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An integrative index of Ecosystem Services provision based on remotely sensed data

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ABSTRACT

We present an approach to generate estimates and to map Ecosystem Services (ES) related to C and water dynamics (Soil Carbon sequestration, evapotranspiration and groundwater recharge) and biodiversity (Avian Richness) from remotely sensed data in two ecoregions of South America: the Semiarid Chaco woodlands and the Rio de la Plata grasslands. Two attributes of the seasonal dynamics of the Normalized Difference Vegetation Index (NDVI); the annual mean ($NDVI_{mean}$), an indicator of light interception and hence of total C gains and the intra-annual Coefficient of Variation of the NDVI ($NDVI_{CV}$), a descriptor of seasonality; were combined into an ES provision index ($ESPI = NDVI_{mean} * (1\text{-}NDVI_{CV})$). The proportion of the variance in ES provision explained by the ESPI varied from 0.484 for avian richness up to 0.662 for C sequestration. A relatively large proportion of the studied area presented changes in ES provision. A 32,4% of the Semiarid Chaco and the Rio de la Plata grasslands presented significant ($p < 0.01$) trends. Most of the trends (30.2%) were negative, showing a decrease in ESPI. An index like the one proposed here can be used as an aggregated indicator of the status and/or trends of ES supply at large spatial scales (subcontinental in our case).

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1. Introduction

1.1. Turning operative a good idea

The Ecosystem Services (ES) concept has a potentially important role in the diagnosis, planning and management phases of land use policies. Though many initiatives promoted the use of the ES idea (i.e. the Millennium Ecosystem Assessment (MEA), 2005, or the Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES, Turnhout et al., 2012) its impact on defining policies was limited (Norgaard, 2010).

Measuring ES supply is often complicated because of the temporal and spatial scale of the providing units (Kremen, 2005). Measurements need to be able to capture the supply at the level of the stand, plot or watershed and to integrate it over, at least, a growing season (the period of active growing of vegetation). For academic purposes such as comparing the supply or value of ES among biomes (e.g. Costanza et al., 1997) rough data are enough. However in decision-making processes, monitoring ES requires much more accuracy and resolution. Daily and Matson (2008) identify the generation of production functions to quantify and map services as one of the major constraints to implement policies based on ES. Such production functions describe the spatial and temporal variation of an ES from a marginal change in a series of ecosystem processes and climatic, edaphic and management variables.

1.2. Definitions and the "Cascade Model" of ES provision

One of the classification schemes of ES, the "cascade" model presented by de Groot et al. (2010) and Haines-Young and Potschin (2010) (Fig. 1), allows to link ecosystem structure and functioning with human well-being. The idea of production functions is embedded into the cascade model. In this scheme the Intermediate ES (IES, the ecosystem structure and processes in itself) are separated from the final ES (FES, the processes that are directly associated to the generation of human benefits) (Boyd and Banzhaf, 2007; Fisher and Turner, 2009) (Fig. 1). The limit between intermediate and final services is often ambiguous. For example, biodiversity or

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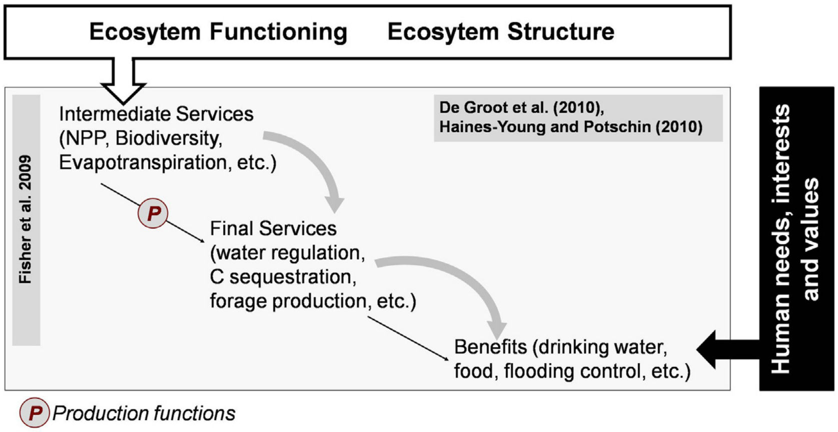

Fig. 1. General scheme of the connection between ecosystem structure and functioning, intermediate and final services. Based on Boyd and Banzhaf (2007) "Cascade Model".

hydrological yield can be considered either final or intermediate services depending on the context and the benefits on which stakeholders are focusing.

Some authors (i.e. Costanza et al., 1997; Millennium Ecosystem Assessment (MEA), 2005) defined ES as the benefits that the society derives from ecosystems. Fisher and Turner (2008) strongly advocates for separating the concept of benefits from the idea of services, as it facilitates the integration of the biophysical and sociocultural dimensions of the ES framework (Mastrangelo et al., 2015). Benefits are associated to interests, ideological frameworks, beliefs and social needs. The generation of benefits depend on the actual demand of ES (Tallis and Polasky 2011; Yahdjian et al., 2015). A way to represent the magnitude of the benefits is through the valuation of the ES that generate such benefit. In this article, we define ES as the aspects of ecosystems utilized (actively or passively) to produce human well-being (Boyd and Banzhaf, 2007; Fisher and Turner, 2009).

1.3. Production functions

Ecologists and agronomists were able to develop a number of models that connect ecosystem and/or biophysical processes to products/services that directly generates benefits to the society or, at least, to part of it. Most of such models were generated in a completely independent framework than the one provided by the ES concept. Crop production functions (i.e. Parish and Dillon, 1955; Vaux and Pruitt, 1983; Tafteh et al., 2013), soil erosion (Wischmeier and Smith, 1960; Hudson, 1993; Laflen et al., 1997), physiological crop (Jones et al., 2003; Hoogenboom et al., 2012), biogeochemical (i.e Century or DAYCENT, Parton et al., 1987, 1994, 2001) and hydrological (Zhang et al., 2001; Walker et al., 2001; Govender and Everson, 2005) models are some examples of them. Such models provide valuable information on how environmental and management factors relate to ES supply. Some of the problems associated to the use of several of these models are that often they required a large amount of field and management data that are laborious/expensive to obtain and/or that they require site specific calibration.

Viglizzo et al. (2011) proposed a set of equations to estimate the supply of several ES (Supplementary Material). The equations are based on proxies of annual net primary production (NPP) and of its intra-annual variability. Barral and Maceira (2012) used the equations to map ES supply at the county level and Viglizzo et al. (2011) mapped ES supply at the national level for Argentina. The equations presented by Viglizzo et al. (2011) represent a first tier to generate estimates of ES supply for land planning processes (Paruelo et al., 2015). The equations proposed have a solid conceptual foundation. On one hand, the idea that the energy captured by the ecosystem determines ES supply has been already proposed by Richmond et al. (2007). On the other, the intra-annual variation of NPP is a key descriptor of ecosystem functioning (Pettorelli et al., 2005), particularly sensitive to land use changes (Paruelo et al., 2001; Guerschman et al., 2003). Do the descriptors of the C gains dynamics capture the spatial and temporal variation in ES supply? In other terms, is the approach proposed by Viglizzo (using NPP and its intra-annual variation) supported by empirical data? No evidence is available yet for such relationships. In this article we sought to answer these questions.

1.4. The use of remote sensed data to estimate and map ES

Different proxies or indicators of ES supply have been proposed (Stephens et al., 2015). Those associated to land cover/land use types have been the most widely used (Ayanu et al., 2012), but species richness (see Balvanera et al., 2006) and functional diversity (Cadotte et al., 2011) were also considered. Ayanu et al. (2012) presented a comprehensive review of the use of remotely sensed data to map ES. Most of the cases reviewed were attempts to connect ES supply to land cover patterns (mapped from spectral data (i.e. Burkhard et al., 2012). Scenarios of ES supply can be built by altering land cover patterns (Swetnam et al., 2011). Some of the most common protocols (e.g., InVEST, ARIES, ECOSER) use a look up table for each land cover type to estimate ES supply (Nelson and Daily, 2010; Nelson et al., 2009; Laterra et al., 2011, 2012). However the same land use or land cover change may impact differentially on ES supply depending on legacy factors, management or the influence of factors such as climate or landscape structure. For example, in the Argentine Pampas the effect on C sequestration of transforming the original cover (grasslands) into croplands depends not only on the sowed crop but also on the tillage system and the fertilization level (Caride et al., 2012).

Remotely sensed data allow not only the description of land cover spatial patterns but also a direct estimation of functional attributes of the ecosystems (Pettorelli et al., 2005). Data on the surface reflectance and/or emission recorded by sensors on board of satellites provide direct measurements of surface temperature, albedo (Prata et al., 1995; Liang, 2000) and allow to calculate indices that estimate the fraction of the photosynthetic active radiation absorbed by green tissues and radiation use efficiency (Gamon et al., 1995; Petorelli, 2013). The data provided by sensors on board of satellites can be integrated into biophysical models able to provide reliable estimates of critical ecosystem processes or intermediate services. Two well known examples are those models that estimate Primary Production (Running et al., 2000) and Evapotranspiration

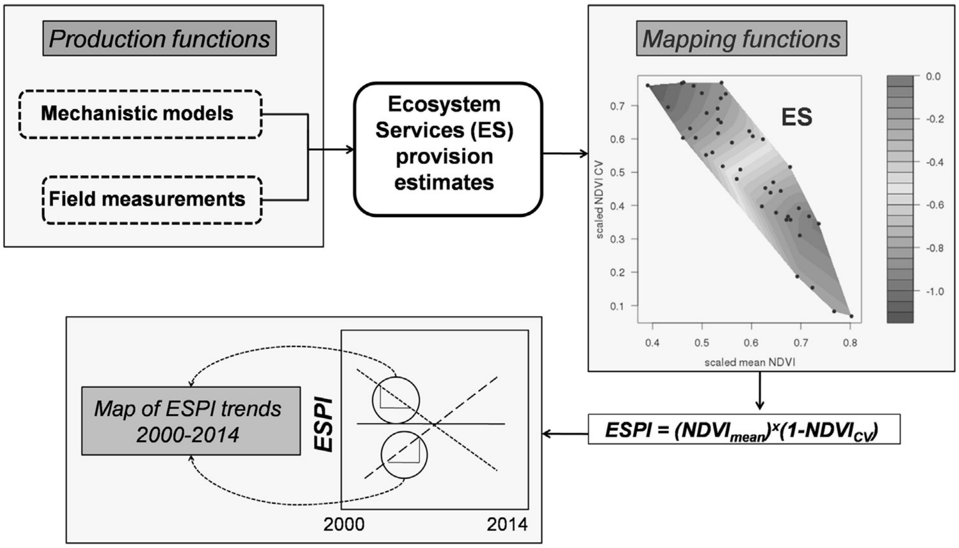

Fig. 2. General scheme of the generation of the Ecosystem Services Provision Index (ESPI) from estimates of ES provision derived from either filed data or mechanistic models and remotely sensed data (mean NDVI and NDVI CV). The ESPI can be derived annually and to derive maps temporal trends.

(Courault et al., 2003). Primary Production and Evapotranspiration are key terms to calculate C and water balances.

A particularly important characteristic of the functional attributes derived from satellite data is the ability to track both phenological changes and the interannual variability of ecosystem processes (Cabello et al., 2012). As several studies suggested C gains and its temporal dynamics are integrative attributes of the ecosystem (McNaughton et al., 1989; Paruelo and Lauenroth, 1995; Running, 2012). Several structural and functional aspects of ecosystems are tightly related to C gain dynamics: water losses, biodiversity, harvestable biomass, soil protection, Soil Organic Carbon (SOC) stocks. All these ecosystem aspects are, following Fisher and Turner (2009), Intermediate Ecosystem Service in the cascade model (Fig. 1). Paruelo and Lauenroth (1995) showed that most of the spatial heterogeneity in C gains can be captured by two attributes of the seasonal dynamics of the Normalized Difference Vegetation Index (NDVI, an spectral index calculated from the red and infrared reflectance of the surface, Pettorelli, 2013): the annual average ($\text{NDVI}_{\text{mean}}$) and a measure of the intra-annual seasonality. Volante et al. (2012) used the annual integral of the NDVI or $\text{NDVI}_{\text{mean}}$ and the coefficient of variation (CV) of monthly NDVI (NDVI_{CV}) as Intermediate Ecosystem Service related to C gains to describe the impact of land cover changes in the Argentine Chaco. The intra-annual NDVI_{CV} was particularly sensitive to land use change: the transition to agriculture (annual crops) always increased seasonality (Paruelo et al., 2001; Guerschman et al., 2003; Guerschman and Paruelo, 2005; Volante et al., 2012) (see supplementary material). The NDVI_{CV} was also sensitive to another land cover transition observed in temperate and subtropical South America: the conversion of grasslands into tree plantations (Jobbágy et al., 2006). In this case NDVI_{CV} decreased (Vassallo et al., 2013) (see supplementary material). Agriculture, except in the case of irrigated crops, in general reduce or do not change mean NDVI (Paruelo et al., 2001; Guerschman et al., 2003; Volante et al., 2012) while afforestation increase this attribute compare to the original vegetation (see supplementary material).

1.5. *Questions and objectives*

Changes in the supply of a final ES could affect the magnitude of the benefits perceived by some stakeholders and the socio-ecological vulnerability of the system. Some of the relevant questions for making decisions on land planning include: how much does a particular ES changed during a giving period? Is the rate of change of ES provision accelerating or decreasing? How does the change in ES supply compare among landscapes occupied by different land covers or land uses? Based on historical trends, how much an increment of a given land cover would impact on a particular ES? To answer these questions repeatable estimates of ES that cover a long time period and the whole territory under analyses is needed.

In this article we present an approach to generate estimates and map ES related to C and water dynamics (Soil Carbon sequestration, evapotranspiration and groundwater recharge) and biodiversity (Avian Richness) from remotely sensed data. We based our analysis on case studies from the Chaco-Pampean Plains of South America. We evaluated how ES estimated from direct measurements or using mechanistic production functions can be described from ecosystem functional attributes derived from remotely sensed data (the annual mean of the NDVI and its intra-annual coefficient of variation) (Fig. 2). Based on an integrative index derived from ecosystem functional attributes we mapped the temporal trends in ES provision in two South American ecoregions, the Rio de la Plata grasslands and the Chaco dry forests. Though we recognize the importance of characterizing ES demand, this article focus on the provision.

2. Materials and methods

2.1. *Study region*

Our study region include the Chaco-pampean plains of South America. The area extends from 16° to 40° S, from Bolivia and Paraguay to the central part of Argentina and Uruguay (Fig. 3). The study area include both subtropical (the semiarid Chaco plains) and temperate (the Río de la Plata grasslands) climates with mean annual precipitation ranging from less than 400 mm in the central-North part to more than 1300 mm, in the eastern portion (Minetti et al., 1999; Paruelo et al., 2007). Mean annual temperatures decreased southward from 29 °C to 14 °C. The southern plains are dominated by temperate grasslands and the northern part by woodlands or thorn forests, interspersed with grasslands (Paruelo et al., 2007; Morello et al., 2012). The whole area is experiencing an intensification and expansion of annual (mainly soybean) and

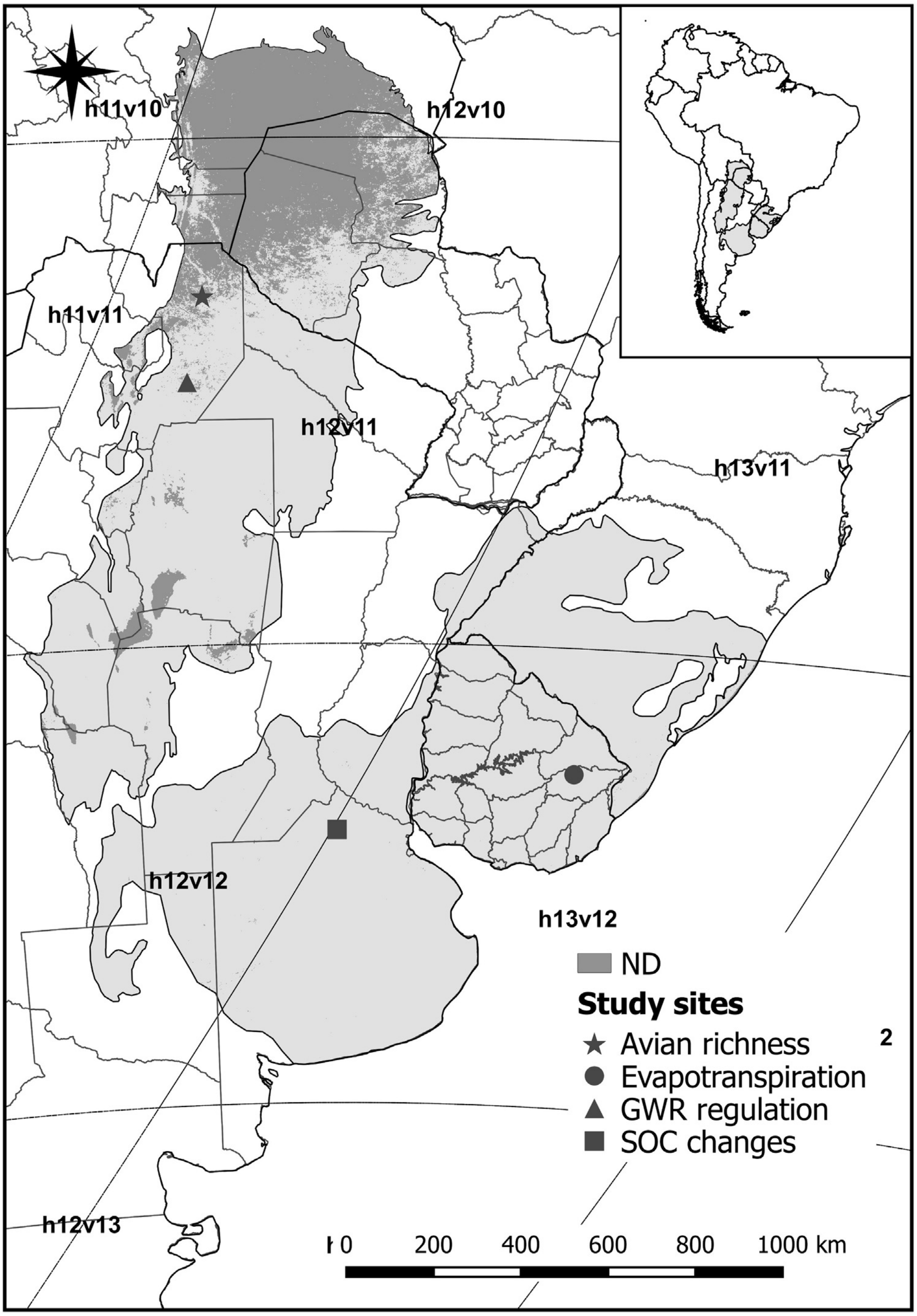

Fig. 3. Ecoregions analyzed, the Semiarid Chaco and the Rio de la Plata Grasslands. Dots indicated the areas where the field (Avian richness, evapotranspiration, Groundwater recharge (GWR)) or model (Soil Organic Carbon (SOC) changes) estimates were derived. The dark grey areas (no data, ND) corresponded to pixels for which less than 9 years of data were available.

perennial (mainly eucalyptus) crops that determines some of the highest rates of land cover transformation in the world (Volante et al., 2012; Baeza et al., 2014; Vallejos et al., 2014; Volante et al., 2015).

2.2. Estimates of Ecosystem Services provision in the Chaco-Pampean plains

Several local efforts quantified specific ES in the Chaco-Pampean region (Laterra et al., 2011; Altesor, 2011). The focus was on ES particularly affected by land cover changes and with a high social, environmental and economic impact. The ES estimated included biodiversity, groundwater recharge, C sequestration and evapotranspiration. In this article we used data on the supply of these ES derived from direct estimates or mechanistic simulation models.

Macchi et al. (2013) and Mastrangelo and Gavin (2014) studied the supply of the ES Biodiversity Conservation by sampling avian diversity across the Chaco Region. Data were derived from field observations on different land covers. Both studies found a relationship between their descriptor of the ES (avian diversity) and land use types. In our analysis we used the richness data compiled by Mastrangelo and Gavin (2014). Though we recognize the role of biodiversity in determining ecosystem multifunctionality we considered in this case the avian richness as a final ecosystem service (Hector and Bagchi, 2007).

Amdan et al. (2013) analyzed a critical regulation service for the sustainability of the Chaco Region: groundwater recharge. As in many semi-arid woodlands of the world (Jobbágy et al., 2008) the replacement of dry forest by agriculture determines a sustained raise of the water table level that may lead to serious and almost irreversible salinization processes. Based on the analysis of Cl^- profiles and water content profiles, Amdan et al. (2013) found that the magnitude of groundwater recharge differed among land uses (croplands > pastures > dry forests) and with the time since the plot was deforested. From Amdan et al. studies we used the regulation of the recharge assuming 0 recharge as maximum regulation and > 0 recharge values as losses of the regulation capacity.

Carbon sequestration has been identified as one of the most important ES provided by grasslands (Sala and Paruelo, 1997; Costanza et al., 1997). Using one of the complex production functions described above, the Century Model, Caride et al. (2012) described how different combination of land management (sequence of crops, rotation with pastures, fertilization and tillage systems) affect Soil Organic Carbon (SOC) dynamics in the most productive soils of the Chaco-pampean plains. SOC losses after 60 years varied from 9% for native grasslands grazed by domestic herbivores up to 37% for continuous agriculture based on summer crops. C regulation was expressed as percentages of change from the original situation and it went from positive (C accumulation respect the reference situation) to negative (C losses respect to the original situation).

In the portion of the Chaco-pampean plains dominated by shallow soils (Southern Brazil and Uruguay) water supply is highly dependent on superficial and sub-superficial runoff. Small reservoirs and spring depends on rainfall able to "escape" plant water absorption. Evapotranspiration at the watershed level is the main determinant of the available water. Land covers differed in the total amount of water transpired. *Pinus* and *Eucalytus* plantations evapotraspirated 80% more than the native grasslands that they replaced (Nosetto et al., 2005). Gallego (2014) evaluated, using Landsat 8 imagery, the evapotranspiration of 132 individual watersheds in Uruguay. Evapotranspiration ($mm\,d^{-1}$) increased with the afforested proportion of the watershed. Changes in evapotranspiration (an intermediate service) allow to estimate changes in water supply or hydrological yield (final services). For our analyses we used the evapotranspiration values reported by Gallego (2014). In the supplementary material we provide additional details on the four data sets used.

2.3. Mapping ES from NDVI functional attributes

We extracted NDVI from the MODIS sensor onboard the EOS Terra satellite (MOD13Q1 products, which have been available since February 2000, have a 250-m spatial resolution and a 16-day temporal resolution) for the sites described in the previous section. The use of NDVI data instead of more elaborated products (i.e. NPP or GPP data from MODIS) would allow to calculate the ESPI from different platforms. On the other hand, in temperate and subtropical areas the seasonal patterns are more evident in the NDVI than in the NPP annual curves (Baeza et al., 2010). NDVI is a reliable estimator of the seasonal dynamics of PAR interception by green tissues (Di Bella et al., 2004). For the areas studied and for the scale of the analysis we discard serious biases in the estimation of the ESPI based on the behavior of NDVI in the extremes of a NPP/biomass gradient (Piñeiro et al., 2006; Baeza et al., 2010; Guido et al., 2014; Blanco et al., 2008; Blanco et al., in press). We calculated the NDVI annual mean ($NDVI_{mean}$) and the intra-annual Coefficient of Variation ($NDVI_{CV}$) of the monthly values for the sites studied. The NDVI is a widely used spectral index, linearly related to the fraction of the photosynthetic active radiation absorbed by green tissues (fAPAR) (Sellers et al., 1992; Di Bella et al., 2004). fAPAR is the main determinant of net primary production (Monteith 1972; Running et al., 2000). We evaluated the proportion of the variance of the provision of the different ES explained by a Provision Index (ESPI) defined as the product of $NDVI_{mean}$ and ($1\text{-}NDVI_{CV}$). It can be easily demonstrated that the product ($NDVI_{mean}$) x ($1\text{-}NDVI_{CV}$) is equivalent to ($NDVI_{mean} - NDVI_{SD}$), where $NDVI_{SD}$ is the intra annual standard deviation of NDVI monthly values. Thus, we used this difference as an ESPI. To map ESPI and its temporal trends over the whole study area we used NDVI data at a 1 km^2 resolution corresponding to tiles h11v10, h12v10, h12v11, h12v12, h13v11 and h13v12 (see Fig. 3).The values were normalized considering the highest and lowest values of the index to scale it up to the 0–1 range. Those areas where, due to cloud cover, data were not available for at least 9 years (mainly in the Northern portion of the Semiarid Chaco, see dark grey areas in Fig. 3) were not included in the analysis.

2.4. Models on the relationship between ES estimates and the NDVI attributes

We built models for each of the ES studied and the $NDVI_{mean}$ and $NDVI_{CV}$, on one hand, and the ESPI, on the other, as independent variables. Previous to any analysis, the four ES descriptors were normalized to remove the effects of scale differences among variables. Normalization was based on the maximum and minimum values observed on each data set, in order to express valuesin a 0–1 scale. Mean annual NDVI and intra annual NDVI coefficient of variation were also re-scaled considering regional minimum and maximum values.

To analyze the relationship between each ES descriptor and the annual mean of the NDVI and its intra-annual coefficient of variation we applied multiple linear regressions (MLR, Quinn and Keough, 2002) and generalized additive models (GAM, Hastie and Tibshirani, 1990; Wood 2006). Whereas MLR assume a linear relationship between the dependent variable (i. e. each ES descriptor) and the independent ones (NDVI mean and NDVI intra annual CV), GAM do not assume any specific shape for the relationships. The complexity of the curve (the number of effective degrees of freedom) was determined by penalised regression splines and generalized cross validation (GCV, Wood, 2006). For each ES descriptor the model with the best compromise between fit and complexity (MLR vs different GAMs) was selected by means of the small sample Akaike information criteria (AICc, Burnham and Anderson, 1998). These analyses were performed in an exploratory way, given that as $NDVI_{mean}$ and intra annual $NDVI_{CV}$ are correlated, we cannot unambiguously partition the effect of these variables on the ES descriptors (i.e. Zuur et al., 2009, chapter 3). A second set of models explored the bivariate relationship between each ES descriptor and ESPI. Here we also applied (simple) linear regression and generalized additive models, and selected the most parsimonious models by means of AICc.

The predictive ability of the fitted models was evaluated by means of jackknife (Crowley, 1992). In this cross validation method, subsamples are generated leaving out one of the original observations, re-fitting the model, and obtaining predictions for the excluded observation with the re-fitted model. This process is repeated for all the observations. The predictive ability of the models was evaluated by means of the square of the correlation coefficient between observed data and jackknife predictions ("R2 jackknife" in Table 2). We evaluated the trends in ESPI for the period 2000 and 2014 by means of linear regression against time.

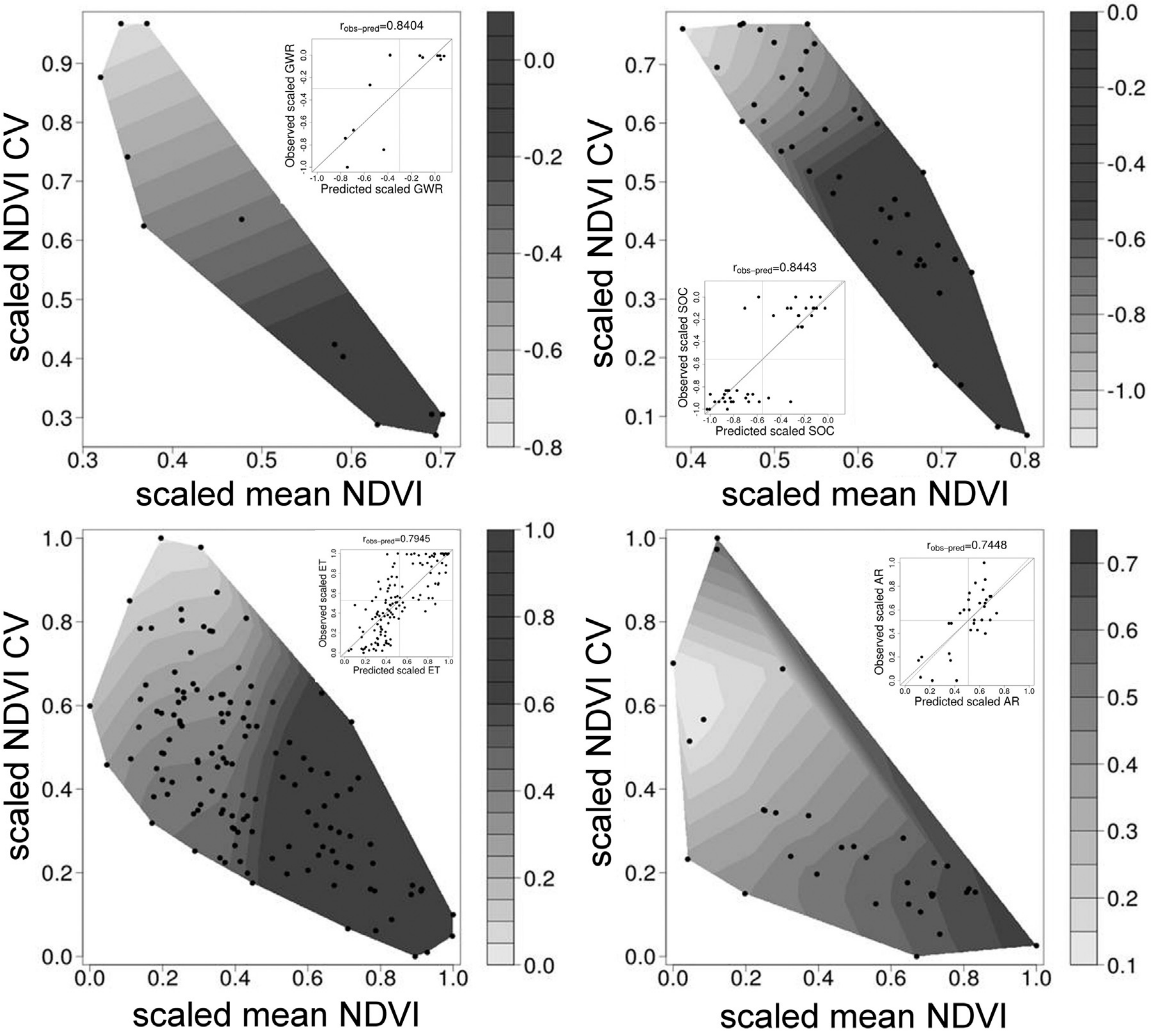

Fig. 4. Model (surface responses) fitted to the scaled ES provision data (Groundwater recharge (GWR), Soil organic Carbon changes (SOC), evapotranspiration (ET) and Avian Richness (AR)) as a function of scaled mean NDVI and NDVI CV). The type of models fitted are presented in Table 1. The inset presents the values predicted by the models and the observed data.

3. Results

3.1. Models— response surfaces

The combination of NDVI $_{mean}$ and NDVI $_{CV}$ described a substantial proportion of the spatial differences in ES provision. All bivariate models, relating specific ES with $NDVI_{mean}$ and $NDVI_{CV}$, exhibited a positive effect (linear or non linear, depending on the ES, Fig. 4, Table 1) of $NDVI_{mean}$, and a negative effect of $NDVI_{CV}$ (again linear or non linear, depending on the ES, Fig. 4, Table 1). The proportion of the variance in ES provision explained by the ESPI, the index that combined both attributes $NDVI_{mean}$ * ($1-NDVI_{CV}$), or equivalently ($NDVI_{mean} - NDVI_{SD}$), varied from 0.484 for avian richness up to 0.662 for C sequestration (Table 2). For all ES considered, the effect of ESPI was positive, linear for ground water recharge and evapotranspiration and non-linear for soil organic carbon losses and avian richness (Table 2). The R^2 coefficients between observed and predicted values, generated through Jackknife, varied between 0.392 and 0.629 and were always significant ($p < 0.01$).

3.2. Patterns and trend in ES supply

Both environmental factors (mainly precipitation) and land use/land cover patterns controlled the heterogeneity in the mean (2000–2014) ES provision index (Fig. 5, left). Areas with low transformation (Vallejos et al., 2014; Baeza et al., 2014) had a marked contrast in ESPI. The eastern portion of both the Chaco woodlands and the Rio de la Plata Grasslands (RPG) presented areas with high values of the ESPI associated to the highest precipitation (that increase NDVI $_{mean}$) and temperature (that decrease NDVI $_{CV}$) values of the regional gradient (**Gh** in Fig. 5, left). The other extreme of the gradient corresponds to the western portion of the RPG (**Gl**). Some intermediate values of the index (**Gi**) were associated to the central portion of the semiarid Chaco region (constrained by precipitation, Volante et al., 2012) and the SE portion of the RPG (constrained by soils characteristics, Paruelo et al., 2007). Areas with low values of ESPI clearly corresponded to agricultural foci (**Af**) (Vallejos et al., 2014; Baeza et al., 2014; Volante et al., 2015). The arrows indicated either biophysical or political boundaries where a marked contrast in ESPI was evident due to profound differences in land use/land cover.

The analysis of the absolute values of the ESPI trends over the period 2001–2014 showed some important features (Fig. 5, right). First, a relatively large area presented changes in ES provision. A 32.4% of the Semiarid Chaco and the Rio de la Plata grasslands presented significant ($p < 0.01$) trends. Most of the trends (30.2%) were negative showing a decrease in ESPI. Many of those areas corresponded to the agricultural foci identified in both regions The **Af**

Table 1
Models (surface responses) fitted to the scaled ES provision data (Groundwater recharge (GWR), Soil organic Carbon changes (SOC), evapotranspiration (ET) and Avian Richness (AR)) as a function of scaled mean NDVI and NDVI CV). Mean NDVI represents the form of the relationship beetwen each ES and mean NDVI (linear with positive (+) or negative (−) slope or non-linear) whereas NDVI CV represents the same, but for the intra anual CV of NDVI. The effective degrees of freedom for each independent variable (edf_{mean}, edf_{cv}), the sample size (n), the Akaike information criteria corrected for small samples (AIC_C) and the adjusted determination coefficient (R^2_{adj}) are also shown.

	Mean NDVI	NDVI CV	edf_{mean}	edf_{CV}	n	AIC_C	R^2_{adj}
GWR	Linear (+)	Linear (−)	1	1	12	9.62	0.706
SOC	Linear (+)	Non linear	1	3.9	44	3.42	0.682
ET	Non linear	Non linear	4.4	2.1	132	−35.9	0.612
AR	Linear (+)	Non linear	1	3.2	33	−9.67	0.485

Table 2
Models (surface responses) fitted to the scaled ES provision data (Groundwater recharge (GWR), Soil organic Carbon changes (SOC), evapotranspiration (ET) and Avian Richness (AR)) as a function of ESPI. Best fit model (linear with positive (+) or negative (−) slope or non-linear) for each ES is shown. The effective degrees of freedom (edf), the sample size (n), the Akaike information criteria corrected for small samples (AIC_C) the adjusted determination coefficient (R^2_{adj}) and the jackknife coefficient of determination ($R^2_{jackknife}$) are also shown.

	Best model	edf_{mean}	n	AIC_C	R^2_{adj}	$R^2_{jackknife}$
GWR	Linear (+)	1	12	6.5	0.645	0.582
SOC	Non linear	3.9	44	1.57	0.662	0.629
ET	Non linear	2.5	132	−25.7	0.529	0.521
AR	Non linear	1.5	33	−6.0	0.484	0.339

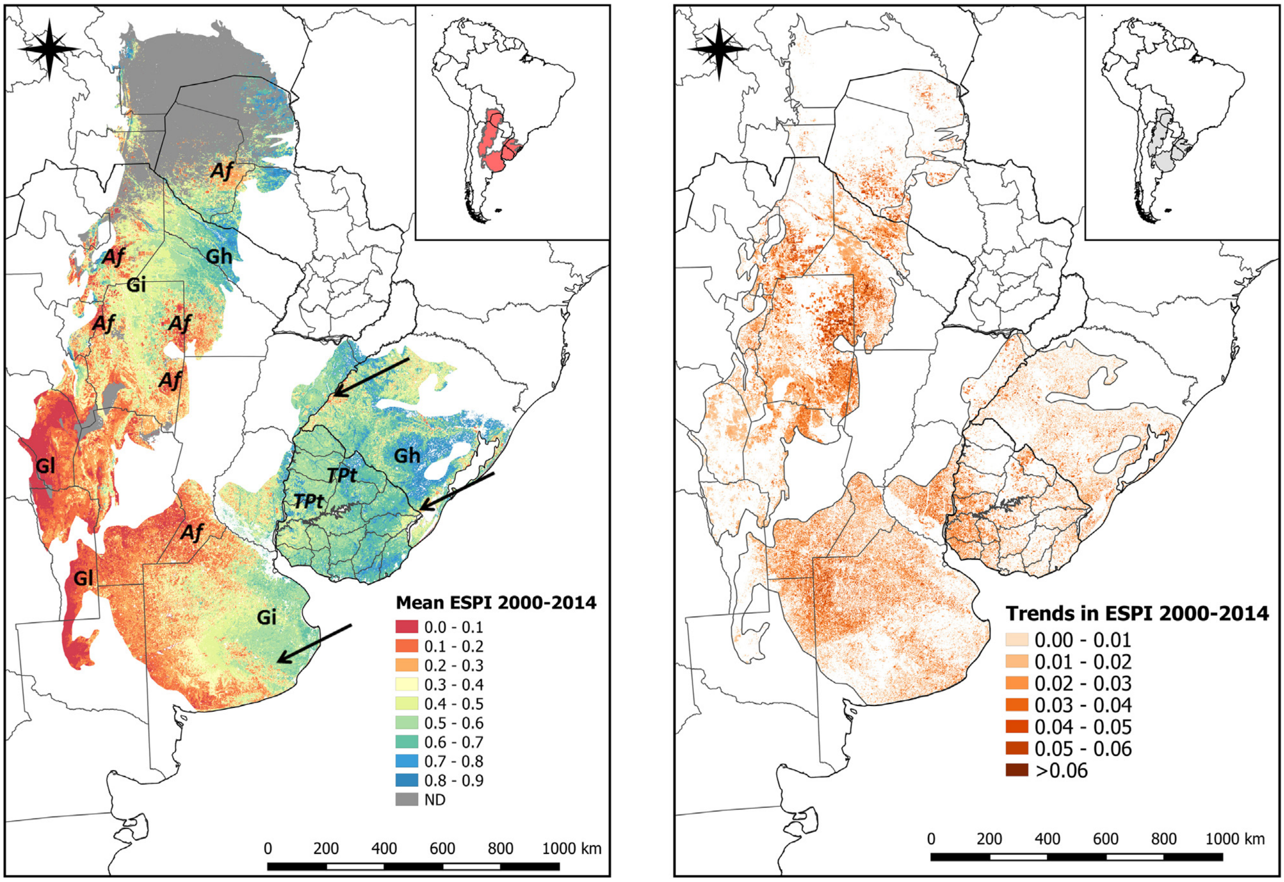

Fig. 5. Left: Mean values of the ESPI for the two ecoregions studied. The arrows indicate frontiers with contrasting land uses due to environmental or political factors. Gh, Gi and Gl, correspond to low transformed areas with high, intermediate and low values of ESPI across a resource gradient (mainly precipitation). Af and TPf corresponded to agricultural and tree plantation foci respectively. Right: Absolute values of the ESPI trends (slope of the regression ESPI vs time) for the period studied.

corresponded to the rainfed transition toward a low seasonality and high productivity (supplementary material). A 2.2% of the area presented positive trends. Most of the pixels showing positive changes corresponded to a particular case: the transition from grasslands to tree plantations (the afforestation transition, indicated as **TPt** in Fig. 5, right). In this case the ESPI presented higher values than the native vegetation that replaced.

4. Discussion

The index (ESPI) proposed provided important insights on the spatial and temporal changes in ES provision. Such information is critical in designing, implementing and monitoring environmental policies. For the ES analyzed, those ecosystems showing higher photosynthetic active radiation interception and less seasonality presented, in general, a higher level of ES provision. Mapping trends in the index allows us to identify areas where the potential change in ES provision deserves particular attention. Clearly the agricultural and afforestation foci indicated in Fig. 5 are some of them.

The actual relationship with the two functional attributes derived from the seasonal dynamics of the NDVI differed among the ES considered. However, for all of them the two descriptors of ecosystem functioning used were able to capture an important proportion of the variance of the spatial variability in ES provision. The models, though, were based on a restricted set of data both in number and spatial extent. Though the index may provide a description of the level of provision and temporal trends of bundles of ES, actual estimates of the absolute values of specific ES need to consider additional information not captured by the functional attributes considered (i.e topography, vegetation structure, management, etc). In each of the local cases considered, measuring or estimating ES was possible but difficult. Procedures were costly, time consuming or demand highly trained personnel. Even simulation models (as CENTURY) required input data that were hard to obtain (i.e. data on fertilization or tillage system at the plot level). Moreover, point measurements need to be intra/extrapolated over the region generating additional uncertainty. An index based on remotely sensed data (i.e. the ES Provision Index proposed in this article) will expand the temporal and spatial scale and the areal coverage of ES estimates as well as reducing cost and time lags on information availability.

Wong et al. (2015) identified a number of problems in ES evaluation methods. The first one is related to the lack of well established criteria to select the ecosystem attributes and services to be evaluated. Secondly, they found no clear indication on how production functions determine changes in ES provision from marginal changes in ecosystem characteristics. Finally, these authors did not identify in current methods a way to estimate ES at different spatial and temporal scales. The scheme outlined in Fig. 2 and the results presented showed a way to overcome, at least partially, the shortcomings of current ES evaluation methods. The index is based on two functional attributes clearly connected to C gains that can be monitored continuously over time and space by remote sensing. The possibility of deriving the two functional attributes ($NDVI_{mean}$ and $NDVI_{CV}$) from different satellite sensors and platforms permit the use of the same protocol to evaluate ES provision at different spatial scales, with a resolution of meters to km and a global extension. Texeira et al. (2015) showed, i.e., the possibility of expanding the temporal extension in the analysis of these attributes to a period of more than 30 years combining databases generated from different sensors.

The index proposed was particularly sensitive to changes associated to land use/land cover transformations. The sensitivity results from the functional attributes used to define it. The seasonality of light interception (and hence of C gains), one of the factors included in the index, is tightly associated to land transformation (Paruelo et al., 2001; Guerschman et al., 2003). The descriptor of seasonality (the intra annual NDVI CV) is higher for annual crops than for native grasslands or forests (Guerschman and Paruelo, 2005; Volante et al., 2012). However, the replacement of grasslands by tree plantations, another common land cover transition, reduced markedly the seasonality (Vassallo et al., 2013). So, as for many indicators, the deviance of ESPI respect to a reference situation is more indicative than the absolute value.

As in human health, the use of single indicators may be risky and can lead to misleading diagnoses. As with many clinical analysis the interpretation depends on the context and on subjective values, needs to consider additional indicators and requires reference situations for comparison. As Byrnes et al. (2014) acknowledged, the definition of the desirable value or change of a given indicator is inherently subjective. Subjectivity would be defined in terms of ideology and political relationships and would be influenced by contextual factors as international agreements or geopolitical issues. The actual magnitude of ES changes and its importance in decision making would depend on many other additional factors (management, landscape configuration, etc.). The identification and quantification of such factors would allow to increase the proportion of the spatial variability of ES provision explained. Defining reference situations become a critical aspect in using indices; the network of protected areas may provide reference situations to discriminate the influence of global and local drivers on ES provision changes (Garbulsky and Paruelo, 2004; Cabello et al., 2012). In our case the implicit reference situation was the past condition.

The index proposed is describing the level of provision of either intermediate and/or final ES, not benefits. As several authors acknowledge (see Kareiva et al., 2011), benefits will depend not only on the ES supply but also on the demand, cultural values and interest (Fig. 1). Moreover, the benefits may be associated to an increase or a decrease in the ES depending on the context. For example, an increase in the hydrological yield may represent a benefit in terms of water supply or a harm in terms of water regulation.

As Orians and Policansky (2009) suggested, an index like the one proposed here can be used as an aggregated indicator of the status and/or trends of ES at large spatial scales (subcontinental in our case). The empirical and functional relationships between the variables and different ES provide a solid conceptual basis for the index. The temporal trend in the index is providing a strong evidence of changes in the provision of ES. These information could be helpful to get aware on time of possible future conflicts.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecolind.2016.06.054.

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