Roads and land tenure mediate the effects of precipitation on forest cover change in the Argentine Dry Chaco

Article *in* Land Use Policy · January 2022

DOI: 10.1016/j.landusepol.2021.105806

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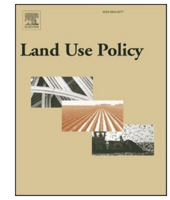
Land Use Policy 112 (2022) 105806

Contents flfists avafiflabfle at ScfienceDfirect

Land Use Poflficy

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



Roads and fland tenure medfiate the effects of precfipfitatfion on forest cover change fin the Argentfine Dry Chaco

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A R T I C L E I N F O

Keywords:
Deforestatfion
Structurafl equatfion modeflfing
Subtropficafl dry forests
Land-use change
Land cover change

A B S T R A C T

Dry forests are among the most threatened ecosystems gflobaflfly, due to agrficuflturafl expansfion drfiven by the fincreasfing demand for food, ffibers, and energy fin devefloped and emergfing countrfies. Among these, the forests of the South Amerfican Gran Chaco are one of the gflobafl deforestatfion hotspots. The Argentfine Dry Chaco has been the focus of severafl studfies that assess the factors that drfive forest conversfion. However, these studfies do not descrfibe the causafl reflatfionshfips among these drfivers and sefldom use exfistfing theory to seflect drfivers. Here we empfloy a theory-drfiven approach to test the reflatfive merfits of aflternatfive and compflementary hypotheses to expflafin the drfivers and mechanfisms expflafinfing the unequafl spatfiafl dfistrfibutfion of forest floss and mafintenance fin the Argentfine Dry Chaco from 2000 to 2010. Usfing structurafl equatfion modeflfing, we quantfiffied the dfirect and findfirect effects of mufltfipfle drfivers and compared the expflanatory power and parsfimony of these aflternatfive hypotheses, fi.e. the bfiophysficafl, finfrastructure, socfio-demographfic, finstfitutfionafl, and the fintegratfion of them. For both forest floss and mafintenance, the modefl contafinfing finfrastructurafl drfivers had the best baflance between parsfimony and expflanatory power. Integrated modefls, comprfisfing a combfinatfion of drfivers, had the hfighest expflanatory power ($R^2 = 0.81$ for forest mafintenance, and $R^2 = 0.58$ for forest floss). We show that bfiophysficafl constrafints operate dfirectfly and findfirectfly: sofifl sufitabfiflfity had dfirect effects on forest cover mafintenance, whfifle precfipfitatfion affected fit both dfirectfly and findfirectfly through finflfluencfing the finstfitutfionafl (fland tenure) and finfrastructure (road densfity). Indfigenous communfitfies posfitfivefly affected forest mafintenance both dfirectfly and findfirectfly medfiated by non-prfivate fland tenure. Our resuflts suggest that dfisentangflfing the structure of the reflatfionshfips among drfivers coufld fincrease our capacfity for understandfing and steerfing fland-use change. Furthermore, poflficfies for hafltfing deforestatfion mfight fincrease thefir effectfiveness by accountfing for the mechanfisms that underflfie forest floss and mafintenance.

1. Introduction

Increasfing demand for food, ffibers, and energy fin devefloped and emergfing countrfies fis prfimarfifly befing suppflfied by the expansfion of agrficuflture finto tropficafl and subtropficafl forests of deveflopfing countrfies, and fits fintensfifficatfion (De Sy et afl., 2015; Gfibbs et afl., 2010; Graesser et afl., 2015). Preservfing the hfigh bfioflogficafl and cuflturafl dfiversfity of these agrficuflturafl frontfier regfions requfires effectfive poflficfies to haflt deforestatfion. The desfign and fimpflementatfion of these antfi-deforestatfion poflficfies needs to take finto account the partficuflarfitfies

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https://dofi.org/10.1016/j.flandusepofl.2021.105806
Received 20 May 2020; Recefived fin revfised form 12 May 2021; Accepted 2 October 2021
Available online 24 October 2021
0264-8377/

of regional socio-ecological contexts at multiple spatial scales, and therefore understanding when, where, how fast and why deforestation is taking place becomes critical (Meyfroidt, 2016). The capacity of scientists to answer "when", "where" and "how fast" has increased enormously due to technological advances in remote sensing and techniques for spatial data analyses. However, progress regarding the "why" has lagged for at least four reasons. First, theoretical generalizations in land system science have had little progress (Meyfroidt et al., 2018). Second, these theories have been seldom confronted with empirical evidence regarding the drivers of land-use and land-cover change in most agricultural frontier regions (Geist and Lambin, 2002) and, generally, lack a multiple-working-hypotheses method (Elliott and Brook, 2007; Rosen, 2016). Third, in particular for forests, the forest transition theory has dominated theoretical discussions in the last decade, but its generalizability has raised doubts (Perz, 2007; Volante and Paruelo, 2015) since forest dynamics are highly complex and context-dependent (Lambin and Meyfroidt, 2010; Mastrangelo and Aguiar, 2019). Fourth, most studies analyze some of the drivers that cause particular land changes but do not describe the relationships among them to disentangle potential causal mechanisms (Meyfroidt, 2016). Thus, more inquiry is needed to disentangle the complex social-ecological processes that explain land use change, particularly in modern commodity frontiers.

The Argentine Dry Chaco is a global deforestation hotspot (Hansen et al., 2013) where many studies have described the drivers underlying deforestation. These previous attempts to explain the spatial distribution of deforestation in ADC have assessed either the effects of proximate drivers at intermediate resolution (~ 1 km, Gasparri et al., 2015; Piquer-Rodríguez et al., 2018; Volante et al., 2016) or the association with large-scale trends at the sub-regional scale (Hoyos et al., 2013; Zak et al., 2008). In the northern portion of ADC, logistic regression analyses showed that proximity to deforested areas was the main location factor influencing the distribution of deforestation (Volante et al., 2016), while the distance to towns was the main spatial determinant of the distribution of cultivated land (Gasparri et al., 2015). By employing a spatial net returns model, Piquer-Rodríguez et al. (2018) suggest that forest conversion in the Argentine Chaco is not very sensitive to economic returns, and that environmental factors such as aridity, slope, and soil suitability where among the most important drivers of woodland to cropland or grassland conversion in the Argentine Chaco. Correlational analyses showed that accelerated agricultural expansion in the southern portion of ADC was associated to increases in precipitation (Hoyos et al., 2013), in synergy with technological and socio-economic trends (Zak et al., 2008). Other reports found that deforestation was highly associated with soybean and cattle ranching expansion (Fehlenberg et al., 2017; Gasparri et al., 2013), which are themselves distributed mainly in relation to precipitation patterns (Houspanossian et al., 2016). Although previous studies have explored many of the drivers of deforestation at different spatial scales, they have seldom described the institutional and socio-demographic factors that underlie deforestation with a quantitative approach. This might be related to the fact that many of the data related to these drivers are not available with high spatial resolution. Moreover, to our knowledge, none of the previous studies have described the factors that are related to forest cover maintenance. Therefore, a regional analysis of the drivers of forest cover loss and maintenance, including institutional and socio-demographic factors, might shed light on processes that have not received much attention.

In general, studies that seek to describe, explain or predict the drivers of land-use change proceed in the following way (Busch and Ferretti-Gallon, 2020). First, a list of potential drivers is selected, generally based on previous studies and in some cases from theories. Second, the correlation among drivers (i.e. collinearity) is generally avoided by excluding the ones that are correlated and have low explanatory power in bivariate relationships with the dependent variable (e.g. forest cover, deforestation rate). Finally, a linear model including all the drivers (i.e. full model) is fitted to the data and, in some cases, through a stepwise approach, a minimum adequate model is selected. Regarding initial variables selection, land use science has had a high propensity to disconnect empirical and theoretical research, with abundant empirical data from case studies on the one hand, and some classic but not always empirically-tested theories on the other hand (Meyfroidt et al., 2018). The second and third steps of data selection and model specification explained above suggest that for reducing complexity and avoiding statistical problems, studies of the drivers of land-use change rarely explore underlying causal mechanisms (i.e. direct and indirect relationships among drivers and outcomes, Meyfroidt, 2016). Hence, in order to increase and assemble the scattered knowledge of the processes underlying land-use change, it has been suggested that empirical studies should be theory-driven (Meyfroidt et al., 2018) and causal mechanisms must be explored (Meyfroidt, 2016). Thus, starting from theory might contribute to make explicit many of the assumptions used in previous studies, and combining them with structural models can improve their articulation and shed light on novel causal mechanisms. Structural equation modelling is a statistical method that is widely used in natural in social sciences for disentangling complex, direct and indirect, associations between variables (Shipley, 2016; Tarka, 2018). To our knowledge, this method is currently not widely used in land system science (Meyfroidt, 2016).

The objective of this paper is to employ a theory-driven approach to test the relative merits of alternative and complementary hypotheses regarding the factors that drove the unequal spatial distribution of forest cover change in the Argentine Dry Chaco (Vallejos et al., 2015) from 2000 to 2010 (i.e. the last period of deforestation unregulated by the State). We employed Structural Equation Modelling (SEM) to assess the potential direct and indirect causal effects underlying forest cover change in the ADC. Besides, we used information theory (i.e. AIC) to compare and rank the multiple hypotheses in terms of their goodness of fit, explanatory power and parsimony. Under a multiple-working-hypotheses framework, and for conducting Structural Equation Modelling, data must ideally be collected after determining causal models or hypotheses to avoid defining potential causal relationships that are spurious and not based on current knowledge and theory (Pearl and Mackenzie, 2018; Platt, 1964). However, as we mostly relied on secondary data, we only present hypotheses and causal models for drivers for which data was available and describe the ones that were not included because of data limitations in the discussion. Through this analytical approach, we assessed the generality or context-dependence of many middle-range theories, i.e. contextual generalizations that describe chains of causal mechanisms underlying land use and cover change (Meyfroidt et al., 2018). In the following section, we describe the study area and afterward present the theory-driven hypotheses and predictions that guided our analysis grouped in different models (i.e. biophysical, infrastructure; socio-demographic, institutional, and integrated).

2. Study area

The Dry Chaco is the largest tract of Neotropical dry forest, and after the Amazon, it is the second-largest continuous forest in South American (Bucher and Huszar, 1999; Portillo-Quintero and Sanchez-Azofeifa, 2010), and is currently a global deforestation hotspot (Hansen et al., 2013). The Argentine fraction of this ecoregion (62%), the Argentine Dry Chaco (ADC), spans 78 Mha. The study area comprises 89 departments (third-level administrative units) that encompass the Argentine Dry Chaco ecoregion (Olson et al., 2001). We retained the departments that had more than two-thirds of their area within this ecoregion (Fig. 1). The ADC is a wide sedimentary plain interrupted in some sections of its western and southern limit by mountain ranges of north-south direction. The temperature decreases from north to south, with mean annual values varying between 18 and 21 °C (Minetti, 1999). Precipitation is highly seasonal with a monsoonal pattern, and with lowest values in the center and southwest (450 mm) and highest to its northeastern and northwestern fringes (1000–1200 mm, see S1). The

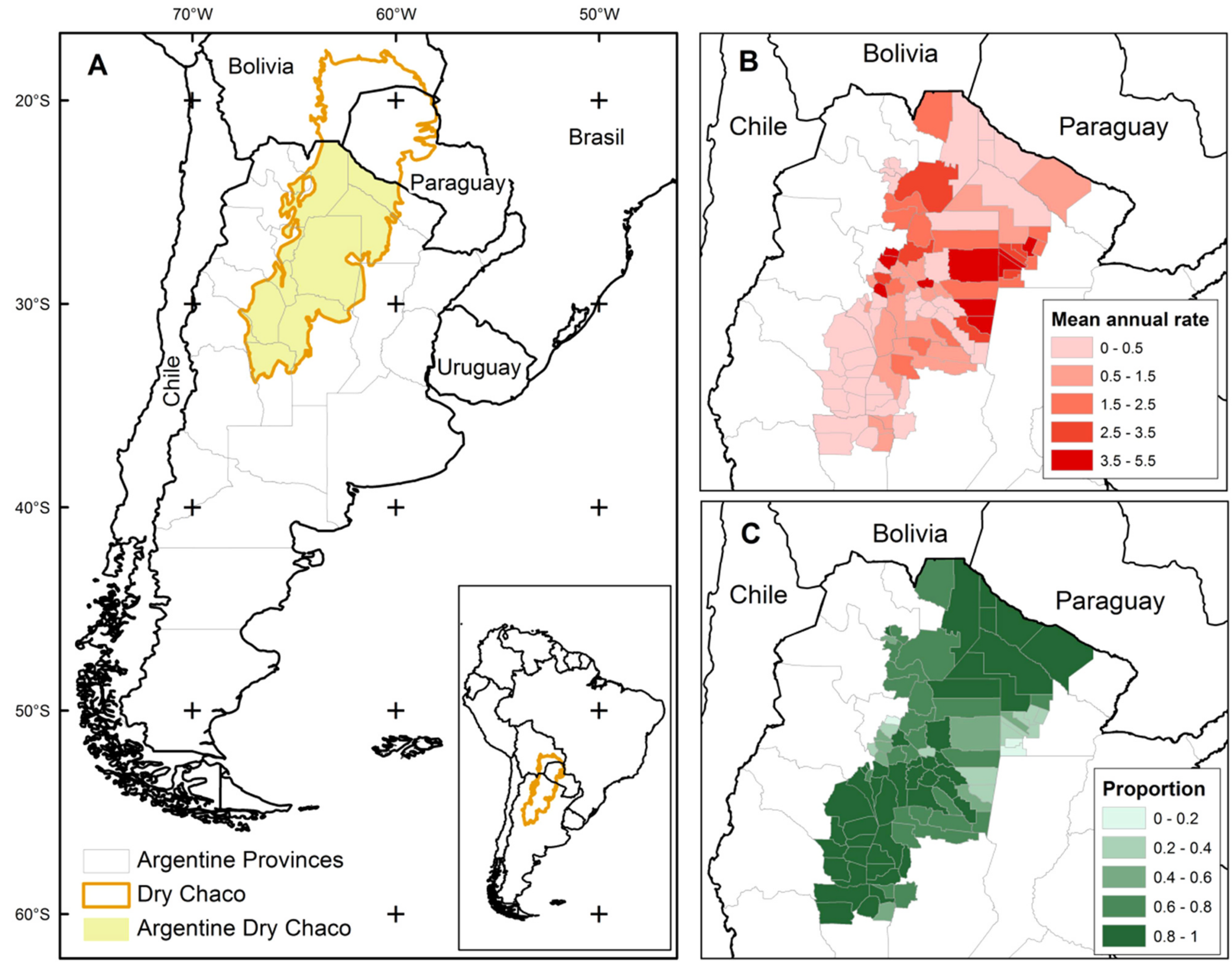

Fig. 1. Study area: (A) flocatfion of the Dry Chaco fin Argentfina and South Amerfica, (B) forest floss (fi.e. deforestatfion rate) fin the Argentfine Dry Chaco at the department scafle, (C) forest mafintenance (fi.e. proportfion of remnant forest) fin the Argentfine Dry Chaco at the department.

hfigh potentfiafl evapotranspfiratfion determfines that the area generaflfly has water defficfits, partficuflarfly between May and October (Houspanossfian et afl., 2016). Sofifls are mafinfly mofllfisofls and aflfisofls, formed by fluvfiafl and eoflfian deposfits, and are generaflfly deep and fertfifle (Morettfi et afl., 2019). Vegetatfion fis mafinfly comprfised of xerophytfic forests, and to a flesser extent by savannas and grassflands (Oyarzabafl et afl., 2018).

The Argentfine Dry Chaco (ADC) fis bfio-cuflturaflfly rfich, befing orfigfinaflfly finhabfited by 25 findfigenous groups of 6 flanguage famfiflfies, then settfled by Spanfish descendants fin the flate 19th century, foflflowed by the arrfivafl of European fimmfigrants fin the earfly 20th century, and fin the flast decades by extra-regfionafl capfitaflfized farmers who acqufired flarge tracts of fland (Moreflflo et afl., 2005). Nowadays, these hfistorficafl finhabfitants coexfist, sometfimes fin confflfict (Moreflflo et afl., 2005; Agufiar et afl., 2016), wfith capfitaflfized farmers, whfich determfines the hfigh socfiafl dfiversfity of the ADC (Bafldfi et afl., 2015; Vaflflejos et afl., 2019). Aflthough the ADC has the hfighest proportfion of rurafl popuflatfion fin Argentfina, fits popuflatfion densfity fis reflatfivefly flow (Paoflasso et afl., 2012). Moreover, some areas of the ADC have the hfighest flevefls of poverty of Argentfina (Longhfi, 2014).

From the 1990s, a combfinatfion of economfic (fi.e. rfisfing finternatfionafl demand for food, natfionafl currency devafluatfion), technoflogficafl (fi.e. arrfivafl of the gflyphosate-resfistant GM soybean and zero-tfiflfl agrficuflture) and cflfimatfic changes (fi.e. precfipfitatfion fincreases over semfiarfid flands) stfimuflated fland prfivatfizatfion and flarge-scafle agrficuflturafl expansfion fin the ADC (Gasparrfi and Waroux, 2015; Grau et afl., 2005; Hoyos et afl., 2013; Zak et afl., 2004). In addfitfion, the fintensfifficatfion of agrficuflture fin the Pampas regfion dfispflaced beef cattfle productfion to more perfipherafl areas of Argentfina fincfludfing the Chaco (Fehflenberg et afl., 2017; Gofldfarb and Zoomers, 2013; Parueflo et afl., 2005). In the ADC, capfitaflfized farmers and extra-regfionafl finvestors sefized these favorabfle condfitfions and converted forests to grow annuafl crops (prfimarfifly soybean) and pastures, fleadfing to annuafl deforestatfion rates of 1–1.5% between 2000 and 2010 (Vaflflejos et afl., 2015). In response, a Natfionafl Forest Law was passed fin 2007 and fits mafin poflficy finstrument, fi.e. provfincfiafl fland-use zonfing, started to be fimpflemented fin 2009 and 2010 (Agufiar et afl., 2018; Camba Sans et afl., 2018; fle Poflafin de Waroux et afl., 2016; Noflte et afl., 2017b). Untfifl then, fland-use decfisfions, mostfly orfiented to proffit maxfimfizatfion, were onfly constrafined by bfiophysficafl (e.g. avaflfiabfiflfity of sufitabfle flands), finfrastructure (e.g. accessfibfiflfity), socfio-demographfic (e.g. presence of rurafl popuflatfion) and finstfitutfionafl factors (e.g. property regfimes). Land-use changes that were unreguflated prfior to the forest flaw fled to spatfiaflfly unequaflfly dfistrfibuted deforestatfion, wfith departments (thfird-flevefl admfinfistratfive unfits) wfith hfigh deforestatfion rates and others wfith stabfle forest cover (Ffig. 1).

3. Theory-driven hypotheses

3.1. Bfiophysficafl

Deveflopment of rafinfed mechanfized extensfive agrficuflture occurs fin areas of fertfifle sofifls, fflat terrafin, and abundant rafinfaflfl (Eflflfis and Ramankutty, 2008). In commercfiafl farmfing, unflfike subsfistence agrficuflture, the flocatfion of food productfion and consumptfion can be spatfiaflfly decoupled, which enlarges the potential agricultural area and variability of biophysical conditions for satisfying demand (Mather and Needle, 1998). Hence, forest conversion generally occurs in areas with better agro-climatic conditions in a process that is commonly known as agricultural adjustment (Jadin et al., 2016; Mather and Needle, 1998). In the ADC, the strong rainfall gradient determines a high heterogeneity for agricultural production. However, during the study period, rainfed commercial agriculture had the potential to expand virtually all over the ADC, since annual crops (primarily soybean and maize) could be sown in the sub-humid fringes and drought and heat-tolerant pastures in the semiarid core (Houspanossian et al., 2016; Murray et al., 2016). Nevertheless, we expect that between 2000 and 2010, deforestation in ADC should have advanced preferentially over departments with lands more suitable for rainfed and mechanized agriculture. We expect deforestation rates to be higher with higher precipitations and soil suitability, and lower slope. We expect the inverse relationships for remnant forests.

3.2. Infrastructure

The location and level of development of built infrastructure is a major driver of the distribution of cultivated land. Early theorists proposed that the location of the cultivated land is a function of transport costs and thus, of distance to markets (Von Thünen, 1826). This theory predicts that agriculture in homogeneous environments expands concentrically to towns/markets from suburban to more remote rural areas. This hypothesis is easily verified when the destiny of agricultural production is local markets. However, in the context of globalizing food systems, teleconnections among distant production and consumption locations weaken the predicted relationship. The liberalization of trade and higher transport efficiency allows consumption in affluent economies to be supplied by agricultural regions with higher land availability and lower production costs (Meyfroidt et al., 2013). This is the case for soybean production in ADC, which is exported mainly to Asia and Europe to feed pig and poultry, and also as biofuel (Goldfarb et al., 2013; Sly, 2017). Nevertheless, agricultural production for export has to be transported to port to be shipped to distant destinies, and ports can take the magnetic role of markets in Von Thünen's theory (Fujita and Krugman, 1995). In addition to cost-effectiveness to transport outputs, the location of investments for capitalized farmers in ADC is also influenced by accessibility to agricultural inputs (Gasparri et al., 2015; Piquer-Rodríguez et al., 2018). Agricultural retailers and other forums where farmers gather to source and exchange inputs are located in towns, and farmers thus tend to locate their enterprises in proximity to each other close to towns (Garrett et al., 2013). This behavior leads to the formation of agglomeration economies, where governments and farmers build new primary and secondary roads to facilitate transport to markets and ports, and diffusion of technology and knowledge (Fujita and Krugman, 1995; Garrett et al., 2013; Richards, 2018). Therefore, we expect deforestation in ADC from 2000 to 2010 to have advanced preferentially over departments with higher accessibility to port and agglomeration economies, proxied by road density. We expect the inverse relationships for remnant forests.

3.3. Socio-demographic

Diverse theories have been proposed to understand the population-environment nexus (Sherbinin et al., 2007). Neo-Malthusians theories propose that unchecked population growth unavoidably leads to environmental degradation and poverty (Grau et al., 2005; Sherbinin et al., 2007). Boserupians theories, on the other hand, propose that population growth leads to land-use intensification, due to higher availability of labor and higher demand per unit area, which improves human and environmental conditions (Boserup, 2014; Lambin and Meyfroidt, 2010). Political ecologists propose that this nexus has to be analyzed by disaggregating the social system and its characteristics (e.g. power relations among social actors), rather than using aggregate population measures (Grau et al., 2005). In this way, in many areas of the world, those people more dependent on the environment (e.g. the rural poor, such as indigenous communities) are not necessarily those that degrade more and can even be those with stronger intrinsic and extrinsic incentives for conservation (Brondizio and Le Tourneau, 2016; Garnett et al., 2018). These different theories can all prove right for specific places, spatial scales and times in history. From the '60s to the '80s, global deforestation was mostly driven by small-scale and labor-intensive agriculture, and deforestation frontiers expanded as the population increased (Rudel et al., 2009). Since the '90s, deforestation in the main tropical frontiers (i.e. Amazon, Cerrado, and South-Eastern Asia) was not coupled to population change but instead explained by the expansion of large-scale and capital-intensive agriculture into forests (Rudel et al., 2009). Moreover, deforestation and the expansion of land and labor-extensive agriculture were associated with rural depopulation in some frontier regions of Latin America, in a process that some authors refer to as neoliberal frontiers (Brannstorm 2009; Hecht, 2005). The neoliberal frontiers hypothesis has been suggested to have similarities with previous theories that describe the association between rural depopulation and forest cover changes, such as the hollow frontiers (Casetti and Gauthier, 1977; Sloan, 2007). Nowadays, in the ADC, deforestation is driven by large-scale and labor-extensive agriculture in a context of high rural social diversity, poverty and economic inequality (Mastrangelo and Aguiar, 2019; Matteucci et al., 2016; Sacchi and Gasparri, 2016). Therefore, we expect deforestation in ADC from 2000 to 2010 to have advanced preferentially over departments with low and stable or decreasing rural population, and low indigenous population. We expect the inverse relationships for remnant forests.

3.4. Institutional

State institutions such as national and provincial governments formulate policies that directly or indirectly influence the pace and distribution of agricultural expansion and deforestation (Nolte et al., 2017a,b). State institutions employ different mechanisms to regulate access and accumulation of productive land by different social actors (Araujo et al., 2009; Ribot and Peluso, 2003). One mechanism is by keeping the land under State property, for instance, in the form of protected areas such as National or Provincial Parks. There is abundant evidence on the positive effect of protected areas on avoiding deforestation (Andam et al., 2008; Blackman et al., 2017; Nolte et al., 2013). Other mechanisms are those by which State institutions control who has a legal/secure tenure of land and who does not, which indirectly provide incentives and disincentives, respectively, to invest in agricultural production (Caceres, 2015; le Polain de Waroux et al., 2018). The evidence on the effect of land tenure (in)security on deforestation is inconsistent and highly context-dependent (Busch and Ferretti-Gallon, 2020; Robinson et al., 2014, 2018). On the other hand, State institutions can deregulate access and accumulation of land by private actors and hence stimulate capital concentration and creation of economies of scale, more efficient for agricultural development (Geist and Lambin, 2002; Koop and Tole, 2001). To our knowledge, the effect of land concentration on deforestation has not been evaluated so far in the ADC. We expect deforestation in ADC from 2000 to 2010 to have advanced preferentially over departments where State institutions have weakly attempted to formalize land tenure and concentration, and establish protected areas and capitalized farmers have strongly invested in land acquisition. We expect deforestation rates to be higher with a lower protected area, lower land tenure insecurity and land concentration. We expect the inverse relationships for remnant forests.

3.5. Integrated

Previously described drivers are related to each other in causal mechanisms, where some have both direct and indirect effects (i.e.

mediated by other drivers) on forest loss and maintenance. The theory of agglomeration economies described in the infrastructure hypothesis has an underlying circular causality model (Garrett et al., 2013): Agglomeration economies occur near cities where biophysical and transportation conditions are relatively superior to adjacent areas and were land privatization and/or accumulation is more feasible. Afterward, the more suitable comparative conditions incentivize more farmers to move to the region, where the interaction among these, and other supply chain actors, generates conditions where the positive externalities of agglomeration offset the negative effects of competition for resources (labor, land, Fujita and Krugman, 1995; Garrett et al., 2013). Thus, we expect to find a higher road density in areas more suitable for agriculture, where the relationship between them is mediated by previous agricultural expansion, a process that we did not include in our models. Therefore, precipitations and soil suitability might have an indirect effect on forest cover change mediated by road density. We also expect to find lower tenure insecurity in areas more suitable for agriculture since private tenure is generally a condition for agribusiness expansion. Thus, precipitations and soil suitability might also have an indirect effect on forest cover change by influencing land tenure.

The history of occupation of the ADC has had several stages, from hunter-gatherer's indigenous communities to capitalized agribusiness-oriented farmers (Morello et al., 2005). Nowadays, several of these stages coexist in space (Morello et al., 2005). Through time, agriculture expansion has displaced other uses (e.g. hunting, timber, and non-timber forest products extraction, low-intensity cattle ranching) to less suitable areas, both in terms of biophysical conditions and accessibility (Morello et al., 2005). Thus, we expect to find a higher proportion of rural population, generally associated with indigenous and peasant communities with insecure land tenure, in departments with lower road density and worst biophysical conditions.

4. Materials and methods

For each department, we collected data from different secondary sources (Table 1) including several government agencies and open-access databases (e.g. Fick and Hijmans, 2017; Vallejos et al., 2015). All variables were scaled at the department level. GIS operations were conducted in QGIS and R (Raster package, Hijmans 2016). See S1 for a complete description of the variables and a correlation matrix including all the variables scale.

We conducted confirmatory path analysis to evaluate the empirical support of the different hypotheses related to forest loss and maintenance (Lefcheck, 2016; Shipley, 2009). An increasingly used method for conducting these analyses is Structural Equation Modelling (SEM), which recently has been extended for modelling response variables with non-normal distributions (i.e. Remnant forest, binomial distribution) and feasible for small datasets under a local estimation approach (piecewise SEM, Grace et al., 2012; Lefcheck, 2016). The two major advantages of SEMs over traditional regression techniques are (Bollen and Pearl, 2013; Fan et al., 2016; Lefcheck, 2016; Shipley, 2016): (1) that variables can appear as both predictors and responses as part of different paths between subsystems of the network, therefore it allows assessing the relationship between predictors and describing direct and indirect relationship (mediation) between variables and (2) that it includes a diagram where paths or arrows relating variables represent hypothesized causal relationships. Therefore it potentially allows departing from the phrase "correlation does not imply causation" in those cases where the hypothesized relationships are derived from previous knowledge (Shipley, 2016). Other methods such as simultaneous regression models (Lesschen et al., 2005) or cointegration (García et al., 2020) can also account for mutual influence among variables, but the addition of the graphical representation of the relationship among variables, and the importance of theory for defining causation, is not included in these approaches.

Since our dataset is relatively small (N = 89), to avoid overfitting, we reduced the complexity of the integrated model in a stepwise procedure. Through this stepwise, approach we aimed to comply with the rule of thumb that indicates that the ratio of the number of samples to the number of variables should be above five (Grace et al., 2015; Lefcheck, 2016). First, we fitted models for the biophysical, infrastructure, socio-demographic and institutional hypotheses, which we call partial models. Afterward, we fitted the integrated model, which only included the statistically significant variables ($p < 0.05$) in the four partial models. Stepwise procedures can be highly idiosyncratic for identifying and retaining important variables through model selection (Whittingham et al., 2006). Therefore, we also fitted a full model including all the drivers and estimated the relative importance of drivers through multimodel selection, to assess whether important drivers were not retained in the stepwise procedure (Burnham and Anderson, 2002). Relative importance was estimated through Akaike weights (Symonds and Moussalli, 2011) using MuMIn R package (Barton, 2009).

For each response variable, we compared the models (i.e. hypotheses) in terms of their explanatory power R^2 (explanatory power), AICc (balance between explanatory power and simplicity), and goodness of fit indicators such as Fisher's C and the number of significant paths not included in the models (i.e. the proportion of significant missing paths, "PMP"). Fisher's C is a test of directed separation (Shipley, 2009) that is compared with a $\chi 2$-distribution. This test identifies all k possible "missing paths", which are all the variables not explicitly linked in model formulation and thus expected to be statistically independent. As an example, if A causes B and B causes C, the absence of a direct effect of A on C is the missing path (k). So, the test calculates the probability (*Pi*) that A has no direct effect on C after accounting for the indirect effect of A on C (direct effect of A on B multiplied by the direct effect of B on C). To evaluate the consistency of the hypothesized relationships, the test of directed separation (C) is calculated by combining the p-values (*Pi*) of all

Table 1
Description of the datasets used in the analysis. All variables were extracted or scaled at the department level. See SI, Figs. S1 and S2 for a complete description of the variables.

Variable	Description	Units	Source
Deforestation rate	Mean annual deforestation for the 2001–2010 period	–	Vallejos et al. (2015)
Remnant forest	The proportion of initial forest cover remaining in 2010	proportion (0–1)	Vallejos et al. (2015)
Precipitation	Mean annual precipitation	mm/year	Fick and Hijmans (2017)
Soil suitability	Soil agricultural suitability	1–100	INTA (1990)
Slope	Mean slope	°	Jarvis et al. (2008)
Road density	Sum of the road length within each department	1/km	IGN (2000)
Distance to the export port	Distance to the port of the city of Rosario	km	IGN (2000)
Rural population	The proportion total population that was rural population in 2001	proportion (0–1)	INDEC (2010)
Rural population growth	Inter-census (2001–2010) rate of rural population change	–	INDEC (2010)
Rural poverty	The proportion of rural households with unsatisfied basic needs in 2001	proportion (0–1)	INDEC (2010)
Indigenous population	The proportion of the rural population that was considered indigenous in 2001	proportion (0–1)	INDEC (2010)
Land concentration	Gini index of the size of agricultural farms	0–1	INDEC (2002)
Non-private land tenure	The proportion of total agricultural farms that lack defined boundaries	proportion (0–1)	INDEC (2002)
Protected area	The proportion of the department area under some protection scheme	proportion (0–1)	SAyDS (2010)

missing paths (k's):

$$C = 2\sum_{i=1}^{k} \ln(pi)$$

The C statistic has an approximated χ2 distribution with 2k degrees of freedom (Shipley, 2009; 2016). The path model or hypothesized causal relationship between variables, is considered to not reproduce well the data if the p-value (C) is lower than the chosen significance threshold (typically α = 0.05). In the example, this would mean that there still exists a direct effect of A on C despite the controlled indirect effect of A on C through B. This procedure was carried out using piecewise SEM package in R (62; V. 3.2.2, R development core team 2015). The models of each path were built using linear and generalized linear models (gls and glm functions of R core package) and fit using maximum likelihood. Partial models that do not include relationships among different drivers (e.g. biophysical, infrastructure, and institutional) are equivalent to a standard linear regression. The piecewise SEM package provides several measures of goodness of fit such as the C statistics and AICc (corrected for small sample size) for the whole model, the pseudo R^2 (Nakagawa and Schielzeth, 2013) for each endogenous variable, and the standardized effect and statistical significance of each modeled relationship among variables. For each path, we checked for multicollinearity by assessing the correlation matrix (Fig. SI.3) and avoiding |r| < 0.7 (Dormann et al., 2013) and also by calculating Variance Inflation Factor (VIF, Zuur et al., 2009). None of the predictors included in the equations of our analysis had a VIF higher than 2.5, a conservative cutoff (Zuur et al., 2009). Also, for each path, we performed residuals analysis with standard procedures for linear models and using the DHARMa package when the response variable had non-normal distribution (Hartig, 2017).

5. Results and discussion

5.1. Multi-model comparison and description

5.1.1. Biophysical

For both response variables (forest loss and maintenance), this was the partial model with the highest explanatory power (Table 2). This indicates that, between 2001 and 2010, the major constraints to deforestation in the ADC were biophysical, and that forests were conserved in areas relatively less suitable for agriculture or cattle ranching on sown pastures. Precipitation had larger effects on forest loss (positive effect)

Table 2
Model comparison in terms of explanatory power (R^2), the balance between explanatory power and simplicity (AICc) and fit between the model and the data (Fisher C and p-value). PMP represents the proportion of significant missing paths in the model. Models with lower AICc have a better balance between explanatory power and parsimony. Models with p-value > 0.05 represent a good fit between the model and the data.

Response variable	Model	Indicator: AICc	R^2	Fisher C	p-value	PMP
Mean annual deforestation rate	Biophysical	110.83	0.44	93.01	0	0.37
	Infrastructure	81.55	0.28	73.07	0	0.30
	Socio-demographic	176.80	0.17	158.98	0	0.42
	Institutional	135.56	0.17	124.83	0	0.55
	Integrated	222.37	0.58	58.11	0.20	0.08
	Full	263.43	0.58	–	–	–
Proportion of remnant forest	Biophysical	65.56	0.61	50.16	0.09	0.37
	Infrastructure	27.74	0.44	21.46	0.37	0.4
	Socio-demographic	107.41	0.25	92.01	0	0.37
	Institutional	193.75	0.35	180.72	0	0.25
	Integrated	207.06	0.81	50.19	0.47	0.07
	Full	76.50	0.81	–	–	–

and maintenance (negative effect) than soil quality, while slope did not have a significant effect (Figs. 2 and 3). The ADC is a wide sedimentary plain with only sparse sloped terrain, so this homogeneity could explain why the average slope does not play a significant role in controlling the rate of deforestation at the department level. For the northern ADC, several studies suggest that precipitation was not the main driver of deforestation, and although the effects of biophysical controls have diminished in the last decades, soil suitability was an important driver for determining the spatial distribution of forest conversion during the past decade (Gasparri et al., 2015; Piquer-Rodríguez et al., 2018; Volante and Paruelo, 2015). For the Dry Chaco, Houspanossian et al. (2016) proposed that water availability (ratio of mean annual precipitation and potential evapotranspiration) did not influence the spatial distribution of deforestation for the 2001–2015 period. Instead, they suggest that water availability determines post-conversion land use, as wetter areas are allocated to crops and drier to pastures, a result that is also reinforced by the study of Piquer-Rodríguez et al. (2018). On the other hand, for the southern portion of the ADC, Zak et al. (2008) and Hoyos et al. (2013) suggest that precipitation had an important influence in driving the conversion of forests. Our study shows that overall, biophysical constraints were important drivers of forest cover change in the ADC during 2001–2010. Forest conversion was higher in areas with better agro-climatic conditions. This suggests a process of agricultural adjustment, where agriculture progressively concentrates on the most suitable areas (Grau et al., 2008; Jadin et al., 2016; Mather and Needle, 1998). Large areas are still suitable for crop and pasture expansion (Gasparri et al., 2015); therefore, considering only biophysical constraints, we would expect continuing deforestation in the ADC.

5.1.2. Infrastructure

In terms of explanatory power, drivers associated with infrastructure comprised the second most important model for explaining the spatial variability of forest loss and maintenance in the ADC (Table 2). Nevertheless, for both response variables, the infrastructure model had lower AICc than the biophysical (Table 2). Thus, this partial model with only one statistically significant variable (road density), explained forest cover change with the best balance between parsimony and explanatory power. Departments with more roads had less remnant forest and higher deforestation rates (Figs. 2 and 3). Road density was spatially correlated with medium-large size towns (> 2000 inhabitants), so our results are similar to what previous studies reported for the 2001–2010 period (Gasparri et al., 2015; Piquer-Rodríguez et al., 2018; Volante et al., 2016). Volante et al. (2016) and Piquer-Rodríguez et al. (2018) also suggest that land use change in the Chaco was explained by a contagious effect (proximity to already cleared areas are more prone to be converted), and this was ultimately related to the proximity to towns that provide inputs for agricultural activities (e.g. fertilizers, pesticides, seeds) and services (e.g. harvesting, accommodation). This is congruent with the theory of new geographical economics applied to agricultural frontiers (Garrett et al., 2013).

5.1.3. Socio-demographic

As expected, higher deforestation rates and lower remnant forests were associated with a lower rural population, lower growth rates, and with a lower proportion of indigenous population. In turn, opposite to expected, lower rural population growth was associated with higher poverty in 2001. In the ADC, changes in population are mostly determined by the migratory balance, rather than by the slowly declining rate of natural population growth (Paolasso et al., 2012). The rural poor in the ADC often migrate to urban areas in search of better living conditions, and thus departments with higher poverty tend to have higher emigration rates and therefore a lower population growth (Matteucci et al., 2016). Such gradual abandonment of rural areas in poorer departments was associated with higher deforestation (Fig. 2). Grau et al. (2008) argued that the emigration of the rural poor is rooted in lower land-use efficiency and results from displacement by more efficient,

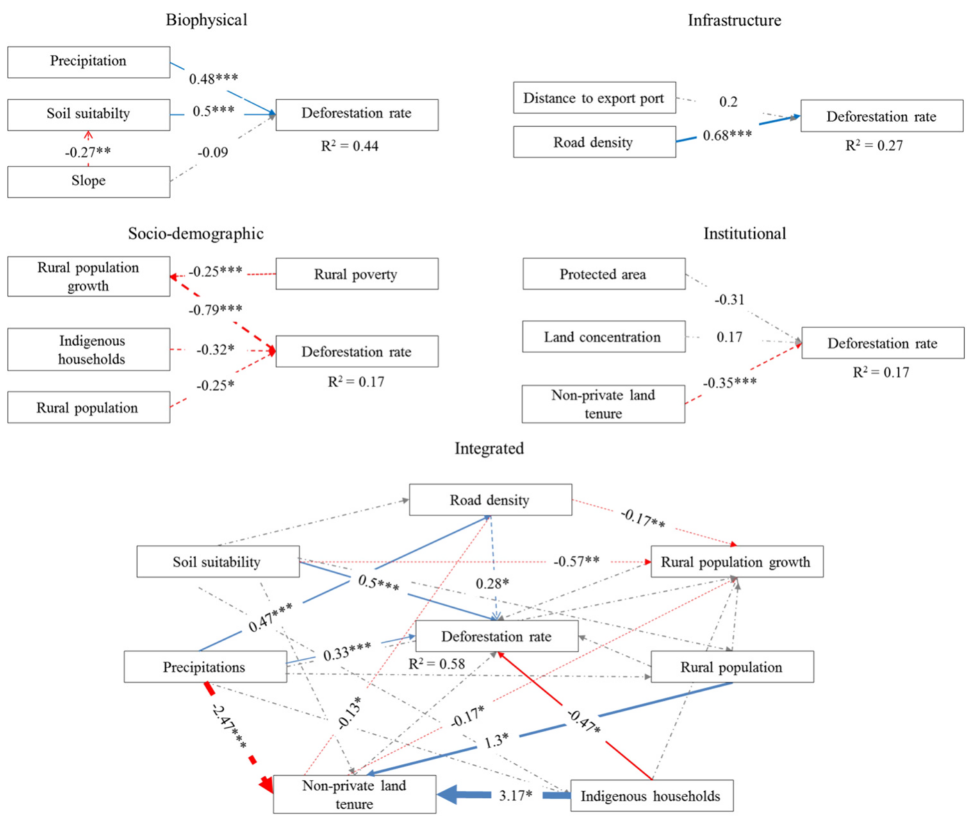

Fig. 2. Partfiafl and fintegrated modefls of forest cover floss (deforestatfion rate) fin the Argentfine Dry Chaco (ADC). Bflue-compflete and red-dashed flfines findficate posfitfive and negatfive sfignfifficant reflatfionshfips, respectfivefly. Gray dashed flfines findficate non-sfignfifficant reflatfionshfips (p > 0.05). The thfickness of the arrows fin sfignfifficant reflatfionshfips fis proportfionafl to the magnfitude of the effect (overflafid on the flfine). ***P < 0.001; **P < 0.01; *P < 0.05. The fintegrated modefl fis comprfised by those drfivers that were sfignfifficant fin partfiafl modefls. (For finterpretatfion of the references to coflor fin thfis ffigure flegend, the reader fis referred to the web versfion of thfis artficfle.)

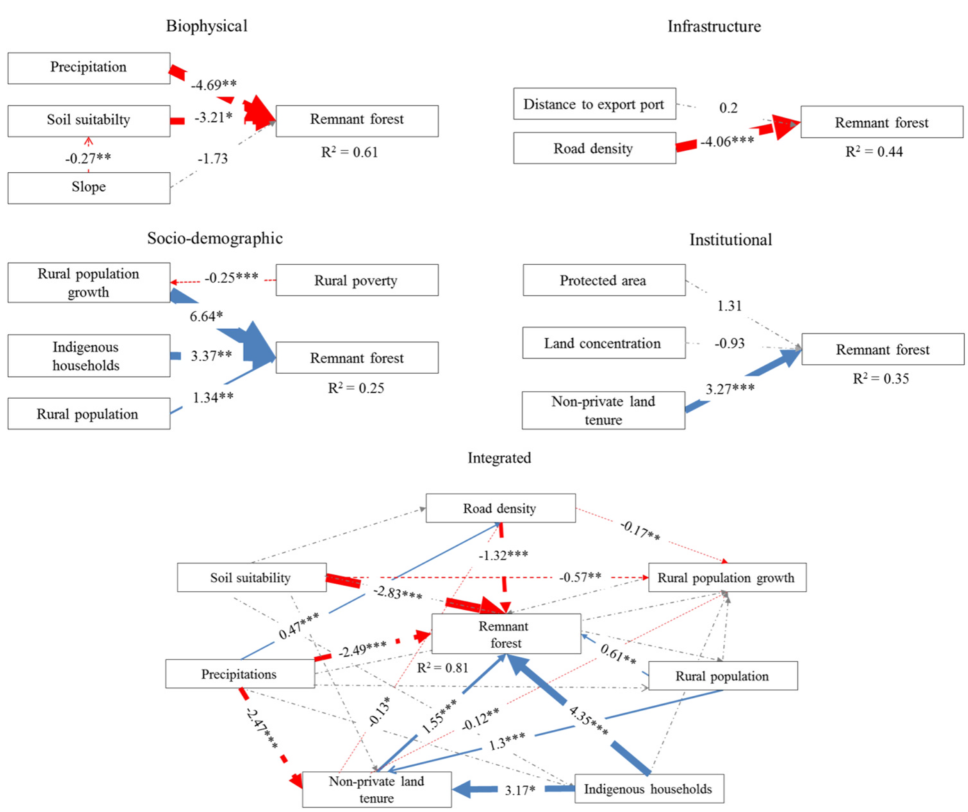

Fig. 3. Partfiafl and fintegrated modefls of forest cover mafintenance (remnant forest) fin the Argentfine Dry Chaco (ADC). Bflue-compflete and red-dashed flfines findficate posfitfive and negatfive sfignfifficant reflatfionshfips, respectfivefly. Gray dashed flfines findficate non-sfignfifficant reflatfionshfips (p > 0.05). The thfickness of the arrows fin sfignfifficant reflatfionshfips fis proportfionafl to the magnfitude of the effect (overflafid on the flfine). ***P < 0.001; **P < 0.01; *P < 0.05. The fintegrated modefl fis comprfised by those drfivers that were sfignfifficant fin partfiafl modefls. (For finterpretatfion of the references to coflor fin thfis ffigure flegend, the reader fis referred to the web versfion of thfis artficfle.)

flarge-scafle capfitaflfized farmers. Other studfies suggested that ruralf-urban mfigratfions were mafinfly the resuflt of dfirect evfictfions of poor peasants and findfigenous communfitfies from thefir flands by capfitaflfized farmers searchfing for new flands to expand thefir actfivfitfies (Caceres et afl., 2010; Cáceres, 2015; Gofldfarb and van der Haar, 2016).

The aggregated scafle of our anaflysfis does not afllow sheddfing flfight on the processes described above and others occurring at smaller scales such as the movement of rural population to other frontiers or near cities within a given district. However, the facts that a widespread decrease in rural population has taken place during the study period (Fig. SI.1), and that deforestation was higher in areas where rural population decreased (Fig. 2) and remnant forest is higher in departments with higher rural population (Fig. 3), might suggest that rural population changes in the ADC appear to be more a consequence of land-use dynamics rather than a cause. Moreover, as suggested by other studies, rural population in the ADC is mainly comprised of smallholders that generally do not convert forests (Baldi et al., 2015; Marinaro et al., 2017), and deforestation is highly driven by extra-local actors which employ little rural labor force (le Polain de Waroux et al., 2018; Mastrangelo et al., 2019). Thus, our findings, and those of previous studies, suggest that the neoliberal frontier dynamics might take place in the ADC (Brannstrom 2009; Hecht, 2005). Despite the low explanatory capacity of the socio-demographic models, the evidence suggests that Neo-Malthusian and Boserupian models are not very useful for explaining the population-deforestation nexus in the ADC as suggested by other studies (le Polain de Waroux et al., 2018; Mastrangelo and Aguiar, 2019; Sacchi and Gasparri 2016). However, more studies at higher spatial resolution are needed to better understand the relationship between socio-demographic conditions and land use change in the ADC, such as rural-urban migrations and land-use displacement by different actors.

5.1.4. Institutional

Institutional aspects related to land governance and its private appropriation and accumulation had a low influence on forest loss and moderate influence on forest maintenance in the ADC (Figs. 2 and 3). Departments with lower non-private land tenure had higher forest conversion. This suggests that more forest is maintained in departments where agricultural farms do not have defined limits by fences, which has also been recently described at a more detailed spatial scale (Marinaro et al., 2020). In the ADC, the areas with non-private land tenure represent approximately 27% of the agricultural farms (the remaining is fenced, see Fig. S1.1 for its spatial distribution) and are generally associated with peasant and indigenous communities (Goldfarb and van der Haar, 2016). In some cases, these lands are claimed by extra-regional (national and international) capitalized farmers (i.e. land grabbing, Caceres, 2015; Goldfarb and van der Haar, 2016), which often increases social unrest in rural areas (Aguiar et al., 2016). Other forms of land tenure, such as the recognition of ancestral and communal land tenure of the areas currently occupied by peasant and indigenous communities, could contribute to conserve forests and address social unrest by avoiding violent evictions. Conservation policies similar to this have proven to be one of the most effective alternatives for reducing deforestation in the Amazon (Blackman et al., 2017; Hajjar et al., 2020; Nolte et al., 2013). A recent study has suggested that these strategies have not been effective in a portion of the ADC for reducing deforestation (Ceddia, 2019). However, this study alerts that the amount of land-titled to indigenous communities was small and that titling may have induced preventive deforestation to prevent external land claims (Ceddia, 2019). Therefore, although our results suggest that the presence of indigenous communities may have positive conservation outcomes as suggested by a recent global study (Garnett et al., 2018), the recognition of their ancestral land tenure as a way to inhibit deforestation requires further inquiry in the ADC.

Regarding conservation through protected areas, the non-significance of this driver on reducing deforestation rates could reflect the low proportion of protected areas in the ADC (Brown et al., 2006) and be also related to the motivations that underlie conservation. Baldi et al. (2017) suggest that the primary motivation determining the spatial location of protected areas in South America was "opportunity", i.e. where agricultural suitability is low. Moreover, the National Forest Law that was enacted after the period that we analyzed here has the same flaws, and which effectiveness for reducing deforestation remains unclear (Aguiar et al., 2018). Areas with higher restrictions to deforestation are generally located in regions with low opportunity costs (Aguiar et al., 2018; Nolte et al., 2017b). Therefore, the expansion of protected areas through National Parks or the Forest Law in areas with high agricultural suitability could be an effective mechanism for halting deforestation as suggested by the finding of several studies (Camba Sans et al., 2018; Nolte et al., 2017b; Piquer-Rodríguez et al., 2018). This has proven to be an effective strategy in other commodity frontiers, such as the Amazon (Nolte et al., 2013), but is highly contingent on government interests and ideology (Abessa et al., 2019). Finally, our analysis suggests that the level of land concentration (i.e. Gini index) at the department scale did not have a significant effect on forest loss or maintenance. Yet, this does not necessarily suggest that the size of the landholdings is unrelated to the rates of deforestation, since other studies report that there was a positive relationship between them in the Gran Chaco and other agricultural frontiers of South America during the past 25 years (Graesser et al., 2018). However, a comprehensive analysis of the relationship between farm size, land concentration, and land cover change would require a more spatially detailed analysis for which updated cadastral information is lacking for most of the ADC. Moreover, a historical perspective on land tenure dynamics in the region is needed to better understand the differences among provinces, departments, and frontiers.

5.1.5. Integrated

Geographical patterns of forest cover change in the ADC resulted from the interaction between multiple drivers. These results reinforce and expand the theoretical implications of the partial models. As expected, precipitations had a negative significant effect on non-private land tenure, and this had a positive significant effect on remnant for-est (Fig. 3), but not on deforestation rate (Fig. 2). Departments with higher precipitation also had higher road density and, similarly to the partial infrastructure model, more roads were positively associated with higher deforestation rates and less remnant forest. Thus, the effect of precipitation on remnant forests occurred in a direct way, but also indirectly mediated by non-private land tenure and road density. These associations were partially the same for forest loss, on which non-private land-tenure did not have a significant effect (Fig. 2), and therefore non-private land tenure might not be a driver that stops deforestation. The associations among drivers might suggest that the circular causality model of agglomeration, derived from the new economic geography theory (Fujita and Krugman, 1995), could be taking place in the ADC (Garrett et al., 2013; Richards, 2018). According to this theory, agglomeration economies occur near cities where biophysical and transportation conditions are relatively superior to adjacent areas, and therefore give place to a circular causality model or positive feedback loop of agglomeration (Garrett et al., 2013). Moreover, the indirect effect of precipitation on remnant forest mediated by non-private tenure and road density suggest that the circular causality model of agglomeration might include institutional aspects besides infrastructure (Fig. 3). Thus, the favorable conditions that trigger the agglomeration feedback loop might include institutional aspects related to land tenure besides biophysical and infrastructure (Faingerch et al., 2021; Marinaro et al., 2020). Under this context, the agglomeration might be directly or indirectly promoted by private (e.g. farmers and other supply chain actors) and government actors (e.g. land "colonization" offices), where their interaction might promote land privatization (Faingerch et al., 2021) and road expansion. It is important to remark that the association between precipitation and land tenure is probably mediated by past agricultural expansion, a process that we did not include in our models. Hence, untangling these processes requires further inquiry and a long-term perspective, since they are the result of historical changes in land tenure and accessibility for which, unfortunately, open-access and good quality cadaster information, is still lacking in the ADC. Hence, further studies should assess the processes and spatial determinants that underlie land privatization. In the context of recent studies that assess the fencing and privatization of land (Aguiar et al., 2021; Marinaro et al., 2020), this would imply assessing if land with better agroclimatic accessibility conditions is privatized first and also understanding the cognitions of different actors involved in the process. A similar approach could be employed for better understanding the processes that explain the spatial distribution of road expansion and the paving of existing.

Non-private land tenure was higher in departments with a higher rural population with an important proportion of indigenous households (Figs. 2 and 3). However, the data did not support our expectation that rural population, and indigenous households, are associated to worst biophysical and accessibility conditions (Figs. 2 and 3). Thus, at the regional scale, indigenous communities are not necessarily occupying lands with low suitability for the expansion of agriculture and cattle ranching. However, within departments, particularly large ones, there can be many contrasting situations and thus, more detailed spatial analyses are needed. Overall, the absence of an indirect effect of biophysical conditions on forest loss and maintenance mediated by socio-demographic drivers suggests that the neoliberal frontier hypothesis in the ADC might be independent of the environmental conditions (Hecht, 2005; Sloan, 2007). This means that socio-demographic conditions (i.e. proportion of rural population) and dynamics (i.e. rural population change) in the ADC might not be necessarily determined by biophysical conditions as many Neo-Malthusian theories suggest (Sherbinin et al., 2007). However, the magnitude of the effect of socio-demographic drivers on forest cover change is not very strong. Therefore, our study sheds light on potential associations that require further inquiry for understanding the complex relationships between them, land cover and biophysical factors at more detailed spatial and temporal scales.

5.1.6. Multi-model comparison

The comparison of the individual effects of drivers in the partial and integrated models allows assessing their relative contribution when different sets of drivers are included. For example, the effect of precipitation on remnant forest in the integrated model (2.49, Fig. 3) is almost half of that in the biophysical model (4.69, Fig. 3), suggesting that about half of the effect of that variable might be mediated by its effects on land tenure and road density. The opposite occurs with the influence of indigenous population on remnant forests, which is higher in the full model (4.35, Fig. 3) than in the partial model (3.37, Fig. 3), reinforcing the contribution of this variable to forest maintenance. Thus, excluding correlated drivers, as most studies do, might avoid multi-collinearity but also reduces our comprehension of land use changes which are complex processes determined by the interaction of multiple factors.

Accounting for indirect effects through SEM increased the capacity to explain forest loss and maintenance in comparison to partial models (Table 2). However, as expected, there was a clear trade-off between explanatory power and simplicity of models, as models containing all statistically significant drivers (integrated models) were those with higher R^2 but also AICc. Furthermore, they also presented the highest fit to data (p-value, and Fisher's C, Table 2), whereas partial models are incomplete descriptions of the mechanisms that drive forest change in the ADC since they have missing relationships among drivers (PMP, Table 2). For both forest cover loss and maintenance, the explanatory capacity of the integrated models is not equal to the additive contribution of the partial models (Table 2). This indicates that multiple drivers interact in a non-additive fashion and that some of them influence forest loss and maintenance in the ADC in both direct and indirect ways. This study reports an explanatory capacity (range pseudo R^2 = 0.17–0.81) that is within the range of previous studies regarding the drivers of deforestation in the ADC [Gasparri et al. (2015): R^2 = 0.13–0.31; Piquer-Rodríguez et al. (2018): R^2 = (0.11–0.25); Volante et al. (2016): R^2 = (0.19–0.61)]. However, these comparisons should be interpreted with caution since the differences in spatial scale and methods (e.g. statistical method, goodness of fit estimation) preclude a comprehensive analysis.

The set of drivers that significantly affected forest loss and maintenance were generally similar, although with an opposite sign. This suggests that the drivers of forest cover change over the short term (rates of deforestation) and over the medium term (remnant forest) might be similar (Figs. 2 and 3). However, there were some notable differences. While non-private land tenure was a significant driver explaining forest maintenance, it did not have an inhibitory effect on the deforestation rate. Both models were also similar in terms of the order of partial models concerning their explanatory capacity (R^2) and balance between this and simplicity (AICc, Table 2). The infrastructure model had the lowest AICc while the biophysical had the highest explanatory capacity. This corresponds to most of the previous literature suggesting that biophysical conditions (precipitation and soil suitability) and infrastructure (roads and distance to markets) are the main direct spatial determinants of deforestation in the ADC between 2001 and 2010 (Fehlenberg et al., 2017; Gasparri et al., 2015; Hoyos et al., 2013; Piquer-Rodríguez et al., 2018; Volante et al., 2016; Zak et al., 2008). The previous analyses of drivers at a finer resolution (1 km^2, e.g. Gasparri et al., 2015; Piquer-Rodríguez et al., 2018; Volante et al., 2016) allowed for a more accurate spatial match between forest cover changes and biophysical and infrastructure factors. However, analyses at coarser resolution (e.g. department), such as ours, allow for including underlying drivers such as institutional and socio-demographic ones. Therefore, studies investigating multi-scalar drivers of land-use change should be encouraged in the ADC and other commodity frontiers. Moreover, there are other drivers, not included in our analysis, which should be explored in further studies such as land prices, and grain storage infrastructure, and also a description of some drivers (e.g. land tenure, road density) for not only the beginning of the study period but also its temporal change.

5.2. Novelty, limitations, and caveats of the analytical approach

In this study, we employed a theory-driven approach to evaluate the merits of multiple hypotheses regarding the causal mechanisms underlying forest cover change in the ADC. For this, we used Structural Equation Modelling and information theory. The explicit derivation of hypotheses from theory and previous knowledge is a way for land system science to organize knowledge and for assessing the generality or context-dependence, of middle range theories (Meyfroidt et al., 2018). The comparison of multiple hypotheses is important for understanding complex phenomena such as land-use change, which are generally the result of multiple interacting drivers. Thus, our approach might be useful for avoiding biased support for theories, and for promoting a better balance between the theoretical and empirical developments within land system science. Although our results might not be easily extrapolated to other modern commodity frontiers, further studies should explore the similitudes and differences among them regarding the causal mechanism underlying forest cover change. These multi-region studies are fundamental for better understanding and governing deforestation in a global, telecoupled world (Magliocca et al., 2018).

Structural Equation Modelling has been scarcely used in land system science for assessing the drivers of land-use change (Meyfroidt, 2016, e.g. Lang et al., 2018). Although this modelling approach allows describing the complex association among drivers, it is impossible to capture all the processes and, therefore, some confounders may exist as in other modelling approaches. The main limitations and caveats of this approach, in comparison with traditional linear modelling, are potentially higher endogeneity (i.e. the order of causal relationships could be inverse), higher model complexity, and the risk of wrongly inferring causation from correlation. Endogeneity should not be a major concern in our models, as our explanatory variables are generally chronologically ordered or clearly exogenous (e.g. precipitations, soil suitability). One specific relationship where causation might by reciprocal is the one between non-private land tenure and road density. Therefore, while explaining the relationship between these drivers, and forest cover change, we have not assumed any order of causation. Moreover, although endogeneity is an important statistical concern, the explicit derivation of the order of causation from theories and previous knowledge, might be a first step towards reducing it. Regarding complexity, our integrated models could be poorly fitted since the sample size is relatively small for models with so many parameters. To reduce the complexity of the integrated models we employed a stepwise procedure that did not exclude important drivers (Supporting information 2). Finally, concerning causation, structural equation modelling explicitly defines a direction of causation among variables, for which previous knowledge is critical, as in other approaches used for describing causation in social-ecological systems (e.g. counterfactual analysis, cointegration) which all rely on an a-priori causal model (Ferraro et al., 2019; García et al., 2020). The best strategy to tackle causation in land system science likely results from combining a multiple-working-hypothesis framework with methodological pluralism, at different spatial and temporal scales. For example, for enhancing the inferences of our study, it could be complemented with other assessments at finer spatial scales, such as counterfactual analysis and matching (Ferraro et al., 2019), and surveys and interviews with stakeholders to understand the cognitions underlying their land-use decisions (Meyfroidt, 2013). Some of these approaches and studies have already been conducted in the ADC (Mastrángelo et al., 2014; Nolte et al., 2017b).

5.3. Implications for forest conservation

The evidence obtained here provides two main contributions relevant to territorial planning and public policies in the ADC, which could also be further explored in other commodity frontier regions. First, it allows identifying areas where deforestation is expected to expand in the future constrained by biophysical and infrastructural factors. The reduced effect of precipitation on deforestation in the integrated model suggests that although some regions are particularly prone to deforestation due to their biophysical conditions, this risk would be mitigated by appropriate policies for regulating land tenure and for planning infrastructure (Laurance et al., 2014; Robinson et al., 2018). In line with previous studies (Gasparri et al., 2015; Piquer-Rodríguez et al., 2018; Volante et al., 2016), we showed that both precipitation and soil suitability have a strong and independent effect on deforestation. In recent years, the main destiny of deforested areas has been shifting from soybean cropping to pasture sowing (Gasparri et al., 2013), which is more tolerant to water stress. Hence, areas with suitable soils are prone to deforestation despite low precipitation (Houspanossian et al., 2016). As also suggested by previous studies, the important effect of road density on forest cover change suggests that the expansion of roads, and the paving of existing ones, should be planned considering their environmental consequences (Gasparri et al., 2015). Conversely, as roads continue to expand and be paved in the northern Argentine Dry Chaco, specifically in the "Impenetrable region" (western Formosa, northwestern Chaco, east Salta, northeastern Santiago del Estero), deforestation there is expected to continue since the region has biophysical conditions for the expansion of pastureland. Moreover, the Impenetrable region is one with the highest rates of rural depopulation (Fig. SI.1) and our results suggest that higher forest cover is maintained in areas with higher rural populations. Thus, the maintenance of forests on agriculturally suitable soils requires specific policies such as assigning them a higher conservation status in the ongoing upgrade of the National Forest Law (Aguiar et al., 2018), expanding protected areas in areas with agricultural suitability and corridors among pre-existing, and designing and implementing public policies that increase rural entrenchment and promote economic activities that balance production and conservation. These issues are currently discussed in government offices (e.g. Monaco et al., 2020), and some of them, such as expanding protected areas and promoting sustainable ranching are taking place (e.g. Tschopp et al., 2020).

Second, our study allows identifying socio-demographic and institutional conditions compatible with the maintenance of forest cover, which can be fostered and enhanced to promote long-term forest maintenance. The positive effect of the lack of land use privatization, and of the presence of indigenous communities, on remnant forest in the integrated model, suggests that the use of land by peasant and indigenous families under non-private land tenure is associated with forest maintenance. This association arises from the fact that in the ADC, peasant and indigenous land-use systems depend on goods and services provided by native forests (e.g. forage, timber, charcoal), and thus do not usually clear the forest for their livelihoods (Altrichter and Basurto, 2008; Baldi et al., 2015; Marinaro et al., 2017). However, in some cases, this may be related to financial capital limitations rather than to intrinsic motivations for maintaining forests for their livelihoods and culture. These associations suggest that under the current livelihoods and productive activities of peasant and indigenous communities, the relationship between forest maintenance and rural population might be reciprocal (Altrichter and Basurto, 2008; Baldi et al., 2015; Marinaro et al., 2017). Overall, these insights suggest that policies supporting rural to urban migrations to relieve pressure on forests in the ADC may fail to be effective or even be counter-productive for forest maintenance. However, the maintenance of forest cover associated with non-private land tenure may be fragile because large farmers and land investors tend to grab lands with insecure tenure and dispossess less powerful actors (Caceres, 2015; Goldfarb and van der Haar, 2016). Therefore, for peasant and indigenous families to become enduring stewards of the forests, and ensure their permanence in rural areas, land-use policies should empower them by protecting them from land grabbing, and therefore securing their access to land and their livelihoods (Blackman et al., 2017; Brondizio and Le Tourneau, 2016; Piquer-Rodríguez et al., 2018; Robinson et al., 2014). Thus, for increasing the effectiveness and legitimacy of the National Forest Law, its upgrade should explicitly account for social conflicts related to land tenure (Seghezzo et al., 2017). Within this context, recent upgrades in the National Forest Law have started to include the social perception of indigenous and peasant communities regarding forest zoning schemes, although the legitimacy of this process has not been assessed. Moreover, although most of these communities have insecure land tenure, they are still eligible to access payment for ecosystem services for forest conservation (Aguiar et al., 2018). In parallel to public policies, the political organization of communities has been suggested to be a driver that halts deforestation (Aguiar et al., 2016), and therefore, it could be an alternative pathway for reducing deforestation that is not led by the government. Finally, since the rural population in the ADC not only has insecure land tenure but also high levels of poverty (Paolasso et al., 2012), integrated public policies oriented towards increasing their quality of life (e.g. sanitation, health, education) are critical and urgent.

6. Conclusion

To our knowledge, this is the first study that uses a structural multimodel approach for comparing alternative theoretical explanations of the processes driving forest loss and maintenance in a global deforestation hotspot. During 2001–2010, forest conversion in the ADC resulted from the interaction of multiple drivers operating at different spatial scales in the ADC. Our results suggest that at the regional scale, the spatial distribution of forest conversion was explained mainly by precipitation, soil suitability for agriculture, and accessibility, whereas forest cover was maintained in areas with a higher rural population generally comprised of indigenous and peasant communities lacking land titles. Our findings support the notion of agricultural adjustment since areas with better biophysical conditions had higher forest conversion. Moreover, in these areas with better environmental conditions, we also found higher road density and land privatization, which suggest that the circular causality model of economic agglomeration is taking place, and that besides infrastructure and biophysical drivers it may also include institutional aspects related to land privatization. However, some of these effects were not very strong and statistically significant for both forest conversion and maintenance. Therefore some of these processes require further inquiry. Finally, our study supports the neoliberal frontiers hypothesis, since in the ADC, changes in rural demography appear to be more a consequence than a cause of forest cover dynamics, as areas with higher rural depopulation had higher deforestation. These findings might be useful for enhancing the effectiveness and equity of the National Forest Law. A more widespread use of structural models and, more broadly, causal diagrams in land system science could contribute to a better understanding of the complex interactions, moderations and mediating effects among direct and indirect drivers of land system changes.

Acknowledgments

We thank Marcela Román and Martín Aguiar for providing valuable comments on a previous version of this manuscript. Jim Grace, Jonathan Lefcheck, and Bill Shipley provided assistance with statistical modelling. This work was carried out with the aid of grants from ANPCyT and the Inter-American Institute for Global Change Research (IAI) CRN3095 (Bridging Ecosystem Services and Territorial Planning (BEST-P)): A southern South American initiative, which is supported by the US National Science Foundation (Grant GEO-1128040). CONICET and UBA also provided funding through undergraduate and graduate scholarships to S.A. All authors declare no conflict of interest.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.landusepol.2021.105806.

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