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Carbon emissions from agricultural expansion and intensification in the Chaco

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Abstract

Carbon emissions from land-use changes in tropical dry forest systems are poorly understood, although they are likely globally significant. The South American Chaco has recently emerged as a hot spot of agricultural expansion and intensification, as cattle ranching and soybean cultivation expand into forests, and as soybean cultivation replaces grazing lands. Still, our knowledge of the rates and spatial patterns of these land-use changes and how they affected carbon emissions remains partial. We used the Landsat satellite image archive to reconstruct land-use change over the past 30 years and applied a carbon bookkeeping model to quantify how these changes affected carbon budgets. Between 1985 and 2013, more than 142 000 km^2 of the Chaco's forests, equaling 20% of all forest, was replaced by croplands (38.9%) or grazing lands (61.1%). Of those grazing lands that existed in 1985, about 40% were subsequently converted to cropland. These land-use changes resulted in substantial carbon emissions, totaling 824 Tg C between 1985 and 2013, and 46.2 Tg C for 2013 alone. The majority of these emissions came from forest-to-grazing-land conversions (68%), but post-deforestation land-use change triggered an additional 52.6 Tg C. Although tropical dry forests are less carbon-dense than moist tropical forests, carbon emissions from land-use change in the Chaco were similar in magnitude to those from other major tropical deforestation frontiers. Our study thus highlights the urgent need for an improved monitoring of the often overlooked tropical dry forests and savannas, and more broadly speaking the value of the Landsat image archive for quantifying carbon fluxes from land change.

Keywords: carbon bookkeeping model, deforestation frontiers, Landsat image archive, land-use change, post-deforestation dynamics, savannas, tropical dry forests

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Introduction

The expansion of agriculture into tropical forests releases substantial amounts of carbon to the atmosphere (Baccini *et al*., 2012; Earles *et al*., 2012) and is one of the major contributors to climate change (Reay *et al*., 2012; Lawrence & Vandecar, 2015; Tubiello *et al*., 2015). Understanding where and at which rates tropical forests are converted to agriculture, and what the associated carbon emissions are, is therefore critical (Houghton, 2005). The forests of the humid tropics, especially in Amazonia and South-East Asia, have been in focus in this regard, because these forests are extremely carbon-rich and harbor astonishing biodiversity, and because agricultural expansion has triggered widespread deforestation in these forests recently (Geist & Lambin, 2002; Achard *et al*., 2014; Margono *et al*., 2014).

Deforestation, however, is also widespread in tropical dry forests and savannas. Much of the last undeveloped fertile lands are found in these ecosystems (Lambin *et al*., 2013), and as a result, rapid agricultural expansion has caused substantial forest loss and carbon emissions there (Miles *et al*., 2006; Lehmann, 2010; Parr *et al*., 2014). However, compared to the tropical humid forests, tropical dry forests and savannas remain under-researched (Blackie *et al*., 2014), and our understanding of deforestation dynamics and associated carbon emissions is partial (Lehmann, 2010; Parr *et al*., 2014). This knowledge gap is worrisome, as tropical dry forests and savannas cover about 20% of the global terrestrial surface, contribute 30% of the global primary productivity, sustain about 20% of the world's human population, and harbor high and unique biodiversity – suggesting much is at stake as these systems undergo rapid transformation.

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Tropical dry forests and savannas in South America have suffered disproportionally from deforestation recently (Hansen *et al.*, 2013), including the Cerrado (Klink & Machado, 2005), the Chiquitano (Müller *et al.*, 2012), and the Chaco forests (Gasparri & Grau, 2009). In addition to deforestation, agricultural intensification is widespread, as traditional grazing systems are increasingly replaced by intensified ranching, which itself often makes way to industrialized cropping, mainly soybean and maize (le Polain de Waroux *et al.*, 2016). These intensification trends have major ecological effects. For example, bird densities are almost twice as high on grazing lands compared to soybean fields after deforestation (Macchi *et al.*, 2013), soil erosion is much higher on intensified croplands compared to grazing lands (Abril *et al.*, 2005), and cropland is at higher risk of salinization compared to grazing lands (Silburn *et al.*, 2007). In terms of carbon, total organic soil carbon is substantially higher in grazing lands compared to croplands (Fujisaki *et al.*, 2015), and grazing lands also retain more aboveground biomass (Tonucci *et al.*, 2011). Thus, understanding the rates and spatial patterns of post-deforestation land-use dynamics is critical for understanding carbon budgets, yet consistent maps on the extent of grazing lands and cropland, and conversions among them, are scarce for many dry forest and savanna regions.

Remote sensing is a powerful tool to assess land-use dynamics, including in South America's dry forests, with most existing studies focusing on small study regions though (Grau *et al.*, 2005; Müller *et al.*, 2016). The few studies that assessed larger areas have predominantly mapped deforestation only, including at the global scale (Hansen *et al.*, 2013) or for the Brazilian Cerrado (Garcia & Ballester, 2016), or Paraguay's dry forests (Caldas *et al.*, 2015). We know of only three studies that have assessed deforestation and post-deforestation land uses in South America, for the Cerrado (Grecchi *et al.*, 2014), and all of South America (Graesser *et al.*, 2015; Sy *et al.*, 2015). Land-use change assessments are particularly scarce for the Gran Chaco, a 1 100 000-km² ecoregion extending into Argentina, Paraguay, and Bolivia which has been particularly heavily affected by deforestation (Hansen *et al.*, 2013) and grazing-lands-to-cropland conversions (Gasparri *et al.*, 2013, 2015). As elsewhere in South America, land-use change assessments for the region have predominantly focused on deforestation only (Gasparri & Grau, 2009; Vallejos *et al.*, 2015) or, when assessing post-deforestation dynamics, done so for smaller regions, short time periods, or at coarse scale only (Clark *et al.*, 2010; Caldas *et al.*, 2015). This means that important land-use change processes are potentially missed. The opening of the Landsat archives (Woodcock *et al.*, 2008), along with algorithmic advances and increasing computational power (Gómez *et al.*, 2016), provides new opportunities to reconstruct land-use histories back to 1984 at fine spatial resolution and across large geographic extents (Wulder *et al.*, 2012; Griffiths *et al.*, 2014; Potapov *et al.*, 2015), and thus to analyze deforestation and post-deforestation dynamics. These opportunities, however, have not been fully capitalized on for the Chaco.

The paucity of consistent land-use/cover change maps for the Chaco over longer time periods has also translated into major knowledge gaps regarding carbon emission associated with land-use change. While some maps of carbon stocks cover the Chaco, they were modeled using no or only little field data from inside the Chaco and outdated land-use/cover maps for the Chaco, translating into large uncertainties of close to 50% in estimating aboveground biomass (Saatchi *et al.*, 2011; Baccini *et al.*, 2012). To our knowledge, there is only one study quantifying carbon emissions from deforestation for an area in the Chaco (Gasparri *et al.*, 2008). While this assessment was based on extensive field data on carbon stocks, and highlighted the substantial emission that deforestation can trigger, it also covered only a small region in northern Argentina and did not consider post-deforestation land-use dynamics. What is missing is a coherent estimation of carbon fluxes due to deforestation and post-deforestation land-use change across the Chaco.

Our goal was therefore to map deforestation and post-deforestation dynamics for the entire South American Chaco for the period 1985–2013, and to assess the carbon fluxes associated with these land-use changes. Specifically, we addressed three research questions:

1 What were the rates and spatial patterns of deforestation in the Chaco since 1985, and how did deforestation patterns differ between 1985–2000 and 2000–2013?
2 Were forests primarily cleared for grazing lands or agriculture, and what were the post-deforestation land-use changes in the region?
3 What were the carbon emissions associated with deforestation and post-deforestation land-use changes?

Materials and methods

Study region

The Chaco is a large tropical dry forest region, climatically characterized by a distinct dry season between May and September, followed by a wet season from November to April. Mean annual temperature is about 22 °C, with an average monthly maximum of 28 °C. Annual precipitation ranges from 1200 mm in the east (wet Chaco, Chaco humedo) to 450 mm in the west (dry Chaco, Chaco seco; Bucher, 1982;

Volante *et al.*, 2012). Terrain is mainly flat, except for the west and southwest of the Chaco (Fig. S1).

The rainfall gradient, edaphic conditions, and fire have resulted in a varied mosaic of vegetation formations in the Chaco, mainly consisting of xerophilous and subxerophilous forests, with interspersed gallery forests, savannas, and grasslands (Cabrera, 1976; Bucher, 1982). The most characteristic trees of the Chaco are quebrachos (*Schinopsis balansae*, *S. lorentzii*, and *Aspidosperma quebracho*), which co-occur in the wet Chaco with urunday (*Astronium balansae*) and palo lanza (*Phyllostylon rhamnoides*), and in the dry Chaco with algarrobos (Prosopis spp.), palo santo (*Bulnesia sarmientoi*), and itín (*Prosopis kuntzei*). The shrub layer is dominated by Acacia, Mimosa, Prosopis, and Celtis species. Cacti, grasses, and bromeliads are abundant in the understory (Prado, 1993). The southern dry Chaco has less rainfall than the rest of the dry Chaco (hereafter: very dry Chaco), resulting in less trees and a more dominant shrub layer that includes *Mimozyganthus carinatus* (Conti *et al.*, 2014).

Historically, land use in the Chaco consisted of subsistence agriculture, with small-scale cropping and silvocultural grazing where livestock roams freely in the forest around watering points, often resulting in local overgrazing. In addition, wood extraction and charcoal production put further pressure on the Chaco's natural forests (Fatecha, 1989; Bucher & Huszar, 1999). Since the 1980s, land use has changed in all Chaco an countries in major ways. In Argentina, much of the area is suitable for soybean, maize, and wheat, and these crops have expanded rapidly in recent decades, especially after the introduction of genetically modified soybean variants due to rising global crop prices during the 2000s (Reenberg & Fenger, 2011; Richards et al., 2012). In the Paraguayan Chaco, widespread deforestation for cattle ranching occurred since the early 1990s (Ramirez & Laneri, 1989; Caldas *et al.*, 2015), as a consequence of rising global demands for beef, and optimized production techniques, primarily in the area around Filadelfia (Vidal, 2010; Vazquez, 2013). In Bolivia, the Chaco is largely uninhabited, except for the Andean foothills in the west (Müller *et al.*, 2012) where rainfall permits cattle ranching and cropping (Killeen *et al.*, 2007, 2008).

Mapping land-use change between 1985 and 2013

We mapped land-use/cover change for the entire Chaco using Landsat TM and ETM+ image composites (Roy *et al.*, 2010; Potapov *et al.*, 2011). Image composites are mosaic-type surface reflectance images that make use of large numbers of Landsat images and that are radiometrically corrected, cloud- and gap-free, and thus consistent across large areas. Composites can be generated for key phenological time windows that help separating land-use/cover classes reliably (Griffiths *et al.*, 2013, 2014). Moreover, composites can contain a range of spectral metrics (e.g., the bandwise mean, min, max reflectance of all cloud-free observations) and meta-information (e.g., the number of cloud-free observations, zenith and azimuth), which further improves land-use/cover mapping (Griffiths *et al.*, 2013).

We generated composites for three target years: 1985, 2000, and 2013. We considered imagery from ±1 year around these target years, except for 1985 where we considered images from ±2 years, as data availability was lower (Table 1). All Landsat images were already terrain-corrected (Wulder *et al.*, 2012), and we converted them to surface reflectance using the Landsat Ecosystem Disturbance Adaptive Processing System (LEDAPS; Masek *et al.*, 2006) and masked clouds and cloud shadows using Fmask (Zhu *et al.*, 2012). For each target year, we calculated (1) a best pixel composite (BPC) centered on the day of year 258 (i.e., mid-September), (2) ten image metrics per spectral band, and (3) nine meta-information layers on the images used in the compositing. We combined all these layers into one multitemporal image composite stack.

To map changes in land use/cover, we classified these image composite stacks into classes of 'stable forest', 'stable cropland', 'stable grazing lands', and 'other' as well as change classes describing transitions (Table 2). The forest class included all woody vegetation that forms a closed canopy cover of more than 50%; croplands were in our case predominantly intensified soybean, maize, and cotton fields; and grazing included natural grasslands (almost all of which are grazed), implanted pastures, and silvopastures. The 'other' class contained all nonvegetation land covers such as water, urban areas, and salt planes, as well as wetlands, natural grasslands along rivers that are temporarily flooded, and palm savannas.

We digitized training data for our classification by interpreting high-resolution imagery in Google Earth (~85% of our study area is covered by high-resolution imagery) and the Landsat imagery itself, complemented by field data. Reliable on-screen digitizing was possible because our land-cover classes are spectrally and structurally very distinct, and our team of collaborators has more than 30 years of field experience in the region. We identified forest training data through visual inspection of high-resolution imagery, and croplands via their shape and structure (e.g., plowlines) in high-resolution imagery and their spectral signature in Landsat composites. Grazing lands were identified due to their distinct spectral signature as well as trees or shrubs clearly visible in high-resolution imagery. To capture spectral variability, we digitized entire plots [e.g., an entire field, the entire deforestation plot (Baumann *et al.*, 2012)]. Once training polygons were collected, we randomly selected 3000 points per class as input for a random forest classifier (Waske *et al.*, 2012), which can handle spectrally complex classes, often outperforms other classifiers, and is computationally efficient (Breiman, 2001). We applied a minimum mapping unit of 3 pixels (~0.25 ha) and summarized land-use/cover change for the entire Chaco, for the dry and wet Chaco separately, and for each of the three countries individually.

We assessed the accuracy of our map based on a stratified random sample of 100 points per class, collected fully

Table 1 Number of Landsat imagery used for each year in our analysis

Target year	Minimum year	Maximum year	# Images
1985	1984	1987	3794
2000	1999	2001	7673
2013	2012	2014	4157

Table 2 Class catalog of stable classes and transition classes mapped for our study region

	Class name	Class (short)	Land use/land cover in period		
			1985	2000	2013
Stable classes	Stable Forest	FFF	Forest	Forest	Forest
	Stable Cropland	CCC	Cropland	Cropland	Cropland
	Stable Grazing land	GGG	Grazing land	Grazing land	Grazing land
	Other	OOO	Other	Other	Other
Deforestation classes	Forest to Cropland 1985–2000	FCC	Forest	Cropland	Cropland
	Forest to Grazing land 1985–2000	FGG	Forest	Grazing land	Grazing land
	Forest to Cropland 2000–2013	FFC	Forest	Forest	Cropland
	Forest to Grazing land 2000–2013	FFG	Forest	Forest	Grazing land
Intensification classes	Grazing land to Cropland 1985–2000	GCC	Grazing land	Cropland	Cropland
	Grazing land to Cropland 2000–2013	GGC	Grazing land	Grazing land	Cropland
Deforestation/Intensification classes	Forest to Grazing land to Cropland 1985–2013	FGC	Forest	Grazing land	Cropland
Other classes	Water, urban areas, salt planes, wetlands, natural grasslands, palm savannas	O	Other	Other	Other

independently from the training data. For each point, we visually inspected the Landsat imagery and high-resolution imagery in Google Earth and assigned the respective class label. We then generated a confusion matrix (Table S1), from which we calculated overall classification accuracy, and classwise user's and producer's accuracies. We also corrected our class area estimates and error estimates for potential sampling bias and calculated confidence intervals around them (Olofsson *et al.*, 2014).

Estimating carbon emissions

To estimate carbon emissions from land-use/cover changes in the Chaco, we applied a carbon bookkeeping modeling framework (CBKM; Houghton, 1995; Houghton & Hackler, 2001). CBKMs are widely used for this purpose (Kuemmerle *et al.*, 2011; Olofsson *et al.*, 2011; Houghton *et al.*, 2012; Carlson *et al.*, 2013), and for our study had several advantages over more mechanistic tools such as dynamic vegetation models. First, CBKMs are less data-hungry and do not require spatially explicit parameter estimates. As such, CBKMs are particularly well suited in data-sparse regions, such as the Chaco. Second, we were able to build upon a large sample of plot-level inventory data of aboveground biomass (AGB) in forests (Gasparri *et al.*, 2008), collected to estimate parameters for a regionalized CBKM version. Third, while mechanistic models such as dynamic vegetation models are powerful to assess ecosystem response to environmental change, bookkeeping models can assess and isolate the effects of land-use change on carbon fluxes (Houghton, 1995; Houghton *et al.*, 2012).

We used the CBKM to estimate the carbon fluxes associated with three land-use/cover conversions: (1) forests to croplands (F-C), (2) forests to grazing lands (F-G), and (3) grazing lands to croplands (G-C). We assumed that land-use/cover changes occurred gradually between our target years, and applied a linear interpolation of land-use/cover conversion rates between two years to estimate two emissions: (1) emissions related to AGB changes, and (2) soil organic carbon (SOC) emissions.

Aboveground biomass values for our three land-use/cover types came from field-inventory plots and existing literature. For the AGB of intact forests, we integrated five carbon stock components (Table 3): (1) carbon in the AGB of trees with diameter at breast height (dbh) >10 cm [Mg ha^{-1}], (2) carbon in the AGB of trees with dbh <10 cm [Mg ha^{-1}], (3) carbon in belowground biomass, (4) carbon in biomass in deadwood [Mg ha^{-1}], and (5) carbon in litter [Mg ha^{-1}]. These parameters were based on 96 field plots from the wet and the dry Chaco. For the dry Chaco, we had 55 sample plots of 0.8 ha size from our own previous work, arranged in a grid of 50×50 km across an area of about 200 000 km^2 (for more details on the survey methods, see Gasparri *et al.* (2008)). For the wet Chaco, we surveyed 41 plots using the same sampling methodology. AGB values for the very dry Chaco (i.e., the south of the dry Chaco) were taken from a recent study that contained plot-based biomass surveys for that region (Conti *et al.*, 2014). We assumed that 50% of the AGB in the Chaco is carbon (Gasparri *et al.*, 2008), and calculated the total carbon in forest biomass as the sum of components (1)–(4), divided by two, plus (5).

Aboveground biomass values for croplands came from the Intergovernmental Panel on Climate Change Agriculture, Forestry, and Other Land Use (IPCC-AFOLU) manual for the region (IPCC, 2006), and we assumed 5 Mg C ha^{-1} in all croplands across our study area (i.e., very dry, dry, or wet Chaco; Table 3).

Estimating AGB for grazing lands was more complex, as two main types of grazing lands are prevalent in the Chaco, which differ in terms of their carbon stocks: (1) silvopastures, which have a substantial amount of trees left at the site, and (2) grazing lands without trees remaining. As our satellite-based land-use/cover map did not differentiate between these two types of grazing lands, we used two alternative scenarios for the AGB of grazing lands. First, we

Table 3 Aboveground biomass and carbon stored in biomass for the three land-use/cover types in our analysis. These parameters provided the input for the CBKM to estimate carbon emissions from land-use change

Class	Variable	Region	Value	Source
Forest	Aboveground biomass in trees dbh>10 cm (AGB_{10+}) [Mg C ha^{-1}]	Very dry Chaco	37.5	Conti *et al.* (2014)
		Dry Chaco	78.0	Gasparri *et al.* (2008)
		Wet Chaco	130.6	Based on 41 sample plots of 1000 m^2 in the Wet Chaco of Argentina using the same method as in Gasparri *et al.* (2008)
	Aboveground biomass in trees dbh<10 cm (AGB_{10-}) [Mg C ha^{-1}]	Very dry Chaco	1.5	Conti *et al.* (2014)
		Dry Chaco	2.3	Gasparri *et al.* (2008)
		Wet Chaco	3.9	Based on 41 sample plots of 1000 m^2 in the Wet Chaco of Argentina using the same method as in Gasparri *et al.* (2008)
	Belowground biomass (BGB) [Mg C ha^{-1}]	Very dry Chaco	17.0	Calculated as 32.2% of AGB biomass for very dry Chaco and dry Chaco, and as 42% in wet Chaco (Mokany *et al.*, 2006)
		Dry Chaco	25.9	
		Wet Chaco	54.9	
	Biomass in dead wood (BDW) [Mg C ha^{-1}]	Very dry Chaco	7.4	Conti *et al.* (2014)
		Dry Chaco	10.9	Calculated as 14% of the AGB (IPCC, 2003)
		Wet Chaco	14.4	Calculated as 14% of the AGB (IPCC, 2003)
	Carbon in litter (CL) [Mg C ha^{-1}]	Very dry Chaco	3.9	Conti *et al.* (2014)
		Dry Chaco	2.3	Abril *et al.* (2005)
		Wet Chaco	2.3	Abril *et al.* (2005)
	Total carbon in Biomass [Mg C ha^{-1}]	Very dry Chaco	42.6	Calculated as: ((AGB_{10+} + AGB_{10-} +BGB + BDW)/2) + CL
		Dry Chaco	60.8	
		Wet Chaco	104.2	
Grazing lands	Total carbon in Biomass [Mg C ha^{-1}]	Very dry Chaco	6.1	Parameterization I: IPCC (2006) for cold temperate dry regions
			28	Parameterization II: Houghton & Hackler (2001)
		Dry Chaco	6.5	Parameterization I: IPCC (2006) for warm temperate dry regions
			28	Parameterization II: Houghton & Hackler (2001)
		Wet Chaco	13.5	Parameterization I: IPCC (2006) for warm temperate wet regions
			28	Parameterization II: Houghton & Hackler (2001)
Cropland	Total carbon in Biomass [Mg C ha^{-1}]	Very dry Chaco	5	IPCC (2006) for cold temperate dry regions
		Dry Chaco	5	IPCC (2006) for warm temperate dry regions
		Wet Chaco	5	IPCC (2006) for warm temperate wet regions

used estimates from the IPCC-AFOLU manual (IPCC, 2006), assuming an AGB in grazing lands of 6.1, 6.5, and 13.5 Mg C ha^{-1} for the very dry Chaco, dry Chaco, and wet Chaco, respectively. Second, we used the Houghton & Hackler (2001) estimate, assuming an AGB of 28 Mg C ha^{-1} for grazing lands. Thus, the IPCC estimate represents a scenario of grazing lands with few trees, which is the case for most grazing lands in the Chaco, while the higher Houghton & Hackler (2001) estimate accounts for more trees and thus higher AGB on grazing lands.

Soil organic carbon in our context referred to humus, excluding litter and mineral carbon, and we generally assumed higher (but constant) SOC emissions during the first 5 years after conversion, and lower (and constant) emissions for the next 15 years (Houghton, 1999). Regarding F-C conversions, we assumed constant emissions of 2.80 Mg C ha^{-1} during the first five years after conversion, and constant emissions of 0.87 Mg C ha^{-1} for the next 15 years (Houghton, 1999). Regarding SOC emissions from F-G conversions, we assumed two scenarios. On the one hand, we assumed no

SOC emissions during F-G conversions, in accordance with Houghton & Hackler (2001). However, F-G conversions in the Chaco are increasingly being carried out by first removing all natural vegetation and litter with tractors and caterpillars, not different from preparing a forest site for croplands, followed by planting exotic, perennial grasses. Assuming no SOC release may therefore underestimate emissions. We therefore used an alternative scenario where SOC emissions during the first 2 years were assumed to be equal to a conversion from forest to cropland, until perennial grasses have established and SOC release stops. Together with our two AGB scenarios for grazing land, this resulted in four alternative estimations for the F-G conversion: (1) high AGB/no SOC emissions, (2) low AGB/no SOC emissions, (3) high AGB/2-year SOC emissions, and (4) low AGB/2-year SOC emissions.

A second set of parameters required by the CBKM to estimate carbon fluxes following deforestation events (i.e., in our case F-C and F-G conversions) concerns the turnover rates for different carbon components after a land-use transition has happened. We used different turnover rates before and after 1990 to account for technological change that has happened in the Chaco (Table 4; Gasparri *et al.*, 2008). Today, heavy machinery is used for deforestation and soil preparation (i.e., heavy tractors or caterpillars to remove most of the aboveground biomass and litter, including large roots). Removed woody biomass is most commonly burned on-site and thus released quickly (Gasparri *et al.*, 2008). However, until 1990 this was uncommon. Most of the wood was used for forest products (e.g., fire wood), and thus, less biomass was burned on-site (Table 4). To account for this, we applied the carbon fractions and turnover rates from Houghton *et al.* (1991) before 1990, and the rates from Gasparri *et al.* (2008) thereafter, which in contrast to the prior rates assume a higher carbon release right after conversion (i.e., 0.55 from Gasparri *et al.* (2008) compared to 0.30 from Houghton *et al.* (1991)), a lower carbon fraction left on-site as debris and slash (0.35 compared to 0.38), and a lower share of carbon removed from the site as wood products (0.08 vs. 0.30; Table 4). This corresponds well with turnover rates suggested for Mato Grosso (Brazil), with similar land-use and deforestation practices (Morton *et al.*, 2006).

Table 4 Carbon fractions and turnover rates in the CBKM parameterization

#	Parameter	Before 1990	After 1990
1	Fraction of C returning to soils as mineral carbon after burning	0.02	0.02
2	Fraction of C removed from site as products	0.30	0.08
3	Fraction of C burnt on-site	0.30	0.55
4	Fraction of C remaining on-site after clearing	0.38	0.35
5	Constant of decomposition for elemental carbon	0.001	0.001
6	Constant of turnover for carbon in products removed from site	0.1	0.1
7	Decomposition for carbon left on-site as slash and debris	0.30	0.30

The extent of land-use/cover change for 1985–2013 came from our Landsat-based change analyses. To assess the impact of remaining uncertainty in these maps on the carbon flux estimations, we calculated an upper and a lower limit around the estimated areas corresponding to two standard errors of the producer's accuracy of our classes (Olofsson *et al.*, 2013). We thus performed three CBKM runs for each scenario: one run using the area estimates from the classification, and one run each using the upper and lower confidence interval boundaries of these estimates (Olofsson *et al.*, 2011).

We also explored carbon emissions for a number of hypothetical 'what/if' land-use change scenarios until 2050: (1) conversion of all grazing lands into croplands without additional deforestation (intensification without deforestation), (2) conversion of all grazing lands in 2013 into croplands, and deforestation for grazing lands at 2000–2013 rates (intensification with deforestation 2000–2013), (3) land-use changes as in our first period 1985–2000 (extrapolation 1985–2000), (4) continuation of land conversions at rates from 2000 to 2013 (extrapolation 2000–2013), and (5) continuation of land conversions at rates from 1985 to 2013 (extrapolation 1985–2013). For comparison, we also summarized post-2013 emissions when assuming no land-use change after 2013, thus capturing the emission legacy of pre-2013 land-use change.

Results

Land-use change in the Chaco, 1985–2013

Our Landsat-based land-use/cover change maps highlight widespread and rapid forest loss across the Chaco (Fig. 1). By 2013, 18.4% of all forests in 1985 were lost (142 000 km^2 in total, 5070 km^2 annually between 1985 and 2013, equaling an annual deforestation rate of 0.66%). Forest loss accelerated markedly in the second period (i.e., 2000–2013), when we found up to 7232 $km^2\ yr^{-1}$ of forest loss (deforestation rate: 1.00% yr^{-1}), compared to 3190 $km^2\ yr^{-1}$ between 1985 and 2000 (0.41% yr^{-1}). Relative forest loss was higher in the wet Chaco (19.8% of the total forest in 1985 for the entire period, 7.0% 1985–2000, 13.7% 2000–2013), compared to the dry Chaco (18.2%, 6.1%, and 12.9%, respectively; Fig. 2 (a)), and so where deforestation rates (0.7% yr^{-1} for the entire time period, 0.47% yr^{-1} for 1985–2000, and 1.05% yr^{-1} for 2000–2013 for the wet Chaco; 0.65% yr^{-1}, 0.4% yr^{-1}, and 0.99% yr^{-1}, respectively, for the dry Chaco).

Comparing among countries showed that forest loss was highest in Paraguay at 22.4% (0.8% yr^{-1}) of the forest cover of 1985 (23.9% in the dry vs. 18.4% in the wet Paraguayan Chaco). This was closely followed by Argentina (all of the Argentine Chaco: 20.0% (0.72% yr^{-1}), dry Chaco: 19.9%, wet Chaco: 21.1%; Fig. 2a). The Bolivian Chaco experienced much lower rates of forest loss (dry Chaco: 4.9% (0.17% yr^{-1}); there is no wet Chaco in Bolivia).

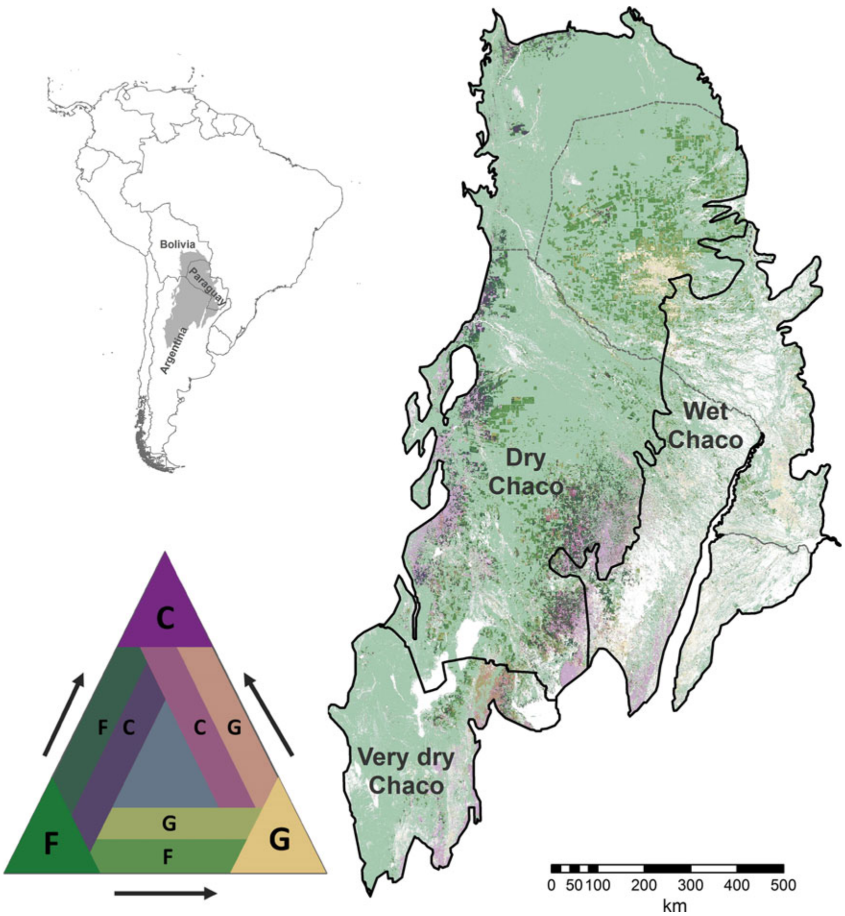

Fig. 1 Large map – deforestation and agricultural expansion in the South American Chaco for the period 1985–2013. The class colors are chosen such that the corners represent stable classes stable forest (F), stable grazing lands (G), and stable croplands (C), whereas colors along the sides of the triangle represent transitions. Lighter colors present transitions between 1985 and 2000, whereas darker colors represent transitions between 2000 and 2013 (e.g., light green represents transition from forests to grazing lands 1985–2000; darker green represents transitions from forests to grazing lands 2000–2013). Gray indicates transitions from forest to grazing lands to cropland (1985–2000–2013), and white indicates other classes or change trajectories (see Supporting Information). Small map – location of the study region in South America. [Colour figure can be viewed at wileyonlinelibrary.com].

Grazing lands were the most common land use following deforestation (82.0%; Fig. 2c, d). Yet, this differed when comparing between the dry and wet Chaco, with F-C conversions being more prevalent in the dry Chaco (19.6% of all deforestation for cropland) compared to the wet Chaco (only 10.5% of deforestation for cropland). We also found important variation across countries, with F-C conversions being more prevalent in Argentina (27.6%) and Bolivia (41.7%), whereas such conversions were insignificant in Paraguay (0.8%). G-C conversions were also widespread across the Chaco (29 680 km^2 total, of which 86% occurred between 2000 and 2013), but much more so in the dry Chaco (26 421 km^2, 86.6% of which occurred between 2000 and 2013) than in the wet Chaco (3253 km^2, 79.8% between 2000 and 2013). Similarly, G-C conversions were more common in Argentina (28 265 km^2) than in Bolivia (465 km^2) and Paraguay (944 km^2; Fig. 2d).

Our accuracy assessment showed that our change map was reliable (overall accuracy = 89%), with general high user's and producer's accuracies. The stable classes had the highest user's accuracies (stable forest (F-F-F) = 92%, stable cropland (C-C-C) = 83%, stable grazing lands (G-G-G) = 79%), followed by the conversion classes from forest to cropland (i.e., user's accuracies of 75% and 88.0% for F-F-C and F-C-C, respectively) and to grazing land (i.e., 86% and 80% for the F-F-G and F-G-G conversions, respectively) (Table S1). The G-G-C conversion class had a lower user's accuracy (59%) and was fairly frequently confused with the permanent grazing land (G-G-G) class. The 'other' (O) class, entailing land covers not in focus here (e.g., urban, water, bare areas), had a user's accuracy of 89% (Table S1). Producer's accuracies were generally

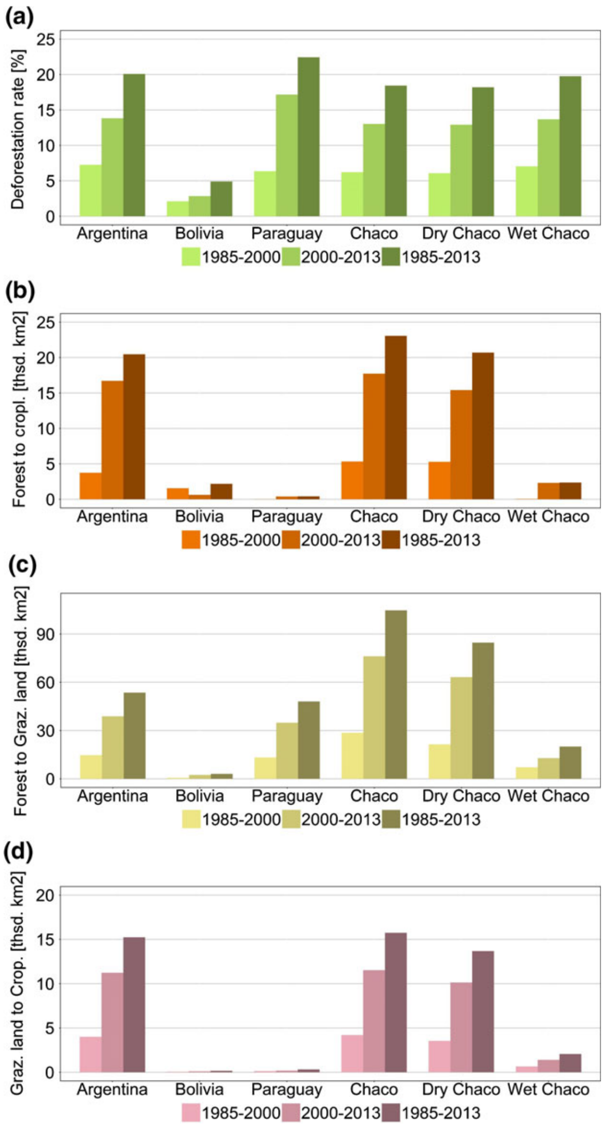

Fig. 2 Overview over the transition classes in our map, divided by country and ecoregion. (a): deforestation rate [%]; (b): overall area [km^2] of all forest-to-cropland conversions; (c): overall area [km2] of all forest-to-grazing-land conversions; (d): overall area [km2] of all grazing-land-to-cropland conversion. The column groups represent the subdivision of the study area into *dry (incl. very dry)* and *wet* Chaco, as well as by the Chacoan countries (i.e., Argentina, Bolivia, Paraguay). [Colour figure can be viewed at wileyonlinelibrary.com].

higher than the user's accuracies, with the conversion classes showing very high accuracies (e.g., F-F-C of 93%, G-C-C of 94%; Table S1). The resulting confidence intervals around the area estimates were narrow (Fig. 3).

Carbon emissions from land-use change in the Chaco

Land-use changes in the Chaco between 1985 and 2013 resulted in major carbon emissions of up to 824 Tg C

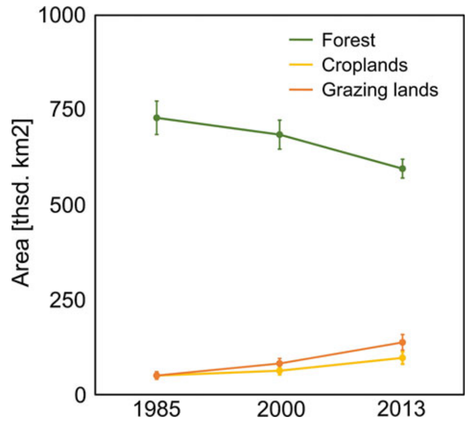

Fig. 3 Area estimates for the three land-use/cover classes at each of the three time points of the analysis. The lines represent error bars, resulting from the uncertainty entailed in our classification (after correcting for possible sampling bias). [Colour figure can be viewed at wileyonlinelibrary.com].

(29.4 Tg C yr^{-1}; Table 5) across the entire Chaco for the period 1985–2013 for our scenario of low grazing lands AGB and no SOC emissions after F-G conversion. About 13% of these emissions came from the soil (109 Tg). Of the total emissions, 208.1 Tg (23%) came from F-C conversions, 563.2 Tg (71%) came from F-G conversions, and 52.6 Tg (6%) from P-C conversions. For the year 2013 (i.e., the last year of our study) alone we estimated carbon emissions of 46.3 Tg C yr^{-1} (10.1 Tg from F-C conversions, 32.7 Tg from F-G conversions, 3.5 Tg from P-C conversion, respectively; Table 5 and Fig. 4).

Comparing among our subregions (i.e., *very dry Chaco*, *dry Chaco*, and *wet Chaco*), showed highest emissions for the dry Chaco at 577 Tg C for the 1985–2013 (159.9 Tg from F-C conversions, 383.3 Tg C from F-G conversions, 33.8 Tg from G-C conversions), followed by the wet Chaco with 211.6 Tg C for the entire period (39.5 Tg, 161.3 Tg, 10.8 Tg, respectively), and the very dry Chaco with 35.3 Tg C (8.6 Tg, 18.6 Tg, and 8.1 Tg, respectively). Comparing among time periods revealed that carbon emissions strongly accelerated after 2000. While 20.2 Tg C yr^{-1} were emitted between 1985 and 2000, we estimated more than twice these emissions during 2000–2013 (40.1 Tg C yr^{-1}).

When comparing among countries, Argentina was the main emitter of carbon from land-use change in our study period (56.6% of all emissions, equaling 466.0 Tg), followed by Paraguay (30.1%, 248.4 Tg) and Bolivia (13.3%, 109.6 Tg), and in all three countries emissions in 2000–2013 (Argentina 295.6 Tg, Paraguay 156.5 Tg, Bolivia .69.3 Tg) were higher than in 1985–2000 (Argentina 170.4 Tg, Paraguay 91.8 Tg, Bolivia 40.3 Tg). In Argentina, 25.2% of all emissions from land-use change came from F-C conversions, 67.9% from F-G conversions and

Table 5 Carbon emissions from land-use change under the four scenarios: (1) high AGB in grazing lands, no SOC emissions during F-G conversions; (2) low AGB in grazing lands, no SOC emissions during F-G conversions; (3) high AGB in grazing lands, with SOC emissions during F-G conversions; and (4) low AGB in grazing lands, with SOC emissions during F-G conversions. All numbers represent Tg C per year, except the relative emissions which are % values from all emissions

		(1) High AGB, no SOC emissions				(2) Low AGB, no SOC emissions				(3) High AGB, with SOC emissions				(4) Low AGB, with SOC emissions			
		All	F-C	F-G	G-C	All	F-C	F-G	G-C	All	F-C	F-G	G-C	All	F-C	F-G	G-C
All emissions	1985–2000	245.5	89.5	125.9	29.9	302.5	89.5	197.7	15.2	266.3	89.6	146.8	30.0	323.4	89.6	218.5	15.3
	2001–2013	400.4	118.5	206.2	75.6	521.4	118.5	365.4	37.3	443.9	118.6	249.7	75.6	564.8	118.6	408.9	37.4
	1985–2013	646.0	208.1	332.2	105.5	824.0	208.1	563.2	52.6	710.2	208.1	396.5	105.6	888.2	208.1	627.4	52.6
From Soil Tg C	1985–2000	41.4	27.3	0.0	14.2	41.4	27.3	0.0	14.2	62.2	27.3	20.8	14.2	62.2	27.3	20.8	14.2
(%of all)		(17%)	(30%)	(0%)	(47%)	(14%)	(30%)	(0%)	(93%)	(23%)	(30%)	(14%)	(47%)	(19%)	(30%)	(10%)	(93%)
	2001–2013	68.1	34.7	0.0	33.4	68.1	34.7	0.0	33.4	111.5	34.7	43.4	33.4	111.5	34.7	43.4	33.4
		(17%)	(29%)	(0%)	(44%)	(13%)	(29%)	(0%)	(89%)	(25%)	(29%)	(17%)	(44%)	(20%)	(29%)	(11%)	(89%)
	1985–2013	109.5	62.0	0.0	47.5	109.5	62.0	0.0	47.5	173.7	62.0	64.2	47.5	173.7	62.0	64.2	47.5
		(17%)	(30%)	(0%)	(45%)	(13%)	(30%)	(0%)	(90%)	(24%)	(30%)	(16%)	(45%)	(20%)	(30%)	(10%)	(90%)
Mean	1985–2000	16.3	5.9	8.4	2.0	20.1	5.9	13.1	1.0	17.8	6.0	9.8	2.0	21.6	6.0	14.6	1.0
Emissions	2001–2013	30.8	9.1	15.8	5.8	40.1	9.1	28.1	2.8	34.1	9.1	19.2	5.8	43.4	9.1	31.5	2.9
	1985–2013	23.0	7.4	11.8	3.7	29.4	7.4	20.1	1.8	25.4	7.4	14.2	3.8	31.7	7.4	22.4	1.9
	2013 only	36.9	10.1	20.4	6.4	46.2	10.1	32.6	3.4	40.3	10.1	23.8	6.4	49.6	10.1	36.0	3.5
	2010 only	35.3	9.8	19.3	6.1	44.6	9.8	31.6	3.2	38.7	9.8	22.7	6.2	48.0	9.8	34.9	3.2
Relative	1985–2000		0.36	0.51	0.12		0.29	0.65	0.05		0.34	0.55	0.11		0.27	0.68	0.05
Emissions	2001–2013		0.30	0.52	0.19		0.22	0.70	0.07		0.27	0.56	0.17		0.21	0.72	0.07
	1985–2013		0.32	0.51	0.16		0.25	0.68	0.06		0.29	0.56	0.15		0.23	0.71	0.06
	2013 only		0.27	0.55	0.17		0.21	0.70	0.07		0.25	0.59	0.16		0.20	0.73	0.07
	2010 only		0.28	0.55	0.17		0.21	0.70	0.07		0.25	0.59	0.16		0.20	0.73	0.07

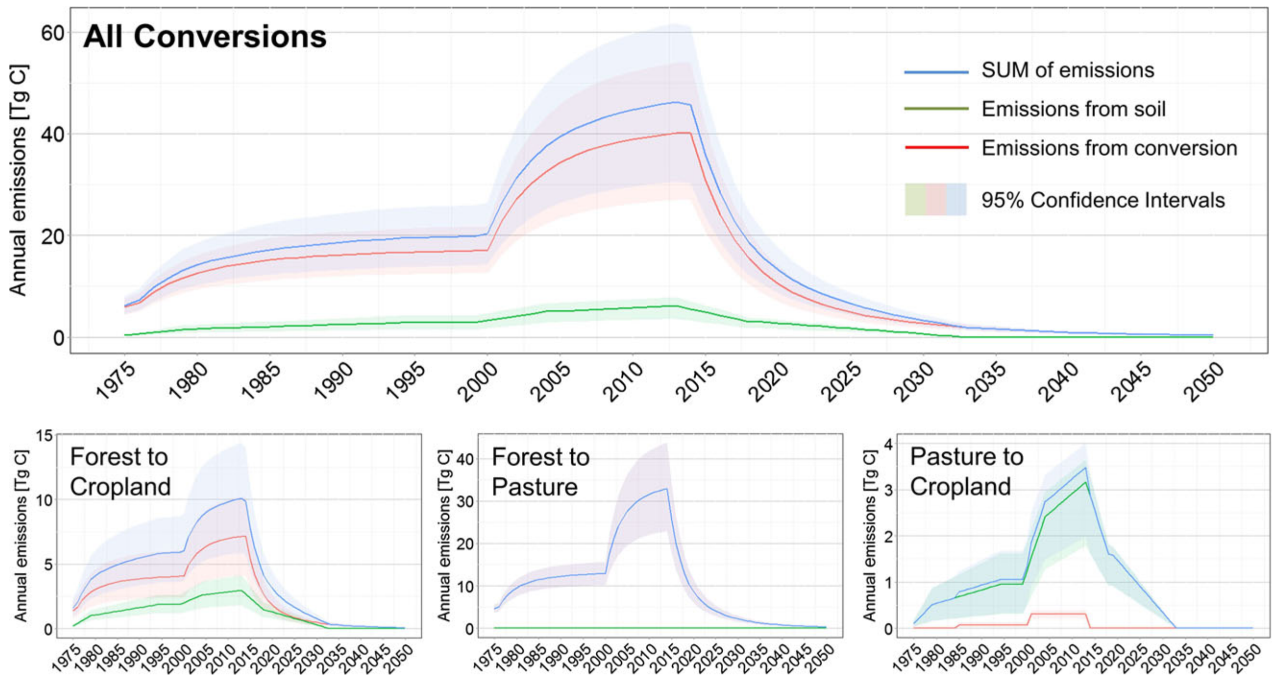

Fig. 4 Carbon emissions from land-use change in the Chaco, estimated using a carbon bookkeeping model (total carbon emissions, carbon emissions from aboveground biomass change only, emissions from changes in soil organic carbon). Emissions between 2013 and 2050 represent the legacy from land-use changes between 1985 and 2013 (i.e., continuing emissions even if land-use change would stop immediately). The top graph represents the sum of all three conversion types, and the bottom graphs represent emissions associated with each of the three land-cover conversions we mapped (please note the differences in the scales of the y-axes). The graph represents the assumption of low AGB in grazing lands and no SOC emissions during forest-to-grazing-land conversions. For the results of the other assumptions, please refer to the Supporting Information. [Colour figure can be viewed at wileyonlinelibrary.com].

6.9% from G-C conversions, and these proportions were only slightly different for Paraguay (24.3%, 70.1%, 5.6% for F-C conversions, F-G conversions and G-C conversions, respectively) and Bolivia (27.7%, 66.4%, 5.9%).

Our sensitivity analyses regarding grazing lands AGB and SOC emissions following F-C conversions showed a moderate influence of these parameters on overall carbon emissions (Table 5). The lowest emissions occurred when assuming high AGB in grazing lands and no SOC emissions (646 Tg C overall, 23.1 Tg C yr^{-1}), whereas the highest emissions resulted under the assumption of low AGB in grazing lands and SOC emissions from F-G conversions (888.2 Tg C overall, 31.7 Tg C yr^{-1}). Thus, the emissions under the conditions we considered most plausible based on 30 years of field experience in the region, that is, low AGB in grazing lands and without SOC emissions, fall between these extremes and are ~10% lower than our highest emissions estimate and ~25% higher than the lowest emission estimates (Table 5). Exploring how the uncertainties in our remote sensing analyses affected carbon emissions estimates showed that uncertainty due to possible error propagation was overall low, amounting to ± 12.5% in carbon emission (Fig. 4).

Our what/if scenarios resulted in very different emission trajectories until 2050 (Fig. 5). If all land-use changes stopped after 2013, previous land-use changes would still result in legacy emissions of 268 Tg. If deforestation halted and all grazing lands were converted into croplands (scenario I), the resulting emissions would add up to 495.4 Tg. Assuming continued deforestation at the rates of 2000–2013 and a conversion of all existing grazing lands into croplands (scenario II) would trigger emissions across the Chaco of 1591.4 Tg. If land-use changes (i.e., deforestation and conversion rate of grazing lands to cropland) continued at the rates of 1985–2000 (scenario III), 876.7 Tg C would be emitted until 2050. Conversely, if land-use changes continued at the rates of 2000–2013 (scenario IV), carbon emissions of 1813.0 Tg would be the result. Finally, if future land-use changes occurred at the past rates for the entire study period of 1985–2013 (scenario V) would result in carbon emissions of 1264.4 Tg C (Fig. 5 and Table S2).

Discussion

Carbon emissions from tropical deforestation are a key factor contributing to global climate change. We

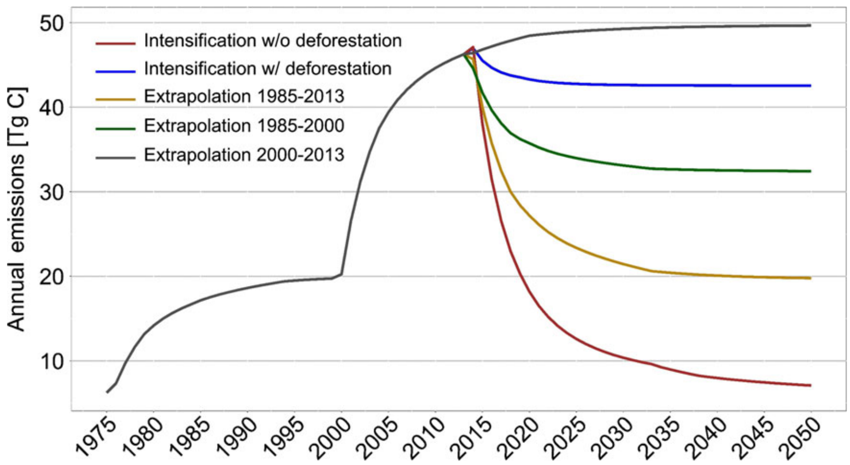

Fig. 5 Estimated carbon emissions from land-use changes until 2050 under five hypothetical land-use change scenarios: (1) conversion of all grazing lands into croplands without additional deforestation (intensification without deforestation); (2) conversion of all grazing lands in 2013 into croplands, and deforestation for grazing lands at 2000–2013 rates (intensification with deforestation 2000–2013); (3) land-use changes as in our first period 1985–2000 (extrapolation 1985–2000); (4) continuation of land conversions at rates from 2000 to 2013 (extrapolation 2000–2013); and (5) continuation of land conversions at rates from 1985 to 2013 (extrapolation 1985–2013). The graph represents the assumption of low AGB in grazing lands and no SOC during forest-to-grazing-land conversions. For the summaries of the other assumptions, please refer to Table S2 in the Supporting Information. [Colour figure can be viewed at wileyonlinelibrary.com].

quantified carbon fluxes for the South American Chaco for the period 1985–2013 using the Landsat satellite image archive to reconstruct land-use changes, and a carbon bookkeeping model to assess carbon budgets. Our analyses provide a number of key insights. First, land-use change in the Chaco has been rampant, with almost 20% of the forest being replaced by croplands or grazing lands between 1985 and 2013, at increasing pace since 2000. These land-use changes, the combined result of rising global prices, an increasing export orientation, and technological innovation, render the Chaco a global land-use change hot spot. Second, associated carbon emissions were substantial, totaling to 824 Tg C for the Chaco as a whole. Emissions mainly came from forest losses, highlighting the importance of forest protection if safeguarding the region's carbon stocks is a goal. Third, although less carbon-dense than tropical moist forests, carbon emissions from the Chaco's tropical dry forests were comparable to those from other major global deforestation frontiers such as in the Amazon or South-East Asia. Fourth, considering what happens after initial deforestation is important for understanding land-use-related carbon fluxes, and in our case led to markedly different carbon trajectories among countries. Finally, if land-use changes continued unabated in the Chaco, the region's importance as a global carbon source is likely to amplify further.

Our satellite-based assessment of deforestation dynamics, to our knowledge the first covering the Chaco in its entirety over the last 30 years, highlighted substantial deforestation, with increasing forest-loss rates since 2000. Our deforestation estimates rates correspond well with studies that covered the entire Chaco, for example, in the context of a global analysis (7800 km^2 yr^{-1} Chaco between 2001 and 2012 in Hansen *et al.* (2013), compared to our 7200 km^2 yr^{-1} between 2000 and 2013), but also compared to more regional studies such as in the Paraguayan Chaco (2472 km^2 yr^{-1} between 2005 and 2011 in Caldas *et al.* (2015) compared to our 2708 km^2 yr^{-1} between 2000 and 2013). Most of the deforestation in the Chaco was for establishing grazing lands (82%), in line with studies across entire South America (Graesser *et al.*, 2015; Sy *et al.*, 2015) or more regionalized studies in Northern Argentina (Volante *et al.*, 2016).

Deforestation rates increased in all three Chaco countries between 2000 and 2013 compared to between 1985 and 2000, likely due to a combination of rising world market prices for beef and soy, especially during the 2000s (Reenberg & Fenger, 2011; Leguizamon, 2014), an increasing export orientation and market liberalization during the 1990s and 2000s in these countries, and technological innovation (e.g., development of new soybean variants, introduction of exotic grasses to increase pasture productivity) in agriculture (Zak *et al.*, 2008; Newell, 2009; Vazquez, 2013). These general trends were superposed by important country-specific factors, explaining the marked differences in deforestation rates and the post-deforestation land uses we found. In Argentina, cropland expansion became a more dominant proximate driver of deforestation, especially after the introduction of GM soybean varieties in the early

2000s (Reenberg & Fenger, 2011) and massive support for soybean cultivation by the Argentine government after 2001 (Goldfarb & Zoomers, 2013). Soybean expansion was widespread both into forests and grazing lands, in line with previous findings based on coarse-resolution data (Clark *et al*., 2010; Graesser *et al*., 2015). To the contrary, deforestation rates were much lower in the Bolivian Chaco, possibly because the more active deforestation frontier is currently located in northern Bolivia with much undeveloped left, and actors have therefore not yet turned their attention to the Chaco (Müller *et al*., 2012). In Paraguay, deforestation was highest among the countries we assessed, but almost exclusively for establishing grazing lands. This is likely due to the introduction of productive exotic grasses (Hecht, 1975; Cabrera *et al*., 2001) and new cattle breeds (Vazquez, 2013), mainly in the Mennonite colonies around Filadelfia (Dana & Dana, 2007; Vidal, 2010), and a general rise in foreign land acquisitions (Bertello, 2008; Gonzales, 2013).

To our surprise, we found higher deforestation rates for the wet Chaco compared to the dry Chaco. While forest losses in the dry Chaco were more extensive, the high deforestation rates we found for the wet Chaco are worrying in light of the lower level of forest cover there, the importance of these forests for wildlife (riparian forests such as in the wet Chaco often function as corridors (Naiman *et al*., 1993)), and because wet Chaco forests are more carbon-dense than those in the dry Chaco. Grazing lands were the dominant post-deforestation land use in the wet Chaco, likely because many areas there are inundated during parts of the year and thus not suitable for crop cultivation (Lemaire *et al*., 2000). However, as in case of the dry Chaco, the global demand for beef and soybean seems to be driving deforestation in the wet Chaco as well (Caldas *et al*., 2015).

The land-use changes we mapped resulted in high carbon emissions of up to 31.7 Tg C annually. Most of these emissions from the Chaco came, expectedly, from AGB loss (87%), highlighting that if protecting carbon stocks and avoiding emissions is a goal, maintaining larger swaths of natural forests will be essential. The Chaco's protected area network is unfortunately still sparse, for example, covering only 2.8% of the Argentine Chaco (7.6% in the wet Chaco), 32.8% in Bolivia and 5.5% in Paraguay. Expanding protected areas, assigning larger areas where the maintenance of carbon stocks is an explicit land-use target (e.g., via zoning as in the case of Argentina's forest law, or via payment for ecosystem services schemes such as REDD+) and enforcing existing regulation are all important elements for slowing down the currently high carbon emissions from forest loss in the Chaco.

Comparing our carbon emissions estimates for the Chaco (37.2 Tg yr^{-1} between 2000 and 2013) to those from other tropical deforestation frontiers, for example, for Amazonia [172 Tg C yr^{-1} for an area almost 10 times the Chaco for 2006–2010 (Numata *et al*., 2011)], or Kalimantan [31.6 Tg yr^{-1} for an area of similar size than the Chaco (Carlson *et al*., 2013)], shows that emissions were of similar magnitude. Thus, although tropical dry forests are less carbon-dense than moist forest, carbon emissions from dry forests are contributing to climate change in comparable ways due to the rapid rates of land-use change these regions experience. Still they remain under the radar of policymakers and scientists alike, which is worrisome considering that many forests in the Chaco and other tropical dry forest regions remain weakly protected, and the underlying drivers of forest conversions continue to intensify (Kissinger & Herold, 2012).

Our study also showed that different post-deforestation dynamics are important to consider when assessing carbon fluxes, in line with work based on assessing carbon emissions from land using a sampling-based approach (Sy *et al*., 2015). Post-deforestation land-use dynamics remain often overlooked when quantifying land-use-related emissions, including in the only study for the Chaco we know of (Gasparri *et al*. (2008)). Had we quantified deforestation only, we would have neglected emissions in the order of 52.6 Tg from P-C conversions (i.e., up to 10% of the total emissions). More importantly, G-C conversions are increasingly more prevalent in the Chaco, especially in Argentina where we found 61.5% of all G-C conversions) due to technological innovations that allow farmer to intensify (e.g., new soybean strains allowing for soybean cultivation in areas historically only suitable for ranching). Moreover, farmers also often initially convert forests to grazing land with the ultimate intention of intensifying to cropland later (Baumann *et al*. 2016; le Polain de Waroux *et al*., 2016), because deforesting to grazing lands is cheaper than preparing land for crop cultivation right away, because farmers treat grazing lands as a land reservoir (Macedo *et al*., 2012), or because current zoning only allows for establishing grazing lands (Ley De Proteccion Ambiental De Bosques Nativos, 2007). Our findings thus also emphasize that policies aiming at slowing emissions from deforestation must consider that stricter zoning and stronger enforcement may incentivize landowners to convert grazing lands into croplands, thus possibly leading to overspill emissions, such as in the case of the Paraguay's Atlantic Forests deforestation ban during the 2000s (World Wildlife Fund, 2015).

The importance of considering different post-deforestation dynamics is also highlighted by our hypothetical what/if scenarios. Under all scenarios, the

Chaco will remain a significant global carbon source until 2050, even under the most conservative scenario assumptions. If deforestation halted and only post-deforestation land use would change, emissions from past land-use changes and post-deforestation agricultural intensification could still add up to almost 500 Tg C until 2050, an amount that is higher than the total carbon emissions [excluding land-use/cover change of the three countries between 1990 and 2012 (420 Tg C for the three countries (World Resources Institute, 2015)]. However, a more realistic assumption is that land-use change continues and the resulting emissions of 1562 Tg C until 2050 of our most drastic scenario would equal fourfold carbon emissions other than from land-use change between 1990 and 2012. This emphasizes the need for international and national actors and climate policymakers to focus more on the Chaco, and other dry forests, if curbing these carbon emissions is a goal.

While we here provide, to our knowledge, the first areawide assessment of land-use change and associated carbon emissions for the Chaco as a whole, using well-established analytical tools and a large sample of ground data on carbon stocks, a number of uncertainties need mentioning. First, we focused on three major land-use changes only, but did not include carbon emissions from forest degradation (e.g., from charcoal production, logging, or forest grazing), from wetland drainage, or from land management (e.g., fuel, industrial fertilizer production). This means our land-use-related carbon emissions are likely conservative. Second, we considered only carbon emissions, but did not quantify emissions of other greenhouse gases such as methane and NO_x, all of which can be expected to increase due to the land-use changes we assessed in the Chaco. Third, we used all available imagery to reconstruct land-use change back to 1985, resulting in a reliably change map, but some classes had higher accuracies than others. Especially the mapping of G-C conversions would benefit from time series analyses, to better determine the timing of these conversions which would likely increase the user's accuracy of this class further. Dense time series of imagery and the resulting higher temporal resolution, and likely also thematic detail, would thus result in smaller confidence intervals around our carbon flux estimates. Fourth, we parameterized our CBKM using three subregions (i.e., dry, very dry, wet Chaco) for our model parameters using a large sample of ground surveyed plots. However, a more fine-scale regionalization of our parameters would be beneficial, including ground data from areas (e.g., Bolivia) and vegetation formations (e.g., savannas) where our sample was sparse would be desirable and would likely improved the emission estimates further. Fifth, although our results highlight the importance of SOC emissions (13% of all emissions), the knowledge base on SOC losses (e.g., from G-C conversions vs. F-G conversions) and SOC gains (e.g., stabilization and accumulation in areas with exotic grasses) remains limited, highlighting the need for further research in this area. Lastly, while our assumptions of high and low biomass in grazing areas represent the nature of different types of grazing areas in the Chaco (i.e., grassland pastures vs. silvopastures), it represents the extreme conditions (i.e., 100% grassland pastures vs. 100% silvopastures) and spatially explicit information on the distribution of silvopastures in the Chaco would have been desirable.

In summary, our comprehensive assessment of land-use dynamics and associated carbon emissions between 1985 and 2013 for the entire Chaco highlights that these emissions were substantial and of similar magnitude than those from Amazonia or South-East Asia. This adds to voices that the disproportional focus of researchers, conservationists, and policymakers alike on deforestation in moist tropical forests may not be justified and that dry forests remain neglected – despite rapid emissions from deforestation. Moreover, our study highlights that post-deforestation land-use dynamics, especially the intensification of grazing lands to cropland, are important to consider, although often ignored. Finally, our study shows that the Landsat archives can be very valuable in understanding these dynamics and associated carbon emissions. More proactive land-use and conservation planning in the Chaco is urgently needed to curb carbon emissions, and to better balance agricultural production and carbon conservation goals.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Figure S1. Study region including country-boundaries, ecoregion-subdivision and Landsat footprint coverage.
Figure S2. Carbon emissions from land-use change between 1985 and 2013 under the four different assumptions: (1) high AGB/no SOC release, (2) low AGB/no SOC release, (3) high AGB/2-year SOC release, and (4) low AGB/2-year SOC release.
Table S1. Error matrix and accuracy measures for the remote sensing classification.
Table S2. Carbon estimation results from our five future land-use change scenarios.