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Ecological and environmental footprint of 50 years of agricultural expansion in Argentina

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Abstract

Agriculture expanded during the last 50 years from the Pampas to NW Argentina at the expense of natural forests and rangelands. In parallel, productivity was boosted through the increasing application of external inputs, modern technology and management practices. This study evaluated the impact of agricultural expansion between 1960 and 2005 by assessing the implications of land use, technology and management changes on (i) carbon (C), nitrogen (N) and phosphorous (P) stocks in soil and biomass, (ii) energy, C, N, P and water fluxes and (iii) water pollution, soil erosion, habitat intervention and greenhouse gas (GHG) emissions (impacts). Based on different data sources, these issues were assessed over $\sim$1.5 million km^2 (63% of Argentina), involving 399 political districts during three representative periods: 1956–1960, 1986–1990 and 2001–2005. The ecological and environmental performance of 1197 farming system types was evaluated through the *AgroEcoIndex* model, which quantified the stocks, fluxes and impacts mentioned above. Cultivation of natural ecosystems and farming intensification caused a noticeable increase of productivity, a strengthening of energy flux, an opening of matter cycles (C, N, P) and a negative impact on habitats and GHGs emission. However, due to the improved tillage practices and the application of less aggressive pesticides, erosion and pollution risk are today lower than those of the mid-20th century. The consistency of some assumptions and results were checked through uncertainty analysis. Comparing our results with international figures, some impacts (e.g. soil erosion, nutrient balance, energy use) were less significant than those recorded in intensive-farming countries like China, Japan, New Zealand, USA, or those of Western Europe, showing that farmers in Argentina developed the capacity to produce under relatively low-input/low-impact schemes during the last decades. [Correction added after online publication 4 October 2010: In the first sentence of the Abstract, NE was corrected to NW.]

Keywords: Argentine agriculture, cultivation boundaries, environmental impacts, fluxes, stocks

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Introduction

Since the 1960s, as an intensive model of agriculture expanded across industrialized countries, its environmental impact became a source of controversy (Tilman *et al.*, 2002; Ewers *et al.*, 2009; IAASTD, 2009). Meanwhile, low-input, rotational cattle-crop production schemes prevailed in the Argentine Pampas (Solbrig, 1997). Until the early 1980s, production in the Pampas increased through expansion on natural lands, but once this possibility was exhausted, additional increases were achieved through more intensive use of external inputs, technology and management (Viglizzo *et al.*, 2001). The production model of the Pampas later expanded over other regions dominated by natural (mostly woody) vegetation in the north of Argentina (Carreño & Viglizzo, 2007).

Several authors studied the impacts of agriculture in the Pampas (Viglizzo *et al.*, 2001; Rabinovich & Torres, 2004; Satorre, 2005), but only few (Paruelo *et al.*, 2004; Adámoli, 2006) in the rest of Argentina. Traditional ecological perspectives (Odum, 1975; Ehrlich *et al.*, 1977) on the impacts of agriculture on energy flow, material cycles and pollution lack development in Argentina and are critical to guide and manage current and future transformations.

The main objective of this study was to assess the ecological and environmental consequences of agricultural expansion in Argentina between 1960 and 2005. We focused on its impacts on (i) nutrient stocks of carbon (C), nitrogen (N) and phosphorous (P), (ii) energy, C, N, P and water flows in ecosystems and

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(iii) water pollution, soil erosion, habitat intervention and greenhouse gas (GHG) emissions.

Materials and methods

Extent of the study

The study extended over 1.47 million km^2 (63% of Argentina's continental area) (Table 1), including the most important rainfed agricultural and cattle production areas represented by 15 eco-regions (Fig. 1). This area concentrates 89% of human population, 98% of bovine cattle heads and over 90% of annual and perennial crops (INDEC, 2004). Dominant biomes in the area are grasslands; tropical, subtropical and temperate forests and shrublands; with many areas being replaced by croplands. Agricultural exports from this area include soybean, sunflower, maize, wheat and beef.

To study a time span of 50 years, we divided it into three subperiods: 1960, which represented the farming conditions of the 1950 decade when the traditional extensive agricultural model prevailed; 1986–1990, which represented the transition from the traditional to modern model; and 2001–2005, when the modern model intensified and expanded to new regions.

Data sources and calculation procedure

Three land-use types were considered: (i) forest lands (native forests and tree plantations); (ii) grasslands and pastures (savannas, shrublands and cultivated pastures) and (iii) rainfed croplands (cereals like wheat, maize, and rice; oil crops like linseed, soybean and sunflower; cereal forages like oats, rye and triticale; and industrial crops like sugarcane and cotton). Irrigated land, covering $<0.5\%$ of the country, was not considered. We used databases that covered 399 political districts (Table 1) and included the national agricultural censuses of 1960, 1988 and 2002 (INDEC, 1964, 1991, 2004), covering all legal farm units and the annual enquiry of the National Secretary of Agriculture (SAGPyA, 2009), recording cultivated areas from farm samples since 1970. Data before 1970 were provided by the 1960 national census. To reduce biases, we replaced the 1-year records of cultivation provided by INDEC for 1988 and 2002 by 5-year averages estimated from SAGPyA since 1970.

Given that cattle productivity was not estimated by censuses and enquiries, it was indirectly calculated through equations based on stocking rates, as done before in Argentina (Cerqueira *et al.*, 1986; Viglizzo, 1993) and other countries (Mott, 1960; Jones & Sandlands, 1974; Holmes, 1980; Doyle & Lazenby, 1984). Beef production was estimated using non-linear equations developed by Viglizzo (1982) for calving and fattening areas, where $Y = -27.0 + 258.4X - 15.4X^2$ ($R = 0.86$) and $Y = -32.0 + 252.9X - 62.6X^2$ ($R = 0.86$), respectively, where Y is the average beef production (kg ha^{-1} yr^{-1}) and X the average stocking rate (animal units ha^{-1} yr^{-1}) obtained from national agricultural censuses. Animal units represent one bovine of 450 kg live weight. Given that 95% of dairy farms are located in the Pampas (Sanmartino, 2006), milk production figures refer to that region and where based on the population of dairy cows. Milk production was sustained only by grazing in 1960 but supplementation with concentrate increased during the following two periods (Viglizzo *et al.*, 2001). Average milk productivity of dairy farms were 1230, 3765 and 5733 L ha^{-1} yr^{-1} for 1960, 1986–1990 and 2001–2005, respectively, which was the result of dividing total milk production in the Pampas by the total area of dairies.

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 systems, being, respectively, 100–0–0, 60–30–10 and

Table 1 Areas and districts involved in the studied ecoregions and subregions

Ecoregion	Subregion	Area (km^2)	% of the total area	Number of districts
Pampa		426 160	28.92	135
	Rolling	74 399	5.05	41
	Subhumid	129 350	8.78	32
	Southern	82 530	5.60	21
	Semiarid	14 682	1.00	4
	Flooding	93 161	6.32	31
	Mesopotamian	32 038	2.17	6
Espinal and Campos		246 981	16.76	37
Chaco		638 187	43.32	143
	Humid Subhumid	111 180	7.55	21
	Central Subhumid	97 063	6.59	21
	Dry	360 131	24.44	80
	Western Subhumid	69 813	4.74	21
Atlantic Forest		29 801	2.02	17
Iberá Marshes		40 441	2.74	14
Paraná Delta		45 387	3.08	9
Yungas Region		46 468	3.15	44
Total		1 473 425	100	399

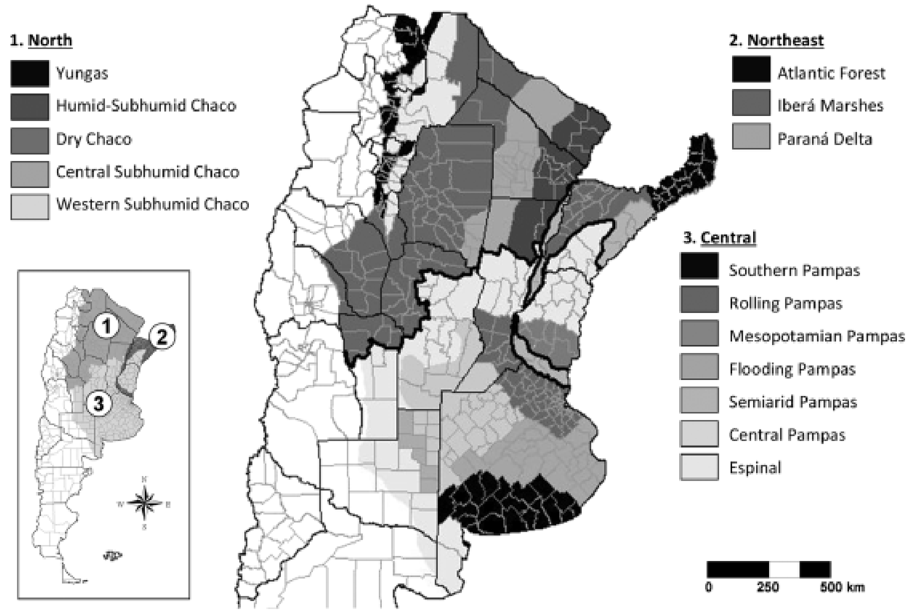

Fig. 1 Location of the studied regions and subregions in the Argentine territory.

20–30–50 for the three selected periods (Salvador, 2002). Pesticides also varied, being dominated by chlorinated products in 1960, replaced by pyrethroids and organo-phosphorus pesticides in 1986–1990, and by pyrethroids in 2001–2005. Herbicides, various combinations of 2,4-D, 2,4-DB, Piclorane, Atrazine, Trifuraline, Bromoxynil y Glyphosate (SENASA, 2004) were applied.

In all cases, we assumed that the doses proposed by manufacturers. Regarding N and P fertilizers, we assumed that they were used at the farm level keeping a proportion to national statistics of consumption. Thus, 0%, 20% and 100% of the recommended dose were, respectively, attributed to 1960, 1986–1990 and 2001–2005 (SENASA, 2004). Especially during the last 20 years, genetically modified (GM) soybean was the dominant crop, which expanded very quickly at the expense of other crops on already cultivated lands. The so-called 'soybean package' – which is cheap, simple and effective – was extensively adopted across the country (Rabinovich & Torres, 2004; Satorre, 2005).

Land use and land-use change in each district were analyzed from censuses and enquiries. Contradictory data on forest area (INDEC, 1964, 1991, 2004; SayDS, 2007a, b) was avoided and replaced by recent assessment of C stocks and emissions in forests of north Argentina for 1900–2005 (Gasparri *et al*., 2008). We distributed absolute figures from Gasparri *et al*. (2008) among districts by using the percentage of forest that censuses attributed to each district.

Stocks. Our estimates of C stocks included biomass and soils. In forests we took into account: (i) aboveground biomass, (ii) an extra fraction of biomass (under-story vegetation, belowground biomass, biomass in dead wood and litter, and (ii) soil organic C (SOC). The initial aboveground stock of biomass C (Mg ha^{-1}) in forests of different climate zones (tropical, subtropical and temperate, wet and dry) was estimated from *default* factors of IPCC (2006) Tier 1 after successful cross-checking with those in Gasparri *et al*. (2008): 78 vs. 74, 256 vs. 300 and 198 vs. 180 Mg DM ha^{-1} for Chaco, Atlantic and Yungas forests, respectively. Extra biomass fraction based on Gasparri *et al*. (2008) (as a fraction of aerial biomass) was 0.49, 0.41 and 0.38, respectively, for the same three forest types. An average of the three systems was applied for the Espinal-Campos forests, which have no available data. A factor of 0.47 was used to convert DM to C stock (IPCC, 2006). Regarding the total study area, the area planted with forests (SAGPyA, 2009) was 0.0%, 0.43% and 0.74% in 1960, 1986–1990 and 2001–2005, respectively. Its contribution to the C budget was calculated by summing plantations in Atlantic Forest and the Iberá and Parana Delta wetlands (0.0, 4.1 and 8.0 for periods 1960, 1986–1990 and 2001–2005, respectively). Planted forest coverage in other regions of Argentina like Pampas and Espinal was always $<0.2\%$ (SAyDS, 2004). Following IPCC (2006) Tier 1 procedure, average values of aboveground biomass of 222.50, 157.5 and 88.75 Mg DM ha^{-1} for planted forests were adopted for tropical, subtropical and subtropical mountain biomes. Likewise, IPCC (2006) criteria for estimating biomass stock in grasslands/pastures and croplands were followed. Biomass stocks in grasslands/pastures, croplands and wetlands were assumed to be under steady-state, with losses equalling gains. Estimates of SOC stocks for the top 30 cm of mineral soil assumed 58% C in organic matter (Alvarez *et al*., 1995; Steinbach & Alvarez, 2006; Gasparri *et al*., 2008). Bulk density (Mg m^{-3}) was estimated for all soils from organic matter content, as proposed by Ruehlmann & Körschens (2009). These authors provided a universally valid equation for soils different in their genesis, compaction and type of land use (R^2 of 0.99 in a comparison of calculated versus measured values). Based on IPCC (2006) for Tier 1 and 2 methods and survey data, the initial C stocks in soil (Mg C ha^{-1}) were set for grasslands and croplands. In general, values were higher in temperate than in tropical climate regions, and in soils with high cation-exchange capacity than in sandy soils. Likewise, several *default* values were used to estimate temporal changes in soil C stocks (see section on 'Fluxes'). Forest SOC stocks were obtained from Gasparri *et al*. (2008), and had values of 31, 35, 65 and 56 Mg SOC ha^{-1} for Chaco, Atlantic Forest, Yungas Forest and Espinal, respectively. A similar value to that of the Atlantic Forest (35 Mg ha^{-1}) was adopted for the forest soils of the Iberá Marshes and Paraná Delta.

Biomass and soil N and P stocks (Mg ha^{-1}) were calculated from C stocks and C : N : P stoichiometric values from the literature. Regarding C : N, we used specific ratios (kg N kg C^{-1}) for crop residues (0.02458 ± 0.00256) (Givens & Moss, 1990), forests (0.0359 ± 0.0126) (Zak & Pregitzer, 1990; Nadkrmiz & Matelson, 1992; Hooker & Compton, 2003), grasslands (0.0603 ± 0.0072) (Zak *et al*., 1990) and pastures (0.0796 ± 0.0234) (NRC, 1978; Givens & Moss, 1990). According to data from Huan (1996), the N : P mass ratio (kg P kg N^{-1}) is 0.1184 ± 0.0620 for biomass in general. To estimate soil contents of N and P, they were related to SOC content. According to field data from Steinbach & Alvarez (2006) and Galantini & Suñer (2008), a C : N : P relation of 100 : 11 : 1 was adopted.

Fluxes. Classical works in ecological science state that matter (C, N, P, water) cycles within ecosystems and energy flows across them (Odum, 1969, 1971, 1975). However, given that matter closes their cycles on very large spatial scales, at the limited scale of our study we assumed that matter, in practice, behaves as a flow.

The analysis of energy fluxes (MJ ha^{-1} yr^{-1}) involved estimates of inputs in the form of fossil energy consumed for the synthesis of pesticides, fertilizers, concentrates, seeds, etc. and by agricultural activities (ploughing, harrowing, seeding, spraying, harvesting, water pumping, etc.); and outputs in the form of agricultural products (see Viglizzo *et al*., 2003 for details). Net C fluxes (Mg ha^{-1} yr^{-1}) were estimated through changes in plant biomass and SOC. Average values of forests biomass change (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). The annual C balance of biomass resulted from a *Gain–Loss method* that can be applied to all C gains or losses:

$$\Delta C_B = \Delta C_G - \Delta C_L, \qquad (1)$$

where ΔC_B is the annual change of C stock in biomass, ΔC_G the annual biomass gain and ΔC_L the biomass loss. We assumed, as IPCC (2006) Tier 1 suggests, no annual change in above- and belowground biomass in grasslands/pastures because they are in an approximate steady state. A similar criterion was applied to biomass in croplands and wetlands; we assumed that biomass increases in a single year is equal to biomass losses.

The SOC annual flow (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_{REF}) was assigned to each region based on climate and soil type. Then, a land-use factor (F_{LU}), a management factor (F_{MG}) and a C input level factor (F_I) was assigned to each land use at each time period, except for woodlands, for which we assumed no change in their mineral soil C stocks as recommended by IPCC (2006). Finally, the factors (F_{LU}, F_{MG}, F_I) were multiplied by SOC_{REF} to estimate an initial [$SOC_{(0\text{-}T)}$] and final [$SOC_{(0)}$] SOC stock for each time period. Thus, the SOC annual flow was estimated by subtracting $SOC_{(0-T)}$ from $SOC_{(0)}$, which was divided by 20 years according to IPCC (2006) recommendation. The annual loss of C due to soil erosion (C_{SE}) was also subtracted from C stocks [see equations for soil wind erosion (WEQ) and soil water erosion (USLE) in section on 'Impacts'].

Regarding the balance of N and P due to natural events and human activities, various ways of gain and loss were considered. N inputs included (i) atmospheric deposition of 0.6 kg N per 100 mm of rainfall (Panigatti & de Hein, 1985), (ii) Fertilizers N (kg ha^{-1}), (iii) biological N fixation (Baethgen, 1992; Brenzoni & Rivero, 1996) by legumes: (70–120 kg ha^{-1} yr^{-1} depending on species), and (iv) input through animal feeds (Viglizzo *et al*., 2003). N 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, where 0.68 is approximately 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 'Impacts'. P inputs included (i) fertilizers and (ii) supplementary feeds for cattle. 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 on 'Stocks'). Data on N and P concentration in agricultural inputs and outputs were provided by NRC (1978) and Givens & Moss (1990).

Our analysis of water fluxes considered water gains through rainfall and water losses through evapotranspiration. While precipitation information was ready available from meteorological records, evapotranspiration estimates required different approaches in relatively homogeneous croplands and more heterogeneous natural vegetation. In the first case, actual evapotranspiration (ET) values were based on crop-specific K_C coefficient (FAO, 1992) and potential evapotranspiration estimates based on meteorological records. Evapotranspiration in forests and grasslands was estimated based on the empirical models linking evapotranspiration to precipitation proposed by Zhang *et al*. (2001), which included data from over 250 catchments worldwide. Taking into account the annual precipitation (PRE) in mm yr^{-1} for different climate regions, the following quadratic equations were estimated from Zhang *et al*. (2001) data to estimate actual EVT (mm yr^{-1}) in forest (EVT_f), cultivated pasture and native grassland (EVT_{pg}) and mix cattle-crop (EVT_m) ecosystems: (i) $EEV_f = 50.0 + 0.9195PRE - 0.0001PRE^2$, (ii) $EVT_{pg} = 69.875 + 0.6263PRE - 0.0001PRE^2$, (iii) $EVT_m = 59.938 + 0.7729PRE - 0.0001PRE^2$.

Water consumption by cattle considered drinking water and a much greater component which is the water consumed in the production process of feedstock. 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 & Renault (2002). Data on rainfall and EVT, water retention capacity of soils were, respectively, obtained from Murphy (2008) and INTA (1990).

Impacts. The environmental impact of agriculture expansion included (i) nutrient and (ii) pesticide pollution, (iii) soil erosion and (iv) GHGs emissions. The risk of nutrients pollution (mg L^{-1} of runoff/infiltration water) was directly related to N and P balances as estimated in section on 'Fluxes'. According to McRae *et al*. (2000), we assumed pollution risk only when N and/or P excesses (based on N and P balances) coexisted with water excesses (based on rainfall and evapotranspiration values).

Pollution by pesticides (relative index) was estimated based on the following equation of pollution risk (PR)

$$\mathrm{PR} = \frac{1000}{DL_{50}}\left[\frac{K_{\mathrm{sp}} + R}{2} + K_{\mathrm{oc}} + T1/2\right] \times \mathrm{Dose} \times \mathrm{Area}, \quad (2)$$

where, for any given pesticide, based on CASAFE (1997) data, DL_{50} is the oral lethal dose for rodents of commercial pesticides, K_{sp} is the partition coefficient that represents solubility of pesticide in water, R is the water recharge capacity of soils (infiltration), K_{oc} is a soil adsorption coefficient and $T1/2$ is the mean lifetime. Relative numerical factors were obtained from Weber (1994). Total pollution risk included the contribution of all the pesticides used in one farming year. Precursors and intermediate metabolites were not considered in our calculations.

Soil erosion (Mg sediments $ha^{-1} yr^{-1}$) was estimated based on the universal equations proposed by Woodruff & Siddoway (1965) and Hagen (1991) to estimate wind and water erosion, assuming no erosion on noncultivated lands. Local parameters used in both equations were obtained from INTA (1990). Wind erosion (WEQ) considered the potential erodibility (E) of soils, plant coverage (C), and roughness (R), as influenced by the tillage system:

$$\mathrm{WEQ} = (34.8183 - 23.1874 \times \mathrm{C} + 0.5718 \times \mathrm{E}) \times \mathrm{R}. \quad (3)$$

Water erosion (USLE, Wischmeyer & Smith, 1978) estimates took into account rainfall erodibility (R), susceptibility of soil (K) to water erosion, topography features (LS), plant coverage (C) and conservation practices (P)

$$\mathrm{USLE} = R \times K \times LS \times C \times P. \quad (4)$$

The values of parameters used for E, LS and P were obtained from the Atlas of Soils of Argentina (INTA, 1990), rainfall data from Murphy (2008), and R and K from Michelena *et al*. (1989). In addition to the previous indicators, we included in our analysis an index that assessed the relative anthropic intervention (I) in habitats, which was the result of the product of land-use changes (L), tillage operations (T) and pesticide pollution (PR).

GHGs emissions were calculated by following IPCC (2006) generic methods (Tier 1) applicable for multiple land-use categories. Annual emissions were estimated as the sum of emissions of croplands, grasslands, woodlands and wetlands, plus emissions related to fossil energy use and livestock. Within each land-use category, GHGs emissions were estimated as the sum of carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) emissions. Factors of 21 and 310 were, respectively, applied to the latter two gases in order to express them as CO_2 equivalent (Mg CO_2-Eq $ha^{-1} yr^{-1}$).

The basis to estimate CO_2 emissions were the gain–loss and/or the stock-difference methods proposed by IPCC (2006) for each land-use type. Changes of C stocks in biomass, SOC of mineral soils and dead organic matter were included, as well as changes in land use. Annual estimations of CO_2 emissions also included fossil fuels used in rural activities and fuels used for manufacturing fertilizers, herbicides and machinery, which were roughly estimated through fuel-based emission factors.

Methane emissions included biomass burning, enteric fermentation, faecal losses and rice cultivation (CH_4). Non-CO_2 emissions from biomass burning were estimated from deforestation and wood removal (mostly leaves and branches). Livestock related emissions were calculated from livestock population and feed characterization, combined with climate-based emission factors. CH_4 were also estimated for rice fields in NE Argentina using generic emission factors according to water regime.

Finally, losses of N_2O to the atmosphere were estimated through (i) cattle faeces and urine emissions, (ii) synthetic N fertilizer (urea) use, (iii) biological N fixing and (iv) crop residues. In all cases, we adopted the *default* factors provided by IPCC (2006) taking into account the historical period: factors for Oceania were used in 1960, those for eastern Europe in 1986–1990 and those for western Europe in 2001–2005. The rationale for those choices was that they, respectively, represent acceptably well the productive and technological farming conditions of Argentina during the periods.

Analysis of results and mapping

The effect of agriculture expansion was assessed by means of simple regression analysis, using linear and nonlinear models. To detect geographical patterns and gradients, and temporal trends as well, results were transferred to a geographical information system (Arc-View) in order to display dot-density maps.

Uncertainty analysis

Based on the IPCC (2003) practice guidance, we adopted a semiquantitative method to assess the uncertainty of our results. Our procedure included estimates of standard deviation, comparisons with independent statistical data. Probability distributions were reconstructed through expert judgment when data were missing or not consistent. Soil and ecology experts were consulted to support the reconstruction of curves for C, N and P stocks. All of them, cited in this work (Hepper *et al*., 1996; Viglizzo *et al*., 2001; Cruzate & Casas, 2003; Gutierrez Boem *et al*., 2008), were members of well-known and experienced soil-research teams in Argentina. Results were cross-checked with research results in order to estimate overlapping between the probability distribution of estimated and research results, assuming that a high degree of overlapping indicates low uncertainty. Uncertainty was semiquantitatively expressed in terms of very low ($P < 20\%$), low ($20\% < P < 40\%$), moderate

(40% < P < 60%), high (60% < P < 80%) and very high probability (P > 80%).

Results and discussion

Land-use/cover change

While a significant increase in annual crops (around 60%) occurred in the whole region (Table 2, Fig. 2). The boundaries of cultivation did not extend homogenously throughout the country (Fig. 3), with a rapid expansion towards the NW, a stable situation in the W and a retraction in the Flooding Pampas. On the other hand, due to recent migration of cattle from the Pampas and Espinal (Rearte, 2007; SENASA, 2008), pastures showed a persistent increase in the Chaco, Atlantic Forest and Iberá wetlands. The natural forests area suffered a significant reduction during the study period, accounting for 42%, 28% and 16% of the initial area for the Atlantic, Chaco and Yungas forests, respectively. Only in Espinal, we found a slight increase of the woody area, which agree with data from Dussart *et al*. (1998).

Table 2 Land use (%) in ecoregions and subregions during the three studied periods

		Average area (%) of								
		Annual crops			Grasslands/pastures			Forests and shrubs		
Ecoregion	Subregion	1960	1986–1990	2001–2005	1960	1986–1990	2001–2005	1960	1986–1990	2001–2005
Pampa		33.93	34.26	44.55	66.07	65.74	55.45	–	–	–
		± 26.64	± 42.98	± 47.22	± 26.64	± 42.98	± 47.22			
	Rolling	36.80	56.76	70.08	63.20	43.24	29.92	–	–	–
		± 13.91	± 31.26	± 30.07	± 13.91	± 31.26	± 30.07			
	Subhumid	44.19	39.65	50.73	55.81	60.35	49.27	–	–	–
		± 13.21	± 29.57	± 28.26	± 13.21	± 29.57	± 28.26			
	Southern	39.08	39.17	52.59	60.92	60.83	47.41	–	–	–
		± 11.74	± 13.17	± 22.88	± 11.74	± 13.17	± 22.88			
	Semiarid	38.98	41.93	43.16	61.02	58.07	56.84	–	–	–
		± 4.43	± 15.15	± 7.30	± 4.43	± 15.15	± 7.30			
	Flooding	17.48	9.50	10.77	82.52	90.50	89.23	–	–	–
		± 13.14	± 8.13	± 10.55	± 13.14	± 8.13	± 10.55			
	Mesopotamian	18.08	16.12	38.51	81.92	83.88	61.49	–	–	–
		± 7.79	± 5.31	± 10.46	± 7.79	± 5.31	± 10.46			
Espinal and Campos		18.15	17.34	23.41	68.55	58.39	53.01	13.30	24.28	23.58
		± 20.72	± 19.30	± 26.05	± 17.76	± 20.32	± 26.33	± 13.33	± 23.20	± 26.84
Chaco		2.07	3.90	8.57	60.31	63.23	59.35	37.62	32.87	32.08
		± 8.43	± 10.95	± 24.49	± 24.83	± 23.08	± 29.28	± 26.71	± 19.86	± 22.91
	Humid Subhumid	3.64	5.64	7.92	68.85	72.35	69.32	27.51	22.00	22,75
		± 5.08	± 4.89	± 8.18	± 6.81	± 8.36	± 10.22	± 8.14	± 8.31	± 6,99
	Central Subhumid	2.54	6.26	16.59	59.72	60.48	50.67	37.74	33.26	32,74
		± 5.03	± 8.30	± 19.23	± 10.47	± 16.87	± 20.87	± 11.39	± 13.01	± 14,40
	Dry	1.55	2.91	6.93	58.24	61.49	58.73	40.21	35.60	34,34
		± 4.46	± 5.22	± 12.76	± 21.46	± 13.34	± 17.80	± 22.75	± 12.50	± 16,40
	Western Subhumid	1.85	3.87	9.09	59.48	62.03	58.81	38.67	34.10	32,10
		± 4.86	± 9.53	± 21.14	± 23.43	± 20.41	± 24.68	± 24.92	± 16.60	± 19,06
Atlantic Forest		1.90	1.53	0.96	21.36	37.52	44.69	76.74	60.94	54.35
		± 1.87	± 1.46	± 0.94	± 14.59	± 15.87	± 21.43	± 15.27	± 16.64	± 21.83
Iberá Marshes*		1.20	0.72	0.60	38.76	45.27	46.07	4.11	7.96	12.59
		± 1.02	± 0.89	± 0.65	± 10.09	± 13.46	± 15.60	± 3.70	± 3.44	± 3.81
Paraná Delta*		5.60	4.17	8.91	50.09	44.75	49.30	5.41	7.54	13.27
		± 6.58	± 5.88	± 15.98	± 19.78	± 21.85	± 18.38	± 4.99	± 3.81	± 7.07
Yungas Region		2.11	2.66	9.15	32.19	40.67	43.34	65.70	56.67	47.51
		± 3.44	± 4.47	± 14.43	± 24.30	± 21.67	± 23.16	± 24.66	± 21.21	± 17.76
Total		14.06	14.77	21.12	60.78	60.85	55.68	22.43	21.78	21.20
		± 35.64	± 48.96	± 63.03	± 54.21	± 64.60	± 73.19	± 42.09	± 41.06	± 45.84

*Ecoregions with a high percentage of the area covered by water. Figures preceded by ± are standard deviations.

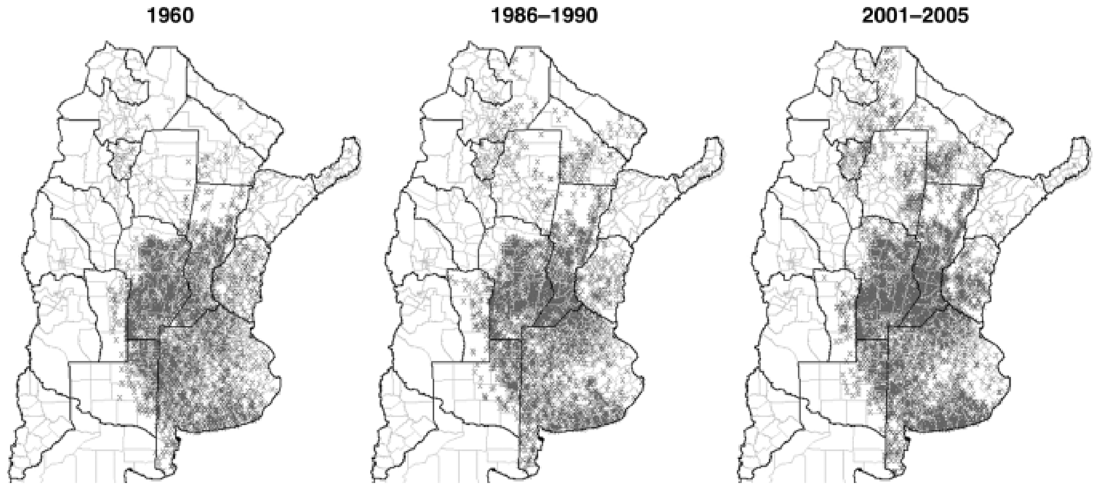

Fig. 2 Changes on cropland area in the ecoregions of Argentina during the three studied periods. 1 dot = 7500 ha.

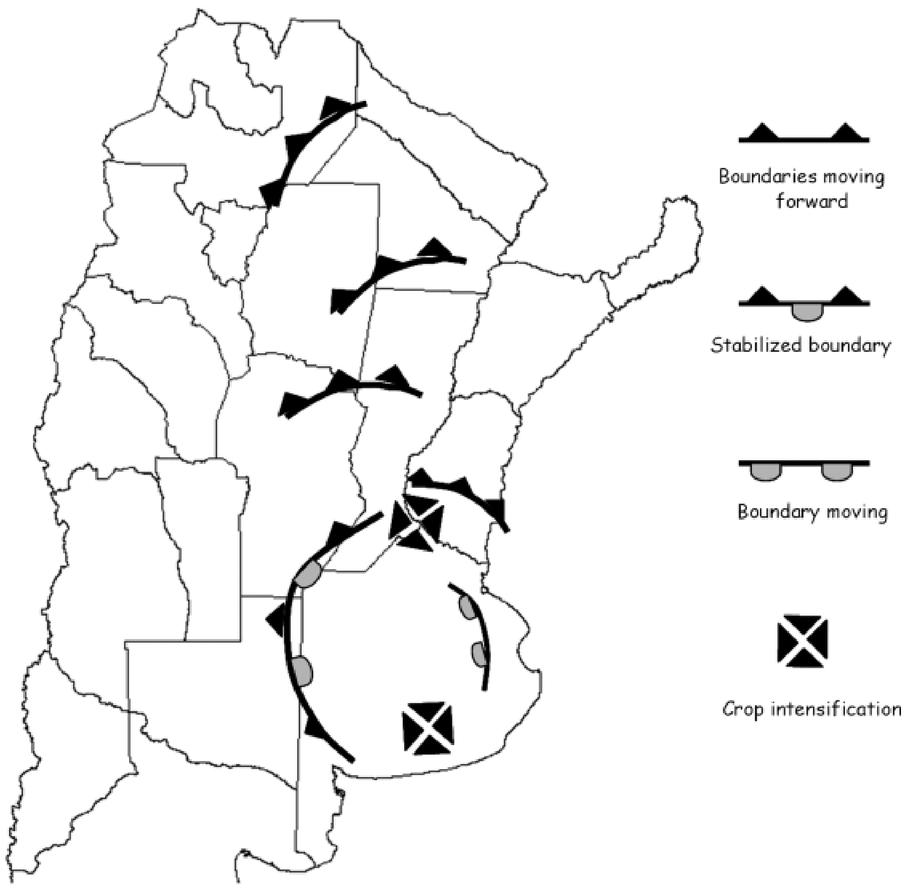

Fig. 3 Dynamics of the agricultural boundaries during the 1960–2005 period.

Changes in C, N and P stocks

C stocks in biomass and soil organic fraction varied among regions and from period to other (Fig. 4). SOC seemed to be more stable than biomass C across time and space, so the latter appears to be more vulnerable to human action. More than 50% of the total C is stored in biomass in the Atlantic Forest and Yungas, which become vulnerable because of their easily harvestable C. Given the C : N : P relationships that we have used in

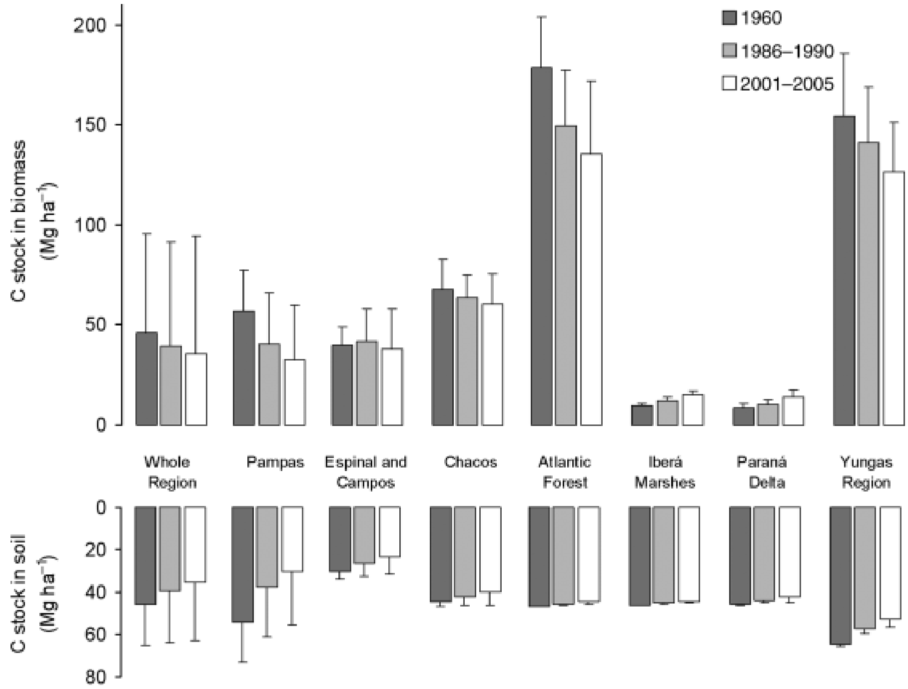

Fig. 4 Estimated carbon (C) stock average in biomass and in soil in different ecoregions during the three studied periods. Error bars are standard deviations.

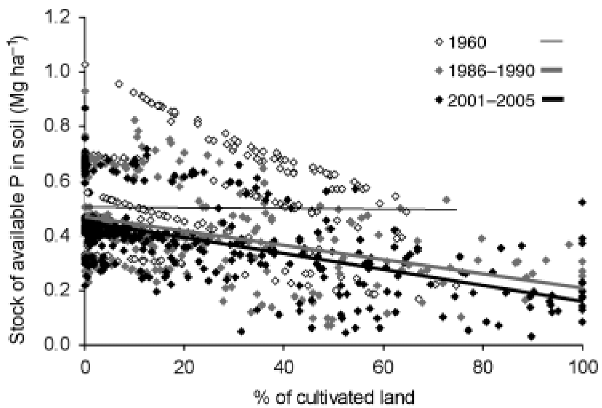

Fig. 5 Relationship between the percentage of cultivated land and the estimated stock of available phosphorus (P) in soil in 399 districts during the three analyzed periods. R^2 and P-values are 0.02 and 0.01 (1960); 0.22 and 0.01 (1986—3090); 0.34 and 0.01 (2001–2005).

our calculations for biomass and soil (see section on 'Stocks'), the loss of C in biomass and soil caused, accordingly, a proportional loss of N and P. However, because of its low renewable capacity, P exhaustion is more concerning than the N one. P stocks tend to decline persistently (Fig. 5) because the accumulated extraction form soil was not counterbalanced by P fertilization (Suñer *et al.*, 2005; Galantini & Suñer, 2008; Gutierrez Boem *et al.*, 2008). Current P stocks in traditional croplands of the Pampas seem to be reaching soil concentrations (20 ppm) that are not suitable to maintain crop productivity. A map showing the depletion of P stocks in the study area is shown in Fig. 6. Our estimations were confirmed by data of Hepper *et al.* (1996) and Cruzate & Casas (2003) from field measurements.

Energy, C, N, P and water flows

A progressive increase of fossil fuel consumption and a relatively higher increase of energy productivity were recorded in most regions (Fig. 7), leading to widespread increase in energy efficiency reaching an annual average of 1.5 GJ were produced per each GJ of fossil energy consumed. This ratio was higher in regions where crop cultivation expanded quickly (e.g. the Pampas) or in areas like Yungas where energy from biomass (e.g. sugarcane) surpassed that achievable with grains. This ratio deteriorates in areas where cattle production prevailed. Because of deforestation, Yungas and Atlantic forests had the highest C losses during the study period. In the Pampas, SOC loss tended to decrease because of the generalization of reduced to no-till practices (Steinbach & Alvarez, 2006). Overall, elemental losses showed the ranking P > N > C and negative trends for P balance agreed with results by Michelena *et al.* (1989) and Cruzate & Casas (2003). Water-use efficiency

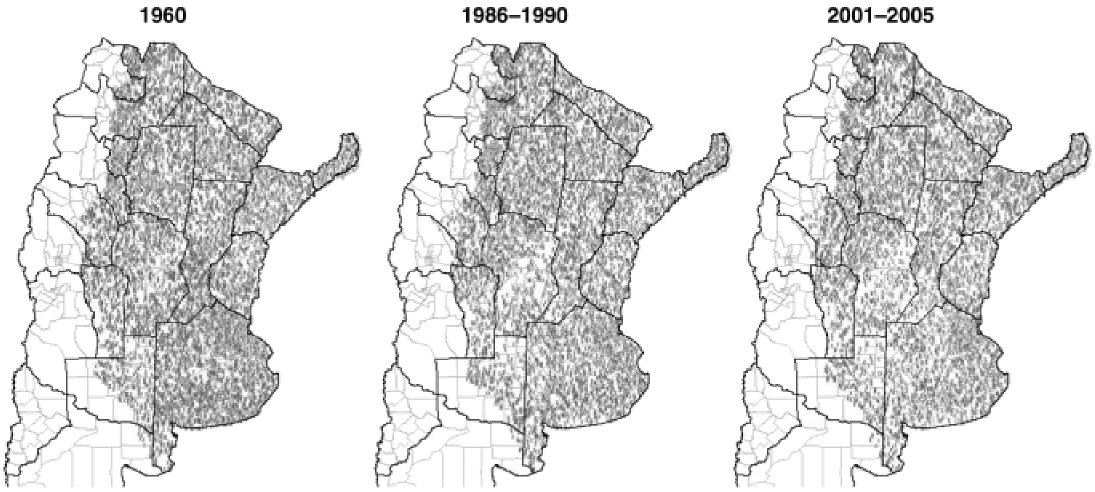

Fig. 6 Changes in the stock of available phosphorus P) in soil in the ecoregions of Argentina during the three studied periods. 1 dot = 1×10^4 Mg.

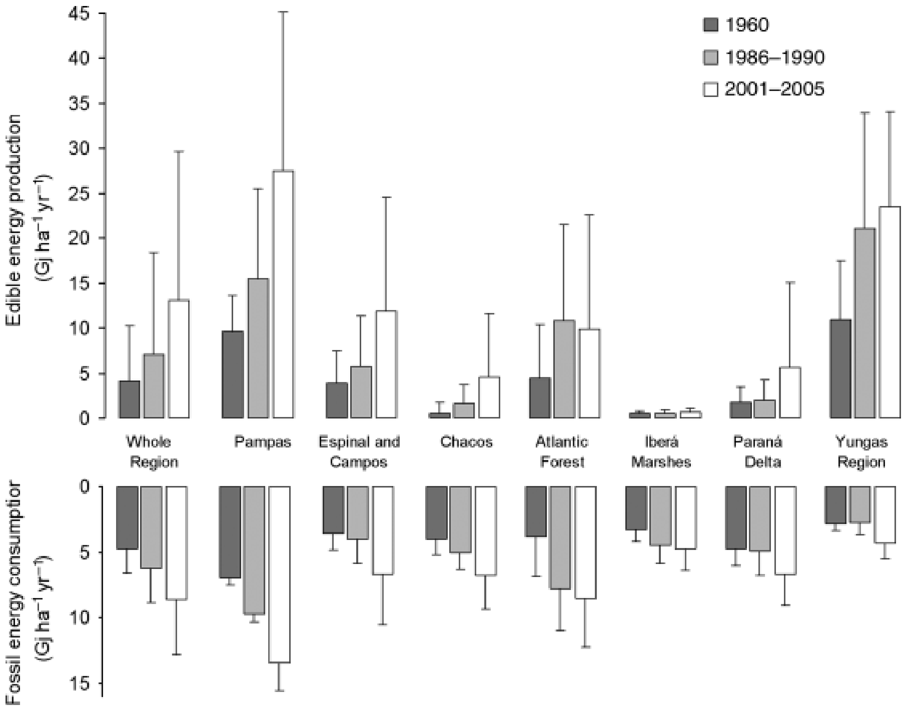

Fig. 7 Estimated relationship between the rate of fossil energy consumption and edible energy production in different ecoregions during the three studied periods. Error bars are standard deviations.

declined in all eco-regions (Table 3) because biomes of high water demand vegetation (forests and shrublands) were replaced by croplands and pastures with lower demand (Wullschleger *et al*., 1998). Empirical evidence demonstrates that forest removal alters the surface and groundwater dynamics (Jobbágy & Jackson, 2004; Jackson *et al*., 2005).

Environmental impact risk

The balances for soil N and P suggested nil pollution risk by these nutrients, so regressions were not included in Table 4. Beyond positive N balances in our calculation method, the amounts of residual N were too small to cause N pollution. However, despite that an

Table 3 Estimated average values of water-use efficiency

	Water-use efficiency (%)					
	1960		1986–1990		2001–2005	
Ecoregion	Mean	SD	Mean	SD	Mean	SD
Pampa	64.83	32.37	65.71	67.75	58.99	73.96
Espinal and Campos	74.71	34.32	76.69	61.07	72.32	68.81
Chaco	87.70	97.94	85.67	128.04	81.75	141.28
Atlantic Forest	74.83	12.12	71.65	17.94	70.59	22.40
Iberá Marshes	88.11	24.08	84.15	30.45	82.51	35.25
Paraná Delta	80.95	20.33	84.43	32.85	75.55	32.65
Yungas Region	86.61	24.34	84.89	32.73	78.61	34.83
Total	78.42	116.41	78.00	167.67	73.09	184.90

underestimation of pollution risk cannot be discarded, it is likely that nutrient pollution episodes can expand quickly across croplands if productivity is boosted through increased fertilization. A decline of the specific pesticide pollution risk took place throughout the whole period as organo-chlorinated products became replaced by phosphorated ones and, later, by hypermetrines (Fig. 8a). Even though the specific toxicity of pesticides decreased with time (Environment Agency, 1999), pollution risk increased in 2001–2005 as a result of the expansion of agriculture (Fig. 9). Erosion risk (water and wind) showed a dramatic decrease in response to the expansion of no-till agriculture in the last two periods of our study (Fig. 8b) (Alvarez *et al.*, 1998). Conversely, habitat intervention increased as a result of crop expansion. Furness & Greenwood (1993) demonstrated that bird species decline when natural lands were converted into croplands. Although some erratic results (Bilenca *et al.*, 2008), recent investigations in the Pampas (Schrag *et al.*, 2009) showed that bird species richness tends to correlate positively with natural vegetation, and negatively with annual crops.

With the exception of the Pampas, where no-till involved less use of fossil fuels, GHGs emission increased across time in all eco-regions (Fig. 10), being larger where they were subjected to high deforestation (Atlantic Forest and Yungas) and/or where burning was used to manage grasslands and shrublands (Espinal and Campos). Regression analysis showed stocks, fluxes and impacts responded to the proportion of cultivated area (Table 4). The aggregated general trends were used to elaborate a hypothetical model (Fig. 11) that showed some outstanding patterns including (i) C, N, P stocks in biomass and soil declined, particularly in the case of P, (ii) flux of energy, C, N and P increased significantly in response to the increasing agricultural productivity, depleting nutrient stocks, (iii) the risk of pesticide pollution and soil erosion decreased

Table 4 Regression analysis between stocks, flows and impacts versus the percentage of annual crops

	1960				1986–1990				2001–2005			
Stock/flow/impact	Intercept	Slope	R^2/P	SE	Intercept	Slope	R^2/P	SE	Intercept	Slope	R^2/P	SE
Total C stock	86.14	−0.76	0.13 (<0.01)	36.24	77.20	−0.74	0.22 (<0.01)	31.75	72.89	−0.63	0.27 (<0.01)	29.01
Total N stock	6.42	−0.03	0.06 (<0.01)	1.99	5.87	−0.04	0.23 (<0.01)	1.77	5.61	−0.04	0.33 (<0.01)	1.67
Total P stock	0.56	−0.01	0.02 (0.01)	0.16	0.52	−0.01	0.22 (<0.01)	0.14	0.50	−0.01	0.34 (<0.01)	0.13
Fossil energy input	4090.1	68.68	0.36 (<0.01)	1,708.7	5339.9	78.34	0.34 (<0.01)	2,448.4	6407.4	122.11	0.58 (<0.01)	2,909.2
Productive energy output	2555.9	217.76	0.18 (<0.01)	8,575.4	5412.4	338.86	0.12 (<0.01)	20,399.7	3148.1	589.70	0.64 (<0.01)	12,409.5
C flow	−0.03	−0.01	0.73 (<0.01)	0.13	−0.10	−0.01	0.81 (<0.01)	0.15	−0.13	<0.01	0.59 (<0.01)	0.10
N flow	7.91	0.36	0.30 (<0.01)	10.43	6.46	0.20	0.15 (<0.01)	10.82	11.46	−0.09	0.11 (<0.01)	7.37
P flow	−0.65	−0.05	0.25 (<0.01)	1.66	−1.17	−0.08	0.20 (<0.01)	3.62	−0.55	−0.09	0.54 (<0.01)	2.26
Soil erosion	1.05	0.22	0.80 (<0.01)	2.08	2.55	0.16	0.55 (<0.01)	3.24	2.63	0.07	0.26 (<0.01)	3.14
Pesticide risk	11.93	2.98	0.65 (<0.01)	40.66	4.68	0.60	0.48 (<0.01)	14.10	4.12	0.92	0.81 (<0.01)	12.50
Habitat intervention	0.11	0.01	0.62 (<0.01)	0.11	0.19	0.01	0.70 (<0.01)	0.12	0.19	0.01	0.81 (<0.01)	0.11
GHGs emission	−0.36	0.11	0.44 (<0.01)	2.33	1.12	0.06	0.21 (<0.01)	2.49	1.85	0.05	0.15 (<0.01)	3.28

Intercept, slope, determination coefficients (R^2), *P*-values (*P*) and standard error (SE) of the relationships between stocks, flows and impacts and the percentage of annual crops in the three studied periods. GHG, greenhouse gas; C, carbon; N, nitrogen; P, phosphorus.

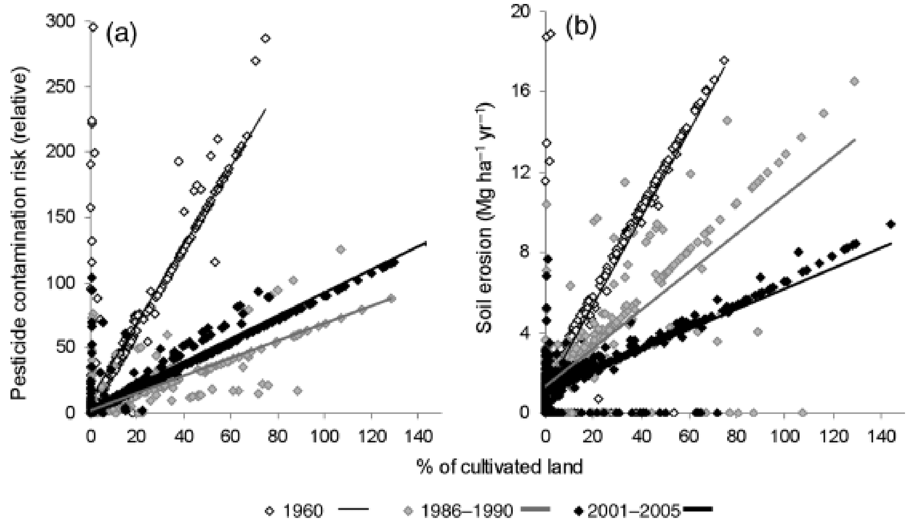

Fig. 8 Relationships between (a) pesticide contamination risk, (b) soil erosion and cultivation intensity for the three studied periods. R^2- and P-values are 0.65 and 0.01 (1960); 0.48 and 0.01 (1986–1990); 0.81 and 0.01 (2001–2005) for pesticide contamination risk; and 0.80 and 0.01 (1960); 0.55 and 0.01 (1986–1990); 0.26 and 0.01 (2001–2005) for soil erosion.

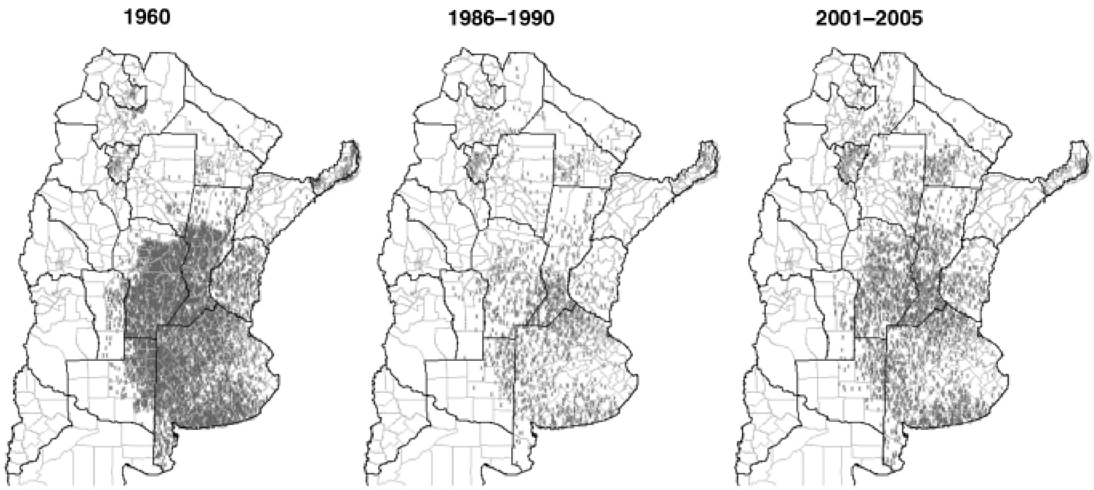

Fig. 9 Geographical display of estimations of pesticide contamination risk across the Argentine territory during the three studied periods. 1 dot $= 1 \times 10^6$ relative units ha^{-1}.

dramatically as a consequence of massive adoption of technology innovations, (iv) however, due to cultivation expansion, the negative effect of human intervention on habitat and biodiversity would have increased, (v) water-use efficiency decreased as the result of the increasing expansion of annual crops.

Uncertainty assessment

Various constraints affect certainty in this study: systems were characterized by dominant farming activities only, a mix of data and information sources were utilized, field validation of results was unviable at our low-resolution level of analysis, methods may not be sensitive enough to detect small change of inputs, all calculation procedures were not equally sound, and some initial assumptions may be questionable. However, beyond limitations, results allowed the display of patterns, gradients and trends across time and space that, in turn, may support strategic decisions for policy-making.

Results of our uncertainty assessment were summarized in Table 5. The uncertainty of land-use area values for grasslands, pastures and croplands was low but

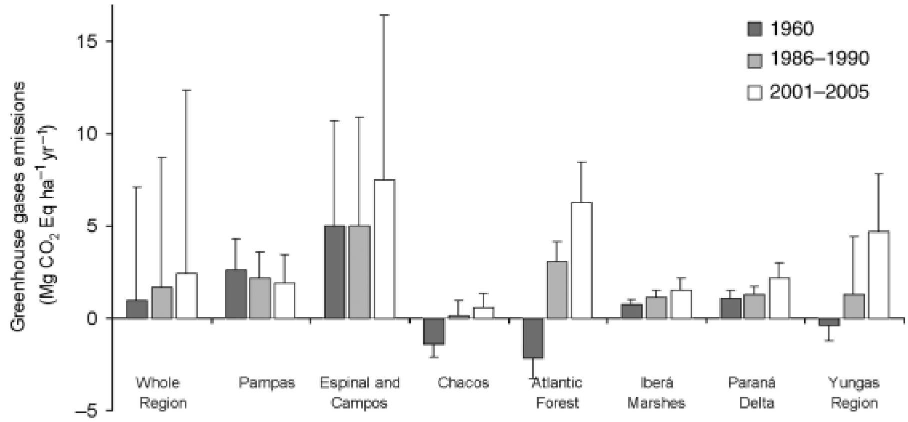

Fig. 10 Estimated greenhouse gases emissions in different ecoregions during the three studied periods. Error bars are standard deviations.

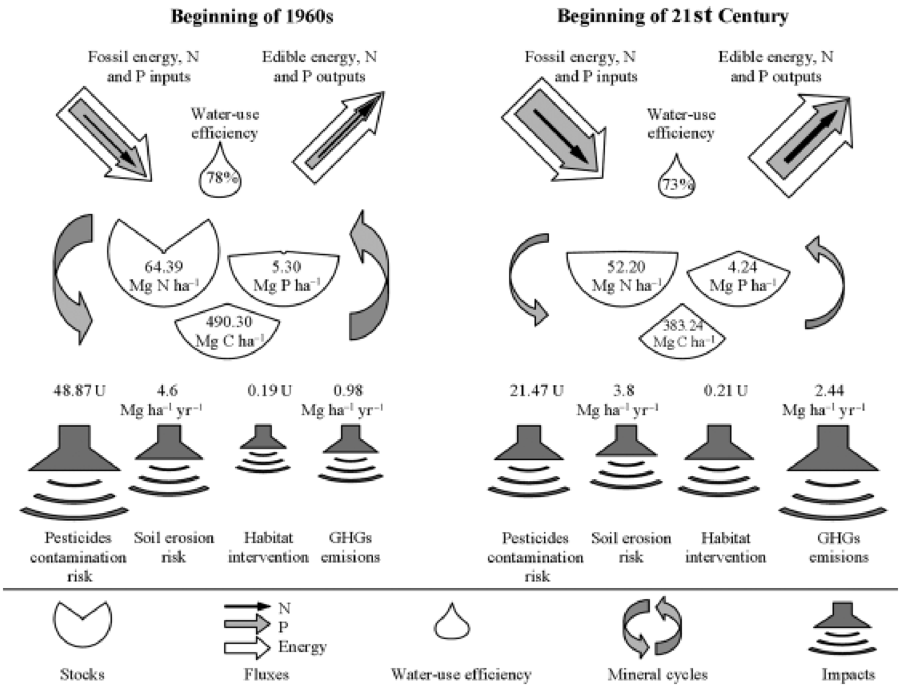

Fig. 11 Functional alterations on ecosystems and environmental impacts due to cultivation expansion in Argentina during the last 50 years.

higher for forest, as shown for the ratio of estimates derived from two independent statistics (agricultural censuses of INDEC and national forest inventories of SAyDS) tested at the provincial level These ratios were close to the unit (1.04 in average) in most provinces, but departed from it in Catamarca (1.25), Corrientes (0.61), Entre Ríos (0.65), La Rioja (1.60) and San Luis (1.85). Beyond figures in Table 5, our interpretation may be questioned if we consider values of standard deviation in Table 2, which in general are high. Therefore, we accept that there is room for different interpretations according to the method applied to assess uncertainty. Uncertainties for soil stocks of C, N and P, and of water consumption, ranged between low and moderate. However, because of missing field studies, we assigned very high uncertainty to estimations of nutrient and pesticide pollution risk.

Vertical patterns of SOC are also a source of uncertainty because they have implications on the balance of C and related nutrients. Most evidence shows that the effects of land-use and management practices on SOC primarily occurred within the top 30 cm of the soil (Zhang *et al.*, 2006). Effects in deeper soil layers have been poorly explored, although globally, the depth

Table 5 Uncertainty assessment for selected variables

		Estimations*		Reference values		
Variable	Units	Mean	SD	Mean	SD	Uncertainty estimation
Land use	%	44.55	47.22	ND	ND	From low to high†
Soil C stock	$Mg\,ha^{-1}$	39.02	14.31	45.03	16.93	32 % – Low‡
Soil N stock	$Mg\,ha^{-1}$	4.29	1.43	4.95	1.86	48 % – Medium‡
Soil P stock	$Mg\,ha^{-1}$	0.39	0.14	0.5	0.19	50 % – Medium‡
Biomass C–N–P stocks	$Mg\,ha^{-1}$	ND	ND	ND	ND	High§
Water-use efficiency	%	71.65	19.33	68.93	26.46	27 % – Low¶
Nutrient pollution risk	Relative index	ND	ND	ND	ND	High‖
Pesticide pollution risk	Relative index	21.47	71.66	ND	ND	High§

*Mean and SD values for the whole agricultural area of Argentina.
†This estimation depends on the method applied to assess uncertainty (see text).
‡After review data from Steinbach & Alvarez (2006).
§Data lacking for comparison.
¶Data from various sources.
‖Data from this study did not show signs of N or P pollution.
SD, standard deviation; C, carbon; N, nitrogen; P, phosphorus; ND, no data.

Table 6 Energy, N and P balances and soil erosion in Argentina and other countries

Country	Fossil energy consumption ($GJ\,ha^{-1}\,yr^{-1}$)	Edible energy production ($GJ\,ha^{-1}\,yr^{-1}$)	Nitrogen balance ($kg\,ha^{-1}\,yr^{-1}$)	Phosphorus balance ($kg\,ha^{-1}\,yr^{-1}$)	Soil erosion ($Mg\,ha^{-1}\,yr^{-1}$)	Comments
Argentina[1, 10]	5.0[a]	5.5[a]	+ 12.6[a]	−1.2[a]	8.5[a]	Erosion data from pampas only
	6.6[b]	10.7[b]	+ 9.3[b]	−2.9[b]	11.3[b]	
	9.0[c]	15.9[c]	+ 9.6[c]	−2.1[c]	8.8[c]	
	2.0	21.2				
The Netherlands[2]		62.4	+ 115.0		> 50.0	
UK[3, 4, 5, 9]	26.4[a]	58.6[a]	+ 20.0[c]		0.1–0.4	[a]Wheat
	30.1[b]	83.7[b]				[b]Maize
	21.4[c]	3.9[c]				[c]Beef
China[6, 9, 10]	25.4	62.2	+ 227.0	+ 53.0	220.0–536.0	Inland China
Japan[8, 10]	115.8	47.5	+ 135.0		50.0–250.0	
Scandinavia[8, 9, 10]	15.3	30.3	+ 19.0		0.5–2.5	Sweden and Denmark
France[8]			+ 53.0		50.0–250.0	
Canada[7, 8, 9, 10]	6.0	10.8	+ 13.0		> 50.0	
	6.9	17.4				
USA[6, 8, 9, 10]	12.6	25.6	+ 10.0	−9.0	50.0–250.0	Mid-West states
New Zealand[8, 10]	60.2	37.3	+ 6.0			
Brazil[8, 9, 10]	5.4	25.0	−8.6		> 50.0	Brazilian Cerrado
Nigeria[2, 5, 9, 10]	1.3	12.0	−22.0		10.0–50.0	
Kenya[6, 9]			−52.0	+ 1.0	195.0	

[1]Results from this study: [a]1960, [b]1986–1990 and [c]2001–2005 periods; [2]Giampietro *et al*. (1999); [3]Spedding (1979); [4]Spedding & Walsingham (1975); [5]Stoorvogel & Smaling (1990) and Frissel (1978); [6]Vitousek *et al*. (2009); [7]McRae *et al*. (2000); [8]Organisation for economic co-operation and development (2001); [9]Lal (1994); [10]Estimations from Giampietro *et al*. (1999) and Conforti & Giampietro (1997) for grains only.

interval of 30–100 cm represents 47% of the total 0–100 cm stock (Jobbágy & Jackson, 2000; Meersmans *et al*., 2009). Slower turnover rates with depth suggest less intense SOC changes with land use for this deep stock, yet measurable responses have been reported (Osher *et al*., 2003; Yang *et al*., 2008). Available syntheses indicate that average SOC losses following the onset of cultivation may be around 50% and 30% for the top 20 and 100 cm of the soil, respectively (Post & Kwon, 2000). Considering that SOC declines exponentially with depth (IPCC, 2000), and C : N ratio does not exhibit significant change across the soil profile (Yang *et al*., 2008), we can accept that most uncertainty sources regarding nutrient stocks in croplands locate on the top 0–30 cm layer.

Conclusions

Results showed that 50 years of agricultural transformation in Argentina caused significant structural and functional changes on the rural environment. In the mid- and long term, the most concerning issues to be considered are biomass C loss and habitat destruction due to deforestation, and an increasing depletion of soil P stocks. Two positive consequences are the reducing risk of pollution and soil erosion because of the generalized adoption of less aggressive pesticides and no-till practices. Despite the decreasing water-use efficiency of agriculture can be seen as a disadvantage, it should be noted that ecosystems may be yielding more water under current conditions, what could represent an opportunity in some environments.

Comparing our results for Argentina with international figures (Table 6), it can be appreciated that energy consumption, energy productivity, and N and P balance, and soil erosion estimations were lower than those of some European countries, China, Japan, New Zealand and the United States. Argentine farmers produced food under a relatively low-input and low-impact production scheme during the last decades. This, however, may change in the near future if the global food demand drives additional expansion and intensification.

Although the study has analytical limitations and looks ambitious both in terms of spatial/temporal coverage and comprehensiveness of the processes involved, it provides information that sensibly used can potentially be helpful to orientate land-use policies and agricultural strategies.

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References

Adámoli J (2006) Problemas ambientales de la agricultura en la región chaqueña. In: *La Situación Ambiental Argentina 2005* (eds Brown A, Martinez Ortiz U, Acerbi M, Corcuera J), pp. 436–442. Fundación Vida Silvestre Argentina, Buenos Aires.

Alvarez R, Díaz RA, Barbero N, Santanatoglia OJ, Blotta L (1995) Soil organic carbon, microbial biomass and CO_2–C production from three tillage systems. *Soil Tillage Research*, **33**, 17–28.

Alvarez R, Russo ME, Prystupa P, Scheiner JD, Blotta L (1998) Soil carbon pools under conventional and no-tillage systems in the Argentine Rolling Pampa. *Agronomy Journal*, **90**, 138–143.

Baethgen WE (1992) Dinámica del nitrógeno en sistemas de rotación cultivo-pasturas. *Revistas de Investigaciones Agropecuarias INIA*, **1**, 3–25.

Bilenca D, Codesido M, González Fischer C (2008) Cambios en la fauna pampeana. *Ciencia Hoy*, **18**, 8–17.

Brenzoni E, Rivero ET (1996) *Di-nitrogen fixation measurements in Medigaco sativa (L) using ^{15}N isotope dilution*. Proceedings of the XV Argentine Congress on Soil Science, Santa Rosa LP, Argentina, pp. 78.

Carreño L, Viglizzo EF (2007) *Provisión de Servicios Ecológicos y Gestión de los Ambientes Rurales en Argentina* (eds INTA), Buenos Aires.

CASAFE (1997) *Guía de Productos Fitosanitarios para* Cámara de Sanidad Agropecuaria y Fertilizantes, Buenos Aires.

Cerqueira E, Lesser A, Roberto ZE, Viglizzo EF (1986) Sistemas zonales de producción lechera en la región pampeana argentina. II Factores relacionados con el rendimiento de leche por hectárea. *Revista Argentina de Producción Animal*, **6**, 353–360.

Conforti P, Giampietro M (1997) Fossil energy use in agriculture: an international comparison. *Agriculture, Ecosystems & Environment*, **65**, 231–243.

Cruzate G, Casas RR (2003) Balance de nutrientes. *Fertilizar*, Año 8, Número Especial Sostenibilidad, pp. 7–13.

Doyle CJ, Lazenby A (1984) The effect of stocking rate and fertilizer usage on income variability for dairy farms in England and Wales. *Grass and Forage Science*, **39**, 117–127.

Dussart E, Lerner P, Peinetti R (1998) Long term dynamics of two populations of *Prosopis caldenia* Burkart. *Journal of Range Management*, **51**, 685–691.

Ehrlich PR, Ehrlich AH, Holdren JP (1977) *Ecoscience: Population, Resources, Environment*. WH Freeman and Company, San Francisco, CA.

Environment Agency (1999) *Pesticides in the Aquatic Environment*. Environment Agency, Wallingford, UK.

Ewers RM, Scharlemann JPW, Balmford A (2009) Do increases in agricultural yield spare land for nature? *Global Change Biology*, **15**, 1716–1726.

FAO (1992) *CROPWAT: a computer program for irrigation planning and management*. FAO Irrigation and Drainage, Technical Paper No. 46, Rome.

Frissel MJ (ed.) (1978) *Cycling of mineral nutrients in agricultural systems*. Elsevier Sci. Publ. Co., Amsterdam, Holland.

Furness RW, Greenwood JJD (1993) *Birds as Monitors of Environmental Change*. Chapman & Hall, London.

Galantini JA, Suñer L (2008) Las fracciones orgánicas del suelo: Análisis en los suelos de la Argentina. *Agriscientia*, **45**, 41–55.

Gasparri NI, Grau HR, Manghi E (2008) Carbon pools and emissions from deforestation in extra-tropical forests of Northern Argentina between 1900 and 2005. *Ecosystems*, **11**, 1247–1261.

Giampietro M, Bukkens SGF, Pimentel D (1999) General trends of technological change in agriculture. *Critical Reviews in Plant Sciences*, **18**, 261–282.

Givens DI, Moss AR (1990) *UK Tables of Nutritive Value and Chemical Composition of Feeding Stuffs*. Blacklaws Printers, Aberdeen, UK.

Gutierrez Boem FH, Alvarez CA, Cabello MJ, Fernández PL, Prystupa P, Taboada MA (2008) Phosphorus retention on soil surface of tilled and no-tilled soils. *Soil Science Society of America Journal*, **72**, 1158–1162.

Hagen LJ (1991) Wind erosion mechanisms: abrasion of aggregated soil. *Transactions of the American Society of Agricultural Engineers*, **34**, 891–937.

Hepper G, Hevia G, Buschiazzo DE, Urioste AM, Bono A (1996) Efectos de la agricultura sobre fracciones de fósforo en suelos de la región semiárida pampeana central. *Ciencia del Suelo*, **14**, 96–99.

Holmes W (1980) *Grass, its Production and Utilization*. The British Grassland Society, Blackwell Scientific Publications, Oxford.

Hooker TD, Compton JE (2003) Forest ecosystem carbon and nitrogen accumulation during the first century after agricultural abandonment. *Ecological Applications*, **13**, 299–313.

Huan WZ (1996) *Dynamics of carbon, nitrogen, phosphorus and sulphur in a boreal aspen forest soil*. PhD thesis, University of Saskatchewan, Canada, 188 pp.

IAASTD (2009) *Agriculture at a Crossroads: Global Report*. International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD), Island Press, Washington, DC.

INDEC (1964, 1991, 2004) *Censos Nacionales Agropecuarios 1960, 1988 y 2002*. Instituto Nacional de Estadística y Censos de la República Argentina, Buenos Aires. Available at http://www.indec.mecon.gov.ar/ (accessed 2008).

INTA (1990) *Atlas de Suelos de la República Argentina*. Ediciones INTA, Instituto Nacional de Tecnología Agropecuaria, Buenos Aires, 388 pp.

IPCC (2000) Land use, land-use change and forestry. Chapter 2: implications of different definitions and generic issues. In: *Summary for Policymakers. A Special Report of the Intergovernmental Panel on Climate Change* (eds Watson RT, Noble IR, Bolin B, Ravindranath NH, Verardo DJ, Dokken DJ), pp. 24–28. Cambridge University Press, UK.

IPCC (2003) *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories*. WMO-UNEP Edition, Kamiyagamuchi, Japan.

IPCC (2006) Revised 2006 IPCC Guidelines for National Greenhouse Gas Inventories. In: *Reference Manual, Prepared by the National Greenhouse Gas Inventories Programme* (eds Eggleston HS, Buendia L, Miwa K, Ngara T , Tanabe K), , Vol. 4, Chapters 2–7, pp. 2.1–7.24. IGES, Japan. Available at http://www.ipcc-nggip.iges.or.jp/

Jackson RB, Jobbágy EG, Avissar R *et al*. (2005) Trading water for carbon with biological carbon sequestration. *Science*, **310**, 1944–1947.

Jobbágy EG, Jackson RB (2000) The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications*, **10**, 423–436.

Jobbágy EG, Jackson RB (2004) Groundwater use and salinization with grassland afforestation. *Global Change Biology*, **10**, 1299–1312.

Jones RJ, Sandlands RL (1974) The relation between animal gain and stocking rate. *Journal of Agricultural Science*, **83**, 335–342.

Lal R (1994) Sustainable land use systems and soil resilience. In: *Soil Resilience and Sustainable Land Use* (eds Greenland DJ, Szabolcs I), pp. 1–9. CAB International, Wallingford, UK.

McRae T, Smith CAC, Gregorich LJ (ed.) (2000) *Environmental Sustainability of Canadian Agriculture: Report of the Agri-Environmental Indicator Project*. Agriculture and Agri-Food Canada, Ottawa, ON.

Meersmans J, van Wesemael B, De Ridder F, Fallas Dotti M, De Baets S, Van Molle SM (2009) Changes in organic carbon distribution with depth in agricultural soils in Northern Belgium, 1960–2006. *Global Change Biology*, **15**, 2739–2750.

Michelena RO, Irurtia CB, Vavruska F, Mon R, Pittaluga A (1989) *Degradación de suelos en el Norte de la Región Pampeana. INTA. Proyecto de Agricultura Conservacionista*. Publicación Técnica No. 6. Estación Experimental Agropecuaria INTA – Pergamino.

Mott GO (1960) Grazing pressure and the measurement of pasture production. In: *Proceedings of the 8th International Grassland Congress*. Alden Press, Oxford pp. 606–611.

Murphy GM (ed.) (2008) *Atlas Agroclimático de la Argentina*, pp. 58–82. Editorial Facultad de Agronomía, Universidad de Buenos Aires, Buenos Aires.

Nadkrmiz NM, Matelson J (1992) Biomass and nutrient dynamics of epiphytic litterfall in a neotropical montane forest, Costa Rica. *Biotropica*, **24**, 24–30.

National Research Council (NRC) (1978) *Nutrient Requirements of Domestic Animals*. National Academy of Sciences, Washington, DC.

Odum EP (1969) The strategy of ecosystems development. *Science*, **164**, 262–270.

Odum EP (1975) *Ecology: The Link Between the Natural and Social Sciences*. Holt, Reinhart and Winston, San Francisco, CA.

Odum HT (1971) *Environment, Power and Society*. Wiley-Interscience, New York.

Organisation for economic co-operation and development (ed.) (2001) Use of farm inputs and natural resources. In: *Environmental Indicators of Agruculture: Methods and Results*, Vol. 3, pp. 117–139. OECD Report, Paris, France.

Osher LJ, Matson PA, Amundson R (2003) Effect of land use change on soil carbon in Hawaii. *Biogeochemistry*, **65**, 213–232.

Panigatti JL, de Hein WIH (1985) Mineralización potencial del nitrógeno en epipedones del norte de la región pampeana. *Ciencia del Suelo*, **3**, 54–60.

Paruelo JM, Oesterheld M, Del Pino F *et al*. (2004) *Patrones Espaciales y Temporales de la Expansión de Soja en Argentina: Relación con Factores Socio-Económicos y Ambientales*. Informe final LART/FAUBA al Banco Mundial.

Post WH, Kwon KC (2000) Soil carbon sequestration and land-use change: processes and potential. *Global Change Biology*, **6**, 317–328.

Rabinovich JE, Torres F (2004) *Caracterización de los Síndromes de Sostenibilidad del Desarrollo: El Caso de Argentina*. CEPAL/Naciones Unidas, Santiago de Chile.

Rearte D (2007) *Situación Actual de la Producción de Carne Vacuna*. Documento SAGPyA/INTA. Available at http://www.sagpya.mecon.gov.ar/(accessed 2008).

Ruehlmann J, Körschens M (2009) Calculating the effect of soil organic matter concentration on soil bulk density. *Soil Physics*, **73**, 876–885.

SAGPyA (2009) *Estimaciones Agrícolas de Cereales, Oleaginosas, Algodón, Maní y Poroto*. Secretaría de Agricultura, Ganadería, Pesca y Alimentos de la República Argentina. Available at http://www.sagpya.mecon.gov.ar/(accessed 2009).

Salvador C (2002) The environmentally positive aspects of the agrochemical industry in Argentina. In: *The Impact of Global Change and Information on the Rural Environment* (eds Solbrig OT, Di Castri F, Paarlberg R), pp. 201–215. Harvard University Press, USA.

Sanmartino F (2006) *Anuario del Campo Argentino 2005/2006*. Ediciones La Nación, año 2, número 2, Buenos Aires.

Satorre EH (2005) Cambios tecnológicos en la agricultura actual. *Ciencia Hoy*, **15**, 24–31.

SAyDS (2007a) *Monitoreo de la Superficie de Bosque Nativo de Argentina*. Secretaría de Ambiente y Desarrollo Sustentable de la República Argentina. Available at http://www.ambiente.gov.ar/?idarticulo=311 (accessed 2008).

SAyDS (2007b) *Informes Nacional y Regionales del Primer Inventario Nacional de Bosques Nativos*. Secretaría de Ambiente y Desarrollo Sustentable de la República Argentina. Available at http://www.ambiente.gov.ar/?idarticulo=316 (accessed 2008).

Schrag AM, Zaccagnini ME, Calamari N, Canavelli S (2009) Climate and land-use influences on avifauna in central Argentina: broad-scale patterns and implications of agricultural conversion for biodiversity. *Agriculture, Ecosystems and Environment*, **132**, 135–142.

SENASA (2004) *Mercado de Fertilizantes*. Servicio Nacional de Sanidad Agropecuaria de la República Argentina. Available at http://www.sagpya.mecon.gov.ar/ (accessed 2006).

SENASA (2008) *Movimiento de Hacienda 2007*, Informe Estadístico N° 2. Servicio Nacional de Sanidad Agropecuaria de la República Argentina. Available at http://www.sagpya.mecon.gov.ar/ (accessed 2009).

Solbrig OT (1997) Ubicación histórica: Desarrollo y problemas de la pampa húmeda. In: *¿Argentina Granero del Mundo? ¿Hasta Cuándo?* (eds Morello J, Solbrig OT), pp. 29–40. Orientación Gráfica Editora, Buenos Aires.

Spedding CRW (1979) *An Introduction to Agricultural Systems*. Applied Science Publishers, London.

Spedding CRW, Walsingham JM (1975) The production and use of energy in agriculture. *Journal of Agriculture Economics*, **27**, 19–30.

Steinbach HS, Alvarez R (2006) Changes in soil organic carbon contents and nitrous oxide emissions after introduction of no-till in Pampean agroecosystems. *Journal of Environmental Quality*, **35**, 3–13.

Stoorvogel JJ, Smaling EMA (1990) *Assessment of soil nutrient depletion in sub-Saharian Africa, 1983–2000*. Report 28. Winan Staring Center for Integrated Land, Soil and Water Research (SC-DLO), Wageningeng, Holland.

Suñer L, Galantini J, Rosell R (2005) Cambios del fósforo en suelos bajo diferentes rotaciones de cultivos. *Informaciones Agronómicas*, March 2005, pp. 5–8.

Tilman D, Cassman KG, Matson PA, Naylor RL, Polasky S (2002) Agricultural sustainability and intensive production practices. *Nature*, **418**, 671–677.

Verdegem MC, Bosma RH, Verreth JA (2006) Reducing water use for animal production through aquaculture. *Water Resources Development*, **22**, 101–113.

Viglizzo EF (1982) *Los potenciales de producción de carne en la región pampeana semiárida*. Primeras Jornadas Técnicas sobre Producción Animal en la Región Pampeana Semiárida. Universidad Nacional de La Pampa, Santa Rosa, pp. 233–269.

Viglizzo EF (1993) La productividad de la ganadería argentina: La brecha tecnológica. *Agromercado*, **80**, 16–21.

Viglizzo EF, Lértora FA, Pordomingo AJ, Bernardos J, Roberto ZE, Del Valle H (2001) Ecological lessons and applications from one century of low external-input farming in the pampas of Argentina. *Agriculture, Ecosystems and Environment*, **81**, 65–81.

Viglizzo EF, Pordomingo AJ, Castro MG, Lértora F (2003) Environmental assessment of agriculture at a regional scale in the pampas of Argentina. *Environmental Monitoring and Assessment*, **87**, 169–195.

Vitousek PM, Naylor R, Crews T *et al*. (2009) Nutrient imbalances in agricultural development. *Science*, **324**, 1519–1520.

Weber J (1994) Properties and behavior of pesticides in soil. *Journal of Environmental Quality*, **24**, 684–690.

Wischmeyer WH, Smith DD (1978) *Predicting rainfall erosion losses. A guide to conservation planning. Agric. Handbook No. 537*. US Department of Agriculture, Washington, DC.

Woodruff NF, Siddoway FH (1965) A wind erosion equation. *Soil Science Society of American Proceedings*, **29**, 602–608.

Wullschleger SD, Meinzer FC, Vertessy RA (1998) A review of whole-plant water use studies in trees. *Tree Physiology*, **18**, 499–512.

Yang YH, Fang JY, Tang YH, Ji CJ, Zheng CY, He JS, Zhu B (2008) Storage, patterns and controls of soil organic carbon in the Tibetan grasslands. *Global Change Biology*, **14**, 1592–1599.

Zak DR, Grigal DF, Gleeson S, Tilman D (1990) Carbon and nitrogen cycling during old-field succession: constraints on plant and microbial biomass. *Biogeochemistry*, **11**, 111–129.

Zak DR, Pregitzer KS (1990) Spatial and temporal variability of nitrogen cycling in northern lower Michigan. *Forest Science*, **36**, 367–380.

Zhang L, Dawes WR, Walker GR (2001) Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resources Research*, **37**, 701–708.

Zhang X, Chen L, Fu B, Li O, Oi Z, Ma Y (2006) Soil organic carbon changes as influenced by agricultural land use and management: a case study in Yanhuai Basin, Beijing, China. *Acta Ecologica Sinica*, **26**, 3198–3203.

Zimmer D, Renault D (2002) *Virtual water in food production and global trade: review of methodological issues and preliminary results*. Proceeding Expert Meeting on Virtual Water No. 11. Delft, the Netherlands, pp. 93–109.