within habitat types. The region is then subjected to habitat conversion, climate change, and species invasions.

There are four native species types within the model: habitat specialists, habitat generalists, disturbance tolerants, and climate sensitives. Each species type has an affinity for its native habitat (Figure 1). Habitat specialists can exist only inside their native habitat, while other types can persist in other natural habitats or in the converted matrix if other criteria are met. Assuming that resources are more limiting in the matrix (Felton *et al.* 2003), only a limited density of conspecific generalists can be supported in the matrix, while disturbance-tolerant species can only exist within a threshold distance of their native habitat. There was no density dependence in native habitats. Climate-sensitive organisms can exist anywhere as long as the climatic conditions are within their tolerance range. Each natural habitat is initialized with 20 individuals of each species type. The model also incorporates two additional species types: exotic generalists and exotic climate-sensitive organisms. These are not present in the initial model landscape but are probabilistically introduced following anthropogenic change. The simulation moves forward in time intervals of 25 years, and at each step, habitat is converted following a random walk from a randomly chosen start location at a rate of 0.2% per year, approximately equivalent to the global rate of conversion of pristine habitat to human-dominated area documented in the HYDE reconstruction of land-use change (Goldewijk 2001) and observed in global land conversion data (Balmford *et al.* 2003). The ambient temperature of the region increases at a rate of 0.01 °C per year. Source code for the model can be obtained from https://github.com/mikeharfoot/Reconciling-Biodiversity-Indicators

The model was used to generate biodiversity metrics comparable to those most often used to characterize biodiversity (see above): species richness, population abundance, and extinction rates. Richness and abundance were measured over time at two different spatial scales: a local scale comparable to a single plot (which was sited randomly) and a regional scale (i.e., across the whole landscape) such as may be targeted for national, regional, or ecosystem-wide assessments. Extinction rate was only measured at the regional scale since it has little meaning at the plot scale. We also calculated a metric of compositional similarity through time—Bray Curtis similarity, which takes into account both species identities and their abundances (Bray & Curtis 1957)—to measure the overall similarity of the regional community at each time step to the initial state. A value of 1 indicates that the community has a composition of species in identical relative abundances to that of the initial state, and a value of 0 indicates that no initial species are present in the current community. The necessary data are seldom available to calculate compositional similarity for real ecosystems.

Full details of the model are provided in Section S2.

Results

The narrative

Most experts described change to global biodiversity in terms of loss of species (extinctions), loss of species abundance, and spatial changes due to invasions, loss of habitat, geographic range shifts, and homogenization processes (see Table 1). Experts were more confident when making statements at a global scale than at regional or local scales, and were more confident about the state of biodiversity than the causes of change, with particularly high uncertainty over how anthropogenic pressures interact and the consequences for ecosystems. Experts were more confident when making statements about vertebrates, and identified gaps in knowledge concerning invertebrates, some plants, and microbes (see Section S3 for details).

The key threats (habitat degradation and loss, climate change, and invasive species) were identified as affecting species differentially so that the simulation included species with differing sensitivities (see Methods section). Protected areas were identified as a key response and were therefore included within the simulation landscape.

The simulation results

Figure 2 shows how the four metrics (species richness (a), species abundance (b), extinction rate (c), and compositional similarity (d)) respond over a 400-year period. Regional richness increases in the early stages of the simulation due to the introduction of exotic species to the region while native specialists persist. Regional richness shows a clearer response than the plot-based metrics that are subject to sampling variation across the landscape. After around 1900, however, many habitat specialists and climate-sensitive species are extirpated leading to rapid declines in regional richness and increases in extinction rate. Plot-level richness declines monotonically throughout these simulations as species are lost from the sites faster than exotics establish. Both regional and plot-level abundances decline but asymptote once all sensitive species are filtered out. Because the original assemblage is known precisely in these simulations (as opposed to in real life), we can also track change in compositional similarity to the starting condition over time. Unlike other metrics, compositional similarity has declined sharply, linearly, and continuously.

Table 1 Summary of major findings from the survey of experts used to create the narrative. Details of the questions posed and answers given are provided in the Sections S1 and S3, respectively

Main generalizations	Further comments
There are widespread global losses in species abundance and range size	We are more certain of the status of vertebrates than other taxonomic groups All species, including common species, may be impacted Large-bodied mammal populations are rebounding from a very low baseline in North America and northern Europe. Large predators and all medium-sized animals are declining in Africa and other developing nations outside protected areas due to persecution and hunting Large-bodied mammals have declined (on average) in the past 50 years Freshwater species are faring worse than other groups, everywhere and including most taxa The marine environment is really suffering in nearshore parts of Africa due to intensive fishing.
Many species are threatened with extinction and the situation is not improving	Invertebrates are just as threatened as vertebrates Specialist species are worse off than generalist species
Local species richness is not declining	Locally, across sites at plot scale, there is no overall change in species richness over time for plant communities In time series, there is no overall loss of species richness within sites This may be temporary and due to extinction debt or introduced species
Local species richness is declining	Selected species are being removed from ecosystems
Homogenization is occurring –species communities are becoming more similar	Invasions of nonnative species are very significant in this process This can lead to losing diversity globally but not locally
Climate change is set to further impact biodiversity	Species ranges are moving consistent with climate change Climate change is already affecting species in the oceans and at high latitudes on land As climate change increases in scope and severity, it will affect susceptible species and those subject to other threats We do not understand how pressures from climate change will interact with other pressures such as hunting and land conversion
Invasive species pose a threat to native species	Currently, this is especially evident on islands and increasingly in continental areas Invasive species are greatly underreported in Africa and the tropics
The establishment of protected areas is preventing species loss in some places	The rate of habitat loss has increased over the past 50 years—this has been the primary driver of wildlife decline
Biodiversity has been detrimentally impacted by loss and degradation of habitat, human presence, and harvesting	Reduction of area of natural habitats causes "overcrowding" of habitat specialists, causing an extinction debt of unknown size and duration We need ecosystem-level analyses of how these pressures interact

Discussion

While the narrative is simplistic and the simulation stylized, we suggest that the coupling of these approaches is valuable in a number of different ways. First, our approach acts as a heuristic tool. The narrative of change derived from expert judgment can be encoded in a simulation model that can be used by biodiversity experts, policy makers, and general public to better understand how responses emerge. As a result, our approach provides a straightforward means to explain and even enhance the messages derived from the suite of biodiversity metrics, which may be confusing to policy makers and the general public, or even be interpreted as conflicting. For example, evidence that there is no local loss of diversity on average (Vellend *et al.* 2013; Dornelas *et al.* 2014) may seem to be inconsistent with evidence for overall loss of abundance (Dirzo *et al.* 2014) (though see Gonzalez *et al.* 2016). However, our simulation shows that short-term stability in species richness can be consistent with significant decreases in abundance (Figure 2). Similarly, greatly elevated global extinction rates (Barnosky *et al.* 2011; Pimm *et al.* 2014) are consistent with much lower levels of net loss—or even gain—in local species

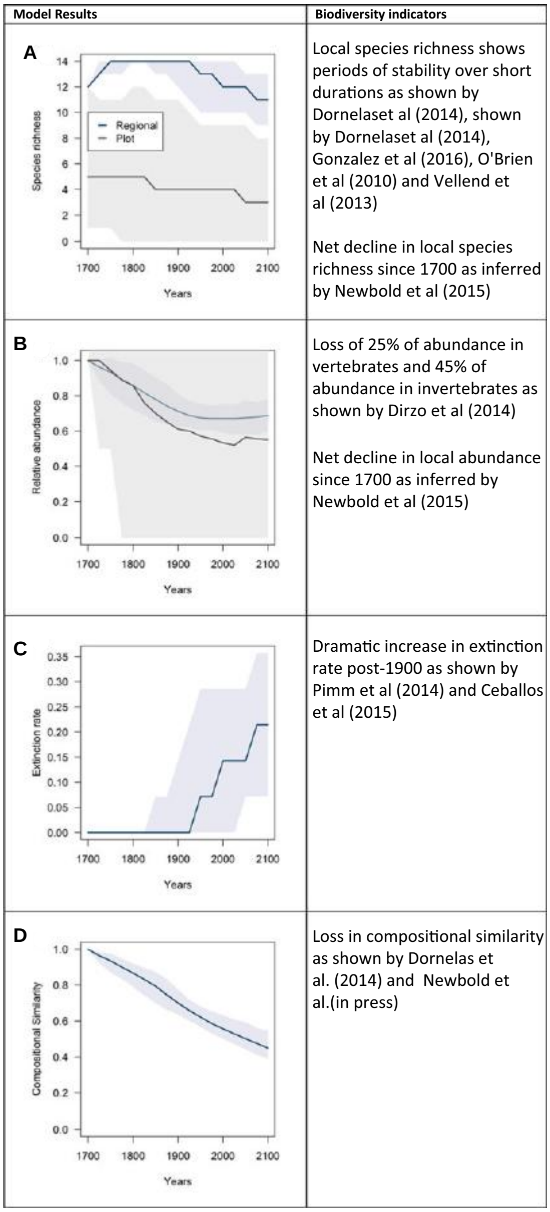

Figure 2 Comparison of time series of biodiversity metrics emerging from an ensemble of 125 simulations of the stylized biodiversity model with recent biodiversity indicators. Dark lines on plots indicate median responses and shaded regions show minimum and maximum ranges. (a) Species richness is a count of the number of species in the entire region (blue) or measured at the plot scale (black). (b) Abundance is the total number of individuals—irrespective of species—in the entire region (blue) or measured at the plot scale (black). (c) Extinction rate is the proportion of all species present at the start of the simulation that have been lost, recorded across the entire region. (d) Compositional similarity between the ecological community in the simulation at each time step and its initial state. The compositional similarity index has a value of 1 when the community composition is the same as the initial state and zero when none of the same species are present, and is based on 125 simulations with the line showing the median result and the shaded regions indicating maximum and minimum observed values.

richness, both of the net outcomes of local invasions and homogenization, and because of the longer term stabilization. The failure of commonly used diversity metrics, such as local species richness and abundance, (Figures 2a and b) to capture fully the rapid ongoing degradation in the composition of the ecological assemblage (Figure 2d) is a worrying feature for indicators that might be used as a basis for policy decisions. However, the ability to present a consensus view based upon evidence from a variety of indicators, capturing differing aspects of biodiversity, is advantageous when communicating with policy makers, and it is encouraging that we did not observe any metrics that diverged from the narrative.

Second, the simulation results demonstrate that, with recognition of cause and effect, it is possible to link biodiversity indicators dynamically. Though currently poorly understood, these relationships can be improved dynamically as we gain more ecological understanding. Future knowledge may support functional or even phylogenetic diversity metrics that could underpin the development of more efficient and informative indicators providing information better linked to decisions (Jones *et al.* 2011), for example, when comparing community changes resulting from two different patterns of habitat loss (Keil *et al.* 2015).

Third, a more comprehensive dynamic framework would permit more meaningful integration of biodiversity models into 1) decision-analysis tools, for example, to demonstrate the consequence of climate change affecting only the most sensitive species, or the conservation interventions that might best mitigate the impacts of a particular anthropogenic pressure on biodiversity, and 2) IAMs to demonstrate biodiversity feedback to socioeconomic futures. This would allow robust and evidence-based biodiversity goals to be produced. Until such integration takes place, it is hard to see how biodiversity can be mainstreamed into the development agenda.

There are many candidate models to support such developments, such as those that are being reviewed for the IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, see http://www.ipbes.net/) guide for scenario analysis and modeling of biodiversity and ecosystem services. Here, for illustration, we focus on the potential benefits of integration across two particular types of model that lie at opposite ends of the pattern- to process-based model continuum and differ in the characteristic units of ecological representation: taxonomic-versus functional-trait based. The PREDICTS model is a statistical model that currently focuses on land-use change, and allows various biodiversity metrics to be predicted based on a global compilation of studies of local ecological communities (Hudson *et al.* 2014; Newbold *et al.* 2015). Already

PREDICTS can be used to illustrate how different metrics of local biodiversity may behave under different extrinsic forces at different scales (Newbold *et al.* in press; Newbold *et al.* 2015). However, PREDICTS contains no ecological processes, it is entirely empirical based on a large number of observations, and currently cannot be used to predict changes in regional or global biodiversity. Spatial and temporal dynamics relevant to real ecological systems are most directly incorporated through process-based models, such as the Madingley model (Harfoot *et al.* 2014b) or the Ecosim model (Walters *et al.* 2002); such models have the additional advantage that they can report directly on some aspects of ecosystem function and services, such as biomass production or aspects of ecosystem dynamics such as stability or resilience, but do not currently report on the species-based biodiversity variables that are used to calculate currently mainstream indicators.

By coordinating these different modeling approaches, it will be possible to substantially strengthen simulations that establish cause and effect through the DPSIR framework. This could refine the process of indicator production, from focusing data collection toward key metrics, to defining and implementing indicators that more comprehensively describe the changing state of the system. As a whole, this development would allow the conservation community to more strategically and effectively evaluate how the biodiversity and ecosystem targets of the SDGs can be met simultaneously with those for socioeconomic development.

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Author contributions

The project was designed by GM, DP, and AP; MH and DP developed the simulation; and SH and GM developed the narrative. GM, SH, and MH led the writing of the article with contributions from all authors.

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LETTER

Persistent Disparities between Recent Rates of Habitat Conversion and Protection and Implications for Future Global Conservation Targets

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Abstract

Anthropogenic conversion of natural habitats is the greatest threat to biodiversity and one of the primary reasons for establishing protected areas (PAs). Here, we show that PA establishment outpaced habitat conversion between 1993 and 2009 across all biomes and the majority ($n = 567$, 71.4%) of ecoregions globally. However, high historic rates of conversion meant that 447 (56.2%) ecoregions still exhibit a high ratio of conversion to protection, and of these, 127 (15.9%) experienced further increases in this ratio between 1993 and 2009. We identify 41 "crisis ecoregions" in 45 countries where recent habitat conversion is severe and PA coverage remains extremely low. While the recent growth in PAs is a notable conservation achievement, international conventions and associated finance mechanisms should prioritize areas where habitat is being lost rapidly relative to protection, such as the crisis ecoregions identified here.

Introduction

Humans have reshaped patterns and processes in ecosystems across the terrestrial biosphere, both intentionally and unintentionally, for millennia (Ellis *et al.* 2010; Dirzo *et al.* 2014). This reshaping has accelerated over time (Steffen *et al.* 2015), with a human footprint now obvious in most parts of the terrestrial realm (Sanderson *et al.* 2002). One of the more severe impacts of this anthropogenic transformation of the biosphere is the loss of natural habitats. Three decades of conservation science have extensively documented the impacts of habitat transformation on genetic diversity, species survival, and ecosystem function (Fischer & Lindenmayer 2007). In many cases, these impacts have proved insurmountable, making habitat loss the greatest driver of postindustrial species endangerment and extinctions (Venter *et al.* 2006; Hoffmann *et al.* 2010).

Protected areas (PAs) spearhead global efforts to conserve nature, and when properly managed they are particularly effective for combating habitat loss (Bruner *et al.* 2001; Gaston *et al.* 2008). Since 1992, the Convention on Biological Diversity (CBD) has catalyzed a global proliferation of PAs, including through a commitment in 2010 to protect 17% of terrestrial and 10% of marine environments globally by 2020, especially "areas of particular importance for biodiversity and ecosystem services" through "ecologically representative" PA systems or other "area-based conservation measures" (CBD 2011). Some nations have set national PA commitments even greater than the global target (Butchart *et al.* 2015), and there has been a pronounced expansion of the global PA estate over the past two decades (Juffe-Bignoli *et al.* 2014).

Yet, many nations are also expanding their use of natural resources as a primary means of achieving economic development targets (Brunnschweiler 2008). Consequently, rates of anthropogenic habitat conversion are climbing alongside PA growth (Hansen *et al.* 2013). This situation has led to the establishment of a dedicated CBD Aichi target (Target 5) under which nations committed to halve and, where feasible, bring close to zero the rate of habitats loss (CBD 2011). To be effective at slowing habitat conversion, it is widely recognized that PAs need to be placed in areas at risk of loss in the absence of protection (Pressey *et al.* 2015; Visconti *et al.* 2015). However, despite increasing recognition by nations of the importance of PAs in abating habitat loss (Watson *et al.* 2014), there has been no assessment of which areas have experienced the greatest rates of recent anthropogenic habitat conversion, nor whether the recent growth in the PA estate is located in high conversion areas. This is critical baseline information that will not only allow nations to report on their progress toward achieving CBD targets (CBD 2011), but also inform the priorities of financial mechanisms (e.g., the Global Environment Facility) that fund PA establishment (Watson *et al.* 2016).

Here, we examine the extent of habitat conversion across the world's biomes and ecoregions in 1993 and 2009 using a novel and temporally explicit cumulative threat map (Venter *et al.* 2016). We compare the relationship between habitat conversion and PA establishment during this period and identify those ecoregions (and the nations that contain them) that need urgent attention if the 2020 CBD's strategic goal to "improve the status of biodiversity by safeguarding ecosystems, species and genetic diversity" is to be achieved.

Methods

Biome and ecoregion classification

Biomes and, at a finer spatial scale, ecoregions, represent relevant environmentally and ecologically distinct spatial units at the global scale and are used by international funding institutions and conservation organizations to guide broad-scale global conservation investments and action (Olson & Dinerstein 2002; Funk & Fa 2010). Following previous global analyses (Hoekstra *et al.* 2005; Segan *et al.* 2016), we used the global biomes ($n = 14$) and ecoregions ($n = 825$) identified by Olson *et al.* (2001) as the basis for our analysis.

Measures of habitat modification

We used the revised Human Footprint map (Venter *et al.* 2016) to measure habitat conversion. The revision takes advantage of recently available datasets to provide a cumulative score of eight in-situ anthropogenic pressures. These pressures include urban centers, intensive agriculture, pasture lands, human population density, night-time lights, roads, railways, and navigable waterways. Following Sanderson *et al.* (2002), individual pressures were placed on a 0–10 scale and then summed to create the cumulative measure of the Human Footprint. We note that the presence of a human pressure and its actual impact on biodiversity is assumed, but these pressures are considered among the greatest threats to biodiversity (Maxwell *et al.* 2016), and previous analyses have shown the Human Footprint is an important predictor of extinction risk (Di Marco *et al.* 2013).

For our purposes, a threshold criterion for habitat conversion was set at a Human Footprint value of 4 or greater. This value equates to a human pressure score equal to pasture lands, representing a reasonable approximation of when anthropogenic land conversion has occurred to an extent that the land can be considered human dominated and can no longer be considered "natural." Previous analyses show that this threshold is where species are far more likely to be threatened by habitat loss (Di Marco *et al.* 2013).

We note that there is no universal threshold for habitat conversion, because there is no single level at which the environmental values we associate with habitat "intactness" are suddenly lost (Tulloch *et al.* 2016). We therefore explore the sensitivity of our results using different thresholds (see Supplementary Materials). Here, we present only the results using the threshold of "4 or greater," as the sensitivity analysis revealed only minor variation in the results.

Protected areas

We estimated temporal trends in PA coverage using data on the year of PA establishment recorded in the 2014 version of the World Database on Protected Areas (UNEP-WCMC 2014). As this was unknown for 15% of the area of the terrestrial PA estate, we followed Butchart *et al.* (2012) and assigned a year by randomly selecting a year (with replacement) from all PAs within the same country with a known date of establishment. For countries with fewer than five PAs with known year of establishment, a year was randomly selected from all terrestrial PAs with a known date of establishment. The random assignment was repeated 1,000 times to identify the median and 95% confidence intervals.

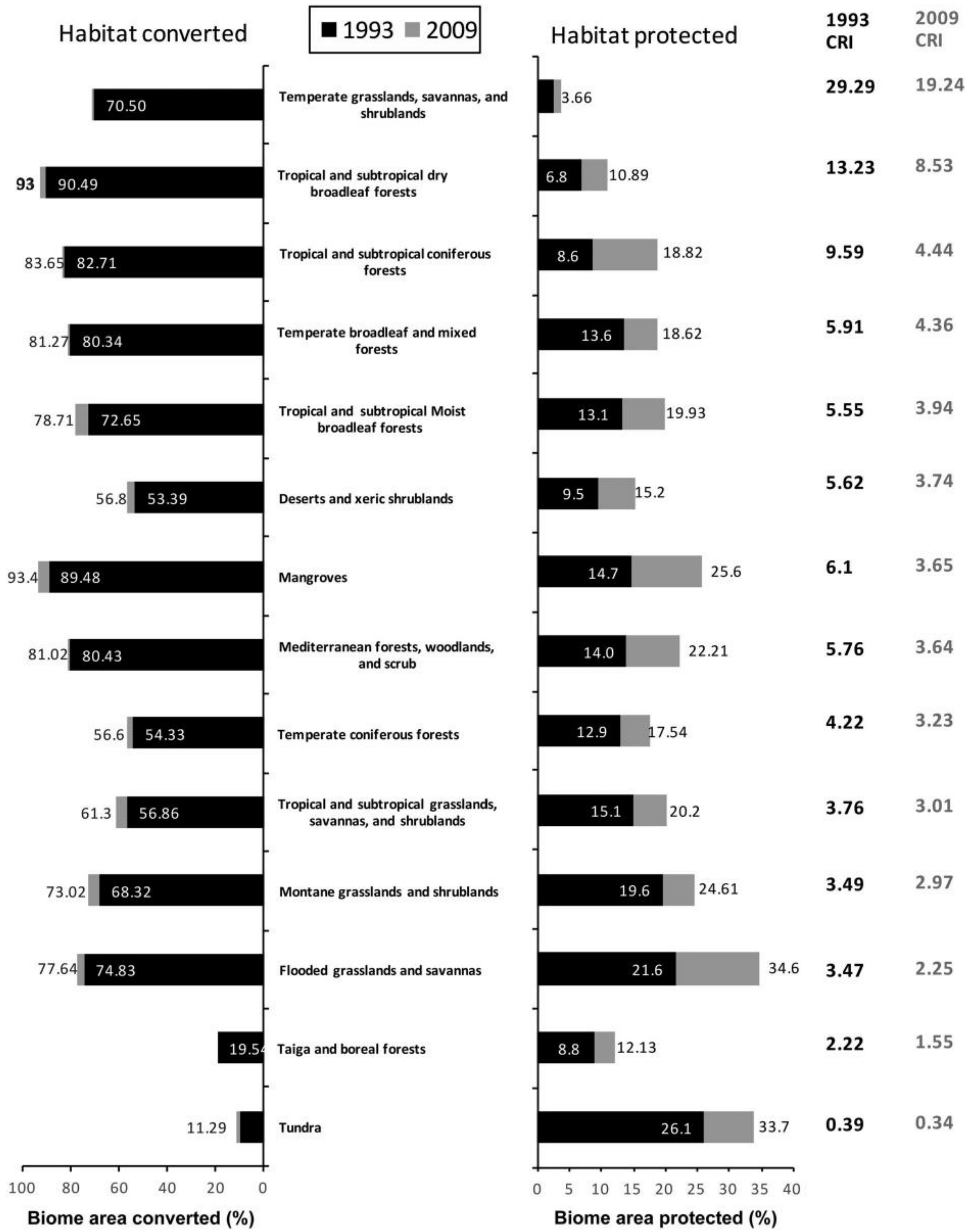

Figure 1 Percentage of habitat conversion and PA coverage among the world's 14 terrestrial biomes in 1993 (black bars) and 2009 (gray bars). The baseline assumption is full habitat extent across all biomes. Numbers inside the black bars show the value as of 1993, while numbers at the end of the bars show the value as of 2009. Biomes are ordered by their conservation risk index (CRI) for 2009 (which was calculated as the ratio of percentage area converted to percentage area covered by PAs, following Hoekstra *et al.* 2005).

We followed the methods of previous global assessments (Rodrigues *et al.* 2004; Jenkins & Joppa 2009; Venter *et al.* 2014) and included only PAs with a national designation, excluding areas protected only by international agreements and all PAs with a status other than "designated." For PAs that met the above criteria, but for which only central coordinates and total area were available ($n = 15{,}404$), a circular buffer of the appropriate area was generated around the central coordinates to depict the spatial extent of the PA. PAs that lacked polygonal representation or a specified areal extent were excluded from the analysis ($n = 7{,}311$).

Analysis of spatial data

All spatial data were processed in vector format using ESRI ArcGIS v10 and Mollweide equal-area projection. For all terrestrial coverage statistics, we followed established practice (Juffe-Bignoli *et al.* 2014; Venter *et al.* 2014) by excluding terrestrial Antarctic ecoregions, "Rock and Ice" and "Lakes." We also excluded ecoregions that had an area < 5,000 km^2, because of discrepancies in spatially referenced information across datasets over small areas, which left 794 ecoregions out of a possible 825.

Assessing relationship between habitat conversion and protection

Habitat conversion rates over time are related both to the availability of unconverted land and to the rates of land protection. We explored the role of these two factors by building a generalized linear model in which conversion rates were predicted as a function of "original proportion of converted land" and "current proportion of Pas." We also calculated the habitat conversion-to-protection ratio between percentage area converted and percentage area covered by PAs (following Hoekstra *et al.* 2005) for 1993 and 2009. We call this ratio the "conversion risk index" (CRI) because it relates to the risk of conversion of remaining intact habitat for ecoregions and biomes (Hoekstra *et al.* 2005). We categorized the threat risk of ecoregions based on their CRI using the following criteria. First, any ecoregion that met the 17% PA target outlined in the 2010 CBD strategic plan was considered not at risk, albeit only in the sense that it meets the current globally accepted target for PA extent (CBD 2011). Second, for all those ecoregions with <17% PA coverage, we identified "at-risk" ecoregions: *moderate,* those ecoregions having CRI > 2 or total areal habitat conversion > 20%; *high,* those with CRI >10 or total areal habitat conversion >40%; and *very high,* those with CRI >25 or total areal habitat conversion > 50%. Finally, of the ecoregions at very high risk in 2009, a further subset of "*crisis* ecoregions" that have also experienced high rates (>10%) of recent habitat conversion since 1993 was identified. We labeled all ecoregions that do not meet any of these "at-risk" categories as "*low risk,*" recognizing that biodiversity in these areas is of course not free from threat and that PAs are just one form of conservation response.

Results

Habitat loss across biomes and ecoregions

Globally, over half (51.4%) of the world's land area was converted to human-dominated land-uses in 2009, of which 9.3% (4,406,769 km^2) was converted between 1993 and 2009. Two biomes (mangroves and tropical and subtropical broadleaf forests) were > 90% converted by 2009 (Figure 1). During the period 1993–2009, all biomes experienced some degree of habitat conversion, with *tropical and subtropical moist broadleaf forests*, *montane grasslands and shrublands*, *tropical and subtropical grasslands, savannas and shrublands*, and *mangroves* experiencing the most change (Figure 1). Within biomes, there was considerable variation in habitat conversion across ecoregions. The extent of habitat conversion in 2009 ranged from <1% in 13 ecoregions (1.6%) to > 75% in 426 ecoregions (53.7%) (Figure 2a). Our assessment of habitat conversion since 1993 shows that 91 ecoregions (11.6%) underwent >10% habitat loss during the 16-year period, but the majority (52.5%) underwent relatively small losses (<1%).

Rates of PA growth across biomes and ecoregions

The terrestrial PA network almost doubled between 1993 and 2009, growing by 7,004,035 km^2 (9.0% of land) to cover 18,874,488 km^2 (14.2%). This has led to substantial increases in protection levels at the biome scale, with 10 of the 14 biomes achieving >17% protection in 2009 (Figure 1). Two biomes (*temperate grasslands, savannahs and shrublands*, and *tropical and subtropical dry broadleaf forests*) stand out as still having relatively low levels of protection (Figure 1). Habitat protection exceeded > 17% coverage in 314 (39.5%) ecoregions in 2009, a large increase with respect to 1993 ($n = 184$, 23.2%; Figure 2).

PAs are not necessarily immune to habitat conversion (or indeed other important threatening processes such as overharvesting, invasive species, and climate change); however, we found on average, very little habitat conversion has occurred within PAs during the study period, with an increase in average Human Footprint values of just 0.15.

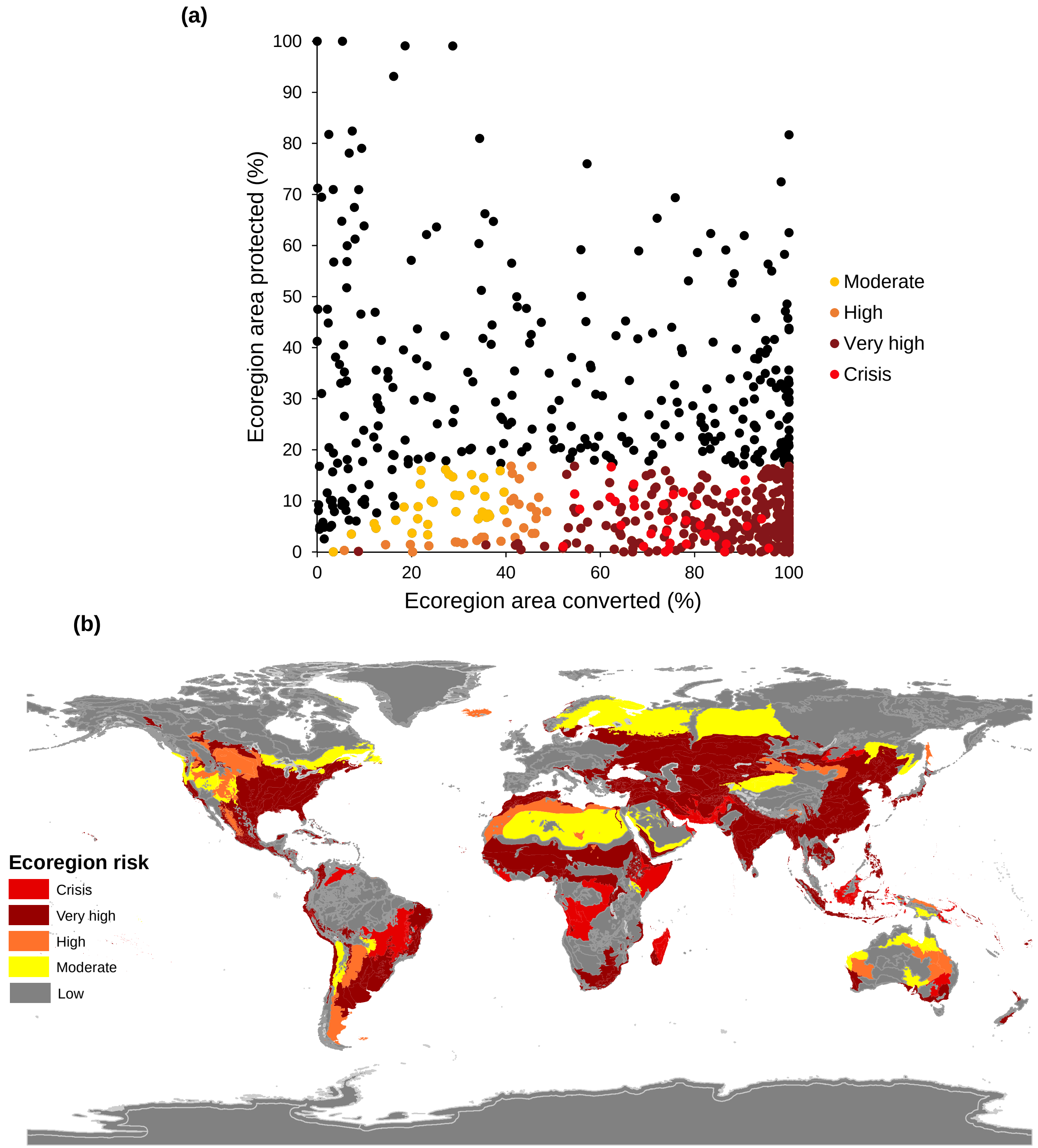

Figure 2 The relationship between degree of habitat conversion and PA coverage across the world's terrestrial ecoregions in 2009 as a scatterplot (a) and their locations (b). Ecoregions with > 50% habitat conversion or conservation risk index (CRI) > 25, and with >10% change in habitat conversion from 1993 to 2009, are classified as crisis ecoregions (red); ecoregions with > 50% habitat conversion or CRI > 25 are classified as very highly at risk (maroon); ecoregions with > 40% conversion or CRI > 10 are classified as highly at risk (orange); and those ecoregions with > 20% conversion or CRI > 2 are classified as moderately at risk (yellow). CRI for each ecoregion was calculated as the ratio of % area converted to % area covered by PAs.

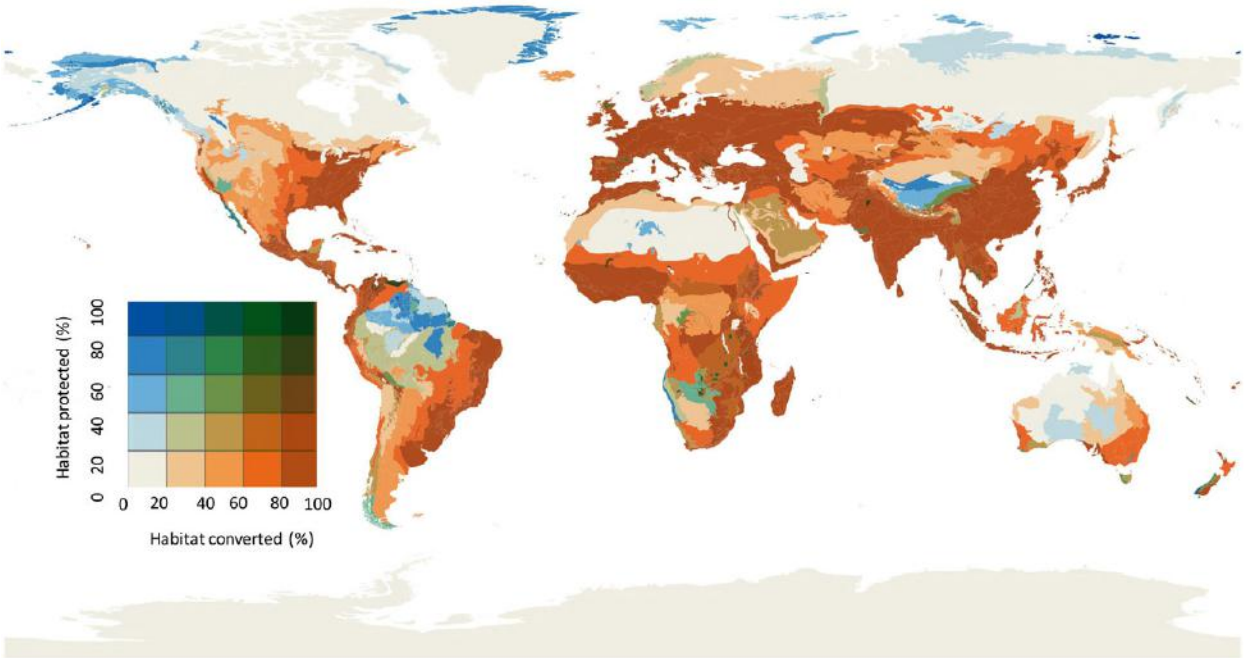

Figure 3 The spatial relationship between degree of habitat conversion versus PA coverage in terrestrial ecoregions in 2009.

Relationship between habitat loss and protection

The vast majority of ecoregions have very high levels of habitat conversion compared to their overall areal protection (Figures 1 and 3). These highly converted and poorly protected ecoregions occur across all continents and dominate Europe, south and south-east Asia, western South and North America, western Africa, and Madagascar. The small number of ecoregions that contain high levels of protection and low levels of conversion are primarily located in the Arctic, the northern Amazon, North Asia, and central Australia (Figure 3).

Encouragingly, all biomes had a lower CRI in 2009, indicating that the rate of new protection exceeded the rate of habitat conversion at the biome level during the period (Figure 1). At a finer scale, 567 (71.4%) ecoregions also showed a lower CRI in 2009 than in 1993 (Figure 4). On the other hand, 203 (25.3%) ecoregions showed a higher CRI in 2009 than in 1993, indicating that habitat conversion outpaced protection. These latter ecoregions occurred in all biomes and on all continents, but were concentrated in eastern and western Africa, north-western Madagascar, northern and southern South America, north Asia, Indonesia, Papua New Guinea, and in many parts of Australia, United States, and New Zealand (Figure 4).

Conversion over the 16-year time period was negatively and significantly correlated with the extent of converted land in 1993 ($r = -0.06$, $P < 0.05$), but there was no significant relationship between the extent of PA coverage in 1993 and in 2009 ($r = -0.02$, $P = 0.11$).

At-risk ecoregions

We identified 447 "at-risk" ecoregions based on their CRI and high levels of conversion in 2009, of which 341 were "very high" (Figure 2). These very high-risk ecoregions were found on every continent and biome, and were represented in 67 nations (Figure 2b). In addition, 41 *crisis* ecoregions were identified, as they had experienced >10% conversion between 1993 and 2009 (Figure 2b). These crisis ecoregions are located in 45 nations, but are especially concentrated in Indonesia (8), Papua New Guinea (6), Madagascar (5), Angola, DR Congo, and Pakistan (4 each).

While the majority of ecoregions remained in the same risk category in both 1993 and 2009, 79 ecoregions were downgraded from either very high or high risk to low risk (Table 1). The ecoregions that moved from imperiled categories to low-risk categories were generally located in Europe and Africa (Table 1; Figure S2). Of the "at-risk" ecoregions identified in 2009, 121 (27%) had a CRI ratio that worsened from 1993 to 2009 (Figure S1; Table S1),

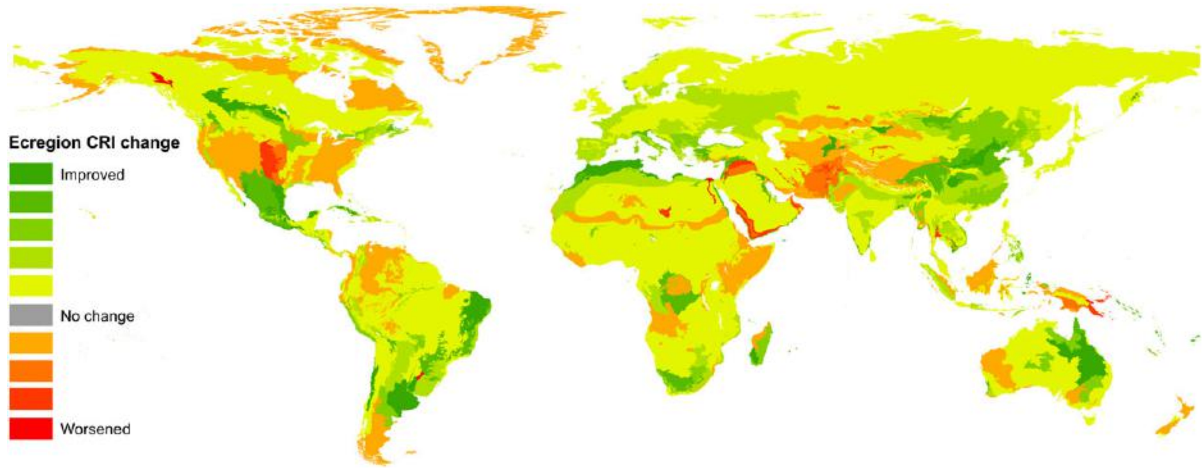

Figure 4 The spatial patterns of the changing ratios between habitat conversion and protected area coverage in 1993 and 2009 across the world's ecoregions. Ecoregions that experienced worsening ratios are shown in red and those in which the ratio improved (i.e., slower habitat conversion and/or greater PA expansion) are shown in green. Those in which there was zero change are shown in gray.

Table 1 Ecoregion status in 1993 and 2009, based on ratios between habitat conversion and PA coverage, and the degree of total habitat conversion (see methods for how ecoregions were categorized)

		2009 Risk level					
		Low	Moderate	High	Very high	Crisis	Total
1993 Risk level	Low	246	1	0	0	0	247
	Moderate	22	34	8	0	1	65
	High	9	1	18	4	6	38
	Very High	70	0	3	337	34	444
	Total	347	36	29	341	41	

Note: There were a total of 688 "at-risk ecoregions" (those not categorized as low risk) in 1993, and 447 in 2009.

of which 66 (54.5%) were considered very highly at risk and 22 (18%) were identified as crisis ecoregions.

Discussion

The past two decades have seen alarming rates of global habitat conversion (Bianchi & Haig 2013; Parr *et al.* 2014). This is particularly concerning considering that habitat loss is the largest driver of biodiversity loss globally (Hoffmann *et al.* 2010). Our results reveal a significant continued disparity between the overall amount of habitat converted versus the amount protected at both the biome and ecoregional scales over the past two decades. In 2005, Hoekstra and colleagues argued that a global habitat crisis was upon us based on the ratio of habitat lost versus protected (Hoekstra *et al.* 2005). While direct comparison between the studies is limited by differences in the data used, our temporal analyses support the argument presented by Hoekstra *et al.* (2005) and show that the crisis is not yet averted. The vast majority of terrestrial ecoregions still have dangerously high levels of habitat conversion relative to their levels of protection (Figures 1 and 3).

Encouragingly, we discovered that recent increases in protection are substantially outpacing rates of habitat conversion over the past two decades in all biomes and in >70% of ecoregions (Figures 1 and 4). This has led to a decreasing number of "at-risk" ecoregions between the two time periods, down 35% from 569 in 1993 to 431 in 2009 (Table 1). These results support studies reporting recent positive progress toward achieving a more representative PA system by at least some nations (Juffe-Bignoli *et al.* 2014; Di Marco *et al.* 2015). However, we also found that the availability of unconverted land played a major role in predicting habitat conversion rates when compared with PA extent over the time period. If this trend continues, those ecoregions with large proportions of remaining habitat are more likely to suffer future high conversion rates. This result speaks to the need for an expansion of PAs in ecoregions with relatively high availability of natural habitats, even if they are currently undergoing low rates of conversion.

While some ecoregions have shown recent improvements in PA coverage relative to habitat conversion, the fact that the majority of all ecoregions are still considered "at risk" owing to high habitat conversion relative to protection highlights the scale of the issue. Nearly, 30% ($n = 127$) of ecoregions that were "at-risk" in 1993 experienced a further worsening in their ratio of habitat conversion to PA coverage. Of these, 69 were considered at very high risk in 2009 and two were classified as crisis ecoregions. Clearly, strategic protection is urgently needed in these highly converted and underprotected ecoregions, especially those we classify as "very high" risk and "crisis" (Figure 2b). Achieving this protection will be complicated by the fact that many ecoregions, which are defined by biophysical characteristics, cross international boundaries, and the fact that there can be considerable spatial variation within ecoregions in habitat conversion rates. We identify 45 nations that contain all the crisis ecoregions and 67 nations that contain very high-risk ecoregions; coordinated implementation of new PAs across these countries is needed. To avert further biodiversity losses, global and regional PA finance mechanisms should be directed toward these nations as a priority, to catalyze PA establishment where it is needed most (Pressey *et al.* 2015; Visconti *et al.* 2015).

Ecoregions represent biophysically and climatically distinct units, and are often used in assessments of the representativeness of PAs for biodiversity targets (Jenkins & Joppa 2009). However, a focus on ecoregions may hide nuanced but important conservation implications of habitat clearance. In particular, as ecoregions vary in their size across six orders of magnitude, even small percentage conversion rates in large ecoregions, such as Africa's Sahelian Acacia Savanna or the Brazilian Cerrado, can have major implications for species loss and the disruption of ecosystem services. Species loss can occur in areas where there have been only relatively small amounts of habitat loss (He & Hubbell 2011), and this can have significant impacts on important ecosystem processes, such as net primary production (Cardinale *et al.* 2012). Significant scope exists for subsequent analyses aimed at quantifying the biodiversity and ecosystem service implications of the habitat conversion mapped in this study.

When targeting future protection, we urge that nations move beyond simply improving ecological representation, and attempt to capture those specific sites and locations that are important for imperiled biodiversity and at high risk of future clearance (Butchart *et al.* 2012; Venter *et al.* 2014). This will not only necessitate nuanced planning techniques (Groves & Game 2015), but also a substantial change in direction in how the global community next sets PA targets in international conventions.

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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:

Main document. Sensitivity analysis around habitat conversion thresholds

Figure S1. The location of ecoregions that have improved habitat conversion/protection ratios from 1993 to 2009, but are still categorized as at risk ecoregions.

Figure S2. The location of the ecoregions that changed their "at risk" status from 1993 to 2009, based on changes in habitat conversion to protection ratios, and the level of habitat conversion.

Table S1. The number of "at-risk" ecoregions that have decreased habitat conversion/protection ratios from 1993 to 2009.

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LETTER

Equitable Representation of Ecoregions is Slowly Improving Despite Strategic Planning Shortfalls

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Abstract

Representing all ecosystem types in protected areas (PAs) is central to international conservation agreements (i.e., Aichi Target 11) and ensuring the persistence of biodiversity. In response to these agreements, we have seen rapid growth of PA networks, but we do not know how this affects ecosystem representation. We explored this question by investigating drivers and trends of representation during periods of rapid land acquisition using the protection equality metric. We found that 90.9% of the studied countries have improved protection equality through time. Periods of rapid area expansion resulted in greater increases in protection equality, particularly through multiple, smaller PAs as opposed to fewer, larger PAs. However, observed increases may not be due to strategic planning, as protection equality from random PA allocation was statistically similar to observed values within six country-level simulations. Future international agreements should hold countries accountable to meeting multiple objectives and prioritize conservation outcomes over individual targets.

Introduction

Protected areas (PAs) have experienced marked expansion in recent decades and remain the primary focus of global conservation efforts (Chape *et al.* 2008; Watson *et al.* 2014). For example, the Convention on Biological Diversity (CBD) Aichi Target 11 requires signatory countries to protect 17% of terrestrial environments in effectively and equitably managed, *ecologically representative* and well-connected systems by 2020 (Secretariat of the CBD 2010). For most countries, only the terrestrial percent coverage target is projected to be achieved by the current deadline (Tittensor *et al.* 2014), while the other targets lack definition and transparent, comparable metrics. The continued loss of habitats and species despite over 32.8 million km^2 of conservation areas (Deguignet *et al.* 2014; WWF Living Planet Report 2014) questions our true progress in meeting conservation objectives (McDonald-Madden *et al.* 2009) and the role of land accumulation alone in conserving biodiversity (Ferraro & Pattanayak 2006). Better performance metrics are needed to shift the focus of PA expansion from the quantity of area protected to the quality of that PA system (Barnes 2015; Watson *et al.* 2015).

Spatial conservation planning principles prescribe that well-designed, effective PA networks ensure the inclusion of each biodiversity feature of interest (comprehensiveness), as well as the variation within each feature (representativeness) (Margules & Pressey 2000; Possingham *et al.* 2006), which are often referred to together as "representation." Historically, PA selection was not systematic, leaving many habitats and species underrepresented (Rodrigues *et al.* 2004; Watson *et al.* 2014; Butchart *et al.* 2015). Recently, Barr *et al.* (2011) introduced one of the first metrics to evaluate ecological representation called protection equality (PE). Moving beyond uniform targets and percent-based measures, PE uses a modified version of the Gini coefficient to quantify the difference between a perfectly equitable distribution and the actual distribution of a biodiversity feature within a

PA network (Barr *et al.* 2011). A value of 1 signifies perfect equality in protection, while 0 signifies complete inequality.

The near exponential increase in the global PA network is well documented (McDonald & Boucher 2011; Watson *et al.* 2014; Butchart *et al.* 2015), including periods of substantial growth. Radeloff *et al.* (2013) identified "hot moments in conservation," where countries established more than 33% of their total area protected in a single year, which have played a major role in shaping PA networks. Large land acquisitions for conservation (i.e., "hot moments" and/or "green grabbing" [Fairhead *et al.* 2012]) may become more prevalent as countries race to meet percent coverage targets (Blomley *et al.* 2013). As representation is cited as such an important component of effective PA systems (Margules & Pressey 2000), it is critical to identify how rapid PA expansion impacts ecological representation at a global scale to inform future conservation strategies and achieve greater biodiversity outcomes.

Here, we provide the first explicit test of trade-offs between PA expansion and equality of representation. We aimed to determine whether PE has increased over the past 60 years (1954–2013) and whether large land acquisitions have positively or negatively impacted representation. We then tested whether observed patterns could be attributed to deliberate action (i.e., systematic planning) or whether they were an inevitable consequence of PA expansion by benchmarking observed PE within six countries (Australia, Brazil, Canada, Indonesia, Mongolia, and Peru) against optimal and random protection scenarios in the last two decades. Finally, we investigated the impact of country-level economic and social factors, as well as differences in PA implementation strategy, on annual change in PE.

Methods

Data

We used the World Database on Protected Areas (WDPAs) to extract information on terrestrial PAs of IUCN categories I–IV (IUCN & UNEP-WCMC 2015) and terrestrial ecoregions developed by the World Wildlife Fund to represent global biodiversity features (Olson *et al.* 2001; World) (see Supplementary Appendix S1). Countries that had at least 70% of PAs with delineated boundary and establishment year data, five ecoregions, and protected at least 1% of one ecoregion were selected for further analysis. Total ecoregion area (km^2) and total area protected (km^2) of each ecoregion within each country were calculated to assess PE, which was calculated annually as in Barr *et al.* (2011) (see Supplementary Appendix S2).

Patterns of PE through time

We assessed patterns of PE from 1954 to 2013 and within six 10-year increments (e.g., 1954–1963, 1964–1973, etc.), capturing the major period of PA expansion (Watson *et al.* 2014). We used Mann–Kendall nonparametric trend tests to determine trends in PE for each time period. Data were pre-whitened to account for potential temporal autocorrelation (Kulkarni & von Storch 1995). The Mann–Kendall function of the Kendall package (McLeod 2011) in the software R v. 3.2.2 (R Core Team 2016) was used to calculate Kendall's Tau. Trends were calculated from the date of the first established PA in each country (i.e., PE > 0).

Spearman's rank correlations were used to assess the impact of change in area protected (area protected in a given year/total country area) on change in PE (difference in PE from one year to the next) in each decade. Years when there was no change in area, by definition, had no change in PE and were excluded from the analysis. To determine the overall effects between countries, we also tested the correlation between total area protected and total PE as of 2013.

Drivers of change in PE: Inevitable or deliberate?

To understand whether changes in PE are a result of better planning or could be achieved randomly, we compared observed PE values against PE from random and optimal protection scenarios within six countries in the last two decades. We chose Australia, Brazil, Canada, Indonesia, Mongolia, and Peru because they all protected substantial amounts of area (>27,500 km^2) in both decades, which represent time before and after representation became an international target. For each country, we calculated the amount of area protected within each decade and allocated the same amount randomly or optimally.

We determined optimal PE by assuming countries would always protect the proportionally least-protected ecoregions first, as it results in the largest increase in PE. We took a "greedy" approach, solely aiming to maximize PE without considering the quality or availability of land for protection. For random simulations, we considered land quality and availability by removing PAs designated before each decade, as well as degraded land types that were considered unsuitable for protection (croplands and urban and built-up areas; Friedl *et al.* 2010; Channan *et al.* 2014). We randomly selected planning units equal to the average PA size (rounded up to the nearest 100 km^2; Supplementary Table S1) in each country and decade over 1,000 simulations and calculated PE. Random PE was considered as the average PE of all simulations. A sensitivity analysis was performed to determine how planning unit size impacts random PE scores within the last decade by randomly allocating 100, 2,000, and 6,000 km^2 planning units within each country and calculating PE as above.

Economic, social, and ecological drivers of change in PE

We built linear mixed-effects models, with country, world region, and year as random effects, to investigate the relationship between periods of rapid PA expansion, PE, and economic and social covariates. These included annual change in total area protected, time, a binary variable representing rapid PA expansion ("hot moments," where countries protected ≥33% of their area in a single year; Radeloff *et al.* 2013), and economic, social, and environmental variables (see Supplementary Appendix S3). All variables were included as additive effects except for the interaction between "hot moments" and the number of PAs designated in each year, which was included to investigate the effects of rapid expansion through large or small PAs. All models were run in R v. 3.2.2 and compared using the AICc (Burnham and Anderson 2002). The top models (ΔAICc $\leq$ 4) were averaged to obtain estimates of the effect of each variable on change in PE.

Results

In total, 68 countries met our selection criteria. However, Eritrea and Iraq did not protect enough area within our time period and were removed from all analyses. Bhutan, Guyana, and Suriname were removed from the multivariate model because reliable economic and social time series data were unavailable. As a result, 66 countries were included in our trend and correlation analyses and 63 countries in our multivariate model, all of which are accountable to the goals outlined under the CBD.

PE within the studied countries ranged from 0.025 in Bangladesh (1.23% protected) to 0.743 in Greece (8.84% protected) (Supplementary Table S2). More countries protected area in 1984–1993 than any other period and had the most "hot moments." There was a significant positive relationship between total area protected and total PE within each country in 2013 ($r_s = 0.46$, $P \leq 0.0001$; Figure 1).

Patterns in PE through time

Overall, 60 (90.9%) countries exhibited significant trends in PE over the past 60 years; all of which were increasing (Supplementary Figure S1A). No significant overall trend

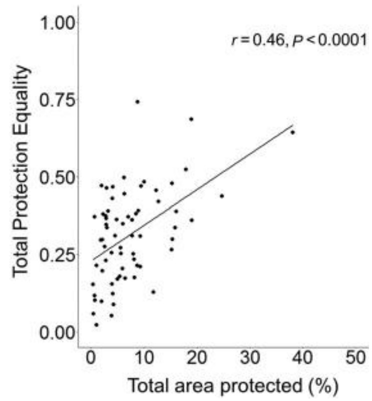

Figure 1 The correlation between the total area protected and total PE in each studied country as of 2013 ($n = 66$).

was detected within Japan, Myanmar, Nepal, Oman, Pakistan, and Uzbekistan, although Myanmar had the only overall reduction in PE over this time period. Eight countries (Afghanistan, Angola, Bangladesh, Central African Republic, Germany, Mali, Morocco, and Slovenia) had significant positive trends despite relatively small overall increases in PE (<0.1), while Greece, Botswana, and Bhutan had the largest increases (0.73, 0.61, and 0.64, respectively). In a typical decade, approximately 46.7% of countries exhibited an increasing trend in PE while nearly 3% had a significant decreasing trend (Figure 2A and Supplementary Table S3 and Figure S2).

Twenty-six (39.4%) countries had an overall significantly positive correlation between change in PE and the amount of area protected, while the rest did not exhibit a significant relationship (Supplementary Figure S1B). The percent of positive correlations steadily increased, within each decade with the two most recent decades having the greatest percentage of significantly positive correlations between change in area protected and change in PE (26.8 and 21.9%, respectively; Figure 2B and Supplementary Table S4 and Figure S3). Japan had the only negative correlations.

Drivers of change in PE: Inevitable or deliberate?

All six countries for which we simulated random and optimal protection scenarios had below optimal PE values in both decades. Canada, Indonesia, Mongolia, and Peru achieved PE values closer to optimal in the second period than in the first (Figure 3). Only Australia expanded its PA system in a way that was significantly greater than random PE from 1994 to 2003, while Peru and Australia had significantly lower than random PE in 1994–2003 and 2004–2013, respectively. All other countries had PE

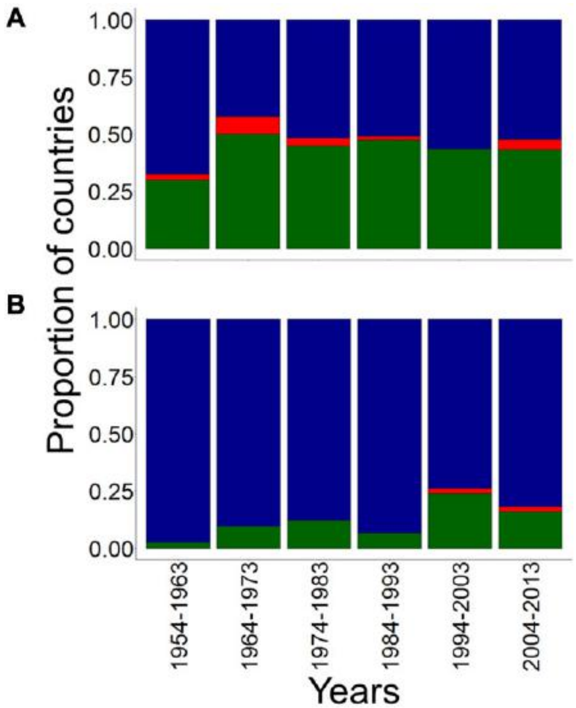

Figure 2 (A) Mann–Kendall trend results for each 10-year time interval showing the proportion of countries with significant increasing (green), significant decreasing (red), and nonsignificant (blue) trends in PE and (B) Spearman's correlation results for each 10-year time interval showing the proportion of countries with significant positive (green), significant negative (red), and nonsignificant (blue) correlations between the change in PE and the change in area protected. Proportions are calculated from the number of countries which protected enough area to detect a trend or correlation in each time period.

values that were not statistically different from random in both decades. Results were not dependent on planning unit size within Brazil, Canada, Indonesia, and Mongolia. In Australia and Peru, observed PE was significantly less than random when 100 km^2 planning units were used but became nonsignificantly different as planning unit size increased. In all cases, smaller planning units resulted in higher random PE scores (Supplementary Table S5).

Economic, social, and ecological drivers of change in PE

Our complete model set for explaining drivers of change in PE contained 1,024 models, with 19 considered to have good fit to the data (ΔAICc $\leq$ 4; Supplementary Table S6). Model selection revealed annual change in PE has decreased since 1954 (Table 1). Large increases in the amount of area protected had positive effects on the annual change in PE, as well as "hot moments" and the interaction between "hot moments" and the number of

Table 1 Averaged model output from multivariate model to search for drivers of annual change in PE (*denotes significance)

Parameter	Estimate	95% CI
(Intercept)	0.020	(0.0123, 0.0272)
Hot moments	0.070*	(0.026, 0.114)
Number of protected areas	0.001	(−0.0012, 0.003)
Change in area protected	0.011*	(0.0087, 0.0136)
Hot moments × number of protected areas	0.228*	(0.0789, 0.3779)
Population density	−0.002	(−0.004, 0.0003)
Number of ecoregions	−0.001	(−0.0035, 0.0010)
Gross Domestic Product per capita	0.002	(−0.0008, 0.0054)
Political status (Non-independent)	−0.004	(−0.0115, 0.0035)
Political status (Democratic)	−0.007*	(−0.0128, −0.0016)
Start category (Late)	0.015*	(0.0079, 0.0227)
Start category (Mid)	0.011*	(0.0044, 0.0173)
Start category (Mid-early)	0.006	(−0.00009, 0.012)
Protection gap	0.001*	(0.00002, 0.0011)
Time since 1954	−0.0003*	(−0.0004, −0.0001)

designated PAs each year. Countries that are not democratic, that started protecting area relatively later, and that have a greater number of years between periods of area protection (i.e., changes in PE) showed significantly greater increases in annual change in PE.

Discussion

Ecological representation is a cornerstone of international conservation agreements aimed to safeguard the world's biodiversity. It ensures that all biodiversity features of interest are included within a PA network. Testing trade-offs between actions is common practice in conservation, and trade-offs *between* Aichi Targets (i.e., 11 and 12) have already been identified (Marco *et al.* 2015). Evaluating trade-offs and synergies *within* targets, such as rapid land acquisitions and achieving representation, is imperative for determining the impacts of conservation commitments and achieving desired outcomes. Our initial results found that PE has increased through time and may be playing a larger role in conservation planning. However, further analyses questioned the strategic nature of these trends and identified potential disconnects between theory and practice.

A trade-off does not appear to exist between rapid PA expansion and achieving equitable representation. Our *a priori* assumption was that large land acquisitions would result in little to no improvement in PE due to shortfalls in the time and resources needed to implement representative PA networks. In contrast, rapid PA growth resulted in larger changes in PE, and most countries exhibited positive trends in PE through time. Nevertheless, we found

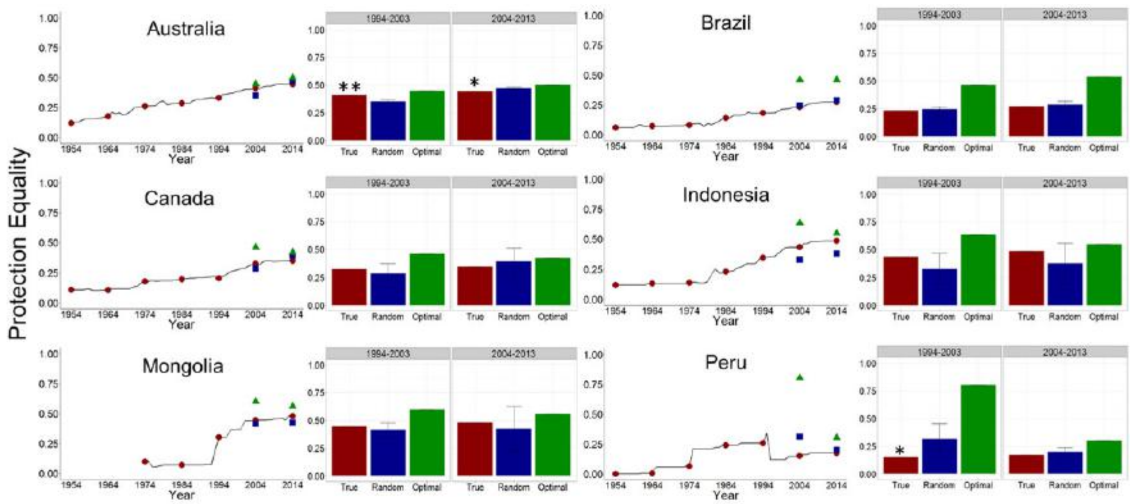

Figure 3 Random and optimal PE results within six simulated countries. Observed ("true") trends in PE from 1954–2013 with simulated optimal (green), random (blue), and "true" (red) PE (±95% CI) values in the last two decades. **An observed PE value significantly greater than random. *An observed PE value significantly less than random.

that, in most cases, these increases are likely not driven by deliberate consideration of representation principles but are the fortuitous result of protecting more area.

Our random PA simulations suggest that positive trends in PE are due to chance rather than choice; driven by the increased probability of representing more ecoregions as more area is protected rather than strategic planning. The positive relationship between total area protected and total PE further supports this point, as well as our model, which revealed that despite the overall increasing trends in PE, the annual change in PE has decreased through time. Regardless of inevitable inefficiencies and lags between the introduction of theories and their execution, it is surprising that changes in PE today are less than those in the 1950s, before representation was defined. Some countries did achieve closer to optimal PE in the last decade, after the introduction of the first international representation target that required at least 10% of each of the world's ecological regions to be conserved by 2010 (Secretariat of the CBD 2002). However, the purposeful consideration of representation could have likely resulted in even greater PE.

The increasing proportion of countries with positive correlations between large land acquisitions and large changes in PE through time may suggest that representation and conservation planning are playing an increasing role during rapid PA growth. Unlike the previous trends, these correlations do not seem to be driven by the amount of area protected. For example, Chile exhibited significant positive correlations between PA expansion and change in PE in the last two decades, even though more or equal area had been protected in the four previous decades where no significant correlation was found. While isolating the factors behind this pattern is difficult, the average size of newly designated PAs in Chile was the smallest during these last two decades, indicating that the size of individual PAs during large PA network expansion may play an important role in achieving equitable representation.

Our multivariate model and sensitivity analyses support this point, revealing that rapid area accumulation improves changes in PE when implemented through multiple, smaller PAs rather than fewer, larger PAs. Previous studies have found similar results, showing that large selection units (in our case, PAs) drive the overrepresentation of features (Pressey & Logan 1998) or allow entire features to fall into unprotected gaps (Kendall *et al.* 2015). With large PA expansion predicted to increase in coming years in an effort to meet percent coverage targets (Blomley *et al.* 2013), and others promoting the need for "mega reserves" (Laurance 2005), it will be important to consider the role of scale in achieving conservation outcomes.

Our model also revealed that countries that started protecting area relatively later tend to have greater annual change in PE. Historical biases in representation (Pressey 1994; Joppa & Pfaff 2009; Watson *et al.* 2011) likely create a significant disadvantage for countries that have a longer history of area protection, while countries that started protecting area later may have incorporated new knowledge in PA design. The positive impact of breaks between PA designations may signify that these periods are spent planning the strategic placement of PAs. However, this relationship, as well as other potential mechanisms driving changes in PE (i.e., the use of conservation planning tools, education, funding, etc.), may be difficult to quantify and should be investigated further at a finer scale.

We used the best publicly available global data, but it has some limitations, which we discuss briefly below (see Supplementary Appendix S4 and Table S7 for additional caveats). Due to data availability, we included only a subset of countries that reported sufficient PA establishment year and boundary data. Every major world region was represented in our final selection; however, some countries (i.e., the United States, Russia, and China) could not be included due to this constraint. This may have limited our ability to identify significant drivers of annual change in PE in our multivariate model. Additionally, reported establishment year may reflect the date of reclassification or reporting not PA designation, which could skew PA expansion to later dates and ultimately affect the accumulation of PE through time. Complementing our findings with regional analyses is likely to uncover further insight that we were unable to capture, or inadvertently missed, at this scale. For example, fine-scale data on conservation funding/aid, land use change, and political structure may uncover additional factors governing the effective implementation of PE.

Uncertainty surrounding biodiversity makes equitable ecological representation appealing, as it safeguards every feature to the same degree. However, PE is just one potential metric to measure representation. Threats and the importance of features are often uneven in the landscape, which may prioritize protection of one feature over another (Myers *et al.* 2000; Dirzo & Raven 2003; Brooks *et al.* 2006). For example, Aichi Target 11 highlights the need to protect areas of "particular importance for biodiversity" (Secretariat of the CBD 2010), which may impact the equality of representation and require a different approach.

Implementing clear, quantifiable, and achievable targets will be instrumental in conserving biodiversity. Representation, no matter the definition, will only be effective if other objectives within PA networks are met (i.e., management, connectivity, etc.). Global conservation agreements need to simultaneously consider representation with other conservation targets and balance trade-offs to maximize the overarching goal: halting biodiversity loss.

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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 S1. Data sources, selection criteria, and spatial analysis.

Appendix S2. Equations to calculate PE.

Appendix S3. Description and data sources of variables used to search for drivers of annual change in PE in our multivariate model.

Appendix S4. Additional data limitations and caveats.

Table S1. Area protected, average PA size between 1994–2003 and 2004–2013 and corresponding planning unit (PU) size used for random simulations.

Table S2. Summary table of area protected and PE in each studied country as of 2013.

Table S3. Mann–Kendall trend results by country in each decade.

Table S4. Spearman's correlation results by country in each decade.

Table S5. Random PA allocation sensitivity analysis results using 100, 2,000, and 6,000 km^2 planning units and the average PA size within each country from 2004 to 2013.

Table S6. Model comparison statistics for the list of models that were considered as a good fit to our data (ΔAICc ≤ 4).

Table S7. Results from performing a correlated Bonferroni technique to account for potential increases in type I errors. Uncorrected results are presented in the main text as corrected results did not change the narrative of our results.

Figure S1. (A) Overall (1954–2013) trends in PE and (B) overall Spearman's correlation results between annual change in area protected and annual change in PE.

Figure S2. Maps of decadal Mann–Kendall trend results in PE in each country.

Figure S3. Maps of decadal Spearman's correlation results between PE and the change in the proportion of area protected in each country.

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LETTER

Status and Trends in Global Ecosystem Services and Natural Capital: Assessing Progress Toward Aichi Biodiversity Target 14

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Abstract

The Convention on Biological Diversity uses six indicators to assess progress toward Aichi Biodiversity Target 14 (ecosystem services), leaving many elements of the target untracked. We identify 13 ecosystem services as directly essential for human well-being, and select a set of 21 datasets as indicators of the state of natural capital underpinning those services, the benefits derived from them, and distribution of access to those benefits. Analysis of these indicators supports previous conclusions that there is no overall progress toward Target 14. Sixty percent of our "benefit" indicators have positive trends, whereas 86% of our "state" indicators show a decline in natural capital. This suggests that well-being is increasing in the near-term despite environmental degradation, and that unsustainable use of natural capital may fuel human development. As regulating services such as "soil fertility" continue to decline, however, it seems unlikely that this trend can continue without future negative impacts on humanity.

Introduction

In response to biodiversity declines (Butchart *et al.* 2010) and an increasingly well-understood relationship between biodiversity and human well-being (Mace *et al.* 2012), the Parties to the Convention on Biological Diversity (CBD) adopted the Strategic Plan for Biodiversity 2011–2020 (Strategic Plan), including the 20 Aichi Biodiversity Targets. Human interactions with nature can be framed using the language of natural capital (NC) and ecosystem services (ES; Figure 1). ES contributions to human well-being are complex and sometimes poorly understood (The Economics of Ecosystems and Biodiversity 2010). A growing human population and a shift toward more resource intensive lifestyles are increasing the demands on ES. This appears to be degrading reserves of NC, potentially reducing ES available to future generations (Millennium Ecosystem Assessment 2005).

Global Biodiversity Outlook 4 (GBO-4; Secretariat of the Convention on Biological Diversity 2014), which provided a mid-term assessment of progress toward the Aichi Targets, concluded that, while significant progress had been made toward "some components of the majority of" the targets, generally progress was insufficient to ensure that targets would be met by 2020. GBO-4 was underpinned by sources including statistical analysis of global indicators (Tittensor *et al.* 2014), the 5th National Reports to the CBD, and global indicators compiled by the Biodiversity Indicators Partnership (BIP; Leadley *et al.* 2014).

GBO-4 divided Target 14, which focuses on ES, into two elements: (1) "Ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded . . . "; and, (2) " . . . taking into account the needs of women, indigenous and local communities and the poor and vulnerable." Element one was assessed as

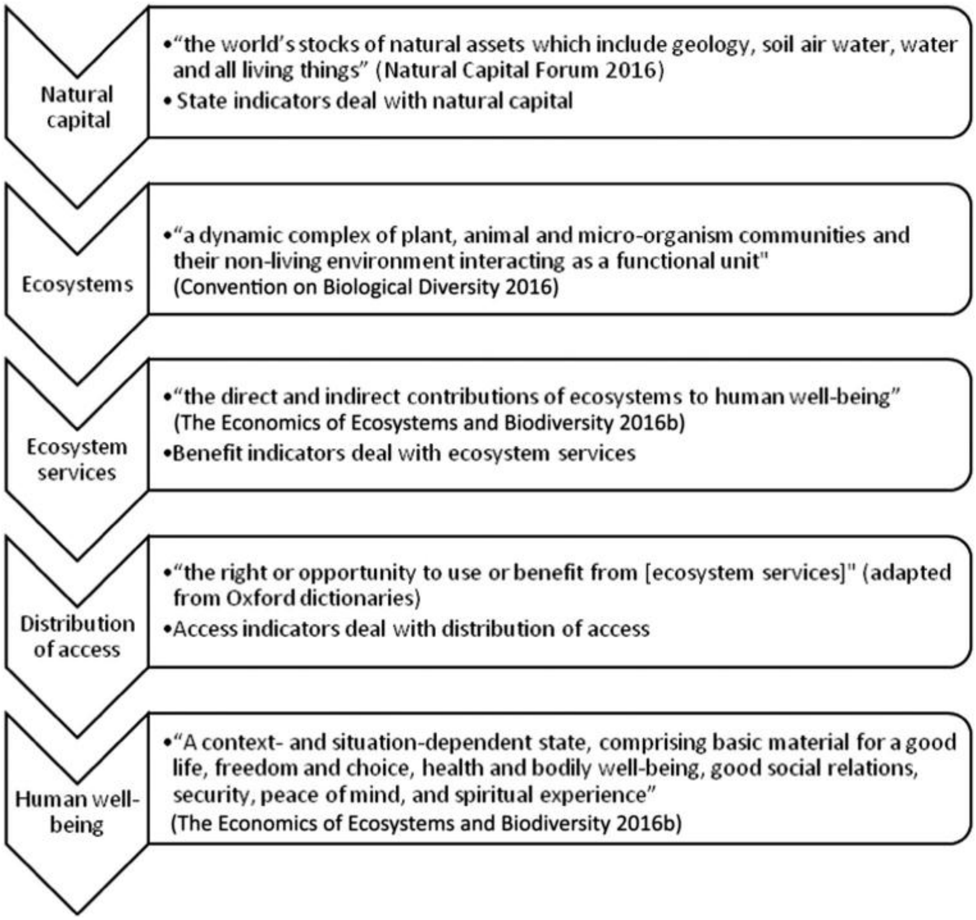

Figure 1 Basic flows from natural capital to human well-being (Convention on Biological Diversity 2016; Natural Capital Forum 2016; The Economics of Ecosystems and Biodiversity 2016b).

"moving away from the target" (i.e., ES were declining), while element two had "no significant overall progress." Both were given the lowest available level of confidence in the assessment "based on the available evidence" (Secretariat of the Convention on Biological Diversity 2014). Challenges in identifying indicators arise from the inherent complexity of ES: definitions of ES overlap (The Economics of Ecosystems and Biodiversity 2016a), for example, pollination also contributes to food provision; the same ES are provided in different ways in different areas; and the same NC can provide multiple ES. Trade-offs between ES are also important (Millennium Ecosystem Assessment 2005). For example, deforestation for agriculture may increase food provision at the expense of climate regulation and carbon storage (Coe *et al.* 2013).

Recent analysis for the Ad Hoc Technical Expert Group on Indicators for the Strategic Plan found that the set of available indicators for Target 14 was inadequate (Chenery *et al.* 2015). We address the challenges of availability of NC and ES datasets, and of assessment of progress toward Target 14. We identify relevant datasets, select an indicator set, and assess their trends to provide a preliminary evaluation of progress toward Target 14.

Methods

Step 1: Essential ES

As Target 14 focuses on "essential services," our first step was to define essential ES. For each ES defined by the Economics of Ecosystems and Biodiversity (The Economics of Ecosystems and Biodiversity 2010), we assessed its contribution to the components of three human well-being frameworks (Millennium Ecosystem Assessment 2003; Gough 2014; United Nations Development Programme 2015). These were selected for their differing approaches to defining well-being, to ensure we took a broad perspective. As Target 14 also focuses on local communities, we considered an ES "essential" if it contributed directly to well-being under any of these frameworks, while acknowledging that other ES are indirectly essential (e.g., climate regulation). Analysis of the Aichi Targets suggested that indirect ES were covered by other targets (e.g., "carbon sequestration" is covered by Target 15 [carbon stocks]). To focus on ES which would otherwise not be measured, indirect ES were excluded from our analysis (Table S1).

Step 2: Dataset compilation

We compiled an extensive list of datasets. While the majority of these did not meet our criteria for a Target 14 indicator set, we believe that this collection of datasets is a valuable contribution in itself. For element 1 of Target 14 (ecosystems are restored and safeguarded), we sought global datasets that could indicate trends in the state of NC underpinning each essential ES, and in the total benefits derived from those ES. For element 2 (accounting for the needs of vulnerable groups), we sought global datasets that could indicate trends in the distribution of access to those benefits across the human population (Tables S2 and S3). Although this approach does not explicitly consider the different groups mentioned in Target 14, people without sufficient access to essential ES are likely to be those considered to be poor and vulnerable.

We reviewed the literature and contacted experts to identify existing datasets, ongoing projects, and unpublished datasets (Tables S2 and S3). We excluded certain abiotic NC components such as fossil fuels and metals from this review (United Nations Environment Programme 2012). While we recognize that this excludes sources of energy, fertilizers, and raw materials, this is only after extensive processing using human capital and in combination with other NC; therefore, we believe that they are not the intended focus of Target 14. NC datasets were categorized as biodiversity, carbon, atmosphere, land, oceans, soil, and freshwater (Tables S2 and S3).

Step 3: Target 14 indicator set

Datasets were selected for each essential ES in three categories: (1) **state** of the underlying NC (which underpins the long-term sustainability of ES), (2) global **benefits** derived from the ES (measuring the status of element 1 of the Target), and (3) distribution of **access** to those benefits (measuring the status of element 2). Figure 1 shows how these categories map onto the flows from NC to human well-being.

Datasets that did not fit into these categories were rejected, as were datasets that were not from a credible source (scientific publications or institutionally branded reports or datasets). Datasets scoring "poor" in the following three criteria were also rejected: (1) spatial extent, (2) number of data points in time series, and (3) ability to detect trends in the ES or underlying NC. Additional criteria were: (4) date range, (5) likely continuation of data collection, and (6) data available online (Table S4). This includes all criteria for global biodiversity indicators considered by Tittensor *et al.* (2014) and Chenery *et al.* (2015), plus additional criteria recommended by the BIP (Biodiversity Indicators Partnership 2012). There are other criteria for environmental indicators (e.g., the Organisation for Economic Co-operation and Development list in Ruffing [2007]), however, we considered these the minimum criteria for a functional global-scale indicator, which could realistically be met by some of the available datasets.

Some datasets relate to multiple ES, for example, the "Wetlands Extent Index" could be a state indicator for "moderation of extreme events" as well as "waste-water processing." To avoid overweighting any datasets when interpreting our indicator set, we included each dataset only once. Those that relate to more than one ES are shown against the most relevant ES. To minimize gaps in our indicator set, in some cases, we selected datasets with a moderate relationship to the ES. We did this only where no stronger eligible datasets were identified, and highlight these indicators as particularly in need of improvement (Table S5).

Step 4: Analysis of trends

To identify trends over time, we calculated linear regressions of annual global averages for the selected datasets. Although actual trends may not be linear, Target 14 is concerned with the long-term trajectory rather than the dynamics of interannual change. Linear regression is the simplest approach to identifying an overall long-term trend, particularly as there is significant uncertainty within the data, as many datasets selected are necessarily based to some extent on estimates. For two of our selected datasets, countries were excluded from our analysis if they had data missing for any year, to ensure we compared like with like when calculating trends. Sensitivity analysis shows that this approach did not materially affect our results (Annex S2). A linear regression with P-value ≤ 0.05 was considered to indicate a statistically significant trend. All available years from 1980 onward were included in the analysis for each dataset, to identify reasonably long-term trends.

For one dataset, "disability adjusted life years (DALYs) lost to parasitic and vector-borne diseases," data were only available for 2000 and 2012. To be precautionary about inferring trends given the lack of data, we considered a change of over 20% between the two years to be substantive. For seven datasets, we relied on previously published analyses to determine trends (Table S6). We define a positive trend as one that results in an increase in well-being, regardless of the direction of the statistical trend.

To derive overall ratings for each ES, if all selected indicators for that ES had the same trend (positive, negative, or no trend), a matching overall rating was given. If the indicators were a combination of positive trend and no trend, an overall rating of positive trend was given (similarly for negative trend). If an ES had indicators showing both positive and negative trends, an ambiguous overall rating was given.

Baseline states against which to assess performance have not been identified, a limitation raised by Tittensor *et al.* (2014). Therefore, our analysis does not indicate proximity to meeting Target 14, only whether the gap is closing or widening.

Results

We classified 13 of the ES defined by TEEB as "essential" in the context of Target 14 (Table S1). In total, we identified 153 datasets (Tables S2 and S3). Most have global coverage, with nine regional datasets included that are useful for global analyses. For example, Saatchi *et al.* (2011) provide globally important information about forest carbon stocks in tropical regions.

From these datasets, we selected a set of 21 indicators (Tables 1 and S3). Annex S1 describes the selection process for each ES. Gaps remain in our indicator set, in particular no suitable datasets were identified for "aesthetic appreciation and inspiration" or "spiritual experience and sense of place" (Table S5).

Of these 21 datasets, we assessed 13 as having high ability to detect trends in the ES or underlying NC. All selected datasets have global coverage. The four "extreme events" datasets have the longest complete time series, with annual data from 1980 to 2014. In contrast, two of the selected datasets have just two or three data points in the time series. Additionally, five of the selected datasets ended before 2011, whereas nine had a value for 2014 or 2015 (Table S4).

Six of the seven state indicators had a negative trend. The other, "mangrove extent," had no trend. Thus, overall, analysis of our indicator set suggests that the state of the NC underpinning ES is declining. Of the 10 benefit indicators, six show a positive trend, two have a negative trend, and the trends for two are ambiguous, depending on the time period over which they are analyzed (Figure 2). Thus, overall, the benefits obtained from ES appear to be improving, although this conclusion hides substantial variation. "Prevalence of undernourishment" has a positive (decreasing) trend, "population affected by fires, floods and storms" has a negative (increasing) trend, and the other two access indicators have no discernible trend, indicating no overall improvement in access to ES among the poor and vulnerable (Tables 2 and S6).

Overall trends (Table 2) were positive for two ES ("biological control" and "recreation and physical and mental health"), ambiguous for six, and negative for three ("freshwater," "waste-water treatment," and "erosion prevention and soil fertility"), suggesting little or no progress toward Target 14. This broadly confirms the GBO-4 assessment (Secretariat of the Convention on Biological Diversity 2014). Of the 16 linear regressions calculated, six had an $R^2 \geq 0.9$, indicating strong trends with little variation, while others had relatively noisy datasets, and three showed no trend (Figure 3; Table S6).

Discussion

Our approach captures the essence of Target 14, by focusing on the state of the NC that underpins essential ES, together with the benefits derived from them, and the distribution of access to those benefits. However, the Target explicitly mentions livelihoods, and access to ES for specific vulnerable groups. These components are hard to include in a global-scale indicator set, as that requires large amounts of disaggregated local-scale data.

Our proposed set of 21 indicators is more comprehensive than those used previously for Target 14. GBO-4 used just six indicators alongside a selection of case studies (Secretariat of the Convention on Biological Diversity 2014). Tittensor *et al.* (2014) identified just one indicator that met their criteria for analysis, "the Red List Index for pollinators," which we included in our set. Tittensor *et al.* considered and rejected eight datasets for Target 14, of which three are included in our indicator set: "inland water resources" was rejected on the grounds of geographic coverage, although our dataset "total inland water resources per capita" covers 180 countries over five continents. "Production of forest products" and "inadequate access to food" were rejected by Tittensor *et al.* on the grounds of relevance to the target, but both fitted well into the categories of indicators we sought. Tittensor *et al.* did not set out any structure within which to identify datasets, making it difficult to assess relevance.

Currently, global indicators for ES and NC are inadequate for detailed trend analysis, highlighting an important challenge for all the Aichi Targets. Our work also demonstrates potential difficulties for the measurement

Table 1 State, benefit, and access indicators selected for the Target 14 assessment, organized by ecosystem service (for details, see Table S2)

Ecosystem service	State indicators	Benefit indicators	Access indicators
Provisioning services:			
Food	S1. State of world marine fish stocks	B1. Average dietary energy supply adequacy	A1. Prevalence of undernourishment
Raw materials	S2. Forest extent	B2. Production of forest products	–
Freshwater	S3. Nitrogen and phosphate fertilizers	B3. Renewable water resources per capita	–
Medicinal resources	S4. Red List Index (RLI) for food and medicine	B4. Estimated export volumes of medicinal plants	–
Regulating services:			
Local climate and air quality	–	B5. Population weighted exposure to particulate matter <2.5 μm in width ($PM_{2.5}$)	A2. Proportion of the population exposed to a $PM_{2.5}$ concentration of 10 $\mu g/m^3$
Moderation of extreme events	S5. Mangroves extent	B6. Occurrence of fires, floods, and storms	A3. Population affected by fires, floods, and storms
Waste–water treatment	S6. Wetlands Extent Index	–	–
Erosion prevention and soil fertility	–	B7. Occurrence of drought and landslides	A4. Population affected by drought and landslides
Pollination	S7. RLI for: pollinators	B8. Production of pollinator–dependent crops	–
Biological control	–	B9. Disability adjusted life years (DALYs) lost to parasitic and vector diseases	–
Cultural services:			
Recreation and physical and mental health	–	B10. Global average healthy life expectancy (HALE)	–
Aesthetic appreciation and inspiration	–	–	–
Spiritual experience and sense of place	–	–	–

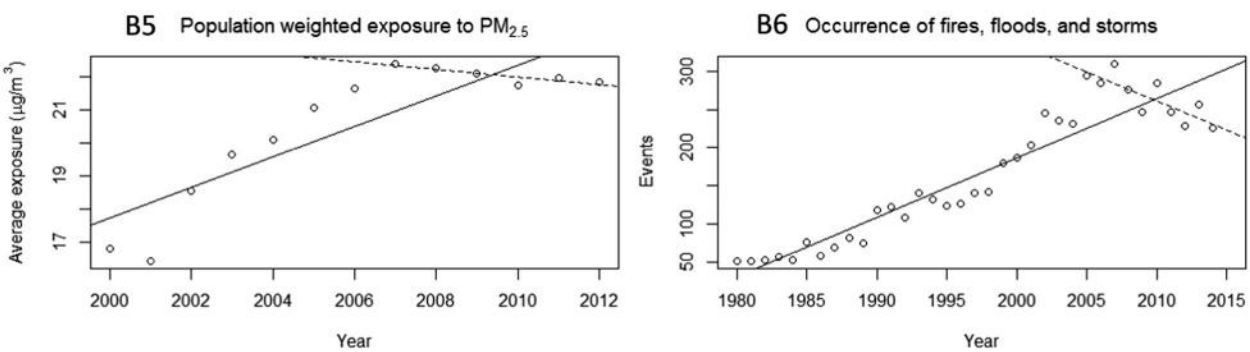

Figure 2 Trends in indicators showing reversed trends depending on the time periods analyzed. Solid lines show the regression line for the whole time series and dashed lines show the regression line for a subset of the time series ending at the final available data point: 2007–2012 for B5; 2005–2014 for B6. For full details of data sources and analysis, see Tables S2 and S6.

Table 2 Results of analysis organized by ecosystem service (for full analysis of datasets, see Tables S5 and S6)

Ecosystem service	State	Benefit	Access	Overall
Provisioning services:				
Food	↘	↗	↗	(↘↗)
Raw materials	↘	↗	–	(↘↗)
Fresh water	↘	↘	–	↘
Medicinal resources	↘	↗	–	(↘↗)
Regulating services:				
Local climate and air quality	–	(↘↗)	→	(↘↗)
Moderation of extreme events	→	(↘↗)	↘	(↘↗)
Waste–water treatment	↘	–	–	↘
Erosion prevention and soil fertility	–	↘	→	↘
Pollination	↘	↗	–	(↘↗)
Biological control	–	↗	–	↗
Cultural services:				
Recreation and physical and mental health	–	↗	–	↗
Aesthetic appreciation and inspiration	–	–	–	–
Spiritual experience and sense of place	–	–	–	–

↗ positive trend; ↘ negative trend; → no trend; (↘↗) trend ambiguous;—no indicators identified.

of progress against some of the Sustainable Development Goals (SDGs; United Nations Department of Economic and Social Affairs 2015). Series length and series fullness (i.e., how many years have data within the series length), and the interaction of these, impact the analysis of overall trends in time series data (Collen *et al.* 2009). There was great variability in these aspects for our indicator set. The approach taken by Tittensor *et al.* (2014), to fit statistical models and forecast levels at 2020, would provide a more powerful analysis, but given levels of uncertainty in the selected datasets, and the variability in series length and fullness, we preferred a simpler approach. Future researchers may consider establishing appropriate benchmarks or thresholds for each indicator, to assess proximity to meeting Target 14 as well as overall trends.

Gaps remain in our indicator set, particularly for cultural services, for which only one dataset was selected (Table S5). Development of new methods for assessing cultural ES would be beneficial, for example, UN Habitat have proposed an SDG indicator, "Proportion of residents within 0.5km of accessible green and public space," based on existing remote sensing data. No suitable datasets were identified for 18 of the 39 categories for which we sought indicators. Furthermore, only 13 (62%) of our selected indicators have a strong ability to detect trends in the relevant ES or underlying NC. Data on the state of NC are particularly important to provide information on the sustainability of ES for future generations (Millennium Ecosystem Assessment 2005). The state indicators for "freshwater," "medicinal resources," and "pollination" were all assessed as only moderately able to detect trends in the underlying NC, and no suitable state indicators were identified for "local climate and air quality," "erosion prevention and soil fertility," "biological control," or any cultural ES. Additionally, no suitable state indicators were identified for food grown on the land. Developing datasets to monitor the state of the NC underlying these ES should be the focus of further work. Some of these gaps are being addressed through initiatives including the Water Quality Index for Biodiversity (United Nations Environment Programme 2015) and the Global Action on Pollination Services for Sustainable Agriculture (Food and Agriculture Organization of the United Nations 2015). However, it will take time to generate sufficient data to identify trends, and resources for generating data will only be made available where there is clear utility from delivering results (Bubb *et al.* 2011).

For transparency, we show the results of analysis of each dataset individually (Table 2). To avoid opposing trends canceling each other out, we use an "ambiguous" overall ES rating in these cases. In assessing progress toward Target 14, we give all datasets equal weighting regardless of the strength of their ability to reflect relevant trends or their relative importance to ES. Some of these indicators might have more importance than others for global ES, potentially adding bias.

Positive trends can be seen in benefits, at least in the short term, even where the underlying NC is being degraded. For example, the energy supply to the urban poor can be improved by clearing forests and making charcoal for cooking fuel. In contrast, all but one of our state indicators has a negative trend, suggesting that benefits are being extracted today at the expense of future generations.

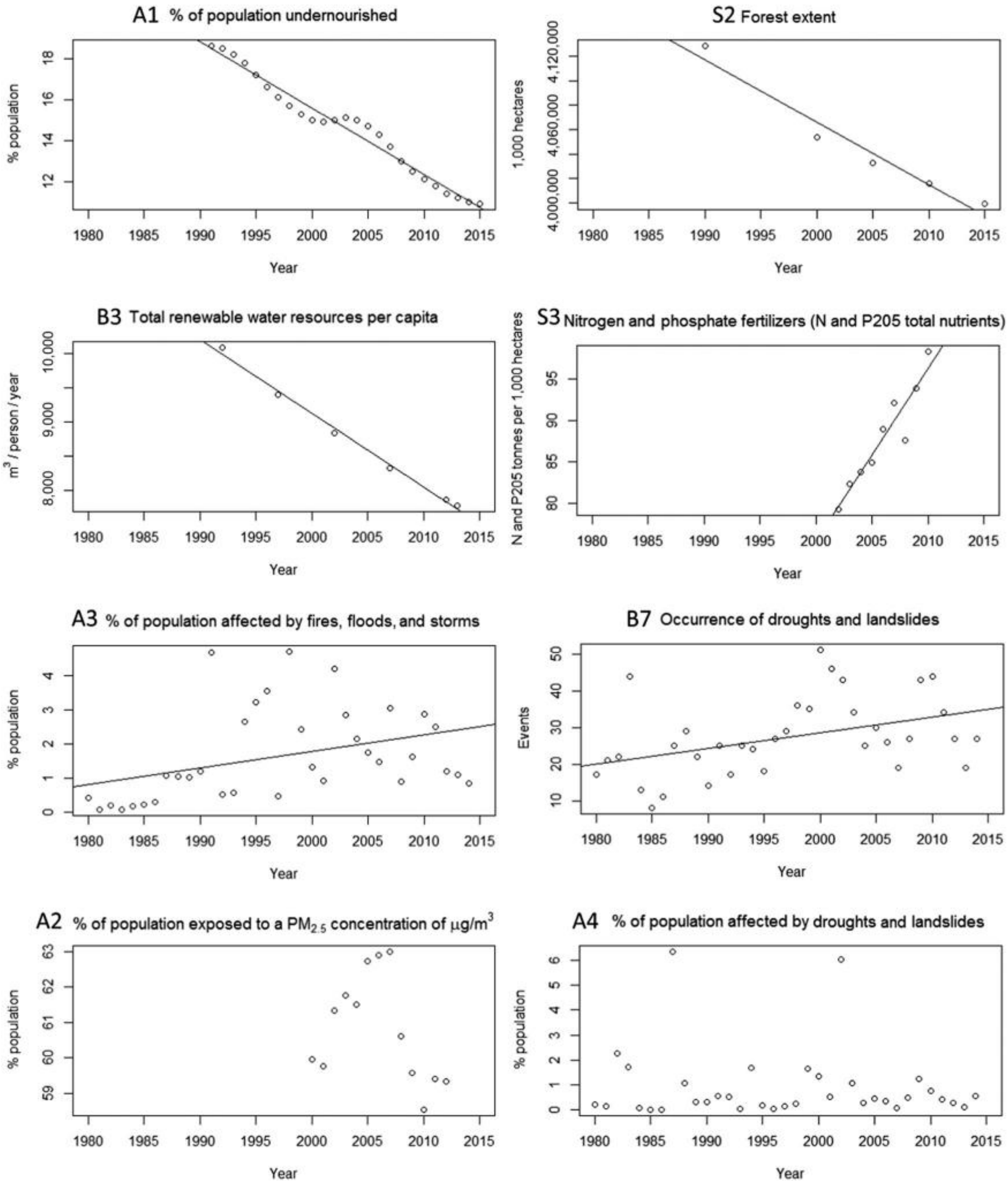

Figure 3 Examples of trends in indicators of relevance to Aichi Biodiversity Target 14 (for full details of data sources and analysis, see Tables S2 and S6).

In general, positive trends were identified in areas important for human development (e.g., access to sufficient food), which can be generated from human capital and infrastructure and are not solely dependent on the state of the underlying NC. Negative trends were found in areas with less immediate influence on well-being, such as "erosion prevention and soil fertility," suggesting that NC providing such benefits is at particular risk. Less easily quantified ES such as "sense of place" may also be more difficult to replace in the absence of the NC that underpins them.

In conclusion, determining a structure against which to identify datasets enabled selection of a more complete indicator set than previously used to assess progress toward Target 14. Our analysis supports previous conclusions that the global community is not making any real progress toward Target 14. The pattern of trends within our indicator set highlights the "environmentalist's paradox," that human well-being is increasing, together with access to ES including freshwater and disease prevention, despite the degradation of the NC that underpins those ES. Negative trends in ES such as "moderation of extreme events" highlight the risk that the paradox may not hold true forever. An increased ability to monitor the interactions between human well-being, ES, and NC is needed, to support the generation of policy options and their testing within an indicator-policy cycle (Nicholson *et al.* 2012), supporting a move toward a more sustainable future.

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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:

Table S1: Determination of "essential" ecosystem services (ES). We identified essential ES by assessing the likely contribution to well-being of each ES categorized by The Economics of Ecosystems and Biodiversity (The Economics of Ecosystems and Biodiversity 2016). To ensure that we took a broad perspective, we considered contribution to well-being in terms of the components of three different human well-being frameworks as outlined in columns 2–4 in the table below.

Table S2: Target 14 indicator set, comprised of state, benefit, and access datasets, identified through literature review and contact with experts. "State" datasets are grouped into seven categories of natural capital (NC): biodiversity (including species biodiversity and stocks, genetic resources, and ecological communities); carbon (including Net Primary Productivity, soil organic carbon, and ocean carbon); atmosphere (including precipitation, temperature, and air quality); land; oceans; soil (including fertility and productivity); and freshwater (including quality, extent, and rivers). "Benefit" and "access" datasets are categorized by the most relevant ES.

Our categories of NC are based on the categories proposed by the Natural Capital Committee (2014), adapted to better fit the scope of our review. We combined "species" and "ecological communities" into "biodiversity," included "coasts" within our "oceans" category, and excluded "minerals" and "subsoil assets" as these datasets were not a focus of our study, with the exception of "carbon" which we added as a separate category.

Table S3: Additional state, benefit, and access datasets, identified through literature review and contact with experts. "State" datasets are grouped into seven categories of natural capital: biodiversity (including species biodiversity and stocks, genetic resources, and ecological communities); carbon (including Net Primary Productivity, soil organic carbon, and ocean carbon); atmosphere (including precipitation, temperature, and air quality); land; oceans; soil (including fertility and productivity); and freshwater (including quality, extent, and rivers). "Benefit" and "access" datasets are categorized by ecosystem service.

Table S4: Assessment of selected Target 14 indicators.

a) Spatial extent: rated poor if extent is fewer than 10 countries or fewer than three continents; rated good if range covers at least five continents and at least 20 countries (Chenery *et al.* 2015);

b) Number of data points in the time series: rated moderate if two to four data points available, rated high if five or more data points available;

c) Rated high if end point is 2014 or 2015, rated poor if end point is 2010 or earlier.

Table S5: Key data gaps in our indicator set by indicator category and ES. We highlight where no datasets were identified, and where the best available datasets were assessed as "poor" in any of our assessment criteria (Table S4). These indicators are particularly in need of better, or extended, datasets in the future.

Table S6: Analysis of our Target 14 indicator set. Results of linear regressions to estimate trends in selected Target 14 indicators, for all selected indicators with accessible data and at least three data points in the time series. For all other indicators, we have summarized the analysis completed.

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LETTER

Balancing Ecosystem and Threatened Species Representation in Protected Areas and Implications for Nations Achieving Global Conservation Goals

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Abstract

Balancing the representation of ecosystems and threatened species habitats is critical for optimizing protected area (PA) networks and achieving the Convention on Biological Diversity strategic goals. Here we provide a systematic approach for maximizing representativeness of ecosystems and threatened species within a constrained total PA network size, using Australia as a case study. We show that protection of 24.4% of Australia is needed to achieve 17% representation for each ecosystem and all threatened species habitat targets. When the size of the PA estate is constrained, trade-off curves between ecosystem and species targets are J-shaped, indicating potential "win-win" configurations. For example, optimally increasing the current PA network to 17% could protect 9% of each ecosystem and ensure that all threatened species achieve at least 78% of their targets. This method of integrating species and ecosystem targets in PA planning allows nations to maximize different PA goals under financial and geographical constraints.

Introduction

Systematically planned protected areas (PAs) aim to ensure representative samples of ecosystems are protected and threatened species' habitats are retained (Barr *et al.* 2011; Watson *et al.* 2014; Butchart *et al.* 2015), yet global and national level analyses indicate that neither of these biodiversity conservation goals has yet been achieved (e.g., Rodrigues *et al.* 2004b; Dietz & Czech 2005; Venter *et al.* 2014). Gaps in PA coverage occur because of past biases in PA placement toward remote and unproductive areas with low land use conflicts, coupled with a more recent focus on achieving areal targets without considering the underlying conservation objectives (Rodrigues *et al.* 2004a; Watson *et al.* 2011; Watson *et al.* in 2016a). Future expansion of PAs can only fill these gaps if fine resolution data on species and ecosystem distributions are systematically included (Moilanen *et al.* 2009; Polak *et al.* 2015).

The Convention on Biological Diversity (CBD)'s strategic plan (CBD Secretariat 2010) provides systematic guidance for a global expansion of PAs. The 2010 CBD's Aichi Target 11 stipulates a quantitative goal to protect 17% of terrestrial and inland water area and 10% of marine and coastal ecosystems in areas of particular importance for biodiversity. These PAs should be ecologically representative, effectively managed, and connected. The CBD advocates the use of ecosystems as the primary targets for the placement of PAs to achieve ecological representation (CBD Secretariat 2010; Woodley *et al.* 2012) and the phrase "areas of particular importance for biodiversity" has often been operationalized as protecting threatened species' habitats (CBD Secretariat 2010; Watson *et al.* 2014; Butchart *et al.* 2015). In addition, Aichi Target 12, which refers specifically to preventing the extinction of threatened species, also refers to protecting habitat as one of the means to achieving this goal.

While the CBD plays an important role in bringing nations together to secure global biodiversity, its guidance is somewhat open to interpretation regarding the exact amounts of each ecosystem and threatened species range that should be protected. A common interpretation of the representation element of Target 11 is that 17% of each terrestrial ecosystem should be represented in PAs (Woodley *et al.* 2012; Venter *et al.* 2014). The guidelines for threatened species under Target 12 are even less specific (Butchart *et al.* 2015; Watson *et al.* 2016b). More quantitative guidance would assist countries in expanding their PAs in a way that provides maximum protection for threatened species as well as ecosystems.

As PA networks across the world continue to expand in response to the CBD targets, it is crucial that we understand the trade-offs between targets focused on ecosystem representation (Target 11) and those focused on threatened species habitat requirements (Target 12; Marques *et al.* 2014; Venter *et al.* 2014; Di Marco *et al.* 2015). Here, we address this challenge and provide a systematic approach for simultaneously maximizing representation of threatened species and ecosystems within fixed-size PA networks, using Australia as a case study. We start with a set of area-based targets for the country's 85 major ecosystems and 1,320 listed threatened species, following Polak *et al.* (2015). We use trade-off curves and cost-effectiveness analysis to explore the possible representation of ecosystems and threatened species as PA coverage expands. For each of four PA network sizes (15%, 17%, 19%, and 21% of Australia's total terrestrial area) we identify the optimal combination of ecosystem and species target sizes that can make the best use of limited conservation resources, offering key insights for PA expansion.

Methods

Biodiversity datasets and targets

We divided Australia into 85 bioregions, based on the Interim Biogeographic Regionalization of Australia (Figure 1, IBRA bioregions, version 6.1, Steffen *et al.* 2009), using a spatial resolution of approximately 10 km^2. Australia's bioregions were derived by compiling geographic information on continental scale gradients and patterns in climate, substrate, landform, vegetation, and fauna, and each bioregion is considered a distinct ecologically and geographically defined area (Natural Resource Management Ministerial Council 2004). Bioregions are the unit used by Australia's National Reserve System strategy (Commonwealth of Australia 2009) to represent ecosystems as referred to by the CBD, whereby the goal is to represent 17% of each bioregion to meet the CBD's ecosystem representation goal (Commonwealth of Australia 2015). Other types of data may be used to best represent "ecosystems" in other national contexts. We refer hereafter to our selected units as "ecosystems" to allow for a more universal interpretation. Each ecosystem received an upper target representing 17% of its area, and a range of smaller target sizes was also explored.

We considered the distributions of 1,320 extant terrestrial species listed under the Environmental Protection and Biodiversity Conservation Act (EPBCA). We used maps of species' distributions at a resolution of approximately 10 km^2, developed for extant threatened species available in the Species of National Environmental Significance (SNES) database (Commonwealth of Australia 2012). Species-specific targets for each of the 1,320 threatened species were set based on geographic range size and level of endangerment (Watson *et al.* 2011; Polak *et al.* 2015). These targets scale with geographic range size, requiring species with smaller ranges to be increasingly well protected (Rodrigues *et al.* 2004a). Critically endangered species and/or those with a geographic range size of < 1,000 km^2 were set a target of complete coverage (i.e., 100% of remaining distribution area). For species with large range sizes (> 10,000 km^2), the target was set to cover 10% of current range. For species with geographic ranges of intermediate size (between 1,000 km^2 and 10,000 km^2), the target was linearly interpolated between these two extremes (see Polak *et al.* 2015 for details).

For both ecosystems and species we masked out distributions that occurred in cleared areas devoid of native vegetation. Approximately 7% (0.5 million km^2) of Australia is covered by "cleared areas" which are largely developed for urban or intensive agricultural land use (using a cleared land layer at 100 m^2 resolution in Arc GIS 10.2.2; ESRI 1996).These areas are not currently suitable for conservation through PAs and we were not able to consider the opportunity and financial costs and feasibility of improving their conservation value. For some species, the area of remaining available intact habitat was smaller than their representation target. In such cases, we reduced the target for these species to represent 100%

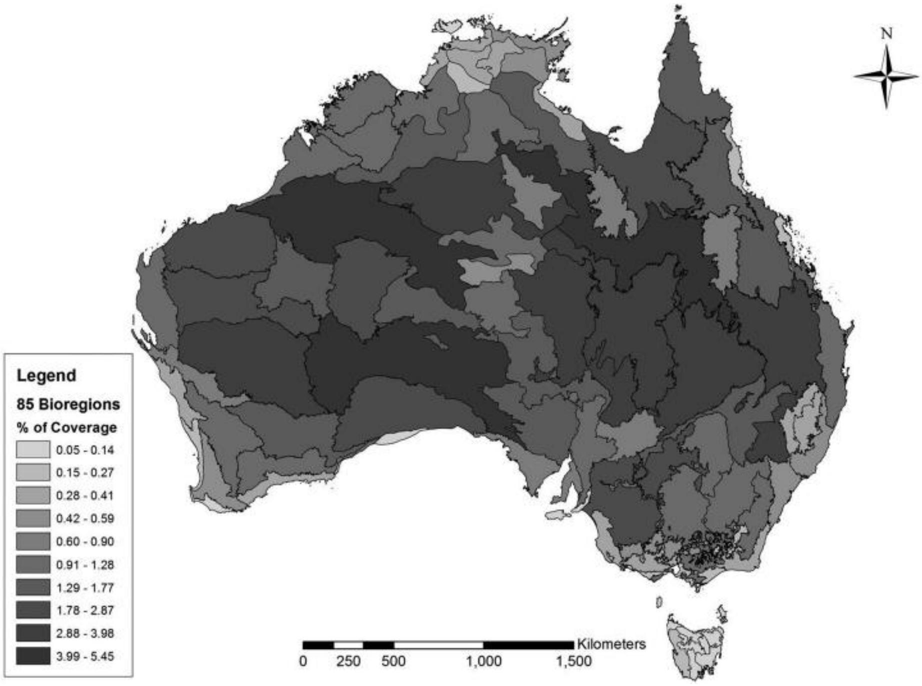

Figure 1 The spatial distribution and cover of Australia's 85 bioregions based on the Interim Biogeographic Regionalization of Australia (IBRA bioregions, version 6.1, Steffen *et al.* 2009).

of remaining available intact habitat. Thirteen of our 1,320 species had none of their distribution within areas that were considered intact and available for conservation. These were counted as gap species and their targets were set to zero. This left 1,307 species as our threatened species target set.

We created a planning unit layer of 10×10 km grid cells covering Australia, and intersected it with the Collaborative Australian Protected Area Database using PAs with IUCN categories I-IV. This resolution approximately matches the scale of the maps of threatened species (Watson *et al.* 2011) and ecosystems (Fuller *et al.* 2010). We intersected the planning unit layer with the PAs, species distribution and ecosystems layers, to determine the amount of each biodiversity feature in each planning unit and the amount already protected based on spatial overlap.

Trade-off and cost-effectiveness analyses

We used the systematic conservation planning software Marxan (Ball *et al.* 2009) to identify the efficiency frontier for the trade-off between representation targets for ecosystems and threatened species coverage when expanding Australia's PA network. Marxan is typically used to select multiple alternative sets of areas that meet prespecified biodiversity targets while minimizing overall cost (e.g., Carwardine *et al.* 2008; Smith *et al.* 2008; Klein *et al.* 2009). When investigating trade-offs between the two sets of targets, we locked in the current PA estate (Watson *et al.* 2011; Polak *et al.* 2015) and set the cost of each planning unit as the total area potentially suitable for conservation within the planning unit. We assumed that only nondeveloped areas would be suitable for inclusion in the PA estate and we used area as a surrogate for the costs of PA management (Ball *et al.* 2009).

To test the trade-off between the target of each ecosystem and threatened species that could be represented, we varied the size of target selected for each feature from 1 to 100% of the original target size, in 1% increments, for all features of the same type (ecosystem or threatened species). We evaluated all combinations of target percentages (e.g., 50% of the original target size for the ecosystem and 10% of the original targets size for the species), giving us 10,000 combinations of target size for the two kinds of features. These percentages of target size are only the minimum level of protection for each run, as Marxan will allow for more protection if it comes at no extra cost. Since there are ~1,300 biodiversity features, many with overlapping distributions, representation above a target level is common because some planning units containing an over-represented feature are critical for meeting targets for other features. Lastly, for ease of interpretation of the results, we translated the percentage of target size for ecosystems to percentage of ecosystem area (i.e., 60% of the 17% target is 10.2% of the size of the ecosystem). We did this for ecosystems only as their target is uniform (17% of each ecosystem's area), while species targets are species-specific (see above).

For each of the target combination runs we identified 100 alternative PA networks and used the most efficient solution (i.e., the one that meets all targets at the lowest cost) in our analysis. We built trade-off curves between the protection of ecosystems and threatened species under four scenarios based on differing sizes of PA networks: 15%, 17%, 19%, and 21% of the land area of Australia. For each scenario we only recorded the unique combinations (out of 10,000) of target percentages that met the scenario's area constraints. Of those, we recorded how many of the targets for each set were met to 99.9% or above for each unique combination of target percentage. These results created a trade-off curve that provides the efficiency frontiers of the nondominated solutions: all points on the top edge of the curve cannot be outperformed by any other point. We also tested how much area of terrestrial Australia is needed to reach every target in full for both kinds of conservation features.

A J-shaped trade-off curve can indicate the existence of a "win-win" solution, where we can achieve relatively high percentages of both targets within the limitation of the set reserve area. To find the points that represent the most cost-efficient "win-win" solutions, we calculated the cost-efficiency of each point, which is the benefit (sum of the two percentages of targets met for species and ecosystems) divided by the area-based cost (i.e., the percentage of Australia's terrestrial area that was used to limit the analysis). Although each point on the efficiency frontier is optimal for the set of targets it meets, the most cost-efficient points provide the greatest feature coverage per unit area protected. The most cost-efficient points were compared within and between the scenarios. We plotted the benefit/cost value of each point along each efficiency frontier against the area constraint of each scenario to compare each area constraint in terms of overall value for investment.

Results

Expanding Australia's current PA network to meet 100% of all species and ecosystem targets requires 24.4% of the total land area, which is much higher than the minimum 17% recommended by the CBD and the area constraints we tested (15–21%). We identified clear trade-offs between target sizes for threatened species and ecosystems, for all four area-constrained scenarios. For each scenario only a few hundred (out of the 10,000) runs met both the area constraints' restrictions and all their targets (Figure 2a–d).

All scenarios displayed J-shaped efficiency frontiers, indicating the potential for finding win-win combinations of target sizes for ecosystems and threatened species (Figure 3a). When the analysis was limited to 15% of Australia's land area, the most cost-efficient points corresponded to protecting between 7.14% and 7.8% of the area for each ecosystem and 54–58% of each species' area target. When following a common interpretation of Aichi Target 11's areal goal of protecting 17% of terrestrial area, the most cost-efficient points corresponded to ecosystem protection of at least 8.7–9.5% of the area for each ecosystem and threatened species protection of at least 75–80% of each species' target. A higher total PA network size of 19% of Australia improves representation of features to at least 81–82% of each species' area target and 12.5–12.8% of the area for each ecosystem. Finally, when the size constraint is at 21% of the land area of Australia, 88–90% of species targets could be met along with the coverage of 15–15.3% of the total extent of each ecosystem.

The cost-effectiveness of the optimal points along each efficiency frontier varied with the area constraint and the combination of target percentages represented (Figure 2b). The area constraint that gives the point with the highest cost-effectiveness ratio is 21% of Australia, at the point of representing ~15% of the ecosystems. While a PA of 24.4% could meet all targets, the targets met per unit PA were slightly lower.

Discussion

We provide a clear and systematic approach to show how to maximize both ecological representativeness and threatened species' coverage in a PA network within a constraint on the total size of the PA system within a country. This enables decision makers to operationalize the dual goal of adequately protecting important habitats for threatened species and achieving ecosystem representation in the global PA network, which is at the heart of the CBD strategic plan (CBD Secretariat 2010). Our approach provides trade-off curves for a wide range of optimal solutions (Polasky *et al.* 2005) allowing decision makers to choose between different configurations of target sizes within the constraints they set on the size of their PA network. For example, placing 17% of terrestrial Australia in PA can at best achieve 9% representation of all ecosystems and at the same time achieve at least 78% of each threatened species' habitat target. This is well short of what is needed to meet Australia's obligations under

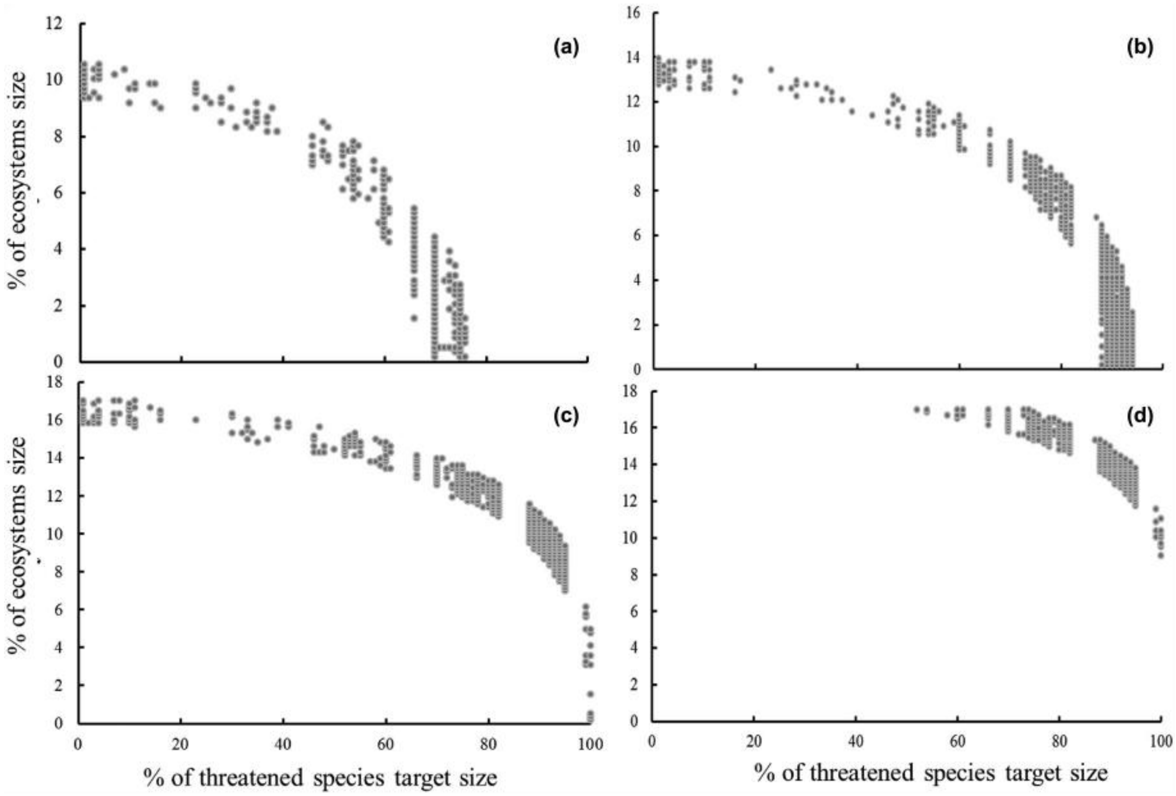

Figure 2 Trade-off curves between target size of species versus target size of ecosystems in a protected area network the size of: (a) 15%, (b) 17%, (c) 19%, and (d) 21% of Australia's land area. Gray points represent solutions that met the area constraints and met all their targets to a level of 99.9%.

the CBD, but maximizes the benefits of a size-limited total PA estate.

We found that among the multiple solutions along the efficiency frontier for each area-constrained scenario, there was a large range of cost-efficiency in terms of how many targets can be met in a limited area. Within each area-constrained scenario, the most cost-efficient points are the ones nearest the inflection point of the J-shaped frontiers, and cost-efficiency declines away from these win-win points (Polasky *et al.* 2005). When comparing between the different area-constraint scenarios (Figure 2b), we can see that the maximum cost-effectiveness increases slightly as more area is available for PA expansion, up to 21%, and then declines. This is likely because as more area becomes available there is more opportunity to select efficient areas that can protect multiple biodiversity features. Once the PA is above 21%, the more efficient and compact options for meeting targets will already be protected and gaining the remaining land required to meet the final parts of targets will require larger areas, resulting in less efficient PA networks.

Although the work we present here is based on information from one country, many of the same challenges faced by land management agencies in Australia occur in other countries (Waldron *et al.* 2013; Venter *et al.* 2014; Di Marco *et al.* 2015). This is because countries are challenged by the goals of meeting their current CBD and country-level PA targets (Waldron *et al.* 2013; Walsh *et al.* 2013). There is a clear need for systematic thinking around targets for species and ecosystem representation, and transparent analysis of the likely compromises between species-based and area-based objectives (Di Marco *et al.* 2015). The overall lesson from our study is that even when countries cannot reach full protection, it is still possible to make progress toward the targets logically and efficiently. Our approach can assist countries in deciding where and how to focus PA expansion efforts given a particular set of geographical and financial constraints.

Where possible, countries should employ spatial information on both ecosystems and threatened species to create their country-specific trade-off curves. If the two sets of targets (Targets 11 and 12) are relatively well aligned, the shape of the curve will exhibit a strong J-shape, making both targets easier to meet. However, if the two sets are relatively discordant, the curves will be closer to linear, and it will be more challenging and area-intensive to

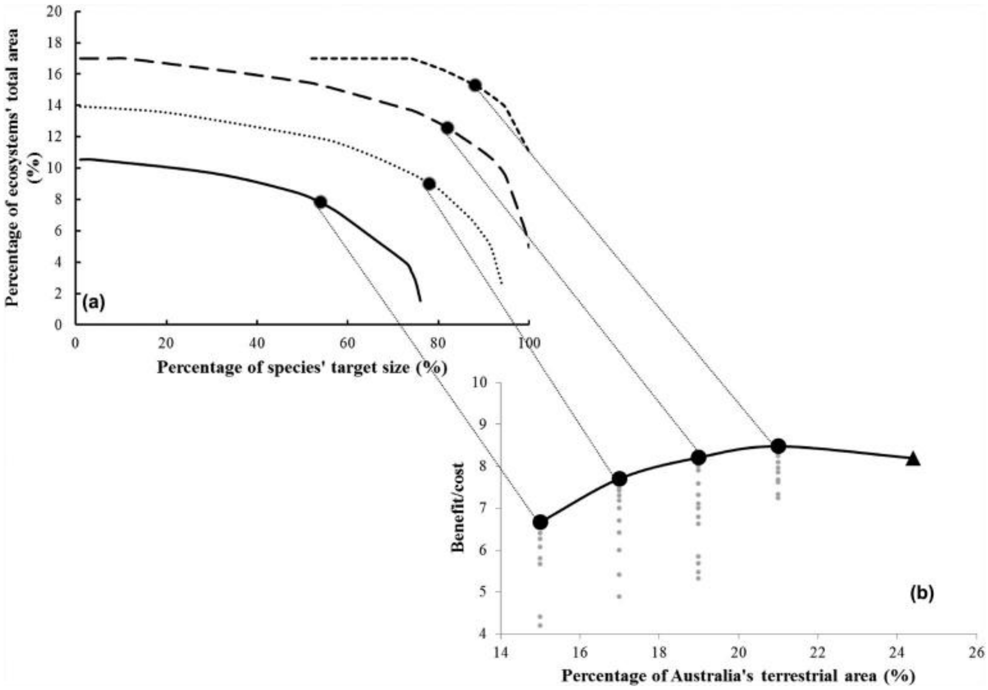

Figure 3 Efficiency frontiers and benefit/cost curves. (a) Efficiency frontiers of the nondominated solutions for the four trade-off curves (Figure 1), where x-axis is the percentage of threatened species' target size and y-axis is the percentage of ecosystems' size. Solid line represents 15% of Australia's land area; dotted line represents 17%; large dashed line represents 19%; and fine dashed line represents 21%. Black circles represent the configuration with the highest benefit/cost ratio for each frontier. (b) Benefit/cost curve of the five most cost-efficient percentage configurations for each of the area constraint curves from Figure 2a (the black circles): x-axis is the percentage of Australia's land area and y-axis is the combined target size percentages over the percentage of Australia's land area. Small light gray circles below each of these represent the benefit/cost values of the rest of the points along the efficiency frontier. Black triangle represents the point where both sets of targets are at 100% (24.4% of Australia's land area).

represent both sets of targets in a PA network. In many countries, distributions of threatened species reflect current and past land-use histories (Taylor *et al.* 2011). This may lead to spatial alignments between the two target types, where remaining threatened species' habitats overlap with the remnants of heavily impacted ecosystems. However, the financial and/or opportunity costs of PA networks in these cases may be relatively high due to the fact that heavily impacted ecosystems are often productive for other uses. In such cases it may be useful to investigate trade-off curves that consider costs as well as area.

Protecting threatened species typically requires a range of management actions, including PA establishment. Decisions on allocating resources among threatened species should consider how important PAs are for ensuring threatened species' persistence. For example, in New Zealand the highest priority action to conserve threatened species is predator control (Dowding & Murphy 2001; McGuinness & Carl 2001). Expanding PAs alone will not adequately protect threatened species unless resources for predator control are built into PA management plans. As such, when planning PA expansion, New Zealand may prioritize the representation of ecosystems in PAs to meet Target 11 (i.e., points on the upper left of the efficiency frontier in Figure 3), while constructing separate threatened species management plans to meet the goals of Target 12.

We have addressed the trade-off between two of the fundamental goals of the CBD, the representation of ecosystems and threatened species through PA expansion. Further trade-offs also exist in the PA planning process. For example, a potential trade-off exists within Target 11 between representing ecosystems and "areas of importance for biodiversity and ecosystem services." Further, a synergy exists between protecting threatened species (Target 12) and reducing the rate of loss of natural habitat (Target 5), as identified by Di Marco *et al.* (2015). There is also a trade-off between protecting existing habitat and restoring some currently unsuitable habitat, especially for species with very limited suitable habitat remaining. Understanding and incorporating multiple trade-offs will improve the effectiveness of implementing the CBD targets. Our approach can be modified to include additional criteria for making informed and optimal decisions when reality demands compromises among various biodiversity goals in PA estates.

Planning for PAs is a dynamic process. The CBD targets have adjusted with time—between 2004 and 2010, CBD targets for global PAs increased from 10% to 17%—and are likely to continue to change, along with countries' capacity to meet them (Noss *et al.* 2012). Changes in species' conservation status will occur and better data on species distributions and their responses to management are likely to become available in the future. For example, there is currently a taxonomic bias in threatened species listing, with many invertebrates missing (Walsh *et al.* 2013). The inclusion of more invertebrate species targets is likely to increase the area or resources required for their protection; however, further research is required to determine how well aligned the important areas for invertebrates with existing priority areas. Countries can use our approach to accommodate such changes, by re-evaluating the progress of PAs against new goals and information as they arise.

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CORRESPONDENCE

Nugatory Targets Lead to Nugatory Reserve Systems that will not Staunch Biodiversity Loss: Commentary on Polak *et al.* (2016)

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Two recent articles (Polak *et al.* 2015, 2016) explore conservation planning for Australian threatened species and ecosystems, concluding that "to meet 100% of all species and ecosystem targets requires 24.4% of the total land area." There is risk in such statements that policy-makers will assume that a reserve network of such extent will ensure conservation security for all threatened species, and be adequate to maintain environmental variation. It will not. The value of the conservation outcomes that such analyses deliver is largely contingent on the ecological sense of the targets used. Responding to Polak *et al.* (2016)'s invitation, that "there is a clear need for systematic thinking around targets for species and ecosystem representation," I note some shortcomings in the targets applied in their analysis.

Both articles use two targets: 17% representation of all ecosystems and inclusion (to variable target levels) of threatened species. The former target, consistent with the UN's sustainable development goals, is without robust ecological foundations and likely to be insufficient to retain many components of biodiversity (Scott & Tear 2007). But in this case, the attribute to which the target is applied is suboptimal. Both articles equate Australia's 85 defined bioregions as ecosystems, an equivalence nowhere intended in the bioregion concept (Thackway & Cresswell 1995). Rather, Australia's long-established policy seeks to reserve the range of ecosystems *within* bioregions (Australian & New Zealand Environment and Conservation Council 1999). Bioregions contain very many distinct ecosystems; e.g., >1,000 ecosystems are defined in the 13 Queensland bioregions (Sattler & Williams 1999). A network designed simply to represent bioregions will leave very many ecosystems unreserved.

The Polak *et al.* articles also claim that their design will meet targets for "all [threatened] species." However, many (several hundreds) of Australia's ca. 1,800 listed threatened species were omitted from their analyses, including freshwater and migratory species, those "whose distributions are only estimated with low certainty," and those now occurring mostly in largely modified environments. Second, modeled distributions were used. Nominal protected areas that represent modeled distributions may well not actually have those species, especially so where ranges are contracting rapidly, as for many Australian species (Woinarski *et al.* 2014). Third, Australia's formal list of threatened species represents only a subset of the actual number of species at risk of extinction. It has major deficiencies, particularly for invertebrates. Even among well-known groups, it is incomplete: for example, of 133 Australian mammals that meet International Union for the Conservation of Nature (IUCN) criteria as threatened, 44 are not included in Australia's threatened species list (Woinarski *et al.* 2014). Fourth, for many (ca. 150) threatened species, the reservation target was for only 10% of their distribution. It is highly unlikely that this nugatory level (i.e., up to 90% of the current distribution left outside reserves) will be adequate to prevent extinction and improve and sustain their status.

The Polak *et al.* (2016) design meets arbitrary targets for representation of bioregions. But, it will fail to provide *any* reservation for many Australian ecosystems and many threatened species, and will provide inadequate reservation—and hence inadequate conservation security—for many other threatened species.

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POLICY PERSPECTIVE

Conservation (In)Action: Renewing the Relevance of UNESCO Biosphere Reserves

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Abstract

The research and policy landscape for biodiversity conservation is changing. Protected areas are now expected to meet a broad range of objectives including effective and equitable management. In this new landscape, organizations strive to find ways to ensure the rights of local and Indigenous peoples are respected while conservation scientists have endorsed the need for platforms for international research and practice. For 40 years, a growing international network of sites support such research and practice, yet, it has been underutilized and largely ignored by scientists and decision-makers alike. To better understand this paradox, this article explores the evolution of the World Network of UNESCO Biosphere Reserves internationally and its application in Canada. Analysis of archived materials, a national survey of practitioners, and interviews with past and present members of Canada's national committee reveals an expanded mandate for biosphere reserves beyond conservation science and biodiversity protection. The article recommends that to support the expanded conservation agenda, biosphere reserves work with governments and conservation scientists to connect more effectively with global concerns and initiatives such as the Convention on Biological Diversity and Sustainable Development Goals; establish appropriate, reliable, and active transdisciplinary partnerships; and meaningfully engage a broader range of knowledge holders.

Introduction

The research and policy landscape for biodiversity conservation is changing. Biodiversity conservation is no longer solely a matter of nature protection; protected areas today are expected to make a broad range of contributions to human society including maintaining critical ecosystem "goods and services" such as water, food, carbon storage; mitigating climate change; alleviating poverty; and even providing opportunities for economic development (Watson *et al.* 2014). Conservation scientists who seek to protect biodiversity have called on one another to participate in a new social contract that supports "effective and just conservation" (IUCN 2014: 38) and to connect conservation science and practice through effective long-term collaborations with practitioners and other stakeholders (Pressey *et al.* 2007). International initiatives such as the IUCN-led Conservation Initiative on Human Rights, and the Aichi Targets associated with the Strategic Plan of the Convention on Biological Diversity (CBD) call for protected areas that are effectively and equitably managed taking into account the rights and needs of local and Indigenous peoples (Woodley *et al.* 2012). Additionally, scientists studying socioecological systems have argued the need for an international network to provide platforms for interdisciplinary, longitudinal, and comparative research to better understand human-environment relations at multiple scales (Liu *et al.* 2007). These initiatives and calls also reinforce the potential relevance of an existing and longstanding international network of sites dedicated to conserving biodiversity, demonstrating sustainable development, and conducting research and education—the World Network of Biosphere Reserves (BRs) of the United Nations Educational, Cultural and Scientific Organization (UNESCO).

Described as "groundbreaking" and "innovative" when first introduced (Bridgewater 2016), UNESCO's Man and

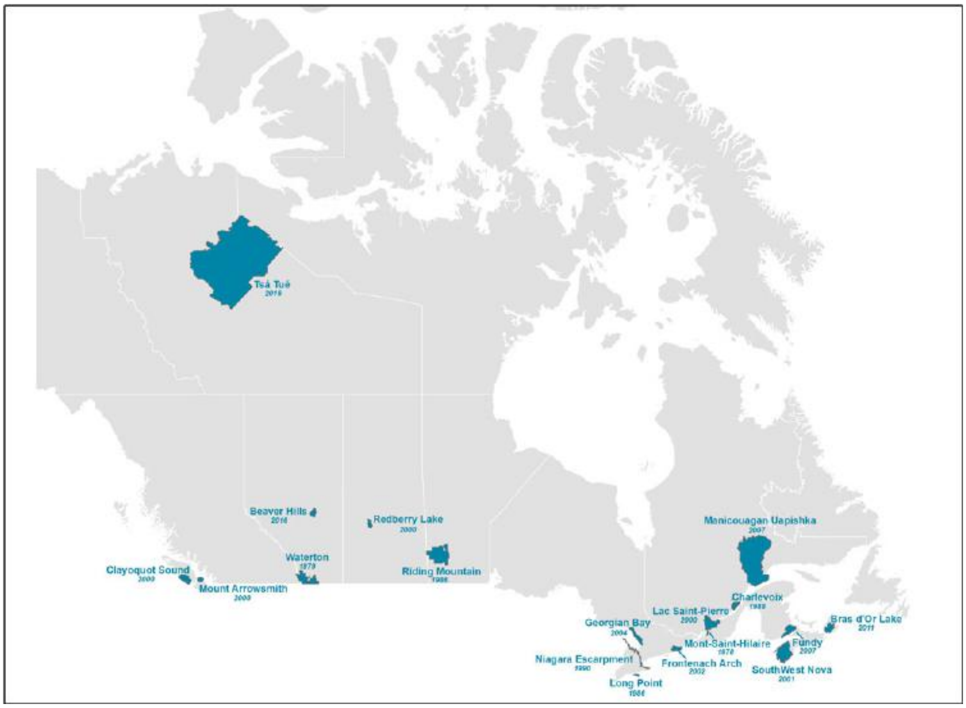

Figure 1 Location of Canada's 18 biosphere reserves (2016). Credit: Mont Saint-Hilaire Biosphere Reserve.

Biosphere (MAB) program was conceived as an interdisciplinary (involving physical, biological and social sciences), intergovernmental program of problem-based research and action focused on human-environment interaction. Its primary tool became BRs—representative ecological sites where environmental change could be monitored, policies or practices could be "tested," and lessons could be learned to inform environmental policy and management practice (Batisse 1982). Success was measured by the ability of sites to address local management priorities and share results across an international scientific network (Franklin 1977; Sokolov 1981). In 1995, the Statutory Framework for Biosphere Reserves included sustainable development as an official objective, involving the inclusion of local and Indigenous peoples and knowledge in research and management. Subsequent strategic plans reinforced these priorities. Indeed, BRs' stated objectives fit with the emerging consensus for conservation through protected areas that are equitably and effectively managed (Woodley *et al.* 2012; MacKinnon *et al.* 2015).

And yet, despite the 2016 distribution of BRs of 669 sites across 120 countries, the network operates in relative obscurity. Scientists, policy and decision-makers, and even local communities where they are situated, remain unaware of the purpose, activities and potential benefits of BRs. For example, a review of *Conservation Letters* from 2008 to 2015 reveals only nine articles that mention BRs. Of these, only two make passing reference to BRs as potential conservation tools (Eigenbrod *et al.* 2010; Tscharntke *et al.* 2015) and none focuses on BRs as key supports for better understanding of biodiversity conservation, sustainability or networked research. Indeed, their value as a network has been significantly underutilized (Reed & Egunyu 2013).

With a need to conduct action-based research that supports the conservation of biological diversity so pressing, and a ready network first formed by conservation scientists in the 1970s, why are BRs such minor notations in the research and action agendas of scientists and policymakers? How did this network evolve and what is its current mandate? How can this network be utilized more fully to address contemporary conservation issues? To address these questions, I provide a short history of the evolving philosophy of BRs internationally, explain their application in Canada, and consider the potential they offer to scientists, policy-makers, and local people united in the interests of conservation.

Methods

Canada was selected for more intensive scrutiny because of its key conceptual and logistical contributions to the MAB program and BR formation. Canadians served on the International Coordinating Council of MAB six separate times between 1970 and 1983. Canada was the first country to establish national procedures and nomination processes for BRs that were subsequently adopted elsewhere. Today, the country hosts 18 BRs—the largest, active national network with historical records available in English (Figure 1).

Data for this research included documentary materials archived at UNESCO (Paris, France), available online, and the Wilfrid Laurier Archives (Waterloo, Canada). Records of Canada's national program include the Francis Fonds (a collection of records now 50.6 m long), Roots Fonds, Birch Fonds, and Canadian BRs Association Fonds. Additionally, since 1995, BRs have been subject to periodic review approximately once every 10 years. The accompanying reports provide extensive information on the social and ecological characteristics of each BR, as well as the research, conservation, sustainability and public education programs undertaken during the review period. Only UNESCO documents cited directly are listed here; Supplementary Material provides a list of periodic review reports consulted and full citations for UNESCO documents mentioned in this article. Additionally, I draw on a 2011 survey of the then-15 BR managers to better understand how they interpreted their mandate. The survey was conducted at the beginning of a 4-year action research project designed to help Canadian BR managers become more effective through social learning and networking strategies (Reed *et al.* 2014). Managers came to their positions with backgrounds in natural and social science education, community organizing, and business. Sixteen interviews were also conducted in 2011 with past and present members of the Canadian-MAB committee. Former Canadian-MAB committee members included active and retired academic or government scientists, civil servants and private consultants. I have served as a member of the Canadian-MAB committee since 2010, allowing me to understand the connections between the international directives and national and local implementation. Three themes emerge from these diverse sources: (a) a programmatic evolution from research for biodiversity conservation to management for sustainability; (b) a shift away from selecting representative ecosystems as BR sites; and (c) an enlarged scope of activities for BR managers.

Results

The evolving mandate of BRs internationally

In 1968, an international "Biosphere Conference" in Paris sparked the creation of an international network of research sites that would examine ecological questions from the perspective of human use. BRs were first created in 1976, modeled philosophically and practically on the experience and expertise of North American and European scientists of the early and mid-20th Century. Russian zapovedniks, British conservancies, and American experimental forests (Bocking 2012) allowed for long-term field observations and experimentation and drove the intellectual foundation of the original primary objective of BRs – to establish sites for research related to biodiversity conservation.

In the mid-20th Century, conservation scientists such as Arthur Tansley (1945) and Aldo Leopod (1949) promoted conservation for scientific, moral, and aesthetic reasons, contributing to the second objective of BRs—to grapple with complex issues where humans are embedded in nature. The introduction of "big science" in the 1960s—particularly the International Biological Program—helped produce reliable ecological research at a global scale and raised awareness among scientists and citizens of the global extent of challenges at the human–environment interface (McCormick 1995). This experience contributed to the third objective of BRs—to build a scientific network to expand knowledge and action about the effects of human activities on or in the natural environment. Hence, BRs were created to "to safeguard the genetic diversity of species, . . . provide areas for ecological and environmental research, and provide facilities for education and training" (UNESCO 1974).

The evolution of BRs can be considered in two phases (Reed & Massie 2013). In Phase 1 (1974–1994), BRs were to be *representative* ecosystems based on an international classification of biogeographical provinces developed by Miklos Udvardy (1975). Although this ideal was never achieved, the focus on representativeness over uniqueness was aimed at understanding and redressing widespread environmental challenges across a diversity of landscapes rather than focusing on biological exceptions (Batisse 1982). The International Council of Scientific Unions (ICSU) (now the International Council for Science) endorsed this approach, identifying the value of BRs as field laboratories wherein scientific research could serve humanity and address regional and global problems.

The network was to include natural and seminatural ecosystems; individual sites were to have a strictly protected area at their core with zones of increasing human influence, thereby allowing for manipulative research (Figure 2). Related to this vision, scientists were encouraged to investigate the human-use system rather than the ecosystem, a concept that invited human–nature interaction and, importantly, encouraged the involvement of local people in learning how conservation and development might be reconciled (di Castri *et al.* 1980).

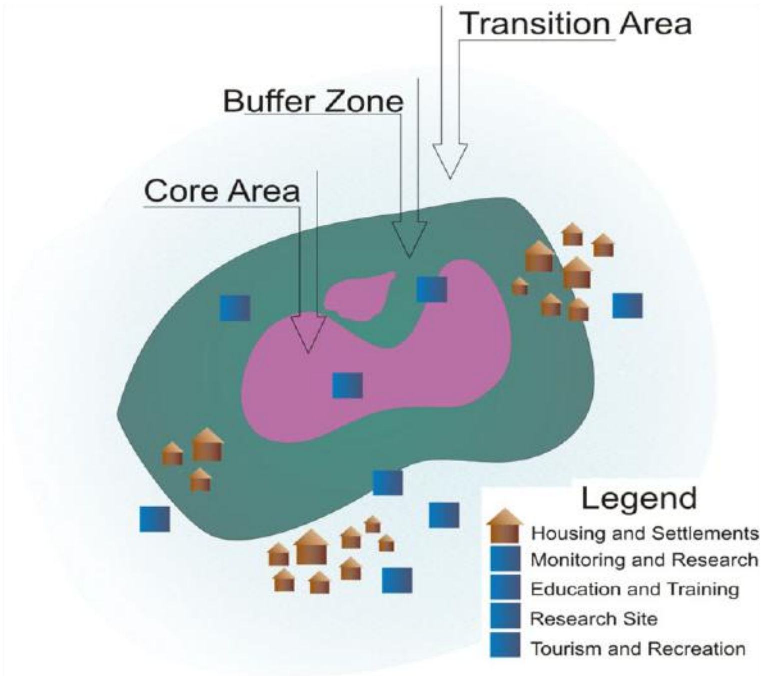

Figure 2 Classical configuration of a biosphere reserves. Credit Colleen George.

Some MAB scientists suggested that in Phase 1, BRs were also created to support a development function, although this function was not well-practiced (Batisse 1986). The development function became prominent in Phase 2 (1995 to present) once the Statutory Framework for BRs was introduced. The framework dictated that BRs become sites of excellence to demonstrate approaches to conservation *and* sustainable development. Hence, by 1995, the original objectives were translated into three official functions: biodiversity conservation, sustainable development, and logistical support for research and capacity building. The 2015 MAB Strategy and the 2016 Lima Action Plan suggest that BRs are to be model regions for sustainable development. These changes exemplify a gradual shift in orientation from "a research-driven to a management-driven program" (UNESCO 2007) and a broadening of focus from conservation science to sustainability science. This shift is also evident in Canadian practice.

Canadian experiences: Shifting away from representative ecosystems

In Canada, BRs do not have legal jurisdiction over lands, waters, or resources, or dedicated government funding, but work with public, private, and civic sectors to identify mutually beneficial research and action initiatives. In 1987, Canada adopted a national action plan that reinforced UNESCO's 1984 International Action Plan. Both called for systematic ecological representation as a criterion for designating BRs. Canada's plan was never implemented, yet the archival records show that the idea resurfaced periodically. In 2007, for example, BR practitioners roundly rejected a Parks Canada proposal based on ecological representation.

When asked "Do you think that the network of BRs should be representative of ecosystems like national parks?," 5 of 16 MAB committee interviewees responded "yes," 1 responded "maybe," and 9 responded "no." Three interviewees who said "yes" are still very active with BRs. All interviewees, however noted that the absence of BRs north of the 60th parallel was a "gap" in the network (a gap that was filled in 2016 with the designation of Tsú Tsé BR). This gap also featured in discussion documents that circulated across the network in 2007 and 2008. One, in particular, recommended that the Canadian BR Association continue the process of opportunistic planning, based on the coincidence of a local organizing committee with a plan to meet the UNESCO criteria such as diversity of ecosystem type and encouraging a geographic spread of BRs across the country and networking opportunities. This opportunistic approach remains today.

Canadian experiences: enlarging the scope of BRs

Research and monitoring once dominated Canadian BR functions. For example, McGill University Archives reported that scientists had completed 698 research outputs at Mont Saint-Hilaire BR between 1978 and 2006 (Reed 2009) and the original six BRs were part of the national Ecological Monitoring and Assessment Network (EMAN) of Environment Canada between 1994 and 2010 (Vaughn *et al.* 2001). Additionally, these BRs conducted a collective study of landscape change in their regions to inform knowledge users of the drivers of landscape change and develop a common database (Canada MAB 2000).

Yet, in the 2011 national survey of BR managers, respondents rated the importance and effectiveness of the conservation function lower than the sustainable development or logistical support functions (Table 1).

Only 53% (8/15) of respondents indicated *biodiversity conservation* and associated research was a strong or very strong priority and two-thirds saw themselves as effective in this regard. The other Phase 1 function, providing logistical support for research, was rated as a high priority in 10 BRs. However, fewer BR managers saw themselves as effective in this regard. Fostering economic development was a high or very high priority for 93% (14/15) of BRs; though only half rated themselves as effective. With respect to facilitating collaboration between organizations and generating regional awareness of BRs, 93% (14/15) rated these objectives as high priorities. Seventy-three percent (11/15) rated themselves as effective in facilitating collaboration and 40% (6/15) rated themselves as effective in generating regional awareness. Logistical support for monitoring and education was ranked as a high or very high priority in about two-thirds of BRs, with about one-third rating themselves as highly effective (Table 1). At least two-thirds of BRs in Canada are located in regions with significant resident Indigenous populations and/or traditional territories. The survey determined that Indigenous organizations participated in the events of eight BRs, and partnered with seven. Yet, only three of the then-15 BRs indicated that they maintained communication with Indigenous organizations about their activities and only two BRs reported having Indigenous representatives on their management boards.

Discussion: reframing conservation in UNESCO BRs

The ratings of importance and effectiveness in protecting biodiversity were lower than anticipated. Periodic review reports between 1998 and 2015 reveal that many Canadian BRs have longstanding conservation and research programs involving scientists and citizens including for amphibians and reptiles (e.g., Clayoquot Sound, Long Point, Georgian Bay), birds (Redberry Lake, Mont Saint Hilaire, Long Point), large mammals (Waterton, Riding Mountain), and forest ecosystems (Long Point, Mont Saint Hilaire). Some of these were documented in a compilation of "proven good practices" that was first shared across the national and international network in 2013 (Godmaire *et al.* 2013).

The low ratings, however, could be interpreted in a variety of ways. Respondents may view that the conservation mandate is taken up by organizations or agencies, such as Parks Canada, that have a legislative mandate for conservation in the core areas of BRs. Many of the activities in BRs are conducted in partnership with such organizations. BRs may have rated their effectiveness low because of the general difficulty in assessing how specific interventions affect long-term conservation success. Schultz *et al.* (2011), who conducted a similar survey internationally, suggested that low ratings of effectiveness in reaching conservation goals may be an indication that "management relying mainly on volunteer efforts is not sufficient in reaching the ambitious objectives of BRs." This concern may also be at play in Canadian BRs.

Another possible explanation is that conservation is no longer viewed as separable from economic development. For example, periodic review reports and the development of the best practices booklet revealed that the longstanding mammal conservation programs at Waterton and Riding Mountain and the forest corridor projects at Fundy and Mont Saint Hilaire involved extensive negotiations and debates among local peoples whose livelihoods depended on resource use. These same sources indicated that other BRs are facing difficult economic pressures such as human population declines from agricultural intensification (e.g., Redberry Lake, Riding Mountain) and development pressures from ex-urban growth (e.g., Mount Arrowsmith, Mont Saint Hilaire) or tourism (e.g., Georgian Bay, Niagara Escarpment). Respondents identified some projects involving associated land use conflicts as *economic* projects rather than conservation or sustainability projects. Hence, they may be not-so-subtly breaking away from the earlier label that they are primarily *environmental* organizations.

The high ratings of importance and effectiveness in stimulating and facilitating regional collaboration suggest that BR organizations are undertaking tasks as honest brokers in regional efforts to advance conservation and sustainable development. Indeed, the good practices guide provided examples from every contributing BR (Godmaire *et al.* 2013). Notably absent, however, was the reporting of systematic and regular collaboration with Indigenous peoples. Furthermore, awareness of, and adherence to, the 2008 Madrid Action Plan and international

Table 1 Self assessment of priorities and effectiveness across biosphere reserve objectives

	Priority		Effectiveness	
	Average rating of priority (*n*)	Number of BRs that rated this priority high (4–5)	Average rating of effectiveness (*n*)	Number of BRs that rated their effectiveness high (4–5)
Biodiversity conservation	3.8 (14)	8	3.7 (13)	10
Logistical support for research	3.5 (15)	10	3.2 (13)	6
Logistical support for monitoring	3.2 (14)	8	3.2 (13)	4
Logistical support for education	3.7 (15)	9	3.2 (12)	5
Fostering economic development	4.1 (13)	10	4.1 (13)	10
Fostering social and cultural development	2.2 (15)	1	2.4 (12)	3
Fostering dialogue within organization	3.7 (14)	7	3.6 (13)	7
Facilitating collaboration between organizations	4.6 (15)	14	4.2 (13)	11

Note: 1 = very low priority/effectiveness; 2 = low priority/effectiveness; 3 = medium priority/effectiveness; 4 = high priority/effectiveness; 5 = very high priority/effectiveness.
Total number of possible responses = 15 (number actually responding to the question).
The dotted horizontal line separates "first generation" (before 1995) from "second generation" priorities (1996 to present).

protocols for biodiversity protection were highly variable, with strongest connections made by BRs located in Québec and those that had recently been subject to a periodic review.

It is difficult to tell if Canada is representative of BRs around the world, but it does reflect UNESCO's observed shift from a science to a management agenda. Two key points demonstrate this shift internationally and in Canada. First, the commitment to biodiversity conservation has become embedded within a broader sustainable development mission. At the international level, the initial draft Strategic Action Plan, released February 2015, set its first strategic objective for BRs to, "conserve biodiversity, maintain ecosystem services and foster the sustainable use of natural resources" (UNESCO 2015). However, in the revised plan, (released February 2016), conservation became subsumed under Strategic Action Area A in which "BRs [are to be] recognized as models contributing to the implementation of sustainable development goals (SDGs) and multilateral environmental agreements (MEAs)" (UNESCO 2016). In the final plan, approved March 2016, the associated actions include contributing to implementing MEAs, with explicit reference to the Aichi Targets of the CBD and establishing alliances for biodiversity conservation and benefits to local people.

Second, the goal of selecting representative ecosystems as sites for BRs no longer prevails. Although never realized, this transition in aspiration began with the 1995 Seville Strategy that suggested that BRs "promote a comprehensive approach to biogeographical classification that takes into account such ideas as vulnerability analysis, in order to develop a system encompassing socio-ecological factors" (UNESCO 1996). The Madrid Action Plan contained only one target for designation – that individual BRs must engage in open and participatory procedures. This theme pervades the 2016 Action Plan.

The "opportunistic" or "grassroots" approach now adopted in Canada appears synchronized with international requirements. Eleven of the 18 BRs have national parks as part of their core areas. Although Canada has had an ecologically based systems plan for establishing national parks since 1971, its application has been criticized as simplistic and incomplete (Wright & Rollins 2009). Furthermore, Canada has no national biodiversity strategic plan to guide site selection for the protection of biodiversity (MacKinnon *et al.* 2015). Since at least 2010, new BR applications in Canada have been judged by their community commitment and governance arrangements rather than their ecological representation. Without equal consideration to ecological representation, BRs will lose the potential to serve as reference sites for understanding the biogeography of biodiversity change or loss.

Canadian BRs today are active organizations engaged in collaborative management for regional sustainability but they face significant challenges to operating as a national network including the lack of core funding, large geographic distance and multiple time zones, and cultural differences. However, their engagement in a national partnership initiative helped them identify common concerns, systematically evaluate and practices related to the provision of ecosystem goods and services, sustainability education, and sustainable tourism (Reed *et al.* 2014). Canadian BRs raised their international profile in 2013 when they hosted the EuroMAB conference, showcased the outcomes of their partnership and demonstrated leadership in two key international networks—NORDMAB (a consortium of northern countries with BRs) and the Working Group for Indigenous Peoples. NORDMAB has been raising awareness of climate change by participating in programs such as "Students on Ice" (http://studentsonice.com/) while the Working Group has consistently encouraged BRs to take more action related to climate change and the CBD (e.g., McDermott *et al.* 2015). Both the Canadian and international networks can offer platforms for networked governance, research, training, and colearning among scientists and community members. Their contribution to international initiatives for effective and equitable conservation and sustainability initiatives, however, requires strengthening their connection with international priorities and programs.

In March 2016, the 4th World Congress of BRs convened in Lima, Peru and approved the Action Plan for 2016–2025. To shift from conservation inaction to conservation *in* action, BR networks across Canada and internationally, must address three interrelated challenges:

(1) Demonstrate tangible contributions of BRs to conservation and sustainability among researchers, citizens, private sector interests, Indigenous peoples, and public agencies. For Canadian and international BRs, grappling with a conservation agenda more directly will require BRs to reconnect more effectively with global concerns and initiatives such as long-term monitoring, the CBD and the SDGs. For Canadian BRs, this also will mean renewed federal leadership to translate international objectives into national targets and local action (see Lemieux *et al.* 2011). These connections can then be used as a platform to raise the profile of the program both within and beyond the scientific community.

(2) Establish appropriate, reliable, and active partnerships that retain action research agendas. Bridgewater (2016) documented several missed opportunities at the international level for BRs to connect with other initiatives such as World Heritage Sites and the CBD while Hadley observed that MAB has suffered from being situated "in an institution structured on program sectors based on *nineteenth-century* disciplinary lines." (Hadley 2006). The 2016 plan appears poised to break down the walls of sectoral governance in UNESCO, while the emergence of sustainability science in academia suggests that biologists will likely have to work on transdisciplinary research teams to maintain the conservation agenda. Action for researchers may include renewing the role of BRs as monitoring sites for ecosystem change and governance approaches, and undertaking networked research programs; for practitioners it may mean developing uncommon allies to secure funding. The greatest financial stability for Canadian BRs has come to those operating as social enterprises with a clear mission, new partners, focused governance structure and a service-oriented delivery model (George & Reed 2016). The international network has adopted this approach in the Lima Action Plan; success may rely on its widespread application and ability to meet its targets for broadening engagement and outreach.

(3) Engage knowledge-holders with specialized western scientific knowledge, and local experiential and Indigenous knowledge might be brought into productive conversation towards achieving mutually desirable conservation and sustainability goals (e.g., Haenn 2014). Partnering with, and learning from, other initiatives such as the IUCN-led Conservation Initiative on Human Rights and/or Indigenous Peoples' and Community Conserved Territories and Areas offers such opportunities.

Through the BR network, there is an opportunity to marry a critical research need with existing platforms to promote research that is scientifically sound and socially desirable. Conservation scientists, with their extensive experience of national and international conservation practice, can enhance the mission of BRs to advance knowledge and action towards a sustainable and socially beneficial biosphere without compromising the conservation agenda. Reconnecting with the mission and practices of BRs can be of mutual benefit—offering scientists unparalleled access to research sites and a global network across which lessons can be shared, while breathing new life into the network. By addressing these challenges, BRs can shift once again from a "nice to know" program into a "need to have" model for understanding socioecological systems (Liu *et al.* 2007), connecting conservation science and practice (Pressey *et al.* 2007), and demonstrating effective and just conservation (IUCN 2014).

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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:

UNESCO and Canadian MAB Committee Documents Described and Used in the Text

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POLICY PERSPECTIVE

Formulating Smart Commitments on Biodiversity: Lessons from the Aichi Targets

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Abstract

The world is currently not on course to achieve most of the Convention on Biological Diversity's Aichi Targets to address biodiversity loss. One challenge for those implementing actions to achieve them may be the complexity and lack of clarity in the wording of the targets, which also make it difficult to stimulate and quantify progress. Drawing on experience in developing and measuring indicators to assess progress toward targets, we identify four key issues: ambiguity, quantifiability, complexity, and redundancy. The magnitude of required commitments under some targets is rendered ambiguous by the use of imprecise terms (e.g., "substantially"), while many targets contain poorly defined operational terms (e.g., "essential services"). Seventy percent of targets lack quantifiable elements, meaning that there is no clear binary or numeric threshold to be met in order for the target to be achieved. Most targets are excessively complex, containing up to seven different elements, while one-third of them contain redundancies. In combination, these four issues make it difficult to operationalize the targets and to ensure consistent interpretation by signatories. For future policy commitments, we recommend the adoption of a smaller number of more focused headline targets (alongside subsidiary targets) that are specific, quantified, simple, succinct, and unambiguous.

Introduction

In 2002, world governments adopted a global commitment to address biodiversity loss through the Convention on Biological Diversity (CBD), setting themselves a target "to achieve by 2010 a significant reduction of the current rate of biodiversity loss" (CBD 2003). This "2010 target" was also incorporated into the United Nations Millennium Development Goals (United Nations 2008). By 2010, it was widely accepted that the world had failed to achieve the target (Butchart *et al.* 2010; CBD 2010a).

Recognizing that the condition of biodiversity is influenced by multiple pressures and underlying drivers that must be counteracted by diverse policy responses, the CBD adopted a more sophisticated approach for the decade following 2010, developing a Strategic Plan on Biodiversity that included 20 Aichi Targets (CBD 2010b). Halfway toward the end-date for achieving these targets, it is clear that despite accelerating policy and management responses, trends in the state of biodiversity are unlikely to improve by 2020 without both a substantial scaling up and refocussing of efforts (Tittensor *et al.* 2014; CBD 2014) and a better consideration of the synergies and trade-offs in achieving multiple targets (Perrings *et al.* 2010; Di Marco *et al.* 2016a). Moreover, the articulation and specification of the Aichi Targets themselves may also constitute an additional challenge for those implementing actions to achieve them. It is likely that further targets will be set after 2020 and after the United

Nation's Sustainable Development Goals (SDGs) expire in 2030; (United Nations General Assembly 2015). What, therefore, can we learn from the wording of the Aichi Targets to ensure that future targets are formulated more effectively?

The shortcomings of the Aichi Targets

The Aichi Targets are, in some respects, a vast improvement over the 2010 Biodiversity Target. For example, as well addressing the state of biodiversity, they also focus on pressures on biodiversity, underlying drivers, policy responses, and integration of biodiversity issues across sectors. However, we argue that they have a number of shortcomings. We draw on our experience in attempting to identify indicators with which to measure progress against biodiversity targets, including in relation to the Streamlining European Biodiversity Indicators initiative (European Environment Agency 2012), the Aichi Targets through two CBD Ad Hoc Technical Expert Groups on indicators (CBD 2004, 2015), the SDGs (Sustainable Development Solutions Network 2015), and the Ramsar Strategic Plan (Convention on Wetlands 2015). Identifying meaningful and effective indicators requires forensic analysis of the wording of targets and their meaning, from which we have drawn some of the insights covered below.

Similarly, we also draw on efforts to synthesize evidence across multiple indicators to quantify progress in achieving such targets (Butchart *et al.* 2010; Juffe-Bignoli *et al.* 2014; Tittensor *et al.* 2014), which poses similar challenges. We, therefore, attempt to identify problems with the Aichi Targets that may hinder their ease of understanding and interpretation, as well as their measurability and intercomparability (between countries and targets), leading to ineffective efforts to identify and implement the actions they are intended to stimulate.

While there have been previous general calls for smarter, less vague environmental targets with greater quantification (Perrings *et al.* 2010; Stafford-Smith 2014; Maxwell *et al.* 2015), we provide the first detailed analysis of each element in each of the 20 Aichi Targets (Table 1). We define "elements" as clauses or components of the targets that address different aspects of the status of biodiversity, threats to it, or actions needed for it, or that require very different indicators or datasets to monitor progress toward their achievement. We argue that the Aichi Targets would be more effective if they contained fewer elements, ambiguities, redundancies and unnecessary complications, were less complex, and contained more quantification. We then propose some general recommendations for future target setting.

Ambiguously worded

Some of the targets contain wording that is difficult to interpret because of its ambiguity. For example, caveats like "as appropriate" (target 2) and "where feasible" (target 5) render the target so subjective that individual Parties could defend almost any action and outcome as being sufficient, greatly weakening the value of their commitment. Similarly, the magnitude of required commitments under some targets is rendered ambiguous by the use of imprecise terms such as "significantly" (target 5), "substantially" (target 20), "minimized" (target 10), or by language such as "taken steps to achieve" (sustainable production and consumption, in target 4). Such ambiguities make it impossible to define and quantify what achievement of these targets would comprise, and they make it difficult for Parties to ensure consistency of response.

Some terms used in the targets remain undefined and can be interpreted in different ways within different scientific contexts or by different Parties, making it difficult to measure global target achievement. For example, in relation to target 11, there are multiple approaches to defining what comprises an "ecologically representative" protected area system (Watson *et al.* 2016) and a "well-connected" system (Fischer and Lindenmayer 2007). Similarly, "safe ecological limits" (in relation to production and consumption systems other than fisheries; target 4), "areas of importance for biodiversity and ecosystem services" and "other effective area-based conservation measures" (target 11), and "degraded ecosystems" and "restoration" (target 15) are not easily defined. For the some of these, work is underway to reach consensus (Watson *et al.* 2016). For example, IUCN has recently established a Task Force to develop guidance on the definition of "other effective area-based conservation measures." Similarly, many of the approaches to identify "areas of importance for biodiversity," such as Important Bird and Biodiversity Areas (BirdLife International 2014) and Alliance for Zero Extinction sites (Ricketts *et al.* 2005) have now been brought together under a single umbrella with the development of unified standard for the identification of Key Biodiversity Areas (IUCN 2016), while potential protected areas to conserve biodiversity in the marine realm have been identified worldwide through delineation of "Ecologically or Biologically Significant Areas" (CBD 2009). However, while the inclusion of such terms in the Aichi Targets has stimulated new work to advance biodiversity conservation, when target wording is not carefully defined, it is likely that different Parties will use different definitions and interpretations, with the probable outcome of less coherent global conservation responses than would be achieved otherwise (Di Marco *et al.* 2016b).

Table 1 The multiple elements and shortcomings of the Aichi Biodiversity Targets

Target	Elements	Quantifiable	Ambiguities	Unnecessary complexities	Redundancies
1. By 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably.	People are aware of the values of biodiversity	No			
	People are aware of [. . .] the steps they can take to conserve and use it sustainably	No			
2. By 2020, at the latest, biodiversity values have been integrated into national and local development and poverty reduction strategies and planning processes and are being incorporated into national accounting, as appropriate, and reporting systems.	Biodiversity values have been integrated into national and local development and poverty reduction strategies	No	"As appropriate"		
	Biodiversity values have been [. . .] integrated into national and local planning processes	No			
	Biodiversity values have been [. . .] integrated into national [. . .] reporting systems	No			
	Biodiversity values have been [. . .] integrated into reporting systems	No			
3. By 2020, at the latest, incentives, including subsidies, harmful to biodiversity are eliminated, phased out, or reformed in order to minimize or avoid negative impacts, and positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the convention and other relevant international obligations, taking into account national socioeconomic conditions.	Incentives, including subsidies, harmful to biodiversity are eliminated, phased out, or reformed in order to minimize or avoid negative impacts	No		"consistent and in harmony with the convention and other relevant international obligations, taking into account national so-cioeconomic conditions"	
	Positive incentives for the conservation and sustainable use of biodiversity are developed and applied, consistent and in harmony with the convention and other relevant international obligations, taking into account national socioeconomic conditions.	No			
4. By 2020, at the latest, Governments, business, and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption and have kept the impacts of use of natural resources well within safe ecological limits.	Governments, business, and stakeholders at all levels have taken steps to achieve or have implemented plans for sustainable production and consumption	No	"Taken steps to achieve or have implemented plans for"; "safe ecological limits."		"Taken steps to achieve" = "have implemented plans for"
	Governments, business, and stakeholders at all levels [. . .] have kept the impacts of use of natural resources well within safe ecological limits.	No			
5. By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.	The rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero	Yes (numeric)	"Where feasible"; "significantly reduced"		"including forests" is encompassed by "all natural habitats"
	Degradation and fragmentation [of natural habitats] is significantly reduced	No			

Continued

Table 1 Continued

Target	Elements	Quantifiable	Ambiguities	Unnecessary complexities	Redundancies
6. By 2020, all fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally, and applying ecosystem-based approaches, so that overfishing is avoided, recovery plans and measures are in place for all depleted species, fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems, and the impacts of fisheries on stocks, species, and ecosystems are within safe ecological limits.	All fish and invertebrate stocks and aquatic plants are managed and harvested sustainably, legally, and applying ecosystem-based approaches	No	"Safe ecological limits"		"Fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems" is encompassed by "the impacts of fisheries on stocks, species, and ecosystems are within safe ecological limits"
	Overfishing is avoided	No			
	Recovery plans and measures are in place for all depleted species	No			
	Fisheries have no significant adverse impacts on threatened species and vulnerable ecosystems	No			
	The impacts of fisheries on stocks, species, and ecosystems are within safe ecological limits	No			
7. By 2020, areas under agriculture, aquaculture, and forestry are managed sustainably, ensuring conservation of biodiversity.	Areas under aquaculture [. . .] are managed sustainably	No			
	Areas under agriculture [. . .] are managed sustainably	No			
	Areas under forestry [. . .] are managed sustainably	No			
8. By 2020, pollution, including from excess nutrients, has been brought to levels that are not detrimental to ecosystem function and biodiversity.	Pollution [. . .] has been brought to levels that are not detrimental to ecosystem function and biodiversity.	No			"Including from excess nutrients" is encompassed by "pollution"
	Pollution [. . .] from excess nutrients should be brought to levels that are not detrimental to ecosystem function and biodiversity	No			
9. By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment.	Invasive alien species are identified and prioritized	No			
	Invasive alien [. . .] pathways are identified and prioritized	No			
	Priority [invasive] species are controlled or eradicated	No			
	Measures are in place to manage pathways to prevent their introduction and establishment	No			
10. By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning.	The multiple anthropogenic pressures on coral reefs [. . .] are minimized, so as to maintain their integrity and functioning	No	"Vulnerable ecosystems impacted by climate change or ocean acidification"; "minimized"		"Or ocean acidification"

Continued

Table 1 Continued

Target	Elements	Quantifiable	Ambiguities	Unnecessary complexities	Redundancies
	The multiple anthropogenic pressures on [. . .] other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning	No			
11. By 2020, at least 17% of terrestrial and inland water, and 10% of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well-connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes.	At least 10% of coastal and marine areas [. . .] are conserved	Yes (numeric)	"Areas of particular importance for biodiversity and ecosystem services"; "other effective area-based conservation measures"; "integrated into the wider landscapes and seascapes"		
	At least 17% of terrestrial and inland water areas [. . .] are conserved	Yes (numeric)			
	[. . .] Areas of particular importance for biodiversity and ecosystem services, are conserved	No			
	[Areas are conserved through] ecologically representative . . . protected areas and other effective area-based conservation measures	No			
	[Areas are conserved through] effectively and equitably managed . . . protected areas and other effective area-based conservation measures	No			
	[Areas are conserved through] well-connected systems of protected areas and other effective area-based conservation measures	No			
	Areas are conserved [. . .]and integrated into the wider landscapes and seascapes	No			
12. By 2020, the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained.	The extinction of known threatened species has been prevented	Yes (binary)			
	The conservation status [of known threatened species, particularly of those most in decline] has been improved and sustained	Yes (binary)			
13. By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socioeconomically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.	The genetic diversity of cultivated plants [. . .] is maintained	No			
	The genetic diversity of [. . .] farmed and domesticated animals [. . .] is maintained	No			
	The genetic diversity of [. . .] wild relatives [. . .]is maintained	No			
	The genetic diversity of [. . .] socioeconomically as well as culturally valuable species, is maintained	No			
	[. . .] Strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity	No			

Continued

Table 1 Continued

Target	Elements	Quantifiable	Ambiguities	Unnecessary complexities	Redundancies
14. By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods, and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable.	Ecosystems that provide essential services, including services related to water, and contributing to health, livelihoods, and wellbeing, are restored	No	"Essential" services	"Taking into account the needs of women, indigenous and local communi-ties, and the poor and vulnerable"	"Including services related to water, and contribute to health, livelihoods, and well-being"
	Ecosystems that provide essential services, including services related to water, and contributing to health, livelihoods, and wellbeing, are safeguarded	No			
15. By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15% of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification.	Ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration [. . .] thereby contributing to climate change mitigation and adaptation and to combating desertification	No	"Enhanced"; "degraded ecosystems"; "restoration"		
	[. . .] Including restoration of at least 15% of degraded ecosystems [. . .]	Yes (numeric)			
16. By 2015, the Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force and operational, consistent with national legislation.	The Nagoya Protocol on Access to Genetic Resources and the Fair and Equitable Sharing of Benefits Arising from their Utilization is in force [by 2015]	Yes (binary)		"Consistent with national legislation"	
	The Nagoya Protocol [. . .] is operational [and] consistent with national legislation [by 2015]	Yes (binary)			
17. By 2015, each party has developed, adopted as a policy instrument, and has commenced implementing an effective, participatory and updated national biodiversity strategy and action plan.	Each Party has developed[. . .] an effective, participatory, and updated national biodiversity strategy and action plan (NBSAP)	Yes (binary)	"Effective"		
	Each party has [. . .] adopted as a policy instrument [. . .] an effective, participatory, and updated national biodiversity strategy and action plan (NBSAP)	Yes (binary)			
	Each party has commenced implementing [. . .] an effective, participatory, and updated national biodiversity strategy and action plan (NBSAP)	Yes (binary)			

Continued

Table 1 Continued

Target	Elements	Quantifiable	Ambiguities	Unnecessary complexities	Redundancies
18. By 2020, the traditional knowledge, innovation, and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are respected, subject to national legislation and relevant international obligations, and fully integrated and reflected in the implementation of the convention with the full and effective participation of indigenous and local communities, at all relevant levels.	The traditional knowledge, innovations and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are respected, subject to national legislation and relevant international obligations [...] at all relevant levels.	No		"At all relevant levels"	"Innovations and practices" is encompassed by "knowledge" "and reflected" is encompassed by "integrated"
	The traditional knowledge, innovations, and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, are [...] fully integrated and reflected in the implementation of the convention [...] at all relevant levels.	No			
	The traditional knowledge, innovations, and practices of indigenous and local communities relevant for the conservation and sustainable use of biodiversity, and their customary use of biological resources, [are respected, integrated, and reflected] with the full and effective participation of indigenous and local communities, at all relevant levels.	No			
19. By 2020, knowledge, the science base, and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared and transferred, and applied.	The science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are improved, widely shared, and transferred [. . .]	No			
	The science base and technologies relating to biodiversity, its values, functioning, status and trends, and the consequences of its loss, are [. . .] applied.	No			
	Information and tools at the disposal of policy is applied	No			

Continued

Table 1 Continued

Target	Elements	Quantifiable	Ambiguities	Unnecessary complexities	Redundancies
20. By 2020, at the latest, the mobilization of financial resources for effectively implementing the Strategic Plan for Biodiversity 2011–2020 from all sources, and in accordance with the consolidated and agreed process in the strategy for resource mobilization, should increase substantially from the current levels. This target will be subject to changes contingent to resource needs, assessments to be developed and reported by Parties.	The mobilization of financial resources for effectively implementing the Strategic Plan for Biodiversity 2011–2020 from all sources, and in accordance with the consolidated and agreed process in the Strategy for resource mobilization, should increase substantially from the current levels [...]	No	"Substantially"; "current levels"	"Consolidated and." "This target will be subject to changes contingent to resource needs assessments to be developed and reported by Parties."	

In other cases, the intended meaning can be presumed, but clearer wording would be helpful to avoid potential ambiguity. For example, "safe ecological limits" (under target 4) in relation to fisheries may refer to "limit reference points" (maximum values of fishing mortality or minimum values of the biomass which must not be exceeded in order to ensure harvests are sustainable; Cadima 2003). Similarly, under target 11, "areas of particular importance for biodiversity and ecosystem services" presumably refers to areas that are important for at least one of these features, rather than being restricted to areas that are important for both. The latter interpretation would be highly problematic and potentially inherently contradictory given that some essential ecosystem services (e.g., timber extraction or fishing) can have a negative influence on biodiversity if the levels of extraction are unsustainable (Mace *et al.* 2012). Indeed for some services associated with particular biodiversity features (e.g., coastal defense by coral reefs), areas of high importance for biodiversity (e.g., where the habitats are most intact and richest in diversity) do not generally coincide with the areas of high ecosystem service value (i.e., adjacent to the largest coastal populations and associated infrastructure) (Mora *et al.* 2011).

Target 10 is perhaps the most problematic to interpret: "the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized." The vast majority of ecosystems (and conceivably all of them) will be impacted by climate change, so most could be argued to be vulnerable, which, in combination with the all-encompassing "multiple anthropogenic pressures" to be minimized, arguably means that achievement of target 10 requires addressing the entire biodiversity crisis, and almost all elements of the other 19 targets. Furthermore, ocean acidification is a consequence of greenhouse gas emissions and climate change, so singling it out for extra emphasis is somewhat confusing.

Unquantifiable

A major difficulty with the wording of the Aichi Targets is that most (14 of 20) lack quantified elements (Table 1), meaning that there is not a clear, binary or numeric, threshold to be met in order for the target to be achieved. Without such clarity and quantification, it is difficult to determine progress toward targets (Stafford-Smith 2014). Three targets contain explicit numeric thresholds for at least some of their elements: habitat loss is "at least halved" (target 5), conservation of "at least 17% of terrestrial and inland water, and 10% of coastal and marine areas" (target 11), and "restoration of at least 15% of degraded ecosystems" (target 15). A further three targets contain clear binary thresholds that can objectively be met or not, for example, "the extinction of known threatened species has been prevented" (target 12), "the Nagoya Protocol . . . is in force" (target 16), and each Party has "commenced implementing . . . a national biodiversity strategy and action plan" (target 17). However, only two targets (16 and 17) have all their elements quantifiable, and these are both measurement of human responses rather than underlying biodiversity status or pressures. This lack of quantifiability for most targets proved a major difficulty when reviewing progress at the midpoint of the Aichi target's lifespan (Tittensor *et al.* 2014; Secretariat of the Convention on Biological Diversity 2014). It should be noted that for some elements of some targets, quantification may not require a particular number to be specified in the target text itself. For example, "safe ecological limits" for fisheries (under target 4) should be quantified for each fishery individually, as a universal value across all fisheries would be meaningless.

A related issue is the availability of indicators. While adoption of targets can stimulate development of indicators to meet measurement needs, it would seem ineffective to create a target for which indicators are presently unavailable and unlikely to be developed. Tittensor *et al.* (2014) found that indicators suitable for assessing progress were unavailable for 23 elements across 12 of the Aichi targets. The availability of relevant existing indicators and the feasibility of developing new ones must be borne in mind when formulating future biodiversity targets.

We acknowledge the fact that it is difficult at present to determine objectively a meaningful number for some aspects of some targets, for example, the level of habitat connectivity required for a protected area network, or the degree of ecological integrity needed to maintain essential ecosystem services. This is closely connected to the difficulty in finding universal measures of these elements. Nevertheless, it would be more coherent and efficient to adopt a standard approach with a common aim rather than leaving this to the interpretation of individual Parties or to those scientists attempting to quantify progress in achieving the relevant targets. At the same time, it is important to be explicit about the basis of the quantification: for the Aichi targets, these are largely politically rather than scientifically derived, and in some cases may only partly achieve the overall aspiration to achieve sustainable development and to halt or significantly reduce biodiversity declines. A stronger scientific basis for the values adopted in future quantified targets is desirable.

Excessively complex

Although there are 20 targets, most have multiple elements, each of which requires different actions to address and indicators with which to measure progress. Only one target has a single element: target 20 on increasing the mobilization of financial resources for effectively implementing the CBD strategic plan. Other targets typically contain two or three elements, with an overall mean of 2.8 elements per target (Table 1). Target 11 is particularly complex, having at least seven distinct elements (some of which arguably could be subdivided further; Table 1). With so many elements, it is not straightforward to identify the actions and solutions required to achieve the target as a whole, nor to develop indicators for measuring progresses toward its achievement (Juffe-Bignoli *et al*. 2014; Watson *et al.* 2016). Indeed, no single indicator is able to incorporate all the seven elements of target 11, with some of the elements (e.g., the percentage area targets) being arguably easier to measure than others, and some elements being objectively difficult to quantify (e.g., "integration with the wider landscape and seascape"). Furthermore, it is unclear how many elements need to be met before the target is considered to have been achieved: arguably, all of them. We suggest that it would be more effective to have headline targets that are less complicated, with different elements separated out into specific subsidiary targets.

Containing redundancies and unnecessary complications

The wording of one-third of the targets could be shortened and simplified without changing their meaning because of redundancies within the text. For example, "all natural habitats, including forests" (target 5), and "pollution, including from excess nutrients" (target 8). If particular emphasis is to be placed on a particular aspect, then these may be better addressed in a separate target. Similarly, "implemented plans for" is arguably encompassed by, or at least confounded with, "taken steps to achieve" (target 4) while ecosystems that "contribute to health, livelihoods, and well-being" are arguably a subset of those that "provide essential services" (target 14).

Several of the targets contain text that would arguably be better placed in background documentation and guidance. For example, under target 3, positive incentives are to be developed and applied "consistent and in harmony with the convention and other relevant international obligations, taking into account national socioeconomic [sic] conditions." However, these riders arguably apply to all targets, not just this one. This is also true for the text "taking into account the needs of women, indigenous and local communities, and the poor and vulnerable" (in relation to safeguarding ecosystems that provide essential services under target 14), and "consistent with national legislation" (in relation to the Nagoya Protocol being operational under target 16). The final clause of target 20 ("this target will be subject to changes contingent to resource needs assessments to be developed and reported by Parties") is an explanatory caveat that arguably belongs in the preamble, not the target. We are not the first to call for less ambiguous and more quantified environmental targets (Stafford-Smith 2014; Maxwell *et al.* 2015), but our more detailed target-by-target analysis also highlights the unnecessary complexities, redundancies, and complications in the structure and wording of the Aichi Targets. While these are potentially not as problematic as the other issues, they nonetheless reduce the ease with which the targets are interpreted and communicated, and hence may impact the degree to which they are adopted and applied.

Lessons for future target setting

The wording of biodiversity targets are typically negotiated in intergovernmental policy fora through protracted and tortuous discussions (Maxwell *et al.* 2015). This renders them susceptible to the introduction of redundancies, complications, ambiguities, and contradictions, and to the inclusion of references reflecting the agendas of particular groups, whether their focus is forests, fisheries, water, indigenous peoples, or other aspects.

In constructing targets to address biodiversity loss in future, efforts should be made to keep target language as simple and succinct as possible, using background documents, guidance, and preamble text to cover explanations, definitions, and caveats rather than incorporating these into the wording. In addition, targets should be worded as specifically as possible (the "S" in the mnemonic acronym "SMART," which is often used in relation to targets; Doran 1981), and with quantified components as far as possible (Stafford-Smith 2014). This makes the magnitude of required actions unambiguous and transparent. These considerations should be revisited throughout the process of constructing future targets to ensure that they are reflected in the final wording adopted.

It is critical that national biodiversity strategies and action plans (which set out CBD Parties' plans for implementing the actions needed to achieve the 20 Aichi Targets) take into account the potential synergies and trade-offs between targets (Stafford-Smith 2014; Di Marco *et al.* 2016a). For example, actions to expand terrestrial protected area coverage (target 11) could also contribute to reducing habitat loss (target 5) avoiding extinctions (target 12), and maintaining carbon stocks (target 15; Di Marco *et al.* 2016a).

We suggest that there may be merit in selecting a smaller number of more focused headline targets, alongside specific subsidiary targets capturing other elements. The former might highlight a set of specific actions, which if implemented in full, could together produce a major reduction in the rate of biodiversity loss. For example, ambitious, specific, quantified targets to reduce deforestation and wetland degradation, increase the sustainability of fisheries, minimize agricultural expansion, tackle invasive alien species, increase the scale and effectiveness of protected areas (and their coverage of important sites for biodiversity and large areas of intact habitat such as primary forest), address ocean acidification, recover threatened species, and augment financing. This set of headline targets could be sufficiently focused as to concentrate efforts while being adequately broad in impact as to advance biodiversity conservation substantially. They could be underpinned by more specific subsidiary targets covering the other aspects and elements of the Aichi Targets.

In conclusion, we suggest that future biodiversity targets should be specific, simple, succinct, quantified, unambiguous, relatively few in number, and set through a process involving greater collaboration between scientists and policy makers. Ultimately, however, the success of such targets in stimulating effective action to tackle the biodiversity crisis, as with the Aichi Targets and relevant SDGs, will be largely determined by the degree to which progress or lack thereof is transparent, and the degree to which national governments prioritize the needs of nature and of future generations of people over short-term aspirations.

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POLICY PERSPECTIVE

"As Far as Possible and as Appropriate": Implementing the Aichi Biodiversity Targets

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Keywords
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Abstract

Past shortfalls to meet global biodiversity targets have simultaneously prompted questions about the relevance of global environmental conventions, and sparked renewed ambition, for example, in the form of the Aichi Biodiversity Targets. While progress toward the Aichi Targets through the Convention on Biological Diversity is well-documented globally, less is known at the national level. We conducted a systematic content analysis of 154 documents to assess the nature and extent of national implementation of the Aichi Targets using Canada as a case study. Results indicate that most responses are aspirational, with only 28% of responses implemented. Implemented responses tend to be associated with targets with specified levels of ambition that emphasize biophysical values, or targets that are relatively straightforward to achieve in this context (e.g., knowledge capacity and awareness). In contrast, targets focused on equity, rights, or policy reform were associated with fewer actions. Implementation of this latter class of targets is arguably stalled not solely because of a lack of effective target design, but because of lack of fit within existing institutional commitments. This suggests that solutions—in terms of improving implementation—lie not only in overcoming known dilemmas of quantifiability, but also in fostering institutional transformation.

Introduction

From the Rio Conventions, including the Convention on Biological Diversity (CBD) to the Millennium Development Goals (SDG), and the recent Sustainable Development Goals, the use of targets in global environmental governance (Harrop & Pritchard 2011; Campbell *et al.* 2014; Velazquez Gomar 2014) and international development (Roberts 2005; Le Blanc 2015) has risen markedly in recent decades. Proponents argue that measurable and time-bound objectives are essential to meet the commitments of multilateral agreements (MEAs) and achieve sustainable development (Dernbach 2005). The promise of a targets approach is to inspire "broad-based action by parties and stakeholders" (CBD/COP6 2002), to set a coherent agenda for action, and to raise the international and national profile of progress (Roberts 2005). Others argue that the effective use of targets signals meaningful commitment within MEAs, and thus credibility of the agreement itself (Dernbach 2005).

Established in 1993, the CBD adopted a targets approach in 2000 with the decision to "develop a Strategic Plan for the Convention" for 2002–2010 (CBD/COP5 2000). Prompted by the "need for more effective and coherent implementation," this decision included the commitment to identify a "set of operational goals," and 2 years later, the agreement "to achieve by 2010 a significant reduction of the current rate of biodiversity loss at the global, regional, and national level as a contribution to poverty alleviation and to the benefit of all life on Earth" (CBD/COP6 2002). This commitment is known as the *2010 Biodiversity Target*. Yet, repeating an inauspicious trend of falling short of global targets in international conventions (Roberts 2005), recognition of the lack of progress toward the 2010 target was confirmed in the spring of 2010 (Butchart *et al.* 2010). News reports

Table 1 Summarized descriptions of the 20 Aichi Biodiversity Targets, adapted from the Global Biodiversity Outlook (2014) and the CBD Quick Guides for the Aichi Biodiversity Targets CBD. Unless otherwise noted, each target has 2020 as the end date

Strategic goal/ Aichi Target #	Description of target
Strategic goal A	*Address the underlying causes of biodiversity loss by mainstreaming biodiversity across government and society*
1	Public awareness of the values of biodiversity increased
2	Biodiversity values integrated into national and local development and poverty reduction strategies
3	Incentives, including subsidies, harmful to biodiversity are eliminated, phased out, or reformed
4	Sustainable consumption and production
Strategic goal B	*Reduce the direct pressures on biodiversity and promote sustainable use*
5	Rate of loss of all natural habitats is at least halved or where feasible brought close to zero; degradation and fragmentation significantly reduced
6	Sustainable management of marine living resources
7	Sustainable management of areas under agriculture, aquaculture, and forestry
8	Pollution has been brought to levels that are not detrimental to ecosystem function and biodiversity
9	Invasive alien species prevented and controlled
10	Anthropogenic pressures on vulnerable ecosystems minimized
Strategic goal C	*Improve status of biodiversity by safeguarding ecosystems, species, and genetic diversity*
11	Protected areas increased (17% terrestrial and inland water areas; 10% coastal and marine areas) and conserved through effectively and equitably managed, ecologically representative, and well-connected areas
12	Extinction of known threatened species prevented
13	Genetic diversity of cultivated, farmed, and domesticated species maintained
Strategic goal D	*Enhance the benefits to all from biodiversity and ecosystems services*
14	Ecosystems that provide essential services are restored and safeguarded
15	Ecosystem resilience and contribution to carbon stocks enhanced through conservation and restoration
16	Nagoya Protocol on access and benefit sharing is in force and operational (2015)
Strategic goal E	*Enhance implementation through participatory planning, knowledge management, and capacity building*
17	National Biodiversity Strategies and Action Plans (NBSAPs) are developed, adopted and being implemented (2015)
18	Traditional knowledge, innovations, and practices of indigenous and local communities are respected; full and effective participation at all relevant levels
19	Knowledge, the science base, and related technologies are improved, widely shared, and applied
20	Mobilization of financial resources from all sources increased substantially from current levels

declared the convention's effort a failure (Black 2010), and a few months later, plenary speakers at the 10th Conference of the Parties to the CBD underscored the relevance of the Convention.[1] Paradoxically, efforts to motivate action and assert the credibility of the convention through a targets approach threatened the opposite effect.

The implementation of MEAs faces myriad, known challenges. These include lack of scientific knowledge, lack of political will, political instability, inadequate economic incentives, poor involvement of civil society, and funding limitations to name a few (Bille *et al.* 2010; Gagnon-Legare & Prestre 2014; Adenle *et al.* 2015). On the heels of the perceived 2010 failure, and in effort to overcome known challenges, parties to the CBD agreed to 20 new and ambitious targets—the 2020 Aichi Targets (Table 1). Reflecting broader trends in the use of "global targetry" in sustainable development (Roberts 2005), and conservation (Carwardine *et al.* 2009), the Aichi Targets were designed to be SMART (specific, measureable, ambitious, realistic, and time-bound) (Maxwell *et al.* 2015). This emphasis on SMART targets is constitutive of broader trends toward the use of market-based instruments in conservation (Muradian *et al.* 2013) and associated inclinations toward a governance logic of "measurementality" based on managerial principles (Turnhout *et al.* 2014). Yet, despite their intended SMART-ness (Maxwell *et al.* 2015), midway global assessments indicate that the majority of targets are unlikely to be met (Secretariat of the CBD 2014; Tittensor *et al.* 2014).

In the vernacular of SMART targets, the uneasy trade-offs that pertain to a convention's credibility lie at the nexus of (A)mbition and (R)eality. That is, set the level of ambition too low, and the target becomes inconsequential and ineffective in addressing the problem the convention was designed for. Set the level too high, and risk failure to implement. Both pathways carry risks for a convention's perceived credibility.

One of the defining insights from the past decade of scholarship on global environmental governance is that MEAs are part of a complex network of diverse actors and institutions that interact across local, national, and international scales (Biermann *et al.* 2012). Yet, scholarly examinations of progress toward the Aichi Targets at national scales (i.e., beyond the submission of National Biodiversity Actions Plans, NBSAPs) remain relatively underexamined. Notable exceptions include a handful of target-specific studies. Aspects of implementation for Target 11 have been examined in Canada (MacKinnon *et al.* 2015), France (Meinesz & Blanfune 2015), Japan (Naoe *et al.* 2015), and the Philippines (Mallari *et al.* 2016). Target 12 has been examined in Italy (Fenu *et al.* 2015). Further, despite the increasing role of nonstate actors such as environmental nongovernmental organizations (ENGOs) (Gagnon-Legare & Prestre 2014) and business (Pistorius & Freiberg 2014) in biodiversity governance, relatively little is known about how different actors within nation states are seeking to address or align their activities with the Aichi Targets. This is a problematic gap considering that the Aichi Targets (and thus the convention as a whole) are implemented at the national level, and the fact that the full collection of targets is deemed essential to achieve the mission of the CBD. Finally, the use of global targets is often invoked as a means to motivate action, communicate high standards, and assert credibility. Yet, the CBD is implemented through activities taken at the national level, with commitments that are highly qualified. NBSAPs, for example, are the primary mechanism for implementing the CBD, with submittal of an NBSAP[2] being the one legal commitment required of parties (Harrop & Pritchard 2011). Article 6 of the convention text places this requirement in context:

> Each Contracting Party shall, in accordance with its particular conditions and capabilities: a) Develop national strategies, plans or programmes for the conservation and sustainable use of biological diversity . . . and b) Integrate, *as far as possible and as appropriate*, the conservation and sustainable use of biological diversity into relevant sectoral or cross-sectoral plans, programmes and polices [emphasis added] (United Nations 1992).

Thus, success or failure to implement is strongly influenced by social–political context at national levels.

The aim of this article is to deepen understanding about national implementation of global targets in the context of multilevel environmental governance. The specific objectives are to: (1) apply systematic content analysis within a case study approach to examine the nature and extent of engagement with the Aichi Targets in Canada from 2011 to 2016 in terms of actors involved, types of actions pursued, representation of actions across spatial scales and biomes, and relative emphasis of targets; (2) draw policy recommendations to inform the final years toward implementation in the Canadian context; and (3) develop insights to foster policy-relevant dialog about the roles and possibilities of global targets in environmental governance.

We selected Canada as a case given its consequential role in contributing to global biodiversity. Covering almost 10 million square kilometers of land and water, Canada contains 28% and 15% of world's boreal forests and temperature forests, respectively, 25% of wetlands globally, and the world's longest coastline (Natural Resources Canada 2016). Canada is home to an assessed 70,000 species, including 110 threatened species (Environment and Climate Change Canada 2016). As the first developed country to ratify the CBD, and host of the CBD Secretariat, Canada possesses the apparent institutional capacity and political stability that one might expect to contribute to successful implementation.

Methods

Document selection

We identified and reviewed publicly accessible English language policy, planning, public relations, and technical documents that addressed the Aichi Targets in Canada between January 2011 and April 2016. Our systematic search included manual and keyword-driven approaches. For the former, inclusion criteria included all official documents produced by the National Focal Point (Environment and Climate Change Canada) as well as Natural Resources Canada, and provincial and territorial environment agencies. For the latter, we used Google-advanced search tools using the terms "Aichi Target," "biodiversity target," and "Convention on Biological Diversity" in all possible combinations using the domains ".ca," ".gc.ca," ".org," ".bc.ca," ".alberta.ca," ".sk.ca," ".mb.ca," ".ontario.ca," ".qc.ca," ".nl.ca," ".gnb.ca," ".ns.ca," ".pe.ca," ".yk.ca," ".nt.ca," and ".nu.ca." This strategy identified 237 documents. After duplicates were removed, 230 documents were screened for our inclusion criteria. We subsequently reviewed 184 full-text documents and removed 30 that did not meet the inclusion criteria. Through this process, we identified and analyzed 154 documents. Figure S1 summarizes the document inclusion process. Table S1 contains a complete list of documents analyzed.

Analysis

The 154 documents were systematically analyzed using content analysis. Content analysis describes a class of qualitative and quantitative analytical approaches for analyzing textual data that are sometimes classified in terms of conventional, directed and summative approaches (Hsieh & Shannon 2005). Here, we apply directed content analysis following established methodological protocols to summarize trends in key categories of interest. Specifically, we developed a set of ex ante coding categories to guide our inquiry. The codes included: type of action, actor group, document type, status, biome, scale, and target. Details of this typology are described in Table S2. Each document was added to QSR International's NVivo qualitative data analysis software (QSR International Pty Ltd. Version 10, 2012). Text was coded line by line according to the typology described above. Frequencies for each category were calculated to explore trends in this context, not to infer generalizability (Hsieh & Shannon 2005). One author (RP) coded all documents. The lead author (SH) independently reviewed a selection of documents to ensure reliability.

Results

Actors

Seven different actor groups produced the 154 documents analyzed in this study. More than half of the documents were produced by federal agencies (31%) and ENGOs (29%) (Figure 1). The majority (50%) of documents produced by federal agencies were progress reports. ENGOs produced a range of documents including progress reports (39%), annual reports (20%), and policy briefs (20%). Figure S2 details the types of documents produced by different actors.

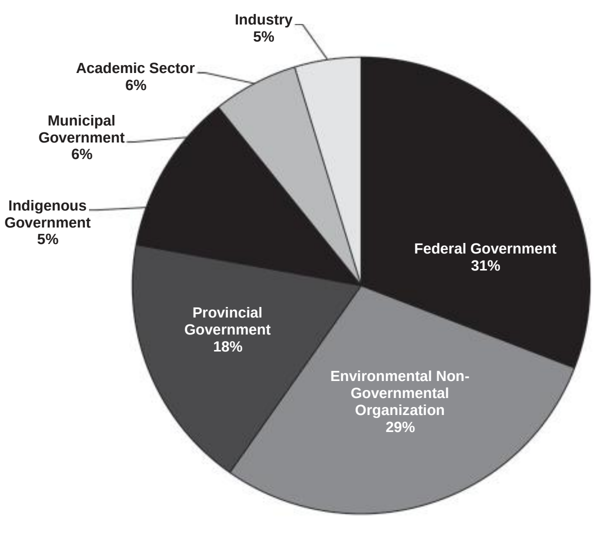

Figure 1 Distribution of documents produced by different actors ($N = 154$). For documents authored by more than one actor group ($N = 4$), each actor group received attribution. Note: The term "indigenous government" is used here to reflect a "nation-to-nation" or "government-to-government" perspective of the relationship between federal or provincial governments and indigenous peoples (not nation-to-stakeholder).

Responses

Our analysis identified 2,222 responses related to the Aichi Targets. Seventy-two percent of these (1,593) were aspirational. Twenty-eight percent (629) were implemented (Figure 2a). We identified seven types of implemented responses: information support (35%), resource mobilization (20%), institutional planning (14%), collaboration (12%), public awareness (11%), monitoring (6%), and consultation (3%) (Figure 2b). Descriptions and examples for each category are summarized in Table S3.

Geographic and socioecological focus

To varying extents, responses were detected in all Canadian provinces and territories. British Columbia (BC) was associated with the greatest number of responses (both aspirational and implemented) (Figure S3). Figure S4 illustrates patterns of engagement with specific targets for each province/territory. The majority of responses (both aspirational and implemented) were associated with nonforest-specific terrestrial (39%) and marine (37%) ecosystems (Figure 2c).

Extent

Targets 11, 19, 1, 12, and 14 were associated with the greatest number of implemented actions (between 121 and 57). The remaining 15 targets were associated with less than 43 (to 0) implemented responses each (Figure 3). Informed by previous scholarship and known dilemmas associated with the use of targets in MEAs (e.g., as relates to lack of quantifiability), we present the number of responses for each target by clarity in level of ambition and focal emphasis of the target (Table 2). As detailed by Butchart *et al.* (2016), each individual Aichi Target contains multiple elements some of which may be quantifiable, while others may not be (e.g., T5, T11, T15). Accordingly, we use three categories to describe the clarity in level of ambition for each target: all elements quantifiable, at least one element quantifiable, and no specified level of ambition. For emphasis, we developed and applied a set of categories drawn from an understanding of the literature on policy implementation and global governance.

Across the 20 targets, three are characterized as fully quantifiable, three as partially quantifiable, and 14 as having no specified level of ambition. In this context, the six quantifiable or partially quantifiable targets were associated with targets located across the spectrum in terms of the number of implemented actions (Table 2). For instance, quantifiable or partially quantifiable targets were associated with targets with relatively high levels of

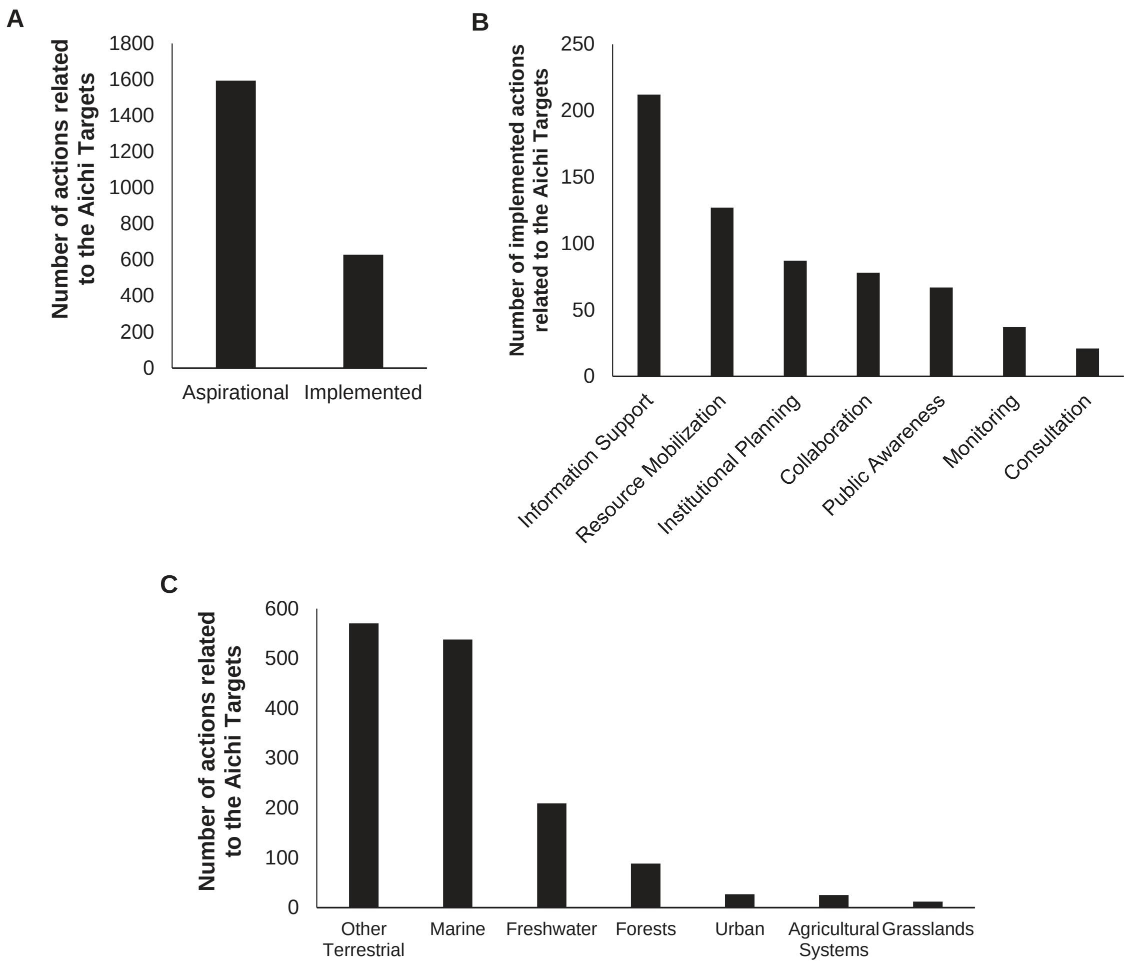

Figure 2 Number of actions related to the Aichi Targets by (a) status of implementation, (b) type of action (considering implemented actions only), and (c) focal biome (aspirational and implemented actions). Actions may belong to more than category (e.g., projects designed to achieve both collaboration and public awareness objectives) or more than one biome (e.g., funding for both terrestrial and aquatic protected areas), in which case counts are made for each relevant category.

implemented actions (e.g., T11), and relatively low levels of implemented actions (e.g., T16). Six of the top 10 targets included those focused on alleviating impacts and pressures or sustaining biophysical values. Five of the six targets focused on equity, rights, and policy reform occurred in the bottom 10.

Discussion

From critical to normative perspectives, most conservation scholars and practitioners recognize the potential of MEAs to contribute to conserving biodiversity and the equitable use and sharing of benefits. Some argue that MEAs are increasingly relevant as a governance response given the magnitude of global change and the need for "planetary stewardship" (Biermann *et al.* 2012). Yet, cast in the light of persistent shortfalls to meet targets, MEAs face a potential crisis of credibility—particularly in the eyes of those who may question whether MEAs "matter" (Andresen & Hey 2005). Our finding that engagement with the Aichi Targets in Canada tends to be mostly aspirational, thus raises concerns voiced by others, that while nation states proclaim their membership to conventions like the CBD, substantive efforts remain insufficient (Morgera & Tsioumani 2011).

Which targets left behind?

The finding that some quantifiable or semiquantifiable targets that tend to emphasize biophysical values and impacts (e.g., T11 and T12) are associated with higher numbers of implemented responses in this context in

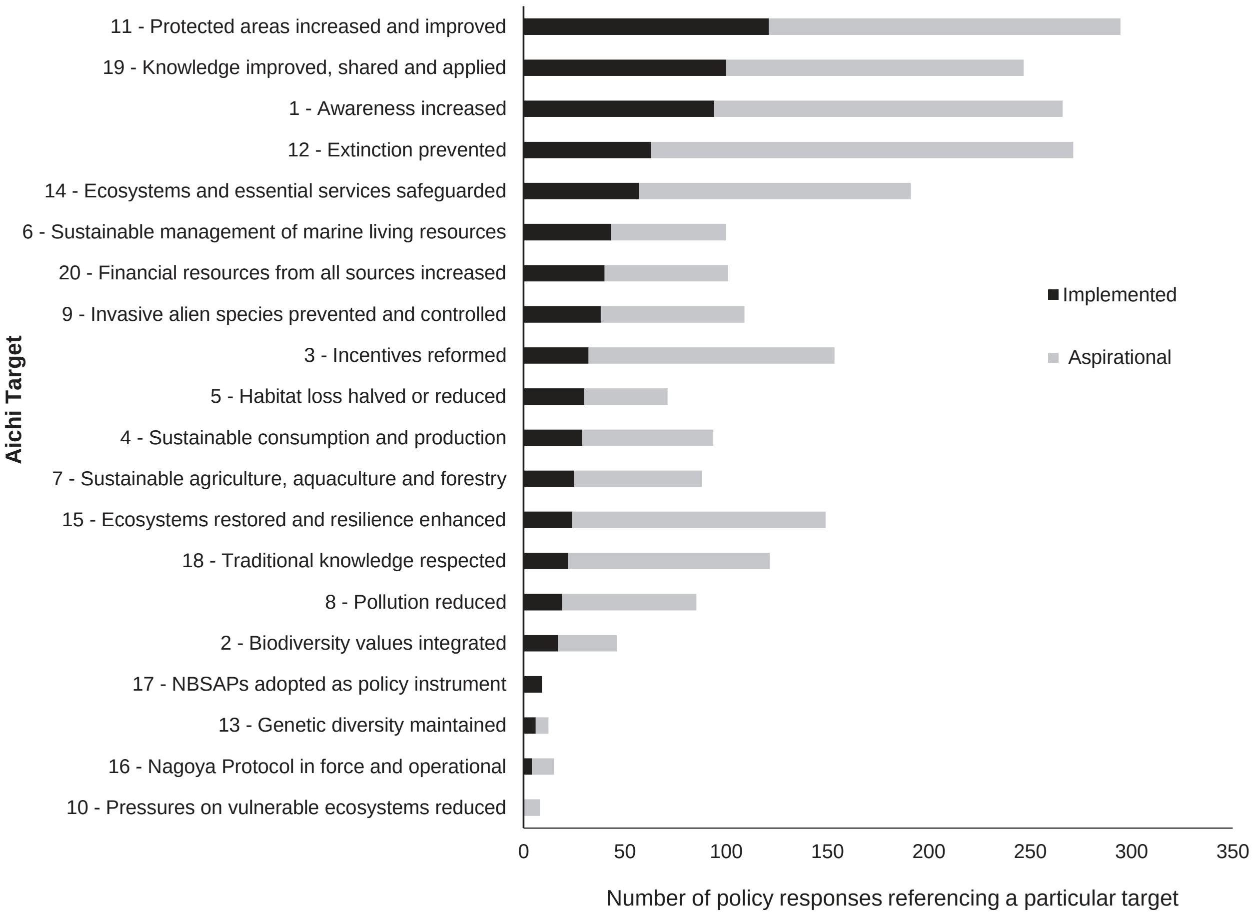

Figure 3 Number of implemented and aspirational actions for each of the 20 Aichi Biodiversity Targets. Some actions are associated with more than one target (e.g., providing funding [Target 20] to establish a protected area [Target 11]) in which case counts are made for each relevant category.

contrast with those that are focused on equity, rights, or policy reform is not surprising, but cause for concern. It is not surprising given the volume of conversation about the need for quantifiable ecosystem-service-focused accounting (e.g., The Economics of Ecosystems and Biodiversity initiative), in conservation governance (Macdonald & Corson 2012). This finding is also consistent with recent CBD analyses that found that national reports tend to use indicators for some targets (specifically those that are quantifiable) (e.g., 5, 11, and 12) more than others (CBD/SBSTTA20 2016).

The degree of quantifiability, however, provides only a partial explanation as the targets moving forward in this case are those that are aligned with and reinforced by conventional institutional commitments and norms, including those that are relatively straightforward to achieve in this context. The relatively high frequency of responses associated with increasing protected areas (PAs) (T11) is a prime example of the former where in addition to being buoyed by a legacy of institutional commitments, PAs are further reinforced as the "natural solution" in a world increasingly affected by global change (Dudley *et al.* 2010). Capitalizing on the institutional centrality of PAs, diverse actors frame their objectives (e.g., biodiversity, carbon, and livelihoods) through PAs, thus making PAs, "everyone's solution" (Corson *et al.* 2014). Similarly, while having no specified level of ambition, progress for other targets with relatively high levels of implemented actions (e.g., knowledge sharing [T19] and public awareness [T1]) is arguably enabled by an institutional context in which science capacity is relatively high, and "understanding awareness" reasonably easy to claim progress toward.

By comparison, targets that are "left behind," such as the explicitly quantifiable target on access and benefit sharing (T16),[3] integrating biodiversity into national accounts (T2), or ensuring rights and customary use of biological resources by indigenous and local communities

Table 2 Aichi Targets presented in order of the most (Target 11) to least (Target 10) number of implemented actions, and characterized by specificity of level of ambition, and target emphasis. For specificity in level of ambition, filled circles indicate targets for which all target elements are quantifiable, half-circles indicate targets that have at least one quantifiable element, and empty circles indicate targets for which there is no specified level of ambition. Asterisks denote the primary emphasis of the target

Target	Specificity of level of ambition			Emphasis					
	All elements quantifiable	At least one element quantifiable	No specified level of ambition	Alleviating impacts and pressures	Sustaining biophysical values	Enabling policies, protocols and planning processes	Knowledge capacity and resource mobilization	Ensuring equity and rights	Promoting increased public awareness
11		◖			*				
19			○				*		
1			○						*
12	●				*				
14			○		*				
6			○	*					
20			○				*		
9			○	*					
3			○			*			
5		◖		*					
4			○			*			
7			○	*					
15		◖			*				
18			○					*	
8			○	*					
2			○			*			
17	●					*			
13			○		*				
16	●							*	
10			○	*					

are respected (T18), include those that challenge prevailing institutional norms and governance arrangements. Despite landmark decisions within Canadian law (e.g., Delgamuuk v. British Columbia - 1997, Tsilhqot'in Nation v. British Columbia - 2014), and cogovernance arrangements for some conservation areas (e.g., Gwaii Haanas), there remain deeply contested and competing claims regarding jurisdiction and management authority for extensive areas where Crown and Indigenous governments assert ownership, rights, and responsibilities over the same area. It is amidst this backdrop that progress toward the target relating to ensuring that rights and customary use of biological resources are respected (T18) can be viewed as posing a challenge to prevailing social–political structures of jurisdiction and management authority.

Thus, we argue that implementation for these targets may be stalled not necessarily because of their lack of SMART-ness—specifically measurability in terms of quantifiability, but because of their institutional disruptiveness. This suggests that solutions—in terms of improving implementation—lie not only in overcoming known dilemmas and challenges related to the lack of quantification of targets (Butchart *et al.* 2016), but also in fostering institutional transformation and change.

Our review invites consideration of a number of potential pathways for policy action, and questions for future research. First, the relative underrepresentation of implemented actions associated with targets related to equity (T16) and rights (T18) is significant considering that these components are central to the mandate of the CBD. Further, equity dimensions of otherwise high-profile targets—notably the PAs target (T11)—have also received relatively little attention, where the overriding focus has rested instead on the (quantifiable) element of spatial coverage. The prospect of continued slow progress in this realm poses potential risks for indigenous and local communities, as well as for the perceived credibility of the convention itself. While outreach initiatives tend to emphasize the need to increase public awareness of biodiversity values, our analysis suggests the need to increase awareness of the importance of rights and equity in biodiversity conservation, and to translate this awareness into progress for equity-related targets, and equity dimensions of targets like Target 11. Outstanding questions to be addressed include: To what extent, if at all, is this observation reproduced in other social-ecological contexts? What explains variation where it is observed? What institutional arrangements or policy interventions might enhance progress for those targets that do not confirm easily within existing governance arrangements? What potential levers toward transformation might be applied to achieve these crucial dimensions? Given that successful implementation of complex MEAs requires time (Andresen & Hey 2005), what is the relationship between protracted and marginal progress toward global targets and the perceived credibility of the convention?

Second, the engagement of state and nonstate actors with the Aichi Targets is revealing both in terms of who is, and is not engaged. Relatively high levels of engagement by ENGOs are expected given the increased involvement of nonstate actors in conservation governance generally. The finding of low levels of engagement by indigenous governments requires deeper scrutiny. In Canada and worldwide, indigenous governments have engaged in conservation for millennia, although formal recognition by global conservation institutions—for example, in the form of Indigenous and Community Conserved Areas (ICCAs)—is a relatively recent development. Further, indigenous communities are asserting their rights to their traditional territories for which nation states may or may not recognize. Therefore, while our analysis suggests thin engagement by indigenous governments with the Aichi Targets, there are historical and sociopolitical reasons why indigenous governments may choose not to engage directly with state-sanctioned initiatives like the Aichi Targets. Lastly, as the CBD seeks to maximize alignment and synergies with mechanisms including the SDGs and within the UNFCCC, outstanding questions include: How are these efforts shaping the development and pursuit of targets at the national level, particular as relates to the rights, involvement and participation of indigenous peoples and local communities?

The analysis presented here provides an overview of actions at a particular point in time within the bounds of our sample frame and methodology. There very likely exist documents related to the Aichi Targets that we did not have access to. Further, this study does not offer insights into the sociopolitical processes that shape how specific implementation activities are designed, adopted, and pursued at different levels of governance and by different actors. By applying analytical tools from the social sciences to an interdisciplinary challenge, this work offers a novel and systematically derived snapshot of the types on actions moving forward in the Canadian context that serve to highlight persistent blind spots in the implementation of the CBD, and tensions inherent to the use of global targets in multilevel governance.

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1. Based on participant observation field notes taken at the 10th Conference of the Parties to the Convention on Biological Diversity (COP 10 - CBD) by the first author in October, 2010, Nagoya, Japan
2. As of April 2016, 79 of 196 (40%) Parties had submitted NBSAPs that take the Strategic Plan, and thus the Aichi Targets into account (https://www.cbd.int/nbsap/).

3. Canada is not a party or signatory to Nagoya protocol. As reported in Canada's 5th National Report to the Convention on Biological Diversity (2014), "Canada is engaging provinces, territories, Aboriginal groups and other key stakeholders to provide them with an opportunity to consider possible elements of a domestic ABS policy and contribute to an increased understanding of the potential impacts of the Nagoya Protocol in Canadian jurisdictions" (p. 98).

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POLICY PERSPECTIVE

The Contributions of the EU Nature Directives to the CBD and Other Multilateral Environmental Agreements

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Abstract

Through a review of published studies and new analyses of publicly available data, we assess how the European Union (EU) Nature Directives complements the CBD strategic goals for 2020 as set out in the 20 Aichi Targets, thereby addressing a question posed by the European Commission about the coherence of the Directives with other international biodiversity commitments. We find evidence that the Directives complement several Aichi Targets and other Multilateral Environmental Agreements (MEAs). For example, 92% of the EU's Important Bird and Biodiversity Areas (IBAs), many of them otherwise unprotected, are partly or wholly covered by the Natura 2000 network of protected areas (contributing to Aichi Target 11). Species listed on Annex I of the Birds Directive have fared better than other species (Aichi Target 12). As 65% of EU citizens live within 5 km of a Natura 2000 site, and 98% within 20 km, these sites have the potential to raise awareness of biodiversity (Aichi Target 1) and to deliver ecosystem services to a high proportion of the EU's population (Aichi Target 14). The Nature Directives provide a regulatory framework that, with fuller implementation, will help EU Member States to meet their obligations under the CBD and other MEAs.

Introduction

The 28 Member States of the European Union (EU) are legally bound by the 1979 (amended in 2009) Birds Directive and the 1992 Habitats Directive, jointly referred to as the Nature Directives, which form the cornerstone of EU conservation law. These Directives oblige all Member States to safeguard and restore threatened species and habitats. Countries joining the EU are required to bring their national legislation into line with these Directives, so that the aims of the Directives are codified in national laws relating to wildlife conservation and other policy areas that impact on target sites and species.

The Directives have two main pillars, a strict system of species protection and the Natura 2000 network of protected sites. Natura 2000 comprises over 27,300 protected sites (around 3,000 of them having a significant marine component) covering 18% of the land area of the EU (http://ec.europa.eu/environment/nature/natura2000/barometer/index˙en.htm). It forms the world's largest network of conservation sites under a single regulatory framework (Evans 2012). Natura 2000 sites are highly heterogeneous and most contain human populations, agricultural land, and forestry (Tsiafouli *et al.* 2013). The Directives also contain Annexes of species for which Member States must implement special conservation measures. This may take the form of legal protection from persecution and disturbance, habitat protection and restoration, and monitoring and research. The EU LIFE funding instrument provides targeted, albeit modest, financial support to conservation, and additional resources are available through the Common Agricultural Policy in the form of agri-environment measures (Matthews 2013), which in many countries are targeted toward Natura 2000 sites, and through EU Structural Funds and the new EU Natural Capital Financing Facility (Kettunen *et al.* 2014).

There is empirical evidence that both the Birds Directive (Donald *et al.* 2007; Deinet *et al.* 2013; Sanderson *et al.* 2015) and the Habitats Directive (Pellissier *et al.*

2013, 2014; Brodier *et al*. 2014; Kallimanis *et al*. 2015, but see Santana *et al*. 2013) have had a positive impact on the EU's biodiversity. However, there remain considerable challenges to achieving full implementation of the Directives (Křenová & Kindlmann 2015), which could be improved with more effective planning and enforcement, more effective, consistent and policy-relevant monitoring (Davis *et al*. 2014) and, perhaps most important, increased funding (Kettunen *et al*. 2011; Hochkirch *et al*. 2013; Louette *et al*. 2015).

In 2014, the EC commissioned a "Fitness Check" of the EU Nature Directives under the Regulatory Fitness and Performance Programme (REFIT), which explicitly poses the question: "How coherent are the Directives with international and global commitments on nature and biodiversity?" (http://ec.europa.eu/environment/nature/legislation/fitness˙check/docs/Mandate%20for%20Nature%20Legislation.pdf).

We assess the extent to which the Directives complement or directly contribute to a wider multilateral environmental agreement, the Convention on Biological Diversity (CBD). All EU Member States, and the EU in its own right, have ratified the CBD. The CBD Strategic Plan for 2011–2020 contains five strategic goals under which are organized the 20 Aichi Biodiversity Targets. The Nature Directives and the CBD both emphasize the conservation of threatened species, the protection of important habitats and the integration of societal considerations in conservation management. These commonalities are recognized in the EU Biodiversity Strategy (http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri = CELEX:52011SC0540&from = EN), which links the full implementation of the Nature Directives with progress toward Aichi Targets 11 (site protection) and 12 (species protection).

Rationale

Despite these commonalities, and much recent attention on measuring progress toward the Aichi Targets (Tittensor *et al*. 2014; Butchart *et al*. 2015), there has been no quantitative assessment of the extent to which the Directives contribute to the full set of Aichi Targets, either directly or by complementarity (i.e., creating an environment within which the Aichi Targets are more likely to be met). Here, we map the objectives and successes of the Nature Directives onto the 20 Aichi Targets to assess their complementarity. We identify six Aichi Targets whose complementarity with the EU Nature Directives can be assessed empirically (Table 1).

Counterfactual assessments of what the EU conservation landscape would look like in the absence of the Directives are difficult because the Directives are codified in national law in ways unique to each Member State (European Environment Agency 2012). However, there is ample evidence that the Directives have yielded additional benefits. For example, over 50% of Natura 2000 sites are not covered by any other form of protected area designation (see below). Countries acceding to the EU show substantial increases in their coverage by protected areas around the time of accession (European Environment Agency 2012), and increases in the populations of target species thereafter (Sanderson *et al*. 2015). Even in the United Kingdom, which has a longer history of conservation legislation than most countries, the "Directives have added a layer of protection for nature . . . above and beyond that provided in previous national legislation" (Institute for European Environmental Policy 2013). However, some of these conservation gains may have accrued in the absence of EU legislation, and the unique contribution of the Directives cannot be quantified.

Species protection is explicitly addressed in just one Aichi target (Target 12), and the contribution of the Directives to species protection have already been quantified (e.g., Donald *et al*. 2007; Sanderson *et al*. 2015). The complementarity of Natura 2000 is less well understood, yet the Aichi targets are divided in such a way that site protection, and the wider benefits thereof, are spread across several targets. We therefore undertake new analyses to assess the role of Natura 2000 in meeting Aichi targets.

Full details of the analyses are presented in Appendix S1. Data sources for each analysis are referenced in Table 1, which also summarizes some of the limitations of the data used.

Complimentarily and contribution of the EU Nature Directives to the CBD Aichi Targets

Aichi Target 1—awareness of biodiversity

As an assessment of the extent to which the Natura 2000 network might complement the target that "people are aware of the value of biodiversity," we quantified the extent to which people in the EU live in proximity to Natura 2000 sites, since proximity to wildlife has been shown to promote environmental awareness (e.g., Miller 2005) and there is a correlation between the time spent in nature as a child and the level of support to environmental protection as an adult (Wells & Lekies 2006). We used two measures of human population distribution, one based on commune-level census data, and the other based on satellite-detected night

Table 1 Summary of assessment of the contribution of Natura 2000 to the CBD Aichi Targets

Aichi Target	Test metrics	Test of contribution or complementarity, and limitations of metric	Sources of data
1: "By 2020, at the latest, people are aware of the values of biodiversity and the steps they can take to conserve and use it sustainably"	The proportion of the EU's population living within accessible distance of a Natura 2000 site	Complementarity Remoteness/disconnection from wild nature hinders an awareness of biodiversity, but proximity to wild nature does not on its own necessarily increase it.	Human population: Population density disaggregated with Corine land cover 2000, European Environment Agency[a,b] http://www.eea.europa.eu/data-and-maps/data/population-density-disaggregated-with-corine-land-cover-2000-2 Night lights: Version 4 DMSP-OLS Night-time Lights Time Series, NOAA Earth Observation Group[a,b] http://ngdc.noaa.gov/eog/dmsp/downloadV4composites.html#AVSLCFC Accessibility: Natural England (2015)
5: "By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced"	Rates of forest loss between 2000 and 2012 within Natura 2000 sites compared with rates of forest loss outside Natura 2000 sites	Contribution The forest loss statistics used do not differentiate between natural forest and plantations, nor between different types of natural forest	Forest loss and tree cover: Hansen *et al*. (2013) Human population: see Target 1 Altitude: SRTM Digital Elevation Model, USGS http://glcf.umd.edu/data/srtm/ Distance to roads: Vector map Level O,[a,b]www.mapability.com/index1.html?http&&www.mapability.com/info/vmap0_intro.html
11: "By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through . . . . protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes"	The increase in the number of ecoregions having less than 17% coverage by protected areas when areas protected only by Natura 2000 are excluded	Contribution Natura 2000 sites do not necessarily cover the most representative or pristine elements of each ecoregion	Terrestrial ecoregions: WWF terrestrial ecoregions of the world (Olson *et al*. 2001) IBAs: http://www.birdlife.org/datazone/site Distribution of protected areas: http://www.protectedplanet.net/
	The proportion of all Important Bird and Biodiversity Areas (IBAs) inside Natura 2000 sites in each country and across the EU as a whole	Contribution	Distribution of UNESCO World Heritage Sites: http://whc.unesco.org/en/list/
12: "By 2020 the extinction of known threatened species has been prevented and their conservation status, particularly of those most in decline, has been improved and sustained"	The proportion of all Important Bird and Biodiversity Areas (IBAs) listed for globally threatened species or species threatened at a European level that are contained within Natura 2000 sites across the EU	Complementarity Substantial proportions of some threatened species' ranges may fall outside IBAs	As for Aichi Target 11, plus: Global conservation status: IUCN Red List (http://www.iucnredlist.org/) European conservation status: BirdLife International (2004)

Continued

Table 1 Continued

Aichi Target	Test metrics	Test of contribution or complementarity, and limitations of metric	Sources of data
14: "By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded, taking into account the needs of women, indigenous and local communities, and the poor and vulnerable"	The proportion of the EU population living close to Natura 2000 sites	Complementarity Proximity to Natura 2000 sites does not necessarily guarantee access to/benefits of the ecosystem services they provide	As for Aichi Target 1
15: "By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification"	Carbon stocks within the Natura 2000 sites as a proportion of total EU carbon stocks	Complementarity The metric used relates only to estimates of current carbon stocks, not to their enhancement or restoration	Carbon stocks: Kapos *et al*. (2008)[a,b]

Note: Under "Sources of data", [a]indicates data of uncertain accuracy and [b]indicates data with coarse spatial resolution.

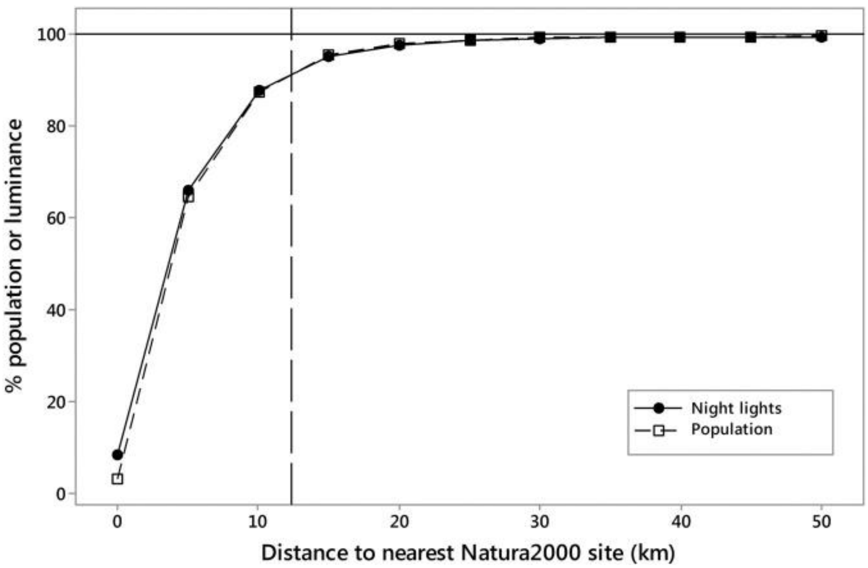

Figure 1 Percentages of the total EU population (estimated directly and from luminance from night lights) living within different distances of Natura 2000 sites. The vertical line at 12.35 km represents the mean distance traveled by participants in the MENE survey (Natural England 2015) to visit a number of wildlife habitats for recreational purposes.

lights (Appendix S1). We estimate that 65% of EU citizens live within 5 km of a Natura 2000 site, and 98% live within 20 km (Figures 1 and S1). This accessibility is reflected in visitor numbers; Natura 2000 sites are estimated to receive between 1.2 and 2.2 billion visitor days each year (ten Brink *et al.* 2011). The Directives also offer the opportunity for the general public and national and international NGOs to be involved in the designation, monitoring and management of Natura 2000, with the potential for greater engagement by the public and stakeholder groups (landowners, hunters, farmers etc.) in nature conservation and the growth of volunteer networks of site support groups and citizen scientists (Unnerstall 2008).

Aichi Target 5—loss of natural habitats

We are unaware of any attempt to quantify the extent to which Natura 2000 designation slows habitat change across multiple countries. Using a global assessment of tree cover loss (Hansen *et al.* 2013), we estimate that the probability of significant forest loss between 2000 and 2012 was lower in forested or partly forested 1 km^2 within Natura 2000 than outside (Figure 2; Appendix S1). Some forest loss within Natura 2000 is likely to result from management for nature conservation, making our assessment conservative. Data on the loss of other natural habitats within and outside Natura 2000 are not available at a resolution appropriate for analysis.

Aichi Target 11—protected area coverage

Target 11 aims to protect 17% of terrestrial areas in "ecologically representative" and "well-connected" systems and to include "areas of particular importance for biodiversity." We quantified the spatial overlap between Natura 2000 and (1) the EU's terrestrial ecoregions and (2) all the EU's Important Bird and Biodiversity Areas (IBAs), both at a country level and across the EU as a whole. The European Court of Justice has indicated that the coverage of IBAs by the Natura 2000 network should be used in assessing whether Member States have met their obligations regarding the designation of key sites (http://ec.europa.eu/environment/nature/info/pubs/docs/others/ecj_rulings_en.pdf). We also intersected the Natura 2000 and IBA networks with all other protected (Appendix S1).

Of the EU's 42 ecoregions, 37 (88%) had at least 17% coverage by area by all classes of protected area designation, falling to 23 (55%) when areas protected only by Natura 2000 were excluded (Figure 3a). At the national level, 84 of the 107 country/ecoregion combinations (79%) had at least 17% coverage by all classes of protected area, falling to 56 (52%) when areas protected only by Natura 2000 were excluded (Figure 3b). Across the EU, 94% of all IBAs are wholly or partly covered by Natura 2000 (Figure S2), with 72% coverage by area (Figures 4 and S3). Of the area of IBAs within Natura 2000, 47% is otherwise unprotected (Figure 4). The designation of marine Natura 2000 sites lags behind that of terrestrial designations, and is unlikely to meet the 10% coverage of key marine areas required by Aichi Target 11. Around half the area covered by Natura 2000 is otherwise unprotected (Figure 4). This is likely to underestimate the contribution of the Directives, since Natura 2000 designation often triggers further protective designation (European Environment Agency 2012).

The connectivity of the Natura 2000 network was assessed with a nearest-neighbor analysis, which indicated that nearly half (>11,000; 44.5%) of all sites were contiguous with one or more other sites, and that sites not contiguous with other sites were a mean of just 2.7 km from the closest site, suggesting that the network is well-connected and thus spatially aligned to provide a useful role in mitigating climate-change induced range shifts (Kettunen *et al.* 2007).

Aichi Target 12—species extinction

The performance of the EU Directives in preventing the "extinction of known threatened species," and ensuring that their "conservation status... has been improved," has previously been assessed by Donald *et al.* (2007) and Sanderson *et al.* (2015), who showed that species listed on Annex I of the Birds Directive showed greater improvements in their population trajectories than did non-Annex I species after, but not before, the introduction of the Directive, and within, but not outside, the EU. Natura 2000 effectively captures the ranges of a high proportion of threatened species (Gruber *et al.* 2012; Trochet & Schmeller 2013). Over 90% of IBAs designated for their importance to threatened species are included in Natura 2000 (Figure 4). The species protection measures of the Directives have also been important in the recovery of many species (Deinet *et al.* 2013, European Commission 2015).

Aichi Target 14—ecosystem services

Safeguarding sites important for biodiversity conservation provides substantial benefits to human well-being (e.g., European Environment Agency 2012; Larsen *et al.* 2012). It has been estimated that the value of the ecosystems services delivered by Natura 2000 is approximately €200–300 billion per year (2–3% of the EU's GDP), without taking account of benefits to human health (ten Brink *et al.* 2011). The growing public recognition of these

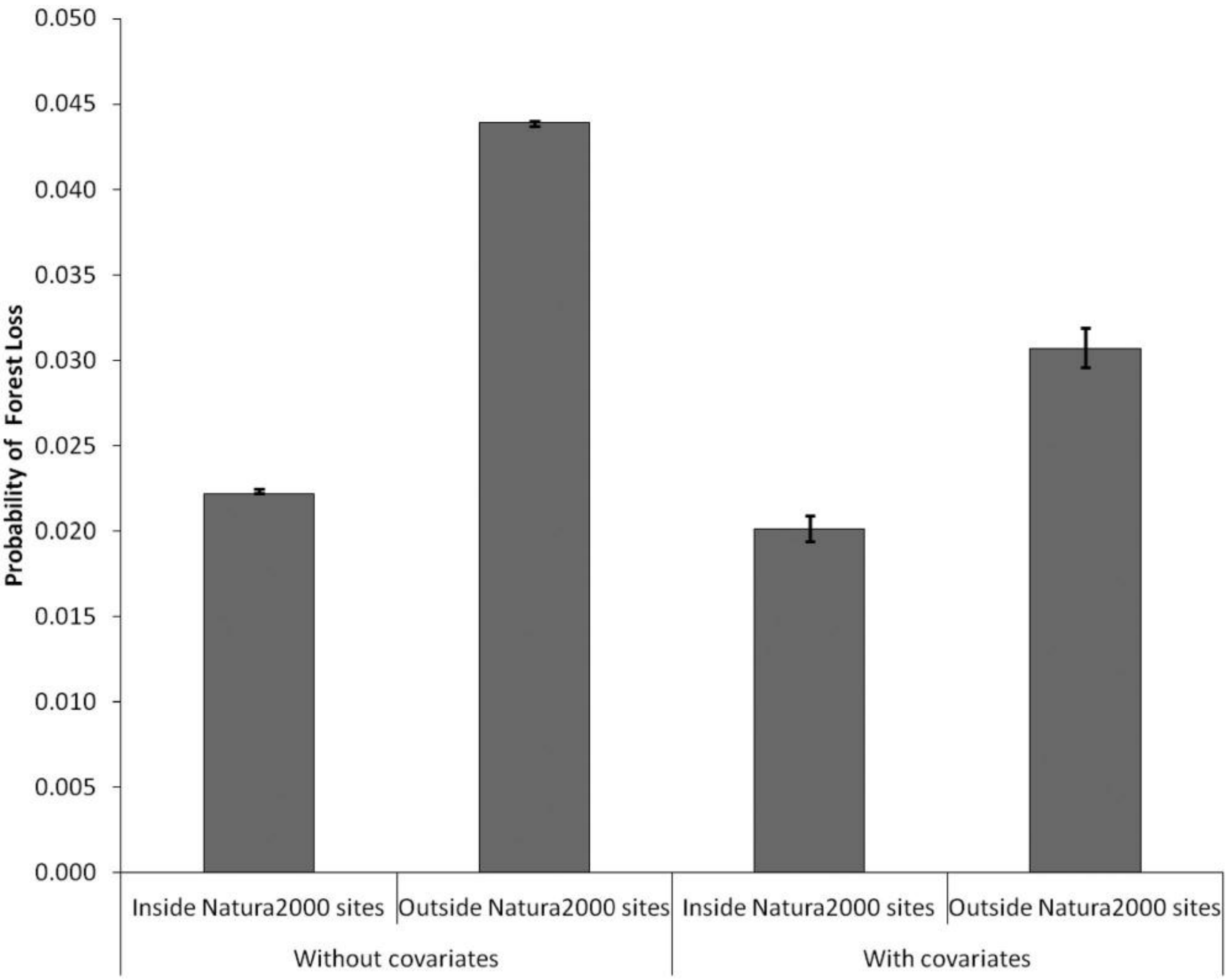

Figure 2 Probability (±1 SE) of significant forest loss between 2000 and 2012 in forested or partially forested 1 km^2 within and outside the Natura 2000 network, modeled with and without covariates (altitude, distance to nearest major or primary road, and human population density). See Appendix S1 for full methods.

benefits is evidenced by increased property values close to such sites (Mourato *et al*. 2010). Most EU citizens live close to Natura 2000 sites (Figure 1), which receive 1.2–2.2 billion visitor days per year yielding annual recreational benefits worth €5–9 billion (ten Brink *et al*. 2011), suggesting that they can "contribute to health, livelihoods and well-being." In one region of Spain, increases in human well-being between 1989 and 2009 were significantly higher within Natura 2000 sites than outside (Bonet-García *et al*. 2015).

Aichi Target 15—carbon and climate change

Estimated below and above ground carbon stocks per unit area in Natura 2000 sites are 43% higher than the average across the rest of the EU (Figure 5), "thereby contributing to climate change mitigation." This importance has been recognized by the EU in its assessment of the economic benefits of Natura 2000 (ten Brink *et al*. 2011). Requirements in the Directives to restore habitats are likely to increase the amount of carbon they store. As some of this carbon is stored in trees, there is a clear synergy with Aichi Target 5. Many areas of high-carbon peatland are captured by the Natura 2000 network (e.g., in Caithness, UK), and are benefiting from EU funding aimed specifically at bogs and mires, and hence carbon storage. No data are available on changes in carbon stocks over time, so it is not possible to assess whether Natura 2000 contributes to the Aichi target to "enhance" carbon stocks.

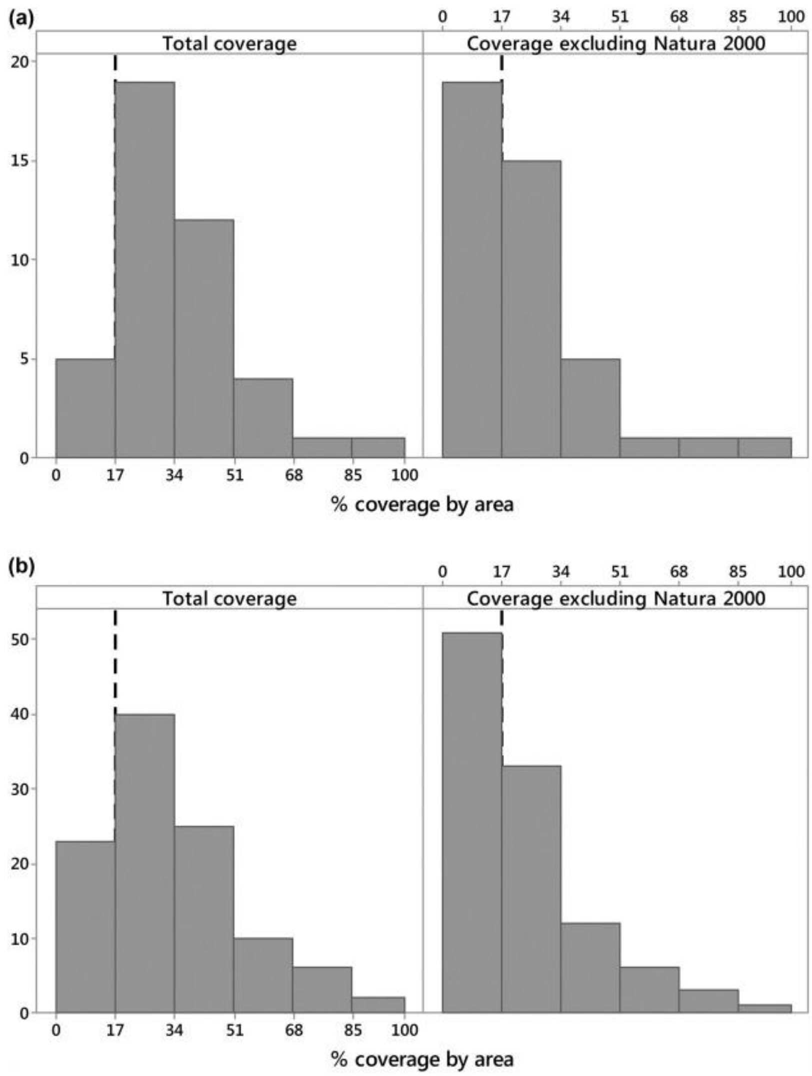

Figure 3 Histograms of percentage coverage (by area) of different ecoregions (Olson *et al*. 2001) by all classes of protected area (left) and by all classes of protected area excluding Natura 2000 (right), (a) across the entire EU and (b) by country/ecoregion combinations. The reference line indicates the 17% protected area coverage target of Aichi Target 11.

Other Aichi Targets

The Nature Directives require EU Member States to account for biodiversity in development and planning processes, and the reporting requirements of the Directives mean that status, threats, and trends in biodiversity are included within national reporting systems (Aichi Target 2). Most Natura 2000 sites contain agriculture and forestry (Tsiafouli *et al*. 2013) which designation requires be sensitively managed, thus contributing to Aichi Targets 7 and 8. The Directives encourage the preparation of management plans for selected sites and species, thus contributing to the preparation of national action plans (Aichi Target 17). Both Directives require Member States to collect scientific information on the status and trends of target species and habitats and changes in them (Aichi Target 19). The EU LIFE funding instrument, established in part to support the implementation of the Nature Directives, has since 1992 provided around €3.4 billion to support over 4,000 biodiversity conservation projects, and Article 8 of the Habitats Directive requires Member States to establish prioritized action frameworks to support the financing of nature conservation (Aichi Target 20).

Other multilateral environmental agreements

The EU and its Member States are bound by a number of other agreements, including the Convention on the Conservation of European Wildlife and Natural Habitats (Bern Convention; 1979), the Convention on Migratory Species (CMS, or Bonn Convention; 1979) and the

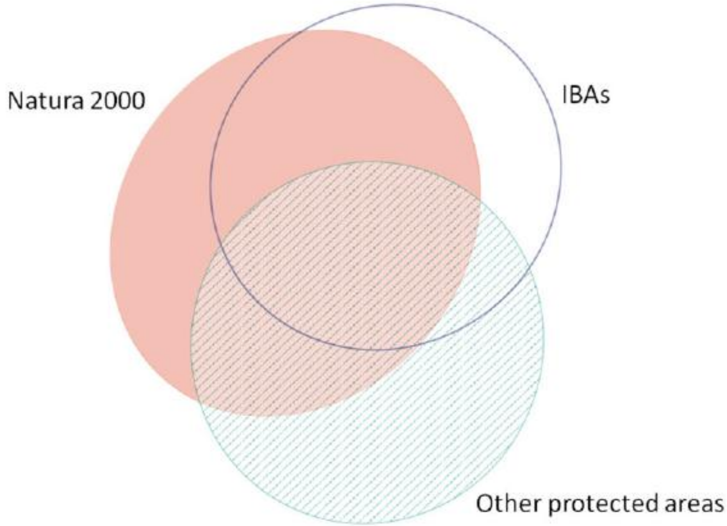

Figure 4 Proportional-area Venn diagram of the relative coverage by area of Natura 2000, Important Bird and Biodiversity Areas (IBAs) and other protected areas (all IUCN categories, not necessarily designated for nature conservation), and their respective intersections. Plotted using "eulerAPE" (Micallef & Rodgers 2014), which uses ellipses to enable the exact proportions, by area, of each segment to be represented.

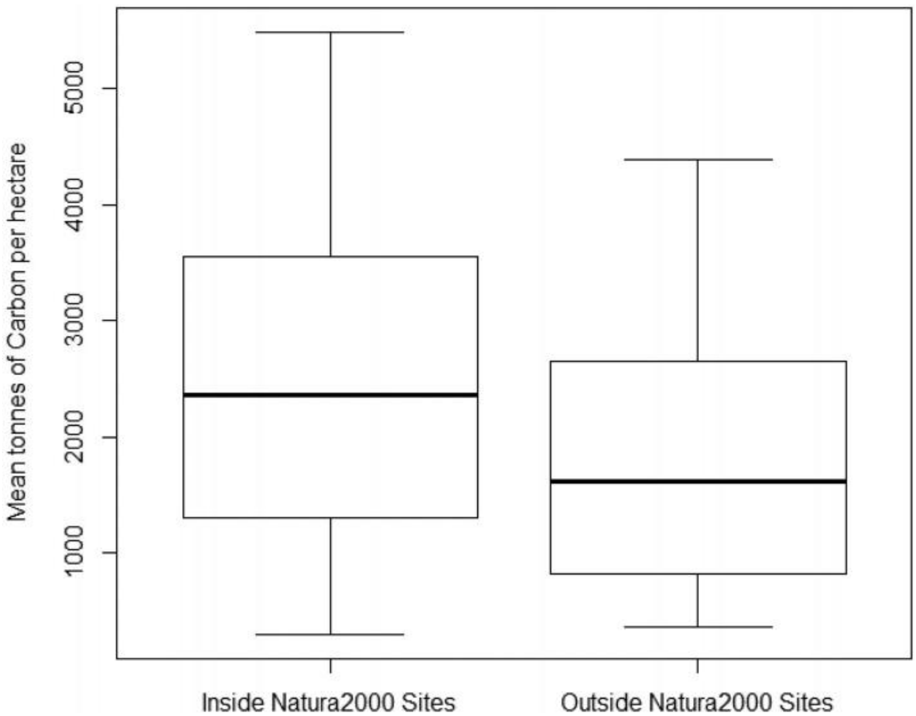

Figure 5 Box-and-whisker plot of total above and below ground carbon per hectare inside and outside Natura 2000. The horizontal bar indicates the median, the box contains the interquartile range and the whiskers the 95 percentile limits.

Convention on Wetlands (Ramsar Convention; 1971). The Directives are the means by which the EU meets its obligations under the Bern Convention, and Natura 2000 is the contribution from EU Member States to the Bern Convention's Emerald Network. The Birds Directive makes special mention of the importance of migratory species and Article 4 requires Member States to protect areas of importance for migratory species, thus contributing directly to the CMS and its agreements, such as the Agreement on the Conservation of African-Eurasian Migratory Waterbirds (AEWA). Article 4 also requires "Member States to pay particular attention to the protection of wetlands of international importance," as defined under the Ramsar Convention; indeed, 83.3% of the area of the 659 Ramsar sites in the EU for which digital boundaries are available falls inside Natura 2000. The 106 natural and mixed UNESCO World Heritage Sites in the EU whose digital boundaries were available for analysis have 91.7% of their total area captured by the Natura 2000 network.

Policy implications

The Directives set out principles and targets that are enshrined in national law, thereby creating the legal framework necessary for meeting the requirements of other international obligations. We suggest that fuller implementation of the EU Nature Directives will help the EU and its Member States to meet their commitments under a number of agreements. This will require the wider designation and better management of new sites, particularly IBAs, a significant improvement in the management of existing sites, particularly in more recently acceded states (Křenová & Kindlmann 2015), significantly better representation in the Appendices of the Directives of currently underrepresented taxa (e.g., Gruber *et al.* 2012; Rubio-Salcedo *et al.* 2013) and better capture of threatened species (Maiorano *et al.* 2015), and substantially increased funding. Furthermore, the extent to which the Nature Directives achieve their aims and contribute to other international agreements will continue to depend largely on the degree to which they are undermined by other policy frameworks, such as those relating to agriculture (European Environment Agency 2015).

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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:

Figure S1. Percentages of the each EU Member State's population living within different distances of Natura 2000 sites.

Figure S2. Number of IBAs qualifying under different criteria that are wholly (>98%), partially (2-98%) or not (<2%) covered by Natura 2000 sites across the EU.

Figure S3. Percentage coverage by area of all IBAs by Natura 2000 sites in each EU country.

Appendix S1.

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POLICY PERSPECTIVE

Sufficiency and Suitability of Global Biodiversity Indicators for Monitoring Progress to 2020 Targets

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Aichi Targets; biodiversity; biodiversity indicators partnership; conservation; convention on biological diversity; indicators; strategic plan.

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Abstract

Biodiversity indicators are widely used tools to help determine rates of biodiversity change and the success or failure of efforts to conserve it. However, their sufficiency and suitability in providing information for decision-makers is unclear. Here, we review the indicators brought together under the Biodiversity Indicator Partnership to monitor progress towards the Aichi Targets to determine where there are gaps. Of the 20 Aichi Biodiversity Targets, Targets 2, 3, and 15 are missing indicators entirely. Scoring the indicators in relation to their alignment, temporal relevance and spatial scale shows additional gaps under Targets 1, 13, and 16–20. Predominately, gaps were found to be socio-economic in nature (i.e., benefits, pressures, and responses) rather than status-related (i.e., states), principally due to a poor alignment between the indicator and the text of the Aichi Target. Hence, it is critical that existing indicators are properly resourced and maintained and new indicators developed to be able to effectively monitor biodiversity and its influencing factors to 2020 and beyond.

Introduction

Indicators have become essential for effective policy formation and political decision-making (Mace & Baillie 2007; Nicholson *et al.* 2012). In 2010, The Parties to the Convention on Biological Diversity (CBD) committed to a significant reduction in the rate of biodiversity loss by 2010 (Conference of the Parties to the Convention on Biological Diversity 2010); however, tracking progress towards the target was hindered by an underdevelopment of, and underinvestment in, biodiversity indicators (Walpole *et al.* 2009). In 2010, renewed commitments to halt biodiversity were made as the new Strategic Plan for Biodiversity 2011–2020 was adopted. The plan is supported by 20 "Aichi Biodiversity Targets" (henceforth, Aichi Targets) covering "pressures" on, "states" of, and "benefits" from biodiversity and "responses" to the biodiversity crisis. As the new Strategic Plan considers a number of subjects not covered under the 2010 Target (Conference of the Parties to the Convention on Biological Diversity 2010) the sufficiency and suitability of global biodiversity indicators for monitoring progress remains open to debate.

Whilst a large number of biodiversity indictors exist, differing levels of spatial and/or temporal coverage means that not all are applicable to monitor progress toward the 2020 target (Tittensor *et al.* 2014). The Biodiversity Indicators Partnership (BIP) is the principle mechanism supporting the delivery of indicators for international governance via the CBD's Strategic Plan (e.g., Conference of the Parties to the Convention on Biological Diversity 2010). Since its establishment, BIP partner organizations and their indicators have been successfully mobilized to track how biodiversity has changed (Butchart *et al.* 2010; Tittensor *et al.* 2014); demonstrate that biodiversity loss is continuing (e.g., WWF 2014; Regan 2015); and show that society needs to mobilize greater resources, or allocate them more effectively, to tackle the biodiversity crisis (Tittensor *et al.* 2014). Nevertheless, despite their widespread use, the current set of global biodiversity indicators that form the BIP remain largely unevaluated in their capacity to report meaningfully on global targets (Collen & Nicholson 2014).

The aim of this study is to objectively assess the sufficiency and suitability of the global BIP indicator suite, highlighting areas in need of additional development, thereby aiding data-informed decision making and effective conservation interventions.

Methods

The current BIP indicator suite (http://www.bipindicators.net/globalindicators)[1] was scored according to three criteria: their alignment to the relevant Target, temporal relevance to the Strategic Plan and their spatial coverage. Given the broad and multifaceted nature of the Aichi Targets, scoring was conducted against target "elements" which represent discrete textual aims within each Target. The elements are based on those used in Global Biodiversity Outlook 4 (Secretariat of the Convention on Biological Diversity 2014) (Table S1). This scoring scheme enabled us to identify two types of information insufficiency: (1) situations where there are no suitable indicators and (2) situations where there is one or more indicators but their poor alignment, spatial coverage or temporal relevance limit their utility.

1. ***Alignment to Aichi Target***: *how well does an indicator align to the text of the relevant Aichi Biodiversity Target elements*? The level of alignment for each indicator with a Target element varies; where there was overlap in the indicators the scores came from Tittensor *et al.* (2014), for new indicators the same method as used in Tittensor *et al.* (2014) (Table S2) was repeated in order to assess qualitatively whether we consider them to be of "low," "medium," or "high" alignment (Table 1).
2. ***Temporal relevance***: *are there sufficient pre- and post-2010 data points and planned data points for the period 2010–2020 to enable accurate assessment of implementation of the Strategic Plan?* Scoring was based upon the number of annual data points available during the Strategic Plan period. Estimates of the number of data points available from 2011 to 2020 were based on the temporal spacing of data points to date and in some cases in consultation with indicator partners (Table 1).
3. ***Spatial coverage***: *what is the spatial scale of the indicator*? The scores given to each of the indicators were assigned according to the criteria adopted in Tittensor *et al.* (2014) (Table 1).

The scores from each criteria were combined for each indicator, and then across indicators, to produce one score per Aichi Target. Combined scores were generated per indicator based on weightings: alignment scores were multiplied by 2, temporal relevance by 1.5 and spatial scores were unweighted (see Table S2 for justification). In order to aid visualization, the indicators were categorized as high, medium, or low relative to the scores of the other targets using the Natural Jenks method (Jenks 1967).

Results

For three of the twenty Aichi Targets (2, 3, and 15) there were no global indicators within the Biodiversity Indicator Partnership (BIP) (Figure 1), all of which are related to "responses" to biodiversity loss. Aichi Target 2 considers integrating biodiversity values into development and poverty reduction processes, while Aichi Target 3 aims to eliminate or reform incentives that are harmful to biodiversity. Aichi Target 15 relates to ecosystem resilience, conservation, and restoration. Of the 54 elements that form the 20 Aichi Targets, under half (46%) have indicators; Aichi Target 13 (genetic diversity) has the highest proportion of gaps, with only one of its five elements having an indicator (Figure S1).

For the seventeen Aichi Targets that do have indicators, the indicators varied in regards to their alignment to the text of the Aichi Targets, spatial coverage and temporal relevance:

Alignment: Thirteen Aichi Targets had at least one indicator that aligned well with the corresponding text (65%) (Figure 2), while all of the indicators that represent Aichi Targets 8, 14, 18, and 19 had low alignment. For example, the Ocean Health Index, Red List Index (species used for food and medicine) and the Red List Index (pollinating species) are indicators under Aichi Target 14 but were not considered robust proxies for the Aichi Target text. Over the 54 elements, 12 (33%) of the indicators scored low in regards to alignment; 5 (9%) had a medium score; 18 (22%) scored highly, and 19 (35%) had no indicators. (Figure S1).

Temporal relevance: Sixteen of the Aichi Targets had at least one indicator with high temporal relevance and

Table 1 The scoring system applied through this review to support indicator gap analyses. Amended from Tittensor *et al.* (2014)

Score	Alignment to Aichi Target element	Temporal relevance	Spatial coverage
High/good	The level of alignment for each indicator with an Aichi Target element was determined (qualitatively) to be of "low," "medium," or "high" alignment	According to current trends 5 ≥ data points are projected between 2010 and 2020. This gives greater sensitivity to change than indicators that scored medium	"Good," as defined by Tittensor *et al.* (2014): 5 + continents (more than 20 countries total)
Medium/ moderate	The level of alignment for each indicator with an Aichi Target element was determined (qualitatively) to be of "low," "medium," or "high" alignment	According to current trends 3–4 data points are projected between 2010 and 2020. A trend can be inferred but with large uncertainty	"Moderate," as defined by Tittensor *et al.* (2014): 3–4 continents (more than 10 countries total); 5 + continents (fewer than 20 countries total)
Low/poor	The level of alignment for each indicator with an Aichi Target element was determined (qualitatively) to be of "low," "medium," or "high" alignment	According to current trends 2 ≤ data points are projected between 2010 and 2020. This is insufficient information to analyze a trend	"Poor," as defined by Tittensor *et al.* (2014): 1–2 continents (no matter how many countries); 3–4 continents (less than 10 countries total)

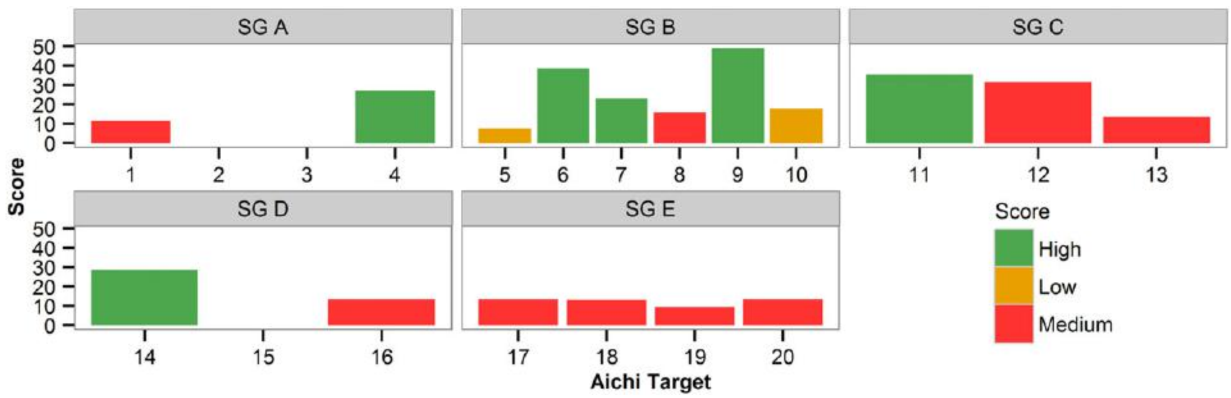

Figure 1 The score for the global indicators available for each Aichi target combined for alignment, temporal relevance, and spatial coverage.

19 (35%) had no indicators (Figure 2), with the indicator for Aichi Target 18 (Index of Linguistic Diversity) scoring low due to the more irregular frequency in which it is updated. At the element level, 5 (9%) indicators scored low; 2 (4%) had a medium score; 28 (52%) scored highly, and 19 (35%) had no indicators. (Figure S2).

Spatial coverage: Sixteen Aichi Targets had at least one indicator with high spatial coverage (80%) (Figure 2), Aichi Target 1 (awareness of biodiversity) has a single indicator, which scored "low" for spatial coverage. The indicator in question is the Biodiversity Barometer, which uses data from only six countries—Brazil, China, France, Germany, the United Kingdom, and the United States. At the element level, 4 (7%) of the indicators scored low in regards to spatial coverage; 2 (4%) had a medium score; 29 (53%) scored highly; and 19 (35%) had no indicators (Figure S3).

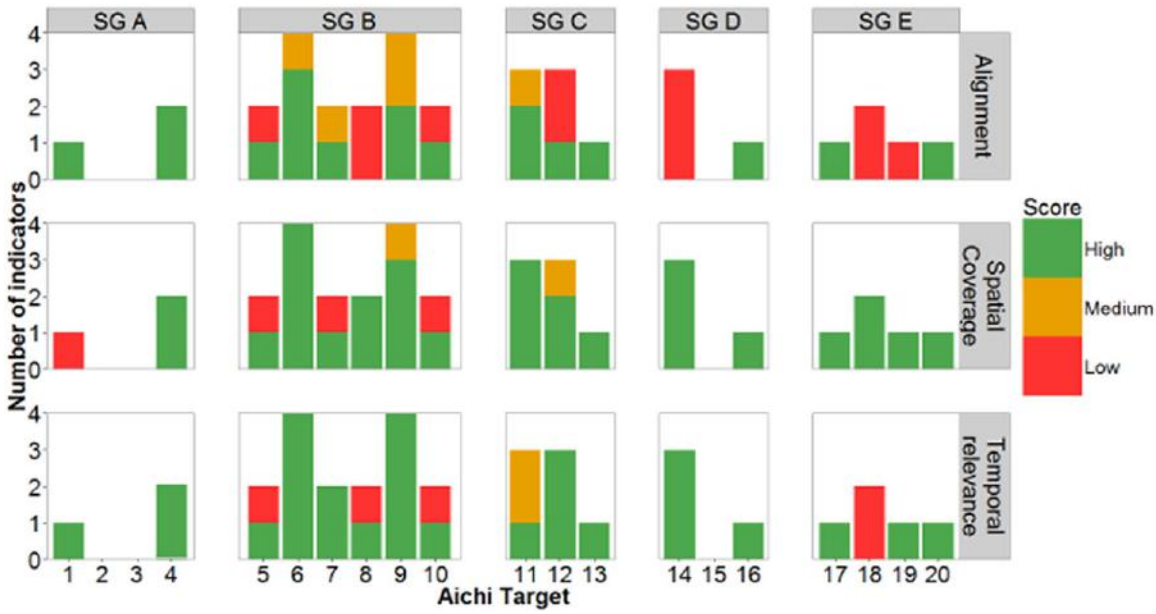

Figure 2 The score for the global indicators brought together under the Biodiversity Indicators Partnership (BIP) available for each Aichi target in relation to their alignment, temporal relevance, and spatial coverage.

Combined scores

Three Aichi Targets had relatively high indicator sufficiency scores (6, 9, and 11) whilst ten had relatively low scores (1, 5, 8, 10, 13, 16, and 17–20), with the largest number (four) sitting under Strategic goal E "Enhance implementation through participatory planning, knowledge management and capacity building." For a number of targets (13, 16, 17, and 20), a low score was a result of the Aichi Target having only one indicator (Figure 1). A separate analysis was conducted in which weightings were not applied in order to test the sensitivity of the findings, the results show the relative score for each Aichi Target is almost identical with the only expectation being Aichi Target 11, which was classified as "medium" when no weightings were used as opposed to "high" using weightings (Supplementary material 2).

Discussion

The rapid development of online databases, indicators and indicator partnerships continues to improve our ability to quantify progress toward international biodiversity targets (Collen & Nicholson 2014; Tittensor *et al.* 2014). Nevertheless, there remain three Aichi Targets (2, 3, and 15) for which no global indicators have been developed. For Aichi Target 2, the difficulty lies in the undefined nature of "biodiversity values," the challenge of measuring integration and the lack of universally accepted ecosystem accounting and reporting frameworks. In particular, the diverse nature of national and subnational processes and plans relating to development, poverty reduction and planning poses a challenge. For many countries, these will not form just three distinct documents but may encompass a huge number of different policies, plans and strategies, hindering both national level reporting and the development of a uniform global indicator. With respect to Aichi Target 3, the majority of incentives occur at the national to regional scale, and again may vary greatly in nature. Global indicators for both these targets would therefore need to collate together information on various national/regional incentives relevant to each individual country, which would be a resource-intensive activity. For Aichi Target 15, a major challenge is that the target is so vague—what does resilience relate to, do we mean resilience of ecosystems to climate change or to threats in general? Either way the concepts are multifaceted and broad, not lending themselves to easy measurement. Furthermore, the definition of degradation is also difficult and not standardized, and will vary between ecosystem types (Leadley *et al.* 2014).

For those seventeen Aichi Targets with indicators, a sizeable proportion had shortcomings relating to their alignment, spatial coverage or temporal relevance. That is, while we do have indicators, their ability as useful proxies remains insufficient, thereby leading to a limited or circumscribed view of progress. Specifically, indicators for Aichi Targets 1, 13, 16, 17, 18, 19, and 20 need to be enhanced in order to improve our ability to monitor biodiversity and to obtain a comprehensive picture of status and trends that can accurately inform decision-making. For the majority of these targets, the primary challenges lie in their openness to diverse interpretations due to both their unspecific nature and their broad and multifaceted coverage. For example, Target 15 covers resilience, degradation, climate change mitigation, and the contribution of biodiversity to carbon stocks; these are four, in many ways distinct and not necessarily related, ideas. As such one recommendation that comes from this work is that future target wording should be as unambiguous as possible and should at least theoretically be possible to measure.

The majority of gaps and insufficiencies relate to our ability to track sociological / economic process (i.e., benefits, pressures and responses) rather than the status of biodiversity. In particular, there is a clear gap in our ability to track and monitor the effectiveness of capacity building, with all indicators under Strategic Goal E (Enhance implementation through participatory planning, knowledge management and capacity building) scoring poorly. This differentiation likely reflect the long history of biologists collecting information compared to those disciplines seeking to link socio-economic factors to biodiversity (Balmford *et al.* 2005). A lack of capacity at the national level in formulating regulatory policies for biodiversity conservation has long been a significant challenge (Adenle 2012). Furthermore, insufficient funding has been shown to be one of the main barriers to achieving the 2010 Target (Waldron *et al.* 2013); based on the results of this study, it is apparent that in the context of biodiversity indicators, similar challenges remain.

Conclusion

The indicators brought together under the BIP provide the best possible framework from which to monitor progress toward the Strategic Plan for Biodiversity 2011–2020, with at least one global indicator available for 17 of the 20 Aichi Biodiversity Targets at present. Multipurpose Indicators, such as the Red List Index, were found to be of particular value due to their ability to be disaggregated to report against various targets. This review has highlighted the following issues and recommendations. In addition to adequately maintaining and resourcing the existing indicators, there is a need to develop more meaningful indicators to track societal and demographic changes, which are driving forces that exert pressures on the environment. Furthermore, building capacity, finance, and technological innovation, in conjunction with providing an enabling policy environment, is a particular need. In order to effectively overcome these limitations and challenges, the international biodiversity community and policy and decision makers need to come together, to work across multiple sectors and jointly develop solutions to ensure all aspects of quantifying international biodiversity policy effectiveness are covered to 2020 and beyond.

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1. With the exception of "Trends in potentially environmentally harmful elements of government support to agriculture (producer support estimate)," which was added to the BIP subsequent to the completion of this manuscript.

Supporting Information

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

Figure S1. The relative Aichi Target scores with and without the weighting factor.

Table S1. The breakdown of each Aichi Target into its respective elements and their alignment, temporal relevance and spatial coverage scores

Table S2. Justification of the multipliers used in the analysis

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