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Assessing the cross-scale impact of 50 years of agricultural transformation in Argentina

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ABSTRACT

Given the increasing number of actors related to farming that make decisions at different scales (plot, farm, region, etc.), knowledge about patterns and processes that behave hierarchically is increasingly needed. This is necessary in countries like Argentina, where cultivation expanded at increasing pace during the last 50 years over an area of 1.47 million km^2. Relying on different sources of existing data, the purpose of this work was to assess the cross-scale dependence of patterns and processes related to the expansion of cultivation in Argentina. The study involved indicators of (i) carbon, nitrogen and phosphorous stocks, (ii) energy productivity, fossil energy consumption, C, N and P balances, water consumption and greenhouse gases fluxes, and (iii) impacts related to pesticide contamination, habitat intervention and soil erosion. Three scales involving (i) regions, (ii) macro-regions and (iii) the whole country were analyzed. Principal Components, Correlation and Regression Analysis were used to identify and quantify meaningful relationships between the different scales. The expansion of annual crops affected C–N–P stocks significantly at the regional scale, whereas it influenced energy and matter flows, and contamination across all scales. This finding explains conflictive responses to land use and management when different scales are considered and shows that scale dependency needs to be considered when their effects on the environment are explored and quantified.

1. Introduction

Agriculture expanded during the last 50 years from the Pampas to NW and NE of Argentina at the expense of natural forests and rangelands (Viglizzo et al., 2011). In parallel, productivity increased in response to the application of external inputs, modern technology and management practices (Satorre, 2005). Several authors studied the impacts of agriculture on the environment at regional scales (Viglizzo et al., 2001, 2011; Rabinovich and Torres, 2004), but there are no works that looked at such impacts across scales. Given the increasing number of agronomists, farmers, policy makers and development agents that make decisions at different scales (plot, farm, region, country), knowledge about patterns and processes that behave hierarchically is increasingly needed (Viglizzo et al., 2005).

Ecologists are often asked to contribute to solutions for broad-scale problems, but scale extrapolation faces various limitations (Miller et al., 2004). Among other reasons, cross-scale studies are justified because knowledge at one scale is normally insufficient to explain the behaviour of processes that occur at other scales. In spite of the fact that 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}, U\$S ha^{-1}), many others (e.g. soil sediment losses, contamination hotspots) that are meaningful only within discrete scales may break their integrity with a switch in the scale (Allen and Holling, 2002). In practice, most studies in agriculture are focused on smaller (e.g., the plot) more than on broader scales (e.g., the landscape or the agro-ecosystem) and this asymmetry may lead to wrong decisions when the study process can not be linearly scaled up or down

Abbreviations: AET, actual evapotranspiration; C, carbon; CSC, carbon stock change; DM, dry matter; EP, energy productivity; ET, evapotranspiration; FEC, fossil energy consumption; GHG, greenhouse gases; HI, habitat intervention; L, milk liters; N, nitrogen; NB, nitrogen balance; NS, nitrogen stock; P, phosphorus; PB, phosphorus balance; PC, pesticide contamination; PCA, principal components analysis; PET, potential evapotranspiration; PS, phosphorus stock; SE, soil erosion; SOC, soil organic carbon; WC, water consumption.

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from one level to another. Not surprisingly, there is a frequent gap between the scale at which most agricultural research is done and the scale at which knowledge outcomes are applied (Dalgaard et al., 2003). Small-scale experiments are usually too small and too short to explain processes that occur at larger spatial and temporal scales (Hobbs, 1998), but broad-scale experiments are normally unviable because of practical constraints such as the number and replication of treatments (Carpenter et al., 1995). 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.

Recently, Sachs et al. (2010) emphasized the need of a global network to monitor the effects of agriculture on the environment across ecological and climatic zones worldwide. They recognized that one major limitation to do this is that agriculture must be assessed at different scales, but the cross-scale comparison of data collected is not always viable because of methodological inconsistencies in the spatial scale at which observations are made. They suggested developing a common set of metrics to collect comparable data at different scales. As such, principal components analysis is suitable for data sets in multiple dimensions finding relevant components, or predominant patterns, across scales and to uncover unknown trends in the data. This contribution is in line with this purpose.

Relying on statistical databases and models, the main objective of this study was to assess the cross-scale dependence of patterns and processes related to the expansion of cultivation in Argentina between the end of the 1950s and the beginning of the 21st century. There is a long history of ecological studies that provided the basis for the study of spatial patterns and processes (Turner, 1989), which is what today differentiates landscape ecology from other branches in the ecological science. However, the explicit effects of spatial patterns on ecological processes are not still well understood. The concept of patterns as constant patches of vegetation was originally presented by Watt (1947), a plant ecologist who assumed the existence of biotic processes that are related to those spatial patterns. In this work, while spatial patterns refer to the dominant vegetation characteristics of the study biomes, processes refer to functional changes that were triggered by agricultural expansion. Thus, this study comprised the analysis of indicators of stocks (CS, NS and PS) that represent pattern conditions, and indicators of processes related to fluxes (EP, FEC, NB, PB, WC and GHG) and impacts (PC, HI and SE). Three scales involving (i) regions, (ii) macro-regions and (iii) the whole country were also analyzed.

One relevant question to be answered was how those changes affected patterns and processes across scales. One initial hypothesis in this study was that the components of the variance in the set of indicators of flux, impact and stock decline as the analysis moves from the smaller to the larger spatial scales. Another hypothesis was that some indicators of flux, impact and stock can maintain their integrity across scales better than others.

2. Materials and methods

2.1. Study area and data sources

The whole study area, which today comprises 63% of the Argentine territory and includes more than 90% of annual and perennial crops, covers around 1.47 million km^2 (Fig. 1) that were divided into fifteen dominant biomes (INDEC, 2004). Biomes include croplands, grasslands, shrublands, and tropical, subtropical, and temperate forests. Most croplands are devoted to soybean (*Glycine max* L.), sunflower (*Helianthus annuus* L.), maize (*Zea mays*), wheat (*Triticum aestivum* L.) and beef (*Bos taurus*) production. Two periods (1956–60 and 2001–05) that represent the traditional and the modern agricultural models were analyzed within a 50-year time span.

Three biomes and various land-use/land-cover types under rainfed conditions were studied: (i) native forest lands; (ii) native grasslands (savannas and shrublands) and cultivated perennial pastures and (iii) croplands (dominant annual crops like wheat, maize, rice, soybean, sunflower and cotton in Chaco). Lineseed was considered during the first and second period only because its importance was negligible during the first decade of the 21st century. Irrigated lands (less than 0.5% in Argentina) were not considered. The study relied on databases that covered 399 political districts reconstructed from the agricultural censuses of 1960 and 2002 (INDEC, 1964, 2004) and the annual survey of the National Secretary of Agriculture (SAGPyA, 2009), which provided data on cultivated areas at the farm level since 1970. Contradictory data on the forest area (INDEC, 1964, 1991, 2004; SAyDS, 2007a,b) were amended through recent assessment of carbon stocks and emissions in forests of Northern Argentine for 1900–2005 (Gasparri et al., 2008). Their estimations were based on forest inventories, deforestation estimates from satellite images and historical data on forest and agriculture. By means of simulation outputs, they calculated the accumulated C emissions due to deforestation during the 105-year studied period.

Cattle productivity was indirectly calculated through equations based on stocking rates. Beef production was estimated using quadratic equations developed by Viglizzo (1982) for calving and fattening areas: $Y = -27.0 + 258.4X - 15.4X^2$ ($R^2 = 0.74$) and $Y = -32.0 + 252.9X - 62.6X^2$ ($R^2 = 0.74$), respectively, where Y is the average beef production (kg ha^{-1} yr^{-1}) and X average stocking rate (animal units ha^{-1} yr^{-1}), both obtained from national agricultural censuses. The animal unit was represented by a bovine of 450 kg live weight. Given that 95% of dairy farms are located in the Pampas (Sanmartino, 2006), milk production figures were assumed to be restricted to that region and where based on the population of dairy cows. Milk production was obtained under grazing conditions during the 1950s and 60s, but supplementation with concentrates was extensively used during the period 2001–05 (Viglizzo et al., 2001). Milk productivity per dairy cow was 1230 L ha^{-1} yr^{-1} for 1956–60 and 5733 L ha^{-1} yr^{-1} for 2001–05, which was the result of dividing total milk production in the Pampas by the total area of dairy farms.

Crop technology was characterized by means of tillage, pesticide, herbicide and fertilizer use based on technical reports (CASAFE, 1997; SENASA, 2004). Tillage was characterized by the proportion of the area cultivated with conventional, reduced, and no-till methods being, in that order, 100–0–0 and 20–30–50 for the two selected periods (Salvador, 2002). Pesticides also varied, being dominated by chlorinated products in 1960, replaced by pyrethroids in 2001–05. In terms of herbicides, various combinations of 2,4-D, 2,4-DB, Piclorane, Atrazine, Trifuraline, Bromoxynil and Glyphosate (SENASA, 2004) were applied across the study time span. For our calculations, we used the doses proposed by manufacturers in all cases. Regarding N and P fertilizers, we assumed that they were used at the farm level keeping a proportion to the national statistics of consumption – 0% and 100% of the recommended doses was respectively applied in 1956–60 and 2001–05 (SENASA, 2004). Genetically modified soybean was the dominant crop since the 1990s, which expanded very quickly at the expense of other crops on already cultivated lands (Satorre, 2005).

2.2. Calculation of stocks, fluxes and impacts

A detailed description of methods applied in this work to calculate stocks, fluxes and impacts can be found in Viglizzo et al. (2011).

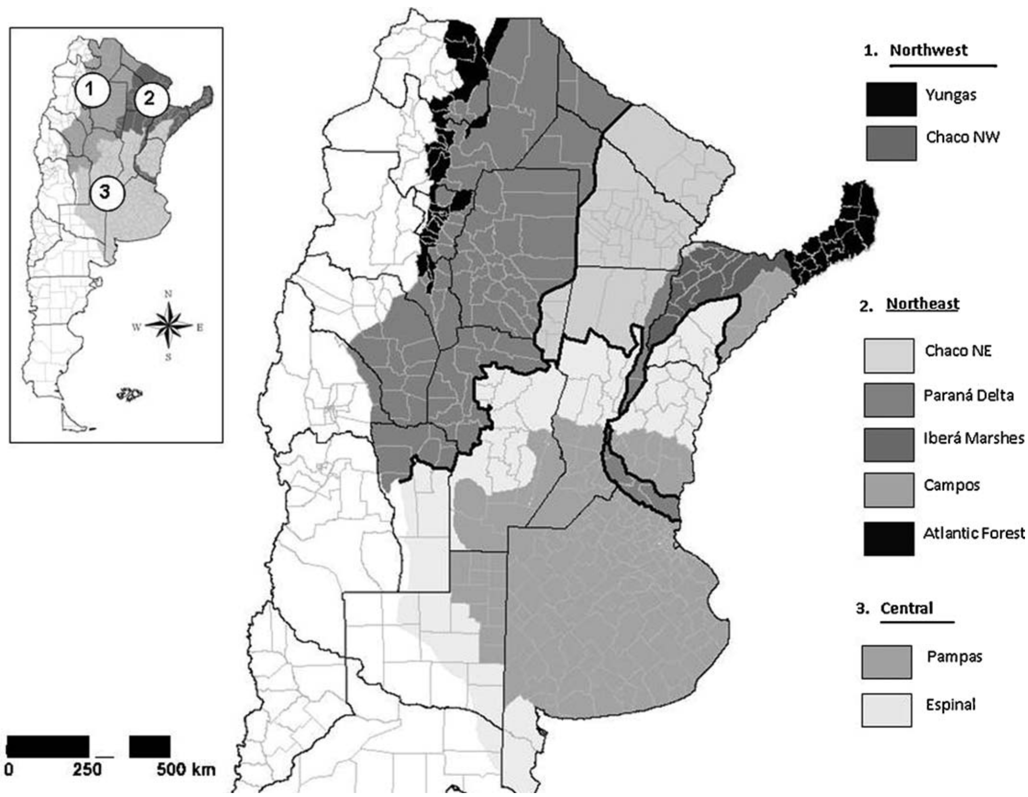

Fig. 1. Location of the study areas in the Argentine territory.

2.2.1. C, N and P stocks

Following IPCC (2006) guidelines, the three biomes mentioned above (forest lands, perennial grasslands and pastures, and rainfed annual crops) that represent different land-use/land-cover types were considered in this study. Carbon stocks estimations included biomass and soil compartments.

In the case of forest lands, three compartments were considered (i) aerial biomass, (ii) an extra fraction of biomass (under-story vegetation, below-ground biomass, biomass in dead wood and litter), and (iii) SOC. A factor of 0.47 was used to convert DM to CS (IPCC, 2006). The initial aboveground stock of biomass C ($\mathrm{Mg\,ha^{-1}}$) in forests for different climate zones (tropical, subtropical and temperate, wet and dry) was estimated from *default* factors of IPCC (2006), which were later cross-checked with those of Gasparri et al. (2008). An extra biomass fraction (estimated as a percentage of the aerial biomass) based on Gasparri et al. (2008) was 0.49, 0.41 and 0.38, respectively, for Chaco, Atlantic and Yungas forests. Because of data lacking, an average of the three systems was applied for the Espinal and Campos forests. Changes in biomass were only estimated for perennial woody crops. On the other hand, in line with IPCC (2006) criteria, biomass stocks in grasslands, pastures, croplands and wetlands were assumed to be under steady-state, with C gains equalling C losses. Thus, for grasslands, pastures, croplands and wetlands, increase in biomass stocks in a period is assumed equal to biomass losses from harvest and/or mortality in that same period. Hence, there is no net accumulation of biomass C stocks.

Regarding SOC, assuming an average soil bulk density of $1.2\,\mathrm{g\,cm^{-3}}$ and 58% C in organic matter (Alvarez et al., 1995; Steinbach and Alvarez, 2006; Gasparri et al., 2008), SOC stocks for the top 30 cm of mineral soil were estimated. Based on IPCC (2006) Tier 1 and 2 methods and survey data (Viglizzo et al., 2011), the initial C stocks in soil ($\mathrm{Mg\,C\,ha^{-1}}$) were set for each soil. Several empirical (Steinbach and Alvarez, 2006) and default values (IPCC, 2006) were used to estimate temporal changes in soil C stocks (see Section 2.2.2). Forest SOC stocks were obtained from Gasparri et al. (2008), and had values of 31, 35, 65 and 56 $\mathrm{Mg\,SOC\,ha^{-1}}$ for Chaco, Atlantic Forest, Yungas Forest and Espinal, correspondingly. A similar value to that of the Atlantic Forest ($35\,\mathrm{Mg\,ha^{-1}}$) was adopted for the forest soils of the Iberá Marshes and Paraná Delta.

Biomass and soil N and P stocks ($\mathrm{Mg\,ha^{-1}}$) were calculated from C stocks and C:N:P stoichiometric values from the literature. Specific N:C ratios (0.02458 ± 0.00256) were used for crop residues (Givens and Moss, 1990), forests (Zak and Pregitzer, 1990; Nadkrmiz and Matelson, 1992; Hooker and Compton, 2003), grasslands (Zak et al., 1990) and pastures (Givens and Moss, 1990; NRC, 1978). According to data from Huan (1996), the N:P mass ratio ($\mathrm{kgP\,kgN^{-1}}$) is 0.1184 for biomass in general. To estimate soil contents of N and P, they were related to SOC content. According to field data from Steinbach and Alvarez (2006) and Galantini and Suñer (2008), a C:N:P relation of 100:11:1 was adopted.

2.2.2. Energy, C, N, P, WC and GHG fluxes

The analysis of FEC and EP (expressed in $\mathrm{MJ\,ha^{-1}\,year^{-1}}$) involved (i) estimates of inputs in the form of fossil energy consumed for the synthesis of pesticides, fertilizers, concentrates, seeds, etc. and (ii) estimates of FEC by agricultural activities (ploughing, harrowing, seeding, spraying, harvesting, water pumping, etc.). On the other hand, outputs were estimated in terms of energy contained in agricultural products (Viglizzo et al., 2003).

Net C fluxes ($\mathrm{Mg\,ha^{-1}\,yr^{-1}}$) were estimated through changes in plant biomass and SOC. Average values for forests biomass change ($\mathrm{Mg\,DM\,ha^{-1}\,yr^{-1}}$) were obtained from default data of IPCC (2006) for tropical, sub-tropical and temperate forests (both natural and cultivated). As IPCC (2006) Tier 1 recommends, it was assumed no annual change in above- and belowground biomass in grasslands/pastures, croplands and wetlands because they are in an approximate steady state: biomass increases in a single year are equal to biomass losses. The SOC annual flow ($\mathrm{Mg\,ha^{-1}\,yr^{-1}}$) was calculated by a simplified methodology proposed by IPCC (2006) Tier 1. To estimate annual changes in SOC stock, land-use data were organized into inventory time periods of 20 years, and a native reference SOC stock value was assigned to each region based on climate and soil type. A land-use factor, a management factor and a C input level factor was assigned to each land use at each time period, except for woodlands, for which no change was assumed in their mineral soil C stocks as recommended by IPCC (2006). Regarding the balance of N and P due to natural events and human activities, various ways of gain and loss were considered. Nitrogen inputs included (i) atmospheric deposition of 0.6 kg N per 100 mm of rainfall (Panigatti and de Hein, 1985), (ii) Fertilizers N (kg ha^{-1}), (iii) biological N fixation (Baethgen, 1992; Brenzoni and Rivero, 1996) by legumes: (70–120 kg ha^{-1} yr^{-1} depending on species), and (iv) input through animal feeds (Viglizzo et al., 2003). Nutrient inputs through animal feeds comprise all nutrients supplied by supplementary feeds coming from outside the study system. In this case, the political district was the study system and therefore it was the only one scale unit considered in calculations. Nitrogen outputs were estimated by (i) product outputs, (ii) N lost through SOC removal and soil erosion and (iii) N emitted as N_2O to atmosphere. To estimate N_2 emission, N_2O was multiplied by 0.68, which is the relative weight of N in the molecule of N_2O. To estimate N_2O emissions, we relied on data from IPCC (2006) as described later in Section 2.2.3. It should be noted that this work undertook a low-resolution analysis of N gains and losses. Given the large variability of soils and climate conditions across the large study area, many details regarding local characteristics of N balance had to be set aside and it was assumed that this inevitably led to some degree of calculus imprecision both, in N gains and losses. This constraint also applies in the P outputs. Phosphorus inputs included (i) fertilizers and (ii) supplementary feeds for cattle, and P outputs comprised losses through (i) products (grain, meat or milk), (ii) SOC and soil erosion losses and (iii) runoff and leaching. The last two were estimated assuming a constant N: P relationship (see Section 2.2.1). Data on N and P concentration in agricultural inputs and outputs were provided by NRC (1978) and Givens and Moss (1990).

The analysis of water fluxes considered water gains through rainfall and water losses through ET. While precipitation information was readily available from meteorological records, ET estimates required different approaches in relatively homogeneous croplands and more heterogeneous natural vegetation. In the first case, AET values were based on crop-specific K_C coefficient (FAO, 1992) and PET estimates were based on meteorological records. Potential ET in forests and grasslands was estimated through an empirical model linking ET to precipitation proposed by Zhang et al. (2001), which included data from over 250 catchments worldwide. Water consumption by cattle considered drinking water and a larger component of the water consumed in the production process. Drinking water was estimated as 50 L head^{-1} day^{-1} for bovine cattle (Verdegem et al., 2006). Despite the daily consumption of forage being affected by various factors (type and size of animals, physiological condition, forage quality, etc.) water consumption through the intake of forage was roughly estimated from water consumed to produce it. Literature estimations were provided by FAO (1992), Wullschleger et al. (1998) and Zimmer and Renault (2002). Data on rainfall and ET, and on water retention capacity of soils were obtained from Murphy (2008) and INTA (1990), respectively.

The emission of GHG was estimated by following the IPCC (2006) Tier 1 recommendations. Multiplicative factors of 1, 21 and 310 were applied in that order to carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) emissions to get CO_2 equivalents (Mg CO_2-Eq ha^{-1} yr^{-1}). Carbon dioxide balances involved calculations of the biomass and SOC change on the one hand, and the fossil fuel consumption due to inputs used to farming operations. Methane emissions were calculated by means of livestock density and management systems on rice field coverage. We shifted the *default* factors provided by IPCC (2006) according to the period of our estimates using those for Oceania in 1960 and those for W Europe in 2001–05. The reason for choosing such areas and time periods was that they respectively represent well the productive models and technological farming conditions of Argentina during the same periods. Nitrous oxide emissions were estimated from IPCC (2006) *default* factors based on cattle faeces and urine emissions, synthetic N fertilizer use, biological N fixing and crop residues.

2.2.3. Environmental impact of pesticide pollution, habitat intervention and soil erosion

We considered the environmental impacts of (i) PC, (ii) HI and (iii) SE. First, the environmental risk impact of a given pesticide was estimated through various factors (Viglizzo et al., 2011): the oral lethal dose for rodents of commercial pesticides, an index for solubility in water, the water recharge capacity of soils (infiltration), a soil adsorption coefficient and the mean lifetime of the pesticide in the environment. Relative numerical factors were obtained from Weber (1994). The total pollution impact included the contribution of all the pesticides used in one farming year.

The estimation of the HI index assesses the degree of human interference on the habitat through (a) land use, (b) tillage practice, and (c) pesticide contamination. Land use was the proportion (%) of land cultivated annually with grain crops. The tillage practice factor used was the same "management factor" used for estimating SOC fluxes (IPCC, 2006). The same coefficients obtained from the estimation of PC risks were used here as well. The combined HI factor was the result of the simple multiplication of those three intervention factors. Thus, the higher the proportion of annual crops, the aggressiveness of tilling practices, and the toxicity of pesticides, the greater the detrimental effect of humans on habitats (Viglizzo et al., 2003).

The impact of SE (Mg sediments ha^{-1} yr^{-1}) was calculated through the universal equations proposed by Woodruff and Siddoway (1965) and Wischmeyer and Smith (1978) to estimate wind and water erosion, respectively. No erosion was assumed on non-cultivated lands. Local parameters used in both equations were obtained from INTA (1990). Wind erosion estimations considered the potential erodibility of soils, the plant coverage of soil and the soil roughness, as influenced by the tillage system. Water erosion (Wischmeyer and Smith, 1978) estimations took into account the rainfall erodibility, the susceptibility of soil to water erosion, the soil topography, the plant coverage and the application of conservation practices. Parameters used in both estimations were obtained from the Atlas of Soils of Argentina (INTA, 1990), Murphy (2008) and Michelena et al. (1989).

2.2.4. Scale approach and analytical procedures

Three spatial scales: (i) regions, (ii) macro-regions and (iii) the whole-country were analyzed in two historical periods (1956–60 and 2001–05). In this work, the political district was the basic scale unit considered for calculations. Attribute values for different scales were then estimated by means of simple data aggregation. Potential interactions between attributes within and across scales were not considered in this study because methods and empirical data to estimate this were not available. The arrangement of data into scales responded to the basic statistical structure of available data sets provided by censuses and surveys. The cross-scale impact of agriculture expansion was assessed by means of multivariate and conventional bivariate (correlation and regression) analyses (Cooley and Lohnes, 1971). The applied procedures of multivariate analysis concerned about the need of reducing the multidimensional complexity of data to a minimum number of dimensions needed to describe the relevant information contained in data sets. In this study, PCA was used to determine the minimum number of

Table 1
Proportional area devoted to annual crops (%) in regions involved in the study.

Scales			Area		% of annual crops	
Whole country	Macro-region	Region	km^2	%	1955–60	2001–05
Country			1,473,425	100.0	14.06	21.12
	Centre		634,650	43,07	29,71	39,67
		Pampas	426,160	28.9	33.93	44.55
		Espinal	208,490	14.2	21.11	27.77
	NW		573,475	38.9	1.94	9.48
		Chaco NW	527,007	35.77	1.92	9.51
		Yungas	46,468	3.2	2.11	9.15
	NE		265,300	18.1	3.19	5.01
		Chaco NE	90,755	6.2	1.31	5.60
		Campos	58,916	4.0	6.26	6.18
		Paraná Delta	45,387	3.1	5.60	8.91
		Iberá Marshes	40,441	2.7	1.20	0.60
		Atlantic Forest	29,801	2.0	1.90	0.96

independent dimensions needed to account for most of the variance in the original set of variables. The purpose of this procedure was to asses how the variance of bio-physical data on fluxes, stocks and impacts on the one hand, and geographical scales on the other hand, scatter and relate across a two-dimensional space comprised by axis *X* (PC 1) and *Y* (PC 2). The PCA procedure is relevant to reduce the complexity of standardized data to a few dimensions by identifying redundancy among variables (i.e. highly inter-correlated variables). Data standardization was automatically done by the applied statistical software SAS 9.0. It consists of a process by which the absolute value of each variable in a dataset is converted into a new standard value with an arithmetic mean equal zero, and a standard deviation equal one. Thus all variables can be compared despite disparity in their absolute values, units and expressions.

As mentioned above, a hypothesis in this study was that the components of the variance in the set of indicators associated with patterns and processes decline as the analysis moves from the smaller to the larger spatial scales. In this work, PCA was used to look for changes in data variance linking them to three spatial scales and two periods of time. Thus, if the variance declines as the spatial scale enlarged due to a mathematical compensation (variance estimation can not increase with scale), it also indicates that the scale is a critical factor to be considered in land-use strategies. Similarly, a temporal displacement of the variance would be indicative of the tendency of regions to become similar or dissimilar to each other across time. Outcomes will provide conceptual orientation to explore the underlying mechanisms that could explain how data variance that can be associated with changes in patterns and processes across time and space scales.

Correlation and regression analysis were later applied to quantify meaningful relationships between cultivation and pattern/process attributes across scales.

3. Results and discussion

As shown in Table 1, a rapid expansion of annual crops under rainfed conditions occurred in most of the territory of Argentina between the late 1950 and the early 2010 decade (Viglizzo et al., 2011). The proportional area devoted to annual crops in the Pampas and Espinal was higher than in the other regions. However, in percentage terms, the cultivated area increases at higher rates in Chaco, Yungas and Paraná Delta, and this expansion was achieved at the expense of losing native forests and grasslands. In ecological and environmental terms, as Viglizzo et al. (2011) have demonstrated, such expansion of annual crops resulted in substantial changes in the energy flow, matter cycle, emission of GHG, C and other minerals stocks, and the erosion risk, pesticide pollution and human intervention on the natural habitat.

3.1. Scales and components of variance

The complexity of the analyzed dataset is illustrated in Fig. 2 where indicators were displayed across *X* and *Y* axis that represent the components 1 and 2 of variance in the 1956–60 and 2001–05 periods. The PCA clearly discriminated between the analyzed attributes: while PC1, which explained 58.97% and 49.56% of total variance in 1956–60 and 2001–05, respectively, PC2 explained 26.26% and 28.39% of total variance in each period. Besides, PC1 largely represented the indicators of flux and impact, while PC2 represented indicators of stock. Thus, jointly PC1 plus PC2 explained 85.23% and 77.95% of the total variance in the two study periods, respectively. To facilitate the interpretation of the results, a sequence of three Fig. 3a–c was displayed to successively disentangle the knot of information regarding increasing spatial scales and two periods of time.

When the analysis moved from smaller to larger scales (from regions to macro-regions and whole-country) an expected decrease of the variance was detected in the multivariate analysis, with scattering across *X* and *Y* axis becoming lower with larger aggregation (Fig. 2). This agrees with Gardner (1998), who had already pointed out that one simple method to assess the variance of patterns and processes across scales in ecosystems is to plot the change in variance with changes in the spatial extent. He concluded that this type of analysis generally shows that estimates of the variance (S^2) are inversely proportional to the sample size, which is confirmed by the results in this work.

At the smaller regional scale, both attributes (fluxes/impacts and stocks) on the one hand, and regions (Pampas, Chaco, etc.)

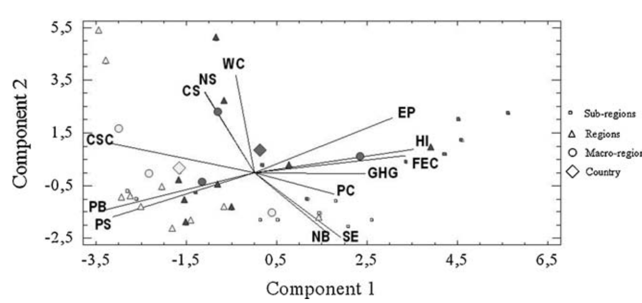

Fig. 2. Principal components analysis for mean values of stocks, fluxes and impacts at different cultivation scales in Argentina. *References*: Values on *X* (PC1) and *Y* (PC2) axes indicate the percentage of total variation explained by each component. References: Carbon (CS), nitrogen and (NS), phosphorous (PS) stocks, fossil energy consumption (FEC), energy production (EP), nitrogen (NB) and phosphorous (PB) balance, carbon stock change (CSC), pesticide contamination (PC), soil erosion (SE), greenhouse gas balance (GHG), water consumption (WC) and habitat intervention (HI). Bright gray symbols represent 1956–60 period while dark gray symbols represent 2001–05 period.

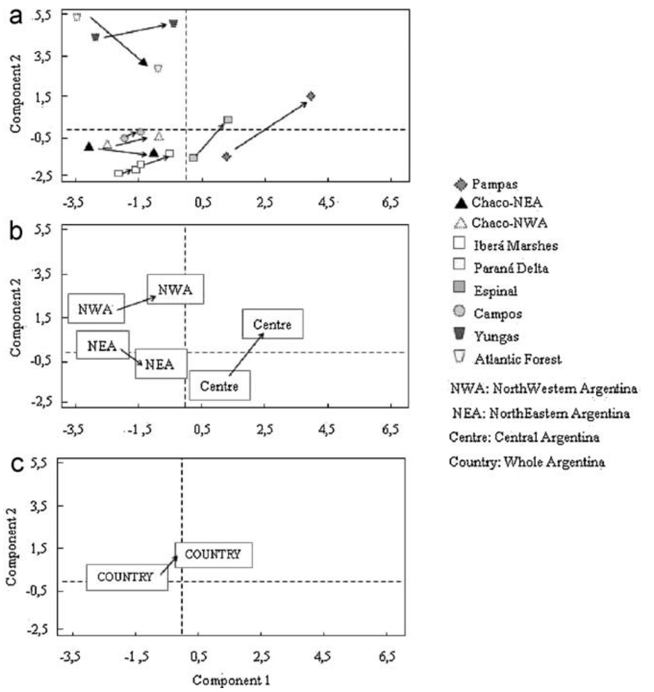

Fig. 3. Principal components analysis for cultivated lands in Argentina at three spatial scales: (a) regional, (b) macro-regional and (c) whole country in two time periods. The arrows indicate the change of situation from 1956–1960 to 2001–2005.

on the other hand, were scattered across *X* and *Y* axes (Fig. 3a). Differences (in terms of distance) between average standardized values of fluxes/impacts and regions were the largest and the most significant on *X*, and such difference, on the other hand, was lower on *Y*. Because of the noticeable expansion of cultivation between 1956–60 and 2001–05 (Table 1), when changes are observed from the PC1 (*X*) both fluxes/impacts on the one hand, and regions like the Pampas, Espinal, Atlantic Forest, Yungas and Chaco on the other hand, show a rightward displacement on axis *X*. This means that most of the variance of datasets is explained by flux/impact variations in first term, and by stocks variation in second term. Eigenvalues for PC1 and PC2 were 5.8045 and 3.5261, respectively. Then, the accumulated variance of both fluxes/impacts and stocks amounted around 71.77%. Considering the whole 1956–2005 period, this is a relevant finding because it indicates that fluxes and impacts tended to increase basically due to cropland expansion in Pampas and Espinal, and the stocks of material tended to decrease in Atlantic Forest, Chacos and Yungas due to deforestation. Jointly, both components can account for most the functional variability of the study biomes Across the whole study period, data reflect a significant expansion of croplands in the cases of the Pampas and Espinal (increasing variability on *X* axis), and a slow but progressive tendency (displacement toward a central position on *X* axis) of the Atlantic Forest, Yungas and Chacos to resemble functional characteristics of cultivated landscapes.

In the case of macro-regions, Fig. 3b shows that displacements were again more noticeable in the case of fluxes/impacts (PC1) than in the case stocks (PC2). Changes in fluxes/impacts appeared to be the largest in NW Argentina. A further reduction of variance is appreciated when the analysis focused on the broadest, whole-country scale. As Fig. 3c shows, while Argentina as a whole had a significant rightward movement of the *X* axis, such movement has been almost imperceptible on the *Y* axis. The analysis at this broader scale shows that changes in flows and impacts were by far more significant than those of stock.

The outcomes of this investigation appear to be relevant: (i) in all cases, regardless the scale, the underlying mechanism that explains the variance increase in flows and impacts over time seems to be related to fluxes and impacts, which are associated, as Viglizzo et al. (2011) have shown, with the increasing input of fossil energy that results from the expansion of cultivation across regions; (ii) the variance at the three studied scales tended to increase with time or, in other terms, as cultivation in regions tended to resemble the same cropping scheme of the Pampas; (iii) the variance of flows and impacts tended to decrease as the analysis displaced from smaller to larger scales.

3.2. Functional relationships across scales

Taking into account the three scales of analysis, Table 2a and b shows correlation values between FEC, which is a reliable estimate of energy fluxes (Dalgaard et al., 2003), and the rest of the study flux/impact variables for the 1956–60 and 2001–05 periods, respectively. As expected, correlation between FEC and EP was in general high and positive in most cases. The correlation between FEC and GHG was positive across scales in both periods. Correlation values were ambiguous (positive and negative) in the case of NB. Negative correlations were associated with N removal in areas where extractive cultivation predominates on total land, while the positive ones occurred in biomes dominated by woody species and water bodies where N removal due to crop production is negligible. The negative correlation between FEC and PB unequivocally shows that soil P removal increased as cultivation increases since the beginning to the end of the 50-year period. In general, the practice of P fertilization has not been extensively adopted because farmers still relied on the soil natural stocks of P. Thus, the higher the productivity of croplands the greater is the removal of soil P stocks. While an uncertain relationship was found in the case of WC, with only one exception in the case of Campos, positive correlations were mostly obtained between FEC and PC on the one hand and FEC and HI on the other hand. The ambiguous relation between FEC and SE risk in both periods can be explained by the drastic change in tillage technology between 1956–60 and 2001–2005. While conventional tillage was extensively used during the first period, its replacement by no-till methods in 2001–05 drastically modified the historical patterns of soil erosion risk, especially in the Pampas. The impact of no till cannot be perceived in regions where natural lands still predominate and croplands expansion was very low. The negative relation between FEC and SE in the Pampas during the period 2001–05 can be explained by the low fossil energy consumption of no-till methods.

FEC was used as reference indicator to examine cause–effect relationships across scales. Relying on regression analysis, Table 3 shows values for slope and R^2 across scales and regions that have a different cultivation index. Determination coefficient values and positive slopes decreased progressively toward NW and NE Argentina where most lands were predominately covered by natural vegetation, even in 2001–05. In comparison with 1956–60, such values became higher during the period 2001–05.

As Table 3 shows, in areas where croplands had large expansion (Pampas and Espinal) the strength of the relationship tended to decrease as the analysis moved from smaller to broader scales (from region, to macro-region and whole country). In general, such decrease was most noticeable during the end than during the beginning of the 50-year study period. This is consistent with previous findings by Viglizzo et al. (2005), who analyzed cross-scale relations and interaction in highly cultivated lands of the Argentine Pampas. Likewise, looking for rules to scale up soil properties in savannas, Ludwig et al. (2000) found that N concentrations in soil suffered a disjunction when measurements were done on patches of increasing size. They realized that direct scaling, as proposed by King (1991), is not always possible because of novel properties emerging at different scales. In other words, beyond the necessity of

Table 2
Correlation matrix (Pearson test) between fossil energy consumption and other indicators.

Scales			1956–60							
Whole country	Macro-region	Region	EP	GHG	NB	PB	PC	HI	SE	WC
Country			0.38	0.48	0.49	−0.60	0.79	0.81	0.22	−0.36
	Centre		0.77	0.08	0.70	−0.79	0.63	0.74	0.74	−0.26
		Pampas	1.00	1.00	−0.91	−1.00	1.00	1.00	1.00	−0.05
		Espinal	0.77	0.66	−0.58	−0.78	0.76	0.80	0.79	−0.34
	NW		0.19	0.27	−0.27	−0.35	0.81	0.79	0.08	−0.34
		Chaco NW	0.93	0.64	−0.87	−0.94	0.93	0.93	0.95	−0.03
		Yungas	0.47	0.64	−0.32	−0.47	0.47	0.49	−0.46	−0.93
	NE		0.73	0.21	0.21	−0.70	0.72	0.78	−0.64	−0.01
		Chaco NE	−0.23	1.00	−0.87	0.15	−0.34	−0.43	0.57	−0.99
		Campos	0.61	0.48	0.36	−0.75	0.45	0.51	0.48	0.60
		Paraná Delta	−0.23	0.82	0.79	0.25	−0.25	0.70	0.31	0.05
		Iberá Marshes	0.56	0.91	0.99	−0.41	0.43	0.99	−0.01	0.36
		Atlantic Forest	0.94	0.85	0.01	−0.94	0.94	0.98	−0.93	−0.11
Scales			2001–05							
Whole country	Macro-region	Region	EP	GHG	NB	PB	PC	HI	SE	WC
			0.71	0.37	0.16	−0.75	0.81	0.94	0.07	0.63
Country										
			0.84	0.20	−0.15	−0.82	0.81	0.94	0.55	0.15
	Centre									
			0.97	0.97	−0.92	−0.97	0.97	0.99	−0.74	1.00
		Pampas	0.90	0.86	0.36	−0.90	0.90	0.87	0.99	−0.48
		Espinal								
			0.56	0.26	0.07	−0.61	0.80	0.93	−0.26	−0.20
	NW									
			0.83	0.92	−0.03	−0.81	0.83	0.93	0.98	−0.11
		Chaco NW	0.95	0.89	−0.83	−0.95	0.95	0.96	−0.94	0.17
		Yungas								
			0.74	0.41	0.73	−0.72	0.74	0.87	−0.62	0.26
	NE									
			0.70	0.92	0.46	−0.66	0.69	0.89	0.98	−0.39
		Chaco NE	−0.11	−0.10	0.96	0.21	−0.30	−0.12	0.99	−0.68
		Campos	0.87	0.80	0.63	−0.87	0.86	0.97	−0.85	0.70
		Paraná Delta	0.60	0.87	1.00	−0.62	0.49	1.00	0.07	0.67
		Iberá Marshes	0.85	0.32	0.77	−0.85	0.85	0.95	−0.83	0.23
		Atlantic Forest								

References: energy production (EP), greenhouse gases (GHG) emission, balance of nitrogen (NB) and phosphorus (PB), pesticide contamination (PC), habitat intervention (HI), soil erosion (SE) and water consumption (WC).

reliable scaling rules, the scaling equations for extrapolation have a relatively narrow range of applicability (Miller et al., 2004).

3.3. *Cross-scale patterns and processes*

An up-scaling analysis involving the Pampas region, the Centre macro-region and the whole country was done to assess the impact of cultivation on fluxes/impacts across scales. Taking into account the results in Table 2a and b, EP, PC and the HI indices were chosen to build a hypothetical model to explain how fluxes and impacts behave across scales. Such variables were chosen because of their high correlation with FEC, especially in most areas where cultivation expanded across the study period. On the other hand, regarding stocks, Figs. 2 and 3a show that the variance of CS was low and its importance was constrained to the regional scale in forest lands like those of the Atlantic Forest and Yungas. Thus, the hypothet-

Table 3
Regression analysis between cultivation (% crops) and energy productivity (Mj ha^{-1} year^{-1}) at three different spatial scales during the periods 1956–60 and 2001–05 in Argentina.

Scales			1956–60			2001–05		
Whole country	Macro-region	Region	Slope	R^2	*P* value	Slope	R^2	*P* value
Country			68.68	0.366	0.0000	115.88	0.585	0.0000
	Centre		52.66	0.461	0.0000	88.43	0.652	0.0000
			34.25	0.999	0.0000	68.56	0.948	0.0000
		Pampas Espinal	56.32	0.583	0.0000	137.62	0.757	0.0000
	NW		85.14	0.052	0.0031	113.81	0.286	0.0031
		Chaco NW	75.83	0.046	0.0172	83.75	0.207	0.0000
		Yungas	140.30	0.142	0.0000	229.86	0.737	0.0000
	NE		−44.66	0.010	0.4031	73.36	0.056	0.0464
		Chaco NE	−177.14	0.117	0.0118	−67.57	0.056	0.3307
		Campos	8.08	0.004	0.8478	−18.25	0.088	0.3495
		Paraná Delta	−38.67	0.063	0.5148	130.51	0.746	0.0027
		Iberá Marshes	448.85	0.186	0.1241	835.39	0.235	0.0792
		Atlantic Forest	−457.62	0.064	0.3292	−703.93	0.040	0.4405

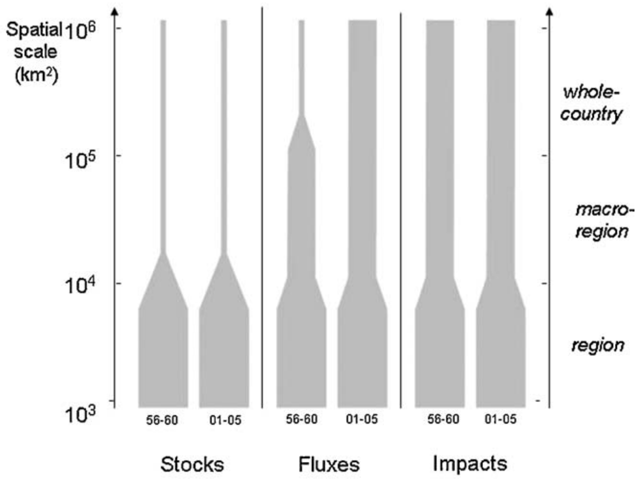

Fig. 4. The scale dependence of C stocks, energy fluxes and the human impact on the habitat. The thickness of bars indicates the relative importance of factors across regional, macro-regional and whole-country scales.

ical model of Fig. 4 was the result of combining the outcomes of Figs. 2 and 3a, and Table 2a and b. The thickness of the bar at a given scale indicates the relative importance of the study factor (stock, flux or impact) at that scale. As McGill (2010) pointed out, ecologists began drawing diagrams like that of Fig. 4 more than 20 years ago, but few empirical studies were done to interpret meaningful scale-dependent relationships.

The results in Fig. 4 show that the CS, being a pattern component of the landscape, was significant only in woody regions (e.g., Atlantic Forest, Yungas), no detectable relationships with cultivation were found beyond such regional scales. Although relations decreased at broader scales, fluxes/impacts (represented by FEC, EP, PC and HI) keep a positive relation with cultivation that maintains across scales. Such relation even tended to strengthen in response to cultivation expansion during the period 2001–2005 in relation to 1956–60. It is noticeable, on the other hand, that the impact indicator of HI kept a strong and positive relation with cultivation across the study scales and time periods (Table 2a and b). This finding related to scale-dependence processes could help interpreting some seemingly conflicting responses in agriculture and ecology. For example, relying on Table 2a and b, while it could be expected a positive cross-scale relation between FEC on the one hand, and GHG emission, PC and HI on the other hand, a strong negative relation was expectable between FEC and PB, in agreement to what normally happened in low-input farming ecosystems (Vitousek et al., 2009). As demonstrated by previous evidence provided by this research group (Viglizzo et al., 2001, 2003, 2011), such relations across scales became unclear or conflicting in the case of NB, SE and WC. Thus, fluxes related to FEC were influential at the regional scale, but such influence was projected to the macro-regional and the whole-country scale as cultivation reached an extensive geographical expansion. It is noticeable that impacts strongly related to FEC such as PC and HI on the other hand, appear to be equally important across the three scales in lands that were already extensively cultivated.

4. Conclusions

A novel methodological approach was proposed in this work to explore cross-scale relations in agriculture and ecology. The work rendered quantitative results to deliver at least three tentative conclusions that can be applied to different biomes in Argentina: (i) in general, positive relationships between cultivation and fluxes/impacts arose during the whole studied period; (ii) while the relevance of stocks is constrained to specific regional scales (e.g. Atlantic Forest and Yungas regions), fluxes and impacts tend to project their influence across different spatial scales; (iii) the magnitude of fluxes and impacts across scales tended to be strengthened in those regions like Pampas and Espinal where the cultivated area expanded significantly during the last decades.

The outcomes on this research also indicated that it would be possible to set tentative bands on scales (see Fig. 4) at which pattern and processes seem to be important, letting decision-makers know what factors may gravitate at a given scale and across scales. This information core may be particularly useful to support sound land-use policies because it can help explaining why centralized decisions made at the national scale (the broader one) may potentially fail or be ineffective to resolve problems at regional or local scales. For example, the promotion of farming systems that minimize the consumption of fossil fuels may trigger various positive impacts that can keep its integrity across scales. On the hand, decisions affecting stocks (e.g., CS) may have an impact that can be constrained to some specific regional scales only.

For agricultural and ecological scientists, the identification of scale dependencies is a key step to look for data to explain what factors control different patterns and processes at different scales and across scales. Soil erosion at the site scale, for example, may be indicating the application of inappropriate tillage operations, but soil erosion at the national scale may be revealing, on the other hand, a generalized technological lateness. In order to avoid confusing conclusions a gap should be filled in terms of the metrics (quantitative indicators) that are necessary to assess cross-scale patterns and processes. It is likely that explanations to some ambiguous relations in ecology and agriculture may depend on the scale of analysis. This scale dependency could help explaining, for example, why the relation between fossil energy input and N balance in soil can be negative in intensely cultivated land at a site scale, and neutral or positive at broader scales where natural vegetation or biological N-fixing pastures coexists with crops.

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