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Review

Multiple-use protected areas are critical to equitable and effective conservation

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SUMMARY

Protected areas are widely used to stem biodiversity loss, with global conservation policy mandating their use in meeting area-based conservation targets, such as the recent target to protect 30% of Earth's surface by 2030. Over the last two decades, the role of multiple-use and other protected areas in meeting conservation targets has increased, now representing 55% of terrestrial and 75% of marine protected areas globally. Given their substantial contributions to meeting area-based targets, we review evidence of multiple-use protected areas in meeting global policy goals. We found that multiple-use protected areas can be effective in conserving biodiversity and are more likely to be under equitable governance arrangements than their strict counterparts. Our results show that multiple-use and other protected areas provide important environmental and socio-economic benefits that are needed to achieve the 2030 target. Closing knowledge gaps around when and where they are most effective and equitable is needed to ensure that further growth in the protected and conserved estate meets global policy goals.

INTRODUCTION

Protected areas are frequently called the "cornerstone" of global biodiversity conservation strategies. These areas represent one of the most significant resource use allocations on the planet and the largest planned land uses,[1,2] covering approximately 17% of land mass and 9% of marine area.[3] Over the last two decades, there has been growth in global policy and commitments that emphasize the role of area-based measures in achieving global biodiversity targets. For example, the area-based targets for 2010, 2020, and 2030 have sought to expand conservation extent as well as to recognize a larger set of governance mechanisms for area-based conservation including both IUCN-class protected areas (termed protected) and other effective area-based conservation measures (OECMs) (termed conserved). Since 2000, the overall coverage in protected and conserved areas has seen a 50% growth on land and more than a 10-fold increase in the sea (Figure 1). Much of the growth in coverage has come from multiple-use and other protected areas (those without an assigned IUCN category); today, these categories constitute 55% of protected and conserved areas on land and 75% in the oceans (Figure 1).

The push over the coming 8 years to meet the "30 × 30" target to protect 30% of the planet by 2030, as set in the Kunming-Montreal Global Biodiversity Framework (GBF),[4] would result in a further ~50% growth of terrestrial protected and conserved areas and further tripling of marine areas. Drawing from the strongest published evidence (impact evaluations and surveillance monitoring studies that endeavor to control for confounding factors), the published literature suggests that protected areas can be effective in reducing threats[5–7] and conserving biodiversity within their boundaries.[8] However, their ability to do so varies significantly, and thus local variation in outcomes can be both positive and negative for nature and people. Given this evidence of variable outcomes, it remains unclear if protected areas are fully delivering on their promises, let alone on the potential benefits (or lack thereof) of such a speedy increase by 2030. Furthermore, evidence on the relative benefits of different protected area types (i.e., strict IUCN I–IV vs. multiple-use IUCN V–VI) is limited, with only a small number of studies designed to detect relative impacts of each (e.g., Ferraro et al.,[9] Gill et al.,[10] and Nelson and Chomitz[11]). To guide global conservation policy that ensures that the growth of protected and conserved areas is meaningful and delivers the desired outcomes, an understanding of the relative effectiveness of protected area types as well as the contexts in which they are most suitable is needed.

In this review, we take a data-driven approach to explore the current role that strict (IUCN categories I–IV) and multiple-use (IUCN V–VI) protected areas have played in terms of the growth of the global protected area estate and the context in which

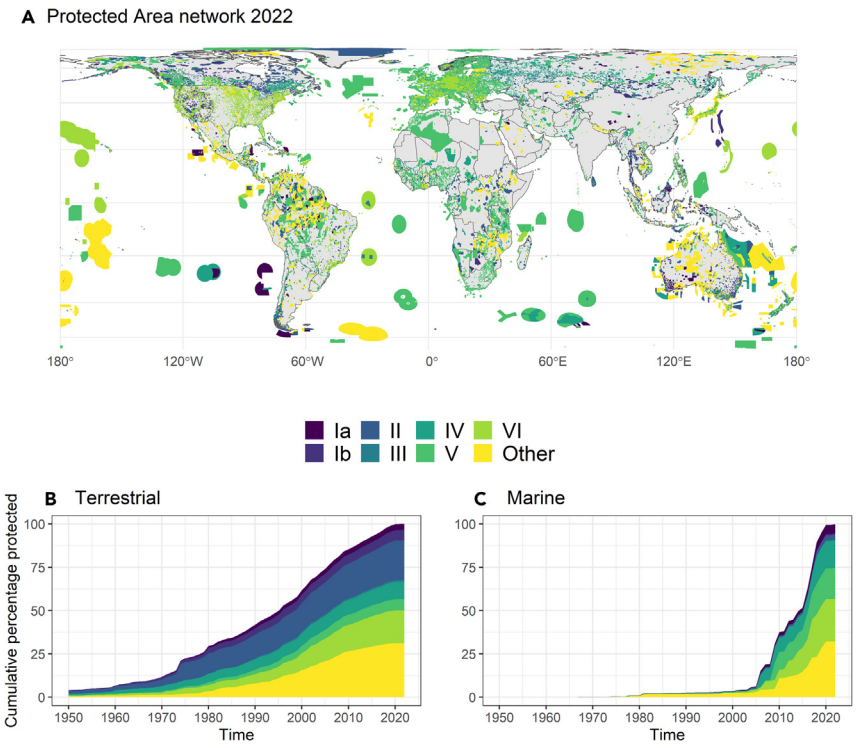

Figure 1. Current global protected area estate across terrestrial and marine realms

(A) Geographic distribution of protected areas by IUCN category.

(B) Growth in terrestrial protected areas over time by IUCN category displayed as cumulative percentage protected.

(C) Growth in MPAs over time by IUCN category displayed as cumulative percentage protected.

In general, IUCN I–IV are considered strict protection, while V–VI are multiple-use zones. "Other" refers to protected areas without a nominated IUCN category.

benefits (or harms) have been delivered to both nature and people. To this end, we synthesize key historical moments related to global protected area policy (such as the 2010, 2020, and 2030 area-based targets and their contexts), explore the relationship that global policy has had with shaping protected area growth, and then analyze patterns of growth in protected areas across IUCN categories and the evidence of impacts of these on the environment and people. We draw upon the published literature and supplement this with re-analysis of global data to compare the overall performance of protected areas across categories. We find that multiple-use (V–VI) and other (those without an assigned category) protected areas have grown substantially as a proportion of the global protected area estate. Given their growing importance, in terms of meeting targets, we further explore evidence of effectiveness and equity. We find that multiple-use protected areas can deliver benefits to nature and people hand in hand, rather than in trade-off, but that the evidence points to local variation in the nature of benefits and harms. Based on our findings, we look to the coming decade and what additional evidence is required to guide the proposed growth in global protected areas and OECMs to ensure that they contribute to effective and equitable conservation.

A SHORT HISTORY OF PROTECTED AREA DEBATES

The growth of protected areas has slowly gathered momentum over the twentieth century.[12] A focus on protected areas at the global level accompanied the birth of the UN and helped give rise to the IUCN in 1948.[13] The centrality of protected areas as a tool to address biodiversity loss (over the very many other available tools) was formalized in global policy in the 2010 biodiversity target adopted by the 7th Conference of the Parties (COP) to the Convention on Biological Diversity (CBD) in 2002. Specifically, goal 1 had associated targets 1.1 and 1.2 that read the following: "At least 10% of each of the world's ecological regions effectively conserved; and Areas of particular importance to biodiversity protected." This goal formalized and extended the related aim in the Caracas Action Plan, a key outcome of the 1992 World Parks Congress, which called for the protected area network to cover at least 10% of each major biome by 2000. In the subsequent two decades, further iterations on these goals and targets have occurred alongside other hallmark policies and public debates. We highlight here a brief history of major moments in global protected area policy (see Table S1 for a more comprehensive list). We emphasize those policies that relate to the interplay between environmental and social outcomes because these reflect current debates related to effective and equitable conservation and the roles of multiple-use protected areas, OECMs, and other alternatives to strict protected areas (e.g., those that countries choose to not match to an IUCN category).

The biodiversity target for 2010, adopted in 2002 at COP-7, was "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." This target attempted to resolve heated debates over the apparent trade-offs between development and conservation through its coupled articulation to reduce biodiversity loss and alleviate poverty.[14] The following year, the 5th IUCN World Parks Congress 2003 was held in Durban. This meeting was notable, as it attracted an unprecedented attendance of Indigenous and local community groups and representatives. The debate around the role and types of protected areas was at the forefront of this congress with a voiced concern that expansion of protected areas would exacerbate harms to local livelihoods already in play due to conservation.[14]

An important outcome from this public debate was the Durban Accord, which put forward a number of goals and actions to increase the benefits of protected areas to society and to ensure they contribute significantly to global agendas on sustainable development. Related to this, a resolution was passed asking the IUCN to update the IUCN protected area categories.

Published in 1994, the IUCN protected area category system has been widely used to guide policy, planning, and reporting. The typology is based on protected area management objectives and has six categories spanning a continuum of strictness in terms of allowed resource extraction: strict nature reserves (Ia), wilderness areas (Ib), national parks (II), natural monuments or features (III), habitat/species management areas (IV), protected landscapes (V), and protected areas with sustainable use of natural resources (VI). At the 2004 World Conservation Congress, the conservation-poverty debate continued, and a resolution was passed to undertake a 4-year consultation to identify points of contention and possible changes in the categories system. The definitions of, roles for, and management of IUCN categories V and VI protected areas (i.e., multiple-use protected areas) were at the center of continued debate during this consultation process.[1] The debate emerging from the Durban Accord was about whether less strict forms of protection (i.e., multiple-use protected areas) deserved the term "protected area." In the end, the IUCN provided guidance stating that protected areas can have multiple objectives but "in the case of conflict, nature conservation must be the priority."[15]

The 2020 and 2030 CBD area-based conservation targets, Aichi target 11[16,17] and GBF target 3,[4] outline a role for both protected areas and OECMs in meeting expanded area-based conservation targets. Guidance for OECMs and their criteria were adopted by CBD parties in 2018, and their contribution to targets are now formally tracked alongside protected areas[18,19] (see Box 1 for a discussion on the role of OECMs and their key differences from protected areas). Additionally, GBF target 3 explicitly recognizes the role of Indigenous and traditional territories in meeting the target. This text was added during the 2022 COP in Montreal in response to concerns about the implications of the 30 × 30 for Indigenous peoples and local communities given that some protected areas have perpetuated colonial ideologies and resulted in injustices to these groups.[20] Although the text falls short of Indigenous representatives' call for their territories to be a distinct category separate from protected areas and OECMs, the importance of recognizing these groups' rights and the role of their territories in meeting conservation outcomes is clear in the target (representing a significant variation to previous iterations):

> Ensure and enable that by 2030 at least 30% of terrestrial, inland water, and coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem functions and services, are effectively conserved and managed through ecologically representative, well-connected, and equitably governed systems of protected areas and other effective area-based conservation measures, recognizing Indigenous and traditional territories where applicable, and integrated into wider landscapes, seascapes, and the ocean while ensuring that any sustainable use, where appropriate in such areas, is fully consistent with conservation outcomes, recognizing and respecting the rights of Indigenous peoples and local communities, including over their traditional territories.

Target 3 reflects the continued move by the conservation community beyond a sole focus on strict protected areas toward a range of complementary area-based protection measures that can be implemented to suit diverse contexts (see Box 1 for further discussion around complementarity of measures).

This short historical account of global protected area policy emphasizes that area-based targets have accelerated over the last two decades and that the types of area-based measures have diversified. Have these changes in policies influenced the rate at which countries are protecting land and sea or the types of mechanisms they are using? Multiple-use and other protected areas (those without an assigned IUCN category) have grown substantially over the last two decades; today, these categories constitute 55% of protected and conserved areas on land and 75% in the oceans (Figure 1; see Figure S1 for further details).

Using the concept of "hot moments,"[21] we explored the rate of change of growth of protected areas and tested for hot moments (time periods above the mean rate of increase) around key policy time periods (Box 2). We found that the global protected area has grown substantially since 2000, and the majority of that growth has come from multiple-use and other protected area categories over the last decade, with these forms of protected areas forming the largest proportion of protected areas (Figure 1). There are significant differences in growth across terrestrial and marine protected area estates, and so we further consider these patterns specific to realm.

Within the terrestrial estate, considering cumulative growth by IUCN category, the proportion of area in IUCN II has had a linear rate of increase and in VI and other (those in the WDPA database with no IUCN category recorded) have had larger growth rates, and the remaining categories have been stable (Figure S1). Considering annual rates of change within the growth curves by category, we found limited evidence of hot moments around global political landmarks, constrained primarily to IUCN V and VI and other protected areas in the 2000–2005 period (indicated by peaks in annual changes within shaded policy periods; Figure 2; Box 2) and within IUCN II in the 2000–2005 and 2018–2020 periods.

Within the marine estate, the proportion of area protected as IUCN category I and III has been stable over the last two decades, but all other categories have had steep increases in area, with the largest growth occurring in IV–VI and other areas (Figure S1). Most IUCN categories also had some alignment with peaks in annual growth rates within policy periods, with the strongest evidence of hot moments being for Ib between 2018 and 2020 and for V and other peaking in 2008–2010, which align with the lead up to meeting global target dates of 2010 and 2020 (Figure 2; Box 2).

Outside of the limited number of hot moments identified above, rates of change were highly stochastic and thus do not support consistent identification of global-policy-driven hot moments. This is consistent with findings from Radeloff et al.[21] and suggests that while there are certainly hot moments in protected area growth, global targets are not a direct primary driver. There is strong evidence in the literature of national priorities driving hot moments[12,21,22]; however, the role of global targets in driving national priorities remains unclear.

PROGRESS TOWARD GLOBAL TARGETS

If achieved, GBF target 3 will see a speedy increase in the protection of 30% of land and sea by 2030. To draw lessons for

Box 1. OECMs as area-based conservation policy tools

Other effective area-based conservation measures (OECMs) first appeared in CBD policy in 2010, with Aichi target 11 specifying "17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas … are conserved through … protected areas and other effective area-based conservation measures." However, little attention was given to OECMs until 2018, when CBD parties adopted a formal definition for the policy tool as

> a geographically defined area other than a Protected Area, which is governed and managed in ways that achieve positive and sustained long-term outcomes for the *in situ* conservation of biodiversity with associated ecosystem functions and services and where applicable, cultural, spiritual, socio–economic, and other locally relevant values.

Unlike protected areas, which must be "dedicated" or "designated or regulated and managed to achieve specific conservation objectives," according to definitions provided by the IUCN and CBD, OECMs are defined by effectiveness in conserving biodiversity rather than their objectives. The OECM policy tool thus provides opportunities to recognize or enhance existing or new managed areas that sustain biodiversity, irrespective of their objectives.[88] Managed areas that could be recognized as OECMs are diverse and include lands and seas managed by Indigenous peoples and local communities, historic wreck reserves, sacred sites, and areas managed by production sectors such as fisheries and forestry. Designation of these managed areas as OECMs depends on the consent of relevant governing bodies and whether the area meets CBD's definition for OECMs.[89]

The addition of OECMs to the conservation toolbox is suggested to advance two tenets for area-based conservation enshrined in Aichi target 11: equitable governance and biodiversity conservation effectiveness.[18] By enabling recognition of the value of managed areas other than protected areas to sustaining biodiversity, the OECM tool enhances recognition equity, which relates to respect for socio-cultural diversity, including identities, rights, and values. While IUCN V and VI protected areas allow multiple uses, the IUCN guidelines for applying these management categories stipulate that "in the case of conflict, nature conservation will be the priority."[15] This primacy of nature conservation can alienate and disempower actors who manage areas for other objectives or where definitions of what constitutes "nature" and what constitutes "conservation" differ. For example, it is suggested that even in the relatively successful case of Indigenous protected areas in Australia, misalignment of Indigenous values and those espoused by the IUCN protected area definition can undermine Indigenous power and authority.[90] Advancing recognition equity paves the way for equity in distribution and procedure. Recognizing local governance via OECMs helps ensure that benefit-sharing mechanisms and decision-making processes are suited to the socio-cultural context and are perceived as fair by local actors.[91] At the level of the global conservation system, because the OECM policy can be used to recognize managed areas governed to meet local needs and values, they can help alleviate distributional inequities related to the burdens of conservation being experienced locally but the benefits shared globally. Further, OECMs can promote procedural equity by facilitating the participation of those groups who are governing areas that sustain biodiversity but who are not currently involved in conservation decision-making.

Alongside protected areas, OECMs are expected to help bolster the effectiveness of the global conservation system in several ways.[18] The OECM tool provides a means to support or develop management that is "fit" to the relevant social-ecological context, fostering local support, leadership, and compliance.[20] By facilitating institutional fit, OCEMs provide new opportunities to support and incentivize management in working land and seascapes. Further, by increasing the coverage of managed areas delivering biodiversity benefits, OECMs can collectively yield cumulative conservation benefits, including the underachieved qualitative elements of Aichi target 11, for example, ecological connectivity and representativeness.

Ensuring that OECM policy tool delivers on its purported promise to advance equitable and effective conservation will require addressing several key concerns related to its implementation.[18,92] In particular, these relate to misusing the tool to meet area-based conservation commitments without actually stemming biodiversity loss and recognition processes substantially altering or displacing existing governance, disempowering local actors. Addressing these concerns is of pressing importance as CBD parties move swiftly to incorporate OECMs into their conservation portfolios, with 817 cases documented in the World Database on OECMs as of November 2022.

the coming decade, we review progress in meeting Aichi target 11, including its key subcomponents that persist in target 3 (namely coverage, representation, management effectiveness, connectivity, and equitable management). Alongside the review of available evidence, we re-analyze data to draw inferences for each of the target subcomponents across land and sea. Considering the relative growth in multiple-use protected areas over the past two decades, and the signals of hot moments within these IUCN categories in the 2000–2005 policy period, we focus our analysis on comparisons across strict and multiple-use protected areas. These can be difficult to make because what different protected areas are trying to achieve can vary greatly in different circumstances. The meanings of "success," "equity," or "well-being" can be highly context specific, both for biodiversity and for people. Nevertheless, to the extent that comparison is possible, we seek better to evidence the roles that each type have played to date and might be expected to play in meeting the 30 × 30 targets. Based on our analysis, we discuss aspects that require further attention to create a pathway for meeting target 3 and achieving broader desired outcomes for biodiversity and people.

Coverage and representation

The area-based protection elements of the 2020 target have been met for terrestrial targets and have been close to being met for marine areas: 16.98% terrestrial and 8.65% marine areas

Box 2. Do protected area debates motivate growth in the protected and conserved estate?

The global discourse and policy settings for protected areas over the last two decades have emphasized protected areas as a central conservation mechanism, with a particular focus on codifying increasing protected area targets. This emphasis on target setting to achieve conservation activity leads to a natural question of have targets spurred the desired change in protected area designation? Alongside the increased protected area targets, there has been an evolving debate and a set of associated area-based mechanisms designed to address equitable management goals—in particular, the introduction of multiple-use IUCN categories and the term "conserved" areas alongside "protected" as equal tools for achieving protected area targets. A secondary question is, then, has this period of debate resulted in greater support for, and thus an increase in, the declaration of multiple-use protected areas and their overall contributions to meeting global targets? In other words—are there observable changes in the rates of protected and conserved area growth and the type of areas declared that relate to these political debates and global targets?

Our analysis of hot moments (based on slope of growth curves) finds that there is some evidence of hot moments for protected areas around political landmarks; however, these signals are stochastic, and evidence of hot moments is strongest in MPAs and in particular for categories V and VI and other protected areas (Figure 2). For example, IUCN V and VI and other protected areas (those in the WDPA database with no IUCN category recorded or nominated as OECMs) all had maximum rates of change for terrestrial protected areas in the 2000–2005 period and have had stochastic peaks and troughs in rates of change since then, with no major peaks in the 2010 or 2020 target due dates, and MPAs within these classes had hot moments in, or very close to, the 2010 target due date (Figure 2). Other classes are highly stochastic, such as IUCN II and III, which display frequent rate changes for terrestrial protected areas.

Thus, while there are certainly peaks and troughs in rates of protected area growth, hot moments are only sparsely timed alongside global protected area policy periods. Our findings are consistent with, and confirm those of, Radeloff et al.,[21] who also concluded that global factors are likely to have a limited direct role in driving protected area growth. These results point to the question of "how do global target-setting processes influence on-ground protected area activity, if at all?" The lack of direct influence of global targets on protected area growth found in our analysis suggests a hypothesis that the distance is quite large between negotiators engaging in the global policy discussions, to policy makers aligning national policy with global obligations, and onward to the local communities that must then confront changes in their rights structures to deliver on these targets. Thus, ultimately translating a global target to a local action is not directly linked. This is particularly true for terrestrial protected areas. In the case of MPAs, often established in exclusive economic zonesfurther offshore and with fewer rightsholders outside of national governments that must be consulted or compensated, the distance from global target setting to local action is plausibly much shorter and hence why our evidence of global-policy-driven hot moments is much stronger. These are hypotheses worth testing, as answers to these questions would highlight where conservation policy and practice can invest to truly influence growth of protected areas in effective and equitable ways.

are currently protected and conserved[3,23] (Figure 1A). However, the overall representativeness of the protected area estate remains low. At an ecoregion scale, only 44.5% terrestrial and 47.4% marine ecoregions have met their protection targets of 17% and 10%, respectively.[3] Furthermore, some ecoregions have been so seriously degraded that targets can only be met through restoration.[24,25]

Considering the types of protected areas used across time, growth in the global protected area estate has been substantial over the last two decades, with particular growth in IUCN category V and VI and other protected areas (Figures 1B and 1C). The relative use of strict, multiple-use, and other protected areas varies regionally, with multiple-use and other types being a dominant portion of protected areas (>50% of total area) in Africa, Europe, and Oceania (Figure 3). This increase in multiple-use protected areas appears to accelerate around 2010.[26]

Outside the existing protected area estate, there is substantial overlap between terrestrial areas important for biodiversity and areas of human use,[19] including Indigenous territories.[27] Policy makers may decide to meet conservation goals by expanding the protected area estate in these places in order to meet area-based targets and to fill gaps in representation across both land and sea. If so, then this will require embracing management approaches that do not exclude existing human uses and governance modes that recognize and support existing governing actors, especially rightsholders. This emphasizes the role that multiple-use protected areas and OECMs can play in regions such as Africa, Asia, and Central America, where predicted overlap in conservation priority areas and human populations are particularly high.[28] Considering these areas of high overlap, it is worth noting that multiple-use and other types of protected areas are already a dominant form of terrestrial governance in some regions, such as Africa, and they are the dominant form of marine governance in all regions (Figure 3). This suggests that further emphasis on other approaches to conservation outside of area-based mechanisms may be appropriate in these regions (e.g., fishing industry regulations).

Management effectiveness

Within the existing protected and conserved estate, effective management of protected areas is critical for combatting threats to biodiversity.[5–7] Management effectiveness is generally understood as whether protected and conserved areas make a positive difference on biodiversity or on some other metric of interest, and several broad approaches have been employed to assess it. The agreed-upon approach for tracking progress toward the CBD area-based target at the global level is a protected area management effectiveness (PAME) evaluation, which is aligned

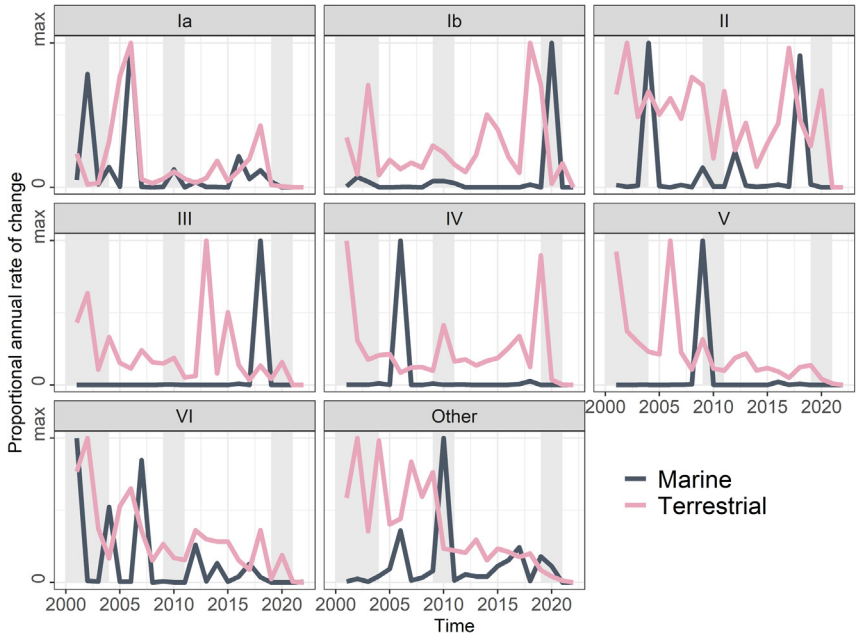

Figure 2. Detecting hot moments based on annual rates of change for protected area growth

Proportional annual rate of change scaled from 0 to 1 by realm and IUCN category where 0 = no increase in protected area for that year and max = maximum rate of change. Peaks in rate of change are relative growth periods, with peaks reaching maximum being particularly hot moments. Policy periods hypothesized to be hot moments are shaded in gray and include 2000–2005 due to large numbers of key events as discussed in the historical narrative, 2008–2010 as the leadup to the first target due date of 2010, and 2018–2020 as the leadup to the second target due date of 2020.

with broader program evaluation, focusing on context, planning, inputs, processes, outputs, and outcomes. Only 18.29% of protected areas have even completed management effectiveness assessments according to the PAME database.[3] The low rates of management effectiveness assessment completion are disheartening, as completing an assessment is a very basic indicator of whether a protected area might be effectively managed.

The other two main approaches to assessing management effectiveness are impact evaluation and surveillance monitoring, applied either to biodiversity outcomes or to level of threats.[29] An impact evaluation approach involves estimating the impact of an area in making a difference by comparing against a counterfactual of an area with no protection or other types of protection while controlling for confounding factors. While this approach provides the highest quality evidence, counterfactual data are often not available, especially at a global level. Surveillance monitoring is a commonly employed method that examines the outcomes of protected areas but without comparing them to alternative governance forms. It involves examining a metric of interest over time within a protected or conserved area. Broadly, the literature employing these two approaches shows that protected areas can be effective in reducing threats[5–7] and conserving biodiversity within their boundaries[8] but that their ability to do so varies significantly, with important predictors including the level of permitted extraction, sufficient funding, socio-economic contextual characteristics, and good governance, including the degree to which stakeholders are meaningfully engaged.[30] For example, a study of 447 terrestrial protected areas using a surveillance monitoring approach found that protected areas were maintaining populations of monitored birds and mammals within their boundaries, with trends being more positive in countries with high development scores.[8] In this study, protected area design characteristics (size, shape, and IUCN management category) were not significantly related to bird or mammal populations in any of the models.

In comparing the relative benefits to biodiversity from different protected area IUCN categories, our understanding of the relative impact of multiple-use areas is stymied by the lack of impact evaluation studies.[31] But the welcome exception to this silence is the focus on studies of forest and tree cover loss. Those studies that have estimated the impact of different sorts of protected areas have found that while relative impacts vary by region, multiple-use protected areas can be more effective at stemming deforestation than their strict counterparts.[9,11,32,33] This finding has been attributed in part to placement—that multiple-use protected areas are more likely to be placed in areas of higher deforestation pressure[32]—but also to management capacity and the political will of stakeholders to comply with and enforce rules.[34,35] Importantly, in some of these same protected areas, not only have the relative biodiversity benefits (as measured by avoided deforestation) across IUCN categories been measured but so too have the relative benefits to humans (as measured by reduced poverty).[9,36,37] This emphasizes that multiple-use protected areas can deliver benefits to nature and people hand in hand rather than in trade-off.

An analog to tree cover in terrestrial parks is fish biomass in marine protected areas. One study has conducted a comprehensive impact evaluation of 218 marine protected areas,[10] taking into consideration counterfactuals including protection versus no protection as well as multiple-use marine protected areas (MPAs) versus strict (no take) MPAs. The study finds that fish biomass was 1.6 times higher in MPAs than in matched non-MPA areas but that fish biomass response to protection was up to two times greater for strict protected areas relative to multiple-use areas. Similar to the terrestrial finding that management capacity strongly influences environmental outcomes,[32] this study linked management processes and ecological outcomes (in 62 MPAs) and found that adequate staff capacity followed by adequate budgets were the most important factors in explaining fish responses.[10]

A handful of studies have used a surveillance monitoring approach to examine the relative effectiveness of different protected areas with different IUCN categories. For example, Jones et al.[38] examined how human impacts within different categories of terrestrial protected areas changed over a single time step: 1993–2009. They found that for protected areas ratified before

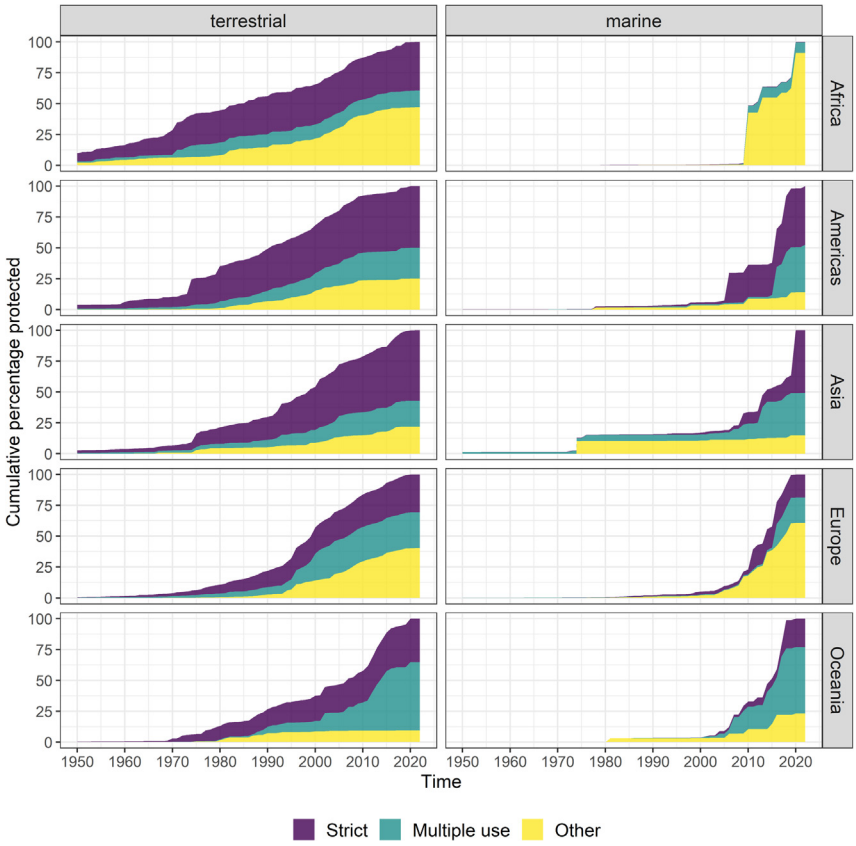

Figure 3. Protected area growth by region and realm

Breakdown of protected area growth by realm and by region for strict (IUCN I–IV), multiple-use (IUCN V and VI), and other (no IUCN category assigned) protected areas.

1992, more than half experienced increases in human pressure, with lower increases in strict (defined by the authors as IUCN categories I and II) protected areas. Given the paucity of studies examining the relative effectiveness of different categories of protected areas, we present a re-analysis of existing data to consider the relative effectiveness of the various IUCN categories of protected areas across realms, in terms of reducing human impacts within their boundaries (Box 3). We note that we do not expect protected areas to eliminate human impacts, particularly where human use is appropriate and consistent with conservation objectives. However, we do expect that human impacts would be either stable or reduce over time within protected areas where management is effective. Importantly, we extend existing analyses, including Jones et al.,[38] by considering both terrestrial and marine areas (over the same period, treating available human impact data similarly across both realms[39,40]) and by exploring the variance in trends from 2009 to 2013 relative to the initial level in 2009. These are important novelties in analyses that allow us to further interpret the variable changes in protected areas across realms, IUCN categories, and regions.

Across regions, in the marine estate, we found that at the start of our analysis, in 2009, strict protected areas had higher cumulative human impacts compared with multiple-use protected areas (Figure 4; Box 3), while the opposite was found on land (Figure 5; Box 3). Europe and Asia had the highest human impacts in protected areas in 2009 in both terrestrial and marine realms. When considering changes in human impacts over 2009–2013, we found that the trend was increasing in many areas but that this pattern varies across regions, countries, and realms. Where trends were decreasing was primarily in multiple-use protected areas; for example, in terrestrial protected areas, the trend was decreasing for multiple-use protected areas in all regions except Asia and South America (where impacts were increasing slightly on average). In MPAs, the trend was close to 0 for most regions and categories but was slightly decreasing for multiple-use protected areas in Asia and Europe (Box 3; Table S2). It is worth noting that we have not estimated the impact of protected areas on avoided human impacts (i.e., we have not compared to the counterfactual of no protection), and thus protected areas may still be effective in terms of avoiding loss.

Connectivity and land and seascape context

Connectivity is notoriously difficult to measure. In general, the application of connectivity criteria is more advanced in land-based conservation[41] relative to marine,[42] but challenges still remain in bridging implementation and high-level conservation targets.[43] As reported in the Protected Planet Report, in 2020, 7.84% of the world's terrestrial surface was both protected and geographically connected.[3] While this estimates geographic connectivity of terrestrial protected areas, it does not consider connectivity outside and between protected areas or functional connectivity, all known to be critical for overall land and seascape health. Brennan et al.[44] investigated functional connectivity of global terrestrial protected areas and found that the most important connectivity routes are currently unprotected and remain threatened by conversion. Formal protection in these areas, to create connectivity, could be contested or have adverse impacts on human populations. Therefore, alternative conservation strategies that incorporate working lands or broader landscape management consistent with existing land uses are recommended. Supporting this, studies have found that private protected areas play an important role in improving land-based connectivity of the protected and conserved estate.[45] This emphasizes the relative importance of managing both within and outside of protected and conserved areas.[46]

Where connected conservation plans have been implemented, important ingredients of success include using a range of interventions tailored to the context (for example, protected areas, wildlife tailored solutions, private land conservation), the existence of enabling legislation and policy, a transparent and repeatable scientific approach, adequate funding, and appropriate public outreach.[41] In particular, flexibility in governance and rapid

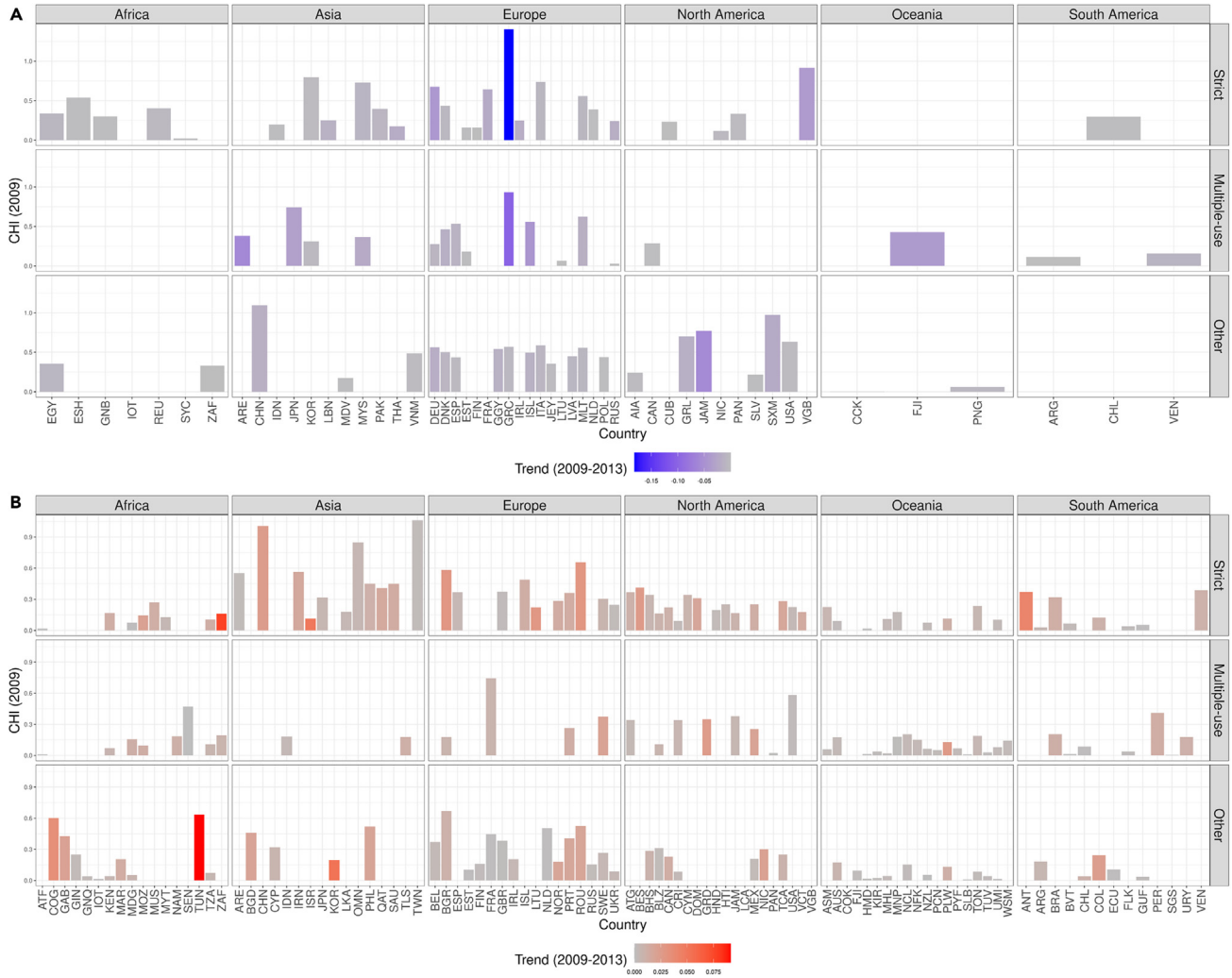

Figure 4. Change in cumulative marine human impact (CHI) within strict, multiple-use, and other protected areas by region

Size of bar represents the mean CHI in 2009 within protected areas, and color is the trend in change in CHI over the time period 2009–2013.

(A) Positive changes over time (blue).

(B) Negative changes over time (red).

deployment of new conservation strategies and resourcing will be critical in the face of climate change.[47,48]

In the marine realm, multiple-use and other protected areas are the dominant IUCN categories currently employed, potentially making MPAs more flexible forms of protection as climate change warrants new and novel practices such as temporary or dynamic arrangements to facilitate species migrations. In Box 4, we find that many MPAs are exposed to high rates of climate velocity, the speed and direction a species would need to shift to maintain its environmental niche under climate change.[49] Species within smaller MPAs with faster climate velocity are most exposed to these changes. Even if these species can adapt, their range is more likely to shift outside of these small, static MPAs, thus losing any benefits of protection. Most of these highly vulnerable MPAs (smaller with faster velocity) were found in the Mediterranean and tropics and in areas of high endemicity for coral reefs (e.g., the coral triangle and the Caribbean; Figure 6; Box 4). In these contexts, including connectivity to facilitate movement of species and flexible governance to support moveable or temporary protected areas are needed. While strict protection has been recognized as important for climate resilience and is potentially easier to manage,[50] multiple-use MPAs (or OECMs, or temporary/dynamic sector closures) are likely to be deployed more quickly and to provide more flexibility to be relocated as steppingstones between strictly protected areas.[48] Temporary or moveable (i.e., dynamic) protected areas are already deployed in some regions where customary approaches to sea country management include resting of areas or use of traditional fishing grounds such as Tabus in Fiji.[51] Expanding our conservation toolbox to support flexible governance arrangements through agile processes and funding is needed.[52,53]

Equitable management and governance

While Aichi target 11 called for equitable management, this subcomponent has been poorly defined and received little attention,

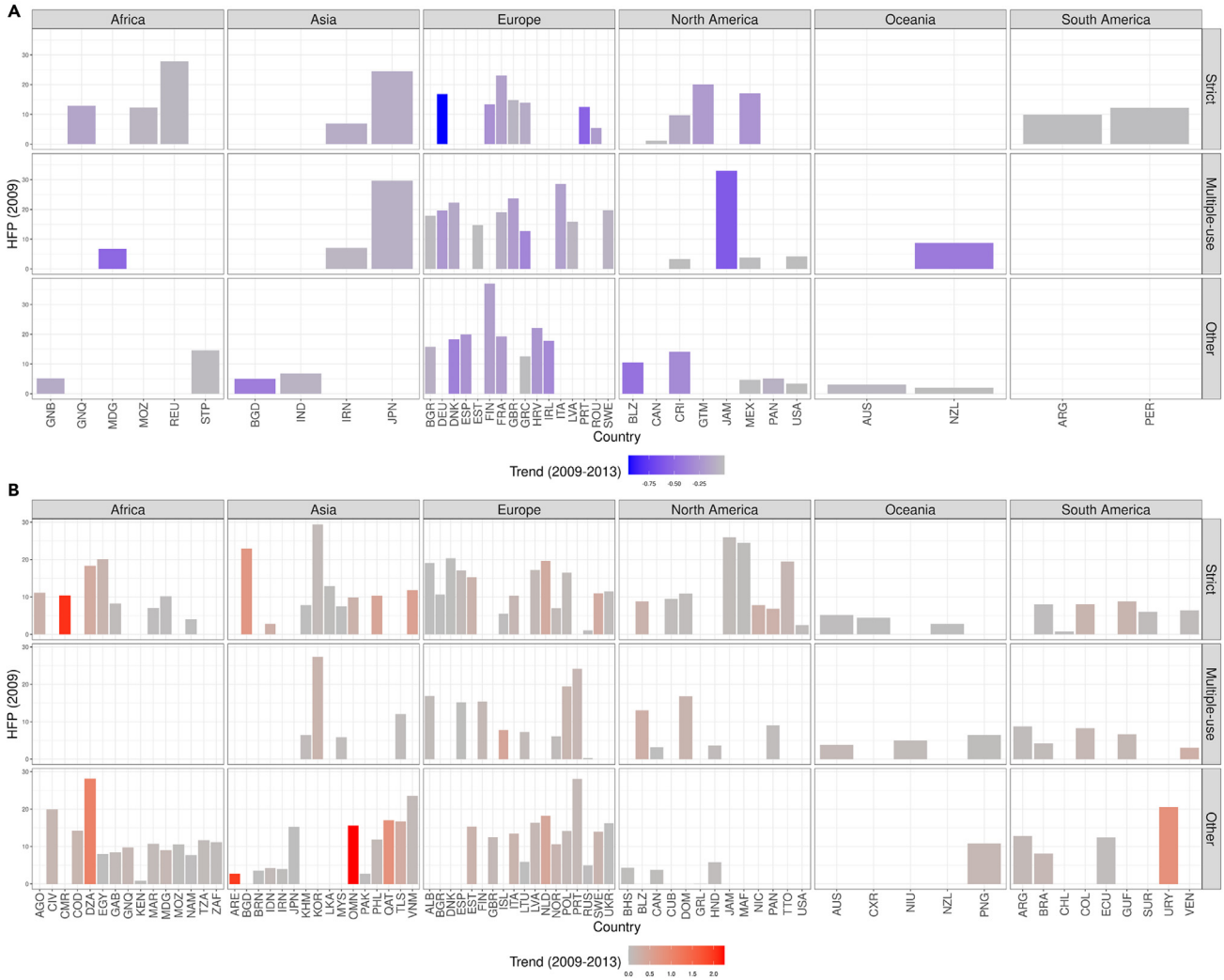

Figure 5. Change in cumulative terrestrial human footprint (HFP) within strict, multiple-use, and other protected areas by region

Size bar represents the mean HFP in 2009 within protected areas, and color is the trend in change in HFP over the time period 2009–2013.

(A) Positive changes over time (blue).

(B) Negative changes over time (red).

with no established global monitoring system. More emphasis has been given to equity in the GBF, with target 3 calling for equitable governance, equity indicators for monitoring being specified for the first time, and equity considerations being evident throughout, including in the implementation advice.[20] Equity is generally defined as what is right and fair and tends to be conceptualized by the conservation community, including by the CBD, as encompassing distribution of costs and benefits, decision-making procedures, and recognition of socio-cultural diversity (e.g., identities, rights, values).[54] Given that studies explicitly examining equity in area-based conservation are rare, we focus on the outcomes for local actors, whether they be benefits or harms.

Research examining the impacts of protected areas on human well-being paint a mixed picture, with protected areas being both beneficial and harmful across a range of human well-being domains, including livelihoods, living standards, health, education, social relations, safety, property rights, political agency, food security, cultural identity, subjective well-being,[55,56] and, more recently, mental health.[57,58] The most severe social impacts of protected areas tend to result from physical displacement—eviction,[59] which can cause multiple harms.[54] Conversely, where people are part of the landscape, socio-economic benefits have been found to be more likely to arise.[60,61] Importantly, impacts have been shown to be heterogeneous across well-being domains, time, social subgroups, and governance and management approaches. For example, in Indonesia, paired community comparisons of MPAs found that MPAs generally appeared to contribute to the alleviation of multidimensional poverty[62] but that impacts were mixed according to poverty domain (e.g., positive impacts on material wealth and negative impacts on governance aspects such as marine resource control). Benefits over time were also variable, with most positive impacts occurring during the implementation period, after which management activities all but ceased and reductions in poverty did not continue to accrue. Impacts can also

Box 3. Changes in cumulative human footprint within various categories of protected areas

The evidence around the effectiveness and impacts of protected areas are mixed and are focused primarily on terrestrial protected areas. To complement the synthesis of the existing literature, we seek to answer the question: have protected areas been effective at reducing human pressures, and what is the relative effectiveness of different types of protected areas? We sought to quantify whether there are detectable changes in human pressures and, if so, how these compared across protected area types, with an expectation that effective management would result in either a stable or declining level of human pressures within protected areas. To address this, we use cumulative human impact data on land[39] and in the sea[40] to assess trends from 2009 to 2013 within established protected areas (including only protected areas established on or before 2009 to isolate effects of existing protected areas on changes). In analyzing existing data, we were able to complete this analysis for comparable time steps across both land and sea to allow for the opportunity for comparisons across regions, realms, and types of protected areas. We note that this is a key distinguishing feature of our analysis compared with other similar analyses[38] that have summarized human impact within and outside of protected areas.

In the marine realm, we found that the 2009 cumulative human footprint (which is primarily reflective of fishing pressures) was higher in strict protected areas compared with multiple-use areas in most regions and that it continues to increase (across 2009–2013) in strict protected areas in Europe and North America, while in multiple-use protected areas, changes are primarily neutral or slightly decreasing (Figure 4).

On land, we found (Figure 5) that the 2009 human footprint (which is an aggregate of a range of terrestrial human impacts) was lower in strict protected areas compared with not strict and other protected areas. In contrast, strict protected areas had slightly increasing human footprint from 2009 to 2013. Not strict and other protected areas had slightly decreasing trends.

vary substantially by social subgroup,[63] with studies that examine the distribution of fortunes and misfortunes identifying pockets of harm, often visited on already marginalized peoples.[64]

Few studies have examined the relative impact of multiple-use protected areas on people. A key exception is Naidoo et al.'s[65] global-scale analysis that found that households living near terrestrial multiple-use protected areas had a lower likelihood of being poor (as measured by stunting in children under 5) as compared with similar households living further than 10 km away. The authors hypothesized that the pathway for this observed positive impact on household poverty is that multiple-use protected areas lead to improved environmental conditions, and due to their accessibility (which is different from strict protected areas), they allow people to benefit from this abundance of plants and animals to take to market or to consume.

This global finding, comparing strict and multiple-use areas quantitatively, supports the overall evidence that multiple-use protected areas where management practices are more inclusive tend to produce more benefits for people and nature.[66] This understanding is supported by the results of a systematic review by Dawson et al.[26]; the authors found that most studies that document positive outcomes for both well-being and conservation were cases where Indigenous people and local communities played a central role, such as having substantial influence over decision-making or a recognized role in the governance of areas. Similarly, a review of MPAs found that community-governed areas were more likely to yield positive outcomes for people than those under other governance arrangements, with 70% of documented impacts being positive for community-governed areas and just 41% for areas governed by states.[56]

Given the paucity of studies examining multiple-use protected areas, we analyze the literature to identify the documented benefits or harms that this type of protected areas have resulted in for local actors. We draw upon a systematically compiled database of existing literature on multiple-use protected areas.[67] We queried the database for those studies that have documented social impacts and further extracted studies that document the "directionality" of outcomes, i.e., whether the authors reported positive, negative, or neutral social impacts from the studied activities. Outcomes were grouped by well-being domain.[56] This resulted in 53 articles from 26 countries for a total of 106 studies globally (articles often presented multiple studies). To visualize relationships between type of activity studied within protected areas (e.g., agriculture, fishing, tourism) and impacts, we plot the volume of studies for each activity-outcome pair and the direction of outcome (positive, negative, neutral) (Figure 7).

Most activities led to a mix of outcomes in terms of positive, negative, or neutral. The exception to this is health (encompassing food security and treatment of ailments), which was uniformly positive, and human-wildlife interactions (within the ecological health category), which were uniformly negative. Food security was shown to be improved by subsistence hunting within multiple-use protected areas in three articles.[68–70] In contrast, human-wildlife interactions were conflictual and therefore negative regardless of type of activity being undertaken (Figure 7). This may reflect the difficulties of using terms like positive and negative, as studies of human-wildlife coexistence often describe situations of sustainable conflict, which do not present major threats to either human communities or valued wildlife but which are also not harmonious.[71,72]

Tourism, recreation, agriculture, and forestry were the most common activities studied. Most articles that focused on tourism or recreational activities found that they brought about positive outcomes such as on education and economic improvements to communities[73–76]; however, negative impacts included visitors creating waste and pollution and increased crime. Even within single-protected areas, these outcomes can be mixed; for example, an article in Belize found growth of ecotourism within the multiple-use protected area to be associated with higher environmental awareness and increased employment as a positive outcome, but it also led to a loss of sense of empowerment and cultural values as a negative outcome.[77] Forestry

Box 4. Patterns of climate velocity in MPAs

Connectivity is critical to sustain ecological processes, functions, and species movements, particularly in the face of climate change. This requires considering the status of conservation values within and around protected areas and engaging a range of stakeholders, which can be facilitated through a diverse set of conservation tools, governance approaches, and, ultimately, procedural flexibility.[47,93] In the marine realm, flexible arrangements to the establishment, movement, and overall governance of protected areas may be more feasible due to use rights and overall management approaches. Temporary or moveable (i.e., dynamic) protected areas are already deployed in some regions where customary approaches to sea country management include resting of areas or use of traditional fishing grounds such as Tabus in Fiji.[51] However, the extent to which these types of flexible or temporary governance measures are needed will largely depend upon the extent to which climate change impacts existing protected areas. To explore this, we quantified climate velocity within existing MPAs. We compared average climate velocity to the amount of area protected under the assumption that slower climate velocities within larger protected areas would allow for greater adaptation than faster climate velocities within smaller protected areas. We found that the majority of MPAs are predicted to experience faster climate velocity, in particular those near shore, while pelagic MPAs had on average lower velocity and were also larger and thus more able to facilitate adaptation in the face of climate change (Figure 6).

and agricultural activities were documented as leading to decreases in availability or use of resources, e.g., the availability of non-timber forest products.[78,79]

While most of the studies in our database could make limited inferences around the relative impacts of protected areas, due to lack of baselines or paired control sites, we noted several exemplar studies whose study designs supported further causal inference, many of which happened to be located in Brazil. These studies find that while development, such as trade in non-timber forest products, can coexist alongside conservation, these can have unforeseen impacts such as upon community resilience, cooperation, and community perceptions.[80,81]

To complement this broad overview of the literature across all countries, we consider whether patterns of outcomes remain variable or become more consistent within a particular country (where protected area context may become more uniform). Therefore, we further analyze the evidence specific to Brazil (Box 5). Brazil represents the country with the greatest number of articles having studied the impacts of activities in multiple-use protected areas on various socio-economic outcomes, with 15 articles having been identified and several of these having high-quality evidence-based results (as discussed above and further described in Box 5).

Our analysis emphasizes that both benefits and harms have eventuated from protected and conserved areas, across regions and governance types. Even within relatively homogeneous contexts, within single countries and types of protected areas, our analysis of the literature from multiple-use protected areas in Brazil (Figure 8; Box 5) demonstrates that outcomes can be heterogeneous, including positive outcomes in one domain of well-being resulting in negative outcomes in another from single-protected areas. Thus, while the evidence suggested that

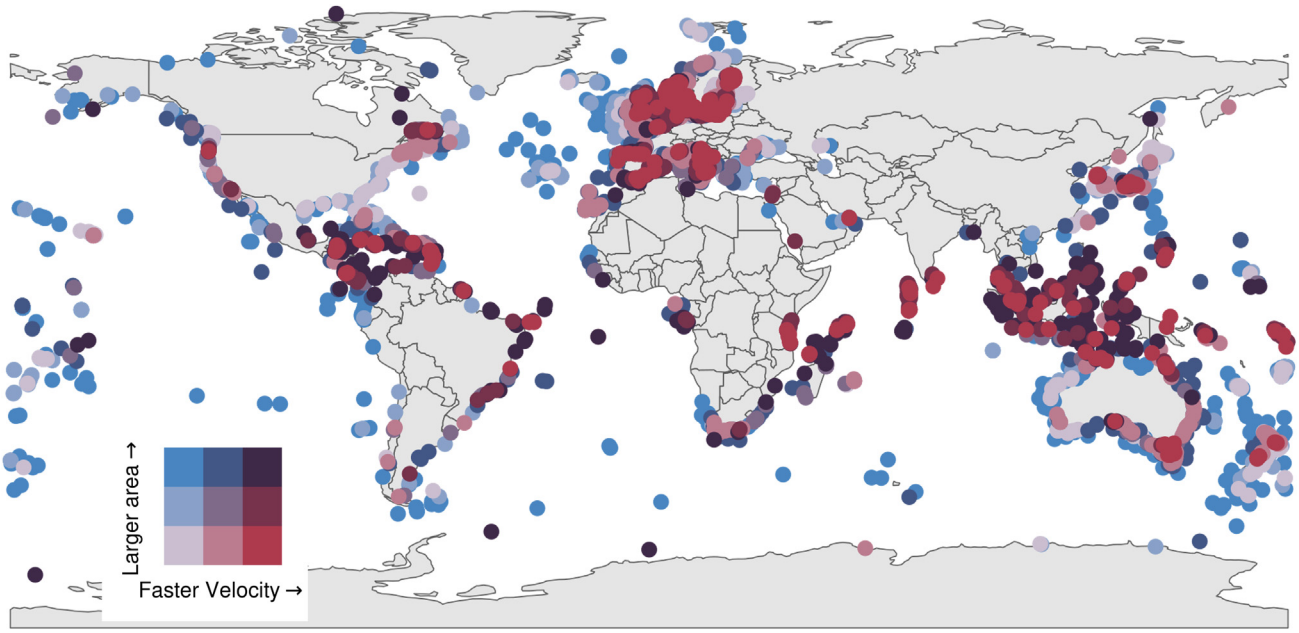

Figure 6. Climate velocity in MPAs
Size of MPA against speed of velocity is plotted for the centroid of all MPAs. Blue indicates larger area and slower velocity, red indicates smaller area and faster velocity, purple indicates larger area and faster velocity, and mauve indicates smaller area and slower velocity.

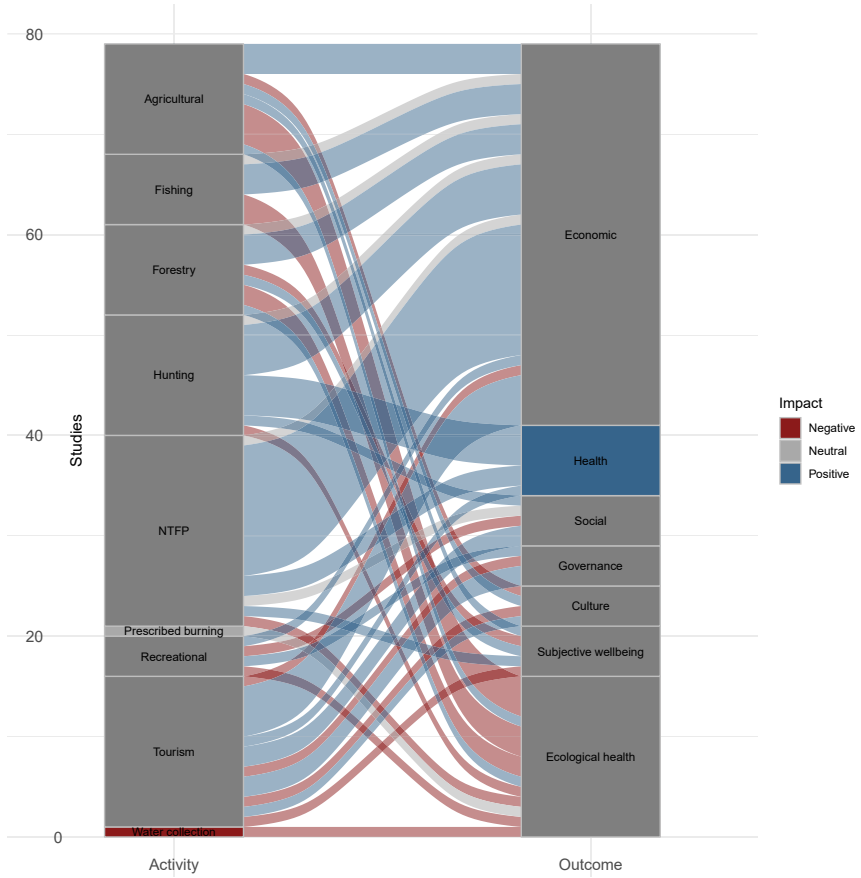

Figure 7. Documented relationships between human well-being and allowed activities in multiple-use protected areas

Activities, outcomes, and directionality of outcomes of reviewed articles that examined the link between human well-being and allowed activities in multiple-use protected areas (53 articles). The y axis shows the number of studied protected areas (n = 106; some articles studied multiple protected areas, outcomes, or activities). Activities and outcomes are colored by direction of impact, where gray indicates mixed directional outcomes. Outcomes are grouped by well-being domain, and activities and outcomes are colored by direction of impact, where gray indicates mixed directional outcomes. Note that NTFP refers to harvesting of non-timber forestry products.

multiple-use protected areas are most appropriate for ensuring equitable governance and social benefits to local actors, these still require careful design to fully deliver on both conservation and social outcomes.

CONCLUSIONS

Expansion of the global protected area system in recent decades has seen a growth in multiple-use protected areas and the introduction of OECMs. These areas collectively represent an important set of tools in the protected and conserved toolbox to not only increase area-based coverage but to contribute to the improved status of biodiversity and deliver benefits to rightsholders and stakeholders, such as communities, private landowners, and production industries. Our synthesis of available analyses, as well as re-analysis of data, emphasizes that multiple-use protected areas are effective at conserving biodiversity (with many studies showing that they can have greater impact than their strict counterparts) and have recorded benefits to people in many contexts. However, we caution that these are not immune to the recorded harms of protected areas and add our voices to increasing calls for growth in the global protected and conserved estate to be underpinned by principles of equity and justice (e.g., Gurney et al.[20]).

Lastly, given the prominence of protected areas in global conservation policy, we need to get serious about tracking performance measures in terms of biodiversity outcomes[82] as well as impacts to well-being of local actors (both benefits and harms) across diverse contexts,[83] ideally through an integrated social-ecological systems approach (e.g., see Gurney et al.[84] in regard to the monitoring of a multicountry coral reef conservation program). This may require regionally contextualized approaches to tracking progress that directly acknowledge different forms of success and failures and the ways these eventuate. For OECMs, monitoring and evaluation approaches could be codeveloped between governing actors and conservation scientists, practitioners, and/or policymakers to ensure that all outcomes of interest and visions of success are included. The rapid planned expansion of the protected estate to 30% of land and sea requires considering how to manage the land and sea outside of the protected and conserved areas to ensure that the matrix surrounding these areas supports their overall management and effectiveness.[46]

Our analysis contrasts the ways in which the different types of protected areas (strict, multiple use, other) have been used to date and reveals the growing role of multiple-use protected areas in meeting current and future targets. This evidence suggests that further growth in the global conserved and protected areas will benefit from continued use of multiple-use protected areas or OECMs and their improved resourcing and political and technical support.

EXPERIMENTAL PROCEDURES

Resource availability

Lead contact

The lead contact for our data and code is Caitlin D. Kuempel.

Materials availability

This study did not generate new unique materials.

Data and code availability

While we did not generate new data, we did synthesize existing data into new datasets as described here. All synthesized data and code are available at https://github.com/cdkuempel/Adams_multiple_use_PAs. We detail each synthesized dataset here.

Our core data for all analyses was the cleaned WDPA data.[85] Where analyses were completed by realm, we included only terrestrial protected areas for terrestrial analyses and, similarly, only marine protected areas (MPAs) for marine analyses.

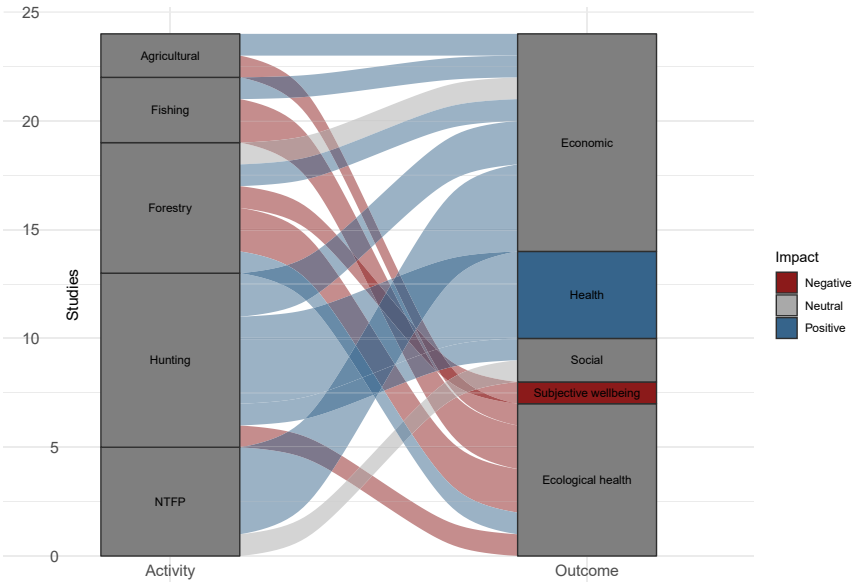

Figure 8. Documented relationships between human well-being and allowed activities in multiple-use protected areas in Brazil

Activities, outcomes, and directionality of outcomes of reviewed articles that examined the link between human well-being and allowed activities in multiple-use protected areas in Brazil (15 articles). The y axis shows the number of protected areas studied (n = 24; some articles studied multiple protected areas, outcomes, or activities). Outcomes are grouped by well-being domain, and activities and outcomes are colored by direction of impact, where gray indicates mixed directional outcomes. Note that NTFP refers to harvesting of non-timber forestry products.

Box 5. Multiple-use protected areas in Brazil as a case study

The implementation of multiple-use protected areas has been geographically clustered, with some countries heavily contributing to the overall growth of IUCN categories V and VI. In particular, Brazil has a large proportion of multiple-use protected areas (Figure 1). To understand the types of socio-economic outcomes in a country-specific context, we therefore further analyzed the literature specific for Brazil.

Overall, most articles found activities to result in positive outcomes (60% positive, 26.7% negative, and 13.4% neutral) (Figure 8). The most frequently researched outcome related to the economic impact of sanctioned activities within protected areas, with all activities being represented and all having found either a positive or neutral impact. Hunting was a dominant sanctioned activity, consistent with the context of these multiple-use protected areas within Brazil, and the impacts of hunting on social outcomes were positive. Two articles found that subsistence hunting contributed to food security,[68,69] and one of these also found that hunting for medicinal resources plays an important role for communities.[68] When looking at human interactions, hunting and related meat sharing was found to strengthen social bonds in one article.[94] The other dominant sanctioned activity studied was forestry, which, in contrast to hunting, had predominantly negative or neutral impacts. For example, one article found logging to lower the availability of non-timber forest products as well as decrease hunting rates,[79] and another found that participating in logging projects was associated with lower levels of well-being.[81]

When considering those studies that met the highest quality standards, via study design such as employing before-after control-impact (BACI) designs, these were primarily within Brazil. Rizek and Morsello's[80] article was interested in determining whether trade in non-timber forest products affected cooperation among community households of Brazilian Amazon Caboclos. The authors compared the frequency of cooperation within two communities, one engaging in non-timber forest product trade and the other not. They used a variety of control variables including other forms of income, demographic factors such as household size, years of residence, and time spent in collective activities. Their dependent variable was the amount of goods obtained through household cooperation. The results showed that goods were acquired less frequently by cooperation in the community that commercialized non-timber forest products, due to multiple factors including increased income inequality, population growth, and a decrease in non-timber forest product transfers. The authors warn that conservation and development projects may reduce community resilience by making certain households more vulnerable to food shortages.

Cooper et al.[81] assessed the impact of timber commercialization within the Brazilian Reserva Extrativista Chico Mendes on well-being through residents' perspectives. The study considered household participation and perceptions of a logging project, comparing participating and non-participating households, supporters and non-supporters, and men and women. The authors conclude that although logging is often seen as a way to improve human well-being while meeting conservation targets within protected areas, their results showed that residents perceived the decision to log or not as a trade-off between conservation or well-being (through income generation). However, logging was perceived as conflicting with non-timber forest product harvesting and the ideology upon which the park was established.

Our focused analysis of the demonstrated social impacts of sanctioned activities within multiple-use protected areas in a single country, Brazil, emphasizes that there are a range of variable outcomes depending on context as well as activities. This emphasizes the importance of context when designing and implementing protected areas and the potential for negative outcomes even where ongoing use of areas is designed as part of a multiple-use designation.

To consider changes in human influences within protected areas, we used human footprint data for land and sea. On land, we used the terrestrial human footprint dataset from Mu et al.[39] that quantifies a cumulative pressure index across eight variables (pasture, roads, railways, population density, navigable waterways, night-time light, built environment, and cropland). We used the full range of pressures in this analysis under the assumption that terrestrial protected areas could reduce intensity across all pressures.

Marine cumulative impacts were quantified using data from Halpern et al.[40] that assessed the pace of change in human impacts of 14 human stressors across 21 marine ecosystems. For our analysis, we only included marine human pressures that could be abated by MPAs, with a particular emphasis on fishing as direct extraction from MPAs. This included all measures of fishing pressure (6 pressures) and shipping (1 pressure) for a total of 7 pressures. We excluded pressures related to climate change and land-based impacts.

To consider connectivity and relative influence of climate change, we calculated climate velocity for marine areas. Climate velocity is a metric that describes the "speed and direction that a species at a given point would need to move to remain within its climatic niche."[52] We calculated climate velocity (km/year) within MPAs using the VoCC R package[86] using sea surface temperature data from 1901–2020.[87]

Protected area policies

We reviewed and compiled a list of key protected area policy dates and events from 2000 to 2022 relating to conservation and development. The full list is detailed in Table S1 and is the basis for the narrative of recent protected area debates in this review. From this summary, we identify 2000–2005, 2008–2010, and 2018–2020 as key policy periods to expect hot moments in protected area growth, in particular for multiple-use areas and OECMs. These periods are further plotted against growth curves by IUCN category of protected area (Figure S1).

Hot moments

To test for hot moments in protected area growth related to global target policies, we plotted cumulative growth curves by IUCN category alongside key policy time periods identified in the review: 2000–2005 due to large numbers of key events as discussed in the historical narrative, 2008–2010 as the leadup to the first target due date of 2010, and 2018–2020 as the leadup to the second target due date of 2020. We then plotted changes in slope in the growth curves to detect potential hot moments. Where there is a peak in annual rate of change aligned within the noted policy periods, we infer a potential global hot moment. Where there is a lack of peaks within policy periods, we identify a lack of support for global hot moments, indicating that while there is certainly global interest in setting targets, as evidenced by country investment in and engagement with the CBD target setting process, this does not necessarily translate to on-the-ground protected area declarations that are significantly different from natural rates of protected area growth.

Trends in human impact

To provide a globally consistent approach to measuring effectiveness of protected areas in reducing human impacts across protected area types and realms, we sought to answer the question "have protected areas been effective in reducing human pressures?" To answer this, we drew upon human footprint data mapped consistently across time steps and with methods specific to land[39] and sea.[40] To harmonize the data approaches across land and sea, we segmented the data to consider only human impacts that a protected area can plausibly reduce (as detailed in data and code availability).

Our analysis differs from other surveillance monitoring studies using human footprint data in protected areas, such as Jones et al.,[38] in nuanced, but very important, ways. Previous analyses have largely sought to answer the question "how does the human footprint vary inside protected areas compared with unprotected areas and across IUCN categories?" This is answered by simply comparing the average human footprint for various categories of the landscape. For our analysis, we wished to explore the changes in human footprint relative to their starting condition across time. Therefore, our analyses are quite different from those currently available in the literature. For example, a protected area may be wrongly inferred as ineffective if we simply rely on a data signal that says there are human impacts within its boundaries. However, even where protected areas allow human use or have visible human impacts, we do expect those impacts to be either stable or decreasing over time if protected area management is effective. Thus, we focused on the change in human footprint relative to initial conditions in 2009 and therefore included protected areas established on or before 2009 to align with human footprint data. Second, we define strict as categories I–IV throughout our analysis, while other analyses such as Jones et al.[38] define strict as categories I and II (rendering it impossible to draw upon their conclusions for the purposes of our review, where we consider differences across I–IV, V–VI, other, and OECMs). Third, previous analyses have used a human footprint dataset with only two time steps (1993 and 2009) and inferred those changes as human footprints within protected areas to be related to management effectiveness. We felt that was not accurate given that many protected areas would have been declared during that time frame. We chose human footprint datasets for land and sea with further time steps and limited our change analysis to a more contemporary time period, which allowed us to use the exact same time frame across land and sea (2009–2013).[39,40]

Using our modified human footprint data, we calculated cumulative human impact for the relevant subset of pressures across all years on land and sea. The trend was estimated within each raster cell using linear regression across all 5 years to determine the magnitude, direction, and significance of change using the methods and code from Halpern et al.[40] We averaged the cumulative human impact and trend results across cells within protected area IUCN categories in each country. Strict protected areas were considered IUCN categories I–IV, non-strict was IUCN V–VI, and other for the remaining categories (not reported, etc.). We plot variance in what happened from 2009 to 2013 relative to the initial level of human footprint in 2009. Results by realm, protected area type, and country are presented in Box 3. Summaries by region are presented in Table S2.

Marine climate velocity and size of protected area

We set the minimum observations of climate velocity within MPAs to calculate a trend to 10 and a lower threshold of 0.0001 for truncating the spatial gradient. We compared average climate velocity to the amount of area protected within each MPA under the assumption that slower climate velocities within larger protected areas would allow for greater adaptation than faster climate velocities within smaller protected areas.

SUPPLEMENTAL INFORMATION

Supplemental information can be found online at https://doi.org/10.1016/j.oneear.2023.08.011.

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DECLARATION OF INTERESTS

The authors declare no competing interests.

REFERENCES

1. Dudley, N., Parrish, J.D., Redford, K.H., and Stolton, S. (2010). The revised IUCN protected area management categories: the debate and ways forward. Oryx *44*, 485–490. https://doi.org/10.1017/S0030605310000566.
2. Chape, S., Harrison, J., Spalding, M., and Lysenko, I. (2005). Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. Phil. Trans. R. Soc. B *360*, 443–455.
3. UNEP-WCMC, and IUCN (2020). Protected Planet Report 2020. https://livereport.protectedplanet.net/.
4. CBD (2022). Kunming-Montreal post-2020 Global Biodiversity Framework. CBD/COP/DEC/15/4. Convention on Biological Diversity. https://www.cbd.int/doc/decisions/cop-15/cop-15-dec-04-en.pdf.
5. Geldmann, J., Manica, A., Burgess, N.D., Coad, L., and Balmford, A. (2019). A global-level assessment of the effectiveness of protected areas at resisting anthropogenic pressures. Proc. Natl. Acad. Sci. USA *116*, 23209–23215.

6. Geldmann, J., Coad, L., Barnes, M., Craigie, I.D., Hockings, M., Knights, K., Leverington, F., Cuadros, I.C., Zamora, C., Woodley, S., and Burgess, N.D. (2015). Changes in protected area management effectiveness over time: A global analysis. Biol. Conserv. *191*, 692–699.

7. Geldmann, J., Coad, L., Barnes, M.D., Craigie, I.D., Woodley, S., Balmford, A., Brooks, T.M., Hockings, M., Knights, K., Mascia, M.B., et al. (2018). A global analysis of management capacity and ecological outcomes in terrestrial protected areas. Conserv. Lett. *11*, e12434.

8. Barnes, M.D., Craigie, I.D., Harrison, L.B., Geldmann, J., Collen, B., Whitmee, S., Balmford, A., Burgess, N.D., Brooks, T., Hockings, M., and Woodley, S. (2016). Wildlife population trends in protected areas predicted by national socio-economic metrics and body size. Nat. Commun. *7*, 12747.

9. Ferraro, P.J., Hanauer, M.M., Miteva, D.a., Canavire-Bacarreza, G.J., Pattanayak, S.K., and Sims, K.R.E. (2013). More strictly protected areas are not necessarily more protective: evidence from Bolivia, Costa Rica, Indonesia, and Thailand. Environ. Res. Lett. *8*, 025011. https://doi.org/10.1088/1748-9326/8/2/025011.

10. Gill, D.A., Mascia, M.B., Ahmadia, G.N., Glew, L., Lester, S.E., Barnes, M., Craigie, I., Darling, E.S., Free, C.M., Geldmann, J., et al. (2017). Capacity shortfalls hinder the performance of marine protected areas globally. Nature *543*, 665–669.

11. Nelson, A., and Chomitz, K.M. (2011). Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: A global analysis using matching methods. PLoS One *6*, e22722. https://doi.org/10.1371/journal.pone.0022722.

12. Brockington, D., Duffy, R., and Igoe, J. (2008). Nature Unbound: Conservation, Capitalism and the Future of Protected Areas (Earthscan).

13. Holdgate, M. (2014). The Green Web: A Union for World Conservation (Routledge).

14. Roe, D. (2008). The origins and evolution of the conservation-poverty debate: a review of key literature, events and policy processes. Oryx *42*, 491–503. https://doi.org/10.1017/S0030605308002032.

15. Dudley, N. (2008). Guidelines for Applying Protected Area Management Categories (IUCN).

16. CBD (2010). Convention on Biological Diversity - Aichi Biodiversity Targets. http://www.cbd.int/sp/targets/.

17. Lopoukhine, N., and de Souza Dias, B.F. (2012). What does Target 11 really mean. Parks *18*, 5.

18. Gurney, G.G., Darling, E.S., Ahmadia, G.N., Agostini, V.N., Ban, N.C., Blythe, J., Claudet, J., Epstein, G., Himes-Cornell, A., Himes-Cornell, A., et al. (2021). Biodiversity needs every tool in the box. use OECMs *595*, 646–649.

19. Jonas, H.D., Ahmadia, G.N., Bingham, H.C., Briggs, J., Butchart, S.H., Cariño, J., Chassot, O., Chaudhary, S., Darlling, E., Degemmis, A., and Dudley, N. (2021). Equitable and effective area-based conservation: towards the conserved areas paradigm. Parks: The International Journal of Protected Areas and Conservation *27*, 71–84.

20. Gurney, G.G., Adams, V.M., Álvarez-Romero, J.G., and Claudet, J. (2023). Area-based conservation: Taking stock and looking ahead. One Earth *6*, 98–104.

21. Radeloff, V.C., Beaudry, F., Brooks, T.M., Butsic, V., Dubinin, M., Kuemmerle, T., and Pidgeon, A.M. (2013). Hot moments for biodiversity conservation. Conserv. Lett. *6*, 58–65. https://doi.org/10.1111/j.1755-263X.2012.00290.x.

22. Adams, W.M., Aveling, R., Brockington, D., Dickson, B., Elliott, J., Hutton, J., Roe, D., Vira, B., and Wolmer, W. (2004). Biodiversity conservation and the eradication of poverty. Science *306*, 1146–1149. https://doi.org/10.1126/science.1097920.

23. Secretariat, C.B.D. (2020). Global Biodiversity Outlook 5. https://www.cbd.int/gbo5.

24. Chauvenet, A.L., Watson, J.E., Adams, V.M., Di Marco, M., Venter, O., Davis, K.J., Mappin, B., Klein, C.J., Kuempel, C.D., and Possingham, H.P. (2020). To achieve big wins for terrestrial conservation, prioritize protection of ecoregions closest to meeting targets. One Earth *2*, 479–486. https://doi.org/10.1016/j.oneear.2020.04.013.

25. Kuempel, C.D., Chauvenet, A.L.M., and Possingham, H.P. (2016). Equitable representation of ecoregions is slowly improving despite strategic planning shortfalls. Conserv. Lett. *9*, 422–428. https://doi.org/10.1111/conl.12298.

26. Dawson, N.M., Coolsaet, B., Sterling, E.J., Loveridge, R., Gross-Camp, N.D., Wongbusarakum, S., Sangha, K.K., Scherl, L.M., Phan, H.P., Zafra-Calvo, N., et al. (2021). The role of Indigenous peoples and local communities in effective and equitable conservation. Ecol. Soc. *26*, 19. https://doi.org/10.5751/ES-12625-260319.

27. Garnett, S.T., Burgess, N.D., Fa, J.E., Fernández-Llamazares, Á., Molnár, Z., Robinson, C.J., Watson, J.E.M., Zander, K.K., Austin, B., Brondizio, E.S., et al. (2018). A spatial overview of the global importance of Indigenous lands for conservation. Nat. Sustain. *1*, 369–374. https://doi.org/10.1038/s41893-018-0100-6.

28. Allan, J.R., Possingham, H.P., Atkinson, S.C., Waldron, A., Di Marco, M., Butchart, S.H.M., Adams, V.M., Kissling, W.D., Worsdell, T., Sandbrook, C., et al. (2022). The minimum land area requiring conservation attention to safeguard biodiversity. Science *376*, 1094–1101.

29. Mascia, M.B., Pailler, S., Thieme, M.L., Rowe, A., Bottrill, M.C., Danielsen, F., Geldmann, J., Naidoo, R., Pullin, A.S., and Burgess, N.D. (2014). Commonalities and complementarities among approaches to conservation monitoring and evaluation. Biol. Conserv. *169*, 258–267.

30. Grorud-Colvert, K., Sullivan-Stack, J., Roberts, C., Constant, V., Horta e Costa, B., Pike, E.P., Kingston, N., Laffoley, D., Sala, E., Claudet, J., et al. (2021). The MPA Guide: A framework to achieve global goals for the ocean. Science *373*, eabf0861. https://doi.org/10.1126/science.abf0861.

31. Zhang, Y., West, P., Thakoli, L., Suryawanshi, K.R., Supuma, M., Straub, D., S. Sithole, Sharma, R., Schleicher, J., Ruli, B., et al. (In review). Governance and Conservation Effectiveness in Protected Areas and Indigenous and Locally Managed Areas.

32. Pfaff, A., Robalino, J., Lima, E., Sandoval, C., and Herrera, L.D. (2014). Governance, location and avoided deforestation from protected areas: Greater restrictions can have lower impact, due to differences in location. World Dev. *55*, 7–20. https://doi.org/10.1016/j.worlddev.2013.01.011.

33. Schleicher, J., Peres, C.A., Amano, T., Llactayo, W., and Leader-Williams, N. (2017). Conservation performance of different conservation governance regimes in the Peruvian Amazon. Sci. Rep. *7*, 11318. https://doi.org/10.1038/s41598-017-10736-w.

34. Blackman, A. (2015). Strict versus mixed-use protected areas: Guatemala's Maya Biosphere Reserve. Ecol. Econ. *112*, 14–24. https://doi.org/10.1016/j.ecolecon.2015.01.009.

35. Radachowsky, J., Ramos, V.H., McNab, R., Baur, E.H., and Kazakov, N. (2012). Forest concessions in the Maya Biosphere Reserve, Guatemala: A decade later. For. Ecol. Manage. *268*, 18–28. https://doi.org/10.1016/j.foreco.2011.08.043.

36. Ferraro, P.J., and Hanauer, M.M. (2014). Quantifying causal mechanisms to determine how protected areas affect poverty through changes in ecosystem services and infrastructure. Proc. Natl. Acad. Sci. USA *111*, 4332–4337. https://doi.org/10.1073/pnas.1307712111.

37. Oldekop, J.A., Sims, K.R.E., Karna, B.K., Whittingham, M.J., and Agrawal, A. (2019). Reductions in deforestation and poverty from decentralized forest management in Nepal. Nat. Sustain. *2*, 421–428. https://doi.org/10.1038/s41893-019-0277-3.

38. Jones, K.R., Venter, O., Fuller, R.A., Allan, J.R., Maxwell, S.L., Negret, P.J., and Watson, J.E.M. (2018). One-third of global protected land is under intense human pressure. Science *360*, 788–791.

39. Mu, H., Li, X., Wen, Y., Huang, J., Du, P., Su, W., Miao, S., and Geng, M. (2022). A global record of annual terrestrial Human Footprint dataset from 2000 to 2018. Sci. Data *9*, 176. https://doi.org/10.1038/s41597-022-01284-8.

40. Halpern, B.S., Frazier, M., Afflerbach, J., Lowndes, J.S., Micheli, F., O'Hara, C., Scarborough, C., and Selkoe, K.A. (2019). Recent pace of change in human impact on the world's ocean. Sci. Rep. *9*, 11609. https://doi.org/10.1038/s41598-019-47201-9.

41. Keeley, A.T.H., Beier, P., Creech, T., Jones, K., Jongman, R.H., Stonecipher, G., and Tabor, G.M. (2019). Thirty years of connectivity conservation planning: an assessment of factors influencing plan implementation. Environ. Res. Lett. *14*, 103001. https://doi.org/10.1088/1748-9326/ab3234.

42. Balbar, A.C., and Metaxas, A. (2019). The current application of ecological connectivity in the design of marine protected areas. Global Ecology and Conservation *17*, e00569. https://doi.org/10.1016/j.gecco.2019.e00569.

43. Beger, M., Metaxas, A., Balbar, A.C., McGowan, J.A., Daigle, R., Kuempel, C.D., Treml, E.A., and Possingham, H.P. (2022). Demystifying ecological connectivity for actionable spatial conservation planning. Trends Ecol. Evol. *37*, 1079–1091. https://doi.org/10.1016/j.tree.2022.09.002.

44. Brennan, A., Naidoo, R., Greenstreet, L., Mehrabi, Z., Ramankutty, N., and Kremen, C. (2022). Functional connectivity of the world's protected areas. Science *376*, 1101–1104. https://doi.org/10.1126/science.abl8974.

45. Palfrey, R., Oldekop, J.A., and Holmes, G. (2022). Privately protected areas increase global protected area coverage and connectivity. Nat. Ecol. Evol. *6*, 730–737. https://doi.org/10.1038/s41559-022-01715-0.

46. Kremen, C., and Merenlender, A.M. (2018). Landscapes that work for biodiversity and people. Science *362*, eaau6020. https://doi.org/10.1126/science.aau6020.

47. Rhodes, J.R., Armsworth, P.R., Iacona, G., Shah, P., Gordon, A., Wilson, K.A., Runting, R.K., and Bryan, B.A. (2022). Flexible conservation decisions for climate adaptation. One Earth *5*, 622–634. https://doi.org/10.1016/j.oneear.2022.05.010.

48. Tittensor, D.P., Beger, M., Boerder, K., Boyce, D.G., Cavanagh, R.D., Cosandey-Godin, A., Crespo, G.O., Dunn, D.C., Ghiffary, W., Grant, S.M., et al. (2019). Integrating climate adaptation and biodiversity conservation in the global ocean. Sci. Adv. *5*, eaay9969. https://doi.org/10.1126/sciadv.aay9969.

49. Loarie, S.R., Duffy, P.B., Hamilton, H., Asner, G.P., Field, C.B., and Ackerly, D.D. (2009). The velocity of climate change. Nature *462*, 1052–1055. https://doi.org/10.1038/nature08649.

50. Hopkins, C.R., Bailey, D.M., and Potts, T. (2016). Perceptions of practitioners: Managing marine protected areas for climate change resilience. Ocean Coast Manag. *128*, 18–28. https://doi.org/10.1016/j.ocecoaman.2016.04.014.

51. Clarke, P., and Jupiter, S.D. (2010). Law, custom and community-based natural resource management in Kubulau District, Republic of Fiji Islands. Environ. Conserv. *37*, 98–106.

52. Arafeh-Dalmau, N., Brito-Morales, I., Schoeman, D.S., Possingham, H.P., Klein, C.J., and Richardson, A.J. (2021). Incorporating climate velocity into the design of climate-smart networks of marine protected areas. Methods Ecol. Evol. *12*, 1969–1983. https://doi.org/10.1111/2041-210X.13675.

53. Cashion, T., Nguyen, T., ten Brink, T., Mook, A., Palacios-Abrantes, J., and Roberts, S.M. (2020). Shifting seas, shifting boundaries: Dynamic marine protected area designs for a changing climate. PLoS One *15*, e0241771. https://doi.org/10.1371/journal.pone.0241771.

54. Schreckenberg, K., Franks, P., Martin, A., and Lang, B. (2016). Unpacking equity for protected area conservation. Parks *22*, 11–28.

55. McKinnon, M.C., Cheng, S.H., Dupre, S., Edmond, J., Garside, R., Glew, L., Holland, M.B., Levine, E., Masuda, Y.J., Miller, D.C., et al. (2016). What are the effects of nature conservation on human well-being? A systematic map of empirical evidence from developing countries. Environ. Evid. *5*, 8–25. https://doi.org/10.1186/s13750-016-0058-7.

56. Ban, N.C., Gurney, G.G., Marshall, N.A., Whitney, C.K., Mills, M., Gelcich, S., Bennett, N.J., Meehan, M.C., Butler, C., Ban, S., et al. (2019). Well-being outcomes of marine protected areas. Nat. Sustain. *2*, 524–532. https://doi.org/10.1038/s41893-019-0306-2.

57. Buckley, R., Brough, P., Hague, L., Chauvenet, A., Fleming, C., Roche, E., Sofija, E., and Harris, N. (2019). Economic value of protected areas via visitor mental health. Nat. Commun. *10*, 5005. https://doi.org/10.1038/s41467-019-12631-6.

58. Buckley, R.C., and Chauvenet, A.L. (2022). Economic value of nature via healthcare savings and productivity increases. Biol. Conserv. *272*, 109665. https://doi.org/10.1016/j.biocon.2022.109665.

59. Brockington, D., and Igoe, J. (2006). Eviction for conservation: a global overview. Conserv. Soc. *4*, 424–470.

60. Andam, K.S., Ferraro, P.J., Sims, K.R.E., Healy, A., and Holland, M.B. (2010). Protected areas reduced poverty in Costa Rica and Thailand. Proc. Natl. Acad. Sci. USA *107*, 9996–10001. https://doi.org/10.1073/pnas.0914177107.

61. Sims, K.R. (2010). Conservation and development: Evidence from Thai protected areas. J. Environ. Econ. Manag. *60*, 94–114. https://doi.org/10.1016/j.jeem.2010.05.003.

62. Gurney, G.G., Cinner, J., Ban, N.C., Pressey, R.L., Pollnac, R., Campbell, S.J., Tasidjawa, S., and Setiawan, F. (2014). Poverty and protected areas: An evaluation of a marine integrated conservation and development project in Indonesia. Global Environ. Change *26*, 98–107. https://doi.org/10.1016/j.gloenvcha.2014.04.003.

63. López-Angarita, J., Tilley, A., Díaz, J.M., Hawkins, J.P., Cagua, E.F., and Roberts, C.M. (2018). Winners and losers in area-based management of a small-scale fishery in the Colombian Pacific. Front. Mar. Sci. *5*.

64. Brockington, D. (2002). Fortress Conservation: The Preservation of the Mkomazi Game Reserve, Tanzania (Indiana University Press).

65. Naidoo, R., Gerkey, D., Hole, D., Pfaff, A., Ellis, A.M., Golden, C.D., Herrera, D., Johnson, K., Mulligan, M., Ricketts, T.H., and Fisher, B. (2019). Evaluating the impacts of protected areas on human well-being across the developing world. Sci. Adv. *5*, eaav3006. https://doi.org/10.1126/sciadv.aav3006.

66. Oldekop, J.A., Holmes, G., Harris, W.E., and Evans, K.L. (2016). A global assessment of the social and conservation outcomes of protected areas. Conserv. Biol. *30*, 133–141. https://doi.org/10.1111/cobi.12568.

67. Stoudmann, N., Savilaakso, S., Waeber, P.O., Wilmé, L., Garcia, C., Byrne, J., and Adams, V.M. (2023). Overview of evidence on mechanisms affecting the outcomes of terrestrial multiple-use protected areas. One Earth *6*, 492–504. https://doi.org/10.1016/j.oneear.2023.04.005.

68. Barros, F.B., Pereira, H.M., and Vicente, L. (2011). Use and knowledge of the razor-billed curassow by a riverine community of the Oriental Amazonia, Brazil. J. Ethnobiol. Ethnomed. *7*, 1–11.

69. Bizri, A.E., Hani, R., Thaís, Q., José, C., Neto, V., and Carlos, F.A. (2020). Social and Biological Correlates of Wild Meat Consumption and Trade by Rural Communities in the Jutaí River Basin , Central Amazonia Social and Biological Correlates of Wild Meat Consumption. J. Ethnobiol. *40*, 183–201.

70. Gardner, C.J., and Davies, Z.G. (2014). Rural bushmeat consumption within multiple-use protected areas: Qualitative evidence from Southwest Madagascar. Hum. Ecol. *42*, 21–34. https://doi.org/10.1007/s10745-013-9629-1.

71. Marino, A., Blanco, J., Cortes-Vazquez, J., López-Bao, J., Bosch, A., and Durant, S. (2022). Environmentalities of coexistence with wolves in the Cantabrian mountains of Spain. Conserv. Soc. *20*, 345–357.

72. Hussain, S. (2019). The Snow Leopard and the Goat: Politics of Conservation in the Western Himalayas (The University of Washington Press).

73. Alexander, S.E., and Whitehouse, J.L. (2004). Challenges for balancing conservation and development through ecotourism: insights and implications from two Belizean case studies. Sustainable Tourism *9*, 129–142.

74. Buyinza, M., and Acabo, J.T. (2009). Ecotourism for environmental conservation and community livelihoods around Budongo Forest in Masindi District, Western Uganda. Res. J. Appl. Sci. *4*, 160–165.

75. Knežević, R., Žiković, R.G., and Magdić, M. (2011). Factors of sustainable tourism development in the Ogulin-Plaski micro region. Sustainable Tourism: Socio-Cultural, Environmental and Economics Impact, 187–203.

76. Maldonado-Oré, E.M., and Custodio, M. (2020). Visitor environmental impact on protected natural areas: An evaluation of the Huaytapallana Regional Conservation Area in Peru. Journal of Outdoor Recreation and Tourism *31*, 100298. https://doi.org/10.1016/j.jort.2020.100298.

77. Alexander, S.E. (2000). Resident attitudes towards conservation and black howler monkeys in Belize: the Community Baboon Sanctuary. Environ. Conserv. *27*, 341–350.

78. Assalé, A.A., Barima, Y.S., Sangne, Y.C., Bleu, D.K., and Kpangui, K.B. (2020). Évaluation des services d'approvisionnement fournis par les espaces domaniaux anthropisés : cas de la forêt classée du Haut-Sassandra (Centre-Ouest de la Côte d'Ivoire). Can. J. For. Res. *50*, 1002–1011. https://doi.org/10.1139/cjfr-2019-0443.

79. Menton, M.C. (2003). Effects of logging on non-timber forest product extraction in the Brazilian Amazon : community perceptions of change. int. forest. rev. *5*, 97–105.

80. Rizek, M.B., and Morsello, C. (2012). Impacts of trade in non-timber forest products on cooperation among Caboclo households of the Brazilian Amazon. Hum. Ecol. *40*, 707–719. https://doi.org/10.1007/s10745-012-9506-3.

81. Cooper, N.A., and Kainer, K.A. (2018). To log or not to log: local perceptions of timber management and its implications for well-being within a sustainable-use protected area. Ecol. Soc. *23*, art4. https://doi.org/10.5751/ES-09995-230204.

82. Adams, V.M., Visconti, P., Graham, V., and Possingham, H.P. (2021). Indicators keep progress honest: A call to track both the quantity and quality of protected areas. One Earth *4*, 901–906. https://doi.org/10.1016/j.oneear.2021.06.014.

83. Brockington, D., and Wilkie, D. (2015). Protected areas and poverty. Philos. Trans. R. Soc. Lond. B Biol. Sci. *370*, 20140271.

84. Gurney, G.G., Darling, E.S., Jupiter, S.D., Mangubhai, S., McClanahan, T.R., Lestari, P., Pardede, S., Campbell, S.J., Fox, M., Naisilisili, W., et al. (2019). Implementing a social-ecological systems framework for conservation monitoring: lessons from a multi-country coral reef program. Biol. Conserv. *240*, 108298. https://doi.org/10.1016/j.biocon.2019.108298.

85. Hanson, J.O. (2022). wdpar: Interface to the World Database on Protected. Areas. R package version 1.3.3. https://CRAN.R-project.org/package=wdpar.

86. García Molinos, J., Schoeman, D.S., Brown, C.J., and Burrows, M.T. (2019). VoCC: An r package for calculating the velocity of climate change and related climatic metrics. Methods Ecol. Evol. *10*, 2195–2202. https://doi.org/10.1111/2041-210X.13295.

87. Rayner, N.A., Parker, D.E., Horton, E.B., Folland, C.K., Alexander, L.V., Rowell, D.P., Kent, E.C., and Kaplan, A. (2003). Global analyses of sea surface temperature, sea ice, and night marine air temperature since the late nineteenth century. J. Geophys. Res. *108*, 4407. https://doi.org/10.1029/2002JD002670.

88. Jonas, H.D., Barbuto, V., Jonas, H.C., Kothari, A., and Nelson, F. (2014). New steps of change: looking beyond protected areas to consider other effective area-based conservation measures. Parks *20*, 111–128.

89. IUCN (2019). IUCN-WCPA Task Force on OECMs: Recognising and reporting other effective area-based conservation measures. IUCN. https://doi.org/10.2305/IUCN.CH.2019.PATRS.3.en.

90. Lee, V.H.F., Chan, W.W.L., Lee, E.Y.P., Choy, T.S., Ho, P.P.Y., Leung, D.K.C., Lam, K.O., Kwong, D.L.W., Leung, T.W., and Khong, P.L. (2016). Protected areas, country and value: The nature–culture tyranny of the IUCN's protected area guidelines for Indigenous Australians. Antipode *39*, 355–362. https://doi.org/10.1111/anti.12180.

91. Gurney, G.G., Mangubhai, S., Fox, M., Kiatkoski Kim, M., and Agrawal, A. (2021). Equity in environmental governance: perceived fairness of distributional justice principles in marine co-management. Environ. Sci. Pol. *124*, 23–32. https://doi.org/10.1016/j.envsci.2021.05.022.

92. Claudet, J., Ban, N.C., Blythe, J., Briggs, J., Darling, E., Gurney, G.G., Palardy, J.E., Pike, E.P., Agostini, V.N., Ahmadia, G.N., et al. (2022). Avoiding the misuse of other effective area-based conservation measures in the wake of the blue economy. One Earth *5*, 969–974. https://doi.org/10.1016/j.oneear.2022.08.010.

93. McGuire, J.L., and Shipley, B.R. (2022). Dynamic priorities for conserving species. Science *376*, 1048–1049. https://doi.org/10.1126/science.abq0788.

94. Valle Nunes, A., Guariento, R.D., Santos, B.A., and Fischer, E. (2019). Wild meat sharing among non-indigenous people in the southwestern Amazon. Behav. Ecol. Sociobiol. *73*, 26. https://doi.org/10.1007/s00265-018-2628-x.