Restoration to offset the impacts of developments at a landscape scale reveals opportunities, challenges and tough choices

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Abstract

When development impacts a broad landscape and causes the loss of multiple ecosystem services, decisions about which of these impacts to offset must be made. We use industrial oil-palm developments in Kalimantan and quantify the potential for restoration to offset oil-palm impacts on carbon storage and biodiversity. We developed a unique backcasting approach combined with a spatial conservation prioritisation framework to identify priority areas for restoration offsetting. We calculated the past impacts of oil-palm development, quantified the future benefits of restoration for carbon storage and biodiversity over one oil-palm planting cycle of 25 years, and prioritised areas for restoration to balance the impacts and benefits for the least cost. We estimate that offsetting the carbon emissions attributable to the existing 4.6 Mha of industrial oil-palm plantation in Kalimantan is most cost-effectively achieved by restoring 0.4–1.6 Mha of degraded peatlands, including failed agricultural projects, at a cost of US$0.7–2.9 billion. On the other hand, offsetting biodiversity losses would require at least 4.7 Mha of degraded areas to be restored (equating to 8.7% of Kalimantan) at a cost of US$7.7 billion. We show that priority areas for offsetting biodiversity losses overlap poorly with those for compensating carbon emissions. Our analysis suggests that reconciling multiple impacts at landscape scales will necessitate difficult choices among contested socio-political preferences. Our findings also clarify the fundamental importance of conserving biodiversity-rich primary forests and peatlands in the tropics and the need to avoid converting these areas in the future.

Keywords: Restoration planning; *Elaeis guineensis*; carbon storage; biodiversity habitat; trade-off; Indonesian Borneo

1. Introduction

Global attention to forest and landscape restoration has been rapidly growing in recent decades as a response to the deterioration of ecosystem services and the acceleration of both species extinction and climate change (Dobson *et al.*, 1997; Lamb *et al.*, 2005; Chazdon *et al.*, 2017). The Convention on Biological Diversity (CBD) through Aichi Target 15 has pledged to restore at least 15% of degraded ecosystems by 2020 for ecosystem resilience, biodiversity conservation and carbon enhancement (Convention on Biological Diversity, 2011). More recently, various global initiatives for restoration have emerged such as the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) through Objective 3(b)(i), and an international commitment through the Bonn Challenge with ambitious targets to restore 350 million ha of degraded and deforested lands by 2030 (IPBES, 2013; IUCN, 2014). The knowledge to support such policy commitments has, however, lagged (Suding, 2011; Menz *et al.*, 2013; Chazdon *et al.*, 2017) and this is an obstacle for the effective implementation of large-scale restoration efforts (Calmon *et al.*, 2011; Murcia *et al.*, 2016; Chazdon *et al.*, 2017). This includes limited knowledge on how financing large-scale restoration, for example, through environmental offsetting (BBOP, 2012; Chazdon *et al.*, 2017).

Environmental offsetting is a policy tool used to mitigate the damaging impacts of development activities such as mining, infrastructure development, and agriculture expansion (Kiesecker *et al.*, 2009b; Madsen *et al.*, 2010; RSPO, 2014; Maron *et al.*, 2015). Offsets aim to counterbalance environmental damage by generating an equivalent benefit elsewhere, such as through protection and/or restoration. Such policies have been gaining popularity, and attracting financing of up to US$4 billion annually (Madsen *et al.*, 2010; OECD, 2013). Its legitimacy is, however, contested, including whether it can contribute to achievement of existing commitments such as the Aichi Targets that otherwise would not be achieved, and unresolved ethical, social, technical and governance issues (McKenney & Kiesecker, 2010; Maron *et al.*, 2016a; Maron *et al.*, 2016b).

Until now, most offsetting studies focus on single impacts, commonly on biodiversity (e.g. Bull *et al.*, 2014; Kormos *et al.*, 2014; Sonter *et al.*, 2014). When offsetting landscape scale developments, such as industrial agriculture, multiple impacts will require consideration simultaneously (OECD, 2013, 2016; Sonter *et al.*, 2018). Furthermore, existing offsetting mechanisms have been focused on a site-scale approach (including that employed in the Remediation and Compensation Procedure by the Roundtable on Sustainable Palm Oil (RSPO, 2014)). This could potentially lead to sub-optimal performance in compensating environmental damage at a landscape scale due to the uneven distribution of biodiversity and services derived from ecosystems, the degree of degradation, the cost of restoration, and the economic value of land (Goldstein *et al.*, 2008; Birch *et al.*, 2010; Wilson *et al.*, 2011; Budiharta *et al.*, 2014a; Budiharta *et al.*, 2016).

Here, we develop a unique decision-making framework to support restoration offsetting and to reveal choices that will be invoked. We illustrate this with the example of extensive oil-palm developments in Kalimantan (Indonesian Borneo). The island of Borneo is a global biodiversity (Rafiqpoor *et al.*, 2005; Kreft *et al.*, 2008) and regional evolutionary hotspot (de Bruyn *et al.*, 2014) with 574 threatened species (IUCN, 2015). The region also has high carbon storage capacity in the form of forest biomass and peat soil carbon (Page *et al.*, 2002; Carlson *et al.*, 2013). In the last four decades, however, Borneo has been undergoing rapid land-use changes with more than one third of its old-growth forests converted into non-forest land-uses (Koh *et al.*, 2011; Carlson *et al.*, 2013; Gaveau *et al.*, 2014a; Gaveau *et al.*, 2016). Oil-palm is a major driver of these processes, with the industrial oil-palm estate estimated to have caused up to 3.9 million ha (20.9%) of natural forest cover loss in Borneo alone (Gaveau *et al.*, 2016).

We employed a backcasting approach combined with a spatial conservation prioritisation framework to identify priority areas for restoration offsetting. We developed our decision-making framework by: (1) calculating the impacts of oil-palm development on carbon emissions and biodiversity losses in terms of native vegetation and mammal habitat using recently-developed maps of land conversion for industrial-scale oil-palm plantation; (2) spatially quantifying the benefits of restoration over one oil-palm planting cycle (25 years) in terms of carbon sequestration and avoided emissions, re-establishment of native vegetation and mammal habitat if degraded areas outside oil-palm plantations are restored; (3) using a spatial decision-support tool to prioritise areas for restoration at a landscape scale with the target that restoration benefits gained were at least equal to the impacts from oil-palm development for the least cost.

2. Materials and methods

2.1. Oil palm-driven land conversion data

Spatial data of oil-palm driven land-cover change over the period 1973–2013 was extracted from Gaveau *et al.* (2016). These data were generated from 357 LANDSAT images using a 5-year interval to detect the trajectory of land-cover change and to determine the existing land cover prior to industrial scale (>100 ha) oil-palm plantation establishment (Gaveau *et al.*, 2016). We cross-checked the oil-palm map (Gaveau *et al.*, 2016) with land-cover maps produced by Indonesian Ministry of Forestry (Ministry of Forestry, 2012b) to delineate existing land-cover being replaced by oil-palm plantation into six classes: intact forest, logged forest, scrub/burned forest, agroforest, non-forest/grassland and uncertain/cloud.

2.2. Impacts by oil-palm plantations

2.2.1. Carbon emissions

We calculated carbon dynamics from oil-palm plantation establishment using a loss-gain method (IPCC, 2006; Murdiyarso *et al.*, 2010). We spatially stratified parameters used in the models (i.e. existing land cover class, mineral or peat soils and peat depth) to allow for better accuracy and to reduce uncertainty (Paoli *et al.*, 2011). For oil-palm plantations occurring on mineral soils/non-peatlands, carbon loss was estimated as the loss of above-ground biomass (AGB) of existing vegetation during land clearing while gain was calculated as AGB stored in oil-palm plantations (Equation 1). We used a 0.5 conversion factor as a fraction of carbon in dry biomass (Brown & Lugo, 1982).

$$\Delta C_{mineral} = C_{AGB(i)} - C_{AGB(OP)} \qquad \text{Eq. (1)}$$

where $\Delta C_{mineral}$ is net carbon emissions in above-ground biomass on mineral soils/non peatlands, $C_{AGB\,(i)}$ is the AGB carbon stock under land cover class *i*, and ($C_{AGB\,(OP)}$) is the AGB carbon of oil-palm plantations. We did not account for the changes in soil carbon from the conversion of forest into oil palm plantation in mineral soils as there are large uncertainties associated with the quantification of this change (Falloon & Smith, 2003). As such, we assumed that soil carbon in mineral soils remained constant before and after oil palm is planted.

For intact forest we used the mean value of AGB carbon (238 $\pm$ 58.5 MgC ha^{-1}) obtained from 62 sites of old growth forest on mineral soils across Borneo (Slik *et al.*, 2010; Budiharta *et al.*, 2014b). The estimates of AGB carbon for logged forest (130.0 $\pm$ 74.67 MgC ha^{-1}) were obtained from a pilot study that measured typical logged forests prior to conversion into oil-palm plantations (Dewi *et al.*, 2009). Gaveau *et al.* (2016) defined scrub as degraded forest following forest fires. For this land cover class, we employed the average AGB carbon of burned forest in East Kalimantan with value of 57.0 $\pm$ 39.61 MgC ha^{-1} (Van der Laan *et al.*, 2014). For agroforest, we extracted a value range of AGB carbon of agroforests and fallow lands across Kalimantan resulting in 41 $\pm$ 16 MgC ha^{-1} (Ziegler *et al.*, 2012). We assumed non-forested land to be severely degraded land dominated by grasses (e.g. *Imperata cylindrica*) and pioneer ferns and shrubs (e.g. *Macaranga* spp.), and assigned an input value of 10 $\pm$ 8 MgC ha^{-1} (Otsamo, 1998; Dewi *et al.*, 2009; Ziegler *et al.*, 2012).

The AGB carbon of oil-palm plantations ($C_{AGB\ (OP)}$) was defined as the time-averaged AGB carbon over a 25-year planting cycle based on field data from Central Kalimantan (Dewi *et al.*, 2009) with a value of 39 $\pm$ 7.4 MgC ha^{-1}, assuming 19% variability of the mean value (Morel *et al.*, 2011). This value applied similarly to both mineral and peat soils (see below).

We added two additional emission sources when calculating carbon dynamics on peatlands (Equation 2): below-ground carbon emissions from peat burning, and oxidation (decomposition) due to draining (Page *et al.*, 2002; Hooijer *et al.*, 2010).

$$\Delta C_{peat} = C_{AGB(i)} + C_{oxid(j)} + C_{burn} - C_{AGB(OP)} \qquad \text{Eq. (2)}$$

where ΔC_{peat} is the net carbon emissions in peat soils, $C_{AGB\ (i)}$ is the AGB carbon stock under land cover class *i*, $C_{oxid\ (j)}$ is carbon emissions from oxidation under peat depth *j* and C_{burn} is carbon emissions from peat burning. We used the average value of AGB carbon of old growth peat swamp forest (174.35 $\pm$ 40.47 MgC ha^{-1}) from seven sites across Borneo as input for intact forest on peat soils (Budiharta *et al.*, 2014b). For logged forest, we assumed that 54.6% AGB carbon is retained as in mineral soils (Dewi *et al.*, 2009) resulting in an input value of 95.2 $\pm$ 54.7 MgC ha^{-1}. We assigned values of AGB carbon for scrub, agroforest and non-forest similar to those in mineral soils with 57.0 $\pm$ 39.61 MgC ha^{-1}, 41 $\pm$ 15 MgC ha^{-1} and 10 $\pm$ 8 MgC ha^{-1} respectively.

As carbon emissions from peat oxidation increase with drainage depth at a rate of 2.5 MgC ha^{-1} yr^{-1} for every 10 cm of additional depth (Couwenberg *et al.*, 2010), we differentiated two levels of emissions from this source. For shallow peat soils (peat depth up to 50 cm), we used carbon emissions of 12.5 MgC ha^{-1} yr^{-1}, while for deep peat soils (peat depth more than 50 cm) we employed 20 MgC ha^{-1} yr^{-1}, assuming the recommended maximum drainage depth was 80 cm (Ministry of Agriculture, 2009). We used the peatlands base map developed by Sekala and Wetland International to assign peat depth (Gingold *et al.*, 2012).

Carbon emissions from peat burning have a large uncertainty as they are heavily influenced by management practices of oil-palm planters and environmental conditions, such as prolonged meteorological and hydrological drought during El Niño events (Casson, 2000; Obidzinski *et al.*, 2012; Taufik *et al.*, 2017). We therefore used estimates of 217.5 MgC ha^{-1} to account for the annual probability of burning on drained peatlands in Southeast Asia (Hooijer *et al.*, 2006; Venter *et al.*, 2009). This value is comparable to the average carbon emissions from peat burning across Indonesia by another study with 203 MgC ha^{-1} (Carlson *et al.*, 2013). We then used the low and high values (72-386 MgC ha^{-1}) to account for uncertainty (Carlson *et al.*, 2013).

2.2.2. The loss of native vegetation

Our first measure of biodiversity loss was the clearing of native vegetation replaced by oil-palm monoculture plantations. We used floristic eco-regions to represent the potential distribution of native vegetation in Kalimantan (Raes, 2009). Raes (2009) classified Borneo into floristic eco-regions based on species distribution modelling using the MaxEnt algorithm (Phillips & Dudík, 2008) of more than 2,270 vascular plant species, using 44,000 herbarium records. Raes (2009) clustered the resultant matrix of species distributions using a hierarchical clustering analysis and generated eleven floristic eco-regions, of which all occur in Kalimantan, using an indicator species analysis.

We masked the floristic eco-region map (Raes, 2009) with the oil-palm driven land conversion map (Gaveau *et al.*, 2016). As the condition of existing vegetation cover varies due to anthropogenic and environmental factors such as logging and forest fires (Klein *et al.*, 2009; Etter *et al.*, 2011), we used 'intactness-adjusted area' (IAA) as the metric for native vegetation loss (Habib *et al.*, 2013). The IAA was calculated as follows:

$$IAA_{(i)} = A_{(i)} \times I_{(j)} \qquad \text{Eq. (3)}$$

where $IAA_{(i)}$ is intactness-adjusted area for floristic eco-region *i*, $A_{(i)}$ is the extent of area lost due to oil-palm establishment under floristic eco-region *i*, and $I_{(j)}$ is the intactness index for land cover class *j*. We used species richness of native trees to generate the parameters of a floristic intactness index with the rationale that Borneo's terrestrial ecosystems were historically composed of tree-dominated ecosystems (i.e. forests) with limited evidence of the prevalence of other vegetation types in the past (e.g. savannahs) (Raes *et al.*, 2014). We assumed that intact forest serves as a baseline system with an intactness index of 1. We assigned an average intactness value of 0.77 to logged forest, as species-area curves per hectare showed that this land cover type retains 74-80% of tree species of intact forest (Cannon *et al.*, 1998; Imai *et al.*, 2012). For scrub, we assumed that burned forest has 30% floristic similarity in trees to intact forest (Slik *et al.*, 2008), resulting in a value of 0.3 for the intactness index. An intactness index of 0.23 was assigned to agroforest according to the average similarity indices between primary forest and forest garden systems in Maluku, Indonesia (Kaya *et al.*, 2002). We assigned a zero value of the intactness index for non-forested areas as tree species richness there is extremely low, especially on *I. cylindrica* grassland (Potter, 1996). We acknowledge that this method does not account for species-area effects or stem density effects, but this was unavoidable due to the large scale of our analysis and limited fine resolution data.

2.2.3. The loss of mammal habitat

A second measure of biodiversity loss was the loss of original habitat of mammal species impacted by oil-palm development. Mammals have been frequently used as conservation flagship species (Fitzherbert *et al.*, 2008), and most mammals are negatively affected by oil-palm plantations (Danielsen *et al.*, 2009). We employed recently-developed habitat suitability maps of 81 mammal species belonging to three groups: carnivores (23 species), primates (13 species) and bats (45 species) that represent a diverse suite of life-history traits and extinction risks (Struebig *et al.*, 2015b). Struebig *et al.* (2015b) employed the Maximum Entropy (MaxEnt) algorithm (Phillips & Dudík, 2008) to map an environmental envelope for each species using bioclimatic variables (i.e. climates, topographic, and distances to water, wetlands and limestone karst). They then corrected the resultant environmental envelope map with mammal sensitivity to land cover following Wilting *et al.* (2010) and consulted 70 experts resulting in habitat suitability maps for all species. For our analysis, we employed a habitat suitability map with strict treatment of possible omission errors (i.e. 25%), reflecting the core habitat inside the known geographical range of the species (Struebig *et al.*, 2015a). We calculated habitat loss for each mammal by masking its habitat suitability map onto the oil-palm plantation map (Gaveau *et al.*, 2016).

2.3. Potential areas for restoration

We defined potential areas for restoration offsetting as areas outside oil-palm plantations that were currently deforested or degraded. To identify deforested and degraded areas, we employed the land cover map generated from ALOS PALSAR data (Gaveau *et al.*, 2014a) which classified land cover into nine categories. For our analysis, deforested and degraded areas were those under the class of non-forest/grassland, agroforest, scrub/burned forest and logged forest. Within these areas, we mapped 'future' landscapes assuming all degraded areas are restored accounting for the benefits of restoration in terms of carbon, reestablishment of native vegetation, and mammal habitat suitability.

2.4. The benefits of restoration

2.4.1. Carbon

We calculated and mapped the carbon benefit based on the difference in value between the initial condition and the restored state (Maron *et al.*, 2013; Evans *et al.*, 2015). We differentiated potential sources of carbon benefits from restoration between mineral soils and peatlands. For mineral soils, the carbon benefit was formulated as:

$$\Delta C_{mineral} = C_{seq(ij)} - C_{AGB(i)} \qquad \text{Eq. (4)}$$

where $\Delta C_{mineral}$ is net carbon gain from restoration arising from $C_{seq(ij)}$, the total AGB carbon sequestered on a restored site currently under land cover class *i* and floristic eco-region *j*, and C_{AGB} the initial AGB carbon stock under land cover class *i*.

We calculated carbon sequestration using the 3-PG (physiological principle for predicting growth) model (Landsberg & Waring, 1997; Budiharta *et al.*, 2014b) over 25 years—equating to one oil-palm cycle. Soil texture classes, fertility ratings and maximum and minimum plant-available soil water were obtained from the Harmonised World Soil Database (FAO/IIASA/ISRIC/ISSCAS/JRC, 2012). Climate variables including monthly temperature, monthly precipitation and vapour pressure deficit were obtained from the WorldClim database (Hijmans *et al.*, 2005), and solar radiation data from the POWER project (The Prediction of Worldwide Energy Resource) (NASA, 2013). We parameterised physiological inputs for the model for each floristic eco-region (Budiharta *et al.*, 2014a; Budiharta *et al.*, 2014b).

Degraded peatlands are generally drained using canal systems (Harrison *et al.*, 2009; Gaveau *et al.*, 2014b). Canalisation lowers the water table and makes degraded peatlands susceptible to repeated burning, especially during drought years (Taufik *et al.*, 2017). As such, for restoration on peatlands, we accounted for carbon benefits from peat burning and peat oxidation in the absence of restoration (i.e. business as usual scenario) (Equation 5):

$$\Delta C_{peat} = C_{seq(i)} + C_{oxid(j)} + C_{burn} - C_{AGB(i)} \qquad \text{Eq. (5)}$$

where ΔC_{peat} is net carbon gain from restoration on peatlands, $C_{seq(i)}$ is the total AGB carbon sequestered on a restored peatland site under land cover class *i*, $C_{oxid(j)}$ is the avoided carbon emissions from peat oxidation under peat depth *j*, C_{burn} is the avoided carbon emissions from peat burning, and $C_{AGB(i)}$ is the initial AGB carbon stock under land cover class *i*. We explored the sensitivity of the assumption that restoration would fail to prevent fires on peat by omitting C_{burn} from Equation 5 (i.e. fires would also occur on restored peatlands and carbon emissions from peat burning would continue). We assigned parameter values for avoided carbon emissions from peat oxidation and burning, similar to when calculating carbon loss.

2.4.2. The establishment of native vegetation

We calculated the benefit of restoration on native vegetation establishment by subtracting the restored state by the initial state before restoration occurs (Evans *et al.*, 2015). We assumed that restoration would fully recover native vegetation on the degraded areas and thus accumulate area in the intact condition class and determined the net gain in extent (measured as the intactness-adjusted area, $\Delta IAA_{(i)}$) for each floristic eco-region *i* (Equation 6):

$$\Delta IAA_{(i)} = A_{(i)} [1 - I_{(j)}] \quad \text{Eq. (6)}$$

where $A_{(i)}$ is the extent of area restored under floristic eco-region *i* and $I_{(j)}$ is intactness index of the initial state for land cover class *j*. We assigned parameter values for the intactness index for each land cover class as per the calculations for native vegetation loss.

2.4.3. The establishment of mammal habitat

The contribution of each potential restoration offset site to mammal habitat was calculated as the extent of degraded areas that occurred within historical suitable habitat (i.e. prior industrialisation in Kalimantan, which commenced in the 1950s). As such, we assumed that restoration would fully recover the degraded areas to their pre-1950 condition. Historical suitable habitat was delineated using the MaxEnt algorithm (Phillips & Dudík, 2008) and bioclimatic variables as predictors (i.e. environmental envelopes) (Struebig *et al.*, 2015b), and was then corrected with historical land cover (Struebig *et al.*, 2015a). We employed a strict commission error threshold of 25% to assign habitat suitability of restored sites into binary categories.

2.5. Restoration approaches and costs

We employed restoration approaches used by Budiharta *et al.* (2014a) for restoring heterogeneous tropical landscapes in Kalimantan which are developed based on scientific papers, technical reports, government regulations and personal communications. Budiharta *et al.* (2014a) divided landscapes into several zones representing various levels of landscape degradation and assigned plausible restoration approach for each zone with the main activity being planting of native tree species. For example, in critically degraded areas intensive-square planting was assigned, while in highly degraded and moderately degraded forest strip planting and gap planting was used respectively.

We calculated the restoration cost as a combination of the implementation and opportunity costs. The implementation cost was based on the standard cost of forest rehabilitation in Indonesia as prescribed by the Ministry of Forestry and differentiated by the restoration approach implemented and the starting degradation level (Ministry of Forestry, 2012a; Budiharta *et al.*, 2014a). This cost captures expenses related to planting activities and maintenance (including fire prevention) up to fourth year after planting as suggested by Hardiansyah (2011). For restoration occurring on degraded peatlands, we also accounted for the cost of rehabilitation of hydrological conditions assuming that dam construction was required to decommission canals (Kalimantan Forest Carbon Partnership, 2009; Budiharta *et al.*, 2014a).

The opportunity costs were defined as the revenues forgone for alternative forms of land management (Table S1). We considered oil-palm plantations, logging and agroforestry as the most relevant alternative land uses in the region (Venter *et al.*, 2009; Carlson *et al.*, 2013; Gaveau *et al.*, 2014a). We employed the Net Present Value (NPV) of oil-palm plantations managed by listed companies as the baseline opportunity cost (Irawan *et al.*, 2013). We differentiated the NPV on the basis of land suitability mapped using 11 biophysical variables (Table S2). We added potential revenues from timber extraction during land clearing, if the areas suitable for oil-palm plantations overlapped with extant forest (Venter *et al.*, 2009). For areas not suitable for oil palm, the opportunity cost was derived from timber revenue if it occurred on logged and burned forest, and from timber and non-timber forest products if it occurred on agroforest.

2.6. Prioritising areas for restoration offsetting

We prioritised potential areas for restoration offsetting using the decision support tool *Zonation v. 4* (Moilanen *et al.*, 2014). For each feature (i.e. carbon, floristic eco-regions and mammal habitat) we determined the loss incurred due to the development of oil-palm plantations (as detailed above) and employed this as a target in the prioritisation analysis. A target-based algorithm sought the most cost-efficient combination of areas to meet these targets. We also investigated the resultant priority areas by seeking to compensate for the loss of: (a) carbon only; (b) floristic eco-regions only; (c) mammal habitat only; (d) floristic eco-regions and mammal habitat; (e) carbon and floristic eco-regions; (f) carbon and mammal habitat. All input layers (and the resultant priority maps) had a spatial resolution of 100 ha to align with the minimum size of industrial-scale oil-palm plantations (Gaveau *et al.*, 2016).

3. Results

3.1. Impacts of oil-palm development on carbon and biodiversity

Extracting data from Gaveau *et al.* (2016) indicated that 4.6 million ha of industrial oil-palm plantations were established in Kalimantan between 1973 and 2013 (Fig. S1). Using a loss-gain method, we estimated net emissions of 0.7 GtC (0.4–1.0 GtC) of carbon over a 25-year planting cycle (Fig. 1). While only 14.3% of the oil-palm plantations were on peatlands (Figs. S1 and S2), they contributed 74.8% of total carbon emissions from oil-palm development (Fig. 1). Net carbon emissions per hectare from peatland conversion (averaged across land-cover classes and peat depths) was 745 tC ha^{-1} (340–1045 tC ha^{-1}) more than five times higher than mean emissions from converting forests (of intact and logged condition) on mineral soils (136 tC ha^{-1}) (Fig. S3). Conversely, establishing oil-palm plantations on mineral soils in non-forested areas, including grasslands, resulted in a net carbon gain (29 tC ha^{-1}).

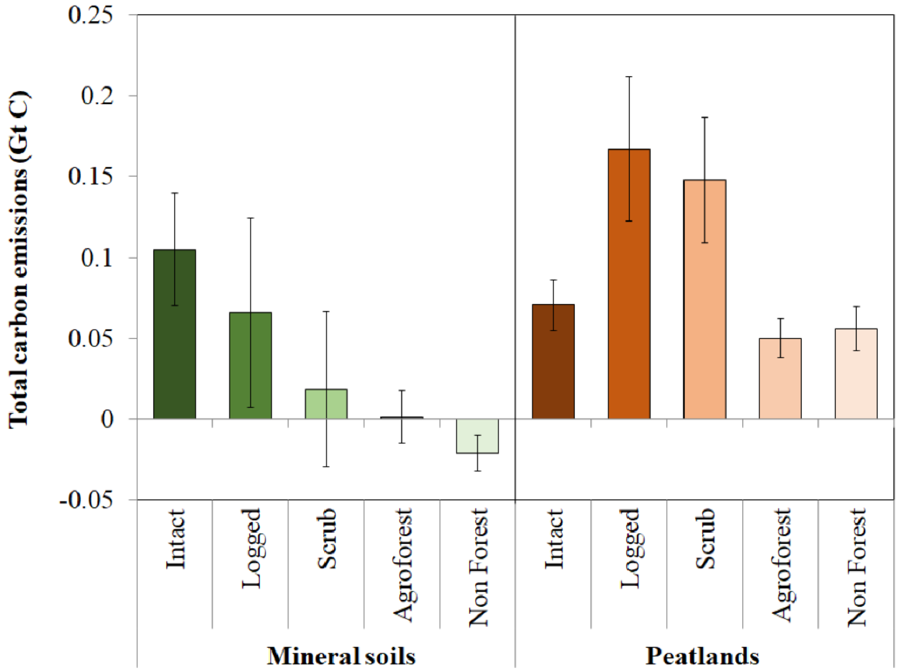

Fig. 1. Oil palm-driven carbon emissions in Kalimantan between 1973 and 2013**.** Total net carbon emissions across land-cover classes and soil types, assuming a 25-year oil-palm planting cycle. Extent of land-cover class per soil type and net emissions per hectare are detailed in Figs. S2 and S3. Scrub refers to degraded forest that have become converted to short vegetation following recurrent forest fires. Negative carbon emissions indicate net carbon gain (carbon sequestered from oil-palm plantation exceeds carbon loss associated with converting non-forested areas). Error bars are lower and upper estimates of net carbon emissions.

Industrial oil-palm plantations of 4.6 million ha have converted the equivalent of 1.9 million ha of intact floristic eco-regions (Table 1; Fig. S4). Lowland forest of 'southern Kalimantan' is the eco-region that has been most extensively replaced with oil-palm with a total extent of 1.0 million ha, equivalent to IAA of 0.5 million ha. Heath forest has had the greatest proportional replacement with 18.7% of the historical extent converted to oil-palm plantations. The most intensely impacted eco-region (with the highest IAA relative to oil-palm extent) is lowland forest of 'northern Kalimantan' as plantations have replaced most of its intact and logged forests.

Table 1. Extent of native vegetation in Kalimantan replaced by oil-palm plantations. Intactness-adjusted area (IAA) represents ecological integrity of extant native vegetation (e.g. logged forest, burned forest) relative to intact forest. Native vegetation is represented by floristic eco-region using clustering analyses (Raes, 2009; Fig. S4). For the purpose of this paper, some nomenclatures of the eco-regions were modified from the original dataset described by Raes (2009).

Floristic eco-region name	Historical extent (000 ha)	Extent occupied by oil-palm plantations (000 ha)								IAA (000 ha)	IAA relative to oil-palm extent (%)
		Intact	Logged	Scrub	Agroforest	Non-forest	Uncertain	Total extent	Per cent occupied (%)		
Freshwater swamp forest	4,971	71	99	156	69	95	23	515	10.38	211	41.0
Peat swamp forest	4,575	67	113	161	152	123	21	639	13.98	238	37.3
Heath forest	4,205	95	137	222	142	174	16	788	18.74	300	38.1
Lowland forest of ‘western Kalimantan’	4,865	14	11	0.308	80	37	41	186	3.82	41	22.5
Lowland forest of ‘central Kalimantan’	4,991	40	45	27	139	34	71	358	7.19	115	32.3
Lowland forest of ‘southern Kalimantan’	10,346	142	301	173	118	226	67	1,030	9.96	453	44.1
Lowland forest of ‘northern Kalimantan’	1,083	32	79	0.450	19	15	0	147	13.63	98	66.9
Lowland forest of ‘eastern Kalimantan’	8,056	132	141	466	44	85	11	881	10.94	391	44.4
Hill forest	445	0	0	0	0	0	0	0	0.00	0	N/A
Montane forest of upper Kapuas	3,529	0	0	0	0	0	0	0	0.00	0	N/A
Montane forest of ‘eastern Kalimantan’	5,765	0	0.479	0	0	0.246	0	0.725	0.01	0.368	50.9

We estimated that the suitable habitat of 78 mammal species (96.3% of the sample) has been planted with oil-palm with an average of 7.6% (± 2.9%) of habitat having been converted to plantations (Table S3). For some charismatic mammals such as Bornean orangutan (*Pongo pygmaeus*) and proboscis monkey (*Nasalis larvatus*), oil-palm plantations have replaced more than 10% of their suitable habitat across Kalimantan. Bornean banded langur (*Presbytis chrysomelas*), a Critically Endangered mammal and one of the rarest primates in Borneo (IUCN, 2015), suffered a loss of 11.3% of its habitat. The most severely-affected mammal was the white-collared fruit bat (*Megaerops wetmorei*), with 16.7% of its habitat lost to oil-palm development.

3.2. Priority areas for restoration to offset carbon emissions

We discovered that to offset the emitted carbon from the creation of industrial oil-palm plantations would require restoration of 0.8 million ha (0.4–1.2 million ha) and incur costs of US$1.3 billion (US$0.7–2.0 billion) assuming that restoration would avoid emissions from further peat fires (Figs. 2a and 2b). The areas selected for offsetting carbon impacts are primarily severely logged and frequently burned peatlands with deep peat including the site of the failed Ex-Mega Rice Project (EMRP) in Central Kalimantan (Fig. 3a). Restoration of these areas avoids carbon emissions from peat oxidation and burning (Fig. S5) while incurring low opportunity costs due to low suitability for timber extraction and palm-oil production, although the cost of hydrological rehabilitation is high (Fig. S6). If restoration failed to prevent peat fires, the required area for compensation would increase to 1.1 million ha (0.6–1.6 million ha) with the cost of restoration rising to US$1.8 billion (US$1.0–2.9 billion) (Figs. S7 and S8).

3.3. Priority areas for restoration to offset biodiversity impacts

To offset the combined biodiversity losses due to industrial oil-palm plantations developed between 1973 and 2013, the oil-palm industry would need to restore vegetation across 8.7% of Kalimantan's landmass. Offsetting the loss of floristic eco-regions measured as intactness-adjusted areas would require the restoration of 2.2 million ha at a cost of US$3.6 billion (Figs. 2a, 2b and 3b). To offset the loss of mammal habitat would require 4.6 million ha to be restored at a predicted cost of US$7.6 billion (Figs. 2a, 2b and 3c). Simultaneously offsetting the losses of floristic eco-regions and mammal habitat slightly increased the total area to restore and cost compared to when targeting mammal habitat with 4.7 million ha at a cost of US$7.7 billion (Figs. 2a and 2b). The relatively similar cost and extent when offsetting the combined biodiversity losses with offsetting only for the loss of mammal habitat indicates that achieving mammal habitat targets would also simultaneously achieve the targets for floristic eco-regions. Priority areas for offsetting the combined biodiversity features would include severely degraded lowland forests in East Kalimantan and logged forests in Central Kalimantan (Fig. 3d). These areas have low suitability for oil-palm plantation, resulting in low opportunity cost (Fig. S5).

3.4. Priority areas for restoration to offset carbon and biodiversity losses

When attempting to achieve the offset targets for restoration of floristic eco-regions and carbon simultaneously, the extent of offsets is similar to that required when compensating the loss of floristic eco-regions alone, but the priority areas change to include degraded peatlands in the EMRP (Fig. 3e). This spatial shift would incur a higher cost of US$3.7 billion (US$3.6–4.1 billion) (Figs. 2a and 2b). When carbon offset targets were included with mammal habitat targets, the extent of offsets increases to 4.6 million ha costing US$7.7 billion (Figs. 2a, 2b and 3f), indicating offsetting the loss of mammal habitat would achieve the target for carbon.

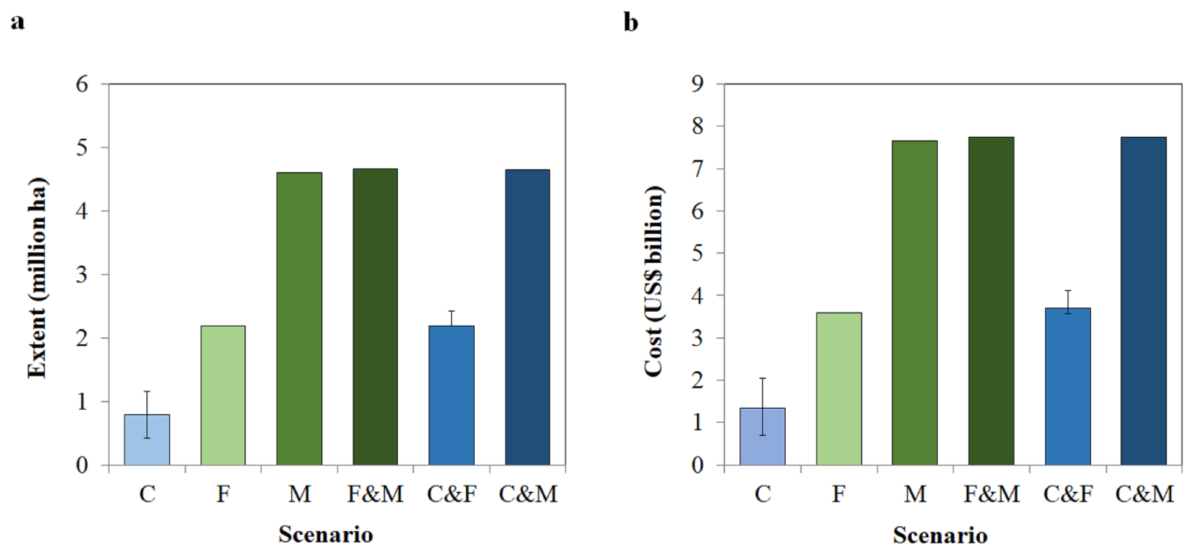

Fig. 2. Resources required to offset the impacts of oil-palm plantation in Kalimantan. **a,** Extent of landscape selected for restoration offsetting. **b,** Total offsetting costs accounting for opportunity and implementation costs. Each offsetting scenario aims to compensate for the loss of: carbon (scenario C); floristic eco-region (scenario F); mammal habitat (scenario M); floristic eco-region and mammal habitat (scenario F&M); carbon and floristic eco-region (scenario C&F); carbon and mammal habitat (scenario C&M). Error bars represent the range of results accounting for lower and higher estimates of total carbon emissions as such the bars only apply for scenarios involving carbon.

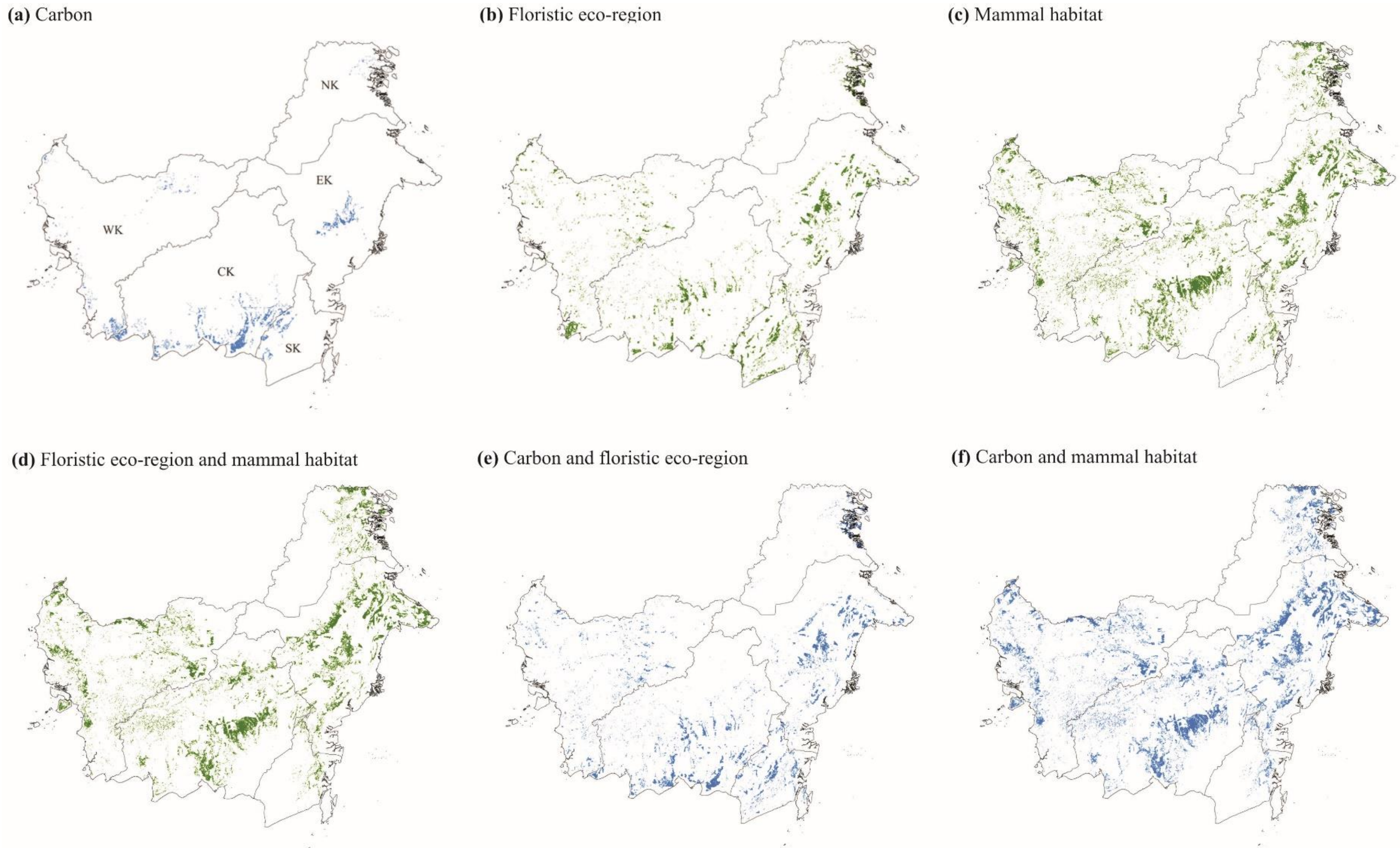

Fig. 3. Priority areas for offsetting the impacts of oil-palm development via restoration in Kalimantan. Each figure represents different scenarios for compensating the loss of: (**a**) carbon; (**b**) floristic eco-region; (**c**) mammal habitat; (**d**) floristic eco-region and mammal habitat; (**e**) carbon and floristic eco-region; (**f**) carbon and mammal habitat. Priority areas were identified through spatial decision support using a target-based algorithm while minimising cost (Supplementary Methods). The target was set to reflect the loss of carbon and biodiversity that has been incurred due to the development of oil-palm plantations. Labels refer to province: West Kalimantan (WK), Central Kalimantan (CK), South Kalimantan (SK), East Kalimantan (EK) and North Kalimantan (NK).

When we overlaid the priority areas for offsetting carbon only (Fig. 2a) and biodiversity combined (Fig. 2d), only 0.2 million ha of the priority areas overlapped (Fig. 4), equivalent to 25% of the extent of priority areas for offsetting carbon and 4% of priority areas when targeting biodiversity. The overlapping areas include the degraded peatlands in Kutai, East Kalimantan and in Sampit, Central Kalimantan (Fig. 4). The small extent of overlap indicates a limited opportunity for synergy between achieving the target for carbon and biodiversity in the context of Kalimantan.

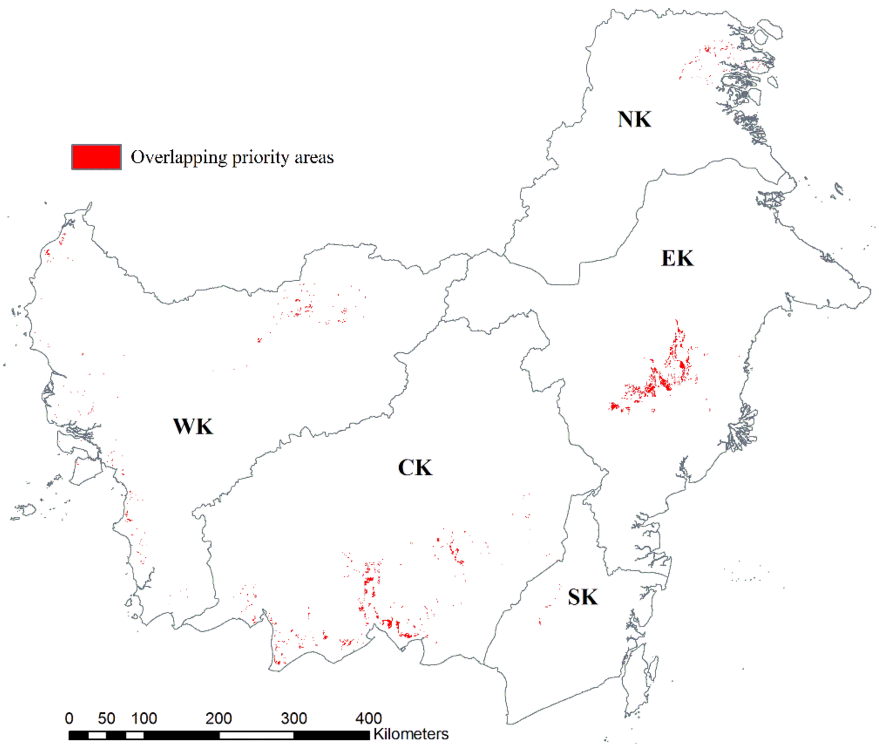

Fig. 4. Overlapping priority areas between offsetting the impacts of oil-palm development on carbon emissions and compensating biodiversity losses (i.e. mammal's habitat and floristic ecoregion combined) in Kalimantan.

4. Discussion

Offsetting the environmental impacts of development may appear to be an opportunity to finance restoration whenever global ambitious targets on forest and landscape restoration are not otherwise likely to be achieved (Maron *et al.*, 2016a; Chazdon *et al.*, 2017). However, we demonstrate here that even assuming perfect restoration effectiveness, difficult decisions are required when offsetting large-scale impacts, with important associated implications for the cost and area requiring restoration. Our findings suggest that reconciling multiple impacts at landscape and larger scales will necessitate difficult choices among contested socio-political preferences.

4.1. What to offset?

When solely targeting for carbon, restoring degraded deep peatlands is the priority strategy to offset carbon emissions from industrial oil-palm development in Kalimantan. To fully offset the emissions from palm oil plantations, an extent in the range of 0.4–1.6 Mha at a cost up to US$2.9 billion would need to be restored. Despite concerns over carbon emissions from conversion to oil-palm plantations (e.g. Koh *et al.*, 2011; Carlson *et al.*, 2013; Busch *et al.*, 2015), offsetting carbon impacts from development is not yet popular in existing offsetting policies and practices. For example, current Remediation and Compensation Procedures developed by the Roundtable on Sustainable Palm Oil do not explicitly state carbon emissions ought to be compensated (RSPO, 2014). Also, if implemented, there are likely further debates in relation to other policy arenas, such as whether carbon offsetting may be included into or should be separated from Reducing Emissions from Deforestation and Forest Degradation (REDD+) mechanism (Solheim & Natalegawa, 2010).

On the other hand, the compensation for two elements of biodiversity loss would require restoration of 4.7 Mha degraded lands, equivalent to the overall extent of Kalimantan's industrial oil-palm plantation estate. This is even assuming perfect restoration success, which is highly implausible (Maron *et al.*, 2012). Restoration of the vast extent required to compensate fully for biodiversity losses would be politically constrained by regional and national development targets aiming for the expansion of oil-palm and industrial timber plantations, logging and mining (Abood *et al.*, 2015; Runting *et al.*, 2015). The high costs incurred (i.e. US$7.7 billion) also raises questions about the capacity of the oil-palm industry to finance the offsetting mechanism. To put this into perspective, the net present value (NPV) of oil-palm plantation per hectare is US$6,355 in one planting cycle (Irawan *et al.*, 2013) – equating to a total NPV for industrial oil-palm plantation in Kalimantan of US$29.6 billion. Considering this economic capacity, covering the cost of biodiversity offsetting is likely not feasible for the oil-palm industry.

The difference in both the amount of restoration required and the locations of priority areas for compensating carbon emissions and biodiversity makes a synergistic solution problematic. Our finding echoes other works that there are trade-offs among desired outcomes when making decisions for restoration of ecosystem services (Budiharta *et al.*, 2014a; Gourevitch *et al.*, 2016). In the context of Kalimantan, the trade-offs between carbon and biodiversity mirrors REDD+ policy implementation, where there is spatial mismatch between areas best targeted for climate change mitigation (by protecting carbon-rich sites such as peatlands) or biodiversity conservation (by focusing on species-rich areas) (Paoli *et al.*, 2010; Murray *et al.*, 2015). While there is opportunity to compensate for carbon emissions by restoring degraded deep peatlands in the region, this choice will have limited co-benefits for biodiversity. Conversely, if restoration is used in an attempt to offset biodiversity loss, the costs and areas required would be much higher, which becomes the hindrance to convince policy makers with limited interest in conservation.

Our analysis illustrates the ethical and social complexity associated with the offsetting mechanism when multiple impacts are considered (Maron *et al.*, 2016b; Sonter *et al.*, 2018). The situation becomes even more complicated if the impacts are mostly intangible, such as socio-cultural values of the forests that have been lost (Ives & Bekessy, 2015). In Kalimantan, local communities perceive forest as important for their spiritual and subsistence needs (Meijaard *et al.*, 2013; Abram *et al.*, 2014), triggering social conflict when the forest is converted to oil-palm plantation (Abram *et al.*, 2017). Further understanding is therefore required to resolve competing preferences held by societies on the choices of what to compensate in environmental offsetting whenever compensating multiple impacts is not feasible.

4.2. Policy implications for the study area

Beside enriching knowledge in restoration and offsetting studies, our analysis provides insights for land-use policy in Kalimantan and Indonesia. Degraded peatlands in the form of grasslands, shrublands and logged forests have been converted to agriculture, either by large companies or small-scale farmers, through draining and burning (Harrison *et al.*, 2009; Gaveau *et al.*, 2014b). These activities led to the release of carbon emissions between 0.81–2.57 GtC in one El Niño event alone (Page *et al.*, 2002). Peat fires also cause up to US$33 billion in economic losses and severe public health problems (Chan, 2015). Our findings suggest that no extractive activities should take place in degraded peatlands and that these areas are potential for carbon offsetting through restoration.

If carbon emissions due to oil-palm plantation development in Kalimantan can be fully compensated through peatland restoration, this strategy alone could reduce by 11% the total emissions from land-use and land-cover change in Indonesia (Busch *et al.*, 2015). Also, through large scale peatland restoration using rewetting and revegetation of drained peatlands, the risks of peat burning could be reduced to mitigate social and economic impacts caused by haze problems (Forsyth, 2014; Taufik *et al.*, 2017; Dohong *et al.*, 2018). To enhance social benefits for local community, restoration offsetting on peatlands could be integrated with emerging policy of community forestry since there are limited restoration investments and capacity building programs currently directed toward community forests located on degraded peatland (Santika *et al.*, 2017). The Peatland Restoration Agency (https://brg.go.id) was formed by the Indonesian Government to coordinate and facilitate restoration of two million hectares of degraded peatlands in Indonesia, mainly in Sumatra and

Kalimantan. The Agency may serve to facilitate the implementation of carbon offsetting through peatland restoration.

Our results also clarify the fundamental role of conserving biodiversity-rich primary forests and peatlands in the tropics (Gibson *et al.*, 2011; Page *et al.*, 2011; Wijedasa *et al.*, 2017) and the costs associated when these areas are damaged. The costs of repairing damaged landscapes to compensate biodiversity loss are extremely high. This reinforces the importance of the early stages of the mitigation hierarchy (i.e. avoid and minimise (Kiesecker *et al.*, 2009a; Pilgrim *et al.*, 2013; Maron *et al.*, 2016b)) when planning for oil-palm development. Oil-palm plantations are predicted to expand between 6.9–9.4 Mha over the next 5 years in Kalimantan alone (Carlson *et al.*, 2013; Abood *et al.*, 2015; Runting *et al.*, 2015). To minimise carbon emissions, there should be no future oil-palm development on peatlands, including those in a degraded condition. Also, expansion should be directed toward degraded lands with limited forest cover (Smit *et al.*, 2013; Santika *et al.*, 2015) to reduce impacts on biodiversity.

4.3. Biases and uncertainties

While we developed a unique framework decision-making in restoration offsetting that account for biophysical and economic heterogeneity of a landscape, our analysis did not consider the social realm. Local communities may not accept the restoration offsetting we describe, especially considering the challenges of community land-claims and conflicts in Kalimantan (Thaler & Anandi, 2016; Abram *et al.*, 2017; Prabowo *et al.*, 2017; Santika *et al.*, 2017). Incorporating social variables, such as community acceptance of restoration, will likely change the priority areas for offsetting with potential increase in costs and/or area extent (Budiharta *et al.*, 2016). As such, the outputs of our analysis should not be used prescriptively.

When calculating the benefits of restoration for biodiversity, we also assumed that restoration will successfully recover native vegetation and the habitat of mammals. In reality, there are long time-lags and uncertainties in restoring sites to the level of intact systems, sometimes requiring centuries (Curran *et al.*, 2013), although there is some evidence in the tropics that some species rapidly colonise the restored sites within three decades (e.g. Edwards *et al.*, 2009; Ansell *et al.*, 2011; Gilroy *et al.*, 2014). Accounting for these constraints in our spatial analysis will likely result in the offset lands needing to be much larger than the original impacted area (Maron *et al.*, 2012) – indeed, it may render full compensation impossible.

4.4. Way forward

Offsetting is an emerging tool for environmental protection and rehabilitation, and has both prospects and limitations. Our study reveals tough choices. To which ecological impacts ought offsetting apply? Who will make these value judgements? If there are trade-offs among outcomes, will civil society accept the compromise? As achieving ambitious global restoration targets is a matter of political will, scientific exercises to answer such questions needs to involve social and political sciences where values and judgments can be incorporated.

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