Article:

Palfrey, R orcid.org/0000-0002-2395-1525, Oldekop, JA and Holmes, G orcid.org/0000-0002-5393-5753 (2022) Privately protected areas increase global protected area coverage and connectivity. Nature Ecology and Evolution, 6 (6). pp. 730-737. ISSN 2397-334X

https://doi.org/10.1038/s41559-022-01715-0

Title: Private protected areas contribute to global protected area coverage and increase PA network connectivity

Full names of all authors, affiliations and email addresses:

1. Miss Rachel Palfrey, The University of Leeds, Leeds, LS2 9JT, eerhp@leeds.ac.uk
2. Dr Johan A. Oldekop, Global Development Institute, The University of Manchester, Manchester M13 9PL, johan.oldekop@manchester.ac.uk
3. Dr George Holmes, The University of Leeds, Leeds, LS2 9JT, g.holmes@leeds.ac.uk

Classification

Major: Biological Sciences

Minor: Environmental Sciences

Corresponding Author:

Rachel Palfrey

eerhp@leeds.ac.uk

+44(0) 7894 24 74 11

Private protected areas contribute to global protected area coverage and increase PA network connectivity

Abstract

Privately protected areas (PPAs) are increasing in number and extent. Yet, we know little about their contribution to conservation, and how this compares to other forms of protected area (PA). We address this gap by assessing the contribution of 17,561 PPAs to the coverage, complementarity and connectivity of existing PA networks in 15 countries across 5 continents. We find that PPAs (i) are three times more likely to be in biomes with <10% of their area protected than other PA governance types and twice as likely to be in areas with the greatest human disturbance; (ii) that they protect a further 1.2% of Key Biodiversity Areas; (iii) that they account for 3.4% of land under protection; and (iv) that they increase PA network connectivity by 7.05%. Our results demonstrate the unique and significant contributions that PPAs can make to the conservation estate and that PPAs deserve more attention, recognition and resources for better design and implementation.

Key words: Privately Protected Areas, Private Land Conservation, Systematic Conservation Planning, Gap Analysis, Ecosystem Representation, Representativeness, Connectivity

Introduction

Terrestrial protected areas (PAs) cover approximately 16% of the world's land mass[1]. However, PAs are disproportionately established in higher and steeper areas that have lower agricultural and economic potential[2]. Therefore, the global PA network underrepresents key species and ecosystems, lacks connectivity, and does not adequately protect areas of high conservation importance. The global PA network thus fails its own goal to comprehensively conserve biodiversity[2, 3]. State governed PAs dominate conservation strategies in most countries[4], but government action alone will be insufficient to reach global PA targets[5, 6]. Co-managed, community governed and privately protected areas (PPAs) are increasingly being used as tools to increase PA coverage and connectivity, and complement existing state PA networks.

PPAs are defined as areas that (i) are governed by private actors; (ii) are primarily engaged in biodiversity conservation activities and have long-term intent to remain in place; and (iii) have legal or other effective means of protection[7]. PPAs vary in landowner types and governance authorities (e.g., individuals, non-governmental organisations (NGOs) or corporate businesses) and protection mechanisms (e.g., conservation easements, NGO freeholds or perpetual landholder agreements). As of November 2018, the World Database on Protected Areas (WDPA) reported 13,250 PPAs representing 5.7% (324,851 km^2) of the total number of all PAs[8], although this is likely to be a significant underestimation because few countries legally recognise or report PPAs[9]. Despite apparent global increases in PPA establishment, recognition and reporting efforts, very little is known about their contribution to the global conservation estate. Moreover, most countries fail to plan or co-ordinate PPA establishment to maximize their conservation benefits[10, 11]. It is thus critical to assess the distribution of PPAs to better understand their contributions to the global conservation estate and identify what potential they have to help achieve global biodiversity targets.

Previous studies suggest that PPAs make different contributions to the conservation estate, compared to PAs under other forms of governance. PPAs tend to be located at lower elevations[10], closer to human settlements[10], in underrepresented ecoregions[11], and in areas of high conservation priority[12]. PPAs have also been found to increase overall PA network connectivity[13, 14]. Yet, these studies have been conducted at national or sub-national levels: to date no international-level analysis has been conducted. Such international-level analyses are needed to provide a more nuanced picture of the current contributions of PPAs to global conservation efforts. This information is critical for better informed global conservation planning, including ecoregion-based conservation strategies that support transnational ecological processes and biodiversity. Furthermore, most studies compare PPAs to state PAs and exclude co-managed or community governed PAs[10, 11]. As conservation approaches continue to diversify, comparisons of different approaches will become more important to determine where, when and why different PA governance types deliver positive biodiversity outcomes.

Here, we conduct the largest study to date on the contributions of PPAs to the global conservation estate. We analyse the contributions of 17,561 PPAs to terrestrial PA networks in 15 countries (Australia, Belize, Brazil, Canada, Chile, Colombia, Finland, Guatemala, Honduras, Kenya, Mexico, Namibia, Peru, South Africa and the USA). Collectively, our case countries represent a wide variety of global ecoregions (377)[15] and biodiversity hotspots (13)[16]. Our study seeks to understand the contributions of PPAs to conservation on an international scale and to assess how these contributions differ to state, co-managed and community governed PAs.

State PAs are governed by federal or national ministries, sub-national ministries or agencies or are areas that have sub-delegated management (e.g., to an NGO)[17]. Co-managed PAs have collaborative management arrangements across different organisations or groups (e.g., La Reserva Nacional Pampa Galeras Bárbara D'Achille, which has a collaborative governance arrangement between the Peruvian government and resident indigenous communities); or transnational boundaries (e.g., Roosevelt Campobello International Park, which is owned and governed by both the American and Canadian government)[17]. Community governed PAs are defined as indigenous peoples' conserved areas and territories, or community conserved areas that are declared and run by local communities[17].

Specifically, we focus on PPA contributions to: (i) PA network coverage; (ii) coverage of threatened or under protected biomes (<10% of biome under protection in case countries); (iii) coverage of Key Biodiversity Areas (KBAs); (iv) coverage of areas of high human disturbance; and (v) PA network connectivity. To generate a better estimate of the relative contribution of PPAs to areas of conservation importance and connectivity, we compare the performance of existing PPAs to a counterfactual dataset with random PPA placement. We limit our analysis to countries with a minimum of 10 PPAs reported to the WDPA to ensure our results offer a more balanced interpretation of the contributions of PPAs. Moreover, to ensure adequate reporting of PPAs, and to justify inclusion within our study, countries in our sample have at least one of the following: (i) legal recognition of PPAs; (ii) national PPA legislation; or (iii) a national PPA database[10]. Although our dataset does not represent a full census of PPAs, it compiles the best currently available data to determine the contributions of PPAs to the global PA estate and provides an important insight into the potential contributions of PPAs in the future (see Methods).

Results & Discussion

Coverage

We find that across our 15 case countries, PPAs cover 246,586 km^2 (an area equivalent to the size of the United Kingdom), accounting for 3.4% of total PA network coverage in these countries. By comparison, state PAs, co-managed, and community governed PAs account for 4,620,065 km^2 (63%), 572,278 km^2 (7.8%) and 1,852,381 km^2 (25%) of total land area under protection, respectively (see Supplementary Information – Table 1). Across the entire WDPA, PPAs account for 1% of the total area of PAs with a reported governance type, with state, co-managed and community governed PAs accounting for 70.5%, 28% and 0.5%, respectively[8].

We find substantial variation in the contribution of PPAs to PA networks in individual countries. South Africa has the highest PPA coverage (25% of total protected land) and Canada the lowest (0.02% of total protected land) (see Supplementary Information – Table 1). This variation is likely the result of historical, environmental, demographic, and economic idiosyncrasies. Across our case countries, differences in PPA distribution could arise from: (i) the difference between the common law system, a legacy of British Colonial Settlement that facilitates private land ownership, and civil law systems used by other European colonial powers, which make private land ownership harder to obtain[18,19]; (ii) presence of established non-governmental PPA networks (e.g. RESNATUR in Colombia and ICMbio in Brazil) that encourage the creation and facilitation of PPAs from a grassroots level; and (iii) presence of and differences in economic incentives. PPAs in South Africa are, at least in part, the result of provincial ordinances. These ordinances have allowed game management and ownership of private land[20], providing an incentive to establish PPAs in grasslands and next to national parks to take advantage of nature-based tourism activities. In the USA, the six largest conservation incentive programs (e.g., The Conservation Reserve Program) target agricultural land (mostly in grassland biomes) and encourage farmers and ranchers to take land out of intensive agricultural production[21]. Financial incentives for grassland conservation also exist in Australia (e.g., plainstender)[22]. Gaining a better understanding of how different incentive mechanisms (both within and between countries) shape the establishment of PPAs will be essential for the creation of more effective management and monitoring systems.

Representation of Biomes, Key Biodiversity Areas and Human Disturbance

In line with Aichi Target 11, we assess the extent to which PPAs contribute to a conservation estate that is ecologically representative (i.e., a conservation estate that contains adequate samples of the full range of existing ecosystems and ecological processes, including at least 10% of each ecoregion within each country) and protects areas important for biological conservation. Biomes represent biodiversity at a broad level and are the most suited biodiversity metric for assessing ecosystem representativeness at an international scale[15]. KBAs highlight sites of global importance for biodiversity that should be prioritised for conservation interventions[23]. We assessed the contribution of PPAs to overall representativeness of PA networks by calculating the area of each biome protected by PPAs and the contribution of PPAs to protecting areas important for biological conservation by calculating the area of KBAs protected by PPAs. We also assess to what extent PPAs protect areas of high human disturbance by calculating the Human Footprint (HF) both within and outside of PAs[24]. We choose the HF because it shows to what extent PPAs are situated in threatened areas and whether they conserve areas of potential conservation concern. PPAs in these areas may protect the last best habitat in a matrix of otherwise degraded lands or be situated in already degraded lands that PPA owners may potentially aim to restore.

Biomes

Within our sample, we find PPAs are three times as likely to be in biomes that do not have 10% of their total area under protection, compared to other PA governance types. We find that 12% of total area of PPAs is in biomes with <10% of their total area protected, compared with 3.9%, 2.3% and 0.5% of state, co-managed and community governed PAs, respectively (Fig 1). We find that 3.2% of randomly placed PPAs are present in biomes with <10% of their total area under protection (Table 1). PPAs contribute most to protecting Mediterranean forests and woodlands (12% of the total area of Mediterranean forests and woodlands under protection is protected by PPAs) (Fig 2). This biome experiences the fourth highest conversion rate from natural vegetation to other land uses (41% of biome area converted globally) and is protected by a skeletal network of PAs (5% of biome protected globally)[25]. Our results suggest PPAs can play a key role in increasing the ecological representativeness of the global PA network and that they are present in biomes that are threatened and underrepresented (<10% of total biome protected).

[Fig 1]

[Fig 2]

We use a complementarity metric (see Methods) to assess whether PPAs protect more or less of a particular biome than would be expected, given the total area of PPAs and that of state, co-managed and community managed PAs. We conduct this analysis to determine if PPAs complement other forms of PAs or if they are generally conserving the same elements of biodiversity. We find that PPAs have greater than expected complementarity for all grassland biomes and for at least seven biomes in total for all other PA governance types (Fig 3). These results show that PPAs are better at representing grasslands than any other PA governance type. This result is critical because grassland biomes are the most significantly degraded biomes globally [25, 26], because habitat conversion in grasslands is exceeding habitat protection by a ratio of 8:1[25], and because grasslands offer a multitude of important ecosystem services[27].

Within our case countries, there is a positive relationship between biomes and ecoregions with large proportions of their area under private ownership (e.g. grasslands biomes and the Atlantic Forest) and the total area protected by PPAs. In Australia, private ownership of grasslands averages 10% across the country but can be as high as 60% in certain states (e.g., Victoria[22]). In the USA, 70% of the Northern Great Plains are privately owned[28]. Within Brazil the majority of PPAs are located within the Atlantic Forest biome, which has 80% of its range under private land ownership[29]. PPAs may be more present in grasslands due to financial incentives for owners of private grasslands[22, 30]. It may also be due to a large number of institutions (e.g., The Nature Conservancy or the Land Trust Alliance) that can support landowners wanting to dedicate their land to private conservation. In addition, the amount of rural development grants and number of NGOs (Non-Governmental Organisations) positively influence the number of conservation easements along the Pacific coast of the USA[31].

[Fig 3]

Key biodiversity areas and areas of high human disturbance

Within our sample, we find that PPAs protect 1.2% of the total area of KBAs, compared with state (32%), co-managed (1.1%) and community governed PAs (2.6%) – (see Supplementary Information – Table 2), and randomly placed PAs (0.68% -Table 1). Twenty percent of the total area of PPAs within our case countries are located within KBAs compared with state (28%), co-managed (1.1%) and community governed PAs (5.8%) (see Supplementary Information – Table 2) and randomly placed PAs (11% - Table 1). Our results show PPAs make a small but nevertheless important contribution to protecting KBAs.

Within our case countries, we found that a greater proportion of the area of PPAs is situated within higher HF areas, compared to other PA types: 47% of the total area of PPAs is in areas with HF ≥3 compared with state (23%), co-managed (11%) and community governed PAs (12% - Fig 4) and randomly placed PAs (43% - Table 1), respectively. We use a human disturbance score of 3 as a threshold, which represents when land can be considered as "human-dominated"[32]. Previous analyses show that this threshold is where species are far more likely to be threatened by habitat loss[33]. To further test that PPAs are situated within areas of higher HFs, rather than being identified as being under high human pressure themselves (as is the case with some PAs)[34], we determine the HF scores of the immediate areas surrounding PPAs (1 km, 5 km and 10 km). We find that 53% of PPAs have a HF score of <3 compared with 34%, 29% and 37% of land within 1 km, 5 km and 10 km of PPAs respectively (see Supplementary Information – Table 3). We also find that 62%, 60% and 58% of individual PPAs have the same or lower HFs than 1 km, 5 km and 10 km buffers surrounding them, respectively. These results show that PPAs have lower human footprint scores than their immediate surroundings. Our findings suggest that PPAs have a key role in conserving areas facing greater pressure from urban and agricultural expansion and other external threats. Furthermore, areas with greater human pressure are also more likely to be substantially degraded[35] and PPAs could thus play a key role in the restoration of degraded lands. PPAs may be more likely to be present in areas of higher HFs due to historic biases in the distribution of private- and state-owned land across high and low productivity landscapes, respectively[36].

We also find that at least twice as much of the total area of PPAs is in areas with the highest HF scores (between 12 - 50) than any other PA governance type: 4% of total area of PPAs compared with 2%, 0.66% and 0.47% for total area of state, co-managed and community governed PAs respectively (Fig 4). We find that PPAs with HF scores between 12 - 50 were situated in large conurbations (e.g., suburbs of São Paulo, Brazil). Urban PAs are distinctively important for two reasons. First, urban PAs can offer key ecological benefits, such as water regulation to reduce flooding, improving air quality and helping to reduce the urban heat island effect[37]. Second, urban PAs can offer experiences in nature to large numbers of people living close to them. Visitors to these areas may be more socially and economically diverse than visitors to more remote PAs[38]. PPAs in urban areas could thus help to broaden and diversify access to nature, promote human health and well-being in under-privileged groups, and help build greater political support for nature conservation within urban populations. However, the proportion of PPAs which allow public access, and the extent to which these potential benefits are realised is unknown. As urban areas and urban populations continue to grow, understanding and protecting biodiversity in cities is of global conservation importance[39].

[Fig 4]

[Table 1 Comparison of current PPA distribution with random placement]

Adjacency

Many species need large areas for roaming distances to reduce genetic bottlenecks, to assist populations in the evasion of natural disasters, and for migration routes[40]. Animal movement can be difficult if PAs are disconnected from one another. We find that on average 11%, 13%, 14% and 18% of PPAs are located within 0 m, 30 m, 100 m and 500 m of a PA of another governance type, respectively. For random placement, 0%, 0.5%, 1.7% and 6% of PPAs were located within 0 m, 30 m, 100 m and 500 m of a PA of another governance type respectively.

We find substantial variation in the percentage of PPAs adjacent to PAs of other governance types across our case countries (Mean = 168, SE = 65). Belize, Kenya, and Namibia have PPAs with the highest levels of adjacency with a PA of another governance type; 60%, 46% and 44% of PPAs at 30 m of state, co-managed and community governed PAs, respectively. However, in Canada, Colombia, Honduras, and Mexico <5% of PPAs are adjacent to PAs of another PA governance type.

Adjacency may be highest in Belize because so much of the country is under some form of protection (37%). Adjacency may also be greater in Kenya and Namibia than other countries due to ecotourism reserves siting along national park boundaries. Adjacency in South Africa is lower than expected perhaps due to the removal of UNESCO biosphere reserves in our analysis. Removal of UNESCO biosphere reserves has a bigger impact in South Africa than other countries because the total area of UNESCO sites in South Africa (109,705 km^2) accounts for 25% of the total area of UNESCO sites across our 15 case countries (429,347 km^2). In contrast, adjacency may be low in Canada, Colombia, Honduras and Mexico because PPAs make up <2% of the total protected area within these countries and therefore there is a reduced probability that they will be located next to PAs of other governance types. Our findings differ to regional or local scale studies showing that PPAs have high adjacency with other PA governance types[13, 14]. This discrepancy may be because these studies have focused on areas of limited size where PPAs are known to be present[14, 41]. Increasing the distance from 0 m to 500 m had no substantial effect on our calculations (See Supplementary Information – Table 4).

Connectivity

Global biodiversity targets call for PAs to be well connected to one another[42]. To determine the contribution of PPAs to total connected protected land in each country, we performed a with and without PPAs scenario analysis using four dispersal distances scenarios of 1 km, 10 km, 30 km, and 100 km[43]. We define dispersal distance as the distance a terrestrial vertebrate species is able to travel between existing populations. We preferentially show results for a dispersal distance of 10 km because that is the median dispersal distance for a terrestrial vertebrate[43]. The exclusion of PPAs decreased contiguous protected land by an average of 7.05% across our case countries (see Supplementary Information – Table 5). This compares to 5.6% for random placement. In our study, the inclusion of PPAs in Kenya made the greatest contributions to PA connectivity increasing the total protected connected land by 29%. This increase may be because PPAs are mainly clustered in one area and are located on the border of national parks. PPAs subsequently connect several national parks together creating one large contiguous patch of connected land. Increases in connectivity are low where there are few PPAs, where PPAs are small are in size, and where they have limited adjacency with other forms of PA. We found that changing dispersal distances (1 km – 100 km) had a limited effect on percentage change of total protected land including/excluding PPAs (see Supplementary Information – Table 6).

Conclusion

Our analysis reveals three important insights. Firstly, PPAs can protect areas that are under-represented by PAs under other forms of governance and contribute to protecting KBAs. Across our case countries, a greater proportion of PPAs are found in biomes that have <10% of their total area protected and areas of higher human pressure. Secondly, PPAs' contribution to PA coverage is significant in some countries but negligible in others. For the five countries in our sample with the greatest contribution to national PA coverage (South Africa, Guatemala, Belize, Namibia, and Peru), PPAs account for 15% of the total area protected in those countries. Thirdly, PPAs make a modest contribution to the connectivity of national PA networks. We found that 38% of PPAs are adjacent to a PA of another form of governance and that the inclusion of PPAs increase protected connected land by 19% for the five countries with highest PPA adjacency and connectivity within our sample. It is important to note that due to underreporting to the WDPA and national-level platforms, our findings represent a "bare minimum" of the contribution of PPAs. Improvements in PPA reporting would likely reflect a greater contribution from PPAs to the global conservation estate.

We suggest that greater legislative, technical and financial support for PPAs and a more co-ordinated approach to their establishment could help maximise their benefits. These forms of support could incentivise and facilitate the establishment of PPAs and help PPA owners implement better land management and restoration practices. Greater co-ordination of PPA establishment could be achieved by: (i) creating frameworks for the inclusion of PPAs into national conservation strategies; (ii) the creation of PPA support networks (such as RESNATUR in Colombia); and (iii) supporting countries with the recording and reporting of PPA boundaries (with the consent of relevant authorities and organisations) to national authorities and the WDPA.

Reporting of PPA boundaries requires time, resources and institutional infrastructure, which some governments may lack. In some countries, political situations may mean PPA land holders and governance authorities and/or governments may be less willing to gather and report data on PPA boundaries[13]. Civil society organisations, land trusts and PPA networks working in these countries can play a key role in facilitating the reporting of PPAs to the WDPA and other authorities with appropriate consent. Additionally, indigenous and local communities may have competing claims to the land contained within some PPAs, often based on customary tenure, which may or may not be recognised by governments. PPA owners and networks therefore have a responsibility and moral duty to ensure that these claims are adequately addressed and resolved in an appropriate and ethical manner, recognising the power disparities that often exist between conservation organisations and indigenous and local communities. Lastly, we encourage future research to assess the spatial contributions of PPAs in other regions to examine the underlying factors and governance structures that lead to specific spatial configurations of PPAs. Such analyses should include efforts to better understand the role of different stakeholders (e.g., private landowners, NGOs and land trusts) and their motivations for the establishment of PPAs, as well as assessments of national policies and incentives that support PPAs.

Methods

Our study uses PA boundaries and global spatial datasets of biome distribution[15], key biodiversity areas (KBAs)[44] and human disturbance[24] to determine the contributions of PPAs to global conservation. We used ArcMap 10.4, Conefor2.6[45] and R[46] for all our analyses.

Spatial layers and processing

We downloaded the November 2018 version of the WDPA from http://www.protectedplanet.net/ as a primary source for PA boundaries[8]. As in other PA assessments[43], we excluded from subsequent analysis 233 PAs with a "proposed" and 439 with a “not reported” status, 29 PAs reported as points without an associated area and 75 UNESCO Man and Biosphere Reserves. When point data was included, we created a circular buffer around the point in ArcMap to account for the total size of the reported PA area. We buffered 16 points. These circular buffers are unlikely to represent the real shape of the PA or their exact location because location points provided by the WDPA can be either in the centre of the PA or on an outer edge. This discrepancy could impact our connectivity analysis by affecting the distance to the nearest PA by up to the half the actual width of the PA (if the point is located on an outer edge of the PA). However, we feel that these discrepancies are likely to have a limited impact on our study because buffered PAs were few and small (mean size = 5 km^2). Remaining PA boundaries were classified into five reported governance types (state, co-managed, community, private and non-reported) using the GOV_TYPE field in the WDPA. We filtered for PPAs using the following GOV_TYPE values; For-profit organisations, Non-profit organisations and Individual landowners. All PA management types (Ia to VI) are included within our study. As of November 2018, the WDPA reported 13,250 designated PPA boundaries. We identified a further 11,074 PPAs that had been incorrectly reported through the DESIG field, which details the designation of a PA at the national level (i.e., Private Natural Heritage Reserves, *Reservas Particulares do Patrimônio Natural*, in Brazil are mistakenly reported as being under government management when they are, in fact, privately protected[9] - see Supplementary Information Table 7). We cross-checked these potential PPAs by consulting with national PPA experts and conducting document analyses of open access materials (e.g., in Honduras, we contacted employees working for the National Institute for Conservation and Forest Development, Protected Areas, and Wildlife (ICF)). We based our definitions of PA governance types on that of the IUCN[7] (see Introduction). We excluded 1,346 PAs with a non-reported governance type. Excluding PAs with non-reported governance type accounted for 6% of the total area of PAs in our 15 case countries. The minimum and maximum size of a PA that we excluded from the study due to no governance being reported was 1 km^2 and was 30,893 km^2, respectively, with a mean size of 306 km^2.

We classified remaining PA boundaries by ISO3 country code. We excluded countries with <10 PPAs reported to the WDPA from subsequent analysis resulting in a global subset of 15 countries as any potential PPA effects would be negligible. We also excluded the UK due to difficulties in establishing the governance structure of potential PPAs because areas under habitats directives and other such initiatives are all reported as government PAs, even when managed by private entities. We obtained additional PPA data for our case countries from multiple sources outside of the WDPA (e.g., Chile's Asociación de Iniciativas de Conservación en Areas Privadas y de Pueblos Originarios (http://asiconservachile.cl/acch/) (see Supplementary Information – Table 7). This resulted in an extra 1,038 PPA boundaries (70,240 km^2) that had not been reported to the WDPA (see Supplementary Information – Table 7). We ensured all additional PPA boundaries met our standard definition of a PPA through consultation with PPA experts in their regions of expertise and document analysis of open access material.

We dissolved PA boundaries with the same governance type to remove overlaps and erased overlaps between PAs of different governance types to avoid double counting[11]. To determine which governance classification to retain, we created a governance hierarchy: state governance, co-management, community governance, and private governance[11]. This hierarchy is based on the strength of legal recognition and environmental protection security that each governance type offers[11]. We designated state PAs as the highest tier because they have the strongest legal standing across all countries and can provide strict environmental protection. We designated PPAs as last in our hierarchy because in some countries (e.g., Chile) PAS have no legal recognition, regulation and no guaranteed permanence. Hereafter these layers are referred to as 'PA governance layers'. While establishing this hierarchy was necessary for the analysis, we recognise that the assumptions made will not reflect reality in all cases, since the level of recognition, strictness of protection and quality of conservation outcomes will all vary within and between governance types. We removed 6% (105,441 km^2), 5% (664,824 km^2) and 3% (439,589 km^2) of private, co-managed and community governed PAs respectively, due to boundaries overlapping with a governance type prioritised by our hierarchy.

The number of reported PPAs is believed to be a significant underestimation of their total number[9]. The quality of available data is highly variable depending on the original data source[47]. Here, we have used the best available data, collected from multiple sources (see Supplementary Information – Table 7), to provide initial insights into the spatial outcomes of PPAs. After the removal of overlaps, our final dataset included 13,206 PPA boundaries originally reported to WDPA, 3,317 PPA boundaries from within the WDPA that had been incorrectly reported as having another PA governance type and 1,038 PPA boundaries from additional sources (see Supplementary Information – Table 7). This resulted in 17,561 PPA boundaries in total.

We used the World Wildlife Fund's (WWF) terrestrial ecoregions layer to assess biome complementarity between PPAs and PAs under other governance types[15]. We used the 14 biomes (i.e., the natural vegetation that would be expected in that area assuming minimal human disturbance) identified by WWF as our unit of analysis because we could not be confident enough in the accuracy of the PA boundaries or ecoregions to make comparisons at the ecoregion level. Hereafter this layer is referred to as the 'biome layer'.

We used the Key Biodiversity Area (KBAs) dataset[44] to assess what degree PAs under difference governance types protect KBAs. KBAs are sites contributing significantly to the global persistence of biodiversity. We used the 2018 released Global Footprint Dataset (V3)[24], which compiles the cumulative human environmental pressure in 2009, to assess to what degree PAs under difference governance types protect areas of greater human disturbance. We used the Global Human Footprint dataset as it the most complete and highest-resolution globally consistent terrestrial dataset on cumulative human pressures on the environment[40]. All data were projected in Mollweide (World) as this is an equal area protection to calculate the total area of PAs within different biomes or degrees of human disturbance.

Analysis

We conducted spatial analyses in ArcMap 10.04 and Conefor 2.6[45] to determine the total area of PAs within different biomes, degrees of human disturbance and their overlap with KBAs and the contribution of PPAs to national PA network connectivity. We determined the total area of PAs within each PA governance layer using the calculate geometry tools. As per previous studies that determine what PAs protect[11], we clipped each of the biome, HPF and KBA layers with the different PA governance layers to determine the overlap between each.

To determine the complementarity of PPAs to other governance types for what biomes they protect we used an adapted a complementarity metric[10]. We define complementarity in this context to mean to what extent PPAs supplement the biome coverage of PAs of other governance types and increase overall biome representation within the PA network of our 15 case countries:

$$Complementarity\ metric\ of\ a\ biome = Mc = \frac{Pp *\ R - Op}{Pp * R + Op}$$

Where Pp = the percentage of a particular biome conserved by PPAs; R = (Area of State or Co-managed or Community governed PA) / PPA Area; Op = the percentage of a particular biome conserved by either state, co-governed or community governed PA. This metric is on a scale of -1 to +1, where negative values indicate less than expected complementarity and positive values indicate greater than expected complementarity between PPAs and PAs under other governance regimes. Expected complementarity is determined by the ratio of the area of PPAs to PAs under other governance types.

We then generated a network of random reserves, equal in area to the current PPA network within each country, to evaluate the coverage of existing PPAs relative to random counterfactuals. We generated this network by randomly selecting cells from a grid until reaching the area of the current reserve network within each country was reached. As per previous studies, the grid size was equal to the average size of each PA governance type within each country[48]. This process was repeated 1,000 times using R[46] to provide an average of the total area of each biome, HF and KBA present within our model. We summed totals for each country to give a general overview for our case countries.

Connectivity. To conduct our connectivity analysis, we used undissolved polygons. We assessed connectivity using two metrics: adjacency and connected protected land.

Adjacency. We measured the adjacency of PPAs to PAs under other governance regimes using the select by location tool. Due to small misalignments in polygon boundaries, PPA adjacency may be inflated because only a small portion (i.e., 1 – 2% of the total area of a PPA) does not overlap with a PA of another governance type. This was the case for 5,102 PPAs (20% of the total number and mostly from Finland) and they were removed from this part of the analysis. To further account for small inaccuracies in the location of PA polygon boundaries we considered four within distance measurements of 0 m, 30 m, 100 m and 500 m to see what difference changing the distance of the buffer made to our results (see Supplementary information – Table 4). As with previous studies[41], we preferentially show a 30 m buffer as we believe that it accounts for most minor inaccuracies in the location of PA boundaries. To test if the placement of PPAs around other forms of PAs occurs by chance or there if there are underlying factors, we generated 20 randomized maps in which the same PPA polygons were moved and rotated at random to new locations within each study country. We used the '*sp*' package in R[46] to generate a new random centroid for each PPA around which the polygon shape was then redrawn. If there were any overlaps between polygons, the script would rerun until a map of non-overlapping PA could be drawn. We re-ran the selection by location tool in ArcMap10.4 for each randomized map and averaged the results and compared that to those for the existing protected area network[14].

Connected protected land. To determine the contribution of PPAs to connected PA networks, we used Conefor2.6[45] in command line (http://www.conefor.org). We performed a with and without PPAs scenario analysis using four dispersal distances of 1 km, 10 km, 30 km, and 100 km (as per previous studies[43]), to determine the equivalent connected area (ECA) of PA networks in each country. The ECA equates to the size of a single patch (PA) that would provide the same value of the probability of connectivity than the actual PA network in a country or continent. In effect, it summarizes the amount of reachable area in the PA network[43]. From the ECA we computed the normalized Equivalent Connected Area (ECA_{norm})[43], a connectivity metric that summarizes the percentage of reachable area in a PA network compared to the total country area, generally referred to as protected connected land. The protected connected land indicator assumes that PAs are effectively managed for connectivity (i.e., there are no important barriers for species movements and other ecological flows within PAs)[43]. We preferentially use a dispersal distance of 10 km as per previous studies[43].

To test if the current PPA network distribution performs better or worse than random at increasing connectivity we also ran scenarios incorporating the 20 randomized PPA maps created for each country (previously created for the adjacency analysis). We averaged the results of the 20 randomized scenarios and compared these to those for the existing protected area network[14].

Limitations

We identify three potential limitations to our analysis. First, this analysis predominately relies of PA boundaries reported to the WDPA. The quality of the data reported to the WDPA can be highly variable depending on the original data source[47]. Until recently, data quality on the WDPA has been measured and reported rather than controlled[47]. Data quality issues may include incorrect or missing attributes (e.g. GOV_TYPE)[9, 47], differences in the reported PA area and the submitted polygon boundaries[47], and presence of PA boundaries that may be degazzetted[50]. These issues can cause both under or over-estimations of the coverage of PAs. Most pertinent to our study is the underreporting of PPAs to the WDPA, which is widely discussed in the literature[9]. An underreporting of PPAs means that results regarding how much PPAs contribute to total PA network coverage, protecting of KBAs and connectivity within our case countries are an under-estimate and should be regarded as a bare minimum. We are also aware that PPAs can be underreported in a biased way. For example, in some countries (e.g., Australia and Canada) certain states or provinces do not report or legally recognise PPAs[7]. We have attempted to mitigate this by contacting local experts who may have access to data currently unpublished at international and national levels. The failure of some states / provinces to report PPAs may lead to a bias of our results regarding the representativeness of PPAs, however the impacts of this may be limited. This is because biomes are mapped at such a large scale that each biome in each country covers multiple states. Therefore, if one or two states fail to report PPAs it is likely that those biomes will still be represented by PPAs within other states. Additionally, the omission of a small subset of states within the country will have a limited impact of the general trend of where PPAs are located. The impact upon HF is harder to determine. However, most states/regions share similar characteristics of having more remote and less remote areas. Therefore, it is not implausible that the characteristics of PPAs in states that fail to report PPAs may be similar to those of PPAs in states or regions that do (see Supplementary Information – Fig. 1). It should also be noted that we have analysed countries with good PPA networks and/or reporting. Therefore, our results cannot be more broadly applied to other countries that have not been included within our study. However, our study shows what may be possible if PPA creation is supported and encouraged by a wider number of countries.

Second, there is a temporal mismatch between HF dataset (2009) and the PPA dataset (2018). With human pressures continuing to rise this could mean that our calculated HFs within and surrounding PAs are an underestimate of the true values in 2018. However, the average global HF only rose by 9% between 1993 and 2009 (16 years) despite an increase of 23% in global population and 153% in the world economy. Therefore, the effects of the temporal mismatch in our data (8 years) are likely to be small[49]. Moreover, for the 60% (10,537) of PPA boundaries for which we have the designation year, 70% (7,376) were established before 2009. Third, we do not assess the extent to which PPAs protect threatened species, beyond their coverage of KBAs (despite some PPAs being set up for specific species) due to a lack of high-quality information on the presence/absence of species in individual PPAs (particularly difficult due to their small size and infrequent use of comprehensive species lists), population densities, minimum viable population sizes of threatened species.

Lastly, although a PPA may be reported in a given location, this does not mean that it is successfully conserving biodiversity, or that it will remain in place in perpetuity[50]. Assessing the effectiveness of PPAs is beyond the remit of this study and future studies should assess the effectiveness of PPAs in different countries. The few studies that have assessed the performance of PPAs to mitigate deforestation and degradation[51] and land cover change[52] found that PPAs are more effective than other forms of PA. Studies of the permanence of PPAs showed that only 6.2% of PPAs were degazetted in a 92 year period (compared with 2.2% for state PAs)[19].

Acknowledgements

This research was supported by ESRC through the White Rose Doctoral Training Programme. The author would like to thank Richard Rigby for the building of the model for random placement. This paper is dedicated to the memory of Elisa Corcuera Vliegenthart, who did so much for privately protected areas.

Author contributions

R.P. designed the study, conducted the analysis and wrote the first draft of the manuscript. GH. and J.A.O. contributed substantially to both the design of the study and revisions of the manuscript.

Data Availability Statement

All data used in this manuscript is openly available online. Protected areas boundaries are available from: http://www.protectedplanet.net/, the WWF ecoregions layer is available from: https://www.worldwildlife.org/publications/terrestrial-ecoregions-of-the-world, Global Human Footprint Dataset V3 (2009) is available from: https://sedac.ciesin.columbia.edu/data/set/wildareas-v3-2009-human-footprint and Key Biodiversity Areas are available on request by filling out a form found at: http://www.keybiodiversityareas.org/kba-data/request

Code Availability Statement

The R script used to rotate and move polygons can be found at: https://github.com/cemac/rotate-move-pas.

The R script for random placement of PPAs can be found here: https://github.com/eerhp/Privately_Protected_Areas_Palfrey_2021

Tables

Table 1: Comparison of the current PPA distribution with random placement

% of PPAs within biomes receiving 0 – 9.99%, 10 – 16.99% and 17% + overall protection		
	Current PPA distribution	Random Placement
0 – 9.99% of biome protected	12	3.2
10 – 16.99% of biome protected	45	42
17 – 100% of biome protected	43	54

% of PPAs within each Human Footprint grouping		
	Current PPA distribution	Random Placement
0	25	26
1 – 2	28	33
3 - 5	28	25
6 – 11	14	12
12 – 50	5	4
% of PPAs within Key Biodiversity Areas		
	Current PPA distribution	Random Placement
% of KBA protected by a PPA	1.2	0.7
% of PPA area within KBAs	20	11

Figure Legends / Captions

Fig 1. Proportion of each biome protection level protected by PA governance types

Proportion of areas of human disturbance (ranked between 0 and 50) protected by state, co-managed, community and privately governed PAs. 0 = no human pressure, 1 – 2 = low human pressure, 3 – 5 = moderate human pressure, 6 – 11 = high human pressure, 12 - 50 = very high human disturbance.

Fig 2. Proportion of each terrestrial biome protected by protected areas

a, Proportion of total area of biome protected across our 15 case countries covered by each governance type (state, co-managed, community, private), ordered by proportion of private governance. A star (*) indicates biomes where <10% of their total area is protected by any form of PA. Biome abbreviations: Flooded g./sav. = Flooded grasslands and savannas; Trop./sub. g./sav./sh. = Tropical and subtropical grasslands, savannas, and shrublands; Montane g./sh. = Montane grasslands and shrublands; Temp. g./sav./sh. = Temperate grasslands, savannas, and shrublands; Deserts/x. sh. = Deserts and xeric shrublands; Med. for./wd./scrub = Mediterranean forests, woodlands, and scrub; Boreal for./taiga = Boreal forests/taiga; Temp. Con. For. = Temperate conifer forests; Temp. br./ mix. for. = Temperate broadleaf and mixed forests; Trop./sub. con. for. =Tropical and subtropical coniferous forests; Trop./sub. dry br. for. = Tropical and subtropical dry broadleaf forests; Trop./sub. moist br. for. = Tropical and subtropical moist broadleaf forests.

Fig 3. Complementarity of PPAs to other governance types in protecting terrestrial biomes

a – c, Relative proportions of biomes protected by **(a)** state, **(b)** co-governed and **(c)** community governed PAs compared with and privately governed PAs**.** + values (0 – 1) = greater than expected complementarity of a given biome between PPAs and either state/co-managed/community PAs accounting for the difference in the total protected area of PPAs and state, co-managed or community PAs. – values (0 - -1) = less than expected complementarity of a given biome between PPAs and either state/co-managed/community PAs accounting for the difference in the total protected area of PPAs and state, co-managed or community PAs. A value of 0.2 indicates twice as much of a biome represented by a PPA than would be expected given the total ratio of PPAs to a PA of a state/co-managed/community PA.

Fig 4 . Proportion of each HF category protected by PA governance types

Proportion of areas of protection level; 0 – 9.99% of biome protected, 10 – 16.99% of biome protected and 17 – 100% of biome protected by state, co-managed, community and privately governed PAs. MH = Million Hectares.

Cited Literature

1. UNEP-WCMC and IUCN (2021), Protected Planet: The World Database on Protected Areas (WDPA) On-line, September 2021, Cambridge, UK: UNEP-WCMC and IUCN. Available at: www.protectedplanet.net'
2. Venter, O., Magrach, A., Outram, N., Klein, C. J., Possingham, H. P., Di Marco, M. & Watson, J. E. M. (2018) Bias in protected-area location and its effects on long-term aspirations of biodiversity conventions, Conservation Biology, 32(1), 127-134
3. Ward, M., Saura, S., Williams, B., Ramirez-Delgado, J. P., Arafeh-Dalmau, N., Allan, J. R., Venter, O., Dubois, G, & Watson, J. E. M. Watson (2020) Just ten percent of the global terrestrial protected area network is structurally connected via intact land, Nat Communications, 11, 4563
4. Adams, W.M. (2004) Against Extinction: The Story of Conservation, Earthscan
5. Watson, J. E. M. Dudley, Segan, N. & Hockings, D. B. (2014) The performance and potential of protected areas, Nature, 515, 67-73
6. Butchart, S. H. M., Clarke, M., Smith, R. J., Sykes, R. E., Scharlemann, J. P. W., Harfoot, M., Buchanan, G. M., Angulo, A. et al., (2015) Shortfalls and solutions for meeting national and global conservation area targets, Conservation Letters, 8: 329–337
7. Stolton, S., Redford, K.H., Dudley, N., Bill, W., Corcuera, E. and Mitchell, B.A. (2014) The Futures of Privately Protected Areas. IUCN, Gland, Switzerland
8. UNEP-WCMC and IUCN (2018), Protected Planet: The World Database on Protected Areas (WDPA) On-line, November 2018, Cambridge, UK: UNEP-WCMC and IUCN. Available at: www.protectedplanet.net'.
9. Bingham, H., Fitzsimons, J.A., Redford, K.H., Mitchell, B.A., Bezaury-Creel, J. & Cumming, T.L. (2017) Privately protected areas: advances and challenges in guidance, policy and documentation. Parks. 23(1), 13–28

10. Gallo, J. Pasquini, L., Reyers, B. & Cowling, R.M (2009) The role of private conservation areas in biodiversity representation and target achievement within the Little Karoo region, South Africa, Biological Conservation, 142(2), 446 – 454
11. SCHUTZ, J. (2018) Creating an integrated protected area network in Chile: A GIS assessment of ecoregion representation and the role of private protected areas. Environmental Conservation, 45(3), 269-277
12. Ielyzaveta, I. M. & Cook, C. N. (2020) The role of privately protected areas in achieving biodiversity representation within a national protected area network, Conservation Science and Practice, 2(12), e307
13. Graves, R. A, Williamson M. A, Belote, R. T, & Brandt J. S. (2019) Quantifying the contribution of conservation easements to large-landscape conservation, Biological Conservation, 232:83–96.
14. De Vos, A., & Cumming, G. S. (2019) The contribution of land tenure diversity to the spatial resilience of protected area networks, People and Nature, 1: 331–346.
15. Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V. N. et al., (2001) Terrestrial Ecoregions of the World: A New Map of Life on Earth: A new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity, *BioScience*, 51(11), 933–938
16. Myers, N., Mittermeier, R. A., Mittermeier, C. G., Da Fonseca, G. A. B. & Kent, J. (2000) Biodiversity hotspots for conservation priorities, Nature, 403, 853-858.
17. Borrini-Feyerabend, G., N. Dudley, N. Jaeger, T. Lassen, B. Broome, A. Phillips, & T. Sandwith. (2013) Governance of protected areas: from understanding to action. Gland: IUCN
18. Lee, A. & Schultz, K. A. (2012), "Comparing British and French Colonial Legacies: A Discontinuity Analysis of Cameroon", Quarterly Journal of Political Science, 7(4), 365-410

19. Acemoglu, D., Johnson, S. & Robinson, J.A. (2001) The Colonial Origins of Comparative Development: An Empirical Investigation, The American Economic Review, 91 (5), 1369-1401
20. De Vos, A., Clements, H S., Biggs, D. & Cumming, G. S. (2019) The Dynamics of proclaimed privately protected areas in South Africa over 83 years, Conservation Letters, 12(6), e12644
21. United States Department of Agriculture (2019) Conservation Programs, [online] Available at: <https://www.ers.usda.gov/topics/natural-resources-environment/conservation-programs/> [Accessed 21st October 2021]
22. Zimmer, H. C., Mavromihalis, J., Turner, V. B., Moxham, C. & Liu, C. (2010) Native grasslands in the PlainsTender incentive scheme: conservation value, management and monitoring, The Rangeland Journal 32(2) 205-214
23. IUCN (2016) A Global Standard for the Identification of Key Biodiversity Area, [pdf] Available at: <https://portals.iucn.org/library/sites/library/files/documents/Rep-2016-005.pdf> [Accessed 21 March 2021]
24. Venter, O., E. W. Sanderson, A. Magrach, J. R. Allan, J. Beher, K. R. Jones, H. P. Possingham, W. F. Laurance, P. Wood, B. M. Fekete, M. A. Levy, and J. E. Watson. 2018. Last of the Wild Project, Version 3 (LWP-3): (2018) Human Footprint, 2018 Release. Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). https://doi.org/10.7927/H46T0JQ4. Accessed 19th October 2021.
25. Hoekstra, J. M., Boucher, T. M., Ricketts, T. H. & Roberts, C. (2005) Confronting a biome crisis: Global disparities of habitat loss and protection, Ecology Letters, 8, 23-29
26. Newbold, T., L. N. Hudson, A. P. Arnell, S. Contu, A. De Palma, S. Ferrier, S. L. L. Hill, A. J. Hoskins, I. Lysenko, H. R. P. Phillips, V. J. Burton, C. W. T. Chng, S. Emerson, D. Gao, G. Pask-Hale, J. Hutton, M. Jung, K. Sanchez-Ortiz, B. I. Simmons, S. Whit-Mee, H. Zhang, J. P. W. Scharlemann, and A. Purvis (2016):

"Has land use pushed terrestrial biodiversity beyond the planetary boundary? A global assessment," Science, 353, 288–291

27. Bengtsson, J., Bullock, J. M., Egoh, B., Everson, C., Everson, T., O'Connor, T., O'Farrell, P. J., Smith, H. G. & Lindborg, R. (2019) Grasslands – more important for ecosystem services than you might think, Ecosphere, 10(2), e02582
28. WWF (2021) Working together for Grasslands. How Ranchers and the WWF help protect the Northern Great Plains, Available at: < https://www.worldwildlife.org/stories/working-together-for-grasslands> [Accessed 28 June 2021]
29. Henderson, K. A., Reis, M., Blanco, C. C., Pillar, V. D., Printes, R. C., Bauch, C. T. & Anand, M (2016) Landowner perceptions of the value of natural forest and natural grassland in a mosaic ecosystem in southern Brazil, Sustainability Science, 11(1), 321–330
30. Kamal, S., Grodzinska-Jurczak, M. & Brown, G. (2015) Conservation on private land: a review of global strategies with a proposed classification system. Journal of Environmental Planning and Management, 58, 576-597
31. Williamson, M. A., Schwartz, M. W & Lubell, M. N. (2018) Spatially Explicit Analytical Models for Social–Ecological Systems, BioScience, 68(11), 885 – 895
32. Watson, J. E. M., Jones, K. R., Fuller, R. A., Di Marco, M., Segan, D. B., Butchart, S. H. M., Allan, J. R., McDonald-Madden, E. & Venter, O. (2016) Persistent Disparities between Recent Rates of Habitat Conversion and Protection and Implications for Future Global Conservation Targets, Conservation Letters, 9(6), 413 – 421
33. Di Marco, M., Brooks, T., Cuttelod, A. et al. (2015). Quantifying the relative irreplaceability of important bird and biodiversity areas. Conserv. Biol., 30, 392-402
34. Jones K. R., Venter, O., Fuller, R. A., Allan, J. R., Maxwell, S. L., Negret, P. J. & Watson (2018) One-third of global protected land is under intense human pressure, Science, 360(6390), 788-791

35. Sanderson, E. W., Jaiteh, M., Levy, M. A., Redford, K. H., Wannebo, A. V. & Woolmer, G. (2002) The Human Footprint and the Last of the Wild: The human footprint is a global map of human influence on the land surface, which suggests that human beings are stewards of nature, whether we like it or not, BioScience, 52(10), 891–904
36. Clements, H. S., Kerley, G. I. H., Cumming, G. S., De Vos, A. & Cook C. N. (2018) Privately protected areas provide key opportunities for the regional persistence of large- and medium-sized mammals, Journal of Applied Ecology, 56(3), 537 – 546
37. Song P, Kim G, Mayer A, He R, & Tian G. (2020) Assessing the Ecosystem Services of Various Types of Urban Green Spaces Based on i-Tree Eco. Sustainability, 12(4),1630
38. Trzyna, T. (2014) Urban Protected Areas: Profiles and best practice guidelines. Best Practice Protected Area Guidelines Series No. 22, Gland, Switzerland: IUCN. xiv + 110pp.
39. Li, E., Parker, S. S., Pauly, G. B., Randall, J. M., Brown, B.V. & Cohen, B. S. (2019) An Urban Biodiversity Assessment Framework That Combines an Urban Habitat Classification Scheme and Citizen Science Data, Frontier Ecology and Evolution, 7
40. Venter, O., Sanderson, E., Magrach, A. et al. (2016) Global terrestrial Human Footprint maps for 1993 and 2009. Sci Data 3, 160067 (doi:10.1038/sdata.2016.67)
41. Rissman, A. R. & Merenlender, A. M. (2008) The Conservation Contributions of Conservation Easements: Analysis of the San Francisco Bay Area Protected Lands Spatial Database. Ecology and Society, 13, 25.
42. CBD (Convention on Biological Diversity) (2011) Strategic Plan for Biodiversity 2011-2020, Including Aichi Biodiversity Targets. https://www.cbd.int/sp/
43. Saura, S., Bastin, L., Battistella, L., Mandrici, A. & Dubois, G. (2017) Protected areas in the world's ecoregions: How well connected are they? Ecological Indicators, 76, 144-158

44. BirdLife International (2020) World Database of Key Biodiversity Areas. Developed by the KBA Partnership: BirdLife International, International Union for the Conservation of Nature, American Bird Conservancy, Amphibian Survival Alliance, Conservation International, Critical Ecosystem Partnership Fund, Global Environment Facility, Global Wildlife Conservation, NatureServe, Rainforest Trust, Royal Society for the Protection of Birds, Wildlife Conservation Society and World Wildlife Fund. September 2020 version. Available at http://www.keybiodiversityareas.org/site/requestgis
45. Saura, S. & J. Torné. (2009) Conefor Sensinode 2.2: a software package for quantifying the importance of habitat patches for landscape connectivity. Environmental Modelling & Software 24: 135-139
46. R Core Team (2014) R: A Language and environment for statistical computing. Vienna, Austria: R foundation for statistical computing. Retrieved from http://www.R-Project.org/
47. Milam, A., Kenney, S., Juffe-Bignoli, D., Bertzky, B., Corrigan, C., MacSharry, B., Burgess, N. D., & Kingston, N. (2016) Maintaining a Global Data Set on Protected Area, In: Joppa et al., ed. 2016 Protected Areas; Are They Safeguarding Biodiversity? Wiley-Blackwell, Oxford UK pp. 81 – 101
48. Mason, C., Alderman, R., McGowan, J., Possingham, H. P., Hobday, A. J., Sumner, M. & Shaw, J. (2018) Telemetry reveals existing marine protected areas are worse than random for protecting the foraging habitat of threatened shy albatross, Diversity and Distributions, 24(12), 1744-1755
49. Venter, O., Sanderson, E. W., Magrach, A., Allan, J. R., Beher, J., Jones, K. R., Possingham, H. P., Laurance, W. F., Wood, P., Fekete, B. M., Levy, M. A. & Watson, J. E. M. (2016) Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation, Nature Communications, 7, 12558

50. Lewis, E., MacSharry, B., Juffe-Bignoli, D., Harris, N., Burrows, G., Kingston, N. & Burgess, N. D. (2017) Dynamics in the global protected-area estate since 2004,. Conservation Biology, 33(3), 570 – 579
51. Schleicher, J., Peres, C. A., Amano, T., Llactayo, W. & Leader-Williams, N. (2017) Conservation performance of different conservation governance regimes in the Peruvian Amazon, Nature, 7(1), 113 – 118
52. Shumba, T., De Vos, A., Biggs, R., Esler, K J., Ament, J. M. & Clements, H. S. (2020) Effectiveness of private land conservation areas in maintaining natural land cover and biodiversity intactness, Global Ecology and Conservation, 22, e00935