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Drivers of agricultural land-use change in the Argentine Pampas and Chaco regions

M. Piquer-Rodríguez[a,*], V. Butsic[b], P. Gärtner[c], L. Macchi[a], M. Baumann[a], G. Gavier Pizarro[d], J.N. Volante[e], I.N. Gasparri[f,g], T. Kuemmerle[a,g]

[a] Geography Department, Humboldt-University Berlin, Unter den Linden 6, 10099 Berlin, Germany
[b] Department of Environmental Science, Policy and Management University of California Berkeley, 130 Mulford Hall #3114, Berkeley, CA 94720, USA
[c] Leibniz Centre for Agricultural Landscape Research (ZALF) e.V., Eberswalder Str. 84, 15374 Müncheberg, Germany
[d] Instituto Nacional de Tecnología Agropecuaria (INTA), Instituto de Recursos Biológicos, Centro de Investigación en Recursos Naturales (CIRN-IRB), De los Reseros y Las Cabañas S/N HB1712WAA, Hurlingham, Buenos Aires, Argentina
[e] Instituto Nacional de Tecnología Agropecuaria (INTA) Estación Experimental Agropecuaria Salta. Ruta Nac. 68 km 172, Cerrillos, Salta, Argentina
[f] Instituto de Ecología Regional, Universidad Nacional de Tucumán and CONICET, CC 34, 4107 Yerba Buena, Argentina
[g] Integrative Research Institute for Transformations in Human Environment Systems (IRI THESys), Humboldt-University Berlin, Unter den Linden 6, 10099 Berlin, Germany

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ABSTRACT

Agricultural expansion and intensification in South America's dry forests and grasslands increase agricultural production, but also result in major environmental trade-offs. The Pampas and Chaco regions of Argentina have been global hotspots of agricultural land-use change since the 2000s, yet our understanding of what drives the spatial patterns of these land-use changes remains partial. We parameterized a net returns model of agricultural land-use change to estimate the probability of agricultural expansion (conversions of woodlands to either cropland or grazing land) and agricultural intensification (conversion of grazing land to cropland) at the 1-km scale for the years 2000 and 2010. Uniquely, our model allowed us to quantify the importance of underlying causes (i.e., changes in agricultural profit) and spatial determinants (i.e., soil fertility, distance to markets, etc.), for Argentina's prime agricultural regions as a whole. We found that cropland and grazing land expansion into woodlands was much less sensitive to changes in profit-related factors than agricultural intensification. Profit-related variables, were a particularly strong cause of intensification in the Pampas, where cropland profits rose by 29% (compared to 18% in the Chaco). This suggests that further conversions of grazing land to cropland in the Pampas and Chaco is likely as long as agricultural demand, and thus returns to agriculture, continue to be high. The moderate impact of profit-related factors on affecting woodland conversion rates also suggests a limited potential of economic policies that affect marginal profits (e.g., taxes or subsidies) to alter deforestation rates and patterns in major ways. Policies that target socio-economic variables not included in our profit-focused framework (e.g., capital availability), area-based interventions (e.g., land zoning), or less-profit oriented actors (e.g., via community-based management) might be more effective in addressing deforestation rates in the Chaco.

1. Introduction

Humans have transformed the Earth for millennia by converting natural areas to agriculture (Foley et al., 2011). These conversions have resulted in a substantial increase in food production, but have also led to major trade-offs in terms of biodiversity, carbon emissions and diminishing non-provisioning ecosystem services (Gibbs et al., 2010; Newbold et al., 2015; West et al., 2010). Today, these trade-offs are especially apparent in tropical dry forest and savannas, which harbor high biodiversity and carbon stocks, yet are under intense land-conversion pressure (Aide et al., 2013; Kuemmerle et al., 2017; Laurance, Sayer, & Cassman, 2014; Portillo-Quintero, Sanchez-Azofeifa, Calvo-Alvarado, Quesada, & Do Espirito Santo, 2015). Understanding the underlying causes of land-use change in these regions, as well as the factors determining the spatial patterns of these changes, is therefore important to foster policy options that balance biodiversity and

* Corresponding author.
E-mail addresses: maria.piquer-rodriguez@geo.hu-berlin.de (M. Piquer-Rodríguez), vanbutsic@berkeley.edu (V. Butsic), gaertner@zalf.de (P. Gärtner), leandro.macchi@geo.hu-berlin.de (L. Macchi), matthias.baumann@geo.hu-berlin.de (M. Baumann), gavierpizarro.g@inta.gob.ar (G. Gavier Pizarro), volante.jose@inta.gob.ar (J.N. Volante), ignacio.gasparri@gmail.com (I.N. Gasparri), tobias.kuemmerle@geo.hu-berlin.de (T. Kuemmerle).

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ecosystem services with agricultural production (Foley et al., 2005; Tilman, Balzer, Hill, & Befort, 2011).

The decisions of individual farmers, agricultural enterprises, and governments to expand or intensify agriculture are taken locally, but depend on a range of underlying causes operating across multiple scales (Angus, Burgess, Morris, & Lingard, 2009; Reidsma, Tekelenburg, Van Den Berg, & Alkemade, 2006). At the global scale, factors such as human population growth (Godfray, 2011; Hazell & Wood, 2008; Tscharntke et al., 2012), changing diets (Alexander et al., 2015; Bajzelj et al., 2014; Lambin & Meyfroidt, 2011; Tilman et al., 2011), as well as bioenergy production influence the demand for agricultural products and thus, international commodity prices (Geist et al., 2006; Lambin, Geist, & Lepers, 2003). At the regional scale, the level of technological development, political instability, or cultural ties to the land can play important roles in how land-use decisions are taken (Ceddia, Gunter, & Corriveau-Bourque, 2015; Gasparri & le Polain de Waroux, 2015; Thomas, Karl-Heinz, & Helmut, 2014). At even finer scales, a range of spatial determinants such as soil quality, climatic patterns, or socio-economic characteristics influence where agricultural expansion and intensification take place (Golub & Hertel, 2008; Lambin et al., 2013; Lubowski, Plantinga, & Stavins, 2008; Meyfroidt, 2015). To identify policies that can effectively influence land-use change toward desired outcomes, it is therefore essential to understand the relative importance of these factors and how they interact across scales to produce the spatial patterns of land-use change we observe (Levers et al., 2014; Meyfroidt, 2015).

Spatial Net Return Models (NRM) are powerful tools for that purpose as they are able to assess the combined impacts of underlying causes of land-use change, such as agricultural profitability, while controlling for spatial determinants influencing the configuration of land use (Bockstael, 1996; Butsic, Lewis, & Ludwig, 2011). The basic intuition of these models is that individual land owners maximize the utility from land use (Capozza & Helsley, 1989). In cases where land is used primarily as an input to production, which should be the case in regions where agricultural expansion takes place (Barbier, 2012; Le Polain De Waroux et al., 2018), utility can be proxied well by economic net returns (i.e., profit or loss). This theoretical concept can be translated to a statistical model via regression techniques (Wooldridge, 2011), allowing to model observed land-use change in terms of economic (including non-market) rents (Irwin & Bockstael, 2004; Lewis, Provencher, & Butsic, 2009). Yet, other factors than changes in marginal profit can have important influence on land-use decision-making. These include both economic factors, for example, land prices and speculation, capital availability, or macro-economic conditions, as well as non-economic factors, such as land tenure, cultural ties to the land by indigenous communities, traditional land uses, or corruption. (Arima, 2016; Ceddia et al., 2015; Dalla-Nora, De Aguiar, Lapola, & Woltjer, 2014; Dent, Edwards-Jones, & Mcgregor, 1995; Gasparri, Grau, & Gutiérrez Angonese, 2013; Henderson, Anand, & Bauch, 2013; Marinaro, Grau, Gasparri, Kuemmerle, & Baumann, 2017). The latter group of factors is particularly challenging to capture in an econometric modelling framework.

South America has recently been a global hotspot of agricultural expansion and intensification, triggering major losses in terms of biodiversity and no-provisioning ecosystem services (Aide et al., 2013; Laurance et al., 2014; Ramankutty, Foley, Norman, & Mcsweeney, 2002). Within South America, dry forests and grasslands are particularly prone to land-use change, especially in Brazil, Paraguay, Bolivia, Uruguay and Argentina (Graesser, Aide, Grau, & Ramankutty, 2015). The Pampas grasslands and Chaco dry forests of Argentina have experienced an especially high increase in agricultural production (Baldi & Paruelo, 2008; Gasparri & Grau, 2009; Grau, Gasparri, & Aide, 2005; Volante, Mosciaro, Gavier-Pizarro, & Paruelo, 2016), particularly of soybean since the 1990s, bolstering Argentina's role as a world-leading agricultural producer (Baumann et al., 2016a; Pengue, 2014). At the same time, this has triggered widespread loss and fragmentation of natural vegetation (Adamoli, Ginzburg, & Torrella, 2011; Aide et al., 2013; Piquer-Rodríguez et al., 2015; Torrella, Ginzburg, & Galetto, 2015; Viglizzo et al., 2010), and cropland is increasingly replacing grazing land in both, the Argentine Pampas and Chaco regions (Gavier-Pizarro et al., 2012; Lende, 2015).

Despite these rapid land-use changes, few studies have assessed the causes of agricultural expansion and intensification in Argentina, and these suffer from one or more of the following shortcomings. First, existing studies have focused only on spatial determinants such as soil quality or climate (Gasparri, Grau, & Sacchi, 2015; Volante et al., 2016), thereby neglecting the underlying causes of land-use change. These models can therefore not or only very indirectly assess the importance of profit-related factors on land-use change. Second, existing work has typically focused on small regions, typically inside a single ecoregion (Bert et al., 2011; Choumert & Phélinas, 2015; Zak, Cabido, Cáceres, & Díaz, 2008), thereby neglecting broad-scale patterns and potential connections between ecoregions and agricultural systems. Third, those studies that have assessed underlying causes have neglected the location factors determining land-use/cover change patterns (Bert et al., 2011), thereby disregarding the substantial spatial heterogeneity that exists inside these ecoregions. Finally, existing work has typically only studied forest loss (Gasparri et al., 2015; Volante et al., 2016), thereby potentially missing the interactions between agricultural expansion and intensification.

This translates into a substantial knowledge gap in our understanding of what causes land-system dynamics in one of the world's prime agricultural regions. To address this research gap, we developed a spatial net returns model of land-use/cover change for the years 2000 and 2010 in order to understand land-use dynamics in the Argentine Pampas and Chaco ecoregions. This decade had the highest agricultural expansion rates and agricultural production since the 1940s (Cáceres, 2015; Pengue, 2014). Our approach is, to the best of our knowledge, novel in that we (a) jointly model agricultural expansion and intensification, (b) assess both underlying causes (using cost and revenue data) and spatial determinants of land-use/cover dynamics, and (c) assess land-use/cover changes at fine resolutions (1 km^2), yet for multiple ecoregions simultaneously (in total 1,300,000 km^2). Specifically, we addressed two main research questions:

1. How did the underlying causes related to agricultural profitability affect land-use/cover change in the Pampas and Chaco regions between 2000 and 2010?
2. Which spatial determinants influenced agricultural land-use/cover change patterns in the Pampas and Chaco regions in that period?

2. Methods

2.1. Study area

Our study area encompassed the main agricultural regions of Argentina: the Pampas, the Espinal and the dry and humid Chaco ecoregions (~1.3 million km^2, Fig. 1). We included all districts that completely fell inside the ecoregions of the Pampas, Espinal, Dry Chaco and Humid Chaco (Olson et al., 2001) but excluded the very dry or mountainous districts in the southwest of the region (Conti et al., 2014). In Argentina, soy accounts for half the grain production and more than half of the cropped area in the country (Lende, 2015). Cattle ranching is also widespread with approximated 3 million tons of meat produced per year (www.minagri.gob.ar/ganaderia), of which 10% is exported (Santarcángelo & Fal, 2009). The Pampas has a longer land-use history than the Chaco, where cattle ranching started in the 16th century with the arrival of the first European settlers. Ranching shifted to cropping in many areas with the introduction and expansion of wheat, corn and sunflower in the 20th century, and the dramatic expansion of soybeans in 1990s displaced most ranching into the Espinal and the more isolated areas of the Chaco (González-Roglich, Swenson, Villarreal, Jobbágy, &

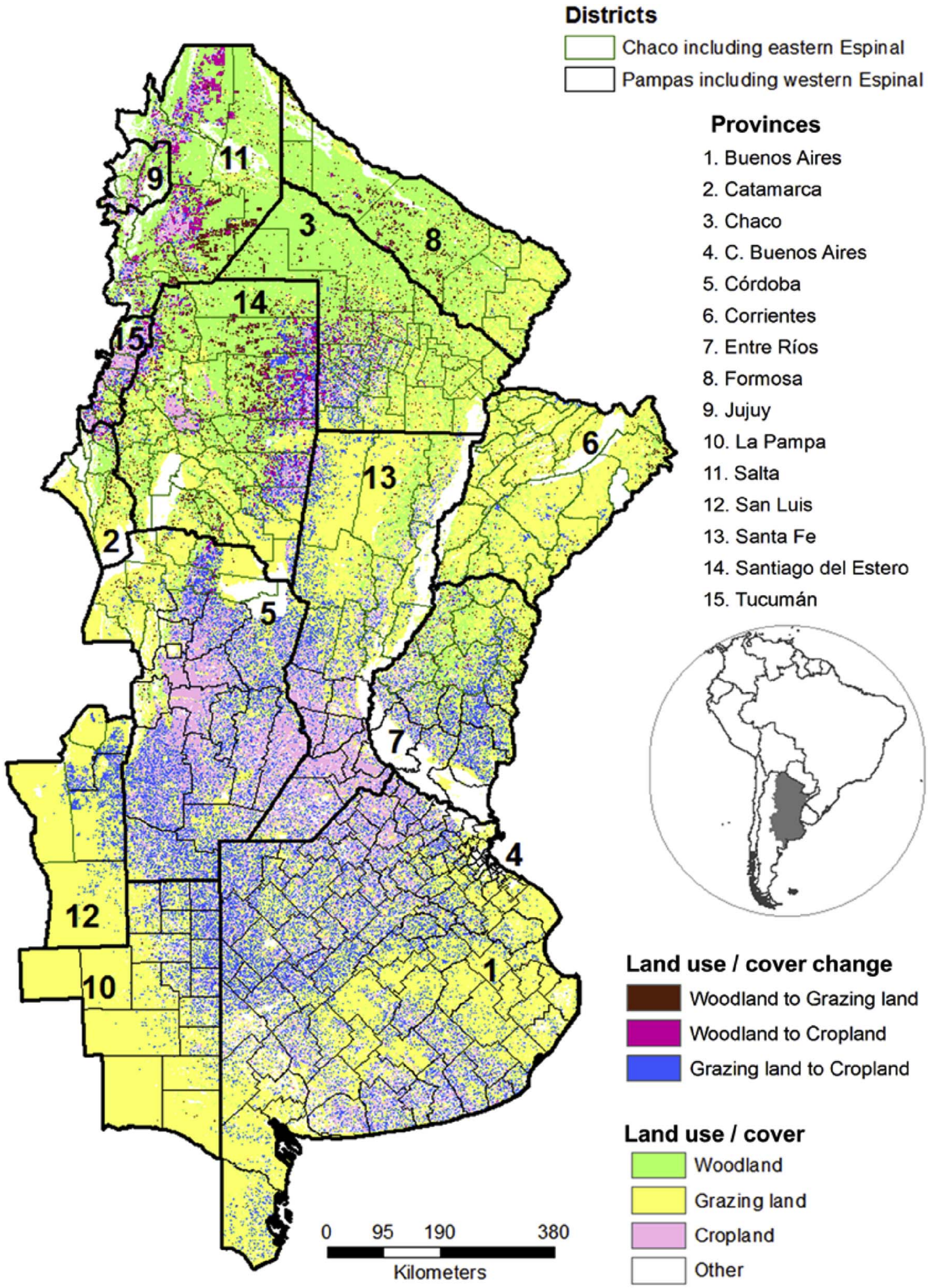

Fig. 1. Agricultural land-use/cover changes between 2000 and 2010 in the study region (i.e., Chaco and Pampas region, including the transitional Espinal region). (For a full color version of this figure see article online.)

Jackson, 2015; Magrin, Travasso, & Rodríguez, 2005; Modernel et al., 2016; Pengue, 2014; Viglizzo et al., 2011). The Chaco ecoregion had a low density of inhabitants until the 1880s, when accessibility improved with railroad expansion, and extensive cattle ranching and wood extraction were made possible (Bucher & Huszar, 1999). Political and economic reforms in Argentina in the early 1990s facilitated the increased production of crops (Pengue, 2005) and by the end of the 1990s, soybean expanded into the Chaco and Espinal regions, as well as in the Pampas, due to increasing soybean prices and new, genetically modified soybean varieties (Cáceres, 2015; Leguizamón, 2016). This led to substantial losses in natural vegetation with about 14% of the Argentine Chaco converted to agriculture during the 2000s (Baumann et al., 2016a). Land-use actors in Argentina range from large-scale producers, who mostly rent the land and use intensified agricultural practices (Baumann, Piquer-Rodríguez, Fehlenberg, Gavier Pizarro, & Kuemmerle, 2016b; Pengue, 2014) to small-scale producers with strong cultural links to the land they own and agricultural practices they conduct (Marinaro et al., 2017). The strong presence of indigenous communities in northern sectors of Argentina, that use the land mainly in traditional ways, enrich the mixture of land-use actors (Le Polain De Waroux et al., 2018).

The Pampas region has flat terrain, subtropical to temperate climate (mean annual temperature is ~15 °C, with an average monthly maximum of 22 °C and annual precipitation ranges 1100 mm in the east to 800 mm in the west) (Bianchi & Cravero, 2010) and soils very rich in organic matter (Herrera, Nabinger, Weyland, & Parera, 2014). Its natural vegetation are grasslands, mainly composed of *Stipa* sp., *Briza* sp., *Bromus* sp., and *Poa* sp. (Cabrera, 1971, p. 43). The Chaco region is generally characterized by flat terrain, except for the west and southwest were terrain is rougher, and a semi-arid and highly seasonal climate (mean annual temperature is ~22 °C, with an average monthly maximum of 28 °C and annual precipitation ranges 1200 mm in the east to 450 mm in the west) (Minetti, 1999). The Chaco is characterized by tree species of the genera Schinopsis and Aspidosperma ("quebrachos") (Prado, 1993). The Espinal constitutes a transition zone between the Pampas and the Chaco and is characterized by tree species such as *Prosopis* sp., *Acacia* sp., and *Aspidosperma* sp., shrubs and grasses (Burkart, Bárbaro, Sánchez, & Gómez, 1999; Guida Johnson & Zuleta, 2013). For our study, we distributed the Espinal among the two other ecoregions based on proximity and ecological characteristics of districts (Fig. 1).

2.2. Data used to characterize underlying causes and spatial determinants of land-use change

We generated land-use/cover maps for the years 2000 and 2010 based on existing maps for cropland at 250 m resolution (Volante et al., 2015) and woodlands, defined as having more than 25% tree cover at 30 m resolution (Hansen et al., 2013). Many woodlands are used by subsistence smallholders for forest grazing (i.e., in so-called *puesto* systems) but these actors are not included in our model. We considered all other lands that were neither water, urban areas, nor had slopes > 5° as suitable for grazing (Volante et al., 2015). Thus, grazing lands potentially contained woody vegetation with less than 25% tree cover, natural grasslands, savannas, and pastures, both with native herbaceous vegetation as well as implanted grasses. Since we homogenized these datasets at a resolution of 1 km^2, all features substantially smaller than that grain were below our minimum mapping unit. Our procedure also lead to some sparse dry forest in the western Espinal being classified as grazing land, not woodland, in line with our definition. Based on overlaying these maps, we mapped conversions of (1) grazing land to cropland, (2) woodland to cropland, and (3) woodland to grazing land between the years 2000 and 2010. These land-use/cover changes formed the dependent variables in our model. The accuracy of our land-use/cover change maps, evaluated using independent data, was > 90% (see Appendix A for details on the land-use/cover change map).

Our independent variables comprised economic factors, mainly variables related to cropland and grazing land profits at the district level (i.e*., departamentos*) in 2010, as well as spatial determinants of agricultural land-use change at the 1-km gridcell level, mainly climatic (i.e., aridity), accessibility (i.e., travel distance to provincial capitals), topographic (i.e., slope), edaphic (i.e., soil productivity) and cropland neighborhood variables (i.e., neighbors and area share in 2000) (Table 1, see Appendix A for details).

2.3. Modelling approach

We modeled three agricultural land-use/cover changes for our study region using two NRM. First, we used a logit model to assess the conversion of grazing land to cropland in the Chaco and Pampas, and second we used a multinomial logit model to assess the conversion of woodland to cropland and woodland to grazing land in the Chaco. We parameterized the multinomial model for the Chaco region only, because the few forests in the Pampas mainly represent commercial forest plantations, and forest loss there thus represents forest management, not land-use change. A few forests in the western Espinal ecoregion that was merged into the Pampas region in our analysis occur, but these are generally small, very sparse, often much smaller than our gridcell size, and in total covered only quite a small area (0,38% of the Pampas region in this study in 2000 and 0.33% in 2010). Using this probabilistic framework, we estimated the likelihood of a parcel of land (represented by a gridcell of 1 × 1 km^2) converting from woodland to either grazing land or cropland, or converting from grazing land to cropland.

Key for estimating a NRM is to calculate the returns of the current land-use alternative to all possible land uses. Because our maps did not distinguish between different crop types, we calculated average district-level net returns for crops in 2010 (Lubowski et al., 2008). Such an approach disregards the spatial distribution of crop types and spatial variability in yields within districts, and thus may miss some fine-scale spatial heterogeneity in profits. We used data on the national average price for main crop types (i.e., "*pricecrop*" in USD/ton (t)*)*, district data on crop yields (i.e., *"yieldcrop"*, t/ha), the percentage of agricultural land in a district in a given crop *(i*.e., *"%crop"),* and the direct cost to produce each crop summarized at the ecoregion level *(*i.e., *"costcrop",* USD/ha). We then calculated the average profit to cropland per ha, per district for all crops under study (i) (i.e., soy, sorghum, sunflower, corn, wheat and cotton) as in Eq. (1):

$$\text{Average profit to cropland} = \sum_{\text{all crops}} (\%crop_i)*((yieldcrop_i*pricecrop_i) - costcrop_i) \quad (1)$$

To calculate average profits to grazing land per district in 2010, we used district-level live meat yield data (*yieldmeat,* t/ha) and multiplied this by the national internal producer price (*pricemeat*, in USD/t) and subtracted the direct costs of production at the ecoregion scale (USD/ha, see Appendix A for detail). We adjusted the profits to account for one of the permanent investment of production, which is the clearance of woody vegetation, or deforestation, prior to production. We did this by dividing the sum of 200USD/ha by 15 (years) and then subtracting this annual investment from the annual profits (Delvalle, Gandara, D'agostini, Balbuena, & Monicault, 2012; INTA, 2013). Our cost data thus represents a simplified cost structure, as we cannot account for spatial variation in some costs that might exist across the region or over time (e.g., for land clearing). It is important to point out that the one-time cost of land clearing is relatively modest compared to the yearly returns from agriculture. It is therefore unlikely that incorporating variation in this cost would change our results qualitatively.

Using these profit calculations, along with our independent variables, we estimated the probability that a parcel of land would convert from grazing to crop as in Eq. (2):

$$\begin{aligned} Y^* = {} & B_0 + B_1 province + B_2*1cropneighbor2000 + B_3*cropneighbor2000 \\ & + B_4*cropprofit + B_{5-8}*soil + B_9*chaco + B_{10}*grazeprofit \\ & + B_{11}*slope + B_{12}*distcapital + B_{13}*\%crop2000 + B_{14}*aridity \\ & + B_{15-22}*Interactions + e_i \end{aligned} \quad (2)$$

where Y^* is the latent variable and the error term is distributed with a standard logistic distribution, e ~ Logistic (0,1). *Cropprofit* and *grazeprofit* represented profits for each use at the district level (as in equation (1)).

Table 1
Description of variables used to parameterize the net returns model.

Variables	Description	Units	Spatial Resolution	Sources
Land-use/cover Conversions				
Grazing land to Cropland	Conversions from grazing land to cropland	0–1	1 km^2	Volante et al., 2015, own data
Woodland to Grazing land	Conversions from woodland to grazing land	0–1	1 km^2	Hansen et al., 2013, own data
Woodland to Cropland	Conversions from woodland to cropland	0–1	1 km^2	Hansen et al., 2013, Volante et al., 2015
Environmental				
Aridity	PP/PEVT in 2010	–	1 km^2	INTA weather stations
Soil	FAO's index of soil agricultural productivity	0–4	1 km^2	Atlas de suelos, INTA
Slope	Degrees of slope	degree	1 km^2	www.landcover.org (SRTM)
Economic				
pricecrop	Producer prices at the first point of sale	USD/t (current $)	Country	FAO stats
pricemeat	Live meat price	USD/t (current $)	Country	FAO stats
Yieldcrop, yield meat	Crop yields meat produced	t/ha	Department	Databases Integrated System of Agricultural Information (SIIA in Spanish) and Stock cattle INTA2010
Costcrop, costmeat	Direct costs for crop and meat production	USD/ha (current $)	Ecoregion	INTA, Margenes Agropecuarios, MAyG
Distcapitals	Cost distance to provincial capitals using roads in 2010	USD (current $)	1 km^2	IGN-SIG250
Structural				
Protected Areas	Network of Protected Areas	0–1	Country	World Database on Protected Areas, www.wdpa.org
Provinces	Control variable (dummy)	character	Province	Database of Global Administrative Areas
Ecoregion	Control variable (dummy)	1,2	Ecoregion	WWF
%cropland2000	Crop area per department in 2000	ha	Department	self-generated
1cropneighbor2000	None or ≥ 1 crop neighbors in 2000	0,1	1 km^2	self-generated
cropneighbor2000	Number of cropland neighbors in 2000	0–8	1 km^2	self-generated

Expansion next to existing cropland was characterized by neighborhood variables (i.e., *1cropneighbor2000, cropneighbor2000)* to account for the fact that expansion near existing cropland may lead to economies of scale (Garrett, Lambin, & Naylor, 2013). Given our sampling strategy, we were able to integrate these variables directly, rather than through the inclusion of a spatial weights matrix, which creates inference problems. To account for district-level unobservable processes that may impact crop expansion (such as unobserved existing infrastructure or knowhow), we included a variable on the amount of cropland in 2000 (*%crop2000)* (see Appendix A for more details). We further interacted *cropneighbor2000* with *crop/grazeprofit* and *soil* for both cropland and grazing land to account for variation in the impact of net returns given the factors that influence yield, soil, the ecoregions and the number of neighbors. In case of the logit model, we additionally interacted with the *chaco* dummy variable. An identical set of covariates was used to estimate a multinomial logit model.

The resulting regressions contained more than 30 independent variables, including the interactions, making the interpretation of model coefficients complex. To facilitate the interpretation of our modelling results, we calculated marginal effects of each variable on the predicted probabilities of conversion (*margins* in Stata), and plotted predicted margins (i.e., probabilities of conversions) across the distributions of our suite of variables, holding all other variables at their mean (*marginsplot* in Stata). The interactions between variables were fully accounted for in these simulations and standard errors were estimated using the delta method (Oehlert, 1992; Williams, 2012).

To minimize the influence of outliers in our model (average net returns to cropland in 2010 were around USD 400 but some districts had maximum of USD 3000), we included only grid-cells where agricultural returns were less than USD 1000 per hectare, though this only reduced the dataset by less than 5%. We parametrized the models avoiding collinear predictors. Specifically, for each of the thematic groups of spatial determinants containing several candidate variables, we ran alternative NRM using only one variable from each group (Table 1), and selected the variable that increased model performance (AIC and pseudo R^2) the most. We excluded protected areas from our analysis because we do not expect land-use changes there to be major neither to be driven by rent theory.

To account for potential spatially correlated error terms, which can bias coefficients in logit and multinomial logit models, we sampled only every second grid-cell for model parameterization. In addition, we clustered standard errors at the district level, in order to allow for inter-district correlation between error terms. Running the model with even greater sampling distance (up to 8-km between gridcells) did not change the results.

For the logit and multinomial logit models, the marginal effects for continuous variables can be interpreted as the change in probability of a gridcell converting from one land use to another, given a one unit change of the independent variable. For example, a marginal effect of cropland net returns of 0.0001 means that for a USD 100 increase in cropland profits, conversions to cropland would be 1 percentage point more likely than to another land use. The marginal effect of factor variables (i.e., categorical and dummy variables) should be interpreted as the change in conversion probability when a gridcell changes from the base level to the level of interest. *Cropneighbor*2000 and *soil* classes are factor variables and the base levels account for no crop neighbor gridcells or no soil productivity, respectively.

2.4. Comparison of actual and predicted land-use change

One major assumption of our models was that land users will maximize the economic profitability of land. However, in reality a number of factors may prevent this from happening such as traditional uses, cultural ties to the land, or existing land-use zonation that prohibit certain conversions (i.e., Argentina's Forest Law, which designates some woodlands where certain land conversions are not allowed). To explore how far actual agricultural land-use changes deviated from those predicted by our NRM, we summarized the actual land-use/cover change data from our maps at the district level, and compared it to the average predicted conversion probabilities at the district level in 2010.

3. Results

In 2000, about 22% of our study area was woodland (of which 96%

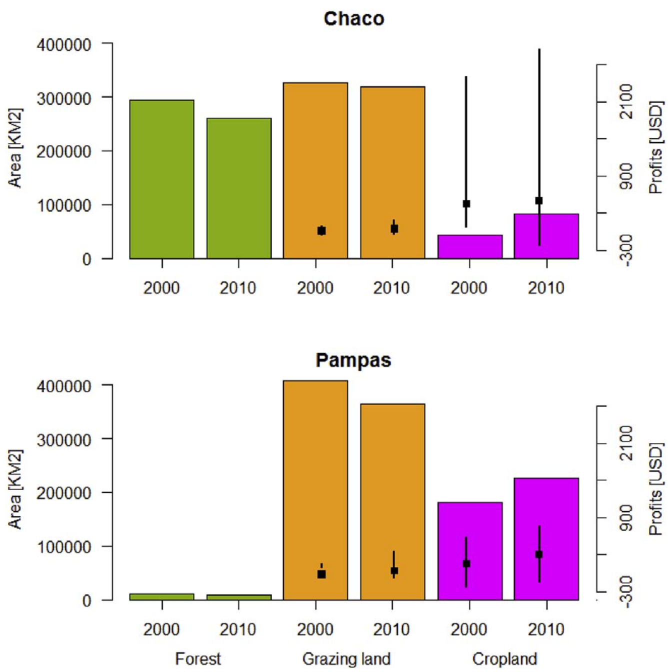

Fig. 2. Extent (km^2) of land-use/cover classes for the studied period (bar plot) and agricultural profits (USD) represented by the mean (dot) and min/max (length of solid vertical lines). (For a full color version of this figure see article online.)

was located in the Chaco), 54% was grazing land and 16% was cropland (Fig. 2). In the Chaco, woodlands decreased from 40% of the region in 2000 to 36% in 2010. This is equivalent to an annual forest loss of 0.4 percentage points or ~3400 km^2, a number almost three times higher than the global annual deforestation rate between 2005 and 2010 (0.14 percentage points) (FAO, 2012). About 40% of all woodland loss was due to conversion from woodland to cropland, while 60% of the change was due to woodland conversion to grazing land. About 19% of the grazing land loss was due to conversion to cropland that happened at some point between 2000 and 2010. Overall, cropland expanded to 23% of the landscape in 2010, almost a doubling compared to 2000 (Fig. 2).

Among the variables that characterized land-use/cover change, crop neighbor effects and environmental characteristics were the strongest determinants of land-use/cover change in both ecoregions (Table 2). *Aridity* positively influenced the likelihood for woodland to grazing land conversions (for a one-unit increase in aridity, the probability that land-use/cover converts increased by 0.12, i.e., 12 percentage points), but had a negative bearing on the conversions to cropland (decreasing 16 and 7 percentage points when converting from woodland and grazing lands respectively; Table 2). Conversions to croplands were more likely in areas of higher soil productivity (increasing 14 percentage points when converting grazing land and 5 percentage points when converting woodland), whereas conversions to grazing land was more likely on soils with lower productivity. Likewise, increasing soil productivity increased the likelihood of conversion from grazing land to cropland over ten percentage points for class two (medium) and four (very high) compared to class zero (no productivity), while increasing from class zero to class three increased the likelihood of conversion by more than 14 percentage points (Table 2). Increasing *slope* or *distcapitals* decreased the likelihood of conversions to both cropland and grazing land (e.g., decreasing between 2 and 5 percentage points when increasing slope by 1°). However, *distcapitals* was not statistically significant for conversions from woodland to cropland (Table 2). Although increasing *crop/graze profits* increased the likelihood of woodland-to-cropland and woodland-to-grazing conversions, this was not a significant cause of land-cover change in the Chaco (Table 2). In the Pampas, *cropprofit* was significant and positively influenced conversions to cropland.

The coefficients of *cropneighbor2000* were significant and positive for conversions to cropland. For example, increasing from zero to eight cropland neighbors in 2000 (*cropneighbor2000*) increased the likelihood of conversion from grazing land to cropland by 30 percentage points (calculated as coeff *1cropeighbor2000* +8*coeff *cropneighbor2000;* Table 2). *1cropneighbor2000* is a factor variable and thus the increase in the likelihood of conversion from grazing land to cropland when changing from zero to 1 or more neighbors in cropland in 2000, was 8.6 percentage points (Table 2). Grid-cells, either in woodland or grazing land, which had at least one neighboring grid-cell in cropland (*1cropeighbor2000)* were 8 percentage points more likely to convert to croplands than those which had no cropland neighbors, when holding all variables at their means (Table 2). The proportion of cropland in 2000 at the department level (*%crop2000*) increased the likelihood of conversions in general and was always positively and significantly correlated with the likelihood of conversion to cropland. Grid-cells that were located in districts with higher cropland proportion in 2000 (*%crop2000*) were 14 and 25 percentage points more likely to convert from grazing land and woodland to cropland respectively, when

Table 2
Logit and multinomial logit regression model estimates with marginal effects (coeff). Statistical significance of $p < .05$ *, $p < .01$ **, $p < .001$***. Profits coefficients are related to the end state of the land-use conversion (e.g., for conversions to cropland, profits coefficients are based on cropland profit).

Variables	Grazing land to Cropland		Woodland to Cropland		Woodland to Grazing land	
Environmental	*coeff*	*p-value*	*coeff*	*p-value*	*coeff*	*p-value*
Aridity	−0,0701	0.008**	−0,1665	0,212	0,12257	0.008**
Slope	−0,0470	0***	−0,0222	0.008**	−0,03079	0.003**
Soil low	0,0455	0***	0,0246	0.001**	0,00965	0,19
Soil medium	0,1000	0***	0,0414	0***	0,00552	0,64
Soil high	0,1438	0***	0,0384	0.007**	0,02967	0.014*
Soil very high	0,1013	0***	0,0502	0***	0,01952	0.02*
Economic						
Distcapitals	−0,0007	0,109	−0,0003	0,698	−0,00085	0.019*
Cropland/Grazing profits	0,0001	0.002*	0,0000	0,499	0,00035	0,15
Spatial						
cropneighbor2000	0,0273	0***	0,0062	0.001**	−0,00673	0,136
%crop2000	0,1425	0***	0,2476	0***	0,06052	0,186
1cropneighbor2000	0,0863	0***	0,0742	0***	0,02603	0.002**
Chaco	−0,0934	0.001**	–	–	–	–
Regression models	**Logistic**		**Multinomial logistic**			
	Numb. Obs.	334597	Numb. Obs.	133400		
	Pseudo R^2	0,25	Pseudo R^2	0,11		

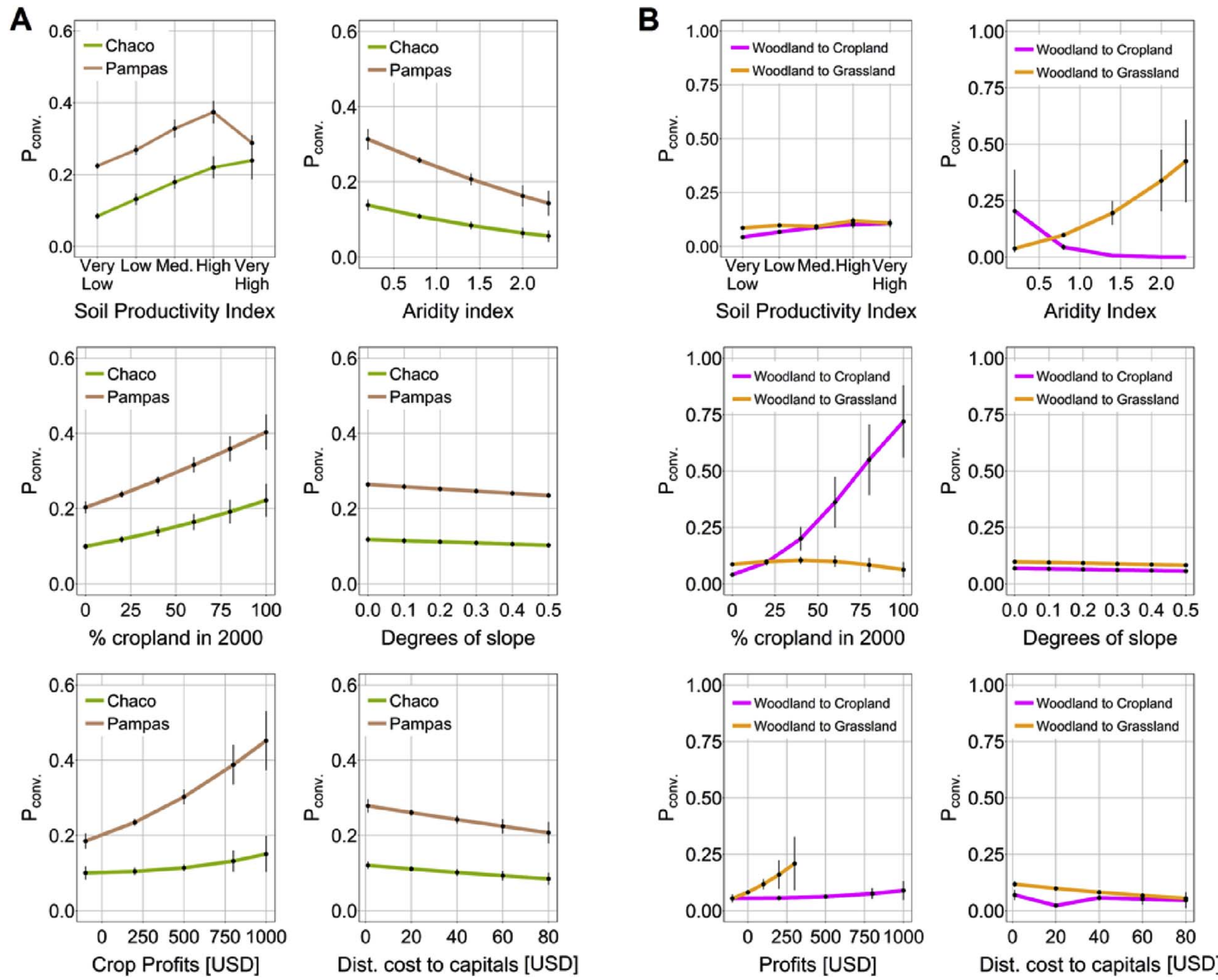

Fig. 3. Predicted probabilities (P_{conv}) of land-use/cover conversions for selected variables across their distribution, while holding all other variables at their means. (A) Conversions from grazing lands to croplands in the Chaco and the Pampas. (B) Conversions from woodland to cropland and grazing lands in the Chaco. Profits shown are those after conversions took place (i.e., for conversions from woodland to cropland, the probabilities are plotted based on changes in cropland profit). (For a full color version of this figure see article online.)

holding all variables at their means (Table 2). Pseudo R-squared measures are not directly comparable to r-squared values from OLS models, and values between 0.2 and 0.4 are generally considered good fits for logit and multinomial logit models (Louviere, Hensher, & Swait, 2000). Our model fit assessment thus showed a very good model fit for the logit model and good model fit for the multinomial logit.

The predicted margins also showed that increasing profits increased the probabilities of conversions from grazing land to cropland in both regions. However, land in the Pampas was both more likely to transition to cropland, and more sensitive to increases in cropland profits than land in the Chaco (Fig. 3A). *Slope*, *soil* productivity, *%crop2000* and *crop/graze profits* had similar impacts in each model (Fig. 3). However, *aridity* had different impacts for conversions to grazing land or cropland, where conversions to grazing land were more likely under more arid conditions than conversions to cropland (Fig. 3). *Distcapitals* showed also an interesting break in the decreasing probabilities of conversions from woodland to cropland at intermediate costs (Fig. 3B).

Comparing the actual and simulated land-use/cover change at the district level for 2000–2010 showed generally high concordance, especially for the Pampas region (Fig. 4A). However, conversions from grazing land to cropland in the Chaco (Fig. 4A), showed high probabilities of land-use conversions for some districts that actually had fairly low conversion rates in 2000–2010, especially in the provinces of Salta and Santiago del Estero. The opposite was the case for conversions from woodland in some districts in the provinces of Chaco, Salta and Cordoba, where actual conversions rates were high but we predicted comparatively low probabilities of conversions (Fig. 4B and C). The logit model had an average predictive accuracy of 83.4% while the multinomial model had a predictive accuracy of 85.3% (estimated using the R package *caret*).

4. Discussion

Many subtropical and tropical dry forests and grasslands are currently undergoing widespread agricultural expansion and intensification, especially in South America. While these land-use changes lead to an increased provisioning of agricultural commodities, they come at the cost of substantial losses in native vegetation, non-provisioning services and biodiversity. Understanding what causes spatial patterns of agricultural land-use change in these systems is therefore important in order to identify policies that allow mitigating these tradeoffs and steering land-use change dynamics towards desired outcomes (Aide et al., 2013; Hill & Southworth, 2016; Kuemmerle et al., 2016). Here, we addressed this research gap by assessing the profit-related and environmental drivers of agricultural expansion and intensification patterns for all of northern Argentina, a prime agricultural region of the world and a global deforestation hotspot. We focused on the 2000s, when agricultural expansion and production was highest since World War II (Cáceres, 2015; Pengue, 2014).

We analyzed agricultural land-use change drivers by parameterizing

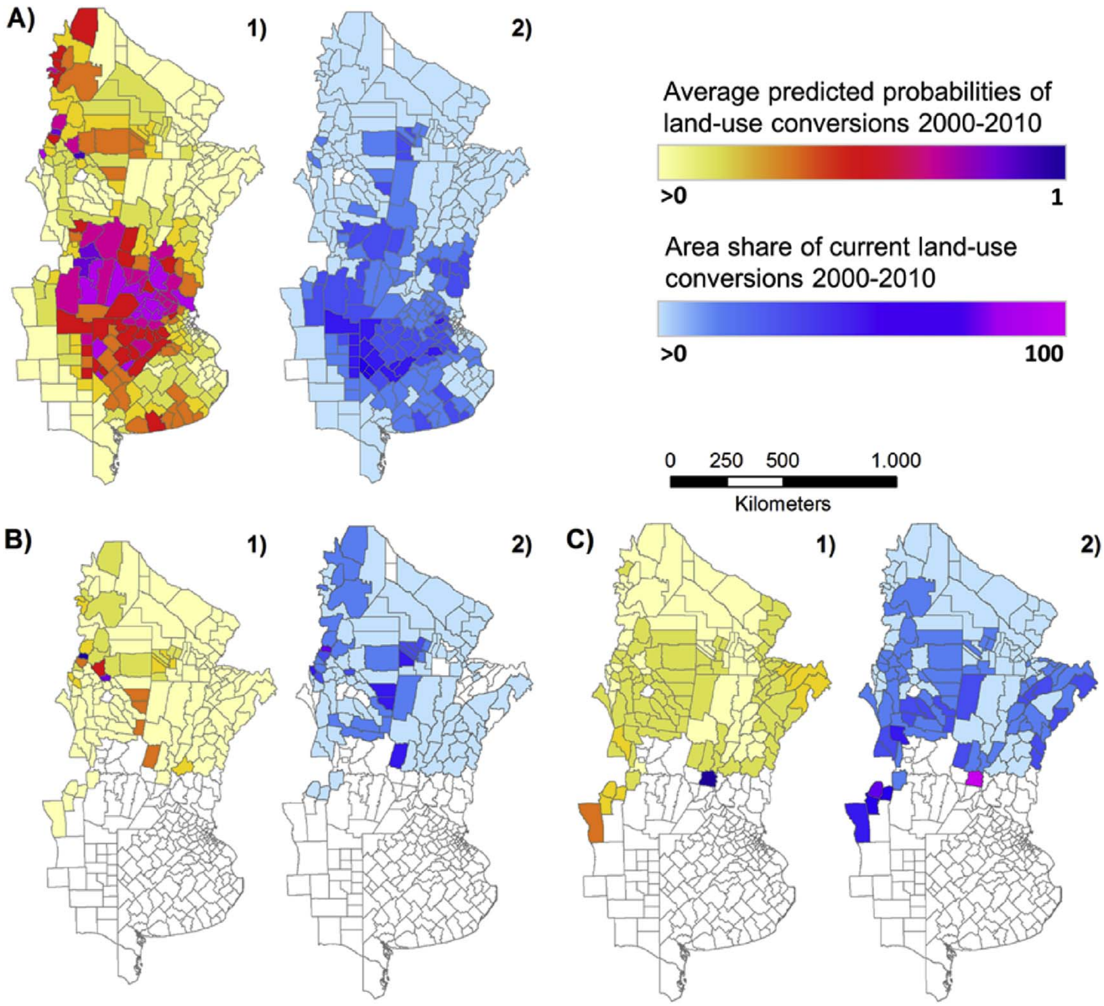

Fig. 4. Predicted (1) and actual (2) land-use/cover conversions at the district level for (A) grazing land to cropland, (B) woodland to cropland and (C) woodland to grazing land. The concordance of these two maps (i.e., districts with high average predicted probability of conversion and a high share of actual land conversions in 2000–2010) point at districts where land-use decisions are captured well by the factors entailed in our net returns model. Where these two maps disagree, factors other than those entailed in our model may have affected land conversion rates. (For a full color version of this figure see article online.)

a spatial net returns model, to our knowledge the first application of this approach for a deforestation frontier in South America. Our model is furthermore novel by explicitly considering district-level profit data. Our analyses provide a number of key insights. First, agricultural expansion patterns were closely related to specific environmental and structural spatial determinants, suggesting that cropland mainly expanded into areas of better agro-environmental conditions, and ranching into areas less suitable for cropping. Second, we found that agricultural expansion into dry forests was less sensitive to changes in profit than agricultural intensification (i.e., conversions from grazing land to cropping), likely because it is nearly always profitable to convert woodlands to croplands or grazing land independently of profit changes and for large-scale actors that have access to capital and focus on maximizing land profitability. This means that woodland conversions were mainly determined by environmental factors and may continue in the future even at small profit changes. Third, the Pampas, which has a longer cropping history, seemed overall more responsive to changes in marginal profits than the Chaco, given the variables included in our model. Finally, comparing actual and predicted patterns of land-use change showed that economic factors were important in some regions (particularly in the Pampas), but could not explain conversion patterns well in others. This suggests that factors not included in our model such as other socio-economic factors (e.g., capital availability or cultural ties to the land) or area-based regulations (e.g, land zoning) are likely also important factors influencing land-use changes.

In our study, environmental factors were more important in determining conversions from woodland to cropland than conversions from woodland to grazing land, whereas transportation costs were not a major determinant of cropland expansion. This likely reflects the economies of scale that exist in the region (i.e., increasing agricultural production resulting in a proportionate saving of transport costs due to expanding infrastructure) (Le Polain De Waroux et al., 2018). Profits are likely less dependent on local logistics and infrastructures, since agribusiness producers own their export facilities and infrastructure (Lende, 2015). This is also apparent in our models, where the likelihood of cropland expansion into woodlands decreased with increasing cost distance only up until 20 USD, but not thereafter (Fig. 3B). This points at a potential cost threshold that may be a constraint for small producers that can amortize transportation investments less than large producers (Sanchez, Cortes, Peralta, & Diaz, 2007). The relatively low importance of profits in determining woodland to cropland conversions can also be explained due to the preference of some land-users for expanding in isolated areas, where lower governmental control on land acquisition and rights exists (Leake & Economo, 2008; Le Polain De Waroux et al., 2018).

Our results also suggest that conversions from woodland to grazing land systems were mainly characterized by the environmental suitability of a location. Grazing mainly expanded into more arid areas, likely because silvopastoral and intensified pasture systems are better adapted to conditions of higher aridity and are more resilient to water scarcity and higher temperatures than cropping systems (Houspanossian, Giménez, Baldi, & Nosetto, 2016; Magrin et al., 2005; Murray, Baldi, Von Bernard, Viglizzo, & Jobbágy, 2016). Moreover, our model showed that higher soil suitability was related to lower likelihood of grazing land expansion, likely because cropping on fertile soils is much more profitable than grazing, and most fertile soils are already used for cropping (Demarco, 2010). Yet, this does not exclude grazing land from expanding over fertile lands in some situations (e.g., where rainfall limits cropping). Cost distances to provincial capitals lowered the likelihood of conversions to grazing land, indicating that transportation costs play a role in expanding agricultural frontiers driven by cattle ranching (Gasparri et al., 2015). Overall, as in the case of cropland expansion, the expansion of grazing land into woodlands was relatively insensitive to grazing profits likely because not all land users that expand agriculture react strongly to price signals in our study. This does not mean that they are not profit maximizers but even when profits decrease, actors are still usually profitable. Moreover, the purposeful stagnation of cattle productivity of the early 2000's by farmers aiming at increasing the value of their cattle stock (Santarcángelo & Fal, 2009) may have strongly influenced the low sensitivity of grazing expansion to profits. Besides, woodland conversions to grazing systems have been suggested to be indirectly connected with soybean expansion elsewhere in the region (Fehlenberg et al., 2017; Gasparri et al., 2013; Murray et al., 2016).

In contrast to agricultural expansion, agricultural intensification (in our case the conversions from grazing land to cropping) was highly sensitive to profit-related variables and occurred mainly in areas characterized by lower aridity and high soil productivity. This is consistent with land rent theory (Lambin, 2012; Le Polain De Waroux et al., 2018), suggesting that the decision to crop or graze on existing agricultural land may indeed be motivated by changes in profit at the margin of these land uses (i.e., the additional profits that motivate investments), as observed elsewhere in South America (Müller, Müller, Schierhorn, Gerold, & Pacheco, 2011; de Espindola, De Aguiar, Pebesma, Câmara, & Fonseca, 2012). In other words, if there were small changes in grazing relative to cropland profits, we would expect large changes in land use, because these systems are both highly profitable, and land-use actors may thus be less capital-constrained in terms of shifting from one land use to another. Our neighbor cropland variables further underline the cropland agglomeration that is taking place in the Pampas due to knowledge and technology transfers, similar as elsewhere in South America (Garrett et al., 2013). Moreover, the longer cropping history of the Pampas (Pengue, 2014) may further explain its higher responsiveness to drivers of agricultural change than the Chaco. Importantly, the fact that the Chaco is less responsive to drivers of agricultural change than the Pampas does not rule out that agricultural intensification may not take place in the future in the Chaco – it is in fact already happening there (Baumann et al., 2016a). It is also noteworthy to highlight, the interrelationship of land-use changes between these two ecoregions is highly interlinked because large-scale land users often own land in both regions and shift uses between regions depending on their interests and investment capacity (Gasparri & le Polain de Waroux, 2015; Pengue, 2014) as observed elsewhere in South America (Macedo et al., 2012).

Comparing predicted and actual land conversions showed that some districts had higher probabilities of land-use/cover conversions than those actually observed between 2000 and 2010, especially in the provinces of Salta and Santiago del Estero (Fig. 4). Under the current national zoning plan (i.e., Forest Law, implemented in 2007; Fig. 4 and Fig. B1 in Appendix), much of these districts fell into the "sustainable use" zones (i.e., yellow zones, where full deforestation is not allowed), or "no use" zones (i.e., red zones, where agriculture is excluded). A second factor explaining the lower-than-expected conversion rates is possibly the presence of indigenous communities that manage forests communally and whose livelihoods depend on non-timber forest products, such as in some areas of Salta province (Fig. B1) as observed elsewhere in Latin America (Barsimantov & Kendall, 2012; Ceddia et al., 2015; Nolte, Agrawal, Silvius, & Soares-Filho, 2013).

Conversely, some districts had higher-than-expected woodland conversion rates, especially in the provinces of Chaco, southern Salta and northern Cordoba. Many of these areas were zoned as "productive" zones in the Forest Law mapping (i.e., green zones, where deforestation is allowed; Fig. 4B and Fig. B1) and deforestation there may also take place in fear of future change in zoning, even if current land utility is not high. Interestingly, areas of highest agreement of actual and simulated conversions from woodlands to grazing lands were mostly located in the Espinal, where expansion of grazing land towards marginal lands from the Pampas took place historically (Pengue, 2014). This may point at distances to regional markets and previous agricultural developments playing an important role, according to land rent theory (Volante et al., 2016; O'kelly & Bryan, 1996; Le Polain De Waroux et al., 2018). High agreement also occurred in 'yellow' zones, where woodland conversion to silvopastures (i.e., pastures with trees) are allowed to some extent in some provinces, especially if conversion to integrated cattle managed systems occurs (Fig. 4C and Fig. B1).

Our study region contains a broad array of land-use actors and different socio-economic contexts suggesting that socio-economic factors others than those included in our model are likely important determinants for explaining agricultural land-use changes in the Chaco and Pampas regions as observed elsewhere in South America (Henderson et al., 2013; Redo, 2013). First, other economic factors not included in our model (e.g., access to land or capital, the existence of agglomeration of economies (Cáceres, 2015; Gasparri et al., 2015; Murray et al., 2016),) may have played an important role in determining land-use change. Our model does not capture well land users' decisions who are mainly interested in securing land rights or claiming land as a commodity for future speculation (Barbier, 2014; Le Polain De Waroux et al., 2018). Second, social factors (e.g., cultural uses and traditional practices (Baldi et al., 2015; Ceddia et al., 2015; Rueda, Baldi, Verón, & Jobbágy, 2013)) may also play an important role in determining land conversions. Our model assumed landowners to maximize their profits and thus disregards landowners that have a preference for cultivating or using the land in traditional ways, not focusing on maximizing profitability (Le Polain De Waroux et al., 2018; Marinaro et al., 2017). However, given the strong presence of large-scale agricultural actors in Argentina, we believe our models, which focus on profit maximization, are a good approximation of decision making in this area. Third, area-based regulations such as spatial planning and land-use zoning (e.g., the Forest Law) can also influence land conversions (Nolte, Le Polain De Waroux, Munger, Reis, & Lambin, 2017b; Le Polain De Waroux et al., 2018)). Although we did not test for the effectiveness of the current land-use zonation in halting deforestation, we can argue that the Forest Law zonation does provide an additional explanation for land-use conversions, because the zones clearly have different agricultural suitability and this influences the likelihood of conversion (Camba Sans et al., 2018; Nolte et al., 2017a).

A few other limitations of our models should be mentioned. First, our broad scale study combined multi-resolution spatial variables that were resampled at a 1-km gridcell. This resulted in a simplification of our dependent and independent spatial variables of coarser resolutions and masked smaller spatial features. For this reason, the woodland definition used in our study (25% tree cover) labelled some sparse dry woodlands with cover below 25% may have been labelled as grazing land in the western Pampa, leading to an underrepresentation of grazing land expansion there and thus of deforestation costs. However, we believe resampling multi-resolution spatial data to a single scale does not hinders our results since our spatial determinants and our modelling approach explicitly accounted for, and minimized, spatial autocorrelation. Second, high quality agricultural production and land-use/cover data for a mid-point in our study would have allowed us to more precisely estimate the impact of (1) changing returns for intermediate land uses (e.g., woodlands converting into grazing lands before a second conversion to croplands), (2) zoning policies (e.g., Forest Law) and (3) landscape configurations on land conversion, but such data was not available. Third, our data does not allow to account for intensification processes occurring within our broad land use classes (e.g., single vs. double cropping, adoption of irrigation, shift from rangeland-based to pasture-based cattle ranching). Fourth, our data also does not account for extra costs derived from climatic variation as a consequence of technological innovation, fertilizers used, and plague or disease incidence. Fifth, our model covers a period with considerable economic perturbations in Argentina, suggesting that landowners had less capacity to react to economic incentives then they would in more stable times. Including access to capital would be a useful extension of our approach to account for this.

4.1. Implications & conclusions

We highlight here that agricultural intensification in the Pampas and Chaco ecoregions of Argentina were more sensitive to marginal profit changes than agriculture expansion into woodlands, as shown by the lower importance of profit-related factors for explaining these conversions (Table 2) and the sometimes high discrepancy of observed and predicted land-use/cover change for the Chaco (Fig. 4). A cautious conclusion from these findings is that as long as global demand for agricultural products continues to grow and net returns of agriculture remain high, both of which is very plausible, continued agricultural intensification in Argentina's Pampas and Chaco regions is likely. Similarly, changes in prices may have only limited impacts on forest conversion rates and patterns, similar to other world regions (Lawler et al., 2014; Radeloff et al., 2012; Stürck et al., 2015), which might question the effectiveness of policy tools that affect marginal profits, such as taxes, subsidies, or payment for ecosystem services schemes. Policies that target socio-economic variables not included in our profit-focused framework (e.g., capital availability) or less-profit oriented actors (e.g., via community-based management of forests or access rights to forest of minority groups), which are partly already in place, might be more promising in curbing deforestation patterns (le Polain de Waroux, Garrett, Heilmayr, & Lambin, 2016; Ceddia et al., 2015). Likewise, area-based interventions such as land zoning or additional protected areas, both of which happened in the region recently (e.g, National Law 26331- Forest Law zonation and National Law 26996 – Creation of the National Park “El impenetrable” Chaco in 2014) might be more promising for curbing deforestation (Bowker, De Vos, Ament, & Cumming, 2017; Heino et al., 2015). This highlights the opportunities connected to the upcoming revision of the Forest Law for steering land conversions away from ecologically and culturally sensitive areas (i.e., indigenous or minority communities). More generally, our study shows how spatial models of net returns can improve understanding of land-use change patterns and what drives them in areas undergoing rapid land-use change.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.apgeog.2018.01.004.

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