Environmental Monitoring and Assessment (2006) **117:** 109–134
DOI: 10.1007/s10661-006-7981-y

A RAPID METHOD FOR ASSESSING THE ENVIRONMENTAL PERFORMANCE OF COMMERCIAL FARMS IN THE PAMPAS OF ARGENTINA

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(Received 25 October 2004; accepted 25 May 2005)

Abstract. The generation of reliable updated information is critical to support the harmonization of socio-economic and environmental issues in a context of sustainable development. The agro-environmental assessment and management of agricultural systems often relies on indicators that are necessary to make sound decisions. This work aims to provide an approach to (a) assess the environmental performance of commercial farms in the Pampas of Argentina, and (b) propose a methodological framework to calculate environmental indicators that can rapidly be applied to practical farming. 120 commercial farms scattered across the Pampas were analyzed in this study during 2002 and 2003. Eleven basic indicators were identified and calculation methods described. Such indicators were fossil energy (FE) use, FE use efficiency, nitrogen (N) balance, phosphorus (P) balance, N contamination risk, P contamination risk, pesticide contamination risk, soil erosion risk, habitat intervention, changes in soil carbon stock, and balance of greenhouse gases. A model named *Agro-Eco-Index* was developed on a Microsoft-Excel support to incorporate on-farm collected data and facilitate the calculation of indicators by users. Different procedures were applied to validate the model and present the results to the users. Regression models (based on linear and non-linear models) were used to validate the comparative performance of the study farms across the Pampas. An environmental dashboard was provided to represent in a graphical way the behavior of farms. The method provides a tool to discriminate environmentally friendly farms from those that do not pay enough attention to environmental issues. Our procedure might be useful for implementing an ecological certification system to reward a good environmental behavior in society (e.g., through tax benefits) and generate a commercial advantage (e.g., through the allocation of green labels) for committed farmers.

Keywords: Argentine Pampas, commercial farms, ecological certification, environmental assessment, sustainability indicators

1. Introduction

In recent years, current and potential environmental problems are frequent cause of concern in Argentina. It is increasingly accepted that the healthy administration of the rural environment is essential to maintain or improve the productivity of land, the economic income, the social condition and the integrity of bio-physical systems that preserve the quality of life. The generation of reliable updated information is critical to support the harmonization of socio-economic and environmental issues in a context of sustainable development.

The agro-environmental assessment and management of agricultural systems often relies on indicators that are necessary to make sound decisions. The purpose of agro-ecological indicators is to facilitate the diagnosis and interpretation of critical processes in order to improve later the decision capacity (Girardin *et al.*, 1999). Normally, they are developed by scientists as quantitative expressions that measure a particular condition of the system in relation to accepted ecological threshold values (Schiller *et al.*, 2001). Users normally calculate indicators to orient decisions and communicate ideas to different actors in the society (Walter and Wilkerson, 1994).

Ideally, scientists, decision-makers and the public should agree on what ecological indicators are needed. But agreement may be difficult because interests may diverge and communication among parties may fail. The selection of universally accepted, cost-effective indicators is necessary to overcome such constraint (Shaeffer *et al.*, 1988; Munasinghe and Shearer, 1995). Frequently, models that simulate complex agricultural systems are considered better than simple indicators to describe a changing reality. But a well-selected set of indicators still appears to be irreplaceable as a tool to synthesize information and orientate farmers' decisions in a rapid way (Girardin *et al.*, 1999).

The use of indicators is neither simple nor automatic. The process of selection, developing and validation of indicators may impose a practical trouble. There is not a unique way to select, develop and validate environmental indicators in agriculture (Smith *et al.*, 2000). Very frequently, indicators arise more in response to occasional local or temporal needs than to universally accepted criteria. But beyond selection, validation is another important step in indicators adoption. Validation is not common because indicator developers tend to consider that a long-term acceptance by users guarantee reliability. But this does not necessarily guarantee usefulness as diagnostic- or decision-support tool. Bockstaller and Girardin (2003) have demonstrated the importance of validation before adoption, and proposed a methodological framework to do it.

The potential provision of technical coefficients for agro-ecological certification should not be underestimated as a by-product of indicators adoption (Viglizzo *et al.*, 2001). Many agricultural firms in the world are considering the environmental consequences of their activities and how can they affect their competitive advantage and social credibility (Wall *et al.*, 2001). The development of reliable indicators can speed-up the willingness of farmers to be monitored by outside, independent organizations (Teisl *et al.*, 1999) and to prove their environmentally friendly behavior. Nowadays, it is widely accepted that indicators can provide to outside, third parties a tool to monitor and certify the application of good-practice principles through standardized environmental codes, such as those of ISO 14000.

Accepting the need and potential benefit of having reliable agro-environmental indicators, the objectives of this work were to provide an approach to (a) assess the environmental performance of commercial farms in the Pampas of Argentina, and (b) propose a methodological framework to calculate environmental indicators that can rapidly be applied to practical farming. As a by-product of this study, we also aimed at discussing the potential application of such indicator to assess the environmental performance of farms and guide first steps to the ecological certification of agricultural processes.

2. Materials and Methods

2.1. The pampas region and the study farms

The Pampas region is a vast, flat region of Argentina that comprises more than 50 million hectares of arable lands for crop and cattle production (Hall *et al.*, 1992). Given that the region is not homogeneous in soils and rainfall patterns (INTA, 1990; Satorre, 2001), it can be divided into five agro-ecologically homogeneous areas (Figure 1) as follows: (1) Rolling Pampas, (2) Central Pampas, which can be subdivided in Sub-humid on the East and Semiarid on the West, (3) Southern Pampas, (4) Flooding Pampas, and (5) Mesopotamian Pampas. Rainfall regimes

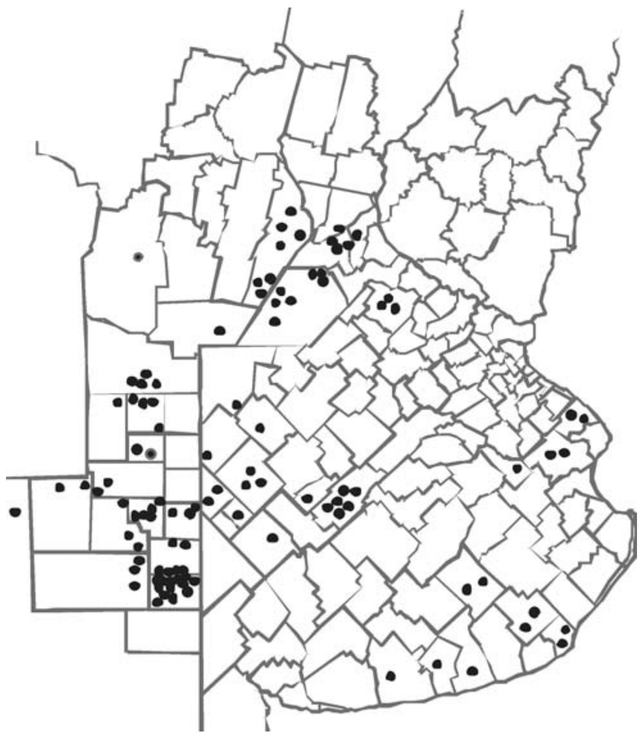

Figure 1. Geographical location of study farms in the Pampas region.

vary across time and space, causing occasional droughts and flood episodes that transitorily can affect both crop and cattle production (Viglizzo *et al.*, 1997).

According to FAO (1989) criteria, deep and well-drained soils predominate on the Rolling Pampas, which provides conditions for continuous cropping (INTA, 1990). Despite increased wind erosion sensitivity towards the West, most of the lands are suitable for mixed cultivation and livestock production in the Central Pampas (Buschiazzo *et al.*, 1999). The Flooding and Mesopotamian Pampas are mostly devoted to cattle production on native and introduced perennial pastures. Limitations for crop production in these areas are normally associated with shallow soil depth, frequent flooding, soil salinity, poor drainage, and water erosion.

120 real farms scattered across the five ecological areas in the Pampas (see dots location in map of Figure 1) were analyzed in this study during years 2002 and 2003. The selection of farms depended strongly on the willingness of farmers to be assessed and to provide requested field data. The geographical spread of farms guarantees the analysis of contrasting production methods and systems. The assessment was not focused on single products but on the whole-farm integrated production process. A standardized procedure was designed to pick up data at the farm level, and to load such data into a model that automatically calculates eleven basic indicators.

2.2. Indicators, models, data collection and statistical analysis

As driving criteria, we assumed that indicators need to be sound, simple to calculate, and easy to interpret and use by decision-makers. The selection of indicators was driven by environmental issues that are frequent cause of concern in agricultural production in the Pampas. The rapid land use change and the increasing use of external inputs during the 1990s have arisen concerns about the consumption of fossil energy, the nitrogen and phosphorus input-output relationship in farms, the risk of contamination by fertilizers and pesticides, the impact on the habitat, the loss of organic matter and soil sediments due to increasing cultivation, and the emission of greenhouse gases (Solbrig and Viglizzo, 1999; Viglizzo *et al.*, 2001, 2002b; Ghersa *et al.*, 2002; Ferraro *et al.*, 2003). Thus, eleven basic indicators were calculated: (1) fossil energy (FE) use, (2) fossil energy use efficiency, (3) nitrogen (N) balance, (4) phosphorus (P) balance, (5) nitrogen contamination risk, (6) phosphorus contamination risk, (7) pesticide contamination risk, (8) soil erosion risk, (9) habitat intervention, (10) changes in soil organic carbon (C) stock, and (11) balance of greenhouse gases (GHG). Methods for calculating this set of indicators were described by Viglizzo *et al.* (2003).

A software named *Agro-Eco-Index*, based on the above mentioned indicators, was developed on an Excel-Microsoft magnetic support in order to facilitate the incorporation of farm data and the calculation of indicators by users. The model was divided into five screens: three for data loading, one for showing the calculated numerical coefficients, and the last one to show the 11 estimated indicators through an understandable graphical device. This magnetic support, developed and calibrated only for the Pampas region at the moment, has already been patented in Argentina.

The assessment comprises a method for data collection at the farm level. At least one visit to each voluntary farmer is unavoidable. Detailed data on land use and land cover (annual crops, annual and perennial pastures, natural pastures, forests, waste areas, etc.), farming activities, use of inputs (fuels, electricity, fertilizers, pesticides and concentrates), livestock categories, stocking rates, local rainfall, evapotranspiration, and the water holding capacity of soils must be carefully recorded. Given that not always farmers have field data on evapotranspiration and water holding capacity of their soils, default data on the study area that we have previously incorporated into the model were used. Later, collected data must be uploaded to the model to calculate the indicators.

Land use, a major driver of change in farming, is a common factor for the calculation of the complete set of indicators. It is well known that land use is highly relevant in environmental assessments because of its direct impact on the environment (van Latesteijn, 1993; Rabbinge *et al.*, 1994; Viglizzo *et al.*, 2001). In our case, the main impact of land use was referred to the proportional allocation (%) of the land to (a) native rangeland, (b) introduced perennial pastures, or (c) annual crops.

The statistical methods were restricted to the use of simple regression analysis through linear and non-linear models. Statistics include the corresponding best-fit equation, the determination coefficient (R^2) and the standard error (SE).

2.2.1. *Fossil Energy Use (Indicator No. 1) and Fossil Energy User Efficiency (Indicator No. 2)*

The use of fossil energy (FE) correlates well with intensification in agriculture. Being usually linked to fuel use for farming activities and to the use synthetic inputs like fertilizers and pesticides, the increasing use of FE is frequently associated with environmental threats like greenhouse gases emission and soil/water contamination episodes (Agriculture and Agri-Food Canada, 2000).

The calculation of this indicator comprised the energy cost (Mj ha^{-1} year^{-1}) of predominant inputs (fertilizers, seeds, concentrates, pesticides) and practices (tillage, planting, weeding, harvesting, etc.). The fossil energy cost of inputs and practices were obtained from different sources (Reed *et al.*, 1986; Stout, 1991; Conforti and Giampietro, 1997; Pimentel, 1999). Although it was not possible to check the original sources and procedures used by those authors, we assume that they provide a reliable, peer-reviewed estimation of all involved fossil energy costs.

The fossil energy use efficiency was calculated by considering the amount of Mj of FE used to get one Mj of product. Calculations were made on annual basis taking into account the proportional participation of each analyzed activity. Under this scheme, the larger the amount of FE used to produce one unit of energy, the less efficient the production process was.

2.2.2. *Input–Output Relationship of Nitrogen (Indicator No. 3) and Phosphorus (Indicator No. 4)*

Among other nutrients, the adequate supply of nitrogen (N) and phosphorus (P) is essential to the plant growth and development. If extraction exceeds supply over the years, the accumulation of negative balances may cause nutrient depletion, declining crop yield and lowering of economic return. Conversely, if supply exceeds extraction, the accumulation of residues can overload the soil with nutrients and increase the risk of soil and water contamination.

A simplistic input-output relationship procedure was used to estimate a farm gate balance. The annual average of N and P in soil was estimated as the annual difference between inputs and outputs per hectare. The ways of nutrient gain in the case of N were (a) precipitation, (b) fertilizers, (c) biologic fixation by legumes, and (d) purchased feed, later excreted and returned to the field as cattle urine and manure. The predominant ways of P gain are fertilizers and purchased feed. The only one way for N and P export in this work is the nutrient extracted by the agricultural product. Because of the unavoidable calculation difficulty, issues of the nutrient cycle like accumulation, depletion, leaching, volatilization, and other pathways in the nutrient dynamics were not considered.

2.2.3. *Contamination Risk by Nitrogen (Indicator No. 5) and Phosphorus (Indicator No. 6)*

The assessment of contamination risk by N and P (expressed as mg l^{-1} of drainage water) is essential for assessing threats in intensive agriculture. The risk of N contamination was calculated by taking into account the residual N when the N balance was positive. The N concentration in drainage water was estimated by dividing the amount of residual N by the amount of water available for nitrogen dilution (water excess). The water excess was calculated on annual basis from a water balance estimation, which takes into account the water gain by rainfall (mm year^{-1}) less the real evapotranspiration in the same period. The contamination risk calculation proceeded only when the excess of water has exceeded the water holding capacity of soils. Where field data were not available, we utilized default values for water holding capacity of average soils according to McDonald (2000). They were: (a) 100 mm for a sandy or a sandy loam soil, (b) 150 mm for a loam soil, (c) 200 mm for a loam clay soil, and (d) 250 mm for a clay soil. Therefore, if the excess of water (rainfall less real evapotranspiration) was less than the water holding capacity of the soil, saturation did not happen, leaching was absent and the water contamination risk did not occur.

A similar procedure was used to calculate the P contamination risk. This was done even when the process involved in P contamination is quite different from that related to N contamination. Such difference can be explained by the opposite mobility of both elements in the soil: while P immobility determines that runoff is the main way of contamination, lixiviation is the main contamination way in the case of N. Both, leaching and runoff coexist in the flat plain of the study region.

2.2.4. *Pesticide Contamination Risk (Indicator No. 7)*

Worldwide, the pesticide contamination risk is cause of concern due to (a) water and soil degradation by pesticides residues, (b) air quality degradation by the volatile fraction of pesticides, and (c) negative impacts on biodiversity. The pesticide contamination risk was quantified by a relative index because absolute values are not meaningful for between farms comparison. Calculation included the most common insecticides, herbicides and fungicides that farmers have extensively used during the assessment period. This indicator was based on the estimation of the relative toxicity of predominant pesticides used in different farming activities. The actual toxicity values (LD-50) were provided by manufacturers, and obtained from a well-known current pesticides guide (CASAFE, 1997). Land use allocation was the base layer of information on which pesticides, and their corresponding toxicity values, were later incorporated to the calculation. Data on application rate of active products were provided by farmers for each crop. The relative index for pesticide contamination was the result of summing-up each pesticide contribution per hectare, after multiplying the proportion (%) of land allocated to each analyzed crop, by the toxicity of each product.

Although our estimation of the contamination risk cannot be expressed in absolute terms, farms can be compared in terms of their relative contribution to toxicity generation by pesticide use.

2.2.5. *Soil Erosion Risk (Indicator No. 8)*

Soil quality was defined here as the soil capacity to sustain agricultural activities over time without affecting its productivity and the quality of the environment. Erosion has negative effects at the farm level, and also can impact larger areas outside the farm. Unsuitable land use schemes and tillage practices can be cause of erosion in fragile lands.

Different tillage operations have different effects on soil conservation. No-till consists of a weed control with herbicides and the seeding of crops without previous tillage operations. This is considered the less aggressive operation on soil. Although conservation tillage minimizes the number of tillage operations, it is more a aggressive alternative than no-tillage. The conventional tillage, the most aggressive one, consists of a mechanical weed control with diskers and harrow disk, repeating the same operation two, three or more times before seeding the crops.

Depending on soil stability, different tillage methods have different effect on sediment production by soil erosion. Accepted criteria (Agriculture and Agri-Food Canada, 2000) classify erosion risk into five categories: tolerable (less than 6 metrictons of sediment loss per hectare and year), low (6 to 11 tons), moderate (11 to 12 tons), high (22 to 33 tons), and heavy (more than 33 tons).

Two methods were utilized for estimating the soil erosion risk: (a) the Wind Erosion Equation (WEQ) for wind erosion, and (b) the Universal Soil Loss Equation (USLE) for water erosion. The first one (Woodruff and Siddo way, 1965; Hagen, 1991) contemplates soil properties (texture, organic matter and $CaCO_3$ contents), historical climate characteristics (mean wind speed, mean prevailing wind direction, annual precipitation and mean temperature), and management characteristics (field length, tillage system and vegetation cover). The latter (Wischmeyer and Smith, 1978) corresponds to water erosion and requires data on rain erosivity, soil susceptibility to water erosion, slope, field length and vegetation cover information. In our model, a unique wind and water erosion indicator was calculated and expressed in terms of t ha^{-1} $year^{-1}$ of soil loss.

2.2.6. *Human Intervention of the Habitat (Indicator No. 9)*

It is accepted that over the centuries agriculture has transformed the natural habitat into a human-designed one. Despite the fact that agriculture has historically benefited from biodiversity, habitat intervention by agriculture greatly reduces natural biodiversity. Land use change seems to be the most relevant factor that impacts on biodiversity (Sala *et al.*, 2000), but tillage operations and pesticides use are causes of habitat aggression as well.

Under the assumption that human action affects habitat and biodiversity, a method was designed in this work to calculate a relative index that estimates the degree of human intervention on the habitat. Intervention occurs through factors like (a) land use change, (b) tillage operations, and (c) pesticide applications. In our case, land use was the proportion (%) of land cultivated annually with annual crops. The tillage impact factor was obtained from the method to estimate soil erosion risk (t of sediments ha^{-1} $year^{-1}$). Likewise, the corresponding relative coefficient was obtained and used from the already estimated indicator on pesticide contamination risk. The emerging combined 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 the habitat. Thus, farms were compared in terms of their estimated relative impact on the habitat.

2.2.7. *Change in Soil Organic Carbon Stock (Indicator No. 10)*

Organic carbon (C) is the main component of soil organic matter, and therefore a factor that strongly determines soil quality. Organic matter decay is associated with soil fertility and soil structure losses, and also with higher soil erosion risk. Depending on organic matter gain or loss, soils can respectively act as a sink or a source of atmospheric C. Thus, C stocks are dynamic and highly sensitive to human action.

The procedure followed in this work to estimate changes in C stock (expressed as t ha^{-1} $year^{-1}$) was based on the IPCC (1996) revised guideline methodology. An initial C stock (obtained from field data or default figures) that varied according to the study area was necessary to initiate the calculation, and 20 years was the basic period considered to estimate changes. Factors associated with crops and pastures vary: while perennial pastures improved the C stocks, annual crops depleted such stock. Imported manure is not incorporated in agricultural systems of the Pampas. Tillage practices were associated with coefficients that represented different C oxidation rates. While conservation- and no-till operations favored soil C sequestration, conventional tillage was cause of C emission. Furthermore, the IPCC method has also provided default coefficients for organic matter enrichment of soils via accumulation of crop residues.

2.2.8. *Greenhouse Gases (GHG) Balance (Indicator No. 11)*

Atmospheric gases, including carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), ozone (O_3) and water vapor are normal components of the atmosphere that have, at the same time, a greenhouse effect. CO_2 and CH_4 account for 90% of the human-driven greenhouse effect (Desjardins and Riznek, 2000). Grain crops and cattle are both significant sources of greenhouse gases emission.

The GHG balance was estimated following the standard guidelines of IPCC (1996). All gases were converted into CO_2 equivalent (ton ha^{-1} year^{-1}). The calculation has included emission and sequestration of carbon in response to land use change, grain cropping and cattle production activities.

The use of fossil fuels was a major source of CO_2. The method included fuels used in rural activities and fuels used for manufacturing fertilizers, herbicides and machinery. Because of the heterogeneity of cases and procedures, the emission due to rural transportation was not included in our calculations. Ruminants were a significant source of GHG. Ruminants emit methane from enteric fermentation and fecal losses. Methane has a greenhouse power that is 21 times greater than CO_2. This figure was used to convert CH_4 into CO_2 equivalents. Nitrogen excreted in feces and distributed with fertilizers was another significant source of nitrous oxide (N_2O) emission. N_2O has a greenhouse power 310 times greater than CO_2 (IPCC, 1996). Losses of N_2O occur via volatilization, leaching and runoff. Arable soils were also a direct source of greenhouse gases through fertilizers, biological N fixation and crop residues.

When data from direct field measurements were unavailable, default values suggested by the IPCC (1996) were used for estimating gains and losses of carbon. The methodology proposed by IPCC (1996) estimated C emission or sequestration through the following components: (1) CO_2 stock exchange in soils over time (CO_2-SC), (2) CO_2 stock exchange in timber biomass (CO_2-BL), (3) conversion of forests and prairies into arable land (CO_2-CTBP), (4) abandonment of intervened lands (CO_2-aband), and (5) emission of CO_2 from fossil fuels burning (CO_2-CF) in different agricultural activities. The procedure estimates CH_4 emission from 3 sources: (1) enteric fermentation (CO_2-FE) from domestic animals, (2) fecal emissions (CO_2-EF), and (3) rice crop emissions (CO_2-EA). The emission of N_2O was the most difficult to estimate because of the complexity of determinations. Emission sources are: (1) Feces and urine (CO_2-EDHO) from domestic animals, (2) volatilization, runoff and infiltration (CO_2-EIVLI) from synthetic fertilizers and animal excrements (urine and feces), and (3) arable soils (CO_2-EDSA), through chemical fertilizers, biological N fixation and crop residues. Therefore, the final equation for estimating the CO_2 balance was:

$$\begin{aligned}CO_2\text{ balance} = {} & (CO_2\text{-SC} + (CO_2\text{-BL} + CO_2\text{-CTBP} + CO_2\text{-aband}) \\ & + CO_2\text{-CF}) + ((CO_2\text{-FE} + CO_2\text{-EF}) \times 21) \\ & + ((CO_2\text{-EDHO} + CO_2\text{-EIVLI} + CO_2\text{-EDSA}) \times 310)\end{aligned}$$

2.3. Model validation

Inspired in validation procedures used in simulation models, Bockstaller and Girardin (2003) have recently proposed a methodological framework to validate indicators. Three types of validation procedures have been identified by these authors: (1) design validation, (2) output validation, and (3) end-use validation. Validation in the first case occurs when the indicator design was supported by the best scientific information available. Output validation is common in modeling science, and it is based on the soundness of results or, at least, on the low probability of refuting them. End-use validation, on the other hand, is based on the usefulness that indicators have demonstrated to provide reliable diagnosis and sound decision-support in practical farming. Given that the third validation involves a long-term period that cannot be fulfilled in our case because our development was very recent, we will focus on the two first validation types.

Within the Bockstaller and Girardin (2003) scheme, the design soundness of our indicators lies on the fact that most indicators were based on peer reviewed procedures and results from well-founded scientific publications. Likewise, the design of some indicators was subjected to the scrutiny of regional researchers that provisionally validate the calculation procedure.

On the other hand, the output validation was done by comparing the outcomes of the model with empirical data from research, experimentation, and field measurements. We consider that this procedure allows a well-founded empirical validation. The only one exception was related to indicators that provide relative indexes that are not commonly used. Model outputs were validated through two main procedures: (a) by comparing the ranges of variability of empirical data with the corresponding range of model outputs, and (b) by analyzing the performance of farms across the region in response to land use change. Land use has a demonstrable and predictable effect (Rabbinge *et al.*, 1994) on the environmental behavior of farms. Then, the rationality of indicators response to land allocation (% of annual crops) in the collection of study farms was used as a means for model validation. A simple regression analysis using linear and non-linear models was applied in order to detect statistically significant relations.

2.4. Output representation and threshold levels

Different procedures were used for presenting results to the users. For the whole sample of farms, trend lines were used to describe the environmental performance of such farms across the whole Pampas in response to a varying proportion of annual crops.

For individual farms, on the other hand, a dashboard that shows the performance of each environmental indicator was designed to facilitate the farmer interpretation of its own performance. Each indicator was represented by a ruler which is ranked into three different areas with colors that resemble the colors of traffic lights. Different tones of green, yellow and red were used to represent the environmental risk in an understandable way. But in this paper we have used different tones in grey to represent the same effect: light grey represents low risk on the left side (equivalent to green), mid-grey represents moderate risk on the middle part of the rule (equivalent to yellow), and dark grey indicates high risk (equivalent to red) on the right side of the rule (Figure 2).

No doubt that determining the environmental performance of farms through color ranges that represent different risk conditions was a trouble that had to be overcome. The first approach to define ranges and threshold limits was to collect quantitative data from scientific literature. But normally, figures from other agricultural areas in the world were rather different from figures that we had been getting from the Pampas. We reached the conclusion that comparisons were not useful in a strict sense because outside data did not adapt well to the agro-environmental conditions of the Pampas. So we finally decided to identify, for each indicator, specific environmental ranges and thresholds that are applicable only to the Pampas. Therefore,

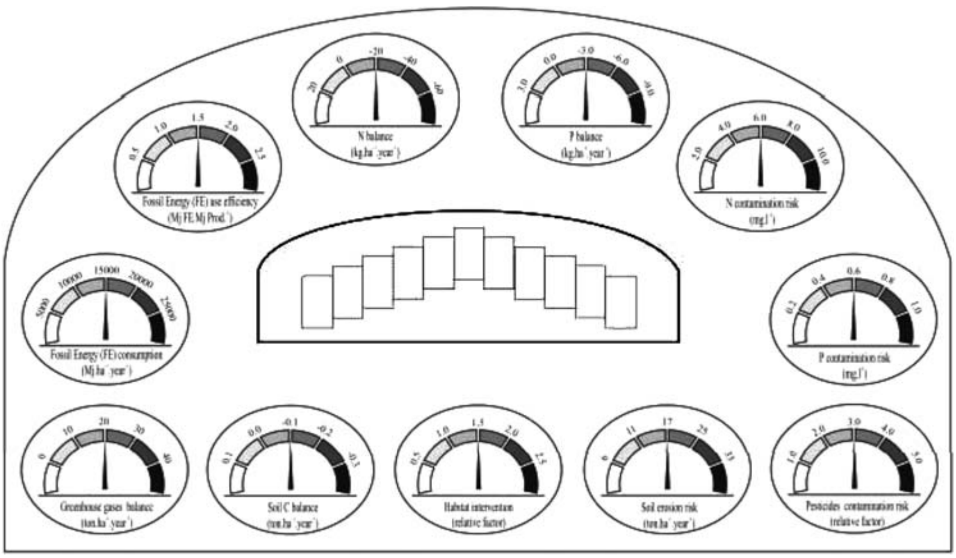

Figure 2. Detail of the dashboard diagram that is used to assess the environmental performance at the farm-scale in the Argentine Pampas. Details about indicators and their units are provided in the text (see Sections 2.2.1–2.2.8).

despite it may be considered subjective, the local adaptation to do between-farm comparisons became useful and reliable.

Once specific ranges suitable for the Pampas were estimated, threshold limits were calculated to identify tolerance limits. For each indicator, tolerance limits were estimated through the variability range of figures within the already study set of 120 farms in the Pampas. Farms with good performance provided figures to set the range and the tolerance limits for the light-grey area. The same procedure was followed to set figures for the mid- and the dark-grey areas, respectively. A clock-hand was used, for each farm, to indicate the exact position of each indicator in the ranked range. Eleven small rectangular boxes were located in the middle of each farm dashboard to set the color determined for each indicator. Thus, the performance of different farms can be compared by the dominance of colors that represent low, medium or high environmental risk. For example, when light grey predominates in a given farm, we assume that such farm has been performing with low environmental risk. On the contrary, when dark colors predominate, a warning signal suggests that corrective actions might be required.

3. Results and Discussion

3.1. Validation through the comparison of variability ranges

A general comparison of outputs from Agro-Eco-Index indicators with empirical evidence from different sources is presented in Table I. Although mean values and variability coefficients of outputs easily arise in our model, the comparison with empirical figures from other sources cannot easily be done. Our figures can be compared with figures from outside sources only for a limited number of indicators (e.g., fossil energy use, use efficiency of fossil energy, N balance and greenhouse gases balance), but not for the remaining indicators. Therefore, our outputs are meaningful only for people living in the Pampas, but not necessarily for people living outside the Pampas. Besides, it must be taken into account that the output range can vary widely both in the case of Agro-Eco-Index and other outside sources. Unavoidably, this conspires against the rigorous validation of the model.

Accepting such constraints, we have assumed that our model would behave acceptably well if the variability range of our estimations falls within the variability range of scientific data from different sources. As it can be appreciated from Table I, the variability range of results from the analysis of 120 farms in the Pampas has fallen within the variability range of data from different scientific sources.

Beyond the already pointed out constraints, some other limitations in indicators calculation must be taken into account: (1) data on fossil energy consumption provided by literature were not necessarily updated in relation to technology change. Occasionally, imprecise description of calculation has raised the risk of double accounting in the case of some inputs, (2) the dynamics of nutrient pools and fluxes

TABLE I

Comparison of Agro-Eco-Index estimations with research and experimental results, and field measurements

Environmental	Agro-Eco-Index model estimations		Research and experimental results, and field measurements in different countries and areas in the Pampas		
indicator	±SD	Range of results	Range of values	Sources	Data from
Fossil energy (FE) use (Gj ha^{-1} year^{-1})	7.15 ± 5.97	0.04–41.20	3.99–100.40	(1)–(4)	Argentina, UK, Nigeria
Use efficiency of FE (Gj FE Gj prod^{-1})	0.69 ± 1.15	0.05–9.47	0.18–5.56	(1)–(4)	Argentina, UK, Nigeria
N balance (kg ha^{-1} year^{-1})	21.82 ± 28.65	−76.62–(−102.87)	−22.00–115.00	(4)–(6)	Argentina, Brazil, France, Netherlands, Nigeria, UK
P balance (kg ha^{-1} year^{-1})	−8.50 ± 7.99	−3.18–1.34	−12.71–(−0.20)	(4) (7)	Whole Pampas
N contamination risk (mg l^{-1})	8.43 ± 22.57	0.00–22.51	0.00–136.00	(8)–(13)	Rolling, Southern, Mesopotamian and Central Pampas
P contamination risk (mg l^{-1})	0.01 ± 0.07	0.00–0.66	0.00–7.60	(9)	Rolling Pampas
Pesticide contamination risk (relative coefficient)	1.34 ± 4.57	0.00–44.55	0.02–39.40	(14)	Whole Pampas
Soil erosion risk (ton sediments ha^{-1} year^{-1})	5.66 ± 8.25	−7.22–27.81	0.00–69.00	(15) (16)	Rolling and Central Pampas involving both, conventional and no-till

(Continued on next page)

TABLE I
(*Continued*)

Environmental indicator	Agro-Eco-Index model estimations		Research and experimental results, and field measurements in different countries and areas in the Pampas		
	±SD	Range of results	Range of values	Sources	Data from
Habitat intervention (relative coefficient)	1.23 ± 1.52	0.00–15.46	1.0–30.20	(14)	Whole Pampas
Change in soil C stock (ton ha^{-1} $year^{-1}$)	−0.13 ± 0.15	−0.47–0.50	−2.11–(−0.1 5)	(17) (18)	Rolling Pampas under continuous cropping
Greenhouse gases balance (ton ha^{-1} $year^{-1}$)	9.07 ± 1130[a]	−2.67–(21. 00)[a]	0.87–2.93	(14) (19) (20)	Argentina, Brasil, Bolivia, Chile, Paraguay and Uruguay

[a]High values of greenhouse gases emissions obtained from various farms analyzed through the Agro-Eco-Index model can be explained by the use of fire as common practice to manage rangelands and cultivated pastures. Source references: (1) Giampietro *et al.* (1999); (2) Spedding (1979); (3) Spedding and Walsingham (1975); (4) Viglizzo *et al.* (2003); (5) Frissel (1978); (6) Stoorvogel and Smiling (1990); (7) Vazquez (2002); (8) Andriulo *et al.* (2002a); (9) Andriulo *et al.* (2002b); (10) Costa *et al.* (2002); (11) Jouli *et al.* (2002); (12) Weir (2002); (13) Papparotti and Machiori (2002); (14) Viglizzo *et al.* (2002a); (15) Bernardos *et al.* (2001); (16) Michelena and Irurtia (1995); (17) Alvarez *et al.* (1999); (18) Andriulo *et al.* (1999); (19) Gonzalez (2000); (20) W. R. I. (1996).

was ignored because of practical reasons; then, simple input-output relations do not accurately reflect nutrient cycling in real farming, (3) estimations of nutrient contamination risk carry away some uncertainty emerging from (2). The same happens with respect to the hydrological dynamics (rainfall, evapotranspiration and soil water holding capacity), (4) in relation to the pesticide contamination risk, aggregated figures at the farm level ignores potential interactions between pesticides. Likewise, we have set aside the specific effect of pesticides on non-target species, (5) although the estimation of soil erosion risk was based on the application of universally accepted models, more field data measurements in the region should had been necessary to validate the calculation, (6) the method used to estimate human intervention on the habitat has resulted from previously calculated indicators, saying nothing about the real impact on biodiversity. Ideally, the method should be focused on habitat fragmentation and its impact on key species, (7) the proposed method to estimate changes in soil organic carbon stock is rather simplistic in relation to the complex dynamics of C in real farming. Land use, climate variability, soil and tillage interactions, plus agronomic practices are sources of uncertainty in our estimations, (8) the revised procedure suggested by IPCC (1996) to estimate greenhouse gases balance is relatively new, and therefore carries away some degree of uncertainty. Ideally, on-farm and field measurements should support such estimations.

3.2. Validation through the analysis of farms performance across the study region

Beyond the evidence from the comparison of output variability ranges, the soundness of results was also checked through between-farm analysis. This validation was based on the selection of one driving factor that has a well-known environmental impact on farms. Scientific evidence (Viglizzo *et al.*, 1997, 2001, 2002a; Bernardos *et al.*, 2001) from the Pampas suggests that land use is an outstanding factor that has affected the functionality (energy flows, carbon dynamics, nutrient cycles and hydrological processes) of ecosystems during the last century.

The results from the regression analysis in which the % of crops (as land use factor) was associated with various dependent variables (X) are presented in Table II. The best fitted model was a second degree equation in all cases. With the exception of phosphorus input-output relation, which showed a statistically significant relation ($P < 0.05$), determination (R^2) and correlation (R) coefficients were highly significant ($P < 0.01$) for the rest of dependent variables.

When the environmental performance of farms was checked against land use change, it was appreciated that indicators behaved according to the expected response (Figure 3). The energy productivity (Figure 3a) and the fossil energy use (Figure 3b) have increased in response to increasing cultivation. In our case, the energy productivity represents the energy yield of edible products that were exported from the farm. To get such expression, all products were previously converted into

TABLE II

Results from regression analysis using a binomial model to assess the relation between the percentage of crops as independent variable (Y) and various dependent variables (X) in 120 farms scattered across the Pampas region

% crops (Y) vs.	Intersection	a	b	R^2	R	P	SE
Energy productivity	10186.10	−462.93	10.40	0.494	0.702	<0.01	22247.50
Fossil energy consumption	4657.07	2.62	0.54	0.099	0.315	<0.01	5716.19
N input–output relation	19.56	0.62	−0.008	0.124	0.351	<0.01	27.05
P input–output relation	3.12	−0.14	0.0007	0.069	0.263	<0.05	7.78
Pesticide contamination risk	0.77	−0.054	0.0009	0.095	0.309	<0.01	4.39
Soil erosion risk	5.03	0.42	−0.0026	0.392	0.626	<0.01	6.03
Change in soil carbon stock	−0.0015	−0.0065	0.00005	0.123	0.351	<0.01	0.14
Greenhouse gases (GHG) balance	18.02	−0.2198	0.0009	0.090	0.300	<0.01	11.85

energy by taking into account their caloric value (Viglizzo *et al.*, 2003). Figures 3a and b show that responses agree with data from different sources (Spedding and Walsingham, 1975; Spedding, 1979; Viglizzo and Roberto, 1998; Giampietro *et al.*, 1999; Pervanchon *et al.*, 2002; Viglizzo *et al.*, 2002b) that assessed the effect of intensification on farming systems. Such evidences have demonstrated that the higher the percentage of annual crops in the study farms the larger their energy productivity and their consumption of fossil energy. This relation is confirmed in Figures 3a and b.

The input-output relation of N and P deserves special attention. As it was mentioned above, cultivation in the Pampas during the 1990's comprised an increasing allocation of land to annual crops, and this was particularly noticeable in the Rolling Pampas. While N fertilization was usual, fertilization with P remained rather low across the region. However, a crop-pasture rotation scheme involving N-fixing legumes still persists in many areas of the Pampas providing significant amounts of N. As a result of this doubled source of N, the model outputs have detected positive N input-output relations all across the region. Higher N surpluses agreed with an intermediate % of crops and a high percentage of legume-based pastures (Figure 3c). However, given that surpluses were not extremely high, they did not represent a significant threat in terms of N contamination. On the other hand, the predomination of negative P balances across the region, and particularly in the Rolling Pampas, was the result of low levels of P fertilization (Figure 3d). Similar trends were found in previous works (Viglizzo *et al.*, 1995, 2001). Given that not all N-surplus is dissolved in water excess (part is lost through gaseous pathways), we admit that these estimations are only a rough and simplistic approach of reality. This constraint was accepted in our model in order to preserve the speed and

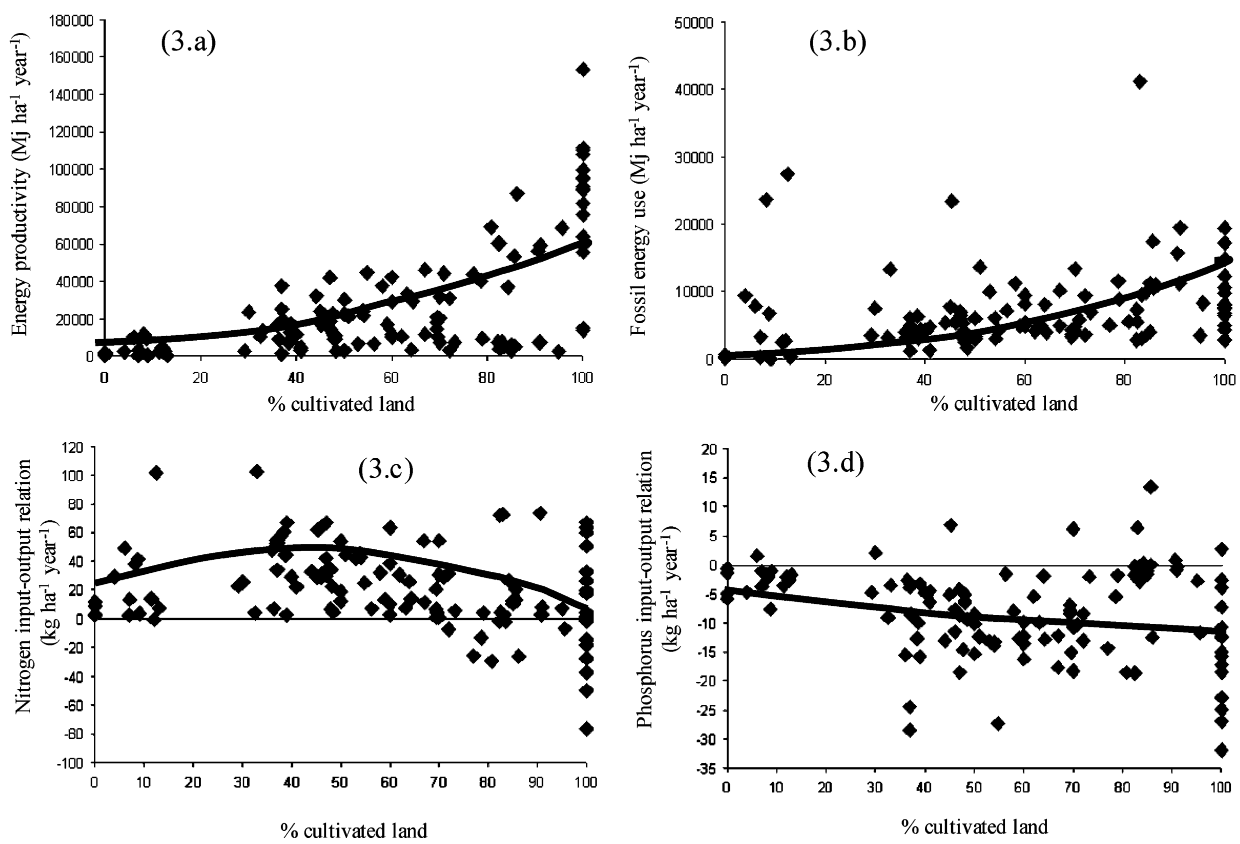

Figure 3. Response of energy productivity (3a), fossil energy use (3b), nitrogen (3c) and phosphorus input-output relations (3d) to the increasing percentage of cultivated land in the study farms. Response of pesticides contamination risk (3e), soil erosion risk (3f), change of the organic carbon stock in soils (3g) and greenhouse gases balance (3h) to the increasing percentage of cultivated land in the study farms. Thick full line: average trend. Thin full line: trend under conventional tillage; thin dashed line: trend under no-tillage.

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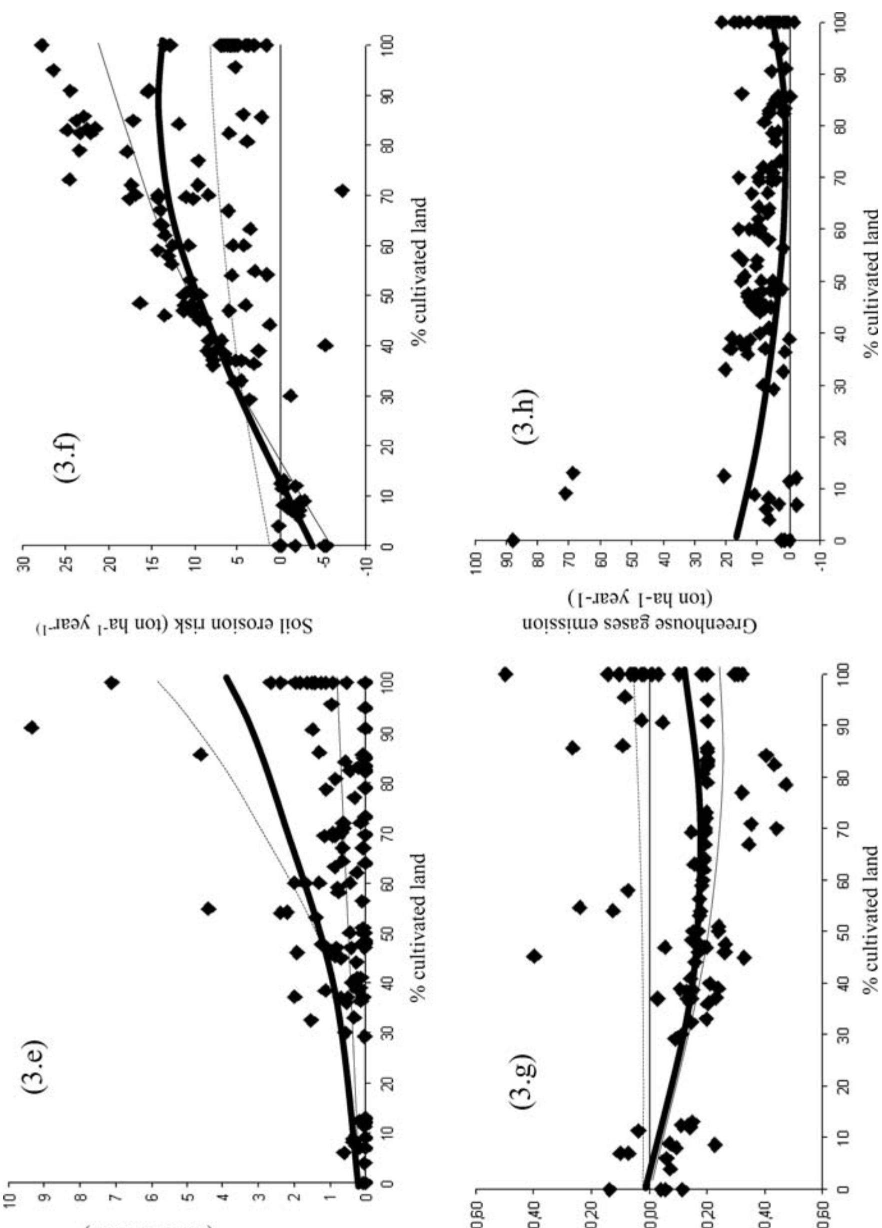

Figure 3. (Continued)

simplicity of calculation under practical farming. The detection of contamination risk episodes due to N and P overloading was rare in this study, and this agreed with field measurements and experimental results from Andriulo *et al.* (2002) in the Rolling Pampas. On the other hand and as it was expected, the contamination risk by pesticide use (Figure 3e) and the soil erosion risk (Figure 3f) were higher in areas where annual crops predominate. The different impact of conventional- and no-tillage on soil erosion risk can be appreciated in Figure 3f. These results were confirmed through experimental evidence and field measurements (Aimar *et al.*, 1996; Buschiazzo *et al.*, 1998, 1999). This demonstrates that the erosion risk increases in response to the relative expansion of the cropping area. However, such risk tends to be lower in farming systems which have already adopted no-till practice. Counter-balancing this, the risk of pesticide contamination tended to increase in no-tilled cropping systems.

The impact of no-till can also be appreciated in the analysis of changes in the stock of soil organic carbon stock (Figure 3g). While C depletion can largely be explained by increasing cropping under conventional tillage, no-till not only seems to prevent C depletion but also to boost a long-term C gain in soils. This behavior can be confirmed through experimental results (Alvarez *et al.*, 1999, 2001; Alvarez, 2001; Chagas *et al.*, 1995; Miglierina *et al.*, 2000; Studdert and Echeverría, 2000).

Greenhouse gases (GHG) emission in response to land use change is also in line with scientific information. GHG emission tends to decrease as far as the area devoted to crop production increases (Figure 3h). This emission has two main sources: (a) methane (CH_4) emission by cattle digestion and (b) GHG emission by rangeland burning in extensive beef cattle production. The last source was particularly relevant in the case of cow-calf farming systems. These results are confirmed through literature information (Viglizzo *et al.*, 2002a; González, 2000; W.R.I., 1996).

It may be argued that this part of the validation process is supported by a circular argumentation. For example, given that specific technical coefficients were used to represent no-till, conservation or conventional tillage operations, the model outcomes will reflect this effect and validation might turn invalid. But not necessarily is this argumentation true because the independent variable was the percentage of annual crops, but not the area allocated to a particular tillage operation.

3.3. Characterization of the Environmental Performance of Single Farms through a Dashboard Analysis

As mentioned above, an indicators dashboard was designed to better visualize the environmental performance of individual farms. Three examples that respectively correspond to a dairy (Figure 4a), a mixed crop-beef (Figure 4b) and a continuous cropping (Figure 4c) real system in the Southern Pampas are showed in Figure 4. According to the position of the clock-hand in the ranked rule, the comparative behavior of each farm in relation to each environmental indicator can be interpreted.

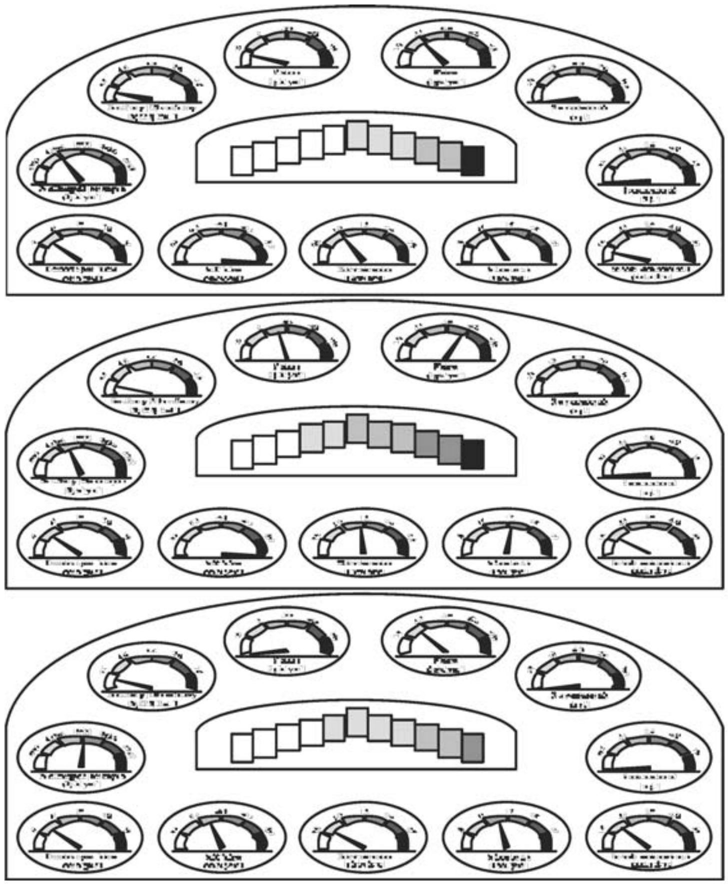

Figure 4. The environmental dashboard applied to three typical farm systems in the Pampas to facilitate the interpretation by users. LG (4a), EA (4b) and ES (4c) are the initial letters of three selected systems in the sample of 120 study farms. The different performance of farms can be appreciated by comparing similar indicators.

A warning signal was detected when the clock-hand tends to the right side. Thus, farms failing in environmental management can be identified and eventual technological solutions can be driven. For example, the application of no-till operations can help to resolve problems of high fossil energy consumption and soil erosion when such problems are detected through the dashboard analysis.

After analyzing a farm sample, between-farm comparisons are possible in order to contrast their respective environmental performance. As an example, and based on the information provided by their respective dashboards, the comparative behavior of nine different farms in the Southern Pampas are represented in Figure 5 by taking into account the central set of colored boxes of each dashboard. Farms identified as LL and ET where light colors predominate (9 over 11) showed a better performance than farms where dark colors dominate.

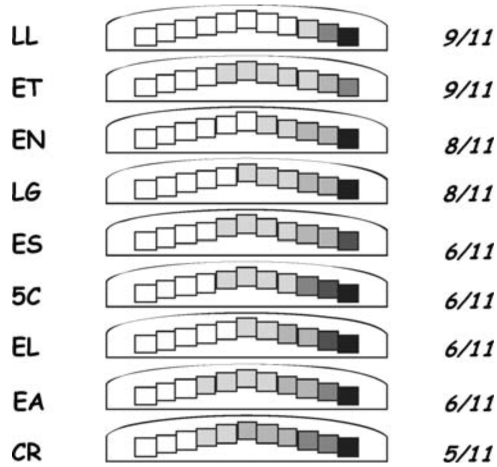

Figure 5. Comparative environmental behavior of nine farms (see initial letters on the left column) in the Southern Pampas. The accumulation of light boxes on total boxes (11) indicates better behavior (for reference of this, see figures on the right column).

4. Conclusions

Beyond limitations in our approach, methods and outcomes, the scheme we propose can help to discriminate among farms in the Pampas of Argentina, separating those that show a friendly behavior from those that do not pay enough attention to the surrounding environment. This discrimination is not a minor challenge in an environmentally sensitive society. An organized community should not ignore the effort of farms and other commercial firms to improve the environmental status of the area where they locate. It sounds sensible to think that such effort should be encouraged by recognizing and rewarding the environmental commitment of farming firms.

Based on the principle that a continuous environmental improvement is essential in modern societies, we believe that ecological certification can help to confer social recognition (for example, through tax benefits) and commercial advantage (for example, through the allocation of "green labels") to the outstanding environmental behavior.

The use of indicators to assess and monitor the agricultural environment can represent a first step in a program aiming at implementing the ecological certification of commercial farms in the region. Despite uncertainties around indicators calculation, this does not invalidate a useful comparison of farms. If we consider that any methodological failure equally affects all farms in the study sample, this still allows a useful inter-farm comparison.

Individual farms can be characterized by means of the dashboard scheme. Indicators in dark grey allow the identification of specific environmental problems, providing a viable way to later drive technical solutions. Looking at a potential ecological certification program, farms which show most indicators within the light grey range are in theory demonstrating a better potential to get advantage from "green labeling" than farms in which dark colors predominate. At the same time, farms showing a poor performance can also benefit from dashboard indicators. It provides a rudimentary diagnosis guide of environmental problems that demand attention. No doubt that the predomination of dark colors in critical indicators could be demanding the support of specialized technological knowledge in specific fields. For example, if the indicator of soil erosion in a farm falls within the dark zone, a soil scientist or soil technician can provide a better diagnosis of the problem and suggest a technological way to overcome such constraint. The Agro-Eco-Index model does not provide a refined diagnosis, but a crude picture of environmental problems in the farm that can potentially become critical.

No doubt that the calculation of our indicators needs to be improved, especially in the case of those that arise doubt around the method applied or the validation process. A continuous review and revision is probably the best way to improve the quality of indicators. However, such improvement should pay special attention to the tradeoffs between the calculation strictness and the calculation complexity. A more strict calculation is needed to make indicators more reliable for representing the reality. But on the other hand, complexity may impose an economic and intellectual cost that might turn indicators unsuitable to practical users. In this case, we have decided that calculation simplicity should be preserved in the model at the expense of a rigorous one.

Acknowledgments

We want to thank the financial support of the National Institute for Agricultural Technology (INTA), the National Research Institute (CONICET) and the National Agency for the Promotion of Science and Technology (FONCYT) in Argentina. Members of two farmer organizations (CREA and Cambio Rural) are specially acknowledged for their willingness to support this study providing on-farm data.

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