Global Change Biology (2008) **14**, 985–997, doi: 10.1111/j.1365-2486.2008.01554.x

Balancing food production and nature conservation in the Neotropical dry forests of northern Argentina

H. RICARDO GRAU*, N. IGNACIO GASPARRI* and T. MITCHELL AIDE†
*CONICET–Laboratorio de Investigaciones Ecológicas de las Yungas, Universidad Nacional de Tucumán. CC 34, Yerba Buena 4107, Tucumán, Argentina, †Department of Biology, University of Puerto Rico, Box 23360, San Juan, Puerto Rico, 00931-3360, USA

Abstract

The growing human population and the increase in per capita food consumption are driving agriculture expansion and affecting natural ecosystems around the world. To balance increasing agriculture production and nature conservation, we must assess the efficiency of land-use strategies. Soybean production, mainly exported to China and Europe, has become the major driver of deforestation in dry forest/savanna ecosystems of South America. In this article we compared land cover patterns (based on satellite imagery) and land-use and human population trends (based on government statistics) in regions with two contrasting development pathways in the Chaco dry forests of northern Argentina, since the early 1970s. The area (ca. 13 million hectares) includes one of the largest continuous patches of tropical dry forests and has experienced rapid land-use change. In the region where land use has been driven by government-sponsored colonization programs, the expansion of extensive grazing has led to a growing rural population, low food production, and widespread environmental degradation. In contrast, in the region dominated by market-driven soybean expansion, the rural population has decreased, food production is between 300% and 800% greater, and low-density extensive cattle production has declined over extensive remaining forested areas, resulting in a land-use trend that appears to better balance food production and nature conservation.

Keywords: agriculture adjustment, Chaco, food production, forest transition, grazing, land sparing, land-use change, land-use efficiency, Neotropical dry forests, soybean

Received 19 June 2007; revised version received 29 October 2007 and accepted 1 November 2007

Introduction

Socioeconomic globalization is producing two opposite large-scale trends in land-use and land-cover change in the Neotropics. First, the relative importance of deforestation as a driving force has moved from shifting agriculture, logging, and cattle ranching to modern agriculture expansion, and a growing proportion of these activities is occurring in dry forest/savanna ecosystems of South America. Of all the major biomes of the world, dry forests have experienced the greatest level of habitat conversion to agriculture lands (Hoekstra *et al*., 2005). This transformation continues at an accelerated rate, making tropical and subtropical dry forests a top conservation priority (Janzen, 1988; Sanchez-Azofeifa *et al*., 2005; Miles *et al*., 2006). The largest remaining area of this biome occurs in South America, covering approximately 1 116 000 km^2, an area equivalent to approximately 20% of the humid/wet forests of the Amazon Basin (Eva *et al*., 2004). The increasing global demand for soy products is the major driver of the extensive deforestation in the neotropical dry forest ecosystems (e.g. Cerrado, Caatinga, Chaco) of Brazil, Bolivia, Paraguay, and Argentina (Fearnside, 2001; Kaimowitz & Smith, 2001; Steininger *et al*., 2001; Zak *et al*., 2004; Grau *et al*., 2005; Silva *et al*., 2006), and this is reflected in the increasing importance of soybean exports for the economy of these countries. The other major trend in neotropical land use, favored by rural–urban migration, is the abandonment of marginal agricultural lands and the initiation of ecosystem recovery (Aide & Grau, 2004), including areas of dry forest ecosystems (Moran *et al*., 1996; Bray & Klepeis, 2005; Jepson, 2005). Although the relative impact of ecosystem recovery is still small in comparison with the extent

Correspondence: H. Ricardo Grau, tel. +54 381 4255123, fax +54 381 4253728, e-mail: chilograu@gmail.com

of deforestation, rural–urban migration is expected to continue (CEPAL, 1999), and this should mitigate the direct human impact over large areas and reduce biodiversity losses (Aide & Grau, 2004; Wright & Muller-Landau, 2006).

The conversion of neotropical dry forests into agricultural lands is a response to the growing human population and the increase in per capita calorie consumption (Smil, 2000; Waggoner & Ausubel, 2001), particularly in Asia. Balancing this increasing global demand for agriculture products with the conservation of the biodiversity and ecological services of natural ecosystems will be a major challenge. To accomplish this, it is essential that we understand the relationship among deforestation, agriculture intensification, and land-use disintensification in marginal lands, and that we evaluate the efficiency of different land-use practices both in terms of food production and nature conservation (De Fries *et al.*, 2004; Mattison & Norris, 2005). Most conservation initiatives promote 'biodiversity-friendly' land uses (agroforestry, silvo-pastoralism, traditional agriculture) mainly because in comparison with modern agricultural systems, they have a higher local biodiversity and maintain a larger proportion of the local flora and fauna (Daily, 2001; Rozenzweig, 2003; Donald & Evans, 2006; Vandermeer & Perfecto, 2007). But these systems often have lower levels of productivity in comparison with modern agriculture (Smil, 2000; Waggoner & Ausubel, 2001; Green *et al.*, 2005), and thus require the conversion of more natural areas to agriculture in order to supply the same food demand. Alternatively, concentrating high-yield modern agriculture systems on the most productive lands may be the most efficient way to balance food production and nature conservation by increasing the amount of land spared for nature (Waggoner & Ausubel, 2001; Balmford *et al.*, 2005; Green *et al.*, 2005). This strategy (land sparing by agriculture adjustment) is supported by land-use change during the last two centuries in Europe and North America. The concentration of agriculture in the most productive areas and the resulting disintensification of agriculture in marginal areas has been the key driver of extensive forest recovery (Mather & Needle, 1998). Consistent with the positive effects of agricultural intensification, during the 1980s, the South American countries with the largest increases in crop yields had lower deforestation rates (Barbier & Burguess, 1997).

The land-sparing argument, based on modern agriculture, has been criticized for neglecting additional environmental impacts. Some case studies showed that an increase in agriculture productivity *per se* does not necessarily reduce deforestation within a region (Hecht, 2005; Morton *et al.*, 2006), and the expansion of modern agriculture can improve access into remote areas, stimulating immigration and promoting additional environmental degradation (Angelsen & Kaimowitz, 2001; Fearnside, 2001). The remaining natural ecosystems are often forest patches with a low conservation value in highly fragmented landscapes (Tscharntke *et al.*, 2005; Donald & Evans, 2006; Vandermeer & Perfecto, 2007), and the degradation of adjacent nondeforested areas may be significant. For example, the area of Amazon forests affected by selective logging and fire is greater than the area deforested (Nepstad *et al.*, 1999; Asner *et al.*, 2005), and what appear to be intact forests have often lost most of the large mammals to hunting (Redford, 1992). Furthermore, modern agriculture frequently leads to soil degradation and watershed contamination (Matson *et al.*, 1997; Tilman *et al.*, 2002).

In summary, different studies provide grounds for both supporting and rejecting the potential of modern agriculture for land sparing and biodiversity conservation. Unfortunately, few studies have simultaneously compared the impact of modern agriculture with traditional productive systems in terms of food production and ecosystem conservation. The Chaco dry forest of northern Argentina, an area of >13 million hectares, includes one of the world's most dynamic agricultural frontiers and provides an opportunity to conduct this comparison because it includes large regions that are experiencing contrasting patterns of land use. In this study, we compare land-use efficiency, in terms of food production and trends in forest conservation over the past 30 years, in a region affected by modern agriculture and another region affected by traditional grazing.

Material and methods

To compare the two development strategies, we analyzed changes in land use/cover between the early 1970s and 2002, and compared changes in the land use, food production, and conservation implications between the two regions. We studied 13.4 million hectares of the Chaco in northern Argentina (Fig. 1). Previous studies (Grau *et al.*, 2005; Volante *et al.*, 2006) and the national forest monitoring program (UMSEF, 2007) have used Landsat satellite images to quantify the expansion of modern agriculture in the study area. In the present study, we expand these analyses, documenting the effects of traditional extensive grazing by quantifying the changes in the number and size of *puestos* between the early 1970s and 2002 using Landsat satellite imagery. *Puestos* are the core of the Chaco grazing system, and consist of a main house, a water reservoir, and corrals, typically surrounded by several hectares of bare soils due to severe vegetation degradation. The area of bare soil of each *puesto* can easily be quantified in remote sensing (Adámoli *et al.*, 1990; Blanco *et al.*,

2005, Figs 2 and 3), and we used this as an index of environmental degradation. The degradation due to grazing extends well beyond the area of each *puesto*, and the density of *puestos* is a good predictor of grazing activity, forest degradation (Blanco *et al*., 2005; Morello, 2005), and habitat quality for fauna (Altritcher & Boaglio, 2004; Altritcher *et al*., 2006).

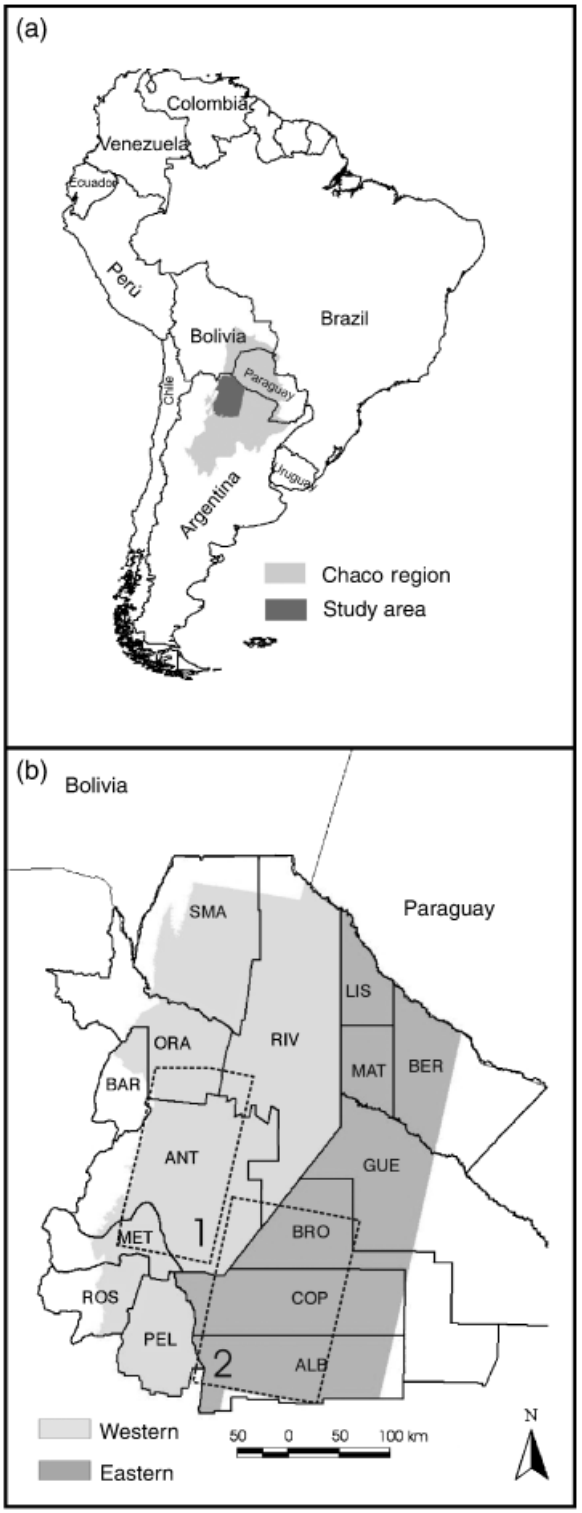

Fig. 1 Study area. (a) Location in South America. (b) Map of the study area showing political boundaries (solid lines) and the location of the two regions and the two sub-regions (1 and 2) used for the detailed analysis (dotted lines). Shaded sectors indicate the area included in the remote-sensing analysis. Human population and land-use data were obtained for the departments included in the two regions: Province of Salta: San Martín (SMA), Rivadavia (RIV), Oran (ORA), Anta (ANT), Metán (MET), Rosario de la Frontera (ROS); Province of Jujuy: Santa Bárbara (BAR); and Province of Santiago del Estero: Copo (COP), Alberdi (ALB), Pellegrini (PEL); Province of Chaco: Almirante Brown (BRO), Guemes (GUE); Province of Formosa: Matacos (MAT), Bermejo (VER), Ramón Lista (LIS). Although Copo and Alberdi represent an intermediate socieconomic situation they were analyzed as part of the eastern region.

Study area

The dry Chaco of northern Argentina (Fig. 1) is an extensive plain, relatively homogeneous in terms of environment and historical land use. The mean annual temperature varies between 20 and 22 °C, the mean January temperature varies between 24 and 27 °C, and the mean July temperature varies between 14.5 and 15.5 °C. Annual rainfall is minimal in the center of the study area (ca. 500 mm), and increases eastward, westward, and southward. Along the western and southern borders, rainfall reaches 700–900 mm, providing the opportunity for rainfed agriculture (Minetti, 1999).

The natural vegetation of the area is seasonally dry forest. Most of the area has typical semiarid Chaco vegetation dominated by *Aspidosperma quebracho-blanco*, *Schinopsis quebracho-colorado*, *Chorisia speciosa*, *Caesalpinea paraguariensis*, and *Prosopis* spp. (Cabrera, 1976). The soils of the region are formed by eolian sediments characteristic of the Chaco plains, and fluvial sediments that originate in the main rivers (Bermejo, Pilcomayo, Dorado, Del Valle, Juramento). The main edaphic limitations for agriculture include slope, salinity, soil texture, and water table depth (Nadir & Chafatinos, 1995).

For centuries, extensive grazing, logging, and the extraction of charcoal and firewood have degraded the Chaco ecosystems (Adámoli *et al*., 1990; Bucher & Huszar, 1999). But the dry Chaco is still the largest continuous area of Neotropical dry forest (Eva *et al*., 2004) and the largest subtropical habitat for many large vertebrates, including jaguars and three species of peccaries, one of them endemic in the region (Altritcher & Boaglio, 2004). Extensive grazing continues to be the most common land-use practice, but expanding global

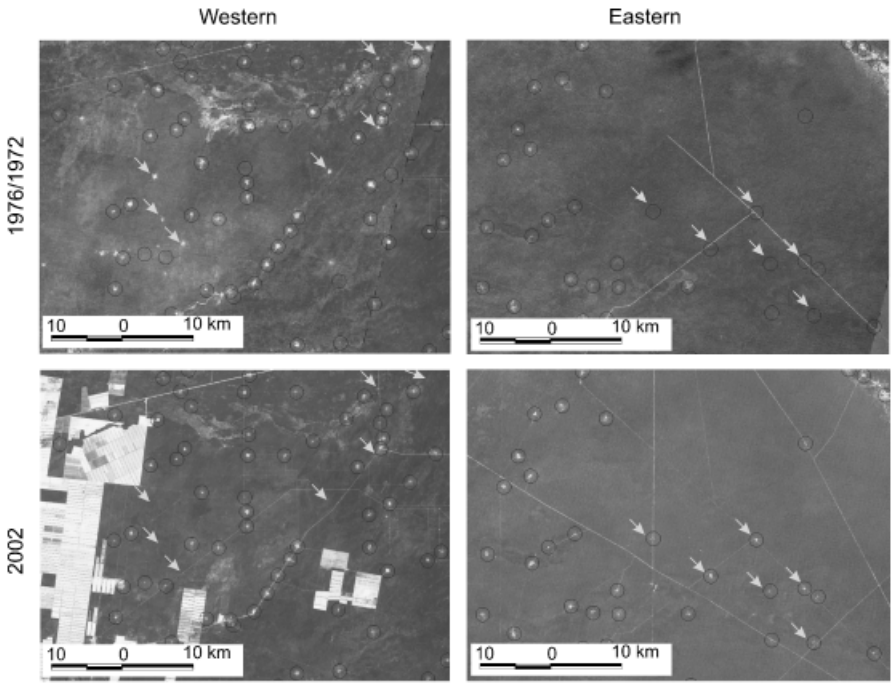

Fig. 2 Visual detection of *puestos* dynamics. Arrows indicate abandoned *puestos* in the western region and newly established *puestos* in the eastern region.

markets for soybean are promoting deforestation for agriculture expansion in areas where increasing rainfall is sufficient for rain-fed agriculture (Grau *et al*., 2005). These external forces have interacted with local provincial policies to generate different patterns of land-use change during the last 30 years. We compared the land-use efficiency of two regions that have experienced contrasting developmental pathways. Land use in the eastern region of our study area is dominated by extensive grazing (Bucher & Huszar, 1999; Altritcher & Boaglio, 2004; Altritcher *et al*., 2006) promoted through government subsides. The western side of the Chaco province has been a major target of government colonization programs, and both western Chaco, western Formosa, and northeastern Santiago del Estero host different development programs sponsored by NGOs, in part motivated by the comparatively large population of indigenous communities and the poor socio-economic conditions. In contrast, the western region (mostly in the province of Salta) did not have an active colonization policy promoting extensive grazing during the decades covered by this study, has a more developed road system, and is characterized by expanding soybean agriculture driven by global markets (Grau *et al*., 2005; Volante *et al*., 2006). This region has had less NGO-lead development projects for indigenous or creole local communities, favoring the promotion of modern agriculture (e.g. Cruz *et al*., 2006).

Puestos *mapping*

For the complete study area (Fig. 1), we determined the change in the number of *puestos* by comparing Landsat MSS images from the early 1970s and Landsat ETM + images for 2001 and 2002 considering only the area not affected by deforestation at the end of the period of analysis (i.e. not considering *puestos* that were converted to agriculture). These images were adequate for quantifying the change in the number of *puestos* (Fig. 2), but because they were taken at different times of the year, they were not used to determine the changes in *puestos* size.

Changes in *puestos* numbers were analyzed separately for the western and eastern regions (Fig. 1). We used Landsat images from 1972 to 1976 (MSS, spatial resolution 80 m by 80 m) and from 2001 to 2002 (ETM + , spatial resolution 30 m by 30 m, Table 1). MSS images were resampled to 30 m by 30 m pixels; and were coregistered (i.e. made spatially comparable) with the ETM + images with an error <30 m, using ground control points consisting of sites clearly visible (e.g. road crossings, field corners) during both periods (Richards & Xiuping, 2006). The map of *puestos* was constructed as a Geographic Information System (GIS) vector layer of points using visual interpretation for both periods on which we determined the location of *puestos* present in both dates, new *puestos*, and

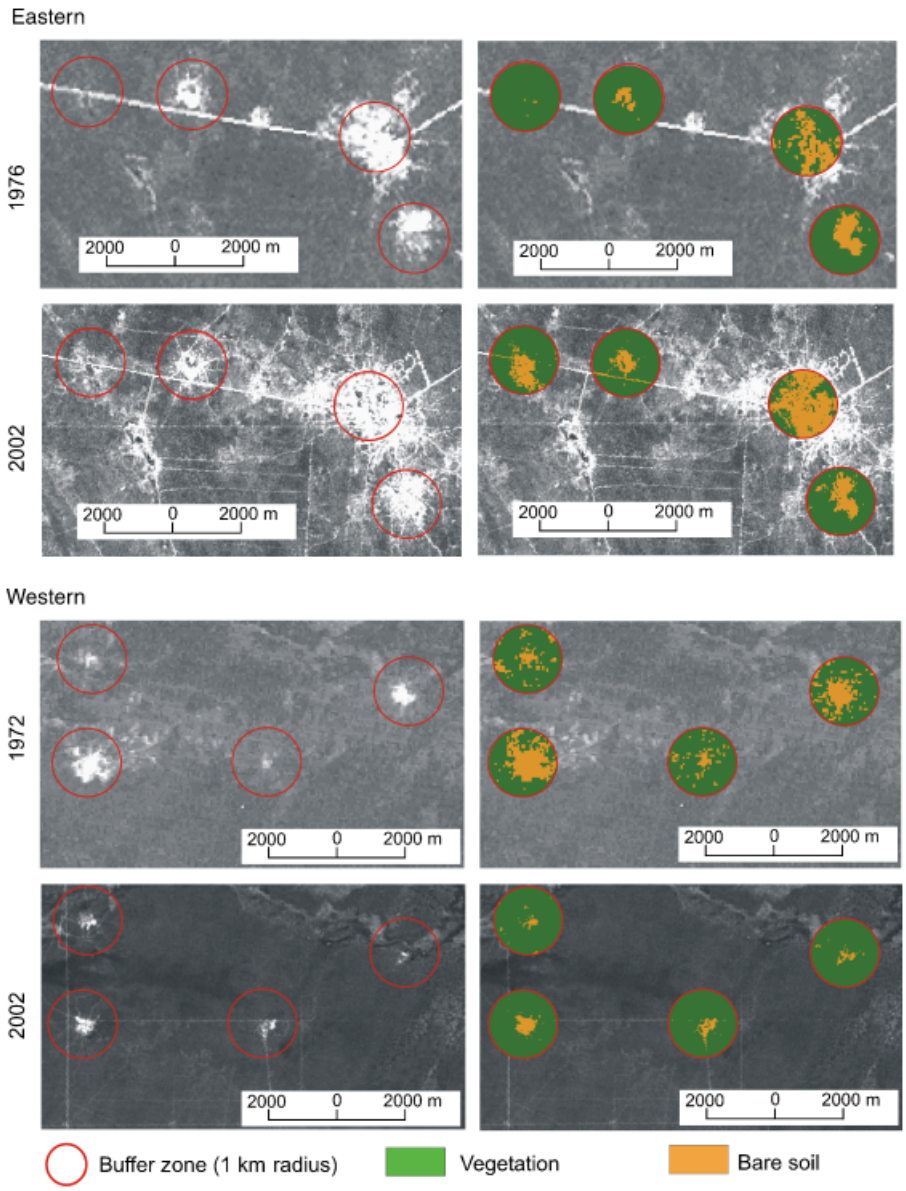

Fig. 3 Method of analysis of change in *puesto* size in the eastern and western regions. Left panels show monochronomatic pictures of band 3 (red) where bare soil appears as bright white and vegetation as dark gray (the MSS images are calibrated to 2002 reflectance values). Right panels indicate the classified image within circles that are centered at each puesto with 1 km of radius: green indicates vegetation and orange indicates bare soil.

abandoned *puestos*. To validate the 2001–2002 classification, in May 2006 we visited 30 sites classified as active *puestos*, and 13 sites classified as abandoned *puestos* (i.e. visible in the 1970s, but not in 2002). All 30 active *puestos* were correctly classified. Ten of the 13 sites classified as abandoned had clearly been abandoned and they were now covered with secondary forests. Two sites still had a house, but interviews with the owners confirmed a substantial decrease in animal density during the past few years. The remaining site was a small water reservoir, which attracts domestic animals and showed similar signs of a decrease in grazing.

Changes in puesto *size*

To analyze changes in *puesto* size, we conducted a more detailed analysis within the eastern and western regions (Fig. 1b) using radiometrically corrected images (Richards & Xiuping, 2006) taken during the same time of the year. To compare the change in *puesto* size

Table 1 Dates, locations and coregistration errors of the Landsat scenes used for the analysis

Path	Row	Date	Coregistration error (m) (GCPs)	Sensor
245	76	03/29/1976	40.8 (7)	MSS
245	77	03/29/1976	38.4 (7)	MSS
245	78	03/11/1976	42.2 (7)	MSS
246	76	03/30/1976	41.1 (10)	MSS
246	77	03/30/1976	40.8 (10)	MSS
246	78	02/23/1976	41.2 (10)	MSS
246	79	02/28/1975	41.5 (10)	MSS
247	76	11/08/1975	40.8 (10)	MSS
247	77	09/03/1972	34.4 (10)	MSS
247	78	09/03/1972	38.7 (10)	MSS
247	79	09/03/1972	29.3 (10)	MSS
229	76	07/28/2001	Reference	ETM +
229	77	07/15/2002	Reference	ETM +
229	78	09/03/2002	Reference	ETM +
230	76	08/23/2002	Reference	ETM +
230	77	09/24/2002	Reference	ETM +
230	78	06/20/2002	Reference	ETM +
230	79	08/04/2001	Reference	ETM +

GCPs is the number of ground control points used for the coregistration.

between the 1970s and 2002, we used images differing in no more than 20 calendar days, so that classification differences should not be attributed to changes in plant phenology. Because of limitations in the availability of good-quality MSS images, this analysis was restricted to a portion of the study area (boxes 1 and 2 in Fig. 1b, for the western and eastern regions, respectively). We used images that correspond to the overlapping areas of Landsat MSS path 247, row 77 images (09-03-1972); and ETM + path 230, row 77 (09-23-2002) for the western region; and the overlapping areas of MSS path 246, row 78 (02-23-1976), and ETM + path 229, row 78 (03-09-2002), for the eastern region (Table 1).

To make the bands from the ETM + and MSS sensors comparable, the values of bands 2 (green), 3 (red), and 4 (near infrared) of MSS were transformed into equivalents of reflectance for the year 2002 using a method of cross calibration assuming a linear relationship between sensors (Song *et al.*, 2001; Chen *et al.*, 2005). This was done by selecting 400 visually nonvariant pixels through time in the different land cover categories (bare soil, grasslands/degraded vegetation, forests) to construct a linear regression using the DN (Digital Numbers) for the MSS as an independent variable and the DN in the 2002 ETM + image as a dependent variable. The regressions obtained were applied to transform the DN of the early 1970s into new values radiometrically equivalent to the images of the year 2002 in the rest of the scenes (i.e. outside the nonvariant pixels). The correlation coefficients for all regressions were >0.84. To assess the quality of the image crosscalibration, we fit a new linear regression to ca. 100 independent points in each pair of images, using the transformed MSS values as an independent variable and the ETM + values as a dependent variable. In all cases, correlation coefficients (r) were >0.8, slopes within the 0.92–1.04 range (1 is optimal), origin close to zero (optimal), and errors <16 over a 0–256 scale.

In most cases a 1 km radius included all the bare soil assigned to one *puesto* and minimized overlapping with the area corresponding to neighboring *puesto*. Hence, *puesto* size was determined as the area of bare soil within a 1 km radius of the center of each *puesto*. To estimate the area of bare soil, we first tested the accuracy of distinguishing it from the other dominant land covers (i.e. grasslands-degraded vegetation and forest). In the area of 1 km of radius centered at each one of 10 *puestos* for each scene, we visually defined the region of interest of three land cover categories: bare soil, grasslands-degraded vegetation, and forests. These areas were used to compute the spectral signatures for each class in 2002. Separability (transformed divergence) between forest and the other classes was always >1.94, whereas separability between bare soil and grassland-degraded vegetation was 1.7 (separability varies between 1 and 2, where 2 is the maximum). These spectral signatures for each class were used to conduct a classification of both the 2002 and the crosscalibrated 1970s' images using a maximum likelihood method (Richards & Xiuping, 2006). Finally, the classes grasslands-degraded vegetation and forests were merged into a single one (vegetation) to obtain a resulting binary image (bare soil-vegetation). The accuracy of the 2002 classification was assessed by groundtruthing 44 points in the 230/77 scene (20 of bare soil and 24 of vegetation) and 51 points in the 229/78 scene (28 of bare soil and 23 of vegetation). The resulting accuracy (percent of cases in which the satellite-defined class corresponded to the ground observation within 30 m of the GPS-based location) was 90% and 82%, respectively. We used the area classified as bare soil within the radius of 1 km from the *puesto* center as an index of *puesto* size. To obtain these data for each individual *puesto*, we used a GIS combining a raster layer to map the cover of bare soil, and a vector layer to define the area of analysis (circles around each *puesto*, Fig. 3). The distributions of *puesto* size for the two dates were compared using a paired t-test (Sokal & Rohlf, 1995).

Population and land-use statistics

Population and land-use data (area of agriculture and planted pastures) were based on national censuses at the department level, the smallest jurisdiction for which data are available (INDEC, 2001, 2002). We report the total for all departments included in each study region. Because the study area represents only a portion of these departments (Fig. 1b), the values of rural population, pastures, and agriculture are greater than the actual values for the study area, but we assumed that they reflect the overall pattern for the region. The changes in population and land use in the departments fully included in the study area (Rivadavia and Pellegrini for the western region, and Ramon Lista, Matacos, and Copo for the eastern region, Fig. 1b) showed trends similar to the overall region trends, supporting this assumption. To correct for the different impact of grazing by cattle, sheep, and goats, we assumed that six goats or sheep were equivalent to one cow (Cocimano *et al*., 1977), and all livestock data are reported as animal units (AU), which is equivalent to cattle units.

Efficiency in food production

To estimate the efficiency of food production in the two regions, we computed two indices: protein production and potential meat production.

Protein production was estimated as the total protein produced in six land-use types: two types of livestock production (planted pastures and *puestos*) and the four most important crops, which represent >95% of the total area under cultivation (INTA, 2006; Volante *et al*., 2006): soybean (72% of the area), corn (5%), wheat (9%), and beans (10%). Based on analyses indicating the livestock affects vegetation up to approximately 5 from the *puesto* (Blanco *et al*., 2005; Morello, 2005), we assumed that the area of forest under livestock production was the area within 5 km of each *puesto* (i.e. 47% in the western sector and 56% of the forests in the eastern region according to our GIS analysis), and in this area we assumed that animal density was the optimal for the region (0.154 AU ha^{-1}, Martin, 1999). The area of planted pastures was obtained from the national censuses, and was assumed to have a density of 0.5 AU ha^{-1}, a conservative density for the region (Martin, 1999). Cropland area was estimated as the deforested area (from previous remote-sensing analyses; Grau *et al*., 2005; UMSEF, 2007) minus the area under planted pastures obtained from the national statistics. Percent of each crop was based on satellite imagery analysis for all the departments including the study area (Volante *et al*., 2006), and we assumed that the same proportion of crops occurred across the area included in the analysis.

Table 2 Productivity of agriculture and livestock land uses. (a) Estimated area, yield and protein content of the most common crops. (b) Estimated area, yield and protein content of livestock management in the two regions

(a)

Crop	Western region area (ha)	Eastern region area (ha)	Average yield (kg ha^{-1})	% protein
Soybean	400 611	39 863	2600	35
Wheat	89 273	2076	3500	16
Beans	140 746	0	1000	22
Corn	67 891	133	4500	10

(b)

Livestock production	Western region Area (ha)	Eastern region Area (ha)	Kg AU^{-1} yr^{-1}	% protein
Puestos	2 258 307	2 538 019	15.4	30
Pastures	110 000	13 000	50	30

Based on INDEC (2002), INTA (2006), Volante *et al*. (2006).

To determine protein production, we estimated the total area of each crop and livestock type, we multiplied this by the average per-hectare yield of each land use, and the average percent protein in the final product, either grain or meat (Table 2). To estimate potential meat production we assumed that all grain was used to feed pork with a feeding efficiency of 5.5 kg of food per kg of living weight, a conservative estimate considering that the other major use of soybean is for poultry production with an efficiency of 2–3 kg per kg of living weight (Smil, 2000).

Results

The analysis showed that the two regions differed clearly in their land-use patterns (Table 3). At the regional scale, the eastern region, which was not affected by agriculture expansion, lost 25 *puestos*, but gained 39 new *puestos*. In contrast, the western region lost 99 *puestos*, and did not gain any new *puesto* (Fig. 4). In the local detailed analysis, 63% of the *puestos* in the eastern region increased in size (paired *t*-test, $P<0.001$, $n=484$), and the mean *puesto* size increased by 25%. In the western region, 91% of the *puestos* decreased in size (paired *t*-test, $P<0.001$, $n=106$) and the mean size of the *puestos* decreased by 41%. This change in the number and size of *puestos* was also reflected in the government census data of the last three decades. The number of livestock increased by 131% in the departments of

Table 3 Changes in land use efficiency in the western and eastern regions of the northern Argentina dry Chaco

Variable	Western region	Eastern region
Total area (ha)	7 475 683	5 954 263
New *puestos*	0	39
Abandoned *puestos*	99	25
Puestos increasing size	8%	63%
Puestos decreasing size	91%	34%
Livestock 1960	523 000	446 000
Livestock 2001	496 000	1 077 000
Pastures 1960 (ha)	19 000	4500
Pastures 2001 (ha)	110 000	13 000
Rural population 1970	113 500	68 200
Rural population 2001	108 679	79 600
Deforestation (ha)	808 521	55 073
Average protein production (kg ha^{-1})	65.6	8.4
Average potential meat production (kg of living weight ha^{-1})	54.1	16.7

The number of new and abandoned *puestos* is for the complete study area (Fig. 4). Changes in *puesto* size were analyzed in two sub-regions of the study area (Fig. 1b). National censuses data on human population and land use are department totals, even if only part of a department is included in the study area (Fig. 1b).

the eastern region and decreased by 5.2% in the departments of the western region, where, in addition, the area of managed pasturelands increased by 91 000 ha, reflecting an intensification of cattle production that contributed to reduction in the density of livestock in forested areas. The decrease in the number and size of *puestos* in the western region was associated with a 4% decrease in the rural population between 1970 and 2001, while during the same period the rural population in the eastern region increased by 17% (Table 3).

Along with the differences in livestock production, agricultural activities also differed between the two regions, and this has important implications for food production. In the eastern region, agriculture is limited, and between 1972 and 2002 only 1.3% of the total area was deforested. In contrast, in the western region, >800 000 ha (ca. 16% of the region) were deforested, mainly for agriculture. This increase in agriculture production, dominated by soybeans with high productivity and protein content, has increased the average protein production of the western region to 65.6 kg ha^{-1}, compared with only 8.4 kg ha^{-1} in the eastern region (Table 3). Even if we assume that all grain production is used for animal feed (i.e. potential meat production), the western region will still have a productivity equivalent to 324% of the productivity of the eastern region (54.1 vs. 16.7 kg of living weight ha^{-1}, Table 3). Furthermore, to produce the equivalent amount of protein or meat currently produced in the 4.7 million hectares of forests surrounding the *puestos* in a 5 km buffer, only 16 000 or 150 000 hectares, respectively, would have to be transformed into soybean fields.

Discussion

The two development pathways differed clearly in their land-use efficiency, assessed both in terms of food production and condition in the remaining natural areas. In the eastern region, government incentives have increased the rural population, as well as the number of *puestos* and livestock, but food production is low and forest degradation has been extensive. The effects of the *puestos* on vegetation cover and biomass have been detected up to 5 km from the *puesto* center (Blanco *et al*., 2005; Morello, 2005), and >50% of the area not deforested (ca. 6 million hectares) is within 5 km of a *puesto*, implying a widespread impact on vegetation. In addition, the *puestos* system can have a large impact on the native fauna. For example, jaguar density in the region shows a clearly decreasing trend in association with the establishment of *puestos* (Altritcher *et al*., 2006) and the density of *puestos* is the best human-related predictor of peccary density, one of the most important game species (Altritcher & Boaglio, 2004). This pattern is consistent with examples of other neotropical regions including Central America (Kaimowitz, 1995), the Amazon basin (Hecht, 1993), and northeastern Brazil (Moran *et al*., 1996), where government incentives for cattle ranching have favored environmental degradation without much economic benefit. In contrast, in the western region, market-driven agriculture expansion has resulted in the deforestation of >800 000 ha. But, at the same time, grazing intensity in the remaining forests reflected in the number and size of *puestos* has decreased, and this will facilitate the recovery of the flora and fauna in >5 million hectares that have not been deforested. A significant proportion of these remaining forests are unlikely to be converted into agriculture under the current climatic and technological conditions, because they are too dry to conduct profitable rainfed agriculture (Grau *et al*., 2005). These conditions could change if rainfall continues to increase, new drought-resistant soybean cultivars are developed, or increasing demands from global markets for meat or biofuels stimulate the expansions of dryland agriculture or grazing. In such cases, the pattern of land-use disintensification observed in this study may only reflect a transient condition leading in the end to a full-scale

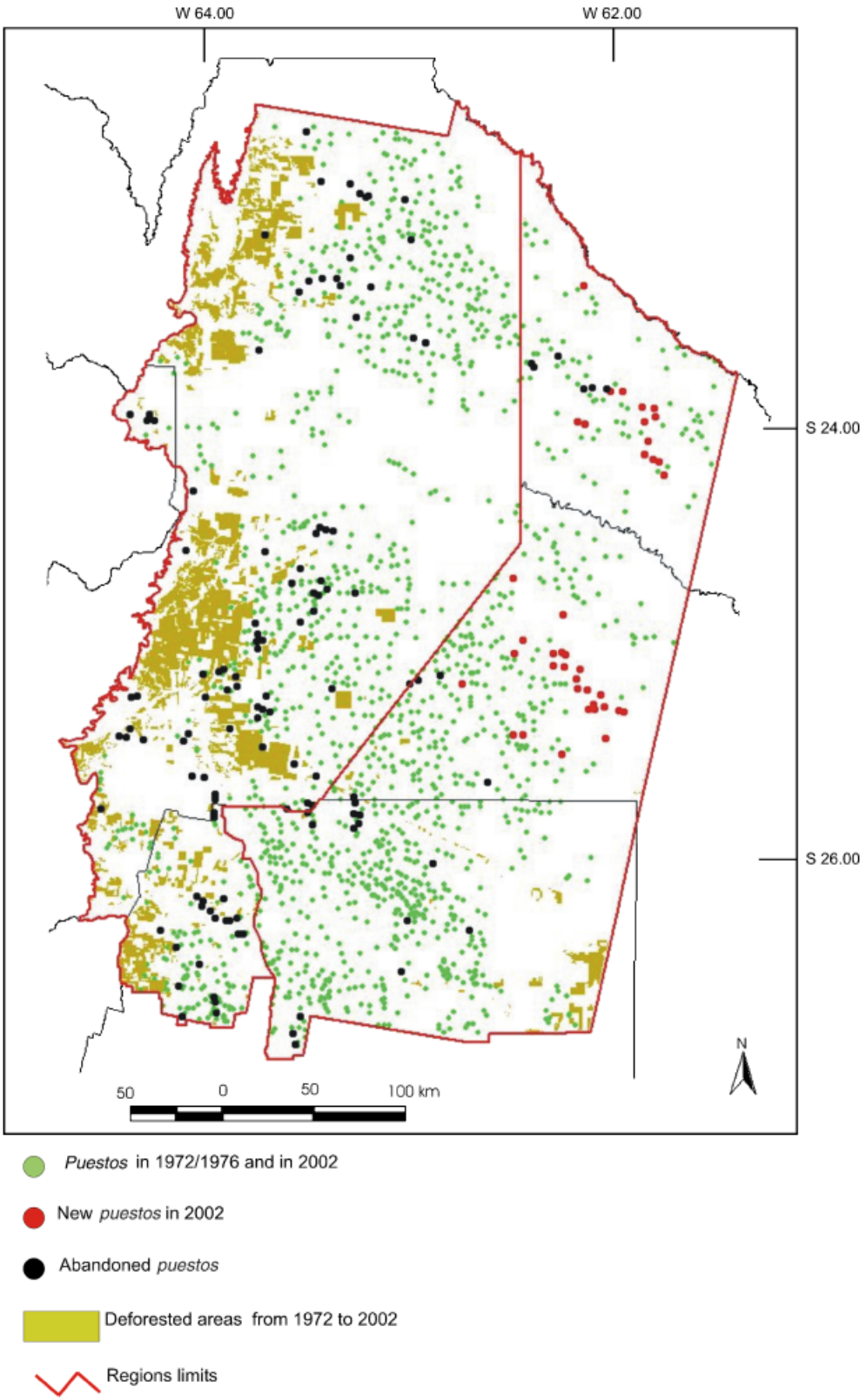

Fig. 4 Distribution of *puestos* in the study area: green = present in the 1970s and 2002; red = present in 2002, but not in the 1970s; blue = present in the 1970s, but not in 2002; maroon = deforested area by 2002 (based on UMSEF, 2007).

transformation of the Chaco ecosystems. But, if the current conditions prevail, and without any incentives derived from government subsides or NGO's support, livestock production based on rural populations living in extreme poverty and lacking basic services should continue to decline, leading to the abandonment of marginal lands, rural–urban migration, and ecosystem recovery, following the process that has been hypothesized as a key factor mitigating biodiversity losses at the continental scale (Aide & Grau, 2004; Wright & Muller-Landau, 2006).

We have focused on two of the most important and frequently conflicting (De Fries *et al*., 2004; Mattison & Norris, 2005) aspects of the land-sparing argument: food production and biodiversity conservation, but there are other factors that should be considered when developing regional- or national-scale land-use policies, including the side effects of human migration patterns and extended forest degradation due to fragmentation and border effects. Contrary to examples from Brazil (Angelsen & Kaimowitz, 2001; Fearnside, 2001), agriculture expansion in our study region has not increased immigration in rural areas. In fact, in the western region, agricultural expansion has been accompanied by rural–urban migration, which should contribute to the recovery of nondeforested areas due to the reduction in grazing, selective logging, fires, and hunting. Furthermore, because most agriculture in the region is rainfed, the deforested areas are concentrated in the western limit of the region in an area of relatively high rainfall (Grau *et al*., 2005). This pattern of deforestation has left large continuous areas of forest (Fig. 4), which is very different from the highly fragmented landscape observed in other regions (e.g. mountains or areas dominated by shifting agriculture, Vandermeer & Perfecto, 2007).

Other factors that were not quantitatively evaluated are soil and water degradation, and social impacts. In the region, soils are deep and fertile, and irrigation is very limited, and thus fertilizer contamination and effects on water courses should be limited (Clair & Ehrman, 1996; Matson *et al*., 1997). In addition, the topography is flat and the dry conditions (i.e. negative water balance) will further reduce runoff. In contrast, pesticides, in particular herbicides, are an important component of the soybean production system that could affect adjacent areas. As expected from any highly productive agriculture crop in the region, soybean exports significant amounts of nutrients through harvesting and soil erosion, and after several decades of soybean cultivation, soils show compaction and depletion of organic matter (Zinck *et al*., 2006). However, soybean yields throughout the region have increased by a factor of 2.5 over the last 30 years (SAGPyA, 2007), suggesting that modern agriculture technology, including the use of fertilizers, and pesticides, overcomes this potential problem for sustainability at least at the scale of decades.

The social impacts of modern agriculture, in this case the transition from extensive grazing to soybean production, are complex, and may include both potentially positive such as a growing economy and improving socioeconomic conditions as people move to urban areas (Polèse, 1998), and potentially negative effects such as the deterioration of local living conditions and the displacement of local communities, including indigenous ethnic groups (Van Damm, 2002). A preliminary analysis at the country scale showed no negative relationship between soybean expansion and local living conditions (Paruelo & Oesterheld, 2005), but, within the Chaco, some indigenous groups have been directly affected (Torrella & Adámoli, 2006). The present study has taken an important first step toward understand how to balance food production and conservation in this region of the Chaco. Overall, the environmental and socioeconomic benefits of modern agriculture and land sparing appear to outweigh the negative impacts, but it is important that the negative impacts (e.g. local communities' rights, ecological externalities due to the use of fertilizers and pesticides) are quantified and compared with these benefits.

Neotropical dry forests are one of the most threatened major biomes in the world (Janzen, 1988; Hoekstra *et al*., 2005; Miles *et al*., 2006). Much of this threat has resulted from historical traditional land uses including timber and firewood extraction, hunting, and, in particular, extensive cattle ranching (Janzen, 1988; Bucher & Huszar, 1999). In recent times, modern agriculture expansion has emerged as a growing threat (Zak *et al*., 2004; Grau *et al*., 2005; Paruelo & Oesterheld, 2005; Silva *et al*., 2006). It is often assumed that this new form of environmental degradation represents an addition to the degradation forces that not only increases the intensity of human impact in the newly deforested areas but also favors environmental degradation in areas not deforested for modern agriculture, by displacing traditional land uses into the shrinking seminatural landscapes (Dros, 2004; Paruelo *et al*., 2005). Our results show that this does not need to be the case. In fact, agriculture expansion into the most productive areas (in our system represented by sectors with higher rainfall) may well be associated with a reduction in the intensity of traditional land use in the remaining areas. The recovery of other neotropical dry forest ecosystems in association with socioeconomic changes (e.g. Bray & Klepeis, 2005; Jepson, 2005) suggests that these dynamics are not restricted to the Argentine Chaco and have important implications for conservation and man-

agement policy of dry forests. Extensive cattle ranching has historically been a major land-use type in tropical dry ecosystems. A shift from a system in which the production of calories and proteins is based on extensive cattle ranching to one based on modern intensive agriculture represents a major increase in the efficiency of food production (Smil, 2000). Our study documents a case in which this shift may contribute to both local conservation of dry forests and to the global balance between food production and nature conservation.

The global population is predicted to increase by approximately 3 billion people during the next 50 years (Lutz *et al*., 2001), and at the same time, per capita food consumption is expected to increase (Smil, 2000; Balmford *et al*., 2005). To meet this growing food demand and the economic benefits for food-producing regions without sacrificing extensive natural areas, we must maximize land-use efficiency (Waggoner & Ausubel, 2001; De Fries *et al*., 2004; Balmford *et al*., 2005; Green *et al*., 2005; Mattison & Norris, 2005). Given the current global trends of increasing demand for soybean, technological advance, and increasing precipitation in the region, deforestation is likely to continue in areas of the Chaco that are suitable for modern agriculture (Grau *et al*., 2005; Paruelo & Oesterheld, 2005). In this study, the expansion of modern agriculture has greatly increased food production and the reduction in extensive grazing has allowed forest recovery over millions of hectares of remaining forest areas. This is a more efficient land-use 'strategy' in comparison with subsides for extensive grazing in the eastern region, providing overall support for the land-sparing hypothesis. But will the benefits of high food production and improved biodiversity conservation continue over the long-term? Although the present socioeconomic forces have produced these results in the absence of active conservation efforts, we believe it is critical that future planning efforts include formal protection and social considerations to maintain these conservation gains in the long-term. This can be accomplished if government agencies and conservation organizations take advantage of the conservation opportunities provided by agriculture modernization and the reduced opportunity costs in areas experiencing land-use disintensification to promote sound land-use policy based on comprehensive analyses of land-use efficiency.

Acknowledgements

Financial support for this research was provided by InterAmerican Institute (IAI), the Argentine Agency of Science and Technology (FONCYT), Fundación Antorchas, Rufford Small Grants for Nature Conservation, and the FIPI program of the University of Puerto Rico. Images and spatial information were obtained from the Global Land Cover Facility of the University of Maryland, the Unidad de Manejo del Sistema de Evaluación Forestal Argentina (UMSEF), and the Comisión Nacional de Asuntos Aeroespaciales (CONAE), Argentina. Forest cover map was provided by the UMSEF. Leo Paolini provided advice on the remote-sensing methods, and the comments of two anonymous reviewers improved the manuscript.

References

Adámoli J, Senhauser E, Acero JM *et al*. (1990) Stress and disturbance: vegetation dynamics in the dry Chaco region. *Journal of Biogeography*, **17**, 491–500.

Aide TM, Grau HR (2004) Globalization, migration, and Latin American ecosystems. *Science*, **305**, 1915–1916.

Altritcher M, Boaglio GI (2004) Distribution and relative abundance of peccaries in the Argentine Chaco: associations with human factors. *Biological Conservation*, **116**, 217–225.

Altritcher M, Boaglio GI, Perovic P (2006) The decline of Jaguars *Panthera onca* in the Argentine Chaco. *Oryx*, **40**, 302–309.

Angelsen A, Kaimowitz D (2001) Introduction: the role of agricultural technology in tropical deforestation. In: *Agricultural Technologies and Tropical Deforestation* (eds Angelsen A, Kaimowitz D), pp. 19–34. CABI Publishing, Wallingford, UK.

Asner GP, Knapp DE, Broadbent EN *et al*. (2005) Selective logging in the Brazilian Amazon. *Science*, **310**, 480–482.

Balmford A, Green RE, Scharlemann JPW (2005) Sparing land for nature: exploring the potential impact of changes in agricultural yield on the area needed for crop production. *Global Change Biology*, **11**, 1594–1605.

Barbier EB, Burguess JC (1997) The economics of tropical land use options. *Land Economics*, **73**, 174–195.

Blanco L, Biurrun F, Ferrando C (2005) *Niveles de Degradación de la Vegetación del Chaco Árido*. Instituto Nacional de Tecnología Agropecuaria, Buenos Aires, Argentina.

Bray DB, Klepeis P (2005) Deforestation, forest transitions, and institutions for sustainability in Southeastern Mexico, 1900–2000. *Enviromental History*, **11**, 195–223.

Bucher EH, Huszar PC (1999) Sustainable management of the Gran Chaco of South America: ecological promise and economic constraints. *Journal of Environmental Management*, **57**, 97–108.

Cabrera AL (1976) *Regiones Fitogeográficas de Argentina*. ACME, Buenos Aires, Argentina.

CEPAL (1999) *Latin America: Urban and Rural Population Projections 1970–2025*. CEPAL, Santiago, Chile. Online version: http://www.eclac.cl/publicaciones/xml/0/5160/BD63.html

Chen X, Vierling L, Deering DA (2005) A simple and effective radiometric correction to improve landscape change detection across sensors and across time. *Remote Sensing of the Environment*, **98**, 63–79.

Clair TA, Ehrman JM (1996) Variation in discharge and dissolved organic carbon and nitrogen export from terrestrial basins: a neural network approach. *Limnology and Oceanography*, **41**, 921–927.

Cocimano M, Lange A, Membielle E (1977) *Equivalencias Ganaderas para Vacunos de Carne y Ovinos. Colección Estudios y*

Métodos. Departamento de Estudios, AA-CREA, Buenos Aires, Argentina.

Cruz N, Casavelos J, Ezcurra E (2006) Caso Pizarro: desafectación y recuperación de una reserva natural. In: *Situación Ambiental Argentina* (eds Brown AD, Martinez-Ortiz U, Acerbi M, Corcuera J), pp. 62–67. Fundación Vida Silvestre, Buenos Aires, Argentina.

Daily GC (2001) Ecological forecast. *Nature*, **411**, 245–245.

De Fries RS, Foley JA, Asner GP (2004) Land use choices: balancing human needs and ecosystem function. *Frontiers in Ecology and the Environment*, **2**, 249–257.

Donald PF, Evans AD (2006) Habitat connectivity and matrix restoration: the wider implications of agri-environment schemes. *Journal of Applied Entomology*, **43**, 209–218.

Dros JM (2004) *Managing the Soy Boom: Two Scenarios for Soy Production Expansion in South America*. Aidenvironment, Amsterdam, the Netherlands.

Eva HD, Belward AS, De Miranda EE *et al*. (2004) A land cover map of South America. *Global Change Biology*, **10**, 731–744.

Fearnside PM (2001) Soybean cultivation as a threat to the environment in Brazil. *Environmental Conservation*, **28**, 23–38.

Grau HR, Gasparri NI, Aide TM (2005) Agriculture expansion and deforestation in seasonally dry forests of northwest Argentina. *Environmental Conservation*, **32**, 140–148.

Green RE, Cornell SJ, Scharlemann JP *et al*. (2005) Farming and the fate of wild nature. *Science*, **307**, 550–555.

Hecht SB (1993) The logic of livestock and deforestation in Amazonia. *Bioscience*, **43**, 687–695.

Hecht SB (2005) Soybeans, development and conservation on the Amazon frontier. *Development and Change*, **36**, 375–404.

Hoekstra JM, Boucher TM, Ricketts TH *et al*. (2005) Confronting a biome crisis: global disparities of habitat loss and protection. *Ecology Letters*, **8**, 23–29.

INDEC (2001) *Censo Nacional de Población y Vivienda 2001. Resultados Generales*. Instituto Nacional de Estadísticas y Censos, Buenos Aires, Argentina. Online version: http://www.indec.gov.ar/wecenso/index.asp

INDEC (2002) *Censo Nacional Agropecuario 2002*. Instituto Nacional de Estadísticas y Censos, Buenos Aires, Argentina. Online version: http://www.indec.mecon.gov.ar/agropecuario/cna_principal.asp

INTA (2006) *Umbral al Chaco con Cultivos de Secano Extensivos.2006*. Instituto Nacional de Tecnología Agropecuaria, Buenos Aires, Argentina. Online version: http://www.inta.gov.ar/tusgo/info/zonas/chacosecano.htm

Janzen DH (1988) Tropical dry forests: the most endangered major tropical ecosystem. In: *Biodiversity* (ed. Wilson EO), pp. 130–137. National Academy Press, Washington, DC.

Jepson WA (2005) A dissapearing biome? Reconsidering land cover change in the Brazilian savanna. *Geographical Journal*, **171**, 99–111.

Kaimowitz D (1995) *Livestock and Deforestation in Central America in the 1980s and 1990s. A Policy Perspective*. International Food Policy Research Institute, Washington, DC.

Kaimowitz D, Smith J (2001) Soybean technology and the loss of natural vegetation in Brazil and Bolivia. In: *Agricultural Technologies and Tropical Deforestation* (eds Angelsen A, Kaimowitz D), pp. 195–211. CABI Publishing, Wallingford, UK.

Lutz W, Scherbow SEDT, Sanderson WC (2001) The end of world population growth. *Nature*, **412**, 543–545.

Martin G (1999) *Hacia un Manejo Sustentable del Pastoreo en el Chaco Semiárido*. Facultad de Agronomía, Universidad Nacional de Tucumán, Argentina.

Mather AS, Needle CL (1998) The forest transition: theoretical basis. *Area*, **30**, 117–124.

Matson PA, Parton WJ, Power AG *et al*. (1997) Agricultural intensification and ecosystem properties. *Science*, **277**, 504–509.

Mattison EHA, Norris K (2005) Bridging the gaps between agricultural policy, land use, and biodiversity. *Trends in Ecology and Evolution*, **20**, 610–616.

Miles L, Newton A, De Fries RS *et al*. (2006) A global overview of the conservation status of tropical dry forests. *Journal of Biogeography*, **33**, 491–505.

Minetti JL (1999) *Atlas Climático del Noroeste Argentino*. Laboratorio Climatológico Sudamericano, Fundación Zon Caldenius, Tucumán, Argentina.

Moran EF, Packer A, Brondizio E *et al*. (1996) Restoration of the vegetation cover in the eastern Amazon. *Ecological Economics*, **18**, 41–54.

Morello J (2005) Entrando en el Chaco con y sin consentimiento de la Naturaleza. *Vida Silvestre*, **92**, 18–23.

Morton DC, De Fries R, Shimabukuro YE *et al*. (2006) Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. *Proceedings of the National Academy of Sciences*, **103**, 14637–14641.

Nadir A, Chafatinos T (1995) *Los Suelos del Noroeste Argentino: Salta y Jujuy*. Universidad Nacional de Salta, Salta, Argentina.

Nepstad DC, Verissimo A, Alencar A *et al*. (1999) Large-scale impoverishmen of Amazonian forests by logging and fire. *Nature*, **398**, 505–508.

Paruelo JM, Guershman JP, Verón SR (2005) Expansión agrícola y cambio de uso del suelo. *Ciencia Hoy*, **15**, 14–23.

Paruelo JM, Oesterheld M (2005) *Patrones Temporales y Espaciales de la Expansión de la Soja en la Argentina. Relación con Factores Socioeconómicos y Ambientales*. Argentina Rural Strategy, World Bank, IFEVA, Buenos Aires.

Polèse M (1998) *Economía Urbana y Regional. Introducción a la Relación entre Territorio y Desarrollo*. Libro Universitario Regional, Cartago, Costa Rica.

Redford KH (1992) The empty forest. *Bioscience*, **42**, 412–422.

Richards JA, Xiuping J (2006) *Digital Image Analysis. An Introduction*. Springer-Verlag, New York.

Rozenzweig M (2003) Reconciliation ecology and the future of species diversity. *Oryx*, **37**, 194–205.

SAGPyA (2007) *Secretaría de Agricultura, Ganadería, Pezca y Alimentos. República Argentina*. Online version: http://www.sagpya.mecon.gov.ar/

Sanchez-Azofeifa GA, Kalacska M, Quesada M, Calvo-Alvarado JC, Nassar JM, Rodríguez JP (2005) Need for integrated research for a sustainable future of tropical dry forests. *Conservation Biology*, **19**, 285–286.

Silva JF, Fariñas MR, Felfili JM, Klink CA (2006) Spatial heterogeneity, land use and conservation in the Cerrado region of Brazil. *Journal of Biogeography*, **33**, 536–548.

Smil V (2000) *Feeding the World: A Challenge for the Twenty-First Century*. MIT Press, Boston.

Sokal RR, Rohlf FJ (1995) *Biometry: the Principles and Practice of Statistics in Biological Research*. Freeman, New York.

Song C, Woodcock CE, Seto KC *et al*. (2001) Classification and change detection using Landsat TM data. When and how to correct atmospheric effects? *Remote Sensing of the Environment*, **75**, 230–244.

Steininger MK, Tucker CJ, Townshend JRG *et al*. (2001) Tropical deforestation in Bolivian Amazon. *Environmental Conservation*, **28**, 127–134.

Tilman D, Gassman KG, Matson PA *et al*. (2002) Agricultural sustainability and intensive production practices. *Nature*, **418**, 671–677.

Torrella SA, Adámoli J (2006) Situación ambiental de la ecorregión de Chaco seco. In: *Situación Ambiental Argentina* (eds Brown AD, Martinez-Ortiz U, Acerbi M, Corcuera J), pp. 75–82. Fundación Vida Silvestre, Buenos Aires, Argentina.

Tscharntke T, Klein AM, Kruess A *et al*. (2005) Landscape perspective on agricultural intensification and biodiversity–ecosystem service management. *Ecology Letters*, **8**, 857–874.

UMSEF (Unidad de manejo del Sistema de Evaluación Forestal) (2007) Monitoreo de la superficie de bosque nativo de Argentina. Online version: http://www.ambiente.gov.ar/?idarticulo=311

Van Damm C (2002) *Ocupación, Degradación Ambiental, Cambio Tecnológico y Desarrollo Sostenible: Los Efectos de la Introducción del Paquete Soja/Siembra Directa en el Chaco Salteño*. Tesis de Maestria, FLACSO, Buenos Aires, Argentina.

Vandermeer J, Perfecto I (2007) The agricultural matrix and the future paradigm for conservation. *Conservation Biology*, **21**, 274–277.

Volante JM, Paoli HP, Bianchi AR *et al*. (2006) *Análisis de la Dinámica de Uso de Suelo Agrícola del Noroeste Argentino Mediante Teledetección y Sistemas de Información Geográfica, Periodo 2000–2005*. Instituto Nacional de Tecnología Agropecuaria, Salta, Argentina.

Waggoner PE, Ausubel JH (2001) How much feeding more and wealthier people will encroach on forests? *Population and Development Review*, **27**, 239–257.

Wright SJ, Muller-Landau HC (2006) The future of tropical forest species. *Biotropica*, **38**, 287–301.

Zak MR, Cabido M, Hodgson JG (2004) Do subtropical seasonal forests in the Gran Chaco, Argentina, have a future? *Biological Conservation*, **120**, 589–593.

Zinck JA, Flores E, Sayago JM (2006) Soil compactation and fertility depletion. Land use change and land degradation in the Western Chaco. *ITC Publication*, **84**, 169–231.