Substantial losses in ecoregion intactness highlight urgency of globally coordinated action

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Abstract

Human activities are altering natural areas worldwide. While our ability to map these activities at fine scales is improving, a simplistic binary characterization of habitat and non-habitat with a focus on change in habitat extent has dominated conservation assessments across different spatial scales. Here, we provide a metric that captures both habitat loss, quality and fragmentation effects which, when combined, we call intactness. We identify nine categories of intactness of the world’s terrestrial ecoregions based on changes in intactness across a 16-year period. We found that highly impacted and degraded categories are predominant (74%) and just 6% of ecoregions are on improving trajectories. It is essential that management of degrading processes be targeted in international agendas in order to ensure that Earth’s remaining intact ecosystems are effectively conserved and restored in order to achieve effective conservation outcomes.

KEYWORDS

Aichi targets, conservation, degradation, fragmentation, habitat configuration, habitat loss, habitat quality, wilderness

1 INTRODUCTION

Agricultural, industrial, and urban development, and other human activities, has resulted in a substantial loss of intact habitat globally (Ellis, Klein Goldewijk, Siebert, Lightman, & Ramankutty, 2010; Hoekstra, Boucher, Ricketts, & Roberts, 2005; Hansen et al., 2013). The remaining intact habitat provides a wide range of values and ecosystem services, including carbon sequestration and climate adaptation, pollination, biological pest control, maintenance of soil structure and fertility, nutrient cycling and hydrological services, refuges for imperiled biodiversity, and indigenous cultural practices (O’Bryan et al., 2018; Power, 2010; Watson et al., 2016). The preservation of the remaining intact habitat, and restoration of degraded habitat, is an urgent priority to address the biodiversity crisis, to mitigate climate change impacts, and to achieve sustainable development goals (Betts et al., 2017; Mackey et al., 2015; Watson et al., 2018a). To identify and target priority areas, improved methods to map impacts on habitats are needed.

To date, global assessments of habitat condition have focused on measurements of the change in habitat extent, such as changes in forest cover (Dinerstein et al., 2017; Hansen et al., 2013; Potapov et al., 2017; Watson et al., 2016). However, quantifying habitat as discrete patches occurring within a matrix of non-habitat is overly simplistic, as some human-modified landscapes might provide intermediate levels of utility and permeability that would be more accurately represented using a continuous metric of habitat condition. Anthropogenic actions, such as logging,
urbanization, and agricultural conversion, alter ecosystem characteristics, including physical structure, species composition, diversity, abundance, and functional organization (Haddad et al., 2015; Lindenmayer, Thorn, & Banks, 2017). As a consequence, they erode many of the ecological and evolutionary processes necessary to sustain the capacity of ecosystems for self-regeneration and to overcome different anthropogenic stressors (Thompson, Mackey, McNulty, & Mosseler, 2009; Watson et al., 2018b). Hence, here, “habitat” refers to the combined biotic and abiotic environment and its ability to support communities of species in an ecosystem.

The ability of habitat to support a diversity of species is influenced by the total area, its fragmentation, and quality (Betts et al., 2017; Fischer, & Lindenmayer, 2007; Hanski, Zurita, Bellocq, & Rybicki, 2013; Ovaskainen, & Hanski, 2003; Saura, & Rubio, 2010). We define loss of habitat intactness as the combined impact of habitat loss, fragmentation, and degradation (loss of habitat quality) arising from anthropogenic disturbance within a specified region, such as an ecoregion (Olson et al., 2001). Although multiple measures can be used to describe intactness, we argue that a single, integrated metric that explicitly quantifies the combined effects of these factors will simplify analyses and interpretation, and foster synthesis and application in policy. Here, we provide the first assessment of global ecoregional intactness, and change in intactness (based on data from 1993 and 2009), to quantify more fully the total extent of anthropogenic impacts on ecoregions.

2 | METHODS

We propose an intactness metric that is sensitive to changes in habitat area, quality, and fragmentation of habitat (Figure 1). It is based only on a continuous, grid-based (raster) representation of variable habitat quality, thereby obviating the need for a binary representation of habitat, and can be calculated at large spatial scales (e.g., ecoregions or other jurisdictional boundaries containing millions of cells). Specifically, intactness (Q) is calculated as:

\[
Q = \frac{\sum_{i=1}^{N} \sum_{j=1}^{N} (w_i w_j) z \exp(-\beta d_{ij})}{\sum_{i=1}^{N} \sum_{j=1}^{N} \exp(-\beta d_{ij})},
\]

where \(d_{ij}\) is the distance between cells \(i\) and \(j\) (km), \(w\) is a measure of the quality of the cell in the range 0–1, \(z\) is an exponent that scales the product of two qualities (Figure S1), and \(N\) is the number of cells within a spatial unit such as an ecoregion. The parameter \(\beta\) determines how the combined value of pairs of cells diminishes as a function of the distance between them. The denominator standardizes the metric such that current state is relative to a hypothetical ideal state in which no habitat loss or degradation has occurred (all habitat weights equal one). Without standardization, the metric would vary according to both ecoregion area and shape, thereby diminishing the ability to make comparisons among ecoregions.

The metric was designed and parameterized to meet the following design criteria: (a) to be proportional to habitat area when there is no habitat fragmentation; (b) to decline monotonically as fragmentation increases and to be sensitive to both the number of patches and the separation between patches; and (c) to be proportional to habitat quality for a given total area of habitat and degree of fragmentation. These criteria were achieved with values of \(\beta = 0.2\) and \(z = 0.5\) (Figure 1; see Supplementary Materials for further justification of the parameterization).

Application of this metric requires only a relative measure of habitat quality among cells. As we are interested in ecosystem assessments, we adopt a generic measure of habitat quality to assess global pattern and change in intactness. The human footprint index for 1993 and 2009 (HFI; Venter et al., 2016b) is a validated map of global human pressure (Venter et al., 2016a) and is used here as a proxy for quality. It is a spatially explicit cumulative index of eight key human pressures on natural ecosystems at a 1-km² resolution. These pressures were quantified through both remotely sensed and survey data, which overcomes drawbacks of stand-alone remotely sensed data, incorporating insidious threats, such as access to remote areas that are difficult to map from satellite imagery alone. Ecoregions smaller than 100 km² (e.g., some Pacific islands) were excluded from the analysis because they cannot be characterized at the mapping resolution of HFI.

The HFI captures a wide range of anthropogenic impacts on ecological systems, including effects related to proximity to disturbance sources and transportation networks that facilitate access (Venter et al., 2016b). For example, an HFI value of zero represents areas that are free of significant human impact (Watson et al., 2016), and an HFI value of four is associated with a landscape that is predominantly pasture (Jones et al., 2018). HFI is a predictor of terrestrial mammals extinction risk (Di Marco, Venter, Possingham, & Watson, 2018), restrictions to natural animal movements (Tucker et al., 2018), and behavioral diversity (Kühl et al., 2019). We use an exponential transformation to scale HFI (range [0, 50]) to quality (range [0, 1]) such that quality is 1 when HFI is 0, and quality is 20% when HFI is 4. We also evaluate alternative transformations (quality is 10% or 30% when HFI is 4) to evaluate the sensitivity of the analysis to these assumptions (see Supplementary Materials for further details). For the purpose of calculating a conventional binary habitat loss metric to compare to the intactness metric, we assume that “habitat” corresponds to HFI values \(\leq 4.0\) and “non-habitat” to HFI values > 4.0.

Two spatial metrics can be derived from Equation (1) that provide further options for integrating intactness into management and planning by describing the contribution of each cell to intactness (\(Q'_i\)), and the expected rate of change
Figure 1: Hypothetical 100x100 km landscapes to illustrate the sensitivity of the intactness metric ($Q$ is the number above each landscape) to changes in habitat area, quality, and fragmentation. A landscape filled completely with maximum quality habitat has an intactness score of 100%. The graphs demonstrate that the intactness metric meets the design criteria in that partial contribution as the quality of habitat in the cell increases or decreases ($\delta Q'$). Specifically:

$$Q'_i = \frac{\sum_{j \in M_i} (w_i w_j)^2 \exp(-\beta d_{ij})}{\sum_{j \in M_i} \exp(-\beta d_{ij})}, \quad (2)$$

$$\delta Q'_i = \frac{\delta w \sum_{j \in M_i} \exp(-\beta d_{ij})}{\sum_{j \in M_i} ((w_i + \delta w) w_j)^2 \exp(-\beta d_{ij}) - \sum_{j \in M_i} (w_i w_j)^2 \exp(-\beta d_{ij})}, \quad (3)$$

where $M_i$ is defined as the set of cells falling with a radius of cell $i$ that corresponds to 99.5% quantile of the exponential distribution (here, 26.5 km) and $\delta w$ is a small value (here, 1.0E-9). Hence, $\delta Q'_i$ is the numerical estimate of the slope of the tangent of the curve representing how $Q'$ changes as the quality of the cell $i$ changes. In practice, if $w_i > 1 - \delta w$, $\delta Q'_i$ must be calculated using $w_i - \delta w$ because the quality score can never be larger than 1.

### 3 RESULTS

Globally, many ecoregions evaluated ($n = 792$) were characterized by low or very low intactness values ($Q_{2009} < 0.33, n = 597, 75.4\%$), indicating that the erosion of intactness has been pervasive and severe (Figure 2a). The highest intactness values generally occurred within ecoregions with more remote areas and low human population density, such as northern boreal and tundra regions, arid regions of Africa and Australia, and portions of the Amazon. The ecoregions that lost the most intactness between 1993 and 2009 were within New Guinea, Borneo, the Middle East, central and east Africa, Brazil, and several ecoregions in Asia (Figure 2b). To more clearly understand the dynamics of change, we classified ecoregions into nine categories (Figure 3) based on the trend in their intactness between 1993 and 2009 (degrading, stable, or improving) and the proportion of habitat remaining in the ecoregion that had an HFI value of less than four (poor: <33%, moderate: 33–67%. high: >67%). The majority
of ecoregions were classified as already severely degraded ($n = 432; 54.5\%$) or in the process of degrading (an additional $n = 201; 25.4\%$). Only 54 ecoregions ($6.8\%$) improved in intactness between 1993 and 2009, primarily in higher latitude regions or Europe, North America, and portions of Asia (Figure 3a).

Geographically, the ecoregions in good condition with no degrading trend ($117, 14.8\%$) and the improving ecoregions were often associated with remote areas, such as the boreal forest and tundra regions of northern North America, northern Europe, and Russia; remote, xeric grassland and shrubland regions of Africa, Australia, and central Asia; and a small
(a) Classification of ecoregions according to their current state and change in state from 1993 to 2009 according to their habitat state in 1993 and whether intactness declines, remained stable, or improved between 1993 and 2009 (see inset in (a)). Colors in the map and graph match. Numbers in the inset represent the proportion of ecoregions occurring within each of the nine categories. Labels A and B refer to two ecoregions that illustrate how ecoregions with a similar proportion of total habitat can vary widely in intactness. Maps A and B represent habitat quality, with values 0–0.2 (red to orange) representing what a binary habitat classification would consider non-habitat, and values 0.2–1 (yellow to green) representing continuous variation in habitat quality.

(b)
number of other remote areas, such as portions of the Amazon (Figure 3b). Sites that were already highly degraded included much of Europe, India, eastern China, eastern United States, and portions of SE Asia, south-eastern Africa, southern Mexico, and eastern Brazil. The majority of remaining ecoregions in South America, the United States, Africa and the Middle East, Asia, and eastern Australia appear to be transitioning to highly degraded states (Figure 3b).

The intactness metric indicated that most ecosystems were considerably more degraded than suggested by the proportion of remaining habitat in both 1993 and 2009 (67.9% and 70.4%, respectively; Figure 3a). The intactness metric and habitat loss metric are almost identical in the absence of fragmentation effects, or loss of quality (Figure 1), hence any difference between them arises from the combined effects of these other factors. The widest difference between the habitat loss and intactness metrics, and the widest variation in intactness, occurred at intermediate-level habitat loss (Figure 3a). For example, at approximately 30% habitat lost, intactness scores ranged from approximately 20% to 50% (e.g., ecoregions A and B, Figure 3a). Intactness is more sensitive to loss of quality than to fragmentation (Figure 1), so much of this variation is likely to arise primarily through variation in habitat quality. Ecoregions in which most of the remaining habitat is of poor quality (HFI > 4), corresponding to the approximately 30% of ecoregions in which habitat loss exceeds approximately 90%, intactness scores are often somewhat higher than the habitat loss scores because the former awards a nonzero value to habitat with an HFI score greater than 4.

There was considerable variation in rates of change among ecoregions over the 16-year period (Figure 2b). The biomes associated with the greatest average losses were tropical and subtropical (moist broadleaf forests, grasslands, savannas, and shrublands), montane grassland and shrublands, and mangroves (Table S2, Supplementary Information). Among all ecoregions, there was a mean loss of 3.2% of habitat and 2.0% of intactness over the 16-year period, although there was large variation among ecoregions (Table S2, Supplementary Information). These values must be interpreted in the context of most ecoregions already being in a severely degraded state with limited opportunities for degrading further (Figures 2a and 3a).

The analysis was insensitive to the choice of parameter governing the translation of HFI values to habitat quality scores. The upper and lower estimates of Q for 1993 and 2009 were strongly correlated with the estimate of Q used in the analysis (Pearson’s r > 0.994 for all comparisons). The less optimistic estimate of habitat qualities resulted in a more rapid decline in intactness in relation to habitat area loss compared to the more optimistic interpretation, but this had a minor impact on the categorization of ecoregions into the nine categories (Figure S4, Supplementary Information) and, importantly, does not alter the inferences regarding the value of the metric relative to the habitat loss metric.

Decomposing the intactness metric into the partial contribution to intactness of each cell (Q’; Figure 4a) reveals considerable finer scale variation within most ecoregions reflecting local patterns of anthropogenic impact. The expected rate of change in Q’ as the quality of habitat in the cell increases or decreases (δQ’) is generally low in regions dominated by low intactness, modest in areas dominated by high intactness, and highest in areas with intermediate levels of intactness (Figure 4b).

4 | DISCUSSION

Our new measure of ecoregional intactness indicates that the degree of human alteration of terrestrial ecosystems worldwide is substantially more severe than that suggested by the loss of habitat area alone, particularly at intermediate levels of habitat area (20–80%). Measures that focus on the binary classification of habitat loss (e.g., Hoekstra et al., 2005; Hansen et al., 2013; Watson et al., 2016) fail to capture fragmentation and degradation (loss of habitat quality) effects, which are detrimental to biodiversity persistence (Fischer, & Lindenmayer, 2007; Hanski et al., 2013; Ovaskainen, & Hanski, 2003; Saura, & Rubio, 2010). A focus on habitat area in policy therefore diminishes our ability to achieve effective conservation outcomes by failing to explicitly recognize the importance of these effects and foster management to address them.

Ecoregions with intactness values near zero and little remaining habitat will only improve through the restoration of habitat as there is limited opportunity to improve habitat quality. Over half of ecoregions globally are already in a severely degraded state and restoration must be the principal focus of conservation in these areas. However, in less degraded ecoregions, improvement can occur through a combination of restoration and rehabilitation, with substantial opportunity to improve conservation outcomes by reducing fragmentation and degradation effects. The wide variation in intactness values at intermediate levels of habitat loss (Figure 3a) arises from these effects and illustrates the importance of the spatial distribution of habitat. Because restoration can be both costly and require long-term management, identifying opportunities for conservation action that retain intact ecosystems is a conservation priority.

The intactness metric facilitates comparisons among ecoregions (or other stratifications of space such as management units) and tracking change over time. It is relevant to monitoring and target-setting in policy and could replace habitat area metrics in that regard. The two spatially explicit metrics derived from intactness are relevant to the prioritization of management actions and resource allocation in space. Q’ identifies areas that are most in need of protection in...
Spatially explicit metrics derived from intactness include (a) the contribution of each cell to intactness ($Q'_i$) and (b) the expected rate of change in that partial contribution as the quality of habitat in the cell increases or decreases ($\delta Q'_i$). Potential uses of these derived metrics include the identification of priority areas for protection and prioritization of areas for restoration or rehabilitation. Insets depict $\delta Q'_i$ around São Paulo and Rio de Janeiro, Brazil (A), western Europe (B), and India (C).

**FIGURE 4** Spatially explicit metrics derived from intactness include (a) the contribution of each cell to intactness ($Q'_i$) and (b) the expected rate of change in that partial contribution as the quality of habitat in the cell increases or decreases ($\delta Q'_i$). Potential uses of these derived metrics include the identification of priority areas for protection and prioritization of areas for restoration or rehabilitation. Insets depict $\delta Q'_i$ around São Paulo and Rio de Janeiro, Brazil (A), western Europe (B), and India (C).
order to maintain intactness, and $\delta Q'$ identifies areas where habitat rehabilitation is expected to have the largest impact on intactness (see Supporting Material for further discussion of these metrics). An assumption in this interpretation of $\delta Q'_i$ is that the benefit provided by action within a cell is independent of actions in other cells. When this is not a reasonable assumption (e.g., when numerous cells in one area would be improved), the intactness metric can be used in a scenario analysis context to evaluate the overall change in intactness under a suite of alternative management scenarios.

The intactness metric can be applied at other scales and extents, within other ecological units, and using other estimates of habitat quality. We used HFI to estimate quality because it was consistently mapped at a global scale (this is essential to facilitate comparisons among ecoregions and time periods) and because it incorporates a suite of anthropogenic stressors that are known to impact ecological systems. However, HFI may not reflect all anthropogenic pressures, such as those that are intermittent (e.g., selective harvesting of forests) or have nonlocal effects (e.g., pollution and pathogens). Other applications of the metric may require an estimate of quality that is tailored to specific ecological units (e.g., ecosystems, to inform IUCN Red List of Ecosystems processes), or mapped at a resolution more relevant to local-scale assessments. The intactness metric could also be implemented with species- or ecosystem-specific quality estimates and parameters.

The loss of ecoregional intactness is antithetical to achieving strategic goals outlined in key multilateral environmental agreements, including the Convention for Biological Diversity, the Paris Climate Accord, and the 2030 Sustainable Development Goals. Two of the Aichi targets, for example, refer to degraded habitats and ecosystems: Target 5 states that by 2020, degradation and fragmentation of habitat should be significantly reduced, and Target 15 refers to the need to restore at least 15% of degraded ecosystems (UN CBD, 2010). A key obstacle to operationalizing these agreements is that quantifiable definitions of ecosystem condition are lacking and have not been integrated in the global environment agenda (Mackey et al., 2015; Kormos et al., 2016). The failure of international policy to acknowledge adequately the benefits arising from intact ecosystem processes versus degraded ones also has implications for targeting by international funding programs, such as the Global Environment Facility and Green Climate Fund and Critical Ecosystems Partnership Fund. Improvements in efforts to retain and restore ecological intactness will only occur if international, national, and subnational policies formally recognize the values of ecologically intact areas and specify policies for their retention (Maron, Simmonds, & Watson, 2018) and central to this is adopting ecosystem-based intactness metrics like the one proposed here.

This work proposes a measure of intactness that extends beyond habitat loss, and that would be feasible to incorporate within national and international policy settings, so as to identify and prioritize important opportunities to achieve improvements in land management that are likely to be beneficial to biodiversity. Those ecoregions that are still relatively free from the damaging impacts of large-scale human activities are rapidly disappearing, despite the fact that they provide extraordinary environmental and social values and have the spatial characteristics that make them resilient to climate change than those ecoregions with low intactness scores. Increased awareness of the scale and urgency of this erosion of ecosystem intactness is now critical, as policy setting must target ecosystem intactness and nations must start monitoring and reporting their losses and gains. Because restoration can be both costly and require long-term management, identifying those opportunities for conservation action that retain intact ecosystems now must become a conservation priority.

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DATA AVAILABILITY

Open access to data products provided through The University of Queensland data repository: https://espace.library.uq.edu.au/view/UQ:51cace.

REFERENCES

Betts, M. G., Wolf, C., Ripple, W. J., Phalan, B., Millers, K. A., Duarte, A., … Levi, T. (2017). Global forest loss disproportionately erodes biodiversity in intact landscapes. Nature, 547, 441.

Di Marco, M., Venter, O., Possingham, H. P., & Watson, J. E. M. (2018). Changes in human footprint drive changes in species extinction risk. Nature Communications, 9, 4621.

Dinerstein, E., Olson, D., Joshi, A., Vynne, C., Burgess, N. D., Wikramanayake, E., … Saleem, M. (2017). An ecoregion-based approach to protecting half the terrestrial realm. Bioscience, 67, 534–545.

Ellis, E. C., Klein Goldewijk, K., Siebert, S., Lightman, D., & Ramankutty, N. (2010). Anthropogenic transformation of the biomes, 1700 to 2000. Global Ecology and Biogeography, 19, 589–606.

Fischer, J., & Lindenmayer, D. B. (2007). Landscape modification and habitat fragmentation: A synthesis. Global Ecology and Biogeography, 16, 265–280.

Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., … Townshend, J. R. (2015). Habitat fragmentation and its lasting impact on earths ecosystems. Science Advances, 1, e1500052.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., … Townshend, J. R. G. (2013). High-resolution global maps of 21st-century forest cover change. Science, 342, 850.
Hanski, I., Zurita, G. A., Bellocq, M. I., & Rybicki, J. (2013). Species-fragmented area relationship. *Proceedings of the National Academy of Sciences of the United States of America, 110*, 12715.

Hoekstra, J. M., Boucher, T. M., Rickets, T. H., & Roberts, C. (2005). Confronting a biome crisis: Global disparities of habitat loss and protection. *Ecology Letters, 8*, 23–29.

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.

Kormos, C. F., Bertzky, B., Jaeger, T., Shi, Y., Badman, T., Hilty, J. A., … Watson, J. E. M. (2016). A wilderness approach under the World Heritage Convention. *Conservation Letters, 9*, 228–235.

Kühl, H. S., Boesch, C., Kulik, L., Haas, F., Arandjelovic, M., Dieguez, P., … Kalan, A. K. (2019). Human impact erodes chimpanzee behavioral diversity. *Science, 363*, 1453.

Lindenmayer, D., Thorn, S., & Banks, S. (2017). Please do not disturb ecosystems further. *Nature Ecology & Evolution, 1*, 0031.

Mackey, B., DellaSala, D. A., Kormos, C., Lindenmayer, D., Kumpel, N., Zimmerman, B., … Watson, J. E. M. (2015). Policy options for the world’s primary forests in multilateral environmental agreements. *Conservation Letters, 8*, 139–147.

Maron, M., Simmonds, J. S., & Watson, J. E. M. (2018). Bold nature retention targets are essential for the global environment agenda. *Nature Ecology & Evolution, 2*, 1194–1195.

O’Byran, C. J., Braczkowski, A. R., Beyer, H. L., Carter, N. H., Watson, J. E. M., & McDonald-Madden, E. (2018). The contribution of predators and scavengers to human well-being. *Nature Ecology & Evolution, 2*, 229–236.

Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V. N., Underwood, E. C., … Kassem, K. R. (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*, 933–938.

Ovaskainen, O., & Hanski, I. (2003). How much does an individual habitat fragment contribute to metapopulation dynamics and persistence? *Theoretical Population Biology, 64*, 481–495.

Potapov, P., Hansen, M. C., Laestadius, L., Turubanova, S., Yaroshenko, A., Thies, C., … Esipova, E. (2017). The last frontiers of wilderness: Tracking loss of intact forest landscapes from 2000 to 2013. *Science Advances, 3*, e1600821.

Power, A. G. (2010). Ecosystem services and agriculture: Tradeoffs and synergies. *Philosophical Transactions of the Royal Society B: Biological Sciences, 365*, 2959–2971.

Saura, S., & Rubio, L. (2010). A common currency for the different ways in which patches and links can contribute to habitat availability and connectivity in the landscape. *Ecography, 33*, 523–537.

Thompson, I., Mackey, B., McNulty, S., & Mosseler, A. (2009). Forest resilience, biodiversity, and climate change: A synthesis of the biodiversity/resilience/stability relationship in forest ecosystems. Technical report, Secretariat of the Convention on Biological Diversity, Montreal. Technical Series no. 43. 1–67.

Tucker, M. A., Böhning-Gaese, K., Fagan, W. F., Fryxell, J. M., Van Moorter, B., Alberts, S. C., … Mueller, T. (2018). Moving in the anthropocene: Global reductions in terrestrial mammalian movements. *Science, 359*, 466.

UN CBD. (2010). *United Nations Convention on Biological Diversity Conference of the Parties 10 Decision X/2. Strategic Plan for Biodiversity 2011–2020*. United Nations.

Venter, O., Sanderson, E. W., Magrach, A., Allan, J. R., Beher, J., Jones, K. R., … Watson, J. E. M. (2016a). Global terrestrial human footprint maps for 1993 and 2009. *Scientific Data, 3*, 160067.

Venter, O., Sanderson, E. W., Magrach, A., Allan, J. R., Beher, J., Jones, K. R., … Watson, J. E. M. (2016b). Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. *Nature Communications, 7*, 12558.

Watson, J. E., Venter, O., Lee, J., Jones, K. R., Robinson, J. G., Possingham, H. P., & Allan, J. R. (2018a). Protect the last of the wild. *Nature, 563*, 27–30.

Watson, J. E. M., Evans, T., Venter, O., Williams, B., Tulloch, A., Stewart, C., … Lindenmayer, D. (2018b). The exceptional value of intact forest ecosystems. *Nature Ecology & Evolution, 2*, 599–610.

Watson, J. M., Shannah, D., Di Marco, M., Allan, J., Laurance, W., Sanderson, E., … Venter, O. (2016). Catastrophic declines in wilderness areas underline global environment targets. *Current Biology, 26*, 2929–2934.

**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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