Ecological Economics 228 (2025) 108441

Contents lists available at ScienceDirect

Ecological Economics

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

Analysis

The impact of downgrading protected areas (PAD) on biodiversity

Yufei Li [a], Lingling Hou [b], Pengfei Liu [c,*]

[a] Lab for Low-carbon Intelligent Governance (LLIG), School of Economics, Beihang University, Haidian District, Beijing, China
[b] School of Advanced Agricultural Sciences, Peking University, Beijing, China
[c] Department of Environmental and Natural Resources Economics, University of Rhode Island, Kingston, RI 02881, United States

ARTICLE INFO

JEL:
Q13
Q48
Q51
Q57

Keywords:
Downgrading protected areas
Biodiversity
Difference in differences
Abundance changes
Environmental conservation

ABSTRACT

We quantitatively assess the impacts of Downgrading Protected Areas (PAD) on biodiversity in the U.S. Results show that PAD events significantly reduce biodiversity. The proximity to PAD events decreases the biodiversity (abundance) by 26.0 % within 50 km compared with records of species further away from the PAD events. We observe an overall 32.3 % decrease in abundance after those nearest PAD events are enacted. Abundance declines more in organisms living in contact with water and non-mammals. Species abundance is more sensitive to the negative impacts in areas where the decisions of PAD events were later reversed, as well as in areas close to protected areas belonging to the International Union for Conservation of Nature (IUCN) category. The enacted PAD events between the period 1903 to 2018 in the U.S. led to economic losses of approximately $689.95 million due to the decrease in abundance. Our results contribute to the understanding on the impact of environmental interventions such as PAD events on broadscale biodiversity change and provide important implications on biodiversity conservation policies.

1. Introduction

Complex land use planning is a critical aspect of addressing sustainability challenges, such as global population expansion (Li et al., 2022) and biodiversity loss (Engist et al., 2023; Ureta et al., 2022). Protected areas (PAs) are geographical regions that have been specifically identified, declared, devoted, and administered by law or other regulations to preserve nature including the ecosystem services and cultural values it supports. Protected area downgrading, downsizing, and degazettement (PADDD) events represent legal modifications that weaken, shrink, or eliminate PAs, which can accelerate forest loss, fragmentation, and carbon emissions. Human activities within a protected area may also significantly affect biodiversity outcomes (see Fig. 1). Human intervention stemming from shift in policy direction has led to adverse impacts and losses to nature and human systems. Growing demand for infrastructure, subsistence, industrial agriculture, minerals, and political pressure impose threats to PAs and the areas around PAs, which transform once-protected landscapes, threatening their biodiversity conservation values and associated ecosystem services (Siqueira-Gay et al., 2022). While many studies have focused on the impacts of multiple types of land conservation expansion on biodiversity (Gray et al., 2016; Kehoe et al., 2017; Li et al., 2022), there is no systematic research to quantify the threat of PAD events on biodiversity. This study empirically evaluates the impacts of PAD events on biodiversity with nationally representative, micro-level data in the U.S.

Broad legal changes often undermine the viability and effectiveness of protected areas (Golden Kroner et al., 2019). Growing economic demand for minerals and other resources, as well as political pressure for related infrastructure such as road accessibility, creates new threats to biodiversity in protected areas. PAs are acknowledged in the Convention on Biological Diversity and the 2030 Agenda for Sustainable Development and are regarded as a pillar of biodiversity conservation (Maxwell et al., 2020). The long-term goal of area-based conservation needs to be supported by policy changes. PAD events may cause a variety of risks for local ecosystems, climate, and human society. Clear, transparent tracking of PAD events will ensure we correctly address current shortfalls in area-based conservation to contribute to biodiversity conservation-related policies.

Despite requests to speed up the creation of protected areas to safeguard biodiversity, some governments have begun to pull down legal protections (Wilson, 2017). High PAs coverage does not guarantee the conservation of biodiversity (Gardner et al., 2023). PADDD can reallocate under-performing protected areas (PAs), reducing PAs in regions that have limited development potential (such as remote areas, or

* Corresponding author.
E-mail addresses: yufeili@buaa.edu.cn (Y. Li), llhou.ccap@pku.edu.cn (L. Hou), pengfei_liu@uri.edu (P. Liu).

https://doi.org/10.1016/j.ecolecon.2024.108441
Received 24 January 2024; Received in revised form 11 August 2024; Accepted 22 October 2024
Available online 31 October 2024
0921-8009/

those with steep slopes or high elevation) (Runting et al., 2015), while developing the technologies and infrastructure required for renewable energy production (Sonter et al., 2020). The first contemporary protected areas, Yellowstone and Yosemite National Parks, are located in the U.S., which has long been an example of global conservation. However, the U.S. government enacted at least 220 PADDD events (including 211 PAD events) between 1905 and 2018 in 195 terrestrial PAs in 46 states, repealing protections for a total of 22,879.32 km^2. The first PADDD event occurred in Yosemite National Park when, in 1905, Yosemite was reduced in size by 30 % to allow for forestry and mining. The cause of the majority of U.S. PADDD events ($n = 186$) was a 2016 National Park Service regulation enabling Native American tribes to harvest plants for customary subsistence purposes if the action will have "no significant ecological impact" (36 Code of Federal Regulations (C.F.R.), 2016). Also, 33 PADDD events were connected to industrial-scale resource exploitation and development, including the downgrading of eight national forests to make room for the expansion of ski infrastructure in 1986.

The U.S. government has proposed more than 700 PADDD events that, if enacted, would affect hundreds of thousands of square kilometers of protected areas. Since 2000, 90 % of U.S. PADDD proposals have been introduced by the government, and industrial-scale development has been involved in 99 % of the proposals (Golden Kroner et al., 2019). The recent PADDD events highlighted the increasingly uncertain future of protected areas and biodiversity in the U.S. After 114 unsuccessful requests over 30 years, oil and gas drilling in the Arctic National Wildlife Refuge was authorized by the US Congress in 2017 (115th U.S. Congress, 2017). Some additional national monuments that favor biodiversity conservation have been downgraded by the U.S. government (U.S. Department of the Interior (DOI), 2017). Given that PAD events constitute the majority of PADDD events and are more frequently located near areas with available biodiversity data in the continental U.S., our analysis focuses on PAD events.

We combine the BioTIME and PADDD tracker data to depict high-resolution distributions of organisms and PAD events across the U.S. We then use a generalized difference-in-differences (DID) model to identify the impacts of PAD events on biodiversity. We show that PAD events have significant negative effects on biodiversity. The proximity to PAD events affects abundance negatively and decreases the abundance by 26.0 % within 50 km. We also observe an overall 32.3 % decrease in abundance after the nearest PAD event is enacted. Abundance declines more in organisms living in contact with water and non-mammals. Species abundance is more sensitive to the negative impacts in areas where PAD events were later reversed, as well as in areas close to protected areas belonging to the International Union for Conservation of Nature (IUCN) category. Our results offer valuable insights and benchmark statistics for policymakers to balance biodiversity protection and PAD in the U.S. and other countries.

This study aims to deepen the understanding of biodiversity change, crucial for informing future biodiversity conservation policies. Biodiversity changes that undermine the resilience of natural ecosystems and their ability to persist in an ecologically fractured world present significant challenges (Gotelli et al., 2017). Utilizing a comprehensive collection of biodiversity data over a long time horizon, previous studies have explored the catalytic effects of landscape-scale forest loss on biodiversity change (Daskalova et al., 2024) and temperature-related biodiversity change (Antão et al., 2020). Whereas PAD events may disrupt the systematic distribution of biodiversity in adjacent regions, thus impairing the local ecological health, there is a dearth of research directly utilizing protected area data to assess these impacts. Past study has shown that protected area downsizing may exacerbate habitat fragmentation (Golden Kroner et al., 2016) and is a key contributor to biodiversity loss globally. The current literature has not systematically examined the effects of PAD events on biodiversity loss, primarily focusing on the scale of PAD events and their habitat implications (Cook et al., 2017; Thieme et al., 2020)'. Our paper fills this gap by directly

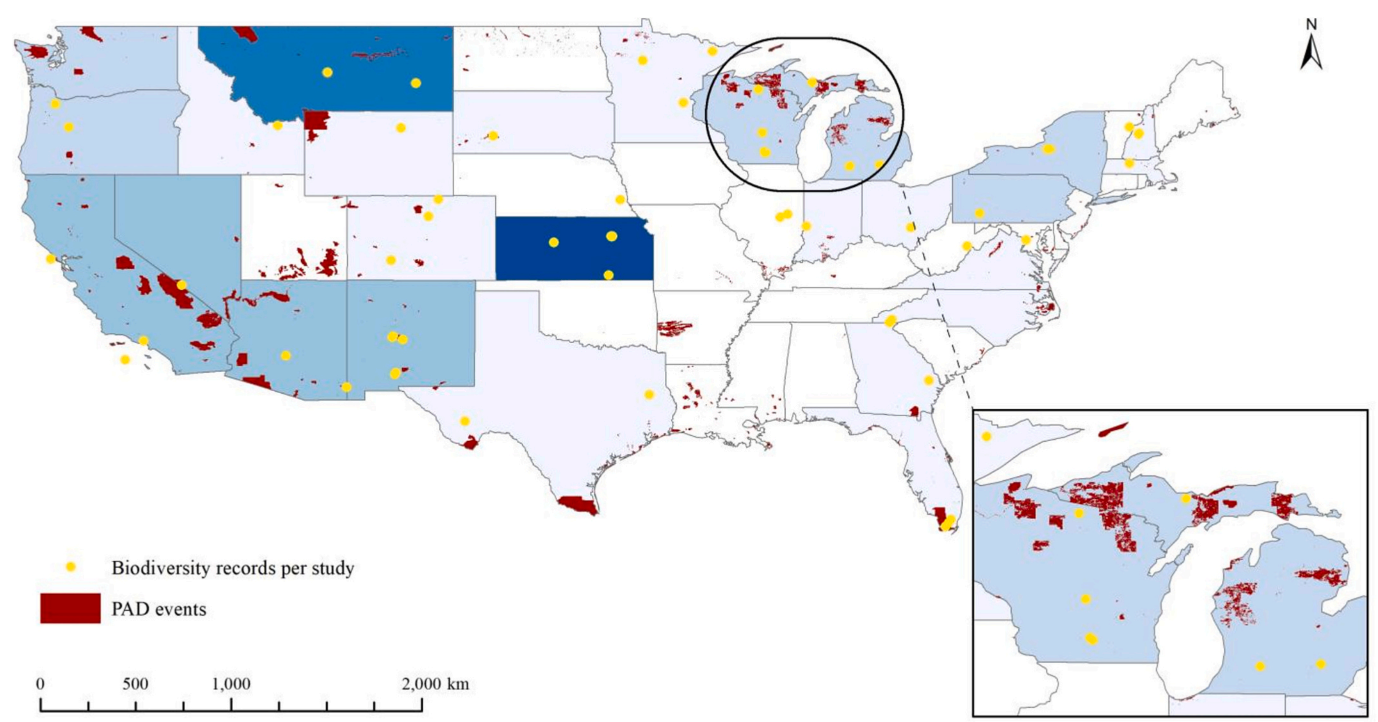

Fig. 1. Spatial distribution of sample PAD events and biodiversity in the continental U.S. from 1903 to 2018. Dots indicate biodiversity records per study in BioTIME, which holds millions of records of species counts at the species-location (latitude and longitude)-year level at more than 10,000 different locations. Red shaded areas show enacted PAD events. The blue background represents the states with biodiversity records affected by adjacent PAD events, with darker blue colors indicating a larger number of specific biodiversity records. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

linking PAD events to changes in biodiversity abundance over a long-time horizon, which is pivotal for understanding the broader role of PAD events and their disruptive effects on biodiversity. The conservation performance can be significantly affected by management practices (Duckworth and Altwegg, 2018). Our contribution lies in providing a quantitative analysis of the impact of PAD events and exploring the role of management practices in conservation effectiveness. Quantifying how differences in management after PAD events influence performance will considerably increase our general understanding of the role played by PAD events, and the disturbance they cause to biodiversity. This not only offers novel insights into the field of ecological economics but also lays the groundwork for more effective biodiversity conservation strategies.

2. Literature review

PAs play a pivotal role in conserving biodiversity by restricting human activities within designated regions, a strategy underscored by early global assessments of their coverage and effectiveness (Chape et al., 2005). The importance of intact ecosystems for conservation has been recognized in various assessments, such as those highlighting the value of primary forests for biodiversity and ecosystem services (Gibbs et al., 2010). However, the phenomenon of PAD, characterized by the reduction or removal of legal protections, poses a significant threat to these conservation efforts (Golden Kroner et al., 2019). PAD has been identified as a global issue with various forms and implications, necessitating a systematic econometric analysis to complement descriptive accounts (Mascia and Pailler, 2011; Symes et al., 2016).

PAD's prevalence and its drivers, such as economic development pressures and political changes, have been examined across different geographical contexts (Mascia et al., 2014). The complex interplay between conservation and development requires a nuanced understanding of PAD drivers (Kareiva and Marvier, 2012). PAD is typically associated with reduced management effectiveness and increased human activities, which can lead to habitat loss, species decline, and ecosystem degradation (Strassburg et al., 2020; Watson et al., 2018). Following PAD, previously protected habitats may undergo fragmentation and conversion to alternative uses, causing significant declines in species richness and diversity, particularly among species with specialized habitat requirements (Cardinale et al., 2012; DeFries et al., 2004; Geldmann et al., 2014; Radeloff et al., 2005). PAD also impacts the provision of critical ecosystem services, such as water regulation and carbon sequestration (Bruner et al., 2001; Laurance et al., 2012).

Empirical studies have begun to quantify the impact of PAD on biodiversity, with case studies examining fragmentation effects within specific national parks (Golden Kroner et al., 2016) and broader analyses of PAD's extent in Australia (Cook et al., 2017). Methodological approaches, such as pre-post comparisons and matched analyses, have been employed to assess biodiversity conservation policy impacts, albeit with limitations regarding data consistency and selection bias (Ferraro and Hanauer, 2014; Ferraro and Pattanayak, 2006). The economic dimensions of diversity within ecosystems are increasingly recognized, with discussions on the economic value of biodiversity in the context of ecosystem services (Bartkowski, 2017) and reviews on the broader economic impacts of biodiversity loss (Hanley and Perrings, 2019).

Despite the growing recognition of biodiversity loss, PAD policies have received significantly less attention compared to PAs policies. Moreover, there is a notable lack of research that integrates PAD policies with biodiversity loss to explore their direct economic consequences and extends the characteristic analysis, especially lacking comprehensive, quantitative assessments of PAD's impacts at the national level and a deeper understanding of its economic and social dimensions. This study addresses these gaps by employing rigorous quantitative methods to evaluate PAD's impact on biodiversity in the U.S., offering a broader perspective and incorporating economic assessments of direct consequences. By doing so, this research contributes to the literature by providing a more comprehensive understanding of the ecological and economic impacts of PAD, which is essential for informed policy-making and conservation strategy development.

3. Data and methods

3.1. Data

We obtained biodiversity data from BioTIME (BioTIME— Global database of biodiversity time series (st-andrews.ac.uk)), the largest database of assemblage time series currently available. BioTIME contains hundreds of academic ecological research studies measuring the abundance (count or biomass) of pertinent species in a given area over time, spanning several decades (Blowes et al., 2019; Dornelas et al., 2018a). BioTIME is composed of species abundance records for assemblages that have been sampled through time using a consistent methodology. Different population size declines are given equal weight. For example, the weights are the same when population size declines from 80 to 75 and from 10 to 5. The dataset includes 381 separate studies (study ID plus additional data sources) that covers a variety of taxonomic groups, including plants, invertebrates, birds, mammals, and fish. As a result, the BioTIME data is useful for investigating the relationship between economic development and biodiversity based on the rich set of features. For our analysis, we used studies mainly from the continental US terrestrial system between 1903 and 2018 (Adler et al., 2007; Lafferty et al., 2013; Webb and Scanga, 2001). Although there are few freshwater and marine studies, we include these two categories in our analysis of biodiversity trends across taxa and geographic regions. The website Padddtracker.org provided detailed information on PAD events, which chronicled legal changes for PAD events, collated available data on a worldwide scale using records that had already been published as well as those that had not, and updated versions created by the World Wildlife Fund and Conservation International. The dataset includes information about the latitude-longitude location of all known PAD events.

Our analysis focuses on downgrading Protected Areas (PAD) events as almost no protected area downsizing events or degazettement events occurred in the U.S. We mapped all the PADDD and biodiversity information on a global scale using the BioTIME and PADDDtracker data. Then we adjusted our dataset to focus on PAD events located within the continental U.S. (district geometries freely available from http://www.gadm.org/, see Fig. 1). To implement this adjustment, a GIS shapefile was imported for the continental U.S. and we matched PAD events that are located within the continental U.S. Information about the biodiversity of protected areas at each PAD event across the continental U.S. was determined using ArcGIS by linking the biodiversity data to the closest PAD events. The distance between biodiversity data and the PAD event is also calculated. When PAD events take place prior to the recording of biodiversity data, the maximum distance between the records and the PAD event is approximately 130 km. Furthermore, Stata is used to merge biodiversity data and information on PAD events with the same unique identifier associated with each biodiversity record. As a result, all the PAD events closest to the biodiversity data are included in our sample. Our data do not include biodiversity records without PAD events or PAs beyond a certain spatial distance.

To compare regions with different climate types and to control for environmental conditions, we added average annual minimum temperature, extremely high temperature (highest daily maximum temperature for the year), and total annual precipitation as covariables. The meteorological data were retrieved from the Global Summary of the Year dataset provided by the National Oceanic and Atmospheric Administration (NOAA)/National Centers for Environmental Information, computed using the Global Historical Climatology Network (GHCN)-Daily Data set. We merged meteorological control variables with the main data set by using latitude-longitude location. A list of summary statistics for the PAD events and related variables is provided in Supplementary Table 1.

Note that the minimum value of the dependents variable before log transformation is positive, or 0.0009 in the Supplementary Table 1, which is consistent with the data provider's clarification that the dataset had been rigorously checked for the presence of duplicates within each study and across the entire database, species with zero abundance, and non-organismal records, all of which were removed (Dornelas et al., 2018b). We also observe the presence of near-zero abundance data in the dependent variable, as shown in Supplementary Table 1, which accurately represents the existence of rare abundances. Hence, the presence of zero values in the log-transformed dependent variable is unlikely to introduce bias in the estimation and the untransformed, original dependent variables all have positive values.

3.2. Empirical framework

A two-way fixed effects model enables us to take advantage of the panel data structure. The two-way fixed effects model is specified in Eq. (1) below:

$$\text{Log}Y_{it} = \beta_0 + \beta_1 post_{it} + \beta_2 post_{it} * distance_{it} + \gamma X_{it} + \mu_t + \theta_p + \delta_s + \epsilon, \tag{1}$$

where Y_{it} is the abundance of species in sample i at year t. The natural logarithm form of the abundance is used as the explanatory variable. The species records are from assemblages consistently sampled for a minimum of 2 years and pool abundance of different life stages, sizes, or sex. About 81 % of the samples were observed for at least ten consecutive years. Additionally, we use the biomass of species in the sample as different dependent variables expressing biodiversity. $post_{it}$ is a time dummy variable to indicate whether the abundance record occurs before or after the closest PAD event, and $post_{it}$ takes 1 if the abundance record occurs after the completion, and 0 otherwise. $distance_{it}$ represents the distance between records of abundance of species and the nearest PAD event. To better control for the differences in properties between the treatment and control groups, a series of meteorological control variables X_{it} are added, including annual minimum temperature, extremely high temperature, and precipitation. Area fixed effects θ_p account for PAD-specific time-invariant factors; study fixed effects δ_s control for the perturbation of the abundance of species by unobservable factors that do not change over time; yearly fixed effects μ_t control for time-invariant characteristics. ϵ is an idiosyncratic error term.

Spatial proximity to PAD events could make a significant difference, and area-fixed effects can capture these structural differences so that we can focus on the comparison of biodiversity near similar PAD events and avoid comparing biological habitats of different characteristics. The binned model uncovers a distanced-based relationship between PAD and biodiversity, which is specified in Eq. (2),

$$\text{Log}Y_{it} = \beta_0 + \beta_1 post_{it} + \beta_{2,b} post_{itb} * vicinity_{itb} + \gamma X_{it} + \mu_t + \theta_p + \delta_s + \epsilon. \tag{2}$$

The variable $vicinity_{itb}$ indicates the proximity of the abundance record to the closest PAD event, with $vicinity_{itb}$ equals 1 if an abundance record is close enough to a PAD event within a distance bin b, and 0 otherwise. We applied 5 distance bins based on an increment of 20–40 km, with the distance higher than 110 km included in the last bin. $\beta_{2,b}$ is the coefficient for the interaction term between $vicinity_{itb}$ and $post_{it}$, and represents the impact difference between the treatment and control groups on biodiversity. $\beta_{2,b}$ identifies the treatment effect changes as the vicinity increases after the PAD.

We also run a quasi-experimental DID analysis to provide more evidence on the choice of the cutoff distance. This analysis confirms that overall significant changes happen within 110 km. Let $LogY_{it}$ be the natural logarithm of the abundance of species in sample i at year t. The treatment group is defined as those that are close to PAD event enough (i.e., closer than 110 km) to be affected. The dummy variable $treat_i$ is equal to 1 if abundance record i belongs to the treatment group (i.e., is located surrounding PAD event less than 110 km), and equal to 0 if it belongs to the control group (i.e., outside 110 km). Let $post_{it}$ takes the value of 1 if the abundance record occurs after the completion, and 0 otherwise. The DID model can be written as Eq. (3):

$$\text{Log}Y_{it} = \beta_0 + \beta_1 post_{it} + \beta_2 treat_i + \pi post_{it} * treat_i + \gamma X_{it} + \mu_t + \theta_p + \delta_s + \epsilon \tag{3}$$

where π represents the treatment effect of the PAD events on the biodiversity changes by comparing the differences between the treatment and control groups before and after the PAD events.

3.3. Event study

To test the plausibility of the parallel trend assumption between the records of abundance with proximate PAD events and those without, we conducted an event study analysis. The event study model is specified as follows:

$$\text{Log}Y_{it} = \beta_0 + \sum_{j=-10}^{j=10} \pi_j [Treat_i \times I(t - T_i = j)] + \gamma X_{it} + \mu_t + \theta_p + \delta_s + \epsilon \tag{4}$$

where $Treat_i$ is a dummy variable that takes the value of one if abundance record i belongs to the treatment group and takes the value of zero otherwise belongs to the control group, which is the same as in Eq. (3). T_i is the specific year t when abundance record i enacted a PAD event. $I(\cdot)$ is an indicator that equals one when $(t - T_i = j)$ and zero otherwise. The baseline, omitted case, is the two years before the PAD event was enacted ($j = -2$ to 0). All other variables carry the same definitions as in Eq. (1). The coefficients π_j measure the effects of PAD events on biodiversity in the relative year j, compared to that in two years before the PAD event was enacted ($j = -2$ to 0). If π_{-10} to π_{-3} are not statistically significant, the abundance in the two groups is statistically indifferent before the PAD events, suggesting the plausibility of the parallel trend assumption.

4. Results

4.1. Abundance changes vary with distance

This study applies a two-way fixed effects model to estimate the relationship between PAD events and biodiversity changes. Results between abundance and the spatial distance of proximate PAD events are in Supplementary Table 2. We find that abundance decreases after the nearby PAD events are enacted. Specifically, PAD events decrease the abundance of organisms by 29.7 %. As the distance between records of abundance and PAD events increases, the effect of PAD events on abundance decreases. The loss of biodiversity threatens ecological stability and also has direct economic repercussions, such as increased costs for ecosystem restoration and loss of natural capital (Costanza et al., 1997). This relationship between spatial proximity and abundance underscores the potential costs associated with the degradation of ecosystem services and the loss of biodiversity. To test for the robustness of our result, we replace the dependent variables with the abundance measurement without taking the natural log and the natural logarithm of the biomass of species in the sample. We find that the results are similar and the detailed regression coefficients replacing the dependent variables are displayed in Supplementary Tables 3 and 4.

In our model, the treatment group consists of abundance records with proximate PAD events within a certain range, while the control group includes those without proximate PAD events. A binned model that connects the proximate PAD events to abundance uncovers a more detailed relationship between PAD and biodiversity. We use 5 distance bins with an increment of 20–40 km to form a balanced distribution of the number of studies in each distance bin. We find that the proximity to the PAD event has decreased the abundance of biodiversity. The magnitudes of the abundance vary across different distance bins. Fig. 2a plots the abundance changes caused by PAD events after controlling for

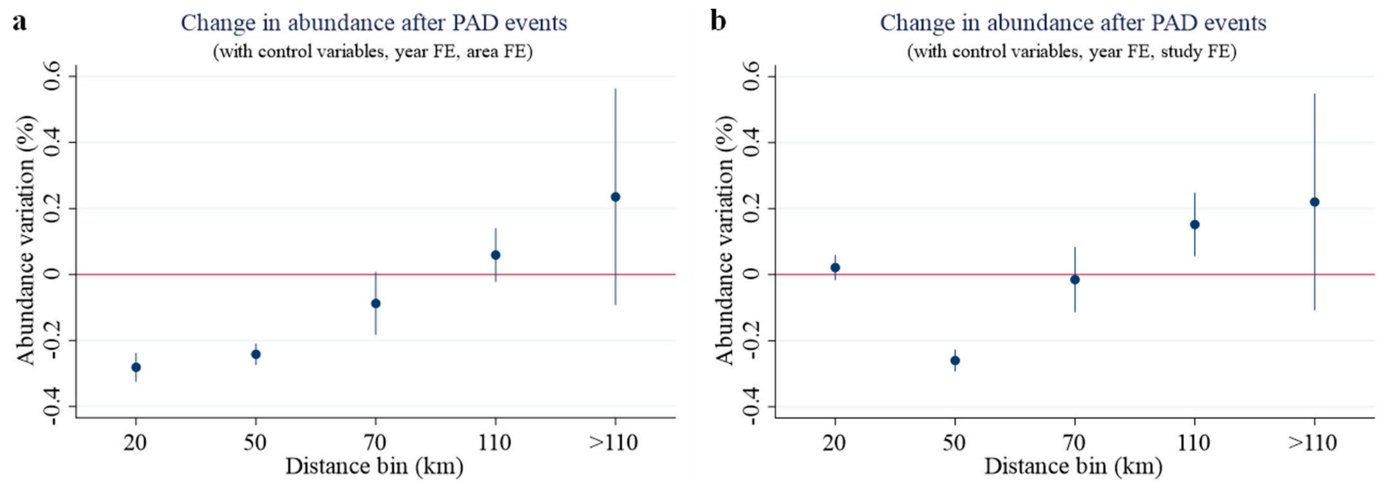

Fig. 2. Impacts of vicinity to PAD events on abundance. a, Impacts with meteorological variables, year, and area fixed effects. b, Impacts with meteorological variables, year, and study fixed effects. The centers of the error bars are the values of the coefficients, which represent point estimates from the regressions and indicate the average effects of the PAD events. Their 95 % confidence intervals are plotted vertically. The dependent variable is the log of species abundance. Each panel is from one regression. For both regressions, the total number of observations is over 1.9 million each.

meteorological variables, area, and yearly fixed effects. Results indicate a 28.1 %, 24.2 %, and 8.8 % reduction in the biodiversity abundance within 0–20 km, 20–50 km, and 50–70 km, respectively. We observe that the negative impacts on the biodiversity abundance diminished with increasing distance from 0 to 70 km and the effect becomes insignificant beyond 70 km. Fig. 2b indicates an abundance reduction of up to 26.0 % when meteorological variables, yearly, and study fixed effects are included. The estimated coefficient becomes slightly smaller in magnitude when meteorological control variables are excluded (Supplementary Table 5). The smallest abundance reduction in magnitude is 21.7 % for proximate PAD events within 20–50 km. In Fig. 2b, we find the negative impact on biodiversity abundance ranges from 20 to 50 km. The impact becomes insignificant after the 50 km. Our results suggest that biodiversity is negatively affected by the proximity to PAD events within a 50 km buffer in general. This degradation of biodiversity may impose direct costs on local economies that rely on healthy ecosystems for tourism, agriculture, and fisheries (Pimentel et al., 2005). Specially, the reduction in biodiversity can lead to a decrease in ecosystem services, such as pollination, water purification, and carbon sequestration, which are critical to local economies. For instance, the loss of pollinator species could directly impact agricultural productivity and increase the costs for farmers who may need to rely on artificial pollination methods (Kremen et al., 2008). We also use the abundance measurement without taking the natural log as the dependent variable. The detailed regression coefficients replacing the dependent variables are displayed in Supplementary Table 6. Our main results remain unchanged.

4.2. *Effects of PAD on nearby abundance*

In the quasi-experimental DID model, we first determine the distance that separates the abundance data into control and treatment groups. Alternatively, matching method is also widely used to control for confounding effects and systematic bias (Schleicher et al., 2020). In our context, we followed the standard methodology in the literature (Haninger et al., 2017; Richardson et al., 2022)˒ and used the least local linear polynomial estimators to determine the distance from a PAD event at which the PAD event stops having a significant impact on the nearby abundance. This cutoff distance was obtained from the intersection of the lower and upper confidence intervals of the residuals of the natural log of abundance versus the distance from the nearest PAD event that was formed before the nearest PAD event and the abundance that was formed after the nearest PAD event (Supplementary Fig. 1). The residuals of the natural log of abundance were determined using an OLS model predicting the natural log of abundance versus the meteorological characteristics, study, and yearly fixed effects. Based on Supplementary Fig. 1, the distance that separates the treatment and control groups is 110 km, suggesting the potential impact range of PAD events on biodiversity.

The results of the DID models are presented in Table 1. Model (1) includes area, and yearly fixed effects, as well as the set of control variables such as temperature and precipitations. Model (2) excludes the meteorological control variables from Model (1) and replaces the area fixed effects with the study fixed effects. Model (3) retains the same specifications as Model (2) with the inclusion of the control variables. Our focus is the coefficient associated with $post_{it}*treat_{it}$. We find that the estimates are statistically significant in Models (1) and (2) at a 1 % level and a 10 % level, respectively. In Model (3), we find that the $post_{it}*treat_{it}$ is still negative and statistically significant at a level close to 5 % (p = 0.054). Model (1) leads to a larger treatment effect estimate. Our preferred specification in Model (3) suggests an overall 32.3 % decrease in abundance after the nearest PAD event is enacted. We also detect a significant negative impact when we replace the dependent variables with the natural logarithm of the biomass of species in the sample to measure biodiversity based on the coefficient of $post_{it}*treat_{it}$ in Supplementary Table 7. The decline in abundance signals a potential weakening of ecosystem resilience, which is critical for maintaining the

Table 1
DID estimation results on abundance.

Variables	(1)	(2)	(3)
Post×treat	−0.525***	−0.287*	−0.323*
	(0.167)	(0.168)	(0.168)
Post	0.230	0.207	0.221
	(0.167)	(0.167)	(0.167)
Treat	−0.661***		
	(0.008)		
Constant	2.007***	1.213***	1.400***
	(0.021)	(0.002)	(0.020)
Control Variables	YES		YES
Yearly FE	YES	YES	YES
Area FE	YES		
Study FE		YES	YES
N	1,902,298	1,902,629	1,902,298
R^2	0.541	0.563	0.563

Notes: The dependent variable is the log of abundance. * $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$. Standard errors are in the parentheses.

stability and health of ecological systems (Lenton et al., 2008). Compromised resilience may increase the susceptibility of ecosystems to environmental perturbations, such as climate variability and invasive species, leading to a heightened risk of ecological tipping points with irreversible economic and ecological costs (Naidoo and Ricketts, 2006). The long-term economic repercussions are particularly concerning, as they can manifest in the form of forgone opportunities for sustainable economic development based on natural capital. This underscores the urgency for policymakers to consider the economic externalities of biodiversity loss when formulating environmental policies and to invest in the conservation and restoration of ecosystems to ensure their capacity to provide essential services over the long term.

We conduct additional robustness checks to enhance the credibility of our main results. The analysis based on the event study model supports the plausibility of the parallel trend assumption. In the event study, we obtain the coefficients before and after the PAD events (Fig. 3). We find that before the PAD events, the effects of PAD events are not statistically different from zero, which is consistent with the parallel trend assumption. After the PAD events, they start to show negative impacts although the magnitude fluctuates, which supports our main results by ruling out the potential influences of differential trends. Another check implemented was a falsifcation check model in which we shifted the year of PAD events, resulting in a change of the post value. We created an artificial year by moving the year of each PAD event K years backward, i.e. if an PAD event was enacted in 2008, and K = 5, we moved its year to 2003. We then updated our post and post × treat variables to refect this move. This robustness check acts as a placebo, showing that the change in abundance is directly linked to the PAD events. The falsifcation test results in Supplementary Table 8 indicate that as we use the artificial year away from the true year, we see a loss of significance, consistent with our main results.

4.3. Heterogenous analyses based on species and PAD characteristics

The abundance changes due to PAD events are heterogeneous across species and PAD characteristics. To investigate potential mechanisms of the PAD event impact, we included additional interaction terms where the $post_{it}*treat_{it}$ is interacted with the species and PAD characteristic variables in Fig. 4 and Supplementary Table 8. We examine how abundance changes over several key species characteristics, including the realm of site, habitat, biome as listed on the WWF (World Wildlife Foundation) site, and taxa. Our results reveal that biodiversity loss near water is more severe in response to PAD events, based on the comparison of organisms in marine and freshwater with those in terrestrial. The aquatic ecosystem is vital for industries such as fisheries and tourism, and its degradation can lead to substantial economic losses through reduced productivity and diminished ecosystem service values. We also observe positive significant effects on abundance changes caused by organisms in the terrestrial realm, habitats such as grasslands and forests, and biome-like shrublands. The negative impacts on abundance are mostly attributed to organisms living in contact with water. PAD events decrease the abundance of organisms living in contact with water by over 70 %, but increase the abundance of terrestrial organisms by around 30 %. The latter live in prime habitat for rich biodiversity (Gardner et al., 2023). PAD events reduce restrictions on overfishing, pollution, invasive species, underwater noise, development activities, climate change, ocean acidification, and others, which have a greater impact on the living environment of marine and freshwater life. We also find PAD events have a negative impact on the abundance in non-mammals while a positive impact on those in mammals. Specifically, enacting PAD events significantly reduces abundance in non-mammals by 47.2 %.

We also examine how abundance variations change if the PAD event is later reversed and when the species belongs to the IUCN category. We find that PAD events have significantly negative impacts on the abundance of organisms close to reversed PAD events and the abundance of organisms near PA belonging to the IUCN category. Enacting PAD events significantly reduces abundance by 36.3 % in areas where PAD events were later reversed as well as in areas that were in the IUCN category before PAD. Possible explanations include policy reversal reduces

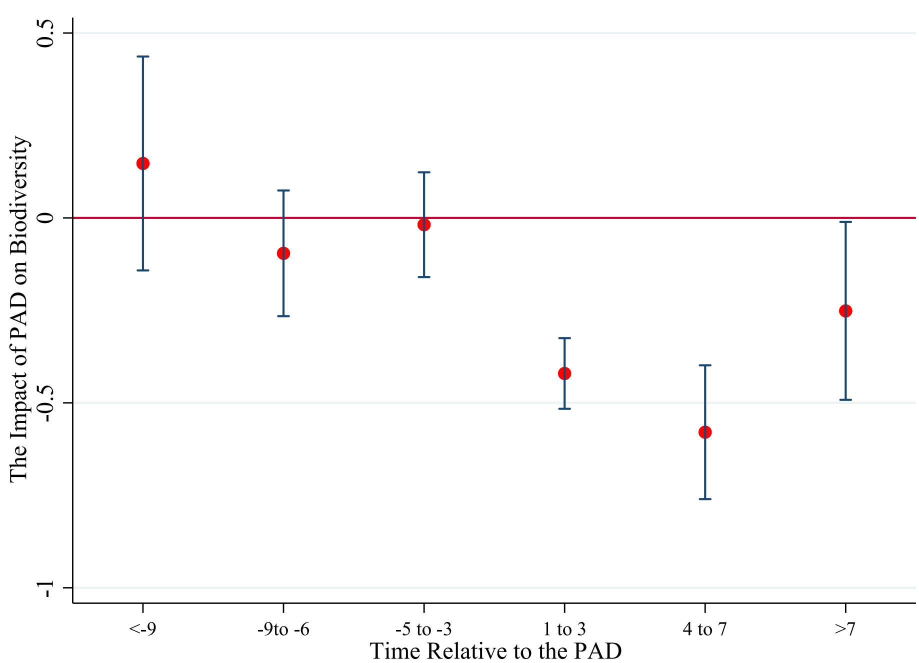

Fig. 3. Abundance changes based on the dynamic event study model. This figure provides the abundance changes for biodiversity relative to the year of the PAD event. The red solid line represents the effect of the year of the PAD event. The horizontal axis is normalized relative to the year of the treatment and the excluded period is $t = -2$ to 0. The blue whiskers indicate the 95 % confidence intervals of point estimates that show average effects. The year fixed effects, area fixed effects, and study fixed effects are included. The number of observations is 207,869. We have dropped the observations before $t = -10$ and after $t = 10$. The effects do not seem to be lasting and fade away after around 7 years. But there is no evidence that biodiversity (or the abundance of certain new species) makes a comeback after seven years. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

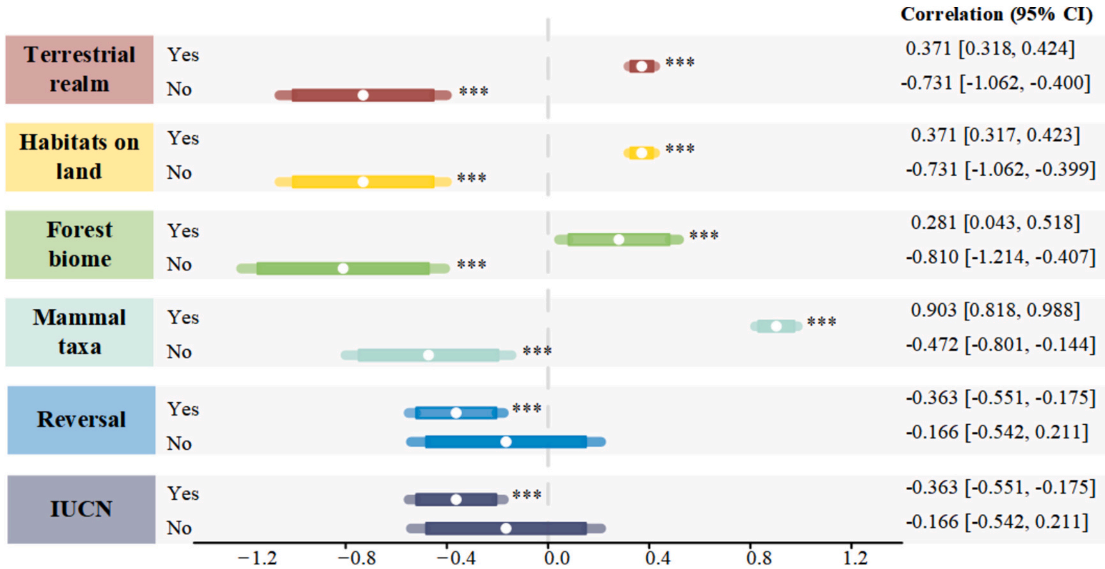

Fig. 4. Heterogenous effects across species and PAD characteristics. Species characteristics include the realm of site (organisms in terrestrial and marine/freshwater), habitat (areas on land like forest/desert/grassland and areas near the water like lakes/ponds/streams), biome as listed on the WWF site (organisms far from water like shrublands/forest/grasslands and organisms living in contact with water like lake ecosystems/river ecosystems/shelf ecoregions), as well as taxa (mammal and non-mammal). As for PAD characteristics, reversal refers to a dummy of whether the species lives in areas where PAD events were later reversed, while IUCN is a dummy of whether the species is in areas that were listed in the IUCN category before PAD events. Estimates of the impact of the PAD events on the abundance are depicted by the white dots. The outer (thin) error bar and inner (thick) error bar for each estimate, respectively, indicate the 95 % and 90 % confidence intervals. The gray dashed line is at an estimated effect of zero. * $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$.

management effectiveness in the short term, the IUCN management category cannot inhibit the aggravation of human pressure caused by PAD events (Li et al., 2022). The areas that receive more attention from the government are vulnerable to events that trigger biodiversity loss. Policy inconsistency can lead to short-term economic dislocation and ecological instability (Victor et al., 2005).

The difference in the conservation importance of species is reflected in the linkage of conservation priorities. We analyze how changes in abundance varied across different IUCN categories before PAD events. In Supplementary Table 9, we find that areas that evolved from national parks and IUCN categories of unknown or unassigned areas to PAD events have no significant effects on nearby abundance after PAD events. While there are significant reductions in abundance at 30.1 %, 44.7 %, and 35.2 %, respectively, after PAD in areas where IUCN categories of PAs before PAD were classified as habitat/species management areas, protected landscapes/seascapes, and protected areas for sustainable use of natural resources. The negative impacts suggest areas that meet the needs of specific species or habitats, or that have unique values for human-nature interactions, are important environmental amenities capable of nurturing ecological benefits that are closely linked to the biodiversity around PAD in these areas. There is a need for a balanced approach to environmental policy that considers both the immediate and long-term economic effects of PAD events. Policymakers should consider the economic benefits of biodiversity conservation, such as the sustained provision of ecosystem services, alongside the costs of ecological degradation and the potential for economic displacement.

4.4. *Direct economic loss assessment by a back-to-envelope model*

Biodiversity generates both direct and indirect economic value in terms of how changes in biodiversity affect human well-being (Bartkowski, 2017). Direct economic value is derived from production and consumption or interaction with environmental resources and services (Kassar and Lasserre, 2004). The willingness to pay (WTP) for conserving a particular species measures the direct economic value of biodiversity (Hanley and Perrings, 2019). The indirect economic value of biodiversity relates to the indirect support for the ecosystem's stability and survival and protection provided to economic activity and property by the ecosystem's natural functions, or regulatory environmental services (Moran and Bann, 2000), which is challenging to quantify (Naeem et al., 2016). In this study, we focus on the direct economic loss of biodiversity and collect the WTP data from the literature (Moran and Bann, 2000). Using a back-of-the-envelope estimation, we calculate the annual direct economic loss in biodiversity caused by the closest PAD event using

$$\text{Loss} = (\pi \times 100\%) \times \overline{\text{WTP}} \tag{7}$$

where *Loss* is the mean economic loss per resident per year due to a change in abundance caused by the nearest PAD event. π is the estimated treatment effect of the nearest PAD event on abundance based on Eq. (3) by a DID estimation. $\overline{WTP}$ is the mean willingness to pay for biodiversity per resident per year, which ranges from \$0 to \$6.39 (Lundhede et al., 2014) and is equivalent to a \$2.06 loss in abundance per resident per year from the nearest PAD event. The national losses in abundance thus add up to $\$2.06 \times 334,282,669 = \$689.95$ million, where 334,282,669 is the population in the U.S. in 2022.

5. Discussion

We find that PAD events have a profound and far-reaching impact on biodiversity, affecting a wide range of species and ecosystems. The proximity to PAD events has a negative significant impact on biodiversity, leading to a 26.0% decrease in abundance within a 50 km radius. Furthermore, we observe an overall 32.3 % decrease in abundance after the nearest PAD event is enacted. The negative impacts on biodiversity are primarily observed among organisms that live in close contact with water, non-mammals, organisms near reversed PAD events, and organisms located within protected areas (PA) belonging to the IUCN category. In this section, we discuss potential causes and consequences of PAD events' significant reduction in abundance, as observed in our micro-level data.

As mentioned in the introduction, PAD can be done to non-performing PAs. While protected areas aim to safeguard regions with a high species diversity and concentrations of protected species (Kremen et al., 2008), they have not always achieved their conservation goals (Cazalis et al., 2021; Venter et al., 2014), suggesting that circumstances or actions leading to PAD may contribute to a reduction in biodiversity. Additionally, post-PAD actions like mining, agriculture, grazing, and energy projects have also been associated with a decline in biodiversity. When monitoring protected areas, it can be challenging to detect subtle changes in land cover. Our findings on the significant impact of PAD events on biodiversity underscore the urgency for policymakers to consider targeted measures that enhance conservation efforts. Specific actions, such as increased funding for protected area management, can bolster monitoring and enforcement capabilities, thereby reducing illegal activities like logging and poaching (Dudley et al., 2010). This could involve deploying advanced monitoring technologies, such as satellite imaging and drones, to detect and respond to illegal activities in real-time (Sanderson et al., 2023). Capacity-building projects for local law enforcement agencies can enhance the efficiency and effectiveness of conservation efforts. The combination of technological tools and strengthened human resources can create a more robust framework for preserving biodiversity and maintaining the integrity of protected areas. Also, establishing buffer zones around protected areas can mitigate the edge effects of habitat degradation and species richness decline, which are often exacerbated by PAD events (Laurance et al., 2012).

The reason for PAD events in the U.S. is mainly infrastructure construction, but the PA network is incidentally optimized in the process of PAD events. Greater conservation benefits might be accrued if protected area management were improved (Sanderson et al., 2023). When ecological protection plays a more important role in the original PAs, the government should choose to improve management around ecological objectives, rather than choose PAD. PAD events in other countries are also generally motivated by economic goals. The direct negative effects of PA mismanagement are compounded by the fact that it is difficult for humanity to derive more benefits from the ecosystem inside the PAs than outside, including the provision of food and water, flood and disease control, spiritual, recreational, and cultural gains, and support nutrient cycle that sustains life on the earth (Duckworth and Altwegg, 2018). Financial incentives that promote conservation, such as Payments for Ecosystem Services (PES), can help align economic and conservation objectives. PES schemes provide economic compensation to landowners and local communities for managing their land in ways that protect ecosystem services (Wunder, 2005). For instance, governments can incentivize sustainable agricultural practices in areas adjacent to protected areas, which can reduce habitat fragmentation and support biodiversity (Ferraro and Kiss, 2002). These programs not only foster environmental protection but also improve local livelihoods, making conservation a socially and economically viable choice.

Differences in biodiversity change are not determined by poor management. In addition, just as changes in species distribution lag behind climate change (Barnes et al., 2023), the occurrence of post-PAD actions such as mining, agriculture, grazing, and energy projects has a time difference over the years compared to PAD events. The aggregate actions of these different industrial structures and pollution characteristics can have complex impacts on biodiversity. Considering that the potential for populations and diversity are initially higher inside PAs than outside (Duckworth and Altwegg, 2018; Sanderson et al., 2023), enacting PAD events has a solid effect on the reduction of abundance.

The observed declines in abundance may also have negative spillover effects on other ecosystem components and processes. For instance, PAD events can disrupt ecological cycling, alter species interactions, and reduce ecosystem resilience (Brudvig et al., 2009). Understanding these processes is crucial for effective conservation planning and intervention to ensure the sustainability of biodiversity in regions affected by PAD events. To support the recovery of biodiversity in regions affected by PAD events, it is essential to consider conservation strategies that account for the observed patterns and mechanisms. Conservation efforts should prioritize species and ecosystems that are particularly vulnerable to PAD events, such as those living in contact with water or located within protected areas.

Many nations have aggressive expansion plans for energy, natural resources, transportation industries, and related infrastructures (Alamgir et al., 2017; Bebbington et al., 2018; Fouquet, 2016; Oldekop et al., 2020). PAD events encourage economic exploitation while facilitating development objectives. The cost of economic activities may be lower in lands that used to be PAs than in other regions. However, damaged ecosystems may hinder sustainable economic development. The degradation of PAs may also influence PAs and have long-term effects on ecosystems (Gray et al., 2016). The loss of biodiversity due to PAD events may lead to an irreversibly degraded ecosystem. Policymakers should consider the potential loss of biodiversity in the cost-benefit analysis of PAD events. Moreover, governments in other countries with some PAD events, such as the United Kingdom, Australia, Brazil, and South Africa, can quantify the impacts of PAD events on biodiversity and design more practical strategies for PA development.

PAD events can also erode biodiversity at a regional level. It is important to set plans and policies aligned with a long-term regional vision for biodiversity conservation to minimize infrastructure expansion, halt extensive biodiversity losses surrounding roads, and manage the landscape-wide consequences. Strategic regional plans, and other conservation policies governing the effects of land-use change like management of the surviving PA network, can be used to control threats to biodiversity and mitigate the cascading effects on organisms, ecosystems, and PAs (Gray et al., 2016).

6. Conclusion

This study makes several important contributions to the understanding of the ecological and economic impacts of PAD events. We extend the current discourse on the value of ecosystem services and the costs of environmental degradation. Through micro-level data analysis, we provide robust empirical evidence that PAD events significantly reduce biodiversity abundance. The proximity to PAD events affects biodiversity negatively and decreases the abundance by 26.0 % within 50 km. We also observe an overall 32.3 % decrease in abundance after the nearest PAD event is enacted. We find that the negative impacts on biodiversity originate from organisms living in contact with water, non-mammals, organisms close to reversed PAD events, and organisms near PA belonging to the IUCN category. The long-term sustainability of our natural resources is emphasized through our findings, which advocate for policies that protect biodiversity while also promoting economic vitality.

We also quantify the economic implications of this biodiversity loss. Existing estimates of the mean willingness to pay for biodiversity range from \$0 to \$6.39 per resident (Ureta et al., 2022). We use a conservative 32.3 % decrease in abundance as our back-of-the-envelope calculation. Our estimations imply that the direct economic value in abundance loss due to the nearest PAD event is approximately up to \$2.06 per resident per year. National losses add up to \$689.95 million in 2022. The large economic loss highlights the importance of addressing potential environmental problems caused by PAD events. This study also calls for the establishment of different forms of nature reserves to protect biodiversity, such as national parks. Adopting such measures is helpful to increase public willingness to pay for biodiversity conservation and other environmental protection initiatives. By linking biodiversity conservation to economic benefits, our study provides compelling reasons for policymakers to prioritize and invest in conservation efforts. Similarly, by revealing the economic benefits of biodiversity conservation, our research encourages individual action on environmental protection, thereby contributing to broader citizen efforts towards sustainable development.

Our study provides significant insights into the implications of biodiversity loss resulting from PAD events. By strengthening regulations, implementing effective monitoring systems, and investing in the establishment and maintenance of nature reserves, we can not only safeguard our natural heritage but also harness the economic benefits that arise from biodiversity conservation. This study highlights the pressing need for proactive measures to address the environmental challenges posed by PAD events. It underscores the importance of recognizing the value of biodiversity and the potential losses incurred due to its decline. The global relevance of our findings transcends the specific national context of our analysis, offering valuable insights for environmental policy and biodiversity management on an international scale. The proactive measures we propose for environmental stewardship are grounded in a robust understanding of the economic benefits of biodiversity, providing a compelling argument for policymakers to prioritize conservation efforts. By integrating these findings into policy decision-making processes, we can pave the way for sustainable and responsible environmental management practices that protect both the ecosystems and economies.

There are several areas for future research. Due to data limitations, this study does not consider PADDD events primarily in marine systems and on private lands (Runting et al., 2015). While we provide estimates of PAD events on abundance, the impacts of PAD events on biodiversity may be more accurate when integrating PADDD tracking data with other area-based conservation databases (such as the World Database on PAs). Considering the critical role of biodiversity in the ecosystem, further research on estimating the impact of PAD events on biodiversity, quantifying the under-appreciated cost-saving benefits of effective biodiversity conservation and the corresponding economic loss in the ecosystem will be valuable. For instance, it would be helpful to calculate the costs of the socioeconomic destruction caused by zoonotic diseases against those of managing PAs which lessens supply to illegal wildlife markets. Furthermore, future research should explore the long-term ecological consequences of PAD events on ecosystem recovery and resilience.

CRediT authorship contribution statement

Yufei Li: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Lingling Hou:** Writing – review & editing, Writing – original draft, Supervision, Investigation, Formal analysis, Conceptualization. **Pengfei Liu:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

The data will be shared through Github upon acceptance.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolecon.2024.108441.

References

115th U.S. Congress, 2017. H.R.1: an act to provide for reconciliation pursuant to titles II and V of the concurrent resolution on the budget for fiscal year 2018. URL. www.congress.gov/bill/115th-congress/house-bill/1.

Adler, P.B., Tyburczy, W.R., Lauenroth, W.K., 2007. Long-term mapped quadrats from kansas prairie: demographic information for herbaceous plants. Ecology 88, 2673. https://doi.org/10.1890/0012-9658(2007)88[2673:LMQFKP]2.0.CO;2.

Alamgir, M., Campbell, M.J., Sloan, S., Goosem, M., Clements, G.R., Mahmoud, M.I., Laurance, W.F., 2017. Economic, socio-political and environmental risks of road development in the tropics. Curr. Biol. 27, R1130–R1140. https://doi.org/10.1016/j.cub.2017.08.067.

Antão, L.H., Bates, A.E., Blowes, S.A., Waldock, C., Supp, S.R., Magurran, A.E., Dornelas, M., Schipper, A.M., 2020. Temperature-related biodiversity change across temperate marine and terrestrial systems. Nat. Ecol. Evol. 4, 927–933. https://doi.org/10.1038/s41559-020-1185-7.

Barnes, A.E., Davies, J.G., Martay, B., Boersch-Supan, P.H., Harris, S.J., Noble, D.G., Pearce-Higgins, J.W., Robinson, R.A., 2023. Rare and declining bird species benefit most from designating protected areas for conservation in the UK. Nat. Ecol. Evol. 7, 92–101. https://doi.org/10.1038/s41559-022-01927-4.

Bartkowski, B., 2017. Are diverse ecosystems more valuable? Economic value of biodiversity as result of uncertainty and spatial interactions in ecosystem service provision. Ecosyst. Serv. 24, 50–57. https://doi.org/10.1016/j.ecoser.2017.02.023.

Bebbington, A.J., Humphreys Bebbington, D., Sauls, L.A., Rogan, J., Agrawal, S., Gamboa, C., Imhof, A., Johnson, K., Rosa, H., Royo, A., Toumbourou, T., Verdum, R., 2018. Resource extraction and infrastructure threaten forest cover and community rights. Proc. Natl. Acad. Sci. 115, 13164–13173. https://doi.org/10.1073/pnas.1812505115.

Blowes, S.A., Supp, S.R., Antão, L.H., Bates, A., Bruelheide, H., Chase, J.M., Moyes, F., Magurran, A., McGill, B., Myers-Smith, I.H., Winter, M., Bjorkman, A.D., Bowler, D. E., Byrnes, J.E.K., Gonzalez, A., Hines, J., Isbell, F., Jones, H.P., Navarro, L.M., Thompson, P.L., Vellend, M., Waldock, C., Dornelas, M., 2019. The geography of biodiversity change in marine and terrestrial assemblages. Science 366, 339–345. https://doi.org/10.1126/science.aaw1620.

Brudvig, L.A., Damschen, E.I., Tewksbury, J.J., Haddad, N.M., Levey, D.J., 2009. Landscape connectivity promotes plant biodiversity spillover into non-target habitats. Proc. Natl. Acad. Sci. 106, 9328–9332. https://doi.org/10.1073/pnas.0809658106.

Bruner, A.G., Gullison, R.E., Rice, R.E., Da Fonseca, G.A.B., 2001. Effectiveness of parks in protecting tropical biodiversity. Science 291, 125–128. https://doi.org/10.1126/science.291.5501.125.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., Naeem, S., 2012. Biodiversity loss and its impact on humanity. Nature 486, 59–67. https://doi.org/10.1038/nature11148.

Cazalis, V., Barnes, M.D., Johnston, A., Watson, J.E.M., Şekercioğlu, C.H., Rodrigues, A. S.L., 2021. Mismatch between bird species sensitivity and the protection of intact habitats across the Americas. Ecol. Lett. 24, 2394–2405. https://doi.org/10.1111/ele.13859.

Chape, S., Harrison, J., Spalding, M., Lysenko, I., 2005. Measuring the extent and effectiveness of protected areas as an indicator for meeting global biodiversity targets. Philos. Trans. R. Soc. B 360, 443–455. https://doi.org/10.1098/rstb.2004.1592.

Code of Federal Regulations (C.F.R.), 2016. Gathering of Certain Plants or Plant Parts by Federally Recognized Indian Tribes for Traditional Purposes. URL. https://www.federalregister.gov/documents/2016/07/12/2016-16434/gathering-of-certain-plants-or-plant-parts-by-federally-recognized-indian-tribes-for-traditional.

Cook, C.N., Valkan, R.S., Mascia, M.B., McGeoch, M.A., 2017. Quantifying the extent of protected-area downgrading, downsizing, and degazettement in Australia. Conserv. Biol. 31, 1039–1052. https://doi.org/10.1111/cobi.12904.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260. https://doi.org/10.1038/387253a0.

Daskalova, G.N., Myers-Smith, I.H., Bjorkman, A.D., Blowes, S.A., Supp, S.R., Magurran, A.E., Dornelas, M., 2024. Landscape-scale forest loss as a catalyst of population and biodiversity change. Science 368. https://doi.org/10.1126/science.aba1289.

DeFries, R.S., Foley, J.A., Asner, G.P., 2004. Land-use choices: balancing human needs and ecosystem function. Front. Ecol. Environ. 2, 249–257.

Dornelas, M., Antão, L.H., Moyes, F., Bates, A.E., Magurran, A.E., Adam, D., Akhmetzhanova, A.A., Appeltans, W., Arcos, J.M., Arnold, H., Ayyappan, N., Badihi, G., Baird, A.H., Barbosa, M., Barreto, T.E., Bässler, C., Bellgrove, A., Belmaker, J., Benedetti-Cecchi, L., Bett, B.J., Bjorkman, A.D., Błażewicz, M., Blowes, S.A., Bloch, C.P., Bonebrake, T.C., Boyd, S., Bradford, M., Brooks, A.J., Brown, J.H., Bruelheide, H., Budy, P., Carvalho, F., Castañeda-Moya, E., Chen, C.A., Chamblee, J.F., Chase, T.J., Siegwart Collier, L., Collinge, S.K., Condit, R., Cooper, E. J., Cornelissen, J.H.C., Cotano, U., Kyle Crow, S., Damasceno, G., Davies, C.H., Davis, R.A., Day, F.P., Degraer, S., Doherty, T.S., Dunn, T.E., Durigan, G., Duffy, J.E., Edelist, D., Edgar, G.J., Elahi, R., Elmendorf, S.C., Enemar, A., Ernest, S.K.M., Escribano, R., Estiarte, M., Evans, B.S., Fan, T., Turini Farah, F., Loureiro Fernandes, L., Farneda, F.Z., Fidelis, A., Fitt, R., Fosaa, A.M., Daher Correa Franco, G.A., Frank, G.E., Fraser, W.R., García, H., Cazzolla Gatti, R., Givan, O., Gorgone-Barbosa, E., Gould, W.A., Gries, C., Grossman, G.D., Gutierréz, J.R., Hale, S., Harmon, M.E., Harte, J., Haskins, G., Henshaw, D.L., Hermanutz, L., Hidalgo, P., Higuchi, P., Hoey, A., Van Hoey, G., Hofgaard, A., Holeck, K., Hollister, R.D., Holmes, R., Hoogenboom, M., Hsieh, C., Hubbell, S.P., Huettmann, F., Huffard, C.L., Hurlbert, A.H., Macedo Ivanauskas, N., Janík, D., Jandt, U., Jażdżewska, A., Johannessen, T., Johnstone, J., Jones, J., Jones, F.A.M., Kang, J., Kartawijaya, T., Keeley, E.C., Kelt, D.A., Kinnear, R., Klanderud, K., Knutsen, H., Koenig, C.C., Kortz, A.R., Král, K., Kuhnz, L.A., Kuo, C., Kushner, D.J., Laguionie-Marchais, C., Lancaster, L.T., Min Lee, C., Lefcheck, J.S., Lévesque, E., Lightfoot, D., Lloret, F., Lloyd, J.D., López-Baucells, A., Louzao, M., Madin, J.S., Magnússon, B., Malamud, S., Matthews, I., McFarland, K.P., McGill, B., McKnight, D., McLarney, W.O., Meador, J., Meserve, P.L., Metcalfe, D.J., Meyer, C.F. J., Michelsen, A., Milchakova, N., Moens, T., Moland, E., Moore, J., Mathias Moreira, C., Müller, J., Murphy, G., Myers-Smith, I.H., Myster, R.W., Naumov, A., Neat, F., Nelson, J.A., Paul Nelson, M., Newton, S.F., Norden, N., Oliver, J.C., Olsen, E.M., Onipchenko, V.G., Pabis, K., Pabst, R.J., Paquette, A., Pardede, S., Paterson, D.M., Pélissier, R., Peñuelas, J., Pérez-Matus, A., Pizarro, O., Pomati, F., Post, E., Prins, H.H.T., Priscu, J.C., Provoost, P., Prudic, K.L., Pulliainen, E., Ramesh, B.R., Mendivil Ramos, O., Rassweiler, A., Rebelo, J.E., Reed, D.C., Reich, P. B., Remillard, S.M., Richardson, A.J., Richardson, J.P., Van Rijn, I., Rocha, R., Rivera-Monroy, V.H., Rixen, C., Robinson, K.P., Ribeiro Rodrigues, R., De Cerqueira Rossa-Feres, D., Rudstam, L., Ruhl, H., Ruz, C.S., Sampaio, E.M., Rybicki, N., Rypel, A., Sal, S., Salgado, B., Santos, F.A.M., Savassi-Coutinho, A.P., Scanga, S., Schmidt, J., Schooley, R., Setiawan, F., Shao, K., Shaver, G.R., Sherman, S., Sherry, T.W., Siciński, J., Sievers, C., Da Silva, A.C., Rodrigues Da Silva, F., Silveira, F.L., Slingsby, J., Smart, T., Snell, S.J., Soudzilovskaia, N.A., Souza, G.B.G., Maluf Souza, F., Castro Souza, V., Stallings, C.D., Stanforth, R., Stanley, E.H., Mauro Sterza, J., Stevens, M., Stuart-Smith, R., Rondon Suarez, Y., Supp, S., Yoshio Tamashiro, J., Tarigan, S., Thiede, G.P., Thorn, S., Tolvanen, A., Teresa Zugliani Toniato, M., Totland, Ø., Twilley, R.R., Vaitkus, G., Valdivia, N., Vallejo, M.I., Valone, T.J., Van Colen, C., Vanaverbeke, J., Venturoli, F., Verheye, H.M., Vianna, M., Vieira, R.P., Vrška, T., Quang Vu, C., Van Vu, L., Waide, R.B., Waldock, C., Watts, D., Webb, S., Wesołowski, T., White, E.P., Widdicombe, C.E., Wilgers, D., Williams, R., Williams, S.B., Williamson, M., Willig, M.R., Willis, T.J., Wipf, S., Woods, K.D., Woehler, E.J., Zawada, K., Zettler, M.L., 2018a. BioTIME: a database of biodiversity time series for the Anthropocene. Glob. Ecol. Biogeogr. 27, 760–786. https://doi.org/10.1111/geb.12729.

Dornelas, M., Antão, L.H., Moyes, F., Bates, A.E., Magurran, A.E., Adam, D., Akhmetzhanova, A.A., Appeltans, W., Arcos, J.M., Arnold, H., Ayyappan, N., Badihi, G., Baird, A.H., Barbosa, M., Barreto, T.E., Bässler, C., Bellgrove, A., Belmaker, J., Benedetti-Cecchi, L., Bett, B.J., Bjorkman, A.D., Błażewicz, M., Blowes, S.A., Bloch, C.P., Bonebrake, T.C., Boyd, S., Bradford, M., Brooks, A.J., Brown, J.H., Bruelheide, H., Budy, P., Carvalho, F., Castañeda-Moya, E., Chen, C.A., Chamblee, J.F., Chase, T.J., Siegwart Collier, L., Collinge, S.K., Condit, R., Cooper, E. J., Cornelissen, J.H.C., Cotano, U., Kyle Crow, S., Damasceno, G., Davies, C.H., Davis, R.A., Day, F.P., Degraer, S., Doherty, T.S., Dunn, T.E., Durigan, G., Duffy, J.E., Edelist, D., Edgar, G.J., Elahi, R., Elmendorf, S.C., Enemar, A., Ernest, S.K.M., Escribano, R., Estiarte, M., Evans, B.S., Fan, T., Turini Farah, F., Loureiro Fernandes, L., Farneda, F.Z., Fidelis, A., Fitt, R., Fosaa, A.M., Daher Correa Franco, G.A., Frank, G.E., Fraser, W.R., García, H., Cazzolla Gatti, R., Givan, O., Gorgone-Barbosa, E., Gould, W.A., Gries, C., Grossman, G.D., Gutierréz, J.R., Hale, S., Harmon, M.E., Harte, J., Haskins, G., Henshaw, D.L., Hermanutz, L., Hidalgo, P., Higuchi, P., Hoey, A., Van Hoey, G., Hofgaard, A., Holeck, K., Hollister, R.D., Holmes, R., Hoogenboom, M., Hsieh, C., Hubbell, S.P., Huettmann, F., Huffard, C.L., Hurlbert, A.H., Macedo Ivanauskas, N., Janík, D., Jandt, U., Jażdżewska, A., Johannessen, T., Johnstone, J., Jones, J., Jones, F.A.M., Kang, J., Kartawijaya, T., Keeley, E.C., Kelt, D.A., Kinnear, R., Klanderud, K., Knutsen, H., Koenig, C.C., Kortz, A.R., Král, K., Kuhnz, L.A., Kuo, C., Kushner, D.J., Laguionie-Marchais, C., Lancaster, L.T., Min Lee, C., Lefcheck, J.S., Lévesque, E., Lightfoot, D., Lloret, F., Lloyd, J.D., López-Baucells, A., Louzao, M., Madin, J.S., Magnússon, B., Malamud, S., Matthews, I., McFarland, K.P., McGill, B., McKnight, D., McLarney, W.O., Meador, J., Meserve, P.L., Metcalfe, D.J., Meyer, C.F. J., Michelsen, A., Milchakova, N., Moens, T., Moland, E., Moore, J., Mathias Moreira, C., Müller, J., Murphy, G., Myers-Smith, I.H., Myster, R.W., Naumov, A., Neat, F., Nelson, J.A., Paul Nelson, M., Newton, S.F., Norden, N., Oliver, J.C., Olsen, E.M., Onipchenko, V.G., Pabis, K., Pabst, R.J., Paquette, A., Pardede, S., Paterson, D.M., Pélissier, R., Peñuelas, J., Pérez-Matus, A., Pizarro, O., Pomati, F., Post, E., Prins, H.H.T., Priscu, J.C., Provoost, P., Prudic, K.L., Pulliainen, E., Ramesh, B.R., Mendivil Ramos, O., Rassweiler, A., Rebelo, J.E., Reed, D.C., Reich, P. B., Remillard, S.M., Richardson, A.J., Richardson, J.P., Van Rijn, I., Rocha, R., Rivera-Monroy, V.H., Rixen, C., Robinson, K.P., Ribeiro Rodrigues, R., De Cerqueira Rossa-Feres, D., Rudstam, L., Ruhl, H., Ruz, C.S., Sampaio, E.M., Rybicki, N., Rypel, A., Sal, S., Salgado, B., Santos, F.A.M., Savassi-Coutinho, A.P., Scanga, S., Schmidt, J., Schooley, R., Setiawan, F., Shao, K., Shaver, G.R., Sherman, S., Sherry, T.W., Siciński, J., Sievers, C., Da Silva, A.C., Rodrigues Da Silva, F., Silveira, F.L., Slingsby, J., Smart, T., Snell, S.J., Soudzilovskaia, N.A., Souza, G.B.G., Maluf Souza, F., Castro Souza, V., Stallings, C.D., Stanforth, R., Stanley, E.H., Mauro Sterza, J., Stevens, M., Stuart-Smith, R., Rondon Suarez, Y., Supp, S., Yoshio Tamashiro, J., Tarigan, S., Thiede, G.P., Thorn, S., Tolvanen, A., Teresa Zugliani Toniato, M., Totland, Ø., Twilley, R.R., Vaitkus, G., Valdivia, N., Vallejo, M.I., Valone, T.J., Van Colen, C., Vanaverbeke, J., Venturoli, F., Verheye, H.M., Vianna, M., Vieira, R.P., Vrška, T., Quang Vu, C., Van Vu, L., Waide, R.B., Waldock, C., Watts, D., Webb, S., Wesołowski, T., White, E.P., Widdicombe, C.E., Wilgers, D., Williams, R., Williams, S.B., Williamson, M., Willig, M.R., Willis, T.J., Wipf, S., Woods, K.D., Woehler, E.J., Zawada, K., Zettler, M.L., 2018b. BioTIME: a database of biodiversity time series for the anthropocene. Glob. Ecol. Biogeogr. 27, 760–786. https://doi.org/10.1111/geb.12729.

Duckworth, G.D., Altwegg, R., 2018. Effectiveness of protected areas for bird conservation depends on guild. Divers. Distrib. 24, 1083–1091. https://doi.org/10.1111/ddi.12756.

Dudley, N., Parrish, J.D., Redford, K.H., Stolton, S., 2010. The revised IUCN protected area management categories: the debate and ways forward. Oryx 44, 485–490. https://doi.org/10.1017/S0030605310000566.

Engist, D., Finger, R., Knaus, P., Guélat, J., Wuepper, D., 2023. Agricultural systems and biodiversity: evidence from european borders and bird populations. Ecol. Econ. 209, 107854. https://doi.org/10.1016/j.ecolecon.2023.107854.

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

Ferraro, P.J., Kiss, A., 2002. Direct payments to conserve biodiversity. Science 298, 1718–1719. https://doi.org/10.1126/science.1078104.

Ferraro, P.J., Pattanayak, S.K., 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biol. 4, e105. https://doi.org/10.1371/journal.pbio.0040105.

Fouquet, R., 2016. Path dependence in energy systems and economic development. Nat. Energy 1, 1–5. https://doi.org/10.1038/nenergy.2016.98.

Gardner, A.S., Baker, D.J., Mosedale, J.R., Gaston, K.J., Maclean, I.M.D., 2023. The effectiveness of UK protected areas in preventing local extinctions. Conserv. Lett. https://doi.org/10.1111/conl.12980 e12980.

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. https://doi.org/10.1111/cobi.12332.

Gibbs, H.K., Ruesch, A.S., Achard, F., Clayton, M.K., Holmgren, P., Ramankutty, N., Foley, J.A., 2010. Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. Proc. Natl. Acad. Sci. 107, 16732–16737. https://doi.org/10.1073/pnas.0910275107.

Golden Kroner, R.E., Krithivasan, R., Mascia, M.B., 2016. Effects of protected area downsizing on habitat fragmentation in yosemite national park (USA), pp. 1864–2014. E&S 21, art22. https://doi.org/10.5751/ES-08679-210322.

Golden Kroner, R.E., Qin, S., Cook, C.N., Krithivasan, R., Pack, S.M., Bonilla, O.D., Cort-Kansinally, K.A., Coutinho, B., Feng, M., Martínez Garcia, M.I., He, Y., Kennedy, C.J., Lebreton, C., Ledezma, J.C., Lovejoy, T.E., Luther, D.A., Parmanand, Y., Ruíz-Agudelo, C.A., Yerena, E., Morón Zambrano, V., Mascia, M.B., 2019. The uncertain future of protected lands and waters. Science 364, 881–886. https://doi.org/10.1126/science.aau5525.

Gotelli, N.J., Shimadzu, H., Dornelas, M., McGill, B., Moyes, F., Magurran, A.E., 2017. Community-level regulation of temporal trends in biodiversity. Sci. Adv. 3, e1700315. https://doi.org/10.1126/sciadv.1700315.

Gray, C.L., Hill, S.L.L., Newbold, T., Hudson, L.N., Börger, L., Contu, S., Hoskins, A.J., Ferrier, S., Purvis, A., Scharlemann, J.P.W., 2016. Local biodiversity is higher inside than outside terrestrial protected areas worldwide. Nat. Commun. 7, 12306. https://doi.org/10.1038/ncomms12306.

Haninger, K., Ma, L., Timmins, C., 2017. The value of brownfield remediation. J. Assoc. Environ. Resour. Econ. 4, 197–241. https://doi.org/10.1086/689743.

Hanley, N., Perrings, C., 2019. The economic value of biodiversity. Ann. Rev. Resour. Econ. 11, 355–375. https://doi.org/10.1146/annurev-resource-100518-093946.

Kareiva, P., Marvier, M., 2012. What is conservation science? BioScience 62, 962–969. https://doi.org/10.1525/bio.2012.62.11.5.

Kassar, I., Lasserre, P., 2004. Species preservation and biodiversity value: a real options approach. J. Environ. Econ. Manag. 48, 857–879. https://doi.org/10.1016/j.jeem.2003.11.005.

Kehoe, L., Romero-Muñoz, A., Polaina, E., Estes, L., Kreft, H., Kuemmerle, T., 2017. Biodiversity at risk under future cropland expansion and intensification. Nat. Ecol. Evol. 1, 1129–1135. https://doi.org/10.1038/s41559-017-0234-3.

Kremen, C., Cameron, A., Moilanen, A., Phillips, S.J., Thomas, C.D., Beentje, H., Dransfield, J., Fisher, B.L., Glaw, F., Good, T.C., Harper, G.J., Hijmans, R.J., Lees, D. C., Louis, E., Nussbaum, R.A., Raxworthy, C.J., Razafimpahanana, A., Schatz, G.E., Vences, M., Vieites, D.R., Wright, P.C., Zjhra, M.L., 2008. Aligning conservation priorities across taxa in Madagascar with high-resolution planning tools. Science 320, 222–226. https://doi.org/10.1126/science.1155193.

Lafferty, K.D., Kenner, M.C., Estes, J.A., Tinker, M.T., Bodkin, J.L., Cowen, R.K., Harrold, C., Novak, M., Rassweiler, A., Reed, D.C., 2013. A multi-decade time series of kelp forest community structure at san nicolas island, California. Ecology 94. https://doi.org/10.1890/13-0561R.1.

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., Arroyo-Rodriguez, V., Ashton, P., Benítez-Malvido, J., Blom, A., Bobo, K.S., Cannon, C.H., Cao, M., Carroll, R., Chapman, C., Coates, R., Cords, M., Danielsen, F., De Dijn, B., Dinerstein, E., Donnelly, M.A., Edwards, D., Edwards, F., Farwig, N., Fashing, P., Forget, P.-M., Foster, M., Gale, G., Harris, D., Harrison, R., Hart, J., Karpanty, S., John Kress, W., Krishnaswamy, J., Logsdon, W., Lovett, J., Magnusson, W., Maisels, F., Marshall, A.R., McClearn, D., Mudappa, D., Nielsen, M.R., Pearson, R., Pitman, N., van der Ploeg, J., Plumptre, A., Poulsen, J., Quesada, M., Rainey, H., Robinson, D., Roetgers, C., Rovero, F., Scatena, F., Schulze, C., Sheil, D., Struhsaker, T., Terborgh, J., Thomas, D., Timm, R., Nicolas Urbina-Cardona, J., Vasudevan, K., Joseph Wright, S., Carlos Arias, G.J., Arroyo, L., Ashton, M., Auzel, P., Babaasa, D., Babweteera, F., Baker, P., Banki, O., Bass, M., Bila-Isia, I., Blake, S., Brockelman, W., Brokaw, N., Brühl, C.A., Bunyavejchewin, S., Chao, J.-T., Chave, J., Chellam, R., Clark, C.J., Clavijo, J., Congdon, R., Corlett, R., Dattaraja, H. S., Dave, C., Davies, G., de Mello Beisiegel, B., de Nazaré Paes da Silva, R., Di Fiore, A., Diesmos, A., Dirzo, R., Doran-Sheehy, D., Eaton, M., Emmons, L., Estrada, A., Ewango, C., Fedigan, L., Feer, F., Fruth, B., Giacalone Willis, J., Goodale, U., Goodman, S., Guix, J.C., Guthiga, P., Haber, W., Hamer, K., Herbinger, I., Hill, J., Huang, Z., Fang Sun, I., Ickes, K., Itoh, A., Ivanauskas, N., Jackes, B., Janovec, J., Janzen, D., Jiangming, M., Jin, C., Jones, T., Justiniano, H., Kalko, E., Kasangaki, A., Killeen, T., King, H., Klop, E., Knott, C., Koné, I., Kudavidanage, E., Lahoz da Silva Ribeiro, J., Lattke, J., Laval, R., Lawton, R., Leal, M., Leighton, M., Lentino, M., Leonel, C., Lindsell, J., Ling-Ling, L., Eduard Linsenmair, K., Losos, E., Lugo, A., Lwanga, J., Mack, A.L., Martins, M., Scott McGraw, W., McNab, R., Montag, L., Myers Thompson, J., Nabe-Nielsen, J., Nakagawa, M., Nepal, S., Norconk, M., Novotny, V., O'Donnell, S., Opiang, M., Ouboter, P., Parker, K., Parthasarathy, N., Pisciotta, K., Prawiradilaga, D., Pringle, C., Rajathurai, S., Reichard, U., Reinartz, G., Renton, K., Reynolds, G., Reynolds, V., Riley, E., Rödel, M.-O., Rothman, J., Round, P., Sakai, S., Sanaiotti, T., Savini, T., Schaab, G., Seidensticker, J., Siaka, A., Silman, M.R., Smith, T.B., de Almeida, S.S., Sodhi, N., Stanford, C., Stewart, K., Stokes, E., Stoner, K.E., Sukumar, R., Surbeck, M., Tobler, M., Tscharntke, T., Turkalo, A., Umapathy, G., van Weerd, M., Vega Rivera, J., Venkataraman, M., Venn, L., Verea, C., Volkmer de Castilho, C., Waltert, M., Wang, B., Watts, D., Weber, W., West, P., Whitacre, D., Whitney, K., Wilkie, D., Williams, S., Wright, D.D., Wright, P., Xiankai, L., Yonzon, P., Zamzani, F., 2012. Averting biodiversity collapse in tropical forest protected areas. Nature 489, 290–294. https://doi.org/10.1038/nature11318.

Lenton, T.M., Held, H., Kriegler, E., Hall, J.W., Lucht, W., Rahmstorf, S., Schellnhuber, H. J., 2008. Tipping elements in the earth's climate system. Proc. Natl. Acad. Sci. USA 105, 1786–1793. https://doi.org/10.1073/pnas.0705414105.

Li, G., Fang, C., Li, Y., Wang, Z., Sun, S., He, S., Qi, W., Bao, C., Ma, H., Fan, Y., Feng, Y., Liu, X., 2022. Global impacts of future urban expansion on terrestrial vertebrate diversity. Nat. Commun. 13. https://doi.org/10.1038/s41467-022-29324-2.

Lundhede, T.H., Jacobsen, J.B., Hanley, N., Fjeldså, J., Rahbek, C., Strange, N., Thorsen, B.J., 2014. Public support for conserving bird species runs counter to climate change impacts on their distributions. PLoS One 9, e101281. https://doi.org/10.1371/journal.pone.0101281.

Mascia, M.B., Pailler, S., 2011. Protected area downgrading, downsizing, and degazettement (PADDD) and its conservation implications. Conserv. Lett. 4, 9–20. https://doi.org/10.1111/j.1755-263X.2010.00147.x.

Mascia, M.B., Pailler, S., Krithivasan, R., Roshchanka, V., Burns, D., Mlotha, M.J., Murray, D.R., Peng, N., 2014. Protected area downgrading, downsizing, and degazettement (PADDD) in africa, asia, and latin America and the caribbean, 1900–2010. Biol. Conserv. 169, 355–361. https://doi.org/10.1016/j.biocon.2013.11.021.

Maxwell, S.L., Cazalis, V., Dudley, N., Hoffmann, M., Rodrigues, A.S.L., Stolton, S., Visconti, P., Woodley, S., Kingston, N., Lewis, E., Maron, M., Strassburg, B.B.N., Wenger, A., Jonas, H.D., Venter, O., Watson, J.E.M., 2020. Area-based conservation in the twenty-first century. Nature 586, 217–227. https://doi.org/10.1038/s41586-020-2773-z.

Moran, D., Bann, C., 2000. The Valuation of Biological Diversity for National Biodiversity Action Plans and Strategies: A Guide for Trainers.

Naeem, S., Chazdon, R., Duffy, J.E., Prager, C., Worm, B., 2016. Biodiversity and human well-being: an essential link for sustainable development. Proc. R. Soc. B Biol. Sci. https://doi.org/10.1098/rspb.2016.2091.

Naidoo, R., Ricketts, T.H., 2006. Mapping the economic costs and benefits of conservation. PLoS Biol. 4, e360. https://doi.org/10.1371/journal.pbio.0040360.

Oldekop, J.A., Rasmussen, L.V., Agrawal, A., Bebbington, A.J., Meyfroidt, P., Bengston, D.N., Blackman, A., Brooks, S., Davidson-Hunt, I., Davies, P., Dinsi, S.C., Fontana, L.B., Gumucio, T., Kumar, C., Kumar, K., Moran, D., Mwampamba, T.H., Nasi, R., Nilsson, M., Pinedo-Vasquez, M.A., Rhemtulla, J.M., Sutherland, W.J., Watkins, C., Wilson, S.J., 2020. Forest-linked livelihoods in a globalized world. Nat. Plants 6, 1400–1407. https://doi.org/10.1038/s41477-020-00814-9.

Pimentel, D., Zuniga, R., Morrison, D., 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecol. Econ. 52, 273–288. https://doi.org/10.1016/j.ecolecon.2004.10.002.

Radeloff, V.C., Hammer, R.B., Stewart, S.I., Fried, J.S., Holcomb, S.S., McKeefry, J.F., 2005. The wildland–urban interface in the United States. Ecol. Appl. 15, 799–805. https://doi.org/10.1890/04-1413.

Richardson, M., Liu, P., Eggleton, M., 2022. Valuation of wetland restoration: evidence from the housing market in Arkansas. Environ. Resour. Econ. 81, 649–683. https://doi.org/10.1007/s10640-021-00643-0.

Runting, R.K., Meijaard, E., Abram, N.K., Wells, J.A., Gaveau, D.L.A., Ancrenaz, M., Possingham, H.P., Wich, S.A., Ardiansyah, F., Gumal, M.T., Ambu, L.N., Wilson, K. A., 2015. Alternative futures for borneo show the value of integrating economic and conservation targets across borders. Nat. Commun. 6, 6819. https://doi.org/10.1038/ncomms7819.

Sanderson, F.J., Wilson, J.D., Franks, S.E., Buchanan, G.M., 2023. Benefits of protected area networks for breeding bird populations and communities. Anim. Conserv.

Schleicher, J., Eklund, J., Barnes, D., Geldmann, J., Oldekop, J.A., Jones, J.P.G., 2020. Statistical matching for conservation science. Conserv. Biol. 34, 538–549. https://doi.org/10.1111/cobi.13448.

Siqueira-Gay, J., Metzger, J.P., Sánchez, L.E., Sonter, L.J., 2022. Strategic planning to mitigate mining impacts on protected areas in the brazilian amazon. Nat. Sustain. 5, 853–860. https://doi.org/10.1038/s41893-022-00921-9.

Sonter, L.J., Dade, M.C., Watson, J.E.M., Valenta, R.K., 2020. Renewable energy production will exacerbate mining threats to biodiversity. Nat. Commun. 11, 4174. https://doi.org/10.1038/s41467-020-17928-5.

Strassburg, B.B.N., Iribarrem, A., Beyer, H.L., Cordeiro, C.L., Crouzeilles, R., Jakovac, C. C., Braga Junqueira, A., Lacerda, E., Latawiec, A.E., Balmford, A., Brooks, T.M., Butchart, S.H.M., Chazdon, R.L., Erb, K.-H., Brancalion, P., Buchanan, G., Cooper, D., Díaz, S., Donald, P.F., Kapos, V., Leclère, D., Miles, L., Obersteiner, M., Plutzar, C., de Scaramuzza, M.C.A., Scarano, F.R., Visconti, P., 2020. Global priority areas for ecosystem restoration. Nature 586, 724–729. https://doi.org/10.1038/s41586-020-2784-9.

Symes, W.S., Rao, M., Mascia, M.B., Carrasco, L.R., 2016. Why do we lose protected areas? Factors influencing protected area downgrading, downsizing and degazettement in the tropics and subtropics. Glob. Chang. Biol. 22, 656–665. https://doi.org/10.1111/gcb.13089.

Thieme, M.L., Khrystenko, D., Qin, S., Golden Kroner, R.E., Lehner, B., Pack, S., Tockner, K., Zarfl, C., Shahbol, N., Mascia, M.B., 2020. Dams and protected areas: quantifying the spatial and temporal extent of global dam construction within protected areas. Conserv. Lett. 13, e12719. https://doi.org/10.1111/conl.12719.

U.S. Department of the Interior (DOI), 2017. Final report summarizing findings of the review of designations under the antiquities act. URL. www.doi.gov/sites/doi.gov/files/uploads/revised_final_report.pdf.

Ureta, J.C., Motallebi, M., Vassalos, M., Seagle, S., Baldwin, R., 2022. Estimating residents' WTP for ecosystem services improvement in a payments for ecosystem services (PES) program: a choice experiment approach. Ecol. Econ. 201, 107561. https://doi.org/10.1016/j.ecolecon.2022.107561.

Venter, O., Fuller, R.A., Segan, D.B., Carwardine, J., Brooks, T., Butchart, S.H.M., Di Marco, M., Iwamura, T., Joseph, L., O'Grady, D., Possingham, H.P., Rondinini, C., Smith, R.J., Venter, M., Watson, J.E.M., 2014. Targeting global protected area expansion for imperiled biodiversity. PLoS Biol. 12, e1001891. https://doi.org/10.1371/journal.pbio.1001891.

Victor, D.G., House, J.C., Joy, S., 2005. A madisonian approach to climate policy. Science 309, 1820–1821. https://doi.org/10.1126/science.1113180.

Watson, J.E.M., Evans, T., Venter, O., Williams, B., Tulloch, A., Stewart, C., Thompson, I., Ray, J.C., Murray, K., Salazar, A., McAlpine, C., Potapov, P., Walston, J., Robinson, J.G., Painter, M., Wilkie, D., Filardi, C., Laurance, W.F., Houghton, R.A., Maxwell, S., Grantham, H., Samper, C., Wang, S., Laestadius, L., Runting, R.K., Silva-Chávez, G.A., Ervin, J., Lindenmayer, D., 2018. The exceptional value of intact forest ecosystems. Nat. Ecol. Evol. 2, 599–610. https://doi.org/10.1038/s41559-018-0490-x.

Webb, S.L., Scanga, S.E., 2001. Windstorm disturbance without patch dynamics: twelve years of change in a Minnesota forest. Ecology 82, 893–897. https://doi.org/10.1890/0012-9658(2001)082[0893:WDWPDT]2.0.CO;2.

Wilson, E.O., 2017. HALF-EARTH: Our Planet's Fight for Life, p. 14.

Wunder, S., 2005. Payments for environmental services: some nuts and bolts. CIFOR Occasional Paper 42, 3–4.