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Tradeoffs between economic and ecosystem services in Argentina during 50 years of land-use change

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ABSTRACT

Farmland expanded quickly at the expense of natural lands in Argentina since the mid-1950's without consideration of ecological costs. In order to analyze the implications of such changes we aimed to (i) develop a simple biophysical model to estimate the relative (0–100) provision of ecosystem services, (ii) calculate the economic value of food and fiber production derived from farming activities (economic services), and (iii) assess the tradeoffs between the provision of ecosystem and economic services. Land-use/land cover changes were studied through data from agricultural censuses in three historical periods (1956–1960, 1986–1990 and 2001–2005). The model uses biophysical data about biomass, water coverage, slope, soil infiltration capacity, temperature, precipitation and altitude. After testing the consistency of the model, its results were used to assess the relative ecological value of main regions across the country. On the other hand, the annual gross margin per hectare of farming activities was estimated in order to compare 1956–1960 and 2001–2005 periods that greatly differ in their regional farming model. Results showed that different regions respond differently to human intervention, both in economic and ecological terms, and any attempt to apply sole and centralized land-use strategies to different biomes may lead to undesirable outcomes. Economic and ecological criteria should be regionally balanced as a pre-requisite to the application of land-use policies.

1. Introduction

Land-use policies in rural areas can be driven by either utilitarian or moral principles. While utilitarian views, in general, seek to maximize economic benefit, the moral ones look at preserving social and environmental values (Goulder and Kennedy, 1997; Kremen and Ostfeld, 2005). Due to the increasing pressure on land, policy- and decision-makers need to monitor land-use change and how this change affects society needs. Policies and decisions can either mitigate or aggravate land-use conflicts, and this in turn may have social, economic and environmental consequences. The cultivation of new lands produces economic benefits in the short term, but it can neither compensate nor justify the loss of irreplaceable ecological services. Under conflicting circumstances, the consideration of tradeoffs between socio-economic benefits and ecological costs is inevitable to strengthen the decision-making processes.

To shed light on this, Viglizzo and Frank (2006) proposed a methodological approach to assess land-use options through tradeoffs analysis involving both ecosystem and economic services provision. Taking into account critical biomes of the Southern Cone of South America, these authors concluded that due to its low capacity to supply ecosystem services, the environmental cost of cultivation in the Argentine Pampas, for example, seems to be of minor importance in relation to its potential impact on some Brazilian forests that are strong suppliers of eco-services. So, they suggested that the functional complementation of biomes seems to be a smart strategy to explore land-use options on broad scale, regional and temporal basis. Although the value of economic goods and services are reliably established by markets, the value of many goods and services provided by nature are not captured by the market and remains unclear (Balmford et al., 2002). Therefore, exercises on tradeoffs analysis between economic and ecological values have failed because current methods to price nature are still imperfect.

Early references to the economic value of ecosystem services date back to the 1960s and 70s (King, 1966; Helliwell, 1969; Odum, 1971; Odum and Odum, 1972), but beyond such efforts, various pricing attempts to value natural services were not effective (Odum, 1973). Only those natural goods and services that are exchanged for money (such as food, fiber, bio-energy, wood and water) are accepted by the neoclassical economy, but those that are intangible in monetary terms (like soil protection, disturbance control, water purification or habitat provision) are set

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aside, even in demonstrable cases of irreversible loss. In order to value ecological intangibles at a global scale, Costanza et al. (1997) published a synthesis of more than 100 studies that tried to price ecosystem services by applying a variety of economic methods that included hedonic pricing, contingent value and replacement cost. They derived average values per hectare for each of 17 ecosystem services across 16 biomes, which were extrapolated to the whole planet by multiplying by each biome area. However, such global figures were in turn criticized by classical economists like Ropke (2004), who argued that average values ignore microeconomic implications.

Criticism also came from ecologists that stated that monetary valuations may say nothing if money is not related to ecosystem functions tightly associated with service provision (Freeman, 1993). They argued that ecosystems have an intrinsic biophysical value that goes beyond their circumstantial price, and thus values may vary largely from one place or time to another. For example, forests may have high price for rich people that are willing to pay for aesthetical attributes, but aesthetics may have no value to people that depend on forests for biological survival. As expected, many philosophical debates (Goulder and Kennedy, 1997; Sagoff, 1997; Turner et al., 2001; NRC, 2004) grew around the non-economic or intrinsic value of ecosystem services. To deal with this, some biophysical approaches were suggested to face the challenge of non-economic valuations and the need of more objective and consistent valuations. For example, Odum (1988, 1996) and Odum and Odum (2000) introduced the notion of solar energy as a common currency for ecosystems valuation. Costanza et al. (1998) found high correlation between their monetary estimations and the Net Primary Productivity ($g\,m^{-2}\,yr^{-1}$) of terrestrial and marine biomes. Beyond critics, arguments have increased around the idea that biophysical measures are necessary to objectively represent the aggregated value of intangible ecosystem services.

Ecosystem service tradeoffs occur when the provision of one service is enhanced at the cost of diminishing the provision of another one (e.g. actions to enhance the supply of food and timber have declined the nutrient cycling, flood regulation and opportunities for recreation) (Balvanera et al., 2001; Rodríguez et al., 2006). In general, ecosystem services tradeoffs derive from intentionally human management choices, but in some cases, they just happen without premeditation or even awareness that they are taking place (Rodríguez et al., 2006). Raudsepp-Hearne et al. (2010) developed a framework for analyzing the provision of multiple ecosystem services across a peri-urban agricultural landscape and they identified tradeoffs between provisioning and almost all regulating and cultural ecosystem services. Also they showed that a greater diversity of ecosystem services is positively correlated with the provision of regulating ecosystem services.

Considering the conflict between ecological and economic valuations in response to land-use and land-cover change during the 1956–2005 period, in this work we aimed to (i) develop a simple biophysical model to estimate the relative (0–100) provision of ecosystem services, (ii) estimate the annual gross margin (GM) ha^{-1} of farming activities in different regions, and (iii) assess the tradeoffs between the provision of ecosystem and economic services and discuss their implications on land-use policies. The ecosystem services involved in the present study were: (i) soil protection, (ii) production, (iii) water purification, (iv) water provision capacity of land, (v) water provision capacity of wetlands, (vi) disturbance regulation and, (vii) habitat provision. For the tradeoffs analysis we have compared 1956–1960 and 2001–2005 periods that greatly differ in their regional farming model. While the 1956–1960 period represents the traditional extensive agricultural model from Argentina, with low input, rudimentary technology and low productivity schemes; the 2001–2005 period shows a tech model of grain production that has expanded rapidly in the country, with a moderate to high use of inputs, improved agronomic practices and high productivity schemes.

2. Materials and methods

2.1. Area and time span of this study

The study involved two steps: the first looked at assessing the provision of ecosystem services across Argentina; the second, at assessing the tradeoffs between the provision of ecosystem and economic services during 50 years of land-use/land-cover change. To evaluate geographical patterns and gradients of ecosystem services supply, the whole-country was covered during by the first step involving all main regions and dominant biomes (Fig. 1). The second step focused on an area of more than 1.47 million km^2 currently subjected to farming expansion that represents about 63% of the continental area of Argentina. A tradeoffs analysis was undertaken on this changing area to study synergies and conflicts between ecological and economic interests.

The last step comprised fifteen eco-regions that included the Rolling Pampas, Central Pampas, Southern Pampas, Semi-arid Pampas, Flooding Pampas and Mesopotamian Pampas, the Humid-Sub-humid Chaco, Central Sub-humid Chaco, Dry Chaco and Western Sub-humid Chaco, the Espinal, the Atlantic Forest, the Iberá Marshes, the Paraná Delta and the Yungas (Naumann and Madariaga, 2003; Brown et al., 2006) and a wide variety of land covers: cultivated plains, grazing lands and temperate and sub-tropical shrublands, tropical and sub-tropical forests, mountain ecosystems, deserts, rivers and creeks, water bodies and wetlands.

Besides its ecological functions, forests in Argentina are also used for several economic purposes. Yungas, Chacos, Espinal and Atlantic forests provide abundant and varied wood resources (timber, tannin extraction, by-products) and, today, there is an emerging tourism activity associated with these forests. Much of the wood supplies to the largest cities in Argentina during the first century of its existence and much of the wood that Argentina supplied to Europe during the World Wars came from these forests (Brown, 2009). However, for the purposes of this paper, we only considered the livestock production associated to these forests.

The analysis was based on few but dominant farming activities like soybean, sunflower, maize, wheat and beef (Obschatko et al., 2006). The investigation comprised three sub-periods (1956–1960, 1986–1990 and 2001–2005) which represent, respectively, the transition from an extensive, low-input farming in the 50s and 60s to a more intensive and productive one that relied on the increasing use of inputs and the application of modern agronomic practices, especially since the 90s.

2.2. Data and information sources

Records on land-use and land-cover from 657 administrative districts from national agricultural censuses (INDEC, 1964, 1991, 2004) and agricultural surveys (SAGPyA, 2009) were used to estimate the regional provision of ecosystem services across the country. They provided data on croplands, natural grasslands and natural woodlands. Only rainfed-farming lands were considered in our estimations because irrigation covered less than 0.5% of lands in the country.

Data on biomass were used to calculate ecosystems service supply in different climate zones of Argentina. Estimates of biomass for forests were obtained from reports of various authors (Gasparri and Menéndez, 2004; MERNRyT, 2004; Grau et al., 2005; Boletta et al., 2006; SAyDS, 2007a,b; Gasparri et al., 2008) and figures were later cross-checked with default data provided by IPCC (2006). In the case of crops, grasslands and pastures, we also used IPCC

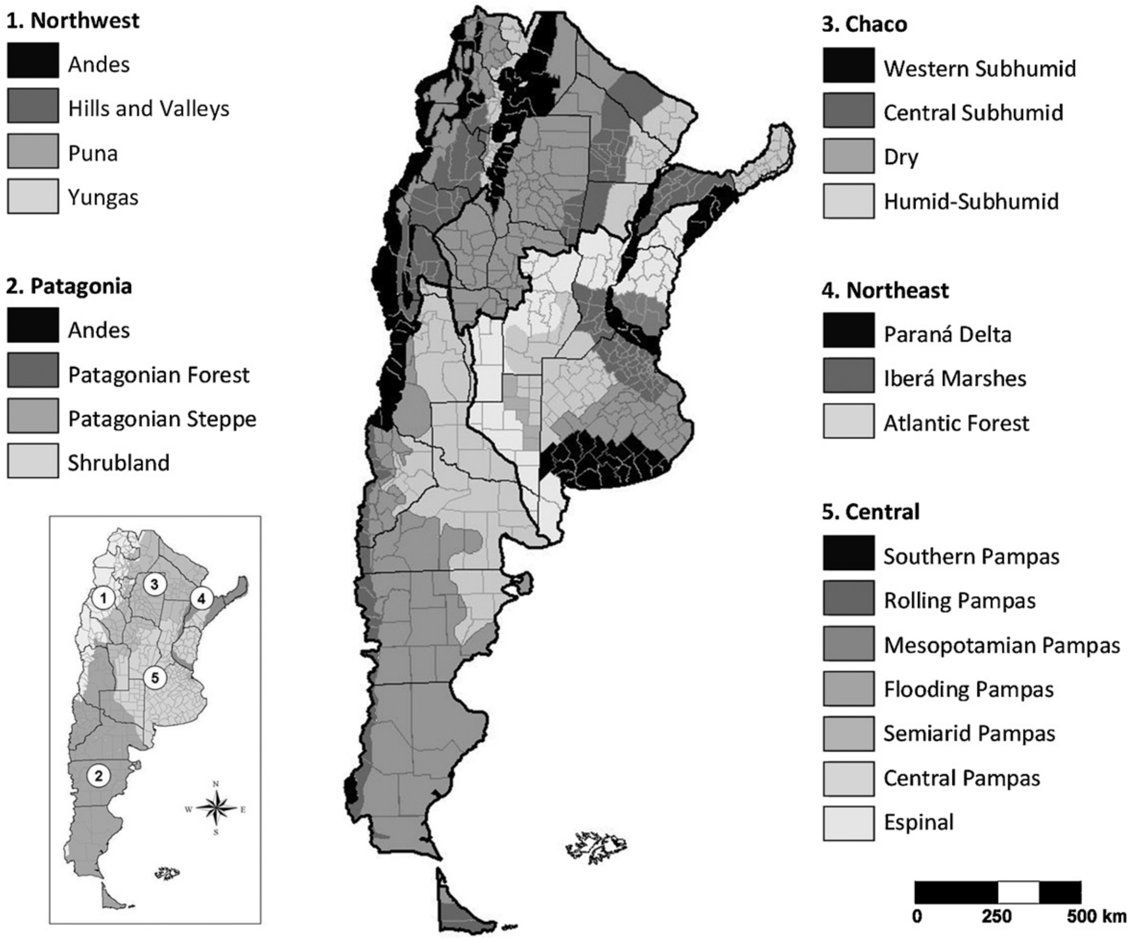

Fig. 1. Location of study eco-regions in the Argentine territory.

(2006) default values for tropical–subtropical and temperate environments and for high and low precipitation conditions for each biome. Data on water bodies were estimated by difference with land-use/-cover records from the same sources.

The most extensively used approaches to assess biomass are based on satellite data, which provide vegetation indexes calculated from the reflectance in different wavelengths of the electromagnetic spectrum. The seasonal variability of vegetation indices is represented by two attributes: the annual mean of the seasonal curve and its seasonal coefficient of variation (Paruelo et al., 1998). Given that satellite data was not used at our low resolution, broad-scale study (see Section 2.2), data variability (VC_B) was empirically estimated through figures on NPP and evapotranspiration from various literature sources (Paruelo and Sala, 1995; Paruelo et al., 1998, 2001; Nosetto et al., 2005; Alcaraz-Segura et al., 2006; Gasparri et al., 2008; Volante et al., 2010, "personal communication").

2.3. Modeling the biophysical provision of ecosystem services

Following a classification by de Groot et al. (2002), it was assumed that the main ecosystem services (soil protection, production, purification and water supply, provision of habitat and shelter) were directly associated with the amount of biomass. Besides, other biophysical factors (slope, water bodies' coverage, precipitation, etc.) were also assumed to be associated with services provision (see each service calculation for details). These biophysical factors ranged between 0 and 1 (Table 1). The more factors are used; values degrade and lose relative weight. Therefore, compensation coefficients that depend on the number of multiplicative factors were applied to maintain the balance of services within the equation.

The combination of these variables allowed a simple but causal estimation of services provision through the following equations:

2.3.1. Soil protection (S_{PRO})

$$S_{PRO} = B \times (1 - VC_B) \times (1 - F_{SLO}) \quad (1)$$

The availability of biomass (B) and its stability over time ($1 - VC_B$) are the main factors of soil protection against erosion (Lal, 1994; Pimentel et al., 1995; Costanza et al., 1998). Therefore, it was assumed that a crop that covers the ground only during six months of the year can offer fewer services than an evergreen forest, and this in turn provides more services than a deciduous one. The protective effect is highest on flat surfaces, and declines exponentially on increasingly sloped ones (Sidle et al., 2006). Thus, the steeper the slope (F_{SLO}), the more important is biomass coverage for soil protection (Eq. (1)).

2.3.2. Production (S_{PRD})

$$S_{PRD} = B \times (1 - VC_B) \times (1 - WB) \quad (2)$$

We assumed that production (valuable material per hectare) was directly associated with biomass (B) and biomass stability ($1 - VC_B$) (Odum, 1971; Odum and Odum, 2000), and decreased with increasing water-body coverage of lands (WB), since these areas are not usually used for production (Eq. (2)).

Table 1
Example of information layers to orientate the application of correction factors to equations in the biophysical model.

Annual Precipitation (mm)	Annual mean temperature (°C)	Soil texture[a]	Altitude above sea level (m)	Terrain slope (%)	Correction factor (0–1)
0–125	<0.0	Clay to clay loam	>5001	>16.1	0.00–0.20
126–275	0.1–5.0	Clay loam to loam	5000–3001	16.0–8.1	0.21–0.40
276–475	5.1–10.0	Loam to silt loam	3000–1501	8.0–6.1	0.41–0.60
476–975	10.1–15.0	Loam to sandy loam	1500–701	6.0–4.1	0.61–0.80
>976	>15.1	Loamy sand to sand	<700	<4.0	0.81–1.00

[a]Texture was used to estimate soil infiltration capacity.

2.3.3. Water-purification and water- provision capacity of land (S_{WPL}) and wetlands (S_{WPW})

$$S_{\mathrm{WPL}} = B \times (1 - \mathrm{VC}_B) \times F_{\mathrm{SIC}} \times F_{\mathrm{PRE}} \times (1 - F_{\mathrm{SLO}}) \tag{3}$$

In dry areas, we assumed that the soil capacity for water purification and provision in a given landscape depends on its ability to process the flows of water entering the system (F_{PRE}) (Norberg, 1999). The greater the biomass (B) the greater capacity of the landscape to intercept, retain and infiltrate the incoming water (Postel and Thompson, 2005). However, this attribute declines with the slope (F_{SLO}) because it reduces the residence time of water within the landscape (Carreño and Viglizzo, 2007). In our case, we also related the infiltration capacity of soil (F_{SIC}) to soil texture (Eq. (3)).

In wetlands, we proposed the following equation:

$$S_{\mathrm{WPW}} = \mathrm{WB} \times (B \times (1 - \mathrm{VC}_B)) \times F_{\mathrm{PRE}} \tag{4}$$

The ability of wetlands for water purification and provision is directly associated with the relative size (WB) of the sedimentation area (Adamus et al., 1991; Verhoeven et al., 2006). According to this, the effective size of a wetland is associated with its functions, hence to its ES provision. These attribute was later confirmed by Keddy et al. (2009) and Orúe et al. (2011), who argued that water regulation, water purification and waste treatments become more important in areas with a greater water flows and large water bodies. S_{WPW} was also related to the capacity of biomass (B) to recycle nutrients from water (Phipps and Crumpton, 1994), and the amount of runoff water (estimated from F_{PPT}) that pour out on water bodies after removing pollutants from the surrounding landscape (Eq. (4)).

2.3.4. Disturbance regulation (S_{REG})

$$S_{\mathrm{REG}} = \mathrm{WB} \times F_{\mathrm{PRE}} \tag{5}$$

Disturbance associated with regulation of water that enters the landscape (e.g. floods) was related to the capacity of water bodies (WB) to expand and absorb the impact of water excess (Pearlsell and Mulamoottil, 1996; Reinelt et al., 1998; Verhoeven et al., 2006). This capacity becomes more important in areas where precipitation (F_{PRE}) (and hence, runoff) is high (Eq. (5)).

2.3.5. Habitat provision (S_{HAB})

$$S_{\mathrm{HAB}} = B \times (1 - \mathrm{VC}_B) \times F_{\mathrm{PRE}} \times F_{\mathrm{TEM}} \times (1 - F_{\mathrm{ALT}}) \tag{6}$$

We assumed that the capacity of the habitat to sustain biodiversity depends basically on biomass stock (B) and biomass stability ($1 - \mathrm{VC}_B$) (Huston, 1997; Costanza et al., 2007) and also water availability (F_{PRE}) (Brown, 1985; Adamus, 2003). On other hand, we also assumed that species richness is directly related to favorable environmental conditions of temperature (F_{TEM}) (Costanza et al., 2007) and inversely related to altitude over sea level (F_{ALT}) (MA, 2005). Thus, the capacity of one biome to provide habitat is high when biomass and water are abundant, temperature is moderate to high, and altitude over sea level is low (Eq. (6)).

2.3.6. Total ecosystem services provision (S_{TOT})

So the relative value of total services provided by the study ecosystem results from the average of the individual services. Estimates were displayed on a scale that ranges from 0 to 100 or, in other terms, between a zero provision of ecosystem services and a maximum provision (the 100 corresponds to the maximum provision calculated in this study). Thus, the relative value of each biome was expressed as a percentage of the maximum value (100) attained by the Atlantic Forest.

2.4. Model evaluation

We compared the regional supply of ecosystem services in different biomes of Argentina through two different methods of estimation: the economic one from Costanza et al. (1997) on the one hand, and the biophysical one proposed here on the other. This was done as an attempt to evaluate the consistency of both models through their relationship with generic values of biodiversity (number of animal families) reported by international literature (MA, 2005) for different biomes across the planet.

First, the biodiversity information came from the report of the Millennium Ecosystem Assessment (2005): "Ecosystems and Human Well-being: Biodiversity Synthesis". We had previously identified the different biomes involved in the study area. Then, we used the information derived from Figure 1.2 (MA, 2005; Section 2, p. 23), which identifies the number of families of amphibians, reptiles, mammals and birds of 14 different type of biomes. These types of biomes were derived from the WWF terrestrial biome classification, based on WWF terrestrial ecoregions.

Second, regarding the economic value of ecosystem services, we used data derived from Table 2 in Costanza et al. (1997). The authors estimated the economic global value of 17 ecosystem services in 16 different biomes around the world (USD 1994 $\mathrm{ha}^{-1}\,\mathrm{yr}^{-1}$). Once we had identified the area corresponding to each type of biome (ha), we multiplied this by the total value per ha (USD $\mathrm{ha}^{-1}\,\mathrm{yr}^{-1}$), so we had the total economic value per year for each type of biome. Records on land-use and land-cover were obtained from national agricultural censuses (INDEC, 1964, 1991, 2004) and agricultural surveys (SAGPyA, 2009) as we had previously explained in Section 2.2.

Our assumption for this approach was that the number of higher families (reptiles, mammals, birds and amphibians) is larger in biomes that provide larger amounts of goods and services related to organic carbon, water and shelter. The comparison between both models was done through correlation analysis.

2.5. Tradeoffs analysis

A tradeoffs analysis (Stoorvogel and Antle, 2001) was undertaken to assess changes in the provision of ecosystem services along 50 years of land use/land cover due to the expansion of the agricultural boundary in Argentina. The analysis aimed at comparing absolute and relative (%) changes in ecosystem and economic services provision at the end (2001–2005) in relation to the start (1956–1960) of the study period.

Economic services were estimated by averaging the annual gross margin (GM) (USD $\mathrm{ha}^{-1}\,\mathrm{yr}^{-1}$) of predominant farming activities (maize, wheat, sunflower, soybean and meat) during the

2001–2005 period. GM data were reported by two agro-economic monthly publications (*Márgenes Económicos* and *Agromercado*) for various regions of Argentina. GM values for different eco-regions during the 2001–2005 period ranged between 136.0 and 177.4 USD ha^{-1} in croplands, and between 45.0 and 118.9 USD ha^{-1} in grazing lands. Due to the lack of GM estimates for 1956–1960, these values were reconstructed through correction factors that reduced GM estimations based on historical records of price and yield of each analyzed product in 1956–1960 regarding 2001–2005. Thus, in all eco-regions, GM changes were explained by the expansion of croplands at the expense of natural lands, and by technology incorporation. Estimated percentages of land cover for maize, wheat, sunflower, and meat during 1956–1960 were, respectively, 20%, 44%, 46% and 37% of those reported for the period 2001–2005. Given that soybean became economically important only at the end of the 1970's, calculations took into account a shorter statistical period (1980–2005).

3. Results and discussion

3.1. Model contrasting and evaluation

The purpose beyond our relative quantification of ES is supporting land-use strategies for different areas that differ in their capacity to provide ES. Given the increasing demand for food, fiber, bioenergy and raw materials, the provision of these services should be concentrated on the best endowed regions to supply them, such as the Pampas. On the other hand, conservation effort should be prioritized on biomes that are not well-endowed to produce provision services but are major providers of regulations services, such as those of the Yungas, the Atlantic Forest and the Iberá Marshes.

As it can be inferred from the equations, NPP (as a proxy for biomass) makes the major contribution to the total ES provision value. According to the number of times it appears in the equations, and its relative weight in them, around 80% of S_{TOT} can be explained by NPP. This fact is in accordance to studies which state that ES provision is directly associated with NPP (Laurance, 2008; Nepstad et al., 2008), and inversely associated with its variability across time.

Therefore, according to various empirical evidences, NNP can be viewed as a flow that maintains the stock of biomass that generates ecosystem services. As Richmond et al. (2008) stated NNP is positively correlated with the flow of many provisioning and regulating services. In general, landscapes with high NPP generate more food, timber, or fiber than less productive landscapes and are positively correlated with the fraction of water supply that is generated by transpiration. According to Imhoff et al. (2004), NPP can be measured in units of elemental carbon and represents the primary food energy source for the world's ecosystems. So, human appropriation of NPP alters the atmosphere dynamics, the energy flows within the food webs, the biodiversity density and the provision of important ecosystem services. They showed that NNP varies spatially and temporally from almost zero to many times the local primary production, thus affecting the provision of essential ES both in space and time.

Gaston (2000) has also demonstrated that the global distribution of biodiversity and the services it provides, such as the availability of genetic resources and biological chemicals, generally increases with net primary production. Likewise, using multiple regression analysis at the site and ecoregion scales in North America, Costanza et al. (2007) estimated relationships between biodiversity (using plant species richness as a proxy) and NPP (as a proxy for ecosystem services). They assumed that biodiversity richness is closely associated with the provision of ES. At the site scale, they found that 57% of the variation in NPP was correlated with variation in biodiversity after effects of temperature and precipitation were accounted for. At the ecoregion scale, 3 temperature ranges were found to be important. While at low temperatures biodiversity was negatively correlated with NPP, at mid-temperatures there was no correlation. But on the other hand, at high temperatures biodiversity was positively correlated with NPP, accounting for approximately 26% of the variation in NPP after discounting the effects of temperature and precipitation. Showing the importance of NNP for ES valuation, they tentatively concluded that a 1% change in biodiversity in the high temperature range (which includes most of the world's biodiversity) corresponds to approximately a 0.5% change in the value of ES. Assuming that ES are spatially correlated with NNP, Ingraham and Foster (2008) have also used NPP as a parameter to map ES provision to reconstruct the boundaries of biodiversity refuge sites in US.

In Fig. 2 we compared the map that resulted from applying our biophysical model with another one elaborated by Carreño and Viglizzo (2007) from economic data provided by Costanza et al. (1997). While maps agreed in identifying the areas of high service supply, differences were evident regarding their relative values. Contrasts among biomes were higher in the Costanza-derived map than in our bio-physical one, especially in areas where wetlands and tropical forests predominate.

When models were compared through a bar chart, our model tended to smooth the difference among biomes. Furthermore, biomes are ordered differently in response to service valuation. For example, as wetlands in Iberá Marshes ranked first in the economic model, such position was occupied by the tropical Atlantic Forest in our biophysical model (Fig. 3).

According to Oreskes et al. (1994), the verification and validation of numerical models of natural systems is impossible. They argued at least two reasons for that: (a) natural systems are never closed and, (b) model results are always non-unique. Although models can be confirmed by the demonstration of agreement between observation and prediction, this confirmation is inherently partial. The authors argued that models can only be "evaluated" in relative terms, and their predictive value is always open to question. Following the rationale behind this discussion, we decided to use the term "model evaluation" in the following sections.

The evaluation through correlation with global biodiversity (MA, 2005) showed a different behavior between both models (Fig. 4). We found a positive and significant correlation coefficients for the economic ($R = 0.44$, $P < 0.05$) and biophysical ($R = 0.72$, $P < 0.01$) model. These results are not surprising since NPP-biodiversity relationship has already been widely reported in literature (Tilman, 1999; Spehn et al., 2005; Costanza et al., 2007; Luck, 2007), and NPP is the variable with the most weight in our model. However, because of the better correlation showed by the biophysical model, this comparison suggests that humans and wildlife species do not show a similar perception of the intrinsic value of natural services. Since we assumed that our biophysical model estimates acceptably well the intangible provision of ecosystem services, we decided to use this biophysical model to proceed with our analysis.

3.2. Tradeoffs between ecosystem and economic services

A set of 50-year trends in ecosystem service provision was displayed in Fig. 5 for different eco-regions and sub-regions. In agreement with previous estimations by Viglizzo and Frank (2006) for Del Plata Basin in South America, a large contrast among eco-regions was appreciated. Provisioning trends over 50 years of land use/land cover change are also presented in Fig. 5. The Atlantic Forest, Yungas, Iberá Marshes and Paraná Delta are, according to our results, the eco-regions that show, regarding the rest of biomes, the higher capacity to deliver services. We infer that they also are

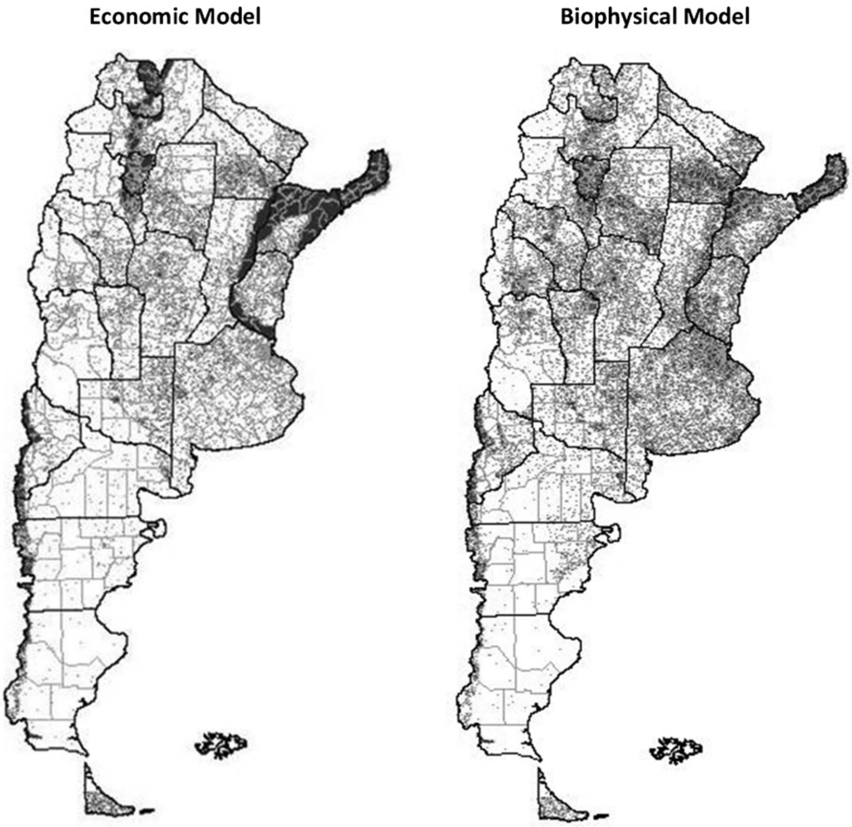

Fig. 2. Maps showing the regional supply of ecosystem services in Argentina estimated from statistical data on land-use/land-cover through two alternative procedures: economic estimations from monetary values provided by Costanza et al. (1997) where 1 dot = 10 USD ha^{-1} yr^{-1} and relative values calculated through our biophysical model, where 1 dot = 0.25 relative units ha^{-1}.

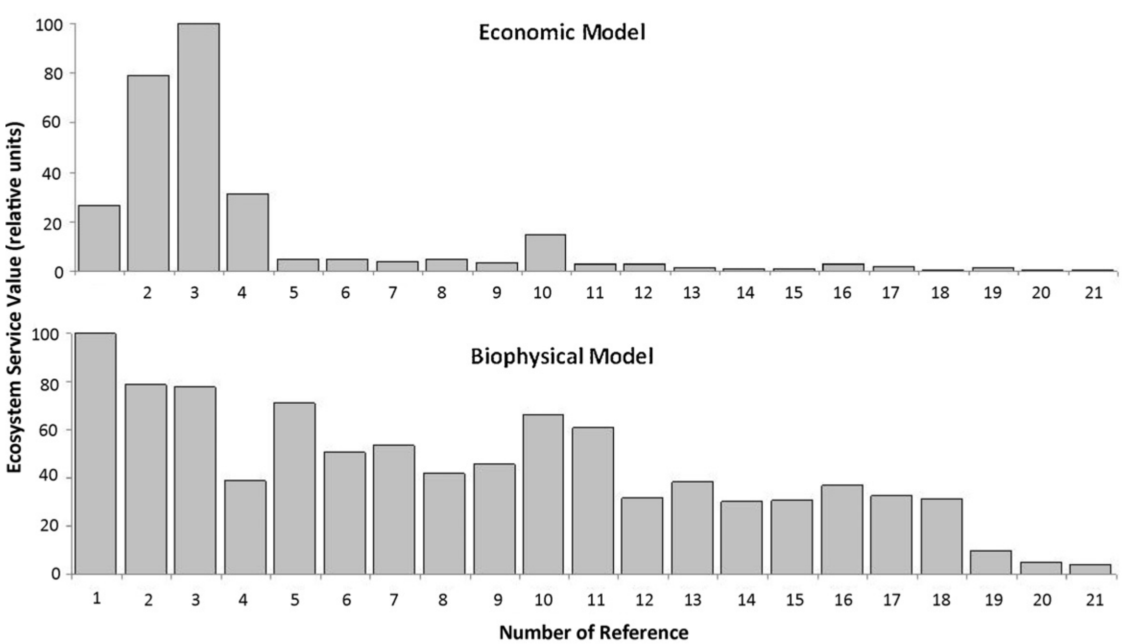

Fig. 3. Regional comparison of two models to estimate the subregional provision of ecosystem services in Argentina: (i) economic estimations (USD 1994 ha^{-1} yr^{-1}) from monetary values by Costanza et al. (1997), and (ii) relative values ha^{-1} calculated through our biophysical model. References: (1) Atlantic Forest; (2) Paraná Delta; (3) Iberá Marshes; (4) Yungas; (5) Humid-Subhumid Chaco; (6) Western Subhumid Chaco; (7) Central Subhumid Chaco; (8) Espinal; (9) Dry Chaco; (10) Patagonian Forest; (11) Mesopotamian Pampas; (12) Central Pampas; (13) Southern Pampas; (14) Rolling Pampas; (15) Shrubland; (16) Semiarid Pampas; (17) Flooding Pampas; (18) Hills and Valleys; (19) Puna; (20) Andes; (21) Patagonian Steppe.

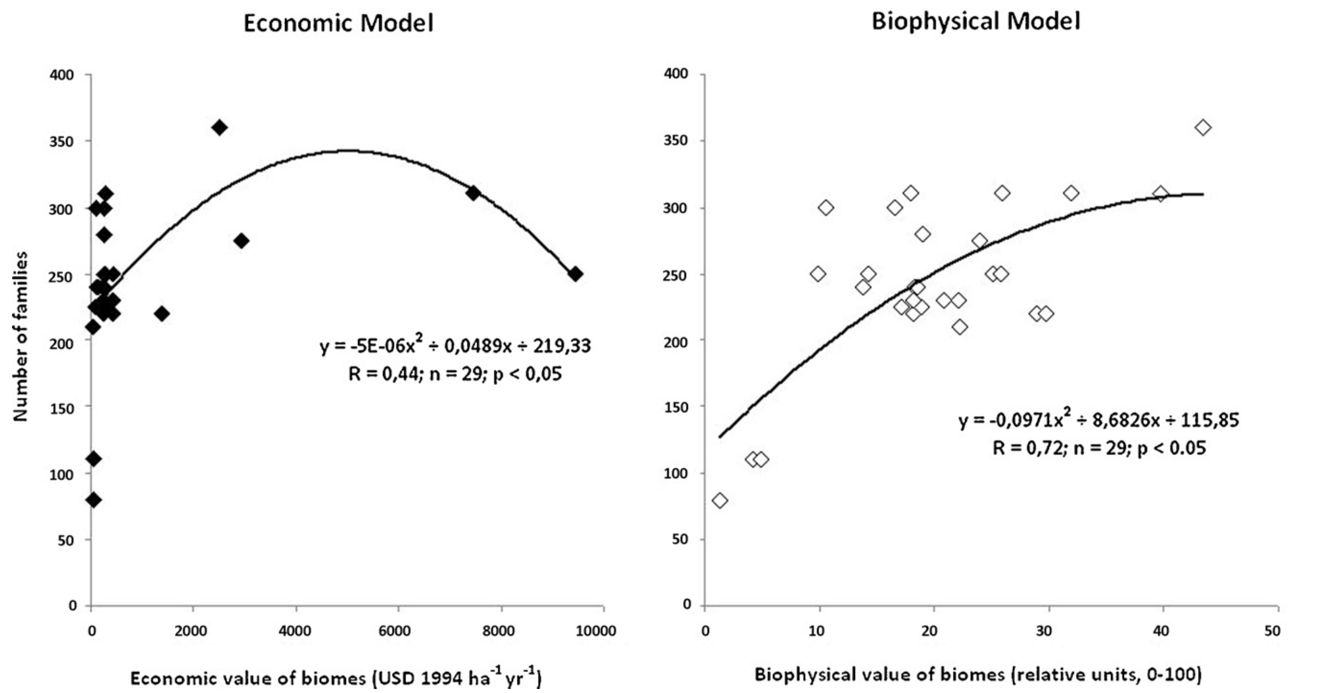

Fig. 4. Evaluation of ecosystem services values estimated through the economic and the biophysical model. Values from both models were correlated with biodiversity data provided by MA (2005) on the number of families of reptiles, mammals, birds and amphibians that are hypothetically hosted in dominant biomes.

more vulnerable because they potentially are exposed to lose larger amounts of essential services. Such are the cases of the Atlantic Forest and Yungas, which would have already experienced the largest loss during the last half 50 years. It is noticeable the relatively low capacity of the Argentine Pampas to provide ecosystem services compared to its capacity to produce food and economic income. Not surprisingly, desert areas such as those of Patagonian Steppe and Puna provide services in very short supply.

Fig. 6 shows the tradeoffs between ecosystem services provision and economic income of the studied districts during the 1956–1960

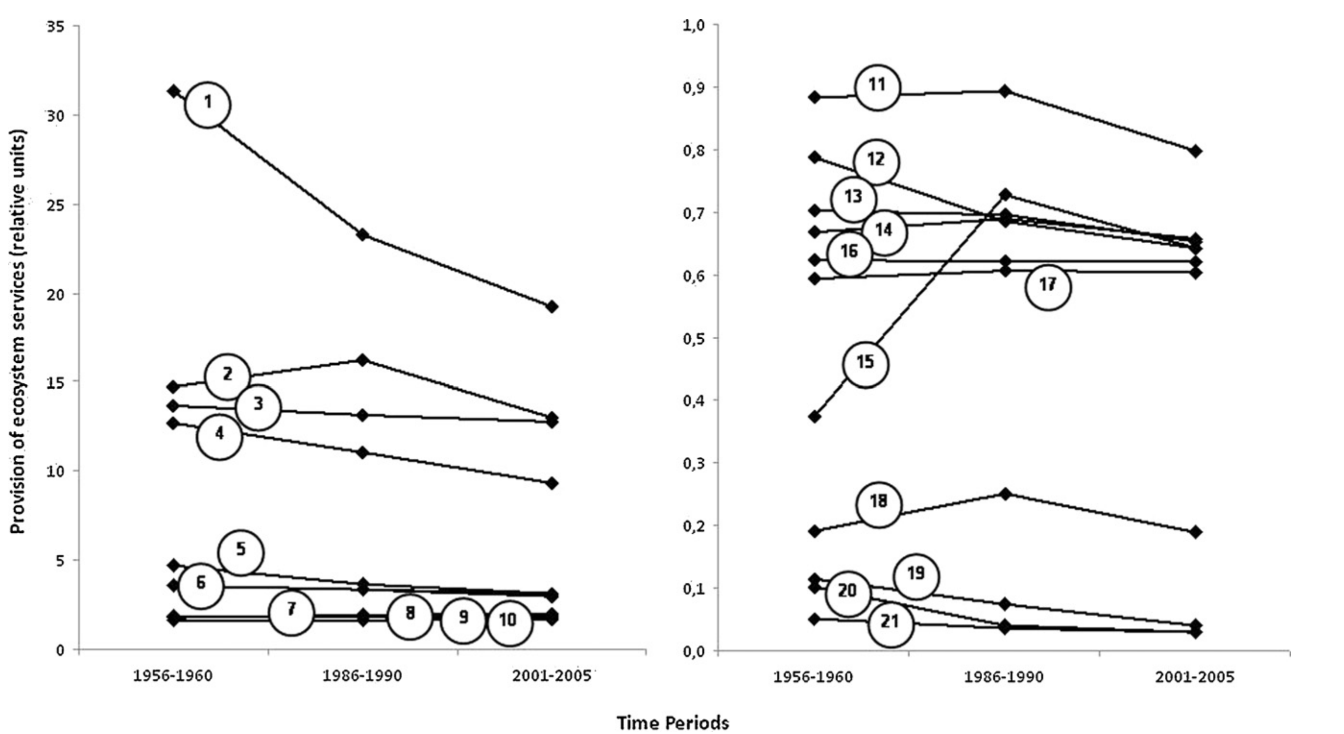

Fig. 5. Long-term trends in the relative provision (0–100) of ecosystem services between 1956 and 2005 due to land-use change in the studied sub-regions of Argentina estimated by the bio-physical model. Graphs are displayed separately to improve the visualization of regional and subregional trends across the relative scale of services provision. References: (1) Atlantic Forest; (2) Paraná Delta; (3) Iberá Marshes; (4) Yungas; (5) Humid-Subhumid Chaco; (6) Western Subhumid Chaco; (7) Central Subhumid Chaco; (8) Espinal; (9) Dry Chaco; (10) Patagonian Forest; (11) Mesopotamian Pampas; (12) Central Pampas; (13) Southern Pampas; (14) Rolling Pampas; (15) Shrubland; (16) Semiarid Pampas; (17) Flooding Pampas; (18) Hills and Valleys; (19) Puna; (20) Andes; (21) Patagonian Steppe.

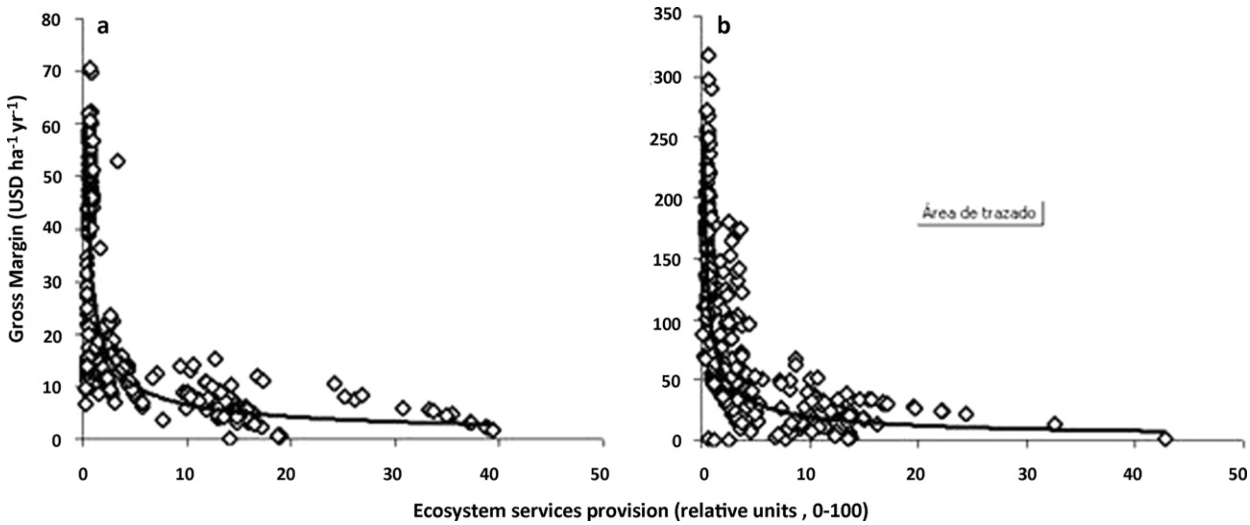

Fig. 6. Tradeoffs between economic income and ecosystem services provision of the studied districts during (a) the 1956–1960 ($y = 28.203 \times \mathrm{EXP}[-0.6333]$; $p < 0.01$; $R^2 = 0.6436$) and (b) the 2001–2005 ($y = 88.448 \times \mathrm{EXP}[-0.6575]$; $p < 0.01$; $R^2 = 0.5419$) periods.

and 2001–2005 periods. The gain of economic services increased allometrically (a power function fitted best) as ecosystem services declined, and the scaling factor in the second period more than doubles the first one. At an eco-regional scale, this inverse relationship is confirmed by estimations on ecosystem services provision and GM at the beginning and the end of the study period are shown, both in absolute and relative terms, in Table 2. The last two columns indicate that the economic benefit clearly outweighs the loss of natural services. The conversion of natural lands into croplands and grazing lands, and the massive introduction of technology between 1950 and 2000, triggered a sharp increase in the biological and economic productivity that clearly surpassed (in % terms) the eco-service loss in most intervened land. The provision of tangible economic benefits by agriculture overwhelmed the loss of ecological benefits that are intangible to human perception. However, caution is required to face hasty and simplistic interpretations. This can be explained not only by land transformation, but also by technology incorporation, which have been driven to maximize the economic income, but not to neutralize the negative ecological impact of agriculture expansion. The argument that income and social benefits more than justifies the conversion of natural into cultivated lands is not easily contradictable at this time, even when it happens at the expense of losing essential ecosystem services. Economic and social priorities have dominated over the environment regarding land-use policies in developing countries. However, although insufficiently assessed, the price to be paid for expanding utilitarian farming models is the irreversible loss of intangible ecological services that cannot be replaced or recovered.

3.3. Limitations of the proposed model

The proposed model has limitations that need further discussion. First, all algorithms were based on assumptions that arose from the ecological theory, which are not necessarily supported by empirical evidence that needs to be subjected to possible exceptions (e.g. higher VC_B can be related to higher biodiversity in some cases). Second, the whole model and utilized equations were simplified through few factors of relatively easy access; in other terms, they were based on information layers that can normally be reconstructed from statistical datasets and field measurements. Nevertheless, we recognize that models of higher complexity would be required to improve the precision of estimations. Third, information sources can be questionable, for example, biomass and its variability in different biomes were estimated from metadata and not empirical assessments. Fourth, the quality of input data was not homogeneous in all regions of Argentina; therefore, uncertainty is higher, for example, in less monitored areas such as those of Chaco in relation to those of the Argentine Pampas.

Finally, there is the issue of the scale. Ecologists are often asked to contribute to solutions for broad-scale problems, but scale extrapolation faces various limitations (Miller et al., 2004): (i) among other reasons, cross-scale studies are justified because knowledge at one scale is normally insufficient to explain the behavior of processes that occur at other scales; (ii) some indicators of agricultural performance are scale-unspecific (Dumanski et al., 1998) and can be scaled up or down linearly without losing their integrity (e.g. kg ha^{-1}, USD ha^{-1}); (iii) but many others

Table 2
Absolute and relative change in the provision of ecosystem and economic services in the studied eco-regions of Argentina in the 1956–1960 and 2001–2005 periods.

Eco-regions	Annual provision of ecosystem services (range 0–100)				Annual Gross Margin (USD ha^{-1})			
	1956–1960	2001–2005	Difference	Percentage	1956–1960	2001–2005	Difference	Percentage
Pampas	0.71	0.66	−0.05	−7.04	50.92	146.43	+95.51	+187.57
Espinal	1.76	2.02	+0.26	+14.77	19.59	53.49	+33.90	+173.05
Chaco	3.44	2.78	−0.66	−19.19	12.54	46.80	+34.26	+273.21
Atl. Forest	31.35	19.23	−12.12	−38.66	5.50	27.91	+22.41	+407.45
Esteros	13.70	12.78	−0.92	−6.72	8.23	24.62	+16.39	+199.15
Delta	14.75	13.03	−1.72	−11.66	10.51	37.91	+27.40	+260.70
Yungas	12.73	9.33	−3.40	−26.71	4.51	20.80	+16.29	+361.20

(e.g. soil sediment losses, contamination hotspots and ecosystem services) are meaningful only within discrete scales, and may lose their integrity with a switch in the scale (Allen and Holling, 2002). Thus, considering that a wrong decision at one scale can produce winners and losers at other scale, policy makers need to understand the cross-scale implications of decisions made on the basis of scale-specific information. In our study, results do not provide detailed site-specific information, but our low-resolution information reports about large-scale geographical patterns and gradients, and temporal trends, that are potentially useful to support decisions by policy makers and land managers.

Beyond the limitations listed above, our approach has much in common with an interesting recent work of Raudsepp-Hearne et al. (2010). Like us, the authors propose a biophysical model to estimate ES expressed on a relative scale (in this case between 0 and 1). Moreover, they also use information from census and agricultural statistics, which determines as in our case, the smallest unit of analysis: the political district. The authors justified this choice arguing that social processes shape the production and use of ES.

4. Conclusions

Beyond the efforts to put value on ecosystem services, the pricing system is still regulated by the free market, creating regional disparities that today reflect large unequal appraisals to value goods and services. The valuation of intangible ecological assets independently of market price appears to be one pressing need to rationally link economy to ecology and avoid the inequalities of price perception.

Considering that the intangibility of most natural assets and services is today a critical dilemma to be resolved, the reliable valuation of ecosystem services is still a source of uncertainty both for biologists and economists. Since the subjectivity of estimates is the weak point of the methods currently applied, they agree on the need of minimizing market failures by increasing objectivity in valuation methods. Models such as the biophysical one described here are a need to improve the consistency of valuations.

Policy makers need the support of objective valuations if ecological assets have to be incorporated in the design of land-use strategies that already comprise the consideration of economic and social variables. Beyond the overwhelming capacity of humans to increase their economic and social benefits in the short term, we have to recognize that the loss of natural services, even at slow rates, may produce irreversible damage. According to our results, irreversibility may become critical in biomes such as those of the Atlantic Forest and Yungas that annually deliver large amounts of intangible services and are thus vulnerable to aggressive human intervention. However, this view would have less practical importance in regions such as those of the Argentine Pampas that have high agricultural value but low capacity to provide ecosystem services. However, it would not be sensible to convert the Pampas sub-regions into novel ecosystems to improve the delivery of ecological services at the expense of their ability to produce agricultural and economic surpluses. So, given the mosaic of highly diverse biomes that spread across the country, any attempt to apply sole and centralized land-use strategies to different biomes may lead to undesirable results. Multiple criteria, including the outcomes of this type of models, should be regionally balanced as a pre-requisite to design sound land-use policies on broad spatial and temporal basis.

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