Journal of Arid Environments 123 (2015) 21–30

Contents lists available at ScienceDirect

Journal of Arid Environments

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

Is forest or Ecological Transition taking place? Evidence for the Semiarid Chaco in Argentina

José N. Volante [a], [*], José M. Paruelo [b]

[a] Laboratorio de Teledetección y SIG, Instituto Nacional de Tecnología Agropecuaria (INTA), Estación Experimental Agropecuaria Salta, Ruta Nacional 68, Km 172 Cerrillos, Salta, Argentina

[b] Laboratorio de Análisis Regional y Teledetección, Instituto de Investigaciones Fisiológicas y Ecológicas vinculadas a la Agricultura, Departamento de Métodos Cuantitativos y Sistemas de Información, Consejo Nacional de Investigaciones Científicas y Técnicas, Facultad de Agronomía, Universidad de Buenos Aires, Av. San Martín 4453, Buenos Aires, Argentina

ARTICLE INFO

Article history:
Received 13 February 2014
Received in revised form
13 April 2015
Accepted 29 April 2015
Available online 7 May 2015

Keywords:
Forest Transition
Land-sparing
Deforestation
Dry Chaco
Remote-sensing

ABSTRACT

The expression *Forest Transition* (FT) and *Ecological Transition* (ET) were coined to describe the recovery path of forested areas (or more general natural vegetation) after heavy conversion into croplands. FT/ET would be driven by two main socio-economic controls: a) agricultural intensification in the most productive areas and the simultaneous reduction of cropped area in the less suitable areas (*land-sparing*); and b) rural population migration from rural to urban areas. In the argentine portion of the Semiarid Chaco a rapid and extensive clearing for industrial agriculture and cattle ranching based on sown pastures is taking place. In this article we evaluated the occurrence, magnitude and localization of FT/ET in the argentine portion of the Semiarid Chaco during the 1977–2007 period using an approach based on remotely sensed data. From land cover maps we derived three diagnosis variables of FT/ET: (1) the area of natural vegetation at the end of the study period; (2) the rate of annual clearance (natural vegetation loss) for the whole period; and (3) the temporal change of the area of natural vegetation in the last portion of the study period. The diagnosis variables were combined to derive 12 classes (landscape types). We observed a systematic loss of the surface occupied by natural vegetation. Industrial agriculture grew in aggregated patches generating a homogenization of the landscape. In only a 4.8% of the study area we observed a pattern of change compatible with a FT/ET. In contrast, in a 34% of the study region (9.57 million ha) a clear negative trend in the cover of natural vegetation was observed during 1977–2007 period. The area that had a negative trend in the cover of natural vegetation was 7 times greater than the portion of the region experiencing positive trends. Such number indicates an imbalance in the landscape dynamics that would further reduce areas covered by natural vegetation.

1. Introduction

In the '90 Alexander Mather starter to use the expression "*Forest Transition*" (Mather, 1990, 1992; Mather and Needle, 1998) to describe the pattern of recovery of forested areas in northern Scotland since the XIX century (Mather, 1992; Mather et al., 1998). Mather proposed that socio-economic development generated, first, a decrease in the area covered by forest, followed by a partial recovery of them. The Forest Transition model describes, then, the temporal sequence of deforestation–afforestation of a region (Fig. 1). This pattern was also observed in other countries (Veblen and Lorenz, 1991; Foster, 1992; Mather et al., 1998, 1999; Mather and Fairbairn, 2000; Perz and Skole, 2003; Aide and Grau, 2004; Grau et al., 2005b; Baptista and Rudel, 2006; Mather, 2007; Yackulic et al., 2011; Lambin and Meyfroidt, 2011). The FT concept was used by other authors to explain the recovery processes, produced by land abandonment and migration, observed in ecosystems others than forest, (Grau and Aide, 2008; Grau et al., 2008a; Izquierdo and Grau, 2009). This mechanism was named Ecological Transition (ET).

Two socio-economic controls would drive FT/ET: a) the agricultural intensification in the most productive areas and the simultaneous reduction of cropped area in the less suitable areas (Mather and Needle, 1998; Lambin and Meyfroidt, 2011); and b) rural population migration from rural to urban areas (Mather and

---

[*] Corresponding author.
E-mail address: volante.jose@inta.gob.ar (J.N. Volante).

http://dx.doi.org/10.1016/j.jaridenv.2015.04.017

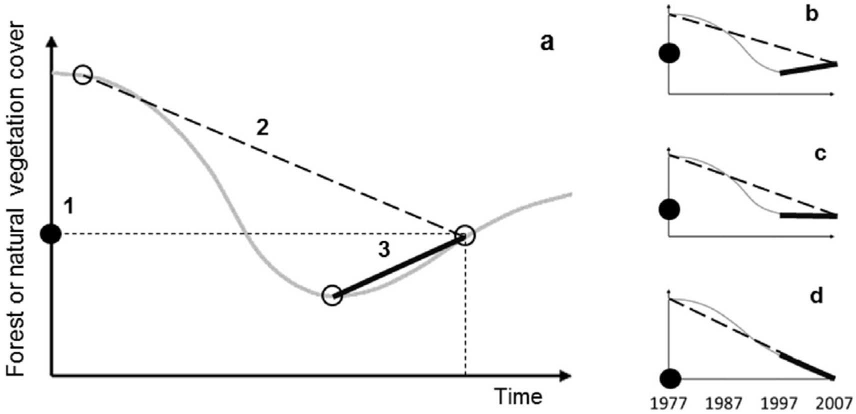

Fig. 1. (a) Hypothetical temporal dynamics (solid gray line) of forest or (more general) natural vegetation cover in a situation of Forest or Ecological Transition. The circles represent observations of natural vegetation cover in 3 times. The numbers represent the "diagnosis variables" used to detect FT/ET: (1) FINAL_AREA_NV: the area occupied by forest or natural vegetation at the end of the study period (black dot); (2) TOTAL_RATE_CHANGE: the annual rate of change of the area of forest or natural vegetation for the whole period (dashed black line); and (3) LAST_RATE_CHANGE: the change of the area occupied by natural vegetation area in the last portion of the study period (slope of the black solid line). The b, c, and d panels show different temporal dynamics of land covers.

Needle, 1998; Aide and Grau, 2004; Grau et al., 2008a; b). The first mechanisms (*"land-sparing"*) has been proposed as a strategy to balance food production and nature conservation throughout increasing the amount of land spared for nature (Fischer et al., 2008; Perfecto and Vandermeer, 2010; Grau et al., 2013).

In the argentine portion of the Semiarid Chaco, extensive ranching for subsistence economy and traditional uses of the forests (hunting and gathering, coal production, timber harvest) are the main activities of native communities (*aborigenes*) and local settlers (*criollos*). Despite that, in recent decades the natural vegetation was rapidly and extensive cleared for industrial agriculture and cattle pastures (Grau et al., 2005a; b; Volante et al., 2006; Gasparri and Grau, 2009). The land-sparing concept was used by Grau et al. (2008a) to explain the recent spatial and temporal dynamics of land use/land cover changes in the Semiarid Chaco of Argentina. They suggested that agricultural intensification in some areas decreased the impact of traditional land use practices in other areas, given place to forest recovery. Such effect would be a consequence of rural-urban migration of local people and the abandonment of "puestos" (this is the typical Chaco subsistence housing unit, composed by a main house, a water reservoir, and corrals, typically surrounded by several hectares of bare soils due to severe vegetation degradation (Grau et al., 2008a)).

The occurrence of land-sparing or FT (or more generally Ecological Transition) phenomena is not neutral in political terms. As a matter of fact is the basis for some "sustainable development" policies aimed to combine an industrialized agriculture and a network of areas for natural conservation (Seghezzo et al., 2011). Is the FT/ET process actually taking place in the Semiarid Chaco? Answering this question is particularly important in an area that is having nowadays one the largest deforestation rates in the world (Vallejos et al., 2015). Therefore, we proposed two objectives; a) to characterize the dynamics (status and trends) of natural vegetation and agriculture land covers; and b) to evaluate the occurrence, magnitude and localization of TF/ET. We carried out our research in the argentine portion of the Semiarid Chaco during the 1977–2007 period using an approach based on remotely sensed data.

2. Materials and methods

2.1. Study area

The study region was located in Subtropical South America and it (22° and 32° S and 61° and 66° W) covers 27.6 10^6 ha of the Santiago del Estero, Salta, Tucumán, Jujuy, Catamarca, Chaco and Formosa provinces of Argentina. Most of the study region corresponded to the Semiarid Chaco, an area dominated by Xerophytic Subtropical Forests characterized by the presence of *Aspidosperma quebracho-blanco*, *Schinopsis quebracho-colorado, Chorisia speciosa, Caesalpinea paraguariensis,* and *Prosopis spp.* (Cabrera, 1976; Morello et al., 2012). The northwestern part of the area a small proportion corresponded to the Yungas ecoregion, covered by Humid Subtropical Forests (Cabrera, 1976; Morello et al., 2012) (Fig. 2). The mean annual temperature ranges between 20 and 22 °C, with maximum mean values (24–27 °C) in January, and minimum mean values in July (14.5 °C–15.5 °C). The lowest annual rainfall is found in the center of the study area (ca. 500 mm) increases eastward, westward, and southward. Along the western and southern borders, rainfall reaches 700–900 mm, providing the opportunity for rainfed agriculture (Morello et al., 2012).

The area experienced an extensive and intense deforestation processes (Vallejos et al., 2015). Two factors drove this clearing process: 1) an increase in global demand and prices of soybeans products; and, 2) a 20–30% increase in regional mean annual precipitation (Zak et al., 2004; Grau et al., 2005b; Boletta et al., 2006). Other factors such as the introduction of transgenic or "Roundup Ready" (RR) soybean, the diffusion of no-tillage systems and macroeconomic changes occurred in Argentina in 2001 (changes in currency exchange rates) operated as "trigger factors" (*sensu* Geist and Lambin (2001)) of extensive clearance.

Criollos are the local inhabitants, predominantly of European origin. They depend to a great extent on small-scale extensive cattle ranching that make use of both public and private land. They have been present in the province of Salta since colonial times but they have intensified their presence in the Chaco region by the end of the 19th century (Gordillo and Leguizamón, 2002). Indigenous communities living in the study area include Wichí, Guaraní, Chané, Qom [Toba], Iyojwa'ja [Chorote], Niwaclé [Chulupí], Tapu'i [Tapieté], and scattered Kolla families. Their livelihoods consist of extensive forms of subsistence agriculture, hunting and gathering, and the widespread use of non-timber forest products (Leake and Ecónomo, 2008). These activities are highly dependent on open access to and the good health of local ecosystems. This is especially important because the level of communal environmental management in some of these ethnic groups is generally low, mostly limited to the abandonment of gathering, hunting, and/or fishing

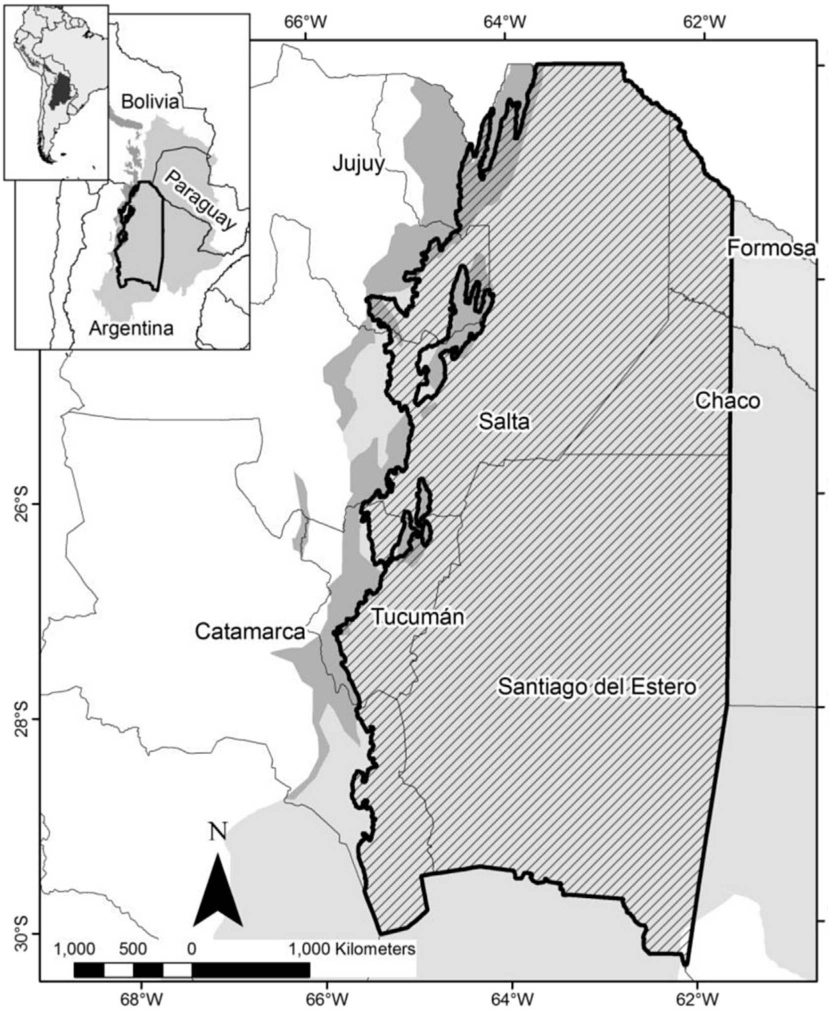

Fig. 2. Study area (shredded frame); Gran Chaco (light gray) and Las Yungas (dark gray) ecorregions.

areas in the low season. Agriculture and cattle ranching are very incipient and spatially circumscribed. There are not ancestral practices for the majority of these indigenous communities. Many natural goods and services have specific survival or cultural purposes. For that reason, they cannot be easily substituted or "traded-off" for other alternatives, in part because indigenous cultures do not accept the notion of valuation of environmental goods and services in monetary terms. The concentration of land in a small number of large-scale agricultural firms (Van Dam, 2008) reduces the space and resources available for criollos and indigenous communities, which threatens their livelihoods, and is a potential source of land tenure conflicts.

2.2. Forest/Ecological Transition detection

The detection of FT/ET required the analysis of the natural vegetation cover through time. In this study we described the temporal dynamics of natural vegetation using three variables (diagnosis variables). These variables summarize the dynamic of vegetation cover and describe the shape of the temporal profile of change. They included: (1) the area occupied by natural vegetation at the end of the study period (2007) (FINAL_AREA_NV); (2) the annual rate of change of the area of natural vegetation for the whole period (1977–2007) (TOTAL_RATE_CHANGE); and (3) the change of the area occupied by natural vegetation in the last portion of the study period (1997–2007) (LAST_RATE_CHANGE) (Fig. 1).

The three variables combined could detect different temporal dynamics of land covers (Fig. 1b–d). Some of them (i.e. panel b in Fig. 1) would correspond to a FT/ET situation. LAST_RATE_CHANGE (the change of the natural cover in the last portion of the study period) is the key variable to detect FT. The other two variables provide a characterization of the current state of the land cover and further details of the temporal changes.

2.3. Maps of vegetation cover

Using Landsat imagery, we mapped three covers: natural vegetation, agriculture and bare soil. The *Natural vegetation* class comprised different physonomical types such as grasslands, shrublands and forest. *Agriculture* class corresponded to regular patches devoted to annual crops or sowed pastures for cattle ranching. Subsistence agriculture, due to the small size of the patches, was not considered as a class by its own. *Bare soil* class included both natural (i.e. salines, water bodies, river beaches) and anthropic areas without plant cover (bares soil areas associated to "puestos", urban areas, and infrastructure).

The spatial resolution of the maps corresponded to the coarsest pixel size of the available images for the 1977–2007 period (80 m, 1977 Landsat MSS images). The maps were elaborated from both digital classification and visual interpretation of Landsat MSS

(*Multi-spectral Scanner Sensor*) (1977) and TM (*ThematicMapper*) (1997 and 2007) mosaics including 19 scenes. Images were acquired from GLOVIS server (*United State Geological Service - Global Visualization Viewer*: http://glovis.usgs.gov/) and INPE server (*Instituto Nacional de Pesquisas Espaciais, Brasil*: http://www.dgi.inpe.br/CDSR/) (Appendix A). We used the methodological approach proposed by Cohen et al. (1998, 2002), which involved both digital classification and visual interpretation of *Tasseled Cap* transformed images (Kauth and Thomas, 1976; Crist and Cicone, 1984). Such approach is particularly useful to detected changes associated to large disturbances (i.e. deforestation or fire). It assumes that the signal associated to the disturbance is larger than the noise generated by differences among images of different dates (Cohen et al., 1998, 2002, 2010). To minimize differences due to phenology, the images included in a given mosaic corresponded to the same season (June to August). To match the resolution between different sensor types, TM images were resampled to the MSS pixel size (80 m) using cubic convolution and coregistered to the GLOVIS images of 2007. We used Relative Radiometric Normalization (Yuan and Elvidge, 1996) to radiometric match contiguous images. To assign pixels to each of the three classes we started by a visual identification of agricultural areas (croplands and pastures) higher than 5 ha on the 2007 mosaic using a 4-5-3 (R-G-B) band combination to maximize contrast between natural vegetation and agriculture (Chuvieco, 2002). The map of agricultural areas was compared with the previous mosaic (1997) to identify the new plots generated in the period. The procedure was repeated for 1977 mosaic to end up with three maps of agricultural areas. Such maps were used as masks in the digital classification stage. Following Cohen et al. (1998, 2002) each MSS and TM mosaic was *Tasseled Cap* transformed (Kauth and Thomas, 1976; Crist and Cicone, 1984). We obtained, from the 6 original reflective bands, 3 synthetic bands (brightness, greenness and wetness). Then, we did a digital classification using ISODATA algorithm (30 classes, 10 iterations and 95% of convergence). The 30 classes were labeled either as natural vegetation or bare soil. To evaluate the accuracy of the maps we compared the classification results and visual interpretations on randomly distributed points (Appendix B).

2.4. Maps of diagnosis variables

The vegetation maps were compiled in a unique geo-spatial database. Then, a grid of 8726 regular cells of 5.5 km side was overlaid (Mitchell, 2005). Each cell of 3025 ha summarizes information (about 4726pixels of 80 m) of each land cover map. The cell size used captures landscape heterogeneity properly because it includes the largest elements (livestock activities around *puestos* affects vegetation up to 5 km) (Grau et al., 2008a).

The cell size used avoided problems associated with spatial autocorrelation among plots deforested for agriculture (see Volante et al., 2012 for analyses of autocorrelation of cleared plots). On the other hand the cell size selected allowed one to integrate the results with other sources of information (i.e.; NDVI LTDR Series, Long Term Data Record). For each cell, we calculated the three diagnosis variables:

a) *Natural vegetation's area in 2007* (FINAL_AREA_NV): Represented the area occupied by forests, shrublands or grasslands quantified as a percentage of the surface of the 3025 ha cell. This variable was categorized into two levels: *High (H)*, where the natural vegetation area was greater than 50% of the cell surface, and *Low (l)* when natural vegetation occupied less than 50%.

b) *Natural vegetation annual change rate for the all period (1977–2007)* (TOTAL_RATE_CHANGE): This variable described the annual loss or gain of natural vegetation cover. We used the annual rate "q" proposed by Food Agriculture Organization (FAO, 1995) to assess deforestation:

$$q = 100 \times [(A_2/A_1)^{1/(t_2-t_1)} - 1]$$

Where "q" is the rate of change in natural vegetation as a percentage; A_1 and A_2 represent the areas of natural vegetation in the years t_1 and t_2 (1977 and 2007), respectively. This variable was also categorized into 2 levels: *High (H)*, where $q \leq -0.51$ (average for South America for the period 2000–2005 according to FAO (2009)) and *Low (l):* for values of $q > -0.51$.

c) *Trend of the natural vegetation in the last period (1997– 2007)* (LAST_RATE_CHANGE): It was estimated as the slope of the relationships between the natural vegetation cover and time during the 1997/2007 period:

$$T = (A_2 - A_1) / (t_2 - t_1)$$

Where T is the trend in the 1997–2007 period, and A_1 and A_2 represent the areas of natural vegetation at$t_1$and t_2 (1997 and 2007 respectively). This variable took 3 levels: *Positive (+)* if $T > 0.01$, *Negative (−)* if $T < -0.01$) or *Neutral (0)* if, $-0.01 > T < 0.01$.

Based on the combination of classes defined for the three diagnosis variables we obtained 12 new categories that represented landscape types based on the natural vegetation dynamics (Table 1). We calculated 'join-count' statistics using "Rookcase" software (Sawada, 1999) to test whether the occurrence of landscape types at the cell level was randomly distributed in space. Join-count statistics were used for testing spatial autocorrelation for binary (presence/absence) and nominal data (Mitchell, 2005). The methods count the number of join encounters in adjacent cells having or not the same category and it assess the presence of spatial autocorrelation. We use a Queen's case (8 neighboring cells) definition for assess common boundary.

3. Results

3.1. Spatial characterization

Land cover maps (Fig. 3a–c) presented an overall accuracy of 85.2, 87.3 and 88.4% for 1977, 1997 and 2007 respectively (See Appendix B, Tables B.1, B.2 and B.3). The matrix of natural vegetation in the study area, was perforated and interrupted by large agricultural patches located in two longitudinal stripes, to the East and West of the study area (Fig. 3a–c). The pattern observed changed through time in the number and sizes of agricultural patches. The distribution of the major agricultural areas was consistent with the rainfall distribution in the region, leaving a central strip of natural vegetation in the driest area of region (Fig. 3a and d). In the Eastern and Western bands mentioned above, occurred the highest rates of change during the past 30 years (Fig. 3e). The changes observed in the whole study period were still evident in the latest period (Fig. 3f). Fifty three percent (14.8 million ha) of the study area showed a negative change while 39% (10.8 million ha) showed a positive one from 1997 through 2007.

All the landscape types showed a very high degree of spatial autocorrelation evidenced by high values of Z-random join-count statistic (greater than 2.33 for a confidence level of 99%) (Fig. 4b). Much of the study area (86%) was characterized by four landscape types (Hl0, Hl+, Hl− and lH−). These types had a high degree of spatial aggregation forming patches (contiguous cells of the same type) of different sizes (Fig. 4a and b). The most abundant category was Hl0 (High natural vegetation in 2007, low rate of change in the

Table 1

Landscape types based on 3 diagnosis variables: a) Natural vegetation's area in 2007 (FINAL_AREA_NV), with two levels: high (H) and low (l); b) Natural vegetation annual change rate for the all period (1977–2007) (TOTAL_RATE_CHANGE) with two levels: high (H) and low (l); and c) Trend of the natural vegetation in the last period (1997–2007) (LAST_RATE_CHANGE), with tree levels: positive (+), neutral (0), and negative (−).

Variables	Categories											
FINAL_AREA_NV	H						l					
TOTAL_RATE_CHANGE	H			l			H			l		
LAST_RATE_CHANGE	+	0	-	+	0	-	+	0	-	+	0	-
Landscapes Types	HH+	HH0	HH-	Hl+	Hl0	Hl-	lH+	lH0	lH-	ll+	ll0	ll-

long term and short term neutral change) covering 42% of the study area (11.7 million ha) (Figs. 3–5).

The lH− type (low natural vegetation in 2007, high rates of deforestation and negative short-term change) was the next most abundant type occupying 17.2% (4.8 million ha) of the study area followed by the Hl− (high natural vegetation, low long-term rates of clearing and short-term negative change) and the Hl+ (high natural vegetation, low long term rates of clearing positive short-term change) types, covering14.2% (3.9 million ha) and 13% (3.6 million ha) of the study area respectively (Figs. 3–5).

The next three types in terms of area occupied were ll+, HH−, and lH+, accounting for by approximately 10% of the study area. ll+ and lH+ types were geographically associated (Fig. 4). The HH−type was a mix of agriculture and natural vegetation. It was located in the advancing agricultural frontier, as a transition between lH− and Hl−. The remaining types (HH0, HH+, ll0, ll− and lH0) had a low presence in the territory. All together they accounted for by 4.3% of the study area.

3.2. Spatial and temporal dynamics

Fig. 4 described the dynamics of agricultural advance in the region both in space and time. We observed a spatial sequence of landscape from the type with the highest proportion of agriculture (lH−) to the less transformed category (Hl−), which is in contact with the natural landscape (Hl0). An intermediate situation (HH−) occurred between lH− and Hl−types. In time, the sequence of landscape type occurred from types dominated by natural vegetation (Hl0) to the agriculture-dominated type (lH−) (Fig. 6).

3.3. Potential Forest Transition's situations

The landscape types that may correspond to possible Forest/Ecological Transition situations are lH+, ll+; HH+, and Hl+. These types covered a 20.8% of the study area. The Hl+ is the most extended type (14.2%) (Fig. 4). This type was geographically associated to the dynamics of wetlands areas or floodplains of the main rivers of the region (Pilcomayo, Bermejo and Salado) (Fig. 4). The positive change in this particular type during the last decade cannot be associated with anthropic activities (i.e. agricultural field abandonment, afforestation) because the areas corresponded to non-agricultural soils (INTA, 1990) and agricultural transitions in the first part of the study period were not detected (*InstitutoNacional de TecnologíaAgropecuaria*; http://inta.gob.ar/documentos/monitoreo-de-cultivos-del-noroeste-argentino-a-traves-de-sensores-remotos/). Consequently, positive changes during the 1997–2007 period in the areas covered by the Hl+ type cannot be associated to a Forest/Ecological Transition dynamics, *sensu* Mather (1992).

The lH+ and ll+ types, covering 5.7% of the study area, were associated with both the oldest areas of intensive farming of the region (4.2%) and salt flats (1.5%). In the case of the oldest agricultural fields a Forest/Ecological Transition processes is likely. The category HH+ (High proportion of natural vegetation, high rates of clearing, and positive changes during the last decade), had a U-shaped temporal trajectory also compatible with a Forest/Ecological Transition processes (Fig. 5a). This type is poorly represented in the study area covering only 0.6% (equivalent to 168,000 ha).

4. Discussion

Our results showed that only a small fraction of the Chaco region presented land cover dynamics compatible with a Forest/Ecological Transition during the last 30 years. Though several of the landscape types HH+, Hl+, lH+ and ll+ presented an increase in the natural vegetation cover during the last decade two additional features are required to provide evidence for FT/ET: (a) the historical presence of forests (or natural vegetation, i.e. savannas or grasslands); and (b) a human induced land use change during the first stages of the period. This was the case of three out of the twelve landscape types defined have some chances to correspond to FT/ET (lH+, ll+ and HH+), with an occupancy of 6.3% of the study area (1.66 million ha). If we subtract the area of these landscape types that corresponds to salt flats (included in types lH+ and ll+), only a 4.8% of the study presented a dynamics compatible with a FT/ET processes. The Hl+ type with occupancy of 14.3% had a very low agricultural occupation (0.5% of study area) and changes in the proportion of natural vegetation could be due to climatic and hydrological dynamics. In contrast to this, 34% of the territory (9.57 million ha) was occupied by types with a clear negative change in the natural vegetation cover during the last years (HH−, Hl−, lH−and ll−). More than 70% of the type lH− was covered by agriculture, and the area devoted to soybean and sowed pastures is still growing (Figs. 3–5) (Vallejos et al., 2015).

It is probable that the state variable *"Natural vegetation's area in 2007"* had some degree of dependence on transitions variables *"Natural vegetation annual change rate for the all period (1977-2007)"* and *"Trend of the native vegetation in the last period (1997–2007)"* but this does not invalidate that can be used to characterize the landscape units in term of their status and their transitions. Is worth to highlight that there is twelve possible

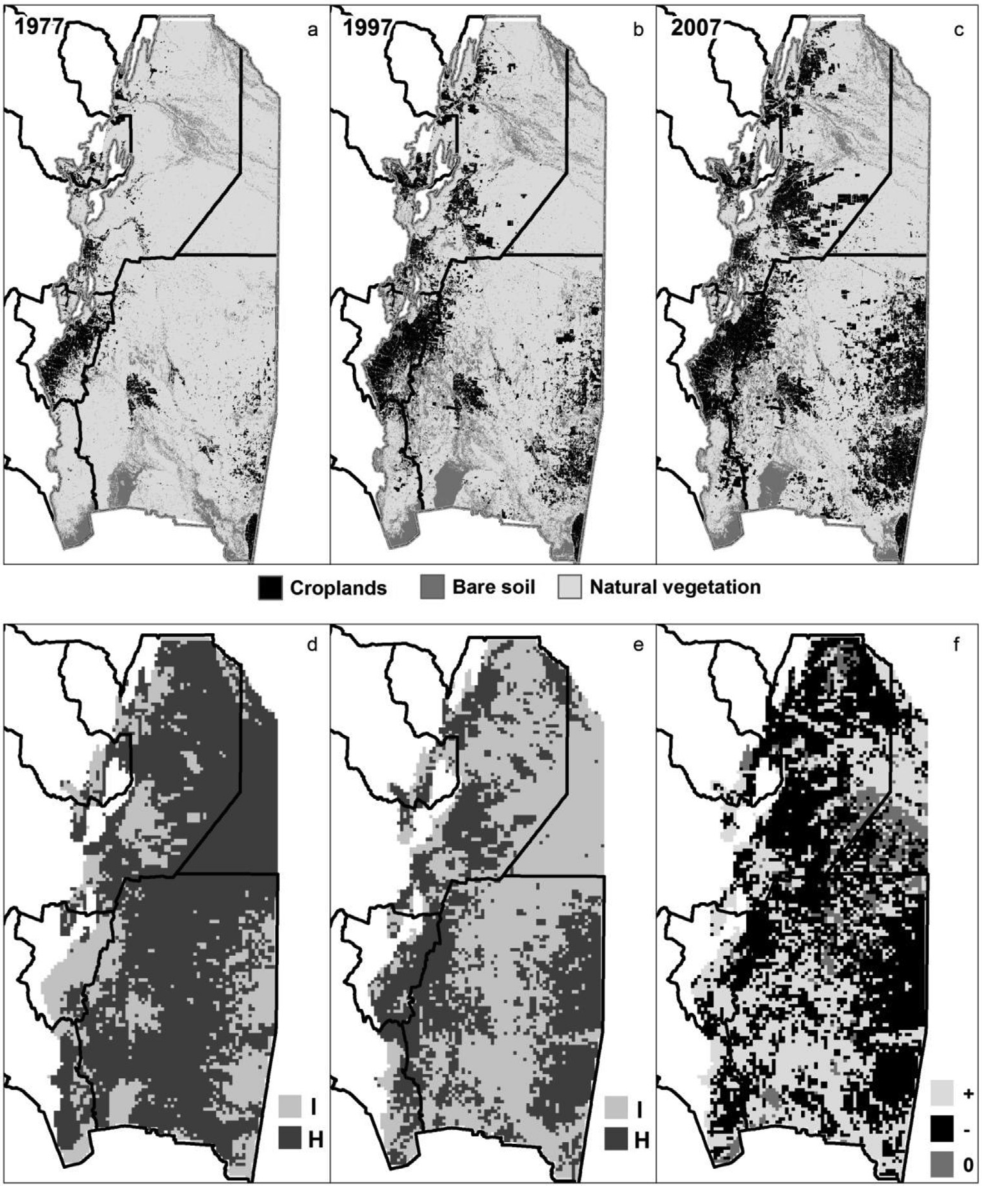

Fig. 3. (a), (b), (c) Land cover maps in 1997, 1987 and 2007; (d) the area occupied by natural vegetation at 2007 (FINAL_AREA_NV); (e) the annual rate of change of the area of natural vegetation for the 1977–2007 period (TOTAL_RATE_CHANGE); and (c) the change of the area occupied by natural vegetation in the 1997–2007 period (LAST_RATE_CHANGE).

landscape types, and all of them were present in the study area allowing us to make a detailed characterization of the landscape in terms of their state and dynamics.

Moreover, the landscape types seem to have a strong association with particular land uses. For example, the Hl0 type, the most abundant category, corresponded to areas with a low degree of transformation occupied by dry forests, shrublands, grasslands and floodplain vegetation. These areas are used by criollos and indigenous communities for subsistence agriculture, livestock production, hunting and gathering. The landscape type lH− corresponded to areas where the highest expansion of industrial agriculture in the past 30 years. The Hl− and HH− types were located in the “advancing agricultural frontier”, as a transition types between Hl0 and lH− categories. The Hl+ category included areas with a high proportion of natural vegetation located in floodplains influenced by the hydrological dynamics of wetlands and the main rivers of the region (Pilcomayo, Bermejo and Salado). Finally, the ll+ and lH+ types were located in the oldest agricultural production zones, mainly devoted to sugarcane production (*Valle de Siancas* in Salta and Jujuy, and *Llanura Deprimida Cañera* in Tucumán), horticulture (*Río Dulce* in Santiago del Estero) or dairy farms (SW of Santiago del Estero). Surprising, these types were also associated to the large salines or marshes.

The ratio between the area of landscape types where land use changes occurred with negative (HH−, Hl−, lH− and ll−) and positive (lH+, ll+ and HH+) changes during the last period was 7.0. Such ratio indicates an imbalance in the landscape dynamics presented in Fig. 6 that would further reduce areas covered by natural vegetation. Such imbalance is indicating that the expansion of industrial agriculture is taking place at a faster rate than the restoration of areas prone to be devoted to conservation. At the regional level there is a clear advance of the area devoted to agriculture but not a land-sparing process.

Based on the clustered spatial configuration of agricultural

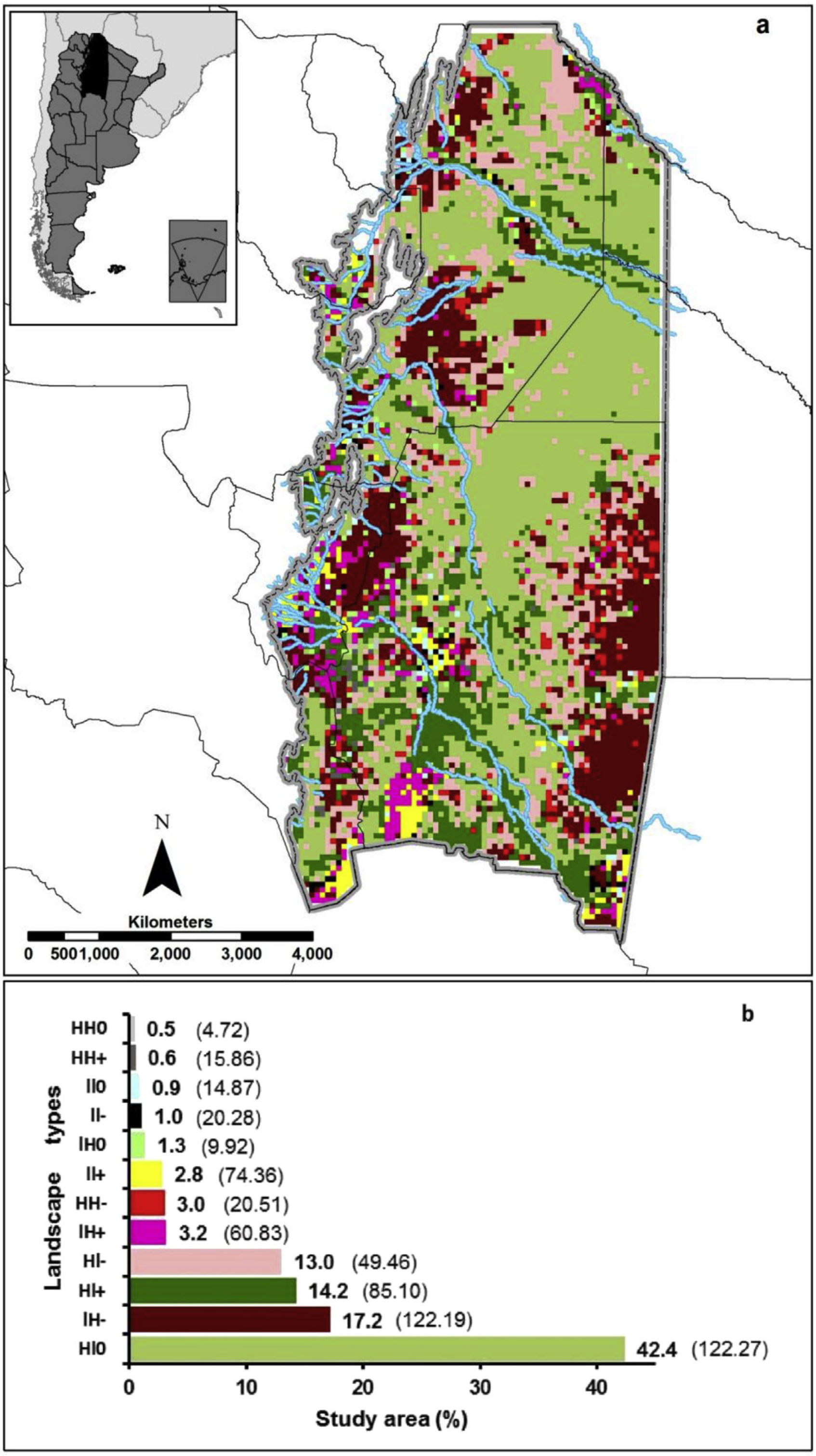

Fig. 4. (a) Distribution of landscape types in the study area; (b) landscape types areas as a percentage of the study area (bold numbers). Numbers in brackets are Z-random join-count values (there is autocorrelation with Z-values greater than 2.33, p-value < 0.01). The hydrological system (IGN, 2007) (light blue lines) was overlaid to visualize the association with the Hl+ type. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

landscapes and the detection of recovery of degraded areas, Grau et al. (2008a) proposed that industrial agriculture associates to agribusiness induced urban-rural migration and, consequently, frees zones for biodiversity conservation (land-sparing model). An important assumption of this model is the existence of a strong environmental limitation for industrial agriculture (Grau et al., 2008a). This assumption should be valid only if annual crops (mainly soybean and maize) are considered. However, deforestation and agricultural expansion is being driven by the expansion of two combined activities, soybean production and cattle ranching

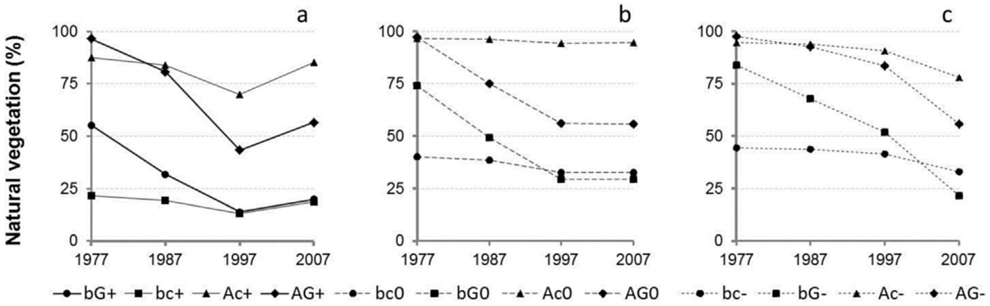

Fig. 5. The temporal trajectory of the natural vegetation from 1997 to 2007, (expressed as a percentage of area of each landscape type). (a) Landscape types with positive trend of the natural vegetation in the last period; (b) those with a neutral trend in the last period; and (c) those with negative trends in the last period.

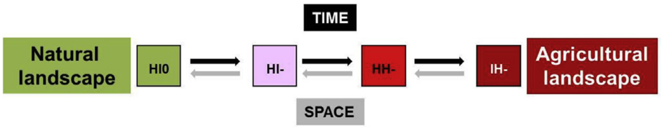

Fig. 6. Spatial and temporal conceptual model of landscape types dynamics. In gray arrows (spatial trajectory) the agricultural landscapes (core areas) are advancing towards to the natural landscapes. In black arrows (temporal trajectory) the natural landscapes are transformed into agricultural landscape type.

based on subtropical tropical pastures (mainly *Cenchrus ciliaris* and *Panicum maximum*) (Gasparri et al., 2013). The relative proportion of these two activities varied according the authors (30–40% soybeans and 60–70% pastures) (Volante et al., 2006; Gasparri et al., 2013).

The analysis performed; based on an objective and evaluated definition of land cover types over a period of thirty years and covering an area of more than 27.6 10^6 ha, provide a synoptic view of the whole region. This allows us to describe two components of agricultural expansion: croplands and pastures. Croplands occupied areas with better environmental conditions (e.g. soils, climate) than those where pastures replaced the natural cover. Of course these two components of the "industrial agriculture" syndrome are not taking place with equal intensity all over the Chaco region. Croplands are mainly concentrated in the more humid sub-region, covering the east and west portion of the study area, while pastures are the dominant type of transformation at the centre of the study area, where rainfall are lower. The semiarid Chaco seems to do not have restrictions for the expansion of subtropical pastures for cattle ranching (Volante et al., 2006; Gasparri et al., 2013). Actually, *Cenchrus cilliaris* (buffel grass) is used in the driest extreme of the Chaco (the Arid Chaco), with annual precipitation lower than 400 mm (Blanco et al., 2009). Though annual crops may have limitations to expand all over the region, pastures may cover the whole Chaco.

Our results suggest that the land-sparing model is not able to describe the dynamics of land use observed in the region. We observed a systematic loss of natural environments for industrial agriculture and cattle ranching that grown in aggregates patches and that homogenizes the landscape from a structural and functional perspective (Volante et al., 2012).

Acknowledgments

This research was supported by INTA (Proyecto PNNAT-1128032), CONICET, UBACyT (0265BA) and FONCyT (PICT 2013-1082). This work was carried out with the aid of a grant from the Inter-American Institute for Global Change Research (IAI) CRN3095, which is supported by the US National Science Foundation (Grant GEO-1128040).

Appendix A

Table A.1
Landsat imagery used to perform land cover maps.

Sensor	Path	Row	Date	Server	Sensor	Path	Row	Date	Server
MSS	248	77	18/06/1975	GLOVIS	TM	231	77	21/08/1996	INPE
MSS	248	78	18/06/1975	GLOVIS	TM	231	78	21/08/1996	INPE
MSS	247	75	17/08/1977	INPE	TM	231	79	21/08/1996	INPE
MSS	247	76	17/08/1977	INPE	TM	230	75	29/07/1996	INPE
MSS	247	77	17/08/1977	INPE	TM	230	76	29/07/1996	INPE
MSS	247	78	17/08/1977	INPE	TM	230	77	29/07/1996	INPE
MSS	247	79	17/08/1977	INPE	TM	230	78	29/07/1996	INPE
MSS	247	80	17/08/1977	INPE	TM	230	79	29/07/1996	INPE
MSS	247	81	17/08/1977	INPE	TM	230	80	29/07/1996	INPE
MSS	246	76	27/10/1977	INPE	TM	230	81	29/07/1996	INPE
MSS	246	77	28/10/1977	INPE	TM	229	76	26/08/1997	INPE
MSS	246	78	29/10/1977	INPE	TM	229	77	26/08/1997	INPE
MSS	246	79	30/10/1977	INPE	TM	229	78	26/08/1997	INPE
MSS	246	80	31/10/1977	INPE	TM	229	79	26/08/1997	INPE
MSS	245	77	17/05/1977	INPE	TM	229	80	26/08/1997	INPE
MSS	245	78	20/09/1977	INPE	TM	228	78	02/07/1997	GLOVIS
MSS	245	79	20/09/1977	INPE	TM	228	79	02/07/1997	GLOVIS
MSS	245	80	17/05/1977	INPE	TM	228	80	02/07/1997	GLOVIS
MSS	245	81	17/05/1977	INPE	TM	228	81	06/10/1997	GLOVIS
TM	231	77	25/07/1986	GLOVIS	TM	231	77	03/07/2007	GLOVIS
TM	231	78	25/07/1986	GLOVIS	TM	231	78	03/07/2007	GLOVIS
TM	231	79	25/07/1986	GLOVIS	TM	231	79	02/09/2006	GLOVIS

Table A.1 (*continued*)

Sensor	Path	Row	Date	Server	Sensor	Path	Row	Date	Server
TM	230	75	23/09/1987	INPE	TM	230	75	26/08/2006	INPE
TM	230	76	23/09/1987	INPE	TM	230	76	26/08/2006	INPE
TM	230	77	23/09/1987	INPE	TM	230	77	26/08/2006	INPE
TM	230	78	23/09/1987	INPE	TM	230	78	26/08/2006	INPE
TM	230	79	23/09/1987	INPE	TM	230	79	26/08/2006	INPE
TM	230	80	23/09/1987	INPE	TM	230	80	26/08/2006	INPE
TM	230	81	23/09/1987	INPE	TM	230	81	26/08/2006	INPE
TM	229	76	15/08/1987	INPE	TM	229	76	22/08/2007	INPE
TM	229	77	31/08/1987	INPE	TM	229	77	22/08/2007	INPE
TM	229	78	31/08/1987	INPE	TM	229	78	22/08/2007	INPE
TM	229	79	31/08/1987	INPE	TM	229	79	22/08/2007	INPE
TM	229	80	16/09/1987	GLOVIS	TM	229	80	22/08/2007	INPE
TM	228	78	06/09/1986	INPE	TM	228	78	31/08/2007	INPE
TM	228	79	06/09/1986	INPE	TM	228	79	31/08/2007	INPE
TM	228	80	21/06/1987	INPE	TM	228	80	31/08/2007	INPE
TM	228	81	21/06/1987	INPE	TM	228	81	31/08/2007	INPE

Appendix B

The accuracy assessment of the cartographic quality of land cover maps was based on Cohen et al. (1998, 2002), Healey et al. (2005) and Kennedy et al. (2007). A comparison of randomly distributed points was performed between land cover maps based on visual interpretations of Tasseled Cap transformations. We randomly selected 40 “focal points” in the study area, on which an area of 40 × 40 km (segment) was defined. Each segment fulfilled the condition of having at least 75% of the category “natural vegetation” and each of the remaining classes were represented in at least one segment. Within each segment, 40 pixels were randomly selected with the following conditions: a) up to 60% for category (agriculture, bare soil, natural vegetation); b) just one point by patch for the category agriculture, defining “patch” as a group of contiguous pixels of the same category; and c) the pixels classified as “natural vegetation” were randomly redistributed geographically to maximize the separation between points within each segment. The conditions (b) and (c) were set to reduce the risk of pseudo-replication (Hurlbert, 1984) produced by autocorrelation (Dormann et al., 2013). This technique was applied separately for each of the analyzed dates (1977, 1997, and 2007). Each set of 1600 evaluation points (40 segments with 40 sample points), were labeled with the categories of the map (natural vegetation, bare soil, agriculture) by visual interpretation of the Tasseled Cap mosaics. The visual interpretations did not consider the results obtained from the digital classification. Subsequently both results were compared by a confusion matrix (Chuvieco, 2002). We found an overall accuracy of 85.2, 87.3 and88.4% for maps of 1977, 1997 and 2007 respectively (Table B.1, B.2, and B.3). These values were consistent with the results reported by (Cohen et al., 1998, 2002) using de same methodology.

Table B.1
Error matrix of 1977's land cover map.

		Reference					
		Agriculture	Bare soil	Natural vegetation	Total	Omission error (%)	Commission error (%)
Map	Agriculture	237	18	36	291	0.19	0.10
	Bare soil	6	371	76	453	0.18	0.21
	Natural vegetation	20	81	755	856	0.12	0.13
	Total	263	470	867	1600		

Table B.2
Error matrix of 1997's land cover map.

		Reference					
		Agriculture	Bare soil	Natural vegetation	Total	Omission error (%)	Commission error (%)
Map	Agriculture	254	28	20	302	0.16	0.10
	Bare soil						
	5	377	73	455	0.17	0.18	
	Natural vegetation	23	54	766	843	0.09	0.11
	Total	282	459	859	1600		

Table B.3
Error matrix of 2007's land cover map.

		Reference					
		Agriculture	Bare soil	Natural vegetation	Total	Omission error (%)	Commission error (%)
Map	Agriculture	317	12	9	338	0.06	0.12
	Bare soil	12	370	77	459	0.19	0.13
	Natural vegetation	32	43	728	803	0.09	0.11
	Total	361	425	814	1600		

References

Aide, T.M., Grau, H.R., 2004. Globalization, migration, and Latin American ecosystems. Policy Forum Sci. 305, 1915–1916.

Baptista, S.R., Rudel, T.K., 2006. A re-emerging Atlantic Forest? Urbanization, industrialization and the forest transition in Santa Catarina, southern Brazil. Environ. Conserv. 33, 195–202.

Blanco, L.J., Ferrando, C.A., Biurrun, F.N., 2009. Remote sensing of spatial and temporal vegetation patterns in two grazing systems. Rangel. Ecol. Manag. 62 (5), 445–451.

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.

Cabrera, A., 1976. Regiones fitogeográficas argentinas. Encicl. Argent. Agric. Jard 2 (5), 85.

Chuvieco, E., 2002. In: Ciencia, Ariel (Ed.), Teledetección ambiental. La observación de la Tierra desde el espacio, 1°. España, Barcelona. ISBN: 84-344-8047-6.

Cohen, W.B., Fiorella, M., Gray, J., Helmer, E., Anderson, K., 1998. An efficient and accurate method for mapping forest clearcuts in the Pacific Northwest using Landsat imagery. Photogramm. Eng. Remote Sens. 64 (4), 293–300.

Cohen, W.B., Spies, T. a, Alig, R.J., Oetter, D.R., Maiersperger, T.K., Fiorella, M., 2002. Characterizing 23 Years (1972-95) of stand replacement disturbance in western Oregon forests with Landsat imagery. Ecosystems 5 (2), 122–137.

Cohen, W.B., Yang, Z., Kennedy, R., 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 2. TimeSync — Tools for calibration and validation. Remote Sens. Environ. 114 (12), 2911–2924.

Crist, E.P., Cicone, R.C., 1984. A physically-based transformation of thematic mapper data – the TM tasseled Cap. IEEE Trans. Geosci. Remote Sens. 22 (3), 256–263.

Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carr, G., Garc, J.R., Gruber, B., Lafourcade, B., Leit, P.J., Tamara, M., Mcclean, C., Osborne, P.E., Der, B.S., Skidmore, A.K., Zurell, D., Lautenbach, S., 2013. Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. Ecography (Cop.) 36, 27–46.

FAO, 1995. Forest Recourses Assessments 1990. Global Synthesis. Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO, 2009. State of the World's Forests 2009 (Rome, Italy).

Fischer, J., Brosi, B., Daily, G.C., Ehrlich, P.R., Goldman, R., Goldstein, J., Lindenmayer, D.B., Manning, A.D., Mooney, H. a, Pejchar, L., Ranganathan, J., Tallis, H., 2008. Should agricultural policies encourage land sparing or wildlife-friendly farming? Front. Ecol. Environ. 6 (7), 380–385.

Foster, D., 1992. Land use history (1730–1990) and vegetation dynamics in central New England. J. Ecol. 80, 753–771.

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

Gasparri, N.I., Grau, H.R., Gutiérrez Angonese, J., 2013. Linkages between soybean and neotropical deforestation: coupling and transient decoupling dynamics in a multi-decadal analysis. Glob. Environ. Change 23 (6), 1605–1614.

Geist, H.J., Lambin, E.F., 2001. What Drives Tropical Deforestation? a Meta-analysis of Proximate and Underlying Causes of Deforestation Based on Subnational Case Study Evidence. In: (LUCC Report Series; 4). Land-Use and Land-Cover Change (LUCC) Project. International Human Dimensions Programme on Global Environmental Change (IHDP). International Geosphere-Biosphere Programme (IGBP), Belgium.

Gordillo, G., Leguizamón, J.M., 2002. El río y la frontera: movilizaciones aborígenes, obras públicas y Mercosur en el Pilcomayo. Editorial Biblos., Buenos Aires, Argentina.

Grau, H.R., Aide, M.T., 2008. Globalization and land-use transitions in Latin America. Ecol. Soc. 13 (2), 16.

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

Grau, H.R., Gasparri, N.I., Aide, T.M., 2005b. Agriculture expansion and deforestation in seasonally dry forests of north-west Argentina. Environ. Conserv. 32 (02), 140.

Grau, H.R., Gasparri, N.I., Aide, T.M., 2008a. Balancing food production and nature conservation in the Neotropical dry forests of northern Argentina. Glob. Change Biol. 14 (5), 985–997.

Grau, H.R., Hernández, M.E., Gutierrez, J., Gasparri, N.I., Casavecchia, M.C., Flores-ivaldi, E.E., Paolini, L., 2008b. A peri-urban neotropical Forest transition and its consequences for environmental services. Ecol. Soc. 13 (1), 35.

Grau, R., Kuemmerle, T., Macchi, L., 2013. Beyond land sparing versus land sharing': environmental heterogeneity, globalization and the balance between agricultural production and nature conservation. Curr. Opin. Environ. Sustain. 5, 1–7. http://dx.doi.org/10.1016/j.cosust.2013.06.001.

Healey, S., Cohen, W., Zhiqiang, Y., Krankina, O., 2005. Comparison of tasseled cap-based Landsat data structures for use in forest disturbance detection. Remote Sens. Environ. 97 (3), 301–310.

Hurlbert, S.H., 1984. Pseudoreplication and the design of ecological field experiments. Ecol. Soc. Am. 54 (2), 187–211.

INTA., 1990. Atlas de Suelos de la República Argentina, Sistema de Información Geográfico. Instituto Nacional de Tecnología Agropecuaria, Buenos Aires, Argentina. Available at: http://geointa.inta.gov.ar/web/index.php/suelos-de-la-republica-argentina/.

Izquierdo, A.E., Grau, H.R., 2009. Agriculture adjustment, land-use transition and protected areas in Northwestern Argentina. J. Environ. Manag. 90 (2), 858–865.

Kauth, R.J., Thomas, G.S., 1976. The tasselled Cap. A graphic description of the spectral-temporal development of agricultural crops as seen by Landsat. p. 13. In: Symposium on Machine Processing of Remotely Sensed Data. LARS Symposia. Data; 6 June–2 July 1976. West Lafayette. Purdue University, Indiana.

Kennedy, R.E., Cohen, W.B., Schroeder, T. a, 2007. Trajectory-based change detection for automated characterization of forest disturbance dynamics. Remote Sens. Environ. 110 (3), 370–386.

Lambin, E.F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. Proc. Natl. Acad. Sci. U. S. A. 108 (9), 3465–3472.

Leake, A., Ecónomo, M., 2008. La deforestación de Salta 2004-2007. Fundación ASOCIANA, Salta, Argentina.

Mather, A., 1990. Global Forest Resources. Bellhaven Press, London.

Mather, A., 1992. The forest transition. Area 24, 367–379.

Mather, A., 2007. Recent Asian forest transitions in relation to forest transition theory. Int. For. Rev. 9, 491–502.

Mather, A., Fairbairn, J., 2000. From floods to reforestation: the forest transition in Switzerland. Environ. Hist. Camb. 6, 399–421.

Mather, A., Fairbairn, J., Needle, C., 1999. The course and drivers of the forest transition: the case of France. J. Rural Stud. 15, 65–90.

Mather, A., Needle, C.L., 1998. The forest transition: theoretical basis. Area 30, 117–124.

Mather, A., Needle, C., Coull, J., 1998. From resource crisis to sustainability: the forest transition in Denmark. Int. J. Sustain. Dev. World Ecol. 5, 183–192.

Mitchell, A., 2005. In: . The ESRI Guide to GIS Analysis: Spatial Measurements and Statistics, vol. 2 (Redlands, California, USA).

Morello, J., Matteucci, S.D., Rodriguez, A.F., Silva, M.E., 2012. Ecorregiones y complejos ecosistemicos argentinos. In: Primera (Ed.). Facultad de Arquitectura Desarrollo y Urbanismo, Buenos Aires, Argentina.

Perfecto, I., Vandermeer, J., 2010. The agroecological matrix as alternative to the land-sparing/agriculture intensification model. Proc. Natl. Acad. Sci. U. S. A. 107 (13), 5786–5791.

Perz, S.G., Skole, D.L., 2003. Secondary forest expansion in the Brazilian Amazon and the refinement of forest transition theory. Soc. Nat. Resour. 16 (4), 277–294.

Sawada, M., 1999. Rookcase: an Excel 97/2000 Visual Basica (VB) add-in for exploring global and local spatial autocorrelation. Bull. Ecol. Soc. Am. 80 (4), 231–234.

Seghezzo, L., Volante, J.N., Paruelo, J.M., Somma, D.J., Buliubasich, E.C., Rodriguez, H.E., Gagnon, S., Hufty, M., 2011. Native forests and agriculture in salta (argentina): conflicting visions of development. J. Environ. Dev. 20 (3), 251–277.

Vallejos, M., Volante, J.N., Mosciaro, M.J., Vale, L., Bustamante, M.L., Paruelo, J.M., 2015. Transformation dynamics of the natural cover in the Dry Chaco ecoregion: A plot level geo-database from 1976 to 2012. J. Arid Environ. 123, 3–11.

Van Dam, C., 2008. Tierra, territorio y derechos de los pueblos indígenas, campesinos y pequeños productores de Salta. In: Serie Documentos de Capacitación No 2 (PROFEDER, Ed.) (Buenos Aires, Argentina).

Veblen, T.T., Lorenz, D.C., 1991. The Colorado Front Range. A Century of Ecological Change. University of Utah Press, Salt Lake City, Utah, USA.

Volante, J.N., Alcaraz-Segura, D., Mosciaro, M.J., Viglizzo, E.F., Paruelo, J.M., 2012. Ecosystem functional changes associated with land clearing in NW Argentina. Agric. Ecosyst. Environ. 154, 12–22.

Volante, J.N., Bianchi, A.R., Paoli, H.P., Noé, Y., Elena Elena, H.J., Cabral, C.M., 2006. Análisis de la dinámica del uso del suelo agrícola del Noroeste Argentino mediante teledetección y Sistemas de Información Geográfica. Período 2000-2005. In: Ediciones INTA. Instituto Nacional de Tecnología Agropecuaria, Salta, Argentina.

Yackulic, C.B., Fagan, M., Jain, M., Jina, A., Lim, Y., Marlier, M., Muscarella, R., Adame, P., Defries, R., Uriarte, M., 2011. Biophysical and socioeconomic factors associated with Forest transitions at multiple spatial and temporal scales. Ecol. Soc. 16 (3), 15.

Yuan, D., Elvidge, C.D., 1996. Comparison of relative radiometric normalization techniques. ISPRS J. Photogramm. Remote Sens. 51 (3), 117–126.

Zak, M.R., Cabido, M., Hodgson, J.G., 2004. Do subtropical seasonal forests in the Gran Chaco, Argentina, have a future? Biol. Conserv. 120 (4), 589–598.