Carbon and Biodiversity Impacts of Intensive Versus Extensive Tropical Forestry

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Abstract
How should we meet the demand for wood while minimizing climate and biodiversity impacts? We address this question for tropical forest landscapes designated for timber production. We model carbon and biodiversity outcomes for four archetypal timber production systems that all deliver the same volume of timber but vary in their spatial extent and harvest intensity. We include impacts of variable deforestation risk (secure land tenure or not) and alternative harvesting practices (certified reduced-impact logging methods or not).

We find that low-intensity selective logging offers both the best and the worst overall outcomes per unit wood produced, depending on whether certified reduced-impact logging methods are used and whether land tenure is secure. Medium-to-high-intensity natural forest harvests and conversion to high-yield plantations generate intermediate outcomes. Deforestation risk had the strongest influence on overall outcomes. In the absence of deforestation, logging impacts were lowest at intermediate and high management intensities.

Introduction
Humans now actively manage the majority of land on Earth for the production of food, fiber, and energy (Hooke et al. 2012). The harvest of wood from natural forests is the primary cause of tropical forest degradation (Pearson et al. 2017), and the conversion of natural forests to wood fiber plantations is among major causes of tropical deforestation (Hansen et al. 2013; Abood et al. 2014). Combined, tropical deforestation and degradation generate greenhouse gas emissions equivalent to the global transportation sector (IPCC 2014) and are a major threat to biodiversity (Gibson et al. 2011). Yet, halting timber harvests could have perverse climate impacts to the extent that wood is replaced by higher carbon footprint materials (cement, metal, and fossil fuels) (Oliver et al. 2014).

How do we meet the demand for wood products while minimizing CO2 emissions and biodiversity losses? In the context of forested landscapes already designated for timber production, should we promote the intensification of production in small areas with intent to spare large areas of forest from human impacts, or would it be better to promote best practices for extensive low-intensity harvests from native forests?

Recent studies have addressed this type of “sharing versus sparing” question for food crop yields and biodiversity impacts (Phalan et al. 2011; Lee et al. 2014), and for wood yields and biodiversity impacts for selective logging scenarios (Edwards et al. 2014). Here, we consider this question for both biodiversity and carbon outcomes across a wide range of timber harvest intensities. We do so by combining empirical data from the literature with models of four archetypal timber production systems that vary
from low (selection harvest) to high (plantation) yield, and from conventional (CL) to reduced-impact logging (RIL) methods.

A critical concern is the vulnerability of “shared” and “spared” forests to external drivers of forest conversion (Fischer et al. 2014). Sometimes, loggers catalyze deforestation (Asner et al. 2005), and sometimes they are successful agents of forest protection (Porter-Bolland et al. 2012; Gaveau et al. 2013). With these contrasts in mind, we model alternative relationships between different forestry scenarios and deforestation rates.

Many tropical countries have put forward ambitious targets for reducing emissions from deforestation and forest degradation while demand for wood products continues to increase (Elias & Boucher 2014). Our analysis explores the extent to which sharing, sparing, and improved logging practices offer preferred approaches to balancing wood yield with carbon emissions and biodiversity conservation.

**Methods**

We modeled carbon and biodiversity outcomes for four forest management scenarios with three variables: wood harvest intensity, land tenure security, and the presence or absence of RIL (Table 1). We assumed that forests in which RIL practices are employed are certified to be responsibly managed by a third-party certifier such as the Forest Stewardship Council (FSC). Harvest intensity included: (1) low-intensity selective logging, (2) medium-intensity selective logging, (3) high-intensity selective logging, and (4) medium-intensity selective logging with subsequent conversion to intensive timber plantations. We ran the scenarios with a common starting point: 30,000 ha of intact harvestable terra firme forest from which overall annual timber yields were kept constant (10,000 m$^3$ year$^{-1}$). To maintain this yield, areas used for timber production declined from the full 30,000 ha for low-intensity selective harvesting to only 1,125 ha converted to plantations, as a function of variable levels of management intensity indicated by the literature (Tables S1–S2). Timber production areas were assigned to be a logging concession, while the remainder of the 30,000 ha forest area was “spared” from logging (Table 1).

We ran each scenario with rates of deforestation pressure derived from the literature. We assigned the gross mean pantropical rate of annual forest loss (0.45% Hansen et al. 2013) as the mean deforestation rate for all “spared” areas outside of logging concessions. Adjustments were made to this base rate for logging concession areas depending on two alternative assumptions: (1) concession managers have insecure land tenure and catalyze deforestation by constructing logging roads, or (2) concession managers have secure land tenure and resist deforestation to protect their interests. We included two levels of reduced deforestation rates in concession areas with secure land tenure, depending on the presence or absence of forest certification. All of these deforestation rates were derived from recent studies that employed proper paired comparisons to control for confounding spatial variables. (See Supplementary Materials S1.) Emissions from forest clearance for timber harvest infrastructure (haul roads and log landings) are included as a flat impact per ha logged regardless of harvest intensity (Table 1) and are not counted as “deforestation.” We did not assume that areas spared from logging were turned into effectively protected areas because this would result in unbalanced scenarios with variable levels of government investment in protection, and because we are unaware of a precedent for establishing protected areas as a function of timber harvest intensity.

We modeled annual gross and net carbon fluxes from timber extraction with parameters derived from the literature on logging impacts as they vary with harvest intensity and regrowth rates for natural forests and plantations (see Tables S1–S3). We conservatively assumed that emissions from RIL were 30% lower than CL at a given logging intensity, based on studies from across Latin America, Africa, and SE Asia (Tables S1 and S3). This also assumes that RIL is effectively implemented, given that when RIL is not effectively implemented climate benefits may not occur (Griscom et al. 2014; Martin et al. 2015). See Supplementary Materials S1 for full descriptions of each of the four scenarios.

We modeled the biodiversity outcomes for each scenario in terms of animal and plant species richness, using the matrix-calibrated species area model (Koh & Ghazoul 2010),

$$S_{new} = \left( \frac{A_{new}}{A_{org}} \right)^{\frac{\gamma}{\alpha + \beta}},$$

(1)

in which $S_{org}$ and $S_{new}$ are the species richness before and after forest management, respectively, $A_{org}$ and $A_{new}$ are the total original and new forest areas, and the exponent reflects the habitat suitability of the landscape matrix. We only considered the state of biodiversity after 60 years, as there is too little known about the process of biodiversity recovery after selective logging. See Table 1 for summary data.

To model our uncertainty in the estimates of various carbon and biodiversity parameters, we ran Monte Carlo simulations (10,000 runs), each time randomly drawing from a distribution of potential parameter values (Supplementary Materials S1 and S2).
### Table 1 Scenario parameters for harvested and “spared” areas

| Parameters                        | 1a: Low-intensity CL | 1b: Low-intensity RIL | 2a: Medium-intensity CL | 2b: Medium-intensity RIL | 3a: High-intensity CL | 3b: High-intensity RIL | 4a: Medium-intensity CL then converted to plantation | 4b: Medium-intensity RIL then converted to plantation |
|-----------------------------------|----------------------|-----------------------|-------------------------|--------------------------|-----------------------|------------------------|---------------------------------------------------|---------------------------------------------------|
| Harvested area parameters         |                      |                       |                         |                          |                       |                        | Medium-intensity CL (first rotation)               | Medium-intensity RIL (first rotation)               |
| Harvest intensity (m³ ha⁻¹)       | 10.0                 | 10.0                  | 40.0                    | 40.0                     | 70.0                  | 70.0                   | 40.0                                              | 10.0                                              |
| Annual area in production (ha)    | 1,000.0              | 1,000.0               | 250.0                   | 250.0                    | 142.9                 | 142.9                  | 250.0                                             | 250.0                                             |
| Number of rotations               | 2                    | 2                     | 2                       | 2                        | 2                     | 2                      | 1                                                 | 1                                                 |
| Total area of timber (ha)         | 30,000.0             | 30,000.0              | 7,500.0                 | 7,500.0                  | 4,285.7               | 4,285.7                | 7,500.0                                          | 7,500.0                                          |
| “Spared” area (ha)                | 0.0                  | 0.0                   | 22,500.0                | 22,500.0                 | 25,714.3              | 25,714.3               | 22,500.0                                         | 22,500.0                                         |
| Logging carbon flux parameters    |                      |                       |                         |                          |                       |                        |                                                   |                                                   |
| Emissions intensity (MgC ha⁻¹)    | 24.8                 | 17.3                  | 44.5                    | 44.5                     | 44.9                  | 44.9                   | 174.1                                            | 174.1                                            |
| Lower 95% CI                      | 8.0                  | 5.6                   | 30.9                    | 21.6                     | 47.7                  | 33.4                   | 30.9                                             | 155.4                                            |
| Upper 95% CI                      | 41.6                 | 29.1                  | 58.0                    | 80.6                     | 56.4                  | 58.0                   | 197.8                                            | 197.8                                            |
| Haul roads and landings (MgC ha⁻¹)| 5.4                  | 2.8                   | 5.4                     | 2.8                      | 5.4                   | 2.8                    | 5.4                                              | 2.8                                              |
| Regrowth rate (MgC ha⁻¹ year⁻¹)  | 1.6                  | 3.1                   | 1.6                     | 3.1                      | 1.6                   | 5.6                    | 3.1                                              | 5.6                                              |
| Lower 95% CI                      | 0.7                  | 2.5                   | 0.7                     | 2.5                      | 0.7                   | 2.5                    | 0.7                                              | 2.5                                              |
| Upper 95% CI                      | 2.5                  | 3.7                   | 2.5                     | 3.7                      | 2.5                   | 3.7                    | 2.5                                              | 3.7                                              |
| Annual deforestation rates (k)    |                      |                       |                         |                          |                       |                        |                                                   |                                                   |
| In concessions—secure tenure, logged and not yet logged | 0.31%       | 0.15%                 | 0.31%                   | 0.15%                    | 0.31%                 | 0.15%                  | 0.31%                                            | 0.00%                                            |
| In concessions—insecure tenure, logged | 1.80%    | na                    | 1.80%                   | na                       | 1.80%                 | na                     | 1.80%                                            | 0.00%                                            |
| Continued                         |                      |                       |                         |                          |                       |                        |                                                   |                                                   |

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### Table 1 Continued

| Parameters                          | 1a: Low-intensity CL | 1b: Low-intensity RIL | 2a: Medium-intensity CL | 2b: Medium-intensity RIL | 3a: High-intensity CL | 3b: High-intensity RIL | 4a: Medium-intensity CL then converted to plantation | 4b: Medium-intensity RIL then converted to plantation |
|------------------------------------|----------------------|-----------------------|-------------------------|--------------------------|------------------------|------------------------|--------------------------------------------------|-----------------------------------------------------|
| In concessions— insecure tenure, not yet logged | 0.45% | na | 0.45% | na | 0.45% | na | 0.45% | 0.00% |
| "Spared" areas (never logged)     | na | na | 0.45% | 0.45% | 0.45% | 0.45% | 0.45% | 0.45% |
| Species richness σ values         |                     |                       |                         |                          |                        |                        |                                                  |                                                     |
| Mammals                           | 0.443               | 0.204                 | 0.736                   | 0.622                    | 1.000                  | 1.000                  | 0.669                                           | 0.397                                               |
| Lower 95% CI                      | 0.397               | 0.139                 | 0.497                   | 0.282                    | 0.570                  | 0.385                  | 0.372                                           | 0.316                                               |
| Upper 95% CI                      | 0.488               | 0.269                 | 0.974                   | 0.962                    | 1.000                  | 1.000                  | 0.967                                           | 0.468                                               |
| Birds                             | 0.034               | 0.082                 | 0.337                   | 0.203                    | 0.639                  | 0.326                  | 0.171                                           | 0.363                                               |
| Lower 95% CI                      | 0.000               | 0.000                 | 0.000                   | 0.069                    | 0.259                  | 0.192                  | 0.000                                           | 0.000                                               |
| Upper 95% CI                      | 0.414               | 0.216                 | 0.717                   | 0.337                    | 1.000                  | 0.460                  | 0.551                                           | 0.489                                               |
| Invertebrates                     | 0.413               | 0.161                 | 0.440                   | 0.200                    | 0.467                  | 0.238                  | 0.300                                           | 0.435                                               |
| Lower 95% CI                      | 0.365               | 0.093                 | 0.403                   | 0.147                    | 0.433                  | 0.190                  | 0.254                                           | 0.321                                               |
| Upper 95% CI                      | 0.460               | 0.229                 | 0.477                   | 0.252                    | 0.501                  | 0.287                  | 0.346                                           | 0.530                                               |
| Amphibians                        | 0.345               | 0.064                 | 0.557                   | 0.367                    | 0.768                  | 0.669                  | 0.446                                           | 0.397                                               |
| Lower 95% CI                      | 0.205               | 0.000                 | 0.412                   | 0.160                    | 0.483                  | 0.262                  | 0.265                                           | 0.316                                               |
| Upper 95% CI                      | 0.485               | 0.264                 | 0.701                   | 0.573                    | 1.000                  | 1.000                  | 0.626                                           | 0.468                                               |
| Plants                            | 0.118               | 0.150                 | 0.330                   | 0.203                    | 0.542                  | 0.257                  | 0.163                                           | 0.401                                               |
| Lower 95% CI                      | 0.000               | 0.016                 | 0.070                   | 0.069                    | 0.282                  | 0.123                  | 0.000                                           | 0.158                                               |
| Upper 95% CI                      | 0.378               | 0.284                 | 0.590                   | 0.337                    | 0.802                  | 0.391                  | 0.423                                           | 0.574                                               |
| Total                             | 0.019               | 0.059                 | 0.307                   | 0.193                    | 0.595                  | 0.326                  | 0.134                                           | 0.397                                               |
| Lower 95% CI                      | 0.000               | 0.043                 | 0.000                   | 0.209                    | 0.235                  | 0.310                  | 0.000                                           | 0.316                                               |
| Upper 95% CI                      | 0.379               | 0.075                 | 0.667                   | 0.177                    | 0.955                  | 0.342                  | 0.494                                           | 0.468                                               |

See Figure S1 for source analysis for emissions intensity values used in uncertainty analysis. This includes impacts from felling and skidding, but not haul roads and landings which were assigned separately as a flat rate per ha harvested.

These are the mean (literature derived) values used in uncertainty analysis. Areas converted to plantations are assumed to not be susceptible to deforestation. We assume that insecure land tenure was not applicable to RIL subscenarios, which were associated with independent third-party certification that verifies legality. See Supplementary Materials S1 for derivation of deforestation rates from the literature.

All scenarios produce 10,000 m³ of timber each year from a 30,000 ha forest area with initial intact forest biomass carbon stocks of 226.8 Mg C ha⁻¹. Scenarios vary in terms of harvested area and harvest intensity, deforestation rates, the use of reduced-impact logging (RIL) versus conventional logging (CL) practices, and associated carbon parameters.
Results

Carbon fluxes

Potential carbon emissions from deforestation were about an order-of-magnitude greater than those from timber harvests (Figure 1). Deforestation emissions varied considerably among scenarios depending on the deforestation rates associated with forest use (wood production or not), RIL and certification (presence or absence), or tenure status (secure or not) derived from the literature (Table 1, S1). Emissions from deforestation were smallest and largest in the two low-intensity scenarios. Deforestation emissions did not notably differ among medium intensity, high intensity, and plantation conversion scenarios.

Net cumulative emissions from only direct impacts of timber production over 60 years were not as clearly different among production intensity levels, but are probably highest for low-intensity selective logging (Figure 1b). RIL (as compared to CL) generated the largest emission reductions when logging was low intensity. RIL made little difference for the plantation conversion scenario.

In the absence of deforestation, forest carbon stocks stabilized during the second harvest for the low-intensity selective logging scenario under both CL and RIL, but only under RIL for medium- and high-intensity selective logging scenarios (Figure S2). Under the plantation scenario, net cumulative emissions reached a maximum at about 30 years, then decreased and stabilized as the logged but unconverted areas recovered and the areas converted to plantations collectively reached stable stocks.

Biodiversity

The combined impact of timber production and associated deforestation on relative species loss was the worst with low-intensity CL under insecure land tenure. With secure land tenure and RIL, the trend was reversed: low-intensity RIL scenario appears to be the best, but this trend was inconclusive (overlapping error bars in Figure 2). While species richness was relatively more sensitive than carbon to direct timber production impacts, deforestation still had larger impacts on species richness than timber production— with the exception of the low-intensity scenario with secure tenure. The ranking of scenarios was similar for individual taxa when considering both direct timber production impacts and associated deforestation (Figure S3). RIL made a substantial positive difference only at low intensities, and only for amphibians, invertebrates, and mammals.
When considering only the direct impacts of timber production, we found little difference between the scenarios in terms of overall reductions in species richness (Figure 2). This result holds when we consider birds and plants separately; but for mammals, amphibians, and invertebrates, higher intensity regimes that involved a smaller area were less harmful (Figure S3).

**Discussion**

We find that low-intensity certified RIL offers the best overall conservation outcomes per unit of wood produced, in contrast with a growing body of scientific and policy literature supporting intensified forestry (Paquette & Messier 2010; Edwards et al. 2014; International Sustainability Unit 2015; Ruslandi et al. 2017). Our findings are clearest for climate outcomes, and are driven by the potential for forest managers with secure land tenure to resist deforestation where they have a commercial interest in future timber harvests. Our results also emphasize the risks of low-intensity selective logging when best practices are not employed. Conventional low-intensity logging with insecure land tenure has the highest impacts on both climate and biodiversity compared to all other scenarios—in terms of both direct impacts from logging and increased likelihood of deforestation.

This stark contrast between alternative scenarios of low-intensity logging, which generate either the best or worst overall outcomes, may help explain why the conservation science community is of two minds about the potential for alliances between loggers and conservationists (Putz et al. 2012; Kormos & Zimmerman 2014). In the absence of deforestation pressure, our results are consistent with Edwards et al. (2014) and indicate lower direct impacts from logging under intensification scenarios.

The direct impacts of timber harvest, in the absence of deforestation, are complex. Low-intensity logging requires more infrastructure than other scenarios and tends to have a higher impact per unit wood harvested than more intensive forms of selective logging—for both carbon and species loss. Among more intensive options, higher intensity selective logging RIL scenarios offer less than half the carbon impact as compared with RIL plantation conversion, but the pattern is reversed for species loss. Relative species richness appears to be more sensitive than carbon to the direct impacts of logging.

There is a notable lack of differences in species richness outcomes under CL practices across the production intensity spectrum. This indicates that as production intensifies, the declining footprint of timber production is counter-balanced by an increasing impact per unit area. This finding would likely differ if forest areas of different sizes and different beta and gamma diversities were considered. Furthermore, these findings mask complex outcomes that vary with taxa and geography (Woodcock et al. 2013; Burivalova et al. 2014). Not all taxa are equally affected by forestry operations (Cowlishaw et al. 2009; Burivalova et al. 2015). Our results reflect this, such as the lower sensitivity of some taxa (e.g., birds) to logging as compared to other taxa (e.g., mammals) and the sensitivity of some taxa to harvest intensity thresholds (e.g., amphibians). Not reflected in our results is large likely variation within community composition, most importantly the differential responses of forest specialist and generalist species (Burivalova et al. 2014).

Unsustainable hunting, potentially stimulated by logging, also reduces biodiversity (Brodie et al. 2015). Our model includes hunting implicitly, through higher habitat suitability values for forests under certified RIL (Table 1) which we assume includes effective measures to protect wildlife, although successful implementation of hunting restrictions remains to be verified with field data (Brodie et al. 2015).

Further research is needed to resolve the complex direct impacts of alternative logging systems; however, our results emphasize the importance of indirect impacts from logging under intensification scenarios.
relationships between forestry and deforestation to the resolution of the tropical forestry sharing versus sparing debate. We encourage broader and deeper applications of statistically robust methods that have identified—in some tropical geographies—that (1) legal logging concessions can resist deforestation as effectively as protected areas (Gaveau et al. 2012, 2013; Blackman 2015) and (2) FSC certification is associated with additional deforestation resistance (Blackman 2015; Miteva et al. 2015). Implementing climate and conservation smart forestry depends on expanding our understanding of the geographic contexts in which such findings either hold, or do not (Busch et al. 2014; Ferretti-Gallon & Busch 2014; Blackman et al. 2015).

Regional spatially explicit analyses are needed for locally relevant conclusions

Extending these findings to specific landscapes should be done with consideration of geography-specific parameters, for which we provide a blueprint (see code in S2). Our model is particularly sensitive to (1) the relationship between forest management scenarios and rates of deforestation, (2) the relationship between harvest intensity and forest growth rates, (3) the response of different taxa to forest management and deforestation, and (4) total forest area. Our generic analysis identified low-intensity RIL as providing the best overall outcomes at the forest management unit scale; however, at the landscape scale, multiple scenarios may be optimal due to spatial variation—particularly in the five variables listed above.

Variables other than those we considered are also critical to designing better tropical forestry landscapes, including tree regeneration requirements, soil carbon, preharvest necromass, reforestation, other biodiversity metrics, other ecosystem service and human well-being metrics, costs and profitability, leakage, market access, and community-based forest management. Landscape approaches that attempt to optimize across some of these variables—like the triad approach to forest management developed for temperate forests—may be applicable in the tropics (Tittler et al. 2016).

Regeneration requirements of commercial tree species that dominate timber markets require geography-specific analyses. Low-intensity single tree selection systems restricted to felling large diameter trees favor regeneration of shade-tolerant tree species, yet in many forests the mature trees being removed are relatively light-demanding species like *Swietenia macrophylla*, *Shorea leprosula*, and *Entandrophragma cylindricum* (Fredericksen & Putz 2003). In such cases, more intensive shelterwood cuts could be used to secure regeneration of light-demanding commercial species (Ashton et al. 2001).

Conversion of forests to plantations can reduce soil carbon (Guo & Gifford, 2002), but we judged the evidence for this effect insufficiently consistent to include as a model assumption for this analysis (Powers et al. 2011), and we are not aware of literature offering an empirical basis for assigning generalized differential soil carbon impacts to alternative levels of selective logging intensity and logging practices. An important exception is plantation development involving peat drainage, where soil carbon losses are large and more important to account for (van der Werf et al. 2010).

Even without considering soil carbon, we conclude that replacing extensive native forest management with smaller footprint intensive plantations is unlikely to offer carbon and species richness benefits; however, we did not consider plantations as a reforestation strategy. Reforestation of deforested areas with intensively managed plantations usually involves gains of biomass and soil carbon, rather than losses, offering carbon and other ecosystem service benefits. Also, while we did not find sufficient quantitative data in the literature to parameterize more nuanced improved plantation systems, plantations with multiple tree species can accumulate biomass faster than monospecific plantations (Erskine et al. 2006) and provide other ecosystem service benefits (Montagnini & Porras 1998).

Species richness is a crude biodiversity metric for assessing the tradeoffs between intensive and extensive timber production. We encourage more refined biodiversity metrics for regional applications of our model. For example, species composition or individual species abundance metrics are needed where conservation of a specific set of threatened, endemic, or locally important species is a priority. These metrics require knowledge of the community composition of the locality being considered. No global estimates are yet available for changes in community composition due to selective logging.

While much work remains to understand how forestry landscapes can be designed to achieve conservation and human well-being objectives, we conclude that both the direct and indirect effects of forest management are central to understanding biodiversity and climate impacts. For tropical countries with extensive selective logging, we emphasize the large potential for reduced-impact logging, certification, and improved land tenure to contribute to meeting carbon emissions reduction goals. The argument for a sustainable “sharing” approach to tropical forestry landscapes depends largely on the extent to which loggers can resist, rather than catalyze deforestation. Indeed, natural forest management is one of the few, if not only, widespread business models that can generate a global commodity from the land without displacing the bulk...
of natural carbon and biodiversity—if best practices are used.

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Author contributions

BG designed study, conducted the literature review, piloted modeling analysis, and led writing of the manuscript. RG conducted the carbon modeling analyses, improved the study design, reviewed literature, and contributed to the manuscript. ZB conducted biodiversity and carbon uncertainty modeling, improved the study design, reviewed literature, and contributed to the manuscript. BG, RG, and ZB made similar overall contributions. FP contributed to the study design, model parameters, reviewed literature, and revised the manuscript.

Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher’s web site:

**Figure S1.** We identified a linear relationship (black line, \( P < 0.001 \), adjusted \( R^2 = 0.57 \)) between biomass carbon emissions (MgC ha\(^{-1}\)) and harvest intensity (m\(^3\) ha\(^{-1}\)) as a function of findings from publications (see Table S1 for details).

**Figure S2.** Cumulative net annual committed emissions of biomass carbon during two harvest cycles (60 years) under four alternative scenarios that vary in wood production intensity and extent of timber production across a 30,000 ha forest area.

**Figure S3.** Relative species losses of five taxonomic groups under four alternative scenarios that vary in wood production intensity and extent of timber production across a 30,000 ha forest area.

**Supplemental text 1.** Additional method details including description of six scenarios, estimation of carbon and biodiversity outcomes, and model parameters.

**Table S1.** Selective logging studies used to determine harvest intensities relative to aboveground carbon stocks of the forest and committed emissions per timber volume extracted.

**Table S2.** Timber volume growth rates in tropical forestry plantations.

**Table S3.** Summary of carbon density, logging emissions, harvest intensity, and growth/sequestration rates reported in the literature for selectively logged tropical forests.

**Supplementary Materials 2 (S2):** R-code used for modeling carbon fluxes and biodiversity outcomes.

References

Abood, S., Lee, J.S.H., Burivalova, Z., Garcia-Ulloa, J. & Koh, L.P. (2014). Relative contributions of the logging, fiber, oil palm, and mining industries to forest loss in Indonesia. *Conserv. Lett.*, 8, 58-67.

Ashton, M.S., Mendelsohn, R., Singhakumara, B.M.P., Gunatilleke, C.V.S., Gunatilleke, I.A.U.N. & Evans, A. (2001). A financial analysis of rain forest silviculture in southwestern Sri Lanka. *For. Ecol. Manage.*, 154, 431-441.

Asner, G.P., Knapp, D.E., Broadbent, E.N., Oliveira, P.J.C., Keller, M. & Silva, J.N. (2005). Selective logging in the Brazilian Amazon. *Science*, 310, 480-482.

Blackman, A. (2015). Strict versus mixed-use protected areas: Guatemala’s Maya Biosphere Reserve. *Ecol. Econ.*, 112, 14-24.

Blackman, A., Goff, L. & Planter, M.R. (2015). Does eco-certification stem tropical deforestation? Forest Stewardship Council certification in Mexico. Resources for the Future Discussion Paper, 15-16, 1-45.

Brodie, J.F., Giordano, A.J., Zipkin, E.F., Bernard, H., Mohd-Azlan, J. & Ambu, L. (2015). Correlation and persistence of hunting and logging impacts on tropical rainforest mammals. *Conserv. Biol.*, 29, 110-121.

Burivalova, Z., Lee, T.M., Giam, X., Wilcove, D.S. & Koh, L.P. (2015). Avian responses to selective logging shaped by species traits and logging practices. *Proc. Roy. Soc. B Biol. Sci.*, 282.

Burivalova, Z., Sekercioğlu, C.H. & Koh, L.P. (2014). Thresholds of logging intensity to maintain tropical forest biodiversity. *Curr. Biol.*, 24, 1893-1898.

Busch, J., Ferretti-Gallon, K., Engelmann, J., Wright, M., Austin, K.G. & Stolle, F. (2014). Reductions in emissions from deforestation from Indonesia’s moratorium on new oil palm, timber, and logging concessions. *PNAS*, 112, 1328-1333.

Cowlishaw, G., Pettifor, R.A. & Isaac, N.J.B. (2009). High variability in patterns of population decline: the importance of local processes in species extinctions. *Proc. Biol. Sci.*, 276, 63-69.

Edwards, D.P., Gilroy, J.J., Woodcock, P. et al. (2014). Land-sharing versus land-sparing logging: reconciling
timber extraction with biodiversity conservation. *Glob. Chang. Biol.*, **20**, 183-191.

Elias, P. & Boucher, D. (2014). *Planting for the future*. Union of Concerned Scientists, Washington, D.C.

Erskine, P.D., Lamb, D. & Bristow, M. (2006). Tree species diversity and ecosystem function: can tropical multi-species plantations generate greater productivity? *For. Ecol. Manage.*, **233**, 205-210.

Ferretti-Gallon, K. & Busch, J. (2014). What drives deforestation and what stops it? A meta-analysis of spatially explicit ecometric studies. Center for Global Development. Working Paper, **361**, 1-44

Fischer, J., Abson, D.J., Butsic, V. et al. (2014). Land sparing versus land sharing: moving forward. *Conserv. Lett.*, **7**, 149-157.

Frederiksen, T.S. & Putz, F.E. (2003). Silvicultural intensification for tropical forest conservation. *Biodivers. Conserv.*, **12**, 1445-1453.

Gaveau, D.L., Curran, L.M., Paoli, G.D. et al. (2012). Examining protected area effectiveness in Sumatra: importance of regulations governing unprotected lands. *Conserv. Lett.*, **5**, 142-148.

Gaveau, D.L., Khatriya, M., Sheil, D. et al. (2013). Reconciling forest conservation and logging in Indonesian Borneo. *PLoS One*, **8**, 1–11.

Gibson, L., Lee, T.M., Koh, L.P. et al. (2011). Primary forests are irreplaceable for sustaining tropical biodiversity. *Nature*, **478**, 378-381.

Griscom, B., Ellis, P. & Putz, F.E. (2014). Carbon emissions performance of commercial logging in East Kalimantan, Indonesia. *Glob. Chang. Biol.*, **20**, 923-937.

Guo, L.B. & Gifford, R.M. (2002). Soil carbon stocks and land use change: a meta analysis. *Glob. Chang. Biol.*, **8**, 345-360.

Hansen, M.C., Potapov, P.V, Moore, R. et al. (2013). High-resolution global maps of 21st-century forest cover change. *Science*, **342**, 850-853.

Hooke, R.L.B., Martin-Duque, J.F. & Pedraza, J. (2012). Land transformation by humans: a review. *GSA Today, 22*, 4-10.

International Sustainability Unit. (2015). *Tropical forests—a review*. The Prince’s Charities’ International Sustainability Unit, London.

IPCC. (2014). Climate change 2014: synthesis report. Page 151 in R.K. Pachauri & L.A. Meyer, editors. *Contribution of working groups I, II and III to the fifth assessment report of the intergovernmental panel on climate change*. Intergovernmental Panel on Climate Change (IPCC), Geneva, Switzerland.

Koh, L.P. & Ghazoul, J. (2010). A matrix-calibrated species-area model for predicting biodiversity losses due to land-use change. *Conserv. Biol.*, **24**, 994-1001.

Kormos, C.F. & Zimmerman, B.L. (2014). Response to: Putz et al., Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. Volume 5, Issue 4, August 2012. *Conserv. Lett.*, **7**, 1-2.

Lee, J.S.H., Garcia-Ulloa, J., Ghazoul, J., Obidzinski, K. & Koh, L.P. (2014). Modelling environmental and socio-economic trade-offs associated with land-sparing and land-sharing approaches to oil palm expansion. *J. Appl. Ecol.*, **51**, 1366-1377.

Martin, P.A., Newton, A.C., Pfeifer, M., Khoo, M. & Bullock, J.M. (2015). Impacts of tropical selective logging on carbon storage and tree species richness: a meta-analysis. *For. Ecol. Manage.*, **356**, 224-233.

Miteva, D.A., Loucks, C. & Pattanayak, S.K. (2015). Social and environmental impacts of forest management certification in Indonesia. *PLoS One*, **10**, 1-18.

Montagnini, F. & Porras, C. (1998). Evaluating the role of plantations as carbon sinks: an example of an integrative approach from the humid tropics. *Environ. Manage.*, **22**, 459-470.

Oliver, C.D., Nassar, N.T., Lippke, B.R. & McCarter, J.B. (2014). Carbon, fossil fuel, and biodiversity mitigation with wood and forests. *J. Sustain. For.*, **33**, 248-275.

Paquette, A. & Messier, C. (2010). The role of plantations in managing the world’s forests in the Anthropocene. *Front. Ecol. Environ.*, **8**, 27-34.

Pearson, T.R.H., Brown, S., Murray, L. & Sidman, G. (2017). Greenhouse gas emissions from tropical forest degradation: an underestimated source. *Carbon Balance Manag.*, **12**, 1-11.

Phalan, B., Onial, M., Balmford, A. & Green, R.E. (2011). Reconciling food production and biodiversity conservation: land sharing and land sparing compared. *Science*, **333**, 1289-1291.

Porter-Bolland, L., Ellis, E.A., Guariguata, M.R., Ruiz-Mallén, I., Negrete-Yankelevich, S. & Reyes-Garcia, V. (2012). Community managed forests and forest protected areas: an assessment of their conservation effectiveness across the tropics. *For. Ecol. Manage.*, **268**, 6-17.

Powers, J.S., Corre, M.D., Twine, T.E. & Veldkamp, E. (2011). Geographic bias of field observations of soil carbon stocks with tropical land-use changes precludes spatial extrapolation. *Proc. Natl. Acad. Sci. U. S. A.*, **108**, 6318-6322.

Putz, F.E., Zuidema, P.A., Synamott, T. et al. (2012). Sustaining conservation values in selectively logged tropical forests: the attained and the attainable. *Conserv. Lett.*, **5**, 296-303.

Ruslandi, Cropper, W.P. & Putz, F.E. (2017). Effects of silvicultural intensification on timber yields, carbon dynamics, and tree species composition in a dipterocarp forest in Kalimantan, Indonesia: an individual-tree-based model simulation. *For. Ecol. Manage.*, **390**, 104-118.

Tittler, R., Messier, C. & Goodman, R.C. (2016). Triad forest management: local fix or global solution. Pages 33-45 in G.R. Larocque, editor. *Ecological forest management handbook*. Taylor & Francis Group/CRC Press, Boca Raton, Florida.

van der Werf, G.R., Randerson, J.T., Giglio, L. et al. (2010). Global fire emissions and the contribution of deforestation, savanna, forest, agricultural, and peat fires (1997–2009). *Atmos. Chem. Phys.*, **10**, 11707-11735.

Woodcock, P., Edwards, D.P., Newton, R.J., Vun Khen, C., Bottrell, S.H. & Hamer, K.C. (2013). Impacts of intensive logging on the trophic organisation of ant communities in a biodiversity hotspot. *PLoS One*, **8**, 2-8.