Global patterns of forest loss across IUCN categories of protected areas

Leberger, Roxanne; Rosa, Isabel; Guerra, Carlos A.; Wolf, Florian; Pereira, Henrique M.

Biological Conservation

DOI:
10.1016/j.biocon.2019.108299

Published: 01/01/2020

Publisher's PDF, also known as Version of record

ARTICLE IN PRESS

Biological Conservation xxx (xxxx) xxxx

Contents lists available at ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/biocon

Global patterns of forest loss across IUCN categories of protected areas

Roxanne Leberger[a,b,*], Isabel M.D. Rosa[a,b,d], Carlos A. Guerra[a,b], Florian Wolf[a], Henrique M. Pereira[a,b,c]

[a] German Centre for Integrative Biodiversity Research (iDiv), Deutscher Platz 5e, 04103 Leipzig, Germany
[b] Institute of Biology, Martin Luther University Halle-Wittenberg, Halle, Germany
[c] Centro de Investigação em Biodiversidade e Recursos Genéticos (CIBIO), Universidade do Porto, 4485-661, Vairão, Portugal
[d] School of Natural Sciences, Bangor University, LL57 2DG Bangor, UK

ARTICLE INFO

Keywords:
Forest conservation
Forest loss
IPBES subregion
IUCN protected areas
Protected areas audit

ABSTRACT

Forests are under increasing pressure globally and the establishment of protected areas has long been used as a conservation tool to preserve them. Seven categories of protected areas have been defined by the International Union for Conservation of Nature (IUCN) with different management objectives and protection levels. However, recent studies raised questions over whether protected areas are effective in preventing ecosystem degradation and whether IUCN categories vary in their effectiveness. In this study, we analysed forest loss and trends between 2001 and 2014 within IUCN protected areas at a global scale and within sixteen Intergovernmental Platform for Biodiversity and Ecosystem services (IPBES) subregions, relevant for international policy. As habitat protection can be driven by the location of protected areas and as the amount of forest within protected sites is highly unequal, we reported the forest loss integrating the proximity of roads and population, as well as the amount of initial forest in 2000. Our results show that worldwide, the highest protection categories experienced less forest loss than those allowing more human intervention, although this result was reversed in three IPBES subregions. Moreover, in four subregions there was more forest loss within protected areas than outside. We also found accelerating rates of forest loss in protected areas across all IUCN categories, more pronounced in the highest protection IUCN categories. Our results highlight the importance of moving the discussion of the post-2020 biodiversity framework for protected areas beyond simple general areal targets and that areas with poor implementation effectiveness should benefit from additional support.

1. Introduction

Forests provide diverse ecosystem services and play a key role in the conservation of endangered and endemic species (Gibson et al., 2011; Moura et al., 2013), covering one third of the terrestrial land surface (Keenan et al., 2015), and are of prime importance for human well-being. Due to increasing demand for agricultural and forest products coupled by a significant urban sprawl and infrastructure development (Faria and Almeida, 2016; Schmitz et al., 2015), forests worldwide have experienced pressure over the last decades (Laurance et al., 2014). The consequences of forest loss can be substantial when impacting intact forests that are hosting irreplaceable biodiversity and ecosystem services (Gibson et al., 2011; Foley et al., 2005). Forest loss has been shown to be linked to drivers such as changes in population density, international trade and economic development (Leblois et al., 2017; Faria and Almeida, 2016). Even though it is widely recognised that active anthropogenic deforestation has a major impact on forest cover (Margono et al., 2014), other natural or human-related factors such as diseases (Kurz et al., 2008), wildfires (Potapov et al., 2008), or drought events (Peng et al., 2011; Phillips et al., 2009) are also responsible for a significant amount of forest loss. The importance of these drivers, however, varies greatly across regions and so does the extension of forest loss (Sloan and Sayer, 2015), with tropical rainforests experiencing twice as much net loss between 2000 and 2012 than temperate or boreal forests (Leblois et al., 2017).

Despite the persistent decrease of intact forest area during the 2000–2013 period, the contribution of protected areas (PAs) to minimise this loss was significant (Potapov et al., 2017). The establishment of PAs is indeed one of the most common conservation actions to prevent the degradation of forests. Under the Convention of Biological Diversity, countries have established the goal to extend PAs to cover at least 17 % of the terrestrial area by 2020 (Aichi Target 11, https://www.cbd.int/sp/targets/). However, not all PAs are created equal, as there are many types of PAs with different land-tenure regimes and

* Corresponding author at: German Centre for Integrative Biodiversity Research (iDiv), Deutscher Platz 5e, 04103 Leipzig, Germany.
E-mail address: roxanne.leberger@posteo.net (R. Leberger).

https://doi.org/10.1016/j.biocon.2019.108299
Received 18 January 2019; Received in revised form 17 September 2019; Accepted 21 October 2019

Table 1
Protected areas management categories definition from IUCN (https://www.iucn.org).

IUCN category		Description
Ia	Strict Nature Reserve	- Strictly protected: human visitation, use and impacts are strictly controlled and limited to ensure protection of the conservation values - Protect biodiversity and geological/geomorphical features - Can serve as reference areas for scientific research and monitoring
Ib	Wilderness Area	- Unmodified or slightly modified areas without permanent or significant human habitation - Protected and managed so as to preserve their natural condition
II	National Park	- Large natural or near natural areas - Protect large-scale ecological processes, along with the complement of species and ecosystems characteristic of the area - Provide a foundation for environmentally and culturally compatible, spiritual, scientific, educational, recreational, and visitor opportunities
III	Natural Monument or Feature	- Protect a specific natural monument, which can be a landform, sea mount, submarine cavern, geological feature such as a cave or even a living feature such as an ancient grove - Generally quite small protected areas and often have high visitor value
IV	Habitat Species Management Area	- Protect particular species or habitats and management reflects this priority - Can need regular, active interventions to address the requirements of particular species or to maintain habitats
V	Protected Landscape / Seascape	- Interaction of people and nature over time has produced an area of distinct character with significant, ecological, biological, cultural and scenic value - Safeguarding the integrity of this interaction is vital to protecting and sustaining the area and its associated nature conservation and other values
VI	Protected area with sustainable use of natural resources	- Conserve ecosystems and habitats together with associated cultural values and traditional natural resource management systems - Most of the area in a natural condition, where a proportion is under sustainable natural resource management and where low-level non-industrial use of natural resources compatible with nature conservation

different regulations of resource use and allowed activities. The International Union for the Conservation of Nature (IUCN) has proposed an international classification scheme (IUCN/WCMC, 1994) with six categories of PAs, related to their conservation goals, management objectives, and protection levels (Table 1). These range from category I, being described as access restricted areas, to category VI, allowing the 'sustainable use of natural ecosystems'. The effectiveness of the conservation management of some low-protection categories (e.g., IUCN categories IV to VI) has been a source of debate (e.g., Shafer, 2015; Locke and Dearden, 2005).

Although several studies have shown the important role of PAs in preventing forest loss in different parts of the world (e.g., Soares-Filho et al., 2010; Andam et al., 2008; Laurance et al., 2012; Joppa and Pfaff, 2010), there is growing concern on their efficacy in preventing forest loss and other forms of ecosystem degradation (Jones et al., 2018; Allan et al., 2017). Indeed, it has been shown that, in the field, the implementation of PAs does not always guarantee an improvement in conservation management (Watson et al., 2014). For instance, in several PAs located in Asia, vegetation loss within these areas was indistinguishable from unprotected lands (Clark et al., 2013), and in Tanzania, some of these PAs had higher rates of forest loss than unprotected lands (Rosa et al., 2018). Similarly, oil and gas concessions can overlap park boundaries in the Amazon and in sub-Saharan Africa, threatening indigenous lands (Lessmann et al., 2016; Osti et al., 2011; Finer and Orta-Martínez, 2010; Finer et al., 2008). As a result of the increasing global demand for agricultural and forest resources, PAs have been downgraded, downsised or degazetted, facilitating the exploitation of their resources (Mascia and Pailler, 2011; Pedlowski et al., 2005) when they are not eroded by illegal harvesting (e.g. Kuemmerle et al., 2009). Moreover, it has been showed that the habitat integrity of a PA does not solely depend on its protection level, but also on its isolation to human activities, where PAs remoteness generally reduces the probability of deterioration (Nelson and Chomitz, 2011; Joppa and Pfaff, 2009). Therefore, it has been suggested that the effectiveness of a site in protecting its habitat is rather due to its location than any protection measures (Joppa and Pfaff, 2009).

Even though several studies have compared forest loss inside and outside PAs, the incidence and persistence of forest loss over time inside PAs of different IUCN categories has yet to be assessed. Monitoring the temporal trend of loss is important to investigate which categories are under greater risk of future degradation. With major advances in the field of remote sensing over the last couple of years, we are now able to monitor globally and at high-resolution tree cover change on an annual basis (Hansen et al., 2013) and inform about the state of forests globally (http://www.globalforestwatch.org/). Temporally and spatially explicit forest monitoring has the potential to contribute to a more sustainable management and rapid assessment of governmental policy implementation. In this regard, the Intergovernmental Platform for Biodiversity and Ecosystem Services (IPBES) – that was set up to strengthen the link between science and policy - implemented environmental, regional and subregional assessments to investigate the status and trends of worldwide ecosystems. IPBES world regions and subregions were thus defined to help move towards a homogenisation of studies made at a similar regional scale to facilitate the dialogue between scientists and policy makers, contributing to improve management of natural resources (Brooks et al., 2016).

Using a 15-year time series of annual tree cover loss (Hansen et al., 2013), this study aimed to provide the first global assessment of incidence and persistence of forest loss inside the different IUCN categories globally and per IPBES sub-region. We hypothesised that forest loss occurred mainly in the categories with lower protection status (IV-VI). Secondly, we assessed the temporal trend in this loss and which categories are in a trajectory of increasing/decreasing loss. We used proxies of distance to roads and cities as well as the amount of initial forest within each PA as covariates to test their effect in explaining the patterns found. In its essence, this study offers a framework to monitor forest loss inside PAs, tracking temporal and spatial variation, thus highlighting regions of the world and PAs categories where current practices are insufficient to mitigate ecosystem degradation.

2. Methods

2.1. Datasets

To analyse global forest loss, we used the data produced by Hansen et al. (2013) version 1.2, available at 30 m resolution, and covering a 15-year period (from 2000 to 2014). In particular, we downloaded the overall loss dataset which shows the accumulated loss between 2000–2014 (i.e., pixels are classified as 1 where a forested area changed to non-forested and 0 where there was no change); the loss year dataset, which shows the year in which the loss occurred (i.e., pixels are classified from 1 to 14, corresponding to 2001 through 2014, respectively),

and the initial tree cover dataset for the year 2000 used as reference in all our analyses. To convert from tree cover to forest (i.e., binary map, 1 – forest, 0 – non-forest) we assumed a conservative threshold of 20 % tree cover, based on Heino et al. (2015), and a minimum contiguous area of 0.5 ha (FAO definition of forest, Kenneth, 2012). In this study, we focused on analysing gross forest loss only. Forest gain was not included in the analyses as it was not available on an annual basis in this dataset (only gain over the entire time period is provided), and its interpretation is more subjective; e.g., might result from forest plantation (Tropek et al., 2014) as it is the case in many category V PAs (Dudley, 2008).

To investigate forest loss within PAs we downloaded the World Database on Protected Areas (WDPA), which contains their location and associated information, such as IUCN category and year of implementation (IUCN and UNEP-WCMC, 2017). The IUCN categories (Table 1) provide a gradient of naturalness ranging from the most natural (category I) to the least natural condition (category V) with equivalent naturalness levels for the categories Ia and Ib, II and III as well as IV and VI (Ia = Ib > II = III > IV = VI > V, Dudley, 2008). The WDPA database contains several PAs with missing information (e.g., unreported categories, unverified PAs), therefore, we adopted a conservative approach when selecting the PAs to be included in the analysis. We only considered in our analysis terrestrial PAs larger than 1 km^2 that contained forest area in 2000, were created before the year 2000 (thus avoiding confusion between year of loss and eventual newly created areas within our period of analysis), and had their IUCN category reported and verified. The overlapping of those three criteria resulted in the exclusion of a large proportion of the original number of PAs (initial n = 214,807, with 57 % of PAs smaller than 1 km^2, 20 % unreported, and 47 % of areas more recent than the year 2000 or without year of implementation), resulting in a dataset of 15,281 Pa (Fig. 1, Table S1).

To account for PAs accessibility, we used the average of road and population density within each PA and the eight surrounding cells on a 100 km × 100 km grid cell globally. Population density was obtained from the Center for International Earth Science Information Network - CIESIN - Columbia University (2018), considering the year 2015) and accounts for the global distribution (counts) of the human population on a continuous surface at 30 arc-second resolution consistent with national censuses and population registers, normalised by the maximum count of population. We used road density data from Geofabrik (2015) following Ceia-Hasse et al. (2017) methodology.

We performed our analysis globally and for each IPBES subregion. The delimitation of the IPBES subregions was obtained from Brooks et al. (2016). Sixteen of the IPBES subregions included the selected PAs: 1) Caribbean, 2) Central Africa, 3) Central and Western Europe, 4) Central Asia, 5) East Africa and adjacent islands, 6) Eastern Europe, 7) Mesoamerica, 8) North Africa, 9) North America, 10) North-East Asia, 11) Oceania, 12) South America, 13) South Asia, 14) South-East Asia, 15) Southern Africa, and 16) West Africa, excluding Western Asia.

2.2. Determining forest loss

We calculated the proportion of forest loss per individual PA and the temporal trend of forest loss per protected area. The proportion of loss per individual PA was calculated by dividing forest loss that occurred between 2001 and 2014 by the initial forest area in 2000 within the PA. The resulting percentage was then divided by the maximum percentage of forest loss (0 and 1 data to fit the model). To investigate the temporal trends in forest loss, we calculated the annual loss of forest in each PA relative to the initial forest area in 2000. Then, we calculated the slope of a Generalized Least Squares (GLS) model, using the nmle package, with the scaled annual deforestation values as response variable, and the year as the only explanatory variable. The GLS models accounted for temporal autocorrelation in the forest loss data by including the first lag between residuals as the majority of the temporal autocorrelation

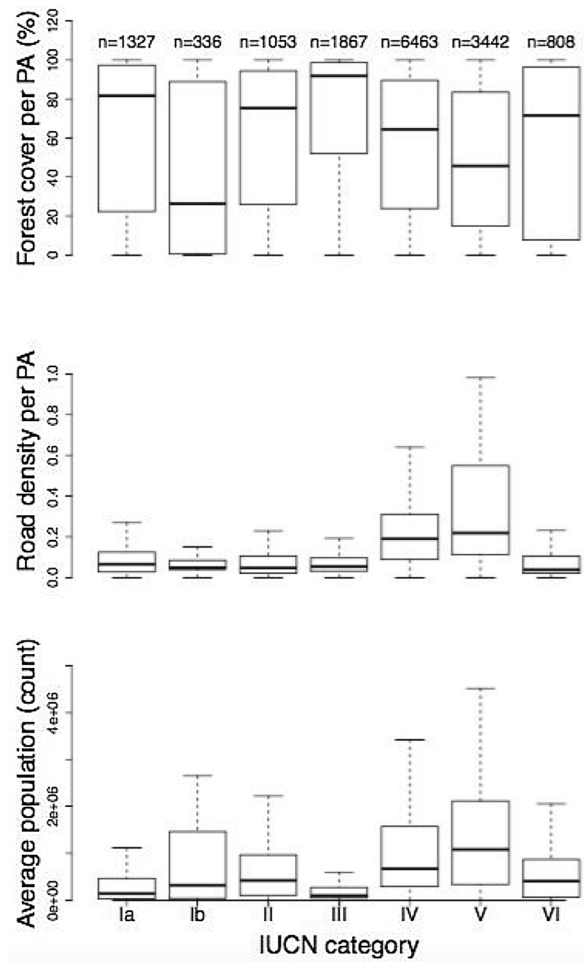

Fig. 1. Percentage of forest cover, road density and average population within the protected areas per IUCN category. The horizontal line shows the median, the top and bottom of the box represent the 25th and 75th percentiles, respectively, and the top and bottom of the bars show the maximum and minimum values. The number of protected areas in each IUCN category is provided by *n*.

Table 2
Median of percentage of forest loss (Loss%) at global scale and for each IPBES subregion, with quantiles in brackets, and ratio of the percentage of forest loss inside PA divided by the percentage of unprotected forest loss relative to the forest area in 2000 (values > 1 mean more loss within protected areas than outside, values close to 0 mean less forest loss within protected areas boundaries compared to outside).

IPBES sub	Loss (%)	Ratio in/out
Global	0.72 [0.12, 0.73] n = 15282	0.63
Caribbean	1.41 [0.26, 4.63] n = 102	1.71
Central Africa	0.31 [0.31, 3.14] n = 21	0.10
Central and Western Europe	0.72 [0.08, 2.64] n = 4499	0.35
Central Asia	0.41 [0.10, 1.80] n = 86	0.41
East Africa	1.94 [1.74, 5.10] n = 194	0.30
Eastern Europe	1.12 [0.23, 3.54] n = 2335	1.17
Mesoamerica	1.76 [0.45, 3.91] n = 224	1.90
North Africa	14.20 [5.38, 87.58] n = 10	2.04
North America	0.38 [0.01, 2.78] n = 1926	0.21
North-East Asia	0.68 [0.15, 1.94] n = 1921	0.57
Oceania	0.24 [0.05, 1.94] n = 1830	0.15
South America	1.13 [0.27, 3.77] n = 888	0.24
South-Asia	0.60 [0.16, 2.36] n = 530	0.65
South-East Asia	1.58 [0.59, 3.87] n = 545	0.36
Southern Africa	3.53 [1.13, 2.36] n = 124	0.31
West Africa	95.72 [8.24, 100] n = 47	0.29

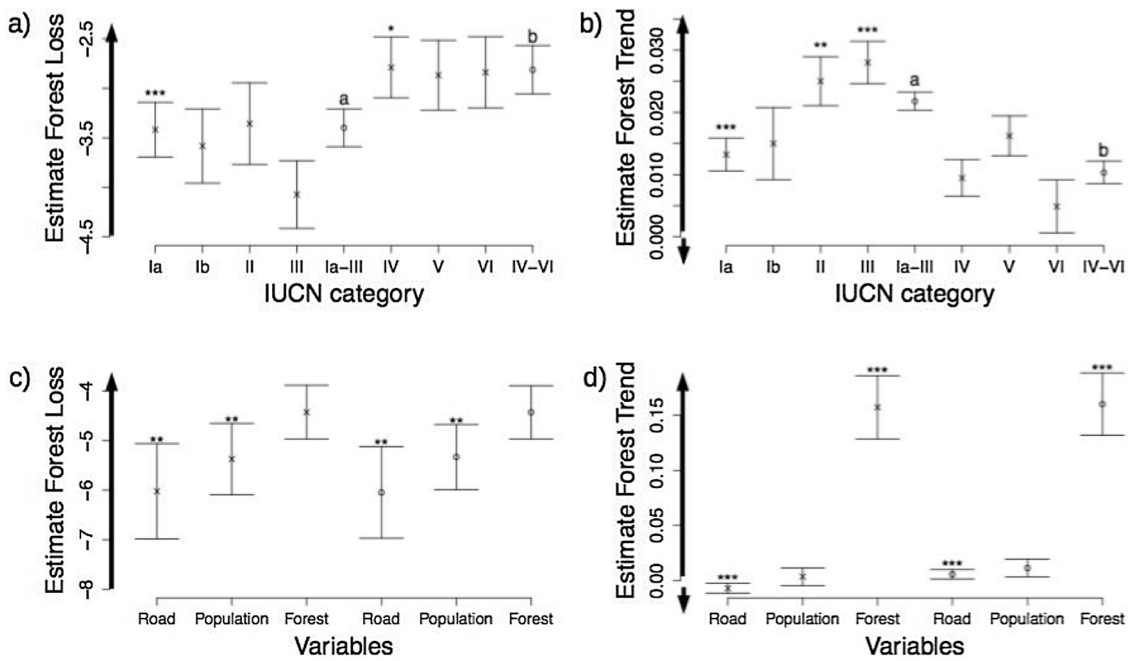

Fig. 2. a) GLM coefficients of the proportion of forest loss per protected area (PA) for each IUCN category between 2000 and 2014 (middle cross). The value of each IUCN category corresponds to the model intercept (corresponding to category Ia) plus the GLM coefficient of each category. The opened dots represent the estimate of the generalized linear model for the merged categories Ia-III and IV-VI. The letters a and b above those two bars show coefficients that are significantly different. b) Trend of forest loss per PA for each IUCN category between 2000 and 2014. Crosses represent the estimate of the linear model for each IUCN category. The opened dots represent the estimate of the linear model for the merged categories Ia-III and IV-VI. c) GLM coefficients of the proportion of road density, average population density and initial forest per protected area (PA) between 2000 and 2014 for the single IUCN category model (middle cross). The value of each variable (road density, population density and initial forest) corresponds to the model intercept (category Ia) plus the GLM coefficient of each variable. d) Trend of forest loss per PA of road density, average population density and initial forest between 2000 and 2014. Crosses represent the estimate of the linear model for the single IUCN category model. The opened dots represent the estimate of the linear model for the merged categories Ia-III and IV-VI. In all plots, the standard error is represented around each estimate and the asterisk above the bar show the significancy of the coefficient relative to the category Ia such as $p < 0.05$*, $p < 0.01$ ** and $p < 0.001$ ***. The black arrow shows the increasing amount of forest loss in a) and c), and the accelerating trend of forest loss when pointing up and decelerating when pointing down in b) and d).

was accounted by a one-year lag.

To assess for differences in the proportion of forest loss between IUCN categories, we fitted a generalized linear model (GLM) with logit function under the binomial family, weighted by the initial forest cover (number of forest cells) in 2000 in each PA. We weighted the different units (PAs) to adjust the standard errors of the model. Differences in the temporal trends in forest loss across IUCN categories were assessed by a linear model using the slopes of the trends produced by the GLS model for each PA. In both cases, predictor variables included the IUCN level, road and population density, as well as the initial forest cover in 2000 for each PA. This last variable was used to account for the effect of forest size that can affect the proportion of forest loss per PA. We used category Ia as a reference, assuming that by definition these will be expected to have the least proportion of forest loss. Therefore, the assessments of the IUCN categories effects on rate of forest loss are relative to category Ia: a negative significant coefficient corresponds to lower deforestation than Ia, while a positive significant coefficient corresponds to higher deforestation than Ia. For visualisation purposes we represented the coefficients as the sum of the intercept plus the coefficient of the respective IUCN category in the GLM. We also performed the same analyses aggregating the categories Ia through III and IV though VI to test whether 'exploitation categories' had higher loss that 'conservation categories' (i.e., Ia-III < IV-VI). Finally, we performed these analyses separately for each IPBES region. In addition, we calculated the ratio of forest loss inside PAs compared to the loss that occurred outside PAs (sum of forest loss in all PAs divided by the sum of forest loss outside all PAs in a subregion) globally and for each IPBES region. All the analyses were performed in R using the packages survey and nmle (R Core Team, 2016). All code and data (with information on specific protected sites) is available at https://github.com/RoxanneLeb/Forest_Loss_IUCN.

3. Results

The 15,281 Pa that matched the selection criteria included 3,216,925 km^2 of forests under protection in 2000 (Table S2). We found that globally, forest cover was unevenly distributed within the IUCN categories. For instance, category Ib had the smallest extent of forest cover within its PAs boundaries with median 26 % forest cover, in comparison with categories Ia and III where most of their PAs were covered by at least 80 % of forest (Fig. 1). Moreover, we found a strong spatial pattern of the distribution of IUCN categories with most of category V located in either Western Europe or North America (44 % and 31 % of category V PAs) (Fig. 3a). Similarly, category IV is mainly distributed across Central and Western Europe, Eastern Europe and North-East Asia (36 %, 27 %, 21 % respectively), category Ia is mostly present in Central and Western Europe and Oceania (55 %), and 50 % of category Ib is in North-America (Fig. 3a). The road network was denser in and around categories IV and V (median of 0.19 and 0.22 respectively), and the average population density was higher in and around categories Ib, II and VI (median Ib = 315,228; II = 417,586; IV = 666,112; V = 1,078,550 and VI = 40,114, Fig. 1).

Globally, areas covered by PAs exhibit less forest loss than areas outside their boundaries (ratio = 0.63, Table 2), and the same in most IPBES subregions. However, four IPBES subregions, had more loss within the PAs than outside: Eastern Europe, the Caribbean, Mesoamerica and North Africa. Moreover, the rate of loss in PA in all those subregions was higher than the global average (median of 1.12 %, 1.41

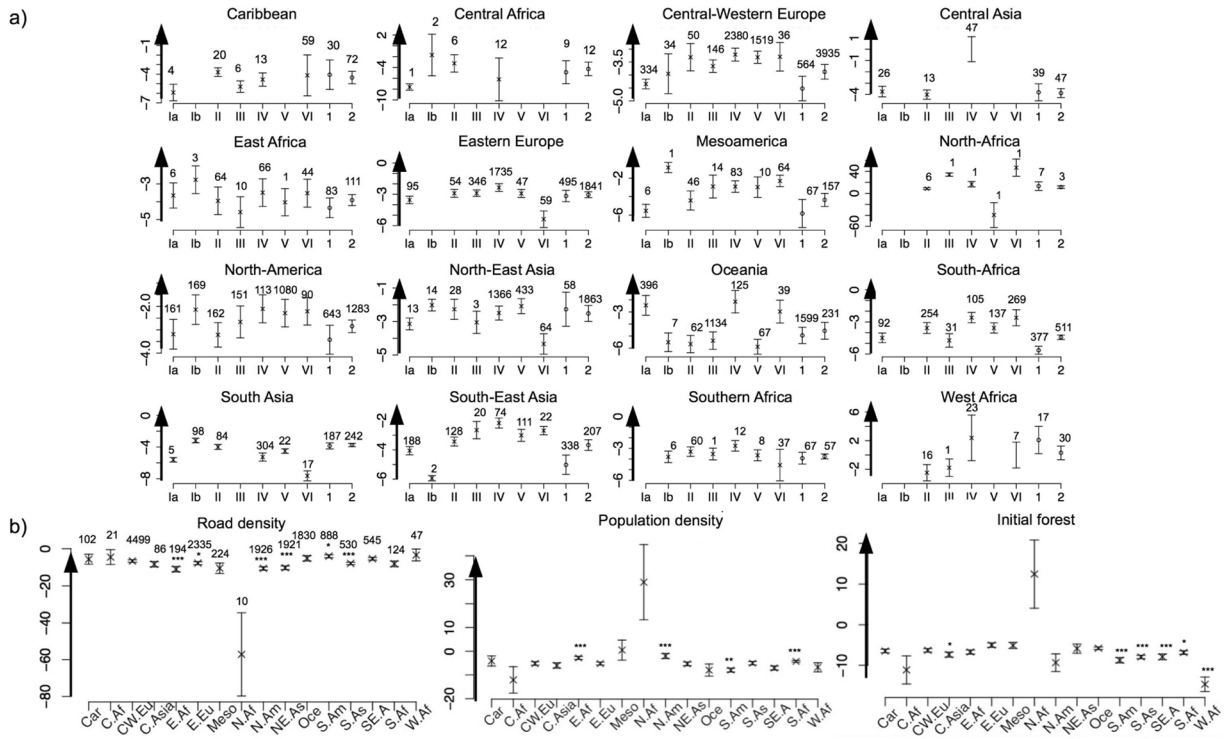

Fig. 3. a) Coefficients of generalized linear model (GLM) of proportion of forest loss for each IUCN category, among the IPBES subregions. The value of each IUCN category corresponds to the model intercept (Ia) plus the estimate of each category, and is represented by the middle cross surrounded by the standard error. The open dots represent the estimate of the linear model for the merged categories 1 (Ia-III) and 2 (IV-VI). The number of protected areas is displayed above. b) Coefficients of generalized linear model (GLM) of road density, population density and initial forest for each IPBES subregion. The value of each variable corresponds either to the model intercept (Ia) of the single IUCN model (middle cross surrounded by the standard error plus the estimate of variable), or to the model intercept (Ia) of the merged IUCN model 1 (Ia-III) and 2 (IV-VI) (middle open dots surrounded by the standard error plus the estimate of variable). The level of significance is indicated above each variable ($p < 0.05$*, $p < 0.01$ ** and $p < 0.001$ ***). The black arrow shows the increasing amount of forest loss.

%, 1.76 % and 14.20 % of forest loss respectively, compared to a global median of 0.72 %, Table 2). From 2000 through 2014, as expected, the forest loss occurring in the PAs belonging to categories I through III was significantly lower than in the categories IV to VI, globally and in most regions, when accounting for the influence of the location and initial forest area in the PA (Fig. 2a). Still, three IPBES subregions presented a reversed pattern, Caribbean, North-East Asia and West Africa (Fig. 3a). The very high proportion of forest loss per PA in West Africa might be inflated by the small size of the forest fragments (Table 2, Fig. 3b).

The global ranking of the effect of the IUCN categories on the forest loss per PA at the global scale, from the least to the most forest loss, was III < Ia = Ib = II < IV = V = VI (Fig. 2a). However, this ranking was not consistent at the IPBES subregions scale (Table S3). For instance, in Eastern Europe and North East Asia few of the Ib-VI IUCN categories presented any differences with the forest loss that occurred in Ia (Fig. 3). This can occur when all PAs categories experience little forest loss or when high forest loss is recorded in all PAs. Indeed, Eastern Europe presented a high percentage of forest loss compared to other IPBES subregions (1.94, Table 2), while North-East Asia presented one of the lowest rates of forest loss (0.68 %, Table 2). Oceania had the smallest percentage of forest loss (0.24 %, Table 2), even though category Ia was one of the most impacted. Central and Western Europe as well as Mesoamerica were the subregions that were the closest to the expected IUCN category ranking, even though the coefficient for category Ib had large variance in Europe and was underrepresented in Mesoamerica. Category Ib presented poor results in preventing forest loss in several subregions, but this effect was confounded by the low initial forest cover of PAs in this category (Fig. 1).

The road and population density have a significant impact on forest loss with the rate of forest loss decreasing in areas where roads network or population are high (Fig. 2b). However, this relationship was not linear with most forest loss occurring in areas with few or no roads or population, and differing across subregions (Fig. S1). This negative pattern with road density was significant in East Africa, Eastern Europe, North America, North-East Asia, South America and South Asia (Fig. 4b). Concerning the average population, the relationship was significantly decreasing with forest loss in East Africa, North America, South America and South Africa.

A visual inspection of the spatial distribution of the rates of forest loss (Fig. 4c) suggests that although high rates of forest loss can occur in all regions, in North America and Western Europe they tend to occur in areas of low initial forest cover (Fig. 4b) while in some developing regions, such as Mesoamerica, South America, Central Africa, Southeast Asia and Eastern Africa, they also occur in areas with high initial forest cover. Regarding the 14-year trend in forest loss, we found accelerating rates of forest loss across all IUCN categories (Fig. 2). Further, PAs from the merged Ia-III categories had higher increases in rates of forest loss than IV-VI PAs, mostly driven by high accelerations in categories II and III (Fig. 2b). Spatially, South-East Asia, South America, East Africa, and Oceania showed accelerating rates of forest loss (Fig. 4d, Tables S3).

4. Discussion

We analysed the status and trends of global forest loss within PAs, considering the difference between IUCN categories and IPBES subregions. Globally, forest loss was lower within PAs boundaries and higher protection categories (I-III) were more effective in preventing forest loss compared to the lower ones (IV-VI). This is congruent with results from previous studies (Jones et al., 2018; Leroux et al., 2010). But our results varied greatly among subregions. For instance, we found that PAs incurred more loss within their boundaries than unprotected forests in Eastern Europe, the Caribbean, Mesoamerica and North Africa

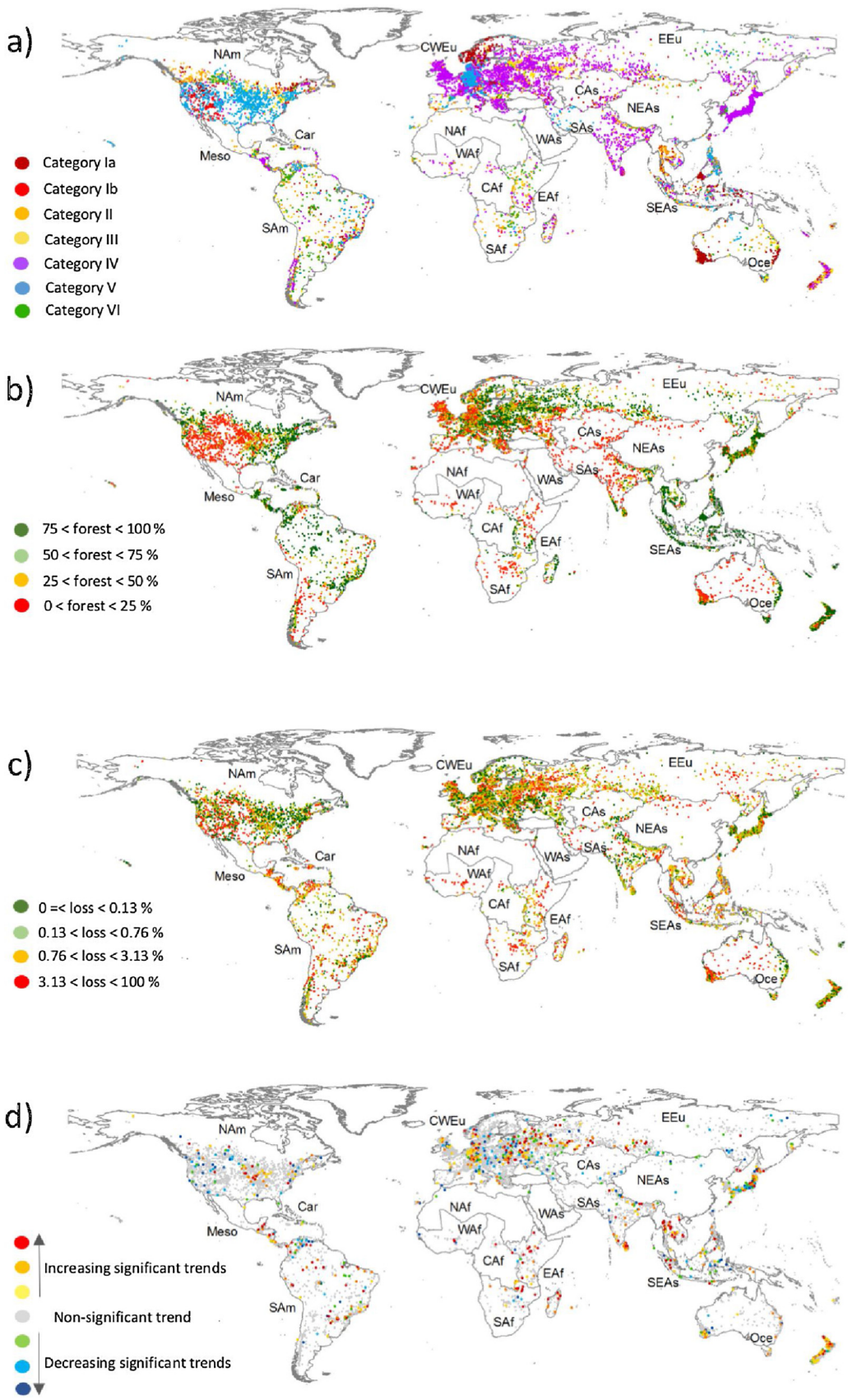

Fig. 4. a) Spatial distribution of the different protected areas included in our analysis, coloured by IUCN category, b) percentage of forest cover per protected area from green highly forested to red poorly forested, c) Percentage of forest loss between 2000 and 2014 within individual protected areas with quantiles from green (low percentage of loss) to red (high percentage of loss, d) Significant trend (increase or decrease) of forest loss between 2000 and 2014 within individual protected areas coloured from yellow to red according to the slope steepness (highest increase in red) and from green to dark blue according to the slope steepness (highest decrease in blue) (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article).

subregions, while categories I-III had lower effectiveness in reducing forest loss than categories IV-VI in the Caribbean, North-East Asia and West Africa.

Previous studies have also found limited effectiveness of PAs in some of these regions. For instance, it has been reported that Eastern Europe PAs can lack efficiency in preventing forest disturbances such as fires or illegal logging (Wendland et al., 2015; Kuemmerle et al., 2009; Achard et al., 2006). Similarly, there have been reports of important forest loss in Jamaica (Caribbean), including in PAs of categories I and II (Chai and Tanner, 2010), due to the rise of yam farming, roads network extension and population density (Newman et al., 2014, 2018). Asia is known for its high rate of forest loss (Hansen et al., 2013) that also occurs within its PAs given the little or absence of protection to forest loss (Spracklen et al., 2015). Further, it has been reported that in North-East Asia, Mongolia suffers from ineffective protection implementation (Tsogtbaatar, 2013; Dorjsuren, 2008). Finally, while forest loss remained quite low in Central Africa with rather efficient protection (Rudel, 2013), we found convergent results with Bowker et al. (2017) regarding a higher forest loss inside than outside PAs in West Africa.

When analysing the protection effectiveness per IUCN category, we found similar patterns to Leroux et al. (2010) who looked at the level of human footprint across categories. They also found that the lowest level of human footprint occurred in category III ('Natural monument or feature'), which is mainly present in Oceania. However, in our study category Ib performed worse than category III concerning forest loss, while they found similar levels of human footprint between Ib and III (in Leroux et al., 2010 the full ranking is Ib = III < Ia < II = VI < IV < V). A difference between our study and Leroux's study is that they look at a single snapshot of land cover/land-use and therefore the effect of the protected area effectiveness is confounded with the effect of protected area placement on landscapes modified prior to the protection. Here, we found that forest loss in PAs is generally highest when they are accessible by a few roads surrounded by the presence of no or small cities. In contrast, few forest losses have been perceived in areas highly occupied by road networks or people. This can tell that PAs surrounded by human activities have either already been exploited by the year 2000 or benefit a better protection than the remote ones from illegal logging.

Although in broad terms, PAs from high IUCN categories (I-III) seemed more effective at protecting against forest loss than lower IUCN categories (IV−VI), within those groups the ranking of effectiveness did not seem to conform with the IUCN definitions (e.g. III < Ia and Ib; no significant differences between IV, V and VI). In addition, the IPBES subregions showed high variation in forest loss among the different IUCN categories, with no consistent pattern. This result suggests that the meaning and the level of deployment of the different IUCN categories may not be consistent across regions and suggests limitations of cross-country comparability of the current IUCN categories.

Worryingly, between 2000 and 2014, the annual rates of forest loss per PA exhibited increasing trends, although there were strong year-to-year fluctuations in the global amount of forest loss. Particularly surprising was that the highest rates of increase occurred in category II and III, although those categories still had low deforestation rates when averaged over the entire period. These results contrast with Geldmann et al. (2014) who found a decrease of human pressure in the categories Ib and III between 1990 and 2010. In addition, our analysis raises concerns about the increasing deforestation rates in PAs in South-East Asia, South America, East Africa, and Oceania which already have higher deforestation rates than the global average across PAs (e.g. Vijay et al., 2016).

Our analysis does not differentiate human-induced from environmentally-induced (e.g. pest or fire) forest loss due to data limitations (Hansen et al., 2013). Therefore, our results should be taken with care. In addition, while IUCN categories are used worldwide their matching to national categories and management practices varies quite widely between countries, i.e. while Ia and Ib are established to have no human interference, in practice that is not the case in different parts of the world (e.g. Osti et al., 2011; Leisher et al., 2013). The WDPA lists the management categories for the PAs as reported by the countries and data providers and no formal verification of the categories is in place (UNEP-WCMC, 2017).

Our results mainly showed that even though at the global scale PAs seem to have been preventing forest loss, we identified regions that have serious issues in protecting their forests, with forest loss occurring in higher proportion within PAs than outside, as well as regions where the strict protection categories were performing worse than the exploitation categories. The underlying causes of those issues should be assessed to implement appropriate solutions. We also highlight the need for regular monitoring of forest loss and change in PAs. Monitoring results could be displayed on online platforms including through web-services, which could be relevant for practitioners. Several projects already in place are developing tools that contribute to this effort (e.g., Digital Observatory of Protected Areas (Joint Research Centre of the European Commission, 2018), Global Forest Watch). In addition, our results call for more research on what renders some PAs more effective than others. Recently, some efforts have been carried out to develop indicators for management effectiveness and equity that can be used to assess progress towards Target 11 for 2020 from the Convention on Biological Diversity (Zafra-Calvo et al., 2019; Leadley et al., 2014). We believe the next step in the context of the post-2020 discussion is to move beyond simple aera targets towards measurable targets on protected area under effective management across a range of categories of naturalness.

Declaration of Competing Interest

We have no conflicts of interest to disclose.

Acknowledgment

Isabel M.D. Rosa has received funding from the European Union's Horizon 2020 research and innovation programme under the Marie Sklodowska-Curie grant agreement No 703862.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.biocon.2019.108299.

References

Achard, F., Mollicone, D., Stibig, H.-J., Aksenov, D., Laestadius, L., Li, Z., Popatov, P., Yaroshenko, A., 2006. Areas of rapid forest-cover change in boreal Eurasia. For. Ecol. Manage. 237, 322–334.

Allan, J.R., Venter, O., Maxwell, S., Bertzky, B., Jones, K., Shi, Y., Watson, J.E.M., 2017. Recent increases in human pressure and forest loss threaten many Natural World Heritage Sites. Biol. Conserv. 206, 47–55.

Andam, K.S., Ferraro, P.J., Pfaff, A., Sanchez-Azofeifa, G.A., Robalino, J.A., 2008. Measuring the effectiveness of protected area networks in reducing deforestation. PNAS 105, 16089–16094.

Bowker, J.N., De Vos, A., Ament, J.M., Cumming, G.S., 2017. Effectiveness of Africa's tropical protected areas for maintaining forest cover. Conserv. Biol. 31, 559–569.

Brooks, T.M., Akçakaya, H.R., Burgess, N.D., Butchart, S.H.M., Hilton-Taylor, C., Hoffmann, M., Juffe-Bignoli, D., Kingston, N., MacSharry, B., Parr, M., et al., 2016. Analysing biodiversity and conservation knowledge products to support regional environmental assessments. Sci. Data 3 sdata20167.

Ceia-Hasse, A., Borda-de-Agua, L., Grilo, C., Pereira, H.M., 2017. Global exposure of carnivores to roads. Glob. Ecol. Biogeogr. 26, 592–600. https://doi.org/10.1111/geb.12564.

Center for International Earth Science Information Network - CIESIN - Columbia University, 2018. Gridded Population of the World, Version 4 (GPWv4): Population Count, Revision 11. NASA Socioeconomic Data and Applications Center (SEDAC), Palisades, NY. https://doi.org/10.7927/H4JW8BX5.

Chai, S.-L., Tanner, E., 2010. Are we losing the best parts of our protected areas in tropical mountains? Biotropica 42, 739–747.

Clark, N.E., Boakes, E.H., McGowan, P.J.K., Mace, G.M., Fuller, R.A., 2013. Protected

Areas in South Asia Have Not Prevented Habitat Loss: A Study Using Historical Models of Land-Use Change. PLoS One 8 (5). https://doi.org/10.1371/journal.pone.0065298. In press.
Dorjsuren, C., 2008. Global Forest Resources Assessment 2010 - Mongolia -. Country Progress Report: Recommendation of Harmonization and Standardization of MAR Terms. Forest Water Reseach Centre (FRC). Ministry of Nature and the Environment of Mongolia (MNE).
Dudley, N., 2008. Guidelines for Applying Protected Area Management Categories (IUCN). 86 p. Available:. IUCN, Gland, Switzerland. www.iucn.org/publications.
Faria, W.R., Almeida, A.N., 2016. Relationship between openness to trade and deforestation: empirical evidence from the Brazilian Amazon. Ecol. Econ. 121, 85–97.
Finer, M., Orta-Martínez, M., 2010. A second hydrocarbon boom threatens the peruvian amazon: trends, projections, and policy implications. Environ. Res. Lett. 5, 014012.
Finer, M., Jenkins, C.N., Pimm, S.L., Keane, B., Ross, C., 2008. Oil and gas projects in the Western Amazon: threats to wilderness, biodiversity, and indigenous peoples. PLoS One 3, e2932.
Foley, J.A., DeFries, R., Asner, G.P., Barford, C., Bonan, G., Carpenter, S.R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., et al., 2005. Global consequences of land use. Science 309, 570–574.
Geldmann, J., Joppa, L.N., Burgess, N.D., 2014. Mapping change in human pressure globally on land and within protected areas. Conserv. Biol. 28, 1604–1616.
Geofabrik, 2015. OpenStreetMap-Shapefiles. Available at:. http://download.geofabrik.de/.
Gibson, L., Lee, T.M., Koh, L.P., Brook, B.W., Gardner, T.A., Barlow, J., Peres, C.A., Bradshaw, C.J.A., Laurance, W.F., Lovejoy, T.E., et al., 2011. Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478, 378–381.
Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R., et al., 2013. High-resolution global maps of 21st-century forest cover change. Science 342, 850–853.
Heino, M., Kummu, M., Makkonen, M., Mulligan, M., Verburg, P.H., Jalava, M., Räsänen, T.A., 2015. Forest loss in protected areas and intact forest landscapes: a global analysis. PLoS One 10, e0138918.
IUCN/WCMC, 1994. Guidelines for Protected Area Management Categories. IUCN Commission on national parks and protected areas with the assistance of the World Conservation Monitoring Centre, Gland, Switzerland.
Joint Research Centre of the European Commission, 2018. The Digital Observatory for Protected Areas (DOPA) Explorer 2.0. Accessed 05/2018, Ispra, Italy. Available at:. http://dopa-explorer.jrc.ec.europa.eu.
Jones, K.R., Venter, O., Fuller, R.A., Allan, J.R., Maxwell, S.L., Negret, P.J., Watson, J.E.M., 2018. One-third of global protected land is under intense human pressure. Science 360, 788–791.
Joppa, L.N., Pfaff, A., 2009. High and far: biases in the location of protected areas. PLoS One 4, e8273.
Joppa, L.N., Pfaff, A., 2010. Global protected area impacts. Proceedings of the Royal Society of London B Biological Sciences rspb20101713.
Keenan, R.J., Reams, G.A., Achard, F., de Freitas, J.V., Grainger, A., Lindquist, E., 2015. Dynamics of global forest area: results from the FAO global forest resources assessment. For. Ecol. Manage. 352 (2015), 9–20.
Kenneth, M., 2012. FRA 2015 Terms and Definitions. Forest Resources Assessment Working Paper 180. Food and Agriculture Organization of the United Nations, Rome 2012.
Kuemmerle, T., Chaskovskyy, O., Knorn, J., Radeloff, V.C., Kruhlov, I., Keeton, W.S., Hostert, P., 2009. Forest cover change and illegal logging in the Ukrainian Carpathians in the transition period from 1988 to 2007. Remote Sens. Environ. 113, 1194–1207.
Kurz, W.A., Dymond, C.C., Stinson, G., Rampley, G.J., Neilson, E.T., Carroll, A.L., Ebata, T., Safranyik, L., 2008. Mountain pine beetle and forest carbon feedback to climate change. Nature 452, 987–990.
Laurance, W.F., Carolina Useche, D., Rendeiro, J., Kalka, M., Bradshaw, C.J.A., Sloan, S.P., Laurance, S.G., Campbell, M., Abernethy, K., Alvarez, P., et al., 2012. Averting biodiversity collapse in tropical forest protected areas. Nature 489, 290–294.
Laurance, W.F., Sayer, J., Cassman, K.G., 2014. Agricultural expansion and its impacts on tropical nature. Trends Ecol. Evol. (Amst.) 29, 107–116.
Leadley, P.W., Krug, C.B., Alkemade, R., Pereira, H.M., Sumaila, U.R., Walpole, M., Marques, A., Newbold, T., Teh, L.S.L., van Kolck, J., Bellard, C., Januchowski-Hartley, S.R., Mumby, P.J., 2014. Progress towards the Aichi Biodiversity Targets. Secretariat of the Convention on Biological Diversity, Montreal, Canada.
Leblois, A., Damette, O., Wolfersberger, J., 2017. What has driven deforestation in developing countries since the 2000s? Evidence from new remote-sensing data. World Dev. 92, 82–102.
Leisher, C., Touval, J., Hess, S.M., Boucher, T.M., Reymondin, L., 2013. Land and forest degradation inside protected areas in latin America. Diversity 5, 779–795.
Leroux, S.J., Krawchuk, M.A., Schmiegelow, F., Cumming, S.G., Lisgo, K., Anderson, L.G., Petkova, M., 2010. Global protected areas and IUCN designations: Do the categories match the conditions? Biol. Conserv. 143, 609–616.
Lessmann, J., Fajardo, J., Muñoz, J., Bonaccorso, E., 2016. Large expansion of oil industry in the Ecuadorian Amazon: biodiversity vulnerability and conservation alternatives. Ecol. Evol. 6, 4997–5012.
Locke, H., Dearden, P., 2005. Rethinking protected area categories and the new paradigm. Environ. Conserv. 32, 1–10.
Margono, B.A., Potapov, P.V., Turubanova, S., Stolle, F., Hansen, M.C., 2014. Primary forest cover loss in Indonesia over 2000-2012. Nature Clim. Change 4, 730–735.
Mascia, M.B., Pailler, S., 2011. Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. Conserv. Lett. 4, 9–20.
Moura, N.G., Lees, A.C., Andretti, C.B., Davis, B.J.W., Solar, R.R.C., Aleixo, A., Barlow, J., Ferreira, J., Gardner, T.A., 2013. Avian biodiversity in multiple-use landscapes of the Brazilian Amazon. Biol. Conserv. 167, 339–348.
Nelson, A., Chomitz, K.M., 2011. Effectiveness of strict vs. Multiple use protected areas in reducing tropical forest fires: a global analysis using matching methods. PLoS One 6 (8), e22722.
Newman, M.E., McLaren, K.P., Wilson, B.S., 2014. Long-term socio-economic and spatial pattern drivers of land cover change in a Caribbean tropical moist forest, the Cockpit Country, Jamaica. Agric. Ecosyst. Environ. 186, 185–200.
Newman, M.E., McLaren, K.P., Wilson, B.S., 2018. Using the forest-transition model and a proximate cause of deforestation to explain long-term forest cover trends in a Caribbean forest. Land Use Policy 71, 395–408.
Osti, M., Coad, L., Fisher, J.B., Bomhard, B., Hutton, J.M., 2011. Oil and gas development in the World Heritage and wider protected area network in sub-Saharan Africa. Biodivers. Conserv. 20, 1863–1877.
Pedlowski, M.A., Matricardi, E.A.T., Skole, D., Cameron, S.R., Chomentowski, W., Fernandes, C., Lisboa, A., 2005. Conservation units: a new deforestation frontier in the Amazonian state of Rondônia, Brazil. Environ. Conserv. 32, 149–155.
Peng, C., Ma, Z., Lei, X., Zhu, Q., Chen, H., Wang, W., Liu, S., Li, W., Fang, X., Zhou, X., 2011. A drought-induced pervasive increase in tree mortality across Canada's boreal forests. Nature Clim. Change 1, 467–471.
Phillips, O.L., Aragão, L.E.O.C., Lewis, S.L., Fisher, J.B., Lloyd, J., López-González, G., Malhi, Y., Monteagudo, A., Peacock, J., Quesada, C.A., et al., 2009. Drought sensitivity of the Amazon rainforest. Science 323, 1344–1347.
Potapov, P., Hansen, M.C., Stehman, S.V., Loveland, T.R., Pittman, K., 2008. Combining MODIS and Landsat imagery to estimate and map boreal forest cover loss. Remote Sens. Environ. 112, 3708–3719.
Potapov, P., Hansen, M.C., Laestadius, L., Turubanova, S., Yaroshenko, A., Thies, C., Smith, W., Zhuravleva, I., Komarova, A., Minnemeyer, S., et al., 2017. The last frontiers of wilderness: tracking loss of intact forest landscapes from 2000 to 2013. Sci. Adv. 3, e1600821.
R Core Team, 2016. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.
Rosa, I.M.D., Rentsch, D., Hopcraft, J.G.C., 2018. Evaluating forest protection strategies: a comparison of land-use systems to preventing forest loss in Tanzania. Sustainability 10 (12), 4476. https://doi.org/10.3390/su10124476.
Rudel, T.K., 2013. The national determinants of deforestation in sub-Saharan Africa. Philos. Trans. Biol. Sci. B 368, 20120405.
Schmitz, C., Kreidenweis, U., Lotze-Campen, H., Popp, A., Krause, M., Dietrich, J.P., Müller, C., 2015. Agricultural trade and tropical deforestation: interactions and related policy options. Reg. Environ. Change 15, 1757–1772.
Shafer, C.L., 2015. Cautionary thoughts on IUCN protected area management categories V–VI. Glob. Ecol. Conserv. 3, 331–348.
Sloan, S., Sayer, J.A., 2015. Forest Resources Assessment of 2015 shows positive global trends but forest loss and degradation persist in poor tropical countries. For. Ecol. Manage. 352, 134–145.
Soares-Filho, B., Moutinho, P., Nepstad, D., Anderson, A., Rodrigues, H., Garcia, R., Dietzsch, L., Merry, F., Bowman, M., Hissa, L., et al., 2010. Role of Brazilian Amazon protected areas in climate change mitigation. Proc. Natl. Acad. Sci. U. S. A. 107, 10821–10826.
Spracklen, B.D., Kalamandeen, M., Galbraith, D., Gloor, E., Spracklen, D.V., 2015. A global analysis of deforestation in moist tropical forest protected areas. PLoS One 10, e0143886.
Tropek, R., Sedláček, O., Beck, J., Keil, P., Musilová, Z., Šímová, I., Storch, D., 2014. Comment on "High-resolution global maps of 21st-century forest cover change.". Science 344, 981.
Tsogtbaatar, J., 2013. Deforestation and reforestation of degraded forestland in Mongolia. In: Yamamura, N., Fujita, N., Maekawa, A. (Eds.), The Mongolian Ecosystem Network Nnvironmental Issues Under Climate and Social Changes, pp. 83–98.
UNEP-WCMC, 2017. World Database on Protected Areas User Manual 1.5. Available at:. UNEP-WCMC, Cambridge, UK. http://wcmc.io/WDPA_Manual.
Vijay, V., Pimm, S.L., Jenkins, C.N., Smith, S.J., 2016. The impacts of oil palm on recent deforestation and biodiversity loss. PLoS One 11 (7). https://doi.org/10.1371/journal.pone.0159668. e0159668.
Watson, J.E.M., Dudley, N., Segan, D.B., Hockings, M., 2014. The performance and potential of protected areas. Nature 515, 67–73.
Wendland, K.J., Baumann, M., Lewis, D.J., Sieber, A., Radeloff, V.C., 2015. Protected area effectiveness in European Russia: a postmatching panel data analysis. Land Econ. 91, 149–168.
Zafra-Calvo, N., Garmendia, E., Pascual, U., Palomo, I., Gross-Camp, N., Brockington, D., Cortes-Vazquez, J.-A., Coolsaet, B., Burgess, N.D., 2019. Progress toward equitably managed protected areas in Aichi Target 11: a Global Survey. BioScience.