Global and Planetary Change 128 (2015) 103–119

Contents lists available at ScienceDirect

Global and Planetary Change

journal homepage: www.elsevier.com/locate/gloplacha

Land use and land cover change impacts on the regional climate of non-Amazonian South America: A review

Alvaro Salazar [a,*], Germán Baldi [b], Marina Hirota [c], Jozef Syktus [a], Clive McAlpine [a]

[a] The University of Queensland, School of Geography Planning and Environmental Management, Brisbane, Australia
[b] Grupo de Estudios Ambientales—IMASL, Universidad Nacional de San Luis & CONICET, San Luis, Argentina
[c] Department of Physics, Federal University of Santa Catarina, Florianopolis, Brazil

ARTICLE INFO

Article history:
Received 26 September 2014
Received in revised form 30 January 2015
Accepted 24 February 2015
Available online 3 March 2015

Keywords:
Dry Chaco
Cerrado
Deforestation
Climate
Tropical Dry Forests
Atlantic forest

ABSTRACT

Land use and land cover change (LUCC) affects regional climate through modifications in the water balance and energy budget. These impacts are frequently expressed by: changes in the amount and frequency of precipitation and alteration of surface temperatures. In South America, most of the studies of the effects of LUCC on the local and regional climate have focused on the Amazon region (54 studies), whereas LUCC within non-Amazonian regions have been largely undermined regardless their potential importance in regulating the regional climate (19 studies). We estimated that 3.6 million km^2 of the original natural vegetation cover in non-Amazonian South America were converted into other types of land use, which is about 4 times greater than the historical Amazon deforestation. Moreover, there is evidence showing that LUCC within such fairly neglected ecosystems cause significant reductions in precipitation and increases in surface temperatures, with occasional impacts affecting neighboring or remote areas. We explore the implications of these findings in the context of water security, climatic extremes and future research priorities.

Contents

* Corresponding author. Tel.: +61 7 3365 6455; fax: +61 7 3365 6899.
E-mail address: Alvaro.salazar.p@gmail.com (A. Salazar).

http://dx.doi.org/10.1016/j.gloplacha.2015.02.009
0921-8181/

1. Introduction

Land use and land cover change (LUCC) affects climate through changes in moisture and energy budgets (Pielke et al., 2011; IPCC, 2014a; Mahmood et al., 2014). In South America, most of the focus on these impacts has been directed at deforestation of the Amazon forests (e.g., Pires and Costa, 2013). By contrast, non-Amazonian South America has received less attention despite experiencing the highest transformation rates in the tropics (Marris, 2005; Hansen et al., 2013). This is a significant problem because the loss of native ecosystems can modify the local and regional surface–climate coupling through feedback processes, and increase the risks imposed by climate extremes in an area that sustains a human population of over 200 million (Grimm and Tedeschi, 2009).

Non-Amazonian South America, also referred to as non-Amazonian ecosystems, covers an area of more than 12 million km^2 and is characterized by a high diversity of biomes including tropical rainforests, tropical savannas, grasslands, shrublands, deserts and a wide array of woodland formations that are distributed according to rainfall, temperature, soil properties and disturbance regimes. Precolonial pressures upon these biomes were expressed through settlement, cultivation, grazing, hunting and burning by indigenous people (Knapp, 2007). However, these changes were temporary and therefore relatively rapidly reversed by ecological succession (Armesto et al., 2010). Since 1500 and especially since 1900, the expansion of European agriculture has resulted in widespread ecosystem transformations. Global demand for food commodities such as soybeans and beef has pushed the expansion of the agro-pastoral frontier into former natural and seminatural areas (Richards et al., 2012). Recent studies have shown high LUCC rates in tropical savannas of Brazil (hereafter referred as Cerrado) (Sano et al., 2010), grasslands in Argentina (Baldi et al., 2006), Atlantic Forests in eastern Brazil (Joly et al., 2014) and the dry forests in the Paraguayan Chaco (Huang et al., 2009). Of the 542,000 km^2 of deforestation in South America between 2000 and 2012, 42% occurred in the Amazon region and 58% in the non-Amazonian region (Hansen et al., 2013).

Changes in land use and land cover can have profound impacts on land surface climate feedbacks by altering the exchange of heat, moisture, momentum, trace-gas fluxes and albedo (Bonan, 2008). Cumulatively, they can impact the climate at a local (Montecinos et al., 2008; Hidalgo et al., 2010; Mohamed et al., 2011), regional (Pitman et al., 2004; Roy et al., 2007; Fairman et al., 2011) and even global scales (Bounoua et al., 2002; Snyder et al., 2004; Avissar and Werth, 2005; Feddema et al., 2005; Lawrence et al., 2012). Many of the studies addressing climatic impacts of LUCC focus on the tropical forests, particularly in the Amazon region. Results suggest that tree removal produces a drier and warmer climate due to reductions in evaporative cooling with implications to vegetation dynamics, river discharge and climate extremes (McGuffie et al., 1995; Rocha et al., 2004; D'Almeida et al., 2007; Sampaio et al., 2007; Costa and Pires, 2010; Pires and Costa, 2013; Stickler et al., 2013). However, evidence relating climate and LUCC in other ecosystems of tropical and subtropical South America is scarce and dispersed. The high conversion rates of natural vegetation and the vulnerability of ecosystems to climate variability create an increasing need to identify signals and patterns of the impacts of LUCC on the regional climate. This will better inform climate science and natural resource management. It's been argued that climate impacts induced by LUCC are significantly comparable to those resulting from anthropogenic greenhouse gases (Pielke et al., 2002), particularly at local to regional scales, in which people and ecosystems are mostly affected (Mahmood et al., 2010). Though there is a good understanding of the major biogeophysical feedbacks of Amazon deforestation, land surface climate interactions and their consequences in non-Amazonian South America are much less understood.

In this paper, we review the modeling and empirical evidence that shows the climatic impacts of LUCC in non-Amazonian ecosystems of South America. First, we estimate the original and remaining amount of natural vegetation in the Amazon and in six non-Amazonian ecosystems. We then assess the impacts of LUCC on the climate of non-Amazonian South America and evaluate the implications and potential risks with regard to climate change and future research priorities.

2. Methods

2.1. Delimiting the Amazon and non-Amazonian South America

We focused on six broad ecosystems, collectively referred as non-Amazonian South America. We also considered the Amazon biome as defined by WWF (2010) to compare surface climate feedback studies between Amazonian and non-Amazonian South America. We defined non-Amazonian ecosystems in South America based on two criteria: 1) they must be located outside the area covered by the Amazon biome and 2) they must exhibit at least one peer-reviewed study describing impacts of land use and land cover change (LUCC) on local or regional climate (see Section 2.4). We geographically delimited them using Olson et al. (2001), MMA/IBAMA (2011a), and MMA/IBAMA (2011b). The final selection covered an area of about 6.3 million km^2 and included: 1) Dry Chaco, 2) Cerrado, 3) Temperate Grasslands, 4) Chilean Matorral, 5) Tropical Dry Forests and 6) Atlantic Forest (Fig. 1). These ecosystems represent a variety of functional groups including moist forests, dry broadleaf forests, grasslands, savannas, shrublands, mediterranean forests, and xeric shrublands. All of them have been subjected to extensive anthropogenic modification (Olson et al., 2001; Friedl et al., 2010).

2.2. Estimating potential and current natural vegetation cover

LUCC information in South America is highly fragmented and localized. For this reason, we estimated potential and current natural vegetation extent for both regions using different sources: 1) peer-reviewed publications, 2) technical reports and 3) the Collection 5 MODIS Global Land Cover Type for year 2012 (Friedl et al., 2010). We first defined potential forest cover (natural) in the Amazon region as the total area described in WWF (2010) without considering savanna ecoregions as classified by Olson et al. (2001). Then we extracted areas covered by evergreen broadleaf forests in these savannas according to Collection 5 MODIS Global Land Cover Type for year 2012. This procedure added those forests (e.g., gallery forests) distributed in areas dominated by savanna vegetation (e.g., Beni Savannas in Fig. 1) inside the Amazon region and gave us the approximate potential extent of dense moist tropical forest in the Amazon.

We obtained the potential historical natural vegetation extent in non-Amazonian South America from regional and local studies. We used Olson et al. (2001) classification for the Dry Chaco, Temperate Grasslands and the Atlantic Forest; MMA/IBAMA (2011b) for the Cerrado; MMA/IBAMA (2011a) for the Caatinga; Portillo-Quintero and Sánchez-Azofeifa (2010) for the Tropical Dry Forests; and Luebert and Pliscoff (2006) for the Chilean Matorral. We included Caatinga into Tropical Dry Forests as suggested by Portillo-Quintero and Sánchez-Azofeifa (2010). However we presented vegetation change

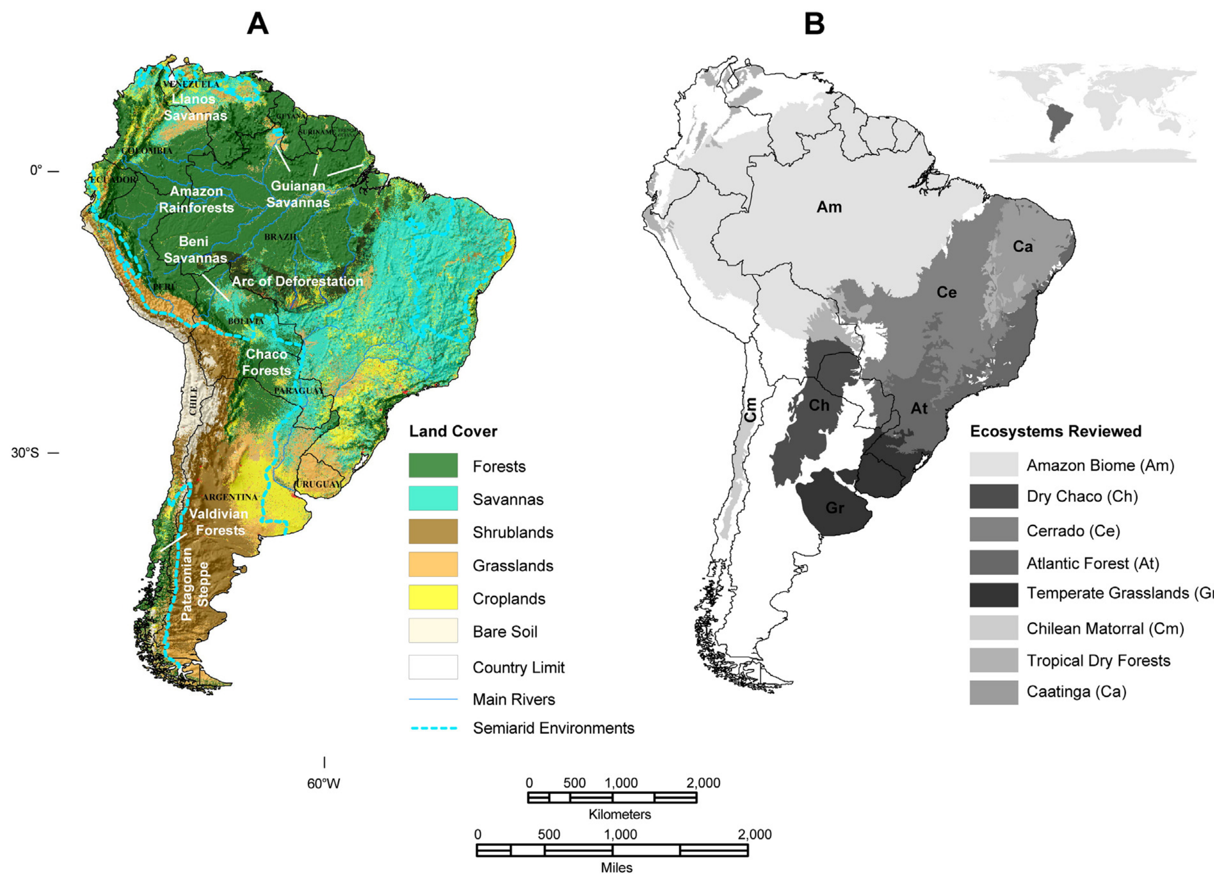

Fig. 1. (A) Land cover map of continental South America based on Collection 5 MODIS Land Cover Type product for year 2012 and (B) ecosystems of non-Amazonian South America reviewed in this article (Amazon biome also shown). Other non-Amazonian ecosystems not included in this study are displayed as white in map (A) (e.g., Llanos Savannas in Colombia and Venezuela). Semiarid environments (Aridity Index <0.65) are derived from Trabucco and Zomer (2009). The southern Amazon arc of deforestation is also shown. Ecosystem limits in map (B) were obtained from Olson et al. (2001) for Dry Chaco, Atlantic Forest, Temperate Grasslands and Chilean Matorral. The maps from MMA/IBAMA (2011a); MMA/IBAMA (2011b) were used to define the Cerrado and the Caatinga ecosystems, whereas Portillo-Quintero and Sánchez-Azofeifa (2010) was used for Tropical Dry Forests.

results separately since we used two different sources for calculating the vegetation cover change for these regions (Table 1).

We obtained current natural vegetation cover from peer-reviewed studies and Collection 5 MODIS Global Land Cover Type for the year 2012. We calculated the current vegetation area in the Amazon biome and the Atlantic Forests as that covered by evergreen broadleaf forest in the MODIS product. We adopted the same procedure to define the current extent of Temperate Grasslands (grasslands in the MODIS product). For the Dry Chaco ecosystem we used the work of Clark et al. (2010) who classified the forest cover at 250 m resolution using

Table 1
Methods for potential and current natural vegetation cover estimation in the Amazon Biome and non-Amazonian South America.

Ecosystem	Reference used to estimate potential vegetation extent	Methods for potential vegetation extent	Reference used to estimate current vegetation extent	Methods for current vegetation extent as described in the references used
Amazon Biome	Olson et al. (2001); Friedl et al. (2010); WWF (2010)	WWF (2010) biome boundary with substracted savanna areas as defined by Olson et al. (2001) not covered by evergreen broadleaf forests according to Collection 5 MODIS for year 2012	Friedl et al. (2010)	Classified as evergreen broadleaf forest in Collection 5 MODIS for year 2012
Chaco	Olson et al. (2001)	Covered by dry forest according to Olson et al. (2001)	Clark et al. (2010)	MODIS 250 m vegetation index product (MOD13Q1) for year 2006
Cerrado	MMA/IBAMA (2011b)	Covered by savanna according to MMA/IBAMA (2011b)	MMA/IBAMA (2011b)	Classification of Landsat TM images for year 2009
Temperate Grasslands	Olson et al. (2001)	Covered by grasslands according to Olson et al. (2001)	Friedl et al. (2010)	Classified as grasslands in Collection 5 MODIS for year 2012
Chilean Matorral	Luebert and Pliscoff (2006)	Covered by forest and shrubland according to Luebert and Pliscoff (2006)	Conaf et al. (1999)	Classification based on aerial photo interpretation
Tropical Dry Forests	Portillo-Quintero and Sánchez-Azofeifa (2010)	Olson et al. (2001) ecoregions defined as Tropical Dry Forests.	Portillo-Quintero and Sánchez-Azofeifa (2010)	Supervised classification of MODIS surface reflectance imagery at 500-m resolution for year 2004
Tropical Dry Forest-Caatinga	MMA/IBAMA (2011a); IBGE (2012)	Savanna and forests given MMA/IBAMA (2011a)	MMA/IBAMA (2011a)	Classification of Landsat TM and CBERS – 2B CCD images for year 2009
Atlantic Forest	Olson et al. (2001)	Covered by forest according to Olson et al. (2001)	Friedl et al. (2010)	Classified as evergreen broadleaf forest in Collection 5 MODIS for year 2012

MODIS. We used IBAMA's (Brazilian Institute of Environment and Renewable Natural Resources) estimation of remanent natural vegetation for the Cerrado (MMA/IBAMA, 2011b) and the Caatinga (MMA/IBAMA, 2011a). Current extent of forests and shrublands in the Chilean Matorral was obtained from Conaf et al. (1999). For Tropical Dry Forests, we used the forest cover area calculated by Portillo-Quintero and Sánchez-Azofeifa (2010). The maps used to calculate current natural vegetation area were the most complete and accurate we had access to (Table 1). We used the Albers Equal Area projection and South American 1969 Datum for all maps.

2.3. Recent change in forest cover

Based on Hansen et al. (2013), we explored the recent changes in forest cover for the Amazon Biome and non-Amazonian South America. Hansen et al. (2013) used Landsat time-series data to quantify global forest loss and gain at a spatial resolution of ca. 30 m for the period 2000–2012. However, this dataset does not discriminate between native forest and exotic tree plantations. Since some regions of South America are now used for intensive forestry practices with high rates of forest loss and gain (Jobbágy et al., 2012), we discuss the Hansen et al. (2013)'s results in conjunction with local studies in order to better understand forest cover dynamics in non-Amazonian South America (see Section 2.4).

2.4. Review of LUCC impacts on climate

We searched peer-reviewed literature in Web of Science database from year 1900 to 2013 using a combination of the following key words: "climate", "land cover change", "South America", "land use change", "deforestation" and "ecosystems' names". Due to the spatial extent of atmospheric studies, the same work might be cited for two or more ecosystems of non-Amazonian South America. We also searched in Web of Science for LUCC climate feedbacks in the Amazon using the key words: "deforestation", "Amazon", "climate", "impact", "land cover change" and "land use change" for the period 1993–2013. We focused the search only on biophysical impacts.

From the bibliographic lists of these articles and from previous literature searches conducted by the authors, we included supplementary articles referring to LUCC and climate feedbacks for non-Amazonian South America. We considered articles only if they met the following criteria:

1) The article must be published, peer reviewed and written in English.
2) The article must have focused on a geographic area outside the Amazon Biome.
3) The article must have explicitly referred to LUCC processes (e.g. conversion from forest to crops).
4) The article must have contained information of LUCC impacts for at least one of the following climatic components: temperature, precipitation and albedo.

Moreover, LUCC articles were also searched to identify the immediate and underlying causes of these dynamics in non-Amazonian South America. The key words used in this search were: "land cover", "South America", "land use", "ecosystem's name", "land cover change", "land use change" and "deforestation". Relevant studies identified from the bibliographic lists of the articles were also included.

Based on the aforementioned criteria, we included a total of 19 LUCC climate feedback related studies for non-Amazonian South America. For each article, we recorded location, study type (modeling or observational), study period, LUCC direction and impacts on temperature, precipitation and albedo (Table 2; an expanded version of this table is shown in the Supplementary Information). The selection of articles in this review was used to choose the regions classified as non-Amazonian South America. This implied that other non-Amazonian ecosystems were not included because we did not find LUCC climate feedback studies for these ecosystems. Examples are represented by the Llanos savanna of Venezuela and Colombia (Etter et al., 2008; Portillo-Quintero et al., 2012; Romero-Ruiz et al., 2012), the Patagonian Steppe in southern Argentina (Paruelo et al., 2001; Bisigato and Laphitz, 2009), the Valdivian forests in southern Chile (Huber et al., 2008) and Ecuadorian Páramo (Farley, 2007), among others.

We acknowledge that land cover datasets and models have varying levels of accuracy and methodologies which limit the ability to make comparisons. We recognize that these differences provide bounds of uncertainty on the major findings, yet they do not invalidate the major conclusions presented here. The main focus of the paper is on the available evidence of LUCC impacts on climate in non-Amazonian South America and is not possible to cover in detail data inaccuracies of the varying approaches.

3. Results

3.1. General trends in LUCC

Historically, the non-Amazonian ecosystems of South America have lost more than 3.6 million km^2 (58% of their potential natural vegetation). This is equivalent to about 4 times the historic Amazon deforestation (918,473 million km^2). Recent forest loss (period 2000–2012) in non-Amazonian ecosystems (excluding Temperate Grasslands) accounts for 45% of the total forest loss in South America (241,551 km^2), compared to the loss of rainforest in the Amazon Biome which represents 42% (227,249 km^2) of total South American deforestation (541,887 km^2).

The ecosystem relatively most impacted by LUCC was the Chilean Matorral, where 83% of its potential natural vegetation had been transformed to other land uses by 1999 (date of the current vegetation map). This ecosystem also showed a high loss and gain in forest area for period 2000–2012 (Table 2; Fig. 3), indicating the presence of exotic tree plantations as reported by different studies (e.g., Niklitschek, 2007). The second most relatively impacted was the Atlantic Forest, with 81% (978,031 km^2) of its potential extent lost by 2012. It also experienced high rates of forest loss and gain between years 2000 and 2012 (Table 2). Together with the Chilean Matorral and the Cerrado, the Atlantic Forest shows the greatest area of exotic tree plantations in South America (Jobbágy et al., 2012). Conversely, the Dry Chaco exhibited the lowest relative extent of historic transformation (34%). Yet, it showed the highest deforestation rate for tropical forests between the years 2000 and 2012 (Hansen et al., 2013) and a very low forest gain in the same period (Table 2). The Dry Tropical Forests have also undergone high historic deforestation and presently cover approximately 40% of their former extension (Sánchez-Azofeifa and Portillo-Quintero, 2011). The most studied areas are the Caatinga in northeast Brazil and the Chiquitano forests in Bolivia, with limited references found for the Tropical Dry Forests of Colombia, Ecuador, Venezuela and Peru, where the remaining forest area is less than 6% of its potential extent (Portillo-Quintero and Sánchez-Azofeifa, 2010). In the case of the Cerrado, 52% has been converted into crops and pastures over an area of about 1 million km^2 (MMA/IBAMA, 2011b). Interestingly, even though the Temperate Grasslands were formerly composed by grassland vegetation, only 30% of which remained by year 2012, they showed high dynamic forest area between 2000 and 2012, presumably due to an increased area of exotic tree plantations (Jobbagy and Jackson, 2007; Nosetto et al., 2012; Hansen et al., 2013).

In terms of vegetation climate feedbacks and associated publications, results showed clear differences between the Amazon and non-Amazonian South America. The Amazon presents a historic land cover change area (loss of rainforests) of about 920,000 km^2 with 54 publications addressing the associated climate impacts. By contrast, historic LUCC in non-Amazonian South America totalled 3.6 million km^2, and its climatic effects were addressed by 19 publications (Fig. 2). Of these studies, 70% focused on the Cerrado and the Tropical Dry

Table 2
Changes in natural vegetation cover for the Amazon Biome and non-Amazonian South America. Total recent forest loss for non-Amazonian South America represents only the ecosystems included in this review.

Region	Main vegetation type	Potential historic area (km^2)	Present area (km^2)	Converted area (km^2)	Converted area (%)	Recent forest cover change for period 2000–2012 based on Hansen et al. (2013)	
						Loss (km^2)	Gain (km^2)
Amazon Biome	Forest	6,546,242	5,627,769	918,473	14	227,249	15,972
Dry Chaco	Forest	786,790	516,011	270,779	34	62,815	695
Cerrado	Savanna	2,039,386	983,348	1,056,038	52	87,274	21,691
Atlantic Forest	Forest	1,204,467	226,436	978,031	81	44,658	33,056
Temperate Grasslands	Grasslands	777,571	236,240	541,331	70	5562	12,181
Chilean Matorral	Forest and shrublands	62,935	10,751	52,184	83	2127	3065
Tropical Dry Forests	Forest	664,191	268,875	395,316	60	27,661	2404
Caatinga	Forest and shrublands	787,968	431,877	356,091	45	17,016	1634
Total Amazon Biome	–	**6,546,242**	**5,627,769**	**918,473**	**14**	**227,249**	**15,972**
Total non-Amazonian	–	**6,323,308**	**2,673,538**	**3,649,770**	**58**	**247,113**	**74,726**
Total South America	–	–	–	–	–	**541,887**	**118,532**

Forests including Caatinga, whereas just one publication was found for the Atlantic Forest, where 978,031 million km^2 of the estimated original forest cover has been cleared. Most of the studies in non-Amazonian ecosystems were accomplished using climate or surface models and only 4 observational-based publications were conducted using remote sensing and weather stations.

In the following sections, we present the results for specific ecosystems. For each non-Amazonian ecosystem, the results are organized based on patterns of LUCC and evidence of climatic impacts according to modeling and observational studies.

3.2. LUCC and its climate impacts in non-Amazonian South America

3.2.1. Dry Chaco

3.2.1.1. LUCC. Former land use of the Dry Chaco was influenced by Aborigines who used fire as a management tool to modify vegetation for hunting, communication and war (Gordillo, 2010). These activities were mainly conducted in grasslands occurring on sandy soils near rivers and in ancient river beds (Morello and Adamoli, 1974). Following European settlement and especially during the first half of the 20th century, extensive cattle ranching, logging, firewood and charcoal extraction led to changes in herbaceous/woody vegetation dynamics and in the forest cover with agriculture occurring in the foothills of the Andes and humid valleys (Adamoli et al., 1990; Bucher and Huszar, 1999).

LUCC accelerated during the second half of the 20th century. From the 1990s, a synergetic combination of global food demand, technology and climatic factors increased the rate of land cover change in the Chaco to those comparable with the Amazon deforestation (e.g., Boletta et al., 2006; Zak et al., 2008). Methods for monitoring Dry Chaco deforestation vary from visual interpretations of aerial photos to digital classifications of multi-temporal satellite imagery (UMSEF, 2007; Gasparri and Grau, 2009; Huang et al., 2009; Clark et al., 2010). All studies document high

Table 3
Summary of LUCC impacts on temperature, rainfall and albedo in non-Amazonian South America for 19 studies reviewed. Numbers represent the amount of peer-reviewed publications and signs represent the direction of change in temperature, rainfall and albedo (e.g., 2+: two publications reporting an increase in temperature for the specific LUCC direction, e.g., from woody to crops in the Dry Chaco). Woody refers to woody vegetation (forests, savannas or shrublands).

Ecosystems	LUCC		Temperature	Rainfall	Albedo	Reference
	From	To				
Dry Chaco	Woody	Crops	2+;1−	1+;1−	2+	1.Houspanossian et al. (2013); 2. Loarie et al. (2011a); 3. Canziani and Carbajal Benitez (2012); 4. Lee and Berbery (2011); 5. Beltrán-Przekurat et al. (2012)
	Grassland	Crops	1+	1−	1+	Beltrán-Przekurat et al. (2012)
	Crops	Woody	1−	1+	No impact	Beltrán-Przekurat et al. (2012)
Cerrado	Woody	Crops	2+	2−	NE[a]	6. Costa and Pires (2010); 7. Lee et al. (2011); 8. Pongratz et al. (2006); 9. Mendes et al. (2010)
	Woody/crops	Sugarcane	1+; 1−	1−	1+	10. Georgescu et al. (2013)
	Woody	Crop/Pasture	1+	NE	1+	11. Loarie et al. (2011b)
	Crop/Pasture	Sugarcane	1−	NE	1+	Loarie et al. (2011b)
	Woody	Pasture	1+	NE	NE	Pongratz et al. (2006)
Temperate Grasslands	Grassland	Crops	2−; 1+	2−; 3+	3+; 1−	Beltrán-Przekurat et al. (2012); Lee and Berbery (2011); Loarie et al. (2011a)
	Grassland	Woody	1−	1+	1+	Beltrán-Przekurat et al. (2012)
Chilean Matorral	Woody	Crops	1−	NE	1−	12. Montecinos et al. (2008)
	Grasslands	Crops	1+	No impact	1+	Beltrán-Przekurat et al. (2012)
Tropical Dry Forests-Chiquitano	Woody	Crops	1+	NE	1+	13. Bounoua et al. (2004)
	Grasslands	Crops	1+	NE	No Impact	Bounoua et al. (2004)
Tropical Dry Forests-Caatinga	Woody	Desert	2+	3−	2+	14. Oyama and Nobre (2004); 15. Castilho de Souza and Oyama (2011); 16. Hirota et al. (2011)
	Savanna albedo	Desert albedo	NE	1−	NE	17. Sud and Fennessy (1982)
	Normal conditions by 1979	Evaporation supressed	NE	1−	NE	18. Sud and Fennessy (1984)
Atlantic Forest	Woody	Crops/pastures	NE	1−	NE	19. Webb et al. (2005)

[a] NE: not evaluated.

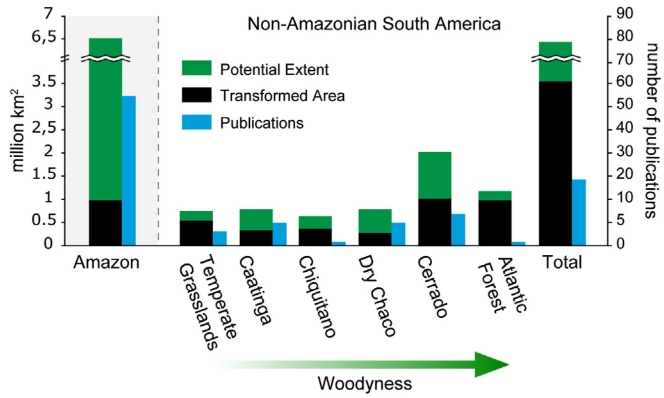

Fig. 2. Comparison of the geographic extent of LUCC and the number of publications documenting the climatic impacts of LUCC for the Amazon and non-Amazonian ecosystems (Chilean Matorral not shown). Dark green and back bars represent potential natural vegetation extent and transformed area, respectively. Blue bars indicate number of publications. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

rates of deforestation. In the Argentinian Chaco, LUCC intensified after the 1980s predominantly over flat terrains where rainfall supported rainfed agriculture (Grau et al., 2005a; Paruelo et al., 2006; Gasparri et al., 2008; Gasparri and Grau, 2009). Most studies from the Chaco show consistent pattern of the replacements of native forests by pastures and croplands, particularly soybean plantations (e.g., Clark et al., 2010; Hoyos et al., 2013). In some areas, such as the Cordoba and Santiago del Estero provinces, deforestation rates of 2% to 5% per year have been reported (Zak et al., 2004; Boletta et al., 2006), which is higher than deforestation in some of the world's humid tropical forests (Achard et al., 2002). Recent observations using Landsat imagery show that between 2010 and 2011, more than 600,000 ha of the Dry Chaco ecosystem was deforested, 86% of which occurred in Paraguay, 12% in Argentina and about 2% in Bolivia (Rodas et al., 2012). This is corroborated by the results of Hansen et al. (2013), who described the Dry Chaco as registering one of the highest rates of tropical forest loss between the years 2000 and 2012, with a total of 62,800 km^2. The continued growth in the global demand for soybean, technological advances and the use of transgenic crop varieties to overcome climate limitations, are expected to increase deforestation in the Dry Chaco, particularly in those areas with more fertile and moister soils (Grau et al., 2005b).

3.2.1.2. Impacts on climate

3.2.1.2.1. Modeling studies. In a study covering southern South America and including the Dry Chaco, Beltrán-Przekurat et al. (2012) applied a regional climate model to evaluate near-surface changes resulting from conversions of pre-European vegetation to present day land cover and under future afforestation scenarios. The conversion of wooded vegetation to soybean plantations decreased surface parameters such us roughness length, leaf area index and rooting depth, and decreased latent and sensible heat fluxes. These changes resulted in an increase in the 2 m surface temperature up to 0.6 °C during dry years, with uncertain effects on rainfall. Similarly, the study of Canziani and Carbajal Benitez (2012) reported a temperature increase in < 1 °C during austral winter (dry season) and spring over the deforested areas of the Chaco and beyond, yet without clear impacts on precipitation. By contrast, Lee and Berbery (2011) using the Weather Research and Forecasting Model (WRF), found less surface temperatures and rainfall when crops replaced savanna and evergreen broadleaf forests. According to the authors, the decrease in temperature was triggered by a significant increase in surface albedo and subsequent decrease in sensible heat fluxes, while the decrease in precipitation was related to a reduction in moisture convergence because of stronger low level winds that favored the advection of large amounts of moisture out of the deforested areas.

3.2.1.2.2. Observational studies. In the Dry Chaco, remote sensing approaches have been used to identify effects of land cover change in surface temperatures and albedo. For instance, Houspanossian et al. (2013) combined the remote sensing and modeling techniques to calculate differences in temperature and albedo between dry forests and crops. Based on satellite images, the authors report a black-sky albedo ca. 50% higher in croplands (mainly soybean but also corn, sunflower, wheat, and rye) compared to dry forests. These results agree with those described by Loarie et al. (2011a), who found that forest–agriculture conversions in the Chaco are responsible for about 7% of albedo increases in South America between 2000 and 2008. Houspanossian et al. (2013) also reported temperatures 1.6 to 5 °C higher in croplands than in dry forests, which they attribute to the cooling effect of the higher evapotranspiration rates of dry forests compared to rainfed croplands.

Other studies not included in this review have also highlighted the potential influences that land surface processes likely associated to LUCC, specifically rainfed agricultural practices, may exert in the Chaco's surface climate. According to Collins et al. (2009), observed increases in surface temperature from 1948 over specific areas of tropical and subtropical South America cannot be explained solely by El Niño or La Niña events and might be the result of human activities such as land use change and/or increased levels of anthropogenic greenhouse gases. Moreover, Nuñez et al. (2008), for the period 1961–2000, applied an "observation minus reanalysis" method to estimate the potential links between land cover change and surface temperature change over Argentina, identifying a warming trend of minimum and maximum temperatures in northern and northeastern areas of the country, which have experienced high land conversion rates during recent decades (Viglizzo et al., 2011). However, in the study of Nuñez et al. (2008) there was no clear link between changes in temperature, precipitation, and changes in land cover.

Overall, studies from the Dry Chaco suggest that the dry forests may induce a cooler and wetter climate as a result of presenting higher latent heat fluxes. Here, the presence of deep-rooted forest and woodland vegetation can produce a shallower, cooler, and moister boundary layer that shifts to warmer and drier conditions after conversion to croplands. This offsets the cooling trend associated with albedo increases when forest (low albedo) is replaced by croplands (higher albedo) (Beltrán-Przekurat et al., 2012; Houspanossian et al., 2013). Yet, the impact on precipitation is not as clear as with surface temperature, showing a positive trend during dry years and negative during wet years (Beltrán-Przekurat et al., 2012). In this regard, a complementary study conducted by Saulo et al. (2010) identified that local feedback effects occur between land and precipitation in subtropical Argentina, and that these are expressed by variations in soil moisture (which is partially controlled by photosynthetic respiration) and consequently potential influences on evaporation, convective available potential energy, and hence, precipitation.

The modeling and observational studies for the Chaco generally agree in the positive trend of surface temperature without clear impacts on precipitation after deforestation. Comparison of the results is limited however, due to differences in modeling settings, period of analysis, spatial resolution and input datasets. In this regard, modeling studies using global climate models better account for land–atmosphere feedbacks and interactions with neighboring areas, and therefore are the most suitable approach for LUCC–climate interactions.

3.2.2. Cerrado

3.2.2.1. LUCC. During the last 30 years, the Brazilian Cerrado has been rapidly transformed through large-scale agriculture into one of the world's most threatened ecoregions (Machado et al., 2004). Before the 1970s, land use in the Cerrado was dominated by low-impact cattle

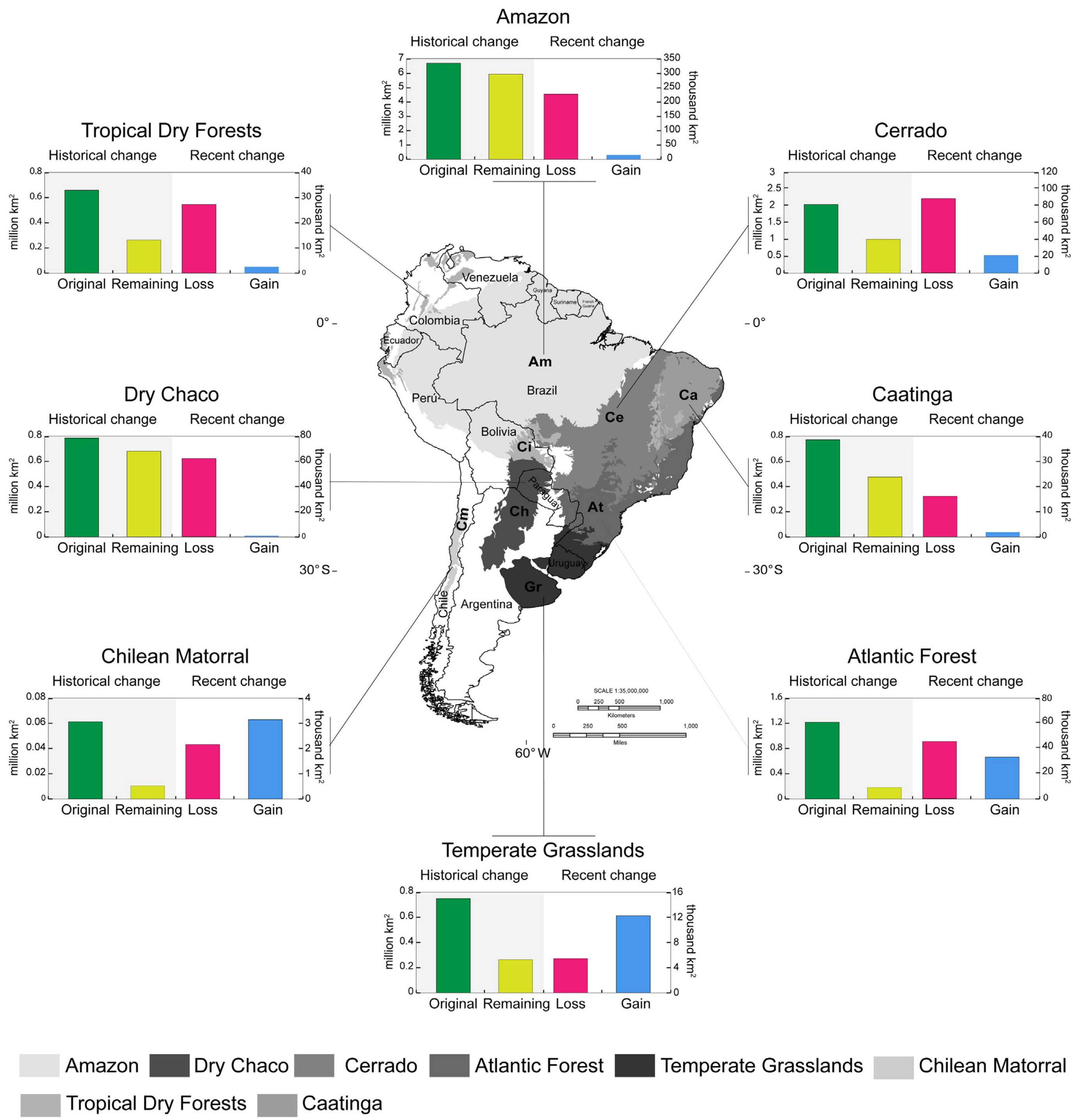

Fig. 3. Distribution and land cover change in the Amazon and non-Amazonian South America: Amazon (Am), Dry Chaco (Ch), Cerrado (Ce), Atlantic Forest (At), Temperate Grasslands (Gr), Chilean Matorral (Cm) and Tropical Dry Forests including Caatinga (Ca) and Chiquitano (Ci). For non-Amazonian South America, original and remaining vegetation area (green and yellow bars) were obtained using ecoregions of Olson et al. (2001) and literature. For the Amazon, the original forest cover was calculated as the total biome area without considering savannas according to Olson et al. (2001), and remaining forest area was calculated from Collection 5 MODIS for year 2012 at 500 m resolution. Recent change (red and blue bars) represents changes in forest cover for period 2000–2012 and was taken from Hansen et al. (2013) datasets. Forest gain, particularly in the Atlantic Forest, Chilean Matorral and Temperate Grasslands are linked to exotic tree plantations (see main text). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

ranching on native vegetation (Sano et al., 2010). However, in recent years, planted pastures and the introduction of extensive and mechanized agriculture including soybean production has transformed the Cerrado savanna into a commercial agro-pastoral landscape (Brannstrom et al., 2008). Brazil is the world's second-largest (after U.S.A.) soybean producer with a production that increased from 1.5 million tonnes in 1970 to 74.8 million tonnes by 2011, 60% of which is concentrated in the Cerrado (Smaling et al., 2008; Jepson et al., 2010; FAO, 2013).

According to Machado et al. (2004), 55% of the Cerrado natural vegetation was cleared by 2002 at rates that would remove all natural vegetation in the region by 2030. However, differences in methodological approaches such as geographic boundaries, mapping scale and remote sensing approaches, have reported land conversion areas between 40% and 80% (e.g., Alho and Martins, 1995; Sano et al., 2010). The southern Cerrado has experienced the highest transformation rate with a clearing frontier expanding north where most of the natural vegetation remains (Diniz-Filho et al., 2009) (Fig. 1). According to MMA/IBAMA (2011b), more than 980,000 km^2 (54%) of the Cerrado's natural vegetation has been converted into other land uses. Hansen et al. (2013)'s datasets show highly dynamic forest area in the ecosystem: from year 2000 to 2012, 87,274 km^2 of the forested area was removed. However, woody cover increased over almost 22,000 km^2 within the same period, probably because of the expansion of exotic tree plantations (Sano et al., 2010). The main causes of Cerrado's LUCC are linked to explicit state development policies (Klink and Moreira, 2002), global soybean demand, and increasing forestry and sugarcane for biofuel production (Loarie et al., 2011b; Jobbágy et al., 2012).

3.2.2.2. Impacts on climate

3.2.2.2.1. Modeling studies. The few modeling studies for the Cerrado have addressed land surface climate feedbacks in the transitional area between the tropical Amazon rainforests and the Cerrado semi-deciduous forests, the zone known as the “arc of deforestation” (Pongratz et al., 2006; Costa and Pires, 2010; Mendes et al., 2010) (Fig. 1). Studies using climate models generally agree in the temperature and rainfall response when woody vegetation is replaced by crops. For instance, Georgescu et al. (2013) simulated the replacement of Cerrado vegetation by sugarcane using the WRF model, reporting a surface cooling up to ca. 1.0 °C during the growing season, and a warming of similar magnitude after harvesting. In their study, the cooling was due to an increased albedo, the warming was influenced by a decline in evapotranspiration and increased sensible heating, while total rainfall also decreased. A drying trend was also described by Costa and Pires (2010) who report a decrease in moisture fluxes and consequently increase the duration of the dry season from the current 5 to 6 months when the cumulative influence of the Amazon deforestation is considered, and by Lee et al. (2011), who project a reduction in total rainfall during the dry season and a warming that increases the risk of more frequent and severe droughts.

In agreement with the warming trend shown by the climate models, land surface modeling also projects an increase in surface temperatures following deforestation in the Cerrado. A study by Pongratz et al. (2006) used a land surface model to evaluate the effects of vegetation changes on the local energy and water balances in the north-central state of Mato Grosso. Here, the conversion of transitional forests, composed by both Amazon and Cerrado vegetation, to cropland resulted in an increased canopy temperature of up to 0.7 °C at midday. Similarly, the conversion of transitional forests to pasture caused an increase in maximum temperature of ca. 0.5 °C, driven by a reduced roughness length and increased aerodynamic resistance. These compensated the cooling trend associated with higher physiological activity of pastures (C4 photosynthetic pathway) compared to transitional forests (C3 photosynthetic pathway). In addition, temperature response is intensified when transitional forest are cleared and converted into bare soils, resulting in a temperature anomaly of 1.2 °C in the dry season (Pongratz et al., 2006). Interestingly, these modeled impacts induced by the loss of woody vegetation in the Cerrado, can potentially affect neighboring areas of the Amazon biome and can enhance the transition from rainforest to Cerrado type vegetation in the next 40 years due to a drier climate associated with Cerrado deforestation (Mendes et al., 2010).

3.2.2.2.2. Observational studies. Evidence from observations of the climatic effects of vegetation loss in the Cerrado generally agrees with those described by modeling studies. Satellite images were used by Loarie et al. (2011b) to evaluate the climate effects of crop/pasture and sugarcane expansion in the Brazilian Cerrado. In their study, transformations from natural vegetation to crop/pasture triggered a decrease in evapotranspiration and an increase in average surface temperature of 1.6 °C. On the other hand, conversions from crop/pasture to sugarcane plantations lead to a mean cooling of 0.93 °C due to an increase in evapotranspiration and in the albedo, with the former exerting the greatest influence on the surface temperature response.

As in the Chaco, the climatic response after land cover change in the Cerrado depends largely on changes in energy fluxes rather than albedo changes. Water uptake by deep-rooted vegetation is released to the lower atmosphere through evapotranspiration and contributes significantly to Cerrado's water balance. Therefore, it is expected that the replacement of woody vegetation by crops and pastures change the hydrological cycle of the Cerrado (Oliveira et al., 2005).

3.2.3. Atlantic Forest

3.2.3.1. LUCC. Even though the Atlantic Forest is recognized as one of the most biodiverse ecoregions in the world, with more than 20,000 plant species, over 1360 vertebrate species and high levels of endemism (Myers et al., 2000), it remains as one of the most threatened tropical forests. Almost 1 million km^2 or 81% of its original extent has been converted with increasing deforestation in Paraguay and Argentina (Table 2).

In Brazil, several studies have estimated a remaining forest cover of between 1% and 12% (Morellato and Haddad, 2000; Oliveira-Filho and Fontes, 2000; Saatchi et al., 2001; Câmara, 2003). Recently, Ribeiro et al. (2009) suggested that the Atlantic Forest extends over between 11% and 16% of its original cover in Brazil, most of which is distributed in small patches in a highly fragmented state. According to MMA/IBAMA (2012), the remaining forest by 2002 was 22% of its former extent.

Deforestation of the Atlantic Forest began with the arrival of European colonizers who exploited the commercially-valuable Brazilwood (*Caesalpinia echinata*) and cleared the rainforests for cropping and human settlements (Câmara, 2003). In the 18th century, the introduction of sugar cane plantations triggered rapid deforestation on fertile soils of the northeastern coast, while the introduction of coffee plantations added further pressures to the forest, particularly during the 19th century (Frickmann, 2003). The expansion of cattle pasture, gold mining and hydroelectric projects is also recognized as an important immediate cause of forest loss, with the former continuing to be an important driver of deforestation (Dean, 1997; Metzger, 2009). More recently, the expansion of urban areas and exotic tree plantations are replacing the remaining forest patches (Metzger, 2009).

Atlantic Forest deforestation in Paraguay began after the 1940s when the establishment of settlements, expansion of the agricultural frontier and the introduction of African grasses for pasture occurred, driving deforestation rates to about 2000 km^2 per year in the 1980s and continuing at 1000 km^2 per year through the 1990s (Cartes, 2003; Catterson and Fragano, 2004). Approximately 25% of the original Paraguayan Atlantic Forest remains (Huang et al., 2007), and soybean plantations are an important factor of recent forest loss, particularly since the 1990s (Richards, 2011). In northern Argentina, the Atlantic Forest is located in the Misiones province and represents the largest remnant of continuous forest (Izquierdo et al., 2008). Covering a former area of about 29,800 km^2 (Chebez and Hilgert, 2003), the Argentinean Atlantic Forest has been affected by soybean plantations, cotton, sugar cane, coffee, and more recently exotic tree plantations represented by *Eucalyptus spp.* and *Pinus spp.* (Plací and Di Bitetti, 2006). Currently, approximately 34% of the original forest extent remains (ca. 10,000 km^2) (Chebez and Hilgert, 2003). Between 1973 and 2006, most of the land cover change (2702 km^2) was characterized by an expansion of exotic tree plantations (Izquierdo et al., 2008), representing, along with human population growth, one of the central threats to the future of the Atlantic Forest in Argentina (Izquierdo et al., 2008, 2011).

Exotic tree plantation expansion could explain the 33,056 km^2 of new forests observed by Hansen et al. (2013) for the period 2000 to

2012, the largest absolute increase in forest cover in non-Amazonian South America (Table 2). Considering the decrease of 44,658 km^2 of all forest cover for the same period, the Atlantic Forest of Argentina, Brazil and Paraguay could be considered as the ecosystem's most affected by intense forestry practices in South America.

3.2.3.2. Impacts on climate. Despite the extent of change in the Atlantic Forest, studies addressing related climatic impacts are extremely rare. The only study found was conducted by Webb et al. (2005), who analyzed weather station data to evaluate the potential effect of rainforest clearance on rainfall in the State of São Paulo in Brazil. Although no strong relationships were observed between forest cover and total rainfall, tree cover was significantly correlated with the number of rainy days and with interannual rainfall variability, with more fragmented forests associated to fewer rain days. Webb et al. (2005) argue that large scale factors independent of vegetation cover, such as coastal weather fronts, control the total amount of rainfall in the study area. However, local geographical features (e.g., topography) together with tree cover explain the number of days over which rain falls.

3.2.4. Temperate Grasslands

3.2.4.1. LUCC. Since the arrival of Europeans in the early 16th century (Báez, 1944), large areas of grasslands in Argentina and Uruguay have been converted into crops and pastures. In the last decades, technological improvements, global food, timber and energy demand and climate changes have intensified LUCC in the remaining native grasslands, which have been converted to annual crops such as soybean, maize, sunflowers, wheat and oats at increasing rates in response to demand from Asia (Zak et al., 2008) and more recently to tree plantations (Nosetto et al., 2012). This trend is partially explained by an increase in rainfall with subsequent replacement of natural grassland located in more humid areas (Pérez and Sierra, 2012).

Conversion of Temperate Grasslands to fast growing *Pinus* and *Eucalyptus* plantations in Argentina and Uruguay has increased rapidly during recent decades, expanding from 23,000 ha in 1992 to 125,000 ha in 2001 (Paruelo et al., 2006; Silveira and Alonso, 2009; Nosetto et al., 2012). Between 2000 and 2012, 13,859 km^2 of new forested area was added to the Temperate Grasslands (Hansen et al., 2013).

3.2.4.2. Impacts on climate

3.2.4.2.1. Modeling studies. LUCC in the Temperate Grasslands involves conversion of natural grasslands (C3 and C4 photosynthetic pathways) to croplands and exotic tree plantations (Baldi et al., 2008a, 2008b). Climatic consequences of these changes have been mainly addressed through regional climate models. In central Argentina, the nation's most important agricultural region, the cooling trend observed by Rusticucci and Barrucand (2004) has been linked to albedo changes as a result of the conversions of natural grasslands by croplands (Beltrán-Przekurat et al., 2012). However, the temperature response depends on whether C3 or C4 grasslands are converted. A cooling effect results from converting C3 grasslands and a warming from converting C4 grasslands, arguably explained by differences in evapotranspiration rates. In addition, changes in precipitation were related with those areas where land cover change occurred, particularly during dry years (Beltrán-Przekurat et al., 2012). The climatic response of LUCC in the Temperate Grasslands seems to be sensitive to how vegetation is described in the land surface component of the climate model used. Though this makes modeling comparison more difficult, it gives insights of land surface feedbacks under different LUCC scenarios in terms of changes in the water and the energy budget. For instance, Lee and Berbery (2011) modeled increases in the near-surface temperature when grasslands were replaced by dry croplands in lower La Plata Basin, with the effects extending beyond the areas where the changes occurred. These changes were associated with alterations in heat fluxes after slight reductions of roughness length and low level wind acceleration that determine net positive effects over precipitation. As the authors discuss, their results are not directly comparable with those from Beltrán-Przekurat et al. (2012) because of differences in the vegetation cover inputs of the regional climate models which affect the resulting biophysical processes.

3.2.4.2.2. Observational studies. Observations of albedo changes in the Temperate Grasslands using remote sensing techniques have reported albedo increases up to 16% between years 2000 and 2008, with agricultural expansion and reduced surface water recognized as the main drivers of these increments (Loarie et al., 2011a). The link between changes in albedo and observed net cooling for central Argentina (Rusticucci and Barrucand, 2004; Nuñez et al., 2008) has been recently proposed in the literature (e.g. Beltrán-Przekurat et al., 2012). However, further research is necessary to relate changes in albedo induced by LUCC and observed temperature trends in the region, especially to separate out the effects of global warming and LUCC.

3.2.5. Chilean Matorral

3.2.5.1. LUCC. Patterns and rates of land cover change in the Chilean Matorral are similar to those in the other areas of non-Amazonian South America. Before European settlement, human populations were restricted to coastal areas and river basins with limited impacts on natural vegetation (represented by localized fire) (Armesto et al., 2010). After the arrival of European colonizers, but intensified after the country's independence, extensive loss of forests occurred due to a massive demand of timber extraction for mining, agriculture and cattle grazing (Armesto et al., 2010). From the 1970s, government subsidies for agriculture and exotic forest plantations were responsible for the loss of 42% of native forests between 1975 and 2008 (Niklitschek, 2007; Schulz et al., 2010). These subsidies particularly impacted the sclerophyllous and temperate forests of central and southern Chile (Echeverria et al., 2006, 2008). In the same period, a large proportion of forests were converted into a savanna dominated by the invasive species *Acacia caven*, which is now the most common land cover type in Central Chile (Schulz et al., 2010; Van de Wouw et al., 2011). Recently, the Chilean Matorral, as in the Atlantic Forest and the Temperate Grasslands, has shown a high dynamic forest cover. Between 2000 and 2012, 2127 km^2 and 3065 km^2 of forest were loss and gain, respectively.

3.2.5.2. Impacts on climate

3.2.5.2.1. Modeling studies. LUCC climate interaction studies in the Chilean Matorral are almost absent. Most of these derive from modeling approaches. In Central Chile, the work of Beltrán-Przekurat et al. (2012) found a warmer and drier climate after the conversion of wooded grassland to croplands (wheat), with increased Bowen ratio in spring. However, former vegetation used by Beltrán-Przekurat et al. (2012) does not agree with vegetation classifications in the Chilean Matorral, previously composed by sclerophyllous forests and shrublands with different physiological features and vegetative characteristics (e.g., leaf area index). Most of this vegetation change has been for irrigated crops (Schulz et al., 2010) and only a few available studies report the atmospheric effects resulting from this change. For instance, in the northern Chilean Matorral, Montecinos et al. (2008) used a mesoscale climate model to evaluate the impacts of irrigated agriculture on the local meteorological variables in the semiarid Elqui valley. In this area, the increased soil moisture along the valley's floor due to irrigation facilitates the transport of moist air through advection into the surrounding areas. Moreover, evapotranspirative changes in specific humidity and temperature on the valley's floor increase the relative humidity which in turn can induce fog formation both in early morning and late afternoon. These impacts are not restricted to the irrigated areas alone, but also influence the energy balance components in the surrounding hillsides by modifying thermally-induced winds

(Montecinos et al., 2008). To date, there is no evidence of the potential climatic impacts of LUCC in other areas of the Chilean Matorral.

3.2.5.2.2. Observational studies. The modeling study of Montecinos et al. (2008) agrees with observations from eddy covariance stations in the same valley, where the Bowen ratio of irrigated fields is more than ten times lower than the surrounding semiarid natural vegetation. This is related to the strong differences in the radiation and energy balance between the two land cover types as well as in the increased evapotranspiration caused by irrigation (Kalthoff et al., 2006).

3.2.6. Tropical Dry Forests

3.2.6.1. LUCC. Tropical Dry Forests are considered as one of the most threatened ecosystems in the Neotropics (Pennington et al., 2006). Originally comprising a large and contiguous forest from Mexico to Bolivia, current Tropical Dry Forests in South America are distributed in small patches covering approximately 34% of their former extent (Sánchez-Azofeifa and Portillo-Quintero, 2011). This vegetation type is associated with fertile soils and therefore is one of the most impacted by crop and livestock production (Pennington et al., 2000). Today, cattle ranching, cropping, timber plantations and fuel-wood extraction are important drivers of forest loss (Miles et al., 2006). In Venezuela, for example, only 15% of the original Tropical Dry Forests remains after cattle ranching and agriculture development, with urbanization and fire that are also representing important factors of deforestation (Fajardo et al., 2005). In Colombia, Tropical Dry Forests are one of the most historically impacted ecosystems (Etter et al., 2008). Currently, <1.5% of the original Colombian Tropical Dry Forests remains, although some degree of recovery has been recently observed (Sánchez-Cuervo et al., 2012). In Bolivia, the Chiquitano dry forests, the largest extant areas of dry forests in South America, are considered as one of the most endangered ecoregions in the Neotropics (Dinerstein et al., 1995) with deforestation rates reaching 80,000 ha per year near the city of Santa Cruz as a result of agriculture expansion, highway construction, gas pipelines and mining (Killeen et al., 1998). Deforestation rates between 3 and 5% per year have been reported for the Chiquitano area of Bolivia (Steininger et al., 2001; Mertens et al., 2004). In the Brazilian Caatinga, 30–52% of dry forests have been altered by human activities, ranking third as the most degraded and destroyed ecosystem in Brazil after the Atlantic Forest and the Cerrado (Leal et al., 2005). In southwestern Ecuador, <1% of the formerly Tropical Dry Forests area (about 28,000 km^2) remains in an undisturbed state (Dodson and Gentry, 1991). Similarly, it is estimated that 95% of the original Tropical Dry Forests in Perú has been converted to human land uses such as those mentioned before (Sánchez-Azofeifa and Portillo-Quintero, 2011).

3.2.6.2. Impacts on climate

3.2.6.2.1. Modeling studies. Despite the extensive conversion of Dry Tropical Forests, evidences of LUCC influences on surface climate are restricted to the Chiquitano dry forest in Bolivia and the Caatinga in north-eastern Brazil, all of them presented through modeling studies. In the Caatinga of northeastern Brazil, early studies using climate models have shown changes in the land surface moisture budget and albedo impacts on near atmosphere, particularly regarding to adiabatic heating and precipitation (Sud and Fennessy, 1982). More recently, some studies have found potential climatic influences of desertification in the Caatinga. According to Oyama and Nobre (2004), desertification (change from xerophytic vegetation to bare soil) may weaken the hydrological cycle, with a strong decrease in precipitation, evapotranspiration, atmospheric moisture convergence and runoff, and an additional increase in surface temperature. In agreement with Oyama and Nobre (2004), Hirota et al. (2011) modeled negative precipitation anomalies as a result of Caatinga desertification, affecting also neighboring northwestern Amazon. Recently, Castilho de Souza and Oyama (2011), using a regional climate atmospheric model, assessed progressive desertification influences on climate in the Caatinga, with similar results as Oyama and Nobre (2004). It has been stated that the expansion of desert areas could feedback upon their selves through radiative and heat alterations (Adams, 2007) such as those reported for the Caatinga.

In the Chiquitano dry forest near to the city of Santa Cruz, Bounoua et al. (2004) applied a land surface model to evaluate the sensitivity of local climate to recent vegetation change using the Simple Biosphere Model (SiB2) (Sellers et al., 1996). They found an increase of 0.6 °C in surface temperature when broadleaf dry forest was converted to cropland. This warming was associated to morphological changes such as a decreased surface roughness, increased aerodynamic resistance and decreased stomatal conductance (Table 1 in Supplementary Information). These variations reduced latent heat flux and increased canopy sensible heat flux, and consequently temperature. Similarly, conversions from wooded grasslands to croplands in the Chiquitano produced an increase in mean temperature of 1.5 °C. This warming was driven by physiological changes when C4 wooded grasslands were replaced by C3 croplands, reducing canopy conductance by approximately 50% (Bounoua et al., 2004).

4. Discussion

4.1. Amazon bias

The total loss of natural vegetation in non-Amazonian South America is estimated in 3.6 million km^2. This area is 4 times greater than the historic Amazon deforestation and equivalent to 37% of U.S. land mass or 3 times the total surface area of Germany, France and United Kingdom. The region has experienced consistent LUCC pressures since European colonization, which are expected to increase in the coming years due to advances in technology, access to former remote areas and an increasing global demand of food commodities and biofuels (Liverman and Vilas, 2006).

It is important to note that present vegetation areas estimated here are not accurate because of assumptions made in calculating areas for potential historic natural vegetation that, for example, did not consider vegetation heterogeneity (e.g. grasslands distributed in the Dry Chaco) and imprecisions of satellite images used to obtain the area of the vegetation types. For the last, errors depend, among other factors, on the training data used and/or the ability of the algorithm to differentiate between two classes with similar spectral signatures. For example, the image from Friedl et al. (2010) used to calculate the current evergreen broadleaf forest extent in the Amazon and Atlantic Forest has a producer and user accuracy of 93% and 83%, respectively, which is high compared to the accuracy of all categories in the MODIS product (75%). However, for grasslands (used to calculate vegetation in Temperate Grasslands) the classification from Friedl et al. (2010) shows a moderate producer accuracy (74%) and low user accuracy (60%), which reflects the difficulties to distinguish this class from others, particularly open shrublands and croplands (Friedl et al., 2010). Notwithstanding, Friedl et al. (2010) was the only available dataset that allowed us to calculate present vegetation cover for the Amazon, Atlantic Forest and Temperate Grasslands. In the case of the Dry Chaco, Clark et al. (2010) developed a robust methodology to classify forest (woody vegetation) at 250 m resolution using training datasets taken from high-resolution Quick Bird imagery in Google Earth. Their producer and user accuracy was 96% and 85%, respectively, which is higher than that obtained by Friedl et al. (2010) for deciduous broadleaf forest (69% and 76%) described for the Dry Chaco at 500 m resolution. High accuracy was obtained for the Caatinga and Cerrado with overall classification accuracy between 92% and 97%, respectively (MMA/IBAMA, 2011a, 2011b).

Despite the large magnitude of the natural vegetation loss showed by the various remote sensing approaches for South America, the number of publications addressing later modifications of surface atmospheric feedbacks is relatively very low. Compared to the Amazon region, non-Amazonian South America registers far less peer-reviewed publications in the field of surface atmospheric processes and feedbacks. Although there is some observational and model-based evidence of the LUCC effects on surface temperature and precipitation, changes in atmospheric circulation and links with climate extremes are barely known. In terms of the impacts of LUCC on climate, non-Amazonian South America remains as one of the least studied regions worldwide.

4.2. Patterns and processes of change

By changing the land cover we are modifying land surface attributes that are important in the exchange of heat and momentum between earth's surface and the atmosphere. These alterations ultimately modify moisture and energy budgets and with them surface temperature and precipitation (Bonan, 2008; Pielke et al., 2011; Mahmood et al., 2014). In non-Amazonian South America, evidence suggests that LUCC, expressed as replacement of native forests, savannas and grasslands by crops and pastures, is associated to changes in surface temperature and precipitation (Fig. 4 summarizes the main impacts of LUCC on climate in non-Amazonian South America). These responses are mainly driven by a decline in evapotranspirative cooling due to differences in morphological attributes that influence evapotranspiration and land–atmosphere coupling, including leaf area index, roughness length and rooting depth between natural and non-natural vegetation, particularly in the Dry Chaco and the Cerrado. Other modeling and observational studies have reported similar patterns of temperature and rainfall response after land cover change in dry forests and savanna type biomes. In Australia, replacement of woody vegetation by agriculture was related to increments in surface temperature between 0.4 °C to 2 °C and lower summer rainfall (McAlpine et al., 2007). As in the Dry Chaco dry forests, the replacement of dry forest to crops and grasslands in southwest Western Australia also explains 50% of the observed warming (Pitman et al., 2004). Similar to the Cerrado (Fig. 4b), in the savannas of Australia, southern Africa and northern South America, Hoffmann and Jackson (2000) modeled increases in surface temperature of 0.5 °C and 10% decreases in rainfall when natural vegetation was converted to grasslands, primarily because of reductions in surface roughness length. This pattern was also described by Snyder et al. (2004), who verified that the removal of the world's savannas caused large reductions in precipitation and surface warming due to reduced latent heat.

In non-Amazonian South America, significant albedo variations associated to LUCC were reported (e.g., Loarie et al., 2011a). These variations are of particular interest in semiarid environments, where albedo enhancement has been linked to precipitation suppression via subsidence anomalies (Otterman, 1989). In the Caatinga of northeastern Brazil, desertification increases albedo and therefore diminishes moisture convergence and precipitation, creating a positive feedback that limits moisture recycling in desertified areas (Sud and Fennessy, 1982; Oyama and Nobre, 2004) (Fig. 4e). This mechanism has been described in the early study of Charney (1975) for the Sahel region, where strong increments in albedo after degradation of vegetation produced a sinking motion and an additional drying that would

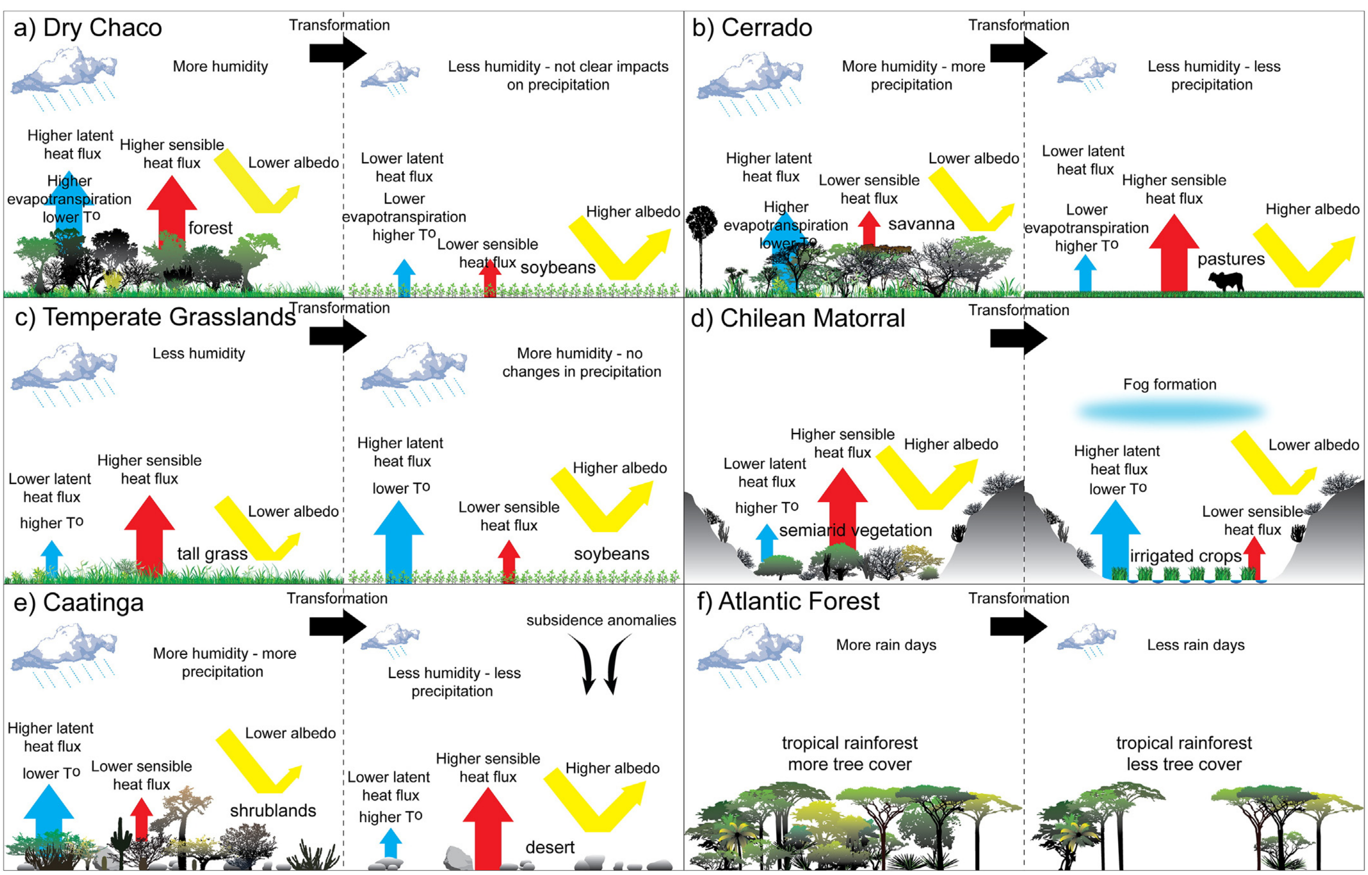

Fig. 4. Summary of the main climatic impacts of LUCC in non-Amazonian South America based on the literature reviewed. Impacts are represented by variations in heat fluxes, temperature, precipitation and albedo, following transformation of natural vegetation to other land uses. The studies of Beltrán-Przekurat et al. (2012); Houspanossian et al. (2013) were used for the Dry Chaco, Loarie et al. (2011b); Costa and Pires (2010) for the Cerrado, Beltrán-Przekurat et al. (2012) for the Temperate Grasslands, Montecinos et al. (2008) for the Chilean Matorral, Castilho de Souza and Oyama (2011) for semiarid Caatinga, and Webb et al. (2005) for the Atlantic Forest. In the Chilean Matorral, changes in precipitation are not shown because they were not evaluated by Montecinos et al. (2008). Similarly, heat fluxes and albedo are not displayed for the Atlantic Forest because Webb et al. (2005) evaluated changes in precipitation only. The trends in climatic responses shown here (heat exchange, temperature, precipitation and albedo) vary greatly between studies and model experiments, and therefore must to be taken carefully because they not necessarily represent unidirectional climatic changes after LUCC.

perpetuate the arid conditions. Conversely, insertion of irrigated agriculture in semiarid environments would increase moisture convergence that can enhance cloudiness and precipitation. In the arid extremity of the Chilean Matorral, Montecinos et al. (2008) report that albedo decreases in irrigated cultivated areas compared to the surrounding semiarid vegetation (Fig. 4d). This increased the net radiation over irrigated areas and consequently the available energy to be transferred into the lower atmosphere through latent heat flux. This process leads to net cooling by up to 2 °C for the irrigated valley compared to semiarid vegetation. Similar examples are found in semiarid environments of India and North America, where the effect of irrigation increases soil moisture levels and consequently latent heat flux, cooling the boundary layer over irrigated areas and increasing atmospheric moisture and cloudiness (Kueppers et al., 2007; Roy et al., 2007; Kharol et al., 2013). In California's Central valley, a region with many biogeographical and climatological similarities to the Chilean Matorral (Zedler et al., 1995), increasing evapotranspiration after irrigation significantly impacts the atmospheric circulation and strengthens the hydrological cycle over southwestern United States (Lo and Famiglietti, 2013). Changes in surface climate after introduction of irrigated agriculture have also been reported for the great plains of North America, where observed maximum daytime temperatures over wheat fields are 2.3 °C cooler than the surrounding grasslands in the growing season but 1.61 °C warmer after its harvest (Ge, 2010). These alterations do not only depend on the direction of land cover change but also on crop phenology, atmospheric moisture content and synoptic scale atmospheric circulation (Mahmood et al., 2014).

In the case of the Atlantic Forest, despite showing very similar vegetation features to the Amazon rainforests (D'Almeida et al., 2007), the only observational study found in this review did not show a strong relationship between the amount of total rainfall and deforestation. However, the strong link between deforestation and the amount of rain days (Webb et al., 2005) (Fig. 4f) suggests that hydrological impacts of deforestation need to be addressed by future research. Since the original distribution of the Atlantic Forest covered an area that presently sustains the highest population density in South America and one of the most populated cities in the world (São Paulo), further studies are required to investigate the influence of historic deforestation on patterns of precipitation and temperature, and the potential role of remaining forest patches in the hydrological cycle and water availability in the region, particularly during climate extremes like droughts.

How local changes can affect the climate of nearby regions and influence neighboring and remote areas through teleconnections are still under discussion in the literature (e.g., Pielke et al., 2011). Most of the studies evaluated here assess only the local climate impacts associated with LUCC processes. However, the impacts should be accounted in a concurrent manner as they occur in reality. This is important because results could differ markedly, and synergistic non-linear processes could occur in order to alter the climatic effects of the combination among LUCC processes in a regional perspective (Costa and Pires, 2010; Hirota et al., 2011).

4.3. Modeling vs observational studies

Most of LUCC impacts on climate in non-Amazonian South America have been addressed through modeling approaches including land surface (2 studies) and climate models (13 studies), while few studies were based on observations (4 studies) (Fig. 5). Land surface models can overestimate the impact of deforestation because they do not take into account land surface atmospheric feedbacks (Pielke et al., 2011) and hence climate models are more robust as they incorporate land surface atmosphere interactions and feedbacks. However, since most of modeling experiments use just one climate or surface model it is very hard to identify whether the results are model dependent or are representative of the climate responses to land cover change. In addition, models differ in their settings and description of land surface processes and atmospheric physics. An example is the difference in the characterization of the land surface properties. Climate models use land cover classifications derived from satellite images with varying classification systems that sometimes are not comparable with others. For instance, in the Temperate Grasslands Beltrán-Przekurat et al. (2012) used a classification that differentiate between C3 and C4 grasses, while Lee and Berbery (2011) applied a climate model with a land cover classification that does not distinguish between photosynthetic pathways, which makes it difficult to compare their results and could explain the difference in the sign of change between the two climate models applied for the same region.

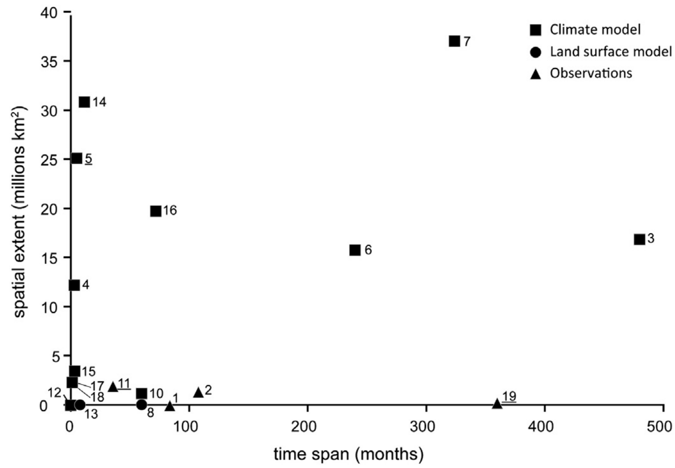

Fig. 5. Spatial and temporal extent of the studies reviewed. Studies approached are categorized as climate models, land surface models and observational studies. Squares represent climate models, circles refer to land surface models and triangles to observations using either satellite images or weather station data. The numbers relate to those shown in the reference column of Table 3 (Mendes et al., 2010 not shown). Underlined numbers correspond to those studies that included autocorrelation in statistical testing.

Another aspect that dampens the reliability of LUCC experiments is the testing for statistical difference. Most of the studies use the Student's t-test that does not take into account autocorrelation issues that affect the independence of climate observations. This problem affects model results because the t-test can overestimate the climate impact of LUCC through Type-I errors or false positives (Zwiers and von Storch, 1995). For non-Amazonian South America, only 3 of 19 studies considered autocorrelation in the statistical analysis and therefore it can be argued that most of the climatic impacts shown could have been overestimated.

4.4. Risks and consequences

According to observations, South America shows different trends in precipitation and temperature depending on the geographic area under analysis. Skansi et al. (2013) describe a wetting trend in many areas of the continent since the mid-20th onwards, mostly in southeastern South America, northern Peru and Ecuador. On the other hand, negative tendencies in evapotranspiration and soil moisture have also been observed between 1982 and 2008 over much of South America (Jung et al., 2010). In southern South America, and contrasting the global tendency, specific humidity has decreased in recent decades (IPCC, 2014a). Central-south Chile and Argentina have registered significant reductions in precipitation and increased surface temperatures (Magrin et al., 2014). Increased warmer days, decreased cold days and more extreme rainfall events have also been observed in many parts of the continent (Alexander et al., 2006; Skansi et al., 2013). These climate extremes (length of drought and/or extreme precipitation events) are projected to increase in future climate change scenarios (Giorgi et al., 2011).

LUCC can exacerbate these regional changes in climate. The removal of native forests and savannas decreases evapotranspiration and moisture flux, enhancing the dryness produced by other drivers of climate change such as increased concentrations of anthropogenic greenhouse gases, with potential increases in extreme events such as droughts and floods. This relationship has been shown in Australia, where LUCC can significantly raise the decile-based drought duration index, particularly during El Niño years (Deo et al., 2009). LUCC can also enhance the negative effects of climate change through alterations in surface hydrology because natural vegetation controls the redistribution of runoff, water table levels and soil moisture by altering soil permeability (D'Odorico et al., 2010; Asbjornsen et al., 2011), which in turn affect water supply for cities, hydropower generation and agriculture (Magrin et al., 2014). This is particularly relevant for semiarid environments of non-Amazonian South America, where drying trends have been observed (Masiokas et al., 2008; Quintana and Aceituno, 2012). Presently, about 200 million people live in these ecosystems (Verbist et al., 2010; WB, 2014), which are already experiencing water stress and reduced agricultural productivity (Magrin et al., 2014). For example, in the Caatinga, current decreased precipitation and river discharge will intensify in the future with strong impacts on crop yields and water security that will force the migration of population from rural to urban areas (Krol and Bronstert, 2007). It is expected that LUCC will intensify these impacts (Montenegro and Ragab, 2010). In the Dry Chaco, increasing precipitation has stimulated the advent of large-scale rainfed crops into areas formerly covered by dry forests (Gasparri and Grau, 2009; Clark et al., 2010). Since this land transformation has locally raised the dryness and surface temperatures (Houspanossian et al., 2013), it could potentially create a negative feedback that will revert the favorable climatic conditions, with significant socioeconomical consequences. In other semiarid environments of non-Amazonian South America such as the central Andes and Chilean Matorral, a weaker hydrological cycle is projected with an associated increased risk of lower water availability (Fiebig-Wittmaack et al., 2012; Vicuña et al., 2012). It is recognized that climate change and LUCC combined with water governance structures, institutional arrangements, societal values and development pathways are the major threats to water security in semiarid South America (Scott et al., 2013).

LUCC climate feedbacks can negatively affect both urban and rural areas. At present, 79% of the South American population live in cities (WB, 2014), many of which are subjected to risks associated with impacts of LUCC, climate change and their feedbacks: increasing flooding and landslides (Andrade and Scarpati, 2007; Marengo et al., 2013), intensification of the heat island effect (Nobre et al., 2011), urban expansion over areas with increased climate risks (e.g. flat terrain), increased food insecurity (Magrin et al., 2014), increase in diseases (CEPAL, 2014) and less water availability (Le Quesne et al., 2009; Little et al., 2009; Winchester and Szalachman, 2009). Compared to urban populations, rural populations show higher poverty levels and therefore are more vulnerable to the adverse impacts of environmental change (IPCC, 2014b). However, increasing evidence shows that both urban and rural population are highly exposed to the negative consequences of climate extreme events and alterations in the moisture budget influenced by LUCC in developing countries (see Magrin et al., 2014 and references therein). Most of the natural disasters in South America between 1972 and 2011 had a hydroclimatic origin, 57% of which were associated with floods, droughts and extreme temperatures (CEPAL, 2014). The potential influences that LUCC could exert upon these events are still not clear.

LUCC in non-Amazonian South America has increased environmental stress and threatens ecosystem resilience. Because of the non-linear relationship between terrestrial ecosystems and climate, changes can exhibit threshold behavior (Zehe and Sivapalan, 2009). These changes may result into an irreversible shift to a drier climate state, in which rainfall would be insufficient to allow for the recovery of ecosystems and their services (Brovkin et al., 1998; Scheffer et al., 2001; Folke et al., 2004).

4.5. Research priorities

Below we outline five key research priories arising out of the findings of the review.

1. *Expand the focus towards understudied regional-scale processes and impacts*. Despite that the high LUCC rates observed in non-Amazonian ecosystems, this paper highlights the need to extend research-oriented activities to quantify the magnitude, climatic consequences and implications of such changes. Examples of these are the Tropical Dry Forests of northern South America, the Cerrado, the Atlantic Forest and the Chilean Matorral. Additional research efforts are required to measure the spatial extent and rate of LUCC and the detection of resulting climatic impacts and risks in these ecosystems. This research effort also needs to be expanded to other regions not included in this review, where evidence of high land cover transformation rates has been reported. Examples of these are the Valdivian Forests, the Llanos Savannas and the Ecuadorian Páramo.
2. *Linking land atmosphere interactions with climate extremes*. There is a need to increase the focus towards the relative contribution of LUCC in regional climate change and its interaction with other forcings such as greenhouse gases. In addition, although climatic extremes are recognized as major threads in South America, there is insufficient evidence of how changes in land cover interact with these climatic phenomena. Though some research has related LUCC with dry/wet El Niño Southern Oscillation conditions (e.g. Beltrán-Przekurat et al., 2012), further research needs to be conducted in order to understand feedbacks and potential societal consequences.
3. *Increasing the surface climate and hydrological observation platform*. It has been recognized that one of the major problems in South America is the lack of long-term homogeneous and continuous climate and hydrological records (Magrin et al., 2014). This makes very difficult to identify historical patterns and trends in local and regional mean climate and in extremes, and hence address hypothesis in relation to the impacts of LUCC over the hydrological cycle. A major investment of resources is required to increase the number and distribution of meteorological and gauge stations, and widen current networks through partnerships between governments, universities, research institutes and programs.
4. *Improving land surface descriptions for regional climate models*. Many of the land surface characterizations used in regional climate models can be improved through the incorporation of more accurate representations of land cover such as different crop varieties, irrigated agriculture, and descriptions of different biomes. In the case of non-Amazonian South America, land surface models embedded in climate models are usually calibrated in regions where the models were developed and do not accurately represent the conditions where such models are applied. Upgrading surface features to local/regional conditions (e.g. leaf area index, vegetation fraction, roughness length, and albedo) will make modeling results more robust.
5. *Statistical testing for LUCC experiments*. There are problems related to the use of discredited statistics to test for differences in LUCC experiments. For example, most of the studies for non-Amazonian South America reviewed used the classical Student's t-test for calculation of differences without considering autocorrelation. Problems associated with it relate to an over-estimation of LUCC impacts on climate, which make results less reliable. The modified Student's t-test (Zwiers and von Storch, 1995) is one of the many available options to overcome this issue.

5. Conclusions

In this study, we reviewed the main patterns of land use and land cover change and subsequent climatic impacts for non-Amazonian South America. Our major findings can be summarized as follows:

- Non-Amazonian South America has been subjected to a consistent historic process of LUCC. Overall, 3.6 million km^2 (58% of the former area) of potential natural vegetation has been converted into anthropogenic land use practices, representing more than 4 times the area of Amazon deforestation, which has lost ~920,000 km^2 or 14% of its former area. The most affected ecosystems are the Chilean Matorral and the Atlantic Forest with 83% (52,000 km^2) and 81% (978,000 km^2) of their former natural vegetation transformed by year 1999 and 2012, respectively. LUCC also affected other ecosystems such as the Cerrado, Temperate Grasslands and Tropical Dry Forests where at least 52% of the original natural vegetation has been converted to anthropogenic land uses. The main drivers behind the conversion of natural vegetation are the expansion of croplands (soybean) and cattle pastures to meet global food demand, technological advances, climatic factors and governmental subsidies to increase production of food commodities.
- Based on the datasets of Hansen et al. (2013) for non-Amazonian South America, the Dry Chaco and the Atlantic Forest showed the highest relative amount of forest loss for the period 2000–2012, followed by the Cerrado and Tropical Dry Forests. While the Dry Chaco deforestation is related to native forest loss, in the other ecosystems, particularly in the Atlantic Forest, the Chilean Matorral and the Temperate Grasslands, forest loss is accompanied by a high proportion of forest gain, suggesting intensive forestry practices (mostly *Eucalyptus* and *Pinus spp*. plantations) as described by local studies.
- Climatic consequences of LUCC based on the studies reviewed are mainly related to an increase in surface temperature and a decrease in precipitation and cloudiness. Even though significant albedo variations are reported, the net change in temperature and precipitation after LUCC is mostly driven by shifts in latent and sensible heat fluxes. However, in semiarid areas albedo seems to play a significant role in reducing precipitation via subsidence anomalies. These impacts can manifest beyond the regions where land cover changes occur and could affect neighboring regions such as the Amazon or even teleconnect beyond South America.
- More studies need to be conducted in order to estimate the magnitude of LUCC in non-Amazonian South America and its related climatic impacts, particularly in the most disturbed and understudied ecosystems. It is also necessary to understand the influence of LUCC on the duration an intensity of climate extremes such as droughts using climate model results supported by increased hydrological and climatic observations. LUCC experiments using such models should be parameterized according to local/regional surface characteristics and appropriate statistical tests need to be applied to make results more robust.

Acknowledgments

This work was funded by the CONICYT PAI/INDUSTRIA 79090016 and CSIRO INRM scholarship (Australia) grant number 605971. This work also benefited from the insights of three anonymous reviewers. Clive McAlpine is supported by an Australian Research Council Future Fellowship (FT100100338).

Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.gloplacha.2015.02.009.

References

Achard, F., et al., 2002. Determination of deforestation rates of the world's humid tropical forests. Science 297 (5583), 999–1002.

Adamoli, J., Sennhauser, E., Acero, J.M., Rescia, A., 1990. Stress and disturbance: vegetation dynamics in the Dry Chaco region of Argentina. J. Biogeogr. 17 (4/5), 491–500.

Adams, J., 2007. The desert makes the desert: climate feedbacks from the vegetation of arid zones vegetation–climate interaction. How Vegetation Makes the Global Environment. Springer Praxis Books. Springer, Berlin Heidelberg, Germany, pp. 101–129.

Alexander, L.V., Alexander, L.V., Zhang, X., Peterson, T.C., Caesar, J., 2006. Global observed changes in daily climate extremes of temperature and precipitation. J. Geophys. Res. 111 (d5).

Alho, C., Martins, E., 1995. De grão em grão, o cerrado perde espaço: cerrado, impactos do processo de ocupação. Fundo Mundial para a Natureza/PROCER, Brasilia, D.F.

Andrade, M.I., Scarpati, O.E., 2007. Recent changes in flood risk in the Gran La Plata, Buenos Aires province, Argentina: causes and management strategy. GeoJournal 70 (4), 245–250.

Armesto, J.J., et al., 2010. From the Holocene to the anthropocene: a historical framework for land cover change in southwestern South America in the past 15,000 years. Land Use Policy 27 (2), 148–160.

Asbjornsen, H., et al., 2011. Ecohydrological advances and applications in plant–water relations research: a review. J. Plant Ecol. 4 (1–2), 3–22.

Avissar, R., Werth, D., 2005. Global hydroclimatological teleconnections resulting from tropical deforestation. J. Hydrometeorol. 6 (2), 134–145.

Báez, J.R., 1944. La primera colonia agro-hispana en el Río de la Plata. Sancti Spiritu, cuna de la agricultura platense. Rev. Argent. Agron. 11, 186–278.

Baldi, G., Guerschman, J., Paruelo, J.M., 2006. Characterizing fragmentation in temperate South America grasslands. Agric. Ecosyst. Environ. 116 (3–4), 197–208.

Baldi, G., et al., 2008a. Long-term satellite NDVI data sets: evaluating their ability to detect ecosystem functional changes in South America. Sensors 8 (9), 5397–5425.

Baldi, G., Nosetto, M., Jobbágy, E., 2008b. El efecto de las plantaciones forestales sobre el funcionamiento de los ecosistemas sudamericanos (The effect of tree plantations on south american ecosystems) functioning. Ambiência 4, 23–34.

Beltrán-Przekurat, A., Pielke Sr., R.A., Eastman, J.L., Coughenour, M.B., 2012. Modeling the effects of land-use/land-cover changes on the near-surface atmosphere in southern South America. Int. J. Climatol. 32 (8), 1206–1225.

Bisigato, A.J., Laphitz, R.M.L., 2009. Ecohydrological effects of grazing-induced degradation in the Patagonian Monte, Argentina. Austral Ecol. 34 (5), 545–557.

Boletta, P.E., Ravelo, A.C., Planchuelo, A.M., Grilli, M., 2006. Assessing deforestation in the Argentine Chaco. For. Ecol. Manag. 228 (1–3), 108–114.

Bonan, G.B., 2008. Forests and climate change: forcings, feedbacks, and the climate benefits of forests. Science 320 (5882), 1444–1449.

Bounoua, L., Defries, R., Collatz, G.J., Sellers, P., Khan, H., 2002. Effects of land cover conversion on surface climate. Clim. Chang. 52 (1–2), 29–64.

Bounoua, L., DeFries, R.S., Imhoff, M.L., Steininger, M.K., 2004. Land use and local climate: a case study near Santa Cruz, Bolivia. Meteorog. Atmos. Phys. 86 (1), 73–85.

Brannstrom, C., et al., 2008. Land change in the Brazilian Savanna (Cerrado), 1986–2002: comparative analysis and implications for land-use policy. Land Use Policy 25 (4), 579–595.

Brovkin, V., Claussen, M., Petoukhov, V., Ganopolski, A., 1998. On the stability of the atmosphere-vegetation system in the Sahara/Sahel region. J. Geophys. Res.-Atmos. 103 (D24), 31613–31624.

Bucher, E.H., Huszar, P.C., 1999. Sustainable management of the Gran Chaco of South America: ecological promise and economic constraints. J. Environ. Manag. 57 (2), 99–108.

Câmara, I.G., 2003. Brief history of conservation of the Atlantic Forest. In: Galindo-Leal, C., Câmara, I.G. (Eds.), The Atlantic Forest of South America: Biodiversity Status, Threats, and Outlook. CABS and Island Press, Washington, DC, pp. 31–42.

Canziani, P.O., Carbajal Benitez, G., 2012. Climate impacts of deforestation/land-use changes in central south america in the PRECIS regional climate model: mean precipitation and temperature response to present and future deforestation scenarios. Sci. World J. 2012, 1–20.

Cartes, J.L., 2003. Brief history of conservation in the Interior Atlantic Forest. In: Galindo-Leal, C., Câmara, I.G. (Eds.), The Atlantic Forest of South America: Biodiversity Status, Threats, and Outlook. CABS and Island Press, Washington, DC, pp. 269–287.

Castilho de Souza, D., Oyama, M.D., 2011. Climatic consequences of gradual desertification in the semi-arid area of Northeast Brazil. Theor. Appl. Climatol. 103 (3), 345–357.

Catterson, T.M., Fragano, F.V., 2004. Tropical Forestry and Biodiversity Conservation in Paraguay: Final Report of a Section 118/119 Assessment EPIQ II Task Order No.1 USAID, Asunción, Paraguay.

CEPAL, 2014. Manual para la evaluación de desastres. United Nations, Santiago de Chile.

Charney, J.G., 1975. Dynamics of deserts and drought in the Sahel. Q. J. R. Meteorol. Soc. 101 (428), 193–202.

Chebez, J.C., Hilgert, N., 2003. Brief history of conservation of the Paraná Forest. In: Galindo-Leal, C., Câmara, I.G. (Eds.), The Atlantic Forest of South America: Biodiversity Status, Threats, and Outlook. CABS and Island Press, Washington, DC, pp. 141–159.

Clark, M.L., Aide, M., Grau, R., Riner, G., 2010. A scalable approach to mapping annual land cover at 250 m using MODIS time series data: a case study in the Dry Chaco ecoregion of South America. Remote Sens. Environ. 114 (11), 2816–2832.

Collins, J.M., Chaves, R.R., da Silva Marques, V., 2009. Temperature variability over South America. J. Clim. 22 (22), 5854–5869.

Conaf, (Corporación Nacional Forestal), Birf, Universidad Australia de Chile, Pointificia Universidad Católica de Chile, Universidad Católica de Temuco, 1999. Catastro y evaluación de los recursos vegetacionales nativos de Chile. Informe Nacional con Variables Ambientales (Santiago, Chile).

Costa, M.H., Pires, G.F., 2010. Effects of Amazon and Central Brazil deforestation scenarios on the duration of the dry season in the arc of deforestation. Int. J. Climatol. 30 (13), 1970–1979.

D'Almeida, C., et al., 2007. The effects of deforestation on the hydrological cycle in Amazonia: a review on scale and resolution. Int. J. Climatol. 27 (5), 633–647.

Dean, W., 1997. With Broadax and Firebrand: The Destruction of the Brazilian Atlantic Forest. University of California Press, Berkeley (482 pp.).

Deo, R.C., et al., 2009. Impact of historical land cover change on daily indices of climate extremes including droughts in eastern Australia. Geophys. Res. Lett. 36 (8), 1–5.

Dinerstein, E.D., et al., 1995. A Conservation Assessment of the Terrestrial Ecoregions of Latin America and the Caribbean. World Wildlife Fund — The World Bank, Washington, D.C. USA.

Diniz-Filho, J.A.F., et al., 2009. Agriculture, habitat loss and spatial patterns of human occupation in a biodiversity hotspot. Sci. Agric. 66, 764–771.

D'Odorico, P., et al., 2010. Ecohydrology of terrestrial ecosystems. Bioscience 60 (11), 898–907.

Dodson, C.H., Gentry, A.H., 1991. Biological extinction in Western Ecuador. Ann. Mo. Bot. Gard. 78 (2), 273–295.

Echeverria, C., et al., 2006. Rapid deforestation and fragmentation of Chilean Temperate Forests. Biol. Conserv. 130 (4), 481–494.

Echeverria, C., Coomes, D.A., Hall, M., Newton, A.C., 2008. Spatially explicit models to analyze forest loss and fragmentation between 1976 and 2020 in southern Chile. Ecol. Model. 212 (3–4), 439–449.

Etter, A., McAlpine, C.A., Possingham, H., 2008. Historical patterns and drivers of landscape change in Colombia since 1500: a regionalized spatial approach. Ann. Assoc. Am. Geogr. 98 (1), 2–23.

Fairman Jr., J.G., Nair, U.S., Christopher, S.A., Mölg, T., 2011. Land use change impacts on regional climate over Kilimanjaro. J. Geophys. Res. 116 (D3), D03110.

Fajardo, L., et al., 2005. Tropical Dry Forests of Venezuela: characterization and current conservation status. Biotropica 37 (4), 531–546.

FAO, 2013. The statistics division. Soybean production. (Data available from). http://faostat.fao.org/.

Farley, K.A., 2007. Grasslands to tree plantations: forest transition in the Andes of Ecuador. Ann. Assoc. Am. Geogr. 97 (4), 755–771.

Feddema, J., et al., 2005. The importance of land-cover change in simulating future climates. Science 310 (5754), 1674–1678.

Fiebig-Wittmaack, M., et al., 2012. Climatic trends and impact of climate change on agriculture in an arid Andean valley. Clim. Chang. 111 (3–4), 819–833.

Folke, C., et al., 2004. Regime shifts, resilience, and biodiversity in ecosystem management. Annu. Rev. Ecol. Evol. Syst. 35 (1), 557–581.

Frickmann, C.E., 2003. Socioeconomic causes of deforestation in the Atlantic Forest of Brazil. In: Galindo-Leal, C., Câmara, I.G. (Eds.), The Atlantic Forest of South America: Biodiversity Status, Threats, and Outlook. CABS and Island Press, Washington, DC, pp. 103–117.

Friedl, M.A., et al., 2010. MODIS Collection 5 global land cover: Algorithm refinements and characterization of new datasets. Remote Sens. Environ. 114 (1), 168–182.

Gasparri, I., Grau, R., 2009. Deforestation and fragmentation of Chaco dry forest in NW Argentina (1972–2007). For. Ecol. Manag. 258 (6), 913–921.

Gasparri, I., Grau, R., Manghi, E., 2008. Carbon pools and emissions from deforestation in extra-tropical forests of Northern Argentina between 1900 and 2005. Ecosystems 11 (8), 1247–1261.

Ge, J., 2010. MODIS observed impacts of intensive agriculture on surface temperature in the southern Great Plains. Int. J. Climatol. 30 (13), 1994–2003.

Georgescu, M., Lobell, D.B., Field, C.B., Mahalov, A., 2013. Simulated hydroclimatic impacts of projected Brazilian sugarcane expansion. Geophys. Res. Lett. 40 (5), 972–977.

Giorgi, F., et al., 2011. Higher hydroclimatic intensity with global warming. J. Clim. 24 (20), 5309–5324.

Gordillo, G., 2010. Historias de los bosques que alguna vez fueron pastizales: la producción de la naturaleza en la frontera Argentino–Paraguaya. Población Soc. 17 (1), 59–79.

Grau, H.R., Aide, T.M., Gasparri, N.I., 2005a. Globalization and soybean expansion into semiarid ecosystems of Argentina. Ambio 34 (3), 265–266.

Grau, R., Gasparri, I., Aide, M., 2005b. Agriculture expansion and deforestation in seasonally dry forests of north-west Argentina. Environ. Conserv. 32 (2), 140–148.

Grimm, A.M., Tedeschi, R.G., 2009. ENSO and extreme rainfall events in South America. J. Clim. 22 (7), 1589–1609.

Hansen, M.C., et al., 2013. High-Resolution Global Maps of 21st-Century Forest Cover Change. Science, 342(6160), 850–853. (Data available on-line from). http://earthenginepartners.appspot.com/science-2013-global-forest.

Hidalgo, J., Masson, V., Gimeno, L., 2010. Scaling the daytime urban heat island and urban-breeze circulation. J. Appl. Meteorol. Climatol. 49 (5), 889–901.

Hirota, M., Oyama, M.D., Nobre, C., 2011. Concurrent climate impacts of tropical South America land-cover change. Atmos. Sci. Lett. 12 (3), 261–267.

Hoffmann, W.A., Jackson, R.B., 2000. Vegetation–climate feedbacks in the conversion of tropical savanna to grassland. J. Clim. 13 (9), 1593–1602.

Houspanossian, J., Nosetto, M., Jobbágy, E., 2013. Radiation budget changes with dry forest clearing in temperate Argentina. Glob. Chang. Biol. 19 (4), 1211–1222.

Hoyos, L.E., et al., 2013. Deforestation and precipitation patterns in the arid Chaco forests of central Argentina. Appl. Veg. Sci. 16 (2), 260–271.

Huang, C., et al., 2007. Rapid loss of Paraguay's Atlantic forest and the status of protected areas — A Landsat assessment. Remote Sens. Environ. 106 (4), 460–466.

Huang, C., et al., 2009. Assessment of Paraguay's forest cover change using Landsat observations. Glob. Planet. Chang. 67 (1–2), 1–12.

Huber, A., Iroume, A., Bathurst, J., 2008. Effect of Pinus radiata plantations on water balance in Chile. Hydrol. Process. 22 (1), 142–148.

IBGE, 2012. Manual técnico da vegetação Brasileira. Second ed. (Available at) ftp://geoftp.ibge.gov.br/documentos/recursos_naturais/manuais_tecnicos/manual_tecnico_vegetacao_brasileira.pdf, Rio de Janeiro.

IPCC, 2014a. Observations: Atmosphere and Surface, Climate Change 2013: The Physical Science Basis. Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel of Climate Change. Cambridge University Press, New York, p. 1535.

IPCC, 2014b. Rural Areas. In: Barros, V.R., et al. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA (XXX pp.).

Izquierdo, A., De Angelo, C., Aide, T., 2008. Thirty years of human demography and land-Use change in the Atlantic forest of Misiones, Argentina: an evaluation of the forest transition model. Ecol. Soc. 13 (2).

Izquierdo, A., Grau, R., Aide, T., 2011. Implications of rural–urban migration for conservation of the Atlantic forest and urban growth in Misiones, Argentina (1970–2030). Ambio: J. Hum. Environ. 40 (3), 298–309.

Jepson, W., Brannstrom, C., Filippi, A., 2010. Access regimes and regional land change in the Brazilian Cerrado, 1972–2002. Ann. Assoc. Am. Geogr. 100 (1), 87–111.

Jobbagy, E.G., Jackson, R.B., 2007. Groundwater and soil chemical changes under phreatophytic tree plantations. J. Geophys. Res. Biogeosci. 112 (G2).

Jobbágy, E., Baldi, G., Nosetto, M., 2012. Tree plantation in south america and the water cycle: impacts and emergent opportunities. In: Schlichter, T., Montes, L. (Eds.), Forests in Development: A Vital Balance. Springer, Netherlands, pp. 53–63.

Joly, C.A., Metzger, J.P., Tabarelli, M., 2014. Experiences from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. New Phytol. 204 (3), 459–473.

Jung, M., et al., 2010. Recent decline in the global land evapotranspiration trend due to limited moisture supply. Nature 467 (7318), 951–954.

Kalthoff, N., et al., 2006. The energy balance, evapo-transpiration and nocturnal dew deposition of an arid valley in the Andes. J. Arid Environ. 65 (3), 420–443.

Kharol, S.K., Kaskaoutis, D.G., Badarinath, K.V.S., Sharma, A.R., Singh, R.P., 2013. Influence of land use/land cover (LULC) changes on atmospheric dynamics over the arid region of Rajasthan state, India. J. Arid Environ. 88, 90–101.

Killeen, T.J., Jardim, A., Mamani, F., Rojas, N., 1998. Diversity, composition and structure of a tropical semideciduous forest in the Chiquitanía region of Santa Cruz, Bolivia. J. Trop. Ecol. 14 (06), 803–827.

Klink, C.A., Moreira, A.G., 2002. Past and current human occupation, and land use. In: Oliveira, P.S., Marquis, R.J. (Eds.), The Cerrados of Brazil: Ecology and Natural History of a Neaotropical Savanna. Columbia Univeristy Press, New York, pp. 69–88.

Knapp, G., 2007. The legacy of european colonialism. In: Veblen, T.T., Young, K.R., Orme, A.R. (Eds.), The Physical Geography of South America. Oxford University Press, New York, p. 361.

Krol, M.S., Bronstert, A., 2007. Regional integrated modeling of climate change impacts on natural resources and resource usage in semi-arid Northeast Brazil. Environ. Model Softw. 22 (2), 259–268.

Kueppers, L.M., Snyder, M.A., Sloan, L.C., 2007. Irrigation cooling effect: regional climate forcing by land-use change. Geophys. Res. Lett. 34 (3), L03703.

Lawrence, P.J., et al., 2012. Simulating the biogeochemical and biogeophysical impacts of transient land cover change and wood harvest in the community climate system model (CCSM4) from 1850 to 2100. J. Clim. 25 (9), 3071–3095.

Le Quesne, C., Acuña, C., Boninsegna, J.A., Rivera, A., Barichivich, J., 2009. Long-term glacier variations in the Central Andes of Argentina and Chile, inferred from historical records and tree-ring reconstructed precipitation. Palaeogeogr. Palaeoclimatol. Palaeoecol. 281 (3–4), 334–344.

Leal, I.R., Da Silva, J.M.C., Tabarelli, M., Lacher, T.E., 2005. Changing the course of biodiversity conservation in the Caatinga of northeastern Brazil. Conserv. Biol. 19 (3), 701–706.

Lee, S.-J., Berbery, E.H., 2011. Land cover change effects on the climate of the La Plata Basin. J. Hydrometeorol. 13 (1), 84–102.

Lee, J.-E., Lintner, B.R., Boyce, C.K., Lawrence, P.J., 2011. Land use change exacerbates tropical South American drought by sea surface temperature variability. Geophys. Res. Lett. 38 (19), L19706.

Little, C., Lara, A., McPhee, J., Urrutia, R., 2009. Revealing the impact of forest exotic plantations on water yield in large scale watersheds in South-Central Chile. J. Hydrol. 374 (1–2), 162–170.

Liverman, D., Vilas, S., 2006. Neoliberalism and the Environment in Latin America. Annu. Rev. Environ. Resour. 31 (1), 327–363.

Lo, M.-H., Famiglietti, J.S., 2013. Irrigation in California's Central Valley strengthens the southwestern U.S. water cycle. Geophys. Res. Lett. 40 (2), 301–306.

Loarie, S., Lobell, D., Asner, G., Field, C., 2011a. Land-cover and surface water change drive large albedo increases in South America. Earth Interact. 15 (7), 1–16.

Loarie, S., Lobell, D., Asner, G., Mu, Q., Field, C., 2011b. Direct impacts on local climate of sugar-cane expansion in Brazil. Nat. Clim. Chang. 1 (2), 105–109.

Luebert, F., Pliscoff, P., 2006. Sinopsis bioclimática y vegetacional de Chile. Editorial Universitaria, Santiago de Chile (316 pp.).

Machado, R.B., et al., 2004. Estimativas de Perda da Área do Cerrado Brasileiro. Relatório técnico não publicado, Conservação Internacional, Brasilia, D.F.

Magrin, G.O., Marengo, J.A., Boulanger, J.-P., Buckeridge, M.S., Castellanos, E., Poveda, G., Scarano, F.R., Vicuña, S., 2014. Central and South America. In: Barros, V.R., Field, C.B., Dokken, D.J., Mastrandrea, M.D., Mach, K.J., Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B., Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., White, L.L. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 1499–1566.

Mahmood, R., et al., 2010. Impacts of land Use/land cover change on climate and future research priorities. Bull. Am. Meteorol. Soc. 91 (1), 37–46.
Mahmood, R., et al., 2014. Land cover changes and their biogeophysical effects on climate. Int. J. Climatol. 34 (4), 929–953.
Marengo, J.A., Valverde, M.C., Obregon, G.O., 2013. Observed and projected changes in rainfall extremes in the Metropolitan Area of São Paulo. Clim. Res. 57 (1), 61–72.
Marris, E., 2005. The forgotten ecosystem. Nature 437 (7061), 944–945.
Masiokas, M.H., et al., 2008. 20th-century glacier recession and regional hydroclimatic changes in northwestern Patagonia. Glob. Planet. Chang. 60 (1–2), 85–100.
McAlpine, C.A., et al., 2007. Modeling the impact of historical land cover change on Australia's regional climate. Geophys. Res. Lett. 34 (22).
McGuffie, K., Henderson-Sellers, A., Zhang, H., Durbidge, T.B., Pitman, A.J., 1995. Global climate sensitivity to tropical deforestation. Glob. Planet. Chang. 10 (1–4), 97–128.
Mendes, A.C., Pires, G.F., Costa, M.H., 2010. Cerrado conservation is essential to protect the Amazon rainforest. Ambio 39 (8), 580–584.
Mertens, B., Kaimowitz, D., Puntodewo, A., Vanclay, J., Mendez, P., 2004. Modeling deforestation at distinct geographic scales and time periods in Santa Cruz, Bolivia. Int. Reg. Sci. Rev. 27 (3), 271–296.
Metzger, J.P., 2009. Conservation issues in the Brazilian Atlantic forest. Biol. Conserv. 142 (6), 1138–1140.
Miles, L., et al., 2006. A global overview of the conservation status of tropical dry forests. J. Biogeogr. 33 (3), 491–505.
MMA/IBAMA, 2011a. Monitoramento do desmatamento nos biomas brasileiros por satélite. Acordo de cooperação técnica MMA/IBAMA. Monitoramento do bioma Caatinga. (Available at), http://siscom.ibama.gov.br/monitorabiomas/caatinga/relatorio_tecnico_caatinga_2008-2009.pdf (Brasília).
MMA/IBAMA, 2011b. Monitoramento do desmatamento nos biomas brasileiros por satélite. Acordo de cooperação técnica MMA/IBAMA. Monitoramento do bioma Cerrado. (Available at), http://siscom.ibama.gov.br/monitorabiomas/cerrado/RELATORIO%20FINAL_CERRADO_2010.pdf (Brasília).
MMA/IBAMA, 2012. Monitoramento do desmatamento nos biomas brasileiros por satélite. Acordo de cooperação técnica MMA/IBAMA. Monitoramento do bioma Mata Atlantica. (Available at), http://siscom.ibama.gov.br/monitorabiomas/mataatlantica/RELATORIO%20MATA%20ATLANTICA%202008%202009.pdf (Brasília).
Mohamed, A., et al., 2011. The influence of large dams on surrounding climate and precipitation patterns. Geophys. Res. Lett. 38 (4), L04405.
Montecinos, S., et al., 2008. The impact of agricultural activities on Fog formation in an arid zone of Chile. Erde 139 (1–2), 77–95.
Montenegro, A., Ragab, R., 2010. Hydrological response of a Brazilian semi-arid catchment to different land use and climate change scenarios: a modeling study. Hydrol. Process. 24 (19), 2705–2723.
Morellato, L.P.C., Haddad, C.F.B., 2000. Introduction: the Brazilian Atlantic Forest. Biotropica 32 (4b), 786–792.
Morello, J., Adamoli, J., 1974. Las grandes unidades de vegetación y ambiente del Chaco Argentino. Segunda Parte: Vegetación y Ambiente de la Provincia del Chaco, INTA Serie Fitogeográfica.
Myers, N., Mittermeier, R.A., Mittermeier, C.G., Gustavo, A.B.d.F., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403 (6772), 853–858.
Niklitschek, M.E., 2007. Trade liberalization and land Use changes: explaining the expansion of afforested land in Chile. For. Sci. 53 (3), 385–394.
Nobre, C.A., et al., 2011. Vulnerabilidade das megacidades Brasileiras as mudanças climáticas: Região Metropolitana de São Paulo. Relatório Final, INPE-UNICAMP-USP-IPTEUNESP, São Paulo, Brasil.
Nosetto, M., Jobbágy, E., Brizuela, A., Jackson, R., 2012. The hydrologic consequences of land cover change in central Argentina. Agric. Ecosyst. Environ. 154, 2–11.
Nuñez, M.N., Ciapessoni, H.H., Rolla, A., Kalnay, E., Cai, M., 2008. Impact of land use and precipitation changes on surface temperature trends in Argentina. J. Geophys. Res.-Atmos. 113 (D6).
Oliveira, R.S., et al., 2005. Deep root function in soil water dynamics in cerrado savannas of central Brazil. Funct. Ecol. 19 (4), 574–581.
Oliveira-Filho, A.T., Fontes, M.A.L., 2000. Patterns of floristic differentiation among Atlantic Forests in southeastern Brazil and the influence of climate. Biotropica 32 (4b), 793–810.
Olson, D.M., et al., 2001. Terrestrial ecoregions of the world: a new map of life on earth. Bioscience 51 (11), 933–938.
Otterman, J., 1989. Enhancement of surface–atmosphere fluxes by desert-fringe vegetation through reduction of surface albedo and of soil heat flux. Theor. Appl. Climatol. 40 (1), 67–79.
Oyama, M.D., Nobre, C.A., 2004. Climatic consequences of a large-scale desertification in northeast Brazil: a GCM simulation study. J. Clim. 17 (16), 3203–3213.
Paruelo, J., Jobbagy, E., Sala, O.E., 2001. Current distribution of ecosystem functional types in temperate South America. Ecosystems 4 (7), 683–698.
Paruelo, J., et al., 2006. Cambios en el uso de la tierra en Argentina y Uruguay: marcos conceptuales para su análisis. Interciencia 10 (2), 47–61.
Pennington, T.R., Prado, D.E., Pendry, C.A., 2000. Neotropical seasonally dry forests and Quaternary vegetation changes. J. Biogeogr. 27 (2), 261–273.
Pennington, T.R., Lewis, G.P., Ratter, J.A., 2006. Neotropical Savannas and Seasonally Dry Forests: Plant Diversity, Biogeography and Conservation. CRC Press, Florida (509 pp.).
Pérez, S., Sierra, E., 2012. Changes in rainfall patterns in the eastern area of La Pampa Province, Argentina. Ambi-Agua 7 (1), 24–35.
Pielke, R., et al., 2002. The influence of land-Use change and landscape dynamics on the climate system: relevance to climate-change policy beyond the radiative effect of greenhouse gases. Phil. Trans.: Math. Phys. Eng. Sci. 360 (1797), 1705–1719.
Pielke, R., et al., 2011. Land use/land cover changes and climate: modeling analysis and observational evidence. Wiley Interdiscip. Rev. Clim. Chang. 2 (6), 828–850.
Pires, G.F., Costa, M.H., 2013. Deforestation causes different subregional effects on the Amazon bioclimatic equilibrium. Geophys. Res. Lett. 40 (14), 3618–3623.
Pitman, A.J., Narisma, G.T., Pielke, R.A., Holbrook, N.J., 2004. Impact of land cover change on the climate of southwest Western Australia. J. Geophys. Res.-Atmos. 109 (D18).
Plací, G., Di Bitetti, M., 2006. Situación ambiental en la Ecoregión del Bosque Atlántico del Alto Paraná (Selva Pranaense). In: Brown, A., Martínez Ortiz, U., Acerbi, M., Corcuera, J. (Eds.), La Situación Ambiental Argentina 2005. Fundación Vida Silvestre Argentina, Buenos Aires, Argentina.
Pongratz, J., et al., 2006. The impact of land cover change on surface energy and water balance in Mato Grosso, Brazil. Earth Interact. 10 (19), 1–17.
Portillo-Quintero, C.A., Sánchez-Azofeifa, G.A., 2010. Extent and conservation of tropical dry forests in the Americas. Biol. Conserv. 143 (1), 144–155.
Portillo-Quintero, C.A., Sanchez, A.M., Valbuena, C.A., Gonzalez, Y.Y., Larreal, J.T., 2012. Forest cover and deforestation patterns in the Northern Andes (Lake Maracaibo Basin): a synoptic assessment using MODIS and Landsat imagery. Appl. Geogr. 35 (1–2), 152–163.
Quintana, J.M., Aceituno, P., 2012. Changes in the rainfall regime along the extratropical west coast of South America (Chile): 30–43°. S. Atmosfera 25 (1), 1–12.
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. Biol. Conserv. 142 (6), 1141–1153.
Richards, P.D., 2011. Soy, cotton, and the final Atlantic Forest frontier. Prof. Geogr. 63 (3), 343–363.
Richards, P.D., Myers, R.J., Swinton, S.M., Walker, R.T., 2012. Exchange rates, soybean supply response, and deforestation in South America. Glob. Environ. Chang. 22 (2), 454–462.
Rocha, H.R.d., et al., 2004. Seasonality of water and heat fluxes over a tropical forest in Eastern Amazonia. Ecol. Appl. 14 (4), S22–S32.
Rodas, O., Aquino, L., Penayo, C., Yanosky, A., 2012. Resultados del Monitoreo de los Cambios de uso de la Tierra, Incendios e Inundaciones Gran Chaco Americano. Primer año de Monitoreo Asociación Guyra Paraguay, Asunción, Paraguay.
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. Appl. Geogr. 32 (2), 766–776.
Roy, S.S., et al., 2007. Impacts of the agricultural Green Revolution-induced land use changes on air temperatures in India. J. Geophys. Res. 112 (D21), D21108.
Rusticucci, M., Barrucand, M., 2004. Observed trends and changes in temperature extremes over Argentina. J. Clim. 17 (20), 4099–4107.
Saatchi, S., Agosti, D., Alger, K., Delabie, J., Musinsky, J., 2001. Examining fragmentation and loss of primary forest in the southern bahian Atlantic Forest of Brazil with radar imagery. Conserv. Biol. 15 (4), 867–875.
Sampaio, G., et al., 2007. Regional climate change over eastern Amazonia caused by pasture and soybean cropland expansion. Geophys. Res. Lett. 34 (17), L17709.
Sánchez-Azofeifa, G.A., Portillo-Quintero, C., 2011. Extent and drivers of change of neotropical seasonally dry tropical forests seasonally dry tropical forests. In: Dirzo, R., Young, H.S., Mooney, H.A., Ceballos, G. (Eds.), Seasonally Dry Tropical Forests Ecology and Conservation. Island Press/Center for Resource Economics, pp. 45–57.
Sánchez-Cuervo, A.M., Aide, 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.
Sano, E., Rosa, R., Brito, J., Ferreira, L., 2010. Land cover mapping of the tropical savanna region in Brazil. Environ. Monit. Assess. 166 (1), 113–124.
Saulo, C., Ferreira, L., Nogués-Paegle, J., Seluchi, M., Ruiz, J., 2010. Land–Atmosphere interactions during a Northwestern Argentina low event. Mon. Weather Rev. 138 (7), 2481–2498.
Scheffer, M., Carpenter, S., Foley, J.A., Folke, C., Walker, B., 2001. Catastrophic shifts in ecosystems. Nature 413 (6856), 591–596.
Schulz, J., Cayuela, L., Echeverria, C., Salas, J., Benayas, J.M.R., 2010. Monitoring land cover change of the dryland forest landscape of Central Chile (1975–2008). Appl. Geogr. 30 (3), 436–447.
Scott, C.A., et al., 2013. Water security and adaptive management in the arid Americas. Ann. Assoc. Am. Geogr. 103 (2), 280–289.
Sellers, P.J., et al., 1996. A revised land surface parameterization (SiB2) for atmospheric GCMS. Part I: model formulation. J. Clim. 9 (4), 676–705.
Silveira, L., Alonso, J., 2009. Runoff modifications due to the conversion of natural grasslands to forests in a large basin in Uruguay. Hydrol. Process. 23 (2), 320–329.
Skansi, M.d.l.M., et al., 2013. Warming and wetting signals emerging from analysis of changes in climate extreme indices over South America. Glob. Planet. Chang. 100 (0), 295–307.
Smaling, E.M.A., Roscoe, R., Lesschen, J.P., Bouwman, A.F., Comunello, E., 2008. From forest to waste: assessment of the Brazilian soybean chain, using nitrogen as a marker. Agric. Ecosyst. Environ. 128 (3), 185–197.
Snyder, P.K., Delire, C., Foley, J.A., 2004. Evaluating the influence of different vegetation biomes on the global climate. Clim. Dyn. 23 (3), 279–302.
Steininger, M.K., et al., 2001. Tropical deforestation in the Bolivian Amazon. Environ. Conserv. 28 (02), 127–134.
Stickler, C.M., et al., 2013. Dependence of hydropower energy generation on forests in the Amazon Basin at local and regional scales. Proc. Natl. Acad. Sci. U. S. A. 110 (23), 9601–9606.
Sud, Y.C., Fennessy, M., 1982. A study of the influence of surface albedo on July circulation in semi-arid regions using the glas GCM. J. Climatol. 2 (2), 105–125.
Sud, Y.C., Fennessy, M.J., 1984. Influence of evaporation in semi-arid regions on the July circulation: a numerical study. J. Climatol. 4 (4), 383–398.
Trabucco, A., Zomer, R.J., 2009. Global Aridity Index (Global-Aridity) and Global Potential Evapo-Transpiration (Global-PET) Geospatial Database. CGIAR Consortium for Spatial

Information. (Published online, available from the CGIAR-CSI GeoPortal at). http://www.csi.cgiar.org/.
UMSEF, 2007. Informe sobre deforestación en Argentina. Unidad de Manejo del Sistema de Evaluación Forestal. Dirección de Bosques. Secretaria de Ambiente y Desarrollo Sustentable, Buenos Aires, Argentina.
Van de Wouw, P., Echeverría, C., Rey-Benayas, J.M., Holmgren, M., 2011. Persistent Acacia savannas replace Mediterranean sclerophyllous forests in South America. For. Ecol. Manag. 262 (6), 1100–1108.
Verbist, K., Santibañez, F., Gabriels, D., Soto, G., 2010. Atlas of arid and semi arid zones of Latin America and the Caribbean, technical documents of the UNESCO PHI-LAC, N25, ISBN 978-92-9089-164-2. Water Center for Arid and Semi-Arid Zones in Latin America and the Caribbean.
Vicuña, S., McPhee, J., Garreaud, R., 2012. Agriculture vulnerability to climate change in a snowmelt-driven basin in semiarid Chile. J. Water Resour. Plan. Manag. 138 (5), 431–441.
Viglizzo, E., et al., 2011. Ecological and environmental footprint of 50 years of agricultural expansion in Argentina. Glob. Chang. Biol. 17 (2), 959–973.
WB, 2014. The World Bank data. World development indicators, urban development, urban population (% of total) and population in the largest city (% of urban population). (Available at). http://data.worldbank.org/topic/urban-development.
Webb, T.J., Woodward, F.I., Hannah, L., Gaston, K.J., 2005. Forest cover-rainfall relationships in a biodiversity hotspot: the Atlantic Forest of Brazil. Ecol. Appl. 15 (6), 1968–1983.
Winchester, L., Szalachman, R., 2009. The Urban Poor's Vulnerability to the Impacts of Climate Change in Latin America and the Caribbean — A Policy Agenda. United Nations (UN), Economic Commission for Latin America and the Caribbean (ECLAC).
WWF, 2010. WWF's living Amazon initiative. A Comprehensive Approach to Conserving the Largest Rainforest and Rivers System on Earth. Worldwide Fund for Nature.
Zak, M., Cabido, M., Hodgson, J., 2004. Do subtropical seasonal forests in the Gran Chaco, Argentina, have a future? Biol. Conserv. 120 (4), 589–598.
Zak, M., Cabido, M., Cáceres, D., Díaz, S., 2008. What Drives Accelerated Land Cover Change in Central Argentina? Synergistic Consequences of Climatic, Socioeconomic, and Technological Factors. Environ. Manag. 42 (2), 181–189.
Zedler, P.H., Fox, M.D., Arroyo, M.K., 1995. Ecology and biogeography of mediterranean ecosystems in Chile, California, and Australia. Springer-Verlag, New York (445 pp.).
Zehe, E., Sivapalan, M., 2009. Threshold behaviour in hydrological systems as (human) geo-ecosystems: manifestations, controls, implications. Hydrol. Earth Syst. Sci. 13 (7), 1273–1297.
Zwiers, F.W., von Storch, H., 1995. Taking serial correlation into account in tests of the mean. J. Clim. 8 (2), 336–351.