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Ecosystem functional changes associated with land clearing in NW Argentina

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ABSTRACT

We assessed the extension of natural habitat conversion into croplands and grazing lands in subtropical NW Argentina and its impact on two key ecosystem functional attributes. We quantified changes in remotely sensed surrogates of aboveground net primary production (ANPP) and seasonality of carbon gains. Both functional attributes are associated with intermediate ecosystem services *sensu* Fisher et al. (2009). Deforestation was estimated based on photointerpretation of Landsat imagery. The seasonal dynamics of the MODIS satellite Enhanced Vegetation Index (EVI) was used to calculate the EVI annual mean as a surrogate of ANPP, and the EVI seasonal coefficient of variation as an indicator of seasonal variability of carbon gains. The 2000–2007 period showed a high rate of land clearing: 5.9% of NW Argentina (1,757,600 ha) was cleared for agriculture and ranching, corresponding to an annual rate of 1.15%. Dry forests experienced the highest rate and humid forests the lowest. Though land clearing for agriculture and ranching had relatively small impacts on total annual ANPP, once deforested, parcels significantly became more seasonal than the natural vegetation replaced. Such increase in seasonality is associated with a reduction of photosynthetic activity during a portion of the year (fallow). Direct consequences of this reduction can be expected on several ecosystem services such as erosion control and water regulation, due to greater exposure of bare soil, and biodiversity, due to the loss or decline in habitat quality and the decrease of green biomass availability for primary consumers during fallow. Land clearing also increased the magnitude of inter-annual differences in C gains, suggesting a greater buffer capacity against climate fluctuations of natural vegetation compared to croplands.

1. Introduction

Land clearing for agriculture and cattle ranching involves the removal of different types of habitats, including forests, savannas, grasslands, and wetlands. The global rate of deforestation during the last decade was 0.18% (FAO, 2009), especially concentrated on the tropical and subtropical regions across South America (0.50%), Africa (0.62%), and Southeast Asia (1.30%) (FAO, 2009). In addition, regions like Latin America are also experiencing acceleration in the loss rate of natural vegetation, i.e. the 0.51% annual loss rate observed from 2000 to 2005 was 10% greater than during the 1990–2000 decade. As Gasparri et al. (2008) and Grau and Aide (2008) pointed out for South America, land clearing impacted fundamentally on three ecoregions: the Brazilian Cerrado (Morton et al., 2006), the Chiquitanos forests in Bolivia (Steininger et al., 2001), and the Gran Chaco in Bolivia, Paraguay, and Argentina (Zak et al., 2004; Grau et al., 2005a,b; Boletta et al., 2006). In the Gran Chaco ecoregion, large areas of subhumid forests were transformed into croplands and pastures of exotic C4 grasses (Hoekstra et al., 2005). Croplands were almost entirely devoted to soybean production to be exported to the European Union and China (Dros, 2004). The Argentine portion of the Gran Chaco ecoregion has been particularly affected with greater deforestation rates than the continental and world averages (0.82% per year in Argentina, 0.51% for South America and 0.2% globally, FAO, 2009; UMSEF, 2007).

A major concern related to natural vegetation conversion into croplands is the change in ecosystem services (ES) provision (Dirzo and Raven, 2003; MEA, 2005). ES have been defined in different ways and, depending on the definition, we can find different classes of ES (Fisher et al., 2009). On the one hand, the Millennium Ecosystem Assessment (MEA, 2005) definition states that ES are the benefits that people obtain from ecosystems. The MEA definition, and other related ones (Costanza et al., 1998; Daily, 1997), consider subjective and cultural elements outside the ecological systems to define the benefits in the characterization of the level of ES provision. The MEA classifies ES into provisioning ES, regulating ES,

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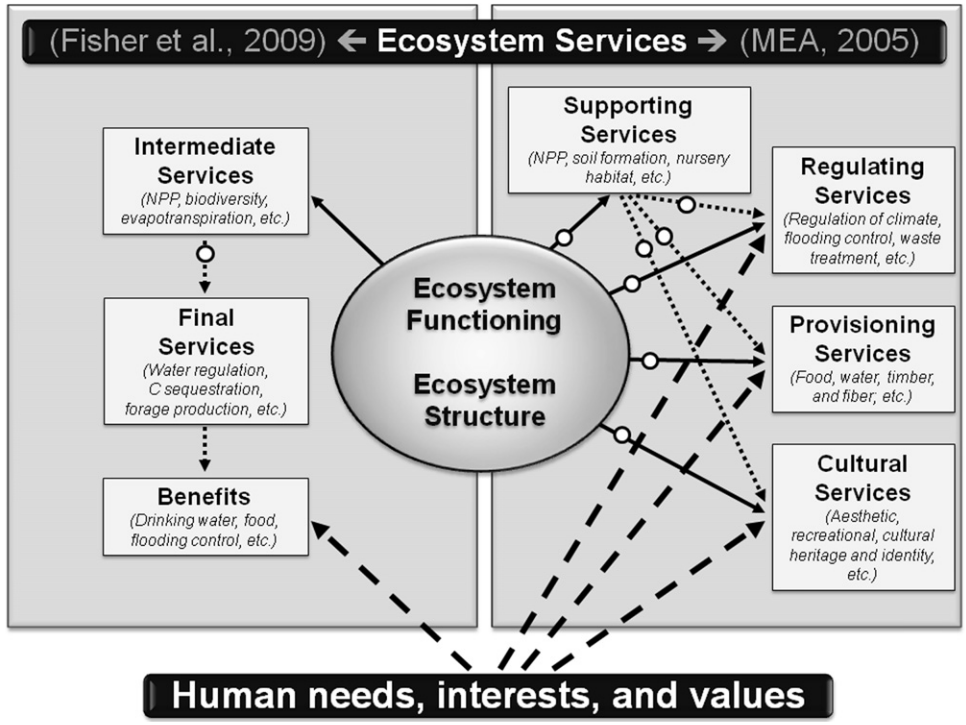


Fig. 1. Main concepts related to the classification schemes of ecosystem services adopted by MEA (2005) and developed by Fisher et al. (2009). Black arrows indicate the relationship between the different categories of ES and the structure and functioning of ecosystems. Such relationship is defined in terms of production functions (circles). Dotted lines represent the relationship between ES categories. Broken lines represent the influence of human needs, interests, and values on the definition of ES and benefits in the two classification schemes.

cultural ES, and supporting ES (Fig. 1). In the MEA scheme, the level of ES provision, regulation, or support is not only linked to basic aspects of ecosystem functioning (e.g. ecosystem exchanges of matter and energy, Virginia and Wall, 2001), but also to the societal context of values, interests, and needs.

On the other hand, Boyd and Banzhaf (2007) referred to ES as the ecological components directly consumed or enjoyed to produce human well-being, without considering the subjective and cultural context. From this perspective, Fisher et al. (2009) defined ES as the aspects of ecosystems utilized (actively or passively) to produce human well-being. We based our analysis on this definition. Fisher et al. (2009) proposed an ES classification scheme where ecosystem functioning and structure are considered "intermediate" services, which in turn determine "final" services (Fig. 1). Several "intermediate" services (e.g. primary production or species composition) may determine the level of provision of a "final" service (e.g. forage production or C sequestration). The link between ecosystem functioning and structure (intermediate services) and final services are defined by "production functions" (Fig. 1). Such production functions are well defined for final ES with market values, such as grain production, where yields are defined by a number of biophysical (water and nutrient availability, temperature, etc.) and management factors (sowing date, cultural practices, etc.). The definition of production functions for final ES (e.g. C sequestration) from intermediate ES (e.g. Net Primary Production, vegetation structure, or soil characteristics) has been identified as an important step to incorporate the ES idea into decision making processes (Laterra and Jobbágy, 2011).

Tradeoffs between ES lead to increases in the level of provision of some ES (e.g. food production) and the reduction in others (e.g. soil protection, water regulation, C sequestration, etc.) (de Groot et al., 2010). Changes in the provision of final ES are mediated by structural and functional changes (intermediate services), such as biodiversity losses and changes in C and water dynamics (Fisher et al., 2009; Guerschman et al., 2003; Guerschman, 2005; Nosetto et al., 2005; Jackson et al., 2005). Hence, to define "impact functions", it would be necessary to identify the main disturbance and stress factors and quantify their effects, for instance, how the level of an ES (e.g. C sequestration) changes with a particular stress or disturbance (e.g. deforested area).

C gains or Net Primary Production (NPP) is one of the most integrative descriptors of ecosystem functioning (McNaughton et al., 1989; Virginia and Wall, 2001). In addition, as an intermediate service (*sensu* Fisher et al., 2009), NPP is a key determinant of several final ES, from the production of commodities to C sequestration. Furthermore, given the same annual C gain, a more even distribution of NPP throughout the year (low seasonality, i.e. low intra-annual coefficient of variation of NPP) has direct positive effects on final ES, such as increases in N retention (Vitousek and Reiners, 1975), reductions of soil losses and runoff, and greater stability of green biomass availability for primary consumers. Annual NPP has also been linked to the economic value of ES at the biome level (Costanza et al., 1998). The carbon gain dynamics has an additional advantage to characterize the level of Intermediate ES provision: NPP can be monitored from remotely sensed data (Running et al., 2000). Satellite images are extensively used to derive spatially continuous estimates of NPP over large areas and with a high temporal frequency, avoiding the use of protocols to inter- or extrapolate point measurements (see i.e. Kerr and Ostrowsky, 2003; Pettorelli et al., 2005). The most widely used approach to characterize carbon gains and ecosystem functioning from satellite data has been the use of the seasonal curves of spectral vegetation indices (VI) such as the Normalized Difference Vegetation Index (NDVI) or the Enhanced Vegetation Index (EVI). These indices are linear estimators of the fraction of photosynthetically active radiation that is absorbed by green tissues (Sellers et al., 1992) and, hence, a key determinant in primary production models (Monteith, 1981; Running et al., 2000). Empirical relationships between vegetation indices and NPP are also well documented in the literature (see e.g. Running et al., 2000; Paruelo et al., 1997; Piñeiro et al., 2006). Two attributes derived from the seasonal dynamics of VIs capture most of the variance in C gain dynamics across vegetation types: the VI annual mean (an estimate of total C gains) and the Coefficient of Variation of the VI seasonal values (a descriptor of the seasonality of C gains) (Paruelo and Lauenroth, 1998; Paruelo et al., 2001; Pettorelli et al., 2005; Alcaraz-Segura et al., 2006). These two ecosystem functional attributes (EFA), can be interpreted (*sensu* Fisher et al., 2009) as intermediate ES related to C gain dynamics and have been widely used to characterize ecosystem functioning and to evaluate the effects of land-use changes on it (Paruelo and Lauenroth, 1998; Paruelo et al., 2001; Guerschman et al., 2003; Roldán et al., 2010).

The effects of land clearing on ecosystem functional attributes (EFA), like primary production and seasonality of carbon gains, can be assessed using both temporal and spatial approaches. The temporal approach requires a comparison of EFA of an area before and after land clearing. The spatial approach is based on the comparison of cleared lands against nearby forested areas at a given time. For instance, protected areas have been frequently proposed as reference areas (Schonewald-Cox, 1988; Stoms and Hargrove, 2000; Cridland and Fitzgerald, 2001; Garbulsky and Paruelo, 2004; Paruelo et al., 2005; Alcaraz-Segura et al., 2008, 2009a,b). The "space × time" approach, has been extensively used in environmental sciences based on the assumption that it is possible to identify both "baseline conditions" and "reference areas". Both temporal and spatial approaches have shortcomings. In the first case, to identify reference areas that correspond to the same vegetation unit and have similar environmental conditions (soil type) can be difficult. In the second one, the baseline environmental conditions (mainly climatic) may change through time.

Linked to the foregoing, we propose the following guiding hypotheses:

(a) Based on the general correspondence between structure complexity and ecosystem functioning (Odum, 1969), we postulate that the greater the structural difference between the vegetation being replaced and the crops introduced in the cleared land, the greater the functional changes. From this hypothesis, we predict that the greatest changes in ecosystem functioning will occur when humid forests are replaced by annual herbaceous croplands.
(b) Transformation of natural vegetation into agriculture not only produces a change on the magnitude of the functional attributes, but it also reduces their inter-annual stability. Our prediction is that the inter-annual coefficient of variation and year-to-year anomalies of the functional attributes will be greater in cleared than in non-cleared plots.
(c) Natural vegetation, a more diverse system than croplands in terms of species, plant functional types, and interactions, has greater capacity than croplands to buffer the impacts of inter-annual fluctuations of precipitation on functional attributes. We predict from this hypothesis that inter-annual anomalies of annual precipitation will generate greater anomalies of carbon gains on cleared lands than on natural areas.

Based on these hypotheses, our objectives were:

(1) To quantify the area of natural vegetation transformed annually into croplands and pastures (land clearing) in NW Argentina during the 2000–2007 period.
(2) To evaluate the effect of land clearing for agriculture on two variables of ecosystem functioning derived from the seasonal dynamics of the Enhanced Vegetation Index: the annual mean and the seasonal coefficient of variation across four vegetation types, going from humid forests (in the Yungas ecoregion) to dry forests, shrublands, and grasslands (in the Gran Chaco ecoregion).
(3) To analyze the difference in the functional attributes response to inter-annual fluctuations of precipitation between croplands and natural vegetation.

2. Materials and methods

Our study area of NW Argentina (Jujuy, Salta, Catamarca, Tucumán, and Santiago del Estero provinces) comprises the entire Argentine portion of the Yungas ecoregion (humid forests) and 35% of the Argentine portion of the Gran Chaco ecoregion (dry forests, shrublands, and grasslands) (Cabrera, 1976) (Fig. 2). The whole area was included within the subtropical belt of South America. Traditionally, natives and settlers practiced subsistence cattle-raising, while during the last decades the area has experienced a rapid and extensive clearing of natural vegetation for market agriculture (mainly soybean and corn) and cattle ranching (mainly pasture) (Grau et al., 2005a,b; Gasparri et al., 2008). Two main factors drove such a vast land clearing process (the largest in Argentine history): (1) the increase in the international demand and prizes of soybean, and (2) a 20–30% increase in precipitation (Boletta et al., 2006; Gasparri and Grau, 2006; Zak et al., 2004). Additional factors included technological changes (the widespread use of the transgenic "Round-up ready" (RR) soybean cultivars and no-till systems) and macroeconomic changes in Argentina (changes in currency exchange rates since 2001–2002).

To quantify the annual surface of natural vegetation transformed into agriculture and ranching, we developed a spatial explicit database of individual plots cleared every year within the 2000–2007 period. For this, we used an annual time-series of summer Landsat 5 and 7 imagery for NW Argentina provided by CONAE (Comisión Nacional de Actividades Espaciales, Argentina). The database was constructed by digitizing the agricultural plots detected by interpretation of image mosaics (RGB band combination: 4-5-3) at a 1:75,000 scale. Each agricultural plot was characterized in the database by the clearing year, vegetation type being replaced, and whether it was irrigated or not. To monitor global deforestation (FAO, 2009), from the annual surface transformed into croplands every year, we estimated the annual rate of change "q", according to the Food and Agriculture Organization (FAO, 1995):

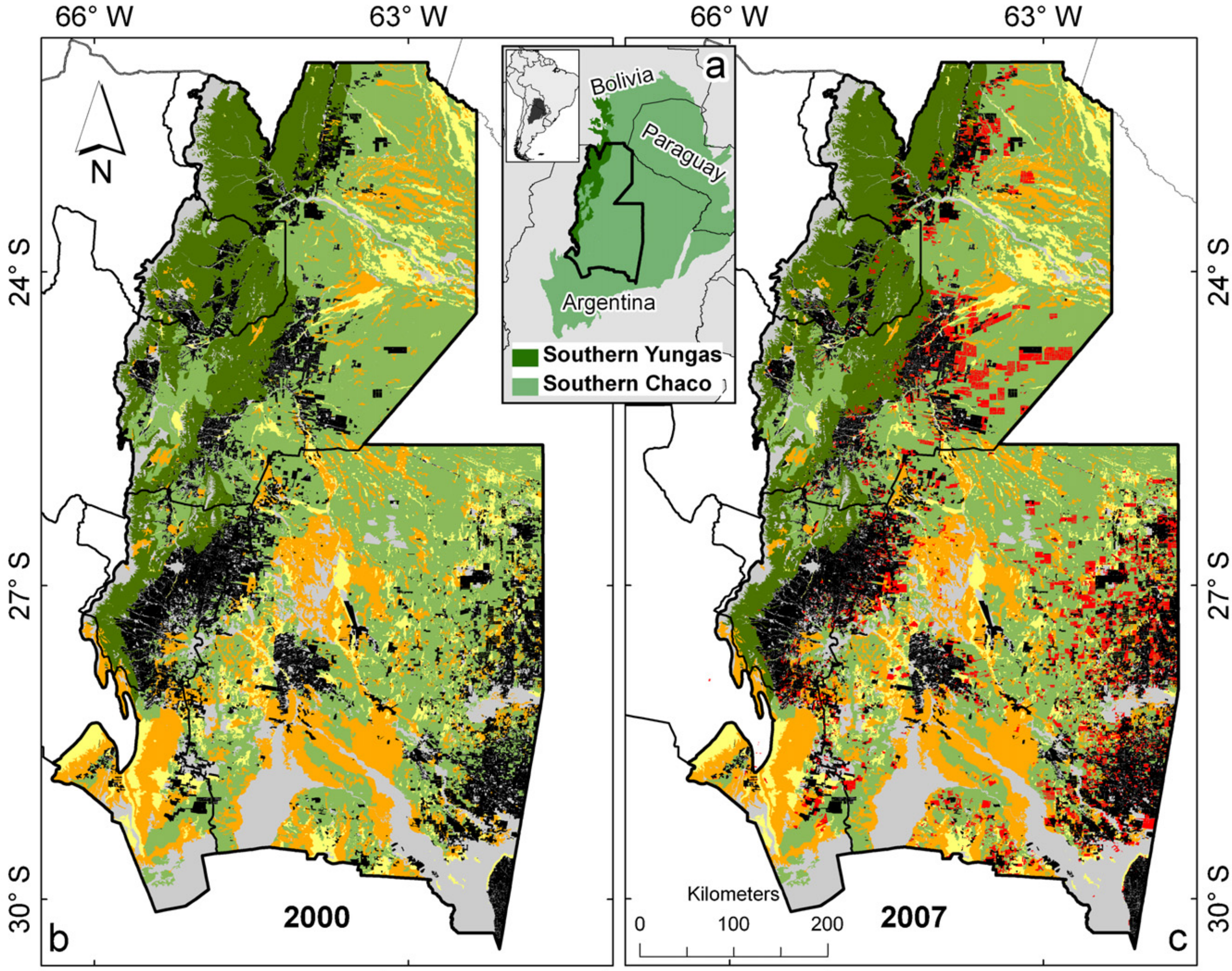
$$q = (A2/A1)^{1/(t2-t1)} - 1$$

where "q" is the Annual Rate of Change in percentage, and $A1$ and $A2$ represent the areas of natural habitats at dates $t1$ and $t2$ respectively.

To characterize ecosystem functioning we used a surrogate of the carbon gain dynamics, the Enhanced Vegetation Index (EVI) (Huete et al., 2002). The EVI is calculated as follows:

$$\mathrm{EVI} = 2.5 \times \frac{\mathrm{IR} - R}{\mathrm{IR} + C1 \times R - C2 \times B + L}$$

where B, R, and IR express atmospherically corrected surface reflectance in the blue, red, and near infrared wavelengths respectively; L (=1) is a correction factor that takes into account the background soil influence; the $C1$ (=6) and $C2$ (=7.5) coefficients consider the presence of aerosols using the blue band to correct the red reflectance band. We used a time-series of MODIS-Terra satellite images (MOD13Q1 product) from 2000 to 2007, with a temporal resolution of 16 days and a pixel size of 230 m × 230 m. The vegetation index quality information was used to filter out those values influenced by clouds, cloud shadows, and aerosols. For each hydrological year (October–September) of the 2000–2007



d	2000		2007		Δ 2000 - 2007		New croplands
	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)
Humid forests	13.0	3,864,000	12.8	3,786,500	-0.3	-77,500	-4.4
Dry forests	38.2	11,347,300	34.0	10,090,800	-4.2	-1,256,500	-71.5
Shrublands	15.1	4,496,000	14.4	4,281,400	-0.7	-214,600	-12.2
Grasslands	7.8	2,322,300	7.3	2,155,800	-0.6	-166,500	-9.5
Other land covers	10.4	3,079,400	10.2	3,036,900	-0.1	-42,500	-2.4
Croplands	15.4	4,571,700	15.4	4,571,700	-	-	-
New croplands	-	-	5.9	1,757,600	5.9	1,757,600	100.0
Total	100	29,680,700	100	29,680,700			

Fig. 2. (a) Study area showing the extension of the Yungas and Chaco ecoregions in NW Argentina. The land-cover maps of the year 2000 (b) and 2007 (c) show the land clearing for agriculture and ranching experienced in the region (SAyDS, 2007a,b,c). The bottom inset (d) shows the percentage of the study region occupied by each land cover and the area cleared from 2000 to 2007.

period, we calculated the EVI annual mean (EVI_mean) as a surrogate of ANPP, and the EVI seasonal coefficient of variation (EVI_sCV) as an indicator of the seasonal variability (Pettorelli et al., 2005).

Changes in ecosystem functional attributes induced by transformation of natural vegetation into croplands were evaluated by comparing paired sites of rainfed agricultural sites (either annual crops or pastures) and their surrounding natural vegetation. From all the agricultural sites photo-interpreted in the study area (over 100,000 agricultural parcels that occupies 6.7 millions of hectares), paired sites were only eligible when the agricultural site was large enough to contain five or more pure MODIS pixels, and there also existed five or more pure pixels of natural vegetation within a distance of 1500 m from the site edge. The 1500 m restriction was imposed to minimize spatial variation in environmental factors such as soil or climate conditions since the Moran's I correlograms (Legendre and Legendre, 1998) of the functional attributes in natural areas maintained high (Moran's $I > 0.5$) significant (Z-value >5; p-value < 0.05) spatial autocorrelation up to this distance. Pixels were considered as pure when more than 95% of their area corresponded to a single vegetation type (Dormann et al., 2007). Paired sites were only eligible when we could select the same number of pixels inside and outside the cleared plot. Then, for each paired site, the spatial mean of EVI_mean and EVI_sCV for the cleared plot and for the paired natural vegetation was calculated from all pixels inside and outside the plot respectively. The process to identify paired sites was repeated for each year (2001–2007) using the digital maps of cleared-land of NW Argentina developed ad hoc (e.g. Fig. 2b and c). This process produced a total number of 27,367 paired sites for the 2001–2007 period (seven years with: 3591; 3614; 3637; 3762; 4221; 4221; 4321 paired sites from 2001 to 2007 respectively).

During the selection of paired sites, we also recorded the vegetation type to test for an effect of vegetation structure (increasing structural complexity from grasslands to forests) on the impact that land clearing had on the functional attributes. The vegetation maps were obtained by reclassifying the First Inventory of Native Forests of Argentina (SAyDS, 2007a,b,c) into five categories: humid forest, dry forest, shrubland, grassland, and other land-covers (Fig. 2b and c).

From the complete set of paired sites, we randomly selected subsets that fulfilled two criteria: they should be independent in time, so only one of the available years was randomly chosen, and in space, so they were far enough from each other. The minimum distance between paired sites was chosen when the Moran's I correlograms for the analyzed variable started to show absence of significant spatial autocorrelation (p-value < 0.01). We also determined the minimum sample size of the subsets necessary to capture most of the variance in the data for each vegetation type and variable. For this, we calculated the increase in the cumulative variance when a new paired site was included in the sample. We stopped when the increase in variance was lower than 5%. Table 1 summarizes the subsets of the studied variables, and the number and characteristics of the samples based on the above criteria.

To explore the effects of land clearing on ecosystem functional attributes (EFA), we first compared frequency histograms of EVI_mean and EVI_sCV between cleared plots and the paired natural vegetation for each vegetation type. To build the histograms, we extracted 1000 random subsets of paired sites and calculated the mean of each random subset. The subset sample size is specified in Table 1 for each variable and vegetation type. 1000 runs were necessary to obtain normal distributions. Then, we compared the differences between the histograms of the natural vegetation and cleared lands by performing one-tailed Student's paired t-tests between their means.

To evaluate whether there existed significant differences in the effect of land clearing on the EFA across different vegetation structures, we first calculated the relative differences in EVI_mean and EVI_sCV between natural vegetation and cleared plots ([natural − cleared]/natural) for all paired sites. Then, we extracted 1000 random subsets of paired sites and calculated the mean of the relative differences for each random subset. The subset sample size is specified in Table 1 for each variable and vegetation type. 1000 runs were necessary to obtain normal distributions. Differences among vegetation types were evaluated by running ANOVAs on the 1000 random subsets. Comparisons between vegetation structures were based on the Sheffe's S procedure, which provides a confidence level for comparisons of means among all vegetation types and it is conservative for comparisons of simple differences of pairs.

To evaluate whether land clearing reduced the inter-annual stability of EVI_mean and EVI_sCV, we only used those sites that had seven complete years of data (i.e., from the initial 6108 sites, only 2338 had 7 years of data). First, we calculated the inter-annual coefficient of variation of EVI_mean and EVI_sCV for cleared plots and paired natural vegetation. Then, we proceeded as in the previous analysis by selecting 1000 subsets to run the ANOVAs. Comparisons between cleared plots and across vegetation types were also based on Sheffe's S procedure.

To evaluate whether natural vegetation has greater capacity than croplands to buffer the impacts that inter-annual fluctuations of precipitation have on C gains (EVI_mean), we evaluated the relationship between the inter-annual anomalies in precipitation and EVI_mean. Monthly precipitation data were obtained from the Tropical Rainfall Measuring Mission (TRMM) archive with a spatial resolution of $0.25^\circ \times 0.25^\circ$ (Product 3B43, V6), distributed by NASA's Goddard Earth Sciences (GES) Data and Information Services Center. Anomalies were calculated as the relative deviation of each hydrological year (from October to September) from the long-term mean (2000–2007 period), as follows: (long term mean − particular year)/(long term mean) × 100. For all paired sites that had seven complete years of data (2338 sites), we estimated the slope and Y intercept of the relationship between the anomalies in precipitation and EVI_mean. We calculated the spatial autocorrelation (see explanation above for Moran's I correlograms) for the slopes and we randomly sampled paired sites with a spatial restriction of 8 km (distance from where the correlograms started to show absence of significant spatial autocorrelation, p-value > 0.01). We ended up with 680 parameter estimates of the regression between the anomalies in EVI_mean and precipitation. We finally calculated the average of the slopes and Y intercepts and compared the differences between natural vegetation and cleared lands by performing one-tailed Student's t-tests.

3. Results

Land clearing for agriculture and ranching transformed 1,757,600 ha of natural vegetation between 2000 (Fig. 2b) and 2007 (Fig. 2c) in NW Argentina, a 5.9% of the region. The greatest relative loss of natural habitats was observed in dry forests (11.1% of their area), followed by grasslands (7.2%), shrublands (4.8%), and humid forests (2.0%) (Fig. 2d). FAO's annual rate of change "q" was −1.15% in the study area, being greater in dry forests (−1.63%) than in grasslands (−1.00%), shrublands (−0.68%), and humid forests (−0.20%).

The change in ecosystem functional attributes (EFA) due to land clearing varied across the vegetation types that were replaced (Fig. 3). In all cases, the effect of land clearing was greater on seasonality than on the total amount of C fixed. For both EVI_mean and EVI_sCV, absolute differences between natural and cleared land increased from grasslands to humid forests, following a gradient of increasing biomass and structural complexity. In all vegetation types (Fig. 3), histograms of EFA showed greater kurtosis in natural vegetation than in cleared land, particularly in the histograms of the seasonal coefficient of variation (EVI_sCV).

The relative changes in EVI_mean and EVI_sCV due to land clearing ([natural − cleared]/natural) also differed among vegetation types, being always greater on seasonality than on the total amount of C fixed (Fig. 4). The relative impact of land clearing on EVI_mean increased along the structural gradient from grasslands to humid forests, being low and similar in grasslands and shrublands but significantly greater in forests; and 3.4 times greater in humid than in dry forests (Fig. 4a). Land clearing significantly increased seasonality of carbon gains (EVI_sCV). Dry forests showed the greatest increases in seasonality and grasslands the lowest (Fig. 4b). On average, land clearing produced a reduction of EVI_mean spatial variability of 24% (the spatial coefficient of variation of EVI_mean

Table 1

Biological meaning, number of records in the initial dataset, sample sizes of random subsets, and spatial restriction to avoid spatial autocorrelation (when Moran's *I* correlograms started to show absence of significant spatial autocorrelation, p-value < 0.01) for the variables used in each analysis (or figure).

Variable	Meaning	Number of records in the initial dataset	Sample size (n) of the random subsets	Minimum distance among sites	Figures
EVI_mean	Enhanced Vegetation Index (EVI) annual mean, as a surrogate of primary production	27,367 paired sites (natural versus cleared).	10 for each vegetation type	60 km	Fig. 2
EVI_sCV	EVI seasonal coefficient of variation, describing seasonal variability of carbon gains	27,367 paired sites (natural versus cleared).	10 for each vegetation type	60 km	Fig. 2
EVI_mean Relative difference (%)	Relative differences in EVI_mean between natural and cleared situations ([natural – cleared]/natural)	27,367 relative differences	50	6.5 km	Fig. 3
EVI_sCV Relative difference (%)	Relative differences in EVI_sCV between natural and cleared situations ([natural – cleared]/natural)	27,367 relative differences	50	6.5 km	Fig. 3
Inter-annual CV of EVI_mean	Inter-annual coefficient of variation of EVI_mean, as an indicator of inter-annual variability of primary production	2338 (paired sites that have 7 years of observations)	50	12.5 km	Fig. 4
Inter-annual CV of EVI_sCV	Inter-annual coefficient of variation of EVI_sCV, as an indicator of inter-annual variability of seasonality	2338 (paired sites that have 7 years of observations)	50	12.5 km	Fig. 4
EVI_mean Anomaly (%)	Relative difference between the EVI_mean of each year and the 7-year average ([long term mean – particular year]/[long term mean])	2338 (paired sites that have 7 years of observations)	630	8 km	Fig. 5
Precipitation Anomaly (%)	Relative difference between the precipitation of each year and the 7-year average ([long term mean – particular year]/[long term mean])	2338 (paired sites that have 7 years of observations)	630	8 km	Fig. 5
Intercept	*Y*-intercept parameter of the linear regression between Precipitation Anomaly (%) and EVI_mean Anomaly (%)	2338 (paired sites that have 7 years of observations)	630	8 km	Fig. 5
Slope	Slope parameter of the linear regression between Precipitation Anomaly (%) and EVI_mean Anomaly (%)	2338 (paired sites that have 7 years of observations)	630	8 km	Fig. 5

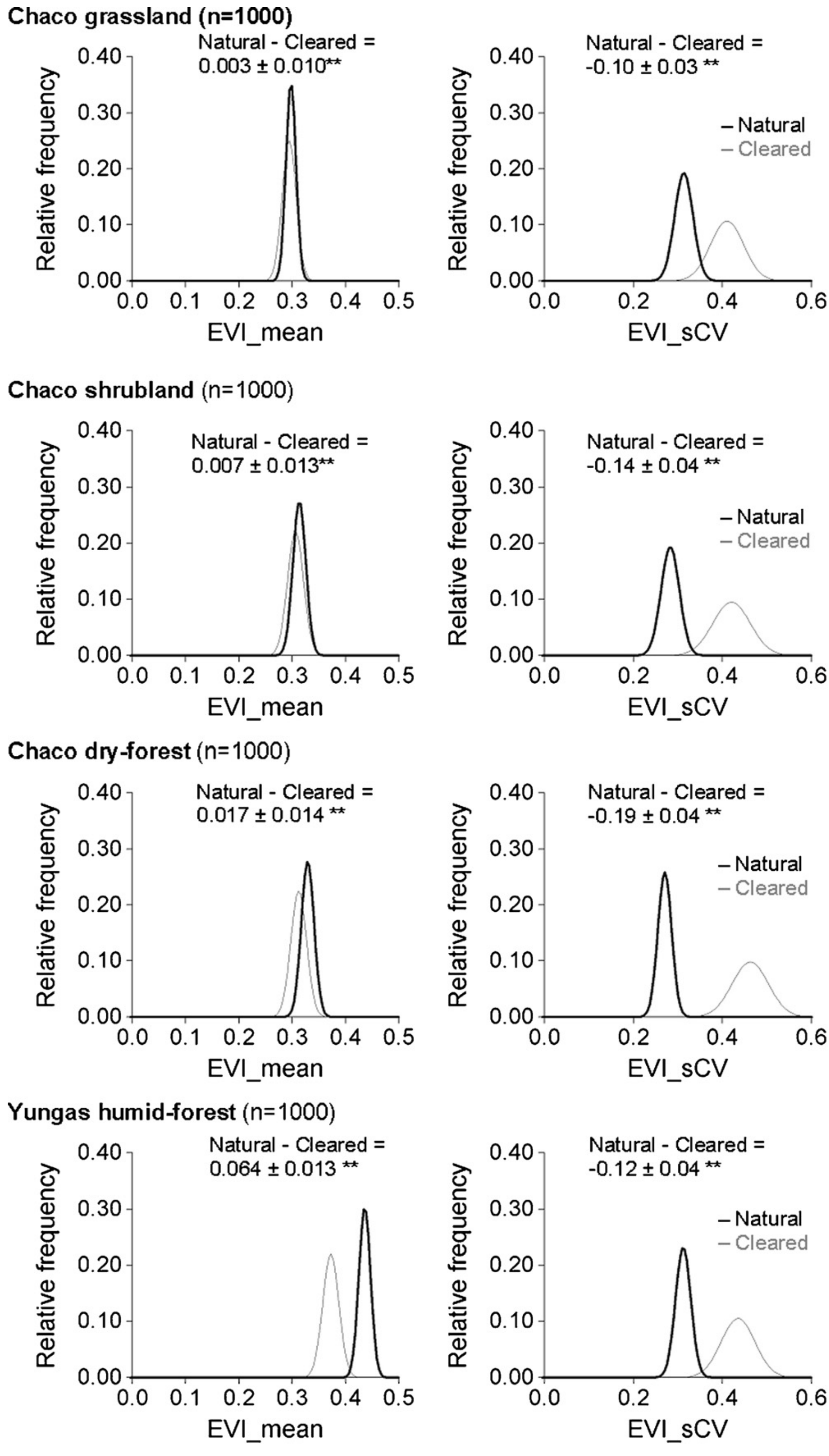


Fig. 3. Changes in the Enhanced Vegetation Index annual mean (EVI_mean) and seasonal coefficient of variation (EVI_sCV) due to land clearing of natural vegetation for agriculture and ranching across different vegetation types in the Chaco and Yungas ecoregions. To build the histograms, we extracted 1000 random subsets of 10 paired sites each (cleared plots versus natural vegetation within a 1500 m buffer around the cleared plots) and calculated the mean of each random subset. The minimum distance among the 10 sites of each random subset was 60 km to avoid spatial autocorrelation (when Moran's *I* correlograms showed absence of significant spatial autocorrelation, p-value < 0.01). 1000 runs were necessary to approximate to normal distributions. **Significant differences between the means were found using one tailed t-tests (p-value < 0.0001, $n = 1000$).

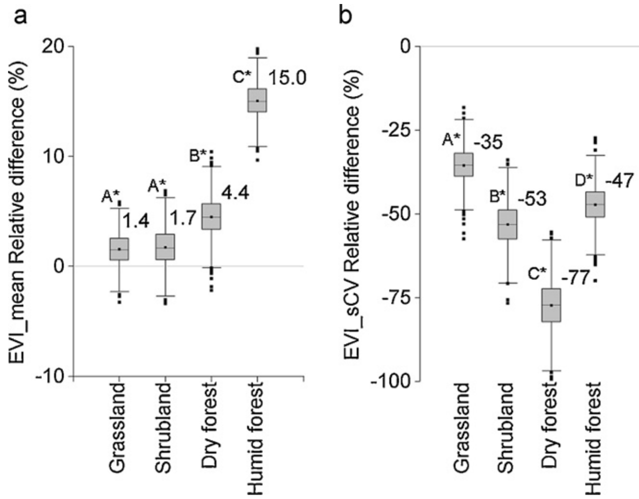


Fig. 4. Relative change (%) of the Enhanced Vegetation Index annual mean (EVI_mean) (a) and seasonal coefficient of variation (EVI_sCV) (b) due to land clearing of natural vegetation for agriculture and ranching across four different vegetation types in the Chaco and Yungas ecoregions. The *Y* axis represents the relative difference between natural vegetation and cleared plots ((Natural − Cleared)/Natural × 100) in 1000 random subsets of 50 paired sites each (cleared plots versus natural vegetation within a 1500 m buffer around the cleared plots). The minimum distance among the 50 sites of each random subset was 6.5 km to avoid spatial autocorrelation (when Moran's *I* correlograms showed absence of significant spatial autocorrelation, *p*-value < 0.01). 1000 runs were necessary to approximate to normal distributions. Different letters indicate significant differences in the ANOVA (*p*-value < 0.05; Sheffe's test; *n* = 1000). *Indicates significantly different from zero (*p*-value < 0.001; *t*-test; *n* = 1000). The bottom and top of the boxes are the 25th and 75th percentiles respectively; the point and the band near the middle of the box are the mean and the median respectively; the bottom and top whiskers represent the 5th and 95th percentiles respectively; external points are extreme values.

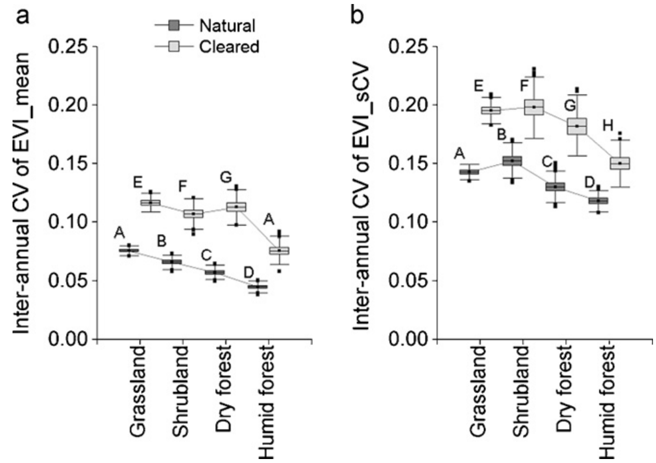


Fig. 5. Increase in the inter-annual variability of the Enhanced Vegetation Index annual mean (EVI_mean) (a) and seasonal coefficient of variation (EVI_sCV) (b) due to land clearing of natural vegetation for agriculture and ranching across four different vegetation types in the Chaco and Yungas ecoregions. The *Y* axis represents the inter-annual coefficient of variation (inter-annual standard deviation/mean calculated from seven years of observations, 2001–2007) of 1000 random subsets of 50 paired sites each (cleared plots versus natural vegetation within a 1500 m buffer around the cleared plots). The minimum distance among the 50 sites of each random subset was 12.5 km to avoid spatial autocorrelation (when Moran's *I* correlograms showed absence of significant spatial autocorrelation, *p*-value < 0.01). 1000 runs were necessary to approximate to normal distributions. Different letters indicate significant differences in the ANOVA (*p*-value < 0.001; Sheffe's test; *n* = 1000). The bottom and top of the boxes are the 25th and 75th percentiles respectively; the point and the band near the middle of the box are the mean and the median respectively; the bottom and top whiskers represent the 5th and 95th percentiles respectively; external points are extreme values.

(the 7-year average) over 2338 sites is 0.17 for natural areas and 0.13 for cleared plots).

EVI_mean and EVI_sCV showed significantly greater inter-annual variability in cleared lands than in natural vegetation. Inter-annual variability was always greater for the seasonality of carbon gains (EVI_sCV) than for primary production (EVI_mean) (Fig. 5). On average, land clearing produced an increase of inter-annual variability of 69% for EVI_mean, and of 34% for EVI_sCV. In both cases, the greatest increases in inter-annual variability occurred in dry forests and the lowest in grasslands and humid forests.

Both cleared and natural areas were able to buffer the effect of climatic fluctuations of precipitation on carbon gains. In 65% of the cleared plots and 79% of the paired natural vegetation, EVI_mean anomalies were lower than precipitation anomalies. However, as we predicted from hypothesis (c), cleared plots presented greater EVI_mean anomalies (both positive and negative) than natural areas and a significantly higher slope (double on average) of the relationship between precipitation and EVI_mean anomalies (Fig. 6). These results indicate that natural areas have a greater capacity to buffer climatic fluctuations than agricultural fields or pastures.

4. Discussion

The transformation of natural habitats into croplands and pastures observed in the region significantly changed ecosystem functional attributes (EFA) related to Intermediate Ecosystem Services associated with carbon gain dynamics. The increase of seasonality after land clearing for agriculture observed in our study was already reported in temperate grasslands (Paruelo et al., 2001, 2006) and in subtropical humid forests of NE Argentina (Roldán et al., 2010). Our results and evidences from literature suggest that the increase in seasonality is the dominant effect of land clearing

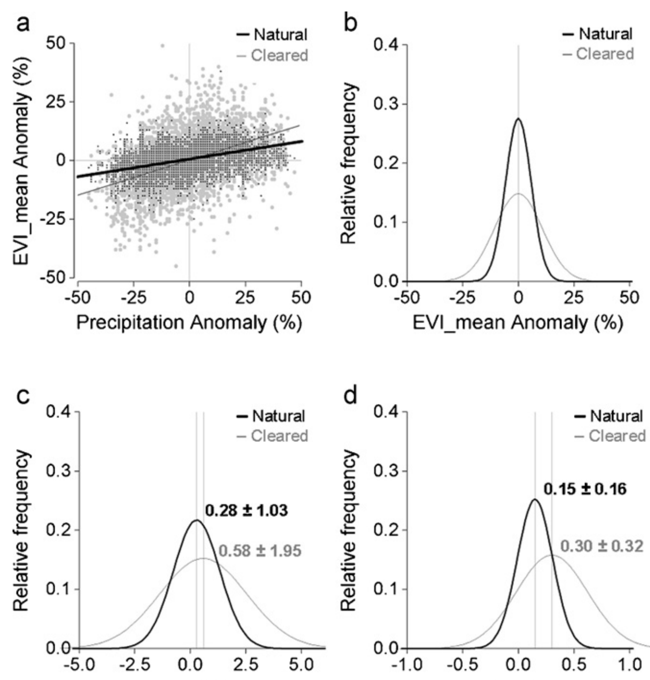


Fig. 6. Differences between cleared plots and natural vegetation in the relationship between the inter-annual anomalies in precipitation and in the EVI_mean (expressed as [long term mean − particular year]/[long term mean] × 100). (a) Relationship between the anomalies along the 2338 paired sites that have seven years of observations from 2001 to 2007. (b) Frequency distributions of EVI_mean anomalies in the 2338 paired sites. Frequency distributions of the *Y*-intercept (c) and the slope (d) of the linear regressions between precipitation anomalies and EVI_mean anomalies for 7 years data (*n* = 7) in a random subset of 630 paired sites (from the initial 2338) sampled with a spatial restriction of 8 km among sites to avoid spatial autocorrelation.

for agriculture regardless the structure of the natural cover being replaced. Such increase results, mainly, from a strong reduction in the minimum values of leaf area index after tillage and during fallow (Guerschman, 2005). Alternatively, total annual C gains can either be increased, maintained, or decreased after land clearing depending on the type of transformation and the vegetation replaced (Paruelo et al., 2001). For instance, Caride et al. (in this issue) found that agricultural managements that included wheat-soybean double cropping had greater C gains than the grasslands replaced, while monocultures of either soybean or maize showed lower C gains.

The magnitude of the impact of land clearing on EFA varied among vegetation types. As we predicted from hypothesis (a), the greatest changes occurred when forests were replaced by annual herbaceous croplands: the greater the structural difference between the areas cleared for agriculture and the vegetation being replaced, the greater the functional changes. Thus, the impact of replacing natural habitats by annual croplands in more structurally complex vegetation types (e.g. forests) would generate greater losses of Intermediate Ecosystem Services related to C gains; not only in absolute terms, but also in relative values (relative to the original value of natural vegetation). Viglizzo and Frank (2006) also found greater impact of land transformation on ecosystem services provision in forests than in grasslands. This has also been observed in economic valuations of ecosystem services, where the greatest losses due to land clearing were observed in forests (Costanza et al., 1998). A rather obvious but interesting result is that the variation among vegetation types in the magnitude of ecosystem functional attributes (EFA) after land clearing results from differences in the EFA value of the natural covers being replaced, since agricultural plots always had a similar level of EFA regardless the original cover. Land clearing, hence, generates a homogenization of the regional landscape in terms of ecosystem functioning at, both, the structural and the functional levels, even across different ecoregions, vegetation types, and precipitation gradients. As hypothesis (b) stated, land clearing for agriculture and ranching not only produced a significant change of EFA, but also increased its inter-annual variability. Our results indicate a greater capacity of natural vegetation than cropped areas to buffer the effects of environmental changes at the functional level. Our quantification of this buffer capacity can be used as an indicator of the resilience of the different systems, a critical descriptor of the system behaviour to face disturbance without collapsing.

A critical point in evaluating the effect of land transformation on ecosystem functioning and ecosystem services provision is the definition of control baseline conditions or control reference sites and whether they refer to time (e.g. a particular year) or space (e.g. a particular plot). This might not only be a technical challenge but also a political issue to define policies for environmental management. On the one hand, both the temporal and the spatial approaches have shortcomings. When comparing the same plot before and after land clearing, the environmental anomalies (e.g. droughts) between years may confuse the effects due to land clearing. Similarly, when comparing in space, there might exist uncertainty whether the cleared lands and the reference or control areas originally corresponded to the same vegetation unit and had similar environmental conditions. On the other hand, it is challenging to find natural areas with similar original environmental conditions than the cleared plots but not subjected to direct human disturbances. In this article, the proximity of reference sites to transformed areas was prioritized, being aware of the pre-existing degree of disturbance due to the practice of subsistence cattle-raising by natives and settlers. National or state parks would provide, of course, a much better description of the non-modified conditions than non-protected wild areas. However, using protected areas may bias the analysis since their extension and spatial distribution may not be representative of the biota, soils, and climate conditions originally present in the transformed lands. Instead, using as reference sites non transformed areas located in the close vicinity of the cleared plots (that maintained a high spatial autocorrelation, so under similar environmental conditions) would minimize this bias. An additional shortcoming of using as reference sites areas nearby agricultural plots is the indirect effect of disturbances related to the activities within the agricultural plots (e.g. trampling, firewood extraction, agrochemical drift). In any case, evaluations based on neighbour sites as reference sites would always provide a conservative estimate of the impact of land clearing on ecosystem functional attributes related to Intermediate Ecosystem Services.

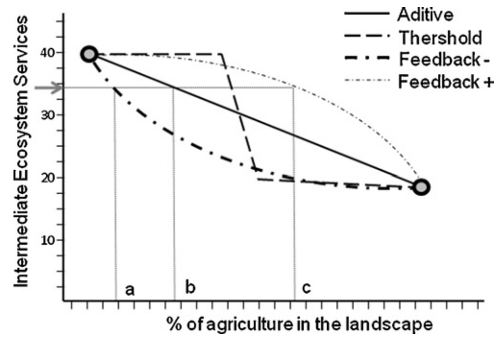


Fig. 7. Hypothetical impact functions of the increase in the agricultural proportion in the landscape on the change in Intermediate Ecosystem Services related to C dynamics (for example the Ecosystem Functional Attributes EVI_mean studied in this article). Circles in the extremes represent the initial and final conditions in our study. The arrow on the *Y* axis indicates a hypothetical level of reduction in Intermediate Ecosystem Services that a local community is willing to tolerate. The letters in the *X* axis show the level of transformation associated to this change in Intermediate Ecosystem Services depending on the shape of the impact functions.

The analyses performed in this study provide the basis to estimate "impact functions" of land clearing. Impact functions may allow one to calculate the mean effect of replacing natural vegetation by agriculture and, even, the variability in time and space of such effect. As we noticed above, in our study the magnitude of the effect differs among vegetation types, which must be considered to define impact functions specific to each vegetation type. The overall impact of land clearing should be observed, though, at the landscape level, and would increase with the spatial extension of the natural habitats removed. Actually, the stress factor (*sensu* Scheffer et al., 2000) will be the simple proportion of the landscape transformed (Fig. 7). To define the actual function that relates the EFAs or the level of provision of Intermediate Ecosystem Services to the area being cleared, studies at the landscape level are needed. As a first approach, an additive effect can be assumed. However, differences in landscape configuration may determine spatial interactions among patches of natural and agricultural plots, leading to non-linear relationships (either positive or negative feedbacks) (Scheffer et al., 2000). A proper definition of these relationships is critical for landscape planning because it allows planners to define the level of landscape transformation based on societal choices (Castro et al., 2011). For example, if the arrow in Fig. 7 indicates the level of change in a Intermediate Ecosystem Service that a local community is willing to tolerate, a linear impact function would allow a medium level of transformation *a*. A non-linear relationship, though, would determine lower or greater levels of transformation, *b* or *c* respectively, depending on the shape of the impact function. In the case of threshold functions, societal decisions are limited to keep the level of transformation within the critical threshold values. Ecosystem functional attributes derived from remotely sensed data are particularly well suited to device such impact functions because they can track changes in Intermediate Ecosystem Services over large areas and at spatial resolutions that include different landscape configurations and structures (e.g. different deforested areas, patch sizes of remnant forests, etc.).

Our analyses focused on ecosystem functional attributes directly linked to Intermediate Ecosystem Services related to C gain dynamics (*sensu* Fisher et al., 2009). Two additional "steps" are needed to derive estimates of goods and services that directly benefit humans. First, to calculate final services (*sensu* Fisher et al., 2009), e.g. water regulation or soil protection. For this, it would be necessary to derive "production functions" (*sensu* Fisher et al., 2009) that yield values for final services (Fig. 1), which would require additional information (e.g. soil types or topography) such as in the model presented by Viglizzo et al. (2011). Second, to estimate direct benefits (e.g. clean water or flood control), it would be needed a detailed characterization of stakeholders, both those playing the role of "affectors" and "enjoyers" (Scheffer et al., 2000). In spite of these needs, evaluating ecosystem functional attributes linked to Intermediate Ecosystem Services, particularly those related to C dynamics, provides a valuable approach since they are both a key piece in the process to calculate final services and a good proxy for benefits. Indeed, Costanza et al. (1998) showed how the economic value of the ecosystem services provided by different biomes was linearly and positively related to Net Primary Production. Once the relationship between land use change and services is known, the consequences of land transformation and management must focus on the total bundle of ecosystem services provided at different spatial scales (Foley et al., 2005; de Groot et al., 2010). This analysis should involve the study of tradeoffs among economic and ecosystem services at different temporal and spatial scales and including stakeholders (Carreño et al., in this issue).

5. Conclusions

Almost 6% of the NW Argentina (1,757,600 ha) was cleared during the 2000–2007 period (at a 1.15% annual rate). The land clearing process for agriculture and ranching occurring in NW Argentina eliminated mainly dry forests and affected key ecosystem functional attributes related to Intermediate Ecosystem Services associated with carbon gain dynamics. Though land-use/land cover changes had relatively small impacts on total annual ANPP, crops and pastures parcels became significantly more seasonal than the natural vegetation replaced. Such increase in seasonality is associated with a reduction of photosynthetic activity during a portion of the year (fallow). Direct consequences of this reduction can be expected on several ecosystem services such as erosion control and water regulation, due to greater exposure of bare soil, and biodiversity, due to the loss or decline in habitat quality and the decrease of green biomass availability for primary consumers during fallow. Land clearing significantly increased inter-annual variability of C gains, suggesting a greater buffer capacity against climate fluctuations of natural vegetation compared to croplands. Our quantification of this buffer capacity can be used as an indicator of the resilience of the different ecosystems, a critical descriptor of the system behaviour to face disturbance without collapsing. The greatest functional changes in the region occurred when forests were replaced by annual herbaceous croplands. Our observations suggest, that the greater the structural difference between the areas cleared for agriculture and the vegetation being replaced, the greater the functional changes. Since the final status is similar across all cleared plots, land clearing tends to generate a homogenization of the regional landscape in terms of ecosystem functioning that operates even across different ecoregions, vegetation types, and precipitation gradients. Our results also provide the basis to estimate "impact functions" of land clearing to calculate the mean effect of replacing natural vegetation by agriculture and, even, the variability in time and space of such effect. As we noticed above, the magnitude of the effect differs among vegetation types, which must be considered to define impact functions specific to each vegetation type.

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