

Is Indonesian peatland loss a cautionary tale for Peru? A two-country comparison of the magnitude and causes of tropical peatland degradation

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Abstract Indonesia and Peru harbor some of the largest lowland tropical peatland areas. Indonesian peatlands are subject to much greater anthropogenic activity than Peru's, including drainage, logging, agricultural conversion, and burning, resulting in high greenhouse gas and particulate emissions. To derive insights from the Indonesian experience, we explored patterns of impact in the two countries, and compared their predisposing factors. Impacts differ greatly among Indonesian regions and the Peruvian Amazon in the following order: Sumatra > Kalimantan > Papua > Peru. All impacts, except fire, are positively related to population density. Factors enhancing Indonesian

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peatlands' susceptibility to disturbance include peat doming that facilitates drainage, coastal location, high local population, road access, government policies permitting peatland use, lack of enforcement of protections, and dry seasons that favor extensive burning. The main factors that could reduce peatland degradation in Peru compared with Indonesia are geographic isolation from coastal population centers, more compact peatland geomorphology, lower population and road density, more peatlands in protected areas, different land tenure policies, and different climatic drivers of fire; whereas factors that could enhance peatland degradation include oil and gas development, road expansion in peatland areas, and an absence of government policies explicitly protecting peatlands. We conclude that current peatland integrity in Peru arises from a confluence of factors that has slowed development, with no absolute barriers protecting Peruvian peatlands from a similar fate to Indonesia's. If the goal is to maintain the integrity of Peruvian peatlands, government policies recognizing unique peatland functions and sensitivities will be necessary.

Keywords Agriculture · Fire · Forest cover loss · Indonesia · Oil palm · Peru · Plantations · Population density · Roads · Tropical peatlands

1 Introduction

Peatlands are globally important for the diversity of ecosystem services they provide, not least of which is their ability to take up large quantities of carbon dioxide (CO₂) and convert it to organically bound forms that are stabilized by the anoxic conditions caused by high water tables, and hence have the potential to mitigate climate change. Peat accumulations over millennial timescales have stored ~1/3 of global soil C on ~3% of the land surface (Page et al. 2011). However, this potential to store carbon (C) depends on whether they remain sinks or are converted to sources of CO₂. The vast stores of C in peatlands are vulnerable to any changes in climate or hydrology that lower water tables, leading to increased oxidation by microorganisms and fire (Turetsky et al. 2015). Peatlands around the world have been drained, usually for conversion to agriculture or tree plantations. This drainage and subsequent disturbances has resulted in massive losses of CO₂ from peatlands, contributing to the rapid rise in concentrations of greenhouse gases (Limpens et al. 2008).

Tropical peatlands represent a significant fraction of global peatlands (Page et al. 2011; Gumbricht et al. 2017), and are under some of the greatest threats from human activities (Crump 2017). The most intensively studied and also most intensively impacted region in the tropics is Indonesia (see, e.g., Rieley et al. 2008 and references therein). In contrast, recently mapped extensive peatlands in Peru are much less well studied and have so far been spared the same impacts, but are on the verge of expanded development pressure (Roucoux et al. 2017). Hence explicit comparison of these two countries could provide insights that guide sustainable development in Peru.

The largest areas of tropical peatlands are in Indonesia, the Congo Basin, and the Amazon Basin (Page et al. 2011; Lahteenoja et al. 2012; Dargie et al. 2017; Gumbricht et al. 2017). Tropical peatlands are most commonly found in lowland areas, and these lowland peatlands are mostly peat swamp forests (wetland type 20 in the Ramsar Classification; Semeniuk and Semeniuk 1997) dominated by angiosperms in Indonesia (Page et al. 2006), and a mixture of angiosperms and palms (Arecaceae) in Peru (Draper et al. 2014). Indonesian peatlands are estimated to hold from ~25 to ~55 Pg C depending on mapped area and assumptions about depth and density (Jaenicke et al. 2008; Warren et al. 2017). This represents more organic

matter than is stored in all the forest biomass in Indonesia. In Peru, lowland peatlands also represent a large C reserve—combining the peatland area (74,644 km²; Gumbrecht et al. 2017) and C density (882 Mg ha⁻¹; Draper et al. 2014), these peatlands are estimated to store ~6.6 Pg C, about 90% of which is in peat. Despite only covering approximately 11% of the area of the Peruvian Amazon, this peatland C equals approximately 75% of mature forest biomass C in the entire Peruvian Amazon (8.80 Pg; Peru, Ministerio del Ambiente 2015b), and so is critical to any efforts to manage the GHG emissions from land use in the Peruvian Amazon.

In Indonesia, peatlands have been subjected to massive development following deforestation and drainage for agriculture, including monoculture pulpwood and oil palm plantations (Miettinen et al. 2012, 2016). In 2015, for the Indonesian regions of Sumatra and Kalimantan, the two regions of Indonesia with the greatest impacts, of a total of about 130,120 km² of peatlands, an estimated 25% were in industrial plantations, 24% were in smallholder agriculture, 42% were degraded to varying degrees or cleared, and only 6% were pristine peat swamp forests (Miettinen et al. 2012). As a consequence of these changes, Indonesian peatlands lose globally significant amounts of C to decomposition, DOC export, and fires, especially in drought years (Page et al. 2009). For Sumatra plus Kalimantan, estimates for 2015 were 119.6 Mt C yr in peat oxidation, and a roughly equivalent annual amount from peatland fires (Miettinen et al. 2016). Although observations indicate that in Peru there is much less human impact on peatlands, with low deforestation, conversion to agriculture, drainage, and fires, there are no change estimates for Peru that are equivalent to those for Indonesia.

Indonesia and Peru both lie in the tropics at similar latitudes: Peru is located between 0 and 18° S latitude, with the majority of its peatlands located north of 8 degrees latitude; Indonesia is located between 6° N and 11° S with peatlands spread across this entire range (Fig. 1). Indonesia has three major regions that harbor the majority of its peatlands: the island of Sumatra; Kalimantan, which is on the island of Borneo; and Papua, which is on the island of New Guinea (Fig. 1). These differ greatly in population and so provide a natural gradient of population impacts within Indonesia that can be compared with that of Peru. Papua is the least populated peatland region in Indonesia and most similar in anthropogenic impacts to the Peruvian Amazon where lowland peatlands are found. There are some significant differences between Indonesia and Peru that might help explain their different peatland development status. Whereas Indonesian peatlands have developed in near-coastal interfluvial regions (Dommain et al. 2014), lowland peatlands in Peru formed away from the coast in the subsiding basins to the east of the Andes Mountains. Indonesia is characterized by both ombrotrophic (rain-fed) peat domes and to a lesser extent minerotrophic (groundwater-fed) peatlands (Dommain et al. 2010, 2014). In Peru, the Pastaza-Marañón Foreland Basin contains the majority of the peatlands (Lähteenoja et al. 2012, Draper et al. 2014) with a mixture of minerotrophic and domed ombrotrophic peatlands (Lähteenoja et al. 2009). The latter are typically palm swamps dominated by aguaje (*Mauritia flexuosa*) whereas the former are dominated by lower-stature pole forests (Draper et al. 2014).

There is a great deal of interest in conservation of Peruvian peatlands in the Amazon Basin, because the Peruvian Amazon is undergoing significant migration, deforestation, and land use change (Ichikawa et al. 2014; Bax et al. 2016). Roucoux et al. (2017) recently published an excellent synthesis of the literature on conservation status of Peruvian peatlands. This study described the nature of certain key impacts (roads and oil and gas development) but lacks a quantitative description of the current status of deforestation and land-use impacts on peatlands in Peru. This makes direct quantitative comparisons with other regions such as Indonesia impossible. Because Indonesia is a region of the world with intensive impacts and a well-

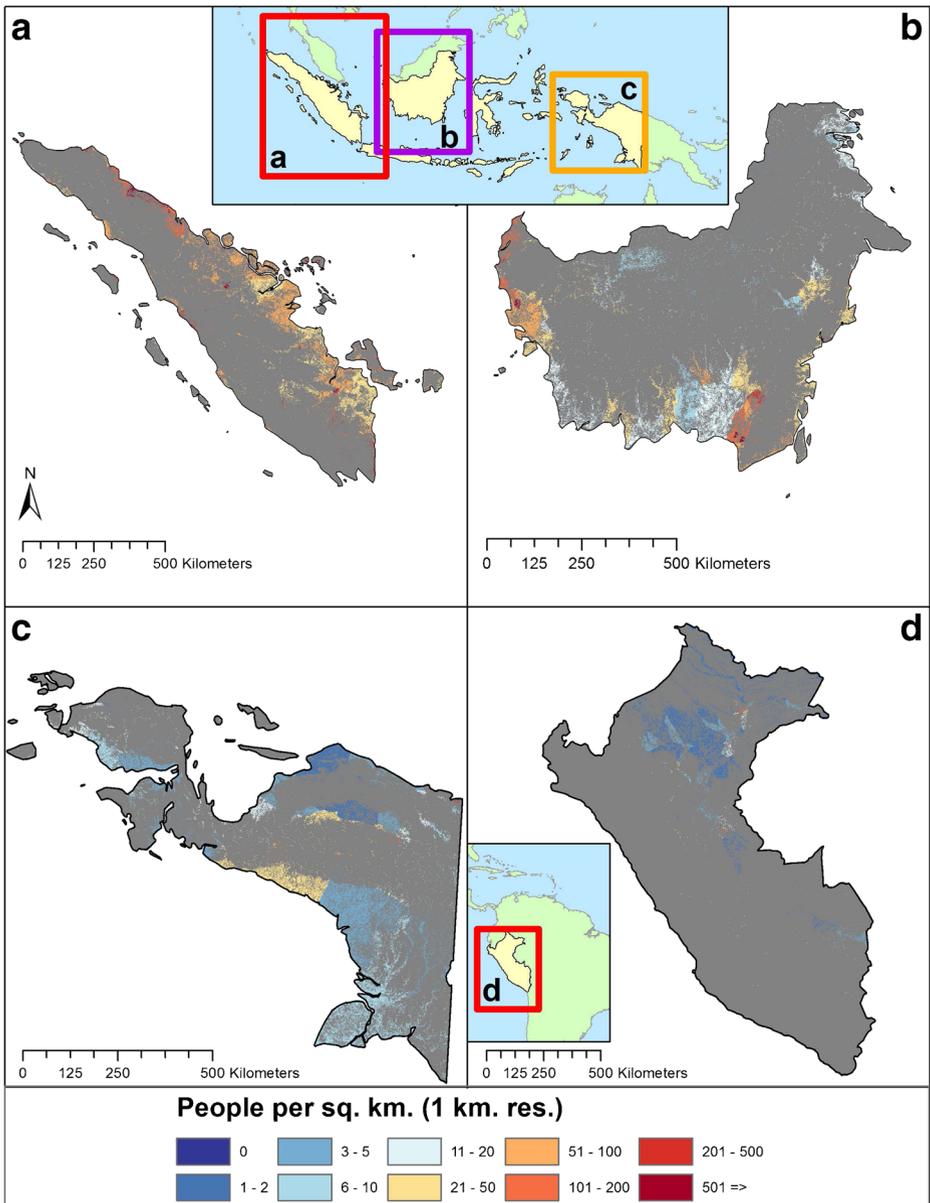


Fig. 1 Map of population density in peatlands of **a** Sumatra, **b** Kalimantan, **c** Papua, and **d** Peru. Inset lettering follows figure pane lettering. Upper inset shows Indonesia (light yellow fill) with the three high peat regions outlined in boxes (**a**, Sumatra; **b**, Kalimantan; **c**, Papua). Lower inset, red outline shows location of Peru (**d**, light yellow fill) in South America

developed literature on the magnitude and causes of anthropogenic impacts, we believed it could be informative to quantitatively contrast the current status of the regions of the two countries, and to ask what we can learn from the Indonesian experience that might inform development in peat-rich regions of Peru. We recognize that intercountry comparisons are

fraught with challenges related to social, economic, political and environmental conditions driving local patterns of human impacts on peatlands. It is our hope that this two-country comparison will reveal similarities and divergences in the factors underlying peatland change, and spur further efforts to compare and contrast patterns and causes of peatland degradation among diverse tropical countries.

Hence, our objectives were to (1) quantify human impacts to lowland peatlands in Indonesia and Peru, (2) explore their possible causes, (3) evaluate whether Peruvian peatlands are likely to experience human impacts similar to those observed in Indonesia given the status quo, and (4) present policy options for peatland protection. Our approach began with a geographic information systems (GIS) analysis of peatland distribution and potential drivers of anthropogenic impacts, followed by a literature synthesis on factors affecting peatland impacts in the two countries. Using existing data layers, we performed our GIS analyses for Peru as a whole, for Indonesia as a whole, as well as for Peru vs. the three main peat-harboring sub-regions of Indonesia: Sumatra, Kalimantan and Papua (Fig. 1). We use “region” from here forward as shorthand for the country of Peru and three insular regions of Indonesia. Our reason for assessing the Indonesian regions separately is that they differ greatly in potential drivers of peatland degradation such as population density, road incursion, and associated human impacts, and hence will allow us to perform statistical analyses of the relationship of variables across the four regions. In each of these regions we examined patterns for the whole region or country, including both uplands and peatlands, as well as for peatlands alone. We tested the hypotheses that population density in each region would predict population density in peatlands, that road density in or adjacent to peatlands would be predicted by road density in the rest of the region, and that population density in peatlands and/or road density in peatlands would be a good predictor for other factors driving peatland loss. Any deviations from these hypothesized relationships suggest that alternate factors (e.g., region-specific differences in government policy, geomorphology, resources, or climate) could be important in structuring patterns of peatland degradation. We then follow the presentation of results of this analysis with an extended discussion and review of the factors examined and how they might be influenced by government policy or conservation efforts.

2 Methods

2.1 Data layers

We gathered existing data layers that provide comparable data for the two countries. Although regionally-developed peatland maps (e.g., Draper et al. 2014) are likely more accurate, we used the global peatland layer from Gumbricht et al. (2017), because it used the same mapping method in both countries, allowing direct comparison among regions. This excludes mountain peatlands which are not mapped by the method of Gumbricht et al. (2017), and so this analysis is focused solely on lowland peatlands. The definition of peat used in Gumbricht et al. (2017) was a soil with $\geq 50\%$ organic matter content to a depth of ≥ 30 cm. Any statistics involving peatlands were for the peat-only areas defined by this layer. Likewise, we selected GIS data layers that included both Peru and Indonesia for quantifiable predictors we expect to be important, either directly or indirectly, in affecting peatland condition. All data were aggregated to 1 km² resolution to make them directly comparable. We recognize that aggregation can lead to some artifacts (Ju et al. 2005), e.g., our estimate of the peatland area in Peru and

Indonesia are 86 and 89%, respectively, of the original Gumbrecht et al. (2017) estimates. For that reason, our results are most useful as relative rather than absolute estimates, which is appropriate for our purpose of comparing patterns among regions.

The human population data from the Gridded Population of the World for 2015 were used (Center for International Earth Science Information Network - CIESIN - Columbia University 2016) to assess population density. The data are in units of individuals per km² representing averages for administrative units (Doxsey-Whitfield et al. 2015), and as a result also include population on uplands adjacent to peatlands when they are contained in the same administrative unit. We view this as a positive attribute for our analysis, because we expect that population in proximity to peatlands is likely to have a large impact on the peatlands.

The road density data are from the Open Street Map project (© OpenStreetMap contributors, <https://www.openstreetmap.org>). This method underestimates actual road density, most notably not capturing the extensive network of private roads that serve plantations. We prioritized getting comparable metrics of road density among regions, and so this density estimate should be considered as a relative estimator. A line density map (road length per unit area) was made for each region at the km² resolution based on a line density radius of 1 km. As a result, the mapped roads may or may not be in peatlands; however, they are at a minimum adjacent to mapped peatlands, i.e., they are found in the same 1 km² pixel.

The agriculture, intact forest, and degraded forest cover data layers are aggregates of relevant classes from the European Space Agency Climate Change Initiative Land Cover Project (Poulter et al. 2015; ESA-CCI, <http://maps.elie.ucl.ac.be/CCI/viewer/>), a remotely sensed product of global land use and land cover (LULC). See Online Resource 1 for details. The degraded forest category might include some naturally open peatlands, although these are typically a small percentage of total peatland area in these regions (Draper et al. 2014).

The plantations data come from Transparent World (2015) plantations dataset (<http://www.transparentworld.ru/en/resources/plantation/>). Land was classified as plantations if the majority of the km² was in any plantation category. The plantations were subdivided into oil palm (*Elaeis guineensis*) plantations, tree plantations, fruit tree plantations, immature plantations, and unknown plantations based on the major crop yield for each km² in the data base. Note that the agricultural layers from ESA-CCI also include plantations. However, these two databases are not always in agreement, so to capture this, we have mapped areas as agriculture (from ESA-CCI) as well as plantations (from Transparent World), indicating whether they are congruent or not.

The forest cover loss data come from the Global Forest Change Mapper (Hansen et al. 2013), and is cumulative for the period 2000 to 2015. The relevant scenes were merged into a single layer to quantify total forest cover loss over the period.

The burned area data come from MODIS Global Monthly Fire Location Product (MCD14ML; <http://modis-fire.umd.edu/pages/ActiveFire.php?target=GetData>). The fire data were aggregated to determine annual coverage of fire pixels over the time period of 2001–2016.

The protected areas were extracted from the World Database on Protected Areas (WDPA) (UNEP-WCMC 2016). The data were identified for each area and converted into raster datasets.

The oil and gas deposits datasets were obtained for the world up to 2009 (Lujala et al. 2007). Depleted areas were removed from the dataset. Note that this under-represents area of oil and gas deposits, especially in Peru where new discoveries have been made since 2009. Others have reported areas in oil and gas lease (e.g., Finer et al. 2013; Roucoux et al. 2017)—we chose instead to map deposits because of the availability of the global database for comparative purposes. Furthermore these deposits would reveal areas of overlap with protected areas, even if oil leases are not presently permitted.

2.2 GIS analysis

All GIS data were converted to 1 km resolution raster format for each area (Peru, Indonesia, Sumatra, Kalimantan, and Papua) to permit the calculations of overlap of peatland and other factors in Table 1. All areas were cut to the political boundaries seen in Fig. 1. All data were re-projected to an Albers equal-area conic projection to provide reliable statistical estimates when compared to a Geographic projection (Slocum et al. 2005). We mapped all data in ArcGIS 10.3.1, using the colorblind safe palettes recommended in Colorbrewer 2.0 (Brewer 2017).

2.3 Statistical analysis

Data were summarized in tabular form for each country and for the three regions of Indonesia. As noted earlier, our goal was largely a descriptive comparison of these regions, as well as application of linear regression for the data from the four regions (Sumatra, Kalimantan, Papua, Peru) to define the relationship between peatland condition and key predictors that are likely to drive peatland alteration. Human population density is an obvious starting point, based on the expectation that many of the impacts observed are anthropogenic in origin.

Table 1 Comparison of key variables affecting peatland degradation for Peru and Indonesia (as a whole and broken out for the three main peat regions; all identified as “region” for brevity). All data were derived from a GIS analysis of available data sources

Variables	Peru	Indonesia			
		Sumatra	Kalimantan	Papua	Total Indonesia
Peatland area (km ²)	66,079	55,914	58,799	66,501	193,346
Peatland area (% region)	5.13	11.84	11.00	16.20	10.20
Pop. density in region (indiv km ⁻² ; 2015)	24	115	29	13	135
Pop. density in peatlands (indiv km ⁻² ; 2015)	3	121	46	8	104
Road density in region (km km ⁻² ; 2017)	0.14	0.16	0.07	0.03	0.19
Road density peatlands (km km ⁻² ; 2017)	0.01	0.16	0.11	0.01	0.15
Forest % region (2015)	62.04	41.92	62.91	89.57	59.90
Forest % peatland (2015)	94.55	43.02	63.15	83.72	62.63
Forest minus plantation % peatland (2015)	94.53	29.59	60.06	83.67	57.62
Shrubland % region (2015)	14.64	21.41	16.54	5.38	14.08
Shrubland % peatland (2015)	1.69	15.02	11.42	9.50	11.95
Shrubland minus plantation % peatland (2015)	1.69	7.38	9.08	9.48	8.78
Agriculture % region (2015)	2.67	34.32	18.14	2.92	22.72
Agriculture % peatland (2015)	0.19	37.86	19.51	3.53	20.12
Oil palm % region (2015)	0.03	17.72	9.08	0.28	7.38
Oil palm % peatland (2015)	0.01	17.97	6.18	0.05	7.31
Tree plantations % region (2015)	0.01	3.73	0.95	0.00	1.35
Tree plantations % peatland (2015)	0.00	8.13	0.59	0.00	2.53
Other/young plantations % region (2015)	0.04	9.85	2.84	0.37	4.04
Other/young plantations % peatland (2015)	0.01	15.78	4.47	0.23	6.20
Total plantations % region (2015)	0.08	31.30	12.87	0.65	12.78
Total plantations % peatland (2015)	0.02	41.88	11.24	0.27	16.03
Forest loss % region (2000–2015)	1.68	20.98	15.96	1.50	11.20
Forest loss % peatland (2000–2015)	1.10	24.20	11.52	1.12	12.46
Area burned % region (2001–2016)	1.02	7.82	9.25	3.06	6.04
Area burned % peatland (2001–2016)	0.28	17.36	20.96	5.90	13.51
Protected area % region (2017)	18.35	11.19	8.88	20.97	11.71
Protected area % peatland (2017)	21.29	8.66	13.60	20.47	14.00
Oil and gas deposit % peatland (2009)	12.24	31.37	6.74	5.17	14.01

Similarly, much of human impact in terrestrial ecosystems is mediated by road access (e.g., Laurance et al. 2014), so understanding the predictive power of road density for impacts on peatlands seemed critical. Using data for these regions, we performed linear regressions between continuous independent (population density, road density) and dependent variables (agriculture, tree plantations, oil palm plantations, burned area, deforested area, oil and gas fields, protected area). Some variables serve in some cases as both independent and dependent variables in different analyses (e.g., road density). We also performed regression analyses for selected variables for the region as a whole, including all upland and peatland areas, as predictor for the same variable in the peatland-only areas from these four regions. The rationale for this comparison was to explore whether degree of development of the larger region is sufficient for understanding impacts on peatlands, or if there are deviations between development in the larger region and the peatlands that might be explained by other, e.g., geographic or climatic, factors. Relationships were considered significant at $p < 0.05$. All plots and regressions were done in SigmaPlot for Windows 12.5 (Systat Software, Inc.). Although we recognize that the data are sparse for regressions, we feel that this is a useful way to display and describe the relationships.

We also summarized fire occurrence patterns in the three Indonesian regions and Peru to visualize long-term trends in fire occurrence. The annual burned peatland area was summarized for each year to determine fire return frequency class, and also to examine relationship with climatic variables. El Niño and the Southern Oscillation in Indonesia (ENSO; Wang et al. 2017) has been identified as a critical driver for fire cycles in Indonesia, with intense fire seasons associated with the combination of deforestation and El Niño events (large-scale oceanic warming events in the tropical Pacific; Murdiyarto and Adiningsih 2007; Yin et al. 2016). The dates for El Niño events were derived from the Oceanic Niño Index at the US National Weather Service Climate Prediction Center (http://origin.cpc.ncep.noaa.gov/products/analysis_monitoring/ensostuff/ONI_v5.php). In the western Amazon, warm tropical North Atlantic sea surface temperature (NTA-SST) anomalies, sometimes in combination with El Niño events, have been identified as likely drivers of drought and fires in the western Amazon (Fernandes et al. 2011; Chen et al. 2015; Marengo and Espinoza 2016; Erfanian et al. 2017) and so were compared with high fire year occurrence.

3 Results

Indonesia has roughly three times more area classified as peatland than Peru. The mapped absolute peatland area for each of the four regions (Sumatra, Kalimantan, Papua, and Peru) are relatively similar (Table 1). However, relative (percent) peatland area of the different regions is lowest for Peru, representing about half that of Indonesia, and one half to one third of the three Indonesian regions. Virtually all of the mapped lowland peatlands in Peru are away from the coast in the Amazonian lowlands to the east of the Andes (Fig. 1d), whereas most of the mapped peatlands in the three Indonesian regions are adjacent to the coast (Fig. 1a–c).

Population density in peatlands decreased in the order Sumatra > Kalimantan > Papua > Peru (Table 1, Fig. 1). Most other human impacts also declined in the same order (Table 1). Although Peru as a whole had a higher population density than Papua, it had a lower population density in peatlands. Similarly, although Peru had a fairly high road density, comparable to Sumatra, it had the lowest road density in peatlands, comparable to Papua (Table 1, Fig. 2). Peru had the lowest area mapped as forest cover loss (Fig. 3), agriculture, tree

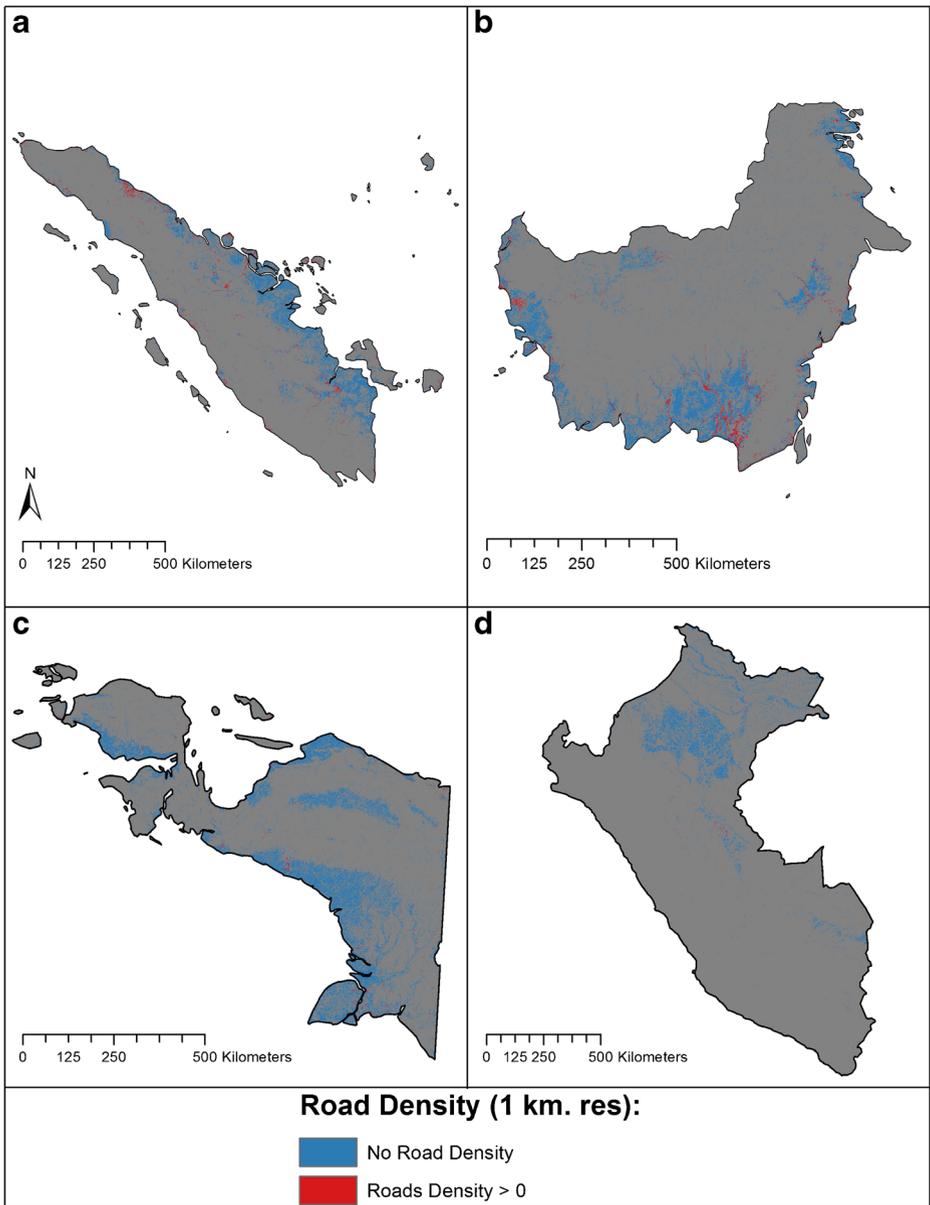


Fig. 2 Map of road density in peatlands of **a** Sumatra, **b** Kalimantan, **c** Papua, and **d** Peru

plantation, and oil palm plantation in both the entire country and in peatlands, similar to or lower than Papua and lower than Sumatra and Kalimantan (Table 1, Fig. 4). Peru had the lowest area burned in the period covered (Table 1, Fig. 5). Peru and Papua had higher percentages of total and peatland area under protected status when compared with Sumatra and Kalimantan (Fig. 6). Sumatra had the highest percentage of peatland area underlain by oil and gas deposits, with Peru being the next highest (Fig. 6).

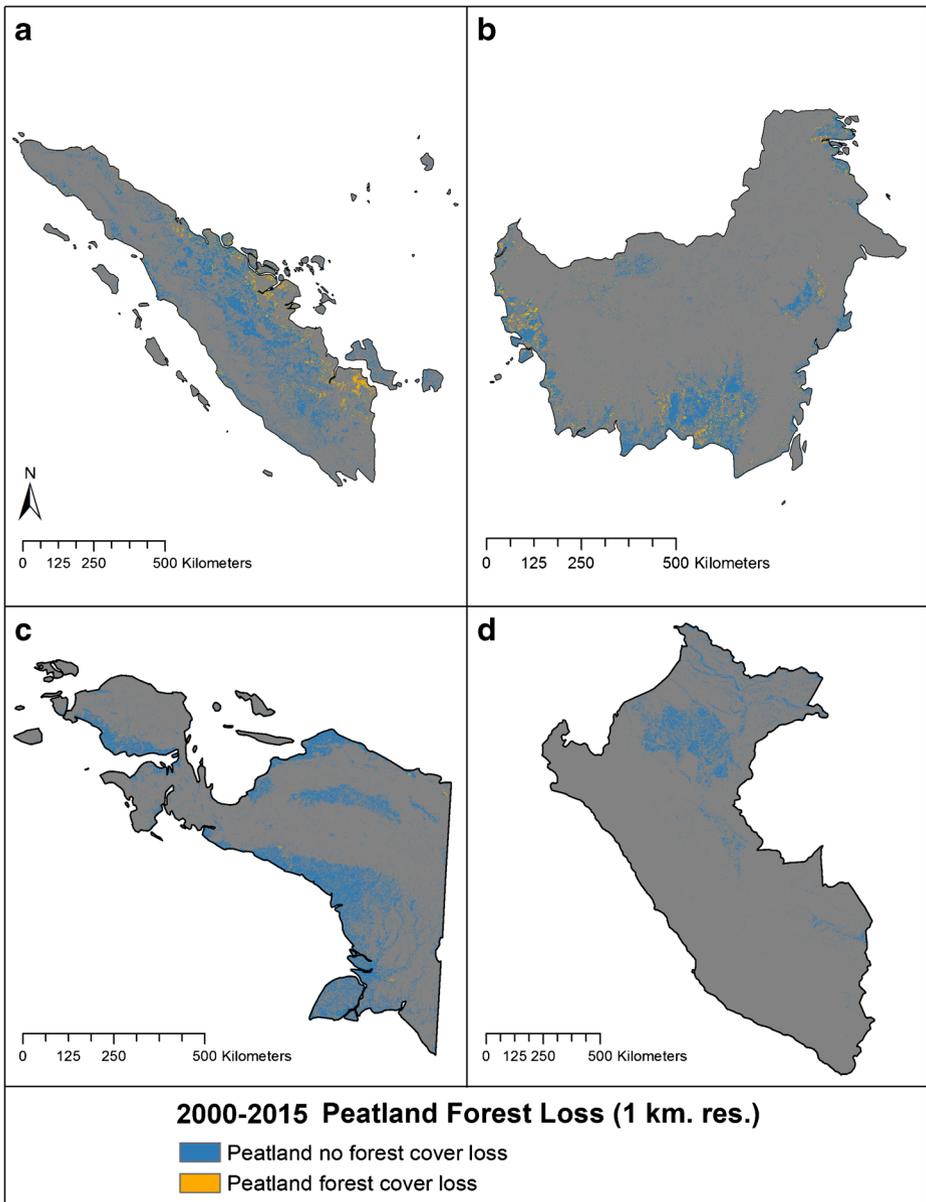


Fig. 3 Map of forest cover loss from 2000 to 2015 in peatlands of **a** Sumatra, **b** Kalimantan, **c** Papua, and **d** Peru. Some of the areas listed as “No forest cover loss” in Sumatra and Kalimantan were already deforested as of 2000. See Fig. 4 for total forest cover as of 2015

Population density in peatlands was predicted by population density in the region (Fig. 7a). Road density in the regions did not predict road density in peatlands, because Peru had lower than expected roads in peatlands based on overall number of roads in the country (Fig. 7b). The relationship between percent area categorized as agriculture, oil palm, tree plantation, total

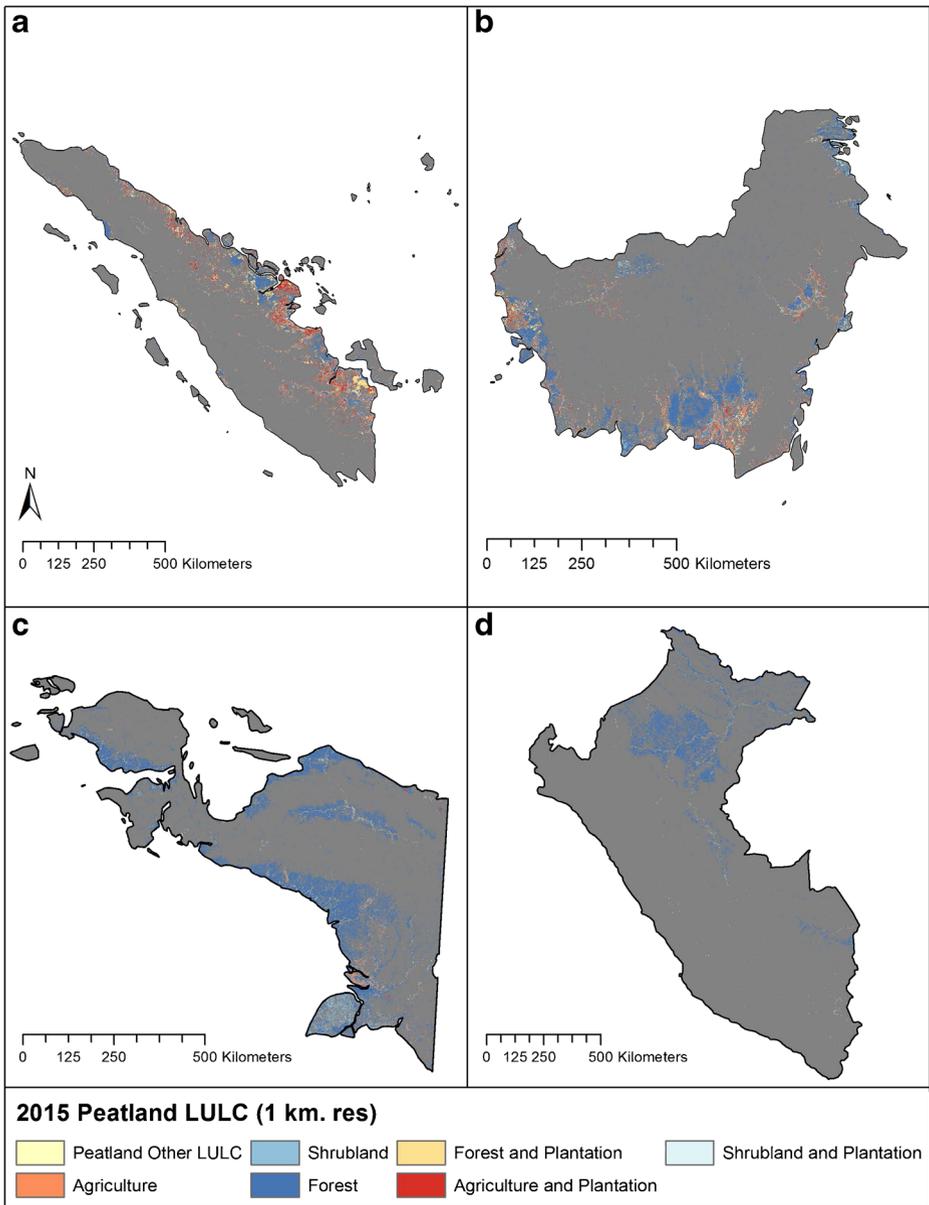


Fig. 4 Map of land use and land cover in peatlands of **a** Sumatra, **b** Kalimantan, **c** Papua, and **d** Peru This map is based on two sources, one LULC map that included forests, shrublands, and agriculture, and another for plantations. When they overlapped we created combined classes of forest and plantation, shrubland and plantation, or agriculture and plantation. The residual “other LULC and plantations” category was too sparse to map, so was mapped in the forest and plantation class. See methods for details on contribution of different formal cover classes to the mapped classes

plantation, deforested, and burned in the regions with their respective percent area in peatlands were all statistically significant (Fig. 7c–h), with Peru representing the low end of each

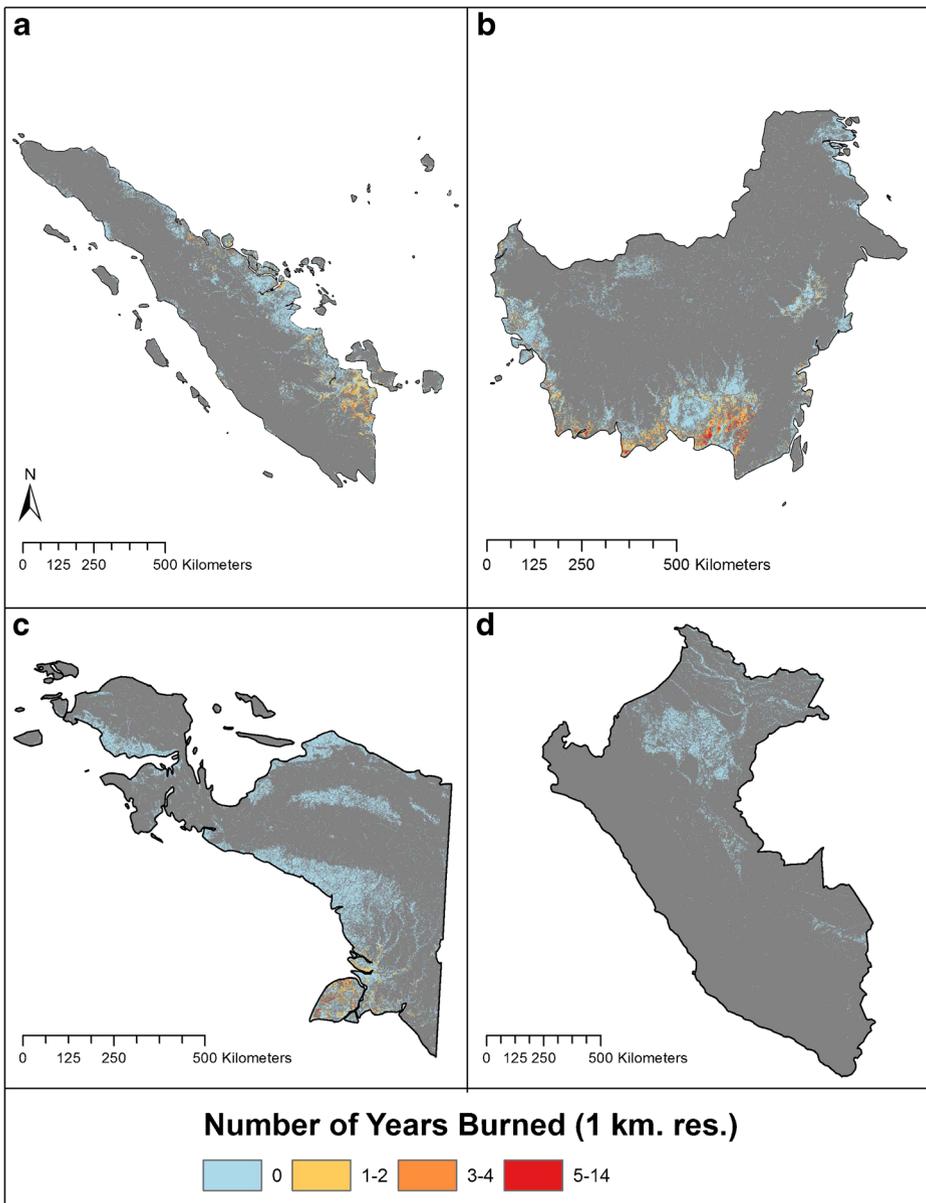


Fig. 5 Map of peatland burned area from 2001 to 2016 for **a** Sumatra, **b** Kalimantan, **c** Papua, and **d** Peru. The data were condensed into four burn frequency classes based on number of years burned, with 0 years burned representing peatland areas that did not burn during the measurement period

correlation, similar to or below Papua. The slope of the significant burned area line was very steep—whereas Peru had a lower percentage burned area in peatlands than in the country as whole, in the Indonesian regions peatlands burned at over twice the percent area of the regions as a whole. Although Papua and Peru had the highest percent protected area in both the regions

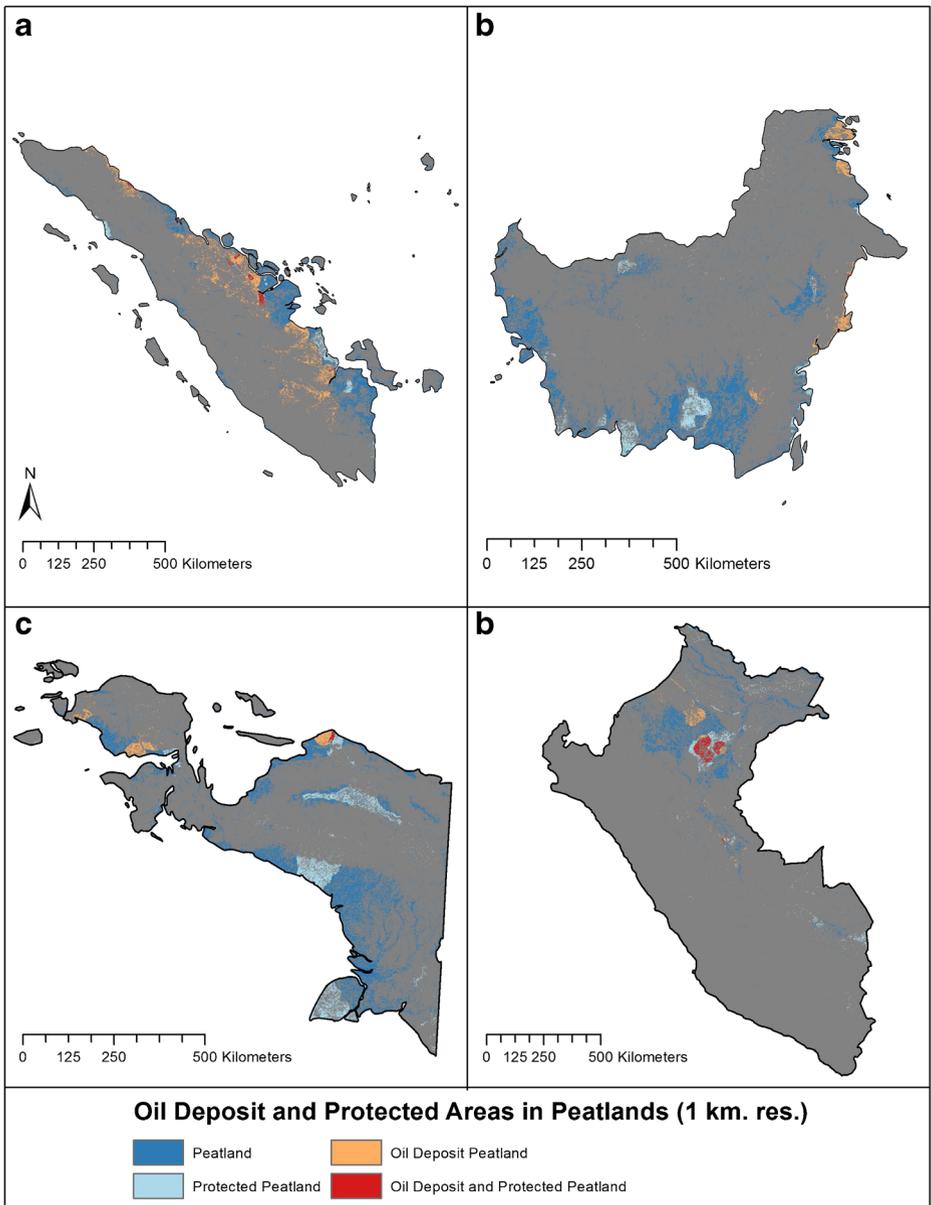


Fig. 6 Map of petroleum deposits, protected areas, and their intersection in peatlands of Sumatra, Kalimantan, Papua, and Peru

and in peatlands (Table 1), protected area in the regions did not predict protected area in peatlands (Fig. 7i).

Population density in peatlands predicted road density in peatlands (Fig. 8a). Similarly, population density in peatlands (Fig. 8) and road density in peatlands (Fig. 9) were significant predictors of peatland percent area in agriculture, oil palm, tree plantations, deforestation (all

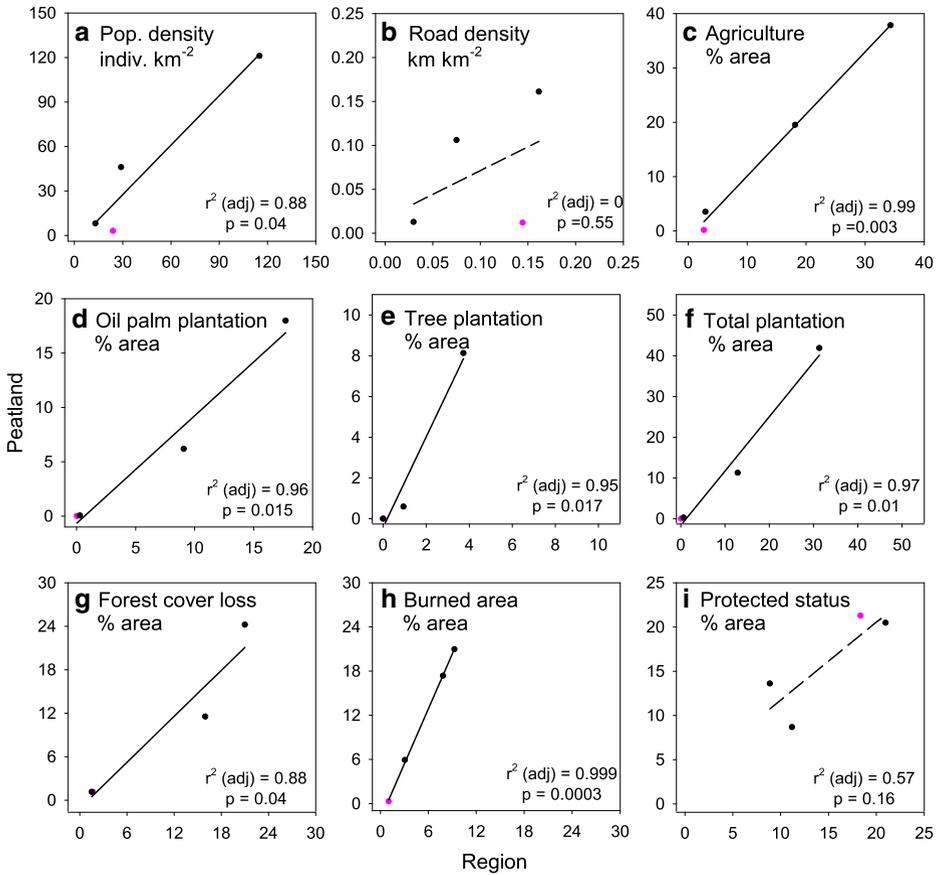


Fig. 7 Regression plots of predictor variables for the four study regions, calculated for the entire region (upland + peatland) vs. in peatlands only. Black symbols represent the three Indonesian regions, pink symbols represent Peru

positive slopes; Figs. 8b–e; 9b–c, e); forest cover and protected area (negative slopes; Figs. 8 and 9f, g); but not oil and gas deposits or burned area (Figs. 8 and 9h, i).

High fire years in Indonesia tended to occur during El Niño events (Fig. 10a). In contrast, high fire years in Peru occurred during 2005, 2007, and 2010, all of which were 1 year after El Niño events, and two of which (2005 and 2010) were regional drought years associated with warm NTS-SST anomalies (Fig. 10b).

4 Discussion

Our finding that population and road density in peatlands are very strong predictors of forest cover loss, agriculture, tree plantations, and oil palm plantations in peatlands reinforces the concept that one of the most important factors to be considered in conserving peatland integrity is policy and practices surrounding road access, population migration and associated agricultural activities in or near peatlands (Roucoux et al. 2017).

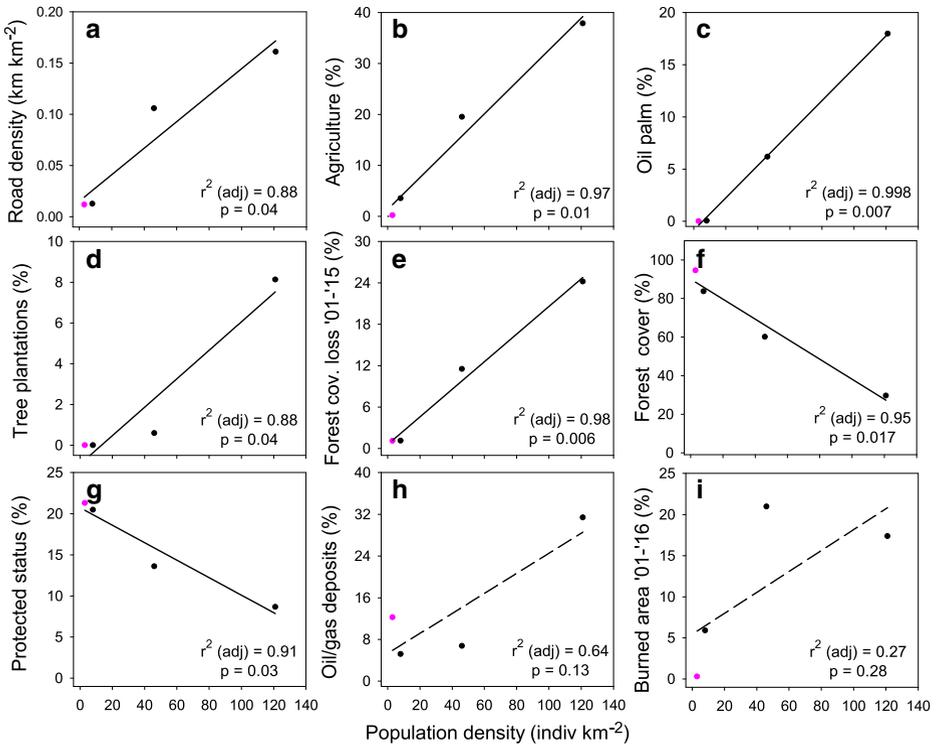


Fig. 8 Regression plots of population density in peatlands vs. other peatland human impact and conservation variables. Black symbols represent the three Indonesian regions, pink symbols represent Peru

4.1 Population and access

In Indonesia, the peat-forming processes favored formation of peatlands in coastal areas with very low relief (Dommain et al. 2010; Fig. 1a-c). Hence, population and road density in peatlands were similar to regional densities, reflecting the absence of major barriers between coastal population centers and peatlands. Most peatland-rich regions in Sumatra and Kalimantan are relatively accessible by road. In addition, because they are formed interfluentially, almost all peatlands in Indonesia are close to rivers which can serve as transport routes out of peatlands over relatively short distances. Finally, a large area of peatlands in Sumatra and Kalimantan is penetrated by a network of canals that are used as access routes for activities such as agriculture and logging operations (Jaenicke et al. 2010). Thus, multiple routes with rapid access to shipping by roads and to a lesser extent rivers (Lubis et al. 2005) favor economic exploitation.

In contrast, Peruvian peatlands have low population and road density despite the relatively high population in the country. This is a consequence of geographic isolation, because the peatlands formed in the Amazon basin on the east side of the Andes, away from the major coastal population centers. This geographic isolation helps explain why, in these interior lowland peatlands of Peru, both population and road density in peatlands are much lower than in the region, in contrast with Indonesia (Fig. 7a, b). As a result of this geographic barrier, there is presently only one major road linking the coastal population centers of Peru to the

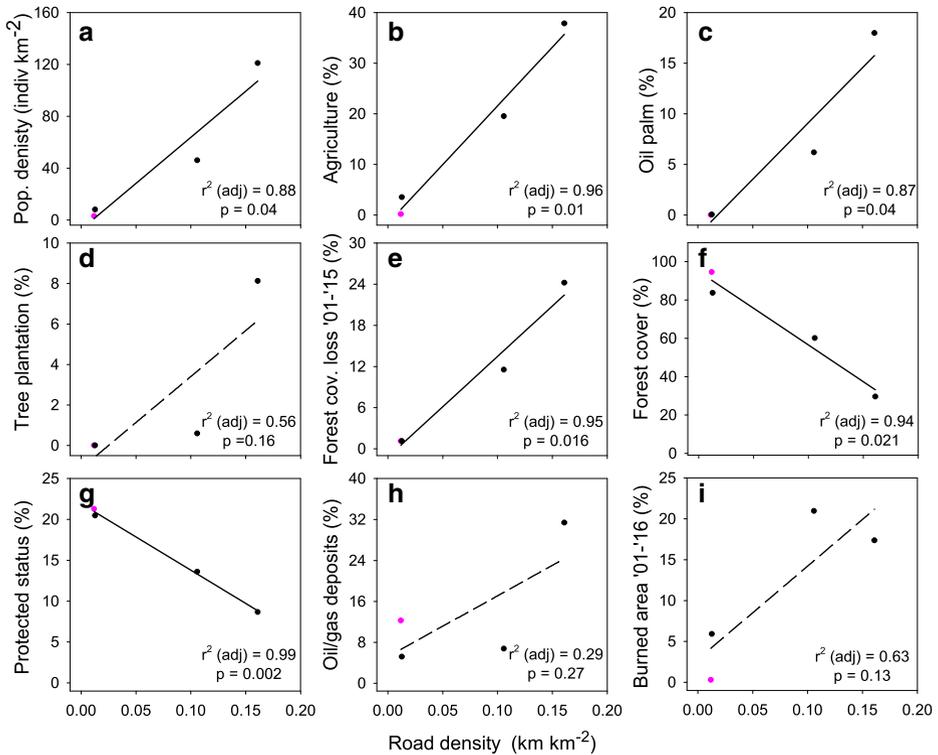


Fig. 9 Regression plots of road density in peatlands vs. other peatland human impact and conservation variables. Black symbols represent the three Indonesian regions, pink symbols represents Peru

margin of the large peatland expanse in the Pastaza-Marañon foreland basin in the province of Loreto. In Peru, as in Indonesia, there is access to large areas of peatlands by river as well. In areas where the distances to markets are relatively short and that are colonized by immigrant communities, rivers are predictors of deforestation (Bax et al. 2016). However, for much of the peatlands in the present study, the distances to markets along these river networks are quite long, which appears to increase transport costs and reduce riparian deforestation and other economic activity (Salonen et al. 2014). If these peatlands become more fully linked to the coast or other population centers via the national road network (see section 4.5), these distances will decline, which is likely to accelerate development and economic expansion along river corridors.

4.2 Peatland fires

Burned area in peatlands was not well-predicted by population or road density, suggesting other factors are more important in regulating area burned. Fire impacts were much lower in Peruvian peatlands than in Indonesian peatlands (Table 1; Fig. 10a). This may be in part because of the lower intra-annual variability in precipitation in northern Peru (Espinoza Villar et al. 2009), where the majority of Peruvian peatlands are located, compared with Indonesia. However, even within Indonesia, burned area was not well-predicted by population or road density, likely because climatic drivers interact with land use. For example, fire varies latitudinally in Indonesia (Aldrian and

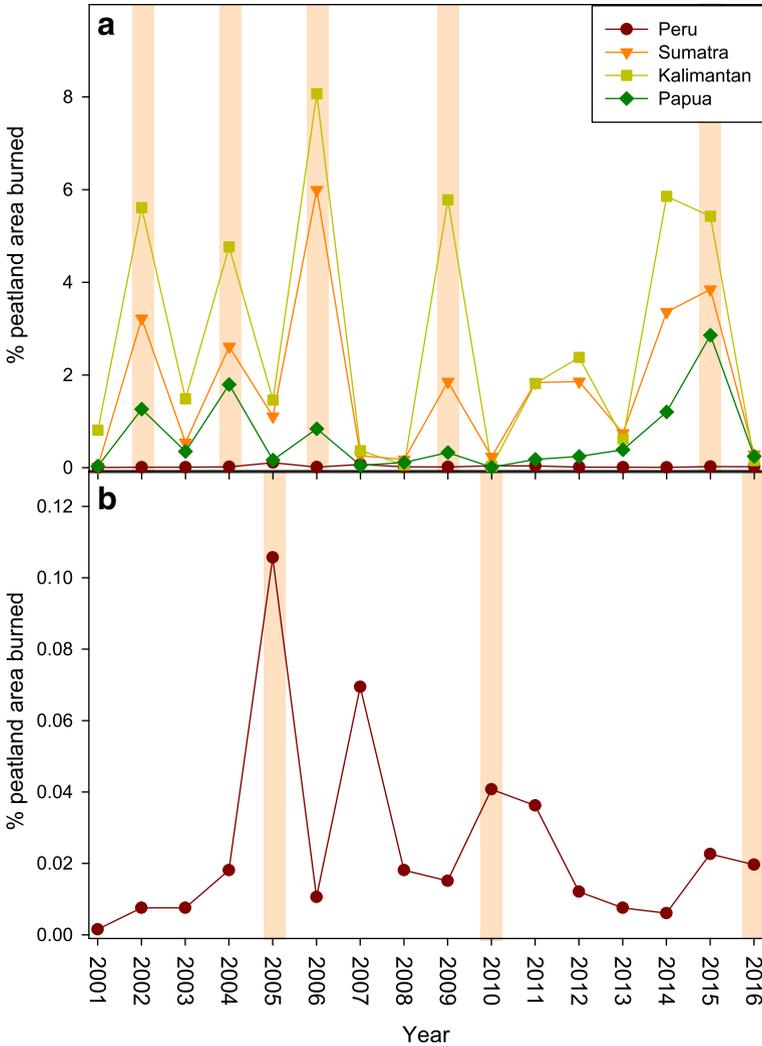


Fig. 10 Annual % peatland area burned **a** in the four regions and **b** in Peru alone. Vertical bars in **a** indicate El Niño events. Note the co-occurrence of El Niño and high fire years in all three regions of Indonesia, but not in Peru. Vertical bars in **b** indicate Tropical North Atlantic sea surface temperature anomalies, which were associated with drought years in the western Amazon basin and with two of the three highest fire years in the Peruvian Amazon. Burned area in Peru was ~70 times higher in the 2005 drought year than the lowest year, and over 5× average of other years

Susanto 2003; Fig. 5) and so climate may obscure land-use impacts on fire occurrence. Hence we might speculate that human activity increases the probability of fire initiation, but spatiotemporal climatic variation likely determines their spread.

Most Indonesian fires are anthropogenic, as fire is often used to clear lands for agriculture (Cochrane 2003). Historic land conversion patterns have dictated fire regimes in each episode of land development (Murdiyarso and Lebel 2007b). When occurring on peatlands, fires commonly consume large quantities of peat and convert it to greenhouse gases, and create noxious smoke that is a major public health hazard (e.g., Gaveau et al. 2014). This has led to

the recognition of the need for peatland fire mitigation via incentives and regulations (Murdiyarso and Lebel 2007a) and was one of the driving factors behind the recent formation of the Peatland Restoration Agency and the banning of the use of fire for peatland clearing (Hergoualc'h et al. 2017a).

Given the low fire frequency in Peru and generally lower seasonality, it is uncertain if Peruvian peatlands will be as strongly susceptible to fires if subjected to increased human population and economic activity. Reconstructions of Holocene fire histories suggest fire has been rare in northern Peru (Bush et al. 2008). Although current climate might constrain intensity of peatland fires in Peru, it is of concern that interannual variability in precipitation is significant in the Pastaza-Marañon Basin of northern Peru where most peatlands are located (Espinoza Villar et al. 2009), leading to the potential for severe fire during drought years, as in Indonesia, if other factors such as anthropogenic sources of ignition or drought increase (Bush et al. 2008). Temporal dynamics of fire in Amazonian peatlands indicate that unlike Indonesia, Peruvian Amazonian peatlands do not burn more during El Niño events (Fig. 10), because the western Amazon basin climate is also regulated by the NTA-SST, which is associated with droughts and enhanced fire activity in the Amazon (Fernandes et al. 2011; Chen et al. 2015). The drought years of 2005 and 2010 in the western Amazon basin (Espinoza et al. 2011; Marengo and Espinoza 2016) were two of the three highest fire years in Peruvian lowland peatlands in our period of measurement, with 2005 rates 70 times higher than the lowest year (2001) and over 5 times higher than the 15-year average (excluding 2005; Fig. 10 b). However, the pan-Amazonian drought year of 2016 did not lead to a high fire year in the Peruvian Amazon, perhaps because in contrast with 2005 and 2010 the drought was weak over Peru during the dry season months of June–August (Erfanian et al. 2017) that seem to be strongest predictors of fire (Fernandes et al. 2011).

This indicates that periodic drought could favor major fire-mediated losses of C from Peruvian Amazon peatlands, and could interact with any anthropogenic increases in fire ignition, fuels, or drainage of peat, as seen in Indonesia (e.g., Siegert et al. 2001). Despite the fact that our data suggest road and population density are not sufficient to explain burned area *among* regions, when fires do occur in the Amazon of Peru they are, as in Indonesia, most strongly associated with both drought and human sources of ignition, as indicated by their proximity to rivers, roads, forest edges, and croplands (Uriarte et al. 2012; Armenteras et al. 2017). Most of the fires in Peru appear to occur to the south of the Pastaza Marañon Basin in the Province of Ucayali, where predisposing factors of greater seasonality, higher population, higher agricultural activity, and higher deforestation likely contribute to higher rates of burning. Thus, as in Indonesia, increased anthropogenic activity, in combination with drought years, could drive increased peatland burning *within* the Peruvian peatlands.

One practice that has greatly increased the extent of fires in Indonesia is peatland drainage by ditching and canals (Page et al. 2002). Domed tropical peatlands are especially susceptible to drainage by ditching (Baird et al. 2017). Therefore, adoption of similar practices in Peru would also greatly increase the likelihood of fire impacts, especially in domed forests. Peruvian peatlands are a mixture of domed and undomed peatlands (Lähteenoja et al. 2009), and so likely vary in their susceptibility to drainage. Although peatland drainage is not common presently in the Peruvian Amazon, we are not aware of any prohibition of such activities (see Section 4.5).

4.3 Peatlands and oil and gas development

Oil and gas extraction is permitted from under peatlands in both Indonesia and Peru. In Indonesia, most of the oil production is in Central Sumatra, but there are also proven reserves

of oil and natural gas in South Sumatra, East Kalimantan and West Papua (Hasan et al. 2012). Known oil and gas deposits as of 2009 were found in similar proportions under Indonesian and Peruvian peatlands. The deposits listed in the global dataset used in the present study underestimated the area of land included in oil and gas leases, as new deposits have been discovered, and the area of the Peruvian Amazon under oil and gas lease is expanding (Finer et al. 2008; Finer et al. 2015; Roucoux et al. 2017). In Peru, almost the entire Amazon lowlands outside of selected protected areas are in concessions that have been leased or permitted for leasing for oil and gas production (Finer et al. 2008; Finer and Orta-Martínez 2010; Roucoux et al. 2017). There is active oil and gas extraction in the northeastern part of the Pastaza Marañon Basin in Peru (Finer et al. 2015).

Despite extensive oil and gas deposits in Indonesia, the major driver of human activity in peatlands appears to have been road development (Sabandar 2004) in parallel with planned and unplanned transmigration from Java which accelerated population growth on outlying islands (Leinbach 1989) followed by widespread deforestation and agriculture (Miettinen et al. 2012), and was not evidently linked to oil and gas activities. There are clear local exceptions, e.g., some road networks in Papua appear to have been built specifically for access to oil wells (Sabandar 2004). Thus, although we cannot rule out the possibility that this occurred more widely than has been documented, there is no evidence that it has been a major driver of peatland degradation.

In contrast, the relatively large area of Peruvian Amazon peatlands underlain by oil and gas deposits, in combination with low population density, creates the potential for a fundamentally different development trajectory, i.e., the potential for expanding development of these deposits to stimulate migrant population incursions into the region, especially if linked to continuous road networks (Finer et al. 2008; Roucoux et al. 2017). One approach that minimizes such incursions is roadless oil and gas extraction projects, also known as the “offshore inland development model”, which utilize alternative forms of transportation such as helicopters to access drilling sites and eliminates access roads associated with pipelines (Finer et al. 2015). Finer et al. (2013) evaluated and found such practices feasible for the province of Loreto which contains the majority of peatland area.

4.4 Peatland land-use change

Consistent with our hypotheses, population and road density in peatlands were excellent predictors of forest cover loss and agricultural activity in the four regions studied. Although our dataset is small and correlation does not prove causation, there is a logical causal linkage between the two, as well as a strong spatiotemporal association found in many regions between road access and deforestation, increase in agricultural land use, and other land use changes in the Brazilian Amazon and elsewhere (e.g., Laurance et al. 2002, 2009, 2014).

There is good reason to believe that if the Peruvian Amazon peatlands were more extensively penetrated by roads they would experience a similar fate to Indonesia. For example, the areas of Peru with greatest peatland land-use change are those smaller areas of peatland in closer proximity to road-associated population centers in the south (e.g., Fig. 11). Deforestation is occurring in areas of the Amazon accessible by roads (Naughton-Treves 2004), but is presently rare in the roadless area of Amazonian peatlands in the Pastaza-Marañon Basin, except for small-scale clearing adjacent to rivers. There are very few legal logging concessions that intersect with peatlands (Finer et al. 2014). Similarly, oil palm plantations have expanded on uplands in the last decades (Gutiérrez-Vélez et al. 2011), but

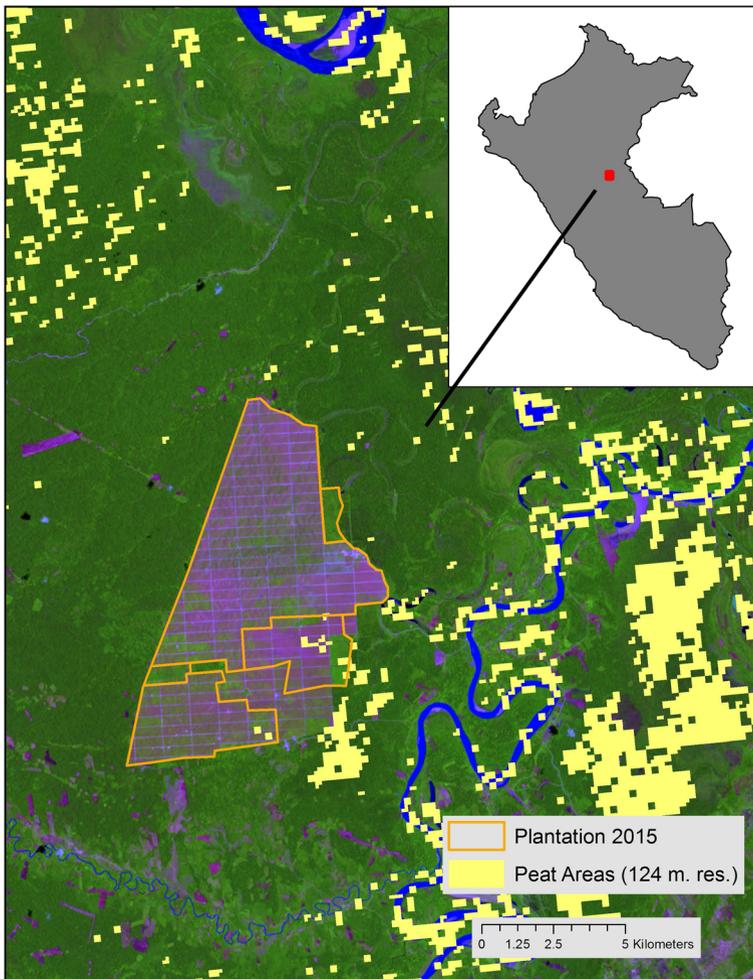


Fig. 11 Landsat image of an area ~35 km NW of Pucallpa, Peru, overlain with boundaries of an oil palm plantation (from Transparent World) and the peatland layer (from Gumbricht et al. 2017). The purple areas are a false color representation of cleared areas

at present there is no expansion into peatland areas, except possibly in some smaller marginal peatlands mapped by Gumbricht et al. (2017) (Fig. 11). Although it is likely that improved access would lead to increased deforestation and agricultural expansion, the hydrology of Peruvian peatlands located along the major Amazon tributaries might constrain the types of viable crops in the expansion of agriculture onto peatlands. Ombrotrophic domed peatlands are the dominant class of peatlands in much of Indonesia (Dommain et al. 2010), and so their water tables can be lowered effectively by ditching, which permits survival of perennial plants intolerant of prolonged flooding while simultaneously increasing fire probability and carbon loss. In Peru, longer or deeper floods than typical of Indonesia's ombrotrophic peatlands might reduce the suitability of the widespread minerotrophic peatlands for oil palm, which is sensitive to prolonged high water tables, especially when immature (Henson et al. 2008). This might hydrologically limit oil palm to ombrotrophic peatlands, which in Peru are typically

dominated by pole forests that do not flood as regularly as minerotrophic peatlands, and make up only about 10% of peatlands in the Pastaza Marañon Basin (Draper et al. 2014). Relatively high hydraulic conductivity of these peatlands could render them susceptible to drainage by ditching, although because of methodological differences it is unclear if conductivity is as fast in Peru as in Indonesia (Kelly et al. 2014).

That is not to say that even the roadless regions of the Peruvian Amazon peatlands are not already changing. Even in the absence of road incursion, forest degradation not captured in our analysis is occurring along river corridors (Hergoualc'h et al. 2017b). This involves both selective logging (Kvist and Nebel 2001) and selective cutting of female *Mauritia flexuosa* (Aguaje) palms for fruit harvest (Horn et al. 2012), which could endanger this species if not curtailed. Given its widespread dominance in minerotrophic peatlands and seasonally flooded forests (e.g., Draper et al. 2014), its effect on microtopography, and its abundant pneumatophorous roots (del Aguila-Pasquel 2017), *M. flexuosa* can be considered a foundation species, i.e., “a single species that defines much of the structure of a community by creating locally stable conditions for other species, and by modulating and stabilizing fundamental ecosystem processes” (Dayton 1972 as cited in Ellison et al. 2005). Hence changes to its abundance could have significant ecosystem consequences. Efforts are underway to encourage use of sustainable harvesting practices, with mixed success (Manzi and Coomes 2009). Yet these changes are subtle compared with the widespread land use change in Indonesian peatlands.

When comparing the four regions, peatland land-cover change declined dramatically with decreasing population, but even in Peru there was still a small amount of land cover change detected. For example, the present paper estimates the area of oil palm in Peruvian peatlands to be only ~ 7 km², all in areas mapped as smaller peatlands nearer population centers and outside of the Pastaza Marañon Basin (see, e.g., Fig. 11). It is not clear how much of this was due to misattribution vs. very low-level natural disturbance, deforestation and plantation activity on peatlands. The global peatland map by Gumbrecht et al. (2017) which was used in this study has not been extensively validated at the country level via ground-truthing, and so although the major conclusions about land use patterns among countries and regions are likely correct (see comparison with published estimate in Online Resource 2) the estimated area of land-use change on peatlands should be considered provisional.

4.5 Government policy

Although population and road access were excellent predictors of peatland degradation, we do not mean to imply an immutable causal relationship. Clearly the trajectory of development and other impacts can be modulated by other factors, including government policy. Peru could differ from Indonesia in development of peatland forests via government policies that affect rate of population movement, road building, peatland drainage, and land-use change in the most important areas of Amazonian peatlands.

Historically, forest exploitation and agricultural development on Indonesian peatlands was favored by government policy. In 1967 almost 75% of the country was designated as forested areas, the exploitation of which was promoted to balance budget deficits (Brockhaus et al. 2012). Around that time, spontaneous transmigration from Java led to population pressures on Sumatran peatlands (Page et al. 2009). In the 1980s, official transmigration policies led to increased pressure from agriculture and illegal logging on peatlands in both Sumatra and Kalimantan. In addition, government-led drainage of peatlands was undertaken, the most

extensive and best-known example being the Mega Rice project, which drained large areas in central Kalimantan, with extreme ecological consequences (Page et al. 2009). Following those projects, government policy favored planting of oil palm and other commercial tree species on peatlands. In 2009, the Indonesian government permitted development of oil palm plantations on peatlands that were < 3 m deep (Murdiyarso et al. 2010), amounting to approximately 80% of Indonesian peatlands (Warren et al. 2017).

Recently, recognizing the negative environmental and public health consequences of drainage and land use change in peatlands, the Indonesian government has begun a major effort to reverse these trends. It initiated a moratorium on conversion of peatlands to agriculture, and a ban on the use of fire for land clearing. Peatland conservation has become one of the major foci of REDD+ (Brockhaus et al. 2012), and peatland restoration has become a government priority, with the undertaking of an ambitious restoration effort led by the new Peatland Restoration Agency (Hergoualc'h et al. 2017a).

Peru seems to be following a similar set of development policies to those of Indonesia in the late twentieth century in regard to the lack of explicit legal or regulatory protection of peatlands from development. Other than in formally protected areas (see Section 4.6), policies addressing agricultural development of the Amazon do not explicitly protect peatlands. Of particular importance, since 2000 the Peruvian government has promoted oil palm expansion, with a series of policies in place in support its development (Peru, Ministerio de Agricultura 2012, 2016). Plantation coverage as of 2015 was almost 580 km² (Potter 2015). In 2012, 11,350 km² of Peru were deemed suitable for oil palm, of which 1800 km² were marked out in the peat-rich Pastaza-Tigre region (Peru, Ministerio de Agricultura 2012). Peru, Ministerio de Agricultura (2012) further notes that flooded lands can have excellent yields of oil palm when drained, with no mention of negative environmental impacts, nor of the special properties or sensitivity of peatlands to drainage. If peatlands are thus identified as areas suitable for plantations, Peru might follow the Indonesian model in the absence of alternative economic and development models that are sustainable (Roucoux et al. 2017). In the new *National Plan for the Sustainable Development of Oil Palm in Peru 2016–2025* (Peru, Ministerio de Agricultura 2016), wetlands are not explicitly excluded as areas for oil palm unless they are part of a Ramsar site. This latter category does include a large portion of the peatlands in the Peruvian Amazon, especially the Pacaya-Samiria National Reserve, and the Abanico del Pastaza, a 3,827,329 ha site which is not formally protected. Although these and other criteria, notably the criterion of zero deforestation for plantations, presently limit most appropriate areas for plantations to outside of the large expanse of peatlands in the Pastaza Marañon Basin, peatlands are not explicitly mentioned in the plan, so it is not clear if these criteria will provide long-term protection to peatlands, especially those outside of Ramsar sites or other protected areas. This is surprising because Glave and Vergara (2016), which was the source of the environmental criteria used in Peru, Ministerio de Agricultura (2016), cite as the basis for their environmental criteria guidelines developed with heavy reliance on experience in Indonesia (RSB 2010; Gingold et al. 2012; RSPO 2013). These previous efforts include criteria to avoid peat soils for new plantings, yet this criterion was excluded from the criteria of Glave and Vergara (2016). If the challenge preventing inclusion was absence of adequate maps of peatland distribution, then mapping efforts cited here could serve as a suitable basis, e.g., Draper et al. (2014) combined with Gumbrecht et al. (2017) for areas not covered by Draper et al. (2014). Therefore, if the Government of Peru wishes to align with previous efforts delineating best practices in sustainable oil palm, information on peatlands can now be incorporated into a revised plan.

This plan did recognize that “it is relevant to complete the design of a nationally appropriate measure of mitigation (NAMA) for its implementation, which will ensure that palm farming provides long-term sustainability for producers and also meets with the global commitments linked to the reduction of GHG emissions” (Peru, Ministerio de Agricultura 2016, P. 59; our translation). NAMAs are “any action that reduces emissions in developing countries and is prepared under the umbrella of a national governmental initiative.” (<http://unfccc.int/focus/mitigation/items/7172.php>). A NAMA on this topic is under development (K. Hergoualc’h, personal communication), and so there is the potential to explicitly exclude oil palm plantations from peatlands in the context of greenhouse gas mitigation, which would align these activities with the Roundtable on Sustainable Palm Oil (RSPO 2013) and Roundtable on Sustainable Biomaterials (RSB 2010) criteria regarding preservation of peatlands and high carbon stocks.

Similar to agricultural policy, we did not find that the environmental sensitivities of peatlands were included in road planning decisions. Recent legislation indicates that the Peruvian government plans to expand road infrastructure in proximity to the largest expanse of intact peatlands in the country, declaring the construction of the Iquitos-Saramiriza highway along the entire northern edge of the largest blocks of peatlands in the Peruvian Amazon (the Abanico del Pastaza area, the largest Ramsar site in the Peruvian Amazon), to be of “public necessity and national interest” (Peru, Congreso de la Republica 2017). If this project goes forward, it would link this entire northern Amazonian region to the coastal road networks, which could lead to either intentional or illegal expansion of deforestation and agricultural activities into these peatlands, following the Indonesian pattern of peatland degradation via population migration, deforestation and intensified land uses. In a global analysis delineating a rational plan for global road development, this region was identified as having high environmental values (without even considering peat carbon stocks) and low agricultural value, making it a high priority for maintenance as a roadless area (Laurance et al. 2014).

Although Peru is working on policies relating to both greenhouse gas emissions and wetland conservation, these policies may not address peatlands’ unique sensitivity to disturbance from drainage and intensive land use. For example, the 2015 *National Strategy for Wetlands* (Peru, Ministerio del Ambiente 2015a) covers many functions of wetlands, but does not mention the unique carbon sequestration and other properties (nor even the existence) of peatlands as a wetland type, nor does it mention the ability of any Peruvian wetlands to store carbon or regulate greenhouse gases. Furthermore, the new *Peruvian National Strategy for Forest and Climate Change* seems to encourage peatland drainage and conversion to agriculture, stating that “providing advice to implement... technologies...to drain wetlands...can reduce the migration of the indigenous and peasant population to fertile soils and/or non-deforested areas” (Peru, Ministerio del Ambiente 2016a; our translation).

However, it appears that there is a growing recognition of unique properties of peatlands within the Ministry of Environment, because C storage was recognized as a justification in recent national government decision-making on protected area status for Amazonian peatlands. The founding documents for the recently created Yaguas National Park recognized the importance of peatlands in storing C and regulating greenhouse gases as one of the important features protected by the park (Peru, Ministerio del Ambiente 2016b).

Thus, to our knowledge, at the national policy level the unique carbon-sequestering properties of peatlands and their sensitivity to intensive land uses are only recognized within the specific framework of protected areas designation. There is a standing National Committee on Wetlands (Comité Nacional de Humedales) in Peru that was initiated in 2013 (Peru, El

Presidente de la República 2013). This committee is represented by all the major land management and environmental agencies and is tasked to, among other things: “Review and propose the modification and adaptation of the current legal framework, in order to improve the performance of environmental management for the conservation of wetlands.” (Peru, El Presidente de la República 2013; our translation). Should the Peruvian Government wish to increase protections for peatlands, this would seem to be an appropriate place to review the unique environmental services and sensitivities of peatlands in order to make recommendations for any needed changes into the *National Strategy for Wetlands* or other policies that require coordination among multiple parties across the government.

The absence of reference to peatlands in most relevant policies is surprising, given the growing body of literature available to decision makers on the important role of tropical peatland ecosystems in long-term climate change mitigation. As noted earlier, studies have quantified large C stocks (Lähteenoja et al. 2012; Draper et al. 2014) and other conservation values and ecosystem services in Amazonian peat swamps (Roucoux et al. 2017) that have, largely by isolation, been protected from extensive degradation. In parallel, working in Indonesia and other parts of Southeast Asia scientists have documented the negative environmental consequences of peatland land use change (Rieley et al. 2008; Hergoualc’h and Verchot 2011; Miettinen et al. 2012, 2016, 2017; Graham et al. 2017). Given our current understanding of the unique ecosystem services of peatlands, especially their potential to store vast quantities of C and to lose that C to the atmosphere and accelerate climate change if disturbed by drainage, logging, plantations, or other agricultural activities (Hergoualc’h and Verchot 2011; Miettinen et al. 2017), it is clear that Peru faces both serious challenges regarding how best to manage development in peatlands, and opportunities to learn from the Indonesian experience. Specifically, national and provincial governments’ decisions on whether and how to protect peatland C reserves when drafting new laws and regulations for the appropriate use of the tens of thousands of square kilometers of peatlands in Peru could have a major impact on the magnitude of ecological disturbance.

Direct exchanges between Peruvian government policy makers and experts on peatland ecosystem science and management would facilitate transmission of lessons learned from Indonesia and other countries. One option would be to form an advisory council on peatland ecosystem science and management that could advise the National Committee on Wetlands and other appropriate bodies. This body, as well as individual government agencies, could also obtain access to expertise via exchanges of knowledge and information on peatlands issues across pan-tropical countries, e.g., in the Global Landscape Forum (<https://archive.globallandscapesforum.org/peatlands/>) and the Global Peatlands Initiative (<https://www.globalpeatlands.org/>; Crump 2017), as well as via direct exchanges with the Peatland Restoration Agency of Indonesia (BRG).

4.6 Protected areas, indigenous territories, and other conservation measures

According to our analysis, Peru has approximately 50% more of its peatlands under formal conservation status than Indonesia (21 vs. 14%). This in itself could have a significant limiting influence on the peatland development trajectory for Peru, assuming that these protections are enforced. The peatland area in land conservation decreased significantly across the four study regions as a function of population density, suggesting it might be politically easier to establish protected areas when there are fewer competing land use pressures, consistent with previous findings of a negative relationship between human population density and protected area size (Chown et al.

2003). Whatever the cause, Peru has established a network of formal protected areas in the Amazon basin that appear to reduce deforestation (Oliveira et al. 2007). Most notable in the context of peatland conservation is the 21,000 km² Pacaya-Samiria National Reserve with a high density of peatlands in the Pastaza Marañon Basin (Roucoux et al. 2017), but there are many other reserves with smaller areas of mapped peatland (e.g., Yaguas National Park, Bahuaja Sonense National Park, Cordillera Azul National Park, El Sira Communal Reserve, Amaraakeri Communal Reserve, Huimeki Communal Reserve, Pukakuru National Reserve, Allpahuayo Michana National Reserve, and Alto Nanay-Pintuyacu Chambriira Regional Conservation Area). Their mapped peatlands are based on the global Gumbrecht et al. (2017) map, and therefore require further validation. Assuming mapped areas of peatlands are reasonably accurate, this network of sites, if properly managed, will form an important component of long-term peatland conservation in Peru. Given peatlands' limited potential for other sustainable uses, expansion of protected areas is a viable mechanism for minimizing peatland losses, and is still feasible given continued low population density in large expanses of peatlands.

Although indigenous territories were not included in our GIS analysis of protected lands because they are not considered formal protected lands, land tenure policy can have a significant impact on land use, and hence is worth comparing among regions. In Indonesia, land tenure was historically governed by a system of customary tenure (*adat*). This customary tenure was largely ignored by national forestry policies, under which approximately 70% of the territory was designated as state forest land (Krishna et al. 2017). Private land title, which can also be acquired by a formal titling process, was granted to transmigrants from Java, often at the expense of indigenous populations (Krishna et al. 2017). As customary rights are only available to indigenous populations, this sets up a conflict between customary and formal land tenure (Krishna et al. 2017), and leads to a high rate of tenure insecurity (Sunderlin et al. 2014). This insecurity is amplified by governmental resistance to recognition of customary tenure to forest lands (Sunderlin et al. 2014). Illegal appropriation and clearing of state forest land is one mechanism by which local indigenous populations obtain de facto tenure for agriculture, leading to deforestation and land use change (Krishna et al. 2017). Hence there are no incentives in this system of tenure for indigenous populations to retain intact forests.

In Peru, there is a stronger tradition of formal assignment of land tenure to indigenous communities (Blackman et al. 2017) than in Indonesia. Significant areas of forested land have been placed in indigenous territories which may provide some protection against deforestation and forest disturbance (Oliveira et al. 2007, Blackman et al. 2017). Blackman et al. (2017) found that titling of indigenous communities in the Peruvian Amazon significantly reduced clearing and disturbance, at least in the short term. It is not clear whether this effect would continue longer-term or under different socioeconomic conditions, or is limited to indigenous communities that are not reliant on permanent forest clearing for livelihoods. For example, it is possible that greater population and agricultural pressures such as are seen in Indonesia would remove or reduce the protective effect of indigenous land title. Additionally, there is debate as to whether indigenous territories are as effective as formal protected areas for forest conservation. Indigenous territories near roads have been found in one study to be more susceptible to deforestation and disturbance than formal protected areas (Oliveira et al. 2007). In contrast, a recent study found that after controlling for other factors, indigenous territories were more effective than protected areas in avoiding deforestation and degradation (Schleicher et al. 2017). In summary, Peru is already on a very different indigenous land tenure trajectory from Indonesia, and expansion of this form of governance might slow deforestation and degradation on peatlands.

However, even protected areas and indigenous territories are susceptible to illegal logging and other impacts, so understanding factors that influence enforcement of protection is critical. For example, the majority of the protected areas of lowland forest in Kalimantan were deforested by the early 2000s (Curran et al. 2004). Protected areas have been more effective at minimizing deforestation in other countries, including Peru (Oliveira et al. 2007) and Costa Rica (Sánchez-Azofeifa et al. 2003). In fact, the large majority of tropical protected areas seem to have a net positive effect on amount of natural vegetation (Bruner et al. 2001; Joppa and Pfaff 2011). Curran et al. (2004) attribute the high rate of deforestation in Indonesian protected areas to illegal logging that occurred after legal government concessions were exhausted, which was enhanced by conversion of unprotected lands to oil palm and decentralized regulation of land use. In Peru, legal logging concessions have also been associated with illegal logging activities outside of concession boundaries (Finer et al. 2014). Therefore, one of the key factors in maintaining peatland integrity in protected areas would be effective management of logging concessions outside of protected areas, and adequate resource allocation for effective management of the protected areas themselves (Watson et al. 2014), especially if and when pressures increase.

There is a growing recognition of the need to consider peatland C storage and accumulation as part of systems of C credits (Pearce 2007; Dunn and Freeman 2011; Tanneberger and Wichtmann 2011; Morel and Morel 2012). Given the especially high belowground C density of these ecosystems, there is high potential for C-based conservation, e.g., via REDD+ (Yamamoto and Takeuchi 2016; Graham et al. 2017) and the Green Climate Fund (Roucoux et al. 2017). The success of these efforts depends on complex global, national, and regional socioeconomic factors that drive both adoption and impacts of these initiatives on climate and people, fueling a continued debate about how to improve the efficacy and equity of these initiatives (e.g., McAfee 2016; Angelsen et al. 2017). For example, in Sumatra and Kalimantan, the high population pressure, competing economic interests, and land tenure issues complicate efforts such as REDD+ (Resosudarmo et al. 2014; Sunderlin et al. 2014). In Peru, given lower population density and associated lower economic activity, combined with ongoing efforts to address indigenous rights (White 2014), these efforts might have a higher probability of success.

Efforts to value the carbon sequestration in peatlands rely on our ability to estimate the greenhouse gas consequences of peatland drainage and land use change. Previous studies, largely in Indonesia and neighboring Southeast Asia, have led to estimates of the greenhouse gas impacts of tropical peatland drainage combined with other land use changes. These form the scientific basis for IPCC guidelines that provide default (Tier 1) CO₂ emissions factors from oxidation and burning of drained tropical peatlands under a variety of land uses (Dröslér et al. 2014). These range from 5.3 Mg CO₂-C ha⁻¹ yr⁻¹ for drained forest, to as high as 20 Mg CO₂-C ha⁻¹ yr⁻¹ for drained short rotation tree plantations. These can be refined (Tier 2 and 3) with more resolved country-specific data and/or models. Default (Tier 1) values for additional organic matter consumption from wildfires on tropical peatlands are 353 Mg dry matter ha⁻¹ with annual average losses dependent on return interval of fires. Estimates can also be derived for losses via dissolved organic carbon in streams and rivers, as well as emissions of other greenhouse gases. Tier 1 estimates could be used as a starting point in Peru to develop cost-benefit analyses of different land use scenarios, but it is good practice to develop Tier 2 and 3 estimates for the unique conditions encountered in Peruvian peatlands.

5 Conclusions

Because Peruvian lowland peatlands are at present sparsely populated and relatively undeveloped, there is an opportunity to follow a different trajectory of peatland development from Indonesia. We

Table 2 Candidate policy options that would align with sustainable management and development of peatlands in Peru, including enabling conditions, and key next steps

Candidate policy option	Enabling conditions needed for policy to be effective	Key next steps needed to assess option
1. New or revised comprehensive legislation to protect peatland functions and services	Coordination among land-management agencies; recognition of value of peatlands	Charge National Committee on Wetlands to lead synthesis of state of tropical peatland science; development impacts on greenhouse gases, public health, etc.; and policy options
2. Rational road network planning that includes environmental impacts	Planning infrastructure. Institutional ability to enact road planning process	Identify areas that should remain roadless based on a rational planning process, e.g., Laurance et al. (2014). Include peatlands in decision process
3. Oil & gas exploration via roadless development in sensitive areas	Regulatory authority or corporate commitments	Based on #1 and 2 above, identify key regions for implementation of roadless approach
4. Quantify value of carbon and other ecosystem services in peatlands & reflect those values in land use decision-making	Scientific expertise available to evaluate peatland ecosystem services, emissions.	Estimates of C accumulation in intact peatlands and greenhouse gas emissions from different land uses using default (Tier 1) factors. Development and application of higher tier (2,3) approaches
5. Incorporate fire risk into the peatland development planning process.	Planning infrastructure, adequate models, information to parameterize models	Experimental and modeling approaches to assess current and future fire risk, environmental and public health consequences
6. Severely limit or ban drainage of peatlands for forestry and agriculture	Cross-talk between land use planning agencies	Summarize literature on drainage impacts on peatland ecosystem function, emissions, as basis for regulations
7. Include unique sensitivities of peatland ecosystem services in agricultural and forestry planning	Adequate assessment under #4 & #5 above	Apply Tier 1 (2,3) estimates of emissions and different scenarios of carbon valuation from #4 to peatland land use planning process
8. Encourage development of markets for sustainable products from peatlands	Sustainability certification for peatland land use, knowledge of products	Identify potential certifiers. Review literature on current products, markets, strategies. Avoid perverse incentives, e.g. for road development
9. Improve enforcement of laws against illegal logging or other degradation of forested peatlands	Funds to allocate toward enforcement	Evaluate resource allocation to, and effectiveness of, enforcement in Peru. Perform risk analysis. Base staffing on risk analysis
10. Protected Area (PA) management for peatland conservation and to prevent illegal logging or extraction, etc.	Funds to allocate toward PAs	Evaluate resource allocation to PA management. Perform risk analysis to guide staff allocation
11. Expansion of indigenous territories combined with sustainable natural resource management in peatland-rich areas	Information on causal linkages between indigenous territory and peatland land use	Evaluate mechanisms by which sustainability is integrated in management of indigenous territories; e.g. Green Climate Fund effort
12. Participate in carbon markets or other international mechanism that value the C sequestration in peatlands	Valuation of intact peatland C in markets or by partners. Willing partners, existing markets; coordination with other conservation efforts	Continue to develop international partnerships. Support efforts to include peatland carbon in markets
13. Create science and policy advisory council to share and guide best science basis for peatland policy and management	Perceived need for more information on peatland management	Charge National Committee on Wetlands to form advisory council. Advisory council could also work with regional governments to advise on regional issues

include a summary of policy options to consider that could have a large impact on the trajectory of development in peatlands of Peru (Table 2). The Indonesian experience shows that population pressures driven by transmigration or local population growth could have a large impact on peatland integrity, especially if national policy toward peatland areas permits or encourages road incursion, population migration, and exploitation of peatlands for agriculture; or lax enforcement allows illegal logging or agricultural ventures to expand into protected peatlands. As in Indonesia until recently, current Peruvian policy appears to provide no explicit barriers to peatland road development, agricultural development, and drainage, which could lead to a recapitulation of the Indonesian experience of peatland degradation and loss of function.

Indonesian history reveals that if peatland drainage is permitted the result is a large loss of C stocks to the atmosphere and extensive damage to human health from smoke, as drainage leads to enhanced greenhouse gas emissions from both decomposition and fire. Fire impacts are presently low in the Peruvian Amazon because of low population densities and associated risk factors. Although fire is likely to go up with more intensive land use, whether Peruvian peatlands under intensive land use would be as susceptible to fires as Indonesia's remains an open question because of differences in climate. Answers to this question could be found by modeling of fire response to land-use and climate.

In contrast with Indonesia, where oil and gas development followed other land use changes, any roads associated with oil and gas development in unpopulated roadless areas of Peru could unintentionally stimulate population migration and peatland degradation, with roadless development as a lower-impact alternative. Also in contrast with Indonesia, Peru's greater investment in protected areas and progress in settling land tenure issues could be a strong component of an effective sustainable management program.

Putting this all together, if Peru seeks to maintain the functional integrity and ecosystem services of these peatland ecosystems, especially their long-term potential to remove and store atmospheric carbon in organic matter, the lessons learned from Indonesia's experience point toward an alternative development model built on policies that explicitly recognize the unique fragility and value of intact peatland ecosystems. Options include limiting or prohibiting road access, agricultural development on peat, and peatland drainage; developing efforts for valuation of intact peatlands; expanding peatland-rich formal protected areas and indigenous territories; and determining priority areas for expanded enforcement. Existing Peruvian institutions such as the National Committee on Wetlands and government agencies could partner with the Indonesian Peatland Restoration Agency (BRG) as well as the international scientific community, serving as a conduit for expert advice on peatland science and management that will allow Peru to benefit from the experience of Indonesia.

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