Pervasive cropland in protected areas highlight trade-offs between conservation and food security

Varsha Vijay[a,1] and Paul R. Armsworth[b]

[a]National Socio-Environmental Synthesis Center, University of Maryland, Annapolis, MD 21401; and [b]Department of Ecology and Evolutionary Biology and National Institute for Mathematical and Biological Synthesis, University of Tennessee, Knoxville, TN 37996

Edited by Hugh P. Possingham, The Nature Conservancy, Sherwood, QLD, Australia, and approved November 23, 2020 (received for review May 19, 2020)

Global cropland expansion over the last century caused widespread habitat loss and degradation. Establishment of protected areas aims to counteract the loss of habitats and to slow species extinctions. However, many protected areas also include high levels of habitat disturbance and conversion for uses such as cropland. Understanding where and why this occurs may realign conservation priorities and inform protected area policy in light of competing priorities such as food security. Here, we use our global synthesis cropland dataset to quantify cropland in protected areas globally and assess their relationship to conservation aims and socio-environmental context. We estimate that cropland occupies 1.4 million km^2 or 6% of global protected area. Cropland occurs across all protected area management types, with 22% occurring in strictly protected areas. Cropland inside protected areas is more prevalent in countries with higher population density, lower income inequality, and with higher agricultural suitability of protected lands. While this phenomenon is dominant in midnorthern latitudes, areas of cropland in protected areas of the tropics and subtropics may present greater trade-offs due to higher levels of both biodiversity and food insecurity. Although area-based targets are prominent in biodiversity goal-setting, our results show that they can mask persistent anthropogenic land uses detrimental to native ecosystem conservation. To ensure the long-term efficacy of protected areas, post-2020 goal setting must link aims for biodiversity and human health and improve monitoring of conservation outcomes in cropland-impacted protected areas.

conservation | food security | protected areas | area-based targets | CBD

Global cropland has more than doubled since 1850 (1), with tremendous consequences for both human health and the environment. The increase in food production led to marked decreases in global hunger, despite exponential growth of human populations over the same period (2, 3). However, habitat loss to make way for cropland expansion has caused extinctions of native species and transformed ecosystem structure and function (4–11). Without significant dietary shifts or food waste reduction, agricultural intensification on existing croplands may not be sufficient to meet food demands for increasing human populations, driving further land clearing (12, 13).

Protected areas provide the backbone for conservation efforts, allowing for the protection of species and ecosystems in an increasingly human-dominated matrix (14). Protected areas have rapidly expanded to now cover an estimated 15% of the Earth's terrestrial surface (15) with further expansion of total area protected called for under the recent Strategic Plan for Biodiversity, adopted by the Convention on Biological Diversity (CBD). Area-based targets of this type for protected areas are common. However, recent studies have questioned the success of these targets (16), pointing to the continued presence of anthropogenic threats (17–19) and species' population declines inside protected areas (20–22). For example, cropland represents one of the most impacted land use types, and yet is known to occur and be expanding inside many protected areas (23).

The United Nations 2030 Sustainable Development Goals (SDGs) provides a shared framework on which to develop a comprehensive approach to managing food production inside current protected areas and creating new protected areas in the broader agricultural matrix. The SDGs state the need to "end hunger and achieve food security" (Goal 2: Zero Hunger) but also to "halt and reverse land degradation and halt biodiversity loss" (Goal 15: Life on Land). The obvious interdependence between these goals creates complex trade-offs to be navigated at local, regional, and international scales. Recently, this has spurred a growing recognition of the need to simultaneously pursue goals to minimize perverse outcomes and maximize cobenefits between the SDGs (24–26).

In recognition of the interconnectedness between biodiversity and human health, the SDGs emphasize the need for considerations of environmental sustainability when meeting food production goals. Corresponding post-2020 strategic planning efforts for biodiversity also need to codify these linkages in global criteria for protected areas, which can be consistently measured using quantifiable metrics. Here, we quantify the extent and configuration of agriculture within current global terrestrial protected areas in order to develop relevant indicators and a baseline to monitor progress toward simultaneous achievement and management for both food security and conservation goals. We utilize a synthesis approach, bringing together multiple remotely sensed estimates of cropland extent in a spatially hierarchical analytical framework, to produce a globally consistent dataset of cropland in protected areas at fine spatial resolutions sufficient for decision making by conservation end users (27). We compare the distribution of croplands in protected areas in a variety of ecosystems and under

Significance

Biodiversity conservation strategies emphasize protected area expansion to mitigate species losses by safeguarding habitat. However, demand for land for food production is also increasing. We establish a baseline estimate of where and why cropland occurs in protected areas. Our estimates would indicate cropland represents around 18% of all human impacts inside protected areas. Cropland will not be effective in conserving many species, particularly habitat specialists, rare, and threatened species. This suggests a reexamination of the effectiveness of area-based protected area planning is needed. The success of post-2020 biodiversity management depends on addressing trade-offs with food production by creating opportunities to integrate ecosystem conservation and restoration with programs for hunger and malnutrition.

Author contributions: V.V. and P.R.A. designed research; V.V. performed research; V.V. contributed new reagents/analytic tools; V.V. and P.R.A. analyzed data; and V.V. and P.R.A. wrote the paper.

The authors declare no competing interest.

This article is a PNAS Direct Submission.

[1]To whom correspondence may be addressed. Email: vvijay@sesync.org.

This article contains supporting information online at https://www.pnas.org/lookup/suppl/doi:10.1073/pnas.2010121118/-/DCSupplemental.

Published January 18, 2021.

differing land-use regulations. We identify countries and regions where cropland occurrence may be of particular concern to conservation outcomes by examining covariation between cropland and biodiversity metrics in protected areas. Finally, we examine the socio-environmental context associated with a high prevalence of cropland in protected areas and test the sensitivity of our findings by modeling the subset of countries with food insecurity.

Results

Using our synthesis cropland data, we identified 24.9 million km^2 of cropland occupying 13.6% of ice-free terrestrial surface (Fig. 1). Of this cropland area, 1.4 million km^2 is found inside protected areas (PAs), constituting 6% of all protected lands. The scale of cultivation in protected areas varies across regions (shown as the proportion of cropland in a given pixel, Fig. 1 *C*–*E*). In Brazil (Fig. 1*C*) and Nigeria (Fig. 1*D*), many protected areas contain low densities of cropland, emphasizing the importance of subpixel detection when monitoring land use at 1-km resolution. In contrast, croplands in protected areas in Germany (Fig. 1*E*) are more easily assessed at coarser scales.

To examine the spatial distribution of cropland in protected areas, we disaggregated these data by latitude (Fig. 1*B*), biome (Fig. 2*A*), and International Union for Conservation of Nature (IUCN) category of the protected areas (Fig. 2*B*). Cropland in protected areas is concentrated between 40° to 60° N, primarily in Europe (Fig. 1*B*), where it occurs more often than would be expected based on the proportion of total cropland area at those latitudes. We also find that cropland in protected areas occurs less than would be expected based on total cropland area between 15° to 35° N in Asia, and between 25° to 45° S across multiple continents (Fig. 1*B*). Cropland in protected areas is found primarily (~80%) within temperate and tropical forest and grassland biomes (Fig. 2*A*). Cropland in closed canopy systems, such as temperate (44%) and tropical (9%) broadleaf forests, may be characterized by existing deforestation data (e.g., ref. 28). However, our analyses indicate cropland also occurs in protected areas within open canopy systems like temperate and tropical savannas and shrublands as well as temperate and tropical grasslands (all savanna and grassland = 34%) requiring specialized land cover data. All IUCN protected area categories contain some cropland (Fig. 2*B*). However, cropland is most associated with the least regulated areas, IUCN V, VI, and uncategorized areas (77%), as well as IUCN IV areas (14%), which are usually considered strictly protected. In both temperate systems and in IUCN categories IV, V, and uncategorized areas, the proportion of cropland in protected areas is greater than the proportion of protected area sited within these categories (Fig. 2 *A* and *B*).

We evaluated the spatial covariation between four vertebrate species richness metrics in protected areas (alpha species richness, threatened species, data deficient species, and crop threatened species) and the proportion of protected area occupied by cropland. The resulting bivariate map (Fig. 3) highlights locations where cropland in protected areas may have the greatest potential impacts on biodiversity. We found consistently high values (≥66% of the distribution) for both biodiversity (across species richness metrics) and cropland proportion in the tropics and subtropics of Africa, Asia, and in temperate Central Asia. We found areas with high biodiversity (≥66%) and moderate levels of cropland (≥33% and ≤66%) in protected areas in East Africa, Madagascar, and Southeast Asia. Covariation patterns were mostly consistent with the four species richness metrics, but additional locations in Eastern Europe may be of concern when focusing on those species for which cropland was identified by the IUCN as a driver of species declines (Fig. 3*D*).

We used statistical regression to determine how socio-environmental context relates to the extent of cropland in protected areas found in different countries ($n = 126$). Three variables showed clear directional relationships with the proportion of cropland found in protected areas: human population density agricultural suitability of protected lands and Gini index (Fig. 4*A*). In particular, greater cropland inside protected areas was associated with higher human population density and lower income inequality. Increased cropland inside protected areas was also associated with higher agricultural suitability. These three variables show a strong association with cropland in protected areas

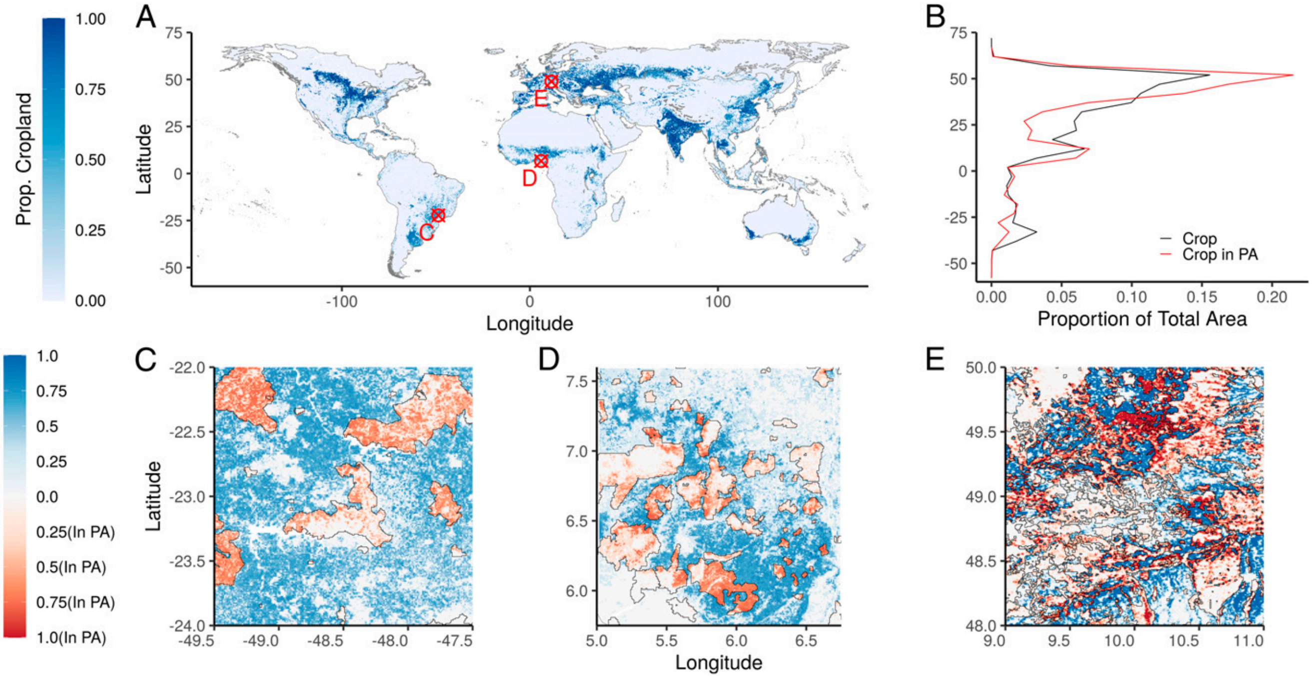

Fig. 1. (*A*) Map showing global map of cropland. Proportion of pixel in cropland from 0 (gray) to 1 (blue). (*B*) Plot showing proportion of total cropland (black) and proportion of cropland in protected areas (red) by latitude. Insets show greater details of cropland patterns from global map (marked in red in *A*) in Brazil (*C*), Nigeria (*D*), and Germany (*E*). Proportion of pixel in cropland from 1 in protected area (red) to 0 (gray) to 1 outside protected area (blue).

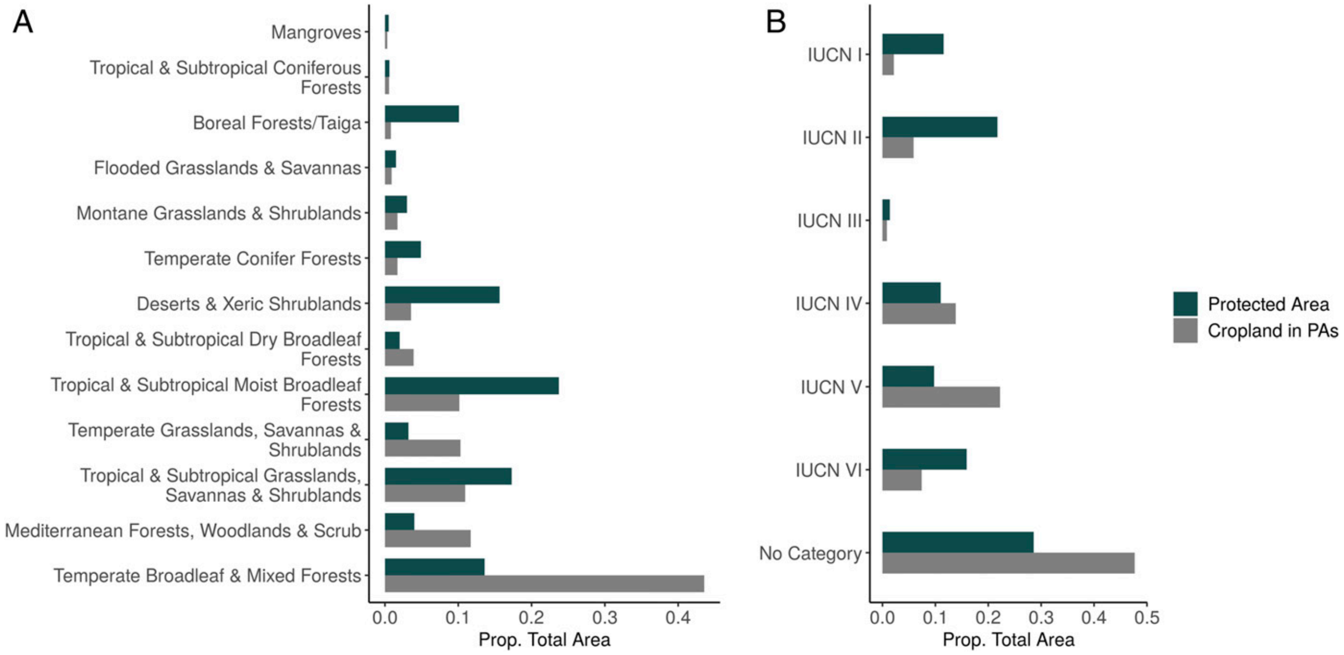

Fig. 2. Cropland in protected areas by biome and by IUCN protected area management category. (*A*) Proportion of total protected area (dark gray) and total cropland in protected area (light gray) by biome. Biomes are ordered in increasing proportions of cropland in protected area. (*B*) Proportion of total protected area (dark gray) and total cropland in protected area (light gray) by IUCN protected area management category.

after controlling for effects of total amount of cropland, protected areas, and country size. Although we may expect to see differences where conflicts with food security might be expected to be greatest, these relationships continue to apply when restricting attention to the subset of countries monitored by the Global Hunger Index (GHI) (*n* = 95, Fig. 4*B*). Regression coefficients adjusted SE (SE) and sum of Akaike weights (SW) for global and hunger subset model average are shown in *SI Appendix*, Table S2.

Discussion

This study represents the most comprehensive assessment of the extent and distribution of global cropland inside protected areas. When prior global studies have considered cropland in protected areas, they have primarily done so using aggregate indices to represent multiple anthropogenic land uses (23, 29, 30), making it difficult to both parse uncertainty associated with cropland data and to closely examine relationships with important contextual predictors. We find that the total area of cropland inside protected areas represents a relatively large proportion of all human impacts on protected areas (18%), based on the estimate that approximately one-third of protected land is under intense human pressure (31). Worryingly, we find that 22% of cropland in protected occurs in areas of strict protection (IUCN I–IV), although cropland is more common in protected areas designated

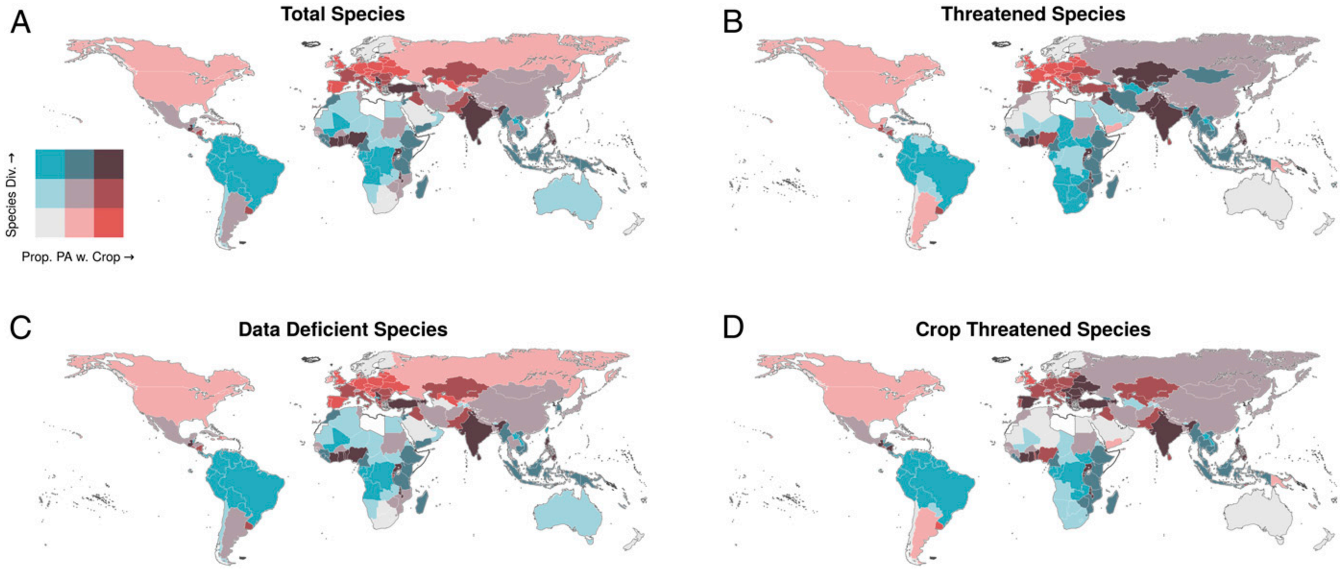

Fig. 3. Bivariate maps of proportion of cropland in protected area and biodiversity in protected areas by country. (*A*) Total terrestrial species diversity. (*B*) Threatened species diversity. (*C*) Data deficient species diversity. (*D*) Crop threatened species diversity. White areas represent missing data.

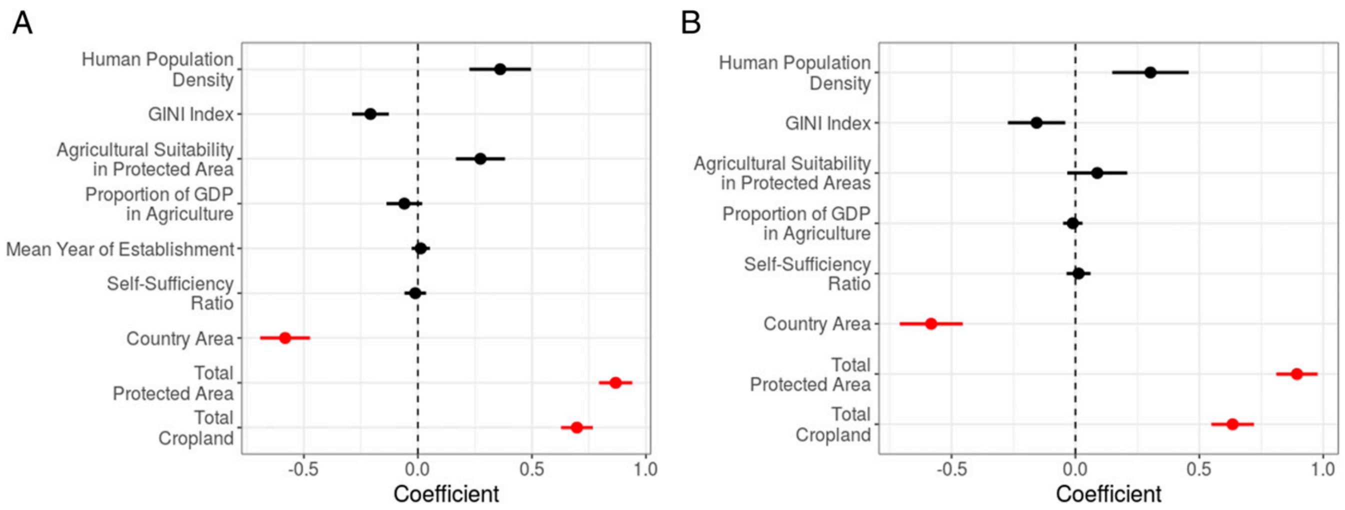

Fig. 4. Standardized model averaged regression coefficients predicting cropland area with error bars showing adjusted SE for global model (*A*) and countries monitored by the Global Hunger Index (*B*). Nonstandardized base model coefficients shown in red, where the null expectation is that these will to be close to −1 for country area and + 1 for total protected area and total cropland.

multiuse. The persistence of many native species, particularly habitat specialists, rare, and threatened species, is not compatible with conversion to cropland (32–36), suggesting conservation goals of protection may often be being compromised. Indeed, those calling for significant expansion of protected areas (37, 38) likely are not envisioning that cropland is one of the major habitat types that would disproportionately benefit from any increase protection if existing protection is an indicator of what is to come.

We found that cropland in protected areas is dominant in midnorthern latitudes, with relatively low levels of food insecurity (39). We also found that tropical and subtropical countries with globally high biodiversity and cropland impacts had, on average, serious hunger and undernutrition issues (GHI = 22.8). These tropical and subtropical regions also experienced some of the largest overall expansions in cropland, raising concerns for cropland expansion into protected and unprotected conservation priority areas (40, 41). Our statistical models suggest greater cropland inside protected areas is associated with higher human population density, lower income inequality, and higher agricultural suitability inside protected areas. Only 14% of current cropland in protected areas occurs in areas established since 2000, potentially reflecting the increasing adoption of systematic conservation planning methods (42). However, we find that the associations between cropland occurrence and protected area establishment vary significantly by continent (*SI Appendix*, Fig. S3). Taken together, our results suggest two primary scenarios behind the appearance of cropland in protected areas. First, in some places, cropland predated the establishment of protected areas, which were designed specifically to include areas of historical cropland production. This scenario occurs in some protected areas of Europe (43–45), where many of the protected areas are designated as being less strictly protected, multiuse sites (45% protected areas are IUCN V, VI, or uncategorized). Supporting this scenario, we find that in Europe almost half (0.45) of the cropland in protected areas is found in older protected areas, established by 1975. The second scenario is one of encroachment of cropland into areas that were previously protected to safeguard other habitats; this scenario is more often associated with some areas of the developing world tropics. In these places, cultivation in protected areas may be a reflection of issues with tenure rights for local and indigenous people who perceive protected areas as community assets (46–48). This is exemplified by patterns of cropland occurrence in South America and in Africa, where recently established areas are more prone to cropland impacts. While the European scenario is more intentional, we would argue neither scenario is particularly helpful to advancing international biodiversity conservation goals based on conserving native species and habitats.

Each scenario also suggests contrasting conservation strategies. In countries with both strong food security and adequate management resources, more attention should be given to how to restore native ecosystems and a smaller fraction of protected area networks should focus on protecting cultivated systems. Encouragingly, many countries which fit this characterization have set targets under the Bonn Challenge, a global effort launched in 2011 to protect biodiversity and ecosystem services through restoration (49). Meanwhile, in more food insecure countries, effective management of protected areas likely involves integrating conservation solutions with programs to address hunger and undernutrition. Exploring relationships with food security creates the potential for deriving cobenefits from food assistance funding, something that could address the underlying causes of cropland expansion as well as noted funding shortfalls in protected management and enforcement (50, 51). To ensure the effectiveness of both strategies, protected area categorization should be more transparent in order to reduce mismatches between regulation and local socio-economic context, a condition which may exacerbate cropland conversions.

The occurrence of cropland in protected areas designed to conserve rare and threatened species and ecosystems is of particular concern. For instance, we note the disproportionate presence of cropland in IUCN category IV. This particular category of protected area is established for the conservation of target species and may contain high levels of habitat degradation and fragmentation. Active monitoring of cropland extent impacts on key species, and, as stated by the IUCN, “regular, active interventions” will be critical to ensuring management aims of these protected areas are met. Cropland impacted protected areas in biomes such as the Mediterranean Forests, Woodland, and Shrublands or forest systems of Madagascar, which are considered biodiversity hotspots, warrant particularly careful monitoring as well (9, 52, 53). The variability in postconversion biodiversity outcomes for different taxa (54) emphasize the importance of ongoing inventory and survey work to supplement remote monitoring approaches.

Our work is subject to several caveats regarding the quantification of cropland extent and the assumptions we make about siting and regulation of protected lands. For example, while we incorporate many estimates of cropland extent to characterize cropland inside and outside of protected areas, our estimates may underestimate small-scale shifting agricultural production and include some grazing land and monoculture pulp and paper production. Including shifting agriculture and grazing lands could lead to estimates of total agricultural land almost three times our estimate (38%; ref. 55), with likely increases in estimates of cropland in protected areas. To give another example, we use overarching IUCN category designations to describe regulation of protected areas. At the same time, we recognize that there is substantial variability within these categories, reflecting differences in management and regulation across countries and regions.

Recent trends suggest that food production goals for the next century may result in cropland expansion (13, 56–58). This is likely to create new challenges for protected areas. As part of ongoing efforts to establish new international targets for protected areas, we suggest a more integrated approach be taken in food insecure countries to address trade-offs between food production and conservation inside protected areas. We also suggest bolder restoration goals for protected areas be established in places like Europe. In addition, moving forward, protected area goals need to move beyond area-based targets, which ignore land use composition inside protected areas and may have contributed to the excessive coverage of cropland that we have today. To do so, we will need to leverage the best available data for monitoring and understanding the drivers and impacts of cropland on species and ecosystems in protected areas. Our study provides a benchmark on which to build monitoring programs for tracking changes in the amount and impact of cropland in protected areas.

Methods

Cropland Data. All spatial analyses were conducted in Google Earth Engine (59) with WGS 84 projection and nominal 1-km spatial resolution (30 arcseconds). We restricted our analysis to terrestrial areas with coastlines, country, and continent boundaries defined by the Database of Global Administrative Areas and removed regions of rock/ice and tundra, defined in the Ecoregions 2017 dataset. To further reduce error introduced by evaluating areas with frequent surface water inundation, we used data from the the European Commission's Joint Research Centre (JRC) Global Surface Water Mapping dataset (60). We filtered those areas where more than half of a 1-km pixel was classified as inundated for more than 90% of the 1984–2015 study period.

In this study, cropland is defined as all land used for permanent or shifting/fallow production of annual or perennial crops. This broad definition includes plantation crops but does not focus on timber and pulp plantations and grazing lands. Many of the land cover products integrated within our analyses used multiple classes to represent cropland. We use guidelines for legend harmonization based on the Land Cover Classification System (LCCS) classification scheme (Food and Agriculture Organization, FAO) to classify each land cover dataset into binary cropland/noncropland classes.

Our cropland data were generated from a combination of five different cropland data products with differing temporal (nominal year listed) and spatial resolutions: European Space Agency Climate Change Initiative (ESACCI) 2015 (300 m), Moderate Resolution Imaging Spectroradiometer Land Cover (MODISLC) 2010–2015 (500 m), Global Land Cover by National Mapping Organizations (GLCNMO) 2013 (500 m), International Institute for Applied Systems Analysis (IIASA)/International Food Policy Research Institute (IFPRI) 2005 (proportion of 1 km), Global Land Cover-SHARE (GLC-SHARE) 2010 (prop. 1 km) (61–65). We selected datasets which represent commonly used estimates of current cropland focused on producing a nominal 2013 product. The datasets encompass different approaches to land cover classification, ranging from classifications of remotely sensed imagery to datasets which incorporate information from regional and national level agricultural statistics. We validated our cropland dataset with FAO cropland data and over 1,850 validation points classified by other researchers (66, 67).

To appropriately incorporate information from these multiple sources, we utilize a spatially hierarchical approach, which evaluates agreement at both an intermediate 500-m spatial resolution as well as at the final 1-km spatial resolution to produce a final dataset representing the proportion of cropland within the pixel. This approach privileges information from high-resolution data products, provides subpixel spatially representative information (particularly important in areas of sparse cropland), and minimizes the computational requirements of global extent analysis.

Our analysis approach (*SI Appendix*, Fig. S1*A*) privileges agreement between commonly used land cover datasets to produce rapid and accurate assessments of land cover to be used in conservation assessments. This approach builds on data integration approaches utilized in a number of recent studies (68, 69), including the IIASA/IFPRI hybrid land cover and FAO-GLCSHARE datasets used in this analysis. Adopting this spatially hierarchical approach to resolving noted spatial inconsistencies between cropland datasets (70) and using multitemporal inputs allows for streamlined dataset validation, replication, and revision as new data become available.

Protected Areas. Terrestrial protected area data were obtained from the November 2019 version of the World Database on Protected Areas (71). We excluded point data and used only polygon data with established boundaries in our analysis to reduce errors introduced by the misclassification of protected land. We followed WDPA recommended practices and removed those protected areas with less than 1-km terrestrial area to improve compatibility with the spatial resolution of our cropland data. To reduce overestimation of protected area coverage, we resolved overlapping protection designations by assigning the strictest IUCN designation present. We analyzed protected areas without IUCN categorization as a separate class (No Category). We calculated mean protected area year of establishment (for areas with documented dates) at the country level. Additionally, to investigate relationships between date of establishment and cropland, we classified protected areas into age classes based on the earliest date of protection (results in *SI Appendix*, *Supplementary Materials*).

Species Distributions and Biomes. We used species distribution maps for birds, mammals, amphibians, and reptiles to determine alpha species richness, threatened species richness, data deficient species richness, and crop threatened species richness (72, 73). We defined biomes using the Ecoregions 2017 dataset (38). For each species, we used only areas where species were classified as Extant or Probably Extant. We defined threatened species as those listed by the IUCN as Vulnerable to extinction or worse. We utilized IUCN classification of threat types to identify species for which crop cultivation was identified as a specific driver of species decline. This included all species which faced threats from annual and perennial nontimber crops (classification 2.1), the category best aligned with agriculture identified in our cropland classification. To increase comparability between species richness metrics, calculated as the sum of species in each category occurring within protected areas, all values were rescaled between 0 and 1. We examined spatial covariation in the country level distributions of species richness and cropland in protected areas, creating a bivariate map based on distribution quantiles (74).

Statistical Analyses. We analyzed the relationship between cropland area in protected areas and a number of socio-economic predictors at the country level.

We first evaluated a null model based on the random distribution of cropland in protected areas (C_P) relative to the total amount of protected area (PA), the total cropland area (CT), and the country area (CA).

$$\log(C_P/PA) = \log(CT/CA) + \varepsilon_j. \qquad [1]$$

Because this process did not conform to null expectations (*SI Appendix*, Fig. S2), we relaxed the specification to consider a base model of the form:

$$\log(C_P) = \beta_0 + \beta_1\log(CT) + \beta_2\log(PA) + \beta_3\log(CA) + \varepsilon_j, \qquad [2]$$

where dimensional analysis allows for the comparison of coefficient values to the expected null values ($\beta_1 = 1$, $\beta_2 = 1$, $\beta_3 = -1$).

We then compared the fit of models that included additional covariates to the base model in Eq. **2** using multimodel inference techniques in the package MuMIn in R (75). These models include variables describing hypotheses related to the Gini index (GI); mean human population density (PD); proportion of GDP associated with agriculture (AG); mean agricultural suitability in protected areas (AS); the Self-Sufficiency Ratio, defined as the amount of food consumed produced domestically (SR); and the mean year of establishment of protected areas (YR). In addition to the base model predictors which supply geometric constraints within our model framework, we hypothesized that protected areas in countries with more recent protected area establishment would contain less cropland. All other predictors, we hypothesized, would be positively correlated with cropland. Sources for predictor variables and more specific hypotheses can be found in *SI Appendix*, Table S1. We evaluated collinearity between predictors and found no strong evidence for multicollinearity. We log transformed our response, all null hypothesis predictors, and human population density to increase normality of the distributions. We standardized all predictors, except for those describing the null hypothesis, by centering and using a z scale transformation to increase comparability of coefficients in the final models. Thus, our full model for cropland in protected areas by country was:

$$\log(C_P) = \beta_0 + \beta_1\log(CT) + \beta_2\log(PA) + \beta_3\log(CA) + \beta_4 GI + \beta_5\log(PD) + \beta_6 AG + \beta_7 AS + \beta_8 SR + \beta_9 YR + \varepsilon_j. \quad [3]$$

We considered all possible combinations of the additional covariates and performed model averaging across the subset of such models with $\Delta AICc \leq 2$. We calculated variable importance using the sum of model weights. We did not include interaction terms having no a priori reason to focus on some interactions from among the many that are possible.

Data Availability. The cropland data developed for this analysis are available for download from https://doi.org/10.5061/dryad.zs7h44j6k (76). All other data need to evaluate the conclusions presented in this paper are publicly available with sources noted in the main text or supplementary materials.

ACKNOWLEDGMENTS. We thank many people for their conversations and insights on both the ecological and social dimensions of protected area management and agricultural cultivation. In particular, we thank E. Ward, E. Fricke, F. Gollnow, D. Graef, L. Mordechai, and Q. Read for their feedback on the analysis and manuscript and B. Parmentier for input on validation of our cropland dataset. V.V. was supported by the National Socio-Environmental Synthesis Center under funding received from NSF Grant DBI-1639145.

1. K. Klein Goldewijk, A. Beusen, J. Doelman, E. Stehfest, New anthropogenic land use estimates for the holocene: HYDE 3.2. *Earth Syst. Sci. Data* **9**, 927–953 (2017).
2. C. B. Barrett, Measuring food insecurity. *Science* **327**, 825–828 (2010).
3. R. W. Fogel, *The Escape from Hunger and Premature Death, 1700-2100: Europe, America, and the Third World* (Cambridge University Press, 2004).
4. E. Crist, C. Mora, R. Engelman, The interaction of human population, food production, and biodiversity protection. *Science* **356**, 260–264 (2017).
5. S. L. Pimm, V. Vijay, "The impact of agriculture on global biodiversity" in *Population, Agriculture, and Biodiversity: Problems and Prospects* (University of Missouri Press, 2020), p. 365.
6. J. A. Foley *et al*., Solutions for a cultivated planet. *Nature* **478**, 337–342 (2011).
7. H. K. Gibbs *et al*., Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. *Proc. Natl. Acad. Sci. U.S.A.* **107**, 16732–16737 (2010).
8. D. Tilman, Global environmental impacts of agricultural expansion: The need for sustainable and efficient practices. *Proc. Natl. Acad. Sci. U.S.A.* **96**, 5995–6000 (1999).
9. O. E. Sala *et al*., Global biodiversity scenarios for the year 2100. *Science* **287**, 1770–1774 (2000).
10. E. Brondizio, J. Settele, S. Díaz, *IPBES. 2019 Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services* (Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Service, Bonn, Germany, 2019).
11. F. S. Chapin 3rd *et al*., Consequences of changing biodiversity. *Nature* **405**, 234–242 (2000).
12. D. Tilman *et al*., Forecasting agriculturally driven global environmental change. *Science* **292**, 281–284 (2001).
13. T. A. Pugh *et al*., Climate analogues suggest limited potential for intensification of production on current croplands under climate change. *Nat. Commun.* **7**, 12608 (2016).
14. M. Di Marco, O. Venter, H. P. Possingham, J. E. M. Watson, Changes in human footprint drive changes in species extinction risk. *Nat. Commun.* **9**, 4621 (2018).
15. UNEP-WCMC, *IUCN, NGS, "Protected planet live report 2020* (UNEP-WCMC, IUCN and NGS, Cambridge, UK, Gland, Switzerland, Washington, D.C., 2020).
16. P. Visconti *et al*., Protected area targets post-2020. *Science* **364**, 239–241 (2019).
17. K. Schulze *et al*., An assessment of threats to terrestrial protected areas. *Conserv. Lett.* **11**, e12435 (2018).
18. A. G. Bruner, R. E. Gullison, R. E. Rice, G. A. B. da Fonseca, Effectiveness of parks in protecting tropical biodiversity. *Science* **291**, 125–128 (2001).
19. A. R. Joshi *et al*., Tracking changes and preventing loss in critical tiger habitat. *Sci. Adv.* **2**, e1501675 (2016).
20. B. W. T. Coetzee, K. J. Gaston, S. L. Chown, Local scale comparisons of biodiversity as a test for global protected area ecological performance: A meta-analysis. *PLoS One* **9**, e105824 (2014).
21. J. E. M. Watson, N. Dudley, D. B. Segan, M. Hockings, The performance and potential of protected areas. *Nature* **515**, 67–73 (2014).
22. J. Geldmann *et al*., Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biol. Conserv.* **161**, 230–238 (2013).
23. J. Geldmann, A. Manica, N. D. Burgess, L. Coad, A. Balmford, A global-level assessment of the effectiveness of protected areas at resisting anthropogenic pressures. *Proc. Natl. Acad. Sci. U.S.A.* **116**, 23209–23215 (2019).
24. P. Katila *et al*., *Sustainable Development Goals: Their Impacts on Forests and People* (Cambridge University Press, 2019).
25. M. Nilsson, D. Griggs, M. Visbeck, Policy: Map the interactions between sustainable development goals. *Nature* **534**, 320–322 (2016).
26. M. Stafford-Smith *et al*., Integration: The key to implementing the sustainable development goals. *Sustain. Sci.* **12**, 911–919 (2017).
27. L. N. Joppa *et al*., BIG DATA AND BIODIVERSITY. Filling in biodiversity threat gaps. *Science* **352**, 416–418 (2016).
28. M. C. Hansen *et al*., High-resolution global maps of 21st-century forest cover change. *Science* **342**, 850–853 (2013).
29. C. M. Kennedy, J. R. Oakleaf, D. M. Theobald, S. Baruch-Mordo, J. Kiesecker, Managing the middle: A shift in conservation priorities based on the global human modification gradient. *Glob. Change Biol.* **25**, 811–826 (2019).
30. O. Venter *et al*., Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. *Nat. Commun.* **7**, 12558 (2016).
31. K. R. Jones *et al*., One-third of global protected land is under intense human pressure. *Science* **360**, 788–791 (2018).
32. B. Phalan, M. Onial, A. Balmford, R. E. Green, Reconciling food production and biodiversity conservation: Land sharing and land sparing compared. *Science* **333**, 1289–1291 (2011).
33. J. A. Foley *et al*., Global consequences of land use. *Science* **309**, 570–574 (2005).
34. Ç. H. Şekercioğlu *et al*., Long-term declines in bird populations in tropical agricultural countryside. *Proc. Natl. Acad. Sci. U.S.A.* **116**, 9903–9912 (2019).
35. T. Newbold *et al*., Global effects of land use on local terrestrial biodiversity. *Nature* **520**, 45–50 (2015).
36. L. Kehoe *et al*., Biodiversity at risk under future cropland expansion and intensification. *Nat. Ecol. Evol.* **1**, 1129–1135 (2017).
37. E. O. Wilson, *Half-earth: Our Planet's Fight for Life* (WW Norton & Company, 2016).
38. E. Dinerstein *et al*., An ecoregion-based approach to protecting half the terrestrial realm. *Bioscience* **67**, 534–545 (2017).
39. K. Von Grebmer *et al*., *2019 Global Hunger Index: The Challenge of Hunger and Climate Change* (Welthungerhilfe, 2019).
40. B. Phalan *et al*., Crop expansion and conservation priorities in tropical countries. *PLoS One* **8**, e51759 (2013).
41. H. M. Jayathilake, G. W. Prescott, L. R. Carrasco, M. Rao, W. S. Symes, Drivers of deforestation and degradation for 28 tropical conservation landscapes. *Ambio* **50**, 215–228 (2020).
42. E. J. McIntosh, R. L. Pressey, S. Lloyd, R. J. Smith, R. Grenyer, The impact of systematic conservation planning. *Annu. Rev. Environ. Resour.* **42**, 677–697 (2017).
43. P. Batáry, L. V. Dicks, D. Kleijn, W. J. Sutherland, The role of agri-environment schemes in conservation and environmental management. *Conserv. Biol.* **29**, 1006–1016 (2015).
44. K. J. Gaston, S. F. Jackson, A. Nagy, L. Cantú-Salazar, M. Johnson, Protected areas in Europe: Principle and practice. *Ann. N. Y. Acad. Sci.* **1134**, 97–119 (2008).
45. C. Hummel *et al*., Ecosystem services in European protected areas: Ambiguity in the views of scientists and managers? *PLoS One* **12**, e0187143 (2017).
46. L. M. Scherl, R. Wild, A. Wilson, *Can Protected Areas Contribute to Poverty Reduction?: Opportunities and Limitations* (IUCN, 2004).
47. C. J. Cavanagh, T. A. Benjaminsen, Guerrilla agriculture? A biopolitical guide to illicit cultivation within an IUCN category II protected area. *J. Peasant Stud.* **42**, 725–745 (2015).
48. R. P. Neumann, Political ecology of wildlife conservation in the Mt. Meru area of Northeast Tanzania. *Land Degrad. Dev.* **3**, 85–98 (1992).
49. R. Dave *et al*., *Second Bonn Challenge progress report. Application of the Barometer in 2018* (IUCN, Gland, Switzerland, 2019).
50. A. Waldron *et al*., Targeting global conservation funding to limit immediate biodiversity declines. *Proc. Natl. Acad. Sci. U.S.A.* **110**, 12144–12148 (2013).
51. L. Coad *et al*., Widespread shortfalls in protected area resourcing undermine efforts to conserve biodiversity. *Front. Ecol. Environ.* **17**, 259–264 (2019).
52. E. C. Underwood, J. H. Viers, K. R. Klausmeyer, R. L. Cox, M. R. Shaw, Threats and biodiversity in the mediterranean biome. *Divers. Distrib.* **15**, 188–197 (2009).
53. B. Mappin *et al*., Restoration priorities to achieve the global protected area target. *Conserv. Lett.* **12**, e12646 (2019).
54. T. Newbold *et al*., Global effects of land use on biodiversity differ among functional groups. *Funct. Ecol.* **34**, 684–693 (2020).
55. N. Ramankutty, A. T. Evan, C. Monfreda, J. A. Foley, Farming the planet: 1. Geographic distribution of global agricultural lands in the year 2000. *Global Biogeochem. Cycles* **22**, GB1003 (2008).
56. M. C. Hunter, R. G. Smith, M. E. Schipanski, L. W. Atwood, D. A. Mortensen, Agriculture in 2050: Recalibrating targets for sustainable intensification. *Bioscience* **67**, 386–391 (2017).
57. N. Alexandratos, J. Bruinsma, *World Agriculture Towards 2030/2050: The 2012 Revision* (FAO, Rome, Italy, 2012).
58. D. Tilman, C. Balzer, J. Hill, B. L. Befort, Global food demand and the sustainable intensification of agriculture. *Proc. Natl. Acad. Sci. U.S.A.* **108**, 20260–20264 (2011).
59. N. Gorelick *et al*., Google Earth engine: Planetary-scale geospatial analysis for everyone. *Remote Sens. Environ.* **202**, 18–27 (2017).
60. J.-F. Pekel, A. Cottam, N. Gorelick, A. S. Belward, High-resolution mapping of global surface water and its long-term changes. *Nature* **540**, 418–422 (2016).
61. ESA. Land Cover CCI Product User Guide Version 2. Tech. Rep. (2017).
62. D. Sulla-Menashe, M. A. Friedl, *User Guide to Collection 6 MODIS Land Cover (MCD12Q1 and MCD12C1) Product* (USGS, Reston, VA, 2018), pp. 1–18.

63. T. Kobayashi *et al*., Production of global land cover data – GLCNMO2013. *J. Geogr. Geol.* **9**, 3 (2017).
64. L. See *et al*., Building a hybrid land cover map with crowdsourcing and geographically weighted regression. *ISPRS J. Photogramm. Remote Sens.* **103**, 48–56 (2015).
65. J. Latham, R. Cumani, I. Rosati, M. Bloise, *Global Land Cover Share (GLC-SHARE) Database Beta-Release Version 1.0-2014* (FAO, Rome, Italy, 2014).
66. J. C. Laso Bayas *et al*., A global reference database of crowdsourced cropland data collected using the Geo-Wiki platform. *Sci. Data* **4**, 170136 (2017).
67. Food and Agriculture Organization of the United Nations, *FAOSTAT statistical database* (FAO, Rome, Italy, 2019).
68. F. Waldner *et al*., A unified cropland layer at 250 m for global agriculture monitoring. *Data (Basel)* **1**, 3 (2016).
69. M. Tuanmu, W. Jetz, A global 1-km consensus land-cover product for biodiversity and ecosystem modelling. *Glob. Ecol. Biogeogr.* **23**, 1031–1045 (2014).
70. A. Pérez-Hoyos, F. Rembold, H. Kerdiles, J. Gallego, Comparison of global land cover datasets for cropland monitoring. *Remote Sens.* **9**, 1118 (2017).
71. UNEP-WCMC, “Protected planet: The world database on protected areas (WDPA)”, https://www.protectedplanet.net/en. Accessed 29 November 2019.
72. IUCN, *The IUCN red list of threatened species* (IUCN, Gland, Switzerland, 2018).
73. BirdLife International, BirdLife International and handbook of the birds of the world (2018). Bird species distribution maps of the world (2018). Version 2018.1. Available at http://datazone.birdlife.org/species/requestdis. Accessed 5 July 2019.
74. C. Prener, T. Grossenbacher, A. Zehr, biscale: Tools and Palettes for Bivariate Thematic Mapping (R package version 0.2.0, 2020).
75. K. Barton, M. K. Barton, Package ‘MuMIn.’ (R package version 1, 2019).
76. V. Vijay, P. Armsworth, Data from: Pervasive cropland in protected areas highlight trade-offs between conservation and food security. *Dryad*. https://doi.org/10.5061/dryad.zs7h44j6k. Deposited 19 August 2020.