Article:

Nanni, A.S., Sloan, S., Aide, T.M. et al. (3 more authors) (2019) The neotropical reforestation hotspots : a biophysical and socioeconomic typology of contemporary forest expansion. Global Environmental Change, 54. pp. 148-159. ISSN 0959-3780

https://doi.org/10.1016/j.gloenvcha.2018.12.001

The neotropical reforestation hotspots: A biophysical and socioeconomic typology of contemporary forest expansion

A. Sofía Nanni[a,*1], Sean Sloan[b,*1], T. Mitchell Aide[c], Jordan Graesser[d], David Edwards[e], H. Ricardo Grau[a]

a Instituto de Ecología Regional and Consejo Nacional de Investigaciones Científicas y Técnicas (CONICET), Universidad Nacional de Tucumán, CC. 34, 1407, Yerba Buena, Tucumán, Argentina

b Center for Tropical Environmental and Sustainability Science, College of Science and Engineering, James Cook University, Cairns, Qld, 4870, Australia

c Department of Biology, University of Puerto Rico, San Juan 00931-3360, Puerto Rico

d Department of Earth and Environment, Boston University, 685 Commonwealth Avenue, Boston, MA, 02215, USA

e Department of Animal and Plant Sciences, University of Sheffield, S10 2TN, UK

Abstract

Tropical reforestation is a significant component of global environmental change that is far less understood than tropical deforestation, despite having apparently increased widely in scale during recent decades. The regional contexts defining such reforestation have not been well described. They are likely to differ significantly from the geographical profiles outlined by site-specific observations that predominate in the literature. In response, this article determines the distribution, extent, and defining contexts of apparently spontaneous reforestation. It delineates regional 'hotspots' of significant net reforestation across Latin America and the Caribbean and defines a typology of these hotspots with reference to the biophysical and socioeconomic characteristics that unite and distinguish amongst them. Fifteen regional hotspots were identified on the basis of spatial criteria pertaining to the area, distribution, and rate of reforestation 2001–2014, observed using a custom continental MODIS satellite land-cover classification. Collectively, these hotspots cover 11% of Latin America and the Caribbean and they include 167,667.7 km2 of new forests. Comparisons with other remotely sensed estimates of reforestation indicate that these hotspots contain a significant amount of tropical reforestation, continentally and pantropically. The extent of reforestation as a proportion of its hotspot was relatively invariable (3–14%) given large disparities in hotspot areas and contexts. An ordination analysis defined a typology of five clusters, distinguished largely by their topographical roughness and related aspects of agro-ecological marginality, climate, population trends, and degree of urbanization: 'Urban lowlands', 'Mountainous populated areas', 'Rural highlands', 'Rural humid lands' and 'Rural dry lands'. The typology highlights that a range of distinct, even oppositional regional biophysical, demographic, and agricultural contexts have equally given rise to

significant, regional net reforestation, urging a concomitant diversification of forest transition science.

1. INTRODUCTION

Changes in tropical forest cover are primary features of global environmental change. Most studies addressing tropical forest cover change have focused on deforestation and its drivers (Gibbs et al., 2010; Hansen et al., 2013; Graesser et al. 2015, Curtis et al., 2018), identifying the loss of ~150 million hectares of tropical forest between 1990 and 2015 (Keenan et al., 2015). Tropical reforestation is, however, also a significant component of global environmental change (Meyfroidt & Lambin, 2008; Aide et al. 2013; Chazdon et al., 2016) that is far less understood, and that has reportedly increased in extent during recent decades (Aide & Grau, 2004; Hecht & Saatchi, 2007). Reforestation would have major implications for global bio-geoclimatic and ecological dynamics, such as carbon sequestration (Chazdon et al., 2016), environmental services (Wilson et al., 2017), and biodiversity conservation (Catterall et al., 2008). Early research on spontaneous tropical reforestation was framed on the "forest transition" model (Mather, 1992), which is based on patters and processes operating during the 19^{th} and 20^{th} centuries. Given the fast socioeconomic changes during the present, 21^{st} century forest expansion patterns and processes are likely to differ. To further understanding of reforestation as an emergent land-cover change, we delineate and characterize the reforestation hotspots of Latin America.

The forest transition narrative is based largely on early European precedents, and anticipates that reforestation arises from an "agriculture land-use adjustment" whereby agricultural modernization over fertile lands coincides with the abandonment of marginal agricultural land use (Mather & Needle, 1998). Localized case studies of recent tropical reforestation similarly purport that reforestation concentrated in agro-economically ‘marginal’ regions (Helmer, 2000; Helmer, 2004; ; Sloan et al., 2016). In Latin America, emerging forests were observed predominantly in topographically steep uplands (Asner et al., 2009; Redo et al., 2012; Aide et al., 2013; Nanni & Grau, 2014), peri-urban zones offering non-farm livelihood alternatives (Grau et al. 2003; Baptista, 2008; Grau et al., 2008; Gutierrez Angonese & Grau 2014), and in areas of land abandonment following major socioeconomic shifts, such as loss of subsides for sugar production in Cuba (Alvarez et al., 2013), or outmigration from Oaxaca, Mexico (Bonilla-Moheno et al., 2012). The land-use adjustment was considered to be induced or otherwise enhanced by urban-economic growth, rural emigration, and the globalization of land-use systems (Aide and Grau, 2004; Hecht and Saatchi, 2007) broadly aligned with modernistic notions of ‘development’ (Perz, 2007; Redo et al. 2012).

However, the direct application of the forest-transition narrative to contemporary tropical reforestation risks its undue corroboration at the expense of alternative or complementary processes (Sloan, 2015). This can occur because studies have focused exclusively on generalized ‘drivers’ nominated by theory, e.g., ‘urbanization’ (DeFries & Pandrey, 2010; DeFries et al., 2010), or on reforesting regions where the expected drivers are known to operate. Comprehensive assessments of reforestation encompassing all possible host contexts would alleviate this issue to some degree. Such assessments across the Neotropics have observed higher rates of reforestation in marginal, high-elevation areas, as well as high rates of deforestation in the lowland moist forest biome (Aide et al., 2013; Hansen et al., 2013; Rudel et al., 2016), suggesting that reforestation and deforestation may arise differentially amongst biomes due to their respective land-use constraints (Redo et al., 2012; Aide et al., 2013).

Although reforestation is increasingly recognized as an emergent regional phenomenon, only recently has it been observed at such scales (Redo et al., 2012; Aide et al., 2013; Hansen et al., 2013, Rudel et al. 2016). The regional contexts defining reforestation, which have not been described well, could to differ significantly from the geographical profiles prominent in the literature (Perz, 2007; Sloan, 2015; Sloan et al., 2016). Case studies provide a tenuous, potentially biased means of articulating overarching regional contexts or dynamics of reforestation (Sloan, 2015), particularly as many conflate small-scale reforestation and localized dynamics with a broader, long-term forest transition (Grau & Aide, 2008). Meta-analyses of case studies similarly extrapolated local observations to regional scales (Rudel et al., 2005) and relied on theoretical suppositions to fill empirical gaps (Meyfroidt & Lambin, 2011). Large-scale assessments of reforestation (e.g. Aide et al., 2013; Hansen et al., 2013) have given scant attention to the contexts of regional net reforestation, instead tending to quantify aggregate gross tree cover gains without differentiating planted from natural forests or ephemeral from sustained trends. Narrative assertions regarding the role of 'development' and 'marginality' and their variation amongst contexts, or indeed other drivers of tropical reforestation thus remain somewhat unrefined.

A definitive characterization of the regional contexts of reforestation across Latin America is critical for three reasons. First, it would provide missing information about the biophysical and socioeconomic conditions under which reforestation occurs. In effect, a comprehensive regional geography of Neotropical reforestation would provide an authoritative complement to the continued reliance on case studies (Sloan, 2015) and narratives based on northern hemisphere systems (Perz, 2007a, b). Improved contextual resolution is also essential for supporting reforestation and conservation initiatives that are frequently assuming ambitious scales (Chazdon & Guariguata, 2016). Amongst these are various continental forest-landscape restoration schemes, such as the 20x20 Initiative (World Resources Institute, 2015) and the Bonn Challenge (The Bonn Challenge, 2015), as well as programs for Reducing Emissions from Deforestation and forest Degradation (REDD+; Sloan, 2015), which are rapidly improvising national-scale schemes (Sloan et al., 2018).

Second, identifying regions of consistent reforestation would help identify the long-term benefits and beneficiaries of new forests (e.g. rural population livelihoods, biodiversity conservation, ecosystem services provision; Rey Benayas et al., 2009; Chazdon & Uriarte 2016), as well as distinguish them from often widespread areas of sporadic or ephemeral reforestation readily visible in satellite classifications (e.g., Hansen et al., 2013). Indeed, the persistence of new forests (Raid et al., 2017) and the scale of forest transitions are major but largely unexplored uncertainties, that regional delineations of contiguous, consistent reforestation would help addressing.

Third, a regional account of Neotropical reforestation would provide a necessary ontological correction to perspectives on the human dimensions of forest-cover change, which are still steeped in the rampant deforestation that characterized the latter half of the 20^{th} century. Significant regional net reforestation is, by definition, the culmination of a longer-term forest transition (Mather, 1992). Thus, the identification of the regional contexts of reforestation would shed light on the generality and diversity of conditions supporting forest transitions.

To improve understanding of reforestation as an emergent regional phenomenon, this article presents the first continental depiction of the significant regional reforestation areas during the early 21^{st} century. It offers two novel insights into Neotropical reforestation to address the uncertainties in its geography and contexts. Drawing upon comprehensive satellite-imagery analysis, it delineates ‘hotspots’ of extensive, significant, and consistent net reforestation across Latin America and the Caribbean between 2001 and 2014. Subsequently, it defines a typology of these hotspots with reference to the biophysical and socioeconomic characteristics that unite and distinguish amongst them. Finally, hotspots types are discussed with reference to case studies elaborating the biophysical and socioeconomic forces shaping regional conditions. In this way, we provide an empirical framework for further exploration of the conditions and processes of contemporary Neotropical reforestation.

2. MATERIALS AND METHODS

2.1. Overview

Four methodological steps defined the reforestation hotspots and their socio-biophysical typology. First, land cover was mapped annually between 2001 and 2014 across the Latin America and the Caribbean via satellite-image classification. Second, reforestation hotspots were delineated based on three spatial criteria ensuring significant rates and patterns of regional reforestation. Third, hotspots were characterized based on 14 social and biophysical attributes from which a socio-biophysical typology was statistically derived. Fourth, the contribution of the hotspots to forest-cover gain by biome was estimated.

2.2. Mapping 2001-2014 annual land cover in Latin America and the Caribbean

Annual land cover across Latin America and the Caribbean (LAC) was mapped over 2001-2014 using MODIS satellite data at 250-m spatial resolution. Following methods outlined elsewhere (Clark et al. 2012; Aide et al., 2013; Graesser et al. 2015), we used MODIS imagery, 60,000 land cover samples collected from visual interpretation of very high-resolution satellite imagery (~1-2 m resolution), and Random Forest (RF) classification models, to classify land cover across LAC. The extensive area and diverse landscapes across LAC limited the success of continental-scale classification test models. Therefore, we defined separate classification models bounded by the 191 terrestrial ecoregions (Olson et al., 2001) to more effectively capture differences in vegetation radiometric characteristics (e.g., dry Chaco forests compared to the Atlantic or Amazon forests) across the study area. A series of trials revealed that this approach improved land cover predictions over global estimates (e.g., MODIS MCD12Q1), with a trade-off of artificial transitions between some ecoregion zones. For each ecoregion, we trained a RF model with intersecting land cover samples from the LAC-wide pool of 60,000 samples to predict eight possible land covers: cropland, pastureland/grassland, natural tree cover, shrubs, tree plantations, barren land, (e.g., ice, snow, rock, sand dunes), built-up structures, and water. This study focuses on natural trees and shrubs (hereafter referred to as "woody") to restrict analyses to spontaneous reforestation to the extent that is possible, though inevitably some planted forests were confused with natural forest predictions (SI Table A).

A post-classification temporal smoothing filter was applied to the annual land-cover predictions to reduce the number of artificial year-to-year fluctuations of land-cover class predictions. Specifically, a three-year moving window was used to average the RF class-conditional posterior probabilities of membership to a given land-cover class, for a given year. For example, for a given pixel initially classified as natural tree cover in 2002 (based on the maximum class RF posterior probability), the three-year (2001—2003) average of RF probabilities for the natural tree-cover class for the pixel in question replaced the RF 2002 class probability. This process was repeated for each of the land-cover classes separately, for each year of our time series, per pixel. A two-year average was used for 2001 (2001 and 2002) and 2014 (2013 and 2014). For a given pixel in a given year, the maximum of the averaged probabilities of land-cover class membership ultimately determined its land-cover class for further analysis.

2.3. Delineating the reforestation hotspots

Rates of woody expansion (reforestation hereafter) between 2001 and 2014 across Latin America and the Caribbean were summarized individually by 15,969 hexagons of 1200 km^2 (average area of municipalities across Latin America and the Caribbean, Aide et al., 2013). These hexagons were subsequently iteratively linked with each other to define larger semi-contiguous networks representing the reforestation hotspots. Two hexagons were linked if: (i) the reforestation rates (2001 to 2014) of both hexagons were statistically significant ($p = 0.001, \text{using F} - \text{test}$); ii) they were within 1 degree (~111 km) of each other; and iii) the reforestation rates of both hexagons were greater than 100 ha yr^{-1} over 2001-2014. The first criterion ensured that hotspots were uniformly characterized by significant reforestation throughout the observation period, while the second condition incorporated disjointed hexagons into nearby developing networks or 'clusters' of hexagons. Developing networks were allowed to merge with other networks as the criteria were iteratively satisfied. The search radius of 1 degree was chosen after an exhaustive examination of alternative radii. An excessively large radius distance would have unduly limited the number of unique hotspots and missed the discrimination between functionally distinctive reforestation regions, while an excessively small radius would have over-segmented biogeographically integral clusters across the continent. The third criterion ensured that hotspots uniformly experienced aerially meaningful reforestation, as by excluding hexagons with statistically significant reforestation but negligible areas of reforestation. Hexagons were linked to progressively develop a hotspot if they met all three criteria. The hotspots are non-overlapping, meaning that a hexagon can only belong to one hotspot. This process was repeated for every hexagon across Latin America, creating an undirected, inductive network of an indeterminate number of reforestation hotspots.

Hotspots with fewer than 10 hexagons were removed from consideration in order to focus on major regional reforestation events. These omitted hotspots were Puerto Rico, another hotspot centered on Macapá city at the mouth of the Amazon river, and a third hotspot spanning the eastern stretch of the border between the Brazilian states of Goiás and Tocantis. Also, two initial hotspots resultant from the network analysis were subsequently sub-divided according to ecoregion boundaries, as these hotspots were relatively extensive, spanned numerous major ecoregions, and had relatively tenuous contiguity between these ecoregions. Such sub-division resulted in three Brazilian hotspots (Atlantic Forests, Cerrado, Caatinga) and three Mexican and Central American hotspots (Southern Mexico & Guatemala, Central America Pine Forests, Costa Rica & Panama). This subdivision was neither appropriate nor realized for the remaining hotspots as it would have resulted in over-segmentation, counteracting the criterion for regional continuity.

2.4. Hotspot accuracy assessment

The classification accuracy of the woody class (i.e., trees + shrubs) in each of the reforestation hotspots was assessed to verify the fidelity of the hotspots (SI Table A). Within the hotspots, 2,233 pixels (250m) from the 2014 land-cover classification were sampled. If a pixel occurred within a high-resolution image from 2010-2015 in Google Earth (typically ~1-2 m resolution) we classified its land cover on the basis of visual interpretation. Pixels interpreted as mixed (e.g., 50% pasture and 50% trees) were excluded from the validation. The average MODIS land-cover classification accuracy within the hotspots was 85% (SI Table A). Accuracy for the woody class alone was 91%, while for plantations it was 83.1%. These are considered to be upper estimates. The sample data consisted of pixels with homogenous land cover, whereas the majority of MODIS pixels are heterogeneous, especially in Mexico and Central America.

2.5. Describing a socioecological Typology of Reforestation Hotspots

A non-metric multidimensional scaling ordination approach (NDMS) was used to define a continental typology of reforestation hotspots on the basis of 14 biophysical and socioeconomic attributes (Table 1). In contrast to other ordination techniques, NMDS makes no assumptions about how variables are distributed along gradients (Kenkel & Orlóci, 1986). The ordination was based on a matrix of euclidean distances (Legendre & Legendre, 1998) calculated using all 14 biophysical and socioeconomic attributes, described below. The final ordination featured two main dimensions of social and biophysical traits. The final "stress" value (an index of agreement between the distances in the graph configuration and the distances in the original data matrix) was 12.3, which is well within the recommended threshold of 20 (Legendre & Legendre, 1998). Pearson correlations between the 14 attributes and the individual hotspot scores in the ordination space were also estimated, and their significance was assessed via 1000 random permutations of the data (Oksanen et al., 2015). All analyses were performed using the vegan package in R software (Oksanen et al., 2015). Once the ordination was performed, clusters or typologies were defined, and hotspots belonging to the same cluster were connected by its group centroid.

The 14 attributes describing the reforestation hotspots capture themes observed or theorized to be relevant to reforestation at different scales (Grau & Aide 2008, Meyfroidt & Lambin, 2011). They include topographic / agro-ecological marginality, rural depopulation, settlement intensity (urbanization), socioeconomic development, and agricultural productivity. Climatic attributes for 1950-2000 provide an additional layer of information to explain the distribution of reforestation. All attributes are spatially explicit, with varying scales/resolutions typically of ~1 km^2 (Table 1). Prior to the NMDS ordination, attributes were summarized (i.e., averaged, summed) and standardized per hotspot.

Attributes related to agricultural productivity were mean agricultural yield, relative change in agricultural area, and relative change in pasture area (2001-2014) (Table 1). The agricultural yield attribute refers to yields of 19 major crops (barley, cassava, cotton, groundnut, maize, millet, oilpalm, potato, rapeseed, rice, rye, sorghum, soybean, sugarbeet, sugarcane, sunflower and wheat), based on a global map of croplands for 2000 and national agriculture yield statistics (Monfreda et al., 2008). Yields for each crop were standardised across the hotspots to derive a summary value of mean standardised yield for all crops combined, per hotspot. The relative areas of agricultural change and pastoral change pertain to agricutlural and pastoral changes over 2001-2014 as proportions of agricultural and pastoral areas in 2001, respectively, as derived from the land-cover estimates. It is assumed that observed grassland changes corresponded mostly to trends in planted pastures rather than natural grasslands.

Four attributes summarized population dynamics within the hotspots: population density, rural/urban population ratio, rural population change, and urban population change. For all these attributes, LandScan (2000 and 2012) 1-km population data (Bhaduri et al., 2002; Bright et al., 2012) were used. Estimates for populaton change in rural and urban areas were performed by overlapping LandScan population data sets of 2001 and 2012 with the urban-extent map of CIESIN (2011). This urban-extent map distinguishes urban from rural areas based on a combination of local population counts (persons), settlement points, and the presence of nightime lights.

Settlement intensity was further estimated with reference to built-up and roaded areas. Satellite-observed nightlight luminosity (Maus et al., 2010), which captures a wide range of persistent electric illumination from dim villages to bright city centers, indicates urban and peri-urban settlement intensity but also indirectly their economic intensity, thus complementing our population density attributes. Road density was calculated by dividing the sum of road length in each hotspot by its area. Road data pertains largely to arterial and inter-urban roadways as of 1980-2010, depending on the country (CIESIN, 2013).

Finally, the Human Development Index (HDI) values were estimated for each reforestation hotspot. HDI values were originally derived directly for individual municipalities, which were then averaged for each encompassing hotspot, with municipality values weighted by the number of hexagons comprising the municipality. The HDI reflects economic income, education, and life expectancy to describe levels of 'development' observed to correlate with reforestation at regional scales (Redo et al., 2012). HDI values for each municipality were obtained from the latest source available, including national and international sources (e.g. Klugman et al., 2009).

Once the hotspot typology was obtained, case studies of land-cover change within the regional hotspots were revised and considered, to elaborate and qualify the local dynamics and conditions that collectively define the regional typology or contexts of reforestation.

Table 1. Biophysical and socioeconomic attributes used to typify reforestation hotspots.

Theme	Description	Spatial Scale	Temporal Scale/Year	Source
Bioclimatic	Mean annual temperature (°C)	1 km^2	1950-2000	Hijmans et al., 2005
	Mean annual precipitation (mm/year)	1 km^2	1950-2000	Hijmans et al., 2005
Topographic Marginality	Elevation (m.a.s.l)	90 m^2	-	Jarvis et al., 2008
	Topographic roughness: SD of Elev. (m.a.s.l)	90 m^2	-	GIS-derived from Elev.
Agriculture production	Mean agriculture yield (T)	10 km^2	2000	Monfreda et al., 2008
	Relative Change in Agricultural Area	250 m	2001-2014	MODIS classification
	Relative Change in Pasture Area	250 m	2001-2014	MODIS classification
Population dynamics	Population density (N° people/km^2)	1 km^2	2012	LandScan, 2012
	Rural-urban ratio	-	2012	LandScan (2000 & 2012) and CIESIN, 2005.

	Rural Population Change	1 km^2	2000-2012	LandScan (2000 & 2012) and CIESIN, 2005.
	Urban Population Change	1 km^2	2000-2012	LandScan (2000 & 2012) and CIESIN, 2005.
Urbanization	NightlightDensity (DN/km^2)	6km^2	2010	NGDC, 2010
	Road Density (km/km^2)	m/km^2	1980-2010	CIESIN, 2013
Socioeconimic development	Human Development Index (0-1)			Various sources

2.5. Contribution of the hotspots to forest cover by biome

Rates of forest loss and gain are variable across biomes (Hansen et al., 2013), possibly reflecting inter-biome differences in predominant land uses, land-use constraints, and remnant-vegetation coverage (Sloan et al., 2014). Therefore, the contribution of the hotspots to reforestation by biome was also evaluated by two comparative measures. First, the extent of reforestation in a given biome within the hotspots (2001-2014) was compared to the continental area of that biome, as defined by Olson et al. (2001). This allowed us to explore whether larger biomes had proportionally large areas of reforestation from the hotspots. Such proportionality was an uncertainty, given that larger biomes (particularly the Tropical and Subtropical Moist Broadleaf Forest biome, and the Tropical and Subtropical Dry Broadleaf Forest biome) have experienced extensive deforestation due to historical agricultural colonization (Achard et al., 2002; Miles et al., 2006; Aide et al., 2013; Rudel et al., 2016). Second, the extent of reforestation within each biome was compared with the representation of the biomes within the hotspots, to explore whether higher reforestation rate in a given biome could be due to its higher representation within the hotspots.

3. RESULTS

3.1. Delineating the reforestation hotspots

Our analysis identified 15 regional hotspots of sustained net reforestation in Latin America and the Caribbean between 2001 and 2014 (Figure 1): Southern Mexico & Guatemala, Central America Pine Forests, the Pacific realm of Costa Rica/Panama, Cuba, Dominican Republic & Haiti, Colombian Andes, uplands of south Ecuador/north Peru, Venezuelan Coast, Roraima of Venezuela/Brazil, Caatinga of Brazil, Atlantic Forests of Brazil, Cerrado of Brazil, Beni of Bolivia, Pantanal & Paraguayan Chaco, and Southern Tropical Andes. These hotspots covered 2,209,930 km^2, representing 11.2% of Latin America and the Caribbean. Collectively, the hotspots accounted for167,667.7 km^2 of net reforestation occurring over 2001-2014, defining a 7.6% reforestation rate for this period.

The extent of reforestation within the hotspots is appreciable. Net reforestation during 2001-2014 added between 7% and 55% of the extant forest area of 2001 across the hotspots. In comparison, the percentages of the hotspot extents recovered by reforestation was relatively constant across the hotspots (3% to 14%), despite notable discrepancies in hotspot extents (Figure 2). High ratios of reforestation to extant forest occurred both in hotspots with low and high extant (2001) woody cover, the latter of which are represented by Cuba and the Southern Mexico & Guatemala hotspots (SI Table B).

3.2. A Socioecological Typology of Reforestation Hotspots

The NMDS ordination defined five overarching types of Neotropical reforestation hotspots, distinguished largely by topographic roughness and related aspects of agro-ecological marginality, climate, population trends, and degree of urbanization. The hotspot types are “Urban lowlands” (Costa Rica/Panama, Atlantic Forests, Cuba, and Venezuela

Coast); “Mountainous populated areas” (Colombian Andes, Central-America Pine Forests, Southern Mexico & Guatemala, and Dominican Republic & Haiti); “Rural highlands” (Southern Tropical Andes, and uplands of south Ecuador-north Peru); “Rural humid lands” (Roraima, Cerrado, and Beni) and “Rural dry lands” (Caatinga and Pantanal & Paraguayan Chaco) (Figure 3).

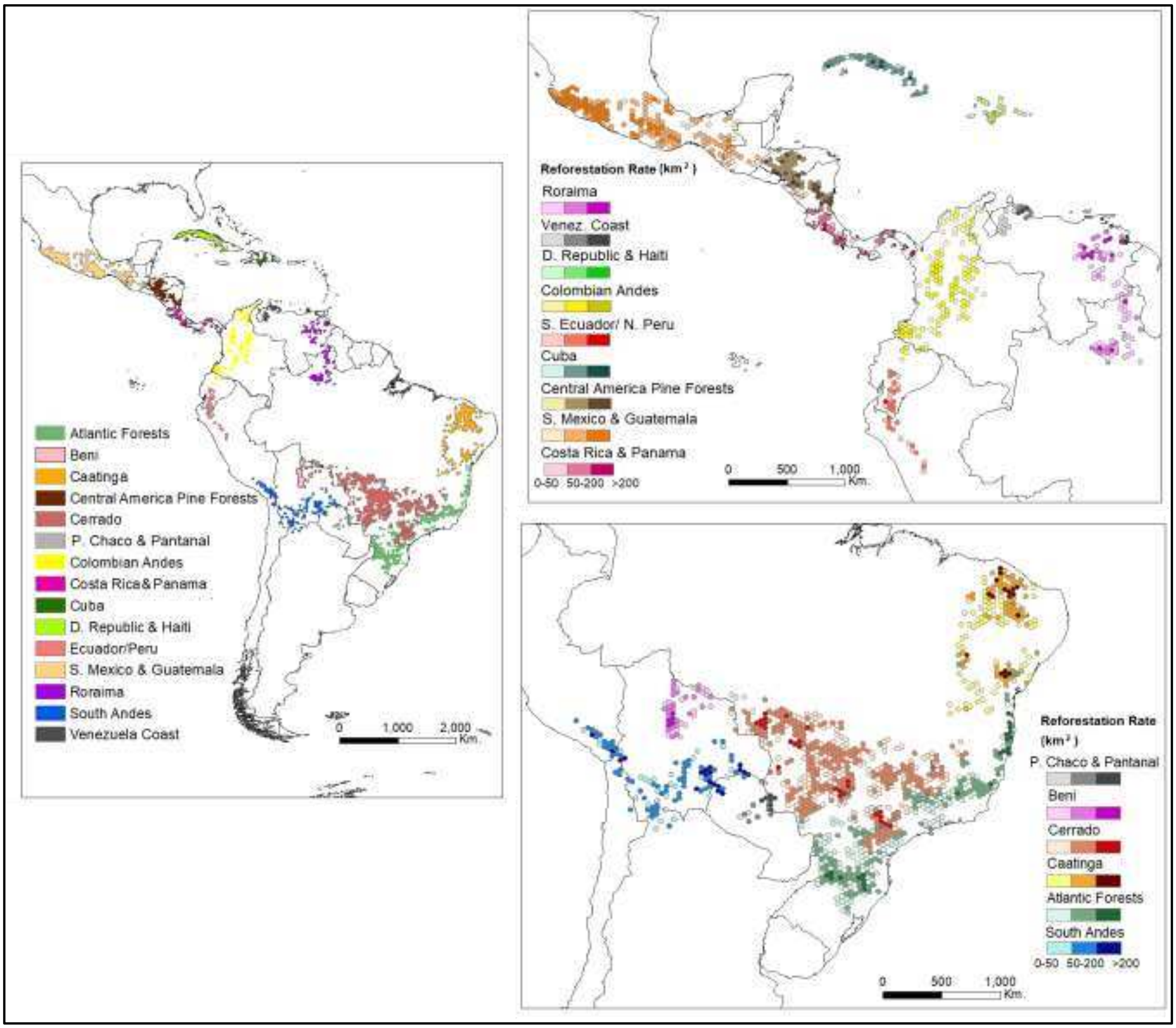

Figure 1. Reforestation hotspots of Latin America and the Caribbean (left side). Right side: Rate of net reforestation (2001-2014) in each hexagon, for northern South America, Central America and North America (top right), and the rest of South America (bottom right). Graduated color pallet indicates the amount of net reforestation between 2001 and 2014 per hexagon (km^2): 0-50 (light); 50-200 (medium) and >200 (dark).

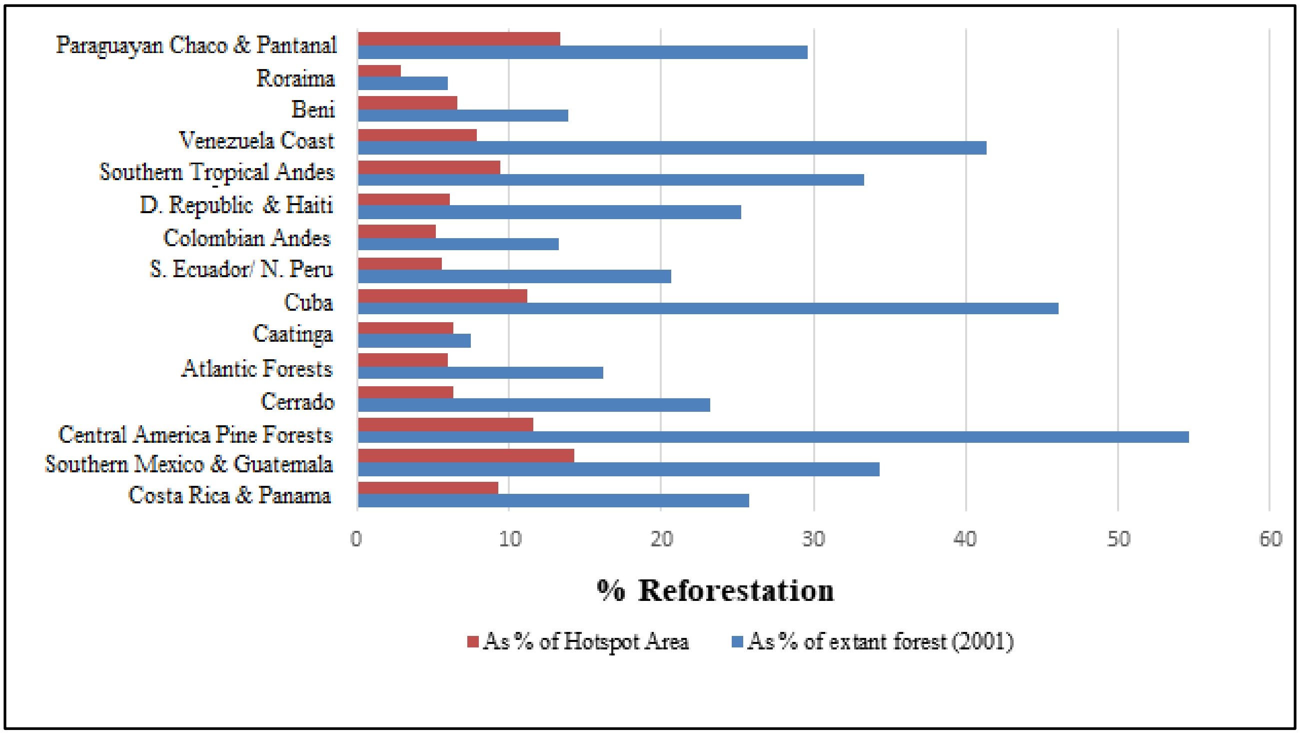

Figure 2. Reforestation in each hotspot, expressed as percent of extant forest area in the hotspot as of 2001, and as a percent of hotspot area.

The first axis of the ordination represents a gradient of 'rurality' and 'dryness' (Figure 3); significantly and negatively correlated with rural-to-urban population ratio, and precipitation. Positively associated hotspots (i.e., rural and dry) also exhibit declining agricultural areas (Table 2) – a trend that is marginally significant ($p<0.1$) but consistent with theoretical expectations of land abandonment in relatively marginal agro-ecological zones. In the ordination space, this axis establishes a spectrum of hotspots, from the relatively urbanized and tropical (e.g., Costa Rica/Panama, Colombian Andes) to the rural and semi-arid (e.g. Southern Tropical Andes, Caatinga). (Figure 3, Table 2).

The second axis of the ordination is a gradient of topographic 'elevation' and 'urbanization'. This axis significantly correlates with rural outmigration and urban population growth, thus distinguishing urbanizing hotspots positively associated with this second axis from the already relatively urban hotspots positively associated with the first axis. This second axis also significantly correlates with settlement intensity (nightlight density population density, road density) and agricultural yield, characterizing hotspots positively associated with this axis as sparsely settled and relatively unproductive (Figure 3, Table 2). A significant positive association with temperature and a negative association with elevation is also evident (Table 2). Accordingly, the hotpots towards the positive side of the second axis correspond with relatively underproductive, lowland, warm rural areas undergoing rural population decline (e.g. Beni, Roraima), including areas affected by frequent flooding (Pantanal, Beni). In contrast, the negative side of the axis corresponds with urbanized regions in lowlands (e.g., Venezuela Coast) and uplands (e.g., Central American Pine Forests) with greater agricultural productivity. Towards the extreme negative end of axis 2, two mountainous hotspots (uplands of south Ecuador/ north Peru, and Southern Tropical Andes) constitute a Rural Highlands cluster, differentiated from the Populated Highlands cluster by even higher elevation, lower temperature, denser and more stable rural population, and greater agricultural productivity.

3.3. Contribution of the hotspots to forest cover by biome

The reforestation hotspots spanned eight of the 11 biomes that comprise Latin America and the Caribbean, excepting the Temperate Grasslands, Savannas and Shrublands, the Temperate and Mixed Forests, and the Mediterranean Forests, Woodlands and Scrub (SI Table B). The contributions of hotspot reforestation to the Neotropical biomes area varied from 0.53% for the Tropical and Subtropical Moist Broadleaf Forests biome to 5.7% for the Tropical and Subtropical Coniferous Forests biome (Figure 4a). The large reforestation rate of this biome is due to the high reforestation rate in the Southern Mexico & Guatemala hotspot (Figure 2; SI Table B).

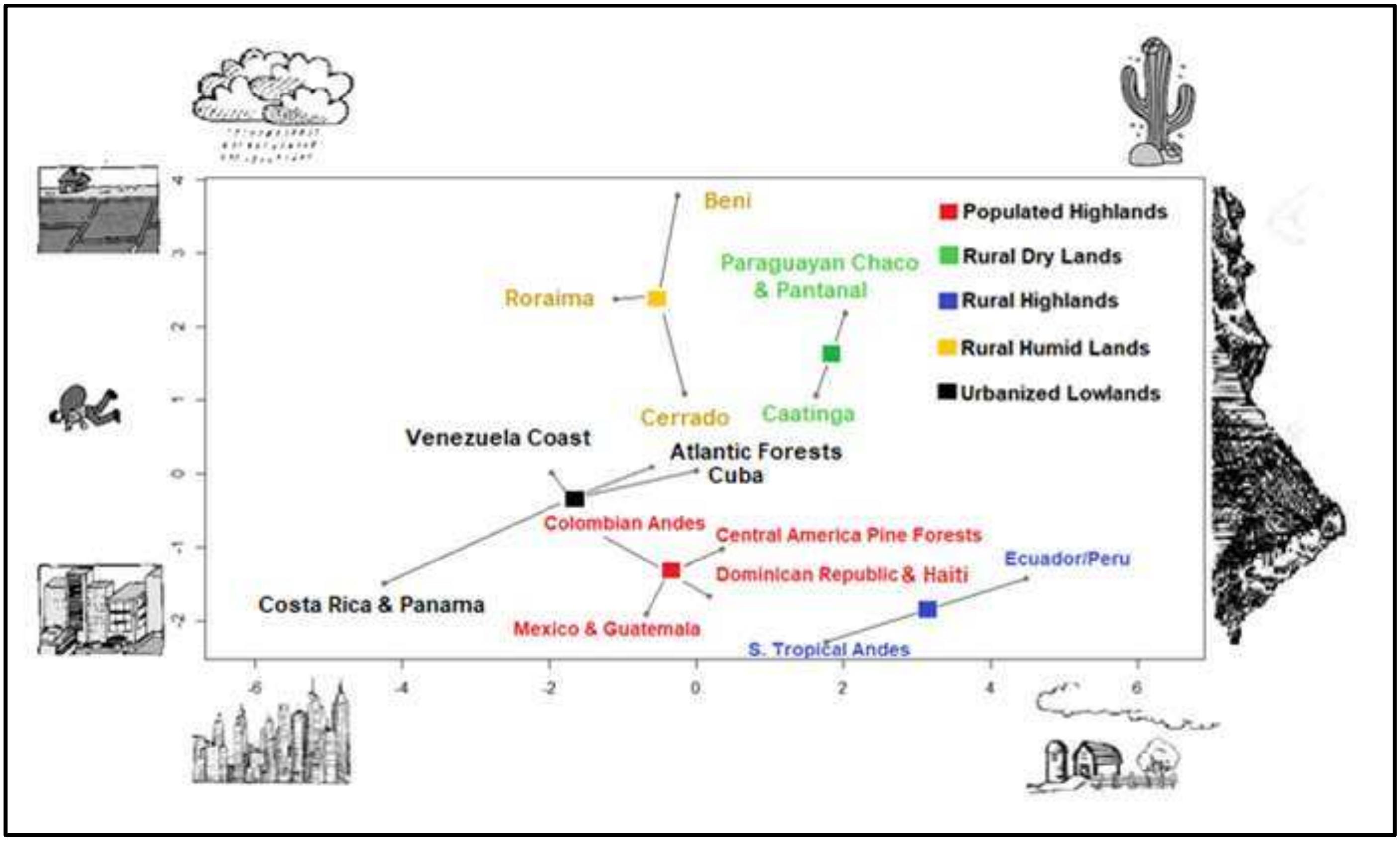

Figure 3. Non-metric multidimensional scaling ordination (NMDS) of the hotspots based on 14 biophysical and socioeconomic attributes. Centroids of the five clusters are represented by colored squares: Rural Dry Lands (green), Rural Humid Lands (orange), Urbanized Lowlands (black), Mountainous Populated (red) and Rural Highlands (blue). Figures in the border of the ordination diagram capture the main attributes correlated with each axis. Axes values are unitless.

A greater representation of a biome within the hotspots did not generally correspond with higher percentage area reforested (Figure 4b). While a subtle correspondence is apparent for some biomes (Figure 4b left side), any overall trend is upset by significant variations in the continental areas of biomes (e.g., mangrove vs. moist forests), and their historical exposure to forest change (e.g., montane grasslands vs. coniferous forests). The area reforested in each biome attributable to the hotspots increased roughly linearly with the continental biome area in all biomes except the moist forest biome (Figure 4c). Upon including the moist forest biome, a nonlinear relationship is observed, reflecting the relatively low reforestation rate of this extensive biome (Figure 4c), much of which is remote and subject to changes in forest cover. Overall, smaller biomes were reforested disproportionately more, considering their continental areas (Figure 4c), particularly the coniferous forest and the dry forest biomes. Otherwise, reforestation within the hotspots appears to have not favored specific biomes, including those well-represented within the hotspots.

Table 2. Pearson correlations for axes 1 and 2 scores and the 14 biophysical and socioeconomic attributes values. Socioeconomic and biophysical attribute loadings on each axis are bold when they are ≥0.75 and significantly correlated at $p<0.05$.

Attribute	Axis 1	Axis 2	Variance Explained (R^2)	Significance (p)
Elevation	0.5994	**-0.8004**	0.668	0.001
Roughness	0.3719	**-0.9283**	0.577	0.007
Mean Yield	-0.4538	**-0.8911**	0.443	0.030
Precipitation	**-0.9977**	0.0670	0.634	0.004
Temperature	-0.6409	**0.7676**	0.688	0.002
Rural Change	0.2784	**-0.9605**	0.411	0.030
Urban Change	0.2279	**0.9737**	0.482	0.020
Rural/Urban Ratio	**0.7978**	0.6029	0.511	0.010
Population Density	-0.4153	**-0.9097**	0.526	0.009
Nightlight Density	-0.6138	**-0.7894**	0.642	0.004
Road Density	-0.6256	**-0.7801**	0.444	0.030
Rel. change in agricultural area	-0.9780	-0.2084	0.386	0.090
Rel. Change in pasture Area	-0.9646	-0.2636	0.259	0.160
HDI	-0.8844	0.4667	0.234	0.208

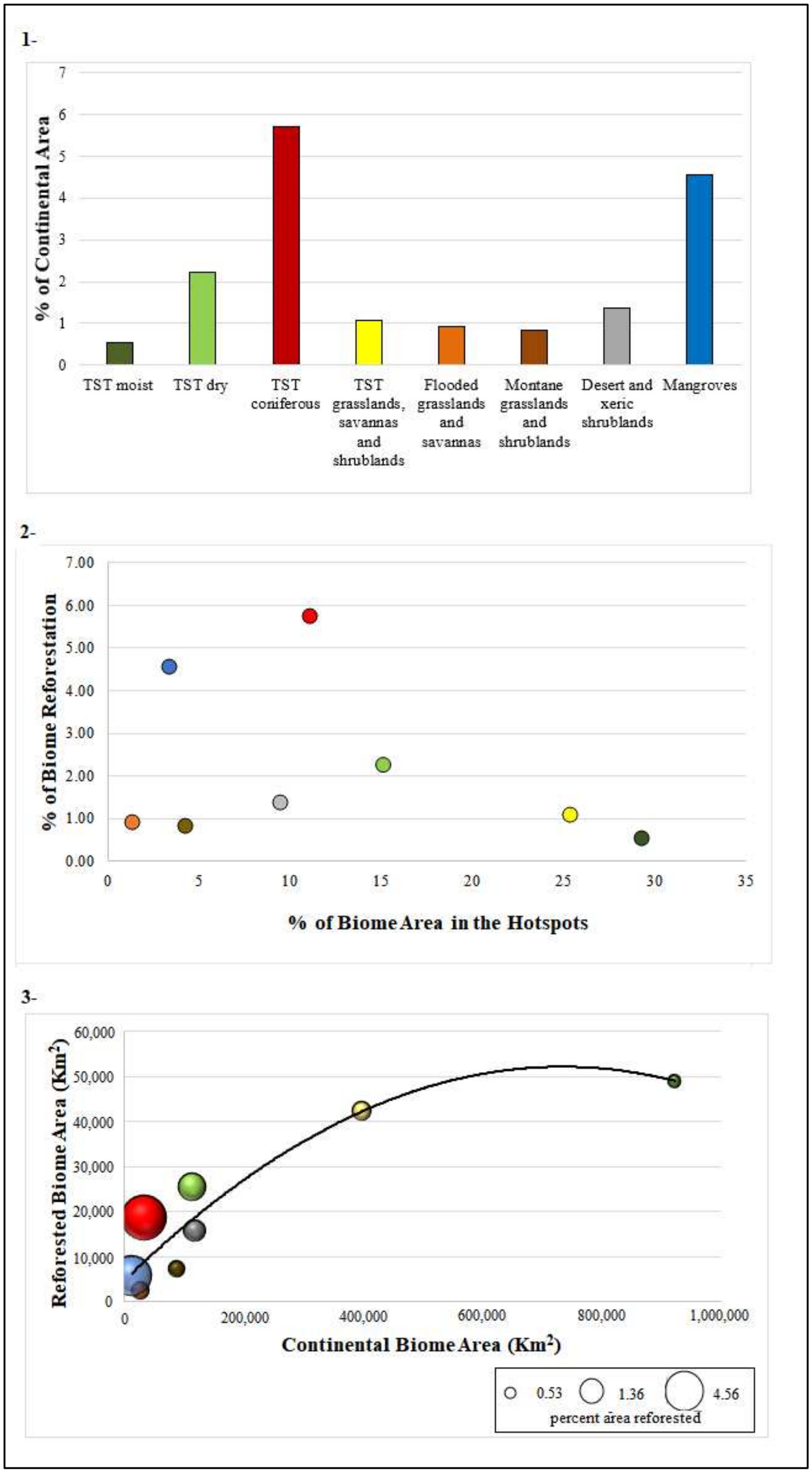

Figure 4. Hotspot reforestation by biome: **(a)** reforestation as percent of continental biome area, for the eight Neotropical biomes coincident with the reforestation hotspots; **(b)** Percent of area reforested per biome versus the percent biome area within the hotspots **(c)** Reforested area per biome versus continental biome area.

4. DISCUSSION

4.1 Regional Concentrations of Reforestation

Despite occurring in a context of extensive deforestation across Latin America (Aide et al. 2013; Hansen et al. 2013; Sloan & Sayer 2015), this study identified regional Neotropical reforestation hotspots defined by significant trends in net expansion of woody cover between 2001 and 2014. These hotspots and their new forest cover represent 11% and 1% of the continental area, respectively. Notwithstanding the challenges of direct comparisons between remotely-sensed estimates, our hotspots apparently account for large proportions of total reforestation, both continentally and pantropically. Although spanning only 11% of Latin America and the Caribbean, the hotspots account for 37% of gross continental reforestation (woody gain) according to our land-cover classification, 50% of similar continental estimates of gross reforestation by Aide et al., 2013, and 67% of finer-scale gross pantropical reforestation estimated by Hansen et al. (2013). Regardless, the proportion of total reforestation confined to our hotspots is likely greater in the long term than such proportions suggest, considering the likely greater persistence of reforestation within the hotspots. Part of gross reforestation observed by Aide et al. (2013), Hansen et al. (2013) and others (Beuchle et al., 2015) is relatively ephemeral and often associated with nearby forest losses (Rudel et al., 2016). In contrast, our hotspots delineate expansive, semi-contiguous, regional zones of net reforestation. As such, their reforestation presumably reflects underlying ecological conditions and societal transformations yielding woody gains that are likely to be relatively enduring.

The relative constancy of reforestation percentages (between 3 and 14%) amongst hotspots of marked geographical and contextual disparities hints at a potential upper limit on the ultimate extent of forest recovery, in keeping with forest-transition narratives. The new forests identified here occurred in all the major Neotropical biomes, with greater extents of reforestation in smaller biomes, which contrasts with continued predominance of deforestation in larger biomes (Sloan et al., 2014), especially the Tropical and Subtropical moist Forests (Aide et al., 2013). The relatively high levels of reforestation in the Tropical and Subtropical Dry Forests and Desert and Xeric Shrublands biomes, particularly in Brazil, are especially noteworthy due to the critical status of these biomes, which harbor less than 10% of their natural area (Sloan et al., 2014). The potential contributions of these new forested areas to ecological recovery are promising but remain uncertain. Continuous, appreciable reforestation relative to the 2001 extant forest across hotspots (average 26%), will likely favor biodiversity conservation. For example, woody expansion in the tropical Andes and Mesoamerican mountains, is particularly important for biodiversity and conservation of water resources. Even more important is the remarkable recovery in the Atlantic forest hotspot, given its extent, biodiversity, and limited remnant forest cover (<15%) (Ribeiro et al., 2009; SOSMA, 2012; Sloan et al., 2014). However, confident assertions to this end ultimately await regional analyses of the coincidence of new forests and threatened species, accounting for species' tolerance of secondary-forest habitat (Gibson et al., 2011), and the persistence and contiguity of reforestation (Latawiec et al., 2016; Reid et al., 2017).

4.2 Limitations and Caveats

While our approach ensured the delineation of hotspots defined by extensive, significant, and potentially persistent regional reforestation, it entails limitations that should not be overlooked. First, by focusing on major regional reforestation events deemed likely to indicate transformative underlying trends, our delineation excluded smaller, dispersed reforestation events, particularly across small Caribbean islands, such as the Dutch Caribbean, Saint Lucia, and Puerto Rico (Rudel et al., 2000; Grau et al., 2003; van Andel et al., 2016; Walters, 2017). Despite their small contribution to continental-scale processes, reforestation in these Caribbean islands is of great conservation importance due to the islands' distinctive biodiversity and the reliance of their populations on forest ecosystem services (Myers et al., 2000).

Second, our analysis observes forest gains only since 2001, due to MODIS satellite image availability. Transitions from deforestation to reforestation were not observable within such a brief period. Any correspondence between the hotspots and forest transitions is therefore implicit. Hotspots are assumed to be indicative of emergent forest transitions, considering that they were all widely characterized by deforestation over most 20th century. Indeed, our focus on 'recent' reforestation allows for historical continuity. By capturing consistent reforestation trends, rather than spurious reforestation events, our hotspots exhibit an affinity with reforestation epicenters of the late 20th century, as in Costa Rica (Calvo-alvarado, 2000), Panama (Sloan, 2015), Brazil (Baptista & Rudel, 2006) and Mexico (Galicia et al., 2008). Reforestation in many hotspots commenced before 2001, and may continue well into the future, as suggested by the case studies discussed below.

Third, potential confusion between natural and planted forest cover cannot be entirely discounted. Our land-cover classification was accurate (SI Table A) and distinguished natural from planted forest cover; yet the nature of our analysis and its coarse pixel size may still allow for confusion among these forest classes. Such confusion is most likely in hotspots where reforestation is known to encompass both planted and natural forest expansion, namely the Atlantic Forests in Brazil (da Silva et al., 2015), or in mountain regions where new forests are interspersed with shade coffee (Redo et al. 2012). In hotspots affected by frequent flooding and wetland dynamic regimes (e.g. Beni, Pantanal & Paraguayan Chaco), forest cover change may actually be associated to changes in water cover.

4.3 A Contextual Typology of Reforestation

Our typology of neotropical reforestation hotspot is a typology of equals. The two gradients of social and biophysical contexts that distinguish amongst hotspot types exhibit marked contextual diversity, even though they were relatively consistent in terms of reforestation rates. This typology implies that a range of distinct, even oppositional regional biophysical, demographic, and agricultural conditions can equally give rise to significant reforestation events. Conceptually, this contextual diversity resonates with theoretical frameworks of multiple socio-agrarian pathways towards the forest transitions (Lambin & Meyfroidt, 2010), while not corroborating any per se theory.

The forest-transition literature has persistently advanced reforestation narratives centered on ‘agro-ecological marginality’ and ‘economic development/modernization’, (Rudel, 2005; Angelsen & Rudel, 2013). The coincidence of outmigration and topographic roughness with higher agricultural yields in our typology conflates, and possibly challenges, these narratives. In particular, topography, a common proxy for marginality, has been considered as a key influencing factor of reforestation, with farmers abandoning remote, sloped lands to cultivate flatter, lower elevation lands (Aide & Grau, 2004; Aide et al 2013); yet our hotspots typology features reforestation also in lowlands. This is possibly the result of the separate manifestation of these narratives within different hotspots, parts of which may be undergoing different dynamics (e.g. lowlands and mountains). For example, in mountains “marginality” (in terms of competitive disadvantage for agriculture production) may not be the result of low soil fertility (reflected in the statistics of per hectare yield) but of the difficulties for mechanization, which results in higher production costs. In lowlands experiencing woodland expansion, this may actually happen in relatively small steep locations (hills, river coasts), not captured by the overall description of topographic roughness at the scale of analysis. However, it is also possible that in other areas absolute agro-ecological marginality is only a coincident or secondary factor of a more complex upland reforestation dynamic. The following subsections discuss case studies of reforestation exploring these processes in each of the five hotspot clusters identified by our typology. Local processes vary amongst hotspots even of a given cluster, challenging the generality of reforestation narratives.

4.3.1 Urban Lowlands (Costa Rica/Panama, Venezuela Coast, Atlantic Forests, and Cuba)

The four hotspots of this cluster occur in urbanized lowland regions. Notwithstanding some common contextual features, the dynamics of reforestation in these hotspots are varied.

Conformant with our typology, case studies within the Atlantic Forests hotspot highlight peri-urban forest transitions promoted by urbanization in Santa Catarina (Baptista

& Rudel, 2006; Baptista, 2008), as well as conservation initiatives for tourism and recreation in Sao Paulo (Ehlers, 2007) and environmental protection policies leading to reforestation (Costa et al., 2017). Other reforestation dynamics are also present, including agroforestry landscapes with Eucalyptus spp., shade coffee, and cocoa in Minas Gerais and Bahia states (Cardoso et al., 2001; Lobão et al., 2007).

In Cuba, extensive reforestation is not necessarily resulting from urbanization. Instead reforestation has followed the loss of Soviet agricultural subsidies and subsequent reforms to lowland agricultural estates, with sugar production particularly affected (Alvarez et al., 2013); a pattern observed in many post-soviet economies (Rudel et al. 2016). Although an increase in woody vegetation occurred in abandoned sugarcane fields, a large proportion of this vegetation is a single exotic species (El Marabu, D. cinerea), which presently covers approximately 18% of Cuba, and that results in limited environmental advantages (Alvarez et al., 2013).

Panama and Costa Rica comprise a single hotspot, but their disparate socio-political dynamics may vary the state of their new forests. In both countries, the main driver of reforestation seems to be the de-agriculturalization of labor and related retractions of agricultural land (Arroyo-Mora et al., 2005; Sloan, 2015); as has been observed in Puerto Rico (Rudel et al., 2000, Grau et al., 2003). In Costa Rica, environmental policy/laws, eco-tourism, and a heightened environmental consciousness apparently enhanced reforestation, as by protecting secondary forests from conversion (Calvo-Alvarado, 2000; Fagan et al., 2014). In Panama, new forests concentrate in populous rural areas host to growing urban hamlets or are otherwise peripheral to the rapidly expanding Panama City (Sloan, 2015). As such, they are presumably more likely to be degraded and re-converted than in Costa Rica.

In the Venezuelan Coast hotspot, the few available studies addressing reforestation ascribe it to woody encroachment in the open savanna, influenced by changes in cattle density and fire regimes (Silva et al., 2001). As in the adjacent llanos of Colombia, the Venezuelan reforestation may also be attributable to the conversion of crops and exotic grasses to palm oil (García-Ulloa et al., 2012; Romero-Ruíz et al., 2012), and avocado plantations (E. Chacon, pers. comm). Nationally, the cultivated area of these crops has increased 60.4% and 65.5%, respectively, over 2000-2015 (FAOSTAT, 2016).

4.3.2 Mountainous Populated Areas (Southern Mexico & Guatemala, Colombian Andes, Dominican Republic, and Central America Pine Forests)

The four hotspots of this typology occur in contexts of high elevation and topographic roughness, high yields, and high population density. Such steep elevation gradient defines heterogeneous areas with a mix of market-oriented and subsistence agricultural practices. Arguably more than elsewhere, forest trends in these hotspots reflect regional changes in economic activities, such as the extensification of marginal agricultural production, in addition to localized population dynamics. Similarly, forest-change trends in these regions are relatively dynamic, with forest redistribution and turnover prevailing over any given forest trend (Redo et al., 2012).

The Colombian and Mexican hotspots are associated with recent decreases in rural population (SI Table B). In both hotspots, reforestation resulted mainly from agricultural abandonment in rural areas, but with varied drivers. In the Colombian Andes, reforestation occurred in tropical and montane forests over pre-existing mixed woody covers (shrubs and herbs) and the abandonment of subsistence agricultural systems is mostly due to recent land conflicts and economic development, with associated migration to urban centres (Sanchez Cuervo et al., 2012; Rubiano et al., 2017). In Oaxaca, reforestation reflects rural outmigration, but also community forest management for certified wood extraction (Gómez-Mendoza et al., 2006; Bray et al., 2009; Robson & Berks, 2011). In Chiapas, the main factor explaining reforestation after a century of forest loss seems to be the expansion of plantation forestry, particularly oil palm, stimulated by government subsidies (Vaca et al., 2012). In Guerrero, secondary dry forests have expanded in the last decades, as a consequence of smallholder farm abandonment (Galicia et al., 2008).

The Central America Pine Forest and Dominican Republic hotspots are associated with negligible rural population change since 2000 (SI Table B). In the former hotspot, coniferous dry forest expansion occurred in Honduras, Nicaragua, and Guatemala to a lesser extent, simultaneously with high deforestation rates in their humid broadleaf forest frontiers (e.g., Guatemalan Peten, Nicaraguan Caribbean), resulting in a forest-redistribution dynamic (Redo et al., 2012). In Honduras, reforestation is due partly to the cultivation of shade-coffee in the uplands, in addition to reforestation through secondary succession (Bass, 2006). In these Central American countries, community forest management also seems to play a role in maintaining forest cover, including secondary forests (Bray & Anderson, 2010), while economic remittances from migrants in the USA have reduced agricultural activities and enhanced forest regrowth (Hecht & Saatchi, 2007; Davis et al., 2010). Such factors may explain the coincidence of reforestation and high rural population density in this region. In the Dominican Republic, reforestation has followed the gradual abandonment of marginal grazing lands and cacao plantations, accompanied by early stages of vegetation succession (Rivera et al., 2000; Slocum et al., 2004; Grau et al. 2008), likely due to rural outmigration and shifts towards non-agriculture activities in rural areas (Castañeda, 2003). Exotic tree species comprise an important proportion of the resultant new forests (20% of all woody basal area) (Alvarez et al., 2013).

4.3.3 Rural Highlands (Ecuador/Peru and South Andes)

These hotspots are characterized by very high elevations (mean 2400-2600 m.a.s.l, SI Table B), lower temperatures, and very rural contexts (i.e., low densities of population, nightlights, and roads). Reforestation there occurred mostly over montane grasslands and shrublands (South Andes) or previously-cleared montane forests (Ecuador/Peru). In both hotspots, reforestation likely corresponds to the expansion of woodlands, including a mix of shrubs and trees, such as Alnus acuminata, Polylepys spp. and Prosopis spp., (Morales et al., 2005; Kintz et al., 2006; Farley, 2007; Weber et al., 2008; Araóz & Grau, 2010). Interactions between fire, land use (especially grazing), and climate influence woodland dynamics in these highlands (Kok et al., 1995), in some cases giving rise to reforestation as rural populations and climatic patterns shift (Morales et al. 2005; Carilla & Grau 2010; Aráoz & Grau, 2010). The South Andes hotspot also includes lower elevation areas of the Bolivian Dry Chaco and Chiquitano Dry Forests, where reforestation has reportedly occurred after the abandonment of fallow agricultural fields close to extant forests, thus allowing for rapid regeneration (Kennard et al., 2002). The wide elevation gradient encompassed by this hotspot (SI Table B) brings it relatively close to the Mountainous Populated Areas cluster in the ordination space (Fig. 3).

4.3.4 Rural Humid Hotspots (Roraima, Beni, and Cerrado)

These hotspots are defined by hot, humid, lowlands, with low rural population densities and settlement intensities. However, their increasing urban populations coupled with high rates of rural outmigration, underlines a nascent urbanization (Table 2).

In Roraima and the Cerrado, rural outmigration has been an important factor of reforestation. In Roraima, reforestation corresponded with forest regeneration in formerly-grazed lands situated within forest mosaics (Kammesheidt, 2000; Feldspauch et al., 2004). In the Cerrado, reforestation came from spontaneous growth of both of trees and shrublands within matrices dominated by pasture, following decreases in grazing as well as burning (Vieira et al., 2006). Resprouting tree species seem to be highly resilient and capable of regenerating even after long periods of disturbance (e.g., more than 40 years; Sampaio et al., 2007). In Beni, in contrast, reforestation appears to have resulted from secondary forest succession under community fallow management (Toledo & Salick, 2006), notwithstanding the aforementioned decreases in rural population. The difference between the landscape matrices of reforestation in Beni and Roraima (reforestation amongst forest patches) and in the Cerrado (reforestation amongst pastures) likely results in very different degrees of forest connectivity.

4.3.5 Rural Dry hotspots (Pantanal & Paraguayan Chaco and Caatinga)

The Pantanal & Paraguayan Chaco, and Caatinga hotspots comprise the Rural Dry cluster due to their low precipitation and high degree or rurality (low rural populations, settlement and road density), again coincident with apparent nascent urbanization (Figure 2, SI Table B). Unlike other hotspot types, reforestation in this type did not occur in forest biomes but almost exclusively in the Tropical and Subtropical Grasslands, Shrublands and Savanna biome in the Pantanal & Paraguayan Chaco; and the Desert and Xeric Shrubland biome in the Caatinga (SI Table C).

In the Pantanal & Paraguayan Chaco hotspot, the observed woody expansion might be mostly attributable to biophysical constrains: in the Paraguayan Chaco, the comparatively low deforestation of the last decades in comparison with other ecoregions within the country, such as the Atlantic forests, has been driven by the Mennonite community dominating the region. However, poor soil quality is a limiting factor for agriculture expansion, thus the resultant agriculture systems are not sustainable in the long-term (Huang et al., 2009; Caldas et al., 2011). This might have led to the observed reforestation in these areas, which overlaps with very low cropland and pastureland changes (Graesser et al., 2015). In the Pantanal, vegetation dynamics are largely influenced by temporal and spatial dynamics of water, with annual and multi-annual wet and dry periods resulting in large-scale changes in vegetation cover that might be the origin of our observed reforestation (Nunes da Cunha et al., 2007).

In the Caatinga, reforestation is associated with the abandonment of indigenous small-scale agriculture and cattle ranching, but the, remaining forested areas are highly degraded due to poor land management, timber extraction, and increasing frequency of severe droughts (Sampaio et al., 1993), retarding the regeneration of nearby abandoned lands (Pereira et al., 2003). The combination of cattle ranching and the use of fire for slash-and-burn agriculture in this region have limited forest propagation upon land abandonment due to a reduction of the seed bank density as well as seedlings (Mamede & Araujo, 2008).

4.4 Conclusion

Reforestation in Latin America and the Caribbean is fairly concentrated in 15 hotspots defining five clusters of varied social and biophysical attributes. Echoing earlier calls (Sloan, 2008), the contextual diversity inherent to our typology of reforestation hotspots urges the exploration of a variety of situations promoting reforestation. Our typology provides an initial framework to this end, and aligns only partially with the preeminent forest-transition pathways. Our clusters differ from one another in important ways, and both biophysical and social attributes equally give origin to such differentiation: hotspots were found in the lowlands and in the highlands, and in rural and peri-urban contexts, and reforestation occurred under decreasing, stable and growing populations (Fig. 3). Despite such variety of socioecological contexts, the reported underlying processes influencing reforestation in each hotspot were in general not as varied, even among clusters. In the majority of the hotspots, reforestation was reported to occur due to socioeconomic changes leading to the abandonment of land, which emphasizes the importance of identifying conditions under which agricultural lands become no profitable even in a context of growing global demand for agriculture products. Other processes such as explicit environmental policies gave place to reforestation in the Atlantic Forests and Costa Rica, and community forest management seemed to have favored the occurrence of reforestation in Central America and Oaxaca. To fully understand the significance of these reforestation hotspots identified by our studies, two main issues remain to be addressed: the identification of the drivers of reforestation at a regional scale; and the implications of these reforested regions for biodiversity conservation and ecosystem service provision. We believe that our identification of the regional Neotropical typology is an important, and purposeful first step towards these ultimate goals.

Acknowledgements

This paper is a product of PARTNERS Research Coordination Network (People and Reforestation in the Tropics, a Network for Research, Education, and Synthesis). Funding for this project was provided by Grant DEB-1313788 from the U.S. NSF Coupled Human and Natural Systems Program; and PICT 2015-0521 from the Argentine Ministry of Science and Technology.

Literature Cited

1. Achard, F., Eva, H. D., Stibig, H. J., Mayaux, P., Gallego, J., Richards, T., & Malingreau, J. P. (2002). Determination of deforestation rates of the world's humid tropical forests. Science, 297(5583), 999-1002.

2. Angelsen, A., & Rudel, T. K. (2013). Designing and implementing effective REDD+ policies: A forest transition approach. Review of Environmental Economics and Policy, 7(1), 91-113.

3. Aide, T. M., & Grau, H. R. (2004). Globalization, migration, and Latin American ecosystems. Science, 305(5692), 1915-1916.

4. Aide, T. M., Clark, M. L., Grau, H. R., López-Carr, D., Levy, M. A., Redo, D., Bonilla-Moheno, M., Riner, G., Andrade-Núñez, M.J & Muñiz, M. (2013). Deforestation and reforestation of Latin America and the Caribbean (2001–2010). Biotropica, 45(2), 262-271.

5. Álvarez-Berríos, N. L., Redo, D. J., Aide, T. M., Clark, M. L., & Grau, R. (2013). Land Change in the Greater Antilles between 2001 and 2010. Land, 2(2), 81-107.

6. Aráoz, E., & Grau, H. R. (2010). Fire-mediated forest encroachment in response to climatic and land-use change in subtropical Andean treelines. Ecosystems, 13(7), 992-1005.

7. Arroyo-Mora, J. P., Sánchez-Azofeifa, G. A., Rivard, B., Calvo, J. C., & Janzen, D. H. (2005). Dynamics in landscape structure and composition for the Chorotega region, Costa Rica from 1960 to 2000. Agriculture, Ecosystems & Environment, 106(1), 27-39.

8. Asner, G. P., Rudel, T. K., Aide, T. M., DeFries, R., & Emerson, R. (2009). A contemporary assessment of change in humid tropical forests. Conservation Biology, 23(6), 1386-1395.

9. Baptista, S. R., & Rudel, T. K. (2006). A re-emerging Atlantic forest? Urbanization, industrialization and the forest transition in Santa Catarina, southern Brazil. Environmental Conservation, 33(3), 195-202.

10. Baptista, S. (2008). Metropolitanization and forest recovery in southern Brazil: a multiscale analysis of the Florianópolis city-region, Santa Catarina State, 1970 to 2005. Ecology and Society, 13(2).

11. Bass, J. J. (2006). Forty years and more trees: Land cover change and coffee production in Honduras. Southeastern Geographer, 46(1), 51-65.

12. Beuchle, R., Grecchi, R. C., Shimabukuro, Y. E., Seliger, R., Eva, H. D., Sano, E., & Achard, F. (2015). Land cover changes in the Brazilian Cerrado and Caatinga biomes from 1990 to 2010 based on a systematic remote sensing sampling approach. Applied Geography, 58, 116-127.

13. Bhaduri, B., Bright, E., Coleman, P., & Dobson, J. (2002). LandScan. Geoinformatics, 5(2), 34-37.

14. Bicudo da Silva, R. F., Batistella, M., Moran, E. F., & Lu, D. (2017). Land changes fostering Atlantic forest transition in Brazil: Evidence from the Paraíba Valley. The Professional Geographer, 69(1), 80-93.

15. Bonilla-Moheno, M., Aide, T. M., & Clark, M. L. (2012). The influence of socioeconomic, environmental, and demographic factors on municipality-scale land-cover change in Mexico. Regional Environmental Change, 12(3), 543-557.

16. Bontemps, S., Defourny, P., Bogaert, E. V., Arino, O., Kalogirou, V., & Perez, J. R. (2011). GLOBCOVER 2009-Products description and validation report.

17. Bray, D. B., & Anderson, A. B. (2005). Global conservation non-governmental organizations and local communities: perspectives on programs and project implementation in Latin America (No. 1). Working Paper.

18. Bray, D. B. (2009). Forest Cover Dynamics and Forest Transitions in Mexico and Central America: Towards a “Great Restoration”? In Reforesting Landscapes (pp. 85-120). Springer, Dordrecht.

19. Bright, E. A., Rose, A. N., & Urban, M. L. (2012). LandScan 2012.

20. Caldas, M. M., Goodin, D., Sherwood, S., Campos Krauer, J. M., & Wisely, S. M. (2015). Land-cover change in the Paraguayan Chaco: 2000–2011. Journal of Land Use Science, 10(1), 1-18.

21. Calvo, J. (2000). Case study: payments for watershed services in Costa Rica. Forests in Sustainable Mountain Development. A State-of-Knowledge Report for, 2000, 428-429.

22. Cardoso, I. M., Guijt, I., Franco, F. S., Carvalho, A. F., & Neto, P. F. (2001). Continual learning for agroforestry system design: university, NGO and farmer partnership in Minas Gerais, Brazil. Agricultural Systems, 69(3), 235-257.

23. Carilla, J., & Grau, H. R. (2010). 150 years of tree establishment, land use and climate change in montane grasslands, northwest Argentina. Biotropica, 42(1), 49-58.

24. Castañeda, A. B. (2003). Desarrollo económico y social de la República Dominicana, 1990-2000. Intec.

25. Catterall, C. P., Kanowski, J., & Wardell-Johnson, G. W. (2008). Biodiversity and new forests: interacting processes, prospects and pitfalls of rainforest restoration. Living in a dynamic tropical forest landscape, 510-525.

26. Center for International Earth Science Information Network – CIESIN. (2013). Columbia University, and Information Technology Outreach Services - ITOS - University of Georgia. Global Roads Open Access Data Set, Version 1 (gROADSv1). Available at http://dx.doi.org/10.7927/H4VD6WCT. Accessed September 15, 2017.

27. Chazdon, R. L., Broadbent, E. N., Rozendaal, D. M., Bongers, F., Zambrano, A. M. A., Aide, T. M., Balvanera, P., Becknell, J.M., Boukili, V., Brancalion, P.H. & Craven, D. (2016). Carbon sequestration potential of second-growth forest regeneration in the Latin American tropics. Science Advances, 2(5), e1501639.

28. Chazdon, R. L., & Guariguata, M. R. (2016). Natural regeneration as a tool for large-scale forest restoration in the tropics: prospects and challenges. Biotropica, 48(6), 716-730.

29. Chazdon, R. L., & Uriarte, M. (2016). Natural regeneration in the context of large-scale forest and landscape restoration in the tropics. Biotropica, 48(6), 709-715.

30. CIESIN, F. (2011). CIAT. (2005). Gridded population of the world [dataset], 23(07).

31. Clark, M. L., Aide, T. M., & Riner, G. (2012). Land change for all municipalities in Latin America and the Caribbean assessed from 250-m MODIS imagery (2001–2010). Remote Sensing of Environment, 126, 84-103.

32. Curtis, P. G., Slay, C. M., Harris, N. L., Tyukavina, A., & Hansen, M. C. (2018). Classifying drivers of global forest loss. Science, 361(6407), 1108-1111.

33. Davis, B., Carletto, G., & Winters, P. C. (2010). Migration, transfers and economic decision making among agricultural households: an introduction. The Journal of Development Studies, 46(1), 1-13.

34. DeFries, R., & Pandey, D. (2010). Urbanization, the energy ladder and forest transitions in India's emerging economy. Land Use Policy, 27(2), 130-138.

35. DeFries, R. S., Rudel, T., Uriarte, M., & Hansen, M. (2010). Deforestation driven by urban population growth and agricultural trade in the twenty-first century. Nature Geoscience, 3(3), 178.

36. Ehlers, E. (2007). Empreendedorismo e conservação ambiental no interior de São Paulo. Revista de Economia e Sociologia Rural, 45(1), 185-203.

37. Fagan, M. E., DeFries, R. S., Sesnie, S. S., Arroyo-Mora, J. P., Soto, C., & Chazdon, R. L. (2014, July). Integrating hyperspectral and multitemporal Landsat imagery to monitor tree plantation expansion in northeastern Costa Rica. In ForestSAT2014 Open Conference System.

38. FAOSTAT (2016). http://www.fao.org/faostat, 16.05.2016. Accessed January 26, 2018.

39. Farley, K. A. (2007). Grasslands to tree plantations: forest transition in the Andes of Ecuador. Annals of the Association of American Geographers, 97(4), 755-771.

40. Feldpausch, T. R., Rondon, M. A., Fernandes, E., Riha, S. J., & Wandelli, E. (2004). Carbon and nutrient accumulation in secondary forests regenerating on pastures in central Amazonia. Ecological applications, 14(sp4), 164-176.

41. Friedl, M. A., McIver, D. K., Hodges, J. C., Zhang, X. Y., Muchoney, D., Strahler, A. H., Woodcock, C.E., Gopal, S., Schneider, A., Cooper, A. & Baccini, A. (2002). Global land cover mapping from MODIS: algorithms and early results. Remote Sensing of Environment, 83(1), 287-302.

42. Friedl, M. A., Sulla-Menashe, D., Tan, B., Schneider, A., Ramankutty, N., Sibley, A., & Huang, X. (2010). MODIS Collection 5 global land cover: Algorithm refinements and characterization of new datasets. Remote sensing of Environment, 114(1), 168-182.

43. Galicia, L., Zarco-Arista, A. E., Mendoza-Robles, K. I., Palacio-Prieto, J. L., & García-Romero, A. (2008). Land use/cover, landforms and fragmentation patterns in a tropical dry forest in the southern Pacific region of Mexico. Singapore Journal of Tropical Geography, 29(2), 137-154.

44. Garcia-Ulloa, J., Sloan, S., Pacheco, P., Ghazoul, J., & Koh, L. P. (2012). Lowering environmental costs of oil-palm expansion in Colombia. Conservation Letters, 5(5), 366-375.

45. Gibbs, H. K., Ruesch, A. S., Achard, F., Clayton, M. K., Holmgren, P., Ramankutty, N., & Foley, J. A. (2010). Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. Proceedings of the National Academy of Sciences, 107(38), 16732-16737.

46. Gibson, L., Lee, T. M., Koh, L. P., Brook, B. W., Gardner, T. A., Barlow, J., Peres, C.A., Bradshaw, C.J., Laurance, W.F., Lovejoy, T.E. & Sodhi, N. S. (2011). Primary forests are irreplaceable for sustaining tropical biodiversity. Nature, 478(7369), 378-381.

47. Gómez-Mendoza, L., Vega-Peña, E., Ramírez, M. I., Palacio-Prieto, J. L., & Galicia, L. (2006). Projecting land-use change processes in the Sierra Norte of Oaxaca, Mexico. Applied Geography, 26(3), 276-290.

48. Graesser, J., Aide, T. M., Grau, H. R., & Ramankutty, N. (2015). Cropland/pastureland dynamics and the slowdown of deforestation in Latin America. Environmental Research Letters, 10(3), 034017.

49. Grau, H. R., Aide, T. M., Zimmerman, J. K., Thomlinson, J. R., Helmer, E., & Zou, X. (2003). The ecological consequences of socioeconomic and land-use changes in post-agriculture Puerto Rico. AIBS Bulletin, 53(12), 1159-1168.

50. Grau, H.R, & Aide, M. (2008). Globalization and land-use transitions in Latin America. Ecology and Society, 13(2).

51. Grau, H.R, Hernández, M., Gutierrez, J., Gasparri, N., Casavecchia, M., Flores-Ivaldi, E., & Paolini, L. (2008). A peri-urban neotropical forest transition and its consequences for environmental services. Ecology and Society, 13(1).

52. Grau, H. R., Pérez Ceballos, M., Martinuzzi, S., Encarnación, X., & Aide, T. M. (2008). Cambios socioeconómicos y regeneración del bosque en la República Dominicana. Restauración de Bosques en América Latina, 211-227.

53. Gutiérrez-Angonese, J., & Grau, H. R. (2014). Assessment of swaps and persistence in land cover changes in a subtropical peri-urban region, NW Argentina. Landscape and Urban Planning, 127, 83-93.

54. Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R. & Kommareddy, A. (2013). High-resolution global maps of 21st-century forest cover change. science, 342(6160), 850-853.

55. Hecht, S. B., & Saatchi, S. S. (2007). Globalization and forest resurgence: changes in forest cover in El Salvador. AIBS Bulletin, 57(8), 663-672.

56. Helmer, E. H. (2000). The landscape ecology of tropical secondary forest in montane Costa Rica. Ecosystems, 3(1), 98-114.

57. Helmer, E. H. (2004). Forest conservation and land development in Puerto Rico. Landscape Ecology, 19(1), 29-40.

58. Huang, C., Kim, S., Song, K., Townshend, J. R., Davis, P., Altstatt, A., ... & Musinsky, J. (2009). Assessment of Paraguay's forest cover change using Landsat observations. Global and Planetary Change, 67(1-2), 1-12.

59. Kammesheidt, L. (2000). Some autecological characteristics of early to late successional tree species in Venezuela. Acta Oecologica, 21(1), 37-48.

60. Keenan, R. J., Reams, G. A., Achard, F., de Freitas, J. V., Grainger, A., & Lindquist, E. (2015). Dynamics of global forest area: results from the FAO Global Forest Resources Assessment 2015. Forest Ecology and Management, 352, 9-20.

61. Kenkel, N. C., & Orlóci, L. (1986). Applying metric and nonmetric multidimensional scaling to ecological studies: some new results. Ecology, 67(4), 919-928.

62. Kennard, D. K. (2002). Secondary forest succession in a tropical dry forest: patterns of development across a 50-year chronosequence in lowland Bolivia. Journal of tropical ecology, 18(1), 53-66.

63. Kintz, D. B., Young, K. R., & Crews-Meyer, K. A. (2006). Implications of land use/land cover change in the buffer zone of a national park in the tropical Andes. Environmental Management, 38(2), 238-252.

64. Klugman, J., Medalho, I., & United, P. (2009). Human Development Reports.

65. Kok, K., Verweij, P. A., & Beukema, H. (1995). Effects of cutting and grazing on Andean tree-line vegetation. Biodiversity and Conservation of Neotropical Montane Forest. The New York Botanical Garden. New York, 527-539.

66. Lambin, E. F., & Meyfroidt, P. (2010). Land use transitions: Socio-ecological feedback versus socio-economic change. Land use policy, 27(2), 108-118.

67. Latawiec, A. E., Crouzeilles, R., Brancalion, P. H., Rodrigues, R. R., Sansevero, J. B., Santos, J. S. D., Mills, M., Nave, A.G. & Strassburg, B. B. (2016). Natural regeneration

and biodiversity: a global meta-analysis and implications for spatial planning. Biotropica, 48(6), 844-855.

68. Legendre, P., & Legendre, L. (1998). Numerical ecology: second English edition. Developments in environmental modelling, 20.

69. Lobão, D. E., Setenta, W. C., Lobão, E. S. P., Curvelo, K., & Valle, R. R. (2007). Cacau cabruca: sistema agrossilvicultural tropical. Ciência, tecnologia e manejo do Cacaueiro, 290-323.

70. Mamede, M. D. A., & de Araújo, F. S. (2008). Effects of slash and burn practices on a soil seed bank of caatinga vegetation in Northeastern Brazil. Journal of Arid Environments, 72(4), 458-470.

71. Mather, A. S. (1992). The forest transition. Area, 367-379.

72. Mather, A. S., & Needle, C. L. (1998). The forest transition: a theoretical basis. Area, 30(2), 117-124.

73. Maus, S., Manoj, C., Rauberg, J., Michaelis, I., & Lühr, H. (2010). NOAA/NGDC candidate models for the 11th generation International Geomagnetic Reference Field and the concurrent release of the 6th generation Pomme magnetic model. Earth, planets and space, 62(10), 2.

74. Myers, N., Mittermeier, R. A., Mittermeier, C. G., Da Fonseca, G. A., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. Nature, 403(6772), 853.

75. Meyfroidt, P., & Lambin, E. F. (2008). The causes of the reforestation in Vietnam. Land use policy, 25(2), 182-197.

76. Meyfroidt, P., & Lambin, E. F. (2011). Global forest transition: prospects for an end to deforestation. Annual review of environment and resources, 36.

77. Miles, L., Newton, A. C., DeFries, R. S., Ravilious, C., May, I., Blyth, S., Kapos, V. & Gordon, J. E. (2006). A global overview of the conservation status of tropical dry forests. Journal of Biogeography, 33(3), 491-505.

78. Monfreda, C., Ramankutty, N., & Foley, J. A. (2008). Farming the planet: 2. Geographic distribution of crop areas, yields, physiological types, and net primary production in the year 2000. Global biogeochemical cycles, 22(1).

79. Morales, M. S., Villalba, R., & Boninsegna, J. A. (2005). Climate, land-use and Prosopis ferox recruitment in the Quebrada de Humahuaca, Jujuy, Argentina. Dendrochronologia, 22(3), 169-174.

80. Nanni, A. S., & Grau, H. R. (2014). Agricultural adjustment, population dynamics and forests redistribution in a subtropical watershed of NW Argentina. Regional environmental change, 14(4), 1641-1649.

81. Nunes da Cunha, C., Junk, W. J., & Leitão-Filho, H. F. (2007). Woody vegetation in the Pantanal of Mato Grosso, Brazil: a preliminary typology. Amazoniana, 19(3/4), 159-184.

82. Oksanen, J., Blanchet, F. G., Kindt, R., Legendre, P., Minchin, P. R., O'hara, R. B., Simpson, G.L., Solymos, P., Henry, M., Stevens, H. & Wagner, H. (2015). vegan: Community Ecology Package. R package version 2.0-10. 2013. There is no corresponding record for this reference.

83. Olson, D. M., Dinerstein, E., Wikramanayake, E. D., Burgess, N. D., Powell, G. V., Underwood, E. C., D'amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C & Loucks, C. J. (2001). Terrestrial Ecoregions of the World: A New Map of Life on Earth: A new global map of terrestrial ecoregions provides an innovative tool for conserving biodiversity. BioScience, 51(11), 933-938.

84. Pereira, I. M., Andrade, L. A., Sampaio, E. S. B. V., & Barbosa, M. R. V. (2003). Use-history effects on structure and flora of caatinga. Biotropica, 35(2), 154-165.

85. Perz, S. G. (2007). Grand Theory and Context-Specificity in the Study of Forest Dynamics: Forest Transition Theory and Other Directions. The Professional Geographer, 59(1), 105-114.

86. Perz, S. G. (2007). Reformulating modernization-based environmental social theories: Challenges on the road to an interdisciplinary environmental science. Society and Natural Resources, 20(5), 415-430.87.

87. Redo, D. J., Grau, H. R., Aide, T. M., & Clark, M. L. (2012). Asymmetric forest transition driven by the interaction of socioeconomic development and environmental heterogeneity in Central America. Proceedings of the National Academy of Sciences, 109(23), 8839-8844.

88. Reid, J. L., Wilson, S. J., Bloomfield, G. S., Cattau, M. E., Fagan, M. E., Holl, K. D., & Zahawi, R. A. (2017). How long do restored ecosystems persist? Annals of the Missouri Botanical Garden, 102(2), 258-265.

89. Rey Benayas, J. M., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. science, 325(5944), 1121-1124.

90. Ribeiro, M. C., Metzger, J. P., Martensen, A. C., Ponzoni, F. J., & Hirota, M. M. (2009). The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biological conservation, 142(6), 1141-1153.

91. Rivera, L. W., Zimmerman, J. K., & Aide, T. M. (2000). Forest recovery in abandoned agricultural lands in a karst region of the Dominican Republic. Plant Ecology, 148(2), 115-125.

92. Robson, J., & Berkes, F. (2011). How does out-migration affect community institutions? A study of two indigenous municipalities in Oaxaca, Mexico. Human Ecology, 39(2), 179-190.

93. Romero-Ruiz, M. H., Flantua, S. G. A., Tansey, K., & Berrio, J. C. (2012). Landscape transformations in savannas of northern South America: Land use/cover changes since 1987 in the Llanos Orientales of Colombia. Applied Geography, 32(2), 766-776.

94. Rubiano, K., Clerici, N., Norden, N., & Etter, A. (2017). Secondary forest and shrubland dynamics in a highly transformed landscape in the Northern Andes of Colombia (1985–2015). Forests, 8(6), 216.

95. Rudel, T. K., Perez-Lugo, M., & Zichal, H. (2000). When fields revert to forest: development and spontaneous reforestation in post-war Puerto Rico. The Professional Geographer, 52(3), 386-397.

96. Rudel, T. K., Coomes, O. T., Moran, E., Achard, F., Angelsen, A., Xu, J., & Lambin, E. (2005). Forest transitions: towards a global understanding of land use change. Global environmental change, 15(1), 23-31.

97. Rudel, T. K., Sloan, S., Chazdon, R., & Grau, R. (2016). The drivers of tree cover expansion: Global, temperate, and tropical zone analyses. Land Use Policy, 58, 502-513.

98. Sampaio, E. D. S., Salcedo, I. H., & Kauffman, J. B. (1993). Effect of different fire severities on coppicing of caatinga vegetation in Serra Talhada, PE, Brazil. Biotropica, 452-460.

99. Sampaio, A. B., Holl, K. D., & Scariot, A. (2007). Does restoration enhance regeneration of seasonal deciduous forests in pastures in central Brazil? Restoration Ecology, 15(3), 462-471.

100. Sánchez-Cuervo, A. M., Aide, T. M., Clark, M. L., & Etter, A. (2012). Land cover change in Colombia: surprising forest recovery trends between 2001 and 2010. PloS one, 7(8), e43943.

101. Silva, J. F., Zambrano, A., & Fariñas, M. R. (2001). Increase in the woody component of seasonal savannas under different fire regimes in Calabozo, Venezuela. Journal of Biogeography, 28(8), 977-983.

102. Sloan, S. (2008). Reforestation amidst deforestation: simultaneity and succession. Global Environmental Change, 18(3), 425-441.

103. Sloan, S., Jenkins, C. N., Joppa, L. N., Gaveau, D. L., & Laurance, W. F. (2014). Remaining natural vegetation in the global biodiversity hotspots. Biological Conservation, 177, 12-24.

104. Sloan, S. (2015). The development-driven forest transition and its utility for REDD+. Ecological Economics, 116, 1-11.

105. Sloan, S., & Sayer, J. A. (2015). Forest Resources Assessment of 2015 shows positive global trends but forest loss and degradation persist in poor tropical countries. Forest Ecology and Management, 352, 134-145.

106. Sloan, S., Goosem, M., & Laurance, S. G. (2016). Tropical forest regeneration following land abandonment is driven by primary rainforest distribution in an old pastoral region. Landscape Ecology, 31(3), 601-618.

107. Sloan, S., Pereira, J. C. Z., Labbate, G., Asner, G. P., & Imbach, P. (2018). The cost and distribution of forest conservation for national emissions reductions. Global Environmental Change, 53, 39-51.

108. Slocum, M. G., Aide, T. M., Zimmerman, J. K., & Navarro, L. (2004). Natural regeneration of subtropical montane forest after clearing fern thickets in the Dominican Republic. Journal of Tropical Ecology, 20(4), 483-486.

109. SOSMA, 2012. SOS Mata Atlântica e INPE divulgam dados do Atlas dos Remanescentes Florestais da Mata Atlântica, no período de 2010 a 2011, Accessed October 2013, http://www.sosma.org.br/5697/sos-mata-atlantica-e-inpe-divulgam-dados-do-atlas-dos-remanescentes-florestais-da-mata-atlantica-no-periodo-de-2010-a-2011/

110. The Bonn Challenge, 2015. The Bonn Challenge, Accessed August 2015, http://www.bonnchallenge.org/

111. Toledo, M., & Salick, J. (2006). Secondary succession and indigenous management in semideciduous forest fallows of the Amazon Basin. Biotropica, 38(2), 161-170.

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

113. van Andel, T., van der Hoorn, B., Stech, M., Arostegui, S. B., & Miller, J. (2016). A quantitative assessment of the vegetation types on the island of St. Eustatius, Dutch Caribbean. Global Ecology and Conservation, 7, 59-69.

114. Vieira, D. L., Scariot, A., Sampaio, A. B., & Holl, K. D. (2006). Tropical dry-forest regeneration from root suckers in Central Brazil. Journal of tropical ecology, 22(3), 353-357.

115. Weber, M., Günter, S., Aguirre, N., Stimm, B., & Mosandl, R. (2008). Reforestation of abandoned pastures: silvicultural means to accelerate forest recovery and biodiversity. Gradients in a tropical mountain ecosystem of Ecuador, 431-441.

116. Wilson, S., Schelhas, J., Grau, R., Nanni, A.S, & Sloan, S. (2017). Forest ecosystem-service transitions: the ecological dimensions of the forest transition. Ecology and Society, 22(4).

117. World Resources Institute, 2015. Initiative 20x20 - Bringing 20 million hectares of degraded land in Latin America and the Caribbean into restoration by 2020, Accessed

November 2017, http://www.wri.org/our-work/project/initiative-20x20/about-initiative-20x20