Thresholds of species loss in Amazonian deforestation frontier landscapes

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Abstract: In the Brazilian Amazon, private land accounts for the majority of remaining native vegetation. Understanding how land-use change affects the composition and distribution of biodiversity in farmlands is critical for improving conservation strategies in the face of rapid agricultural expansion. Working across an area exceeding 3 million ha in the southwestern state of Rondônia, we assessed how the extent and configuration of remnant forest in replicate 10,000-ha landscapes has affected the occurrence of a suite of Amazonian mammals and birds. In each of 31 landscapes, we used field sampling and semistructured interviews with landowners to determine the presence of 28 large and medium sized mammals and birds, as well as a further 7 understory birds. We then combined results of field surveys and interviews with a probabilistic model of deforestation. We found strong evidence for a threshold response of sampled biodiversity to landscape level forest cover; landscapes with < 30–40% forest cover hosted markedly fewer species. Results from field surveys and interviews yielded similar thresholds. These results imply that in partially deforested landscapes many species are susceptible to extirpation following relatively small additional reductions in forest area. In the model of deforestation by 2030 the number of 10,000-ha landscapes under a conservative threshold of 43% forest cover almost doubled, such that only 22% of landscapes would likely to be able to sustain at least 75% of the 35 focal species we sampled. Brazilian law requires rural property owners in the Amazon to retain 80% forest cover, although this is rarely achieved. Prioritizing efforts to ensure that entire landscapes, rather than individual farms, retain at least 50% forest cover may help safeguard native biodiversity in private forest reserves in the Amazon.

Keywords: agricultural expansion, farmlands, landscape extinctions, probabilistic models, species richness

Umbrales de Pérdida de Especies en los Paisajes Fronterizos de Deforestación en el Amazonas Ochoa-Quintero

Resumen: En el Amazonas brasileño, propiedades privadas mantienen la mayoría de la vegetación nativa remanente. Entender cómo el cambio de uso del suelo afecta la composición y distribución de la biodiversidad en estas áreas es necesario para mejorar las estrategias de conservación de frente a la rápida expansión agrícola. Trabajando en un área que excede los 3 millones de hectáreas en el estado de Rondônia (sur-este de Brasil), evaluamos cómo la extensión y la configuración del bosque remanente en réplicas de paisajes de 10,000 hectáreas ha afectado la presencia de un conjunto de mamíferos y aves del Amazonas. En cada uno de los 31 paisajes usamos muestreo en campo y aplicamos entrevistas semi-estructuradas a los propietarios para determinar la presencia de 28 mamíferos y aves, de tamaños grandes y medianos, así como 7 especies más de aves de sotobosque. Después combinamos los resultados de los muestreos de campo y las entrevistas con un modelo probabilístico de deforestación. Encontramos evidencias fuertes de la presencia de un umbral en la respuesta de la biodiversidad muestreada relacionado con cobertura de bosque en los paisajes; los paisajes con cobertura de bosque < 30-40% tuvieron un marcado número menor de especies. Los resultados de los censos en campo y de las entrevistas proporcionaron resultados similares. Estos resultados implican que en los
países parcialmente deforestados muchas especies son susceptibles a la extirpación después de reducciones adicionales relativamente pequeñas en el área de bosque. En el modelo de deforestación, para el año 2030 el número de países de 10,000 hectáreas por debajo de un umbral conservativo de 43% de cobertura forestal casi se duplicó, de tal forma que sólo el 22% de los países podría sostener por lo menos el 75% de las 35 especies focales que muestreamos. La ley brasileña requiere que los propietarios rurales en el Amazonas retengan el 80% de la cobertura forestal, aunque esto rara vez ocurre. Priorizar los esfuerzos para asegurar que países enteros, en lugar de propiedades individuales, retengan por lo menos el 50% de la cobertura forestal puede ayudar a proteger la biodiversidad nativa en propiedades privadas en el Amazonas.

**Palabras Clave:** expansión agrícola, extinción en paisajes, modelos probabilísticos, riqueza de especies, tierras de agrícolas

**Introduction**

The replacement of natural vegetation by agriculture to feed a growing and more affluent human population is the most severe threat facing global biodiversity (Balmford et al. 2012). Creating national parks is often an effective way to counter agricultural expansion (Gaston et al. 2008; Soares-Filho et al. 2010), and there have been multiple calls for an increase in the coverage of protected areas globally following continued reports of species declines (Rodrigues et al. 2004; Butchart et al. 2010). However, limits on the amount of land that can be set aside in reserves means that the long-term success of biodiversity conservation will also depend on the protection and restoration of native vegetation in private lands (Soares-Filho et al. 2006; Chazdon et al. 2009; Gardner et al. 2009).

The Amazon basin is host to a substantial proportion of the world’s terrestrial and freshwater species (Gentry 1988; Peres & Janson 1999), yet ongoing agricultural expansion (Walker et al. 2009) and climate change (Davidson et al. 2012) mean that the fate of the region remains uncertain (Gardner 2013). In Brazil, which harbors nearly two-thirds of the basin, 18% of the Amazon forest has already been cleared through infrastructure development and agricultural expansion, whereas large areas of remaining forest outside the reserve network are in a fragmented and degraded state (Soares-Filho et al. 2006; Pereira et al. 2010). Although rates of deforestation have decreased in the Brazilian Amazon, extensive areas of forest are still being cleared for agriculture (e.g., 4656 km² in 2012 [INPE 2012]), leaving behind human-modified landscapes comprising mosaics of production areas and fragmented regenerating and primary forest (Gardner et al. 2009).

Several studies have found a strong and negative impact of forest fragmentation on biodiversity in the Amazon basin (Laurance et al. 2002b; Lecs & Peres 2006) and elsewhere (Ewers & Didham 2006). However, the majority of this work has been limited to individual patches, and very few studies have evaluated how the number of forest associated species are affected by changes in forest cover at the landscape scale (Pardini et al. 2010) and across a full gradient of deforestation (Gardner et al. 2013). Furthermore we are not aware of any attempts to use models of landscape-scale biodiversity loss derived from field observations to extrapolate the biodiversity impacts of agricultural expansion across entire regions; previous biodiversity scenario work relied on theory (Wearn et al. 2012), secondary data regarding species range sizes (e.g., Soares-Filho et al. 2006; Bird et al. 2012), or species life-history traits (e.g., Dale et al. 1994).

We used a combination of direct observation and interview-based data on the distribution of a subset of mammal and bird species from thirty-one 10,000-ha landscapes in Rondônia to provide one of the first assessments of changes in landscape-level tropical biodiversity across a full gradient of forest loss (12–90% of forest cover). Although we expected that lower levels of forest cover would correspond to a reduction in the number of observed species, we hypothesized that landscape-level patterns of species extinction may be nonlinear. Specifically, we aimed to identify the most important landscape features shaping changes in species richness; identify whether there were threshold changes in the relationship between forest cover and species richness; and develop a model combining observed species-richness responses to changes in forest cover with a probabilistic model of deforestation to estimate likely effects on biodiversity from continued agricultural expansion in northern Rondônia by 2030. Based on these analyses we propose a framework and set of recommendations for landscapes where conservation interventions are most urgently needed in this high conservation-priority region.

**Methods**

**Study Region**

Our study region was in the northern half of the Brazilian Amazon state of Rondônia in the interfluvium of the Mamore-Madeira and Ji-Parana Rivers (Fig. 1). The entire region has a tropical hyper-humid climate, more than 8 wet months, and average wet-season temperatures above 23.5 °C (Cochrane & Cochrane 2006). The settlement of over 30,000 families from the south of Brazil since the start of the 1970s has resulted in substantial forest clearance and resource extraction (Browder 1994).
Figure 1. The (a) Amazon basin, Brazil, and the state of Rondônia, southwest of the Brazilian Amazon, (b) land-cover categories from the forest monitoring system of the Brazilian Amazon Prodes in 2010 (INPE 2012) and landscapes where sampling took place (selected landscapes), and (c) field-work locations in the landscapes and locations where sampling and interviews with land owners were conducted in 1 of the 31 landscapes.

To select study landscapes, we classified Landsat images from 2010 (30-m resolution) (INPE 2012) into areas of forest, open areas (production land), and water. The supervised classification procedure was done using ERDAS. We then divided the region into 1223 squares of 10,000 ha (hereafter landscapes) and calculated the amount and configuration of remaining forest cover in each landscape with Fragstats 3.3 (McGarigal et al. 2002). Fragmentation measures included indicators of fragment clumping and patch density, largest patch index, and number of forest fragments. Thirty-one landscapes were selected across a deforestation gradient (12–90% forest cover [Supporting Information]). Twenty-seven were on private lands, and 4 were within protected areas that were the only areas in the region with almost complete forest cover (Fig. 1). For each landscape, we used the landscape centroid to measure the distance to the closest landscape with >90% forest cover. We determined the difference in mean forest cover from adjoining landscapes by subtracting from the total forest cover of the sampled landscape the mean forest cover value of the 8 adjoining landscapes. We focused on landscape versus patch-scale attributes (e.g., forest fragment size and fragment isolation). We were cognizant that changes in habitat configuration and
amount are strongly correlated between the 2 scales, landscape scale properties are increasingly recognized to have a dominating effect on local biodiversity patterns, and landscapes offer a more appropriate scale of analysis for guiding regional conservation priorities (Gardner et al. 2009; Fahrig 2013).

**Biodiversity Sampling**

We determined species richness and relative abundance of 28 medium or large mammals and birds and 7 understory bird species (Supporting Information) in all 31 landscapes based on direct field observations and semistructured interviews with 216 farmers. Sampled species occur widely across the study area, are associated with forest habitat, and include individual taxa that have distinct responses to habitat loss (e.g., understory insectivorous birds [Ferraz et al. 2007] and hunting pressure [Peres 2001]). Although some of our target species may use nonforest areas (e.g., Capybara), all are associated with forest and most are forest dependent.

Direct sampling was conducted in 1-km transects in forest fragments (total n = 170). Semistructured interviews were conducted with local farmers who owned land around the fragments (total n = 216) during the dry season of 2011. We used a stratified-random procedure following Gardner et al. (2013) to sample in the landscapes. For the direct sampling, we used a standard density of 1 transect/500 ha of forest to determine the number of transects in each landscape according to the amount of remaining forest cover. Transects were distributed randomly in the remaining forest areas, and we maintained a minimum distance of 2 km between transects to minimize interdependence of samples. Each transect was visited once. In landscapes with low forest cover, we sampled a minimum of 3 transects. For medium and large size birds and mammals, species were observed directly with binoculars or presence was determined based on other evidence such as footprints and feces. Using a playback technique similar to Antongiovanni and Metzger (2005), stops of 10 min were made along each transect at 100, 400, 700, and 1000 m to collect evidence of the presence of the 7 understory birds species. Insectivorous bird species are frequently used as sensitive focal species for studies on the effects of fragmentation in the Amazon basin (e.g., Stratford & Stouffer 1999). Landowners of each forest transect were shown color pictures of the medium- to large-size birds and mammals in a random order and asked about their presence and absence. To assess the quality of the information, we asked each respondent for the local name, typical behavior of the species, and what the vocalization of the species sounded like. Both data collection techniques, direct sampling and semistructured interviews, have been widely used to sample these taxonomic groups elsewhere in the Amazon (Michalski & Peres 2005) (Supporting Information).

**Data Analyses**

**MODELING OF BIODIVERSITY RESPONSES TO LANDSCAPE CHANGE**

We used generalized linear models with a Poisson distribution and log link function to model how differences in species richness for both direct observations and interview data may be related to selected explanatory variables at both landscape and regional scales. We used the fragmentation measurements as landscape-scale explanatory variables and the distance to landscape with >90% forest cover and the difference in average forest cover compared with adjoining landscapes as regional-scale explanatory variables. We first evaluated the correlation structure among all explanatory variables at both landscape and regional scales. We discarded variables with a Pearson correlation coefficient exceeding 0.7 based on the expected importance of each candidate variable. Quadratic relationships were included for total amount of forest cover because we expected that patterns of local species extinction may be nonlinear. The number of transects per landscape was included to account for slight differences in sampling effort. The same models were run with a Bootstrap estimator of species richness, which performs well compared with other nonparametric estimators (Colwell & Coddington 1994).

To identify the most plausible models, we performed a model selection procedure based on Akaike information criterion (AIC) in which we compared AIC model weights across models with all possible variable combinations (AIC for sampling data and AICc for interview data). We considered only those models with a ΔAIC or AICc weight below 2 as having strong empirical support from the sample data (Burnham & Anderson 2002). The relative importance of each explanatory variable in explaining differences in species richness was also estimated by summing the Akaike weights of all models in which that variable appeared (Burnham & Anderson 2002).

To identify the possible thresholds in levels of species richness, we used piece-wise regression in the segmented package in R (Muggeo 2004) on the interaction between forest cover and species richness data from both direct observations and interviews. Piece-wise regression splits explanatory variables into 2 or more linear regressions to locate points where the linear relationship changes. The identification of thresholds or break points is estimated using different starting points and identified using the highest R² value (Muggeo 2004). Thus, we used this analysis to detect discrete changes in how species richness relates to differences in the level of deforestation.

**CONSEQUENCES OF FUTURE REGIONAL DEFORESTATION**

Based on our modeled relationships of species richness responses to landscape change, we used a probabilistic model of deforestation (Rosa et al. 2013) to estimate how deforestation under a business-as-usual scenario
(BAU) might affect patterns of focal species richness in northern Rondônia. With this model we estimated the forest cover for each landscape in the study area in 2030 from the mean forest cover estimated after 100 iterations to account for uncertainty in the process (Supporting Information).

To help inform conservation priorities across northern Rondônia, we modeled patterns of focal species richness in 2010 and 2030 by combining the species-threshold and deforestation models. This enabled us to estimate the number of expected species across the unsampled parts of the region in each period based only on changes in forest cover. For this combined analysis, we used the model obtained from the threshold analysis built from the interview-based data because it had the highest sample size and lowest standard error in the threshold analyses. We then categorized landscapes into 4 categories according to the likely marginal biodiversity benefits of targeted efforts to stop further deforestation. All statistical analyses were conducted in R version 2.13.0 (R-Development Core Team 2011).

Results

Forest Loss, Landscape Features, and Changes in Species Richness

The expansion of agriculture in recent decades has led to a loss of approximately 41% of the original forest (2 million ha) in the study region during the last 40 years. As forest cover declined the distance among remaining forest fragments increased. This was evident in the forest aggregation measurements across the region. The largest patch index showed a positive correlation with total forest cover, whereas the number of fragments decreased as forest cover increased in the landscapes. The 31 sampled landscapes followed similar trends in forest configuration patterns; thus, they were representative of patterns across the wider region of northern Rondônia (Supporting Information).

Results from direct observations and semistructured interviews were consistent and robust to differences in sampling effort in both transects and number of interviews (Supporting Information). Both the quadratic and the linear terms of forest cover change were always retained in the best models of observed species richness (ΔAIC < 2.0), but only the quadratic term for models of bootstrap richness were retained in the best models (Supporting Information). Top ranking models for both direct observations and interview data explained the majority of the observed variation in species richness (minimum of 0.77 explained deviance), which demonstrates that the variables included in the best models effectively explained the observed variation. The results of model averaging confirmed that total landscape forest cover (both linear and quadratic forms) was the variable with

Figure 2. Responses of species richness to landscape-scale forest cover on the basis of (a) direct observations, (b) bootstrap richness estimates, and (c) landowner interview data. Threshold analyses are based on piece-wise regression (solid vertical line, location of the break-point; dashed lines, the 95% CI).
Thresholds in Species Richness Levels

We found clear evidence of a threshold in the relationship between species richness and landscape-level forest cover from 19% to 43%; threshold values were similar across the 3 different models (29% [SE 6] raw data, 31% [SE 12] bootstrap richness and 36% [SE 4] interview data) (Fig. 2a-c). This approach produced lower AIC and AICc values than the results obtained from quadratic models with the same set of variables (127.31 < 150.0 AIC for raw data, 140.52 < 156.38 AIC bootstrap richness, and 1003.38 < 1131.7 AICc interview data). Below this threshold species richness declined quickly as forest cover decreased (Fig. 2a-c). Our results suggested that 1–2 species could be lost for every 10% loss of forest cover in landscapes above the threshold (95% CI bounded by 1 to 2 species for transect base and bootstrap data and 1 species for interview data). In contrast, 2–8 species were likely to be extirpated for every 10% loss of forest cover in landscapes below the threshold (95% CI bounded by 2–7 species for transect-base data, 2–8 for bootstrap species richness, and 4–6 species for interview-based data).

Combined Models of Species Richness and Forest Cover

With a conservative threshold of 43% forest cover (maximum threshold value obtained from the threshold model with the Bootstrap species richness plus SE) in 2010, 37% of the 1223 10,000-ha landscapes had a level of forest cover below the threshold, 30% had an intermediate level of forest cover (45% to 70%), and 33% had more than 70% cover (Fig. 3a). Under a BAU scenario forest cover was predicted to be lost at an estimated rate of 1.8%/year. Areas with higher probabilities of deforestation were concentrated in western and northern Rondônia, although southern and central Rondônia retained some areas of high deforestation (Fig. 3b). This was predicted to increase the proportion of landscapes with less forest than the threshold level from 37% to 64% by 2030 (Fig. 3c). Landscapes with intermediate forest cover were predicted to fall from 30% in 2010 to 14% in 2030; 22% of landscapes were expected to retain more than 70% forest cover by the same year (Fig. 3c).

Patterns of landscape-level species loss mirrored the loss of forest cover, demonstrating that the regional biota is likely to be severely impoverished by 2030 (Fig. 4a-b). The area with the greatest projected losses of sampled biodiversity was in the central northern part of the region (Fig. 4c), which was characterized by landscapes closer to the forest cover threshold in 2010. Projections of regional patterns of species loss were similar irrespective of data source (interview or raw data, observed or estimated species richness [Supporting Information]).

By combining models on projected changes in forest cover and species richness, we were able to group landscapes into 4 categories of priority for conservation interventions (Fig. 5). The highest priority group (33% of landscapes) was represented by those landscapes with a level of forest cover just under the 43% threshold in 2010 and by those whose forest cover was expected to cross the threshold within the next 20 years. The group with an intermediate level of priority (19% of landscapes) was landscapes with forest cover from 10 to 30% in 2010; small additional losses of forest cover were predicted to result in disproportionate reductions in species richness. The grouping with a low priority ranking (14% of landscapes) was comprised of landscapes with either a very low level of forest cover (<10%) in 2010 (thereby requiring high investment in restoration, 9% of landscapes) or very low expected deforestation (<10% loss of forest cover over the modeled period; that is, areas located either inside national protected areas or on indigenous lands [25%]) (Fig. 5).

Discussion

Landscape Forest Loss, Biodiversity, and Conservation

We found that total forest cover at the landscape scale was of overriding importance in determining patterns of species richness of medium–large sized birds and mammals and focal understory birds in partially deforested landscapes in the southern Amazon. Similar results have been found in other landscape-scale studies elsewhere, including in the Brazilian Atlantic Forest (Pardini et al. 2010), Australia (Radford & Bennett 2007), and Finland (Heikkinen et al. 2004).

Our most striking finding was evidence of a threshold response in the relationship between levels of forest cover and species richness of sampled taxa. Although the existence of thresholds in ecological responses to landscape change has been contested (Lindenmayer & Luck 2005), the existence of a similar nonlinear pattern in both direct observation and interview-based data strengthens the case for our result being a real effect. We also found very similar results when analyzing a subset of individual threatened species (Supporting Information). Nonlinear responses of biodiversity to landscape-scale deforestation have also been found by others. For example, working...
in an already heavily fragmented region of southeast Australia, Radford et al. (2005) found a threshold change in species richness at about 10% forest cover, and Pardini et al. (2010) and Martensen et al. (2012) reported thresholds comparable to our findings from Rondônia of about 30% forest cover in studies of small mammals and ant-birds in the Atlantic Forest.

Apart from differences in total landscape-level forest cover, other correlated changes are also likely to influence patterns of species richness in farmed landscapes, including increases in fragment isolation and decreases in fragment size in more deforested landscapes. For example, Ferraz et al. (2005) found an increase in fragmentation once landscapes have <35% of forest cover.

Figure 3. Current patterns of deforestation in northern Rondônia, Brazil: (a) forest cover in individual landscapes in 2010 relative to a 43% cover threshold, (b) percent forest loss from 2010 to 2030 according to the probabilistic model of deforestation, (c) percent expected forest cover in each landscape by 2030 relative to 43% cover threshold.
Figure 4. Species richness (a) observed in 2010 and (b) estimated for 2030 and (c) species loss as a result of expected deforestation.
Figure 5. Landscape-scale classification of areas by conservation priority based on expected forest cover change by 2030 (highest priority landscapes, expected deforestation reduces forest cover below the conservative 43% threshold; intermediate priority landscapes, landscapes with 10–30% forest cover in 2010; low-priority landscapes, landscapes expected to retain >43% forest cover by 2030; lower priority landscapes are those with very low forest cover, landscapes with <10% forest cover or, those that are expected to lose <10% forest cover by 2030 in regions with very high forest cover [forests inside protected areas]).

Although disentangling the linked effects of vegetation extent and configuration is challenging (Fahrig 2003), Pardini et al. (2010) demonstrated that species-fragment area effects can be highly dependent on differences in landscape-level forest cover, a finding that suggests reduced connectivity between remaining fragments can be a key part of the mechanism driving local species extinctions in deforested landscapes (see also With & King 2001). In addition to the effects of forest area and fragmentation, the degradation of remaining fragments due to correlated disturbances from logging and fire, as well as unsustainable hunting and extraction of other resources, can also threaten the future of species that depend on relatively undisturbed tracts of forest.

Modeling the Loss of Biodiversity Due to Regional Deforestation

The BAU projections of deforestation from our probabilistic model suggest that the conservation value of farmed landscapes in northern Rondônia is likely to decline substantially. The number of landscapes projected to have <43% forest cover (i.e., our conservative observed threshold) were predicted to double in 20 years (Fig. 3c). Increasing demand for beef (Pacheco & Poccard-Chapuis 2012) and soya (Macedo et al. 2012) and associated infrastructure developments (Garrett et al. 2013), means this deforestation scenario is entirely plausible. Two new dams, Jirau and Santo Antonio, are under construction in the Madeira River, and another was recently approved in the Cachoeira of Ribeirao. These dams will further increase demand for land in the north of the region, an area that until recently was mostly covered by forest.

By calibrating changes in forest cover to changes in species richness it is possible to compile a more accurate assessment of the actual conservation status of landscapes in this dynamic region of the Amazon. The highest deforestation is most likely to occur farther north in landscapes with high forest cover (>60% in 2010), where new infrastructure developments such as roads are anticipated that will increase accessibility and the likelihood of intense forest clearance (Fig. 5b). However, higher decreases in species richness are expected to occur northeast of Rondônia (Fig. 4c). Some of these landscapes coincide with areas of active deforestation in 2010. Landscapes with the lowest projected changes in species richness are in central Rondônia, which has very low forest cover because it received the first settlements in this region (Fig. 4c) (Laurance et al. 2002a; Soler et al. 2009). Landscapes exceeding 70% forest cover are
Guiding Conservation Priorities at Regional Scales

Our finding of a threshold effect of forest loss on Amazon biodiversity has important implications for environmental legislation in the Brazilian Amazon. The Forest Code (Law-No.12.651 2012) is the primary mechanism for protecting native forest and associated biodiversity in farmlands in the Amazon basin. The Forest Code is applied at farm scale and requires farmers to retain at least 80% of their land in native vegetation. Although compliance with this law is assessed at the scale of individual properties, the fact that in 2010 more than 69% of the landscapes outside protected areas in this region had <70% forest cover (Fig. 3a) suggests that many of the farms in the region are not in compliance.

Efforts to maintain and enhance the conservation status of landscapes in northern Rondônia, and agricultural frontiers in the Brazilian Amazon more generally, depend critically on the way in which compliance with the Forest Code is interpreted and the requirements imposed by state zoning legislation already approved in this state (which can adjust the legal requirement for forest protection from 80% to 50% in areas consolidated for agricultural development). For example, the option to compensate for a private reserve deficit through the purchase of forests elsewhere (off farm) could, depending on how it is implemented, either alleviate or exacerbate efforts to maintain forest cover above a minimum landscape level throughout the region. In addition, the process of subdividing existing landholdings may result in large areas of forest being retained only in areas with larger properties (Michalski et al. 2010). We suggest that a shift in focus of efforts to achieve compliance with the Forest Code, from the scale of individual farms to the scale of entire landscapes (e.g., 10,000 ha or larger), could greatly facilitate efforts to maintain and restore the conservation status of farmed landscapes. Additional interventions, such as payments for environmental services (PES) (e.g., through Water Funds [Goldman-Benner et al. 2012]), are likely to be needed to incentivize the kind of collective action necessary to design and implement landscape-scale conservation strategies. Nevertheless, PES markets remain largely voluntary and poorly regulated, and there is little herowithal to differentiate opportunity costs according to social and economic characteristics of farmers (Newton et al. 2012).

At a regional scale, our system of prioritizing individual landscapes identified areas likely to experience a drop in forest cover below the 43% threshold. Although other prioritization processes may select different landscapes as top priorities (e.g., those with extensive forest cover) and be subject to change due to constraints imposed by differences in socioeconomic condition and land use (Ferraz et al. 2009), our focus was on the avoidance of further species extirpation in human-modified landscapes, and our method offers a very cost-effective way to identify priorities to achieve this—through avoided deforestation and restoration efforts.

As for many other areas of the tropics, northern Rondônia requires urgent conservation action in areas outside reserves if widespread biodiversity loss is to be averted (Bird et al. 2012; Wearn et al. 2012). Our analyses revealed the biological consequences of forest loss, and our method can help identify clear priorities for conservation interventions according to differences in current forest cover and expected rates of deforestation. The protection of native forest cover above the conservative threshold of approximately 40% forest cover offers a simple and cost-effective way to limit biodiversity loss in these globally significant forests. We believe a similar approach may also be useful in restoration projects to define native vegetation cover targets on the basis of a threshold of species loss.

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Supporting Information

Figures showing location of the study region (Appendix S1), information on field sampling (Appendix S2), an explanation of the probabilistic model of deforestation (Appendix S3), results of fragmentation and sampling completeness (Appendix S4), results of model selection (Appendix S5), figures on expected species loss (Appendix S6), and additional information on the threshold analysis for threatened species and logistic
regression performed for individual species (Appendix S7) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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