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The role of private lands for conservation: Land cover change analysis in the Caldenal savanna ecosystem, Argentina

Mariano González-Roglich [a,*], Jane Southworth [b], Lyn C. Branch [c]

[a] School of Natural Resources and Environment, University of Florida, 103 Black Hall, PO Box 116455, Gainesville, FL 32611, USA
[b] Department of Geography, Land-use and Environmental Change Institute, University of Florida, 3141 Turlington Hall, Gainesville, FL 32611, USA
[c] Department of Wildlife Ecology and Conservation, University of Florida, Newins-Ziegler Hall, Gainesville, FL 32611, USA

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A B S T R A C T

Protected areas are critical for conservation of the world's biodiversity; however, parks isolated from their surroundings will not assure the maintenance of biodiversity and ecosystem integrity. Private lands have the potential to achieve the dual role of conserving natural habitats while providing goods and services societies need. Therefore, understanding which land use practices on private land contribute to the maintenance of native habitats is important. In this study, a land use and land cover change analysis was developed for the Caldenal savanna ecosystem, Argentina, to determine the effect of private game reserves on landscape scale change. Game reserves were found preferentially located in the areas with highest proportion of forest cover. No differences were found in rates of conversion of native habitat to agriculture between game reserves and cattle ranches for 1987–1999 or 1999–2008. Rates of deforestation differed with landholding size between the first and second period. Deforestation stopped in the second period in the game reserves and big cattle ranches, but increased in intensity in small cattle ranches. Intensity of deforestation was more related to agricultural potential of the ranches, which are correlated with landholding size, than to the land use type. Deforestation was higher in areas with more productive soils and higher annual rainfall. These results suggest that, for the period analyzed, incorporation of game reserves as a land use did not significantly alter processes of land cover change, though if conversion to agriculture increases in marginal areas, game reserves potentially could be beneficial for maintenance of a significant portion of the Caldenal.

Introduction

Protected areas have been the cornerstone of conservation strategies to preserve threatened ecosystems (Margules & Pressey, 2000). However, degradation beyond their boundaries often compromises ecological objectives of protected areas (Clerici et al., 2007; Fynn & Bonyongo, 2011; Soulé & Terborgh, 1999), especially if they are small or isolated (Sanchez-Azofeifa, Rivard, Calvo, & Moorthy, 2002). A variety of mechanisms are being developed to favor conservation practices on private lands, such as conservation easements (Engel, 2007; Rissman et al., 2007; Rissman & Merenlender, 2008), payment for ecosystem services (Engel, Pagiola, & Wunder, 2008; Tallis, Kareiva, Marvier, & Chang, 2008; van der Horst, 2011), and taxes and subsidies to modify behavior (Jenkins, Scherr, & Inbar, 2004). However, in order to correctly implement any of these strategies, information on the conservation value of different land uses is needed.

The importance of wildlife utilization as a source of income in private landholdings has increased in recent years as an alternative to traditional rural production systems (e.g., cattle ranching), providing incentives for wildlife and habitat conservation (Butler, Teaschner, Ballard, & McGee, 2005; Williams & Lathbury, 1996). Wildlife utilization, as a land use, has proven effective in southern Africa from an economic and biodiversity conservation perspective (Bond et al., 2004). Trophy hunting is of major importance to conservation in Africa creating economic incentives for conservation over vast areas unsuitable for other activities (Lindsey, Roulet, & Romanach, 2007), although negative impacts also have been identified (Lindsey, Romanach, & Davies-Mostert, 2009; Packer et al., 2011). In North America, landowners agree to conduct habitat enhancement activities in order to obtain hunting benefits (Williams & Lathbury, 1996), and these benefits function as an incentive for land owners to conserve natural habitats in order to keep wildlife on their properties (Benson, 2001; Butler et al., 2005). Countries in Central and South America, particularly Argentina, are

* Corresponding author. Present address: Nicholas School of the Environment, Box 90328, Duke University, Durham, NC 27708, USA. Tel.: +1 919 613 8000.
E-mail address: mg136@duke.edu (M. González-Roglich).

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becoming important destinations for international sport hunting where hunters pay a fee for hunting on private land. Fee hunting as a conservation strategy has not been evaluated in Argentina, even though private conservation initiatives are especially important in this country because most land is privately owned.

The conservation value of different land uses can be assessed using a variety of methods, although the method selected needs to be able to identify and quantify processes that pose a threat to the maintenance of natural ecosystems. Remote sensing is a particularly effective tool for evaluation of landscape scale processes (Turner et al., 2003) because it offers an important means for detecting and analyzing changes over large temporal and spatial scales (Narumalani, Mishra, & Rothwell, 2004). Remote sensing is used for analyzing consequences of landscape scale land cover change of different management practices, such as regional agricultural policies (Serra, Pons, & Sauri, 2008), tourism development (Gaughan, Binford, & Southworth, 2009), urban development (Dewan & Yamaguchi, 2009), and establishment of protected areas (Southworth, Nagendra, Carlson, & Tucker, 2004). Use of remote sensing for determining effectiveness of protected areas is also common (e.g. Clerici et al., 2007), but we are not aware of similar studies analyzing the effect of fee hunting as a land use on processes of land cover change at landscape scales.

Savannas have been identified as one of the endangered "crisis ecoregions" worldwide (Hoekstra, Boucher, Ricketts, & Roberts, 2005). The Caldenal region in central Argentina is a transitional savanna ecosystem at the margin of the Pampas grasslands, and a reservoir of some of the best examples of perennial grasslands in the country. In the last century more than 40% of its area has been converted to agriculture. In addition to on-going habitat conversion, much of the remaining Caldenal is highly degraded, where the original open savanna characterized by scattered trees and dense cover of perennial grasses (Koutche & Carmelich, 1936) has been replaced by thick thorn scrub (Lell, 2004). Moreover, currently only 0.2% of this region is under formal protection, making this system one of the highest conservation priorities for the national government of Argentina (APN, 2007).

The degree to which different land uses on private landholdings maintain natural land cover is critical for long term sustainability of the threatened Caldenal savanna and its biodiversity because over 95% of the land is privately owned (Gobbi, 1994). The objective of this research was to assess the importance of private game reserves for conservation of native habitats in this region of central Argentina. Processes of land cover change were analyzed at a landscape scale with comparisons of land cover composition, rates of conversion to agriculture and rates of change of naturally vegetated land covers over time among game reserves, cattle ranches and the only protected area. Understanding the effect of different private land uses in the Caldenal region is critical for designing a regional conservation strategy to complement the very limited protected area system with conservation of productive ecosystems on private lands.

Study area

The Caldenal is a semiarid savanna ecosystem of about 170,000 km^2 located in central Argentina, primarily in La Pampa province. This xerophytic open forest system is a transitional ecosystem between the Pampas grasslands, to the east, and the dry Monte shrublands, to the west. It is dominated by the caldén tree (*Prosopis caldenia*) with understory of perennial grasses frequently interrupted by dunes, wetlands and lagoons (Cabrera, 1994). Major changes occurred in the area with the arrival of new settlers at the end of the 19th century and have intensified since then. Replacement of natural systems with agriculture, extractive logging, introduction of non-native species, overgrazing by livestock, and changes in fire regimes are the biggest threats the region has faced historically (Amieva, 1993; Medina, 2007; Mendez, 2007b).

Cattle ranching is the main economic activity in the region followed by crop production (Cano, Fernández, & Montes, 1980; Mendez, 2007a); however, hunting and tourism are growing as complementary activities, occurring simultaneously in these reserves. Fee hunting is organized in private rural enterprises called game reserves (*cotos de caza* in Spanish). In most of these reserves cattle ranching is still the main source of income, while fee hunting is second. Game activities are focused mainly on attraction of international tourists interested in hunting non-native species such as red deer (*Cervus elaphus*), wild boar (*Sus scrofa*), antelope (*Antilope cervicapra*) and buffalo (*Bubalus bubalis*) (Gobbi, 1994). These reserves formally began in La Pampa in 1987, and currently 56 reserves are located in the Caldenal region covering over 3222 km^2 (4.4% of the ecosystem in the province).

The area analyzed in this project (31,000 km^2) is located in the central portion of the Caldenal in La Pampa province. This area was selected based on the large number of game reserves in the area (Fig. 1). The study area, and La Pampa province in general, is

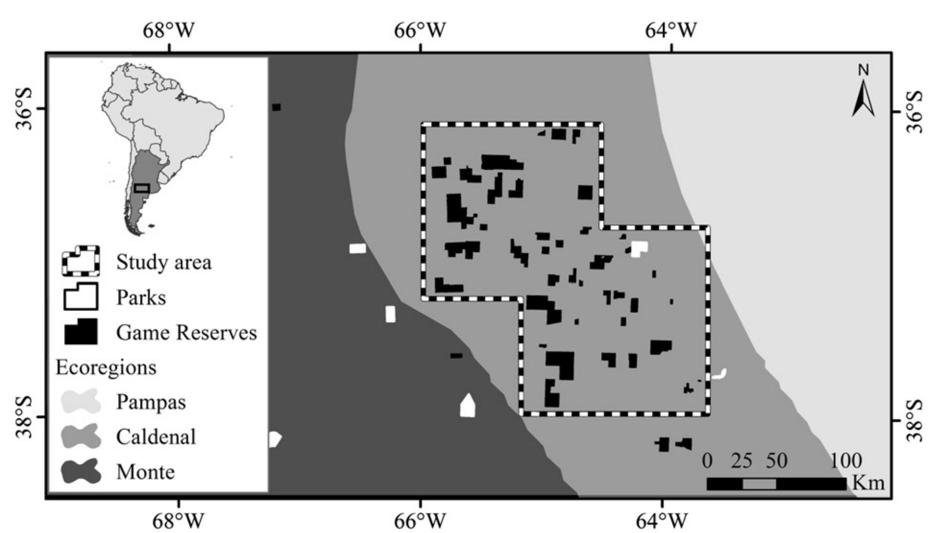

Fig. 1. Location of the study area, parks and game reserves in La Pampa province, Argentina.

characterized by steep northeast-southwest gradients. Rainfall decreases from 900 mm a year to 350 across only 250 km (Supplement Fig. 1). Soil types also follow the same gradient, with the most fertile soils in the northeast (Cano et al., 1980). Agrarian economic unit is the term used in local legislation to refer to the minimum amount of land a family needs to make a living (GLP, 1973). The size of these units follows the same pattern described by the natural gradients (Supplement Fig. 1). In this study, we refer to the land owned or managed by a single person or enterprise as a landholding, and given that management decisions are made in most cases by the direct user of the natural resource, we used the landholding as our unit of analysis.

Methods

Image processing and classification

Landsat 5 TM images were obtained for the end of the summer season for 1987, 1999 and 2008 (Supplement Table 1). All images were geometrically registered to a GIS layer of roads collected with a GPS during fieldwork (error < 15 m). Images were radiometrically calibrated and atmospherically corrected, converting digital numbers to surface reflectance to minimize distortion between dates allowing for more accurate time-series analysis (Green, Schweik, & Randolph, 2005). For each of the three dates, images were mosaicked to create a single image on which to perform classification procedures.

Six land cover classes were identified: agriculture, grassland, shrubland, forest (following the definition from FAO, 2010, p. 378), burned, and lagoon (see Table 1 for detailed descriptions). In this document, we will refer to grassland, shrubland and forest as ‘naturally vegetated land covers’. The term natural is used to differentiate these areas from highly modified agricultural fields, but this should not be interpreted as areas with lack of human influence. For example, grassland includes improved pastures as well as native grasslands, and in some cases shrublands result from anthropogenic changes in grasslands or forests.

During summer 2008, 452 training samples were collected in the field with information registered on vegetation structure and composition, among other variables. In addition, 234 complementary samples were extracted from high resolution imagery of the area (Google Earth). Sixty five percent of the training samples were used during the classification procedure and the remainder for accuracy assessment. Different band combinations were tried for the supervised classification procedure using the maximum likelihood classifier. The best combination in terms of separability among classes was: bands 2, 3, 4, 5, and 7 from Landsat images, together with a digital elevation model of the area (Farr, 2007), and two layers representing the X and Y coordinates. The digital elevation model and X–Y coordinates incorporated information in the model useful to differentiate land cover classes that were not originally separable based on spectral data alone.

The use of ancillary data has been proven to increase the accuracy of land cover classifications (Rozenstein & Karnieli, 2011). Therefore, a rule based approach, as described by Daniels (2006), was later incorporated to improve the final land cover map. Using data mining software (Compumine, 2009) rules were generated and applied in ERDAS Imagine Knowledge Engineer. Rules were based on Tasseled Cap transformation (Crist & Kauth, 1986), normalized difference vegetation index (NDVI), and soil data of the area (soil map modified from Cano et al., 1980). Final accuracy of the classification was 88.5% (overall kappa statistics = 0.85, Supplement Table 2) (Jensen, 2005). The same procedure was applied to 1987 and 1999 images. Lack of reference data for 1987 and 1999 prevented assessment of classification accuracies for those dates. However, given the consistency in the remote sensing data and of the classification procedure applied for 2008 and earlier years, we assumed classification accuracy did not differ significantly from that of 2008. Visual examination of the products (Supplement Figs. 2–4) supported this assumption.

Table 1
Description of land cover classes.

Land cover	Description
Agriculture	Areas in which the land cover resulted from direct human influence through agricultural practices. Three sub-classes composed the agriculture class: active fields, bare soil, and fallow.
Grassland	Areas covered by native grasslands and planted pastures, which could not be distinguished with imagery.
Shrubland	Areas covered with woody vegetation usually having multiple stems branching from or near the ground. This class includes two different units in terms of composition and ecological characteristics. In the southwest of the study area, shrublands are composed mainly of *Larrea* spp., indicating the transition between the Caldenal and the Monte ecoregions. In the rest of the study area, shrublands indicate sites in which caldén (the dominant tree species in the region) is present in early successional stages, both colonizing areas of grassland or re-growing after forest fires.
Forest	Areas covered by at least 10 percent cover of tree crowns (FAO, 2010, p. 378).
Burned:	Burned areas. This includes natural and human-induced fires.
Lagoon:	This class includes both permanent and temporary water bodies.

Landscape partitioning and analysis

For analysis of processes of land cover change and rates of conversion, five landholding types were identified: 1) Parque Luro (Park): the only protected area in Caldenal in La Pampa province (7607 ha, managed by the provincial government), 2) old game reserves: those in operation prior to March 1999; 3) new game reserves: those that have been in operation since March 1999, and 4) two size classes of ranches. Game reserves and the park were significantly larger than the mean size of cattle ranches (Supplement Table 3). Therefore, we split ranches into two size classes: 1) 4860 ha and larger, which is the mean size of game ranches minus one standard deviation and 2) smaller than 4860 ha. Then we took a random sample of ranches ($n = 40$) in each of these classes to represent big and small cattle ranches, respectively. Ranches registered as game reserves in the past but not currently in operation were removed from the analysis. The land cover change and a trajectory analysis were performed at the landholding level for each of the five landholding types: Park, old game reserves, new game reserves, big cattle ranches and small cattle ranches (Fig. 2). The results for the Park are presented and discussed, however they were not included in the statistical analysis given the presence of only one park in the study area.

A set of rates was calculated for analyzing the effect of different landholding types on land cover changes. Given the importance assigned by previous studies to the processes of conversion in the area (Hoekstra et al., 2005; Mendez, 2007b), the annual gross rate of conversion of these naturally vegetated land covers to agriculture was calculated (Dirzo & Garcia, 1992):

$$r_{\text{conversion}} = 1 - \left(1 - \frac{A_{\text{converted\_p}}}{A_{\text{natural\_t1}}}\right)^{1/t}$$

Where $A_{\text{converted\_p}}$ is the extent in hectares converted to agriculture during the period; $A_{\text{natural\_t1}}$ is the extent in hectares of the naturally vegetated land covers (grassland + shrubland + forest) at the beginning of the period; and t is the time in years. The annual rate

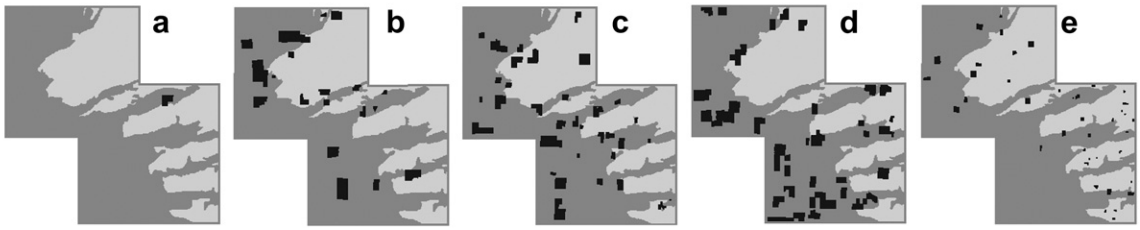

Fig. 2. Spatial distribution of landholdings analyzed in this study (black polygons). Background represents soil agricultural suitability (light gray = suitable, dark gray = unsuitable). (a) Park ($n = 1$), (b) old game reserves ($n = 21$), (c) new game reserves ($n = 38$), (d) big cattle ranches ($n = 40$), and (e) small cattle ranches ($n = 40$).

of change of each land cover was computed at the landholding level using a modified version of the formula above:

$$r_{\text{change}} = 1 - \left[\left(1 - \frac{A_{t1} - A_{t2}}{A_{t1}}\right)^{1/t}\right] * (-1)$$

Where A_{t1} is the extent in hectares of the land cover at the beginning of the period; A_{t2} is the extent of the land cover of interest at the end of the period; and t is the time in years for the period of analysis. This rate was computed on grassland, shrubland and forest covers for each of the landholdings analyzed.

Given that the variables of interest did not meet the assumptions of normality, non-parametric tests were used to analyze differences among landholding types and between periods (1987–1999, 1999–2008). In order to test for preferential establishment of game reserves in areas with higher cover of naturally vegetated land, Kruskal–Wallis tests (Kruskal & Wallis, 1952) were run to test for differences in land cover among landholding types in 1987 before the establishment of game reserves. Differences in rates of change of naturally vegetated land covers and in rates of conversion to agriculture were analyzed between periods for each landholding type with Wilcoxon rank sum tests. Also, within each period, we compared rates of change and conversion of land cover among landholding types with Kruskal–Wallis tests. In all cases, landholdings were used as replicates. When differences were significant, we performed pairwise comparisons between landholding types using Wilcoxon rank-sum tests, and Bonferroni corrections were applied to account for multiple comparisons (Shaffer, 1995). We also examined the relationship between rates of conversion to agriculture in each period and characteristics of the landholdings (size of individual properties, soil agricultural suitability, and average rainfall for the last 30 years) with Wilcoxon rank-sum test for categorical variables, and Spearman's rank correlation for continuous variables.

Trajectory analysis

The original interest was not only to quantify net changes among classes between the dates analyzed, but to understand the nature of the changes occurring in the area. Consequently change trajectories were examined for 1987–1999 and 1999–2008, and across the 21-year period using trajectory maps (Southworth et al., 2004). Trajectory maps are built by incorporating information for each date in the sequence into each pixel, so we get the trajectory of past land covers, or land cover change histories included within the pixel information. As such an area that has been forested on all dates, versus one that underwent transition across dates but is forest again by the final date – clearly are flagged with this difference in history. This allows interpreting the dynamics in land cover at the pixel level, and by aggregation at the regional level. The number of possible combinations of 6 land cover classes for the three dates (216 possible land cover trajectories) made interpretation of trajectories cumbersome, so a more useful and intuitive hierarchical four level trajectory analysis scheme was developed to summarize changes that occurred over the 21-year period (Supplement Table 4).

Results

Preferential location of game reserves

In 1987, before game reserves were formally established, the park exhibited the highest forest cover, followed by areas used for ranching that are now occupied by game reserves, and then by areas in our sample that are big cattle ranches and small cattle ranches (Fig. 3). This indicates that some of the ranches with the highest forest cover were converted to game reserves. The areas now occupied by game reserves and large cattle ranches also had

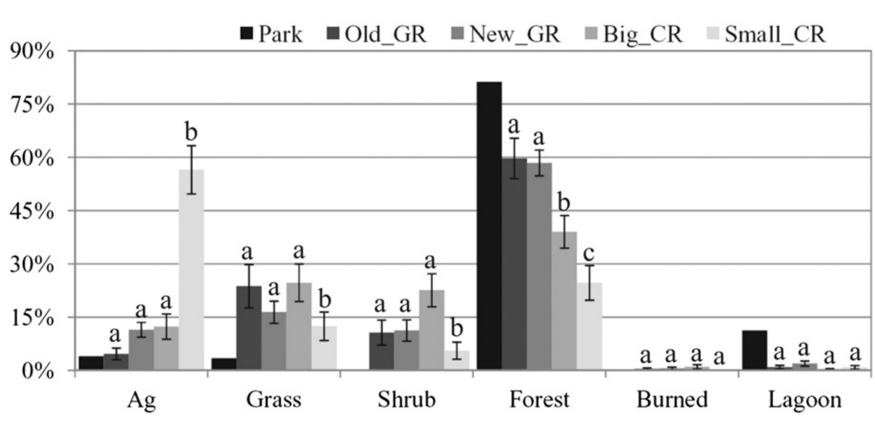

Fig. 3. Land cover composition ($\bar{x} \pm 1$ SE) in 1987 for different landholding types. All areas except the park were ranches in 1987. Areas labeled as game reserves currently are occupied by game reserves. Different letters within each land cover type indicate significantly different means among landholding types ($p < 0.05$).

more grassland and shrub cover than small ranches in 1987, small cattle ranches had a much greater proportion of area in agriculture.

Conversion to agriculture and rates of change

Annual rates of conversion of naturally vegetated land covers to agriculture were lower than 1% for both periods for all landholding types except for small cattle ranches (Table 2). Differences in conversion rates between land uses were not statistically significant for 1987–1999 or 1999–2008 (cattle ranches = 4.5%, game reserves = 0.8%, Wilcoxon rank sum, $W = 2321$, $p = 0.8697$; and cattle ranches = 6.1%, game reserves = 0.8%, $W = 2320$, $p = 0.87$ respectively). However, while rates of conversion were higher for cattle ranches in areas with suitable soils (cattle ranches = 0.8%, game reserves = 0.3%, Wilcoxon rank sum, $W = 284$, $p = 0.14$) in unsuitable areas game reserves presented higher rates of conversion to agriculture (cattle ranches = 0.1%, game reserves = 0.3%, Wilcoxon rank sum, $W = 844$, $p = 0.10$) although differences were not significant in any case. When individual landholdings were classified as suitable or unsuitable for agriculture, regardless of the landholding type, annual rates of conversion were higher in areas with soil suitable for agriculture than areas with unsuitable soil (mean 1987–1999, suitable 7.2%, unsuitable 0.8%, Wilcoxon rank sum, $W = 2646$, $p = 0.03$, Fig. 4a; mean 1999–2008, suitable 9.9%, unsuitable 0.8%, $W = 2803$, $p = 0.004$). For game reserves, no differences in rate of conversion to agriculture were found between those located in suitable and unsuitable soils for agriculture (suitable = 0.2%, unsuitable = 0.3%, Wilcoxon rank sum, $W = 269$, $p = 0.59$), but for cattle ranches rates of conversion to agriculture were significantly higher in the more productive soils (suitable = 0.8%, unsuitable = 0.1%, Wilcoxon rank sum, $W = 1027$, $p = 0.02$). Rates of conversion to agriculture and size of individual landholdings were negatively correlated (Spearman's rank correlation, rho = −0.18, $p = 0.04$, Fig. 4b). The opposite relationship was found between rate of conversion and mean annual rainfall (Spearman's rank correlation, rho = 0.43, $p < 0.001$, Fig. 4c).

Rates of change in land cover types classified as naturally vegetated reflected processes that both remove vegetation (e.g., agricultural conversion or fire) and increase vegetation (e.g., natural regeneration of shrubs and trees). The amount of forest declined in all landholding types, with the exception of the park, during 1987–1999 (Table 3). In contrast, forest cover increased during the second period on large cattle ranches and all game reserves, but declined at higher rates in small cattle ranches from 1999 to 2008 than from 1987 to 1999. Rates of change in forest differed among landholding types based on the Kruskal–Wallis test (1987–1999, $H = 9.6$, d.f. = 3, $p = 0.02$; 1999–2008, $H = 10.1$, d.f. = 3, $p = 0.02$), but the only significant pairwise difference was between big and small cattle ranches (Wilcoxon rank sum, 1987–1999, $W_{(nbig = nsmall = 40)} = 519$, $p = 0.007$; 1999–2008, $W_{(nbig = nsmall = 40)} = 1101$, $p = 0.004$). Grasslands and shrublands showed no significant differences in rates of change over time for any of the landholding types, and also these land cover types did not differ significantly among landholding types for either period of analysis (all $H \leq 4.07$, d.f. = 3, $p \geq 0.25$).

Table 2
Annual rate of conversion of naturally vegetated land covers to agriculture ($\bar{x} \pm 1$ SE).

Landholding type	1987–1999		1999–2008	
	Percentage	Hectares	Percentage	Hectares
Park	0.1% (−)	7.2 (−)	0.3% (−)	20.6 (−)
Old game reserves	0.6% (±0.2)	32.0 (±11.1)	0.5% (±0.2)	24.9 (±7.9)
New game reserves	0.9% (±0.2)	28.7 (±13.1)	0.9% (±0.1)	24.0 (±4.3)
Big cattle ranches	0.7% (±0.2)	27.6 (± 6.1)	0.7% (±0.2)	30.5 (±6.3)
Small cattle ranches	8.4% (±2.6)	3.7 (± 1.2)	11.6% (±3.6)	3.6 (±0.8)

Trajectory analysis

The trajectory analysis showed that the majority of land cover did not change across dates (Supplement Fig. 5a). For all the landholdings analyzed, most changes that occurred during the 21-year period were not agricultural conversion (on average 77% of the changed surface, Supplement Fig. 5c). In game reserves and cattle ranches, an increase in shrub cover was the main change observed (Supplement Fig. 5e). This occurred via two processes: 1) conversion of forest to shrublands through logging and burning, which was distributed throughout the study area with no clear spatial pattern; and 2) invasion of grasslands by shrubs, mainly in the western sector of the study area. Conversion to agriculture represented on average 15% of the changed area, and conversion of agricultural fields to other land covers only comprised 8% of the changed area (Supplement Fig. 5c). The trajectory analysis did not identify clear differences in the pattern of land cover change between game reserves and cattle ranches for the period analyzed.

Discussion

Processes of land cover change

In our study region, game reserves initially were established in areas where forest cover was more abundant, and for the last 21 years game reserves have continued to maintain forest cover. This preferential selection of forest for game reserves may be explained by the high abundance of non-native game species, especially red deer (*C. elaphus*) in forested habitats of La Pampa. This species was introduced from Europe into the Caldenal savanna forests of Argentina for hunting in the first decade of the 20th century and remains the primary game species in the region. The fact that game reserves have been established preferentially in areas where one of the habitats of greatest conservation interest is located is significant and should be considered in regional conservation planning.

Rates of conversion of naturally vegetated land covers to agriculture were expected to be lower in areas where game reserves were established as compared to cattle ranches. However, no such differences were found. Conversion rates for both large cattle ranches and game reserves were small (<1% annually). Thus, these results do not support the initial hypothesis of differential processes of land cover change occurring in the game reserves as compared to the large cattle ranches in the Caldenal region. In contrast, the average rate of conversion to agriculture of small cattle ranches was much higher for 1987–1999 and 1999–2009 (8–11%), and the average annual rate of decline in forest cover on these ranches was very high in the second period (−11.5%). However, in 1987 forest cover already was small in most of these properties (55% of the ranches had < 10% forest cover), and the loss of a small area of forest resulted in a large percent decline in forest cover. Our results show that deforestation on ranches in this region occurred mostly prior to 1987, and thus prior to the establishment of game reserves.

Factors other than land use type have influenced the processes of land cover change in this region. Rates of forest change and conversion to agriculture appear to be related to the size of the property. This relationship likely occurs because the size of individual properties is largely a consequence of environmental factors that determine the capability of the land to produce and generate income. Game reserves and big cattle ranches are located primarily (85%) in areas unsuitable for agriculture and the small cattle ranches are mainly located in the areas suitable for crop production

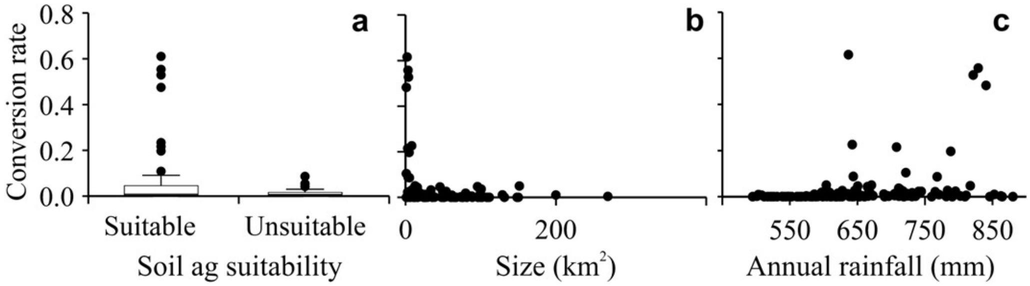

Fig. 4. Relationship between the rate of conversion of naturally vegetated land covers to agriculture from 1987 to 1999 and characteristics of the landholding ($n = 139$). The same patterns were observed from 1999 to 2008.

(73%). Forest loss in La Pampa is occurring most rapidly to the east, where environmental conditions are more appropriate for development of agriculture. These results reinforce the idea of the presence of an active agricultural frontier to the east of the Caldenal (Lell, 2004). Pressure from agriculture, and consequently conversion, is lower the farther west the property is located, regardless of establishment of game reserves. These results indicate that it is not the land use, but the presence of more appropriate environmental conditions, particularly good soils and water availability, that are influencing processes of land cover change in the area. Currently, game reserves and large cattle ranches are not suffering from forest loss. However, if new crops or more efficient farming techniques were developed for the drier environmental conditions of the west, the remainder of the Caldenal may follow the same conversion path as has occurred in the eastern agricultural region.

In the Caldenal region fires kill above ground parts of trees and shrubs, but when fuel loads are not high, below-ground parts of these woody plants often survive and vigorous resprouting occurs (Boo, Pelaez, Bunting, Mayor, & Elia, 1997). Studies in the region have shown that after cattle were introduced Caldén recruitment increased significantly (Dussart, Lerner, & Peinetti, 1998). Cattle also promotes persistence of woody species, through an increase in germination and establishment of Caldén seedlings and a reduction in woody plant mortality due to reduced fuel loads (Villalobos, Pelaez, & Elia, 2005). Introduction of cattle and altered fire regimes in the last century have generated an increase in the shrub cover to the detriment of natural grasslands and open forests (Busso, 1997).

Remote sensing applications offer a very cost effective and objective means to characterize land cover over large areas; however, these applications also have their limitations. In this study in particular, we were not able to differentiate between natural grasslands and improved pastures, a limitation that could have significant implications for the assessment of the final conservation value of the land use types analyzed. The improvement of pastures with an introduced grass, *Ergrostis curvula*, is a very common practice in the area, and even though in these pastures the abundance and diversity of native species increases over time (there are *E. corvula* pastures over 20 years old), they will remain dominated by this introduced species. Moreover, results of surveys developed to understand management differences between cattle ranches and game reserves showed that, although only 15% of the ranches planted pastures, over 80% of the game reserves did (Gonzalez Roglich, unpub.). A more detailed remote sensing analysis (e.g., including phenology differentiation) or intensive field campaigns would allow for a more complete understanding of the effect of game reserves of native grassland conservation.

Game reserves and conservation

Game reserves have been identified as land use types with high conservation value given their relatively low impact on local natural resources and their high profitability, particularly in Africa and North America (Barnes & deJager, 1996; Lindsey, Alexander, Frank, Mathieson, & Romanach, 2006; Tomlinson, Hearne, & Alexander, 2002). However in Argentina, fee hunting in game reserves is a relatively new activity, and the benefits of these activities for the conservation of native ecosystems have not been assessed previously. Our results show that game reserves in central Argentina have been established in the areas of highest forest cover, which potentially could be very beneficial for conservation in a landscape that has suffered high rates of deforestation. Because deforestation is occurring primarily to the east of the game reserves, these reserves are not currently playing a role in reducing deforestation. Nonetheless, because of the prime habitat on most of these reserves (i.e., forest and native grassland), with appropriate management, these reserves could greatly benefit native flora and fauna, and conserve key ecological processes in native ecosystems.

However, the focus on non-native species in hunting enterprises in Argentina likely reduces the conservation value of this land use. Introduced species are one of the greatest threats to conservation of biodiversity worldwide (Butchart et al., 2010). If game reserves in the Caldenal promote increases in populations of non-native species or continue to actively introduce new species for hunting, they could generate negative impacts precisely in areas where the habitat of greatest conservation interest is located. Extensive research is needed to determine how to manage these introduced species to reduce impacts on native ecosystems, and long term research and monitoring are required to determine if

Table 3
Annual rate of change of natural land cover classes ($\bar{x} \pm 1$ SE).

Landholding type	Grassland		Shrubland		Forest	
	1987–1999	1999–2008	1987–1999	1999–2008	1987–1999	1999–2008
Park	−3.9% (−)	1.8% (−)	0.0% (−)	0.0% (-)	0.3% (−)	−0.2% (−)
Old game reserves	−1.8% (±5.3)	−1.8% (±2.5)	6.3% (±9.7)	0.1% (±3.5)	−5.1% (±1.5)	0.7% (±4.5)
New game reserves	1.3% (±1.5)	2.7% (±2.4)	1.8% (±3.8)	5.9% (±3.1)	−3.6% (±1.2)	2.3% (±1.7)
Big cattle ranches	2.7% (±3.7)	1.9% (±1.7)	9.7% (±5.9)	6.9% (±3.0)	−4.7% (±1.5)	4.6% (±2.0)
Small cattle ranches	−6.7% (±4.7)	−11.8% (±6.1)	5.2% (±4.5)	2.7% (±3.2)	−2.1% (±4.0)	−11.5% (±4.4)

intensification in land cover conversion occurs in the area of game reserves and whether game reserves will play a role in preventing the conversion.

Conclusion

The net conservation value of private game reserves in central Argentina is not clear. Establishment of these reserves in the areas with the highest forest cover may make them valuable. However, given their location in the areas of current low pressure of conversion to agriculture, to date they have not affected the processes of land cover change. In addition, game reserves have introduced new threats to the system in the form of new and abundant non-native species for fee hunting. The real contribution of private game reserves to the conservation of the Caldenal region in central Argentina can only be assessed with further monitoring of land cover change and research on effects of introduced species on the native biodiversity and ecosystem functioning of the Caldenal.

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Appendix. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.apgeog.2011.12.002.

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