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LETTER

Equitable Representation of Ecoregions is Slowly Improving Despite Strategic Planning Shortfalls

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Abstract

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

Introduction

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

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

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

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

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

Methods

Data

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

Patterns of PE through time

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

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

Drivers of change in PE: Inevitable or deliberate?

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

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

Economic, social, and ecological drivers of change in PE

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

Results

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

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

Patterns in PE through time

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

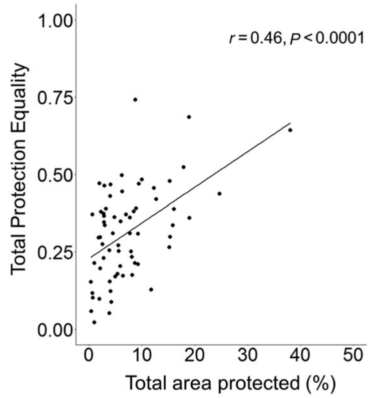

Figure 1 The correlation between the total area protected and total PE in each studied country as of 2013 ($n = 66$).

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

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

Drivers of change in PE: Inevitable or deliberate?

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

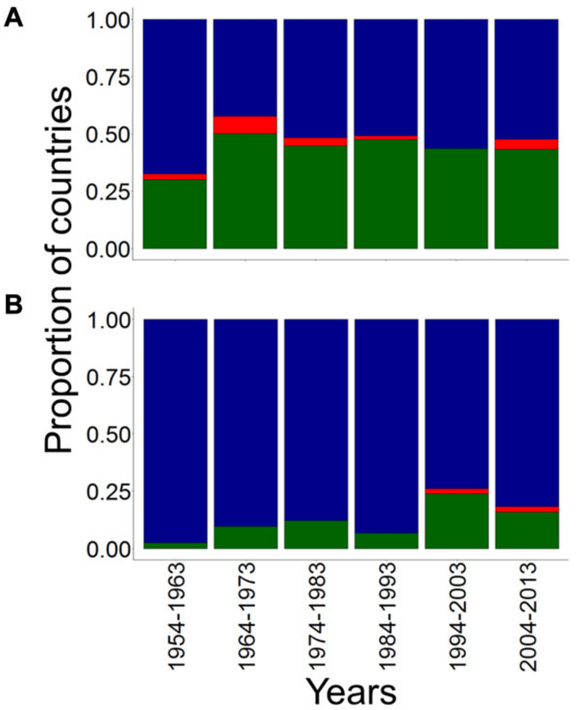

Figure 2 (A) Mann–Kendall trend results for each 10-year time interval showing the proportion of countries with significant increasing (green), significant decreasing (red), and nonsignificant (blue) trends in PE and (B) Spearman's correlation results for each 10-year time interval showing the proportion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

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

Economic, social, and ecological drivers of change in PE

Our complete model set for explaining drivers of change in PE contained 1,024 models, with 19 considered to have good fit to the data (ΔAICc $\leq$ 4; Supplementary Table S6). Model selection revealed annual change in PE has decreased since 1954 (Table 1). Large increases in the amount of area protected had positive effects on the annual change in PE, as well as "hot moments" and the interaction between "hot moments" and the number of designated PAs each year. Countries that are not democratic, that started protecting area relatively later, and that have a greater number of years between periods of area protection (i.e., changes in PE) showed significantly greater increases in annual change in PE.

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

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

Discussion

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

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

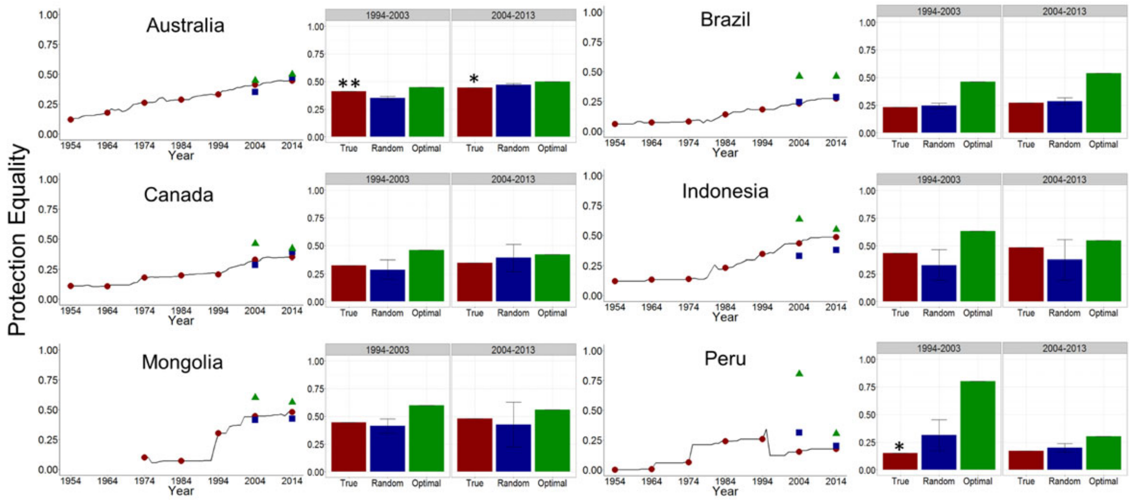

Figure 3 Random and optimal PE results within six simulated countries. Observed ("true") trends in PE from 1954–2013 with simulated optimal (green), random (blue), and "true" (red) PE (±95% CI) values in the last two decades. **An observed PE value significantly greater than random. *An observed PE value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

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

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

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

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

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

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

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

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

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Supporting Information

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

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

Appendix S2. Equations to calculate PE.

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

Appendix S4. Additional data limitations and caveats.

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

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

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

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

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

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

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

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

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

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

References

Barnes, M. (2015). Protect biodiversity, not just area. *Nature*, **526**, 195.

Barr, L.M., Pressey, R.L., Fuller, R.A., Segan, D.B., McDonald-Madden, E. & Possingham, H.P. (2011). A new way to measure the world's protected area coverage. *PLoS ONE*, **6**(9):e24707.

Blomley, T., Flintan, F., Nelson, F. & Roe, D. (2013). *Conservation and land grabbing: part of the problem or part of the* *solution*? Poverty and Conservation Learning Group Symposium. The International Institute for Environment and Development. http://pubs.iied.org/17166IIED.

Brooks, T.M., Mittermeier, R.A., Fonseca, G.A.B., *et al.* (2006). Global biodiversity conservation priorities. *Science*, **313**, 58-61.

Butchart, S.H.M., Clarke, M., Smith, R.J., *et al.* (2015). Shortfalls and solutions for meeting national and global conservation area targets. *Conserv. Lett.*, **8**, 329-337.

Burnham, K.P., Abderon, D.R. (2015). *Model Selection and Multimodel Inference: A Practical Theoretic Approach*. Springer Science and Business Media.

Channan, S., Collins, K. & Emanuel, W.R. (2014). *Global mosaics of the standard MODIS land cover type data*. University of Maryland and the Pacific Northwest National Laboratory, College Park, Maryland.

Chape, S., Spalding, M. & Jenkins, M.D. (2008). *The world's protected areas: status, values and prospects in the 21st century*. University of California Press, Berkley, USA.

Deguignet, M., Juffe-Bignoli, D., Harrison, J., MacSharry, B., Burgess, N. & Kingston, N. (2014). *2014 United Nations list of protected areas. UNEP-WCMC*.

Dirzo, R. & Raven, P.H. (2003). Global state of biodiversity and loss. *Annu. Rev. Environ. Resour.*, **28**, 137-167.

Fairhead, J., Leach, M., Scoones, I., Fairhead, J., Leach, M. & Scoones, I. (2012). Green grabbing: a new appropriation of nature. *J. Peasant Stud.*, **39**, 237-261.

Ferraro, P.J. & Pattanayak, S.K. (2006). Money for nothing? A call for empirical evaluation of biodiversity conservation investments. *PLoS Biol.*, **4**, 482-488.

Friedl, M.A., Sulla-Menashe, D., Tan, B., *et al.* (2010). *MODIS Collection 5 global land cover: algorithm refinements and characterization of new datasets, 2001–2012, Collection 5.1 IGBP Land Cover*. Boston University, Boston, MA, USA.

IUCN & UNEP-WCMC. (2015). *The World Database on Protected Areas (WDPA) [WWW Document]*. Cambridge, UK UNEP-WCMC. URL www.protectedplanet.net

Joppa, L.N. & Pfaff, A. (2009). High and far: biases in the location of protected areas. *PLoS ONE*, **4**, 1-6.

Kendall, B.E., Klein, C.J. & Possingham, H.P. (2015). The role of scale in designing protected area systems to conserve poorly known species. *Ecosphere*, **6**(11):237. http://dx.doi.org/10.1890/ES15-00346.1.

Kulkarni, A. & von Storch, H. (1995). Monte Carlo experiments on the effect of serial correlation on the Mann-Kendall test of trend. *Meteorol. Zeitschrift*, **4**, 82-85.

Laurance, W.F. (2005). When bigger is better: the need for Amazonian mega-reserves. *Trends Ecol. Evol.*, **20**, 643-645.

Di Marco, M., Butchart, S.H.M., Visconti, P., Buchanan, G.M., Ficetola, G.F. & Rondinini, C. (2015). Synergies and trade-offs in achieving global biodiversity targets. *Conserv. Biol.*, **30**, 189-195.

Margules, C.R. & Pressey, R.L. (2000). Systematic conservation planning. *Nature*, **405**, 243-253.

McDonald, R.I. & Boucher, T.M. (2011). Global development and the future of the protected area strategy. *Biol. Conserv.*, **144**, 383-392.

McDonald-Madden, E., Gordon, A., Wintle, B.A., *et al.* (2009). "True" conservation progress. *Science*, **323**, 43–44.

McLeod, A.I. (2011). Kendall: Kendall rank correlation and Mann–Kendall trend test. R package version 2.2. https://CRAN.R-project.org/package=Kendall.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A. & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, **403**, 853-858.

Olson, D.M., Dinerstein, E., Wikramanayake, E.D., *et al.* (2001). Terrestrial ecoregions of the world: a new map of life on earth. *Bioscience*, **51**(11), 933-938.

Possingham, H., Wilson, K.A., Andelman, S.J. & Vynne, C.H. (2006). Protected areas: goals, limitations and designs. Pages 509-533 in M. Groom, G. Meffe, C.R. Carroll, editors. *Principles of Conservation Biology*. Sinauer Associates Inc., Sunderland, MA.

Pressey, R.L. (1994). Ad hoc reservations: forward or backward steps in developing representative reserve systems? *Conserv. Biol.*, **8**, 662-668.

Pressey, R.L. & Logan, V.S. (1998). Size of selection units for future reserves and its influence on actual vs targeted representation of features: a case study in western New South Wales. *Biol. Conserv.*, **85**, 305-319.

R Core Team. (2016). R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Radeloff, V.C., Beaudry, F., Brooks, T.M., *et al.* (2013). Hot moments for biodiversity conservation. *Conserv. Lett.*, **6**, 58-65.

Rodrigues, A.S.L., Akçakaya, H.R., Andelman, S.J., *et al.* (2004). Global gap analysis: priority regions for expanding the global protected-area network. *Bioscience*, **54**, 1092-1100.

Secretariat of the CBD. (2002). Conference of the Parties Decision VI/9. *Glob. Strateg. Plant Conserv.*

Secretariat of the CBD. (2010). Conference of the Parties 10 Decision X/2. *Strateg. Plan Biodivers.*, 1-13.

Tittensor, D.P., Walpole, M., Hill, S.L.L., *et al.* (2014). A mid-term analysis of progress toward international biodiversity targets. *Science*, **346**, 241-244.

Watson, J.E.M., Darling, E.S., Venter, O., *et al.* (2015). Bolder science needed now for protected areas. *Conserv. Biol.*, 4673-4678.

Watson, J.E.M., Dudley, N., Segan, D.B. & Hockings, M. (2014). The performance and potential of protected areas. *Nature*, **515**, 67-73.

Watson, J.E.M., Grantham, H.S., Wilson, K.A. & Possingham, H.P. (2011). Systematic conservation planning: past, present and future, in Conservation Biogeography (eds R. J. Ladle and R. J. Whittaker), John Wiley & Sons, Ltd, Chichester, UK. doi: 10.1002/9781444390001.ch6.

WWF. (2014). *The Living planet Report, 2014*. WWF, Gland, Switzerland.

