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ARTICLE

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Contrasting tree-cover loss and subsequent land cover in two neotropical forest regions: sample-based assessment of the Mexican Yucatán and Argentine Chaco

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ABSTRACT

The neotropical-forest's northern and southern extremes, covering the Mexican Yucatán and the Argentine Chaco, have among the highest rates of recent tree-cover loss in the biome. This study contrasts the character of loss in these regions, estimating proportions of types of loss and subsequent land cover. It is based on two-stage probability sampling design and field and satellite-image surveys. All estimates include uncertainties, which could be further reduced via model-assisted estimation or additional sampling. This approach can be replicated in other regions to estimate types of loss and associated land cover from a definitive, in-situ perspective. The character of loss in the two areas differed greatly. That in the Yucatán was 54% temporary, mostly under fallow or selectively logged, while that in the Chaco was 85% permanent, split nearly equally between crops and pasture. These data contribute to a quantitative basis for studies of socio-economic drivers of neotropical deforestation.

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Remote sensing; deforestation; tropical forest; monitoring; sampling design

1. Introduction

The neotropical forest biome has experienced a continual loss of tree cover since 2000 (Hansen et al., 2013). The biome's northern and southern extremes, primarily covering the Mexican Yucatán and the Argentine Chaco, are of particular importance since they are humid-to-dry forest ecotones, have rich biological endemism and provide critical biological and hydrological resources to their regions' communities (Garcia-Frapolli et al., 2009; Prance, 2006; Smardon & Faust, 2006; Werneck, 2011). These areas also have had among the highest regional rates of tree-cover loss in the biome over these years (Hansen et al., 2013). Previous studies indicate that very different land-use dynamics have contributed to the loss, where the Yucatán peninsula in southeastern Mexico is a region of mostly traditional small-holder, rotational agro-pastoral use, while the Chaco is a region of a relatively recent boom in extensive, rain-fed, mechanized agriculture.

The tropical forests of the Yucatán, extending across the states of Yucatán, Campeche, and Quinana Roo, range from deciduous to semi-deciduous, with rainfall ranging from approximately 900 to 1400 mm y^{-1}. Applying a 50% threshold to the Global Land Analysis and Discovery (GLAD)

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per cent tree-cover product (Hansen et al., 2013), from the University of Maryland (UMD), produces an area estimate of 108,069 km^2 in 2000 for the three states. The GLAD data on tree-cover loss within this extent between 2000 and 2012 indicate 9% of loss over this period, or 0.75% y^{-1}. This compares to official national rates of forest loss of 0.1% y^{-1} to 0.2% y^{-1} during 2000 to 2015 (FAO, 2015). The difference between national and Yucatán rates indicates a concentration of Mexican deforestation there but also may relate to differences in definitions in the two sets of data.

The Yucatán peninsula has a history of dense population settlement and intensive agricultural use during the Mayan period in the middle of first millennia AD, followed by sparse population after the Mayan collapse in the following centuries that remained so until the middle of the nineteenth century, followed by a period of increasing population and deforestation since the latter nineteenth century (Abazaid & Coomes, 2004; Faust, 2001; Turner et al., 2001; Vaca, Golicher, Cayuela, Hewson, & Steininger, 2012). Land use remains mostly traditional shifting cultivation for subsistence, locally called *milpas*, managed by local rural communities, or *ejidos*. Other common land uses are commercial farming, e.g. of corn sugar cane, soybean, sorghum and henequen (Agave fourcroydes), cattle ranching and selective logging, mostly in the extensive forest areas in the southern and eastern portions of the peninsula (Busch & Geoghegan, 2010; Ellis, Romero Montero, & Hernandez Gomez, 2017). The Yucatán forests also experience occasional severe damage from hurricanes (e.g. Rogan et al., 2010) and wildfire. While these forests have been described as very resilient and able to recover from natural and anthropogenic disturbances (Snook, 1993), they are highly anthropogenic rather than pristine (Gómez-Pompa & Kaus, 1992; Romero-Duque, Jaramillo, & Pérez-Jiménez, 2007; Turner, 1978).

The Chaco phytogeographical region is a dry-woodland ecosystem extending over northern Argentina, most of western Paraguay and southeastern Bolivia. The Argentine Chaco extends over the provinces of Formosa, Santiago del Estero, Chaco, northern Santa Fe, northeast San Luis, east Tucuman, northern and western Cordoba and the eastern-lowland portion of Salta (e.g. Frate, Acosta, Cabido, Hoyos, & Carranza, 2015). This lies within in a rainfall gradient from 500 to 1200 mm y^{-1}, increasing to the south (Adamoli, Sennhauser, Acero, & Rescia, 1990). Applying the same threshold to the GLAD product produces an area of 125,971 km^2 in 2000. Tree-cover loss from 2000–2012 indicated by GLAD data was over 2% per year, one of the highest regional rates of tree-cover loss globally in recent years (Hansen et al., 2013).

The Chaco does not have the early history of dense population and intensive agriculture as in the Yucatán, and remained lightly populated throughout most of the centuries following European settlement. Traditional cattle ranching has been the most common use during the past century, while the region still remained mostly forested. A boom in industrial soybean farming has occurred over the past two decades, initially in the more humid areas but extending into all but the driest areas (Grau & Aide, 2008; Boillat et al., 2017). The region has been described as undergoing a transition from one dominated by extensive cattle grazing to one dominated by intensive agriculture (Grau & Aide, 2008).

These two regions demonstrate how very different dynamics can lead to extremely high rates of tree-cover loss and that loss can come in different forms. Automated satellite mapping may not be able to provide reliable estimates of parameters needed to characterize differences between these two regions, such as the specific land-cover types following loss. A sample-based approach with field surveys and image interpretation offers a useful alternative.

2. Background

2.1. Remote sensing and sample-based estimation

Consistent, accurate estimates of land-cover change provide a foundation for understanding the potential effects of change on ecosystems and the resources they provide. These estimates are also an important component of national and subnational resource management and reporting mechanisms, including reporting commitments specified in international agreements such as the

United Nations (UN) Framework Convention on Climate Change, the UN Convention to Combat Desertification and the UN Convention on Biodiversity (UNCBD, 2017; UNCCD, 2017; UNFCCC, 2014). Regarding forests, these commitments require estimates of areas of loss as well as the proportions of conversion for different uses.

Systems for monitoring forests and tree cover have advanced to a level where precise and accurate national to global estimates of loss are generated from automated or semi-automated analyses of satellite imagery (e.g. Hansen et al., 2013; Harper, Steininger, Tucker, Juhn, & Hawkins, 2007; Margono, Potapov, Turubanova, Stolle, & Hansen, 2014; Potapov et al., 2014, 2017). These estimates have mostly been produced using Landsat satellite data, because of their free cost, excellent coverage and archive, and 30-m spatial resolution that is fine enough to reveal sub-hectare changes in land cover.

The raster maps created by the analysis of multi-temporal satellite data provide the most accurate characterization of the spatial distribution of land-cover change; nonetheless, estimates of areas of change derived directly from these raster products have biases. Sampling specific locations, such as those represented by individual cells in these products, can be used to provide an estimate of map bias as well as an unbiased estimator of areas with known uncertainties (Olofsson et al., 2014; Stehman, 2013). For a set of sample locations, satellite imagery can be visually interpreted, depending on the type of imagery and classes being interpreted, or visited in the field. Both approaches can provide highly confident assignments of land-cover type or other attributes for each sample location, in which case sampling variability is the primary source of uncertainty in a sample-based estimate.

The methods described in Olofsson et al. (2014) and Stehman (2013) address sample-based estimation of a class that has been mapped. Sampling can also be used to estimate more specific classes or entirely different classes or attributes than those in a map that is used to support the sample design. For example, mapping crop types over large areas remains mostly in the research domain, and examples of accurate results are mostly of broad classes or of large-scale monocrops (e.g. De Sy et al., 2015; Killeen et al., 2008; Song et al., 2017). However, a map with classes of the probability of crop cover can be used to define strata for locating sample units, and these can be assessed individually for crop types that are not classes in the map. This was done to estimate specific crop areas in the US and elsewhere by Song et al. (2017) and King et al. (2017). Automated classification of bare ground is currently reliable, but not that of more specific cover classes therein, such as mines versus commercial and residential expansion. Ying et al. (2017) used a bare-ground map to target sampling to estimate the land-cover classes that most contributed to the increased area of global bare ground. In studying forests with optical satellite imagery, the spectral distinction among different land-use types following loss or different levels of damage from logging or natural events, is much weaker than that between forest and non-forest land. Potapov et al. (2014) and Tyukavina et al. (2017) used maps of tree-cover loss to target sampling to estimate different land-uses that most contributed to the loss in Peru and Brazil. In another example, McGroddy et al. (2013) stratified the Mexican Yucatán using satellite-derived images of geenness indices following Hurricane Dean to estimate levels of damage from sample points in the field.

Satellite mapping with sample-based estimation is a powerful combination since they can be used to provide the best estimates of the distribution of broad classes from the maps, estimates of map bias for these classes from the samples, best estimates of the areas of these classes from the samples and additional estimates of more-precise classes or attributes again from the samples.

A trade-off usually exists between precision of sample-based estimates and both sampling intensity and thematic detail. Sampling variability decreases with sample size (Cochran, 1977; Lohr, 2010). One expects greater uncertainty in the estimation of more numerous, specific classes, such as specific crop types, compared to fewer, broader classes, such as all cropland, because some of the specific classes will likely be rare and thus poorly represented in the samples. In such cases, stratification based on maps that approach as much as possible the distribution of the rarer classes can help increase sampling efficiency. When travel cost is an issue, two-stage cluster sampling, where points are distributed only within distributed blocks, i.e. clusters, can reduce costs and increase efficiency (e.g. King et al., 2017; Song et al., 2017).

The GLAD data set on tree-cover loss provides the most accurate and globally consistent estimate of all areas of loss from 2000–2012 (Hansen et al., 2013). To estimate loss within forest, users can overlay these data with national or regional maps of forest cover and calculate loss within that extent. Still, much of what is mapped as a tree-cover loss in the GLAD product is not permanent clearance; it includes other forms of clearance or disturbance, such as the re-clearance of tree fallows, selective or rotational logging and damages from wildfire and hurricanes, all examples where the recovery of tree cover is expected. We consider these as forms of temporary loss, where the loss event is likely to soon be followed by regrowth of tree cover. This type of loss has very different implications for the estimation of trends in tree cover and associated ecological impacts than those of permanent loss, where the loss is followed by a land cover that maintains a persistent absence of tree cover. For studying the impacts of loss on ecosystems and climate, for informing land-use planning or for reporting for international commitments, the most critical characterization of tree-cover loss needed is the proportion of loss that is temporary versus permanent. The next most critical is the relative proportions of different land-cover types following loss, beginning with broad categories, such as cropland and grassland, as referred to in the Inter-governmental Panel on Climate Change (IPCC) guidelines for national reporting to the UNFCCC (IPCC, 2006).

2.2. Objectives

The objectives of this study were to quantitatively assess the differences in the character of tree-cover loss in the Mexican Yucatán and Argentine Chaco by estimating the proportions of loss that were temporary versus permanent type and the proportions of subsequent land cover. While there exists qualitative information on which type dominates each and what land-cover types are important in each, we asked: 1) how dominant or rare is temporary versus permanent tree-cover loss and 2) how dominant or rare are different broad land-cover classes as well as specific crop or other cover classes within the areas of recent loss in each region? We sought to answer these questions using a cost-efficient methodology and using spatial data on loss, sampled field observations and visual interpretations of satellite data.

We compare the two regions because they are important in that they are at the geographical extremes of the Neotropical forest biome, of significant to global biodiversity, and have high rates of recent forest-cover loss. Further, they have contrasting land uses and scales of land use. A major benefit of our method is that it uses a generic forest-cover loss layer as the input, after which field data are collected to partition loss into relevant dynamics for reporting purposes. To have two very different case studies, one dominated by degradation dynamics, and the other by deforestation, provides a strong proof of concept for the generic use of data on the forest-cover loss as a stratified for efficient characterization of loss dynamics in different contexts.

3. Methods

3.1. Sampling design

The target population was defined as areas of tree-cover loss from 2000–2012, as determined from the GLAD map. We used all mapped loss regardless of tree-cover in 2000, although the great majority of these loss pixels where in areas where 2000 cover was over 30%, as in areas of mature, natural forest and anthropogenic wooded areas, e.g. secondary shrub and tree fallow, logged forest or plantations. The classes that we sought to estimate do not exist in the map product, as opposed to previous sample studies to refined map-based estimates of class areas (e.g. Olofsson et al., 2014). Rather, the product provided the target population, represented in a raster layer of 30-m resolution cells, where we refer to these cells as pixels since their locations are aligned with the pixel layout of the source Landsat data. The area of loss delineated for each of the two regions was partitioned into blocks to provide the framework for selecting a two-stage cluster sample. In order to target areas where the majority of loss occurred, we included in the population only blocks that had at least 5% tree-cover loss over the 12-year period.

Cluster sampling was motivated because of the need to reduce travel time and cost to field sample locations. The cluster design spatially constrained the sample pixels to a limited number of sample clusters, reducing travel relative to an un-clustered design such as simple random or systematic sampling. Blocks (first-stage sample units) were 6 km × 6 km in the Yucatán and 20 km by 20 km in the Chaco, the larger size for the Chaco chosen because of the greater area of the Chacoan loss population (Figures 1 and 2). For the Yucatán, the two-stage design was implemented by selecting a simple-random sample of 20 blocks and a simple-random sample of 10 pixels within each sample block.

In the Chaco the time to travel between sample blocks was expected to be much greater than in the Yucatán, so fewer blocks were sampled. Also, because of its larger size and greater heterogeneity, the Chaco was partitioned into five geographic strata to better distribute the sample

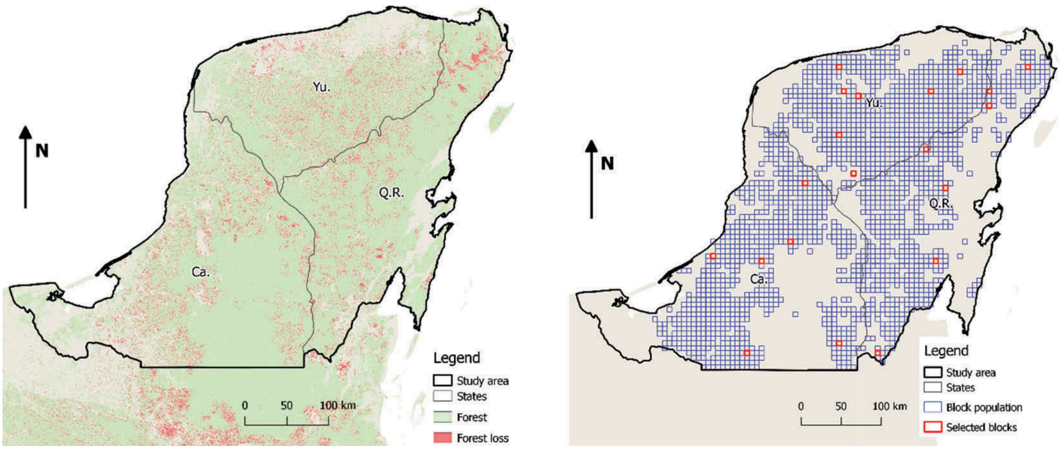

Figure 1. Distribution of forest cover and forest loss from 2000 to 2012 for the Mexican Yucatán study area. Sampled blocks are shown in red. States are Campeche (Ca.), Quintana Roo (Q.R.), and Yucatán (Yu.).

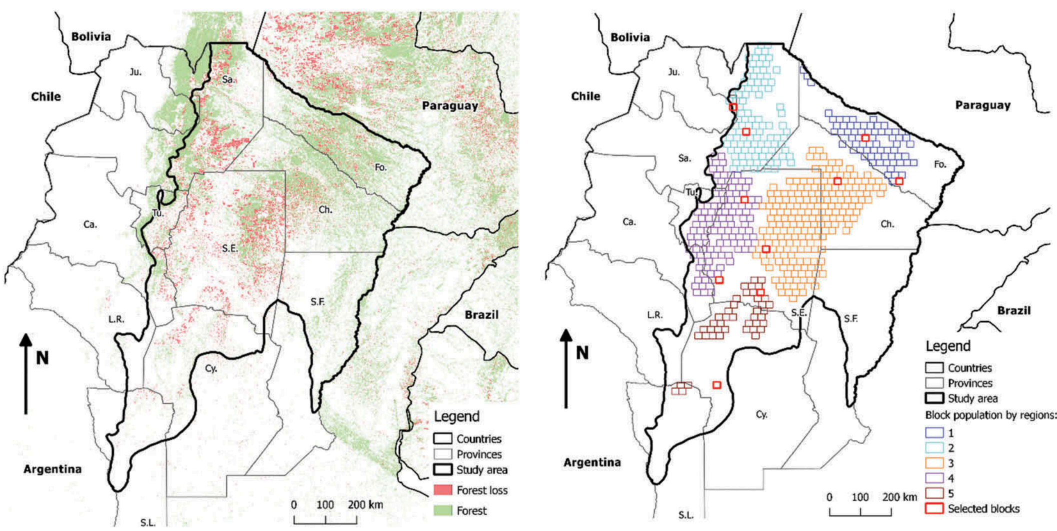

Figure 2. Distribution of forest cover and forest loss from 2000 to 2012 for the Argentine Chaco study area. Sampled blocks are shown in red and sampling strata in the remaining colours. Provinces included in the Chaco study site are Catamarca (Ca.), Chaco (Ch.), Córdoba (Cy.), Formosa (Fo.), Jujuy (Ju.), La Rioja (L.R.), Salta (Sa.), Santiago del Estero (S.E.), San Luis (S.L.), and Tucumán (Tu.).

blocks across this spatial heterogeneity. These strata were defined as regions of tree-cover loss posited to be related to varying land-use dynamics: 1) northern Chaco in Formosa province, dominated by cattle ranching and ranging from tall, sub-humid Chaco in the moister east to shorter, more open, dry Chaco in the centre and west; 2) dry Chaco in central and north-east Santiago del Estero and central Chaco provinces, trending from large-scale pasture and cropland in Santiago del Estero to small-scale mixed-crop agriculture in Chaco province proper; 3) an arc of southern, dry Chaco from southern Santiago del Estero into northern Santa Fe and San Luis provinces, trending from pasture to cropland to forestry; 4) a zone of dry Chaco east of the foothills of the Andes, centred on eastern Tucuman and western Santiago del Estero provinces, with both pasture and cropland; and 5) hilly Chaco Serrano to flat, dry Chaco, principally in Salta province, the site of extensive row crop cultivation, especially soybean. Within each of these regions, blocks were split into two sub-strata of high and low deforestation rates, with roughly equal total deforestation in each stratum. One block was selected at random from each of the high- and low-deforestation sub-strata in each region, yielding 10 total strata and 10 sample blocks.

For each sampled pixel, a polygon surrounding the pixel was delineated to define a field of consistent land cover, based on the patch of tree-cover loss in the loss map and a visual interpretation of the spectral image from 2012. In cases where two selected pixels fell within a single delineated polygon, the same polygon could be identified from two different sample pixels. Selected pixels located within loss patches of fewer than three pixels were labelled as indeterminate. If a sample block or sample pixel within a block could not be reached in the field or interpreted from the imagery, a replacement block or pixel would be selected following the same simple random selection protocol used to select the original sample. For some blocks, the initial sample of 10 pixels was accessed more easily than expected so one or more additional sample pixels were added by the same simple random selection protocol.

The final sample for the Yucatán consisted of 20 blocks with an average of 11 pixels per block and the final sample for the Chaco consisted of 10 blocks with an average of 21 pixels per block. The total effort for the field surveys was approximately 10 days of field surveys in each study area, conducted by four to six people split into two teams and covering different sample subsets. The fieldwork was followed by approximately five person-days of interpretation of Landsat images to augment the sample.

3.2. Field and satellite-data surveys

The objectives of the study were to estimate the proportions of recent tree-cover loss that were temporary versus permanent and to estimate the proportions of different subsequent land-cover types within the extents of loss for the two regions. The types of tree-cover loss and classes of broad and specific land-cover following a loss that we assessed in this study are listed in Table 1.

Field surveys were conducted during the main crop season in each region, June 2014 in the Yucatán and February 2014 in the Chaco. For each sample pixel, we reached the edge of the delineated field closes to the pixel's represented location. We assigned loss type based the land-cover observed at sample locations and the surrounding context. While we acknowledge that a total certainty of the long-term future of a site is not possible, the observed land cover class and context provide evidence of a degree of infrastructure and investment that strongly implies a commitment to the maintenance of continuous cropland or pasture and thus warrants assignment to permanent loss.

In the Yucatán, sample sites were assigned to temporary loss if they were in fallow or part of small-holder, rotational cropping, as identified by the field size, presence of nearby fallows, lack of evidence of mechanized tillage observed in the field surveys or satellite imagery, and location within *ejidos*, as supported by farmer interviews and local experts. Corn occurred in both permanent and temporary contexts, and these samples were assigned to temporary if they were within a fallow-rotation surrounding landscape. Selective logging and hurricane or wildfire damage were also assigned to temporary class because we assume that the regeneration of tree-cover will occur after the disturbance. Any temporary-loss site could in the future be converted to some persistent

Table 1. Types of tree-cover loss and broad and specific land-cover classes following loss surveyed.

Loss type	Broad land cover	Specific land cover
Yucatán		
Permanent	Cropland	Henequen, Permanent corn, Sugar cane
	Grassland	Pasture
	Settlements	Construction
Temporary	Cropland	Temporary corn
	Forest	Burned, Fallow, Hurricane damaged, Logging
Chaco		
Permanent	Cropland	Dry beans, Soy, Corn
	Grassland	Pasture
	Settlements	Construction
Temporary	Forest	Burned, Artisinal logging, Industrial logging

non-treed cover, and if so that would be a new event that in a later study would be assigned to permanent loss. In the Chaco temporary loss includes wildfire damage and logging in managed plantations, as we encountered no rotational agriculture there.

For each pixel, we identified subsequent land cover based on the cover types observed in the field and from satellite imagery. We assigned broad classes and specific sub-classes of cover listed in Table 1. We assigned bush and tree fallow to the forest land class because their structure meets the national forest definition, noting that fallows as a cover class are typically a temporary stage within an agricultural practice.

During field surveys, we visited the edge of each outlined polygon, identified the polygon's current land cover, collected photographs and recorded GPS coordinates. Sites that could not be visited in the field, 36% and 31% of all samples for the Yucatán and Chaco, respectively, were assigned to a loss type and class using the visual interpretation of 2012 Landsat images. Spectral pattern, image texture, spatial context and spectral similarity to other sites visited in the field contributed to image interpretation. The sample selection protocol, at the scales of blocks and 30-m pixels, is thus not biased towards accessible areas, as all blocks and pixels within a stratum have the same inclusion probability.

3.3. Estimation of area proportions and uncertainty

Estimates of land-cover proportions were derived as stratum-weighted means of loss types and cover classes verified for the sample locations. Because of the different sampling designs for the two study regions, the estimation formulas differ slightly between the two regions. For the Yucatán, the proportion of the tree-cover loss attributed to a particular resulting land cover was estimated using a ratio estimator for two-stage cluster sampling. The population parameter is the area of the type or class of fate divided by the area of loss (or equivalently for equal-area pixels, the number of pixels of the type or class of fate divided by the number of pixels of loss in the population). Although the total area of loss can be computed from the loss map (i.e. a census value), the ratio estimator defined subsequently has a smaller SE when the area of loss is estimated from the sample because of the favourable covariance of the estimated areas in the numerator and denominator (Lohr, 2010, Section 5.2.3). Thus, while the area of loss is not one of the parameters we sought to estimate in this study, the sample-based estimate loss is used in the calculation of our desired estimates.

For each sample pixel u in sample cluster (block) i, we defined $y_{i,u} = 1$ if the pixel had the fate of interest and $y_{i,u} = 0$ otherwise. Then for sample cluster i, the estimated proportion of pixels with the fate type of interest is $p_i = \bar{y}_i$. If M_i denotes the number of pixels of tree-cover loss in cluster i, the estimated total number of pixels with the specified fate is

$$\widehat{Y}_i = M_i p_i \tag{1}$$

and the total number of pixels of the fate class of interest in the entire region (population) is estimated by

$$\widehat{Y} = N\sum_{i=1}^{n}\widehat{Y}_i/n \qquad (2)$$

where n is the number of clusters sampled and N is the number of clusters in the region. The total number of pixels of tree-cover loss is estimated by

$$\widehat{X} = N\sum_{i=1}^{n}M_i/n \qquad (3)$$

The proportion of loss associated with the fate class of interest is then estimated as a ratio,

$$\widehat{R} = \widehat{Y}/\widehat{X} \qquad (4)$$

The variance of $\widehat{R}$ is estimated by (Lohr, 2010, equation 5.28)

$$\widehat{V}\left(\widehat{R}\right) = \left(\frac{N-n}{N}\right)\frac{1}{\overline{M}^2}\frac{s_r^2}{n} + \frac{1}{nN\overline{M}^2}\sum_{i=1}^{n}M_i^2\left(\frac{p_iq_i}{m_i-1}\right) \qquad (5)$$

where $q_i = 1 - p_i$, m_i is the number of sample pixels for cluster i, $\overline{M}$ is the average number of pixels per cluster, and s_r^2 is computed as follows (Lohr, 2010, Equation 5.29),

$$s_r^2 = \frac{1}{n-1}\sum_{i=1}^{n}\left(M_ip_i - M_i\widehat{R}\right)^2 \qquad (6)$$

The two-stage cluster sampling design implemented in the Argentina Chaco employed a stratified random sample at the first stage with a single sample cluster selected from each of 10 strata. The stratified estimator of the number of pixels of the fate type of interest is

$$\widehat{Y} = \sum_{h=1}^{H}N_h\widehat{Y}_h \qquad (7)$$

where the subscript h denotes the stratum, $\widehat{Y}_h$ is the estimated total based on the one block sampled in stratum h (Equation 1), H = total number of strata, and N_h = number of clusters in stratum h. The total number of pixels of loss was estimated for the stratified design by

$$\widehat{X} = \sum_{h=1}^{H}N_hM_h \qquad (8)$$

and again the ratio estimator for the proportion of loss associated with a particular land-cover class is $\widehat{R} = \widehat{Y}/\widehat{X}$. Because only one cluster was sampled per stratum, it was not possible to apply a stratified variance estimator. Consequently, we approximated the variance of $\widehat{R}$ using the simple random sample variance (Equations 5 and 6). We calculated the coefficients of variation (CV) as the standard error (SE) divided by the estimate, the SE being the square root of the variance.

4. Results

The Yucatán is a region of mostly rotational milpa agriculture and pasture with fallow: 54% of all tree-cover loss from 2000 and 2012 was temporary (Figure 3). Areas under pasture comprised 36% of loss and fallow 30% of the loss. Forest damage from the fire was another large component of temporary loss, with 23% of all loss reported as burned forest. Land cover in areas of permanent loss was mostly pasture, with modest areas of corn, henequen and sugar cane. Settlements, which included only our construction class, was 2%, and selective logging and hurricane damage were very rare, less than 1% each.

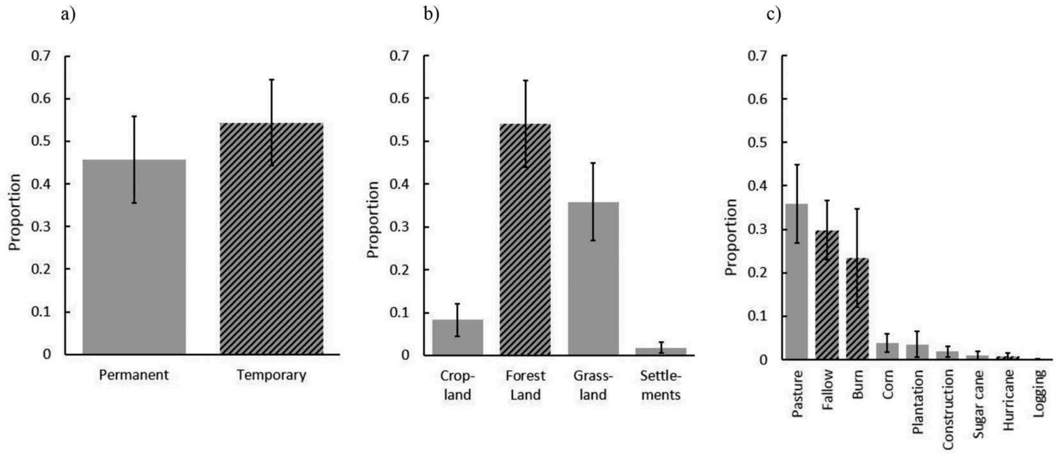

Figure 3. Estimated proportions of loss type and post-disturbance land cover within the total area of forest loss from 2000 to 2012 in the Mexican Yucatán: a) permanent versus temporary loss, b) broad cover classes, c) specific cover classes. 'Hurricane' is hurricane-damaged forest and 'Construction' is road or building structures. Diagonal lines are a temporary loss. Error bars represent standard errors.

Tree-cover loss in the Chaco, in contrast, was dominated by permanent loss, with an estimated 85% of loss attributable to this type (Figure 4). The area of permanent loss was distributed 44% to pasture, 17% to corn, and 17% to soy, with the remaining 7% in dry bean farming. Temporary loss was mostly artisanal-selective logging, followed by industrial-selective logging and damage from fires. In terms of broad land-cover classes, roughly 44% of the area of recent tree-cover loss was under grassland and 42% was under cropland, with the remainder under forest land.

Because of the relatively small number of blocks sampled, particularly in the Chaco, many of the estimated sample proportions have large CVs. CVs are generally smaller for the more common categories of loss type that cover large areas and are well-distributed among sample blocks. The CV is 10.4% for the permanent loss class dominating the Chaco and ranges from 18.6% to 25.2% for the large-area classes of pasture in the Chaco and fallow, pasture and forest in the Yucatán. For the rarer classes such as mono-cropped commercial corn, sugar cane, soy and artisanal logging, the CVs range from 55.9 to 109.3, and it is 61% for the relatively rare temporary loss class.

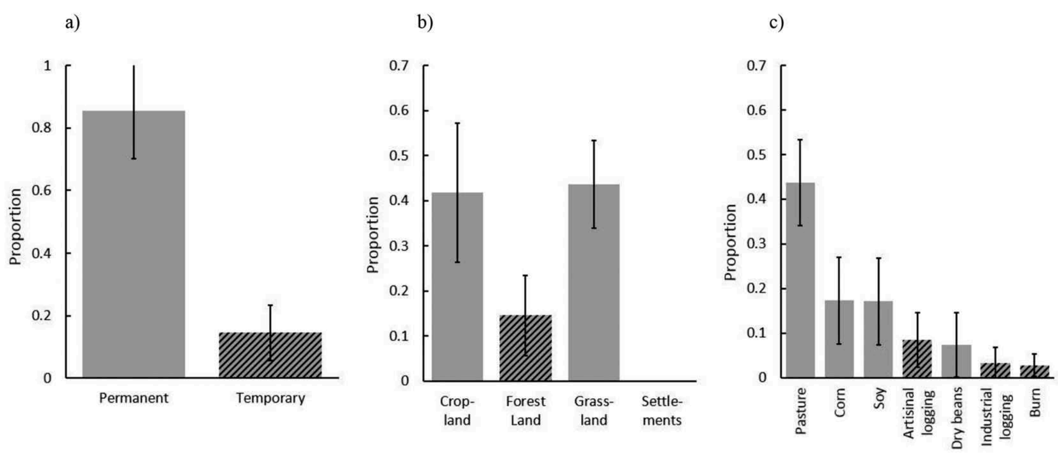

Figure 4. Estimated proportion of loss type and post-disturbance land cover within the total area of forest loss from 2000 to 2012 in the Argentine Chaco: (a) permanent versus temporary loss, (b) broad cover classes, (c) specific cover classes. Diagonal lines are a temporary loss. Error bars represent standard errors.

5. Discussion

5.1. Contrasting contributions to tree-cover loss

Two of the regions with the most concentrated rates of tree-cover loss in the neotropics are its northern and southern forested extremes. Both regions are similar in terms of lying in a climatic gradient and having a significant ranching component to the landscape. However, they have very different histories of land use from the pre-Columbian to today. Data from our 2012 surveys quantify the outcomes of these histories and recent macro-economic trends. Our sample-based study provides estimates of the proportions of different types of tree-cover loss and subsequent land cover in these two regions, quantifying the large differences in the character of tree-cover loss in the Yucatán and Chaco.

Much of the Yucatán's forests are fallow or secondary forests, otherwise descended from 3,000 years of swidden agriculture and selective preference for useful tree species (Faust, 2001). Population densities went from low to rapidly increasing in the 1970s, when state-sponsored settlement to the region began, including a highway connecting Mexico's two coasts built in 1967 (Roy Chowdhury, 2006). Still, the culture there is heavily founded on community-based land management and relatively small land holdings in *ejidos* used to produce goods mostly for local consumption (e.g. Abizaid & Coomes, 2004; García-Frapolli et al., 2009).

Land in the *ejidos* is managed by low-income families with relatively small farms that also have a traditional reliance on forest timber and non-timber products. Soils over much of the region are shallow and with a rocky substrate, and as in other areas of tropical agriculture on deforested land with challenging soils, fallowing is used to permit long-term, acceptable levels of crop production. Roy et al (2006) found that the majority of land-cover changes in sample areas around the Calakmul Biosphere reserve were transitions between cropland and fallows, dwarfing the clearance of mature forest. While our study did not estimate forest type prior to tree-cover loss events, the large proportion of temporary loss, 54%, and proportion of fallow, 30%, suggest that this remains the case. Our study estimates only 1% of the hurricane-damaged forest. This appears in contrast to Rogan et al. (2010), who estimated that potentially 83% of the central Yucatán near the path of Hurricane Dean in 2007 was damaged by it, based on changes in satellite greenness indices before and after the event and validated with stratified field sampling. However, we sampled a much larger region beyond the path and visited the area seven years later.

Corn is a significant component of loss and our estimates indicate it occurred mostly in areas of permanent loss. Corn is central to the Mexican diet, and is the largest recipient crop type of PROCAMPO funds, government subsidies initiated nation-wide to shield farmers from price reductions that were expected under the North American Free Trade Agreement (NAFTA) (Garcia-Salazar, Skaggs, & Crawford, 2011). These funds were made available both to larger landholders and to farmers with cultivated plots as small as a hectare; corn production actually increased in the NAFTA/PROCAMPO years (Garcia-Salazar et al., 2011). Schmook and Vance (2009) note that one difficulty for the majority of low-income farmers is lack of transport to deliver products to market, and farmers must negotiate prices with intermediaries. Pasture is seen as a facile means to increase the productive value of land and provide greater economic, stability, even if many of the small holders with pastures do not own the cattle. We found that pasture far outweighed, crops, at 36% versus 8%, in areas of tree-cover loss over our study period.

The Chaco, while having had indigenous populations before colonization and criollo communities of mostly European descent since, has low-intensity land uses until recent decades (Volante & Paruelo, 2015). During most of the twentieth century, the traditional land use has been cattle ranching. Rotational agriculture, as practiced in the Yucatán and in many regions of traditional agriculture in tropical forest frontiers, is insignificant in the Chaco, and we encountered no agricultural or pastoral fallows during our survey.

We found that the Argentine Chaco strongly contrasts the Yucatán in that permanent loss, 54%, was greater than temporary loss and that the latter was almost entirely logging. Our estimates also corroborate the importance of ranching in the Chaco, where we found pasture in 44% of the area of recent loss. However, large-scale agriculture has become equally important, where we estimate Cropland in 42% of the area of recent loss. This was split equally between soybean and corn, both at 17%, and then followed by dry beans at 7%. This quantifies previously reported descriptions of mechanized cropping displacing pasture in the region (e.g. Piquer-Rodriguez et al., 2018).

There is a well-documented boom in soy farming in the broader Chacoan region (Boillat et al., 2017; Fehlenberg et al., 2017; Grau & Aide, 2008; Killeen et al., 2008), including Paraguay and Bolivia; however, we estimate that corn contributed as much as soy to recent tree-cover loss in the Argentine portion. We did not find in our sample sites any fields of lesser but reportedly significant crops, such as cotton and alfalfa, and our results do not address double cropping that reportedly occurs in some areas. Addressing the former would require a greater sampling intensity, perhaps aided by stratification using seasonal satellite data, and the latter by sampling during two seasons associated with the timing of expected bi-cropping. The high proportion of cropland in the Chaco has serious implications for the region's water resources as cropland has greater water consumption than that of pasture as well as a different seasonality (Gimenez et al., 2016; Houspanossian, Giménez, Jobbagy, & Nosetto, 2017). Logging, both artisanal and industrial, also represents a substantial portion of tree-cover loss in this area, leaving very little Chacoan forest and woodland that has not experienced significant or complete tree-cover loss for some form of land use.

The broader Chaco region has been described as undergoing a transition from a dominance ranching to one of mechanized farming of commodity crops over the past three decades (e.g. Graesser, Ramankutty, & Coomes, 2018), as has been reported in other areas including parts of Brazil and directly north in Bolivia (e.g. Killeen et al., 2008). This reflects a large, international demand for commodity crops that are exported within South America, further incentivized by the Mercosur pact, as well as to China and elsewhere (Boillat et al., 2017; Grau & Aide, 2008). The boom in commercial crops is partly made possible by increases in productivity and resilience to the Chacoan climate of soybeans and other crops, through changes in tillage and through artificially modified genetics (Gasparri, Grau, & Gutierrez Angonese, 2013; Grau & Aide, 2008; Volante & Paruelo, 2015). The economic incentives from international demand have more than offset the effects of national policies on forest protection and export taxes, even when they reached 35% in 2008 (Deese & Reeder, 2007; Gasparri et al., 2013). The latter policy has been criticized internally and internationally, and in 2015 the tax was reduced by 5% with a commitment for further gradual reductions (USDA, 2018). Other forms of deregulation in Argentina have been aimed at encouraging foreign investment. These incentives combined with land prices lower than those in Brazil make the Argentine Chaco is highly attractive to international investors for commercial crops. Their rapid expansion in both countries may be causing displacement of other types of production to other countries with even lower land prices, such as Bolivia and Peru (Fehlenberga et al., 2017).

These findings were quantified via a sampling approach based on unbiased estimation with statistical uncertainty levels. These uncertainties, in terms of CVs, range from 10% to 22% for larger classes to 37% to over 100% for smaller classes. Clearly, the uncertainties are greater for the lesser classes in both regions, yet much useful information is provided via a modest sample size, made possible by increasing the sample size within the loss stratum.

5.2. Potential for reducting uncertainty

Because of the relatively high uncertainties of the sample-based estimates for some classes, approaches to reducing uncertainty are of interest. One approach is a model-assisted estimation, a technique that incorporates map information associated with a response variable to reduce the CV of an estimator of area or proportion (Särndal, Swensson, & Wretman, 1992; Stehman, 2009). These data can then be used in, for example, a regression estimator to improve precision. An example of this can be shown for the Yucatán where we have a sufficient sample size (n = 20 blocks) to provide some assurance that a regression estimator will be unbiased. Temporary and permanent tree-cover loss in the Yucatán is expected to be correlated at the block level with 2012 tree cover and can thus be an auxiliary variable. One can apply a regression model using: y = area of permanent loss (estimated from the second-stage sample within each block) and x = percent tree cover for the block, where $\widehat{Y}_{reg} = N\left[\overline{y} + b_1\left(\overline{X} - \overline{x}\right)\right]$, N is the number of clusters in the study area, b_1 is the ordinary least squares estimate of the slope for y regressed on x, $\overline{X}$ is the population mean per cent tree cover (i.e. mean for all 4193 blocks), $\overline{x}$ is the sample mean of per cent tree cover, and $\overline{y}$ is the sample mean of either temporary or permanent loss. This produces model-assisted estimates for permanent and temporary loss in the Yucatán of 456,673 ± 71,469 ha and 591,476 ± 146,434 ha, respectively, compared to the direct-sample estimates of 472,825 ± 87,038 ha and 562,769 ± 170,991 ha. The model thus changes estimates by approximately 3% and 5% while reducing CVs by 18% and 14%, a desirable improvement for no additional survey cost.

Data on the tree-canopy cover can be obtained globally, but we do not expect it to be correlated to many of the other classes of interest. Other global data sets may be useful for modelling other cover types. For example, data on per cent bare-ground or impervious surfaces (Ying et al., 2017) may be useful to model a settlement class, and similar data on temporal dynamics of water may be useful for modelling wetlands, which were not estimated in this study. While more complex and probably variable over large regions, the annual amplitude of greenness indices or other satellite-based, seasonal metrics may be correlated to pasture or specific crops, especially large-scale monocrops such as soy (King et al., 2017).

Another possibility for reducing uncertainty, if resources exist for a larger survey, is to use a two-step approach to surveying. The first is a relatively rapid survey, as done in this study, which quickly provides confident estimates for larger classes and also provides *a priori* information on class proportions and expected CVs for different classes under different choices of sample size at the block and pixel level (i.e. the first stage and second stage samples). This enables an estimation of resources needed to achieve desired outcomes, in CV levels, of the second survey, or determination of whether a modified approach is needed, e.g. one incorporating targeted sampling, using additional ancillary data, for rarer classes. To simulate CVs from a second-step survey, one can use a resampling of the data from the first step to compute the expected CVs for different combinations of sample sizes.

Figure 5 is an example for the Yucatán, where clusters and pixels were randomly selected with replacement to simulate samples of different sizes over 2,000 iterations. Simulations are for two large-area classes, Forest Land and Grassland, and two rarer classes, Cropland and Settlements. Of these, all classes show improvements in CVs with increasing block number and samples per block, although the amount of improvement varies among classes. Clearly, it was advantageous to have a second-stage sample size of at least 10, as the decrease in SE for 10 pixels is substantial relative to a second-stage sample size of just one or three pixels. However, the decrease in CV gained by doubling the sample size from 10 to 20 pixels is modest, and even five pixels per block could yield acceptable results. The general patterns are as expected from the theory of two-stage sampling. Because the majority of the variance in the proportion estimates will be associated with variation in the area of loss among blocks, increasing the sample size of blocks has the greater effect on decreasing CV. For the rarer classes of Cropland and Settlements the CVs remain large for all of the choices of sample sizes evaluated in Figure 5. To obtain precise estimates for such rare types, it may be necessary to modify the sampling design using targeted sampling, although this remedy may be challenging because it would require relatively accurate *a priori* information on where these rare types may be found. Global products mentioned above, or national data sets, may be used, or the creation of new maps of strata targeted to the rarer classes may be warranted.

6. Conclusion

The sample-based surveys conducted in this study, excluding any improvements with models or additional surveys, have efficiently produced unbiased estimators with known statistical uncertainties

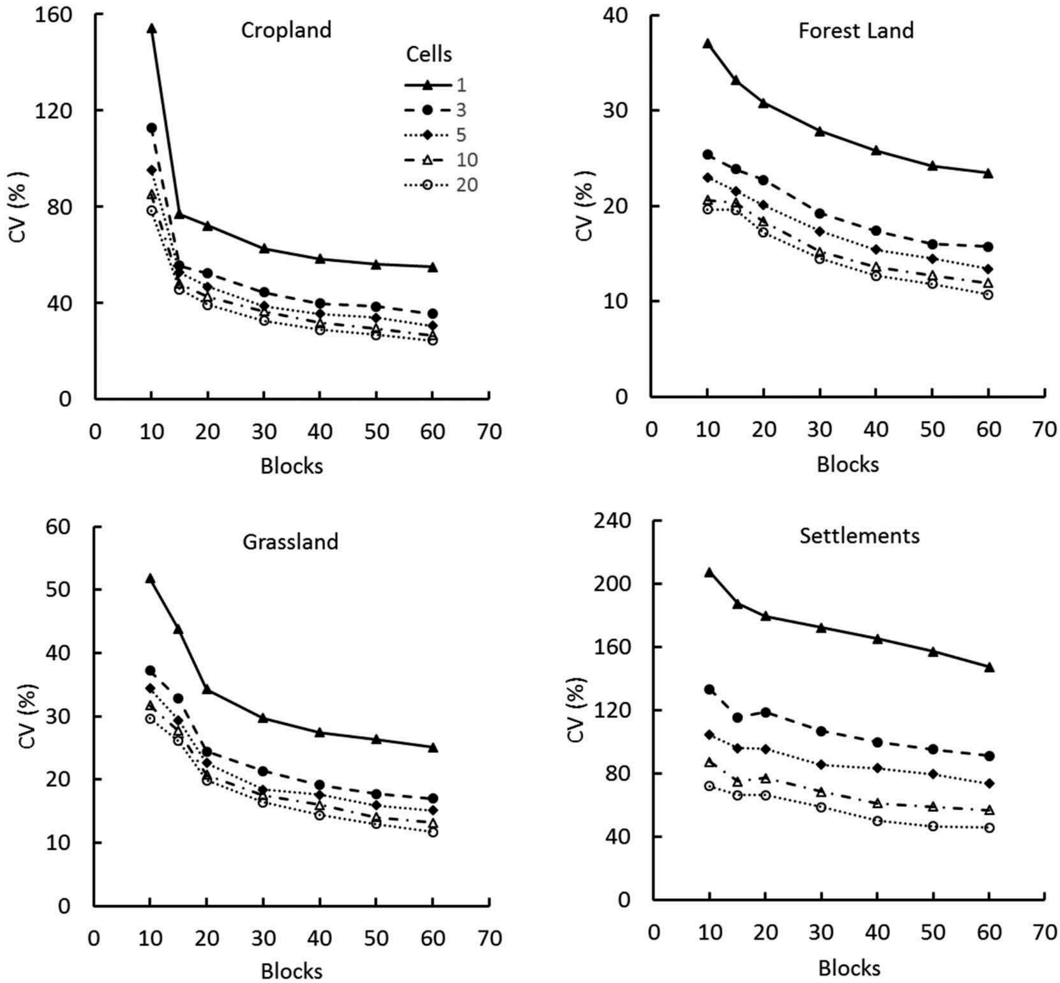

Figure 5. Simulations of coefficients of variation (CVs) for different sample sizes for blocks (first-stage sample) and pixels (second-stage sample) applied to the Yucatán, Mexico study area. The actual sample sizes used for the Yucatán were 20 blocks at the first stage and an average of 11 pixels per block at the second stage.

that shed much light on the differing character of tree-cover loss in two important neotropical regions. The approach we have used can be applied over large areas to provide information about the types of tree-cover loss and fates of the land where these occur for policy and research applications. It can be replicated to efficiently estimate trends from a definitive in-situ perspective, as well as be supported by the visual interpretation of satellite image points. Modest efforts can yield useful information, especially about broader dynamics and larger land-cover classes, over large areas. Data can be collected via field surveys, expert interpretation of specific sites within satellite images, or a combination of both. High uncertainty may be expected for smaller classes, yet this approach is flexible, allowing for a variety of options of stratified sampling design or model-assisted estimation.

Much attention has been given to the expansion of commercial crops in the Chaco. However, for the Argentine Chaco, we find persistence in the importance of ranching: our estimate of the proportion of land cover following a recent tree-cover loss is still slightly higher for pasture than that for all crops combined. Expanded surveys could be used to determine to what degree commercial crops are over-taking pasture throughout the entire Chaco and surrounding areas. Likewise, future surveys in the Yucatán could be used to determine if there is a gradual shift from traditional fallow agriculture, characterized by temporary loss, towards more persistent farming and ranching, characterized by permanent loss and commercial crops. Similar surveys of nearby regions on a regular basis could provide a quantitative basis for exploring inter-related dynamics across larger, international regions, such as evidence of international leakage. This approach can also be used as part of broader strategies to reaching target levels of uncertainty desired for national assessments of the types of change in forests, subsequent land cover and their potential impacts.

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References

Abizaid, C., & Coomes, O.T. (2004). Land use and forest fallowing dynamics in seasonally dry tropical forests of the southern Yucatán Peninsula, Mexico. *Land Use Policy*, *21*, 71–84.

Adamoli, J., Sennhauser, E., Acero, J.M., & Rescia, A. (1990). Stress and disturbance: Vegetation dynamics in the dry Chaco region of Argentina. *Journal of Biogeography*, *17*, 491–500.

Boillat, S., Scarpa, F.M., Robson, J.P., Gasparri, I., Aide, M., Dutra Aguiar, A.P., Brondizio, E. (2017). Land system science in Latin America: Challenges and perspectives. *Current Opinion in Environmental Sustainability*, *26*, 37–46.

Busch, C., & Geoghegan, J. (2010). Labor scarcity as an underlying cause of the increasing prevalence of deforestation due to cattle pasture development in the southern Yucatan region. *Regional Environmental Change*, *10*, 191–203.

Cochran, W. (1977). *Sampling Techniques*. New York, NY: Wiley and Sons.

De Sy, V., Herold, M., Achard, F., Beuchle, R., Clevers, J.G.P.W., Lindquist, E., & Verchot, L. (2015). Land use patterns and related carbon losses following deforestation in South America. *Environmental Research Letters*, *10*, 124004.

Deese, W., & Reeder, J. (2007, September). Export taxes on agricultural products: Recent history and economic modeling of soybean export taxes in Argentina. *United States International Trade Commission Journal of International Commerce and Economics*. https://www.usitc.gov/publications/332/journals/export_taxes_model_soybeans.pdf

Ellis, E., Romero Montero, J., & Hernandez Gomez, I. (2017). Deforestation processes in the state of Quintana Roo, Mexico: The role of land use and community forestry. *Tropical Conservation Science*, *10*, 1–12.

FAO. (2015). *The global forest resources assessment 2015 desk reference*. Rome, Italy: Food and Agriculture Organization of the United Nations.

Faust, B. (2001). Maya environmental successes and failures in the Yucatan Peninsula. *Environmental Science & Policy*, *4*, 153–169.

Fehlenberga, V., Baumanna, M., Ignacio Gasparri, N., Piquer-Rodrigueza, M., Gavier-Pizarrod, G., & Kuemmerlea, T. (2017). The role of soybean production as an underlying driver of deforestation in the South American Chaco. *Global Environmental Change*, *45*, 24–34.

Frate, L., Acosta, A.T.R., Cabido, M., Hoyos, L., & Carranza, M.L. (2015). Temporal changes in forest contexts at multiple extents: Three decades of fragmentation in the Gran Chaco (1979–2010), Central Argentina. *PLoS ONE*, *10*(12), e0142855.

García-Frapolli, E., Ramos-Fernández, G., Galicia, E., & Serrano, A. (2009). The complex reality of biodiversity conservation through natural protected area policy: Three cases from the Yucatan Peninsula, Mexico. *Land Use Policy*, *26*, 715–722.

Garcia-Salazar, J. A., Skaggs, R., & Crawford, T. L. (2011). PROCAMPO, the Mexican corn market, and Mexican food security. *Food Security*, *3*, 383–394. doi:10.1007/s12571-011-0138-z

Gasparri, N.I., Grau, H.R., & Gutierrez Angonese, J. (2013). Linkages between soybean and neotropical deforestation: Coupling and transient decoupling dynamics in a multi-decadal analysis. *Global Environmental Change*, *23*, 1605–1614.

Giménez, R., Mercau, J., Nosetto, M., Páez, R., & Jobbágy, E. (2016). The ecohydrological imprint of deforestation in the semiarid Chaco: Insights from the last forest remnants of a highly cultivated landscape. *Hydrological Processes*, *30*, 2603–2616.

Gomez-Pompa, A., & Kaus, A. (1992). Taming the wilderness myth. *BioScience*, *42*, 271–279.

Graesser, J., Ramankutty, N., & Coomes, O.T. (2018). Increasing expansion of large-scale crop production onto deforested land in sub-Andean South America. *Environmental Research Letters*, *13*, 084021.

Grau, H.R., & Aide, M. (2008). Globalization and land-use transitions in Latin America. *Ecology and Society*, *13*, 16.

Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Loveland, T.R. (2013). High-resolution global maps of 21st-century forest cover change. *Science*, *342*, 850–853.

Harper, G.J., Steininger, M.K., Tucker, C.J., Juhn, D., & Hawkins, F. (2007). Fifty years of deforestation and forest fragmentation in Madagascar. *Environmental Conservation*, *34*, 325–333.

Houspanossian, J., Giménez, R., Jobbagy, E., & Nosetto, M. (2017). Surface albedo raise in the South American Chaco: Combined effects of deforestation and agricultural changes. *Agricultural and Forest Meteorology*, *232*, 118–127.

IPCC. (2006). 2006 IPCC guidelines for national greenhouse gas inventories: Volume 4 agriculture, forestry and other land use. In H. S. Eggleston, L. Buendia, K. Miwa, T. Ngara, & K. Tanabe (Eds.), *National greenhouse gas inventories programme*. Hayama: Institute for Global Environmental Strategies (IGES).

Killeen, T.J., Guerra, A., Calzada, M., Correa, L., Calderon, V., Soria, L., Steininger, M.K. (2008). Total historical land-use change in Eastern Bolivia: Who, where, when, and how much? *Ecology and Society*, *13*, 36.

King, L., Adusei, B., Stehman, S.V., Potapov, P.V., Song, X.P., Krylov, A., … Hansen, M.C. (2017). A multi-resolution approach to national-scale cultivated area estimation of soybean. *Remote Sensing of Environment*, *195*, 13–29.

Lohr, S.L. (2010). *Sampling: Design and analysis*. Boston, MA: Brooks/Cole.

Margono, B.A., Potapov, P.V., Turubanova, S., Stolle, F., & Hansen, M.C. (2014). Primary forest cover loss in Indonesia over 2000–2012. *Nature Climate Change*, *4*, 730–735.

McGroddy, M., Lawrence, D., Schneider, L., Rogan, J., Zager, I., & Schmook, B. (2013). Damage patterns after Hurricane Dean in the southern Yucatán: Has human activity resulted in more resilient forests? *Forest Ecology and Management*, *310*, 812–820.

Olofsson, P., Foody, G.M., Herold, M., Stehman, S.V., Woodcock, C.E., & Wulder, M.A. (2014). Good practices for estimating area and assessing accuracy of land change. *Remote Sensing of Environment*, *48*, 42–57.

Piquer-Rodríguez, M., Butsic, V., Gärtner, P., Macchi, L., Baumann, M., Gavier Pizarro, G., Kuemmerle, T. (2018). Drivers of agricultural land-use change in the Argentine Pampas and Chaco regions. *Applied Geography*, *91*, 111–122.

Potapov, P., Siddiqui, B., Iqbal, Z., Aziz, T., Zzaman, B., Islam, A., … Hansen, M. (2017). Comprehensive monitoring of Bangladesh tree cover inside and outside of forests, 2000–2014. *Environmental Research Letters*, *12*, 104015.

Potapov, P.V., Dempewolf, J., Talero, Y., Hansen, M.C., Stehman, S.V., Vargas, C., & Giudice, R. (2014). National satellite-based humid tropical forest change assessment in Peru in support of REDD+ implementation. *Environmental Research Letters*, *9*, 124012.

Prance, G.T. (2006). Tropical savannas and seasonally dry forests: An introduction. *Journal of Biogeography*, *33*, 385–386.

Rogan, J., Schneider, L., Christmana, Z., Millones, M., Lawrence, D., & Schmook, B. (2010). Hurricane disturbance mapping using MODIS EVI data in the southeastern Yucatán, Mexico. *Remote Sensing Letters*, *2*, 259–267.

Romero-Duque, L.P., Jaramillo, V.J., & Pérez-Jiménez, A. (2007). Structure and diversity of secondary tropical dry forests in Mexico, differing in their prior land-use history. *Forest Ecology and Management*, *253*, 38–47.

Roy Chowdhury, R. (2006). Landscape change in the Calakmul Biosphere Reserve, Mexico: Modeling the driving forces of smallholder deforestation in land parcels. *Applied Geography*, *26*, 129–152. doi:10.1016/j.apgeog.2005.11.004

Särndal, C.E., Swensson, B., & Wretman, J. (1992). *Model-assisted survey sampling*. New York, NY: Springer-Verlag.

Schmook, B., & Vance, C. (2009). Agricultural policy, market barriers, and deforestation: The case of Mexico's southern Yucatán. *World Development*, *37*, 1015–1025. doi:10.1016/j.worlddev.2008.09.006

Smardon, R.C., & Faust, B.B. (2006). Introduction: International policy in the biosphere reserves of Mexico's Yucatan peninsula. *Landscape and Urban Planning*, *74*, 160–192.

Snook, L. (1993). *Stand dynamics of mahogany (Swietenia macrophylla King) and associates species after fire and hurricane in the tropical forest of the Yucatan Peninsula, Mexico* (Doctoral dissertation). New Haven, CT: Yale University.

Song, X.P., Potapov, P.V., Krylov, A., King, L., Di Bella, C.M., Hudson, A., & Hansen, M.C. (2017). National-scale soybean mapping and area estimation in the United States using medium resolution satellite imagery and field survey. *Remote Sensing of Environment*, *190*, 383–395.

Stehman, S.V. (2009). Model-assisted estimation as a unifying framework for estimating the area of land cover and land-cover change from remote sensing. *Remote Sensing of Environment*, *113*, 2455–2462.

Stehman, S.V. (2013). Estimating area from an accuracy assessment error matrix. *Remote Sensing of Environment*, *132*, 202–211.

Turner, B. (1978). Ancient agriculture land use in the Central Maya lowlands. In P.D. Harrison & B.L. Turner (Eds.), *Pre-Hispanic Maya agriculture* (pp. 13–22). Albuquerque, New Mexico: University of New Mexico Press.

Turner, B.L., Villar, S.C., Foster, D., Geoghegan, J., Keys, E., Klepeis, P., & Plotkin, A.B. (2001). Deforestation in the southern Yucatán peninsular region: An integrative approach. *Forest Ecology and Management*, *154*, 353–370.

Tyukavina, A., Hansen, M.C., Potapov, P.V., Stehman, S.V., Smith-Rodriguez, K., Okpa, C., & Aguilar, R. (2017). Types and rates of forest disturbance in Brazilian Legal Amazon, 2000–2013. *Science Advances*, *3*, e1601047.

UNCBD. (2017). United Nations Convention on Biological Diversity. https://www.cbd.int/.

UNCCD. (2017). United Nations Convention to Combat Desertification. http://www2.unccd.int/.

UNFCCC. (2014). The Paris Agreement United Nations Framework Convention on Climate Change. http://unfccc.int/paris_agreement/items/9485.php

USDA. (2018, May 29). *Argentina's delay of Soybean export tax reduction unlikely* (Global Agricultural Information Network report), Buenos Aires, Argentina: United States Department of Agriculture Foreign Agricultural Service. https://gain.fas.usda.gov/Recent%20GAIN%20Publications/Argentina%E2%80%99s%20Delay%20of%20Soybean%20Export%20Tax%20Reduction%20Unlikely%20_Buenos%20Aires_Argentina_5-29-2018.pdf

Vaca, R.A., Golicher, D.J., Cayuela, L., Hewson, J., & Steininger, M.K. (2012). Evidence of incipient forest transition in southern Mexico. *PLoS ONE*, *7*(8), e42309.

Volante, J.N., & Paruelo, J.M. (2015). Is forest or ecological transition taking place? Evidence for the Semiarid Chaco in Argentina. *Journal of Arid Environments*, *123*, 21–30.

Werneck, F.P. (2011). The diversification of eastern South American open vegetation biomes: Historical biogeography and perspectives. *Quaternary Science Reviews*, *30*, 1630–1648.

Ying, Q., Hansen, M.C., Potapov, P.V., Tyukavina, A., Wang, L., Stehman, S.V., Hancher, M. (2017). Global bare ground gain from 2000 to 2012 using Landsat imagery. *Remote Sensing of Environment*, *194*, 161–176.