Docente Guida:
Prof. Tommaso MAGGIORE

Ricercatore di supporto:
Dr. Luca BECHINI

Coordinatore di dottorato:
Prof. Pierluigi GENEVINI

Università degli Studi Di Milano
Facoltà di Agraria
Dipartimento di Produzione Vegetale

Nicola CASTOLDI

Environmental and Economic Assessment of Agricultural Systems at Crop, Field, Farm, and Regional Scale.

Application of Agro-Ecological and Economic Indicators in Northern Italy

PhD Thesis
Department of Crop Science - University of Milan
Via Celoria 2, 20133 Milan - Italy
nicola.castoldi@unimi.it

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Questo desiderio di semplificazione è giustificato, la semplificazione non sempre lo è. È un'ipotesi di lavoro, utile in quanto sia riconosciuta come tale e non scambiata per la realtà; la maggior parte dei fenomeni storici e naturali non sono semplici, o non semplici della semplicità che piacerebbe a noi.[1]

PrimoLevi

Hoc erat in votis: modus agri non ita magnus, hortus ubi et tecto vicinus iugis aquae fons et paulum silvae super his foret.[2]

Orazio

[1] The desire for simplification is justified, but the same does not always apply to simplification itself, which is a working hypothesis, useful as long as it is recognized as such and not mistaken for reality. The greater part of historical and natural phenomena is not simple, or not simple in the way that we would like (The Drowned and the Saved, 1988).

[2] It was all I wanted: piece of land not too big, a garden with a beautiful spring always fresh close to home and even a little woodland (Sermones, Liber II - Sermo VI).

Gracias a la vida

Gracias a la vida que me ha dado tanto
Me dio dos luceros que cuando los abro
Perfecto distingo lo negro del blanco
Y en el alto cielo su fondo estrellado
Y en las multitudes el hombre que yo amo.

Gracias a la vida que me ha dado tanto
Me ha dado el oído que en todo su ancho
Graba noche y día grillos y canarios
Martirios, turbinas, ladridos, chubascos
Y la voz tan tierna de mi bien amado

Gracias a la vida que me ha dado tanto
Me ha dado el sonido y el abecedario
Con el las palabras que pienso y declaro
Madre, amigo, hermano y luz alumbrando,
La ruta del alma del que estoy amando

Gracias a la vida que me ha dado tanto
Me ha dado la marcha de mis pies cansados
Con ellos anduve ciudades y charcos
Playas y desiertos, montañas y llanos
Y la casa tuya, tu calle y tu patio.

Gracias a la vida que me ha dado tanto
Me dio el corazon que agita su marco
Cuando miro el fruto del cerebro humano
Cuando miro el bueno tan lejos del malo
Cuando miro el fondo de tus ojos claros.

Gracias a la vida que me ha dado tanto
Me ha dado la risa y me ha dado el llanto
Asi yo distingo dicha de quebranto
Los dos materiales que forman mi canto
Y el canto de ustedes que es el mismo canto
Y el canto de todos que es mi propio canto.

Violeta Parra

Abstract

Castoldi, N, 2007. Environmental and economic assessment of agricultural systems at crop, field, farm, and regional scale. Application of agro-ecological and economic indicators in Northern Italy. Ph.D. Thesis, University of Milan, Italy, 333 pp, 25 figures, 45 tables, 244 references, 4 annexes.

In the last decades, the perception of relations between agriculture and environment has remarkably changed and concerns have been raised about the sustainability of agricultural production systems, involving consumers, citizens, policy makers and farmers.

As direct measurements are too expensive and time consuming, agricultural indicator should be applied for the evaluation of a large number of farms, because they are based on data already available or easy to collect.

In this work, agro-ecological and economic indicators were selected and applied at different scale (crop, field, farm, and regional) using data available in public agricultural databases or collected by farmer interviews. Indicators synthesize the management effects on the environment and the state of the farming system.

In order to evaluate the effectiveness of the tool used, the uncertainty of a single input variable was tested to quantify the corresponding uncertainty of the indicator.

Reference to the content of Chaptes 3, 4, and 6 should be made by citing the original paper.

CONTENTS

CHAPTER 1

INTRODUCTION AND OVERVIEW

1.1. General context

Sustainability is defined in different ways by different people: Jacobs (1995, as reported in Rigby et al., 2001) has found 386 definitions of sustainable development. We want to report the Bruntland commission (WCED 1987) definition: "(…) *sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs"*. Stückelberger (1999, as reported in Häni et al., 2003) adds two more dimensions of sustainability: the "human dignity" and the "non human environment". He extended the concept of sustainability: "*Sustainable development allows a life in dignity for the present without compromising a life in dignity for future generations or to threaten the natural environment and endangering the global ecosystem"*. Hence we can distinguish three aspects of sustainability: environmental, social, and economic sustainability (Goodland, 1995).

In order to evaluate the land sustainability it is necessary to focus the attention of environmental scientists and decision makers on the agricultural impact and virtuosity: in many European Nations, agriculture represents the main land use (Robinson and Sutherland, 2002). According to EU agricultural statistics (EU, 2006), in the EU Nations, agricultural land covers a large portion of the total area (Table 1.1). Therefore, it is necessary to apply tools that can describe the complexity of the systems through simple and not expensive input datasets, collectable for a large area in an easy way.

In the recent past, the principal function of agriculture was the food and fiber production. Nowadays the farms have assumed an other important function: the production of non-market goods (e.g. environmental services) which becomes increasingly important (Van der Werf et al., 2007b). Consequently the demand for an integrated evaluation of agricultural systems has increased and during the last decade numerous environmental farm management evaluation tools were developed (Rosnoblet et al., 2006). This trend was recently confirmed by the 112 models presented at the first Farming Systems Design symposium (Catania, Italy, 10–12 September 2007).

In order to analyze the environmental sustainability of agro-ecosystems it is possible to choose different methods, like direct measurements, simulation models, simple or composite indicators that have different levels of applicability and potential explanation of the systems (Bockstaller and Girardin, 2002; Castoldi and Bechini, 2006). The tools that evaluate the sustainability of agricultural systems increasingly become complicate, with a corresponding increase of input data quantity and quality (Stoorvogel and Antle, 2007). In many real contests it is economically difficult to collect all the necessary data for a large number of farms, because these data are not available or it is necessary to measure them in an analytical and expensive way.

The term indicator has been defined as a variable which supplies information on other variables which are difficult to access (Bockstaller et al., 1997). Indicators are interesting to analyze systems when it is not possible to carry out direct measurements. They can provide in a relatively short time a synthesis on processes and impacts at different scales and are efficient tools to evaluate the real achievement of agronomic, social, economic, and environmental targets (Silvestri et al.,

Table 1.1. – National and agricultural area in the EU countries

Nation	Tot. area (km^2)	Agricultural area (km^2)	(%)	Nation	Tot. area (km^2)	Agricultural area (km^2)	(%)
Finland	338,150	22,670	7	Czech Rep.	78,868	36,060	46
Sweden	448,474	30,190	7	Bulgaria	110,994	52,650	47
Croatia.	87,660	11,810	13	Germany	357,050	170,350	48
Cyprus	9,251	1,360	15	Italy	301,323	147,100	49
Estonia	45,227	8,340	18	Luxemburg	2,586	1,290	50
Slovenia	20,273	5,090	25	Poland	312,685	159,060	51
Latvia	64,589	17,340	27	Spain	504,878	256,900	51
Greece	131,957	38,050	29	Netherlands	37,358	19,240	52
Malta	316	100	32	France	549,087	295,840	54
Austria	83,858	32,630	39	Rumania	238,391	142,700	60
Slovakia	49,034	19,410	40	Ireland	70,295	43,070	61
Portugal	91,909	37,370	41	Denmark	43,098	27,120	63
Lithuania	65,300	28,370	43	Hungary	93,034	58,630	63
Belgium	30,528	13,860	45	UK	244,101	167,610	69
		UE			4,410,274	1,844,210	42

2002). They are valuable tools for evaluation and decision making as they synthesize information and can thus help to understand a complex system (Mitchell et al., 1995). They are used primarily to evaluate and to monitor the systems and afterwards a more accurate and precise tool would enable to fine-tune the systems gradually (Bellon et al., 2007), considering only the criticisms highlighted by indicators.

1.2. Indicators context

1.2.1. What methods?

In this work we have adopted the indicator methodology in order to i) select the most significant information, ii) simplify complex phenomena, iii) quantify information, and iv) make more simple the interpretation of the information, particularly between data collectors and data users (Rigby et al., 2001).

The first step in the choice of the indicators is the definition of the appropriate detail level: too much detail bears the risk of compromising on modeling the most important processes, whereas over specification and too much complexity of the models has as consequence that data requirements cannot be met. Too few details imply that relevant indicators may not be assessable with the required reliability (Van Ittersum et al., 2007). It is necessary to motivate the selection of the indicators, and to organize them in a framework, avoiding redundancy within a list of candidate indicators (Bockstaller et al., 2007). Two main questions arise in the definition of this framework: what is it the impact on the agriculture sector and what is it the impact on the rest of the word? (Bockstaller et al., 2007).

The assessment of farming systems can be based on the analysis of farmer production practices (“means-based”) or on the effects that these practices have on the state of the farming system or on emissions to the environment (“effect-based”; Van der Werf and Petit, 2002). The means-based approach is less expensive in data collection, but it does not consider the relation among practices and corresponding impacts on the system analyzed, thus makes it difficult to validate the indicator framework. Therefore the effect-based indicator frameworks are preferable.

A second step is the choice of the method that is one of the most critical points in the sustainability assessment: the evaluation depends not only on the characteristics of the systems analyzed, but also on the method used (Van der Werf et al., 2007b). Five major stages of a sustainability assessment method were identified (adapted from Petit and Van der Werf, 2003; Van der Werf et al., 2007b):

i) definition of the global objective of the method, including the choice of the final users (farmers, researchers, policy makers, activists, consumers, citizens, etc.) and the spatial and temporal scales. Evaluation methods should consider a range of objectives covering both local and global effects in order to effectively assess impacts;

ii) definition of a set of more specific objectives (an issue of concern and its associated desired trend). The number of objectives should be sufficiently large to avoid the inadvertent creation of new problems, and as small as possible to maintain feasibility. They should not be redundant and the procedure used for the selection of objectives should be stated;

iii) definition of the system's limits. The definition of the physical and temporal limits is necessary in order to define what the inputs and the outputs of the system are. If the method evaluates only the direct impacts of a system, it considers only the effects of farm operation. On the other hand, it is not possible to restrict the evaluation of agro-ecological sustainability on the analysis of energy, material, and economic fluxes: the farm management plays an important role in environmental sustainability (Bellon et al., 2007; Caporali et al., 2007). The method has to consider the direct and indirect impacts together;

iv) construction or identification of indicator framework. For each objective, one or more indicators are identified or constructed in order to quantify the degree to which the corresponding objective is attained. The quality of an indicator will largely depend on the validity of its calculation and on the precision and accuracy of the inputs (Castoldi et al., 2008b; Post at al., 2008). Van der Verf and Petit (2002) define a guideline in the choice of a good indicator-based model: a) indicators allowing expression of impacts both per unit surface and per unit product are preferable; b) it is possible to express the indicator output by values or by scores. Values are preferable instead of scores that have dimensionless units and can not be balanced against other values or real-world observations; c) if possible, threshold values should be defined for each indicator; d) it is necessary to validate the appropriateness of selected indicators with the objectives previously defined, and to make a comparison among indicator values and real-world data. If this comparison is not possible it is opportune to submit the design of the indicators to a panel of experts;

v) calculation of results. Indicator values are calculated for each of the systems or scenarios to be compared. A partial or total aggregation of results may facilitate their interpretation.

The indicators framework has to be considered always a basis for analysis, and it is necessary to integrate it into a structured assessment approach defined for a specific use (Giupponi and Carpani, 2006).

1.2.2. Spatial scale

The identification of spatial scale is the preliminary step in the definition of indicators framework used in the evaluation of sustainability of agro-ecosystems. For simplicity the agro-ecosystem is considered a composition of interconnected or stratified layers at different hierarchical levels, where each layer is isolated only for a necessity of study and management (Caporali, 2007).

Different spatial scales can be used in the sustainability analysis, with different aims and approaches. The agro-ecological point of view starts from the crop level, considering the entire plant population in a plot, in a single crop season, and its relation with the soil and the atmosphere. The hierarchical upper level is the field considered as the system composed by all crops in a rotation/sequence of crops, and their interaction, in a delimited area (field). Girardin et al. (2000) consider the field level the most relevant to the decision that farmers have to take on cropping systems. The upper level is the farm, considered by Caporali (2007) the most appropriate level for doing research and making decision in favor of sustainability, especially when groups of farms with contrasting managements are involved. At this level it is possible to recognize the results of the farmer's decision making process concerning the balance between the utilization of resources and the biophysical and socioeconomic constraints (Caporali et al., 2007). Higher levels are the river basin catchments and the regional scale, where many farms in a similar condition or with the same sensible target (i.e. a river or a protected area) are involved. An intermediate level is the municipality, but this aggregation level, in our opinion is inappropriate, because it considers only the administrative boundary, without considering the similarities and differences among farms, or a common sensible target. The national and global levels are the highest hierarchical level where the peculiar characteristics of the systems are much smoothed. At each scale different perspectives and goals of sustainability are dominant and different indicators are required for the assessment (Binder and Wiek, 2007).

It is well known that the quality of the indicators relies on the scale which they represent (Stein et al., 2001). The upscaling (to aggregate information from high resolution data towards a coarser resolution) produces less accurate information (i.e. a single indicator value for the entire farm, instead of *n* indicator values for the *n* fields in the farm), but it allows to use many indicators (i.e. landscape indicators) not applicable at the lower scale level.

1.2.3. Time scale

It is possible to focus the sustainability analysis on a single crop season, but the annual variation (i.e. climatic conditions, variability of price) are peculiar every year, and the evaluation of the systems in a single year could be not realistic. For example, in a drought year, the decrease of the yield reduces the energy, nutrient, and economic efficiency, providing a bad judgment on the system analyzed. Eckert et al. (2000) suggest using the mean values over a three years observation period. In our opinion, if the conditions are not particularly unusual (i.e. sharp change in the agricultural policies, significant drought), the reduction of the analysis on two years is suggested, saving money and resource, with a good compromise between costs and benefits.

The sustainability analysis should be carried out for the assessment of (Bellon et al., 2007):

i) past situations, for example in the evaluation of the policy effectiveness or the impact of past cropping systems (*ex-post approach*);
ii) existing situations with the explicit objectives of driving changes in actual farming systems in order to tune the management on the immediate requirement of the systems highlighted by indicator results (*in itinere approach*);
iii) in silico activities in order to make a preliminary evaluation of possible future scenarios or an assessment of policy effects (*ex ante approach*).

1.2.4. Aggregation of indicator in a single index

The set of values of the indicator framework, usually represented by a spider graph, provides a preliminary step in the characterization of the sustainability of the analyzed system (Girardin et al. 2000). A set of aggregated or weighted indicators may produce a more concise and representative value, called index (Giupponi and Carpani, 2006). The index facilitates the use of complex information by non experts. For example decision-makers need a global evaluation of the sustainability of the farming systems in order to drive the definition of policies, but in some case they have not the knowledge necessary to understand the complexity and trade-off among the agro-ecosystems, synthesized by an index.

The aggregation step is a critical point in the evaluation of the systems. Bockstaller and Girardin (2002) identify three methods for aggregation: i) spatial aggregation of indicators, through a weighted mean according to the size of spatial units, ii) aggregation of simple indicators into a composite indicator by the sum of the scores of the different sub-indicators or by calculating the weighted mean, iii) aggregation into a global and unique index.

It is possible to use an index in order to compare the relative threats to sustainability posed by different farming methods, but it should not be regarded as a means to calculate quantitative impacts of a particular farming system (Rigby et al., 2001). The global index does not consider the multidimensional of the sustainability assessment (Tisdell, 1996). Since each indicator represents a different point of view of sustainability, only multi criteria analysis should be applied to jointly analyze different types of indicators (Girardin et al., 2000). Many examples in the aggregation of indicator into an index were found in literature (Leopold et al., 1971; FAO, 1993; Lewis and Bardon, 1998; Girardin et al., 2000; Rigby et al., 2001; Kookana et al., 2005; Castoldi and Bechini, 2007).

1.2.5. Stakeholders

Van der Werf et al. (2007b) recognize three main users of indicators:

i) farmers and extension services (i.e. Goodlass et al., 2003), where indicators offer a tool to monitor the environmental and management performances, giving the base to make a rational choice among possible managements;

ii) researchers (i.e. De Koeijer et al., 2002), who can use the indicators for a preliminary assessment of the economic, environmental, and social effects of farm practices, in order to compare different systems or to drive further researches on the main problems highlighted by indicators;

iii) political decision makers (i.e. Schröder et al., 2004) who use the indicators for two aims: to better understand the problem and hence to define the policies, and to communicate information to general public that has not the knowledge required to understand the complexity of the system.

In addition it is possible to define other three categories of users:

iv) activists who use the indicators as support of their political struggle, to spread information on a large part of the citizens that usually have not a specific knowledge of the problem;

v) consumers who want to know the social, economic and environmental impact of their purchase, and the corresponding sustainability of their choice. A different interest is the definition of the quality and safety of the food and fiber;

vi) citizens who are interested to the effect of the political choice.

It is necessary to consider the final users of the results, in order to provide the clearer and appropriate answer to their questions and needs. It is important to define correct strategies for the communication of the information provided by indicators, avoiding the misinterpretation of the results. The use of scores or simple graphs is necessary to communicate with non expert stakeholders, but it is suggested as supplement information in any communications.

1.3. Limitations of the indicators

1.3.1. Disciplinary approach of the indicators frameworks

Many indicator frameworks provided by literature are not limited to the environmental assessment but only a few works present a comprehensive multi-disciplinary approach (Giupponi and Carpani, 2006). Many of the quantitative methods currently used are biased towards either the economic or environmental aspects and largely miss out the social issues (Van Ittersum et al., 2007).

1.3.2. Subjectivity of the methods

Van der Werf et al. (2007b) have demonstrated that the application of five methods in the evaluation of impacts of three different scenarios (farms type) provides different judgments. The ranking among the farms changes from a method to other. Also the unit used to express the indicator could change the scenarios ranking: using the same method it is possible to obtain a different farm ranking if the indictor results are expressed by area or by kg of live weight produced.

The calculation algorithms of the indicator have to be scientifically supported by experimental data or expert knowledge, and have to be as much objective as possible. The algorithms of a quantifiable variable, such the fluxes of material energy and money, are less subjective than the algorithms that try to describe the performance of the managements: it is difficult or impossible to quantify objectively the appropriateness of a practice (i.e. the quantification of the environment advantage in the pesticide applications when antidrift instruments are used or not). It is possible to define different calibration of the algorithm calculation, applying different parameters or coefficients provided by different stakeholders (Weinstoerffer and Girardin, 2000). This approach is particular effective in the evaluation of non quantifiable aspects expressed by scores (i.e. landscape or rotation suitability). Therefore it is possible to compare the outputs of different calibrations, evaluating the range of responses, and eventually to aggregate them in a single value.

1.3.3. Uncertainty

Unfortunately it is not known to what extent the outcome of studies assessing the impacts of agricultural systems depends on the characteristics of the evaluation method used (Van der Werf et al., 2007b). The prediction of impact invariably engages with condition of complexity and uncertainty (Stirling, 1999), where data are frequently limited and simplified assumptions have been used, in particular in the application of indicators.

It is necessary to gain a good compromise among simple analysis and the possibility to obtain a wrong answer. One way of approaching this challenge may be through case studies of well-known agro-ecosystems in order to evaluate the effects of simplification of methods and their propensity for wrong answers (Van Der Werf and Petit, 2002).

The aim of quantitative uncertainty analysis is to use currently available information to quantify the degree of confidence in the existing data and models, helping to identify how robust the conclusions about model results are (Isukapalli and Georgopoulos, 2001).

It is possible define three main factors that influence the output uncertainty (Post et al., 2008): i) the parameters and coefficients used, ii) the input values, and iii) the model selected.

In literature it is possible to find different values for the same parameter or coefficient (i.e. there are several values for the specific energy content of fertilizers and fuel, and different values of the active ingredient toxicity referred to a specific non-target organism): in this case it is advisable to select the value reported in referenced literature and to use the newest, otherwise it is possible to use the mean or the median value.

In many contexts it is difficult to obtain a quantification of the uncertainty related to the input data used. Usually the data are stored in databases, without to make explicit the method used in the collection and the distribution of the uncertainty around the value provided. It is possible to try to quantify the uncertainty using the available information or draw on the literature the referenced value of the uncertainty associated whit the process analyzed.

The uncertainty of the used model is quantifiable by validation using experimental data. If the indicator model is self built it is necessary to validate it with a new dataset which was not used in the calibration of the model; otherwise it is a good practice to select from literature calibrated and validated indicators, mistrusting the frameworks presented without application in case study.

An other option is the use of the uncertainty of inputs, coefficients and parameter, in order to estimate its propagation in the indicator calculation (uncertainty analysis) (van der Sluijs et al., 2004). The results of the uncertainty analysis could provide broadly speaking, three different conclusions on the outputs of the model (Fig. 1.1):

i) the distribution of the output has a probability density function (PDF) with high kurtosis (low spread of the output), therefore the uncertainty is low and it is possible to use the model output with safety;

ii) the distribution of the output has a PDF with low kurtosis (high spread of the output), and this distribution crosses in a significant way the threshold. In this case the model provides discordant judgments on the case analyzed in relation with the probability level; therefore the effectiveness of the model is low;

iii) the distribution of the output has a PDF with low kurtosis, but it does not cross the threshold, therefore the judgments provided by the model are not discordant at any level of probability, and it is possible to use this output for the sustainability assessment.

Even though significant effort may be needed to incorporate uncertainties into the modeling process, this could potentially result in providing useful information that can aid in decision making (Isukapalli and Georgopoulos, 2001). The quantification of the variability and magnitude due to incomplete knowledge of the real world processes described by models (uncertainty analysis), and the apportion of this uncertainty to different sources in the models (sensitivity analysis) is more and more becoming an integral component for the integrated environmental assessment studies (Post et al., 2008).

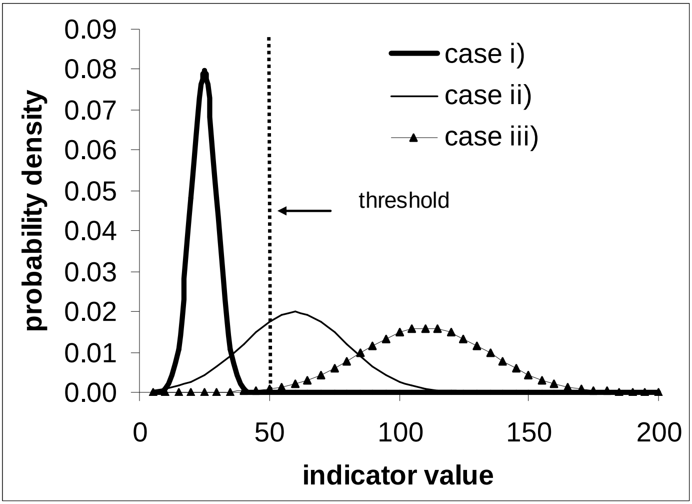

Fig. 1.1. – Possible results of uncertainty analysis.

1.3.4. Trade-offs

In a large part of the frameworks provided by literature, the single aspects of agro-ecosystems are considered and analyzed separately, and therefore integrated in the global evaluation of the systems. This is a simplification of the real word that shows complex relations and trade-offs among different future of the system. A more realistic approach should analyze the different components of the systems together in relation of the possible trade-off that arise when a single factor is changed. For example, when indicators suggest a reduction of the nutrient impact, it is necessary to evaluate also the corresponding dynamics of the organic matter, which in some case should lead in a reduction of soil organic matter. A model that considers the trade-off is the Sustainability Solution Space (Wiek and Binder, 2005).

1.3.5. Conclusions

Farmers are the environment and landscape keepers. They have a double role in the environmental management: they are contemporary the users (as workers) and beneficiaries (as citizens) of the environment, but often this raises conflict among economic and environmental objectives. In many cases they are interested to the environmental quality objective, but a good environmental management is usually expensive or not very profitable. In the intensive agricultural areas in Europe (i.e. Po Valley in Italy, The Netherlands, Denmark), the economic weight of cross-compliance is not enough to compensate the renouncement of intensive agricultural practices (i.e. intensive pesticides and nutrients applications) that often have dangerous effect on the environment. On the other hand it is not correct to consider a simple link among farm management and environmental pollution, because this link is indirect: environmental impacts of farms depend largely on farmer practices, but the causal chain of farmer practices - pollutant emissions - environmental impacts is affected by other factors such as weather and soil characteristics, pollutant fate, and the sensitivity of environmental targets (Van der Werf et al., 2007a).

1.4. Objective of the research

We have defined a framework for the analysis of agro-ecological and economic sustainability of the farming systems at different scale of analysis. We have applied this framework on a set of selected farms.

Our objectives were:

1. to select simple indicators to describe the performance of the agricultural systems at crop, field, farm, and regional scale;
2. to collect data on agricultural management using available data (existing databases) or simple data collectable in a cheap way using a structured questionnaire completed during face-to-face interviews with farmers;
3. to evaluate environmental and economic sustainability, evaluating both fluxes (of energy, material, and economic) and management practices with appropriate indicators;
4. to compare different cropping and farming systems;

5. to evaluate the regional sustainability of the agricultural management;
6. to evaluate the uncertainty of the indicator inputs and the corresponding uncertainty in the outputs, in order to define the robustness of these tools.

1.5. Organization of the research

The flow-chart in Fig. 1.2 shows the flux of our research.

We based our work on the available data in the Agricultural Information System for the Sud Milano Agricultural Park (SITPAS; box 1), and the selected agro-ecological indicators (box 3).

The flow- chart is divided in two parts:

i) in the left side there are the analyses of the agro-ecological sustainability of seven selected farms at crop, field, and farm level (boxes 2, 4, 5, and 6), using new data collected by farmer interviews in the period 2005 – 2006;

ii) in the right side there are the analyses at regional scale using the SITPAS data (Box 7 and 8).

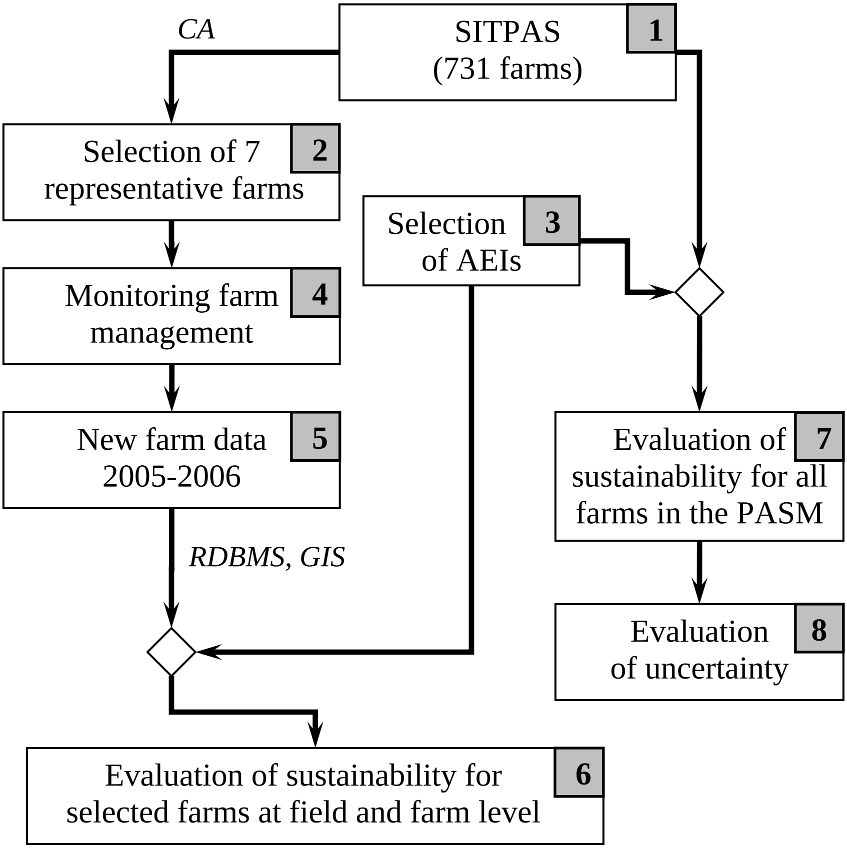

Fig. 1.2. – Flow-chart of this work. CA: cluster analysis; AEIs: Agro-ecological indicators; SITPAS: Agricultural Information System for the Sud Milano Agricultural Park; RDBMS: relational database management systems; GIS: geographical information systems.

1.6. Synopsis

Chapter 2 (*Studied area, SITPAS and farms selection*) introduces the studied area, the Sud Milano Agricultural Park (PASM, 45°N, 9°E, 47,000 ha, including 61 municipalities), which covers an area densely populated, with one of the most intensive agricultures in Europe. In this chapter there is a short presentation of the Agricultural Information System for the Sud Milano Agricultural Park (SITPAS), a large agricultural database used for this sustainability assessment. In this chapter it is presented the statistical methodology (Cluster Analysis) which was used to select seven representative farms that were monitored in this research.

Chapter 3 and 4 (*Agro-ecological indicators of field-farming systems sustainability. I. Energy, landscape and soil management; II. Nutrient and pesticides*) discuss the principles regarding the indicators and our considerations about their use, limit, and virtuosity. A brief review of indicators provided by literature is presented. The selected indicators can be applied at the crop, field, and farm scale, based on data obtainable from the farmers and/or from existing agricultural databases. The principal aspects of the farming system considered are: i) the energy utilization, ii) the effect of agriculture on landscape, iii) the impact of agricultural management on soil quality, iv) the evaluation of nutrient use, and v) the possible effects of pesticides on environment.

Chapter 5 (*Monitoring of cropping systems sustainability in seven farms in Northern Italy: application of environmental and economic indicators*) exposes the results of the monitoring of the seven farms selected. The characteristics of the farms, the monitoring method and the indicator framework used are described. The indicator results obtained using the data collected during farm interviews are presented and discussed. The conclusions on sustainability of different cropping systems are drawn, highlighting the criticisms and virtuosity.

Chapter 6 (*Calculating the soil surface nitrogen balance at regional scale: example application and critical evaluation of tools and data*) regards the analysis of the nitrogen balance calculated in 157 farms in the PASM, using data on nutrient inputs (fertilizer, manure, N returned to soil with residues originating from the previous crop in the rotation, atmospheric deposition, and biological fixation) and outputs (crop aboveground uptake). Only the SITPAS management data that had passed a quality check were used in the calculation of nitrogen balance. The advantages and disadvantages of this approach are discussed.

Chapter 7 (*Geostatistical prediction at the regional scale using existing databases on cropping and farming systems*) contains a description of the procedures used in the estimation of the extractable soil phosphorus (*ESP*) in the PASM. This evaluation was carried out by geostatistical spatial interpolation procedures (kriging). Available data stored in the SITPAS was considered: the *ESP* provided by soil reports and auxiliary information on farm management utilized to perform the estimation of the *ESP*. Three methodologies were compared: Ordinary Kriging, Kriging with External Drift, and a hybrid form. The hybrid form was applied to three interconnected and non-overlapped geographical layers (sections), characterized by different farm management practices. A section-specific interpolation procedure was applied for the three different layers, using separate subsets of soil analyses. The results are maps of the predicted values of *ESP* and corresponding prediction errors.

Chapter 8 (*Evaluation of the spatial uncertainty of agro-ecological assessments at the regional scale: the phosphorus indicator in Northern Italy*) explores the uncertainty of the inputs, and its effect on the Phosphorus Indicator (*IP*) output. The *IP* was applied in the PASM, using data on phosphorus management contained in SITPAS, and predicted *ESP* in order to evaluate the appropriateness of phosphorus fertilization. The uncertainty of a single input variable (*ESP*) was tested to quantify the corresponding uncertainty of the indicator.

Chapter 9 (*General discussion*) attempts to make a synthesis of the approach used in the assessment of the environmental and economic sustainability at the different scales. It focuses on the virtuosity and on the limit of the proposed framework, and suggests the direction for further works.

1.7. Note

Chapter 3 and 4 have been published on the Italian Journal of Agrometeorology. Chapter 6 has been published on Italian Journal of Agronomy. Chapters 7 and 8 have been submitted for publication to Environmental Ecosystems and Environment.

The reference lists from these individual papers have been amalgamated into one list at the end of this thesis. I would like to acknowledge the editorial boards of Italian Journal of Agronomy and Italian Journal of Agrometeorology for their permission to include the papers in this thesis.

CHAPTER 2

STUDY AREA, SITPAS, AND FARMS SELECTION

2.1. Study area

The studied area is the Sud Milano Agricultural Park (PASM: Parco Agricolo Sud Milano; 45°N, 9°E), a regional agricultural metropolitan park, surrounding the town of Milano (Fig. 2.1). It was created by Regional Law 24/90, in order to guide the cohabitation between agricultural and urban area, preserving the agriculture and the agricultural land from the continuous advancing of the urban, industrial and infrastructure activities, particularly strong in the north of Milano (Fig. 2.1c).

The PASM area is about 47,000 ha, including 61 municipalities: it covers one third of the Milano province. The area is densely populated: there are more than 3 million inhabitants totally.

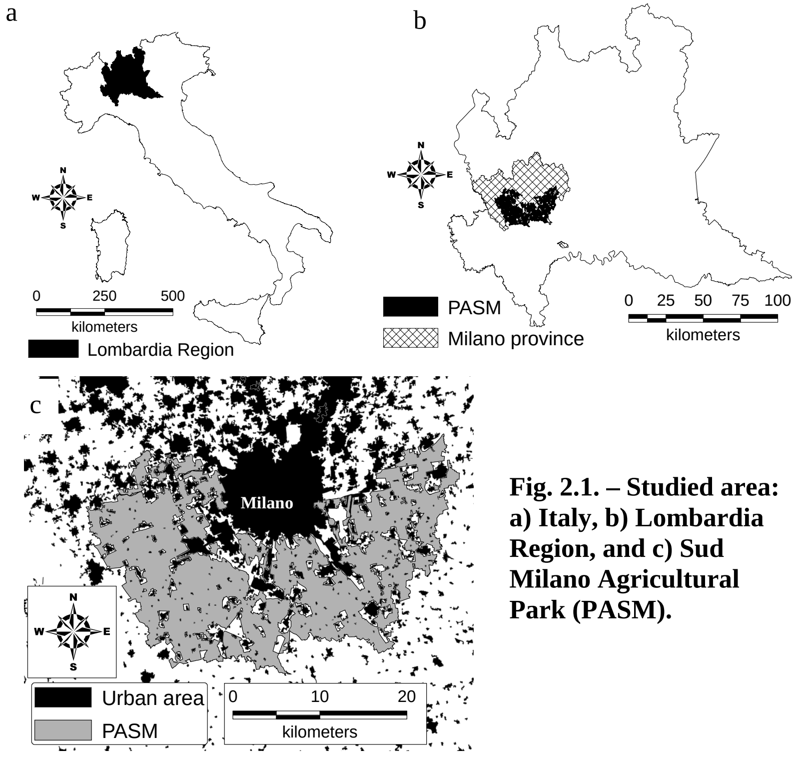

Fig. 2.1. – Studied area: a) Italy, b) Lombardia Region, and c) Sud Milano Agricultural Park (PASM).

2.1.1. Climate

The local climate is subhumid, with average annual rainfall of 996 mm and a large variability (min = 540, 1st quartile = 802, 3rd quartile 1110, max = 1520, SD = 226; data calculated on measured data from 1951 to 2000, Landriano 45°19'N, 9°13'E, 88 m asl). Usually the rainfall is high in May (average 96 mm) and October (average 114 mm) and lowest in winter (averages about 60 mm) and in July (average 64 mm) (Fig. 2.2). Temperatures (Fig. 2.3) increase from January (average T min: –1.2°C, T max: 4.9°C) to July (average T min: 17.7°C, T max: 29.2°C).

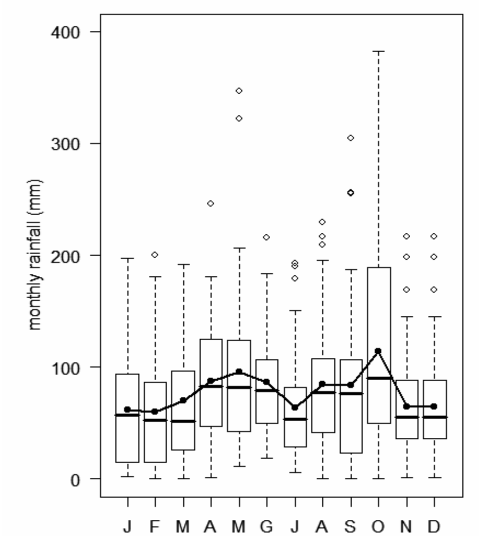

Fig. 2.2. – Monthly rainfall variability calculated on measured dataset from 1951 to 2001; dotted-line: average values.

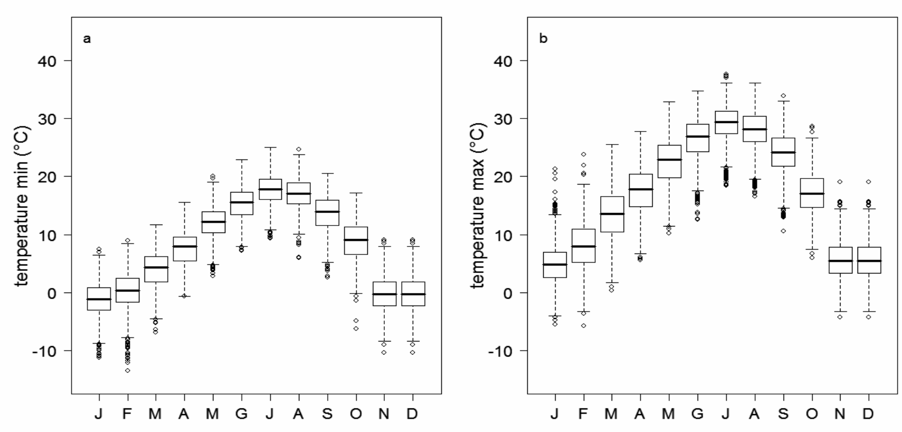

Fig. 2.3. – Distribution of a) minimum and b) maximum daily temperature calculated on measured dataset from 1951 to 2001.

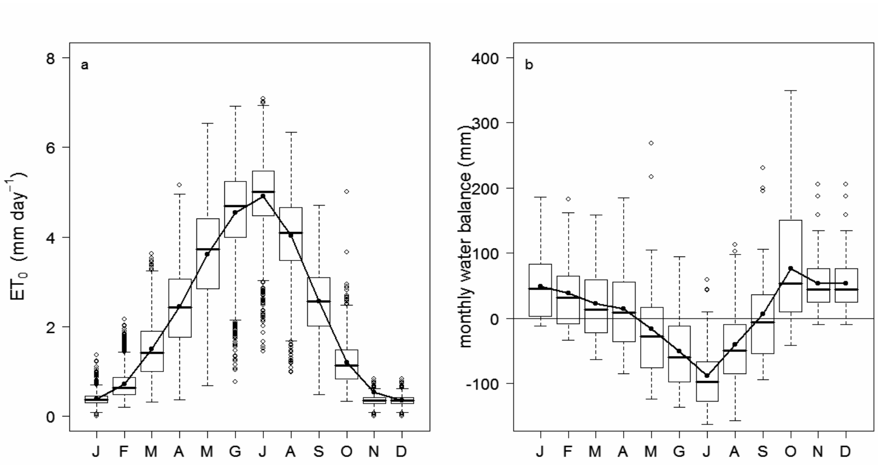

Fig. 2.4. – Distribution of a) daily reference evapotranspiration (ET_0) and b) monthly water balance (monthly rainfall – monthly ET_0); dotted-line: average values.

The annual reference evapotranspiration (ET_0) is on average 818 mm yr^{-1} (SD 62 mm yr^{-1}) with a peak in July (average ET_0 4.9 mm d^{-1}, Fig. 2.4a). ET_0 exceeds rainfall from May to September (Fig. 2.4b).

2.1.2. Morphology, pedology, and hydrology

The Park is located in a plain area with an altitude gradient from North to South of about 160 – 80 m above sea level, and an average slope of 0.3%. The area is characterized by a coarse trend from sandy-skeletal soils in the North and to silt soils in the South. The main soils are: Mollisols, Alfisols, Inceptisols, Entisols. The area is covered by a dense network of natural rivers (Lambro Meridionale, Lambro Settentriomale, Molgora, Olona) and canals (Muzza, Redefossi, Colatore Addetta, Scolmatore Nord Ovest, Deviatore Olona, Redefossi, Naviglio di Bereguardo, Naviglio Grande, Naviglio Martesana, Naviglio Pavese, Roggia Carlesca, Roggia Vettabbia, Roggia Ticinello); the main water source of the canals are two rivers near the Park in the East (Adda) and in the West (Ticino). The groundwater is usually superficial (depth from 16 to 4 m below field surface), with a North-South direction, and it come to the surface in about one hundred resurgence. For details see ERSAL (1993).

2.1.3. Natural vegetation and main crops

In the last centuries, the human activity has deeply changed the natural landscape, replacing the woodland with infrastructure and crops. The main wood species are: oak (*Quercus robur* L.), ash (*Fraxinus oxycarpa* Bieb.), hornbeam (*Carpinus betulus*), box elder (*Acer campestre*), elm (*Ulmus minor* Miller), white poplar (*Populus alba* L.), and hazel (*Corylus avellana* L.). The main shrub are cornel tree (*Cornus mas* L.), hawthorn (C*rataegus monogyna* Jacq.), spindle tree (*Evonymus europaeus* L.), privet (*Ligustrum vulgare* L.), guelder rose (*Viburnum opulus* L.), alder buckthorn (*Frangula alnus* Miller), dogrose (*Rosa canina* L.), honeysuckle (*Lonicera caprifolium* L.), traveller's-joy (*Clematis vitalba* L.), and black bryony (*Tamus communis* L.).

The PASM agricultural area is ca. 35,000 ha, and the most important crops are corn (*Zea mays* L.), rice (*Oryza sativa* L.), permanent meadows, soybean [*Glycine max* (L.) Merr.], barley (*Hordeum* spp.), Italian ryegrass (*Lolium multiflorum* Lam.), and winter wheat (*Triticum aestivum* L.), with moderate to high yields (9.6, 19.5, 5.2, 4.8, 4.9 and 3.0 t DM ha^{-1} for grain maize, silage maize, rice, winter wheat, barley and soybean respectively). Many farms have noticeable breeding (dairy and cattle, swine and poultry). Irrigation is normally performed with surface methods, using water from a dense network of canals and ditches.

2.2. SITPAS

The Agricultural Information System for the Sud Milano Agricultural Park (SITPAS, Bergamo et al., 2007), developed in the period 1999 – 2003, is made of a large relational database connected with a Geographical Information Systems (GIS). It stores detailed and georeferenced information about the agricultural activities, integrated with pedological, climatic and environmental information.

A total of 910 farms has been identified, and 731 were described in detail; of these, animal farms are 348. The information system was built by collecting pre-existing data (regional climatic, pedological and environmental databases, CAP declarations, soil analyses provided by the public administration, private laboratories, and cadastral maps), and by integrating them with those obtained by directly interviewing all the farmers about animal and crop management practices.

The aim of the SITPAS was to collect and order information about the different aspects of the PASM, and producing a dedicate GIS usable as a support in the land planning and in the policy decision process.

2.3. Farms selection

2.3.1. Materials and methods

The selection of representative farms was done using the available data in the SITPAS (Fig. 1.2, box 1 and 2). Cluster Analysis (CA; Bechini et al., 2005a; Fig.1.2, box 2) was applied in order to define homogeneous groups of farms; it allowed to select the seven representative farms in seven different clusters.

Every farm was described using 24 variables: i) the percentage of farm area cultivated with the most important crops (corn, rice, wheat, barley, soybean, Italian ryegrass, and meadows), ii) livestock densities of the principal animal types (dairy, cattle, swine, and poultry), iii) the percentage of farm area with different irrigation systems used for rice (continuous flooding irrigation, delayed-continuous flooding irrigation, periodically flooding irrigation, and flush irrigation), iv) the percentage of farm area where the different agronomic operations are given to contractors (land improvement operations, tillage, sowing, pesticide applications, fodder and grain harvest), v) the power-machinery density (self-propelled machinery), vi) the number of combine harvesters in the farm, and vii) the percentage of area with continuous crop. Before the analysis, these variables were standardized (Z standardization) in order to allow the comparisons between different variables with different ranges and units.

Before CA, Principal Component Analysis was applied in order to find a small set of components usable in CA. However, in order to explain the 87% of the variance, it was necessary 16 principal components, too many compared to 25 variables considered. Hence, it was preferred to use the 25 variables in the CA, instead of the 16 principal components.

The aggregation method used was the Unweighted Pair Group Method with Arithmetic Mean. The distance between samples (farms) were expressed by cosine distance. This analysis was carried-out with SPSS 12.0.1 statistical package.

In the SITPAS, for some farms not all the data are available for the 24 variables used. Therefore only 496 farms with all 24 descriptors were considered in the CA; the others are lacking one ore more variables, and they were excluded from CA.

2.3.2. Results and Discussion

18 clusters (CL) were created in order to separate the 496 farms in homogeneous and representative groups (Table 2.1). The first three CLs (4, 7, and 11) represent the corn farms, where high percentage of farm area is cultivated with corn in continuous crop, and the recourse to contractors is noticeable.

The clusters 5, 12, and 9 represent the farms where soybean (CL5), barley (CL12), and winter wheat (CL9) are the most important crops in the farm. In these clusters the recourse to contractors for harvest operations is high.

Six CLs were defined for the rice farms (CL1, 3, 16, 2, 18, and 17) with different percentage of area cropped with rice (from 67.1 to 28.9% for the CL1 and CL17, respectively). Also in these CLs the continuous crop is elevated and proportional to the percentage of area with rice. This is due to the high costs in the preparation of paddy fields. In CL1, 2, and 18 there are farms that use the continuous flooding irrigation systems; the difference between them is ascribable to the difference in the rice percentage area and the recourse to contractors. CL3 represents the rice farms that use the flooding with a system where irrigation water periodically available on farm (normally every 7 or 14 days); in CL17 the farms use only the delayed-continuous flooding irrigation systems, while in CL16 there are the rice farms that use the flush irrigation systems. In all rice CLs there are breeding with low intensity.

Five CLs (CL10, 6, 8, 13, and 15) represent the livestock farms with densities from middle to high, where the main crops are corn, meadows and barley. In CL15 there are the swine farms, in the CL10, 6, and 8 there are the dairy farms with decreasing livestock densities, and in the CL13 there are the cattle farms. In CL14 there are the farms that have a small area and high power machine densities: this CL represents contractors.

CA has allowed recognizing 18 homogeneous farm groups, with clear distinction of the farm production typology, the livestock type and density, and the mechanization level.

The selection of representative farms was done using these results, selecting farms in different CLs.

Table 2.1 – Homogeneous clusters of farms in PASM (Northern Italy, 45°N, 9°W). Average values for each cluster.

	farms typology																	
Farm descriptors **Main scale**	**corn farms**			**grain farms**			**rice farms**						**livestock farms**					**C[a]**
Cluster number	4	7	11	5	12	9	1	3	16	2	18	17	10	6	8	13	15	14
Number of farms	53	54	41	19	13	12	91	14	5	22	2	9	67	45	21	19	3	6
Average farm area (ha)	39.4	60.9	41.4	53.1	26.5	26.0	110.2	77.1	50.3	95.8	48.4	147.7	56.1	21.5	63.5	34.8	64.5	7.2
CROPS (% farm area)																		
corn	**71.8**	**69.6**	**50.2**	**27.8**	**29.5**	**24.3**	**21.4**	**27.5**	**22.3**	**29.6**	**23.1**	**22.7**	**63.8**	**17.2**	**58.5**	**58.9**	**61.7**	**74.6**
rice	0.0	0.0	6.2	0.0	0.0	0.0	**67.1**	**62.9**	**51.8**	**49.3**	**38.9**	**28.9**	0.0	0.0	0.0	0.0	0.0	0.0
winter wheat	1.7	0.5	1.5	2.6	0.7	**42.2**	0.5	1.5	0.0	2.1	0.0	2.9	0.7	0.4	2.4	0.1	0.0	0.0
barley	0.9	2.0	4.9	2.5	**50.7**	8.8	0.5	0.0	1.4	3.7	0.0	4.9	5.1	3.1	8.4	**10.6**	**12.7**	0.0
meadows	**10.3**	**16.9**	**27.9**	**10.6**	**14.9**	**10.6**	5.7	3.6	7.7	6.6	**32.1**	**11.1**	**25.1**	**76.4**	**26.2**	**28.2**	4.2	**12.7**
soybean	1.6	1.3	3.4	**38.6**	7.4	2.0	1.5	0.0	0.0	1.5	4.7	**16.3**	2.3	0.1	3.4	0.0	2.2	0.0
Italian ryegrass	0.0	2.1	0.8	0.0	1.6	1.2	1.5	2.8	3.4	2.1	0.0	4.4	**18.3**	0.1	2.4	3.1	0.0	0.0
LIVESTOCK (Mg live weight ha^{-1})																		
dairy	0.03	0.40	0.57	0.00	0.17	0.00	0.31	0.31	0.38	0.21	0.00	0.16	**2.15**	**1.36**	**1.12**	0.01	0.00	0.00
cattle	0.02	0.05	0.14	0.01	0.00	0.12	0.04	0.00	0.01	0.01	0.59	0.11	0.05	0.04	0.03	**1.03**	0.00	0.03
swine	0.00	0.07	0.11	0.03	0.23	0.18	0.03	0.13	0.00	0.13	0.00	0.00	0.01	0.01	0.39	0.01	**8.26**	0.00
poultry	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.14	0.00	0.00	0.00	0.00	0.00	0.00	0.00
RICE IRRIGATION SYSTEMS (% farm area)																		
flush	0.0	0.0	0.0	0.0	0.0	0.0	2.3	0.3	100.0	1.5	0.0	0.0	0.0	**0.0**	**0.0**	**0.0**	0.0	0.0
delayed-continuous flooding	0.0	0.0	2.4	0.0	0.0	0.0	24.0	3.6	0.0	19.9	10.0	100.0	0.0	**0.0**	**0.0**	**0.0**	0.0	0.0
continuous flooding	0.0	0.0	2.4	0.0	0.0	0.0	73.1	13.1	0.0	38.7	40.0	0.0	0.0	**0.0**	**0.0**	**0.0**	0.0	0.0
flooding with time-table	0.0	0.0	4.9	0.0	0.0	0.0	0.7	83.1	0.0	3.5	0.0	0.0	0.0	**0.0**	**0.0**	**0.0**	0.0	0.0
AGRONOMIC OPERATIONS CARRIED OUT BY CONTRACTORS (% of farm area treated)																		
land improvement operation	0.0	0.0	4.9	0.0	0.0	0.0	0.0	7.1	0.0	**100.0**	0.0	0.0	1.5	0.0	0.0	0.0	33.3	0.0
tillage	0.0	0.0	29.3	10.2	0.0	0.0	0.0	0.9	10.0	0.0	0.0	0.0	0.1	0.5	0.5	0.9	33.3	0.0
sowing	0.0	0.0	**89.7**	13.2	0.0	0.0	1.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	2.6	33.3	0.0
pesticide applications	0.0	0.0	**69.1**	12.6	0.0	8.3	0.4	0.0	0.0	0.0	0.0	7.2	5.4	0.0	**100.0**	5.3	33.3	0.0
fodder harvest	0.0	1.6	14.3	2.6	0.0	0.0	14.6	4.9	23.1	14.6	0.0	5.7	**55.4**	7.2	39.2	16.0	33.3	0.0
grain harvest	**96.7**	0.5	**89.9**	**85.8**	**61.6**	**50.0**	24.0	14.5	**60.0**	49.1	0.0	**94.7**	**69.9**	31.4	**81.4**	47.4	**89.6**	**66.7**
POWER-MACHINERY DENSITY (kW ha^{-1})	7.5	7.6	6.0	5.7	10.6	7.7	6.6	6.1	6.4	5.4	6.3	4.3	8.2	11.5	7.4	9.6	4.1	66.8
COMBINE HARVESTER (n. $farm^{-1}$)	0.0	0.4	0.0	0.1	0.0	0.3	0.9	0.9	0.2	0.4	0.5	0.0	0.0	0.0	0.0	0.3	0.0	**0.8**
CONTINUOUS CROP (% farm area)	**60.6**	**55.0**	**60.3**	24.2	52.2	49.5	50.8	43.0	43.2	54.0	32.1	18.6	46.0	**91.4**	**60.7**	42.4	38.5	**100.0**

[a]C: contractors

CHAPTER 3

AGRO-ECOLOGICAL INDICATORS OF FIELD-FARMING SYSTEMS SUSTAINABILITY. I. ENERGY, LANDSCAPE AND SOIL MANAGEMENT

Castoldi, N., and L. Bechini. 2006.

Italian Journal of Agrometeorology 1:19–31

Keywords: agroecosystems, environmental impact, rotation, organic matter.

3.1. Abstract

Evaluation of the sustainability of farming systems management can be carried out with direct measurements, simulation models or indicators; the latter have the advantages of requiring a small amount of inputs, being fast to calculate and easy to interpret, allowing comparisons in space and time, and representing a synthesis of processes in complex systems. In this paper we propose a list of indicators which synthesise the state of the farming system or the management effects on the environment related to fossil energy use, landscape and soil management. We selected indicators from the literature which can be applied at the field and farm scale, based on data obtainable from the farmer and / or from existing agricultural databases; we excluded indicators based on direct measurements. In a second paper we will introduce indicators related to nutrients and pesticides use.

The direct and indirect consumption of fossil energy can be calculated at different levels of detail and it is used to calculate the efficiencies of different systems (output/input ratios). Landscape indicators describe the presence and the density of various elements that compose the landscape (crops, linear elements, and isolated shapes), allowing also, in one case, to compare landscape "demand" and "supply". The soil management indicators describe the relation between soil quality and crop management using: i) the crop sequence indicator that evaluates the goodness of each previous-successive crop combination in a rotation, assigning specific scores to the effects of one crop to another in terms of development of pathogens, pests and weeds, soil structure and nitrogen supply; ii) the organic matter indicator that evaluates if the management adopted by the farmer on a specific soil tends to accumulate or deplete soil organic matter; and iii) the soil cover index for evaluating soil protection by crops. Overall the indicators, based on a rather small data set, allow to conduct immediate syntheses of important agro-ecological aspects of farming systems.

3.2. Introduction

The Bruntland commission (World Commission on Environment and Development, 1987) defined the concept of sustainability as "(…) a form of sustainable development which meets the needs of the present without compromising the ability of future generations to meet their own needs." We can distinguish three aspects of sustainability: environmental sustainability, social sustainability and economic sustainability (Goodland, 1995). In this paper we deal with environmental sustainability, defined as the maintenance of the global ecosystem (or of the "natural capital"), both as a "source" of inputs and as a "sink" for wastes (Goodland, 1995). The agricultural system is involved in all these aspects and farmers are guardians of the countryside, of the ecosystem and of the rural landscape (European Commission, 2001). We recall the definition of agro-ecosystem: "an ecosystem constituted by several organism populations that interact one with each other, and with environmental and anthropic factors; man manages the equilibria of this system in order to increase the growth of a few economic-interesting vegetable and animal species." (Borin, 1999).

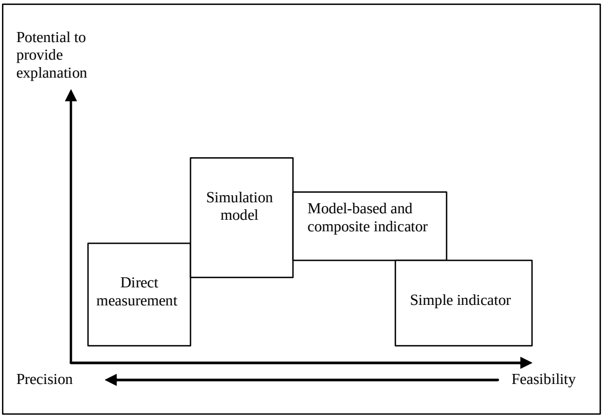

Fig. 3.1. – Potentials an weaknesses of different assessment methods (from Bockstaller and Girardin, 2002, modified).

In order to analyse the environmental sustainability of agroecosystems it is possible to choose different methods, like direct measurements, simulation models, simple or composite indicators that have different levels of applicability and potential explanation of the system (Fig. 3.1).

In many contexts, routine direct measurements are costly and often time consuming, especially if the studied area is large. Often, simulation models require many input data that can be difficult to obtain; moreover, sometimes models are not validated for a wide range of conditions (Bockstaller et al., 1997). Therefore, indicators are interesting to analyse agro-ecological systems when it is not possible to carry out direct measurements.

The term indicator has been defined as a variable which supplies information on other variables which are difficult to access (Bockstaller et al., 1997). Indicators can provide in a relatively short time a synthesis on processes and impacts at different scales.

They are valuable tools for evaluation and decision making as they synthesise information and can thus help to understand a complex system (Mitchell et al., 1995). The indicators can be calculated rapidly and are efficient tools to evaluate the real achievement of agronomic, economic and environmental targets (Silvestri et al., 2002). Since each agro-ecological indicator represents a different point of view on environmental sustainability, the indicators can be included in a multi-criteria evaluation of the sustainability of agricultural systems, which may also include socio-economic indicators of sustainability. In order to build a good indicator it is necessary to take into consideration some properties that influence its potential use: i) independence from the size of the study object; ii) robustness: not highly influenced by extreme or uncommon events; iii) accuracy; iv) precision; v) responsiveness: quick change in response to actions or alterations in the study objects, compared to direct measurements, requiring time for sampling and analysis, and to detect changes in the state of the study objects because of resilience and inertia; vi) measurability: based on planned or available data; vii) ease of interpretation: to communicate essential information in a way that is unambiguous and easy to understand; viii) pertinence: the capacity of identifying the behaviour of studied entity; ix) cost effectiveness: in proportion to the value of the information derived; x) policy relevance: to drive the key environmental issues (European Commission, 2001; Silvestri et al., 2002).

In the set proposed in this paper, not all indicators have all the properties mentioned before. Precision (defined as the variability between replicated measures) is not considered because of the lack of replicated values. The accuracy of indicator (closeness to the real value) is not valuable without a comparison whit other methods or direct measurements. For the measurability property, the planned data are not considered, because the aim of the selected indicators is to provide a judgement on a specific topic using existing and available data. Indicators do not provide an absolute but a relative evaluation of different entities.

Moreover, the errors resulting from lack or inaccuracy of input data are uniformly spread in all alternative study cases (Silvestri et al., 2002). Inputs for calculating indicators are often dissimilar and a semi-quantitative approach can be necessary to integrate all the variables in a unique value (expressed in physical units) or in a judgment (expressed with a qualitative scale). Further conversion to a unit scale (e.g. 0 – 10) is useful in order to compare the result of different indicators. Finally a good indicator should have a benchmark that permits, also for non experts, an easy evaluation.

In the last 10 years the interest for agro-ecological indicators (AEIs) has increased and several sets of AEIs have been proposed. The OECD’s DSR (Drive - State - Response) framework (OECD, 1999a) and the European Environmental Agency’s DPSIR (Drive - Pressure - State - Impact - Response) framework (EEA, 1999) provide the basis for an agro-environmental indicators framework named agricultural DPSIR (European Commission, 2000). The objective of agricultural DPSIR is to provide an harmonised structure of agro-environmental indicators in EU Member States in order to present a common basic level of information that can be aggregated and facilitate comparisons among regions. The agricultural DPSIR identifies a set of 35 agro-environmental indicators (European Commission, 2000) to help monitor and assess agro-environmental policies and programmes, to provide contextual information for rural development, to identify environmental issues, to help target programmes that address agro-environmental issues and to understand the linkages between agricultural practices and the environment (European Commission, 2001).

Within the Italian project "Agriculture for protected areas" (Agripark, 2006; Bisol, 2006) we are evaluating the environmental sustainability of different cropping and farming systems. Our objective in the project is to synthesise the effects of agricultural management using quantities which: i) allow to integrate different aspects of reality, doing a synthesis characterized by a good compromise between the description of the processes and their simplification into single numerical quantities; ii) can be derived from farm characteristics, easily obtainable from the farmer and/or from existing agricultural databases (e.g. Common Agricultural Policy declarations); iii) are easy to interpret and can be used to drive the improvement of environmental performances of agricultural systems. We therefore excluded the indicators constituted by direct measures on soils, waters or crops. We also excluded indicators like the ones used in the IRENA project (OECD, 2002a; EEA, 2005) because they aggregate data at nation- or macro-region level, and do not represent the actual processes occurring in single farms. The indicators proposed by ANPA (2000) were excluded as well, because many of these require an analytical approach (e.g. state and impact indicators are based on measurements of heavy metals, organic matter, pesticides and nutrients in the soil and in the water), while others can be used only at regional scale and not at field-farm scale.

We propose a framework, derived from an extensive literature review, to evaluate the sustainability of agro-ecosystems management at field and farm level, using a set of agro-ecological indicators divided in five categories (energy, landscape, soil, nutrients and pesticides) that describe the environmental sustainability of farming and cropping systems from different points of view. The different categories of agro-ecological indicators are similar to those found in the literature for farm management analysis (Vereijken, 1995; Bockstaller and Girardin, 2000), and for policy analysis (OECD, 2001; EEA, 2005). We did not select categories describing social and economic sustainability. In this paper we report on the first set of indicators, related to fossil energy use, landscape and soil management. In a second paper we will focus on nutrients and pesticides management.

3.3. Energy indicators

Environmental problems due to the intensive use of energy are crucial, especially for CO_2 and NO_x emission due to fossil energy combustion (Pervanchon et al., 2002) and to the limitation of energy sources available nowadays. CO_2 is one the major greenhouse gases; agriculture can contribute to CO_2 emissions with the use of fertilizers, lime and fuel (Robertson et al., 2000). NO_x contributes to acidification and to the generation of ozone in the troposphere (Olivier et al., 1998). Therefore the quantification of fossil energy use is important in order to improve the efficiency of agro-ecosystems and to reduce the emissions and the consumption of limited resources. Different ways are proposed to quantify fossil energy flows in agricultural systems. Dalgaard et al. (2000) use a synthetic approach based on simple description of farm operations carried out for crop and livestock management. Similar approaches are used by many other Authors (e.g. Biondi et al., 1989; Volpi, 1992). Others, like Pervanchon et al. (2002), use a more analytical methodology to estimate energy flows in cropping systems. Once the flows are calculated with one of these methods, they can be interpreted by calculating output/input ratios; Tellarini and Caporali (1999) provide a rich set of possible ratios.

We describe in detail the simple method of Dalgaard et al.(2000), as a recent and complete example of the approaches of the first type. This procedure assesses fossil energy use in different types of farms. The energy balance is divided into two modules: crop module and animal module, each divided in two sub-modules. The energy use (EU) at farm level is calculated as:

$$EU_{farm} = EU_{crop} + EU_{animal} \text{ (MJ)} \qquad (3.1)$$

The crop module is divided in the sub-module for the direct (EU_{direct}) and for the indirect energy use ($EU_{indirect}$). The sub module EU_{direct} is divided in two components: the first for the diesel fuel (EU_{diesel}), the second for other energy use (EU_{other}):

$$EU_{crop} = EU_{direct} + EU_{indirect} = (EU_{diesel} + EU_{other}) + EU_{indirect} \qquad (3.2)$$

EU_{diesel} represents the diesel use for crop management operations:

$$EU_{diesel} = \sum_{i=1}^{N_{oper}} C_i \cdot D_i \cdot k \ \text{(MJ)} \tag{3.3}$$

where N_{oper} is the total number of operations to grow a specific crop, C_i is the area treated (ha) or the amount input factor applied (t) or the weight of crop harvested (t) or the distance to the field (km) on which the i^{-th} operation is carried out, D_i is the norm of diesel use for that operation (L ha^{-1} for field operations, L km^{-1} for transport, L Mg^{-1} for product removed), k is a specific energetic coefficient (35.9 MJ L^{-1} diesel). For the soil preparation, D_i is corrected for soil type by a factor of 1.1 for a loamy soil, a factor of 1.0 for a sandy-loam soil and a factor of 0.9 for a sandy soil. EU_{other} represents other energy forms directly consumed in the farm activity, such as lubrication, drying and irrigation:

$$EU_{other} = \sum_{i=1}^{N_{oper}} C_i \cdot D_i \cdot L + \sum_{j=1}^{N_{dry}} (AD_j \cdot PD_j \cdot R) + \sum_{k=1}^{N_{irr}} (AI_k \cdot I) \ \text{(MJ)} \tag{3.4}$$

where L is the energy consumed for lubrication per unit of fuel used (3.6 MJ L^{-1} diesel), N_{dry} is the total number of drying operations, AD_j is the mass of crops dried (t, wet basis), PD_j is the percentage of drying (t water removed Mg^{-1} wet crop), R is the energy required for drying (5 000 MJ Mg^{-1} water removed), N_{irr} is the total number of irrigations, AI_k is the amount of water used in the k^{-th} irrigation event (mm), I is the energy consumed for a unit volume of water applied (52 MJ mm^{-1}).

$EU_{indirect}$ represents the energy used in the production of inputs, such as machinery, fertilisers and pesticides.

$$EU_{indirect} = \sum_{n=1}^{N_m} CD_n \cdot M + \sum_{i=1}^{5} AE_i \cdot E_i \quad \text{(MJ)} \tag{3.5}$$

where N_m is the number of machines used in the farm, CD_n is therefore the diesel fuel consumed (L), M is the energy incorporated in the machinery (the energy necessary for the construction, averaged per unit of fuel consumed, 12 MJ L^{-1} diesel); and AE_i E_i represents the indirect energy used derived from five types of external inputs: nitrogen (i = 1), phosphorus (i = 2), potassium (i = 3), lime (i = 4), pesticides (i = 5); AE_i is the total amount of input product used (kg for NPK or pesticides, t for lime), and E_i is the energy needed for the production of the input (50 –

12 – 7 – 40 MJ kg^{-1} for N, P, K and formulated spraying agent of herbicides, insecticides and fungicides respectly, and 30 MJ Mg^{-1} for lime).

The animal module, developed for pig and cattle production, is divided in two sub-modules. The first module (EU_{direct}) describes the direct energy use for cattle or pig breeding; it is divided in two components: livestock housing (S), and heating of the livestock housing (H). The second sub-module ($EU_{indirect}$) describes the indirect energy requirement for the cattle/pig breeding, and is divided in three components: farm building (B), imported fodder (F) and self-produced fodder (O).

$$EU_{animal} = EU_{direct} + EU_{indirect} \tag{3.6}$$

$$EU_{animal} = (S + H) \cdot LSU + (B \cdot LSU + F \cdot SFU + O \cdot SFU) \tag{3.7}$$

where S represents the energy required for operation in livestock housing (light, ventilation, milking, milk cooling, fodder milling and pumping), equivalent to 8 – 1.7 – 6.1 – 3.2 – 0.9 – 0.5 GJ LSU^{-1} (Livestock Unit, corresponding to1 large-breed dairy cow, or 30 slaughter pigs) for dairy cows, other cattle, conventional sows, organic sows, conventional slaughter pigs, organic slaughter pigs, respectively; H is the energy required for heating the cattle or pig housing (3.1 – 0.6 GJ LSU^{-1} for conventional sows and conventional slaughter pigs, respectively); B is the energy required for the maintenance of farm buildings and the store (2.5 GJ LSU^{-1}), F is the energy for the imported fodder (5.7 MJ SFU^{-1}, Scandinavian Feed Unit, corresponding to 12 MJ of metabolizable energy, equivalent to the fodder value of 1 kg of barley), O is energy consumption for self-produced fodder (EU_{crop}/harvested yield, MJ SFU^{-1}). Overall, the coefficients proposed by Dalgaard et al. (2000) for converting mass fluxes into energy fluxes are in good agreement with the ones found in other similar works (e.g. Biondi et al., 1989; Jarach, 1985); more specific parameters (e.g. energy content of single active ingredients or pesticide groups) can be found in Volpi (1992). The use of older parameters, however, needs to be carefully evaluated because, as stated by Pervanchon et al. (2002), the efficiency of production of fertilisers and pesticides has increased in the last decades.

An alternative methodology is proposed by Pervanchon et al. (2002) and by Bockstaller and Girardin (2000); they suggest the use of an energy indicator (I_{En}) to evaluate the energy consumption of field crop production calculated with an analytical approach. The indicator provides a value from 0 (worst value) to 10 (best value). A value of 7 represents the achievement of a minimum level.

The energy indicator (I_{En}) is defined as:

$$\begin{cases} I_{En} = 10 & \text{if } 0 < E_t < 3500\, MJ \cdot ha^{-1} \\ I_{En} = a \cdot E_t^{\,2} + b \cdot E_t + c & \text{if } 3500\, MJ/ha < E_t < 34\,900\, MJ \cdot ha^{-1} \\ I_{En} = 0 & \text{if } E_t > 34\,900\, MJ \cdot ha^{-1} \end{cases} \quad (3.8)$$

where E_t is the total amount of energy consumed (MJ ha^{-1}) and a, b, c are coefficients (a = 8.75544 10^{-9}; b = –6.5492 10^{-4}; c = 12.184).

E_t is composed by four modules:

$$E_t = E_m + E_{irr} + E_{fert} + E_{phyto} \quad (3.9)$$

E_m for the fuel consumption, E_{irr} for the irrigation, E_{fert} for fertiliser utilisation, and E_{phyto} for pesticide utilisation. E_m (MJ ha^{-1}) quantifies the direct energy consumed by machinery for each crop management operation, without considering the energy incorporated in the machines during construction.

$$E_m = [(36\ P_a / \eta) / (VLC)] + D / S \quad (3.10)$$

where 36 is a conversion factor, P_a is the tractor power required (kW), η is motor yield (estimated equal to 35%), V is tractor speed (km h^{-1}), L is machine width (m), C is a correction coefficient taking into account the over consumption factor (dimensionless), depending on machine characteristics that increase the energy consumption, D is a correction factor taking into account the distance between the farm and the field (MJ), S is the field area (ha). P_a can be obtained from a database developed by the French Institute for Cereal Crop (ITCF), or can be estimated by a linear correlation:

$$P_a = \alpha VL + \beta V + \alpha' L + \beta' \quad (3.11)$$

where V and L are given by the farmer, and α, β, α’, β’ are coefficients calculated by means of a linear regressions for each machine. The over consumption factor is calculated as:

$$C = C_1 C_2 C_3 C_4 C_5 F \qquad (3.12)$$

where C_1 is 1.00 if the tractor has a driving help systems (e.g. computer) and 0.93 if not, C_2 is 1.00 if the difference between the real tractor power and the power required for the machine used is lower than 15%, 0.85 if the difference is comprised between 15 and 30%, and 0.70 if the difference is greater than 30%, C_3 is 1.00 if the maintenance of the field machines is good, and 0.92 in other cases, C_4 ranges from 0.65 to 1.00 on the basis of the maintenance of the tractor (air filter change, injector and fuel pomp adjustment, tyre’s pressure), C_5 ranges from 0.50 to 1.00 according to the soil wetness during work and the pneumatics characteristics (width and age), F depends on the type of machine and on field size. $D = (35.8\ t_c / 8)\ d$ (MJ), where 35.8 is the energy constant of 1 litre of diesel fuel (MJ L^{-1}), t_c is the specific tractor consumption (L h^{-1}), 8 is a reference tractor speed (km h^{-1}) and d is the farm-field distance (km). E_{irr} (MJ ha^{-1}) accounts for energy consumption used in irrigation:

$$E_{irr} = [36\ P_u\ I / (Q\ G)] + A / S\ (\mathrm{MJ\ ha^{-1}}) \qquad (3.13)$$

where 36 is a conversion factor, P_u is the power absorbed by the pump (kW), I is the irrigation volume (mm), Q is the water flow ($m^3\ h^{-1}$), G (dimensionless) is a correction coefficient for the over consumption factors (related to the type of irrigation, the water transport efficiency, the maintenance and accessories of the irrigation systems), A (MJ) is a correction coefficient taking into account the energetic cost of the implementation of the irrigation system (reservoir or well), S is the area of the irrigated field (ha). The correction coefficient

$$G = G_1 G_2 G_3\ \text{(dimensionless)} \qquad (3.14)$$

where G_1 is a correction coefficient for the application efficiency (0.6 for flooding on a sandy soil, 0.7 for flooding on other soil types, 0.9 for localized irrigation and 0.8 for sprinkler irrigation), G_2 is a coefficient considering the water transport efficiency (0.8 for flooding and 1.0 for localized and sprinkling), G_3 is a coefficient for the maintenance and accessories of the irrigation system (it varies from 0.8 to 1.0 depending on the presence of a sprinkler automatism system and the periodical check of irrigation system state),

$$A = [h_d\,(4000 + 120 + 130)] / 30 \text{ (MJ m}^{-1}) \quad (3.15)$$

where h_d is drilling height expressed in metres, 4000, 120 and 130 (MJ) are the energy consumptions for drilling, cement and steel, respectively, used for 1 m depth. The values of A assume a life of 30 years for the well. In case of a reservoir, A represents 40% of the corresponding drilling cost. For irrigation with surface water (e.g. rivers, lakes), A = 0 (Pervanchon, personal communication) because the water does not need work to be extracted. The indirect energy costs for fertilisers

$$E_{fert} = D_{fert}\,k_{fert} + FPT \text{ (MJ ha}^{-1}) \quad (3.16)$$

is obtained by multiplying the total amount of the product applied, D_{fert} (kg ha^{-1}), by a specific energetic coefficient, k_{fert} (MJ kg^{-1}), which includes the energetic costs for fertiliser production. In order to estimate the energy costs for formulation, packaging and transport of input product used, it is necessary to add the Formulation Packaging Transport Coefficient (FPT) of the specific nutrient in the fertilisers. FPT cost is 1.5 – 9.8 – 7.3, and 5.7 MJ kg^{-1} respectively for N fertilisers, P fertilisers, K fertilisers, and NP fertilisers. For other types of fertilisers (S, etc.) the mean FTP cost is 6 MJ kg^{-1}. The indirect energy costs for pesticides (E_{phyto}) is obtained by multiplying the total amount of active ingredient (D_{phyto}) by a specific energetic cost coefficient k_{phyto}. For example k_{phyto} is 310, 272 and 214 MJ kg^{-1} for generic insecticides, herbicides, and fungicides, respectively. Specific k_{phyto} for several active ingredients are also indicated by Volpi (1992).

After fossil energy inputs have been quantified, the energy content of crop and animal products can be calculates, using coefficients available in the literature (e.g. Biondi et al., 1989; Jarach, 1985; Volpi, 1992). In order to describe the sustainability of crops and farming systems, it is then possible to highlight the relation between inputs and outputs. Based on the classical calculation of output/input ratios, Tellarini and Caporali (1999) have proposed an input/output methodology, providing several indicators to describe and to analyse farming systems in terms of energy and monetary values. The indicators are based on the quantification of input (i) and output (o) flows of energy and money. These flows can be directed from inside to outside the farm (or vice versa), or can be completely internal (recycling). Internal transfers can be classified as “obligatory” (crop roots and part of crop residues left in the soil which, not being removed from the system, are reused) or “voluntary” (all farm products that the farmer chooses to recycle into the production process rather than destine for final consumption). In particular (Table 3.1), internal transfers can derive from current year (i1) or previous year farm production (i2); inputs from outside derive from agriculture (i3), or from other production sectors (i4). Similarly, output flows can recycle production in current year (o1) or for subsequent cycle (o2), or can be destined for final consumption (o3). A set of agro-ecosystem performance indicators (Table 3.2) is then defined to compare homogeneous output/input flows (monetary or energetic: “direct” indicators), or heterogeneous output/input flows (monetary versus energetic: “crossed” indicators). Direct indicators can be structural (describing the most relevant characteristics of agricultural systems) or functional (measuring the efficiency of different systems and the dependence from non-renewable or external inputs).

Table 3.1. – Energy (GJ) and monetary (€) inputs and outputs (from Tellarini and Caporali, 1999)

i1	Total re-use of current year farm production (internal transfers)
i1a	Obligatory re-use of current year farm production
i1b	Voluntary re-use of current year farm production
i2	Total re-use of previous year's farm production
i2a	Obligatory re-use of previous year's farm production
i2b	Voluntary re-use of previous year's farm production
i3	External input produced by agriculture
i4	External input produced by other sectors (non-renewable)
i5	Input produced on the farm (i1+i2)
i6	Input external to the farm (i3+i4)
i7	Input produced by agriculture (i1+i2+i3)
i8	Total input (i1+i2+i3+i4)
o1	Output destined for re-use on the farm in the current year
o1a	Output obligatorily destined for re-use on the farm in the current year
o1b	Output voluntarily destined for re-use on the farm in the current year
o2	Output destined for the subsequent cycle
o2a	Output obligatorily destined for the subsequent cycle
o2b	Output voluntarily destined for the subsequent cycle
o3	Output destined for final consumption
o4	Net output (o2+o3)
o5	Gross output (o1+o2+o3)

Table 3.2. – Sructural, functional and crossed indicators (from Tellarini and Caporali, 1999)

Structural indicators	
1	Indicator of dependence on non-renewable energy sources (i4/i8)
2	Indicator of obligatory re-use [(i1a+i2a)/i8]
3	Indicator of immediate voluntary re-use (i1b/i8)
4	Indicator of deferred voluntary re-use (i2b/i8)
5	Global indicator of voluntary re-use [(i1b+i2b)/i8]
6	Indicator of farm autonomy (i5/i8)
7	Indicator of overall sustainability (i7/i8)
8	Indicator of immediate removal (o3/o5)
9	Indicator of total removal (o4/o5)
10	Indicator of obligatory internal destination [(o1a+o2a)/o5]
11	Indicator of immediate voluntary internal destination (o1b/o5)
12	Global indicator of immediate internal destination (o1/o5)
Functional indicators (GJ/GJ or €/€)	
13	Indicator of gross output from total input (o5/i8)
14	Indicator of gross output from total farm input (o5/i5)
15	Indicator of gross output from annual farm input (o5/i1)
16	Indicator of gross output from external non-renewable input (o5/i4)
17	Indicator of gross output from total external input (o5/i6)
13	Indicator of net output from total input (o4/i8)
14	Indicator of net output from total farm input (o4/i5)
15	Indicator of net output from annual farm input (o4/i1)
16	Indicator of net output from external non-renewable input (o4/i4)
17	Indicator of net output from total external input (o4/i6)
Crossed indicators (€/GJ or GJ/€)	
18	Gross economic productivity of total energy input (o5/i8)
19	Gross economic productivity of energy input from outside the farm (o5/i6)
20	Gross economic productivity of non-renewable energy input (o5/i4)
21	Gross economic productivity of energy input produced by agriculture (o5/i7)
22	Net economic productivity of total energy input (o4/i8)
23	Net economic productivity of energy input from outside the farm (o4/i6)
24	Net economic productivity of non-renewable energy input (o4/i4)
25	Net economic productivity of energy input produced by agriculture (o4/i7)

3.4. Landscape indicators

The agro-ecological network, made of the patch of cultivated fields and interconnected linear elements (such as hedge-rows), has a double function: to build the landscape and to maintain the biological diversity, important for air, water and soil quality. Farm management influences the quality of landscape and consequently the biodiversity and this concept is highlighted by several approaches aimed at studying the importance of agriculture in the evolution of the landscape (Weinstoerffer and Girardin, 2000). Here we focus on indicators describing the relation between farm management and landscape.

In particular, many indicators were developed for application at regional scale, where the action of agriculture takes place on landscape. However, indicators can be useful also to identify the contribution of single farms to landscape quality and biodiversity. We will therefore comment the indicators which can be applied at single farming systems, like the crop diversity indicator proposed by Bockstaller and Girardin (2000), the hedge-row indicator used by Bocchi et al. (2004), applied at regional scale, but applicable also for single farms, and the landscape indicator by Weinstoerffer and Girardin (2000). Other approaches like the Mosaic indicators (Hoffmann and Greef, 2003; Hoffman et al., 2003) consider the presence and abundance of specific indicator species, chosen as representative of the location and the region. This indicator requires several input data, some of which require specific measurements; for this reason, and despite their interest, we will not consider these indicators in our review.

At the farm scale, Bockstaller and Girardin (2000) have proposed a crop diversity indicator (I_{cd}) that evaluates the impact of crop partitioning and field size on landscape and biodiversity. It provides a value from 0 (worst case) to 10 (best case). A value of 7 represents the achievement of a minimum level. The indicator is calculated as:

$$I_{cd} = K\ NC\ D\ T \qquad (3.17)$$

where K is a calibration factor depending on the number of crops (K is equal to 2.00, 1.83, 1.70, 1.59, 1.50, 1.42, 1.36, 1.30, 1.25 if NC is $\leq$ 4, 4.5, 5.0, 5.5, 6.0, 6.5, 7.0, 7.5, 8.0 respectively), NC is the number of crops (from 1 to 8; the intercrop in double-crop systems has a weight of 0.5), D is the crop partitioning factor and T is the field size factor. The crop partitioning factor (D) measures the diversity of crop partitioning; its maximum value is 1, and corresponds to the situation when the areas cultivated with each crop are equal; low values indicate that one or few crops dominate, i.e. occupy most of farm area. The factor D is calculated by dividing the Shannon diversity index (calculated using crop areas instead of species abundances) by the maximum value of the Shannon index which would be obtained in the case of homogeneous partitioning:

$$D = IS / IS_h \qquad (3.18)$$

where

$$IS = \sum_{i=1}^{NC} (p_i \ln p_i) \qquad (3.19)$$

with $p_i = S_i / S_{tot}$ (ratio of S_i the area of the i^{-th} species, to S_{tot}, the total farm area), and $IS_h = \ln (1 / NC)$. The field size factor (T) considers the fragmentation of the field: $T = 1 - SA_{big} / S_{tot}$ where SA_{big} is the area of the fields considered “big” (ha):

$$SA_{big} = \sum_{i=1}^{FN} (c_i F_i) \qquad (3.20)$$

where FN is the number of the fields, F_i is the area of the i^{-th} field (ha), c_i is a factor depending on field size:

$$c_i = 0 \text{ if } F_i < L_{low} \qquad (3.21)$$

$$c_i = 1 \text{ if } F_i > L_{high} \qquad (3.22)$$

$$c_i = \left(\frac{1}{(L_{high} - L_{low})} \right) \cdot (F_i - L_{low}) \text{ if } L_{low} < F_i < L_{high} \qquad (3.23)$$

where L_{high} is the threshold over which the field is considered “big” (proposed value: 15 ha), L_{low} is the threshold under which the field is considered “small” (proposed value: 5 ha).

The hedge-row indicator (Bocchi et al., 2004) describes the evolution and the quality of the landscape, considering the hedges and the rows as important structural elements.

$$I_{hr} = L / A \ (\mathrm{m\ ha^{-1}}) \qquad (3.24)$$

where L is the hedge-row length (m) and A is the total area analysed (ha). This indicator has been created and applied for the regional scale, but it is possible to apply it also at the farm scale.

The landscape indicator (Weinstoerffer and Girardin, 2000) evaluates

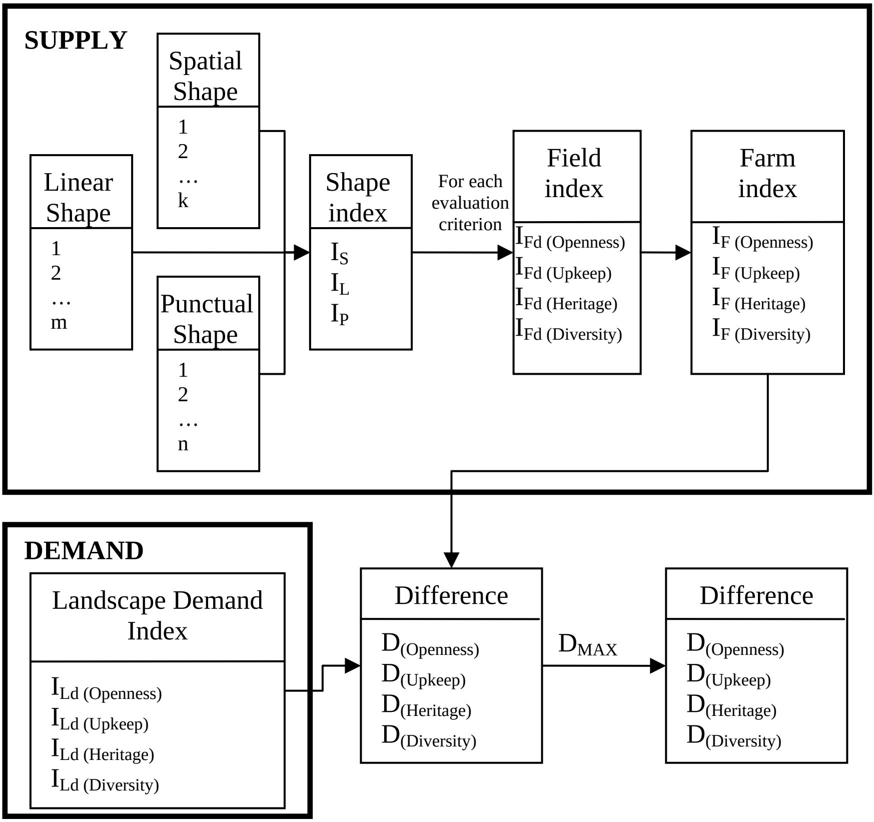

Fig. 3.2. – Calculation of the landscape indicator (I_{LAND}) (from Weinstoerffer and Girardin, 2000).

the correspondence between landscape supply by farmer and landscape demand by social groups. The four main evaluation criteria proposed to calculate demand and supply are: openness (the ease with which an observer can obtain an exhaustive view over the surrounding country), upkeep (the fact that land forms are as uniform and well organised as possible), heritage (the presence of evidence of numerous traces of ancient practices), diversity (the differences in nature, quality and aspect). To calculate the indicator (Fig. 3.2), supply and demand are separately calculated, and then compared, for each evaluation criterion; landscape supply is calculated for each field and averaged for the farm.

In the calculation of the supply, the landscape is described with three different types of shapes that compose the landscape: spatial shapes (crop, permanent grassland, farm yard, woodland, etc.), linear shapes (hedge, row of trees, grassland margin, wall, trench, bank, etc.) and isolated shapes (single tree, agricultural equipment, building, etc.). For every evaluation criterion (openness, upkeep, heritage, diversity) all shapes pertaining to a field are evaluated with a score; the scores are then linearly aggregated:

$$I_S(i) = \frac{\sum_{k=1}^{j} S_k}{j} \tag{3.25}$$

$$I_L(i) = \frac{\sum_{k=1}^{m} \left[L_k \cdot (H_k)\right]}{\sum_{k=1}^{m} L_k} \tag{3.26}$$

$$I_P(i) = \frac{\sum_{k=1}^{n} P_k}{n} \tag{3.27}$$

where $I_S(i)$ is the spatial shape index, $I_L(i)$ is the linear shape index and $I_P(i)$ is the punctual shape index for the i^{-th} field; S_k is the score that characterizes the state of k^{-th} spatial shape (not weighted based on shape area), j is the number of spatial shapes in the i^{-th} field; H_k is the score that characterizes the state of the k^{-th} linear shape, L_k is the length of the

k^{-th} linear shape, m is the number of linear shapes in the i^{-th} field; P_k is the score that characterizes the state of k^{-th} punctual shape, n is the number of punctual shapes in the i^{-th} field. The score is limited between 0 to 4 (0 is given to the shape contributing the least to the criterion), and the authors suggest an expert judgement score for the openness criterion (Table 3.3), the landscape upkeep criterion (Table 3.4), and the heritage criterion (Table 3.5).

For the landscape diversity criterion, the supply of diversity of a farm is integrated into the calculation of the crop diversity indicator, described previously (I_{cs}, Bockstaller and Girardin, 2000). The field index (I_{Fd}) is obtained by combining the three shape indices:

$$I_{Fd(i)} = \frac{I_{S(i)} + I_{L(i)} + I_{P(i)}}{3} \tag{3.28}$$

assuming that each shape group, at this level, has an equal influence in the contribution to the landscape supply. The farm index is obtained by combining the field indices for the total number of fields in the farm. The score for every field is weighted accordingly to its area and it is separately calculated for every evaluation criterion:

$$I_F = \frac{\sum_{i=1}^{n}\left[A_{(i)} \cdot I_{Fd(i)}\right]}{\sum_{i=1}^{n} A_{(i)}} \tag{3.29}$$

where I_F is the farm index for a specific criterion, $A_{(i)}$ is the area of i^{-th} field.

The evaluation of the landscape demand is qualitative and it is done using the judgement of several stakeholders. In a questionnaire all the terms are listed that can be used to describe agricultural landscape in a qualitative way. The stakeholders have to choose the element which they wish to see in the farm area (Table 3.6). The median score is assumed as the indifference evaluation on the part of the observers. For each criterion the absolute value of the difference between the supply and the demand is calculated. It is assumed that one of the criteria cannot compensate for another in the final result; therefore for the evaluation of landscape indicator the least favourable difference between supply and demand (D_{MAX}) is used. The maximum difference is scaled to a maximum of 10 using a coefficient (2.5) to obtain the landscape indicator: $I_{LAND} = 10 - (D_{MAX}\ 2.5)$. The landscape indicator can also be used for an assessment at the regional level.

Table 3.3. – Scores of spatial (S_k) and linear (H_k) shapes contributing to landscape openness, used in the calculation of the landscape indicator (from Weinstoerffer and Girardin, 2000)

Contribution of the spatial shapes to the openness supply			
Forest	0	Wooded orchard	3
Intensive orchard hops	1	"Open" crops[2]	4
"Closed" crops[1]	2		
Contribution of the linear shapes to the openness supply			
Linear wooded margin	0	New hedge[3], line of trees	3
Windbreak hedge	1	Grassland field margin	4
Hedge fence	2		

[1] "Closed" crops: maize, sorghum, sunflower, which close the landscape because of the height of the plants;
[2] "Open" crops, which have no influence on the openness of landscape;
[3] Less than 2 years.

Table 3.4. – Scores of spatial (S_k), linear (H_k) and punctual (P_k) shapes contributing to landscape upkeep, used in the calculation of the landscape indicator (from Weinstoerffer and Girardin, 2000)

Contribution of the spatial shapes to the upkeep supply					
Crops	Tillage and weeding				
	No	Mechanical weeding		Chemical weeding	
	Intervention	No tillage	Tillage	No tillage	Tillage
Winter cereals, rape seed	0.0	1.0	2.0	3.0	4.0
Spring cereals, peas	0.0	1.0	2.0	4.0	
Other spring crops	0.0	0.5	1.5	3.5	
Green manure	2.0	3.0	2.0	3.0	
Set aside	Spontaneous set aside				
	After sunflower, maize, soybean, sugar beet or potatoes		After cereals, rape seed or peas		Cultivated set aside
	0		1		2
Fallow land	0				
Permanent crops					
Meadow	Cutting frequency (yr^{-1})				
	≥3		2		1
	4		2		0
Cut grazed pasture	Cutting or grazing frequency (yr^{-1})				
	≥3		2		1
	4		2		0
Grazed pasture	Intensive pasture				Extensive pasture
	With refusals cutting		Without refusals cutting		
	4		2		0
Arboriculture — forest	Woody plant pruning				
	Regular upkeep		Occasional upkeep		Without upkeep
	4		2		0
Farm yard	Aspect				
	Kept up		Occasional		Untidy
	4		2		0
Contribution of the linear shapes to the upkeep supply					
Upkeep of each linear shape	Regular		Occasional		Without upkeep
	4		2		0
Contribution of the punctual shapes to the upkeep supply					
Upkeep of each punctual shape	Regular		Occasional		Without upkeep
	4		2		0

Table 3.5. – Scores of spatial (S_k), linear (H_k) and punctual (P_k) shapes contributing to landscape heritage, used in the calculation of the landscape indicator (from Weinstoerffer and Girardin, 2000)

Contribution of the spatial shapes to the heritage supply					
The crops					
Species	Usual 4		New 0		
Area	Stable 4		Transformed 2		Changed 0
Shape	Stable 4		Transformed 2		New 0
The Farm					
Site	Identical 4		New 2		
Arrangement	Identical 4		Transformed 2		New 0
Farm yard	Identical 4		Transformed 2		New 0
Contribution of the linear shapes to the heritage supply					
Linear shapes	Stable 4		Transformed 0		
Contribution of the punctual shapes to the heritage supply					
Building	Maintained customs 4	Died out customs 3	Rebuilt according to the traditional style 2	Deeply reshaped 1	Newly built or destroyed 0
Vegetation	Identical 4		New 0		

Table 3.6. – Scores of desired landscape elements used for landscape demand calculation (I_{ld}) (modified from Weinstoerffer and Girardin, 2000)

Evaluation criteria							
Openness		Heritage		Upkeep		Diversity	
Bared	4	Preserved	4	Meticulous	4	Varied	4
Stripped	3	Protected	3	Well kept	3	Heterogeneous	3
Indifferent	2	Indifferent	2	Indifferent	2	Indifferent	2
Obstructed	1	Modified	1	Badly kept	1	Homogeneous	1
Blocked	0	Transformed	0	Disused	0	Uniform	0

3.5. Soil management indicators

These indicators describe how tillage, incorporation of organic materials into the soil, soil cover with crops and residues, and consequently crop rotations influence soil fertility.

Crop rotation is one of the most important factors that influence soil fertility, helping to break the cycle of harmful organisms, improving soil structure, enhancing soil quality and making soil less vulnerable to erosion (Leteinturier et al., 2006). The crop sequence indicator (I_{sc}) was developed by Bockstaller and Girardin (1996, 2000); it provides a value from 0 (worst value) to 10 (best value); a value of 7 represents the achievement of a minimum level. It is a method of global diagnosis applicable at field level; for a single crop it is defined as:

$$I_{sc} = K_p\ K_r\ K_d \qquad (3.30)$$

where K_p is the coefficient describing the effects of the preceding crop on the current crop, K_r depends on the frequency of crop cultivation and K_d is an index of crop diversity.

K_p is derived from the sum of five effect scores (development of pathogens, pests, weeds, soil structure and nitrogen), representing the effect of the previous crop on the current crop. These effects are estimated by an expert group on a semiquantitive scale, from –1 to +1 for soil structure and nitrogen supply, from –3 to +1 for pathogens, from –2 to +1 for weeds and pests. A transformation is then made to convert the sum of scores (S) into the K_p coefficient, obtaining a value on a scale from 1 to 6 ($K_p = S + 5$ whit a minimum and maximum value of 1 and 6 respectively). Examples of K_p values for many previous/actual crop combinations are given in the literature for French and Belgian pedo-climatic and agronomic conditions (Bockstaller and Girardin, 1996, 2000; Leteinturier et al., 2006). K_r is obtained by transforming the difference of the actual return time of crop on a field (t) minus the recommended return time (t_r) which is known to limit the risks of diseases or pests (K_r is equal to 0.3, 0.5, 0.8, 1.0 and 1.2 if $t–t_r$ is –3, –2, –1, 0 and $\geq$ 1 respectively). The quantity K_d is calculated by transforming the number of different crops (NC) cultivated in the last four years (K_d = 0.2 NC + 0.6 with a minimum and maximum value of 1.0 and 1.4 respectively). The indicator I_{sc} can be calculated for every crop in the rotation; the I_{sc} for the entire rotation is calculated as the average of the I_{sc} of single crops.

One of the most important attributes of soil quality is the organic matter (SOM) content; Bockstaller et al. (1997) and Bockstaller and Girardin (2000) have proposed the organic matter indicator in order to detect the negative and the positive effects of different crop management practices on SOM content. The aim of this indicator is to identify and promote the practices that maintain SOM at a satisfactory level. It is an impact indicator applicable at field level and it provides a value from 0 (worst value) to 10 (best value); a value of 7 represents the achievement of a minimum level. The indicator is defined as: $I_{OM} = 7\ (A_x / A_r)$, where A_x (kg ha^{-1} yr^{-1}) is the mean of OM inputs (residues, manure, green manure, etc.) in the four preceding cropping years, A_r (kg ha^{-1} yr^{-1}) is the recommended level of OM inputs needed to maintain a satisfying level of SOM in the long term.

The organic supply (A_x) is defined as (Boiffin et al., 1986):

$$A_x = \sum_{i=1}^{N} (k_{root\,(i)} \cdot m_{root\,(i)} \cdot f_{root\,(i)} + k_{residue\,(i)} \cdot m_{residue\,(i)} \cdot f_{residue\,(i)})$$
$$\sum_{i=1}^{N} (k_{manure\ \ (i)} \cdot m_{manure\ \ (i)} \cdot f_{manure\ \ (i)}) \tag{3.31}$$

where k_{root}, $k_{residue}$, k_{manure} are humification coefficients of roots, residues and manures, respectively (dimensionless), m_{root}, $m_{residue}$, m_{manure}, are the mass applied of roots, residues and manures respectively (kg ha^{-1} yr^{-1}), and f_{root}, $f_{residue}$, f_{manure} are the frequencies of application in the four years. Example of k coefficients are available in Boiffin et al. (1986).

The Hénin and Dupuis model (1945) is used to derive the relationship between the equilibrium level of SOM and OM inputs to a specific soil:

$$A_r = \tau_{es}\ k_2\ M\ P \tag{3.32}$$

where τ_{es} is the SOM concentration (g SOM g^{-1} soil) recommended for a specific textural class, k_2 is the annual mineralization coefficient (yr^{-1}), M is the soil mass at tilled depth (kg soil ha^{-1}), P is a modifier of the mineralization coefficient (dimensionless). The annual mineralization coefficient is estimated on the basis of soil texture, limestone content and air temperature:

$$k_2 = \frac{1200 \cdot f_\theta}{(200 + A) \cdot (200 + 0.3 \cdot C)} \quad \text{(Boiffin et al., 1986)} \qquad (3.33)$$

where f_θ is a temperature factor: $f_\theta = 0.2(T - 5)$, where T is the average annual air temperature (°C), A is clay content (g kg^{-1}), C is limestone content (g kg^{-1}). If no soil analyses are available for every field, soil maps or geostatistical techniques (e.g. Guimaraes Couto et al., 1997; Schloeder et al., 2001; De Ferrari et al., 2002) can be used to estimate clay and limestone contents. The modifier of mineralization coefficient is calculated as:

$$P = f_r \, I \, T_s \qquad (3.34)$$

where f_r is a coefficient considering crop management (Table 3.7), I is a mineralization weight factor (suggested value: 1.25), T_s is a tillage factor (1.0 if the soil is tilled at least once in four years; 0.5 if only no tillage practices were used in the last four years; 0.8 in intermediate cases, with at least one year of minimum tillage).

Other risks related to soil management are structure degradation, erosion, nutrient and pesticide losses and reduction of biodiversity. Vereijken (1995) has proposed the Soil Cover Index (SCI) for evaluation of soil protection by crops. This indicator calculates the percentage of soil cover by crops or residues in a short period (month),

Table 3.7. – Crop management coefficient (f_r) used to consider crop management in the calculation of the modifier of mineralization coefficient (P) in the organic matter indicator (from Bockstaller and Girardin, 2000)

Crop residue management	Organic input frequency (manure, compost, etc.)			
	> 10 years	Between 5 and 10 years	Between 3 and 5 years	< 3 year
Removed or burned	0.8	0.9	1.0	1.1
Incorporated once in two years	0.9	1.0	1.1	1.2
Incorporated every year	1.0	1.1	1.2	1.3

in one year or in a critical period (e.g. autumn): $SCI_{month} = (SCI_{start} + SCI_{end}) / 2$, where SCI_{start} is the percentage of soil surface cover by crops or residues on the first day of the month and SCI_{end} is the percentage on the last day. To avoid direct measurements of soil cover by the crop, the well know crop coefficients (Allen et al., 1998) can be used. SCI is 1 if the soil is completely covered by crops or residues and is 0 if the soil is bare. It is possible to choose intermediate values in proportion to the percentage of cover. For a period longer than a month (e.g. year):

$$SCI_{period} = \left(\sum_{i=1}^{n} SCI_{month} \right) / n \qquad (3.35)$$

where n is the number of months considered. SCI_{month} provides a value between 0 and 1, and SCI_{period} is in the range 0 – 12, if the chosen period is one year. Once SCI is calculated at field scale, it can be averaged for the farm, recalling that it is necessary to calculate the SCI also for the fallow, for the woodland and for the hedge-row. Similar calculations can be done also at regional scale. The OECD (2001) suggested the use of a similar indicator, calculated from agricultural census data, and representing the number of days in a year that agricultural soils are covered with crops. The OECD (2001) proposed also another indicator at national scale, but applicable also at farm scale, in order to represent the winter soil cover; its values are calculated according to the type of cover, and are maximum (100) for fallow land planted before September, intermediate for rapeseed and winter wheat (80 and 40 respectively) and lowest for bare soil (0). The individual values are then aggregated into a single indicator. The risk for soil erosion and nutrient leaching is considered acceptable when the aggregate index is above 50.

For the determination of the risk of soil erosion by water, the OECD (2001) proposed to use the well-known Universal Soil Loss Equation (USLE):

$$E_{water} = R\ K\ LS\ C\ P / T \qquad (3.36)$$

where E_{water} is an indicator of the potential long term average annual soil loss (unitless), R is the rainfall and runoff erosivity (MJ mm ha^{-1} h^{-1} yr^{-1}) considering the intensity, the duration and the frequency of rain storms, K is the soil erodibility factor (Mg h MJ^{-1} mm^{-1}), LS is the slope length-gradient factor (dimensionless), C is the crop/vegetation and management factor (dimensionless), P is the conservation management factor (dimensionless), T is the tolerable soil loss rate (Mg ha^{-1} yr^{-1}), which can be evaluated according to the levels (Table 3.8) of soil erosion risk proposed by the OECD (2001).

We did not find in the literature a simple indicator of the effects of soil management on soil structure and its stability. We believe that this would be a very important indicator, also considering the increasing importance of no tillage and minimum tillage practices.

Works as the ones of Défossez and Richard (2002) or of Roger-Estrade et al. (2000) may constitute a good starting point for the development of such an indicator.

Table 3.8. – Definition of soil water erosion risk based on the total amount of soil loss (from OECD, 2001)

Definition	Mg ha^{-1} yr^{-1}
Tolerable erosion	< 6.0
Low erosion	6.0 – 10.9
Moderate erosion	11.0 – 21.9
High erosion	22.0 – 32.9
Severe erosion	> 33.0

3.6. Discussion and conclusions

The proposed agro-ecological indicators can be calculated at field and farm scale on a relatively small data set describing management, based on farmer's declarations, public databases, or remote-sensed information, without the need of direct measurements. Their calculation is relatively rapid, and interpretation is simple. As such, they represent an excellent tool to rank and classify cropping and farming systems according to their level of sustainability, by exploring a wide range of aspects (fossil energy use, landscape and soil management). After the application of the indicators, additional analyses for particular fields or farms can be carried out, by applying simulation models, or by taking direct measures of the variables of interest for understanding specific processes. Several critical aspects, however, should be considered, namely indicator complexity, input data uncertainty, parameterisation and benchmarks.

3.6.1. Simple vs. complex indicators

First of all, agro-ecological indicators vary widely in the range of complexity and in the associated range of detail of system representation. As shown in this review, indicators range from simple ratios (e.g. the hedge-row indicator) to complex calculations involving detailed aspects of crop management (e.g. the energy indicator proposed by Pervanchon et al., 2002). The question then arises whether one should use a simple or a complex indicator. In general, simple indicators require less input data and are easier to calculate, but the representation of the system they can provide may be poor.

The quantification of fossil energy use with the method of Dalgaard et al. (2000) is based on crop management data at field and animal housing level which can be obtained by interviewing the farmer. Therefore this indicator represents a good compromise between detail and ease of application. On the other hand, the approach proposed by Pervanchon et al. (2002) is relatively more complex (being based on numerous variables about agricultural machineries) and can be used to better calculate and understand energy flows at the cropping system level. Also, their method is more process-based compared to Dalgaard et al. (2000) and is therefore more promising to evaluate alternative management scenarios; a limitation is that it does not consider animal breeding. If one would like to calculate fossil energy use at farm level only, the approach would be much easier: aggregated consumptions of fuel, fertilisers and pesticides (derived for example from documents of purchase) could be multiplied by energy conversion coefficients.

The three landscape indicators are of different level of detail; the one proposed by Bocchi et al. (2004) is the simplest, but does not consider several important factors, as the size and the degree of connection of different vegetated elements; it is therefore adequate for a first screening over large areas, but the results need to be further developed using other approaches. The crop diversity indicator of Bockstaller and Girardin (2000) makes an original synthesis of various important aspects of crop allocation to farm land (number of crops and area occupied, area of single fields), and represents a useful tool to investigate the effects of the crop partitioning scheme on landscape quality. A limitation of this indicator is that the temporal variation of crop appearance and its effects on landscape are not taken into account, i.e. the crops are considered as static entities showing no variation over time. The crop diversity indicator is further developed in the framework of the landscape indicator of Weinstoerffer and Girardin (2000), which also considers non-crop elements of farming systems; it is therefore the most complete landscape indicator revised here, with the additional advantage of considering the point of view of interested stakeholders in the concept of landscape demand.

Soil management indicators represent the effects of various processes on soil fertility. The simplest indicator is the Soil Cover Index, which can be easily calculated based on sowing and harvest dates and literature data on crop cover. A simple but reductionist (Kinnell, 2005) approach is also used to calculate the risk of soil erosion. In spite of their simplicity, these two indicators can rank different cropping systems (e.g. for erosion: Boellstorff and Benito, 2005) and allow further studies with models or direct measurements on soil, water and nutrient dynamics for specific cultivation systems. The organic matter indicator makes a synthesis of different aspects of crop management related to humus formation and mineralization. This complex issue is approached with the simplified annual mineralization and humification coefficients, corrected for climatic, soil and management effects. Again, it is an approach which can be used to integrate existing information about cropping systems management, to estimate trends and to compare cultivation systems. If more insights are needed, the application of a dedicated simulation model, integrated with relevant experimental data, would represent a good way forward. Finally, the crop sequence indicator attempts to compare cropping systems based on the goodness of crop combinations in the rotation. This is a complex issue, involving many different aspects of soil fertility. In this case, an indicator is probably the best approach when a quantitative solution is needed: simulation models do not fully consider the wide range of processes involved (e.g. pests, weeds) and direct measurements would be too expensive, due to the large number of variables to be considered.

Therefore, we believe that in most cases the indicators can be used as a first warning system before other more complex solutions are introduced in the study. And even when the indicators are relatively complex, we think that they still represent a simpler solution compared to the application of simulation models or measurements for the same domain. Also, beyond the definition of simple and complex, the main issue is that the level of complexity and the potential to describe the system of the indicators should be chosen together with the stakeholders, according to the aim of the study, and considering the relevant agronomic and pedo-climatic context. Therefore there are no predefined categories of simple or complex indicators, but a range of possibilities that can be selected according to the study carried out and to the people participating in it. From the research side, an effort should be undertaken to develop indicators with different compromises between the level of system description (processes represented, wide bibliographic support) and simplification (data requirement, ease of interpretation).

3.6.2. Input data uncertainty

Another issue is that input data used in the calculation of indicators are uncertain; this statement applies to parameters and variables used to describe agricultural management. Public administrations can give a strong contribution to the application of indicators by ensuring the availability of good-quality digital databases at farm and field level, including alpha-numerical information and maps. On the other hand, researchers can contribute by developing indicators whose parameters can be clearly and simply calculated, or retrieved from literature. Also, they should quantify the uncertainty in the calculated values of indicators (and the corresponding variations in the ranking of the studied systems) arising from the uncertainty in input data.

3.6.3. Parameterisation

The application of indicators requires in several cases the use of site-specific parameters. Examples taken from the indicators presented in this review include: the thresholds defining “small” and “big” fields for the crop diversity indicator (Bockstaller and Girardin, 2000); the coefficients used to calculate the energy consumed for crop and animal management, and the indirect energy used in the production of inputs; the scores defining the contribution of three types of shapes to the landscape supply (Weinstoerffer and Girardin, 2000); the parameters describing the effect of each previous-successive crop combination in a rotation (Bockstaller and Girardin, 1996, 2000). It is likely that these parameters vary in different study areas; therefore new values need to be defined when the indicators are applied to new situations. Even if this can be seen as a limitation, we believe that in most cases it is a necessary step in the calculation of agro-ecological indicators, as a mean of adapting a general rule (algorithm) to a specific situation. It also should be noted that only the use of very simple indicators, constituted by the direct use of available data (e.g. amount of fuel consumed per kg of output obtained) could avoid this problem, while simulation models and other assessment tools would still require parameterisation. The degree of subjectivity can also be narrowed by selecting parameter values together with the stakeholders, to represent the system using information from all interested groups. Also, the uncertainty analysis should provide indications on the variation of the ranking of different systems generated by the variation of parameter values. If, as in the case of the crop diversity indicator (Bechini, data not published), the ranking of farms does not vary much with the variation of parameter values, parameterisation becomes of smaller importance.

3.6.4. Benchmarks

Finally, one of the most critical aspects of the application of indicators is the level chosen for the threshold benchmark. The value of the benchmarks changes of course depending on the stakeholders involved (e.g. educators, advisors, researchers, farmers, policy makers, food industry, certifying organisations, consumers, supermarkets), and on agro-pedo-climatic conditions. Existing laws or bio-physical considerations can provide useful indications for the development of benchmarks. The development of specific benchmarks for the indicators represent an important field of interaction between researchers and stakeholders.

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CHAPTER 4

AGRO-ECOLOGICAL INDICATORS OF FIELD-FARMING SYSTEMS SUSTAINABILITY. II. NUTRIENTS AND PESTICIDES

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4.1. Abstract

Evaluation of cropping and farming sustainability can be carried out with direct measurements, simulation models or indicators; the latter have the advantage of requiring a small amount of inputs, being fast to calculate and easy to interpret, allowing comparisons in space and time, and representing a synthesis of processes in complex systems. In a previous paper, we proposed a list of indicators related to the use of fossil energy and landscape and soil management. In this paper, we discuss indicators related to the use of nutrients and pesticides. We selected indicators that can be applied on a field and farm scale, based on data obtainable from the farmer and/or from existing agricultural databases; we excluded indicators based on direct measurements.

A nutrient balance is the difference between inputs and outputs of a farm or field (surplus if positive, deficit if negative). Its advantage is its simplicity, the relatively small data requirement, the identification of different inputs, and its applicability to different mineral elements. However, nutrient balances do not indicate how much surplus can actually be lost from the system and in which way. The water quality risk indicator integrates the surplus calculated at field level with simple climatic and pedological information. We also describe two nitrogen management indicators that have been proposed for arable crops and grasslands to overcome the limitations of nutrient balances, and the phosphorus management (P) indicator, which compares the applied P amount with the recommended dose, identifying the risks of spoiling non-renewable resources or depleting soil reserves.

Compared to nutrients, the use of risk indicators for pesticides is more problematic. As a matter of fact, pesticides show a greater variety of potential effects on human health and on different ecosystems; consequently, the analysis of their potential risk requires very complex and varied procedures depending on the environmental compartment considered (ground water, surface water, air and soil). This has led to the development of several pesticide risk indicators that differ greatly in terms of variables considered, field of activity, scale of analysis and methodologies utilized (interactive decision-tree, risk ratio approach,

scoring table, fuzzy system). Some indicators use simple algorithms to estimate the risk, others make use of more complicated models. The simplest and generic indicators require very few data (such as the application rate), but in general they do not consider the fate on the environment and the distribution of the chemicals. On the contrary, more complex indicators require the use of predictive models to evaluate potential exposure of non-target organisms to different active ingredients. We present some pesticide risk indicators with different levels of complexity that can be utilized at farm and field level, in order to obtain a picture of the different approaches available in literature and to point out their values and limitations.

4.2. Introduction

In order to analyse the environmental sustainability of agro-ecosystems, tools with different levels of applicability and potential explanation of the system can be used, such as: direct measurements, simulation models and agro-ecological indicators. If direct measurements and simulation models cannot be applied (due to high costs or data availability), indicators can be used as a first screening method. The term indicator has been defined as a variable that supplies information on other variables that are difficult to access (Bockstaller et al., 1997).

As part of the Italian project "Agriculture for protected areas" (Borin et al., 2005; Agripark, 2006; Bisol, 2006) we are evaluating the environmental sustainability of different cropping and farming systems. Our objective in the project is to synthesise the effects of agronomic management using quantities which: (i) allow the integration of different aspects of reality, doing a synthesis that is a good compromise between the description of the processes and their simplification into single numerical quantities; (ii) can be derived from farm characteristics, easily obtainable from the farmer and/or from existing agricultural databases (e.g. Common Agricultural Policy declarations); (iii) are easily interpreted and can be used to trigger the improvement of environmental sustainability of agricultural systems; (iv) can be calculated at the farming or cropping systems level, because these are the levels where action can be taken by farmers. We excluded the indicators constituted by direct measurements on soils, waters or crops, and the indicators that can be applied only at national or macro-regional level (e.g. OECD, 2002a; EEA, 2005).

In a previous paper (Castoldi and Bechini, 2006), we proposed a list of indicators, derived from an extensive literature review, to evaluate the sustainability of agro-ecosystems management at field and farm level, for the categories "energy", "landscape", and "soil management". In this paper we report a second set of indicators, related to the management of nutrients and pesticides, two issues which have a deep impact on soil, air and water quality.

4.3. Nutrient indicators

Nutrients are fundamental production factors in agriculture, but their inappropriate use may lead to soil and water contamination, and to a waste of the energy consumed for their production. Indicators help to identify and analyse hazardous situations by considering crop management, climate and soils. The indicators selected here range from very simple nutrient balances to more detailed indicators specifically developed for nitrogen and phosphorus.

4.3.1. Nutrient balances

Nutrient balances (Oenema et al., 2003; Öborn et al., 2003) are the simplest and most commonly applied nutrient indicators: they can be calculated with data available on different scales (from field to national) and can be used to analyse various chemical elements. To calculate a balance, the amount of nutrients leaving the system is subtracted from the amount entering the system over a defined period. If positive, the balance indicates a surplus that can be accumulated in the system or lost outside it; if negative, it indicates a deficit that can deplete the system. The most useful indicators for our context are the farm gate balance (Grignani, 1996; Simon et al., 2000) and the soil surface balance (Parris, 1998). They can be calculated using basic data that are easily available on-farm or from official agricultural statistics (e.g. fertiliser use, livestock numbers, areas and quantities of crop and forage production) and by multiplying these estimates by coefficients that convert livestock and crop production data into nutrient equivalents; the coefficients are normally derived from field level research and surveys (OECD, 2001). The two indicators differ as far as the boundaries of the system studied are concerned, because the farm gate balance does not account for the nutrients recycled within the farm (manure and crops produced and reused internally). Surpluses are normally expressed as mass of nutrients per unit area (kg ha^{-1}), but can also be related to the output unit (kg nutrient surplus kg^{-1} nutrient output: Schröder et al., 2003). Additionally, a nutrient efficiency (OECD, 2001; Schröder et al., 2003) can be calculated as an output/input ratio.

The inputs used to calculate the farm gate balance in its complete form are (the list is modified after Schröder et al., 2003): imported organic fertilisers (manure, sewage sludge, compost, etc.), imported feeds, imported animals, imported seeds, chemical fertilisers, biological fixation (only for N), atmospheric deposition, mineralisation, sedimentation. The outputs include: exported manure, exported crop products, exported livestock products, immobilisation, and erosion. All the materials recycled within the farm (crops, livestock products, manures) are not accounted for. The surplus represents: gaseous losses from stables, storage, grazing and spreading, leaching, sub-surface denitrification and stock changes (variation of the nutrient content of supplies of manure, feeds, animals, seeds and fertilisers accumulated inside the farm). This simple balance is normally calculated annually, using the farm gate boundaries (therefore considering the farm as a black box: Grignani, 1996; Simon et al., 2000; Hanegraaf and den Boer, 2003), but can also be calculated on a larger scale using the same set of inputs and outputs (as for example in the OSPARCOM method applied to nations, described by the OECD, 2001). The complete form of the farm gate balance is rarely used: for example, the OECD (2001) does not include imported seeds, biological N fixation, atmospheric deposition, mineralisation and sedimentation in the inputs, and does not mention exported manure, immobilisation and erosion in the outputs. Simon et al. (2000) do not include imported seeds, atmospheric deposition, mineralisation, and sedimentation in the inputs and do not list immobilisation and erosion in the outputs. The Dutch MINerals Accounting System (MINAS; Hanegraaf and den Boer, 2003) does not include as inputs biological fixation, atmospheric deposition, mineralisation, sedimentation and does not include as outputs immobilisation and erosion.

A general definition of the soil surface balance is "the physical difference (surplus/deficit) between nutrient inputs into, and outputs from, an agricultural system, per hectare of agricultural land" (OECD, 2001). A more specific definition states that a soil surface balance "records all nutrients that enter the soil via the surface and that leave the soil via crop uptake" (Oenema et al., 2003). Similarly to the farm gate balance, different Authors calculate the soil surface balance including different variables. Generally, the inputs are: chemical fertilisers, animal manure (net of N losses through ammonia volatilisation to the atmosphere from livestock housing and stored manure), residues remaining in the field from the previous crop, biological nitrogen fixation, atmospheric deposition, recycled organic matter (e.g. sewage sludge, compost), seeds and planting materials, sedimentation. The outputs are: crop residues removed from the field (stems, leaves, straw, roots, etc.), useful products removed from the field (grain, tubers, hay, silage, pasture, etc.), ammonia volatilisation and erosion. The surplus represents the variation of soil nutrient content (accumulation or depletion) and losses through leaching or denitrification. If a term is not included either in the inputs or in the outputs, it will be implicitly incorporated in the calculated surplus.

4.3.2. Water quality risk indicator

It is defined (OECD, 2001) as "the potential concentration of nitrate (or phosphorus) in the water flowing from a given agricultural area, both percolating water and surface run-off". For nitrogen, the indicator estimates the nutrient concentration of water lost from the soil by considering the influence of pedoclimatic condition on the N surplus. The nitrogen surplus is split in two water pools: the soil water holding capacity (WHC; L ha^{-1}) and the excess water (EW; L ha^{-1}). EW is calculated as precipitation less evapotranspiration (per crop type), using either long-term (e.g. 30 years) or annual weather data.

The indicator is the ratio between potential nitrate present (PNP; mg N ha^{-1}) and EW. PNP is calculated as $S \cdot EW / (WHC + EW)$, where S is the N surplus obtained with the soil surface balance (kg N ha^{-1}). Therefore the indicator is:

$$I = \frac{S}{WHC + EW} \tag{4.1}$$

If EW = 0, the soil profile is never saturated and movement of N into surface and ground water is unlikely. In this case, there is no risk of water pollution. If EW > 0 the nitrogen concentration in the water decreases with increasing EW.

The OECD (2001) reports sample calculations of this indicator for Canada and Denmark.

4.3.3. Nitrogen indicators

Bockstaller and Girardin (2000) and Pervanchon et al. (2005) proposed two nitrogen indicators to evaluate the impact of agricultural N management on air and groundwater quality. Compared to nutrient balances, these indicators consider crop, soil and weather and management interactions and therefore provide a more detailed description of the soil-crop system.

The calculation of the first indicator (I_N), specific for annual crops (Bockstaller and Girardin, 2000), is based on an empirical model of N losses. The reference time scale is one year, starting from the beginning of winter (fixed at December 1st). The indicator considers: (i) volatilisation losses immediately after each fertiliser application; (ii) leaching losses during crop growth; (iii) leaching losses after crop harvest (bare soil). The indicator is calculated by summing negative scores (generated by leaching or volatilisation risks) and positive scores (generated by risk reduction measures):

$$I_N = 7 + \sum_{i=1}^{n} kv_i + \sum_{i=1}^{n} kl_i + k_b + k_t \tag{4.2}$$

where kv_i and kl_i are the scores related to volatilisation and leaching after each fertiliser application (i), k_b is a score for the risk of nitrogen winter leaching (bare soil), k_t is a score that takes into account the adoption of good farming practices. Each score point corresponds to a risk increase or decrease of 30 kg N ha^{-1}. The indicator provides a value range from zero (worst value) to ten (best value), where seven is the sufficient value. The indicator is applied on field scale.

Volatilisation scores kv_i are calculated for each fertiliser application i by estimating volatilisation (V_i) as $X_i \cdot v_i$, where X_i (kg ha^{-1}) is the amount of ammonium applied, v_i is the fraction of applied N which is volatilised (depending on soil type, fertiliser type and season) (Table 4.1). Then, $kv_i = - V_i / 30$.

Table 4.1. – Volatilisation coefficients (from Bockstaller and Girardin, 2000)

	No ploughing		Ploughing (by 24 hours)	
For fertilization from 1 September to 31 March				
	Calcareous soil (>5 %)	Non calcareous soil	Calcareous soil (>5 %)	Non calcareous soil
Urea	0.10	0.07	0.05	0
Liquid solution	0.10	0.07	0.05	0
Diammonium phosphate	0.08	0.03	0	0
Ammonium sulphate	0.10	0.03	0	0
Cattle liquid manure	0.55		0.17	
Fettening pig liquid manure	0.40		0.12	
Saw liquid manure	0.45		0.14	
Poultry manure	0.30		0.14	
Turkeys manure	0.30		0.17	
Liquid muds	0.30		0.10	
For the fertilization from 1 April to 31 August				
Urea	0.20	0.14	0.10	0
Liquid solution	0.20	0.14	0.10	0
Diammonium phosphate	0.15	0.05	0	0
Ammonium sulphate	0.20	0.05	0	0
Cattle liquid manure	0.80		0.24	
Fettening pig liquid manure	0.55		0.17	
Saw liquid manure	0.65		0.20	
Poultry manure	0.45		0.14	
Turkeys manure	0.45		0.14	
Liquid muds	0.45		0.14	

Leaching (L_i) that occurs during crop growth and development depends on the amount of N applied, on the timing of N application and on soil water drainage. For each fertiliser application i, it equals

$$L_i = (X_i - V_i) \cdot t_i \cdot f \cdot b \qquad (4.3)$$

where t_i is a factor that takes into account that leaching risk decreases if fertiliser application is done close to the period of high crop N uptake rate, f is the frequency of rainy periods after fertiliser application, and b is the leaching coefficient calculated with the Burns model (for details see Bockstaller and Girardin, 2000). The factor t_i is calculated as:

$$t_i = \frac{D_i - D_{50}}{D_s - D_{50}} \qquad (4.4)$$

where D_i is the date of fertiliser application, D_{50} is the date when the crop has absorbed half of N (values available for France in Bockstaller and Girardin, 2000), D_s is the date of sowing or of the vegetative restart after winter. At D_{50} crop N uptake is so intense (and soil water drainage relatively low) that leaching is considered negligible: fertiliser applications made at or close to D_{50} have a low risk of leaching losses. The frequency of rainfall after fertiliser application (f) can be chosen depending on climatic conditions in the relevant period (0 – 1). The kl_i scores are calculated as: $kl_i = -L_i / 30$.

Excessive N fertilisation may generate accumulation of mineral nitrogen in the soil profile at crop harvest, which can be leached during the fallow period. If total N applied with fertilisers is equal to (or even less than) the recommended dose, this risk is reduced. Nitrate leaching during the fallow period (L_F) is calculated on the basis of a mass N balance related to the fallow period: $L_F = S \cdot b$, where S is N surplus in the period from crop harvest to December 1st, and b is the Burns' leaching coefficient for the fall-winter period (e.g. from October 1st to March 31). It is considered that after December 1st no or scarce mineralisation occurs. The nitrogen surplus S for the fallow period equals:

$$N_u + N_o + N_r + N_h + N_f - N_c \qquad (4.5)$$

where N_u (kg ha^{-1}) is unavoidable mineral N left in the soil at crop harvest (if the right dose is applied), N_o (kg ha^{-1}) is mineral N left in the soil at crop harvest due to over-fertilisation, N_r (kg ha^{-1}) is mineral N produced by net mineralisation of crop residues in the fall (it is equal to zero if net immobilisation occurs), N_h (kg ha^{-1}) is N produced by mineralisation of humus, N_f (kg ha^{-1}) is inorganic N applied with chemical fertilisers (or the inorganic fraction of organic fertilisers), N_c (kg ha^{-1}) is crop N uptake (if present). Tabbed values for N_u, N_r, N_h, and N_c are available for French conditions (Bockstaller and Girardin, 2000). The effect of over-fertilisation ($N_o \geq 0$) is calculated as:

$$\left\{\left(\sum_{i=1}^{n}\left(X_i - V_i - L_i\right)\right) - X_R\right\}/2 \tag{4.6}$$

where X_R (kg ha^{-1}) is the recommended amount of N to be applied with fertilisers (based on the nutrient management plan): it is assumed that only half of N excess is available for leaching, the remainder being used for crop luxury consumption and soil immobilisation. The score k_b is then calculated as – L_F / 30. If $X < X_R$ (under-fertilisation), leaching risk is reduced by the amount

$$\left\{X_R - \left(\sum_{i=1}^{n}\left(X_i - \mathrm{V}_i - L_i\right)\right)\right\}/2 \tag{4.7}$$

which is used (usually divided by 30) to assign a positive score contributing to the final value of the indicator.

Finally, if techniques are used to further reduce X_R (e.g. soil mineral N measurement in spring, crop diagnosis, use of unfertilised control plots, measurement of manure N concentration), a positive score k_t can be calculated as $0.5 \cdot (X_R - X_{RR}) / 30$, where X_{RR} is the reduced dose (lower than the recommended X_R).

In order to evaluate the risks of air and water pollution through N management on grasslands, Pervanchon et al. (2005) developed another nitrogen indicator. The indicator is equal to the lowest score of four sub-indicators that provide information about ammonia volatilisation, nitrous and nitric oxide emissions in the air, and nitrate leaching in groundwater. This is a precautionary principle, because it is not known which impact is most dangerous for the environment or for human health. Each sub-indicator is calculated by comparing the estimated loss with a threshold corresponding to the maximal acceptable emission for air and water; the threshold is expressed per unit of cultivated area. Losses to air of NH_3, N_2O and NO are calculated using dimensionless emission coefficients, representing the ratio of gaseous N losses to applied N . Nitrate leaching is estimated with a procedure similar to that used by Bockstaller and Girardin (2000) for the I_N indicator. For the details of the calculation procedure and values of parameters, the reader is addressed to Pervanchon et al. (2005).

As for the other indicators developed by the French National Institute for Agricultural Research (Institut National de la Recherche Agronomique: INRA) group, this indicator provides a value range from zero (worst value) to ten (best value), where seven is the sufficient value. This indicator is applied on field scale.

4.3.4. Phosphorus indicator

The phosphorus indicator (I_P) (Bockstaller and Girardin, 2000) evaluates the impact of phosphorus fertilisation on the chemical quality of the soil and on the economy of non-renewable resources. The indicator regards both over- and under-fertilisation as negative; in the first case, therefore, it indirectly considers the risk of pollution of ground and surface water. The indicator provides a value from zero (worst value) to ten (best value), where seven is the sufficient value. Every point represents a lack or an excess of 30 kg P_2O_5 ha^{-1}. This indicator is applied on field scale.

The indicator is calculated as:

$$I_P = 7 + \min(P_{res}, P_{sol}) + kt \quad (4.8)$$

where P_{res} is an evaluation of the misuse of non-renewable resources, P_{sol} is an evaluation of the risk of soil P depletion and kt represents the farmer’s efforts in order to improve the effectiveness of P fertilisations.

For the use of non-renewable resources, an excess of P is considered negatively: the total value of the indicator is reduced by one for an excess of 30 kg P_2O_5 ha^{-1}. Excess of organic P is a waste as for inorganic P, because organic P could have been spread instead of non-renewable inorganic P on other fields. A deficit of P is not important for

P_{res}, because it does not involve waste of non-renewable resources (P_{res} equals zero):

$$P_{res} = \begin{cases} -\left(P_a - P_r\right)/30 & if \quad P_a > P_r \\ 0 & if \quad P_a \le P_r \end{cases} \tag{4.9}$$

where P_a is the total amount of P applied to the soil (sum of P applied with all chemical and organic fertilisers; kg P_2O_5 ha^{-1}), P_r is the recommended amount of P to be applied with fertilisers (kg P_2O_5 ha^{-1}), as indicated in the nutrient management plan (P_r is calculated on the basis of the available soil P and the expected crop P uptake, estimated by multiplying the yield by the normal P concentration in the product, and by adding an estimate of P contained in crop residues).

The depletion of soil P occurs when an insufficient amount of P available to the crop is applied to the soil; excess is not relevant because P_{sol} does not evaluate P accumulation in the soil.

$$P_{sol} = \begin{cases} \left(P_{aa} - P_r\right)/30 & if \quad P_{aa} < P_r \\ 0 & if \quad P_{aa} \ge P_r \end{cases} \tag{4.10}$$

where P_{aa} is part of P_a which is available to the crop (kg P_2O_5 ha^{-1}). P_{aa} is calculated by summing the entire amount of P applied with organic fertilisers and P applied using the recommended forms of inorganic P

Table 4.2. – Recommended (+) and non-recommended (–) P applications. Information used in the calculation of the phosphorus indicator I_P (reproduced from Bockstaller and Girardin, 2000)

Form of phosphate	pH≤6.2	6.2<pH≤7.2	pH>7.2	
			Lime ≤ 10%	Lime > 10%
Moderate to high soil phosphate fixing capacity				
Water- or citrate-soluble phosphate	+	+	+	+
Dicalcium phosphate	+	+	+	
Basic slag	+	+	+	–
Al-Ca phosphate	–	+	+	–
Natural phosphate	+ [1]	–	–	–
Very high soil phosphate fixing capacity				
All forms	+	+	+	+

[1] on sandy soils this form is non-recommended.

fertilisers (Table 4.2); non-recommended P inorganic fertilisers do not contribute to P_{aa}.

With a qualitative procedure, it is also possible to take into account farmers' efforts to improve P management: if they use one or more methods to improve P management (fertiliser localisation, measurement of soil phosphate fixing capacity, and/or soil analyses carried out in the last five years), and if $|P_a - P_r| \leq 15$ kg P_2O_5 ha^{-1}, the kt parameter is set at +1 (or +2, +3 if two or three conditions are met), thus increasing the value of the indicator.

4.4. Pesticide indicators

In the last years, several pesticide risk indicators have been developed and applied in different EU countries (Levitan et al., 1995; Hart, 1997; Levitan, 2000; Finizio et al., 2001) aiming at different goals and using different methods. For instance, some indicators evaluate the risk for surface or groundwater systems, others evaluate the risk for terrestrial ecosystems, or for workers, bystanders and consumers. Therefore, they differ greatly in the methodologies and variables considered. Some indicators are interactive (decision trees), while others are based on the risk ratio approach or scoring tables. In some cases also the fuzzy approach has been proposed (van der Werf and Zimmer, 1998; Levitan, 2000; Roussel et al., 2000). In this paper, we selected some pesticide risk indicators with different levels of complexity that can be utilized at farm level even if, in most cases, their original purposes were different (risk classification, risk trend, etc.).

4.4.1. Simple and generic pesticide risk indicators

A simple set of pesticide risk indicators has been proposed by Vereijken (1995), considering the exposure of the environment to pesticides in order to prevent short-term and long-term adverse effects on ecosystems. Three Environmental Exposure-based Pesticide (EEP) indicators in three different environmental compartments (air, soil and groundwater) are calculated using some physical-chemical properties of each active ingredient (a.i.):

$$EEP_{air} = AR_{a.i.} \cdot VP \tag{4.11}$$

$$EEP_{soil} = AR_{a.i.} \cdot DT_{50soil} \tag{4.12}$$

$$EEP_{groundwater} = EEP_{soil} \cdot K_{om}^{-1} \tag{4.13}$$

where $AR_{a.i.}$ is the application rate (kg a.i. ha^{-1}), VP is the vapour pressure at 25°C (Pa), DT_{50soil} is the half life of the chemical in soil (days), K_{om} is the partition coefficient of the pesticide between organic matter and water fractions of the soil.

More recently, the OECD (2005) proposed the Load Index (LI) to evaluate the toxicological effect of an a.i.:

$$LI = \sum_{k=1}^{n} \frac{AR_{a.i.k}}{TOX_k} \tag{4.14}$$

where n is the number of a.i. applied in one year, and TOX_k is the acute or long-term Lethal Dose (LD_{50}, dose required to kill 50% of test organisms) or Lethal Concentration (LC_{50}, concentration required to kill 50% of test organisms) of k^{th} a.i.. This indicator is calculated separately for mammals, birds, earthworms, bees, fish, crustaceans and algae, using a value (average, minimum or maximum) for each a.i..

4.4.2. Eco-rating

Lewis and colleagues (1997a; 1997b) proposed a decision support system (Eco-rating) designed to enhance environmental sustainability at farm level. The system uses an integrated approach to assess all aspects of farming practices individually (modules), such as fertilisation, pesticide use, energy, water efficiency, farmland conservation and livestock management. The value of each module ranges from a positive to a negative score. A positive value reflects an environmental gain; while a negative value is a loss. The zero value should be interpreted as a threshold of sustainability of the farm. Specific areas of the farm where a potential environmental problem exists are highlighted by a greatly negative score.

The eco-rating acts as an expert-system, considering that pesticides can affect different environmental receptors. The scores are calculated with rules derived from best agricultural practices, pesticide regulations and other influencing factors. The eco-rating for pesticides is divided into three different modules: (i) assessment of field applications related to product formulation, label precautions and physical-chemical parameters, (ii) management techniques that consider the method of application, storage, waste disposal and (iii) non-crop use of pesticides such as biocides, sheep dips and rodenticides.

For the purposes of this review the pesticide eco-rating for field assessment (P_f) is more relevant. It can be calculated as:

Table 4.3. – Example of label precautions and assigned scores (L_{SER}) (reproduced from Lewis et al., 1997b)

Label number	Description assigned	Score
1	Product contains an anticholinesterase organophosphate compound	–10
2	Product contains anticholinesterase carbamate compound	–10
3 (a, b, c)	Very toxic	–10
5 (a, b, c)	Harmful	–3
6 (a, b, c)	Irritant	–2
12c	Flammable	–1
36	Keep away from food, drink and animal feedstuffs	–1
37	Keep out of reach of children	–2
43	Keep livestock out of treated areas	–2
45	Dangerous to game, wild birds and animals	–7
46	Harmful to game, wild birds and animals	–5
47	Harmful to animals	–5
48/a	Extremely dangerous to bees	–10
49	Dangerous to bees	–7
50	Harmful to bees	–5
51	Extremely dangerous to fish	–10
52	Dangerous to fish	–7
53	Harmful to fish	–5
54	Do not contaminate ponds, waterways or ditches/Harmful to fish or other aquatic life	–3
58–71	Storage and disposal warnings, score per warning	–1
78	If you feel unwell seek medical advice	–2

$$P_f = f(L_{SER}) + \sum_{k=1}^{n} f(E_k \cdot AR_k) \tag{4.15}$$

which considers label precaution and the ratio between toxicity and exposure.

The function *f* (L_{SER}) is the eco-rating score derived from label precautions (L) (Table 4.3). The majority of these can be associated with one or more specific SER (Sensitive Environmental Receptors), e.g. toxicity of the pesticide to bees, aquatic systems, birds and humans. According to label information, scores (from 0 to – 5) are assigned to each label precaution for each SER (L_{SER}). Furthermore, each score is multiplied by a penalty factor (Fp) chosen according to local-site characteristics. For example, the use of a pesticide with water risk label (score = –5) can be rated with: i) a penalty factor equal to zero if the product is applied far from water bodies (>10 m) (the final score L will be zero); ii) a penalty factor ranging from 0.2 to 1.0 (final score from –1 to –5) when unsprayed margins or buffer zones separate the target zone (field) from any water body; a penalty factor of 0.2 will be given if the distance between the field and the water course is 6 – 10 m, whereas it increases to 1.0 with a decrease in the distance from 6 to 0 m; iii) a penalty factor of 2, if no margins or buffer zones exist and the list of precautions includes the statement, “Extremely dangerous to fish”. The total *f* (L_{SER}) will be obtained by summing all the scores related to relevant SER reported into the label information.

The second term of the equation is obtained by considering some parameters related to the potential environmental distribution of the a.i. and consequently to the potential exposure of organisms in different environmental compartments. The quantity E_k depends on the physical-chemical properties of the a.i. and is the sum of the scores related to potential volatilisation, leaching and bioaccumulation (S_{air} + S_{leach} + S_{bio}) of the a.i.. The subscript k ranges from 1 to n, where n is the number of active ingredients in the product formulation. The loss of pesticide to atmosphere is based upon VP at 20 °C, assuming a loamy soil with a pH of 7. This value is classified and scored (S_{air}) (Table 4.4).

Table 4.4. – Vapour pressure and corresponding volatilisation risk score (S_{air}) (reproduced from Lewis et al., 1997a)

Vapour Pressure (mPa)	$< 10^{-8}$	10^{-8} to 10^{-6}	10^{-6} to 10^{-4}	10^{-4} to 10^{-2}	$> 10^{-2}$
S_{air}	0	–2	–5	–10	–20

The S_{leach} value is obtained using the GUS (Groundwater Ubiquity Score: Gustafson, 1989):

$$GUS = \log DT_{50} \cdot (4 - \log K_{oc}) \tag{4.16}$$

This index is based on the consideration that the potential leaching of an a.i. and consequently its relative risk of contamination to groundwater, depends on its persistence in the soil (measured as the soil half-life, DT_{50}, days) and the soil adsorption capacity expressed with K_{oc} ($m^3\ kg^{-1}$), the sorption coefficient of a.i. to organic carbon. A GUS value below 1.8 represents compounds that do not leach, whereas compounds with a GUS value above 2.8 are potential leachers, and for those between 1.8 and 2.8 the risk will depend on other factors such as soil type and environmental sensitivity. Scores (S_{leach}) range from –10 for potential leachers to 0 for non-leachers.

The score related to potential bioaccumulation (S_{bio}) is obtained by considering the logarithm of the n-octanol/water partition coefficient (log K_{ow} or log P), which is a measure of the distribution of a substance between a lipophilic phase (the n-octanol) and the aqueous phase of the test system, representing potential bioaccumulation of a compound in fatty tissues of animals. Scores (S_{bio}) range from 0 for log P values less than 2.7, –5 for mid-range values and –10 for log P values greater than 3.

The eco-rating (P_f) is determined for each individual pesticide applied to the field. The values calculated for different applications are averaged at field level; field values are then weighted by field size and the arithmetic mean represents the farm value. Field- and farm-average values are then normalized to lie on the scale range –100 to 0 to obtain the final eco-rating. This normalization process simply multiplies the average values by 100 and divides the result by the minimum theoretical eco-rating. Generally, an eco-rating less than –40 can be associated with good practices. Eco-ratings in the range of –40 to –60 may not necessarily represent unapproved applications, but may indicate that an alternative chemical or an adjustment in practices may be environmentally beneficial. Eco-ratings below –70 usually reflect poor practices, an undesirable operation or an illegal application.

4.4.3. p-EMA

An evolution of the eco-rating approach (p-EMA: pesticide-Environmental Management for Agriculture) has recently been proposed to support farmers in optimising the use of agricultural pesticides by means of a computer-based decision support tool (Brown et al., 2003; Hart et al., 2003; Lewis et al., 2003). The overall aim of p-EMA is to support the selection of pesticides that are likely to pose the least risk to the environment within the context of local site conditions and farm practices. The system estimates risks to a wide range of taxonomic groups and environmental compartments using methods consistent with current regulatory assessments, but it also allows adjustments for the local conditions and environmental costs and benefits of varying management practices in the formulation. The methodology requires conventional estimates of exposure, combined with the toxicological properties of the pesticide in the form of toxicity/exposure ratios. It uses simple equations of pesticide dispersion pathways in the local environment to estimate the predicted environmental concentration in the treated field and in the surrounding area, surface water, groundwater and other media to which various organisms (operators, mammals, birds, aquatic organisms, bees, earthworms and non-target arthropods) will be exposed. Concentrations in groundwater are calculated on the basis of a meta-version of the MACRO model linked to environmental and pesticide databases. MACRO is a physically-based one-dimensional, numerical model of water flow and reactive solute transport in field soils. It simulates preferential flow by dividing total soil porosity into two flow domains (macropores and micropores), each characterised by a flow rate and solute concentration (Larsbo and Jarvis, 2003). As this model is complex and cannot be easily adopted in the framework of agro-ecological indicators, a meta version was developed by generating a series of look-up tables (Brown et al., 2003) used in the p-EMA calculation.

Surface water concentrations are taken as the maximum of those arising from inputs via spray drift and drainflow (where installed). Data confidence is determined using a scoring regime considering the data source and the proportion of missing information. Software is available to farmers, advisers and agronomists.

4.4.4. Norwegian indicator (NI)

The Norwegian indicator (Spikkerud, 2000; OECD, 2002b) is an additive scoring system that assigns scores to Toxicity/Exposure ratios (TERs) for earthworms and birds and to Hazard Quotient (HQ) for bees. TERs and HQs are two tools currently utilized for the characterization of the risk of pesticides and are indicated in Council Directive 91/414/EEC for marketing new a.i.. A TER greater than one indicates that the exposure is lower than toxicity, and consequently there is no risk for non-target organisms. High values of HQ indicate risk. Besides non-target organisms of terrestrial ecosystems, the indicator also takes into account the general environmental load by giving scores to persistence and potential bioaccumulation. The exposure for each a.i. is calculated as the PEC (Predicted Environmental Concentration for earthworms) or PIEC (Predicted Initial Environmental Concentration for birds) using standardized models (Hoerger and Kenaga, 1972; FOCUS, 1997a, 1997b; Council Directive 91/414/EEC).

The equation to calculate the indicator is:

$$NI = \sum_{k=1}^{n} \left[(S_E + S_{Bi} + S_{Be} + S_P + S_B)^3 \cdot A_k \right] \tag{4.17}$$

where n is the number of different a.i. applied in one year; S_E is the score attributed to TER for earthworms (TER_{ew}); S_{Bi} is the score attributed to TER for birds (TER_{Bi}); S_{Be} is the score attributed to HQ for bees; S_P is the score attributed to persistence; S_B is the score attributed to bioaccumulation; A_k = area treated (ha) with the k^{th} active ingredient.

Each score ranges from 0 to 4. Because of the compression of these generated results the final value is cubed.

Table 4.5a. – Scores for the Toxicity/Exposure Ratio related to earthworms (TER_{ew}), used in the calculation of the Norwegian Indicator (NI)

TER_{ew}	S_E
> 100	0
10 – 100	2
< 10	4

Table 4.5b. – Scores for the Toxicity/Exposure Ratio related to birds (TER_{Bi}), used in the calculation of the Norwegian Indicator (NI)

TER_{Bi}	S_{Bi}
> 10	0
1 – 10	2
< 1	4

Tab. 5c. – Scores for the Hazard Quotient related to bees (HQ_{Be}), used in the calculation of the Norwegian Indicator (NI) (HQ_o = Hazard Quotient oral; HQ_c = Hazard Quotient contact)

HQ_o and HQ_c	S_{Be}
< 50	0
50 – 100	1
100 – 1,000	2
1,000 – 10,000	3
>10,000	4

The score S_E (Table 4.5a) depends on the ratio between $LC_{50\text{-}14days}$ (14 days Lethal Concentration for earthworms of each a.i.; mg a.i. kg^{-1} soil) and PEC_{acute_soil} (mg a.i. kg^{-1} soil).

$$PEC_{acute\_soil} = AR_{a.i.} 10^2 \frac{1 - f_{int}}{D_e \cdot BD} \quad (4.18)$$

where 10^2 is a conversion factor to transform kg ha^{-1} into mg m^{-2}, f_{int} (unitless) is the fractional interception by crop canopy (default = 0 for bare soil, up to 0.5 when a crop is present), D_e (m) is the soil mixing depth (e.g. 0.05 m depth for surface application, 0.20 m for incorporation) and BD is the bulk density of dry soil (kg m^{-3}; default = 1,500 kg m^{-3}).

The score S_{Bi} (Table 4.5b) is based on TER_{bird}, calculated from PIEC (mg kg^{-1}) and dietary toxicity (dietary $LC_{50\text{-}14days}$, mg kg^{-1} food):

$$TER_{bird} = \frac{LC_{50-14days}}{PIEC} \quad (4.19)$$

For worst case assumption, it is suggested to calculate the PIEC for leaves and small insects as: PIEC = $AR_{a.i.} \cdot 30$. The constant 30 is used because a series of research reports have shown that with an application rate of 1 kg a.i. ha^{-1} the concentration of residues on leaves is approximately 30 mg kg^{-1}. If dietary toxicity data are unavailable, it is suggested to use the acute oral LD_{50} values (mg kg^{-1} body weight) in the calculation of TERs. However, in this case the quantity of contaminated food ingested by birds must be taken into account. European Crop Protection Association (1995) proposed a method in which it is assumed that small birds (weight = 10 g) have a daily food intake of approximately 30% of their body weight, while large birds (weight 100 g or more) have an intake of approximately 10%. Assuming that birds ingest only contaminated food and incorporating such parameters into the previous equation, the daily intake (mg kg body weight^{-1}) for the two categories of birds are: small birds: $AR_{a.i.} \cdot 9$, large birds: $AR_{a.i.} \cdot 3$.

TER values can then be calculated by dividing the toxicity given by acute oral toxicity studies (LD_{50}) by the daily intake. A weakness of this method is that it equates LD_{50} values from research on individual exposures with very crude estimates of exposures that can occur daily over a long period.

The score for bees (S_{Be}) is calculated considering the HQ. According to the EU Uniform Principles (Council Directive 91/414/EEC), the HQ approach is generally utilized to evaluate the risk for bees due to pesticide exposure. Table 4.5c reports the scores (S_{Be}) assigned by the NI to HQ for oral exposure ($HQ_o = AR_{a.i.} / LD_{50\_oral}$) or contact exposure ($HQ_c = 10^3\ AR_{a.i.} / LD_{50\_contact}$) of bees, where 10^3 is a conversion factor to transform the $AR_{a.i.}$ into g ha^{-1} and LD_{50} (µg bee^{-1}) is the median Lethal Dose for bees. The highest of the HQs is used to assign the S_{Be}.

Finally Table 4.5d and 4.5e illustrate the scores for persistence (S_P) and potential bioaccumulation (S_B). The first depends on half-life in soil (DT_{50}) and on the application rate. The second is obtained very roughly, by considering both the log K_{ow} of the chemical and its persistence in soil.

Table 4.5d. – Persistence score (S_P) for pesticide in Norwegian Indicator (NI); DT_{50} is half-life of a.i. in soil

DT_{50} (days)	Pesticide Application Rate (kg a.i. ha^{-1})			
	< 0.1	0.1 – 1	1 – 2	> 2
< 10	0.0	0.0	0.0	0.0
10 – 30	0.0	0.0	0.5	1.0
30 – 60	0.5	1.0	1.5	2.0
60 – 200	1.5	2.0	2.5	3.0
200 – 365	2.5	3.0	3.5	4.0
> 365	4.0	4.0	4.0	4.0

Table 4.5e. – Bioaccumulation score (S_B) for pesticide in Norwegian Indicator (NI)

Persistence in soil, DT_{50}	Log Kow < 3	3 $\leq$ Log Kow $\leq$ 4	Log Kow > 4
< 1 day	0	0	0
1 – 10 days	0	0.5	1
10 – 60 days	0	1	2
60 – 200 days	0	1.5	3
> 200 days	0	2	4

4.4.5. Surface Water Indicator for Pesticides (SWIPE)

Very recently, the SWIPE (Surface Water Indicator for Pesticides) indicator has been proposed (Cassarà et al., 2005; Cassarà et al., 2006) as a tool to help different stakeholders (farmers, agronomists, policy makers) in reaching the goal of sustainable agriculture. The SWIPE indicator operates on different scales, from farm-level to national-level, giving information on pesticide risk for surface water systems. At farm level, this indicator can be utilized to rank pesticides and to identify highest-risk areas. In this way farmers can select pesticides with less environmental impact for surface water systems and choose the most appropriate risk mitigation practices to be applied on critical areas.

SWIPE can be classified as a scoring indicator. The scores are assigned on the basis of ETR values (Exposure/Toxicity Ratio) obtained for selected non-target organisms (NTO) chosen as representative of the aquatic ecosystems (algae, *Daphnia*, fish). Toxicity data are referred to the acute effects on the NTO (EC_{50}: concentration where 50% of its maximal effect is observed, or LC_{50}). The exposure (PEC_{H_2O}: Predicted Environmental Concentration in surface water; mg l^{-1}) is calculated after each pesticide treatment on the basis of $AR_{a.i.}$, of the percentages of pesticides lost by means of drift and runoff processes (L_{drift} and L_{runoff}) and the depth (D_e; m) of the receiving water body:

$$PEC_{H_2O} = \frac{AR_{a.i.} \cdot L}{D_e} \qquad (4.20)$$

with $L = L_{drift} + L_{runoff}$. L_{drift} is calculated according to the Ganzelmeyer tables (Biologische Bundesanstalt, 2000) (Table 4.6), whereas L_{runoff} is calculated using the following equation:

$$L_{runoff} = \frac{Q \cdot f \cdot e^{\frac{3 \cdot \ln 2}{DT_{50soil}}} \cdot 100}{R_a \cdot (1 + K_d)} \tag{4.21}$$

where Q (mm) is the runoff amount calculated according to Lutz (1984) and Maniak (1992), R_a (mm) is the amount of precipitation, K_d is the distribution coefficient:

$$K_d = K_{oc} \cdot f_{OC} \tag{4.22}$$

Table 4.6. – L_{drift} (%) calculated according to Ganzelmeyer tables (from Biologische Bundesanstalt, 2000)

Crop / Technique	Dist. to water (m)	Number of applications per season							
		1	2	3	4	5	6	7	>7
Cereals, grass / alfalfa, legumes, oil-seed rape, potatoes, sugar beet, sunflower, tobacco, vegetables, cotton	1	2.8	2.4	2.0	1.9	1.8	1.6	1.6	1.5
Hops	3	19.3	17.7	15.9	15.4	15.1	14.9	14.6	13.5
Citrus, olives	3	15.7	12.1	11.0	10.1	9.7	9.2	9.1	8.7
Vines (early application)	3	2.7	2.5	2.5	2.5	2.4	2.3	2.3	2.3
Vines (late application)	3	8	7.1	6.9	6.6	6.6	6.4	6.2	6.2
Pome /stone fruit (early application)	3	29.2	25.5	24.0	23.6	23.1	22.8	22.7	22.2
Pome /stone fruit (late application)	3	15.7	12.1	11.0	10.1	9.7	9.2	9.1	8.7
Hand application (crop height < 50 cm)	1	2.8	2.4	2.0	1.9	1.8	1.6	1.6	1.5
Hand application (crop height > 50 cm)	3	8.0	7.1	6.9	6.6	6.6	6.4	6.2	6.2
No drift (incorporation, granular or seed treatment)	1	0	0	0	0	0	0	0	0

where f_{OC} (%) is soil organic carbon concentration, and f is a correction factor:

$$f = f_1 \cdot f_2 \cdot f_3 \qquad (4.23)$$

where f_1 depends on the slope, f_2 depends on plant interception, and f_3 depends on the presence of a buffer zone (OECD, 1999b). If the slope is < 20%, then

$$f_1 = 0.02153 \cdot slope + 0.001423 \cdot slope^2 \qquad (4.24)$$

or else $f_1 = 1$.

$$f_2 = 1 - f_{int} \qquad (4.25)$$

$$f_3 = 0.83^{wbz}/100 \qquad (4.26)$$

where wbz (m) is the width of water buffer zone. On the basis of the PEC/Toxicity ratios, obtained for the three NTO considered, scores and weights are assigned to each a.i. considered (Table 4.7).

Finally, the indicator is calculated according to the following equation:

$$SWIPE = (A + D + F) \cdot P1 \cdot P2 \cdot P3 \cdot WI \cdot T \qquad (4.27)$$

where A, D, and F are the scores assigned to the three NTOs considered multiplied with their weights, P1 is the number of treatments, P2 is the percentage used respect to the recommended dose applied (% of the rate of application), P3 is the correction coefficient in case antidrift instruments are used during application (1.0 no antidrift used, 0.3 antidrift used), WI (unitless) is the Water Index, i.e. the probability of having water surrounding the treated field, T (ha) is the treated area;

Table 4.7. – Scores and weights assigned to each a.i. on the basis of the PEC/TER ratio obtained for the three non-target organisms (from Cassarà et al., 2006)

PEC/EC_{50} (TER)	**Algae (A)**	***Daphnia* (D)**	**Fish (F)**
	Score		
≥1	8	8	8
1 – 0.1	6	6	6
0.1 – 0.01	4	4	4
0.01 – 0.001	2	2	2
<0.001	1	1	1
Weight	3.0	4.0	5.5

$$WI = l_w / (l_{nw} + l_w) \quad (4.28)$$

where l_w (m) is the length of field-water boundary and $l_{nw} + l_w$ (m) is the length of the total field boundary (l_{nw} field no-water boundary).

4.4.6. Environmental Potential Risk Indicator for Pesticides (EPRIP)

According to Padovani and colleagues (2004) the main objective of the EPRIP indicator is to assess the potential environmental risk from pesticide use at farm scale. The index was created to be incorporated in the decision support system Sustainable Supply Agriculture Production (SuSAP) used in the Lombardy region (Italy). The indicator is based upon the Exposure/Toxicity Ratio (ETR) where the exposure (PEC) is estimated on a local scale and toxicity is referred to short term toxicological parameters (acute toxicity). The indicator considers different environmental compartments (surface and ground water, soil and air). Eco-toxicological effects on aquatic organisms (fish algae and crustaceans) and soil organisms (earthworms) are considered with the following procedure: (i) PECs are estimated for different environmental compartments; (ii) one ETR value (PEC/toxicity) is determined for groundwater, soil and air, and six values for surface water resulting from each combination of PEC (drift and runoff) and toxicity (acute toxicity to algae, *Daphnia* and fish); (iii) ETR values are normalised into risk scores (RS) from 1 to 5 (1 if ETR < 0.01, 2 if 0.01 < ETR < 0.10, 3 if 0.10 < ETR < 1.00, 4 if 1.00 < ETR < 10.00 and 5 if ETR > 10.00); (iv) an overall score (EPRIP) is obtained by multiplying the RS obtained in all compartments according to this equation:

$$EPRIP = RS_{groundwater} \cdot RS_{surface\_water} \cdot RS_{soil} \cdot RS_{air} + 25\ N_4 + 50\ N_5 \quad (4.29)$$

where $RS_{surface\_water}$ is the highest score among those obtained for surface water, N_4 is the number of RS values equal to 4 and N_5 is the number of RS values equal to 5. The weighting factors N_4 and N_5 are introduced to magnify higher risk scores; (v) finally, the EPRIP values are translated into six classes of environmental potential risk (no risk if EPRIP = 1, negligible if 2 < EPRIP < 16, low if 17 < EPRIP < 81, intermediate if 82 < EPRIP<256, high if 257 < EPRIP < 400, very high if EPRIP > 400).

The PECs for each compartment are calculated with four models.

$$\mathrm{PEC}_{\mathrm{groundwater}}\,(\mu\mathrm{g\ l}^{-1}) = 2.379 \cdot AF \cdot \frac{AR}{10} \cdot \frac{(1-f_{\mathrm{int}})}{P} \qquad (4.30)$$

Table 4.8. – Crop interception factor for different crops (f_{int}) (from Padovani et al., 2004)

Crop	Interception fraction (–)
Bare soil/pre-emergence	0.00
Green	0.44
Potatoes height <50 cm	0.22
Potatoes height >50 cm	0.89
Orchards early treatment	0.44
Orchards late treatment	0.78
Cereals height <50 cm	0.11
Cereals height >50 cm	0.89

where 2.739 is a conversion factor to transform mg into µg and days into years, f_{int} (unitless) is crop-interception (Table 4.8), P is soil porosity (1 – BD / PD), with PD = soil particle density, equal to 2,650 kg m^{-3} and AF is an attenuation factor:

$$AF = e^{-0.693 \cdot \frac{TR}{DT_{50\,soil}}} \qquad (4.31)$$

where TR is the average residence time:

$$TR = \frac{L \cdot RF \cdot FC}{Q} \qquad (4.32)$$

where

$$RF = 1 + \frac{BD \cdot \mathrm{f}_{OC} \cdot K_{oc}}{FC} + \frac{AC + K_h}{FC} \qquad (4.33)$$

where L (m) is groundwater depth, RF is the retardation factor, Q (m yr^{-1}) is net recharge of groundwater (which depends on rainfall and evapotranspiration), FC (m^3 water m^{-3} soil) is the volumetric soil water content at field capacity, AC is the soil air content (FC – P; m^3 air m^{-3} soil, with P = soil porosity), K_h is Henry's constant.

$$PEC_{acute\_soil} = AR_{a.i.} \cdot 10^2 \, \frac{(1-f_{\mathrm{int}})}{D_e \cdot BD} \qquad (4.34)$$

where 10^2 value is used to convert $AR_{a.i.}$ from kg ha^{-1} to mg m^{-2}. PEC_{acute_soil} for multiple applications is calculated using a simplifying worst-case assumption from the initial PEC_{acute_soil} for one application:

$$PEC_n = PEC_{acute\_soil} \frac{1-e^{-nki}}{1-e^{-ki}} \tag{4.35}$$

where n is the number of applications, i is the number of days between them and k is the dissipation rate constant of the pesticide: k = ln (2 / DT_{50_soil}).

PEC_{air} is estimated as:

$$C_{air} = \frac{J_0}{V_f} \tag{4.36}$$

where C_{air} (g m^{-3}) is the concentration in the air at a height of 1.5 m, V_f is the dilution velocity (324.55 m h^{-1}), and J_o (g m^{-2} h^{-1}) is the boundary-layer flux:

$$J_0 = \frac{D_a \cdot C_{sa}}{d} \tag{4.37}$$

where d is the boundary layer thickness (0.005 m), D_a (m^2 h^{-1}) is the diffusion coefficient in free air and C_{sa} (kg m^{-3}) is the pesticide concentration in the soil atmosphere:

$$D_a = 0.036\ (76 / MW)^{1/2} \tag{4.38}$$

where MW is the molecular weight of a.i.;

$$C_{sa} = \frac{PEC_{acute\_soil} \cdot BD \cdot P_a}{FC \cdot P} \tag{4.39}$$

where P_a (unitless) is the mass fraction of the a.i. in the soil atmosphere:

$$P_a = \frac{Z_a \cdot V_a}{Z_a \cdot V_a + Z_w \cdot V_w + Z_s \cdot V_s} \tag{4.40}$$

where V_a, V_w and V_s are the volume fraction of air, water and soil respectively, and Z_a, Z_w and Z_s are the fugacities in the three compartments:

$$Z_a = \frac{1}{R \cdot T} \tag{4.41}$$

$$Z_w = \frac{S}{V_p} \tag{4.42}$$

$$Z_a = \frac{K_{oc} \cdot BD \cdot Z_w}{1 - P} \tag{4.43}$$

where R (J K^{-1} mol^{-1}) is the gas constant, T (K) is the temperature, and S (mol m^{-3}) is the water solubility of the pesticide.

The PEC for surface water accounts for drift and runoff:

$$PEC_{drift} = \frac{AR_{a.i.} \cdot f_{drift}}{10 \cdot V} \tag{4.44}$$

where 10 is used to convert $AR_{a.i.}$ from kg ha^{-1} to g m^{-2}, f_{drift} is the a.i. drift percentage for different crops (Table 4.9), V is the volume of water in the ditch (per unit of length).

$$V = \frac{h \cdot (b + h)}{b + 2h} \tag{4.45}$$

where h (m) is ditch depth and b (m) is the ditch bottom; the slope is assumed to be 45°. The ditch is assumed to be near the field; the drift for aerial applications is not considered.

$$PEC_{runoff} = \frac{P_r \cdot AR_{3d} \cdot F_{aq}}{D_r} \tag{4.46}$$

where, D_r (mm) is the run-off depth ($D_r = 0.47 \cdot R_{max} - 10$), P_r (%) is the fraction of pesticide lost by runoff:

$$P_r = F_{st} \cdot F_s \cdot F_r \cdot (0.55 \cdot \log K_{oc} + 1.47) \tag{4.47}$$

where F_{st} is the soil type factor (if sand content > 85%, F_{st} = 0.01, if sand = 45 – 85%, F_{st} = 0.5, and if sand < 45%, F_{st} = 1), F_s is the slope factor:

$$F_S = 0.124 \cdot Slope + 0.0082 \cdot Slope^2 \tag{4.48}$$

Table 4.9. – Drift percentage (f_{drift}) for different crops used in PEC_{drift} calculation (from Padovani et al., 2004)

Water distance (m)	Vineyard Early	Vineyard Late	Orchards Early	Orchards Late	Vegetables <50 cm	Vegetables >50 cm	Field
1					4.0		4.0
2					1.6		1.6
3	4.9	7.5	29.6	15.5	1.0	7.5	1.0
4					0.9		0.9
5	1.6	5.0	20.0	10.0	0.6	5.0	0.6
10	0.4	1.5	11.0	4.5	0.4	1.5	0.4
15	0.2	0.8	6.0	2.5	0.2	0.8	0.2
20	0.1	0.4	4.0	1.5	0.1	0.4	0.1
30	0.1	0.2	2.0	0.6	0.1	0.2	0.1
40	0.1	0.2	0.4	0.4		0.2	
50	0.1	0.2	0.2	0.2		0.2	

(the runoff in flat regions is assumed to be zero); F_r is the rainfall factor:

$$F_r = 0.028 \cdot RE + 0.00011 \cdot RE^2 \qquad (4.49)$$

where RE (mm) is rain in excess (RE = R_{max} – 17), and R_{max} (mm) is the maximum daily rainfall average. AR_{3d} is the quantity of applied pesticide remaining on the soil after 3 days (mg kg^{-1}).

$$AR_{3d} = PEC_{soil} \cdot \frac{1 - e^{-k3}}{k3} \qquad (4.50)$$

where k is the dissipation rate constant of the pesticide). F_{aq} is the fraction of pesticide dissolved in runoff water:

$$F_{aq} = 1 / (1 + Q) \qquad (4.51)$$

$$Q = \frac{2 \cdot K_{oc} \cdot \mathrm{f}_{OC}}{100 \cdot R_{off}} \qquad (4.52)$$

where R_{off} is the quantity of water lost by runoff. In herbaceous crops

$$R_{off} = 3.83 - (0.12\ Rf + 0.00056\ Rf^2) \qquad (4.53)$$

and in orchard crops

$$R_{off} = -152.4 + (0.4\ Rf) \qquad (4.54)$$

where Rf (mm) is the annual rainfall.

4.4.7. Environmental Yardstick for Pesticides (EYP)

A holistic approach was proposed by Reus and Leendertse (2000). The Environmental Yardstick for Pesticides (EYP) indicator considers three environmental compartments (groundwater, surface water and soil), producing three output values: (i) risk of groundwater contamination, (ii) acute risk to water organisms (most sensitive organisms), (iii) acute and chronic risk to soil organisms. The score on the yardstick depends on chemical properties (persistence and mobility in soil, toxicity) of both active ingredient(s) and principal metabolites, application rate, organic matter content of the soil (which influences transportation in soil), time of application (which influences degradation and transportation in soil), method of application and distance to surface water (which influence the emissions to surface water). The potential risk is expressed in environmental impact points (EIPs). The EIPs are based on PEC, and on maximum permissible concentration (MPC) set by the Dutch government for that specific compartment:

$$EIP_s = \frac{PEC}{MPC} \cdot AR_{a.i.} \cdot 100 \qquad (4.55)$$

The EIPs are initially assigned for a standard application of 1 kg active ingredient per hectare. For different rates of application, the number of EIPs is multiplied by the actual dose ($AR_{a.i.}$). A score greater than 100 EIPs indicates an unacceptable environmental impact.

$$EIP_{groundwater} = \frac{PEC_{groundwater}}{0.1} \cdot 100 \qquad (4.56)$$

where 0.1 μg l^{-1} value is the Dutch drinking water standard: any chemical that is predicted to exceed this threshold will produce a score greater than 100 EIPs. In order to calculate the $PEC_{groundwater}$ use of the PEARL leaching simulation model is suggested (Tiktak et al., 2000; PEARL, 2006).

$$EIP_{surface\_water} = \frac{PEC_{surface\_water}}{0.01 \cdot LC_{50water\_organisms}} \cdot 100 \qquad (4.57)$$

where $PEC_{surface\_water}$ (mg l^{-1}) is the predicted environmental concentration in surface water, 0.01 is used to express the Dutch environmental standard (1/100 of $LC_{50water\_organisms}$ (mg l^{-1}), the most sensitive LC_{50} among fish, *Daphnia* and algae),

$$PEC_{surface\_water} = 0.1 \cdot AR_{a.i.} \frac{D_r}{D_e} \qquad (4.58)$$

where 0.1 converts $AR_{a.i.}$ from kg ha^{-1} to mg l^{-1}, D_r is the drift percentage of pesticide to surface water, D_e (m) is the ditch depth. When the specific ditch depth is not available, a default value of 0.25 m is used. D_r depends on factors such as distance to the ditch, type of spraying nozzle, spraying pressure, wind speed and other application variables. Drift percentage is assumed to range from 0% for pesticides applied as seed treatments or as granules, to 0.5% for pesticides sprayed on rows, 1% for full field spraying of arable crops, 10% for full field spraying of fruits and 100% for aerial spraying (Levitan, 1997).

In the soil compartment, both the acute ($EIP_{acute\_soil\_risk}$), and chronic ($EIP_{chronic\_soil\_risk}$) risks are evaluated. The acute risk is of concern to soil organisms present immediately after the application event, while the chronic risk is important for soil organisms after two years.

$$EIP_{acute\_soil\_risk} = \frac{PEC_{acute\_soil}}{0.1 \cdot LC_{50worm}} \cdot 100 \qquad (4.59)$$

where 0.1 is used to express Dutch environmental standard (1/10 of $LC_{50worms}$) and LC_{50worm} (mg kg^{-1}) is the acute toxicity to earthworms. PEC_{acute_soil} is calculated in the same way as the Norwegian Indicator.

$$EIP_{chronic\_soil\_risk} = \frac{PEC_{2\_years}}{0.1 \cdot NOEC_{earthworm}} \cdot AR_{a.i.} \cdot 100 \qquad (4.60)$$

where NOEC is the No Observable Effect Concentration (mg kg^{-1}), and

$$PEC_{2\_years}\ (\text{mg kg}^{-1}) = P_2 \cdot PEC_{acute\_soil} \qquad (4.61)$$

where P_2 is the residue of pesticide in the soil after two years. It was often difficult to find $NOEC_{earthworm}$, therefore PEC_{2_years} was replaced with PEC in the soil moisture (mg l^{-1}):

$$PEC_{soil\_moisture} = \frac{PEC_{2\_years}}{K_{om} \cdot f_{OC} + 0.2} \qquad (4.62)$$

and NOEC water organism values are used to determine the $EIP_{chronic\_soil\_risk}$.

4.5. Discussion and conclusions

4.5.1. Nutrient indicators

We have selected several indicators of different complexity (Table 4.10). All of them consider the basic elements of nutrient management, i.e. nutrient inputs and outputs. The water quality risk indicator also takes into account climate and soil properties that influence losses to surface and ground water; it also indirectly uses information on crop growth and development (for the calculation of evapotranspiration). Additionally, the two N indicators and the P indicator make use of crop management information. Climate, soil and management data are always included at the typical level of an indicator calculation, i.e. using simple synthetic properties (e.g. soil water holding capacity, annual precipitation and evapotranspiration, amounts and dates of fertiliser applications). One of the strengths of all the chosen indicators is to allow comparisons in space (among different systems located in a given portion of land) and in time (the same system over years). OECD (2001) reports examples of such comparisons at a national level for soil surface N balance; Hanegraaf and den Boer (2003) and Swensson (2003) report the variations over time of farm gate N surplus. All the indicators (with the exception of the farm gate balance) are applied on the field scale, which is where the losses take place and where it is possible to make changes in nutrient management. The temporal scale is always a year, with the exception of the N indicators, where there are variable stages of time, depending on the dates of fertiliser applications and crop harvest. This is consistent with the dynamic and more detailed nature of this indicator. All the indicators can be calculated using basic data that are easily available on-farm or from agricultural databases (e.g. fertiliser use, livestock numbers and weight, cropped areas, crop yields, etc.). These data need to be integrated with information derived from researches and surveys, such as the nutrient concentration of crops and animal wastes, and the N dynamics data needed for the nitrogen indicator (Bockstaller and Girardin, 2000). In addition, local agricultural offices could provide pedo-climatic information (such as soil hydraulic properties, precipitation, evapotranspiration) with soil maps and climatic data.

The advantage of the simple and easy-to-communicate nutrient balances is that they can be used to create awareness among farmers and to guide improvement in crop and livestock N management, as demonstrated for farm gate balances by Hanegraaf and den Boer (2003) and by Swensson (2003). Schröder et al. (1996) used the soil surface balance to monitor 38 Dutch farms before and after their conversion to integrated farming systems: on average the surplus decreased from 160 kg N ha^{-1} before conversion to 117 kg N ha^{-1} after. Another advantage of nutrient balances is that the relative importance of different inputs can be quickly assessed, as demonstrated for different nations by OECD (2001). Finally, nutrient balances can easily be calculated for different chemical elements (N, P, K, etc.). In the case of the farm gate balance, most of the data can be derived from farm accounts reporting purchased mineral fertilisers and feed, and crop and livestock products sold. The farm gate balance also resolves the weakness of soil surface balance when dealing with the estimates of nutrients contained in animal wastes (internal recycling; OECD, 2001). Simon et al. (2000) show that with the farm gate balance it is possible to identify the determinants of farm surplus and to distinguish different farm types on the basis of the surplus. Schröder et al. (2003), however, have pointed out that the farm gate surplus can be misleading because it is affected by strategic and operational management decisions: a large nutrient surplus is not necessarily associated with low in-farm efficiencies due to operational decisions. They also mention that analyses of the balances of separate compartments (soil, feed, harvestable crops, manure) may be needed to guide improvements in nutrient management. The main disadvantage of nutrient surpluses is that they represent potential (rather than actual) losses (OECD, 2001). Nutrient balances do not indicate where the surplus is stored (outer environment, soil, farmyard), the time scale of its availability, and the pathways of eventual losses (Watson and Atkinson, 1999). Öborn et al. (2003) point out that nutrient balances do not take into account any of the local site conditions such as climate and soil, and that the surplus is an indicator for total N losses only if it is integrated over a relatively long period. Salo and Turtola (2006) show that N losses account only for a part of the surplus integrated over a relatively long period, and that other regressors (e.g. precipitation, runoff, drainage) can better explain surplus variability. Nutrient balances are the most used type of nutrient indicators, and many Authors have applied them in various regions of the world (e.g. Sacco et al., 2003 for northern Italy). As mentioned earlier however, not all the Authors include the same components in the calculation of the surplus therefore a comparison of published results can be difficult.

The water quality risk indicator (OECD, 2001) has the advantage of integrating the soil surface balance with simple climatic and pedological information. It has several limitations: first, it is assumed that all excess nitrogen is lost as nitrate together with excess water; second, volatilisation, denitrification and soil immobilisation (if not subtracted previously) are apparently added to PNP and therefore contribute to the estimate of potential losses; third, land use and management (not considered by this indicator) have a strong influence on the relationship between surplus and nitrate losses; fourth, excess water can leave the soil as surface run-off or deep drainage, therefore the result represents an average nitrate concentration of water lost - as such, it cannot be compared directly to drinking water threshold values; finally, this indicator does not capture nutrient contamination events in semi-arid regions associated with major storms and run-off events, intensive livestock operations or irrigation (OECD, 2001).

The nitrogen indicators proposed by Bockstaller and Girardin (2000) and by Pervanchon et al. (2005) are a more detailed type of indicator. They are an attempt to overcome the limitations of simple balances, without going into the complexity of dynamic simulation models. Pervanchon et al. (2005) compared measured and predicted nitrate concentration of drainage water for grasslands in France, using the predictions carried out with the indicator. The performance of the indicator is not good in absolute terms, but the ranking of observed and calculated cases (number of cases belonging to a given concentration range) is rather similar. The management data needed to calculate the N indicators (e.g. to estimate volatilisation) have to be collected on-farm, as they represent specific information that generally is not available from official databases. The main limitations of the two indicators are the relative complexity of calculation and, for the one by Bockstaller and Girardin (2000), the fact that several parameters are specific for French conditions (e.g. volatilisation coefficients, which depend on temperature; crop and soil N dynamics estimates).

The phosphorus indicator represents an original attempt to include expert knowledge in the calculation of the indicator and to go beyond the calculation of a surplus. The recommended amount of P to be applied should be derived from a nutrient management plan, prepared by an advisor or by the person calculating the indicator. The comparison of the dose of applied P with the recommended dose enables the evaluation of the excess of P fertiliser use (spoil of non-renewable resources) or the deficit that would deplete soil P reserves. The example of an application is provided by Bechini et al. (2004), who used input data obtained by spatial interpolation with ordinary kriging.

4.5.2. Pesticide indicators

Currently there are many pesticide risk indicators published in literature. However, they differ greatly in the purpose for which they have been developed. A lot of pesticide indicators refer to specific topics (pesticide risk classification on a particular environmental compartment; pesticide risk classification for workers, bystanders, consumers; analysis of pesticide risk trends; identification of vulnerable areas to pesticides, etc.). Also the variables utilized and the methodologies differ. In the last years a number of research organizations have started research projects to analyse the state of the art of pesticide risk indicators, to examine the outcome and limitations of different approaches and to harmonize the use of these indicators internationally. For instance, the EU CAPER project (Concerted Action on Pesticide Environmental Risk Indicators; Reus et al., 1999; Reus et al., 2002) compared eight indicators developed for various purposes and created using different approaches for risk evaluation. The Organization for Economic Co-operation and Development also carried out projects concerning pesticide risk indicators, which focus mainly on the analysis and development of indicators for governments (OECD, 1997; 2005). The 6th European Framework Program financed the HAIR project (HArmonised environmental Indicators for pesticide Risk) to provide a harmonised European approach for pesticide risk indicators. The indicators developed in the HAIR project should operate on different scales, from farm-level to the catchments/regional level, up to the national-level. The different aggregation levels can be considered a pyramid with the highest level of aggregation at the top, and the highest level of detail and sophistication at the bottom (Fig. 4.1). At the farm level (bottom of pyramid), indicators are generally applied as decision support systems to help farmers in choosing pest control options and to evaluate the impact of their decisions. These indicators are also frequently utilized as a tool to influence consumers and market behaviour (ecolabelling or green labelling). At the regional or catchment level (middle of pyramid) the indicators are very useful for risk management of the territory. Finally, at the national level (top of pyramid) risk indicators are utilized to track the temporal risk trends on different scales with the aim of evaluating the performances of new or existing agro-environmental policies. In the literature we did not find many pesticide indicators that could be applied on the farm scale. Basically, most of the indicators were developed for comparing and ranking pesticides or to describe the trend of risk at a regional/national level, or to identify vulnerable areas. Moreover, the analysis of literature also allowed the identification of some pesticide risk indicators that could be useful on field-farm scale, even if their original purposes were quite different. These indicators were described earlier. Here we propose some general considerations and conclusions on their applicability and usefulness on farm scale.

The analysed indicators are very different in terms of complexity, inputs required and methodology. The simple and generic pesticide indicators (EEP, LI), require only $AR_{a.i.}$ and some physical-chemical and ecotoxicological properties of the a.i..

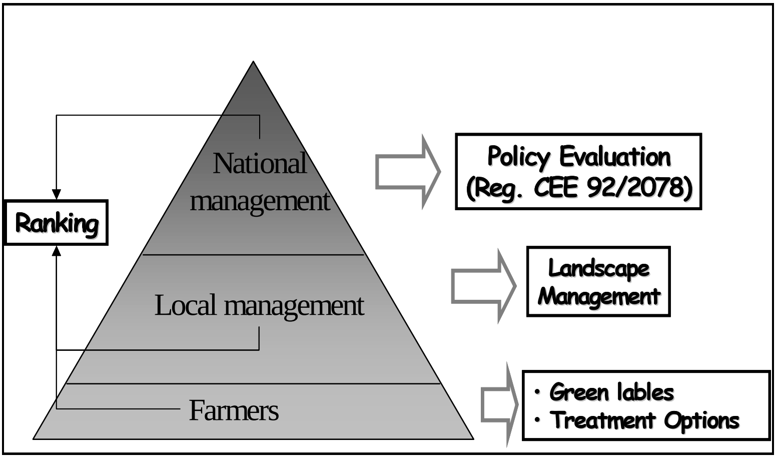

Fig. 4.1. – Use of pesticide risk indicators at different scale levels. Each level has a particular goal. From farm–level to national management, the ranking of pesticides (classification) in terms of their potential risk can be utilized.

These indicators do not consider pedo-climatic conditions and the environmental exposure of pesticides is not taken into account. Other indicators (Eco-rating, NI, SWIPE, EYP, EPRIP) also take into account the fate of a.i. in different environmental compartments (water, groundwater, soil, biota). As a consequence of the different approaches followed by different Authors also the final outputs of the indicators are very different. In fact, in some cases the risk is referred to a generic risk for the environment as a whole or for single organisms. This is the case, for instance, of the simplest indicator (LI), that calculates the number of Toxic Units for different organisms released during a treatment. In other cases, indicators synthesize the risk for a particular environment, as for instance surface water systems (SWIPE and NI). Finally, in one case the indicator calculates the risk for several environmental compartments but aggregates the results in a single score (EPRIP). Thus, when evaluating the outcomes of the indicators, it should be kept in mind that the output reflects the purposes for which the indicator has been developed. Furthermore, by their nature the indicators are crude tools that do not provide an exact measure of real risks. In this way, it does not seem possible to suggest which of these indicators could be better for a field-farming sustainability evaluation. Simple indicators requiring little data are inherently easy to communicate and to be understood by non specialists. On the other hand, if the objective is to compare different farms (or to evaluate the risk over time) more complex indicators that take account of pesticide exposure and of the site-specific conditions of the farm (both pedo-climatic conditions and technological efficiency) are required. For example, the use of anti-drift apparatus can bring about a noticeable reduction in the phenomenon, resulting in a drastic reduction of risk. It is evident that only the more complex indicators (Eco-rating, SWIPE, EYP) take into account management and the characteristics of the environment. The use of complex indicators, however, can be hampered by lack of data (both physical-chemical and ecotoxicological data). To partially overcome this problem we suggest the use of the Pesticide Manual (Tomlin, 2003) or the AGRITOX database from the INRA web site (INRA, 2006).

The scoring methodologies used in different indicators (Eco-rating, NI, SWIPE, EYP, EPRIP) are open to criticism, because these types of techniques are over simplistic (Thompson, 1990). The disadvantages of these techniques are mainly associated with the arbitrary nature of assigning scores (Lewis et al., 1997b). However this subjective choice is necessary in order to provide a starting point in the evaluation of a complex system.

From a careful analysis of the indicators it seems that, with the partial exception of the simplest indicators (LI), the others are more useful to evaluate the performance of single farms in reaching environmental sustainability over time, rather than for a comparison among farms.

On the basis of all these considerations we suggest that the most appropriate indicator is selected case by case. For instance, for a first evaluation and comparison among farms use of the LI could be more appropriate. Even if this indicator does not consider parameters related to the exposure and consequently to risk (the risk is a combination of toxicity and exposure), it gives information on the environmental pressures exerted by pesticides. In fact, this indicator gives the number of Toxic Doses (TU = Toxic Units) released into the environment during a treatment. Thus, for a first screening analysis and for comparative purposes it is possible to compare the number of TUs released from different farms as a consequence of different pest control strategies. Furthermore, this indicator is user friendly and understandable. On the contrary, for a more detailed analysis the use of more complex indicators is suggested. The selection of the indicator should be driven by considerations regarding site specific characteristics of the farm and particularly the relative nearness to the environment system at risk. For instance, the presence of surface water systems could suggest the use of indicators like SWIPE or NI. Finally, for a comprehensive evaluation of the farm it is suggested to use the Eco-Rating or EPRIP. However, such indicators require the use of several input data that are not easily available. Furthermore, EPRIP requires the use of predictive models and thus specific skills for their application.

4.6. Acknowledgments

This work is a part of the project "Un'agricoltura per le aree protette", funded by the Italian Ministry of Education, University and Research and by the University of Milan (Paper number 10).

Table 4.10. – Comparison of the nutrient indicators presented in this paper

<table>
<tr><th>Indicator</th><th>Data availability</th><th>Scales (space, time)</th><th colspan="2">Advantages</th><th colspan="2">Disadvantages</th><th>Space/ time compa-risons</th><th>Sensitive to climate soil</th><th>Sensitive to soil and crop managemen</th><th>Predicts air losses</th><th>Availability of results in literature</th></tr>
<tr><td>Farm gate balance</td><td>Farm statistics
Economic transactions</td><td>Farm, Year</td><td>Creates awareness and guides improvements
No need to estimate manure concentration and amount of manure distributed on each field</td><td rowspan="2">Can be calculated every year
Easy to communicate
Distinguishes the relative importance of different nutrient inputs
Can be applied to multiple nutrients</td><td>Can be misleading when different farms are compared
No description below farm scale
No relation to groundwater N concentration</td><td rowspan="2">Does not tell where the surplus is stored, the time scale of its availability and the losses pathway
Only estimates potential losses
Does not consider management</td><td>Yes</td><td>No</td><td>No</td><td>No</td><td rowspan="2">Yes, but different balance components are included in the calculation by different researchers</td></tr>
<tr><td>Soil surface balance</td><td>Farmer's bookkeeping
Farmer's interview</td><td>Field, Year</td><td>Simple
Describes each field separately</td><td></td><td>Yes</td><td>No</td><td>No</td><td>No</td></tr>
<tr><td>Water quality risk indicator</td><td>Same as soil surface balance + Soil map, climatic data, crop growth and development (to estimate ET_a)</td><td>Field, Year</td><td colspan="2">Includes the effect of climate and soil</td><td colspan="2">Does not separate gaseous losses and soil immobilisation
Does not separate surface– from ground–water losses</td><td>Yes</td><td>Yes</td><td>No</td><td>No</td><td>Few papers</td></tr>
<tr><td>Nitrogen indicators</td><td>Same as soil surface balance + Soil map, climatic data, dynamics of N in the soil–crop system</td><td>Field, Month</td><td colspan="2">Considers the dynamics of N in the soil–crop system and detailed crop N management</td><td colspan="2">More site–specific parameters are needed</td><td>Yes</td><td>Yes</td><td>Yes</td><td>Yes</td><td>Few papers</td></tr>
<tr><td>Phosphorus indicator</td><td>Same as soil surface balance + Soil analysis</td><td>Field, Year</td><td colspan="2">Simple. Considers P management</td><td colspan="2"></td><td>Yes</td><td>Soil</td><td>Yes</td><td>not applicable</td><td>No papers</td></tr>
</table>

ET_a = actual evapotranspiration

Sol

CHAPTER 5

MONITORING OF CROPPING SYSTEMS SUSTAINABILITY IN SEVEN FARMS IN NORTHERN ITALY: APPLICATION OF ENVIRONMENTAL AND ECONOMIC INDICATORS

Keywords: Cost, Nutrient, Energy, Pesticide, Landscape, Organic Matter

5.1. Abstract

The concerns of economically and environmentally sustainable agriculture involve consumers, citizens, policy makers and farmers. A simple tool that could be used for a preliminary evaluation of sustainability of the agricultural systems can be based on indicators.

During the period 2005 – 2006, seven farms in the Sud Milano Agricultural Park (Northern Italy) were monitored, collecting accurate data on farm management by face-to-face interviews with farmers. A set of indicators selected from literature was applied to 266 fields × crops combinations monitored. The indicators describe the economic, nutrients, energy, and plant protection agents management. The indicator values were then aggregated at field and farm level, introducing in the framework other indicators on soil quality, landscape and biodiversity management.

The rice and corn have good economic sustainability, but the former has high potential impact on environment due to intensive use of plant protection agents and low energy production, while the latter has generally high nutrient surpluses. The introduction of barley and winter wheat in crop sequence reduces the economic sustainability of the systems, but increases the environmental sustainability. Generally, fodder crops have very low environmental impact (low energy consumption, and nutrient surpluses, no plant protection agent application), but poor economic performance.

The intensive dairy farm has a good compromise between economic and environmental sustainability, while dairy extensive farms obtained low values in the economic indicators. The two swine farms monitored obtained high value in the economic indicators, but the nutrient surpluses are usually excessive. The two farms with rice obtained high values in the economic indicators, and correct nutrient balances, but the intensive use of plant protection agents and the low amount of carbon left in the soil decrease the environmental sustainability of these farms. The solution adopted in the farm that has differentiated the crops and used low inputs does not provide particularly good results both from the economic and the environmental points of view.

The most important research needed to improve agricultural sustainability assessment with indicators is the analysis of trade offs among different indicators, the integration of different values in a general index, and the quantification of uncertainty related to inputs.

5.2. Abbreviations

a.i.: active ingredient
A.R.: application rate
BI: field boundary indicator
CCf: field with Corn and other winter cereal
CD: crop diversity indicator
CEf: field with winter cereal
CER-RIC: cereal-rice farm
Cf: field with continuous corn
CS: crop sequence indicator
DAI-EXT: dairy extensive farm
DAI-INT: dairy intensive farm
EE: economic efficiency
$EEP_{a, s, w}$: Environmental Exposure-based Pesticide in the air, soil, and groundwater, respectively
EnE: energy efficiency
EnG: energy gain
EnIN: fossil energy input
EnOUT: calorific energy out
GI: gross income
GIS: geographical information systems
GM: gross margin
GM/EnG: economic gain-energy gain (GM / EnG)
GM/EnIN: economic gain-energy input (GM / EnIN)
HR: hedge-row indicator
KS: potassium surplus
$LI_{r, b, e, be, f, c, a}$: Load Index for ratss, birds, earthworms, bees, fish, crustaceans, and algae, respectively
MIX: mixed farm
NE, PE, KE: nutrient efficiency (yield / N_{input}, yield / $P_2O_{5\ input}$, yield / K_2O_{input}
NS: nitrogen surplus
NW, PW, KW: nutrient waste (yield / NS, yield / PS, yield / KS
NW/GM, PW/GM, KW/GM: nutrient waste-economic gain (NS / GM, PS / GM, KS / GM)
NW/EnG, PW/EnG, KW/EnG: nutrient waste -energy gain (NS / EnG, PS / EnG, KS / EnG)
OC: organic carbon
OCI: organic carbon indicator
PASM: Sud Milano Agricultural Park
PMf: field with permanent meadows
PPA: plant protection agent
PS: phosphorus surplus
Rf: field with continuous rice
RIC-POU: rice poultry farm
ROf: field with rice and other crops
SC: soil cover index
SOC: soil organic carbon

SOM: soil organic matter
SWI-INT: swine intensive farm
VC: variable costs
VC/EnG: economic investment-energy gain (VC / EnG)

5.3. Introduction

In the last decades, the perception of relations between agriculture and environment has remarkably changed and concerns have been raised about the sustainability of agricultural production systems, involving consumers, citizens, policy makers and farmers.

Different methods are applicable in the quantification of sustainability of agricultural systems, like direct measurements, simulation models, simple or composite indicators that have different levels of applicability and potential explanation of the system (Bockstaller and Girardin, 2002; Castoldi and Bechini, 2006). As direct measurements and simulation models are too expensive and time consuming, indicators can be applied for a preliminary evaluation of sustainability, because they are based on data already available or easy to collect. Indicators synthesize the effects on the environment and the state of the agricultural systems.

Frequently, sustainability analyses of agricultural systems are carried out using average data on farm management, because specific and detailed data are not available. With this approach it is not possible to highlight the variability of specific managements in different fields, smoothing the maxima and minima, situations that usually represent high potential environmental impacts. Therefore in this work agro-ecological and economic indicators were selected and calculated using data referred to single crops in each field, collected by face-to-face interviews with farmers during two year period. The objective of this work was to monitor different farm managements, collecting and storing accurate data used to evaluate the economic and environmental sustainability of different crop/farm managements. The results obtained were used to suggest possible solutions to improve the systems. It was also possible to highlight the effectiveness and limits of these tools.

The social sustainability of system is an important part of agricultural systems assessment, but it is outside the scope of this research.

5.4. Materials and methods

5.4.1. Study area

The Sud Milano Agricultural Park (PASM, Parco Agricolo Sud Milano; 45°N, 9°E) is a regional metropolitan agricultural Park surrounding the town of Milano (Northern Italy). The Park covers an area of approximately 47,000 ha, of which 35,000 ha are agricultural (Bechini and Castoldi, 2006a). It was created by the Regional Law 24/90, in order to coordinate the cohabitation between agricultural and urban area, preserving the agriculture and the agricultural land from the continuous advancing of the urban, industrial and infrastructure activities. The area is densely populated: there are more than 3 millions inhabitants.

The Park is located in a plain area with an altitude gradient from North to South of about 160 – 80 m above sea level, and an average slope of 0.3%. The main soils are loam, sandy-loam, silt-loam; the area is covered by a dense network of natural rivers and canals. Groundwater table is usually shallow (it has a depth from 16 to 4 m below field surface), with a North-South direction; groundwater reaches the surface in about one hundred resurgences. The climate is sub-humid; the average annual rainfall is about 950 mm, with peaks in May (100 mm) and October (110 mm) and lowest values in winter and in July (60 mm). Temperatures increase from January (average T min: –1.2°C, T max: 4.9°C) to July (average T min: 17.7°C, T max: 29.2°C). The annual reference evapotranspiration (ET_0) has an average of 800 mm yr^{-1} with a peak in July (daily average of 5 mm d^{-1}). ET_0 exceeds rainfall from May to September.

The Park is located in one of the most intensive Italian agricultural production areas. The most important crops are corn (*Zea mays* L.), rice (*Oryza sativa* L.), permanent meadows, barley (*Hordeum* spp.), Italian ryegrass (*Lolium multiflorum* Lam.), and winter wheat (*Triticum aestivum* L.), triticale (*Triticum* × *Secale*), soybean [*Glycine max* (L.) Merr.], with moderate to high yields. The most important irrigation systems are the sprinkler and the surface irrigation, with or without hydraulic turbine. Many farms have breeds (dairy and cattle, swine and poultry).

5.4.2. Farm type and data collection

A database storing detailed and georeferenced information about agricultural activities, was used to select seven representative farms (Fig. 1.2; Table 5.1; Bechini et al., 2005a): two dairy farms (DAI-INT = dairy intensive; DAI-EXT = dairy extensive), two swine farms (SWI-INT = swine intensive; SWI-EXT = swine extensive), a rice farm with a small layer breeding (RIC-POU = rice and poultry); a farm with a large variety of crops with a small cattle and horse breeding (MIX = mixed), and a cereal farms with a large area with rice (CER-RIC = cereals rice).

Table 5.1. – Characteristics of the seven farms monitored

Farm [1]	DAI-INT	DAI-EXT	SWI-INT	SWI-EXT	RIC-POU	MIX	CER-RIC
Total area (ha)	58	134	35	81	115	48	55
n. fields monitored	24	28	8	20	29	20	15
Average field dimension (ha)	2.4	5.0	3.7	4.3	4.3	2.5	3.7
Crop type	% of farm area (n. of crops monitored) [2]						
Corn	58 (26)	19 (16)	75 (14)	90 (34)	13 (12)	41 (13)	31 (10)
Rice					85 (40)		42 (11)
Wheat	4 (1)					18 (6)	17 (5)
Barley	4 (1)	3 (3)				23 (6)	
Meadows	29 (18)	79 (36)				9 (8)	
Triticale	9 (2)						
Ital. ryegrass						9 (2)	
Soybean							4 (2)
Trees			25 (2)				
Set-Aside	5 (2)	2 (4)		10 (6)	3 (6)	9 (7)	5 (2)
Livestock (Mg l.w. ha^{-1})	1.88	0.86	5.69	1.22	0.16	0.31	0.00

[1] DAI-INT = dairy intensive; DAI-EXT = dairy extensive; SWI-INT = swine intensive; SWI-EXT = swine extensive, RIC-POU = rice farm with a small layer breeding; MIX = farm with a large variety of crops and a small cattle and horse breeding, CER-RIC = cereal farm with rice;

[2] the sum of percentage of farm area is higher than 100 when intercrops are present.

The most important crop is corn (cultivated in all the farms monitored): all the farms produce grain corn, and DAI farm harvest a part of corn as silage (76% and 85% in DAI-INT and DAI-EXT, respectively); rice is cultivated in farms located in the South West of the PASM, an important Italian rice districts. SWI-INT is the only farm that does not buy fertilizer, but uses only manure produced in farm.

These farms were periodically visited to collect data about farm management during the period 2005 – 2006, by using a structured questionnaire completed during face-to-face interviews with farmers.

The data collected regard crop management at field level (Table 5.2): generic agronomic operations (date of plowing, harrowing, tillage, etc.), sowing (date, dose, species, cost, variety, and sowing type), fertilizer and manure application (date, dose, nutrients concentration, and cost), pesticide application (date, dose, active ingredients concentration, and cost), irrigation (date and type), harvest (date, yield, humidity, type of harvested produced, and selling price). Unfortunately, it was not possible to define the quantity of water applied. Livestock management was also monitored by collecting data about the number, average weight and type of animals in the farm, the daily ration, and the livestock animal inputs and outputs (animal products as milk and eggs, dead and sold animals). The annual farm inputs and outputs of material and energy were also recorded (fertilizers, manures, pesticides, herbicides, gasoline, lubricants, electricity, and animal feeds). All the amounts of products applied (fertilizers, pesticides) were double-cheched with bills, and furter details requested to farmers if necessary. No precise data of forage crops yields (permanent meadows, Italian ryegrass, and triticale) were obtained during the interviews, hence an indirect method was applied in order to estimate them. The total forage used was obtained by multiplying the per capita daily ration by the number of animal heads, and the sold forage was added (all these data are recorded by farmers with accuracy), obtaining the total amount of forage produced by the farm in one year (TF). The average yield (Y_a) for each single field in each single forage harvesting operation was calculated by dividing TF by the area and the number of haymaking operations. Y_a was increased or decreased according to the farmer judgment on the specific productivity of each single field, maintaining TF constant. These estimated yields are similar to some occasionally direct measurements (data not shown).

All information was stored in a geographical information system (GIS) containing the relational database and farm maps.

A set of indicators at crop and field level was calculated. They are grouped in six categories (economic, nutrient, energy, soil management, pesticide, and cross indicators). These indicators were aggregated by area weighted average in order to provide the indicator value at farm level. At this latter level other indicators were calculated (landscape indicators).

5.4.3. Indicator framework

5.4.3.1. Indicators at crop level

At this level, the indicators are calculated for each crop × field combination (from here reported as crop), using the detailed data declared by farmer during the interviews.

5.4.3.1.1. Economic indicators

To simplify data collection and analysis, we included variable and not fixed costs in the economic indicators. The sum of the variable costs (VC) includes the cost for: i) plant protection agents (PPA: fungicides, herbicides, insecticides, seed dressings, soil fumigants, and pesticide adjuvants), ii) fertilizers, iii) gasoline and lubricants (for the agronomic operations, irrigations, drying, grain milling, fodder ensilages, and transports of machineries and products from farm to field and vice versa), and iv) seeds.

The economic value of the nutrients in manures was not accounted, because manures are considered in some cases as livestock waste, in others a nutrient source. We have considered that all food costs are included in livestock compartment and the excreta nutrients have no economic value if used inside the farm.

Ordinary repair costs of the machineries and buildings were not considered because the redistribution of these costs by area does not represent a significant percentage of total costs. The extraordinary repair costs were also not counted because they have to be redistributed in a long period not quantifiable with our data. The redistribution of human labor time (and corresponding cost) among different crops is not possible due to difficulty in the quantification of: i) time necessary for each field operation (variable depending on different pedo-climatic condition, machinery used, and unforeseen occurred), ii) redistribution of time spent in non field operations (repair operations, hours dedicate to purchasing, selling, planning, reunions, etc.), and iii) time spent in field and in livestock operations. For similar reasons the depreciation allowance or the rent for machinery, lend and building rents, taxes, insurances, trade union fees, were not included in VC. These choices permit to compile a simplified account VC, easy to calculate and to manage.

This indicator is useful to evaluate the economic investment in the short term and the related proneness to risk: high VCs are characteristic of intensive systems where farmers invest many financial resources in order to obtain products with high economic value (high revenues and high profits) or to reduce the product loss and depletion. High VCs represent also high risks of money loss in case of failed or reduced harvest (e.g. in a dry season). Systems with high VCs increase also the allied activities (agrochemicals industry, producers of seeds and machineries).

The gross income (GI) was calculated for each single crop as the sum of the gross proceeds of the sold products or the corresponding economic value of the products re-used in the breeding. The prices were either provided by farmers or estimated by using the annual average price provided by the Milan’s stock exchange quotations. High GI increases the turnover of agricultural compartment, and corresponding gross domestic product, growing the economic importance of agriculture compared to the industry and services.

Therefore gross margin (GM = GI–VC) and the economic efficiency (EE= GI/VC) were calculated. High GM are advisable in region with limited and expensive agricultural area, where the maximum profit per area unit is advisable, while high EEs are preferable in region where the land availability is not a limited factor and the aim of farmers is to obtain the maximum profit (GI) per economic unit invested (VC).

5.4.3.1.2. Nutrient management indicators

The soil surface nutrient balances (Parris, 1998) for the three main macro-nutrients (N, P, and K) were calculated for each crop. These indicators represent the difference between the nutrients entering the soil and those leaving the soil with crop uptake. Positive values of the balance indicate nutrient accumulation in soil and / or nutrient losses, while negative values indicate nutrient depletion from soil. The soil surface balances were calculated as:

$$S = F + M + R_P + A + B - R - U \qquad (5.1)$$

where S is the surplus (NS, PS, and KS, for nitrogen, phosphorus, and potassium, respectively), F is the amount of nutrient applied with chemical fertilizers, R_P is the nutrient returned to soil with residues originating from the previous crop in the crop sequnece, M is the nutrient applied with manures, A is the atmospheric deposition (only for N), B is the biological fixation of leguminous crops (only for N), R is the nutrient removed from soil with crop residues, and U is the nutrient removed from soil with useful product exported from field.

The amount of nutrients in the irrigation water was considered negligible, even if in some case it could be significant. Surface water analyses provided by Milano Province (Provincia di Milano, 2007) for the period 1999 – 2000 report common low values of the nutrient concentration. With a total amount of water provided by irrigation from 100 to 300 mm in each crop season, the nitrogen supply from irrigation was estimated: it ranges from 5 to 10 kg N ha^{-1}, and in a few cases could be between 20 and 30 kg N ha^{-1}. For the P_2O_5, the irrigations could provide on average 1 kg P_2O_5 ha^{-1}, with maxima ranging from 3 to 8 kg P_2O_5 ha^{-1}. In the study area, after the construction of the Nosedo municipal wastewater treatment plant (2003 – 2004), the nutrient contents in the irrigation water are decreased.

Ammonia volatilization and denitrification were not considered because the available informations did not allow their estimate, hence they are part of N surplus.

The amount of fertilizers and manures applied for each crop were declared by farmers during the interviews. Since some declaration appeared not correct, all these data were compared with the bills provided by farmers, and in case of inconsistency between the declared total amount of fertilizer applied and bought, a subsequent interview was carried out, in order to define with security and more accuracy the destiny of all the fertilizers bought. F was estimated by multiplying the amount of fertilizers applied by corresponding nutrient concentrations.

The estimation of M is a critical step because farmers did not have manure analyses, and they have no tools to evaluate the nutrient concentration of manures applied in each operation. The N concentration provided by literature has a high variability because it is heavy influenced by the water used in the livestock management and by the rain collected in the external paddock, mixed with manure and stored in tank. Hence, nutrients excreted by each animal group were estimated using specific live weight – emissions coefficients (NRCS, 1992; Table 5.2). N transformations and C emissions from slurry storage are dynamic over time and are strongly influenced by storage conditions such C/N ratio, temperature and the presence of an adapted microbial community in pre-stored slurry (Sommer et al., 2007). These data are not available and the decreasing of N content, such gas emission (NH_4, N_2 and NO_x), during the storage period was estimated: 18% of total N excreted in anaerobic (liquid manure) processes (Thomsen, 2000).

Table 5.2. – average nutrients and C/N as excreted for each animal categories

	N-tot	P_2O_5	K_2O	C/N
	– kg yr^{-1}	Mg^{-1}	live weight –	
Dairy				
lactating	146	58	114	10
dry	131	42	101	13
heifer	113	33	106	14
Cattle				
veal	73	25	110	2
200 –340 kg	120	84	88	12
340–500 kg	113	92	106	11
> 500 kg	120	100	114	10
Swine				
grower (20 – 100 kg)	153	114	97	7
Poultry				
layer	155	230	132	9
Horse	102	42	84	19

The ammonia volatilization during the manure spread operations was not estimated and it is included in N input. The ammonia loss depends on the crop management such the utilization of systems that reduce the volatilization (ploughing in manure application, band spreading, injection;

Sommer and Hutchings, 2001); therefore heavy losses during manure spread will produce high NBs and corresponding bad judgments. No losses were estimated for P_2O_5 and K_2O. Nutrient concentrations were obtained by dividing the total amount of nutrients after storage by the total volume of manure applied declared by farmers. The nutrient contents provided by literature (Grignani et al., 2003) were used for solid manure because we had no information about the quantity of straw used.

With this procedure, the nutrients allocated in each field from manure are proportional to the declared volume of manure applied. The nutrient concentrations in manure were estimated constant in all manure applications during a year, without to consider the variability among different operations occurred. This coarse estimation could produce uncertainty in the nutrient balances, but it was preferred to the use of fix coefficients provide by literature.

The method adopted provide results close to the nutrient concentration coefficient provide by the official methodology adopted by Regione Lombardia (2006) for the manure nitrogen utilization plan, but it allow to estimate also the P_2O_5 and K_2O concentration.

Atmospheric depositions (A) were set to 30 kg N ha^{-1} yr^{-1}, in accordance with the data reported by Grignani et al. (2003), and 0 for P and K. Biological fixation (B) is 0 for P and K, while for N it was estimated as U – A – 0.5 M – 0.7 F for monospecific leguminous crops (soybean), and equal to 40 kg N ha^{-1} yr^{-1} for permanent meadows (Grignani et al., 2003). U was estimated by multiplying the declared crop yields by the corresponding nutrient concentrations derived from literature (Grignani et al., 2003). For the estimation of R_P and R, the crop residues were estimated using yield and harvest index, and multiplied by corresponding tabbed nutrient concentrations (Grignani et al., 2003).

5.4.3.1.3. Energy indicators

The quantification of fossil energy use is important in order to improve the efficiency of agro-ecosystems and to reduce the green house gasses emissions and the consumption of limited resources. Different ways are proposed in literature to quantify fossil energy flows in agricultural systems. No standardized method (especially for system boundaries and energy equivalents) is available for energy balance in agricultural enterprises. Thus, the energy balance given by various authors are not or only partially comparable (Biermann et al., 1999).

In our study the “economic approach” (Biermann et al., 1999) was adopted: the aim is the determination of the energy consumption for production and handling of agricultural products, analyzing the use and depletion of energy resources. In this approach, only non-renewable fossil energy resources are considered and a synthetic account of the energy flow in the systems is based on specific coefficients used to convert mass fluxes into energy fluxes. A specific energy requirement for the agronomic operation and the energy in the machinery is also accounted.

The equivalent direct and indirect energy inputs (EnI) were estimated. EnI is the sum of energy content of: i) PPAs applied, ii) fertilizers used, iii) seeds, iv) diesel and lubricants consumed (for the agronomic operations, irrigations, drying processes, grain millings, fodder ensilages, and transports of machineries and products from farm to field and vice versa), and v) machinery. Like VC, this indicator is useful to evaluate the intensity of the system, the energy investment, and corresponding loss risk.

The active ingredient (a.i.) dose applied in each crop treatment was calculated for each PPA application. It was not possible to obtain the energy cost for the production of every a.i., but energy input for PPA in the whole energy balance is usually low (Biermann, 1999). Hence the coarse coefficients provided by literature for each specific PPA types were used (Table 5.3). Development expenses have not been considered in energetic assessments of PPA.

For fertilizers, specific energy costs (Table 5.3) provided by literature (Kongshaug, 1998) were used. The fertilizers energy coefficients are strongly influenced by the specific plants where the fertilizers are produced: old plants have higher energy requirement than modern plants (Kongshaug, 1998; Biermann et al., 1999) The specific mine where raw material are extracted influence the energy cost in the production of phosphorus, potassium an lime fertilizers (Kongshaug, 1998). The definition of P_2O_5 and K_2O processes used in the production of compound fertilizer was not possible with our data; hence the energy

Table 5.3. – Specific energy contents of production factors

Product	Unit	Specific energy content	Source
Fertilizers			
$N\text{-}NH_4$	MJ kg^{-1}	39.0	Kongshaug, 1998
N-ureic	MJ kg^{-1}	48.0	Kongshaug, 1998
$N\text{-}NO_3$	MJ kg^{-1}	32.0	Kongshaug, 1998
P_2O_5 [1]	MJ kg^{-1}	4.0	Kongshaug, 1998
K_2O [2]	MJ kg^{-1}	5.0	Kongshaug, 1998
limestone	MJ kg^{-1}	0.8	Kongshaug, 1998
Pesticides			
herbicides	MJ kg^{-1} a.i.	288	Biermann, 1999
insecticides	MJ kg^{-1} a.i.	237	Biermann, 1999
fungicides	MJ kg^{-1} a.i.	196	Biermann, 1999
soil fumigants	MJ kg^{-1} a.i.	196	Estimated
pesticide adjuvants	MJ kg^{-1} a.i.	196	Estimated
seed dressings	MJ kg^{-1} a.i.	196	Estimated
Seeds			
corn	MJ kg^{-1}	113.2	Estimated
rice	MJ kg^{-1}	31.4	Estimated
barley	MJ kg^{-1}	31.4	Estimated
winter wheat	MJ kg^{-1}	31.3	Estimated
triticale	MJ kg^{-1}	31.4	Estimated
soybean	MJ kg^{-1}	40.6	Estimated
Italian ryegrass	MJ kg^{-1}	31.3	Estimated
Irrigation			
sprinkler system	MJ mm^{-1}	34.6	Ribaudo, 2000
surface irrigation			
with hydraulic turbine	MJ mm^{-1}	4.5	Estimated
without hydraulic turbine	MJ mm^{-1}	0.0	Estimated
Other energy consumption			
lubricants	MJ L^{-1} diesel	3.6	Dalgaard et al., 2000
energy embedded in machinery	MJ L^{-1} diesel	12.0	Dalgaard et al., 2000
Fuels			
diesel	MJ L^{-1}	36.4	Patzek, 2004

a.i.: active ingredient; [1] as triple superphosphate; [2] as potassium chloride

Table 5.4. – Diesel use in crop production

	Unit	Diesel consumption rate: Literature	Diesel consumption rate: Declared by farmers
Tilling and sowing			
subsoiling	L ha^{-1}	31.1 [1]	
field leveling	L ha^{-1}	5.0 [1]	17.0
hoeing	L ha^{-1}		17.5
plowing	L ha^{-1}	23.0 [2]	18.0 – 24.0
furrowing	L ha^{-1}		3.0
disc harrowing	L ha^{-1}	7.7 [1]	
spike harrowing	L ha^{-1}	4.3 [1]	
rotary hoeing	L ha^{-1}	14.3 [1]	14.0
harrowing	L ha^{-1}	5.0 [2]	7.0 – 11.5
rolling	L ha^{-1}	2.0 [2]	2.0
sowing	L ha^{-1}	3.0 [2]	3.0
minimum tillage sowing	L ha^{-1}		5.0
combine harrowing and sowing	L ha^{-1}		20.0
combine subsoiling, harrowing, and sowing	L ha^{-1}		35.0
Crop operations			
pesticide application	L ha^{-1}	1.5 [2]	1.5 – 2.0
fertilizer application	L ha^{-1}	2.0 [2]	1.5
weeding and earthing up	L ha^{-1}	10.2 [1]	12.0
cutting of meadows	L ha^{-1}	5.0 [2]	
silage mowing - chopping - loading	L ha^{-1}	23.2 [1]	
haymaking	L ha^{-1}	25.7 [1]	
combine harvesting	L ha^{-1}	14.0 [2]	
straw baling and loading	L ha^{-1}	6.9 [1]	
stalk breaking	L ha^{-1}	11.9 [1]	9.0
weeds mowing	L ha^{-1}	6.2 [1]	
machine transport	L ha^{-1}	0.1 [3]	
manure and fodder transport	L km^{-1}Mg^{-1}	0.1 [2]	
Post harvest operations			
drying	L Mg^{-1} water extracted	91.1 [4]	
milling	L Mg^{-1} product		0.9
silaging	L Mg^{-1} product	0.50 [2]	

[1] Ribaudo, 2000; [2] Dalgaard et al., 2000; [3] Jarach, 1985; [4] Pellizzi, 1996

costs for the production of the most common compound fertilizers (triple superphosphate and potassium chloride) were used.

In accordance with Biermann et al. (1999) the energy content of the manure was evaluated. Without consideration of organic fertilizers in energy balance, their effect on energy recovery in the yield cannot be allocated to the corresponding EnI, and this would mean that additional energy recovery is possible without additional input of fossil energy. In fact, fossil energy is needed for the production of organic fertilizer (e.g. for storage and processing). However, this energy input is hardly quantifiable. For practical reasons mineral fertilizer energy equivalents were adopted to evaluate the nutrient contents: 60 % for nitrogen and 100% for phosphorus and potassium.

Transport routes for the raw materials and intermediates could have been defined. Typical energy requirement for 1,000 km transport is 0.1 MJ kg^{-1} for bulk and liquid sea carriers, 0.4 MJ kg^{-1} for pipeline, 0.7 MJ kg^{-1} for rail, and 1.9 MJ kg^{-1} for truck (28 Mg) (Kongshaug, 1998). The place of origin, the transport used, and the cost of formulation and packaging are not obtainable with farmer interviews; hence a coefficient of 1 MJ kg^{-1} of fertilizer was used to estimate the energy cost of formulation, packaging, and transport.

Seed production uses more energy than regular crop production because there are additional storage, transport, processing, and packaging energy costs (Biermann et al., 1999; Shapoury et al., 2002). Seven times more energy is required to produce hybrid seeds compared with energy in the same mass of corn grain (Patzek, 2004), while for the other seeds, the double of the calorific energy in the grain was used (a humidity of 15% was estimated for seeds applied; Table 5.3). The energy content of seed dressing is not significant (from 0.3 to 0.5 MJ kg of corn seed and about 10^{-3} MJ kg^{-1} for the other seeds); hence the energy content in seeds with or without seed dressing is estimated equal.

The coefficients used for seeds energy content are higher than other values provided by literature: Jarach (1985) use the calorific value of the grain (values from 14 to 15 MJ kg^{-1} seeds), Ceccon et al. (2002) use values of 20 MJ kg^{-1} seeds for sugarbeat and 10 for other crops, Bonari et al. (1992) use values from 17 to 54 MJ kg^{-1} seeds. Other authors do not consider the energy in the seeds (Pervanchon et al., 2002) or subtract the amount of seeds used to the yield (Dalgaard et al., 2001). Biermann et al. (1999) assume the indirect energy for providing seeds (production and transformation) as input, while deduce the calorific value of seeds from the energy output. In our opinion it is necessary to account both indirect and direct energy (calorific value) of the product.

The date of each water application and the method used (sprinkler or surface irrigation, with or without the use of hydraulic turbine), were recorded for each irrigation, but the total amount of water applied was impossible to estimate. The energy required for irrigation was estimated by the use of a specific consumption of 34.6 MJ mm^{-1} of water applied (Ribaudo, 2000), and of 4.5 MJ mm^{-1}, in case of sprinkler system and surface application with hydraulic turbine, respectively. No energy consumption was considerated in case of surface application without recourse to hydraulic turbine. Irrigation water amount was estimated using average values in the studied area: 35 mm for sprinkler, 80 mm for surface application to meadows and 120 mm for surface application to corn and soybean. The lower value for meadows is due to the short interval between two irrigations (from 5 to 8 irrigations, every 20 – 25 days from June to September, are usually carry out in the common management for this crop). Rice flooding does not consume energy and the other crops were not irrigated.

The specific diesel consumption for agronomic operations was provided by farmers or, in case of lack of this information, the coefficients in Table 5.4 were applied. The distance between farm and field, calculated using GIS map, was used to quantify the fuel consumption for the transport. The coefficients reported in Table 5.4. were used. The fuel consumptions were back converted in energy unit using the low calorific value (or net calorific value) of the diesel fuel (36.4 MJ l^{-1}; Patzek, 2004). The low calorific value of a fossil fuel assumes that combustion products contain the water of combustion as vapor. The heat contained in this water is not recovered contrary to the high calorific value (or gross calorific value) that considers also the combustion water energy. The lubricant consumed was estimated multiplying the total fuel consumed (excluding drying) by a specific coefficient (Dalgaard et al., 2000) reported in Table 5.3.

The energy embedded in the machinery was calculated by multiplying the total fuel consumption by a specific coefficient (Dalgaard et al., 2000; Table 5.3). No energy consumption was accounted for buildings and plants because it is not quantifiable with our data. The energy for the human labor is negligible because it has only minor importance in modern agriculture compared with fossil energy (Jarach, 1985; Biermann et al., 1999): 0.7 MJ h^{-1} (Patzek, 2004), with an average labor time of 6.2 h ha^{-1} per growth season for corn (Pimentel, 2003).

Fossil energy is negligible compared to solar energy, therefore, if it is considered in energy balance, production-related differences in the input of fossil energy could not be discovered (Biermann et al., 1999).

The energy output (EnOUT) was obtained by converting the dry matter (DM) of yield (including straw, if exported from field) into energy using equivalents based on calorific values. This indicator quantifies the energy return of different crop, in order to evaluate the best energy production. The carbohydrate, protein and lipid contents for each product were estimated using tabbed values (Succi, 1995; USDA, 2007). The specific calorific energy content of carbohydrates, proteins and lipids are 17.6, 24.6, and 38.9, MJ kg^{-1}, respectively; Merrill and Watt, 1973); the total specific calorific energy contents of each product are reported in Table 5.5. The energy gain (EnG = EnOUT – EnIN) and the energy efficiency (EnE = EnOUT/EnIN) were calculated. The crops with high EnG are recommendable in situation with low land availability, because these crops have the highest energy gain per area unit. The crops with high EnE are recommendable where land availability is not a limitation.

Table 5.5. Energy content of crop products

Crop	Product type	Calorific energy content (MJ kg^{-1} DM)
corn	grain	19.0
	straw	16.8
	mash	18.7
	silage	17.9
Italian ryegrass	hay	17.2
	silage	16.8
meadows	hay	17.6
	silage	17.4
rice	grain	18.5
	straw	16.8
soybean	grain	23.9
triticale	silage	18.0
winter wheat	grain	18.4
and barley	straw	16.8

5.4.3.1.4. Pesticide indicators

In order to describe the complex behavior of PPAs and the corresponding environmental impacts, application of complex predictive models is necessary, but they require many environmental and management informations not obtainable with a simple farmer interview; therefore the use of complex models was rejected. On the other hand, simple indicators provide only the potential environmental impact related to the use of PPAs. In this study, two groups of indicators were calculated: i) the Environmental Exposure-based Pesticide index (EEP; Vereijken, 1995), that considers the potential exposure of different environment compartments, and ii) the Load Index (LI; OECD, 2005) that evaluates the potential effects on non-target organisms. The EEPs indicators are calculated in three different environmental compartments (air, soil, and groundwater) using some physical-chemical properties of each a.i. applied:

$$EEP_{air} = AR_{a.i.} \cdot VP/1000 \tag{5.2}$$

$$EEP_{soil} = AR_{a.i.} \cdot DT_{50soil} \tag{5.3}$$

$$EEP_{groundwater} = EEP_{soil} \cdot K_{oc}^{-1} \tag{5.4}$$

where $AR_{a.i.}$ is the application rate (kg a.i. ha^{-1}), VP is the vapor pressure at 25°C (Pa), DT_{50soil} is the half life of the chemical in soil (days), K_{oc} is the partition coefficient of the a.i. between organic matter and water fractions of the soil: K_{oc} = (conc. adsorbed/conc. dissolved)/ % organic carbon in the soil. It is a measure of a material's tendency to adsorb to soil particles. High K_{oc} values indicate a tendency for the material to be adsorbed by soil particles rather than remain dissolved in the soil solution. Strongly adsorbed molecules will not leach or move unless the soil particle to which they are adsorbed moves (as in erosion). K_{oc} values less than 500 indicate little or no adsorption and a potential for leaching (Extoxnet, 2007).

LI is calculated separately for seven non-target organisms: rats, birds, earthworms, bees, fishes, crustaceans, and algae (LI_r, LI_b, LIi_e, LI_{be}, LI_f, LI_c, and LI_a, respectively). The use of seven non-target organisms in the calculation of LI is necessary due to different effects of a.i. that could be important for one organism but non relevant for others:

the reduction of the number of non-target organisms analyzed could not highlight the potential toxicity of an a.i.. LI is defined as:

$$LI = \sum_{k=1}^{n} \frac{AR_{a.i.k}}{TOX_k} \tag{5.5}$$

where n is the number of active ingredients applied in one crop, and TOX_k is the acute Lethal Dose (LD_{50}, dose required to kill 50% of test organisms) or Lethal Concentration (LC_{50}, concentration required to kill 50% of test organisms) of k^{th} a.i.. The LD_{50} units are mg kg^{-1} for rats and birds, and μg bee^{-1} for bees, while the LC_{50} units are mg L^{-1} for algae, fishes, and crustaceans, and mg kg^{-1} soil for earthworms. In this work, for simplicity all LI units are expressed as TOX units ha^{-1}, instead of the different real units (10^6 $kg_{organism}$ ha^{-1} for birds and rats; 10^9 bee ha^{-1} for bees; 10^6 L ha^{-1} for algae, fishes and crustacean; 10^6 kg ha^{-1} for earthworms). Since the aim of this work is the environmental evaluation of farmer management, the effects of a.i. on workers, bystanders, and consumers (products quality) were not analyzed.

The framework of ten pesticide indicators selected has some limit action: it does not consider the peculiarity of the environment analyzed (pedo-climatic condition) and its effect on the fate of PPA, the interaction among different a.i., and the long term effects. These indicators describe the potential risk of a.i., but not the real dynamics in the environment. On the other hand, this framework does not use subjective scores and the indicators can be calculated with simple data obtained by interviews or provided by literature.

5.4.3.1.5. Cross indicators

Several agro-ecosystem performance indicators (cross indicators) were calculated to analyze the interaction among nutrient, economic and energy indicators previously calculated (Table 5.6).

5.4.3.2. Indicators at field level

The economic, nutrient, energy, pesticide, and cross indicators calculated at crop level were aggregated at field level, considering all crops present in the field during the study period. Moreover, an additional set of indicators that evaluate the appropriateness of soil management was added: these indicators are not applicable at cop level, because the interaction between crops, and the fallow period from previous-next crops are characteristics that influence the soil quality.

Table – 5.6. Cross indicators

Indicator	Acronym	Calculation[1]	Unit
nutrient efficiency	NE	yield / N input	kg DM kg^{-1} N_{input}
	PE	yield / P input	kg DM kg^{-1} P_2O_{5input}
	KE	yield / K input	kg DM kg^{-1} K_2O_{input}
nutrient waste	NW	NS / yield	kg $N_{surplus}$ kg^{-1} DM
	PW	PS / yield	kg $P_2O_{5surplus}$ kg^{-1} DM
	KW	KS / yield	kg $K_2O_{surplus}$ kg^{-1} DM
nutrient waste-economic gain	NW/GM	NS / GM	kg $N_{surplus}$ $k€^{-1}$
	PW/GM	PS / GM	kg $P_2O_{5surplus}$ $k€^{-1}$
	KW/GM	KS / GM	kg $K_2O_{surplus}$ $k€^{-1}$
nutrient waste -energy gain	NW/EnG	NS / EnG	kg $N_{surplus}$ GJ^{-1}
	PW/EnG	PS / EnG	kg $P_2O_{5surplus}$ GJ^{-1}
	KW/EnG	KS / EnG	kg $K_2O_{surplus}$ GJ^{-1}
economic investment-energy gain	VC/EnG	VC / EnG	€ GJ^{-1}
economic gain-energy gain	GM/EnG	GM / EnG	€ GJ^{-1}
economic gain-energy input	GM/EnI	GM /EnIN	€ GJ^{-1}

[1]The nutrient inputs are the sum of fertilizers, manures, and residues from previous crop, excluding biological fixation, and atmospheric deposition.
For abbreviations see § 5.2

5.4.3.2.1. Soil management indicators

The quality of soil management was described by three indicators: Crop Sequence indicator (CS; Bockstaller and Girardin, 2000), Soil Cover index (SC; Vereijken, 1995), and Soil Organic Carbon indicator (OCI, Bockstaller and Girardin, 2000).

The CS provides a value from 0 (worst value) to 10 (best value), with a threshold (7) that represents the achievement of a minimum quality for the rotation. It is defined as:

$$CS = K_p \cdot K_r \cdot K_d \qquad (5.6)$$

where K_p is the coefficient describing the effects of the preceding crop on the current crop (considering the development of pathogens, pests, weeds, and the effect on soil structure and nitrogen left in the soil), K_r depends on the frequency of crop cultivation, and K_d is an index of crop diversity; for details see Bockstaller and Girardin (2000); Castoldi and Bechini (2006). The CS was calculated in each field using the crops present from 2004 to 2007.

The SC is the percentage of soil cover by crops during a year. It evaluates the risks related to structure degradation, erosion, reduction of biodiversity, nutrients and pesticides losses for the purpose of the calculation made here. The soil is considered bare from sowing to crop emergence, covered at 50% from emergence to complete soil cover, and 100 % until harvest. The emergence was estimated by the crop type: 7 days after sowing for Italian ryegrass, 10 days for rice, winter wheat, barley, soybean, and triticale, 20 days for corn sown in the first half of April, 5 days for corn sown after. The complete soil cover is estimated to be reached 23 days after emergence for Italian ryegrass, 30 days for soybean, 35 days for barley, 40 days for rice, winter wheat, and triticale, and 30 days for corn sown before the end of April, 25 days for corn sowed after 1 May. The soil is considered always covered by permanent meadows.

Soil organic matter (SOM) comprise carbon pools having a mean residence time greater than 5 years, to the very recalcitrant carbon, which may have mean residence times on the order of centuries or longer (Stevenson, 1994). SOM is the most often reported attribute from long-term studies and is an important indicator of soil quality and agronomic sustainability because of its impact on other physical, chemical and biological indicators of soil quality (Reeves, 1997). Soil organic carbon (SOC) is the carbon fraction of SOM: the coefficient 0.58 (Boiffin et al., 1986; Ministero per le Politiche Agricole, 1999) was used to convert SOM into SOC value.

Organic carbon (OC) decomposition is influenced by climate, soil texture, quality of crop residues and soil disturbance, but models cannot always simulate the interactions between the different factors (Morari et al., 2006). The OCI proposed by Bockstaller and Girardin (2000) is a simplified model to detect the negative and the positive effects of different crop management practices on SOC content. The aim of this indicator is to identify and promote the practices that maintain SOC at a satisfactory level. It provides a value from 0 (worst value) to 10 (best value); a value of 7 represents the achievement of a minimum level. The indicator is defined as:

$$OCI = 7\ (Ax / Ar) \qquad (5.7)$$

where Ax (kg C ha^{-1} yr^{-1}) is the annual mean of OC inputs (from residues, animal manures, green manures, etc.), Ar (kg C ha^{-1} yr^{-1}) is the recommended level of OC inputs needed to maintain a satisfying level of SOC in the long term. OCI is upper limited (max=10) and it considers equal the soil managements that provide to the soil more than 1.4 (10/7) time the necessary OC, considering not further benefit for high excess of OC input.

For the calculation of Ax, the methodology of Bolinder et al. (2007) was used. They identify four fractions of OC that come from crops: agricultural products (C_P), straw and other above-ground post harvest residues (C_S), root tissues (C_R), and extra-root materials including root exudates (rhizodeposit) and other materials derived from root-turnover (C_E). C_P was calculated assuming that C concentration is about 45% in shoot plant parts (Bolinder et al., 2007) and 40% in root plant part (Boiffin et al., 1986). C_S and C_R were estimated using the tabbed harvest index and shoot:root ratio, respectively (Table 5.7). The biomass lost in the grass harvest was estimated as a fraction of biomass harvested (15%

Table 5.7. – Crops and biosolids parameters used in the estimation of the organic carbon indicator

	Harvest Index	Shoot:Root ratio	Humification rate [3]	
			Shoot residues	Root residues
Crops				
Barley	0.53 [1]	9.46 [1]	0.08	0.15
Corn	0.50 [1]	5.60 [1]	0.12	0.15
Italian ryegrass		1.30 [1]	0.08	0.15
Meadows		0.70 [1]	0.08	0.15
Rice	0.48 [2]	7.43 [1]	0.08	0.15
Soybean	0.40 [1]	5.20 [1]	0.08	0.15
Triticale	0.34 [1]	5.39 [1]	0.08	0.15
Winter wheat	0.40 [1]	6.81 [1]	0.08	0.15
Biosolids				
dairy/cattle/horse farmyard manure			0.30	
liquid dairy manure			0.30	
liquid pig manure			0.30	
layer/broiler litter			0.30	
biosolids from municipal wastewater treatment plants			0.20	

[1] Bolinder et al., 2007; [2] Confalonieri and Bocchi, 2005; [3] Boiffin et al., 1986.

in haymaking, 5% if the grass is cut and used fresh or ensiled); the stubbles for silage crops are estimated as 5% of the above ground biomass (Bolinder et al., 1999). The total root-derived carbon contribution is 1.5 to > 3 times more than above ground biomass derived carbon (Wilts et al., 2004; Johnson et al., 2006), but clear information does not exist about the quantification of rhizodeposition (Kuzyakov and Domanski, 2000; Bolinder et al., 2007). Hence a simplification is introduced (Kuzyakov and Domanski, 2000): about half of the below-ground translocated carbon is incorporated in root tissues; one third is respired by roots and rhizosphere microorganisms utilizing exudates and fine roots, and is evolved as CO_2 during few days after assimilation; the rest, here denoted as stable C_R (SC_E), remains in the soil and microorganisms ($SC_E = 0.33 \cdot C_R$).

Since the farmers have not provided manure analysis, the quantification of C concentration in liquid animal manures is a critical aspect. The concentrations provided by literature are too much variable. Hence total amount of C excreted was estimated case by case utilizing the N excreted previously calculated and the average C/N as excreted for each animal category (NRCS, 1992; Table 5.2). In anaerobic conditions (liquid manure), the decrease of C (loss as CO_2 and CH_4 emissions) during the storage period was estimated equal to 24% of initial carbon (Thomsen, 2000). The final C concentration in liquid manure was determined by dividing the total amount of OC after storage by the total volume of manure applied declared by farmers. The organic matter (OM) in solid manure was set to 16% (Petersen et al., 1998) because we had no information about the quantity of straw used. The OC concentration of solid manure was determined by multiplying OM by 0.35 (Marino Gallina et al., 2005a; Morvan et al., 2006).

In order to quantify the humification of OC provided by crops and biosolids, the plant OC fractions not exported from field and OC applied as biosolids were multiplied by specific humification rates (humified OC / total OC applied; Table 5.7).

A modified version of the Hénin and Dupuis model (1945) was used to quantify the OC inputs necessary to maintain a satisfactory level of SOC in a specific soil (Ar; details in Bockstaller and Girardin, 2000; Castoldi and Bechini, 2006). The satisfactory level of SOC was determined as a function of clay content (Ministero per le Politiche

Agricole, 1999). The inputs of this model are: i) soil texture and carbonate concentration (obtained from soil analyses provided by farmers or from pedological map; ERSAL, 1993), ii) bulk density (calculated from soil texture data; Saxton and Rawls, 2006), iii) average annual temperature (set to 12.5°C), iv) mineralization depth (set equal to the maximum depth of the agronomic operation in the period or equal to 30 cm for permanent meadows (equivalent to the soil layer that contains 77% of the root biomass; Lorenz, 1977), v) number of years with organic application (provided by farmers interview), and vi) number of years with crop residues (straw) harvest (provided by farmers interview).

5.4.3.3. Indicators at farm level

The economic, nutrient, energy, PPA, soil management and cross indicators calculated at the field level were aggregated at the farm level, considering as weight the area of each field. The set-aside and tree fields (SWI-INT) were not accounted for in this aggregation. These indicators evaluate the average management of the crop compartment of the farm. An additional set of indicators that evaluate the landscape was added.

5.4.3.3.1. Landscape and biodiversity indicators

As on the other level, we focus on indicators describing the relation between farm management and landscape (Castoldi and Bechini, 2006).

Bockstaller and Girardin (2000) have proposed a crop diversity indicator (CD) that evaluates the impact of crop partitioning and field size on landscape and biodiversity. It provides a value from 0 (worst case) to 10 (best case) and value of 7 represents the achievement of a minimum level. The indicator is calculated as:

$$CD = K \cdot NC \cdot D \cdot T \qquad (5.8)$$

where K is a calibration factor depending on the number of crops, NC is the number of crops, D is the crop partitioning factor, and T is the field size factor that increase the indicator value when the field are small (for details see Bockstaller and Girardin, 2000; Castoldi and Bechini, 2006).

A hedge-row is defined as a linear feature composed or shrubs and/or trees that forms part of a management unit (Baudry et al., 2000). The hedges and the rows are important structural elements of landscape (aesthetic contribution) and are a resource to wildlife in the agricultural area (Barr and Gillespie, 2000; Baudry et al., 2000). Hedge-rows have an important role in the soil protection and act as barriers and boundaries between management units due to the capacity of reduction of nutrient and PPA drift (Baudry et al., 2000). The hedge-row indicator (HR; Bocchi et al., 2004) is defined as:

$$HR = L / A \ (\mathrm{m\ ha^{-1}}) \qquad (5.9)$$

where L is the hedge-row length (m) and A is the total area analyzed (ha). The shape of hedge-row was defined by direct visit to the field, and mapped in the farm GIS; the tree field in SWI-INT was not considered as hedge-row. This indicator does not consider the characteristics of hedge-rows (e.g. species, height, and management), because this information require specific measures, and accurate survey. The indicator proposed is limited to the evaluation of the presence/ absence of hedge-rows in order to define the proneness of the farmers toward an openness landscape (easy to manage).

As compared with field that is often perturbed (agronomic operations, nutrients and PPAs application), the field boundary is an area with a reduced human pressure. In this area the agrochemicals applications are null or reduced, the agronomic operations are limited, and the self-sown vegetation is usually present during the entire year. These areas are hot-spot of biodiversity, where the animal (especially bugs and small mammals) find a shelter and the vegetation is composed by different species than cropped species. On the other hand it is a source of weed seeds that could infest the field. The field boundary indicator (BI) is defined as the ratio between the field perimeters (m) and the total farm area (ha).

5.5. Results

The results obtained at different scale are aggregated and show from the bottom level (crop) to up level (farm). Details on the distribution of indicators at different level are shown in Appendixes a, b, and c.

5.5.1. Crop level

The indicators were calculated separately for each crop and the results aggregated by crop type are shown in Table 5.8a (simple indicators) and 5.8b (cross indicators).

5.5.1.1. Corn

All farms monitored cultivate corn (Table 5.1). Corn is sown from the end of March to April, but, if it follows a winter forage crop, sowing is at the end of May. The number of seeds used is variable (61,000 to 80,000 seeds ha^{-1}) depending on the variety and destination (grain or silage).

The cropping systems are different according to the availability of manure, the final product obtained, and the use of PPA. The number of irrigations is from 2 to 5 per season, but 2 fields in RIC-POU were not irrigated in 2006 (water table depth was 0.8 – 1.2 m, and precipitations were sufficient).

In DAI and SWI farms, corn yield is high (Table 5.10); in the DAI farms, part of corn is ensilaged, so the average yield is higher than in other farms, but the GI is similar to other situation due to lower price of silage compared to grain.

In RIC-POU the farmer sowed a short-season variety (FAO class 200), with low productivity in order to anticipate the harvest and to avoid overlap with rice harvest. Corn yield in MIX is generally low (about 8 Mg DM ha^{-1}) due to the pedological condition, low input management, and low availability of water in 2006 during flowering that has reduced the production from 30 to 50%.

The PPA represent on average the 18% of VC in corn, the fertilizer 28 %, the fuel and lubricants 28%, while the seeds are 26% (Table 5.9). The economic indicators highlight the good performance of corn (Table 5.8a), especially in DAI-INT and in SWI farms, (Table 5.10), where the necessity of a large amount of animal food drive the farmers towards an intensive cultivation of corn. The strategy to obtain high GM in DAI-INT and SWI-EXT is through high productivity with a considerable economic investment (high VC), while the SWI-INT has reduced VC

Table 5.8a. – Average and standard deviation of indicators calculated for each crop

	CROPS	— corn —	— rice —	— w. wheat —	— barley —	permanent meadows	— triticale —	Italian ryegrass	— soybean —
	n. crops monitored	125	55	12	10	62	2	2	2
ECONOMIC INDICATORS									
VC	€ ha^{-1}	577 (119)	686 (73)	300 (71)	180 (52)	145 (17)	166 (1)	91 (0)	339 (0)
GI	€ ha^{-1}	1,568 (371)	2,063 (450)	1,181 (196)	1,108 (125)	876 (400)	757 (192)	552 (0)	1,231 (0)
GM	€ ha^{-1}	990 (383)	1,377 (470)	882 (185)	927 (157)	731 (384)	592 (192)	461 (0)	893 (0)
EE	–	2.8 (1.1)	3.1 (0.8)	4.1 (1.0)	6.7 (2.3)	5.8 (2.0)	4.6 (1.1)	6.1 (0.0)	3.6 (0.0)
NUTRIENT INDICATORS									
NS	kg N ha^{-1}	159 (134)	73 (41)	61 (84)	32 (54)	23 (57)	26 (26)	81 (22)	50 (0)
PS	kg P_2O_5 ha^{-1}	76 (144)	–12 (17)	8 (41)	7 (56)	17 (27)	8 (10)	37 (14)	–94 (0)
KS	kg K_2O ha^{-1}	140 (146)	65 (85)	188 (143)	67 (56)	–48 (34)	38 (26)	58 (31)	–95 (0)
ENERGY INDICATORS									
EnIN	GJ ha^{-1}	27.3 (5.5)	22.7 (3.3)	16.4 (5.0)	12.0 (2.7)	13.1 (4.9)	14.3 (0.2)	8.8 (0.7)	11.6 (0.0)
EnOUT	GJ ha^{-1}	220.3 (71.6)	138.2 (41.2)	156.0 (22.4)	164.3 (28.1)	139.3 (63.5)	165.9 (28.7)	89.9 (0.0)	100.9 (0.0)
EnG	GJ ha^{-1}	193.0 (72.9)	115.5 (40.8)	139.6 (19.9)	152.3 (28.8)	126.2 (58.9)	151.6 (28.8)	81.1 (0.7)	89.3 (0.0)
EnE	–	8.5 (3.7)	6.2 (1.8)	10.1 (2.6)	14.5 (4.9)	10.6 (2.5)	11.6 (2.2)	10.3 (0.9)	8.7 (0.0)
PPA INDICATORS									
LIa	TOX unit ha^{-1}	122.2 (85.2)	258.7 (145.9)	0.8 (0.6)	0.6 (0.6)	–	–	–	61.5 (0.0)
LIc	TOX unit ha^{-1}	1.5 (7.1)	6.2 (3.1)	47.7 (85.9)	0.0 (0.0)	–	–	–	3.1 (0.0)
LIe[1]	TOX unit ha^{-1}	9.0 (19.3)	11.4 (6.5)	0.4 (0.5)	0.5 (1.4)	–	–	–	1.8 (0.0)
LIf	TOX unit ha^{-1}	2.0 (5.5)	8.4 (4.3)	4.4 (8.0)	0.0 (0.0)	–	–	–	8.5 (0.0)
LIbe	TOX unit ha^{-1}	1.3 (3.7)	0.4 (0.8)	0.2 (0.4)	0.0 (0.0)	–	–	–	0.0 (0.0)
LIr[1]	TOX unit ha^{-1}	1.4 (0.6)	8.3 (5.7)	0.3 (0.3)	0.2 (0.7)	–	–	–	1.5 (0.0)
Lib[1]	TOX unit ha^{-1}	2.0 (3.6)	33.0 (29.9)	0.2 (0.3)	0.5 (1.6)	–	–	–	1.0 (0.0)
EEPa[1]	kg a.i. Pa ha^{-1}	3.3 (2.0)	2.8 (2.3)	0.2 (0.3)	0.1 (0.3)	–	–	–	7.1 (0.0)
EEPs	kg a.i. ha^{-1} d	51.2 (13.9)	131.1 (31.2)	36.5 (45.3)	0.7 (2.2)	–	–	–	158.0 (0.0)
EEPw	kg a.i. ha^{-1} d	0.33 (0.16)	0.18 (0.12)	0.39 (0.47)	0.02 (0.06)	–	–	–	0.46 (0.0)

[1] value multiplied by 10^3

For abbreviations see § 5.2.

Table 5.8b. – Average and standard deviation of cross indicators calculated for each crop

	CROPS	— corn —		— rice —		– w. wheat –		— barley —		permanent meadows		— triticale —		Italian ryegrass		– soybean –	
NUTRIENT EFFICIENCY																	
NE	kg DM kg^{-1} N_{IN}	33	(15)	34	(10)	36	(26)	37	(15)	19	(2)	70	(14)	25	(3)	85	(0)
PE	kg DM kg^{-1} $P_2O_{5\ IN}$	116	(156)	99	(33)	199	(466)	102	(54)	39	(6)	159	(31)	43	(5)	149	(0)
KE	kg DM kg^{-1} K_2O_{IN}	30	(15)	24	(12)	64	(172)	33	(18)	15	(1)	59	(13)	15	(1)	25	(0)
NUTRIENT WASTE																	
NW	kg $N_{surplus}$ kg^{-1} DM	16	(16)	11	(8)	7	(9)	4	(6)	1	(5)	4	(4)	15	(4)	12	(0)
PW	kg $P_2O_{5surplus}$ kg^{-1} DM	9	(18)	–1	(3)	1	(4)	2	(7)	2	(3)	1	(1)	7	(3)	–22	(0)
KW	kg $K_2O_{surplus}$ kg^{-1} DM	14	(18)	11	(15)	22	(17)	8	(8)	–8	(6)	5	(4)	11	(6)	–23	(0)
NW/GM	kg $N_{surplus}$ k€$^{-1}$	194	(219)	64	(54)	75	(96)	40	(58)	11	(53)	55	(62)	175	(49)	55	(0)
PW/GM	kg $P_2O_{5surplus}$ k€$^{-1}$	111	(254)	–8	(14)	13	(40)	13	(77)	24	(33)	17	(22)	81	(31)	–105	(0
KW/GM	kg $K_2O_{surplus}$ k€$^{-1}$	168	(256)	64	(83)	237	(179)	78	(78)	–96	(68)	75	(69)	126	(67)	–107	(0)
NW/EnG	kg $N_{surplus}$ GJ^{-1}	1.04	(1.14)	0.77	(0.65)	0.45	(0.59)	0.25	(0.36)	0.07	(0.31)	0.20	(0.21)	0.99	(0.28)	0.55	(0.00)
PW/EnG	kg $P_2O_{5surplus}$ GJ^{-1}	0.59	(1.21)	–0.08	(0.20)	0.07	(0.27)	0.08	(0.47)	0.14	(0.18)	0.06	(0.08)	0.47	(0.18)	–1.05	(0.00)
KW/EnG	kg $K_2O_{surplus}$ GJ^{-1}	0.90	(1.21)	0.79	(1.07)	1.43	(1.09)	0.49	(0.47)	–0.53	(0.37)	0.27	(0.23)	0.72	(0.39)	–1.07	(0.00)
ECONOMY AND ENERGY																	
VC/EnG	€ GJ^{-1}	3.5	(1.5)	6.6	(2.3)	2.2	(0.5)	1.3	(0.6)	1.3	(0.4)	1.1	(0.2)	1.1	(0.0)	3.8	(0.0)
GM/EnG	€ GJ^{-1}	5.4	(1.7)	13.0	(5.3)	6.3	(0.5)	6.1	(0.4)	5.6	(0.3)	3.9	(0.5)	5.7	(0.1)	10.0	(0.0)
GM/EnIN	€ GJ^{-1}	37	(16)	62	(20)	58	(20)	82	(28)	54	(16)	42	(13)	53	(4)	77	(0)

For abbreviations see § 5.2.

Table 5.9. – Average values of variable costs for corn in different farms

	DAI-INT	DAI-EXT	SWI-INT	SWI-EXT	RIC-POU	MIX	CER-RIC	All corn field
	€ ha^{-1}							
Plant protection agents	137	96	68	121	70	61	84	101
Fertilizers	116	117	0	191	313	167	310	163
Fuel								
Agronomic operations	40	45	57	43	60	39	77	48
Irrigations	22	27	30	30	8	37	19	26
Product transformation	38	37	87	53	46	55	110	56
Transports	7	1	1	1	2	1	3	2
Lubricants	28	29	30	26	23	28	34	28
Seeds	142	151	119	168	173	160	150	153
Total variable costs	530	503	391	635	695	547	786	577

Table 5.10. – Average values of indicators calculated for corn in different farms

FARM		DAI-INT	DAI-EXT	SWI-INT	SWI-EXT	RIC-POU	MIX	CER-RIC
Yield of harvested product [a]	Mg DM ha^{-1}	16.8	13.3	9.5	10.9	8.7	7.4	11.7
VC	€ ha^{-1}	530	503	391	635	695	547	786
GI	€ ha^{-1}	1,706	1,314	1,578	1,777	1,400	1,177	1,598
GM	€ ha^{-1}	1,176	811	1,187	1,142	705	630	812
EE	–	3.2	2.8	4.2	2.8	2.0	2.2	2.0
NS	kg N ha^{-1}	61	111	223	250	240	88	84
PS	kg P_2O_5 ha^{-1}	–28	49	252	74	301	–25	14
KS	kg K_2O ha^{-1}	98	124	173	220	288	–51	23
EnIN	GJ ha^{-1}	23.0	22.9	27.3	30.8	28.4	26.0	34.0
EnOUT	GJ ha^{-1}	305.0	252.0	181.2	205.5	164.7	141.5	223.4
EnG	GJ ha^{-1}	282.0	229.1	153.9	174.7	136.3	115.5	189.4
EnE	–	13.3	11.4	6.7	6.9	5.8	5.4	6.6

[a] grain or entire plant for silage;
For abbreviations see § 5.2.

(no inorganic fertilizers, minimum tillage and lower PPA application), still obtaining a good GM and EE (4.2). The CER-RIC farm has obtained a good GI but the high VC due to fertilizers (all nutrients are bought, with an average cost of 310 € ha^{-1}) and to the high amount of fuel consumed in the operations (average 209 € ha^{-1}) has reduced the GM.

The NS is high in corn (average 159 kg N ha^{-1}, Table 5.8a), especially in SWI farms: in SWI-EXT fertilizer application is not reduced when manure is applied, and this produces a high NS, while in SWI-INT the farmer applies only slurry at the maximum N rate (340 kg N ha^{-1}) allowed by the official manure nitrogen utilization plan (Regione Lombardia, 2006). Probably high ammonia volatilization occurs during manure spreading (no systems to reduce volatilization are applied), therefore a high manure amount is necessary to cover the crop requirements. The RIC-POU farmer concentrates on corn all manure produced on farm (on average 158, 285, and 164 kg ha^{-1} of N, P_2O_5 and K_2O, respectively), in order to avoid over fertilization in rice (sensible diseases caused by over-fertilization). In addition, mineral fertilizer is applied to corn and consequently the N input is very high compared to the crop uptake. Nutrient balances in corn are usually more correct in farms where manures are not applied (CER-RIC) or low amounts are used (MIX, DAIs).

In general, the P fertilization is more correct (lower PS), with the exception of SWI-INT and RIC-POU. Also KS is influenced by manure application: it is high where an elevated dose of manure is applied (DAIs, SWI, and RIC-POU). In these cases a reduction of K mineral fertilization should be possible, with the exception of SWI-INT that does not use mineral fertilizers. Differently to the N and P excess that have direct environmental impact (N leaching, water eutrophication), the K excess does not create direct environmental impact, but high surpluses are a waste of money and energy necessary for the production, transport and application of K mineral fertilizer. In MIX, a nutrient management plan is used, and when previous crops leave high P and K surpluses (residual soil fertility that is not included in the soil surface nutrient balance), the corn fertilization is reduced, providing a negative PS and KS.

The EnIN is usually high in corn (from 15.6 to 51.6 GJ ha^{-1}, average 27.3 GJ ha^{-1}; Table 5.8a), with large differences among farms (Table 5.10). Low values were found in DAI farms, due to relatively low energy embedded in nutrients (10.6 GJ ha^{-1}; data not shown) and low diesel consumption (the ensilage operation requires less energy compared to drying). High EnINs were found in SWI-EXT, RIC-POU and CER-RIC, where high nutrient amounts are applied (17.1, 15.2, and 14.6 GJ ha^{-1}, in SWI-EXT, RIC-POU, and CER-RIC, respectively), and high quantity of fuel is consumed (12 GJ ha^{-1}, in CER-RIC). The diesel consumed for agronomic operations, irrigations, drying, milling, silaging, and transport represents on average 28% of the EnIN, while the energy embedded in fertilizers is the 51% (42% for N fertilizers). The other energy consumptions are represented by seeds (12%), machinery (6%), lubricants (2%), and PPAs (2%).

The EnG is high in DAI farms (more DM is exported from fields, as silage is the preferred destination) and is low in MIX where the low yield has reduced the EnOUT. The EnE is on average good: about 6.8 in grain production and 14.6 in silage production (data not shown).

A large variability of LI and EEP was obtained in corn (Table 5.8a), depending on the a.i. used, but no clear relation with farm management was found. The 10 indicators used to describe PPA management do not highlight marked differences among farms (Table 5.11), probably because the PPA management is similar in all farms: usually only herbicides are used (1 or 2 applications, from 2 to 8 a.i.), while the insecticides are rarely used (only 6 fields in SWI-EXT were treated with insecticides in 2006). Seed dressing is normally applied, but in the 21% of the fields, untreated seeds were used.

Part of the variability is due to the fact that when farmers choose PPA they do not consider the toxicity of a.i., and in many cases they test different products among fields. Hence the indicators calculated in two fields in the same farm provide opposite evaluation, depending on the a.i. property. Usually the average LIs are low but a single a.i. could have a heavy potential effect on non-target organisms, providing LI higher than the average.

The insecticides have strong effects on LIc and LIf, while herbicides impact on LIa. The fumigants increase the LIc, but usually are not used

Table 5.11. – Average values of PPA indicators calculated for corn in different farms

	Units	DAI-INT	DAI-EXT	SWI-INT	SWI-EXT	RIC-POU	MIX	CER-RIC
LIa	TOX unit ha^{-1}	219	211	93	38	45	185	66
LIc	TOX unit ha^{-1}	0.7	0.3	0.4	0.3	2.8	0.9	11.0
LIe	TOX unit ha^{-1}	6.8	6.4	6.4	5.6	7.5	5.1	40.2
LIf	TOX unit ha^{-1}	0.9	0.6	0.6	2.1	6.4	1.1	4.8
LIbe	TOX unit ha^{-1}	0.0	0.4	0.0	0.1	5.8	0.0	8.2
LIr	TOX unit ha^{-1}	1.3	1.2	1.4	1.4	2.0	1.0	1.7
LIb	TOX unit ha^{-1}	1.4	1.3	1.1	1.4	3.5	1.1	7.6
EEPa	kg a.i. Pa ha^{-1}	3.4	1.2	1.5	4.0	4.9	4.4	3.7
EEPs	kg a.i. ha^{-1} d	61.5	56.7	44.5	40.9	56.5	45.2	62.1
EEPw	kg a.i. ha^{-1} d	0.4	0.3	0.3	0.4	0.4	0.1	0.3

For the abbreviations see § 5.2.

(only 2 fields received a soil fumigant application). The seed dressing has a small effect on LI, due to the low A.R. No fungicides were used.

The main potential effect of PPAs used in corn is on algae (LIa = 122; Table 5.8a) however, it should be considered that in these flat fields present in all the farms, the run-off is reduced, and probably small amounts of a.i. may reach rivers or lakes; reasonably the real pollution is lower than the potential risk described by the indicator.

The potential pressure on air is low, because the maximum vapor pressure in PPA used in corn is 6 mPa, with a corresponding EEPa of 5 10^{-3} kg a.i. Pa ha^{-1} for the single a.i. applied in a treatment. The maximum EEPa for a single crop is 6.6 10^{-3} kg a.i. Pa ha^{-1}. The DT_{50soil} is high only for products used for seed dressings (from 15 to 191 d), but the low A.R. reduces the EEPs of this type of products. The DT_{50soil} for the other a.i.s is low (from 2 to 45 d), but the high A.R. produces a noticeable EEPs (56 kg a.i. ha^{-1} d, for a single a.i. applied in a treatment). The K_{oc} has a high variability among a.i. (from 13 to over 10^5 kg a.i. ha^{-1} d), and the EEPw ranges from 0 to 0.9.

These simple indicators were used to provide cross indicators. The NE (Table 5.12) is high in DAI farms where a high yield (silage) is linked to a correct N application. In SWI, NE is low because a large part of N applied with slurry is probably volatilized, and it is necessary to apply large amounts of manure (SWI-INT) or to add mineral N (SWI-EXT). The two averages of NE are not significantly different (t test p= 0.43) in these two farms: this means that NE in grain corn production in SWI farms is the same independently from manure N applied. In RIC-POU the N applied with manure is high and less ammonia volatilization occurs for poultry manure (Sommer and Hutchings, 2001) compared to swine manure, however, this higher N availability is not well accounted in the nutrient management plan and an additional mineral fertilization is applied by farmer. The PE is bad for SWI-INT that use only manure that have and excess of P_2O_5 compared to N (Table 5.12). Also in RIC-POU the poultry manure contains an excess of P_2O_5 and K_2O; even so an extra useless mineral P and K fertilization is applied (about 70 – 80 kg P_2O_5 ha^{-1}; 160 – 180 kg K_2O ha^{-1}). The use of manure produces high values of nutrient waste indicators (NW, PW, and KW), especially when nutrients are applied with manure at high rates (SWIs and RIC-POU).

The good economic performance of corn contrasts with environmental quality, and the good GM does not compensate the nutrient surpluses produced. On average in corn the high GM is related to high nutrient surpluses: for each 1,000 € of GM, the corresponding surpluses are on average 194, 111, 168 kg ha^{-1} of N, P_2O_5 and K_2O, respectively (NW/GM; PW/GM; KW/GM; Table 5.8b). The DAI-INT is the farm that produces the lowest nutrient surplus per euro gained (Table 5.12).

Also EnG is not enough to compensate the environmental pressure of nutrient surplus (high value of NW/EnG, PW/EnG, KW/EnG; Table 5.8b). Only DAI farms and CER-RIC obtain low value of these indicators (Table 5.12). All these cross indicators related with nutrient balances provide generally a non good judgment: the high nutrient surpluses generally are not compensated by yield, GM and EnG, compared to other crops (Table 5.8b). The main problem usually arises from the use of high amount of manure: in order to dispose the manure produced in the farm, SWI-INT and RIC-POU apply high quantity of manure on corn, but the nitrogen applied is greater than N available for crop because part of N is lost during manure spread by ammonia volatilization and nitrate leaching, and this increases the NW. For SWIs and RIC-POU it is not possible to convert the grain production into silage production (obtaining probably better indicator values as in DAI farms), but the application of systems that reduce ammonia volatilization (for SWI farms) and the reduction of mineral fertilization (SWI-INT and RIC-POU) are suggested.

The cross indicators VC/EnG, GM/EnG, and GM/EnIN describe the relation among economic and energy management (Table 5.8b). An investment of about 3.5 € (VC/EnG) is necessary for each GJ^{-1} of energy gained, obtaining about 5.4 € of GM (GM/EnG). The economic surplus for each energy unit invested is on average 37.5 € GJ^{-1} (GM/EnIN). Farms with significant livestock (DAIs and SWIs) obtain the best performance in the indicators that describe the relation between economic and energy management: DAI farms have obtained the best performance in the conversion of economic input (VC) into energy

Table 5.12. – Average values of cross indicators calculated for corn in different farms

	FARM	DAI-INT	DAI-EXT	SWI-INT	SWI-EXT	RIC-POU	MIX	CER-RIC
NUTRIENT EFFICIENCY								
NE	kg DM kg^{-1} N_{IN}	52	48	21	23	19	29	32
PE	kg DM kg^{-1} $P_2O_{5\ IN}$	166	100	25	123	21	234	72
KE	kg DM kg^{-1} K_2O_{IN}	42	40	21	22	16	29	36
NUTRIENT WASTE								
NW	kg $N_{surplus}$ kg^{-1} DM	3.7	7.4	30.5	22.9	29.6	12.8	7.3
PW	kg^{-1} $P_2O_{5surplus}$ kg^{-1} DM	–1.9	3.8	32.0	6.4	37.0	–2.5	1.2
KW	kg^{-1} $K_2O_{surplus}$ kg^{-1} DM	7.3	8.7	25.1	20.2	35.9	–6.6	2.3
NW/GM	kg $N_{surplus}$ k€$^{-1}$	57	130	329	222	438	173	101
PW/GM	kg $P_2O_{5surplus}$ k€$^{-1}$	–23	76	331	58	537	–11	29
KW/GM	kg $K_2O_{surplus}$ k€$^{-1}$	91	164	279	194	528	–87	31
NW/EnG	kg $N_{surplus}$ GJ^{-1}	0.21	0.45	2.06	1.47	1.93	0.86	0.47
PW/EnG	kg $P_2O_{5surplus}$ GJ^{-1}	–0.12	0.23	2.13	0.44	2.40	–0.13	0.07
KW/EnG	kg $K_2O_{surplus}$ GJ^{-1}	0.44	0.54	1.71	1.30	2.33	–0.42	0.14
ECONOMY AND ENERGY								
VC/EnG	€ GJ^{-1}	2.1	2.3	3.1	3.7	5.4	5.0	4.2
GM/EnG	€ GJ^{-1}	4.5	3.6	7.3	6.5	5.1	5.4	4.4
GM/EnIN	€ GJ^{-1}	51	36	44	38	25	24	24

For abbreviations see § 5.2.

output (average VC/EnG equal to 2.1 and 2.3 € GJ^{-1}, for DAI-INT and DAI-EXT, respectively; Table 5.12). The best GM/EnGs are obtained in SWI farms (average GM/EnG equal to 7.3 and 6.5 € GJ^{-1}, for SWI-INT and SWI-EXT, respectively; Table 5.12). The best conversion of energy inputs into economic gain (GM/EnIN) is obtained in intensive farms (average GM/EnIN equal to 51 and 44 € GJ^{-1} in DAI-INT and SWI-INT, respectively; Table 5.12).

5.5.1.2. Rice

Rice cultivation is concentrated in the South West area of the PASM, where the rice cropping systems are a tradition. Only two farms monitored cultivate rice (RIC-POU and CER-RIC; Table 5.1). The cropping systems are similar: no manure application, correct mineral fertilizer application, flooding irrigation system, intensive use of PPAs (herbicides in springtime and fungicides in summer). Sowing was from half April to end of May (in RIC-POU a false seed bed preparation is done in some fields in order to reduce weed pressure, especially red rice (*Oryza sativa* L.). The amount of seeds used is variable depending on the variety and farmer management (from 165 to 245 kg ha^{-1}); all rice seeds used in CER-RIC are treated with seed dressing, while in RIC-POU only in 27 fields, seed dressing was used. The average grain yield is similar in the two farms (5.3 and 5.9 Mg DM ha^{-1} in RIC-POU and CER-RIC, respectively, harvested from the end of September to the end of October). The CER-RIC farmer has dedicated five fields to the production of seed instead of the normal human food. In these fields the yield is lower but the price of product is higher than what normally obtained.

The VCs are high (average 686 € ha^{-1}, Table 5.8a; 705 and 618 € ha^{-1} in RIC-POU, and CER-RIC, respectively; data not shown), with a maximum of 878 € ha^{-1} in RIC-POU, and a minimum of 517 € ha^{-1} in CER-RIC.

The PPAs represent on average 44% of VC, the fertilizers 28%, fuel and lubricants 16%, while the seeds are 12%. Compared to other crops, rice has the highest GI and GM (Table 5.8a), with average GM of 1,284 € ha^{-1} and 1,717 € ha^{-1} in RIC-POU, and CER-RIC, respectively (data not shown). The average EE is similar to corn (3.1; Table 5.8a), with big difference between RIC-POU, and CER-RIC (2.9 and 3.8, respectively; data not shown).

The nutrient balances are close to zero (no manures are used), with moderated NS (average 73 kg N ha^{-1}; Table 5.8a), a very good PS in CER-RIC (2 kg P_2O_5 ha^{-1}; data not shown), and a small deficit in POU-RIC (–16 kg P_2O_5 ha^{-1}; data not shown). The KS is high in CER-RIC (average 114 kg K_2O ha^{-1}; data not shown), but K fertilization could be high in soil with poor K content.

The average EnIN is lower than in corn (22.7 GJ ha^{-1}; Table 5.8a), due to smaller application of nutrients (6.6 GJ ha^{-1}; data not shown). The energy required for nutrients represents the main component of EnIN (29%; 27% for N fertilizers), while the energy for seeds and diesel consumed are 27% and 23%, respectively. PPAs energy is relevant (11%). The other energy consumptions are represented by machinery (5%) and lubricants (1%).

The EnOUT is moderate in rice (average: 138.2 GJ ha^{-1}; Table 5.8a); in particular, it is lower in CER-RIC (108.1 GJ ha^{-1}) than in RIC-POU (146.5 GJ ha^{-1}; data not shown), because in RIC-POU, straw produced is exported from field in 60% of cases. The EnG and EnE are low for grain production (average 83.1 GJ ha^{-1}, and 4.7, respectively; data not shown), but are increased when straw is exported from field (average 152.0 GJ ha^{-1} and 7.8, respectively; max EnG in RIC-POU: 196.8 GJ ha^{-1}; data not shown).

The potential PPA impact is relevant compared to other crops: high LIa and LIf are found, while LIe and LIbe are comparable to what obtained for corn (Table 5.8a). The EEP indicators highlight a high potential risk for soil (average EEPs equal to 131 kg a.i. ha^{-1} d; Table 5.8a). No marked differences are found between the two farms for PPA management.

The nutrient efficiency is comparable with values obtained in corn: the average NE (34 kg DM kg^{-1} N_{IN}; Tab 5.8b) is similar to average NE in corn, while PE and KE are lower than average value in corn. The nutrient waste indicators are low, because nutrient balances are close to zero, due to the high attention of farmer in the use of mineral fertilizers, avoiding the manure application for this sensible crop. The nutrient waste - economic gain indicators (NW/GM, PW/GM, and KW/GM; Table 5.8b) provide very good values. On the other hand, the nutrient waste - energy gain indicators (NW/EnG, PW/EnG, and KW/EnG; Table 5.8b) provide values comparable to corn values: the low nutrient surplus is related to low yield, and corresponding EnG. Some differences arise when straw is exported from field: NW/EnG and KW/EnG are 0.5 kg $N_{surplus}$ GJ^{-1} and 0.2 kg $K_2O_{surplus}$ GJ^{-1}, respectively, when straw is exported, and 1.0 $N_{surplus}$ and GJ^{-1}, 1.4 kg $K_2O_{surplus}$ GJ^{-1} when only grain is exported from field (data not shown). Both VC/EnG, and GM/EnG are high, especially when straw is not harvested. High average values of GM/EnI are obtained in the two managements.

5.5.1.3. Winter cereals

The management of winter wheat and barley is similar in the four farms where these two crops are cultivated (Table 5.1). Sowing is in October or November, depending on pedoclimatic conditions. The amount of seeds used are on average 220 and 180 kg ha^{-1} for winter wheat and barley, respectively; all winter wheat seeds are treated with seed dressing, while only in 2 fields seed dressing is used for barley. The yield is on average 5.2 and 5.0 Mg DM ha^{-1} for winter wheat and barley, respectively (harvested at the end of June or in July). The straw is harvested in all fields. Fertilization is extremely variable: there are cases where the crop is not fertilized (2 fields with barley in MIX) to use the soil fertility left from previous crop is sufficient for barley, and crops that receive autumn (manure or mineral fertilizers before plowing) and/or spring fertilization (mineral nitrogen fertilization) depending on the crop necessity. The use of PPAs is low compared to corn and rice (on a total of 5 field with winter wheat in CER-RIC, 3 were treated with insecticide and fungicide in spring, in DAI farms both winter wheat and barley were treated with herbicides, and in MIX winter wheat and two fields with barley were treated with herbicide).

The VCs are low, especially in barley (180 € ha^{-1}; Table 5.8a); VCs are strongly influenced by fertilizer costs (from 0 to 142 € ha^{-1}), that represent 35% and 19% of VC in winter wheat and barley, respectively, while the seeds cost is 31% and 43%. There are no post harvest transformations of the products, and the fuel, lubricants and PPA costs are about the 20%, 6%, and 10%, respectively for both crops (data not shown). The GI is on average low (1,181 and 1,108 € ha^{-1} for winter wheat and barley, respectively; Table 5.8a), but satisfactory GMs are obtained (average 882 and 927 € ha^{-1} for winter wheat and barley, respectively; Table 5.8a), with excellent maxima: 1,376 and 1,068 € ha^{-1} (data not shown). The EE is high (4.1 and 6.7, for winter wheat and barley, respectively; Table 5.8a) especially in barley that has lower VC.

The NS and PS are on average low, while KS is elevated in winter wheat (188 kg K_2O ha^{-1}; Table 5.8a), probably in order to obtain strong steam and leaf, avoiding pest diseases. in some cases surpluses are elevated (maxima of 173 kg N ha^{-1}, 159 kg P_2O_5 ha^{-1}, and 315 kg K_2O ha^{-1}, for NS, PS, and KS, respectively; data not shown) or very low (minima of –73 kg N ha^{-1}, –88 kg P_2O_5 ha^{-1}, and –113 kg K_2O ha^{-1}, for NS, PS, and KS, respectively; data not shown). Usually the fertilization plans for these crops are related to previous crop: a reduction of nutrient application is introduced in order to exploit the soil nutrient availability left by previous crop (usually corn).

The winter wheat cultivated in CER-RIC is of a bread-making cultivar and it has probably a nutrient concentration higher than the average tabbed values. Therefore the crop uptake was probably underestimated and corresponding NS and KS overestimated (132 kg N ha^{-1} and 232 kg K_2O ha^{-1}, respectively; data not shown).

The EnIN is low because irrigation and drying are not necessary, and nutrient application is low. The seeds are the most important energy input (average 44% of EnIN), the nutrients are 28%, while fuel, machinery and lubricants represent the 19%, 2%, and 6% of EnIN (data not shown).

The EnOUT (average 156.0 and 164.3 GJ ha^{-1}, for winter wheat and barley, respectively; Table 5.8a) and corresponding EnG (average 139.6 and 152.3 GJ ha^{-1}, for winter wheat and barley, respectively; Table 5.8a) are moderate, but a good EnE is obtained in both crops (10.1 and 14.5 for winter wheat and barley, respectively; Table 5.8a).

The potential impact of PPA is low as demonstrated by low values of corresponding indicators (Table 5.8a) with the exception of the crops treated with insecticide and fungicide that have a high environmental toxicity.

The nutrient efficiency (NE, PE, and KE) is slightly better than for other cereals monitored (corn and rice), due to low nutrient inputs and the exploitation of the soil fertility left by previous crop.

All indicators related to nutrient waste provide good results for these crops, with the exception in winter wheat, where all cross indicators related to KS (KW, KW/GM, and KW/EnG) are high, because the K_2O apparently is often used in excess.

The good relation among economic and energy indicators is well described by the three cross indicators that highlight the low VC necessary to obtain an energy unit, especially in barley (average VC/EnG: 2.2 and 1.3 € GJ^{-1}, for winter wheat and barley, respectively; Table 5.8b), and the corresponding high GM obtained per energy unit, higher than values obtained for corn (6.3 and 6.1 € GJ^{-1}, for winter wheat and barley, respectively; Table 5.8b). The GM obtained per EnIN unit for winter wheat (average GM/EnIN: 58 € GJ^{-1}; Table 5.8b) is comparable to values obtained in rice, while it is very high in barley (82 € GJ^{-1}; Table 5.8b).

In case of low input systems (reduction of energy from fuel and fertilizers, water for irrigation, increase of labor cost, etc), barley and winter wheat could be an economic alternative to corn and rice: despite the low GI, they have a noticeable GM and are easier to cultivate compared to corn and rice (less agronomic operations and agrochemical inputs). In the studied area water irrigation availability in summer is lower than optimum crop requirement and in some cases the farmers have not water enough. The fact that winter wheat and barley do not require irrigation (a time and money and energy consuming operation) and the increase of winter wheat price occurred in 2007, could drive the choices of farmers towards the partial replacement of rice and corn with winter wheat and barley.

5.5.1.4. Meadows and other fodder crops

The MIX and DAI farms dedicated a large area to fodder production (Italian ryegrass, triticale, and permanent meadows) in order to produce the feed required by cow breeding. The Italian ryegrass and triticale are intercrops: they are sown in October after corn and harvested in the first ten days of May as hay (Italian ryegrass), or in the end of May as silage (triticale), with average yields of 5.2 and 9.2 Mg DM ha^{-1} for Italian ryegrass and triticale, respectively. For both crops, manure was spread before plowing, and no mineral fertilizer was applied.

In permanent meadows, the number of cuts per year ranges from 3 to 5 (from May to October); in 2006, only 2 harvests were possible in MIX because the irrigation water was not available in June and July. The average annual yield is 8.5, 5.2, and 13.2 Mg DM ha^{-1} in MIX, DAI-EXT, and DAI-INT, respectively (data not shown). The low yield in DAI-EXT is due to low nutrient inputs and to the presence of many hedge-rows and trees around and inside fields (large part of the farm is inside a park of an historical country seat). Hay is the most common product of permanent meadows, but in some cases the grass is used fresh (the earliest and latest cuts in MIX) or ensilaged (part of the earliest and latest cut in DAI farms). Solid manure is applied in autumn/winter (MIX and DAI-INT) and high amounts of slurry are used in the end of winter (DAI farms). In MIX, mineral N and K fertilizers were used because the manure available was not enough. After each cut, a low amount of slurry is applied in DAI farms. The irrigations are from 4 to 8 per year, from June to September.

No PPAs are used in fodder crops.

The VCs, GI, and GM for fodder crops are very low (Table 5.8a). Diesel and lubricants consumptions represent the entire VC for permanent meadows, while they represent the 44 and 54% of the VC in triticale and Italian ryegrass, respectively (other VCs are the seeds). To evaluate the animal economic performance, the GI and GM of Italian ryegrass and triticale have to be added to the corresponding value of previous/next corn, forming an extremely interesting double cropping system. The true economic performance of fodder crops would need to be evaluated in conjunction with the income of the animal breeds: the data reported here show that. in the studied area the cultivation of fodder without a joint livestock is not sustainable from the economic point of view. It is sustainable in marginal areas where the land is not very expensive and productive.

The high uncertainty related to i) the evaluation of nutrient concentrations in the manure, ii) the yield estimation and iii) the variability of flora composition in permanent meadows (inducing different nutrient concentrations) increases the uncertainty in the evaluation of nutrient surpluses. the data presented suggest that nutrient management in fodder crops is correct, with low surpluses (Table 5.8a). The average NS in DAI-INT is relatively high (106 kg N ha^{-1}; data not shown) due to high manure application, while permanent meadows in DAI-EXT and MIX have a K_2O deficit (average values: –64 and –62 kg K2O ha^{-1}, respectively; data not shown).

Like VC, EnIN is low, while the EnOUT in permanent meadows and triticale is comparable to EnOUT in rice and barley, respectively. The EnG is good in triticale with average value over 150 GJ ha^{-1}.

The nutrient efficiency of permanent meadows is low (19 kg DM kg^{-1} N_{IN}, 39 kg DM kg^{-1} $P_2O_{5\ IN}$, and 15 kg DM kg^{-1} K_2O_{IN} for NE, PE, and KE, respectively) compared to other crops. The cross indicators related to nutrient surpluses highlight the good compromise among nutrient waste and production (yield, GM, and EnG).

In fodder crops the VC per energy unit gained is lower than all other crops (Table 5.8a), and the GM/EnG in permanent meadows and Italian ryegrass is similar to value obtained in corn.

5.5.1.5. Soybean

Soybean was cultivated only in CER-RIC in order to break the continuous crop of rice, reducing weed pressure, especially the red rice infestation, controlled with difficulty by PPA. The sowing is in the end of April (85 kg ha^{-1}; data not shown), and the harvest around the half of October (average yield: 4.2 Mg DM ha^{-1}; data not shown). Three herbicide treatments were applied: the first during sowing, the second one month later, and the last in the end of May. A single irrigation (first ten days of July) was necessary, and no fertilizers were applied.

VCs are low (339 € ha^{-1}; Table 5.8a), mainly de to PPAs (50% of VC; data not shown). Other costs are represented by gasoline (30%), seeds (13%), and lubricants (8%). Soybean is not economically profitable if the effect of this crop on the next is not considered: the GM is low compared to corresponding value of rice (893 € ha^{-1}; Table 5.8a), but the cultivation of soybean reduces the seed bank in the soil, increasing the yield for the subsequent rice.

The necessary N is provided by biological fixation, and the P and K requirements are covered by the exploitation of soil nutrients left in the soil by previous crops, hence the PS and KS are negative.

The EnIN and EnOUT for this crop are very low (11.6 GJ ha^{-1}; Table 5.8a), because no fertilizers are required. 50% of EnIN is due to diesel, while seeds represent 29% of EnIN. Other energy inputs are related to machinery (12%), lubricants (4%) and PPA (4%; data not shown). The EnOUT is low (100.9 GJ ha^{-1}; Table 5.8a), even if soybean grain has the highest energy content (23.9 MJ kg DM^{-1}; Table 5.5).

The LIs are generally lower than corresponding values obtained in corn and rice. Only for LIf, soybean has values comparable to rice (8.5; Table 5.8a), and for LIr comparable to corn (1.5; Table 5.8a). The EEPs values obtained for soybean are higher than corresponding values obtained in corn and rice.

The nutrient efficiency indicators are usually higher than corresponding values obtained for other crops, because only the nutrients provided by previous crop residues were used as nutrient inputs (the N biological fixation and nutrient left in the soil from the previous crop are not included in nutrient inputs).

The average VC/EnG (3.8 € GJ^{-1}; Table 5.8b) is similar to average value obtained for corn, but the GM/EnG (10.0 € GJ^{-1}; Table 5.8b) is higher than corresponding average value obtained in corn, even if it is lower then average value in rice.

The distribution of indicators grouped by crop type is shows in Appendix a.

5.5.2. Field level

In the monitored fields, six types of crop combinations are identified: corn and other crops (barley, winter wheat, Italian ryegrass, and triticale; COf), continuous corn (Cf), rice and other crops (soybean or winter wheat; ROf), continuous rice (Rf), permanent meadows (PMf), and winter cereal (barley and wheat; CEf).

The aggregation of indicators at field level has usually produced smoothed values in mixed combination of crop (CCf, and ROf) compared to corresponding values obtained for single crop (Table 5.13a, 5.13b).

As compared with corresponding values obtained in Cf, the alternation of corn with other winter crops (COf) has reduced the GM (840 € ha^{-1} yr^{-1}; Table 5.13a), but also the nutrient surpluses (NS, PS, KS), maintaining similar nutrient efficiency (NE, PE, KE). Therefore the values of nutrient waste indicators are lower in COf than in Cf (Table 5.13b). Also the EnGs were reduced (average 173.5 GJ ha^{-1} yr^{-1};

Table 5.13a), maintaining similar EEs and EnEs. The PPA potential risk indicators are similar in COf and Cf. The nutrient waste indicators show lower value in COf than corresponding values in Cf, while the relations among economic and energy indicators are similar.

The CS is higher in COf than in Cf, due to the presence of different crops with different pathogens, pests and weeds which are contained by the introduction of crops with different biological cycling and management. The SC is higher in COf (average: 0.48; Table 5.13a) than in Cf (average: 0.35), due to i) longer growing season of winter wheat and barley (on average 210 and 199 d, respectively; data not shown) compared to corn, and ii) the introduction of intercrops (triticale and Italian ryegrass) between corn cultivated in two years. The balance of OC in Cf is generally satisfactory (average: 6.7; Table 5.13b) because many corns receive high amounts of animal manure, and leave high amounts of straw, usually not harvested. Low values of OCI are obtained in fields where low amounts of manures were applied in the two year monitored (min: 3.6; data not shown). In COf, the OCI is low (average: 4.8) because winter wheat and barley straws are always harvested and low amounts of residues are by silage corn, Italian ryegrass and triticale, which in these cropping systems are harvested for silage as whole-plant.

The rotation of rice with soybean and winter wheat (ROf), has not reduced to much the average GM of the fields, compared to Rf (1,270 € ha^{-1} yr^{-1} and 1,360 € ha^{-1} yr^{-1}, for ROf and Rf, respectively; Table 5.13a), but has increased the EE (3.8 and 3.0 for ROf and Rf, respectively). The EnGs obtained in ROf are lower than corresponding values in Rf. The potential risk related to the use of PPAs is not significantly reduced, because large amount PPAs were used in soybean and winter wheat cultivated in these fields.

The nutrient waste and efficiency indicators are similar in the two types of crops combination (Table 5.13b). The CS is higher in ROf than in Rf (4.1 and 1, respectively; Table 5.13b). The sequences monitored in 2004 – 2007 were rice-rice-soybean-rice (CS=2.7) and corn-winter wheat-rice-soybean (CS=6.9). The introduction of one soybean crop in the sequence was not sufficient to compensate the low score for rice-rice combination. The SC is very low in Rf due to the short biological cycle of rice (on average the soil is covered by the crop for 122 d), and relatively high in ROf, due to the longer cycle of soybean (153 d) and winter wheat (204 d). The lack of manure applications, the exportation of straw (winter wheat and part of rice) and the low production of crop residues (soybean) have reduced the soil OC inputs, hence low values of OCI in both crop sequences were found (average OCI equal to 4.3 and 2.1 in Rf and ROf, respectively; Table 5.13b).

The CEf have on average low values for the economic indicators (Table 5.13a) and high nutrient efficiencies (Table 5.13b), but the negative nutrient surpluses, which involve in a soil mineral nutrients depletion, produce negative values for the nutrient waste indicators. Energy indicators are low, with a poor EnG (116.7 GJ ha^{-1} yr^{-1}; Table 5.13a), but the GMs obtained per EnG unit are higher then corresponding values obtained in Cf (average GM/EnG equal tot 6.5 € ha^{-1} yr^{-1}; Table 5.13b). The low PPAs applications in CEf have provided low values of LI and EEP indicators. The CS is low because from the agronomic point of view, winter wheat and barley combinations are not recommended, due to similar biological cycle, nutrient requirement, weeds, and diseases. The SC is elevated, and it covers the most sensible period for the risk of nitrogen leaching (autumn and winter). The OCI is very low, because manures are not applied or are applied at low amounts, straw is always exported, and low crop residue productions are estimated based on low yields.

More details on variability of indicators calculated at field level are shown in Appendix b.

5.5.3. Farm level

The average weighted value for indicators calculated in each field are shown in Tables 5.14a, b, c.

The DAI-EXT and MIX have poor GM (515 and 791 € ha^{-1} yr^{-1}, respectively; Table 5.14a), while the other farms obtained similar GM (from 1,147 to 1,264 € ha^{-1} yr^{-1}), despite the VC are very different (average farm VC range from 407 to 704 € ha^{-1} yr^{-1} in DAI-INT and RIC-POU, respectively; Table 5.14a). Hence higher economic inputs do not correspond to high economic gains.

Table 5.13a. – **Average and standard deviation of indicators calculated for each field**

	Unit	— COf —		— Cf —		— ROf —		— Rf —		— PMf —		— CEf —	
	n. fields monitored	21		50		3		24		31		2	
ECONOMIC INDICATORS													
VC	€ ha^{-1} yr^{-1}	445	(118)	583	(109)	466	(34)	692	(45)	145	(17)	188	(98)
GI	€ ha^{-1} yr^{-1}	1,284	(385)	1,648	(210)	1,736	(383)	2,052	(389)	876	(400)	951	(574)
GM	€ ha^{-1} yr^{-1}	840	(303)	1,065	(234)	1,270	(414)	1,360	(389)	731	(384)	763	(476)
EE	–	2.9	(0.6)	2.9	(0.7)	3.8	(1.1)	3.0	(0.6)	5.8	(2.0)	4.9	(0.5)
NUTRIENT INDICATORS													
NS	kg N ha^{-1} yr^{-1}	75	(37)	179	(95)	55	(9)	75	(34)	23	(57)	–18	(26)
PS	kg P_2O_5 ha^{-1} yr^{-1}	7	(40)	96	(122)	–34	(7)	–12	(13)	17	(27)	–28	(51)
KS	kg K_2O ha^{-1} yr^{-1}	68	(84)	178	(87)	33	(6)	65	(54)	–48	(34)	–63	(25)
ENERGY INDICATORS													
EnIN	GJ ha^{-1} yr^{-1}	22.1	(6.0)	27.8	(3.9)	18.9	(2.3)	22.6	(2.5)	13.1	(4.9)	10.7	(7.2)
EnOUT	GJ ha^{-1} yr^{-1}	195.6	(98.2)	226.6	(65.5)	116.0	(30.5)	140.2	(40.3)	139.3	(63.5)	127.4	(77.7)
EnG	GJ ha^{-1} yr^{-1}	173.5	(93.3)	198.8	(67.5)	97.1	(28.1)	117.5	(40.1)	126.2	(58.9)	116.7	(70.6)
EnE	–	8.5	(2.4)	8.5	(3.4)	6.1	(0.8)	6.3	(1.8)	10.6	(2.5)	12.2	(0.9)
PPA INDICATORS													
LIa	TOX unit ha^{-1} yr^{-1}	106.5	(63.3)	108.1	(80.9)	144.5	(16.6)	259.3	(135.1)	–		0.3	(0.4)
LIc	TOX unit ha^{-1} yr^{-1}	15.3	(37.5)	1.2	(4.2)	4.1	(0.7)	6.2	(2.5)	–		0.0	(0.0)
LIe[1]	TOX unit ha^{-1} yr^{-1}	7.7	(16.7)	8.0	(10.9)	11.0	(6.8)	10.8	(3.5)	–		1.1	(1.6)
LIf	TOX unit ha^{-1} yr^{-1}	2.3	(4.3)	2.1	(4.2)	7.6	(2.1)	8.3	(4.3)	–		0.0	(0.0)
LIbe	TOX unit ha^{-1} yr^{-1}	1.1	(3.1)	1.2	(2.5)	0.7	(0.6)	0.4	(0.6)	–		0.0	(0.0)
LIr[1]	TOX unit ha^{-1} yr^{-1}	0.8	(0.5)	1.4	(0.4)	3.6	(1.3)	8.5	(4.1)	–		0.5	(0.7)
Lib[1]	TOX unit ha^{-1} yr^{-1}	1.5	(3.1)	1.9	(2.1)	14.2	(8.3)	33.3	(20.1)	–		1.2	(1.8)
EEP[1]a	kg a.i. Pa ha^{-1} yr^{-1}	2.3	(0.5)	3.2	(1.4)	3.4	(2.0)	2.9	(2.3)	–		0.3	(0.4)
EEPs	kg a.i. ha^{-1} d yr^{-1}	40.6	(24.6)	50.4	(10.8)	132.3	(24.7)	131.1	(26.2)	–		2.0	(2.8)
EEPw	kg a.i. ha^{-1} d yr^{-1}	0.22	(0.22)	0.36	(0.1)	0.37	(0.14)	0.19	(0.11)	–		0.05	(0.06)

[1] value multiplied by 10^3

For abbreviations see § 5.2.

Table 5.13b. – Average and standard deviation of cross and soil management indicators calculated for each field

	Unit	— COf —		— Cf —		— ROf —		— Rf —		— PMf —		— CEf —	
NUTRIENT EFFICIENCY													
NE	kg DM kg^{-1} N_{IN}	34	(10)	30	(15)	41	(5)	33	(7)	19	(2)	73	(47)
PE	kg DM kg^{-1} $P_2O_{5\ IN}$	101	(51)	76	(59)	89	(14)	95	(21)	39	(6)	252	(271)
KE	kg DM kg^{-1} K_2O_{IN}	30	(16)	27	(12)	22	(8)	22	(5)	15	(1)	81	(114)
NUTRIENT WASTE													
NW	kg $N_{surplus}$ kg^{-1} DM	7	(3)	17	(11)	10	(5)	11	(8)	1	(5)	–5	(6)
PW	kg $P_2O_{5surplus}$ kg^{-1} DM	0	(5)	10	(14)	–6	(3)	–1	(2)	2	(3)	–2	(6)
KW	kg $K_2O_{surplus}$ kg^{-1} DM	4	(13)	17	(11)	6	(3)	11	(11)	–8	(6)	–10	(2)
NW/GM	kg $N_{surplus}$ $k€^{-1}$	91	(39)	186	(140)	48	(20)	62	(40)	11	(53)	–43	(60)
PW/GM	kg $P_2O_{5surplus}$ $k€^{-1}$	5	(56)	113	(181)	–30	(13)	–8	(11)	24	(33)	–20	(54)
KW/GM	kg $K_2O_{surplus}$ $k€^{-1}$	50	(166)	189	(156)	28	(12)	55	(54)	–96	(68)	–90	(23)
NW/EnG	kg $N_{surplus}$ GJ^{-1}	0.46	(0.20)	1.07	(0.74)	0.61	(0.24)	0.77	(0.62)	0.07	(0.31)	–0.27	(0.38)
PW/EnG	kg $P_2O_{5surplus}$ GJ^{-1}	0.01	(0.30)	0.66	(0.90)	–0.38	(0.16)	–0.08	(0.16)	0.14	(0.18)	–0.13	(0.35)
KW/EnG	kg $K_2O_{surplus}$ GJ^{-1}	0.26	(0.84)	1.07	(0.72)	0.36	(0.14)	0.79	(0.86)	–0.53	(0.37)	–0.58	(0.14)
ECONOMY AND ENERGY													
VC/EnG	€ GJ^{-1}	3.0	(1.0)	3.2	(1.2)	5.1	(1.6)	6.6	(2.3)	1.3	(0.4)	1.7	(0.2)
GM/EnG	€ GJ^{-1}	5.1	(0.9)	5.7	(1.5)	13.0	(0.5)	12.5	(4.5)	5.6	(0.3)	6.5	(0.1)
GM/EnIN	€ GJ^{-1}	38	(10)	39	(11)	66	(13)	60	(15)	54	(16)	73	(4)
SOIL MANAGEMENT													
CS	1–10	4.6	(1.2)	2.0	(0.6)	4.1	(2.4)	1.0	(0.2)	10.0	(0.0)	3.5	(3.1)
SC	0–1	0.48	(0.14)	0.35	(0.04)	0.41	(0.04)	0.33	(0.03)	1.00	(0.00)	0.45	(0.25)
OCI	1–10	4.8	(2.8)	6.7	(1.7)	2.1	(0.4)	4.3	(1.8)	10.0	(0.0)	1.4	(0.4)

For abbreviations see § 5.2.

Table 5.14a. – Average values (and ranking) of economic, nutrient energy and plant protection agents indicators calculated for each farm

	FARM	DAI-INT	DAI-EXT	SWI-INT	SWI-EXT	RIC-POU	MIX	CER-RIC
ECONOMIC INDICATORS								
VC	€ ha^{-1} yr^{-1}	407 (III)	214 (I)	427 (IV)	634 (VI)	704 (VII)	343 (II)	614 (V)
GI	€ ha^{-1} yr^{-1}	1,671 (IV)	729 (VII)	1,621 (V)	1,781 (III)	1,925 (I)	1,134 (VI)	1,860 (II)
GM	€ ha^{-1} yr^{-1}	1,264 (I)	515 (VII)	1,194 (IV)	1,147 (V)	1,221 (III)	791 (VI)	1,246 (II)
EE	–	5.1 (I)	3.8 (III)	3.8 (II)	2.8 (VI)	2.7 (VII)	3.5 (IV)	3.1 (V)
NUTRIENT INDICATORS								
NS	kg N ha^{-1} yr^{-1}	74 (III)	23 (I)	218 (VI)	254 (VII)	91 (V)	49 (II)	88 (IV)
PS	kg P_2O_5 ha^{-1} yr^{-1}	–9 (I)	23 (IV)	250 (VII)	78 (VI)	25 (V)	–4 (II)	2 (III)
KS	kg K_2O ha^{-1} yr^{-1}	47 (II)	–20 (I)	169 (VI)	223 (VII)	79 (IV)	57 (III)	81 (V)
ENERGY INDICATORS								
EnIN	GJ ha^{-1} yr^{-1}	22.9 (III)	13.1 (I)	27.9 (VI)	31.0 (VII)	23.0 (IV)	17.8 (II)	26.6 (V)
EnOUT	GJ ha^{-1} yr^{-1}	294.4 (I)	123.6 (VII)	186.3 (III)	206.1 (II)	150.1 (VI)	152.8 (V)	156.6 (IV)
EnG	GJ ha^{-1} yr^{-1}	271.4 (I)	110.5 (VII)	158.5 (III)	175.1 (II)	127.1 (VI)	134.9 (IV)	130.1 (V)
EnE	–	12.7 (I)	9.0 (III)	6.7 (V)	6.7 (IV)	6.6 (VI)	9.0 (II)	5.8 (VII)
PPA INDICATORS								
LIa	TOX unit ha^{-1} yr^{-1}	132.8 (V)	40.5 (II)	96.7 (IV)	35.4 (I)	237.6 (VII)	78.4 (III)	146.5 (VI)
LIc	TOX unit ha^{-1} yr^{-1}	0.4 (V)	0.0 (I)	0.3 (III)	0.2 (II)	6.0 (VI)	0.4 (IV)	26.7 (VII)
LIe[1]	TOX unit ha^{-1} yr^{-1}	4.3 (III)	1.2 (I)	8.2 (V)	5.6 (IV)	9.5 (VI)	2.2 (II)	25.0 (VII)
LIf	TOX unit ha^{-1} yr^{-1}	0.6 (III)	0.1 (I)	0.6 (IV)	2.4 (V)	8.1 (VI)	0.5 (II)	8.6 (VII)
LIbe	TOX unit ha^{-1} yr^{-1}	0.0 (II)	0.1 (IV)	0.0 (III)	0.1 (V)	0.9 (VI)	0.0 (I)	2.9 (VII)
LIr[1]	TOX unit ha^{-1} yr^{-1}	0.9 (III)	0.2 (I)	1.8 (V)	1.4 (IV)	8.1 (VII)	0.4 (II)	4.0 (VI)
Lib[1]	TOX unit ha^{-1} yr^{-1}	1.1 (III)	0.3 (I)	1.4 (V)	1.4 (IV)	30.5 (VII)	0.5 (II)	18.5 (VI)
EEP[1]a	kg a.i. Pa ha^{-1} yr^{-1}	2.1 (IV)	0.2 (I)	1.3 (II)	4.0 (VII)	3.4 (VI)	1.9 (III)	2.5 (V)
EEPs	kg a.i. ha^{-1} d yr^{-1}	37.6 (III)	10.9 (I)	57.0 (V)	40.3 (IV)	123.2 (VII)	19.1 (II)	101.8 (VI)
EEPw	kg a.i. ha^{-1} d yr^{-1}	0.3 (IV)	0.1 (II)	0.4 (VII)	0.4 (VI)	0.2 (III)	0.1 (I)	0.3 (V)

[1] value multiplied by 10^3

For abbreviations see § 5.2.

Table 5.14b. – Average values (and ranking) of cross indicators calculated for each farm

	FARM	DAI-INT	DAI-EXT	SWI-INT	SWI-EXT	RIC-POU	MIX	CER-RIC
NUTRIENT EFFICIENCY								
NE	kg DM kg^{-1} N_{IN}	41.2 (I)	24.7 (V)	21.2 (VI)	21.1 (VII)	32.4 (III)	32.6 (II)	30.6 (IV)
PE	kg DM kg^{-1} $P_2O_{5\ IN}$	156.0 (I)	46.9 (VI)	25.7 (VII)	53.5 (V)	91.6 (III)	94.7 (II)	73.3 (IV)
KE	kg DM kg^{-1} K_2O_{IN}	45.6 (I)	20.1 (VII)	21.2 (V)	20.9 (VI)	21.9 (IV)	25.6 (II)	24.0 (III)
NUTRIENT WASTE								
NW	kg $N_{surplus}$ kg^{-1} DM	4.6 (II)	1.4 (I)	23.0 (VI)	23.5 (VII)	11.8 (IV)	5.6 (III)	12.4 (V)
PW	kg $P_2O_{5surplus}$ kg^{-1} DM	–0.6 (I)	3.4 (V)	26.1 (VII)	7.3 (VI)	2.9 (IV)	–0.4 (II)	0.1 (III)
KW	kg $K_2O_{surplus}$ kg^{-1} DM	2.8 (II)	–7.6 (I)	18.0 (VI)	20.6 (VII)	11.2 (IV)	5.1 (III)	12.8 (V)
NW/GM	kg $N_{surplus}$ k€$^{-1}$	57.7 (II)	24.6 (I)	190.1 (VI)	224.9 (VII)	101.3 (V)	61.0 (III)	87.4 (IV)
PW/GM	kg $P_2O_{5surplus}$ k€$^{-1}$	–7.8 (I)	44.5 (IV)	215.9 (VII)	70.2 (VI)	52.6 (V)	–3.6 (II)	5.6 (III)
KW/GM	kg $K_2O_{surplus}$ k€$^{-1}$	38.1 (II)	–82.2 (I)	148.6 (VI)	197.1 (VII)	99.2 (V)	50.7 (III)	75.4 (IV)
NW/EnG	kg $N_{surplus}$ GJ^{-1}	0.3 (II)	0.1 (I)	1.4 (VI)	1.5 (VII)	0.8 (IV)	0.4 (III)	0.9 (V)
PW/EnG	kg $P_2O_{5surplus}$ GJ^{-1}	0.0 (I)	0.2 (V)	1.6 (VII)	0.5 (VI)	0.2 (IV)	0.0 (II)	0.0 (III)
KW/EnG	kg $K_2O_{surplus}$ GJ^{-1}	0.2 (II)	–0.5 (I)	1.1 (VI)	1.3 (VII)	0.7 (IV)	0.3 (III)	0.9 (V)
ECONOMY AND ENERGY								
VC/EnG	€ GJ^{-1}	1.5 (I)	1.9 (II)	2.7 (IV)	3.6 (V)	6.1 (VII)	2.6 (III)	5.6 (VI)
GM/EnG	€ GJ^{-1}	5.1 (VI)	5.1 (VII)	7.5 (III)	6.5 (IV)	10.2 (II)	5.8 (V)	11.7 (I)
GM/EnIN	€ GJ^{-1}	56.6 (I)	40.0 (VI)	42.9 (V)	37.4 (VII)	54.5 (II)	47.0 (IV)	49.8 (III)

For abreviations see § 5.2.

Table 5.14c. – Average values of soil management, landscape, and biodiversity indicators calculated for each farm

		DAI-INT		DAI-EXT		SWI-INT		SWI-EXT		RIC-POU		MIX		CER-RIC	
SOIL MANAGEMENT															
SCI	1-10	0.59	(II)	0.89	(I)	0.40	(V)	0.35	(VI)	0.32	(VII)	0.53	(III)	0.42	(IV)
CS	0-1	4.4	(III)	8.5	(I)	1.6	(VI)	1.8	(V)	1.2	(VII)	5.9	(II)	2.8	(IV)
OCI	1-10	8.1	(II)	9.9	(I)	7.5	(III)	5.1	(IV)	4.5	(V)	4.1	(VII)	4.4	(VI)
LANDSCAPE AND BIODIVERSITY															
CD	1-10	6.2	(III)	2.9	(IV)	2.8	(V)	1.8	(VII)	2.6	(VI)	7.1	(I)	6.9	(II)
HR	m ha^{-1}	89	(III)	142	(I)	88	(IV)	49	(VI)	49	(VII)	92	(II)	70	(V)
VI	m ha^{-1}	266	(II)	190	(VII)	230	(IV)	213	(VI)	217	(V)	267	(I)	241	(III)

For acronyms see § 5.2.

The presence of fodder and winter cereal crops, produce a decrease of average nutrient balances calculated at farm level. The nutrient balances are close to zero in DAI-EXT and MIX, while are very high in SWI farms, due to high manure and fertilizer inputs.

The low yield obtained in DAI-EXT (large area with permanent meadows), in RIC-POU and in CER-RIC (large area with rice) have produced low EnG (Table 5.14a). The PPAs potential risk indicators are high in farm with rice, while are generally low in the other farms (Table 5.14a).

The nutrient efficiencies are generally high in DAI-INT, while are low in SWI farms (Table 5.14b). The indicators that evaluate the nutrient surpluses per yield, GM and EnG unit, show good values for DAI farms, while poor performance in SWI farms. The cultivation of rice in RIC-PUO and CER-RIC have increased the GM/EnG values compared other farms (10.2 and 11.7 € ha^{-1} yr^{-1}, respectively; Table 5.14b).

The average SC is usually low, and the soil is covered by crops less then 6 month per year, with the exception of DAI farms, where the large area with permanent meadows increases SC. The CS is corrected only in DAI-EXT, due to large presence of permanent meadows that is considered a good “sequence of crop”. Only MIX has CS values close to reference threshold, because rarely the same crop is cultivated in the same field in two consecutives years. The OCI is above threshold in farm where large amounts of manures are used (SWI-INT and DAI farms).

The CD is close to threshold in CER-RIC and in MIX, where numerous crops are cultivated in small fields, while is very low in farms that cultivate only corn (SWI farms) or rice and corn (RIC-POU). In DAI-EXT the CD value is low, because despite three crops are cultivated, permanent meadows cover a large part of the farm in large fields. The HR is elevated in DAI-EXT where a dense network of hedges and rows is present around and inside fields, while in SWI-EXT and in RIC-POU these ecological structures are reduced, and concentrated in particular part of the farm. The BI does not highlight marked differences in the farms, because the variability of the field dimensions is low (from 2.4 ha in DAI-INT to 5.0 ha in DAI-EXT; Table 5.1), and usually the field shapes are not elongated.

The variability of indicator values obtained for single crops/field in the seven farms is shown in Appendix c.

5.6. Discussion

The seven farms were selected considering the main production systems in the PASM (Bechini et al. 2005a), but the small number of farm monitored is not completely representative of the 910 farms and corresponding managements present in the studied area. Nevertheless, the data obtained and the sustainability evaluation can be used as a preliminary screening of the agricultural state in the studied area.

The cropping pattern in each farm is oriented by the necessity of maximizing the profit, under a specific human, technical, and economic availability.

The DAI-INT farm has on average a good economic and environmental sustainability, due to a good compromise between intensive (corn) and non intensive crops. The DAI-EXT has a very poor economic sustainability, even if the economic value of crop products will increased when used by the livestock compartment. A fairly good balance between economic and environmental sustainability is obtained by SWI farms, but the bad nutrient management can be improved. The farm managements based on rice production have a very good economic sustainability, but the high PPA potential risk and poor energy and soil management provide a non satisfactory environmental judgment. MIX obtains on average low profit by crops, but the horse and farm tourism activity compensate the low GM obtained from crops. The diversified managements in this farm produce variable environmental judgment, but generally very good performances are not obtained.

The indicator values usually show an economic sustainable management for intensive crops (rice and corn), while the fodder crops are not economically sustainable without considering the related livestock that increase the economic value of field products. Differently to the economic point of view, the environmental sustainability related to nutrients and PPAs management is higher in fodder production unlike intensive crops. The energy indicators show good performances for corn, and moderate in barley, wheat and triticale.

The difficult quantification of nutrients applied with manures is a common characteristic of livestock farms. Extension services could help farmers in the preparation of fertilization plan, reducing the surplus that usually occurs in these farms, improving the environmental quality.

A more correct use of manure is obtainable reducing the ammonia volatilization (ploughing in manure application, band spreading, injection), and monitoring the real amount of nutrients applied with manures by the use of tools that evaluate the real nutrient concentration in each manure application (i.e. near infra red; De Ferrari et al., 2006).

On the other hand, the intense use of manures provide to fields a satisfactory level of OC inputs, necessary to maintain good soil quality. When manure is not applied, the SOC decreases, reducing the soil fertility. The Hénin-Depuis model used in OCI is inadequate to predict the change in SOC, but it is a good tool for ranking cropping systems from the point of view of OC (Boiffin et al., 1986). Accurate harvest index, shoot:root ratio values, and the quantification of source of carbon are a valuable step toward an indirect but easy method of predicting C storage and associated production and environmental benefits (Johnson et al., 2006). Unfortunately the coarse coefficient used in the determination of carbon inputs, especially for the quantification of below ground biomass, and specific pedoclimatic conditions provide indicator values with high uncertainty.

The choice of PPAs applied is made depending on their action and cost, but farmers do not consider the potential environmental pressure of a.i.. Extension services could be help farmers in the selection of correct PPAs, according to specific action, price, and potential effect on environment.

The same crop type has a specific management in each field; indicators calculated with average management data, produce a smoothed judgment, because the average management does not consider the variability inside farm, in particular the maxima/minima values in different fields, where the environmental pressure and corresponding potential risk are high. For example, if the average manure application is used in NS calculation, considering a uniform application of manures in all crops, it is more difficult to highlight the situation where the potential risk is high, because high NSs are masked by the low values obtained in other fields. Therefore it is batter obtain from farmers detailed data at field level instead of an average value for the entire farm, because the field is the most relevant level to the decision that farmers can to take on cropping systems (Girardin et al., 2000).

The imposition of specific managements (i.e. limit in N applied with manure; Regione Lombardia, 2006) or the funding of generic agro-environmental measures (Regione Lombardia, 2005), should be combined with a structured and widespread extension services. In general, farmers would benefit from these services, which will provide technical support and knowledge necessary to drive the management decision making towards a more sustainable agriculture, with case-specific solutions, for example drawing up good fertilization plan with a correct application of manures and fertilizers. Moreover, the extension services could monitor the agricultural sector, recording detailed data on the real farm management, creating a vast database containing information continuously updated, which can be used in the agricultural assessment. Both farmers and Park (and consequently citizens) would benefit from these services.

The indicators are inadequate to predict the exact level of impacts: it is not possible to know the exact nutrient surplus and the environmental fate, but it is possible to define the systems with the higher potential environmental pressure. Indicators are good tool for ranking systems, and it is not correct use the absolute values of indicators for the sustainability evaluation of the systems. These indicators can be used by administrative and technical bodies (e.g. by the Park) as a first screening tool, to compare different management and to identify the most hazardous cropping and farming systems.

Furthermore no interactions among indicators are considered in this work: every indicator is considered individually, while a systems assessment requires an integration of information in a unique analysis. Some preliminary integration of indicator and evaluation of trade-offs were carried out applying Sustainability Functions (Castoldi and Bechini, 2007, Appendix E) and Sustainability Solution Space method (Wiek and Binder, 2005; Castoldi et al., 2007b; Appendix F).

5.7. Conclusions

In this study, seven farms in the Sud Milano Agricultural Park (Northern Italy) were monitored in order to evaluate the crop management sustainability, applying simple tools (indicators) that can be used for a preliminary comparison of economic and environmental systems sustainability.

The data were collected by face-to-face interviews with farmers, checked and stored in a geographical information system. A set of indicators was selected from literature and applied to 266 crop × field combination monitored. The indicators regard the management of economic resources, nutrients, energy, plant protection agents, soil, landscape, and biodiversity, evaluated at different levels (crop, field, and farm).

The rice and corn have good values in economic indicators, but the former has high potential impacts on environment due to intensive use of plant protection agents and low energy productions, while the latter has generally high nutrient surpluses. Alternation of corn with barley or winter wheat reduces the economic performance, but increases the environmental sustainability, with a more corrected nutrient management.

The intensive dairy farm has a good compromise between economic and environmental sustainability, while dairy extensive farm obtained low values for the economic indicators. The nutrient surpluses are the main problem highlighted by indicators in swine farms monitored. The farms with rice obtained high values in the economic indicators, and correct nutrient balances, but the intensive use of plant protection agents and the low amounts of carbon left in the soil, decrease the environmental sustainability of these systems. The cultivation of many crops with low inputs does not provide excellent results both from the economic and the environment points of view.

In order to improve the agricultural sustainability assessment carried out with indicators, the future researches have to be focused on the analysis of trade-offs among indicators and on the evaluation of uncertainty related to inputs.

5.8. Acknowledgments

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CHAPTER 6

CALCULATING THE SOIL SURFACE NITROGEN BALANCE AT REGIONAL SCALE: EXAMPLE APPLICATION AND CRITICAL EVALUATION OF TOOLS AND DATA

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6.1. Abstract

Agro-ecological indicators (AEIs) allow evaluating sustainability for a large number of farms. The SITPAS Information System developed for the agricultural park "Parco Agricolo Sud Milano" (northern Italy) contains detailed farming and cropping systems information for 731 farms that can be used for these analyses. We used the SITPAS database to evaluate N management with an AEI and to evaluate the suitability of the SITPAS data model for this type of applications. The AEI (soil surface N balance) was calculated for each crop at field scale, as the difference between the sum of N inputs (atmospheric depositions, biological fixation, fertilisers, residues from previous crop) and crop N uptake; the results were aggregated at rotation and farm levels. The farming systems with the highest surplus (> 300 kg N ha^{-1}) are dairy, cattle and pig farms, in which chemical N fertilisers are used in addition to animal manures. The crops with the highest surplus are Italian ryegrass and maize (183 and 172 kg N ha^{-1}, respectively), while rice and wheat have the lowest surplus (87 and 85 kg N ha^{-1}). The data model allowed to store and analyse complex information not manageable otherwise; its main limitation was the excessive flexibility, requiring a complicated procedure for the calculations of this example, and the exclusion of most data at the farming systems level (corresponding to 82% of the studied area) for missing, incomplete, out-of-range or inconsistent data. These results suggest to promote actions towards better N management in cropping systems in the Park and to develop simple data models based on minimum data requirements when sustainability evaluations are to be conducted.

6.2. Introduction

In the last decades, there has been an increasing concern about the environmental impact of agricultural activities, involving consumers, citizens, policy makers and farmers. Several policy measures to promote sustainable agriculture were issued by European Union, governments and regional administrations. As a support to ex-post and ex-ante evaluation of these measures, it is important to evaluate the sustainability of agricultural management, to reveal not only the systems which are negatively affecting the environment, but also to identify the positive environmental externalities of agricultural activities. Such evaluations need to be carried out for a large number of farms. To be feasible, they should be based on data already available without the implementation of direct measures, which would be too expensive for non-experimental contexts. Examples of available data include Common Agricultural Policy (CAP) archives, applications for funding under the Rural Development Programmes and demands of agricultural fuel.

The most suitable tools that can be applied in this context are agro-ecological indicators (AEIs). These (OECD, 2001; Castoldi and Bechini, 2006) are synthetic variables representing the cropping or farming systems, based on relatively simple data, allowing evaluating the environmental performance of production systems by benchmarking with thresholds and providing relative comparisons of systems in space and time. Issues that can be faced with AEIs include biodiversity and landscape (Weinstoerffer and Girardin, 2000), water quality, nutrient (Öborn et al., 2003; Parris, 1998) and pesticide management (Reus et al., 2002), and soil quality (Bockstaller and Girardin, 1996).

Our application of AEIs is based on the SITPAS information system ("Sistema Informativo Territoriale per il Parco Agricolo Sud Milano", standing for "Agricultural Information System for the Sud Milano Agricultural Park"; Bechini and Zanichelli, 2000; Provincia di Milano, 2006). The SITPAS information system integrates in the same GIS environment agricultural, pedological, climatic and environmental information collected in the Parco Agricolo Sud Milano (PASM). In particular, it contains detailed and georeferenced crop management information for 731 farms, obtained through direct interviews with farmers.

In this paper (i) we present the SITPAS database, underlining its specific features in relation to the calculation of AEIs indicators of crop management; (ii) we evaluate one aspect of sustainability using a management-based AEI, the soil surface balance for nitrogen (N) (Parris, 1998), applied to the farms described in the SITPAS database; (iii) we finally discuss advantages and disadvantages of this approach.

6.3. Materials and methods

6.3.1. The SITPAS information system

The PASM is a regional metropolitan agricultural Park, surrounding the town of Milano (northern Italy, 45°N, 9°E), and including 61 municipalities. It was created in 1990 with the aim of preserving and improving landscape and natural environment, and, differently than in traditional protected areas, also to safeguard, qualify and promote agro-forestry activities. The agricultural area is ca. 35,000 ha, and the most important crops are maize, rice, permanent meadows, soybean, barley, Italian ryegrass and winter wheat, with moderate to high yields (averages of 9.6, 19.5, 5.2, 4.8, 4.9 and 3.0 t DM ha^{-1} for grain maize, silage maize, rice, winter wheat, barley and soybean, respectively). A total of 910 farms has been identified, and 731 were described in detail; of these, animal farms are 348 and raise bovine (dairy and cattle), swine and poultry livestock. Irrigation is normally performed with surface methods, using water from a dense network of canals.

The aim of the SITPAS project (1999–2003) was to collect, integrate and analyse information about agricultural activities in the PASM to support strategic and operational decisions of the Park. The SITPAS information system was developed in a GIS environment and includes vector maps representing polygons at the cadastral and municipal level and several relational databases. The databases consist of the farming systems database (SITPAS-db), containing data collected through interviews to 731 farmers, and the external pre-existing databases (including databases of public administration, like CAP files).

6.3.2. The SITPAS database

The SITPAS-db contains information collected on purpose during the project and not available in any other external pre-existing database. Its data model was developed with the entity-relationship framework (Garcia-Molina et al., 2002). It is consistent with the questionnaire used to interview the farmers, and is linked with external data sources (databases and maps). It was implemented using the Relational Database

Management System (RDBMS) Microsoft Access. The implementation includes 159 entities, 166 domains (closed lists containing qualitative information, used to avoid typing errors and to ensure data consistency within the working group) and 150 relations. The database contains detailed information related to farm, cultivated parcels, irrigation sources, buildings used for agricultural production, mechanization, crop rotations and management, livestock management (feed, manure). Every crop and animal management information represents the farmer's average behaviour, and therefore is not related to a particular year. All information related to buildings, cultivated land and crop operations are georeferenced at the cadastral parcel level.

The application presented here is related to crop management, therefore details are provided for this section of the data model. A farm may run one or more crop rotations, which can be georeferenced by indicating the cadastral parcel(s) used; if the parcels are not identified, the rotation is referred to the entire farm. Crop rotations are represented as a sequence of crops over time; for each crop, agronomic operations can be recorded (tillage, sowing, fertilisation, irrigation, herbicide, fungicide and insecticide application, harvest). For each crop management operation several types of information can be specified: the type of operation, the date on which an operation is carried out (which can be indicated with the day and month or by specifying the number of days before or after another crop operation or before/after a crop development stage), the percentage of crop area (R_a) interested by a given crop management operation (less than or equal to 100%), the frequency (f) with which a crop management operation is repeated during the growing season (e.g. cutting of meadows), the depth of tillage. For operations involving the application of one or more products (fertilisers or pesticides), the type and amount of product(s) applied are indicated; for each product, the detailed composition is available in the database (e.g. N, P and K contents of fertilisers, active ingredients of pesticides), through a list that can be expanded. For harvest operations, the yield(s) and the fate of harvested product(s) and residues can be specified (sold, recycled within the farm - for example as animal feed -, re-incorporated into the soil).

The flexibility of this data model needs also to be described: farmers presented a wide range of responses, not only in technical terms, but also concerning the amount and the quality of the information provided. This reflects the variability of production systems and the variability of farmers' technical skills and willingness to collaborate. We believe that this variability is difficult to avoid when the number of farms is large and the less co-operative farmers are not excluded from the survey. Therefore, in order to maximize the possibility of using different types of data, the data model described above was kept flexible, so that for example it is possible to: (i) leave several fields empty (e.g. crop yields and the amounts of products applied do not need to be always specified); (ii) avoid inserting the entire set of crop management operations (e.g. specifying a harvest operation is not mandatory, or the harvest operation can be specified but the products obtained do not need to be listed); (iii) use different measurement units for the same variable in different records (e.g. crop yield can expressed on a wet or dry basis); (iv) use hierarchical domains to specify crop types, types of management operation, types of products applied: the hierarchical domains allow to specify the value of a variable at different levels of detail (e.g. crop type can be described as "Cereals", or "Maize", or "Maize - FAO Class 600" or "Maize - Costanza"); (v) store the same information in alternative ways; e.g. straw incorporation can be either described as a distinct crop management operation or as the fate of the straw resulting from a grain harvest operation.

6.3.3. Calculation of N surplus

The soil surface N balance indicator (Parris, 1998) was separately calculated for each single crop of each rotation of each farm, and then a weighted average (based on crop area) was calculated for the rotation and for the farm. This indicator is the difference between the nutrients entering the soil and those leaving the soil with crop uptake annually. Positive values of the balance indicate nutrient accumulation in soil and/or nutrient losses, while negative values indicate nutrient depletion from soil. We calculated the soil surface N balance as:

$$S = F + M + R_P + A + B - R - U \tag{6.1}$$

where: S = N surplus, F = N applied with chemical fertilisers, R_P = N returned to soil with residues originating from the previous crop in the rotation, M = N applied with animal manures, A = atmospheric depositions, B = biological fixation of leguminous crops, R = N removed from soil with crop residues, U = N removed from soil with useful product. Nitrogen contained in irrigation water, ammonia volatilisation and denitrification were not considered because not enough information was available to estimate them. Denitrification and ammonia volatilisation are therefore a part of N surplus, and contribute to possible losses. The quantity F was estimated by multiplying the amount of fertilisers applied by their N concentration. For animal manures, both the amounts applied and the N concentrations were unknown. To estimate the amount of manure-N available at the field level, we calculated for each farm the total livestock weight (using the number of heads in each animal group and their average live weight). On this basis, we then estimated the annual N emissions from livestock, using a conversion coefficient (live weight - emissions), net of losses in the stable and during manure storage; it therefore represents residual N for field distribution (Sacco et al., 2003). Generally, farmers did not indicate the amount of manure applied on each field; for that reason we homogeneously allocated manure-N to crops for which operations of animal manure distribution were declared; if this information was missing, we assumed that the entire farm area was fertilised with animal manures. Atmospheric deposition (A) was set at 15 kg N ha^{-1} crop^{-1} (Grignani et al., 2003). Biological fixation (B) was estimated as U – A – 0.5 M – 0.7 F (Grignani et al., 2003) for monospecific leguminous crops (soybean, meadows), and equal to 40 kg N ha^{-1} yr^{-1} for other rotated or permanent meadows (Grignani et al., 2003). We assumed that crop N uptake (U) could be estimated by multiplying the declared crop yields by their N concentrations derived from literature (Grignani et al., 2003). For the estimation of N contained in crop residues (R_P and R), we first calculated the amount of crop residues using yield and harvest index; we then multiplied this amount by its N concentration (Grignani et al., 2003).

Before carrying out the calculation of the soil surface balance, a detailed data quality check was done. This was particularly important because different persons were involved in data collection and because of the variability in farmers' responses. In particular, we have checked that all the variables required for our calculations were within a proper range and were not missing; otherwise we eliminated from the analysis the corresponding crops, rotations or farms with the exception of average crop yields, used in replacement of missing or out-of-range yields. All the variables related to crop management operations (crop biomass and amounts of products applied) were multiplied by $R_a \times f$ (to consider the percentage of crop area treated with the operation and its frequency).

6.4. Results

6.4.1. Data quality

The original SITPAS-db (dataset A) describes 731 farms, covering 38,095 ha. This figure is higher than the *ca.* 35,000 ha of agricultural area of the Park, because it includes farms with part of the land outside the Park. Average farm-scale properties (Table 6.1) show that maize, rice and meadows are the most important crops (occupying on average 46, 17 and 16% of farm area, respectively), and that livestock mostly belongs to the dairy type, with a moderate average density (0.62 t l.w. ha^{-1}). Contractors carry out harvest operations on 48% of the area, while their contribution is much smaller for sowings and herbicide distributions. Rice is mostly irrigated with the traditional continuous flooding system (on average 14% of farm area).

Three hundred thirty-six farms having at least one record with null, out-of-range or inconsistent data were excluded from analysis. The 395 remaining farms (Table 6.1, dataset B), occupying 20,517 ha (54% of the area of dataset A), maintain similar characteristics compared to the original data set. Farms were excluded for these reasons: missing indication of the amounts of fertilisers applied, incomplete description of livestock density (missing number or missing weight of animals), inconsistent declaration of farm area, crop management described only for a part of the farm, unknown exports of animal manures.

Table 6.1. – Data sets used for the calculation of soil surface nitrogen balance in the Sud Milano Agricultural Park: average (and standard deviations) of farms attributes

Attribute	Units	All the farms of the database (dataset A)		Farms with complete data (dataset B)		Farms with positive crop N balances (dataset C)	
Number of farms		731		395		157	
Total area	ha	38,095 (100%)		20,517 (54%)		6,704 (18%)	
Farm area	ha	52	(65)	52	(66)	43	(43)
Percentage area cultivated with							
Maize	%	46	(33)	46	(32)	48	(35)
Rice	%	17	(32)	17	(33)	19	(35)
Meadows	%	16	(25)	17	(24)	16	(26)
Barley	%	5	(14)	5	(13)	2	(11)
Soybean	%	4	(13)	5	(16)	6	(20)
Italian ryegrass	%	3	(11)	3	(12)	3	(12)
Wheat	%	2	(10)	3	(10)	2	(7)
Livestock intensity							
Dairy	Mg l.w. ha^{-1}	0.62	(1.48)	0.52	(0.94)	0.74	(1.15)
Cattle	Mg l.w. ha^{-1}	0.09	(0.32)	0.09	(0.35)	0.10	(0.47)
Swine	Mg l.w. ha^{-1}	0.14	(1.22)	0.04	(0.30)	0.05	(0.28)
Poultry	Mg l.w. ha^{-1}	0.01	(0.14)	0.00	(0.02)	0.00	(0.03)
Percentage of area treated by contractors for							
Harvest	%	48	(42)	48	(42)	46	(43)
Land	%	5	(22)	4	(20)	5	(22)
Tillage	%	7	(24)	6	(22)	4	(18)
Sowing	%	11	(31)	11	(31)	12	(32)
Herbicide appl.	%	14	(34)	14	(34)	13	(34)
Mechanisation							
Machinery power	kW ha^{-1}	11.0	(11.8)	11.5	(13.7)	12.5	(16.9)
Combine-harvesters	n	0.27	(0.54)	0.25	(0.49)	0.25	(0.51)
Percentage of rice area irrigated with							
Turned irrigation	%	2	(11)	1	(9)	1	(8)
Delayed continuous	%	7	(24)	6	(22)	7	(24)
Traditional	%	14	(33)	14	(32)	14	(33)
Turned flooding	%	3	(16)	3	(16)	3	(16)

6.4.2. Negative nitrogen surpluses

Unexpectedly, several soil surface N balances at crop level on dataset B were negative. Surplus can be negative (equation 1) only when N removed from soil (R + U) exceeds N inputs to soil (F + R_P + M + A + B). Crops with negative surplus show unrealistic amounts (Table 6.2) of N applied with fertilisers (F and M), which appear too low to sustain the reported values of crop N uptake (R + U). These farmers have probably underestimated N inputs, for example by not declaring one or more fertiliser applications. Another error could be an under-estimation of biological fixation (B) for meadows, which represent a relevant fraction of the area affected by this problem. We decided to exclude from analysis the farms with negative surplus for one or more crops: the smaller dataset obtained (C; Table 6.1) includes 157 farms (18% of the area of dataset A) and maintains similar characteristics to A, even with a smaller average farm size (43 vs. 52 ha), a higher portion of farm area cultivated with maize and rice, and different livestock densities (higher for dairy and lower for swine). We will now analyse dataset C at whole-farm level.

6.4.3. Nitrogen surplus

6.4.3.1. Whole-farm level

Nitrogen surpluses of dataset C have large variability (Table 6.3); the number of farms decreases with increasing levels of S. Moving from the first to the last class of N surplus (Table 6.3), the amount of N applied with manure (M) systematically grows; N applied with fertilisers (F) and crop N uptake do not follow a clear trend; at high levels of surplus, F is not increasing and U is always higher than 190 kg N ha^{-1}. The output/input ratio is decreasing from 0.86 to 0.34. The first four classes (0 – 200 kg N ha^{-1}) occupy together 70% of the studied area, while extremely high surpluses (> 300 kg N ha^{-1}) cover 14% only. In the classes with low surplus (0 – 100 kg N ha^{-1}) the quantities of N applied with F and M are similar to crop N uptake (U + R); these farms have negligible or low livestock densities (below the average for the entire Park); a relevant fraction of farm area (29 and 33% on average in the first two classes) is cropped with rice. In the classes with intermediate surplus (100 – 200 kg N ha^{-1}) crop uptake and F increase, together with M. Compared to previous classes, these farms have more intensive dairy farming (0.28 and 1.19 t l.w. ha^{-1}), cultivate more maize, more meadows and less rice. Over 200 kg N ha^{-1} of surplus, the excess is mostly determined by a relevant amount of N applied in the form of manure; this classes show relatively low F amounts and high crop uptakes. They include intensive dairy farms (2.73 t l.w. ha^{-1} in the last class) associated with cattle and / or swine breds. The production of forages is provided by maize, meadows and Italian ryegrass.

6.4.3.2. Rotation level

The results for rotations including crops having positive surplus are presented in Table 6.4; the total area considered is higher than in the case of single farms due to a smaller number of crop excluded. Again, the highest surpluses are driven by relevant amounts of N applied with manures, in rotations including forages. Cereal rotations (with or without rice) occupy a relevant portion of the area and have the lowest surpluses (87 – 141 kg N ha^{-1} on average).

6.4.3.3. Crop level

When the results are analysed at the level of crop types (Table 6.5, describing all crops with positive S), more information can be extracted from the database compared to the whole-farm and rotation levels: the fraction of area analysed compared to total area cultivated in the Park ranges from 20% for barley to 62% for soybean.

Italian ryegrass and maize have the highest surplus (183 and 172 kg N ha^{-1}), deriving from high F and M applications exceeding crop uptake. Rice has a much lower surplus, because farmers tend to use less chemical fertilisers and animal manures. Meadows have high crop uptake and are mostly fertilised with manures, with a small contribution of mineral N. The two winter cereals (wheat and barley) are fertilised with opposite strategies, more based on animal manures for barley and on chemical N fertilisers for wheat. Italian ryegrass is the crop where the highest amount of N from animal manures is used (213 kg N ha^{-1} on average).

Table 6.2. – Soil surface nitrogen balance for crops with negative N surplus in the Sud Milano Agricultural Park: average (and standard deviation) of components and surplus

Crop type	Total area in dataset A	Crop area with negative N surplus	U	R	R_P	F	M	A	B	S
	(ha)		(kg N ha^{-1})							
Maize	16408	2494 (15%)	182 (47)	51 (26)	37 (27)	66 (82)	14 (44)	15 (0)	0 (0)	-100 (70)
Rice	10545	1940 (18%)	74 (12)	36 (7)	27 (15)	23 (25)	1 (7)	15 (0)	0 (0)	-44 (21)
Meadows	4328	1547 (36%)	224 (58)	0 (0)	2 (9)	9 (24)	39 (63)	15 (0)	40 (0)	-119 (68)
Wheat	985	541 (55%)	119 (26)	30 (4)	22 (20)	40 (41)	0 (0)	15 (0)	0 (0)	-72 (50)
Barley	1243	518 (42%)	93 (32)	21 (6)	22 (23)	22 (28)	0 (0)	15 (0)	0 (0)	-54 (44)
Italian ryegrass	1026	296 (29%)	99 (12)	2 (8)	16 (19)	6 (15)	4 (15)	15 (0)	0 (0)	-59 (24)

U = N removed from soil with useful product,
R = N removed from soil with crop residues,
R_P = N returned to soil with residues originating from the previous crop in the rotation,
F = N applied with chemical fertilisers,
M = N applied with animal manures,
A = atmospheric depositions,
B = biological fixation of leguminous crops,
S = N surplus. Output/Input = (U + R) / (R_P + F + M + A + B).

Table 6.3. – Soil surface nitrogen balance for farms without negative crop N surplus in the Sud Milano Agricultural Park: average (and standard deviation) of components and surplus

Surplus class (kg N ha^{-1})	N. of farms	Area covered	Average farm area	U	R	R_P	F	M	A	B	S	Output/Input
		(ha)	(ha)	(kg N ha^{-1})								
< 50	37	1028	28 (27)	130 (57)	37 (19)	32 (21)	91 (67)	13 (40)	15 (0)	42 (73)	27 (13)	0.86
From 50 to 100	35	1648	47 (37)	126 (48)	41 (14)	36 (18)	152 (61)	26 (68)	15 (1)	13 (31)	74 (14)	0.69
From 100 to 150	26	1200	46 (71)	151 (56)	46 (23)	41 (26)	206 (101)	49 (98)	15 (0)	7 (13)	121 (15)	0.62
From 150 to 200	19	831	44 (19)	198 (58)	24 (20)	20 (18)	160 (104)	177 (140)	16 (2)	20 (25)	171 (16)	0.57
From 200 to 250	17	663	39 (27)	202 (50)	29 (20)	26 (20)	178 (118)	221 (140)	16 (2)	15 (16)	225 (14)	0.51
From 250 to 300	8	377	47 (19)	191 (83)	21 (18)	13 (11)	64 (47)	379 (50)	20 (6)	13 (14)	276 (12)	0.43
From 300 to 400	11	820	74 (68)	203 (41)	28 (18)	22 (18)	115 (96)	404 (108)	17 (3)	16 (14)	342 (23)	0.40
> 400	4	138	34 (14)	226 (81)	26 (20)	18 (20)	217 (94)	474 (143)	21 (8)	5 (11)	484 (62)	0.34

Surplus class (kg N ha^{-1})	Maize	Rice	Meadows	Barley	Wheat	Italian ryegrass	Soybean	Dairy	Cattle	Swine	Poultry
	(% of farm area)							(t l.w. ha^{-1})			
< 50	33 (39)	29 (42)	9 (25)	4 (17)	1 (5)	0 (3)	19 (36)	0.07 (0.30)	0.03 (0.10)	0.00 (0.00)	0.00 (0.00)
From 50 to 100	43 (35)	33 (43)	7 (18)	3 (12)	2 (7)	0 (0)	5 (13)	0.17 (0.47)	0.05 (0.23)	0.00 (0.00)	0.00 (0.00)
From 100 to 150	60 (40)	16 (32)	13 (29)	0 (0)	1 (6)	0 (2)	1 (3)	0.28 (0.69)	0.23 (0.88)	0.02 (0.12)	0.00 (0.00)
From 150 to 200	56 (23)	2 (10)	31 (28)	3 (7)	2 (5)	2 (5)	2 (8)	1.19 (1.06)	0.03 (0.08)	0.00 (0.00)	0.02 (0.07)
From 200 to 250	60 (25)	0 (0)	30 (29)	2 (7)	0 (0)	3 (7)	3 (11)	1.22 (1.12)	0.38 (0.79)	0.00 (0.00)	0.00 (0.00)
From 250 to 300	37 (26)	12 (33)	25 (24)	1 (3)	7 (19)	28 (38)	0 (0)	2.37 (0.99)	0.05 (0.14)	0.24 (0.67)	0.01 (0.03)
From 300 to 400	62 (18)	8 (17)	20 (21)	2 (5)	0 (1)	8 (12)	3 (6)	2.43 (1.27)	0.00 (0.00)	0.29 (0.66)	0.00 (0.00)
> 400	65 (29)	16 (31)	14 (27)	0 (0)	0 (0)	25 (29)	0 (0)	2.73 (1.86)	0.00 (0.00)	0.46 (0.92)	0.00 (0.00)

For symbols, see Table 6.2.

Table – 6.4. Soil surface nitrogen balance for rotations without negative crop N surplus in the Sud Milano Agricultural Park: average (and standard deviation) of components and surplus

Type of rotation	Total area in dataset A	Rotation area analysed	U	R	R_P	F	M	A	B	S	Output/ Input
	(ha)		(kg N ha^{-1})								
Cereals	8181	2823 (35%)	152 (47)	52 (19)	43 (24)	218 (86)	68 (124)	15(1)	1 (9)	141(110)	0.59
Cereals and rice	11755	2690 (23%)	77 (35)	33 (11)	28 (16)	99 (59)	53 (115)	15(0)	3 (18)	87(106)	0.56
Permanent meadows	3504	1122 (32%)	192 (88)	0 (0)	0 (0)	20 (52)	229 (144)	15(0)	46 (39)	119(103)	0.62
Cereals and industrial crops	6872	670 (10%)	164 (38)	45 (12)	40 (12)	124 (63)	74 (132)	15(0)	71 (46)	115(98)	0.65
Cereals and forages	2106	656 (31%)	202 (63)	37 (15)	21 (21)	174 (67)	253 (148)	18(4)	9 (19)	237(146)	0.50
Forages and cereals	3201	655 (20%)	239 (53)	28 (11)	21 (13)	175 (134)	325 (139)	20(5)	6 (12)	280(98)	0.49
Forages	1947	517 (27%)	261 (97)	11 (11)	10 (10)	103 (90)	285 (219)	19(6)	80 (140)	226(186)	0.55

Cereals: rotation including maize, winter wheat, barley, oat, rye and eventually rice (less than 10% of the area).
Cereals and rice: from 10 to 100% of rotation area is cropped with rice.
Cereals and industrial crops: at least 10% of rotation area is cropped with sugar beet, oil or protein crops.
Cereals and forages: more than half of the area is cropped with cereals and forages are at least 10% of the area.
Forages and cereals: more than half of the area is cropped with forages and cereals are at least 10% of the area.
Forages: the rotation has only forages.
For symbols, see Table 6.2.

Table – 6.5. Soil surface nitrogen balance for crops having positive surplus in the Sud Milano Agricultural Park: average (and standard deviation) of components and surplus

Crop type	Total area in dataset A	Crop area analysed	U	R	R_P	F	M	A	B	S	Output/ Input
	(ha)		(kg N ha^{-1})								
Maize	16408	6773 (41%)	177 (53)	46 (25)	33 (26)	219 (81)	129 (147)	15 (0)	0 (0)	172 (123)	0.56
Rice	10545	3760 (36%)	69 (15)	32 (10)	25 (17)	86 (50)	63 (97)	15 (0)	0 (0)	87 (84)	0.54
Meadows	4328	1463 (34%)	232 (104)	0 (0)	4 (14)	10 (36)	192 (140)	15 (0)	110 (110)	98 (90)	0.70
Soybean	2287	1427 (62%)	192 (40)	26 (8)	31 (20)	11 (28)	9 (54)	15 (0)	190 (53)	39 (32)	0.85
Italian ryegrass	1026	453 (44%)	96 (28)	0 (0)	20 (21)	31 (44)	213 (132)	15 (0)	0 (0)	183 (116)	0.34
Barley	1243	250 (20%)	77 (27)	20 (4)	25 (27)	65 (53)	140 (120)	15 (0)	0 (0)	148 (103)	0.39
Wheat	985	254 (26%)	109 (17)	30 (8)	35 (26)	133 (33)	41 (93)	15 (0)	0 (0)	85 (86)	0.62

For symbols, see Table 6.2.

6.5. Discussion

6.5.1. Nitrogen surplus

From our results the systems more at risk appear to be the forage and grain crops (Italian ryegrass, maize, barley and meadows) cultivated in dairy and pig farms, where the amounts of N applied with animal manure are in general very high, and chemical N fertilisers not reduced accordingly. Farming systems with high surplus do not represent a large portion of the studied area and could receive technical assistance for better N management. Our results are in agreement with those found by other Authors, both for average and for variability. Schröder et al. (1996) have found an average value for surplus of 160 kg N ha^{-1} for 38 Dutch integrated arable farms, and 117 after the adoption of an integrated nutrient management program. The variability was elevated, with surpluses ranging from less than 50 to more than 200 kg N ha^{-1}. In an area with intensive animal husbandry (3.80 t l.w. ha^{-1} on average for dairy farms), Sacco et al. (2003) have found N surpluses of 41 kg N ha^{-1} for non-livestock farms and of 326 kg N ha^{-1} for dairy farms. There is a good correspondence in terms of surplus between their dairy farms and our surplus class at 300 – 400 kg N ha^{-1}, with lower M values in our work (404 instead of their 426), and similar crop uptakes. Their figures are similar also for non-livestock farms (S=41 kg N ha^{-1}) and for crop balances (S=183, 132, 129, 104 and 58 kg N ha^{-1} for maize, wheat, barley, grassland and soybean, respectively).

In animal farms the uncertain concentration of N in manure and its uncertain availability to crops make the use of chemical N a cheap method to sustain crop production regardless of the fate of the animal N applied (the so called "insurance N": Schröder et al., 2000). For this reason methods should be identified and disseminated to the farmers of the Park to allow them for quick estimates of the nutrient value of animal manures (e.g. Marino Gallina et al., 2005a; Marino Gallina et al., 2005b; Reeves and Van Kessel, 2000; Scotford et al., 1998; Van Kessel and Reeves, 2000). Also, methods to reduce the use of chemical N fertilisers should be applied (e.g. for maize Magdoff, 1991; Schröder et al., 2000).

The variability of S and balance components within farm groups, rotations and crop types was very large (Table 6.3, 6.4 and 6.5), in particular for M and F; the variability decreased at the level of crop types (e.g. for F in maize). Consequently, for further studies attention should be given to balance components with the highest variability, like manure. In addition, the variability demonstrates that a given crop (e.g. maize) or a given farming system (e.g. dairy farming) is not dangerous *per se*, but can be more or less harmful depending on the specific operational and strategic management choices taken by the farmer.

6.5.2. Advantages and disadvantages of the indicator

The interest of the indicator lies in its simplicity, because it allows the calculation using data that can be obtained without carrying out any direct measure, and to integrate aspects of nutrient management that are strictly interconnected (chemical fertiliser and animal manure applications, crop yields and uptake, etc.). The balance allows comparisons in time (the same system in different periods) and space (different cropping / farming systems of a region in the same period). The comparisons can be made on single balance components and on the resulting surplus (e.g. OECD, 2001, for different nations in different periods). OECD (2001) shows also that the balance allows to quickly assess the relative importance of different inputs (e.g. organic vs. inorganic fertilisers) in the determination of the surplus. Finally, nutrient balances can be used to create awareness among farmers and to guide improvements in crop and livestock N management, as demonstrated for example by Schröder et al. (1996) and by Hanegraaf and den Boer (2003).

However, several limitations must be pointed out. Nitrogen losses are the result of complex dynamic processes that are not entirely captured by a simple mass balance: among the factors that a balance does not consider, we may cite water dynamics, initial soil content of inorganic N, soil mineralization rate, type and C : N ratio of crop residues and manures, tillage practices, climate, and soil characteristics. As a result, positive N surplus do not necessarily indicate N losses, mainly because different forms of N accumulation in the soil are possible; also, ammonia volatilisation was not taken into account in our calculation. Therefore, the indicator only shows the potential for environmental damage or unsustainable use of soil resources (OECD, 2001). It has been shown (e.g. Sieling and Kage, 2006; Salo and Turtola, 2006) that the soil surface balance does not estimate the actual N losses in a specific year: N can be accumulated in the soil so that the excess applied to a crop may be actually leached during the fallow or during the cultivation of the subsequent crop, or being absorbed by the next crop, or not being lost if immobilised in organic form. The surplus is an indicator for total N losses only if it is integrated over a relatively long period (Öborn et al., 2003); even then, losses account only for a part (15 – 57%) of actual surplus (Salo and Turtola, 2006). When other regressors (e.g. precipitation, runoff, drainage) are used together with surplus to estimate losses, the variability explained is higher (Salo and Turtola, 2006). In addition, a small excess of N applied is unavoidable, due to the efficiency of chemical and organic fertilisers (Grignani et al., 2003), which is frequently in the range 50 – 70%.

The nitrogen indicators proposed by Bockstaller and Girardin (2000) and by Pervanchon et al. (2005) represent an answer to most of these critical aspects, still avoiding the complexity of dynamic simulation models. They provide a semi-dynamic representation of N cycling in the soil-crop system, using a more process-based approach to estimate N volatilisation and leaching, and over-winter soil N dynamics. Compared to N balances, these indicators allow analysing the interactions that simple balances do not consider. The drawback is that they require more data about climate, soil, and crop management (fertiliser application methods and dates in particular) and that a relatively more complex calculation is needed.

In our application, several sources of uncertainty have arisen: first, the farmers frequently do not know the yields of crops that are neither sold nor weighted (e.g. silage maize and meadows); therefore, these should be measured because their variability is potentially high. Biological fixation was estimated in a simple way, but other methods should be explored to derive figures that are more accurate. Nitrogen in animal excreta was estimated on the basis of live weight, using parameters that do not consider the variability of feed ration. Crop nitrogen concentrations were assumed to be the same for all crops considered, and N uptake could have been overestimated for the less fertilised crops. Ammonia volatilisation was not estimated. A field-by-field estimate of the amounts of manure applied was not available and the homogeneous allocation across farm area (the only solution in this case, as stated also by Sacco et al., 2003) might have introduced a bias for some crops, probably by overestimating M for winter crops and underestimating for summer crops. Finally, other Authors calculate the N balance in a different ways (e.g. residues, biological fixation, atmospheric depositions, and seeds are not always included). It is important therefore to note that the results are more adequate for relative comparisons rather than for estimating absolute values of N losses.

6.5.3. Farm survey and data base management

The use of a relational database for this type of applications is a mandatory requirement, due to the large amount of data to be stored and processed, and to the complexity of the relationships among the objects studied. Our application demonstrates that the data model developed in the SITPAS project is complete and very detailed and that the database contains agro-environmental variables related to a large agricultural area, at the detail of single cadastral parcel.

Possible improvements of the data model are related to three interconnected methodological aspects of survey and data storage: (i) flexibility of the data model, (ii) compromise between direct interviews and reliance on existing databases, and (iii) use of pre-compiled crop management itinerary.

The extreme flexibility of our data model had the great advantage of allowing almost every answer from farmers to be recorded in the database. However, this advantage had two types of adverse consequences: first, some data were very difficult to extract, in particular when the same information could be stored in alternative ways or at different hierarchical levels; second, missing data partly nullified the results of our calculations. As an example of the first consequence, to carry out the simple calculation of the soil surface balance presented here, a total of 135 queries had to be run with the RDBMS; even if this increased the time required to develop the calculations, it did not affect the possibility of using the data. For the second consequence, serious limitations became evident in our work:

82% of the studied area could not be considered at whole-farm level, and 38 – 80% at crop level. Apparently, one would conclude that "flexibility" means actually "lack of structure in the data", and that storing data that cannot be processed is a useless exercise. This is partly true, but it should be mentioned that the problem was particularly evident here due to the high level of integration required by the calculation of the indicator. To properly calculate the balance, the data needed to be completely specified for most agronomic operations of every crop of every farm. For example, even the lack of a single amount of urea applied for one out of 20 crops of a farm would require eliminating the entire farm. When less integrated information is required, much more data become available: for example, only for 15% of fertiliser applications the date was not specified, and only for 8% of inorganic fertiliser applications the amount of fertiliser was not indicated. This means that simpler but useful statistics can be successfully calculated using this database, and the flexibility is not so limiting as in the case of the balance. Our conclusion is that this flexible data model was congruent with the purpose of collecting and integrating as much information as possible on agricultural production systems of the Park during the survey. However, further studies focused on objectives that are more specific would require the development of a simpler database based on a minimum dataset, to simplify data collection and processing (e.g. Sacco et al., 2003).

As Table 6.1 shows, for most of the variables describing farming systems the average and the variability do not differ very much among the three datasets (A, B and C). Therefore, it would have probably been more efficient to concentrate the efforts with the most co-operative farmers to obtain management data, and to integrate this information with existing databases related to the entire set of farms in the Park. This would also answer the question if a calculation of this type can be applied at situations where the resources are not available for conducting so many direct interviews; Sacco et al. (2003), for example, have built a detailed information system on the basis of available data (CAP files, slurry management database, animal livestock register, digital cadastral map) integrated with expert knowledge for specific aspects (chemical fertiliser applications) and have carried out no interviews. This procedure has of course the advantage of being based on official data but may introduce a certain arbitrary homogeneity in some variables: for example, the variability of F (Table 6.5) can be important, while Sacco et al. (2003) have used a fixed crop-specific value derived from expert knowledge.

Finally, the use of a standard crop management itinerary, pre-compiled by experts before the interviews are carried out, would be a useful benchmark to critically evaluate and test the data collected during the interviews, as done for example in the AGENDA project by Giupponi (2002).

6.6. Concluding remarks

The SITPAS-db is an integrated and comprehensive information base to carry out regional-scale calculations of AEIs; the powerful relational data model allows integrating and evaluating the data in a way that would not be possible otherwise. The present application showed that the procedures to collect and store the data for this type of applications can be further improved: (i) before carrying out a survey, determine precisely the objectives; a generic survey may result in too many data collected and a complex database structure; (ii) identify the minimum data required, their scale and their source; (iii) set-up the simplest data model.

The calculated indicator shows that in the study area (the Sud Milano Agricultural Park), intensive dairy and pig farming systems with excessive N fertilisation are potentially at risk of N losses, while cereal farms have lower surplus. Therefore, specific measures should be promoted by the Park for better N management. The most uncertain data were biological fixation, yields of meadows, nutrient emissions from livestock and their apportioning over land area. Scientific rules to determine these quantities at this scale (using easily available data) would be very useful.

The soil surface balance indicator can be used by administrative and technical bodies (e.g. by the Park) as a first screening tool, to identify the most hazardous cropping and farming systems. The balance could be calculated using available databases (CAP, manure distribution, Rural Development Programme), digital maps, remote sensed information, and expert knowledge. The indicator can also potentially be integrated with common soil and climate information (precipitation, evapotranspiration, water holding capacity) to calculate a potential nitrate concentration of water leaving agricultural fields (OECD, 2001). Technical assistance could be delivered to the farmers with the highest surpluses: available scientific knowledge, direct measurements and simulation models should be used to optimise agricultural management, towards reduced N losses and good crop yields.

6.7. Acknowledgments

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CHAPTER 7

GEOSTATISTICAL PREDICTION AT THE REGIONAL SCALE USING EXISTING DATABASES ON CROPPING AND FARMING SYSTEMS

Castoldi, N., A. Stein, and L. Bechini.

Keywords: soil fertility, phosphorus, crop management, spatial interpolation, regional assessment, kriging.

7.1. Abstract

The amount and quality of input data may be a limiting factor when carrying out agri-environmental assessments with complex indicators at the regional scale. Existing information stored in public agricultural databases may be available. Before using this, one has to explore its additional value, to provide complete information layers for sustainability evaluations.

In this study, we estimated the extractable soil P (*ESP*) in the Sud Milano Agricultural Park in Northern Italy. We integrated available information about agricultural activities, with *ESP* concentrations across the area using a large database of measured soil properties and of crop management data collected at individual farms. For unsampled farms, *ESP* was predicted by spatial interpolation. After splitting the dataset into three sections, we used a hybrid and section specific geostatistical interpolation method. We compared this method with Ordinary Kriging (OK) and Kriging with External Drift (KED). In KED we used livestock density and percentage of farm area cropped with corn (*Zea mays* L.) and rice (*Oryza sativa* L.) to describe the external drift. Our hybrid method showed the lowest standard deviation of the prediction error and less smoothing than OK and KED. High *ESP* values are found in corn fields and in animal farms. This is due to large use of P fertilizers in corn, particularly in animal farms where available manure is frequently applied in excess of crop need. *ESP* values are low in rice fields, where fertilizer is more carefully applied. Using a reference threshold of 20 mg P kg^{-1} soil, most of the area can be classified as being very rich in *ESP*. As a consequence, P fertilization could be suspended in many cases for several years without yield decrease. On the basis of this experience, we conclude that additional management information could help interpretation and extrapolation of the target variable. We finally point out how to optimize available resources, avoiding collection of data that are either not usable or needed, considering their spatial and temporal variability.

7.2. Abbreviations

BK: Bray-Kurtz analytical method; ***CDV_i***: *ESP* confidential values corresponding to 99 percentiles calculated at each cadastral parcel; **Co**: percentage of farm area cultivated with corn; ***ESP***: extractable soil P concentration; **KED**: kriging with external drift; ***LnP***: logarithm of *ESP;* **LS**: livestock density; **MK**: mosaic kriging; **OK**: ordinary kriging; **OL**: Olsen analytical method; **PASM**: Sud Milano Agricultural Park; **REML**: Restricted Maximum Likelihood; **Ri**: percentage of farm area cultivated with rice; **SCV**: standard deviation of the cross-validation error; **SITPAS**: Agricultural Information System for the Sud Milano Agricultural Park; **WLS**: Weighted Least Squares.

7.3. Introduction

To evaluate the impact of agriculture on the environment, agri-environmental assessments are conducted. At the regional scale, these assessments are frequently done using simple indicators (e.g. the amount of fertilizers applied: OECD, 2002a; EEA, 2005), as commonly few and approximate data are available at this scale only. The possibility of using realistic and complex indicators or simulation models is thus limited by the associated costs of more intensive data requirements.

Agricultural and environmental databases contain quantitative and qualitative information about cropping and farming systems. This information, however, is frequently not integrated as owners, formats, and methodologies are not homogeneous. Therefore its potential is not fully exploited. In addition, some information is not available for entire regions, as it may be sparsely collected or clustered, as a result of specific or occasional data collection activities. For example, if a fertilizer management extension service is launched for a region, only part of the farmers may decide to be involved, and information is thus recorded from a few fields only. This results into a sparse and possibly clustered availability of information. Other information, such as CAP files describing land use, is complete, however, because these cover entire regions homogeneously as the result of a more systematic administrative procedure.

The integration of sparse information may yield a low and non representative coverage of the region, whereas the use of spatially interpolated information layers allows a complete integration of different types of data and to support the decisions about the region studied.

To carry out more realistic agri-environmental assessments at the regional scale input data have to be estimated by adding as much value as possible to existing information, avoiding additional measurements and integrating what is already available. This helps the decision maker to identify critical areas and to define priorities for policy.

Nutrient management is an important issue in agro-ecological assessments. Phosphorus as a plant nutrient is used for storing and transferring energy, and for building nucleic acids and cytoplasmatic membranes (Hart et al., 2004). P-losses may occur by means of surface runoff, thus affecting the quality of surface waters (Hart et al., 2004) by accelerating freshwater eutrophication with repeated outbreaks of harmful algae blooms (Sharpley et al., 2001). Small additions of P to surface water can lead to water body eutrophication. In addition, P fertilizers are derived from limited, non-renewable phosphate deposits. As their recovery is expected to be more expensive in the future (Colomb et al., 2007), it is important to use P in an efficient way, thus reducing risks of pollution and of overexploitation of non-renewable resources. Extractable soil phosphorus (*ESP*) is fundamental to properly manage P. *ESP* is a standard measure that can be compared with threshold tabbed values to test whether fertilizer application is needed for a given soil/crop combination. *ESP* is usually measured with chemical analyses, like those proposed by Olsen et al. (1954), and Bray and Kurtz (1945).

Evaluation of *ESP* is usually carried out pointwise by soil analyses. To make a regional assessment, it is necessary to carry out spatial interpolation. Geostatistical interpolation, like ordinary kriging (OK), provides a means of predicting environmental variables at unsampled locations using the spatial dependence between observations. Ordinary kriging uses observations of the target variable only, and relies upon its stationarity in the study area. Several hybrid interpolation methods combine kriging with auxiliary information (Hengl et al., 2004) thus improving the prediction quality in non-stationary conditions (Isaaks and Srivastava, 1989). Examples are co-kriging, universal kriging, kriging with external drift (KED) and regression kriging (Hengl et al., 2003).

The objectives of this research were: i) to collect and integrate the information available about agricultural activities at the regional scale in Northern Italy; ii) to estimate the *ESP* across the area by using a large database of measured soil properties and of crop management data collected at individual farms; iii) to assess the adequacy of agricultural P management based on estimated *ESP*; and iv) to quantify the uncertainty of this assessment that is generated by the uncertain knowledge of *ESP*.

This paper deals with the integration of the information and the spatial interpolation of *ESP*. The companion paper (Castoldi et al., 2008b) focuses on the agri-environmental assessment and the quantification of its uncertainty.

7.4. Materials and methods

7.4.1. Study Area and Database

The Sud Milano Agricultural Park (PASM, Parco Agricolo Sud Milano; 45°N, 9°E) is a regional metropolitan agricultural Park surrounding the town of Milano (Northern Italy). The Park covers an area of approximately 47 000 ha, of which 35 000 ha is agricultural (Bechini and Castoldi, 2006). The Park is located in a plain area with an altitude gradient of about 80 – 160 m above sea level, where the main soils are loam, sandy-loam, silt-loam. It is located in one of the most intensive Italian agricultural production areas. It was created in 1990 to protect and improve natural ecosystems and to safeguard, qualify and promote agricultural activities. It was conceived to provide available green areas to people living in town, and to keep farmers in activity, thus avoiding the possible abandonment that could have been favored by the advancement of the surrounding town (Scelsi, 2002). A large agricultural information system for the Park was developed in the period 1999 – 2003 (Bechini and Zanichelli, 2000; Bergamo et al., 2007). Data about agricultural activities were merged with regional climatic, pedological and environmental databases into a GIS-based integrated tool called SITPAS ("Sistema Informativo Territoriale per il Parco

Agricolo Sud Milano", standing for "Agricultural Information System for the Sud Milano Agricultural Park"). Crop and animal management data were obtained by interviewing farm managers in 730 out of 910 farms in the Park. The most important farm types are cereal and dairy farming. All crop management information was georeferenced at the cadastral parcel level. Crop and animal management information represent the average farmer's practices. Pre-existing data provided by the public administration, private laboratories and farmers were integrated into the database (e.g. soil analysis reports). More details are available in Castoldi et al. (2007b).

To analyze the data, we extracted more than 2800 georeferenced soil analyses from the SITPAS database, as well as their corresponding cadastral parcels. These analyses were carried out as a support for the preparation of nutrient management plans. The reference scale of commercial soil analysis is the field, which usually contains one or more cadastral parcels. To obtain a unique representative soil sample for the entire field, sub-samples from a field were mixed and the analytical value of the mixture was applied to the entire field. This procedure was cost-effective although not providing information about the within-field spatial variation. All SITPAS soil analyses are georeferenced, using either geographical coordinates, or a reference to one or more cadastral parcels, in which case coordinates of the parcel centroids were used as proposed by Juang et al. (2004). All soil analyses were carried out in qualified laboratories, according to the Italian official methods for soil analysis (Ministero per le Politiche Agricole, 1999). The *ESP* analyses were carried out with two methods: Olsen et al. (1954) (OL), Bray and Kurtz (1945) (BK). Also, for each cadastral parcel, the livestock density [Mg live weight ha^{-1}] and the percentage of farm area cultivated with corn (*Zea mays* L.), rice (*Oryza sativa* L.), meadow, barley (*Hordeum* spp.), and wheat (*Triticum aestivum* L.), were extracted from the database. This was done regardless of having a soil analysis referred to it.

7.4.2. Geostatistical Procedures

7.4.2.1. Datasets

Only soil analyses obeying specific criteria were used for interpolation (Fig. 7.1). We followed a procedure by creating several datasets successively. First, dataset A (Table 7.1) contained those soil analyses that were located in the PASM area or in a buffer zone of 5 km around it. These analyses were all carried out between 1997 and 2000. This period partially overlapped that of farmers interviews. Data from more than one year could be used and compared because the annual rate of change of *ESP* is small (Russell, 1973; Karpinets et al., 2004; Colomb et al., 2007).

The spatial locations in dataset A are clustered, as those are usually available only in a limited number of farms. In dataset A, 659 (59%) *ESP* analyses were carried out with BK and 466 (41%) with OL. All OL values were multiplied by 1.9 to obtain values that can be directly compared to those measured with BK, following Mallarino and Atia (2005), who observed a strong correlation ($R^2 = 0.89$) between the two. The hypothesis of normality of the log *ESP* values (*LnP*) was not rejected in dataset A following the Shapiro-Wilk normality test ($p = 0.19$). Therefore, *LnP*s values were used because they produce a lesser experimental estimation variance than the kriging built with initial *ESP*s values (Journel and Huijbregts, 1978).

Dataset B emerged from dataset A by selecting those analyses that were located in parcels with recorded farm properties. The linear regression coefficient between *LnP* and several environmental properties (distance from polluted rivers and altitude), soil properties (clay, silt, sand, lime content, pH, organic matter, cation exchange capacity, base saturation, and C/N ratio), and farm management information (livestock density, irrigation water source, percentage of farm area cultivated with corn, rice, meadow, barley, and wheat) were calculated. The main variables correlated with *LnP* were livestock density (LS; Mg live weight ha^{-1}), corn (Co) farm percentage area (0 – 1) and the relation among LS and rice (Ri) farm percentage area (0 – 1) (LS $\times$ Ri). The R^2 was low (0.19) but the regression coefficients were significant (Table 7.2). This allowed to make use of KED for dataset B.

Table – 7.1. Datasets used for spatial interpolation of extractable soil P concentration (*ESP*; expressed as Bray-Kurtz) in the Sud Milano Agricultural Park (PASM, Northern Italy, 45°N, 9°W). Summary of points where the measurements of target variable are available (measurement points) and kriging prediction points. All *ESP* values measured with the Olsen method were multiplied by 1.9 to convert to Bray-Kurtz (details in the text)

	Measurement points								Prediction points		
		Distribution									
Dataset	Number of points	min.	1st quartile	mean	3rd quartile	max.	St.Dev.	Coefficient of variation	Number of points	Area	Percentage of the PASM agricultural area
		mg P kg^{-1} soil								ha	%
A	1125	3.9	24.4	55.3	72.2	324.4	45.8	0.83	14 406	32 616	93.2
B	695	3.9	23.1	49.9	64.4	324.4	41.9	0.84	14 406	32 616	93.2
C	204	5.2	18.7	34.9	40.9	189.2	26.1	0.75	2 917	7 289	20.8
D	212	3.9	23.9	42.1	49.8	190.0	26.9	0.64	4 693	10 033	28.7
E	279	5.7	29.3	66.6	86.3	324.4	53.4	0.80	6 796	15 294	43.7

Table – 7.2. Theoretical models selected for each of the datasets of the logarithm of the extractable soil phosphorus concentration in the Sud Milano Agricultural Park, Northern Italy (45°N, 9°E)

Data-Set	Kriging method [a]	Variogram type [b]	Secondary variables [c]	R^2	Intercept and slope coefficients	Estimated value	SE	p value	Nugget	Sill	Range (m)	Smoothness parameter
A	OK	P–exp	–			–			0	0.5653	612	0.416
B	KED	P–exp	LS, Co, LS × Ri	0.19	intercept	3.31	0.044	$< 2\ 10^{-16}$	0	0.4767	334	0.375
					LS	0.13	0.018	$5.5\ 10^{-14}$				
					Co	0.45	0.091	$7.5\ 10^{-7}$				
					LS × Ri	0.49	0.101	$1.5\ 10^{-6}$				
C	OK	cir	–						0.2451	0.4489	1588	–
D	OK	cir	–						0.0788	0.4096	528	–
E	KED	cir	LS, Co, LS × Ri	0.18	intercept	3.20	0.120	$< 2\ 10^{-16}$	0.1867	0.5120	2976	–
					LS	0.11	0.023	$1.5\ 10^{-6}$				
					Co	0.78	0.214	$3.3\ 10^{-4}$				
					LS × Ri	0.51	0.127	$7.3\ 10^{-5}$				

[a] OK, ordinary kriging; KED, kriging with external drift.
[b] p–exp, power–exponential; cir, circular.
[c] LS, livestock density (Mg live weight ha^{-1}); Co, percentage of farm area cultivated with corn (%); Ri, percentage of farm area cultivated with rice (%), LS × Ri, interaction among LS and Ri.

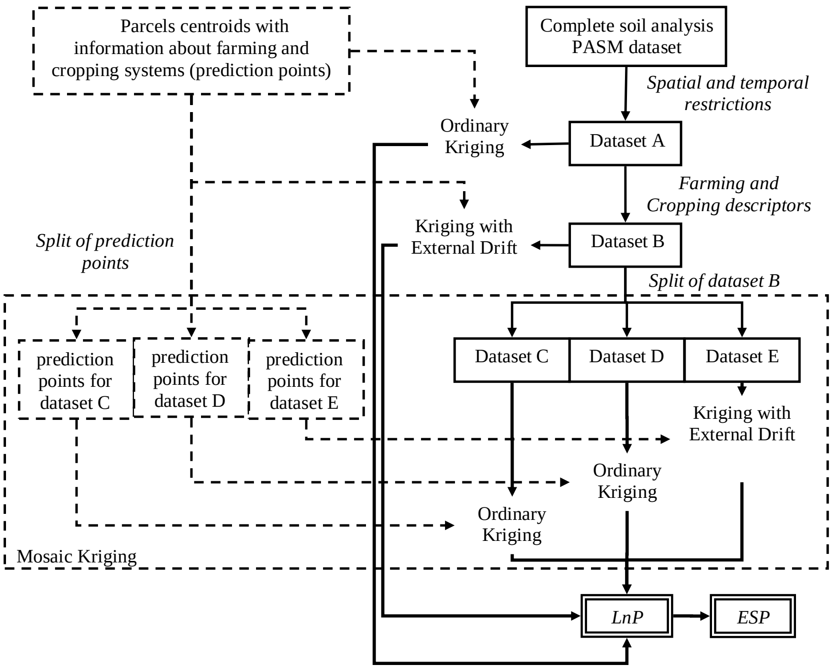

Fig. 7.1. – Flowchart of data and spatial interpolation procedures for the estimation of logarithm of extractable soil phosphorus concentration (*LnP*) and extractable soil P concentration (*ESP*) in the Sud Milano Agricultural Park (PASM, Northern Italy, 45°N, 9°E). Additional information is reported in Table 1 and in the text. Rectangles with solid border = measurement points; rectangles with dashed border = prediction points; rectangles with double border = predictions.

Next, we split dataset B into datasets C, D and E, according to LS and Co values. A threshold of 0.1 Mg live weight ha^{-1} was used for LS in order to separate farms with significant animal breeds from farms producing mostly commodities. A second threshold (Co 20%) was used in order to separate the farms without a significant presence of corn from the others (a Co threshold lower than 10% was not effective due to the presence of corn in nearly all farms). Dataset C (204 reports) contains soil analyses of farms with Co < 20% and LS < 0.1 Mg live weight ha^{-1}. Dataset C includes rice, barley, wheat, soybean and meadow farms, concentrated in the south-western area of the Park. Dataset D (212 reports) contains farms with Co > 20% and LS < 0.1 Mg live weight ha^{-1}. Dataset E (279 reports) contains more intensive animal farms with LS > 0.1 Mg live weight ha^{-1}, regardless of Co. For datasets C and D, no external variables are correlated with *LnP*, while for dataset E the R^2 of the linear regression using LS, Co, and LS × Ri as predictors (Table 7.2) is low (0.18), but the regression coefficients are significant. Therefore, KED was used also for dataset E. The Shapiro-Wilk normality test shows p-value equal to 0.45, 0.24, 0.10, and 0.15 for the dataset B, C, D, and E, respectively, therefore not rejecting the hypothesis of log normality. The frequency distribution of *ESP* (and consequentially of *LnP*) is similar for all datasets in each year (data not shown), and small differences among years existed.

7.4.2.2. Spatial interpolation procedures

We investigated different spatial interpolation procedures: ordinary kriging (OK), external drift kriging (KED), and a hybrid form, here called Mosaic Kriging (MK), to predict *LnP* values. For each procedure, we used i) locations where the target variable (*LnP*) was measured and ii) locations where the prediction was carried out.

Datasets A and B were used in applying OK and KED, respectively. In both cases the prediction points are the centroids of cadastral parcels containing farming and cropping systems information in the SITPAS database (Table 7.1 and Fig. 7.1). Mosaic Kriging is described in terms of a partition of the study area, i.e. by separately processing datasets C, D, and E. It is a modified form of stratified kriging (Stein et al., 1988; Voltz and Webster, 1990). The mosaic thus divides the area *A* into 3 different sections: A_C, A_D, and A_E. These sections have an empty intersection and jointly cover the total area:

$$AC \cup AD \cup AE = A;$$

$$AC \cap AD = AC \cap AE = AD \cap AE = \emptyset$$

$$AC \neq \emptyset,\ AD \neq \emptyset, \text{ and } AE \neq \emptyset$$

Within each section A_M ($M = C, D$ or E) a (soil) property is described by a regionalized variable $Y_M(\boldsymbol{x})$, $M \in \{C, D, E\}$ depending on the location vector $\boldsymbol{x}$. Variograms were constructed for each section.

Prediction points are assigned to sections A_C, A_D, and A_E, yielding three sets of prediction points (Fig. 7.1). A_C contained only farms with Co < 20% and LS < 0.1 Mg live weight ha^{-1}. Identification of prediction points yielded three intermingled and non-overlapping geographical layers, creating a disjoint patchworks of landscape. A section-specific interpolation procedure was applied for the three different sections: OK was applied to A_C and A_D, and KED to A_E (Table 7.2 and Fig. 7.1). Both KED and MK take farm type and soil management into account. MK is most refined, using spatial dependence, farm type and soil management separately for different areas, thus taking crisp boundaries into account when interpolating.

For each procedure *LnP* observations were considered to represent the entire area of the parcel, assumed to be homogeneous. This allowed to predict *ESP* for 93.2% of the agricultural area of the Park (Table 7.1).

For each part, experimental variograms were calculated for *LnP* and a theoretical model was fitted to the data. In datasets B and E, the significant secondary variables (Co, LS, and LS × Ri) were used in order to improve the quality of the empirical variogram and of the theoretical model. Ten models were tested for each dataset using two fitting procedures (Weighted Least Squares, WLS; Restricted Maximum Likelihood, REML) and five different types of correlation function (power-exponential, exponential, spherical, circular, and cubic; Ribeiro and Diggle, 2006).

All theoretical models were validated by cross-validation, where each single measurement point is removed from the data set and the variable at this location is predicted using the remaining locations (Ribeiro and Diggle, 2006). The model with the lowest standard deviation of the cross-validation error (SCV) in each dataset was chosen.

Kriging provides the predicted value (Y_K^*) and error variance ($\sigma_K^{'2}$) of *LnP* at each prediction point. As *LnP* can be assumed to be normally distributed in all datasets, also kriging predictions can be assumed to follow a normal distribution, and the predicted value corresponds to the mean of the *LnP* distribution. Y_K^* and $\sigma_K^{'2}$ were back-converted to *ESP* using the approach proposed by Journel and Huijbregts (1978): if *Z(x)* is a stationary random function with a lognormal distribution, mean *m*, and variance σ^2, the random function $Y(x) = log\ Z(x)$ has a normal distribution with mean m' and variance σ'^2:

$$Z_0^* = K_0\, exp[Y_K^* + \sigma_K'^2 / 2]\ , \qquad (7.1)$$

and

$$\sigma_Z^2 = m^2\, exp[\sigma'^2\ ](1 - exp[-\sigma_K'^2\ ]) \ , \qquad (7.2)$$

where Z_0^* is the estimated value in the original scale (in this case *ESP*), K_0 is a corrective factor used to delete the possible difference between mean of Z_0^* and m: it is determined by equating the arithmetic mean of the Z_0^* to the expectation m; and σ_Z^2 is the kriging error variance in the original scale. In addition, based on Y_K^* and $\sigma_K'^2$, confidential intervals corresponding to 99 percentiles were calculated at each cadastral parcel using the *cumulative distribution function*. The 99 values obtained were back-transformed via a simple exponentiation in order to obtain the 99 corresponding *ESP* values (CDV_i, with i from 1 to 99).

The ESRI ArcMap 9.0 software (ESRI, 2004), Microsoft Access, R statistical package (R Development Core Team, 2006) and geoR geostatistical package (Ribeiro and Diggle, 2001) were used to carry out this work.

7.5. Results

ESP values are generally high in the study area (Table 7.1). The average value for dataset A equals 55.3 mg P kg^{-1} soil, whereas lower average values of 34.9 and 42.1 mg P kg^{-1} soil are observed for datasets C and D containing farms without significant livestock density. In all datasets the 1st quartile of the distribution is higher than most critical levels provided by literature (cited below in the discussion section). For instance, 83, 83, 73, 84, and 89% of measured *ESP* values are above a common critical level (20 mg P kg^{-1} soil), for datasets A, B, C, D, and E, respectively. This confirms the widespread environmental risks due to the high *ESP* concentration, in particular in specialized livestock farms (dataset E).

Prediction errors for *LnP* obtained during cross-validation of variogram models are reported in Fig. 7.2.

Because all mean errors are low (ranging from –0.0014 to 0.0029), predictions of good quality were identified as those with low SCV. Value of SCV are relatively high, ranging from 0.538 to 0.670. Farming systems information helps to improve the predictions, as it is seen when comparing OK applied to dataset A with KED to dataset B, where the SCV decreased by 2.0%. The predictions are further improved on datasets D and E, showing a reduction of 4.7 and 8.4% of the SCV respectively, whereas dataset C remains similar to B, with a decrease of 0.9% in comparison with dataset A.

The choice of the correlation function is important, as it may help to reduce the SCV. For example, with dataset A the maximum SCV (0.670) is obtained with WLS and the cubic correlation function (Fig. 7.2), whereas the minimum value of 0.588 occurs for REML and the

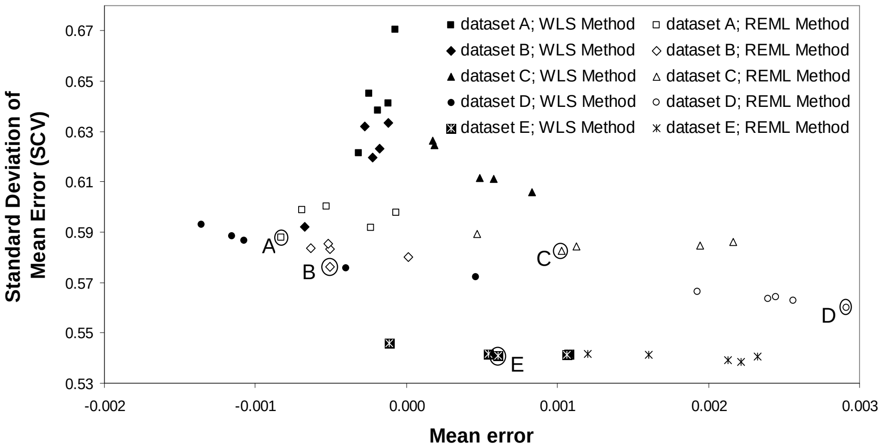

Fig. 7.2. – Mean error and standard deviation provided by cross validation of the fifty theoretical models (5 datasets, 5 model functions, and 2 fitting procedures) used for spatial interpolation of the logarithm of extractable soil P concentration in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E). Each symbol represents a different variogram model. The selected models are indicated with a circle. WLS: Weighted Least Squares method; REML: Restricted Maximum Likelihood.

power-exponential correlation function. Overall, the best approach is that based on split datasets, as dataset E (including 40% of measurement points and 47% of prediction points) has the lowest SCV, and datasets C and D have equal or lower SCV than dataset B.

Selected theoretical models are shown in Table 7.2. All of them were obtained with the REML method. Correlation functions are the power-exponential function for the aggregated datasets A and B and the circular correlation function for the split datasets C, D, and E. For KED applied in datasets B and E the slopes of the secondary variables are positive, as *LnP* is higher when farms cultivate more corn, and when animals are raised. The nugget/sill ratio is 0% for aggregated datasets A and B, and 55, 19, and 36% for the split datasets C, D, and E, respectively. This can be due to the correlation functions used: the power-exponential model allows relatively lower nuggets compared to the circular correlation function, as the smoothness parameter allows to fit experimental variograms with large increases at low lags.

The statistical distributions of predicted *ESP* with OK, KED, and MK are reported in Fig. 7.3. The area with the lowest *ESP* values

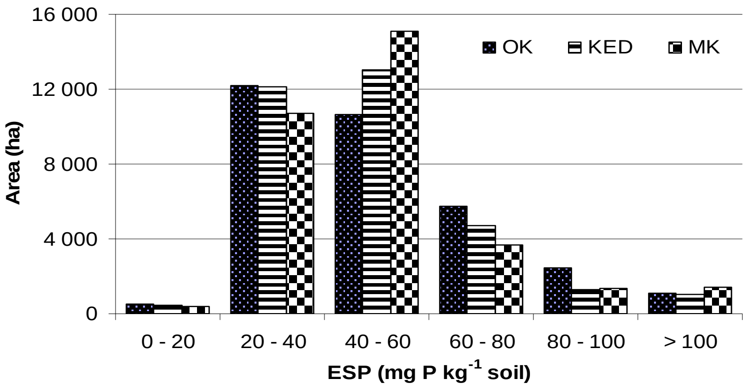

Fig. 7.3. – Comparison of three spatial interpolation methods for extractable soil P concentration (*ESP*) in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E): area in different ranges estimated with ordinary kriging (OK), Kriging with External Drift (KED), and Mosaic Kriging (MK). *ESP*: extractable soil P concentration.

(ranging from 0 to 20 mg P kg^{-1} soil) covers approximately 1.5% of the total area (Fig. 7.3) using either of the three methods. The range 20 – 40 mg P kg^{-1} soil represents 33% of the area using MK, 37% using OK and KED. MK has the largest area in the range 40 – 60 mg P kg^{-1} soil (46%) as compared to OK and KED (33 and 40%, respectively). The high range (> 60 mg P kg^{-1} soil) is generally more common with OK (28%) than with KED (22%) and MK (20%) (Fig. 7.3). The fact that the values predicted with MK have a more narrow distribution is probably due to the fact that MK uses a more homogeneous neighborhood when making predictions, as each dataset is composed of a sub-set of more similar points (the coefficient of variation is lower for datasets C, D and E (75, 64 and 80%, respectively), compared to datasets A and B (83 and 84%; Table 7.1).

As expected, the spatial distribution of the predicted values using the OK method shows a marked smoothing effect (Fig. 7.4a). This effect was reduced by the introduction of the external variables (LS, Co, and LS × Ri), in the KED method (Fig. 7.4b). The map of MK (Fig. 7.4c) shows a more fragmented situation, more realistic as it mimics the variability among different fields. The statistical distributions of kriging prediction error of *ESP*, expressed as the standard deviation obtained with OK, KED, and MK (σ_z, calculated from $\sigma_K^{\prime 2}$ using Eq. 2), have mean values equal to 43.1, 37.4, and 39.1 mg P kg^{-1} soil, respectively, and standard deviation of 3.2, 2.3, and 12.8 mg P kg^{-1}, respectively (data not shown). Therefore, the three methods provide similar average uncertainty in the prediction of the target variable, but with highest variability for MK and lowest for KED. The area with σ_z lower than 35 mg P kg^{-1} soil is the 3.9, 14.0, and 53.1% of the total with OK, KED, and MK, respectively. The range with σ_z from 35 to 40 mg P kg^{-1} soil is observed most frequently in KED (12.3, 84.4, and 0.0 % of the area for OK, KED, and MK, respectively). The area with σ_z greater than 40 mg P kg^{-1} soil is the 83.8, 1.6, 46.9% for OK, KED, and MK, respectively (data not shown). Also the map of spatial distribution of σ_z (not shown) shows a marked smoothing effect for dataset A, using OK. As already observed for the predicted values, introduction of external variables (KED and MK) and splitting the area (MK) reduced the smoothing effect (maps not shown).

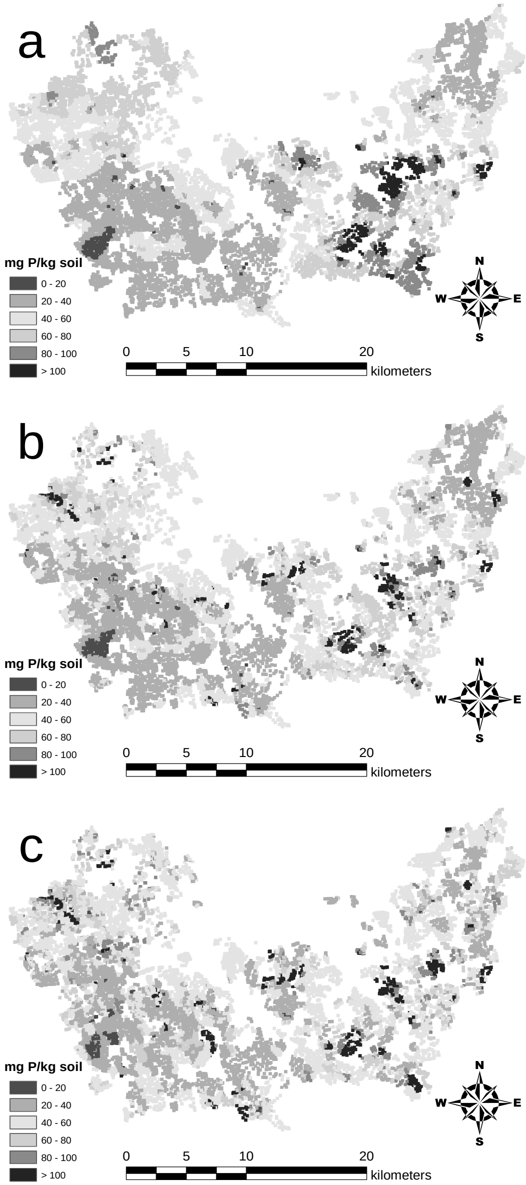

Fig. – 7.4. Spatial interpolation of extractable soil P concentration (mg P kg^{-1} soil) in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E): predicted values provided by (a) Ordinary Kriging, (b) Kriging with External Drift, and (c) Mosaic Kriging.

Predictions of *ESP* with CDV_i with i = 10, 25, 75, and 90% were calculated and mapped (Fig. 7.5). As expected, the values reported in Fig. 7.5c (CDV_{75}) and 5d (CDV_{90}) with cumulative densities higher than 50% are rather elevated, whereas the opposite applies for cumulative densities lower than 50% (Fig. 7.5a and 7.5b).

When the three methods are compared for the capability of discriminating between values below and above a threshold of 20 mg P kg^{-1} soil (Fig. 7.6), no substantial differences exist at the *CDV*s values of 75 and 90%: the area with values below 20 mg P kg^{-1} soil is approximately 0.5 and 0.1% of the total area, respectively. Some differences exist among methods at *CDV* levels of 10 and 25%. We

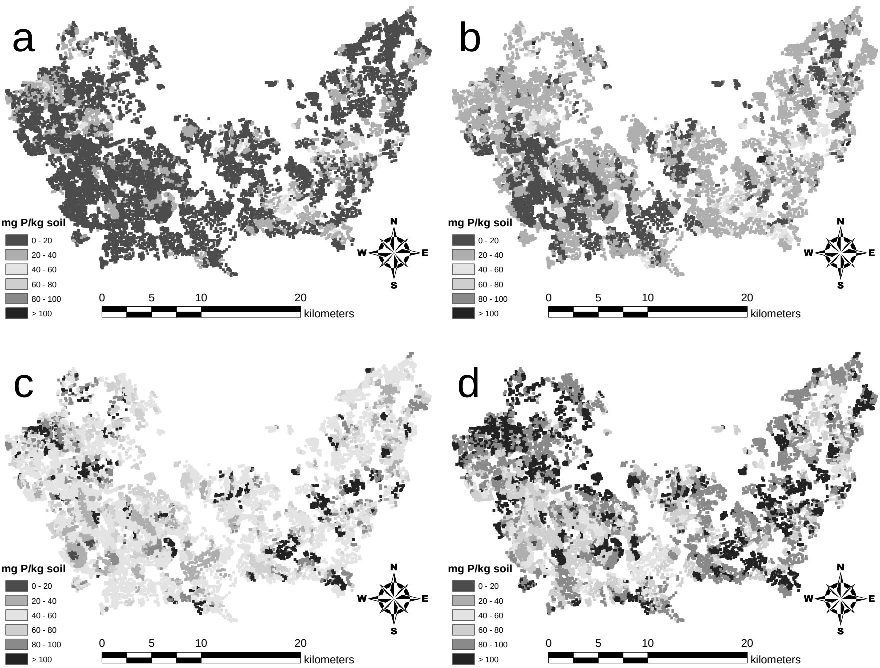

Fig. 7.5. – Spatial interpolation of extractable soil P concentration (mg P kg^{-1} soil) in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E): spatial distribution (Mosaic Kriging) of different *ESP* confidential values (CDV_i) corresponding to percentiles i= 10, 25, 75, and 90%for each cadastral parcel: (a) CDV_{10}, (b) CDV_{25}, (c) CDV_{75}, and (d) CDV_{90}.

conclude that, despite being more precise (as demonstrated by the results of cross-validation), the MK method does not identify areas that are of substantially different size compared to the other methods, when the agronomic threshold of 20 mg P kg^{-1} is used.

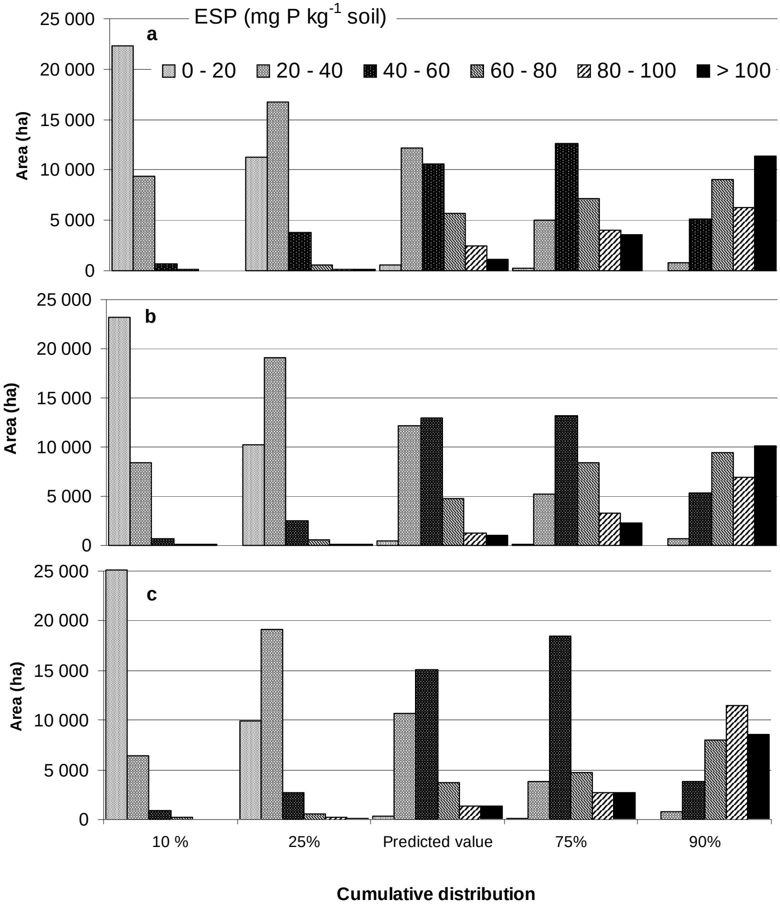

Fig. 7.6. – Spatial interpolation of extractable soil P concentration (*ESP*; mg P kg^{-1} soil) in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E): area (ha) with selected cumulative distribution of probability (%) using (a) Ordinary Kriging, (b) Kriging with External Drift, and (c) Mosaic Kriging.

7.6. Discussion

As the data used here were not collected on purpose to carry out geostatistical analyses, three problems arose when we started this work. First, *ESP* was determined with different analytical methods (OL and BK). This generated an uncertainty when OL analyses were converted to BK to cover a larger area, as the R^2 of the regression between OL and BK is 0.89 (Mallarino and Atia, 2005). Second, for some areas no soil analyses were available. These issues are to be taken into account in a further development and integration of the SITPAS database, for example by loading into the information system the analyses of the soil that are being carried out to comply with manure distribution or cross-compliance regulations. More soil samples should be collected and analyzed in the under sampled and unsampled areas. Third, using log-normally distributed variables like *ESP*, it is necessary to define a good interpolation procedure providing predictions with the lowest possible $\sigma_K^{'2}$. This is needed because when Y_K^* (predicted *LnP*) is high, the back transformation (exponentiation) of confidential interval of predicted values ($Y_K^* \pm \sigma_K^{'}$) to CDV_i amplifies $\sigma_K^{'}$ (even if $\sigma_K^{'}$ is low), and thus increases the uncertainty of *ESP*.

Theoretical models selected for spatial interpolation (Table 7.2) take into account farming systems characteristics for datasets B and E. Regression analysis shows that LS, Co, and LS $\times$ Ri are positively correlated with *ESP*. Higher *ESP* in farms with corn and livestock is explained by the fact that P fertilizers are frequently used on corn at high doses. High doses are applied because corn P uptake is high; also, mineral P fertilizers are used as a starter in spring (Bermudez and Mallarino, 2002). In addition, animal manures are frequently applied in excess of crop requirements in animal farms in this area (Bechini and Castoldi, 2006). In a recent survey conducted in seven farms in this area (Castoldi, unpublished data), it was found that the fields cultivated with corn were fertilized on average with 113 kg P ha^{-1} (sum of mineral fertilizer and manure), with a maximum of 252 kg P ha^{-1}. High amounts of P applied to corn are favored by the fact that this crop is not too sensible to nutrient excesses. All these factors have probably contributed to increase the *ESP* in these soils. In this area, high fertilizations were common for all crops with the exception of rice, where farmers apply fertilizers with extreme care because this crop can be damaged by nutrient surpluses, in particular because they may favor the development of diseases (in the surveyed farms the average is 50 kg P ha^{-1} and the maximum is 87 kg P ha^{-1}; Castoldi, unpublished data). This care in the fertilizer application has maintained low *ESP* values in the area where the rice is the main crop (South West; Fig. 7.4 and 7.5).

In the literature there is no accordance in the definition of the critical level of *ESP*. If we consider corn (the most cultivated crop in this area), some critical levels (BK) proposed for the US Midwest are: 19 mg P kg^{-1} soil (Beegle and Oravec, 1990), 13 (Mallarino and Blackmer, 1992), 11 – 20 (Mallarino, 1997), 16–20 (Mallarino, 2003), and 13 – 26 (Mallarino and Atia, 2005). Colomb et al. (2007) proposed a critical value for France of 4 – 7 (OL), corresponding to 7 – 13 if expressed as BK. Local regulations for the rural development plan (Regione Lombardia, 2005) indicate a threshold of 20 mg P kg^{-1} soil (BK).

Regardless of the spatial interpolation technique, using the critical levels cited above most of the area can be classified as being very rich in *ESP*. In many cases it could be possible to suspend the P fertilization for several years without a yield decrease; this statement is supported by the observation that the decrease of *ESP* in case of no P fertilization is relatively slow, because the *ESP* is released from the soil reserve, which is made of unextractable but active soil P and of soil P that provides long-term buffer (Karpinets et al., 2004). Colomb et al. (2007) reported a decrease of 0.10 mg P kg^{-1} soil yr^{-1} (OL) for 22 yr, in absence of P fertilizations and starting from an initial value of 6.2 mg P kg^{-1} soil. In a 14 yr period with annual cropping and without P fertilizations, Johnston and Poulton (1992 [as cited by Karpinets et al., 2004]) report an average decrease of 2.0 mg P kg^{-1} soil yr^{-1} starting from 47 mg P kg^{-1} soil, and a decrease of 0.17 starting from 7. Heckman et al. (2006) show that critical levels reported in literature are uncertain and could therefore induce an incorrect judgment on the *ESP*, indicating a need of P fertilization when it may not be needed and vice versa. Therefore, particularly where kriging predictions have a high uncertainty, it is difficult to express a sharp agronomic judgment and to take decisions on soil fertility and P management; this is especially true where predicted P values are near the critical level.

If the predicted values are to be used to predict potential P pollution of surface or ground water, the problem arises that *ESP* thresholds are in general very high. For example, Sharpley et al. (2001) report *ESP* above which the enrichment of P in surface runoff is considered unacceptable of about 75 – 200 mg P kg^{-1} soil (BK). With our predictions, having high uncertainty at high *ESP* values, it is difficult to draw any conclusion about the risk of P losses to waters. Also, as losses to surface waters are driven by runoff, without a combined estimate of potential surface runoff and subsurface flow, the high soil P levels alone have a little meaning in the estimation of environmental risk (Sharpley et al., 2001). The risk of losses to surface waters in the studied area could be low as the flat configuration of the land reduces the runoff risk. Acutis et al. (1996) in a similar pedo-climatic condition (Carmagnola, Po Valley, Northern Italy, 44°51’N, 7°51’E, 240 m above see level; loam and sandy-loam soil) in a corn field (slope of 0.5%) in a 6 yr period, have found a moderate runoff (average, minimum, and maximum of 39, 15, and 119 mm yr^{-1}, respectively) and a moderate soil erosion (average, minimum, and maximum of 398, 111, and 718 kg soil ha^{-1} yr^{-1}, respectively).

The maps of the predicted values (Fig. 7.4) show a strip with high values from North West (NW) to South East (SE), and two areas with low values in the South West (SW) and in the North East (NE). The farms in the SW area mainly cultivate rice, usually with small animal breeds and low livestock densities (Bergamo et al., 2007). Besides intensive livestock breeding, another reason for the high *ESP* values in the SE corner could be the past continuous application of irrigation water contaminated with sewage that came from Milano’s sewage system before the construction (in 2003 – 2004) of the Nosedo municipal wastewater treatment plant. Even when we inspect the optimistic values provided with CDV_{10} (Fig. 7.5a), we find high *ESP* values in a large area in the SE and some spots in the NW, confirming that these locations are those where *ESP* is highest. In case of runoff, these locations are probably subject to higher P losses than the others. The maps created with the values of CDV_{75} and CDV_{90} (Fig. 7.5c and 7.5d) show a hazardous situation, where almost all the PASM has very high *ESP*. Over-fertilization with P was a common procedure in the past due to the low cost of fertilizers and the low environmental concerns by farmers and authorities. Recently, several laws and regulations were introduced to monitor and manage manure N, with an indirect improvement also of P management. Furthermore, the increase of fertilizers cost has driven the farmers towards a better nutrient management. These trends allow to make the hypothesis that *ESP* will decrease in the future at a low annual rate.

As shown above, the split of the dataset B in three subsets (C, D, and E) has allowed a lower SCV during cross-validation compared to the use of aggregated dataset (A and B). This has provided a reduction of the uncertainty of the predictions compared to when one or more co-variates were introduced (KED). This confirms that the values of *ESP* do not depend only on the spatial location, but are also substantially influenced by pedo-climatic conditions and by the present and the past management. The fragmentation of this area in fields with different management practices for many years has changed the natural gradient of *ESP* and produced in some cases sharp boundaries between fields (i.e. two adjacent fields with different fertilizer applications for many years have different levels of *ESP*, with a sharp gap corresponding to the fields boundary). This is due to the low mobility of P in the soil that has not allowed to balance the *ESP* gap between two fields.

It is well known that on very large data sets all geostatistical interpolation methods typically perform very similarly. In this study, the best improvement is a 8.4% reduction of SCV when datasets A and E are compared. This reduction could be larger if smaller datasets would be available. If available, and despite the additional effort needed to use them, these data need to be taken into account in the evaluation of the environmental quality. If not available, it may be questionable if the effort of data collection would be worth, as (at least in this case) the decrease in the SCV after the addition of management data was not particularly high.

When the data are stored in a public database, the public administrations has to consider the aim of data collection in order to optimize the use of available resources, avoiding an over collection/production of data either not usable or needed. First it is necessary to plan the temporal and spatial density of data collection in relation to the utilization of these data and to the chemical-physical properties of the entities described by the data: the slow decrease of *ESP* in the soil allows a sparse temporal collection, but its high spatial variability requires a dense and uniform area cover, collecting samples where the interpolation procedure provides high uncertainty. In addition, it is necessary to uniform the analytical methodologies in order to reduce the uncertainty due to the conversion of data. To support spatial interpolation and to increase the quality of the outputs, crop monitoring at field level could improve the correlation between target variable and the co-variables. In this study, the co-variables were expressed as farm average, without considering the variability inside the farms. For example, instead of the farm-average livestock density, it could be better to more accurately define the amount of manure applied to each parcel using the manure utilization plan (unfortunately not available in the SITPAS database).

7.7. Conclusions

In this study we have generated maps of extractable soil P concentration (*ESP*) at the regional scale using soil analyses available in a public database. These data were integrated with crop and farm management information. For spatial interpolation, we compared a hybrid form of spatial interpolation to ordinary kriging and kriging with external drift. Cross-validation showed that this hybrid form resulted in unbiased predictions, i.e. with mean errors close to zero, and with the lowest standard deviation in cross-validation. Most of the area is characterized by high or extremely high *ESP* values (> 20 mg P kg^{-1} soil), presumably due to excessive P fertilizer applications in corn fields, particularly in animal farms. This situation requires crop management practices that minimize P applications only in soil/crop conditions where a real need is established after careful soil analysis and agronomic interpretation. The results of this work can therefore contribute to increased sustainability, i.e. by improved P management at regional scale, optimized fertilizers use, and consequently a better energy and economic balance of farming systems. Future agricultural data collection and storage should optimize the use of available resources, avoid excessive collection of data that are either not usable or needed, and consider the spatial and temporal variability. Also, additional management information could help interpretation and extrapolation of the target variable.

7.8. Acknowledgments

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CHAPTER 8

EVALUATION OF THE SPATIAL UNCERTAINTY OF AGRO-ECOLOGICAL ASSESSMENTS AT THE REGIONAL SCALE: THE PHOSPHORUS INDICATOR IN NORTHERN ITALY

Castoldi, N., A. Stein, and L. Bechini.

Keywords: soil fertility, fertilizer, manure, crop management, spatial interpolation, database, kriging.

8.1. Abstract

Agro-ecological indicators are simple conceptual models to carry out agro- environmental assessments. Their use requires data that can often be obtained at low cost, frequently avoiding direct measurements. The quality of input data thus may limit the usefulness of the indicators. This paper explores the uncertainty of the inputs, and its effect on the output of one indicator, the Phosphorus Indicator (*IP*). The indicator considers both over- and under-fertilization. We evaluated its use for P management in the Sud Milano Agricultural Park (Northern Italy). We used data contained in a large database of soil and farm properties as well as crop management information at the cadastral parcel level to calculate *IP* values. The uncertainty of a single input variable (extractable soil P) was tested to quantify the corresponding uncertainty of the indicator. The results show that within 80% of the analyzed area excessive application of P fertilizers is the main cause for *IP* reduction, in particular in dairy farms. The indicator has a good average value only for soybean, whereas the other crops score badly; rice is the second highest crop after soybean. This is due to the low fertilizer application in soybean and to a more careful fertilizer application in rice, which is sensible to diseases caused by overfertilization. Uncertainty associated with the assessment is often not relevant, as it is either very low, or, if high, it is related to extremely low indicator values: in both cases, judging P management is unaffected by its uncertainty. If uncertainty is relevant, however, it is not possible to provide an accurate judgment. The results show that in this area P fertilizers should be applied at lower doses, or not applied at all. An extension service might help farmers with fertilizer management, reducing resource use, environmental pollution and costs. Uncertainty analysis should be considered a necessary component of environmental assessments, as the importance of uncertain input data needs to be evaluated on a case by case basis.

8.2. Introduction

Agricultural sustainability assessments are frequently carried out to evaluate the impact of agricultural activities and to compare alternative management for cropping and farming systems (van der Werf, 2007). These assessments can be carried out using agro-ecological indicators, which are variables that provide simplified representations of agro-ecosystems processes, like primary production, water and nutrient cycling, energy use, landscape, biodiversity and pesticide fate (Bockstaller and Girardin, 2003; Castoldi and Bechini, 2006; Castoldi et al., 2007a).

As agro-ecological indicators are proposed to save money and resources, direct measurements tend to be avoided, and input data are estimated, derived from existing databases, or obtained from interviews with farmers. Uncertainty of these data may limit the power of these tools (Borrett and Osidele, 2007). This applies also to geographical data used for regional-scale applications of indicators. The uncertainty is an intrinsic property of any geographical data (Duckham, 2002): errors originate in source maps as a result of measurement processes used to construct those maps and then are propagated as consequence operations (Arbia et al., 1999). Also, uncertainty may come either from spatial interpolation (Bechini et al., 2000), from factors that cannot be accounted for at the scale where the work is carried out (e.g. farm-specific management information when simulating processes at the regional scale) or from estimated or non representative data (Rivington et al., 2006), or from non-homogeneous crop management, which is instead assumed to be homogeneous in the calculations.

This uncertainty is commonly described stochastically, where observations are assumed to be drawn from a population of possible observations with predictable characteristics under the central limit theorem. The true value, the accuracy (deviation from the true value), and the precision (spread of observations) can be represented by the mean value of the population, the root mean squared, and standard deviation, respectively (Duckham, 2002). Stochastic simulations provide multiple equally probable maps that can be used in GIS operations (Goovaerts, 2002). One simple stochastic simulation method is probability-field simulation. To apply this, it is necessary i) to know the local conditional cumulative distribution functions describing the behavior of the input data, and ii) to use those in order to generate realizations of the inputs (Dungan, 1998).

Knowledge of input uncertainty may be used to evaluate its impacts on the calculated value of the outputs, providing information about the intrinsic quality of the results and the confidence limits associated (Heuvelink et al., 1989). This uncertainty analysis estimates the probability distribution function of the output (Crosetto et al., 2000).

The uncertainty associated with spatial interpolation of inputs can be estimated with geostatistical procedures. Geostatistical approaches allow calculating the value of an environmental variable at unsampled locations utilizing the spatial correlation between neighbouring observations (Journel and Huijbregts, 1978; Isaaks and Srivastava, 1989; Goovaerts, 1999). Once the target variable has been described with a variogram, several geostatistical approaches can be used to estimate its statistical distribution in each location. For example, the kriging estimate and standard deviation can be used, assuming that the distribution of the kriging estimates is the same as that of the original variable.

Phosphorus (P) losses are the main cause of surface waters eutrophication, a process that produces impairment of water use for recreation, industry and drinking (Withers and Haygarth, 2007). Agriculture contributes to P loads in EU countries (EEA, 2005). In addition, P fertilizers are derived from limited, non-renewable phosphate deposits. Their recovery is expected to be more expensive in the future (Colomb et al., 2007). Therefore, it is important to use P efficiently, thus avoiding the risks of pollution and of overexploitation of non-renewable resources.

To evaluate the impact of phosphorus fertilization on the chemical quality of the soil and on the economy of non-renewable resources, Bockstaller and Girardin (2003) proposed a phosphorus management indicator (*IP*). This indicator is based on the measured extractable soil P concentration (*ESP*). It regards both over- and under-fertilization as negative, considering in the former case the risk of pollution of ground and surface water and the waste of non-renewable resources, and in the latter the risk of soil P depletion. The indicator provides a value from zero (worst value) to ten (best value), with seven as the sufficient value. Every point represents a lack or an excess of P (see below for details).

The objectives of this research were: i) to collect and integrate the information available about agricultural activities in an area in Northern Italy; ii) to estimate the *ESP* across the area by using a large database of measured soil properties and of crop management data collected at individual farms; iii) to evaluate the appropriateness of P management practices using the P indicator; and iv) to evaluate the range of variation of the indicator associated with the spatial uncertainty of *ESP*. The companion paper (Castoldi et al., 2008a) focuses on the first two objectives, while this paper deals with the agro-environmental assessment of P management and the quantification of its uncertainty.

8.3. Materials and methods

8.3.1. The phosphorus indicator

The phosphorus indicator proposed by Bockstaller and Girardin (2003) evaluates P fertilizer management at the crop scale. To calculate the indicator, the amount of P applied by the farmer is compared with the amount suggested in the nutrient management plan. If the two quantities are equal, the management is appropriate and the indicator scores 10. If the fertilizer applied is more than that required, resources are wasted, there is a risk of water pollution and the indicator is below 10; if the fertilizer applied is less than required, there is a risk of soil P depletion and again the value of the indicator is below 10 (Fig. 8.1).

The indicator is calculated as:

$$\begin{cases} I_P = 10 - \dfrac{max(P_{res};P_{sol})}{C} & if\ max(P_{res},P_{sol}) < 2G \\ I_P = 4 - \dfrac{max(P_{res};P_{sol}) - 2 \cdot G}{G} & if\ max(P_{res},P_{sol}) \geq 2G \\ I_P = 0, & if\ max(P_{res},P_{sol}) \geq 6G, \end{cases} \tag{8.1}$$

where P_{res} is an evaluation of the waste of non-renewable resources and protect water from P pollution, P_{sol} is an evaluation of the risk of soil P

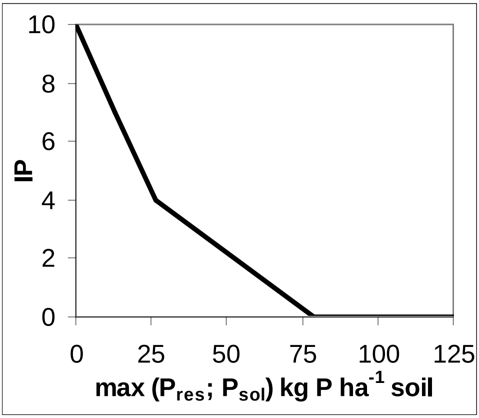

Fig. 8.1. – Phosphorus Indicator (*IP*) as a function of P management, represented by P_{res} and P_{sol}. P_{res}: evaluation of the waste of non–renewable resources corresponding to an over fertilization; P_{sol}: evaluation of the risk of soil P depletion corresponding to an under–fertilization. See text for details.

depletion, G is equal to 13.1 kg P ha^{-1} of P excess or P deficit, and $C = G/3$.

P_{res} is calculated by assuming that, to save non-renewable resources, an excess of P is not appropriate, while a deficit of P is not important:

$$P_{res} = \begin{cases} P_a - P_r & if \quad P_a > P_r \\ 0 & if \quad P_a \le P_r \end{cases}, \tag{8.2}$$

where P_a is the total amount of P applied to the soil (sum of P applied with all chemical and organic fertilizers), and P_r is the recommended amount of P to be applied with fertilizers. P_r was calculated based on *ESP* and on the expected crop P uptake. The methodology to calculate P_r is that officially adopted by Regione Lombardia (2005) for the Rural Development Plan. To calculate P_{sol}, it is assumed that a depletion of soil P occurs when the application of P is not sufficient for the crop (the excess is not relevant in this case):

$$P_{sol} = \begin{cases} -(P_{aa} - P_r) & if \quad P_{aa} < P_r \\ 0 & if \quad P_{aa} \ge P_r \end{cases}, \tag{8.3}$$

where P_{aa} is the part of P_a which is available to the crop. P_{aa} is calculated by summing the entire amount of P applied with organic fertilizers and P applied using the recommended forms of inorganic P fertilizers. These forms depend on pH and lime content; non-recommended P inorganic fertilizers do not contribute to P_{aa} (Bockstaller and Girardin, 2003).

8.3.2. Study area and database

The "Parco Agricolo Sud Milano" (PASM; Sud Milano Agricultural Park; 45°N, 9°E) is a regional metropolitan agricultural Park, surrounding the town of Milano, and occupying about 35,000 ha of agricultural area. The most important farming systems are dairy and cereal farms. The most cultivated crops are corn (*Zea mays* L.), rice (*Oryza sativa* L.), soybean [*Glycine max* (L.) Merr.], barley (*Hordeum* spp.), Italian ryegrass (*Lolium multiflorum* Lam.), winter wheat (*Triticum aestivum* L.), and permanent meadows with moderate to high yields. Out of 730 farms surveyed, 348 were animal farms, with livestock density (average over the entire Park) of 0.62 Mg live weight ha^{-1} for dairy, 0.09 for cattle, 0.14 for swine, and 0.01 for poultry (Bergamo et al., 2007). The flat configuration of the land, the ancient history of cultivation, the ample availability of irrigation water and technical means of production make this Park one of the most intensive Italian agricultural production areas. Agriculture is protected in this area, as it is competing for land use with the neighbouring cities.

The Agricultural Information System for the Sud Milano Agricultural Park (SITPAS, Bergamo et al., 2007), developed in the period 1999 – 2003, is made of a relational database connected with a GIS. It stores detailed and georeferenced information about the agricultural activities carried out in 730 farms of the Park. The information system was built by collecting existing data, and by integrating them with those obtained by directly interviewing all the farmers about crop and animal management practices. Every crop and animal management variable represents the farmer's average behavior, and therefore it is not referred to a particular year. The application presented here is related to crop management, therefore details are provided for this section of the information system. A farm may run one or more crop rotations, which can be georeferenced by indicating the cadastral parcel(s) used. Crop rotations are represented as a sequence of crops over time; for each crop, agronomic operations can be recorded (tillage, sowing, fertilization, irrigation, herbicide, fungicide and insecticide applications, and harvest). For operations involving the application of one or more products (fertilizers or pesticides), the type and amount of product(s) applied are indicated; for each product, the detailed composition is available in the database (e.g. N, P, and K contents of fertilizers, and active ingredients of pesticides). For harvest operations, the yield(s) and the fate of harvested product(s) and residues can be specified (sold, recycled within the farm - for example as animal feed -, re-incorporated into the soil). The 695 *ESP* measurements needed to carry out this work were extracted from the SITPAS database. Each *ESP* measurement is representative of the value for one or more cadastral parcels (the reference scale of commercial soil analysis is the field, which usually contains one or more cadastral parcels). Details about processing of these soil analyses are provided by Castoldi et al. (2008a). Only a subset of crop management variables was used in this study, according Bechini and Castoldi (2006), after a data quality check, have selected only the rotations with values that have passed a check for all the variables of all the crops in the rotations. As a result, a relatively small dataset was obtained (156 farms), that still contains information for a wide range of crop and farm types.

The indicator was calculated (using Eq. 1) separately for each single crop of each rotation of each farm, using measured or estimated values of *ESP* at the cadastral parcel level (see below for details). This calculation unit is called the “farm-rotation-crop-parcel combination” (*FRCPc*). As more than one crop may belong to the rotation insisting on a given cadastral parcel, the total area described at the *FRCPc* level is larger than the area of the parcel. The average *IP* for all the crops belonging to the rotation insisting on a parcel represents the average effect of P fertilizer use over the duration of the rotation for a real hectare. The results were then aggregated by calculating also the average by crop type, rotation type, and farm type. Rotation types (Bergamo et al., 2007) are: i) cereals (C_r; rotation including maize, winter wheat, barley, oat, rye and eventually rice; rice occupies less than 10% of the area); ii) cereals and rice (CR_r; from 10 to 100% of the area is cropped with rice); iii) cereals and industrial crops (CI_r; at least 10% of the area is cropped with sugar beet, oil or protein crops); iv) cereals and forages (CF_r; more than half of the area is cropped with cereals, and forage crops cover at least 10% of the area); v) forages and cereals (FC_r; more than half of the area is cropped with forage crops, and cereals are at least 10% of the area); vi) forages (F_r; the rotation has only forage crops); vii) industrial crops (I_r; the area is cropped with sugar beet, oil or protein crops); and viii) permanent meadows (PM).

To aggregate the results at farm level, farm types were identified with cluster analysis. Every farm was described with the livestock densities for different animal categories (dairy, cattle, swine, and poultry), and the percentage of farm area cultivated with corn, rice, wheat, barley, soybean, Italian ryegrass and meadows. The Ward aggregation method and the Euclidean distance between samples (farms) were used. Twelve farm types were identified, with different livestock densities and cultivated crops (Table 8.1).

8.3.3. Interpolation procedures

To calculate *IP* at the cadastral parcel level, it is necessary know the *ESP* values for each cadastral parcel. At unsampled parcels, *ESP* and its uncertainty were obtained by spatial interpolation (Castoldi et al., 2008a). The spatial interpolation used is a hybrid form of ordinary kriging and kriging with external drift, which are applied separately in three interconnected and non-overlapped geographical layers (sections), characterized by different farm management practices. A section-specific interpolation procedure was applied for the three different layers, using separate subsets of soil analyses. As *ESP* showed a skewed distribution, the logarithm of *ESP* (*LnP*) was used in kriging, and the predicted values were back-converted to *ESP* values (Journel and Huijbregts, 1978). Predictions of *LnP* (and therefore *ESP*) were considered to represent the entire area of the parcel, assumed homogeneous. This allowed predicting *ESP* for 93.2% of the agricultural area of the Park (Castoldi et al., 2008a).

As *LnPs* could be assumed to be normally distributed, their predictions could also be assumed to follow this distribution, with the predicted value corresponding to the mean of the distribution. To describe the local conditional cumulative distribution functions of *ESP* at each prediction point, we generated a frequency distribution of *LnP*

Table 8.1. – Farm types of the Sud Milano Agricultural Parck (Northern Italy, 45°N, 9°E) analyzed in this study, and their characteristics

Cluster [2]	Farms in the cluster (n)	Farm analyzed in this study (n)	Average area (ha)	Average percentage of farm area cultivated with [1] (%) C	R	M	B	W	IR	S	— Average livestock densities — (Mg ha^{-1} live weight) Dairy	Cattle	Swine	Poultry
DAI–INT	35	7	25	76	0	22	1	0	46	0	5.29	0.04	0.00	0.00
DAI–EXT–PM	43	11	21	9	0	87	3	1	0	0	1.35	0.04	0.03	0.00
DAI–EXT–RIC	71	9	84	31	38	5	0	0	2	1	0.58	0.00	0.01	0.00
DAI–EXT	165	31	44	55	1	31	9	1	3	2	1.10	0.05	0.02	0.00
CAT	17	6	22	43	0	50	10	0	3	1	0.00	1.81	0.01	0.01
SWI–POU	10	0	31	68	0	7	20	0	3	2	0.27	0.17	7.68	0.40
COR–SPEC	98	33	22	99	0	0	0	0	0	0	0.00	0.00	0.00	0.00
COR	97	11	61	72	4	7	2	2	1	7	0.09	0.18	0.11	0.00
WHE–COR	38	6	70	38	4	5	3	39	0	6	0.15	0.02	0.10	0.01
BAR	20	2	12	17	0	5	75	0	0	8	0.00	0.00	0.00	0.05
SOY	31	12	47	28	1	8	2	0	0	59	0.09	0.05	0.01	0.00
RIC–SPEC	105	28	71	7	87	1	1	0	1	2	0.04	0.02	0.00	0.00
Total	730	156	50	49	17	17	5	3	4	5	0.67	0.09	0.13	0.01

[1] C: corn; R: rice; M: permanent meadows; B: barley; W: winter wheat; IR: Italian ryegrass; S: soybean.

[2] DAI–INT: dairy farms with high livestock density; DAI–EXT–PM: dairy farms with low livestock density and high percentage of permanent meadows; DAI–EXT–RIC: dairy farms with low livestock density and significant percentage of rice; DAI–EXT: dairy farms with low livestock density and low percentage of permanent meadows; CAT: cattle farms with high livestock density; SWI–POU: swine and poultry farms with high livestock density; COR–SPEC: farms without livestock and only corn; COR: farms with low livestock density and with high percentage of corn; WHE–COR; farms with low livestock density and with high percentage of winter wheat and corn; BAR: farms with low livestock density and with high percentage of barley; SOY: farms with low livestock density and with high percentage of soybean; RIC–SPEC: farms with low livestock density and with high percentage of rice.

for each cadastral parcel, calculating 99 percentiles. The values obtained were back converted to the corresponding 99 percentiles in the *ESP* scale (CDV_m with m from 1 to 99).

8.3.4. Deterministic and stochastic calculation of the indicator

The inputs of *IP* are *ESP* and other management variables. We have carried out two deterministic (Fig. 8.2, procedures A and B) and one stochastic (procedure C) calculation of the indicator. In all the procedures, management variables were considered deterministic (i.e. they assume a fixed value during a given calculation), while the *ESP* is considered deterministic in procedures A and B, and stochastic in procedure C.

For the deterministic calculations (Fig. 8.2), *ESP* values were either measured (procedure A), or estimated with kriging (procedure B). In procedures A and B, crop management variables were those associated

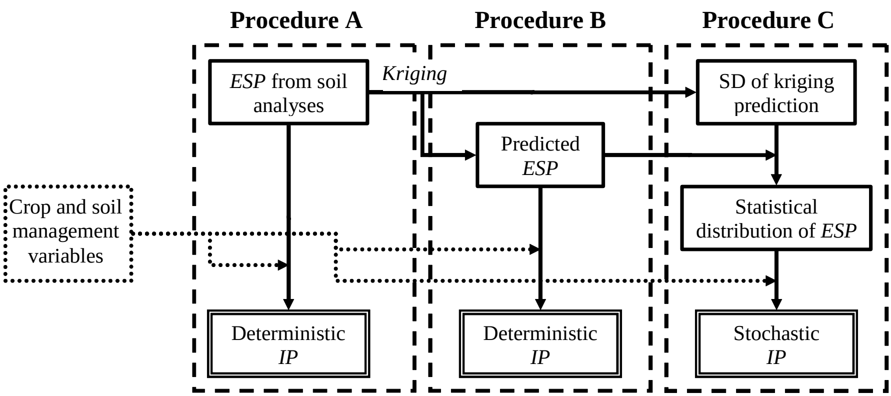

Fig. 8.2. Flow–chart of phosphorus indicator (*IP*) calculation procedures in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E). *ESP*: Extractable Soil Phosphorus concentration. Rectangles and arrows with solid border = soil phosphorus data; rectangles and arrows with dashed border = phosphorus management data; rectangles with double border = *IP*.

to the cadastral parcel in which *ESP* was measured or estimated, respectively. For the stochastic calculation (procedure C), we used the frequency distribution of *ESP* for each cadastral parcel (CDV_m), and we therefore carried out 99 calculations of *IP* for each *FRCPc*.

Procedure B provides a single value of *IP* using the predicted value of *ESP* and other management information; therefore, this value does not take into account the uncertainty of input data. Procedure C considers the variability of *IP* due to the uncertainty of input data (*ESP*). For each *FRCPc*, the average of the 99 *IP* values provided by procedure C is considered the most realistic *IP* value, and it was used in the evaluation of the uncertainty of *IP* in the area.

The result of procedure C is I_{ijklm} where *i, j, k, l* are the farm type, the rotation type, the crop type, and the parcel respectively, while *m* is the *ESP* percentile (CDV_m) used in the calculation of *IP* in the *FRCPc*. As this procedure provides multiple equally probable values, the average of the 99 values of *IP* for each *FRCPc* could be calculated (I_{ijkl}). In order to analyze the variability of *IP* in the area, we have calculated the average value of all I_{ijkl} in the parcel *l* (I_l, with $l = 1$ to 4001). In a similar way, the distributions of I_{ijkl} by farm (I_i), rotation (I_j), and crop type (I_k) were also analyzed.

For each I_{ijklm} the factor that causes *IP* reduction was evaluated (either an excessive, $P_{res} > P_{sol}$, or a deficit, $P_{res} < P_{sol}$, fertilization). To do this, the area corresponding to each *FRCPc* was divided according to to the cases where the excessive or deficit fertilization was the cause of *IP* reduction. Finally, the total area where the excessive or under fertilization is the cause of *IP* value was calculated for each crop, rotation and farm type.

To quantify the uncertainty of *IP*, the inter-quartile range (*R*) in each *FRCPc* was calculated using the 99 values of I_{ijklm}. Subsequently, the statistics of *R* by parcel (R_l), farm (R_i), rotation (R_j), and crop type (R_k) were calculated. The *R* was preferred over the coefficient of variation, because extremely high coefficients of variation can be obtained with a low average value of the indicator (e.g. 0.5) and a low standard deviation (e.g. 1). In these cases, the coefficient of variation, despite assuming a high value (200% in this example), would not indicate a relevant uncertainty of the *IP* estimate.

8.4. Results

8.4.1. Deterministic evaluation in procedure A

In the deterministic calculation of *IP* using measured *ESP* (procedure A; Fig. 8.2), only 79 soil analyses could be used. These are referred to 242 *FRCPc* (139 with rice, 98 with corn, 3 with winter wheat, 1 with barley, and 1 with meadows) corresponding to 391 ha (Table 8.2). The average *IP* at cadastral parcel level is 4.0 (Table 8.3): a large area (74%) has values lower than 7, and a considerable area (52%) has values lower than 3 (data not shown). The average is lower in corn (2.7) than in rice (4.4; Table 8.4). The results aggregated by rotation type show similar average values for C_r, CI_r, and CR_r (3.3, 3.2, and 3.9, respectively; Table 8.5). The factor that limits *IP* in most of the area is the over application of P fertilizers (P_{res}): this happens in 79.6 and 75.5% of the area for corn and rice (Table 8.4), and 73.9, 83.3, and 76.7 in C_r, CI_r, and CR_r, respectively (Table 8.5).

8.4.2. Deterministic versus stochastic evaluation

The Pearson correlation coefficient was calculated 99 times (Fig. 8.3), to compare the *IP* values provided by procedure B for each *FRCPc* with the I_{ijklm} values provided by procedure C (*m* ranging from 1 to 99). There are good correlations (r > 0.80) among *IP* provided by procedure B and

Fig. 8.3. Pearson correlation coefficient among indicator phosphorus (*IP*) values at the level of Farm–Rotation–Crop–Parcel combination (calculated with procedure B) and corresponding *IP*s provided by procedure C (calculated with different percentiles of extractable soil phosphorus distribution, *CDV*).

I_{ijklm} with high CDV_m (m > 37), while there are moderate inverse correlations (r from –0.33 to –0.42) with low CDV_m (m < 5).

The relation between *IP* provided by procedure B and the average *IP* for *FRCPc* calculated with procedure C (I_{ijkl}) is strong (R^2 = 0.86; Fig. 8.4). However, procedure B underestimates the low values of *IP* and overestimates the high values. The I_{ijkl} value calculated with procedure C will be used from now on as the most reliable estimate of *IP* for each *FRCPc* to discuss its variability in the area.

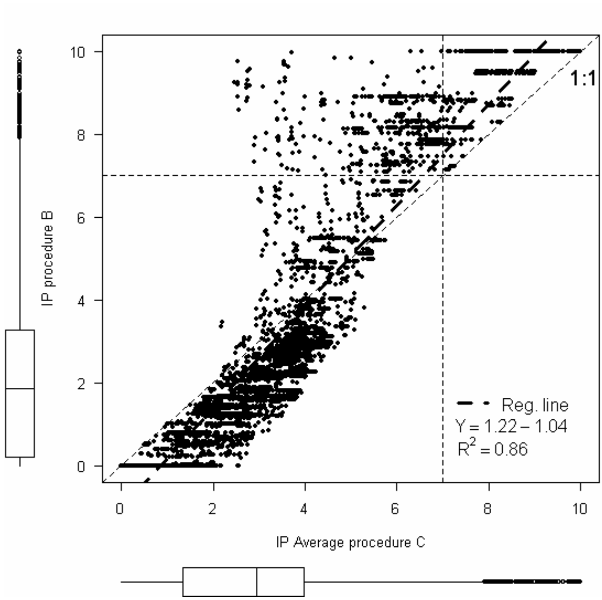

Fig. 8.4. Scatterplot of the relationship between the phosphorus indicator (*IP*) provided by procedure B, and the average of 99 *IP*s provided by procedure C (I_{ijkl}), at Farm–Rotation–Crop–Parcel combination in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E). The two boxplots show the two statistical distributions.

8.4.3. Variability of *IP*

8.4.3.1. Cadastral parcel level

In procedures B and C, the use of *ESP* predictions allows to apply the indicator to 4001 parcels, corresponding to 25.2% of PASM agricultural area (Table 8.2).

As expected from the high correlation between *IP* provided by procedure B and I_{ijkl} provided by procedure C (Fig. 8.4), the statistical distributions of *IP* calculated in procedure B and the average for C (I_l)

are similar at the cadastral parcel level, with low average values (2.8 and 3.1, respectively; Table 8.3). With procedure C, the most frequent I_l range is 3 – 4, covering 21% of the area; the parcels with I_l below the threshold (7) cover 95% of the area, and I_l below 3 represents 47% of the area (Fig. 8.5).

Table 8.2. – Coverage of the three procedures applied to evaluate P management in the Sud Milano Agricultural Park (Northern Italy). See Fig. 2 and text for details

	Procedure A	Procedure B	Procedure C
Number of *FRCPc* [1]	242	12,745	1,261,755
Area covered (ha)	391	8,810	8,810
Agricultural area covered (%)	1.1	25.2	25.2

[1] *FRCPc*: Farm-Rotation-Crop-Cadastral Parcel combination.

Table 8.3. – Variability of the phosphorus indicator calculated at the cadastral parcel level, with three different procedures in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E). See Fig. 2 and text for details

	Procedure A	Procedure B	Procedure C[1]
Average	4.0	2.8	3.1
1st quartile	1.4	0.9	1.9
3rd quartile	7.1	3.8	4.2
Standard deviation	3.2	2.7	2.0

[1] these statistics are calculated on the average of 99 indicator values provided by procedure C (I_l).

8.4.3.2. Crop level

Only for soybean, the indicator has a good average value (6.4), while the other crops have very poor averages (Table 8.4): rice is the crop with the second highest average after soybean. Soybean is the only crop that has the 3rd quartile (7.1) over the threshold. With the exception of rice and wheat (3rd quartile equal to 5.5 and 4.4 respectively), all the other crops have the 3rd quartile below 4. All crops have a minimum *IP* value equal to 0, while the maxima are over the threshold, even if they are not 10 for all crops. For every crop, usually it is the excessive application of fertilizers that reduces *IP*. For rice and soybean, in 32.9 and 28.4% of the area the reduction of *IP* is due to under fertilization (Table 8.4).

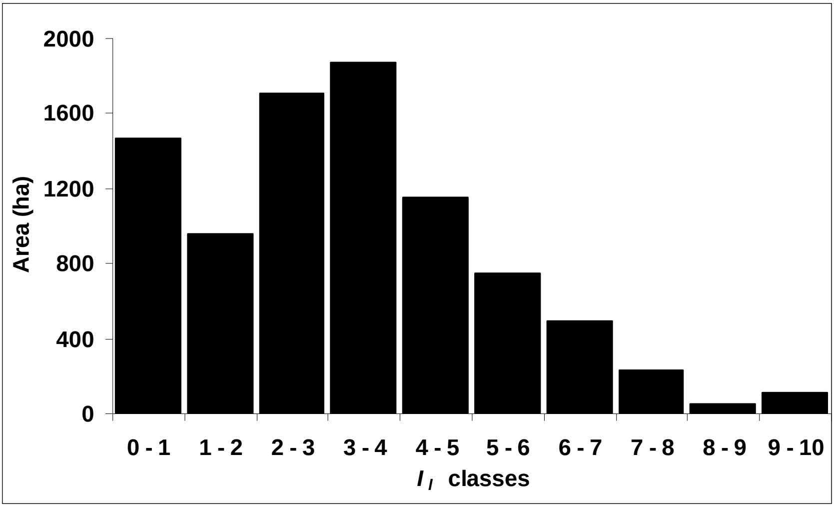

Fig. 8.5. Area covered by different classes of the average of 99 phosphorus indicator values (I_l) calculated in procedure C at the cadastral parcel level in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E).

8.4.3.3. Rotation level

The only rotation with good statistics for the indicator (I_j) is Ir (Table 8.5): this is due to the significant presence of soybean in this type of rotation. CI_r (average 3.8) and CR_r (average 4.0) have higher averages compared to the other rotations, and the 3rd quartiles are not very low (6.0 and 5.2 for CI_r and CR_r, respectively): this is due to the presence of soybean in the former, and rice in the latter. The other rotations (CF_r, C_r, FC_r, F_r, and PM; Table 8.5) have very low averages (with maxima above 7.0). CFr is an exception. P_{res} is the main factor determining *IP* in all rotation types: it is effective on about 80% of the area, with a higher percentage in CF_r (90.3), FC_r (89.8), and F_r (89.2). The P_{sol} factor is high only in I_r (38.7%) and CR_r (27.6%).

Table 8.4. – Variability of the phosphorus indicator calculated at the crop level, with two different procedures in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E). See Fig. 2 and text for details

	Crop type [1]									
	Procedure A		Procedure C							
	C	R	C	R	M	IR	W	S	B	All crops
Number of *FRCPc* [2]	98	139	5,600	3,258	2,166	465	368	335	314	12,745
Area (ha)	530	861	13,192	8,868	4,937	1,106	574	1,052	678	30,838
Average	2.7	4.4	2.3	4.1	2.6	2.6	3.5	6.4	2.6	3.0
Minimum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Maximum	10.0	10.0	8.4	10.0	10.0	9.0	9.9	8.3	9.2	10.0
1st quartile	0.0	1.8	0.8	3.3	0.9	0.3	2.1	6.1	1.3	1.4
3rd quartile	3.5	7.3	3.5	5.2	3.2	3.1	4.4	7.1	3.6	4.0
Standard deviation	3.4	3.1	1.7	1.6	2.2	2.9	2.0	1.3	1.6	2.1
	Reason for the values of *IP* [3]									
IP=10 (% of area)	10.2	7.9	1.0	1.2	6.2	4.0	6.3	3.9	0.4	2.3
IP<10 due to P_{res} (% of area)	79.6	75.5	83.7	70.4	81.1	88.2	83.0	63.1	92.2	79.1
IP<10 due to P_{sol} (% of area)	10.2	16.5	15.4	28.4	12.7	7.8	10.7	32.9	7.4	18.6

[1] C: corn; R: rice; M: permanent meadows; B: barley; W: winter wheat; IR: Italian Ryegrass; S: soybean.
[2] *FRCPc*: Farm–Rotation–Crop–Cadastral Parcel–combination.
[3] *IP*=10: the application of P is correct; *IP*<10 due to P_{res}: excessive application of P fertilizers (risk of over exploitation of non–renewable resources); *IP*<10 due to P_{sol}: too small application of P fertilizers (risk of soil P depletion); percentage of the area analyzed.

Table 8.5. – Variability of the phosphorus indicator calculated at the rotation level, with two different procedures in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E). See Fig. 2 and text for details

	Rotation type [1]										
	– Procedure A –			Procedure C							
	C_r	CI_r	CR_r	CF_r	CI_r	C_r	CR_r	FC_r	F_r	I_r	PM
Number of *FRCPc* [2]	46	36	159	1,365	891	3,463	3,867	1,504	785	58	809
Area (ha)	182	254	974	3,661	2,685	6,810	10,349	3,597	2,099	106	1,519
Average	3.3	3.2	3.9	1.6	3.8	3.1	4.0	1.6	1.8	6.7	2.7
Minimum	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	5.5	0.0
Maximum	10.0	9.8	10.0	5.8	8.5	9.9	10.0	9.5	9.8	7.7	10.0
1st quartile	0.8	0.0	1.3	0.2	2.4	2.1	3.0	0.1	0.9	6.0	0.9
3rd quartile	3.4	7.3	6.2	3.2	6.0	3.8	5.2	2.5	2.2	7.3	3.7
Standard deviation	3.3	3.8	3.2	1.6	2.4	1.7	1.8	2.0	1.6	0.7	2.4
	Reason for the values of *IP* [3]										
IP=10 (% of area)	15.2	0.0	8.8	0.0	1.6	2.4	1.7	2.6	3.7	1.3	9.1
IP<10 due to P_{res} (% of area)	73.9	83.3	76.7	90.3	74.8	79.4	70.6	89.8	89.2	60.0	78.0
IP<10 due to P_{sol} (% of area)	10.9	16.7	14.5	9.7	23.6	18.2	27.6	7.6	7.1	38.7	12.9

[1] C_r: cereals rotation including maize, winter wheat, barley, oat, rye and eventually rice (rice on less than 10% of the area);
CI_r: cereals and industrial crops; at least 10% of the area is cropped with sugar beet, oil or protein crops;
CR_r: cereals and rice; from 10 to 100% of the area is cropped with rice;
CF_r: cereals and forages; more than half of the area is cropped with cereals and forages are at least 10% of the area;
FC_r: forages and cereals; more than half of the area is cropped with forages and cereals are at least 10% of the area;
F_r: forages; the rotation has only forages;
I_r: industrial crops; the area is cropped with sugar beet, oil or protein crops;
PM: permanent meadows.

[2] *FRCPc*: Farm–Rotation–Crop–Cadastral Parcel–combination.

[3] see Table 4.

Table 8.6. – Variability of the phosphorus indicator calculated at the farm level, with procedure C in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E). See Fig. 2 and text for details

	Farm type [1]									
	DAI–INT	DAI–EXT–PM	DAI–EXT–RIC	DAI–EXT	CAT	COR–SPEC	COR	WHE–COR	SOY	RIC–SPEC
Number of *FRCPc* [2]	472	137	1,480	1,913	109	820	529	265	163	2,741
Area (ha)	1,813	714	4,538	7,838	228	1,795	3,823	1,220	850	7,985
Average	1.7	2.4	2.5	1.5	2.6	4.0	2.4	3.0	5.5	4.4
Minimum	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.3	0.1
Maximum	9.5	7.9	9.0	8.8	9.9	8.4	10.0	9.9	9.2	8.3
1st quartile	0.3	1.2	0.3	0.3	1.4	3.2	0.0	1.8	4.0	3.4
3rd quartile	1.4	3.2	3.9	2.2	3.2	5.6	4.0	3.6	7.1	5.3
Standard deviation	2.7	1.7	2.3	1.5	1.8	1.5	2.9	2.3	2.0	1.4
	Reason for the values of *IP* [3]									
IP=10 (% of area)	5.2	1.1	1.5	1.4	1.2	5.5	4.2	5.2	2.1	0.8
IP<10 due to P_{res} (% of area)	88.9	81.0	84.1	89.6	92.6	69.2	84.4	76.4	60.0	65.4
IP<10 due to P_{sol} (% of area)	5.9	18.0	14.4	9.0	6.3	25.3	11.3	18.3	37.9	33.8

[1] details in Table 1.
[2] *FRCPc*: Farm–Rotation–Crop–Cadastral Parcel–combination.
[3] see Table 4.

8.4.3.4. Farm level

At farm level, all clusters have average indicator values (I_i) below 7 (Table 8.6), with a maximum in SOY (5.5), very low values in animal farms, and a minimum in DAI-EXT (1.5). The 1st quartiles are very low in all clusters, and the 3rd quartile is greater than the threshold only in SOY (7.1). As expected, also at farm level the main factor determining *IP* is P_{res}, in particular in CAT (92.6%), DAI-EXT (89.6%), and DAI-INT (88.9%). P_{sol} is high only in RIC-SPEC (33.8%), and in SOY (37.9%).

8.4.4. Uncertainty of *IP*

At the cadastral parcel level, the distribution of *ESP* with different levels of CDV_m, and consequently I_{ijklm}, is peculiar to each single parcel. As it can be seen in the examples of Fig. 8.6, similar distributions of *ESP* may provide very different distributions of *IP*: parcel 1 is cultivated with soybean and, despite a relatively high P uptake, does not receive any P fertilizer, thus receiving a high score at high *CDV*, and a low score at low *CDV*. The fertilizations of crops in parcel 2 are excessive at high *CDV*, thus the *IP* is low, while it is correct at low *CDV*, receiving a high *IP* score. Two crops are cultivated on parcel 3, with different P application and uptake. Both crops receive an amount of P with fertilizers that is excessive at the high *ESP* estimated when *CDV* is high, therefore yielding a low *IP* value. When *CDV* is between 10 and 50, one crop is assigned a higher *IP* and the other a lower *IP*, due to different P requirements and P fertilizations. At the parcel scale, this fact produces the bimodal distribution that can be observed in Fig. 8.6. At low *CDV* (below 10), both crops receive an inadequate P fertilization, receiving a low *IP* score due to the P_{sol} factor.

At the parcel, crop, rotation and farm scale, the uncertainty around I_{ijkl} is on average moderate: for all the parcels, the mean R_l is 1.3 (Fig. 8.7a). To give examples at other scales, the average R_k is 0.4 for Italian ryegrass and wheat, and 1.0 for corn (Fig. 8.7b), Rj is low for Fr (0.4) and FCr (0.5) and CFr (0.7) (Fig. 8.7c), and R_i is 0.4 for DAI-INT and COR (Fig. 8.7d).

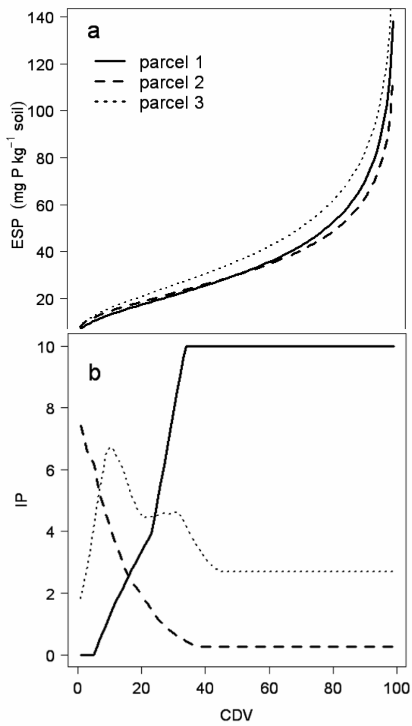

Fig. 8.6. Three examples of the uncertainty of extractable soil phosphorus (*ESP*)(a) and corresponding Phosphorus Indicator (*IP*) values (b) at the cadastral parcel level in the Sud Milano Agricultural Park (Northern Italy, 45°N, 9°E). CDV = Cumulative Distributed Values of *ESP*.

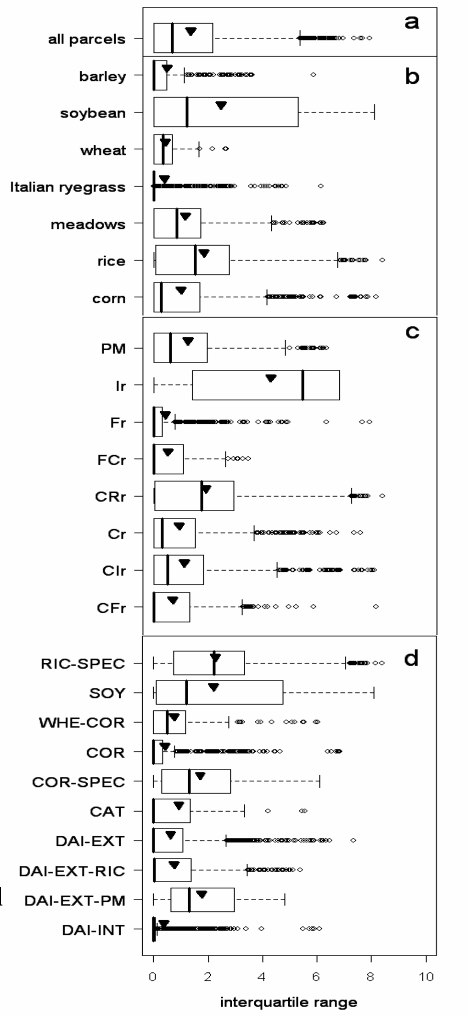

Fig. 8.7. Uncertainty of Phosphorus Indicator (calculated with procedure C), described by boxplot of the distributions of the inter–quartile range at (a) parcel, (b) crops, (c) rotation, and (d) farm level. Triangle = mean. For acronyms see Table 1 and 5.

Despite the relatively low average values of *R*, relevant uncertainty is found in most of the systems studied. There are consistent uncertainties in all classes, as the maxima of *R* are very high in some cases: 7.9 at the parcel scale, 8.4 for rice, CRr and RIC-SPEC. In other cases, however, the maxima of *R* are lower, as in the case of wheat (maximum of 2.6) and FCr (3.5). These results show that, when high, the calculated uncertainty may provide conflicting judgments on P management.

8.5. Discussion

In the deterministic calculation using measured *ESP* (procedure A), only 1.1% (391 ha; Table 8.2) of the agricultural area could be analyzed. This is not sufficient to express a judgment about P management in the Park. However, this procedure provides a preliminary result which is in agreement, for two crops, with procedure B and C. The use of kriging in procedure B and C allows calculating *IP* for a larger area (8810 ha; Table 8.2). For each FRCPc, in procedure B the indicator is calculated based on an average *ESP*, while in procedure C the indicator is the average of 99 calculations, made using 99 *ESP* values. In principle, when models are non-linear, model application should follow the framework of procedure C, i.e. repeated model applications should be carried out using different inputs, and outputs should be averaged. As it can be seen in Fig. 8.4, the indicator values obtained with the two procedures are relatively well correlated (R^2 = 0.86), even if in several cases the difference is high. From the point of view of the application of the indicator, we identify four situations: i) both methods return an insufficient value for the indicator (*IP* < 7); ii) both methods give *IP* > 7 (higher than sufficiency); iii) and iv) one method gives a sufficient value and the other insufficient. The area covered by concordant judgments [cases i) and ii)] is 90.6%, while it is 9.4% for cases iii) and iv) (data not shown in figures or tables). This means that, when only a general screening has to be carried out (i.e. we want to know if *IP* is above or below 7), most of the situations can be properly classified. If a quantitative evaluation is needed (e.g. distinguishing cases when the indicator is 3 or 6 is important), procedure C would be more reliable.

The low values of the 3rd quartile obtained in procedure C (Table 8.3, 8.4, 8.5, and 8.6), indicate that, according to this simplified methodology, P management in the Park is not satisfactory. This is particularly true for corn, the main crop in the Park (average I_k = 2.3; Table 8.4), while rice shows better results (average I_k = 4.1; Table 8.4). It is important that the high values of maxima demonstrate that, despite a generalized excess of P applications, some farmers are able to follow good management practices in this area.

The general bad P management highlighted by *IP* is probably partially due to excess of animal manure application in animal farms (Bechini and Castoldi, 2006). In addition, high P doses are applied to corn, a crop for which mineral P fertilizers are used as a starter in spring (Bermudez and Mallarino, 2002). High amounts of P applied to corn are favored by the fact that this crop is not too sensible to nutrient excesses. The utilization of P fertilizer and manure, and the use of irrigation water rich in nutrients (until 2003, before the construction of the municipal wastewater treatment plant) have created a soil P surplus in this area (Castoldi et al., 2008a). P application, however, was not reduced accordingly: therefore, excessive fertilization (P_{res} factor) is the cause of the low *IP* values in about 80% of the area in all crops (Table 8.4). The too small application of P and the consequent risk of soil P depletion are relevant only for rice and soybean (28.4 and 32.9% of the area, respectively), as for these two crops animal manure is in general not used, and mineral fertilizers are applied with care. Rice farmers in this area apply fertilizers with extreme attention because this crop can be damaged by nutrient surpluses, in particular because they may favor the development of diseases. These results are in accord with Torrent et al. (2007), who have found that in southern Europe on average the inputs of P fertilizer exceed the P exported from field. In addition, it should be considered that, before the construction (in 2003 – 2004), of the Nosedo municipal wastewater treatment plant, there has been a continuous application of irrigation water contaminated with sewage originating from Milano’s sewage system. This fact has contributed to increase *ESP*, especially in the South East area of the Park, and it is likely that this situation was not taken into account when preparing nutrient management plans. The analysis of *IP* variability at rotation level demonstrates that all rotations linked to animal farms (CF_r, FC_r, F_r, and

PM) have bad *IP* values, probably due to the excessive manure applications. The data were collected in the period 1999 – 2003, and today the situation is probably changed. The introduction and adoption of manure regulation (as a consequence of the “Nitrate Directive”, Council Directive 91/676/EEC), the increasing fertilizer costs, the activation of Nosedo wastewater treatment plant, and the increasing environmental sensitivity of farmer have likely improved the P management, reducing the environmental impact. Even if the uncertainty (discussed below) is sometime negligible, the general conclusion for the Park is that technical support should be given to farmers for the improvement of nutrient management, in particular in animal farms.

Use of the stochastic method (procedure C), shows that the uncertainty of input data has a significant effect on the indicator output. For example, the good results of soybean (average = 6.4) are affected by significant uncertainty with an average R_k of 2.4 (Fig. 8.7b). This does not allow expressing an unequivocal judgment on P management for this crop. For other situations, the uncertainties are moderate, and the assessment (expressed by the average *IP*) is relatively more confident. In general, the uncertainty is low with cropping systems using elevated P amounts (in particular in animal farms). The rationale is that the distribution of *CDV* provides in many cases (e.g. parcel 2 in Fig. 8.6) medium to very high *ESP*; in these cases, a very low score of *IP* is assigned, in particular if P applications are elevated. P applications can be so high that they are not even justified at low levels of *ESP*, as they exceed the sum of crop uptake and enrichment dose. As a result, low *IP* scores are obtained with very different *CDV*, and the resulting uncertainty is low. On the contrary, when P applications are low or null (e.g. soybean-based systems), the *IP* may be higher at the right end of the *CDV* curve, while it can be lower when *CDV* (and therefore *ESP*) are low or very low (e.g. parcel 1 in Fig. 8.6). This results in a larger uncertainty of the indicator.

Other authors have concluded that uncertainty in model application can be relevant and need to be carefully dealt with. For example, Rivington et al. (2006), have carried out a comparison of the outputs of a cropping system model fed with different weather data. Their results show that the various data sources may be acceptable or not, depending on local conditions and the assessment metrics used. Similar conclusions are obtained by Van der Werf et al. (2007): their evaluation of three pig production systems with five different assessment methods indicates that the relative ranking of the systems varies depending on the evaluation method and on the fact that the results are expressed per unit area or per unit mass of product obtained. Post et al. (2008) have applied a soil carbon dynamics model, considering the uncertainty of parameters and input data, using a Monte Carlo method. They show that, even if the uncertainty can be relevant, still it is possible to distinguish the trend of soil organic carbon over time of two distinct rotations. These works show that the differences among the systems analyzed can be so relevant that, even if inputs are uncertain, still firm conclusions can be drawn. In other cases, the high uncertainty is larger than the difference among the systems, and it is possible neither to provide a judgment nor to rank them. In conclusion, the effects of uncertainty need to be evaluated on a case-by-case basis.

In our study we have considered only the uncertainty related to a single input (*ESP*), because our database did not provide information to quantify the uncertainty related to the other inputs, i.e. amounts of manure and fertilizers, and crop uptake. Moreover, the conceptual model (Bockstaller and Girardin, 2003) was taken for granted, and its uncertainty not considered.

In conclusion, we recommend that assessments conducted with agro-ecological indicators should include the evaluation of the effects of input data uncertainty. As far as we know, this aspect is more considered in modelling applications (e.g. Acutis et al., 2000; Post et al., 2008). Indicators are simplified tools useful for screening and ranking, but uncertainty should not be overlooked in their application.

8.6. Conclusions

With the aim of evaluating the appropriateness of P management practices in a study area in Northern Italy, we have applied an agro-ecological indicator (the phosphorus indicator; Bockstaller and Girardin, 2003) to the data contained in a large database of soil and farm properties. Crop management information, derived at the cadastral parcel level, was integrated with geostatistical estimates of *ESP*, to evaluate whether the farmers are using the appropriate doses of P fertilizers (either organic or inorganic) or they are over- or under-fertilizing. The results show that, despite very high *ESP* values, in many situations excessive amounts of P fertilizers are applied. The uncertainty associated with this assessment is not always relevant, and allows in several cases to state that in this area P fertilizers should be applied at lower doses (or not applied at all). In general, farmers would benefit from an extension service to receive advice on soil fertility management. Uncertainty analysis should be considered a necessary component of environmental assessments, as the importance of uncertain input data needs to be evaluated on a case by case basis.

8.7. Acknowledgments

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CHAPTER 9

GENERAL DISCUSSION

9.1. Agronomic results

At regional scale, N and P management were evaluated applying the soil surface N balance indicator (chapter 6; Parris, 1998) and the phosphorus indicator (*IP*, chapter 8; Bokcstaller and Girardin; 2003) using the data stored in the SITPAS database (Sistema Informativo Territoriale Parco Agricolo Sud Milano). SITPAS contains detailed information related to average farm management of 730 farms in the Park.

In the study area (the Sud Milano Agricultural Park), excessive N fertilizations occur in intensive dairy and pig farming systems where very high amounts of N are applied with animal manure; moreover, chemical N fertilizers are not reduced accordingly, producing high surplus (> 300 kg N ha^{-1}) and corresponding high potential risk of N losses. The crops with the highest surplus are Italian ryegrass (*Lolium multiflorum* Lam.) and corn (*Zea mays* L.; 183 and 172 kg N ha^{-1}, respectively), while rice (*Oryza sativa* L.) and winter wheat (*Triticum aestivum* L.) have the lowest average surplus (87 and 85 kg N ha^{-1}).

IP considers negative both over- and under-fertilization, and the low values obtained indicate that, according to this simplified methodology, P management in the Park is not satisfactory. Despite very high extractable soil phosphorus values (estimated with geostatistical methodology; chapter 7), in many cropping systems excessive amounts of P are applied. This is particularly evident for corn, the main crop in the Park, while rice shows better results. The high values of maxima *IP* demonstrate that, despite a generalized excess of P applications, some farmers are able to follow good management practices in this area. As for the N balance, the general bad P management is partially due to excess of animal manure application in animal farms. The utilization of P fertilizer and manure has created high concentrations of extractable soil P in this area. P applications, however, were not reduced accordingly, therefore excessive fertilization is the cause of the low *IP* values in about 80% of the area in all crops. The too small application of P, and the consequent risk of soil P depletion, is relevant only for rice and soybean [*Glycine max* (L.) Merr.], as for these two crops usually animal manure is not used and mineral fertilizers are used with care.

The data used in the calculation of these indicators were collected in the period 1999 – 2003, and today the situation is probably changed. The introduction and adoption of manure regulation (as a consequence of the “Nitrate Directive”, Council Directive 91/676/EEC), the increasing fertilizer costs, the activation of Nosedo wastewater treatment plant, and the increasing environmental sensitivity of farmers have likely improved the N and P management, reducing the environmental impact. Even if the uncertainty of the indicator values is not always negligible, the general conclusion for the Park is that technical support should be given to farmers for the improvement of fertilizers use, in particular in animal farms.

In order to evaluate the economic and environmental sustainability of farming systems, during the period 2005 – 2006 more detailed data were collected in seven representative farms by face-to-face interviews with farmers. A set of indicators selected from literature was applied to 266 fields × crops combinations monitored. These indicators describe the management of economic resources, nutrients, energy, and plant protection agents (PPA). The indicator values were then aggregated at field and farm level, introducing in the framework indicators on soil quality, landscape and biodiversity management (chapter 5).

Good economic results were obtained for rice and corn but the former have high potential impact on environment due to intensive use of plant protection agents and low energy production, while the latter has generally high nutrient surpluses. The introduction of barley and winter wheat in crop sequence reduces the economic sustainability, but increases the environmental sustainability. Generally, fodder crops have very low environmental impact (low energy consumption and nutrient surpluses; no PPA application), but poor economic performances. The crop with the best energy gain is corn, despite the high fossil energy input required in the cultivation. The exportation of straw in winter wheat, rice and barley reduces the organic carbon input into the soil. For corn, in many cases the manure applications and the high amount of residues left on the soil represent an important soil organic carbon source.

The intensive dairy farm monitored obtained a good compromise between economic and environmental sustainability. The average gross margin obtained in crop production is high, and the nutrient management is rather correct, with nutrient surpluses not too high. The fossil energy consumed is high, but also the calorific energy contained in products exported from fields. The potential risk related to application of PPA is moderate. The soil quality indicators show a good organic matter management due to the use of manure in all crops and to the presence of permanent meadows. The crop sequence is not completely satisfactory, because in many fields a continuous corn sequence is present. The biodiversity and landscape indicators provide a fairly good judgment due to moderate crop diversity and moderate presence of hedge-rows.

Low values for the economic indicators were obtained in the dairy extensive farm; however, this farm showed an excellent management of nutrients, with nutrient balances close to zero. Both fossil energy consumed and calorific energy incorporated in the products are on average low due to the large area with permanent meadows. The application of PPA in a restricted area (cultivated with corn) reduces the potential impact of agrochemicals. The large presence of permanent meadows and the application of manure increase the value of indicators related to the soil quality. The dense network of hedge-rows increases the landscape differentiation, while the restricted number of crops cultivated in large fields reduces landscape fragmentation.

The intensive and extensive swine farms monitored have high values in economic indicators, but nutrient surpluses are usually excessive, due to the high amount of manure and mineral fertilizer applied. Both energy inputs and energy outputs are high, but the energy gain is lower than the corresponding value obtained in the dairy intensive farm. As concerns PPA management, the potential environmental risk is moderated and referred essentially to the use of herbicides during springtime in corn cultivation. The soil quality indicators provide unsatisfactory judgments in both farms, with the exception of soil organic carbon indicator in the swine intensive farm, where a high amount of organic matter is applied every year to the soil with animal manure. The landscape is simplified by the continuous corn cultivation, and the development of hedge-rows network is reduced in the extensive farm.

The two farms with rice cultivation obtained very good economic gross margin, despite the high variable costs. The nutrient management is generally correct in rice, but high nutrient surpluses are obtained in corn. The energy inputs are high, but the outputs are moderate, due to low yield of rice. Frequent herbicide and fungicide applications occur in rice; hence the potential environmental pressure of PPA is very high in both farms. The soil management is not adequate because a low amount of organic carbon is applied to soil and the continuous crop is preferred to a more diversified crop sequence. The landscape is extremely simplified in the farm with only rice and corn fields, while it is more diversified in the second rice farm, where four crops are cultivated with the presence of a moderate hedge-rows network.

In the last farm monitored (the mixed farm), numerous crops are cultivated in the same year, with low inputs. The solution adopted in this farm does not provide excellent results both from the economic and the environmental points of view. The economic performances are not too high, but the nutrient balances are fairly correct. The energy inputs and outputs are low, but good energy efficiency was obtained. The potential impact of PPA is moderate, and the crop sequence is rather good, but the soil organic carbon management is not satisfactory. A good value for the crop diversity indicator is obtained, but the hedge-row network is not particularly developed.

These indicators can be used by administrative and technical bodies (e.g. by the Park) as a first screening tool, to identify the most hazardous cropping and farming systems.

9.2. Possible solutions for agronomic sustainability

The assessment carried out with indicators provides a picture of the state of agricultural management: many critical situations were highlighted for different aspects, especially in the nutrient management in animal farms.

Often farmers do not have the tools and the knowledge to improve the environmental sustainability of farming systems. The imposition of specific managements (i.e. limit in N applied with manure; Regione Lombardia, 2006) or the funding of generic agro-environmental measures (Regione Lombardia, 2005), should be combined with a structured and widespread extension service.

In general, farmers would benefit from these services, which will provide technical support and knowledge necessary to drive the management decision making towards a more sustainable agriculture, with case-specific solutions. An intense collaboration with qualified agronomists is too expensive for many farms. Therefore, specific measures could be promoted by the Park (or other administrative and technical bodies), funding a public extension service that provides professional environmental advice to farmers, with low cost for them.

As a first step, the services should be oriented towards a restricted number of farms that have high potential environmental pressure, as for example the intensive livestock farms, especially the pig farms. In animal farms the uncertain concentration of nutrients in manure and its uncertain availability to crops make the use of chemical fertilizers a cheap method to sustain crop production regardless of the fate of the animal nutrients applied (Schröder et al., 2000; Bechini and Castoldi, 2006). Available scientific knowledge, direct measurements and simulation models should be used to optimize agricultural managements, reducing the fertilizers (and in general resources) waste, without compromising good crop yields. Moreover, the extension services could monitor the agricultural sector, recording detailed data on the real farm management, creating a vast database containing information continuously updated, which can be used in the agricultural assessment. Both farmers and Park (and consequently citizens) would benefit from these services.

Crop simulation models could be applied with the data collected by extension services, and alternative management solutions could be tested in order to evaluate the best economic and environmental sustainable solutions. Some preliminary simulations on nitrogen management were carried out with data collected in the monitored farms (Bechini and Castoldi, 2006d; Appendix d). The simulations demonstrated that reduction of nitrogen fertilization is possible without compromising the yield, saving money, and reducing the nitrogen leaching.

Field experiments were carried out in 2006 in order to evaluate the correctness of nitrogen management in monitored farms and to evaluate the consistency among experimental data and agro-ecological indicators calculated with data declared by fames (data not published). The presidedress soil nitrate test (PSNT; Magdoff et al., 1984.) and stalk nitrate test (SNT; Binford et al., 1990) were measured in six experimental corn fields with different levels of N application (from manure and/or mineral fertilizers). PSNT was measured in order to evaluate if the preplant N applications were correct, and to indicate if additional N was needed at six-leaf stage (Binford et al., 1992). SNT was proposed as a method of determining if excessive or insufficient N was available to corn during the latter part of the season. Moreover, in these experimental fields, N balance was calculated using declared and measured data: before crop emergence, and after harvest, soil samples were collected for the analytical measure of soil mineral N concentration. Crop samples were collected for the measure of crop yield and N uptake. Inorganic N left in the soil after harvest is also a good risk indicator: high amount of mineral N in the soil after harvest will be probably leached in autumn during the intense rain that usually occurs in this season.

The values obtained from field measures demonstrate that a reduction of N fertilizations is possible without compromising the yield. With N application over a threshold specific for each pedoclimatic condition (about 200 – 250 kg ha^{-1}), there is not evident increase of corn yield (similar results were obtained by Bassanino et al., 2008). The PSNT values measured were rather high indicating correct N applications during the seed bed; a reduction of mineral additional N could be suggested in these fields. After harvest, high SNT (over 2,000 mg N-NO_3 kg^{-1} DM; data not shown) were obtained in fields where high amounts of N (mineral fertilizers and animal manures) were applied. After harvest, high mineral N soil contents were measured in fields where animal manures are applied.

Even if related to a small sample for one year, this experience demonstrates that the PSNT and the SNT are two rapid and relatively cheap tests that could be applied in a large number of farms, because the former requires the analysis of soil nitrate of a soil sample (0 – 30 cm), while the second requires the analysis of nitrate of the corn stalk collected after harvest (20 cm segments of corn stalks between 15 and 35 cm above the soil from about 10 plants). The PSNT can be used to drive the management toward a correct fertilization, while the SNT evaluates ex-post appropriateness of fertilizations. The measured balance is expensive and time consuming and is not applicable in a large area, but the measurement of soil nitrate after harvest is a good indicator of the potential nitrogen leaching. These tests could be organized at large scale by the extension services, in order to improve the nitrogen management and to monitor and to evaluate the real potential N leaching risk.

In addition, as farmers have no tools to evaluate the real amount of nutrients applied with animal manures, laboratory analyses would be necessary for each manure application. However, the analyses are too expensive compared the low economic value of manure. A rapid and cheap evaluation of nutrients concentration in manure can be carried out with near infrared tools (Marino Gallina et al., 2005b, De Ferrari et al., 2006) installed on the machine used to spread slurry or on the storage tank. During each manure application the real amount of nutrients applied could be determined by this tool, and a more correct fertilization plan could be compiled. This tool may require specific knowledge, hence the utilization of this technology is best suited in an organized group comprising a large number of farm.

A development of innovative managements is the way toward to drive the future agriculture in order to maintain elevated levels of environmental quality without compromising the economic sustainability. The high production obtained by intensive crops cultivation should be exploited in fertile areas, optimizing the use of resources and consequently the efficiency of the systems, in order to cover the food/energy demand, while the other areas should be dedicated to different utilization, for example protected natural area, tourism, recreation, etc.

This concentration of intensive agriculture in restricted areas is comparable to industry meta-district. Meta-districts are production areas with heavy links with research and innovation. These districts will represent agricultural development centre with elevated technology, organization, and production, where to promote the collaboration among farmers, industry, and research centers. The aim of these districts is to intensify the production (in an environmental sustainable way) and to improve the economic competitiveness in the home and international market. In these environmental and economic sustainable areas, the waste of resources will be reduced, because the organization is planned at district level, and not at farm level, with a large possibility of resource allocation and reutilization.

9.3. Possibilities and limitations of the indicator framework

9.3.1. Data collection and storage

Farmers provided management information voluntarily and they were not remunerated both in the SITPAS project (data utilized in chapters 6-7-8) and in the interviews carried out in 2005-2006 (data utilized in chapter 5).

During the interviews it is necessary to gain farmers' confidence, in order to obtain also confidential data (i.e. economic data, real amounts of manure and fertilizers applied, all plant protection agent used), and to guarantee the maximum anonymity and protection of data collected. The perception of collaboration instead of an inspection has created good relationships among farmers and researchers allowing collecting good data on farm management. The possibility of a double check on many data (e.g. between seed application declared during interview and seeds bought) has reduced the uncertainty of the assessments even if some doubts and errors persist. Despite the enormous farmers' availability to provide information in these accurate interviews, remuneration for farmers is suggested in a future similar work.

Usually the administrative and technical bodies have a large amount of agricultural and environmental data; however, these are rarely organized in a structured database, and often it is difficult to obtain these data. The SITPAS project (Bergamo et al., 2007; Provincia di Milano 2006) has provided a large amount of organized data, allowing an accurate regional agricultural assessment.

The data storage is a critical step in the assessment process, often undervalued. Examples of problems occurring with data that are not perfectly structured include: errors introduced during data input, missing data, incomplete domains, excessive complexity of database. In our opinion the first step in the data collection is a clear definition of the aim of the work, therefore it is necessary to select the right indicators and define the data required. After these steps it is possible to project and to realize the database and finally to collect data. Unfortunately many assessments start from a data collection and therefore the database is built, or the database is designed before the data collection, but there is not a clear idea of the final utilization of the information collected. In this situation an over collection of information is common, with a clear waste of resources.

The complexity of agricultural systems requires complex databases to store non homogeneous information and all the numerous possible exceptions that occur in farm management. With complex databases, misunderstandings and errors in information management frequently happen. Appropriate data structures are needed for reducing the time necessary for elaborations and the possibility of errors, despite some peculiarity of agro-ecosystems is not recorded.

9.3.2. Limitations of the indicators used

Indicators on green house gases (GHG) production were not included in the framework proposed, because more complex tools (simulation models) are necessary for the evaluation of GHG dynamics, and larger and accurate data are required. The CO_2 production is partially related to fossil energy input, while C sequestration is related to energy output and soil organic matter index. The estimation of CH_4, NO_x, and other GHG emissions is not possible using data obtained during interviews.

Another relevant topic is the water management, but the lack of measurements of the irrigation volumes applied did not allow a comparison among different irrigation managements. The installation of measurement tools on the hydraulic turbine was too expensive for our research.

The method proposed is effective only for the Park or a similar area. The selected indicators were set on the pedoclimatic condition of the Park and on the typical agricultural management of this area. For example, indicators about erosion or deforestation were not included in the framework because these are not significant problems in the study area. The application of this framework for a comparison among these cropping systems with other systems (i.e. horticultural, wine-farming, or fruit-farming systems) is not correct.

Social sustainability is an important part of agricultural systems assessment, but this was outside the aim of this research.

Many indicators are very simple and coarse, and do not allow the evaluation of real environmental and economic impacts. Our framework evaluates the potential impact of the management, highlighting the situations with higher potential risk. For examples high nutrient surpluses were found, but it is not possible to evaluate the environmental fate of nutrients applied in excess. The indicators proposed are useful to drive more accurate research and for a preliminary comparison among different management systems.

9.3.3. Parameters and coefficients

In the calculation of indicators, measured data are often replaced with parameters and coefficients (e.g. the real N concentration in the harvested product is estimated with data provided by literature). The variability of these coefficients and parameters provided by literature is sometimes high. Many factors influence the values of parameters and coefficients:

i) the system's boundary: for example the definition of boundary is essential in the quantification of indirect energy costs of materials (e.g. the energy in the seed is higher than the corresponding calorific value if the indirect energy spent in the cultivation of the seed is accounted);
ii) the environmental condition where parameters and coefficient were estimated (the crop N content depends on pedoclimatic condition, nutrient availability, variety, etc.);
iii) the subjective definition based on expert knowledge.

The choice of correct parameters and coefficients is a critical step in the agricultural systems assessment, because the application of a specific value for a parameter may change the judgment. In this work, the choice of the coefficients has been based on the value provided by the most comparable studies published in referenced paper selected after an accurate review.

9.3.4. Assessment uncertainty

During the last decade, numerous environmental farm management tools and evaluation methods based on indicators were developed (reviewed for example by Rosnoblet et al., 2006). This development raised several questions among potential users, e.g. about the strengths and limitations of each method, and about the convergence of the results provided by different methods (Bockstaller et al., 2007a).

In this context, uncertainty analysis should be considered a necessary component of environmental assessments, as the importance of uncertain input data needs to be evaluated on a case by case basis. The uncertainty associated with indicator assessment is not always relevant, and allows in many cases to state that in a particular area, or for a particular aspect analyzed, the management is correct (or not) even if the uncertainty is elevated.

9.3.5. Interactions and trade offs

The framework proposed does not consider interactions and trade-offs among different indicators (and corresponding farming systems aspects). Every indicator is considered individually, while a systems assessment requires an integration of information in a unique analysis. Interactions among farmer practices themselves also can influence their impacts; for example, incorporating manure into soil soon after spreading tends to decrease nitrogen losses through ammonia volatilization but to increase nitrate leaching (van der Werf et al., 2007a).

A preliminary evaluation of trade offs (Castoldi et al., 2007b; Appendix F) was carried out by applying the Sustainability Solution Space method (Wiek and Binder, 2005), using 14 indicator values calculated at field level (131 fields analyzed). The analysis of trade-offs allows to provide a systemic evaluation of the systems, and it should be considered a necessary component of environmental assessments.

Moreover, a preliminary integration of different indicators values into a single index was carried out by applying Sustainability Functions (Castoldi and Bechini, 2007; Appendix E).

9.4. Future research

During the research period many problems and ideas on how to proceed in the evaluation of sustainability of agricultural systems came out. The main objectives for further researches are:

i) a more efficient and effectiveness data collection. The farmer interviews are time consuming and the data collected require a subsequent check, because during the interview some information may not be recorded or may contain errors (oversight or carelessness of interviewer or farmer). Automated remote sensing tools applied to tractors and other machines allow to record accurate and precise data on farm management. The single farm operation should be recorded, including the time necessary for the operation, the combination tractor-machinery, the exact position of the operation and a good estimation of fuel consumed (Mazzetto, F; personal communication);

ii) a more precise evaluation of nutrients concentration in animal manures, and of yield in fodder crops;

iii) the evaluation of alternative scenarios, in order to drive the farm management toward a more sustainable solution. Some preliminary evaluation in the N fertilization in corn was carried out in Bechini and Castoldi (2006d; Appendix d);

iv) the evaluation of uncertainty related to several inputs analyzed simultaneously with appropriate statistical methods;

v) a more accurate evaluation of trade-offs using Sustainability Solution Space and other tools;

vi) the integration of different indicators into a single index toward a more precise and rigorous definition of Sustainability Function or using other tools.

When all these objectives will be reached, the assessment tool proposed will be considered suited for an efficient and accurate evaluation of the sustainability of agricultural management, and it could be used in the decision making process.

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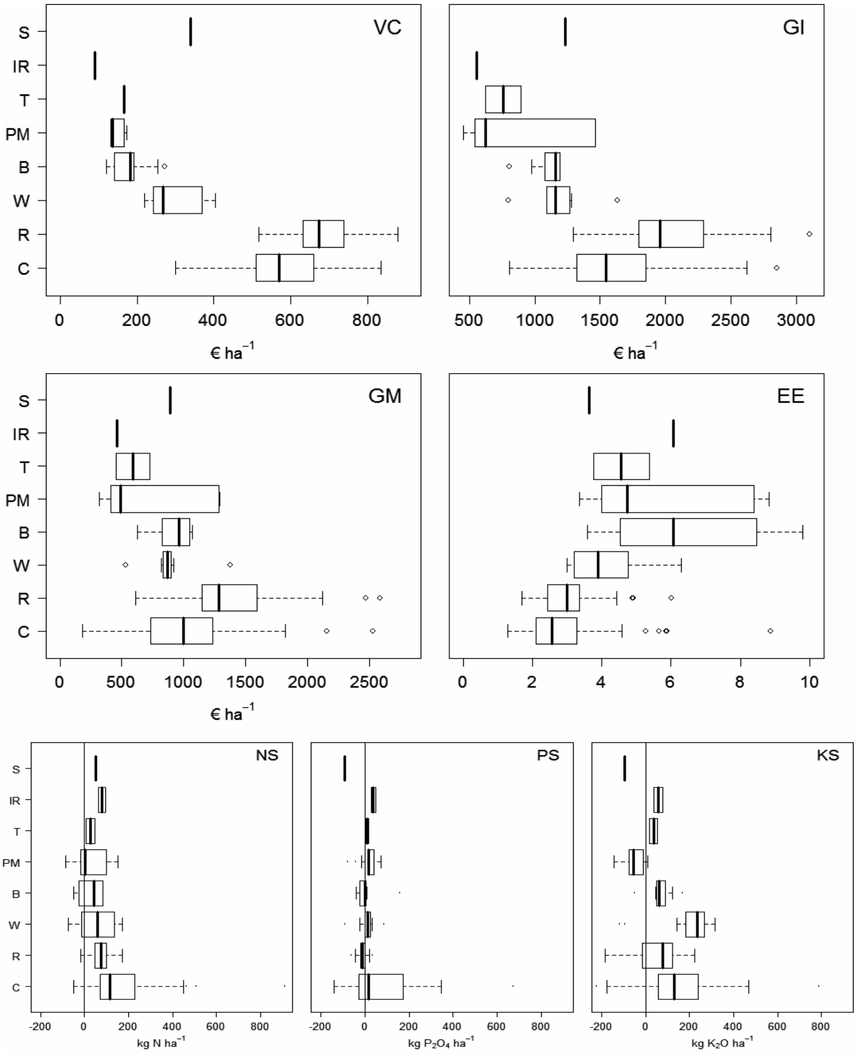
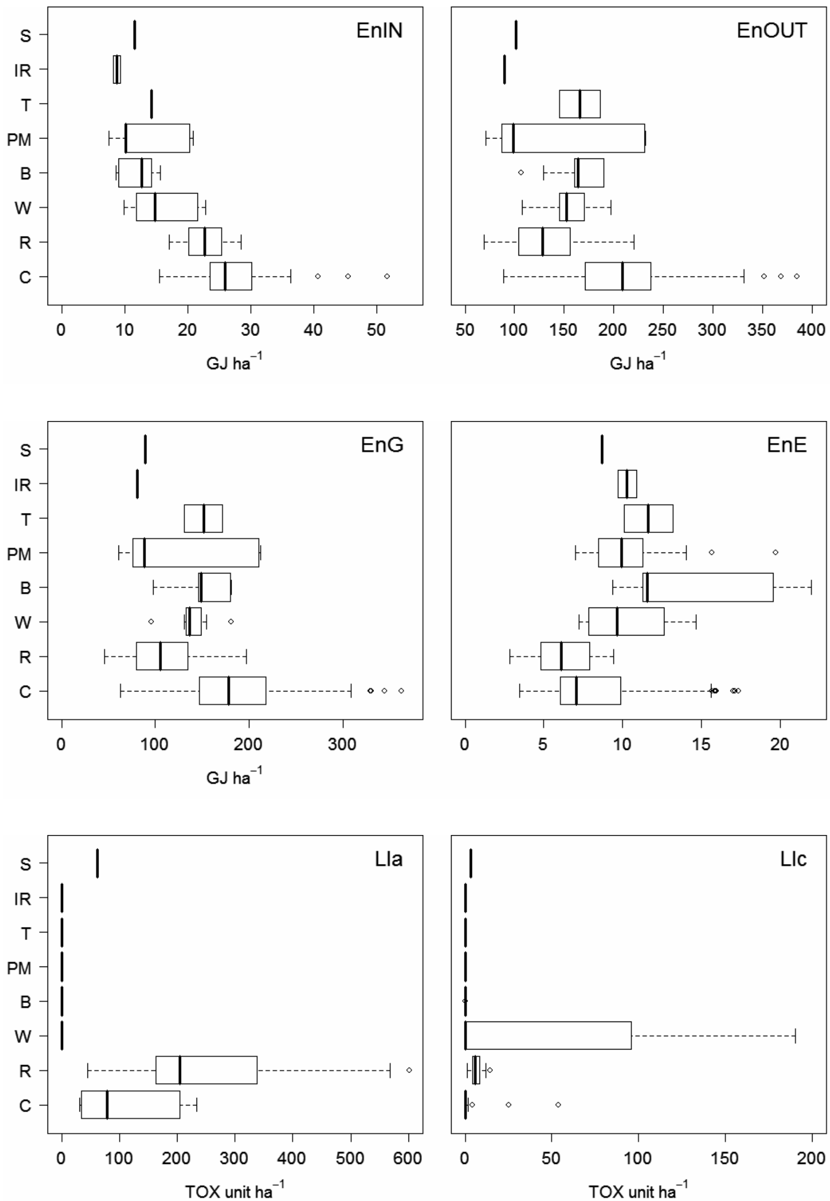
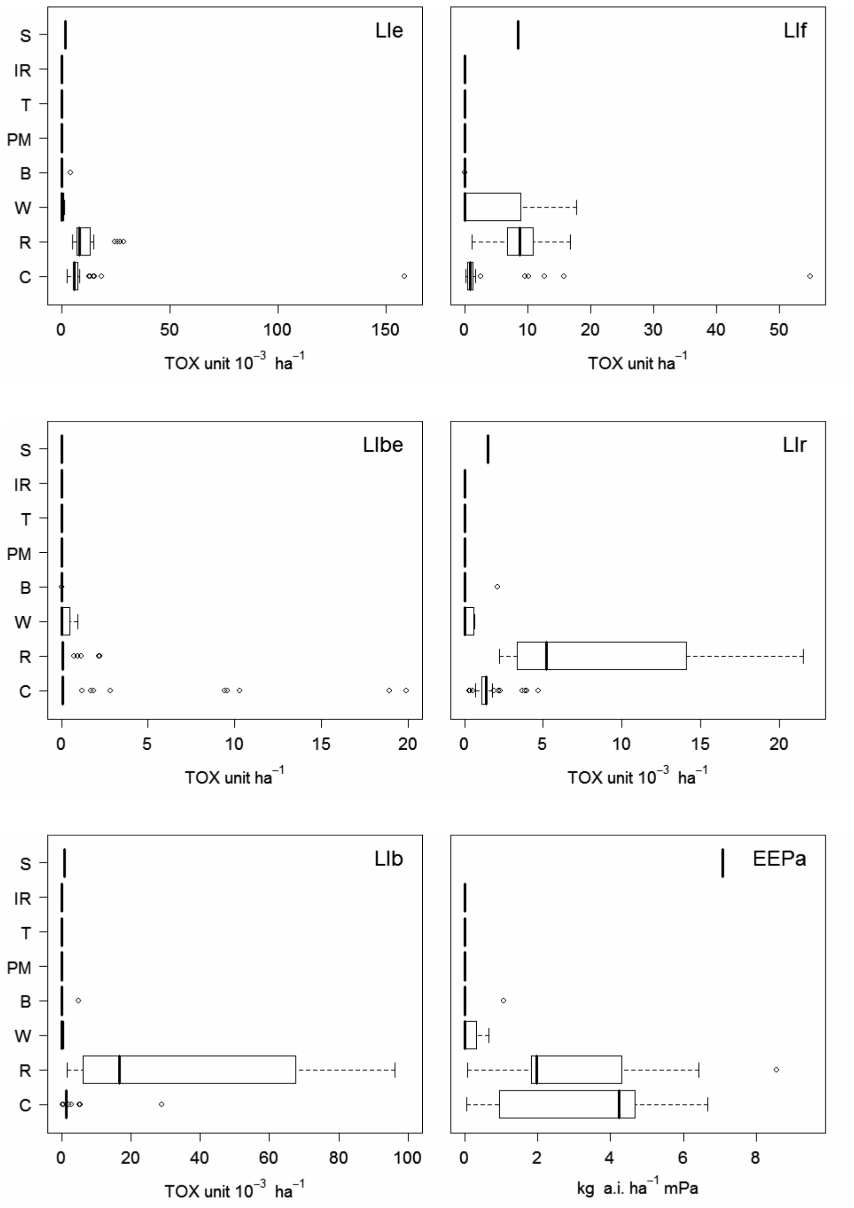
APPENDIXES

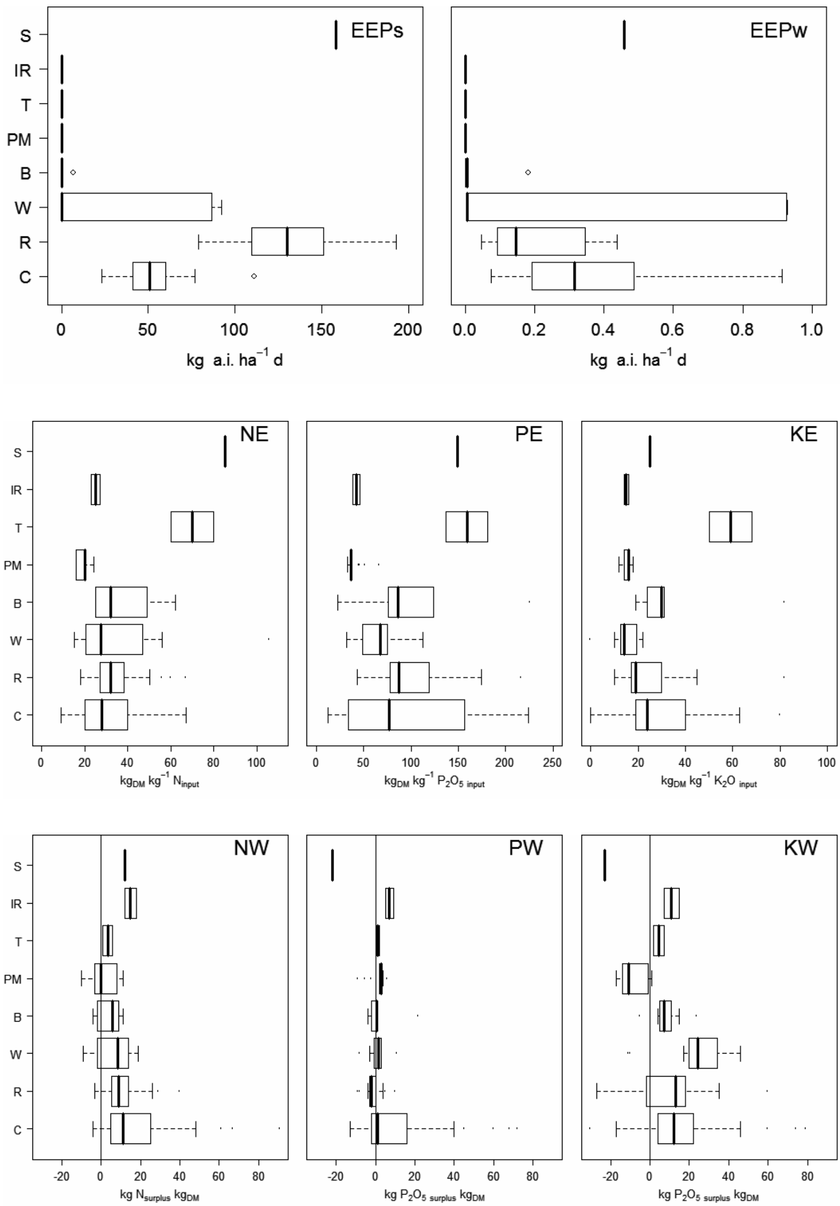
Appendix A.

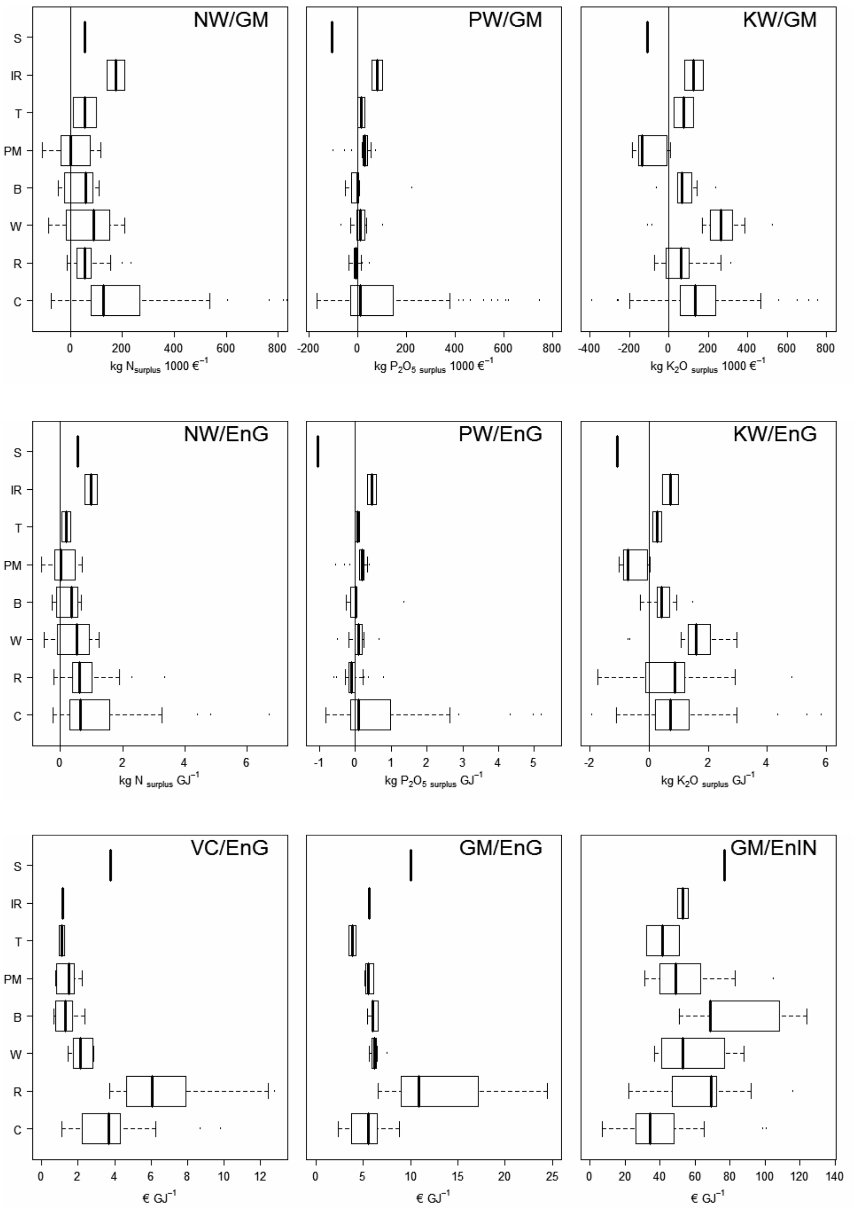
Distribution of indicators calculated at crop level




EnIN
EnOUT
EnG
EnE
LIa
LIc
S
IR
T
PM
B
W
R
C
GJ ha^{-1}
TOX unit ha^{-1}

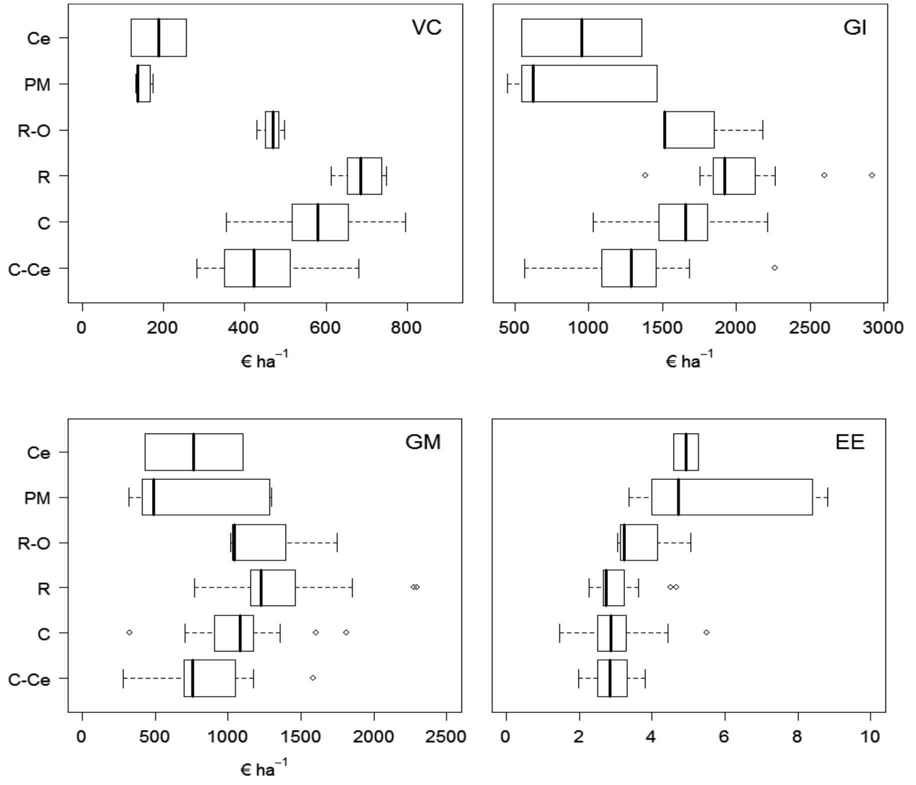

LIe
LIf
LIbe
LIr
LIb
EEPa
S
IR
T
PM
B
W
R
C
TOX unit 10^{-3} ha^{-1}
TOX unit ha^{-1}
TOX unit ha^{-1}
TOX unit 10^{-3} ha^{-1}
TOX unit 10^{-3} ha^{-1}
kg a.i. ha^{-1} mPa

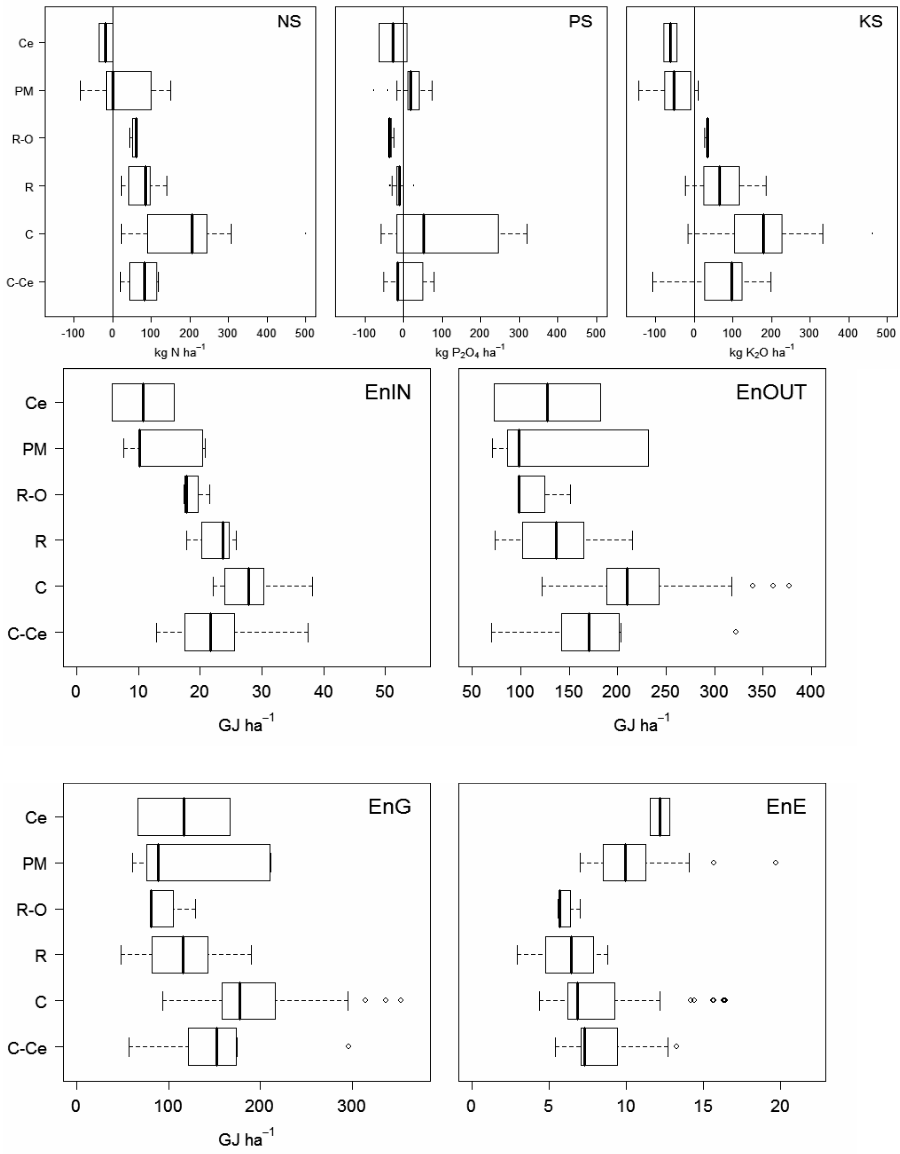

EEPs
EEPw
S
IR
T
PM
B
W
R
C
kg a.i. ha^{-1} d
NE
PE
KE
kg$_{DM}$ kg^{-1} N$_{input}$
kg$_{DM}$ kg^{-1} P$_2$O$_5$ $_{input}$
kg$_{DM}$ kg^{-1} K$_2$O $_{input}$
NW
PW
KW
kg N$_{surplus}$ kg$_{DM}$
kg P$_2$O$_5$ $_{surplus}$ kg$_{DM}$

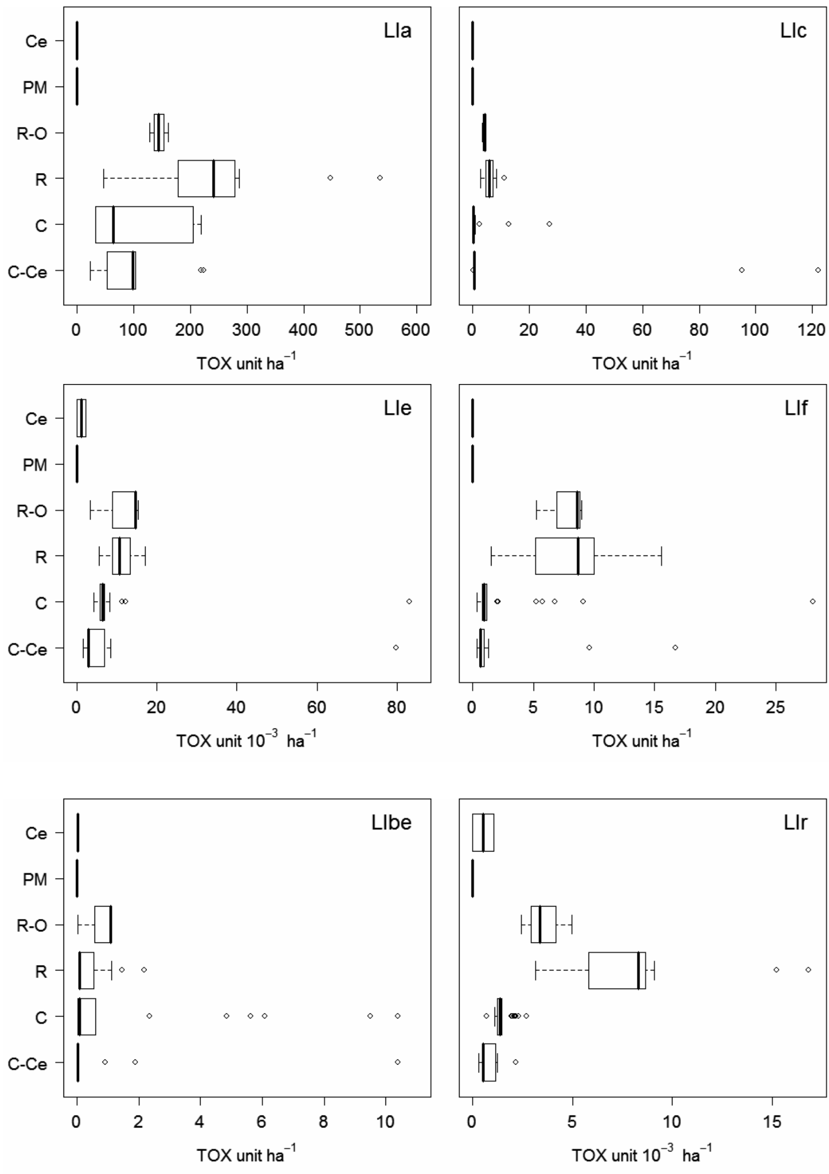

NW/GM
PW/GM
KW/GM
kg N surplus 1000 €⁻¹
kg P2O5 surplus 1000 €⁻¹
kg K2O surplus 1000 €⁻¹
NW/EnG
PW/EnG
KW/EnG
kg N surplus GJ⁻¹
kg P2O5 surplus GJ⁻¹
kg K2O surplus GJ⁻¹
VC/EnG
GM/EnG
GM/EnIN
€ GJ⁻¹
S
IR
T
PM
B
W
R
C

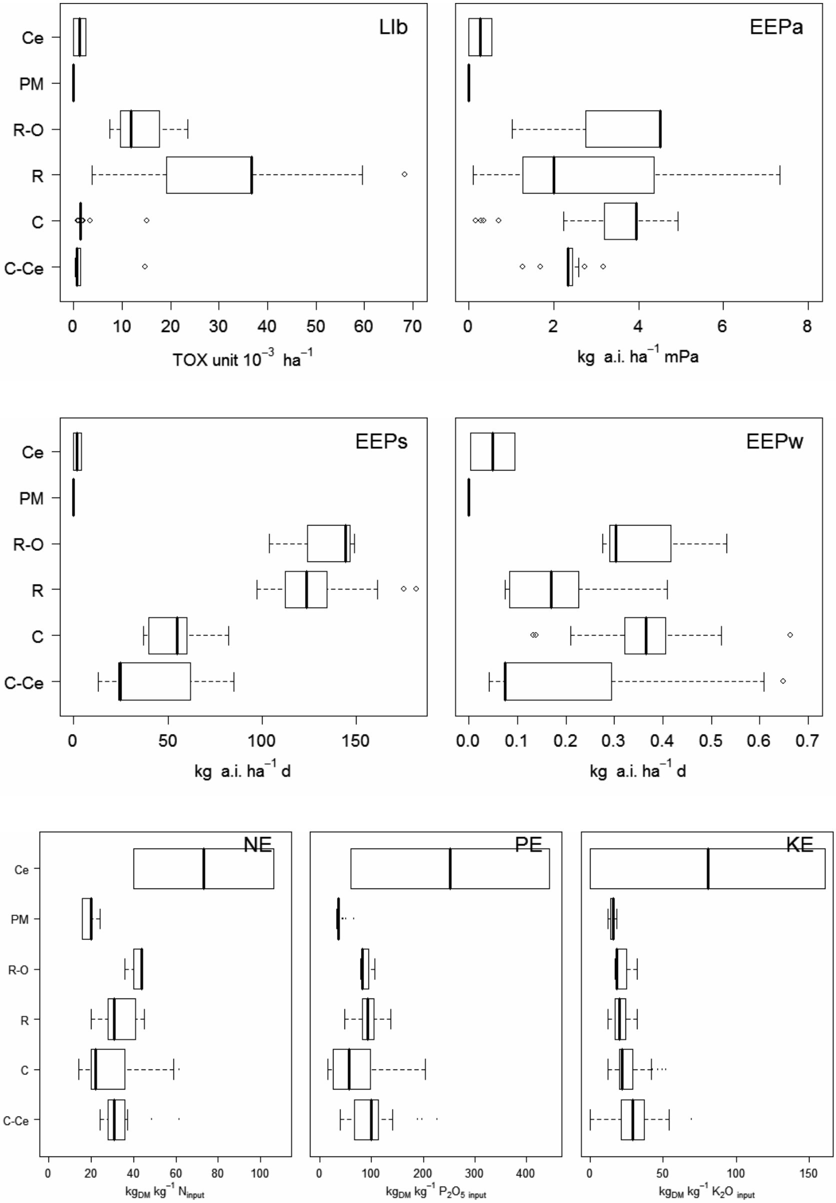
Appendix B.

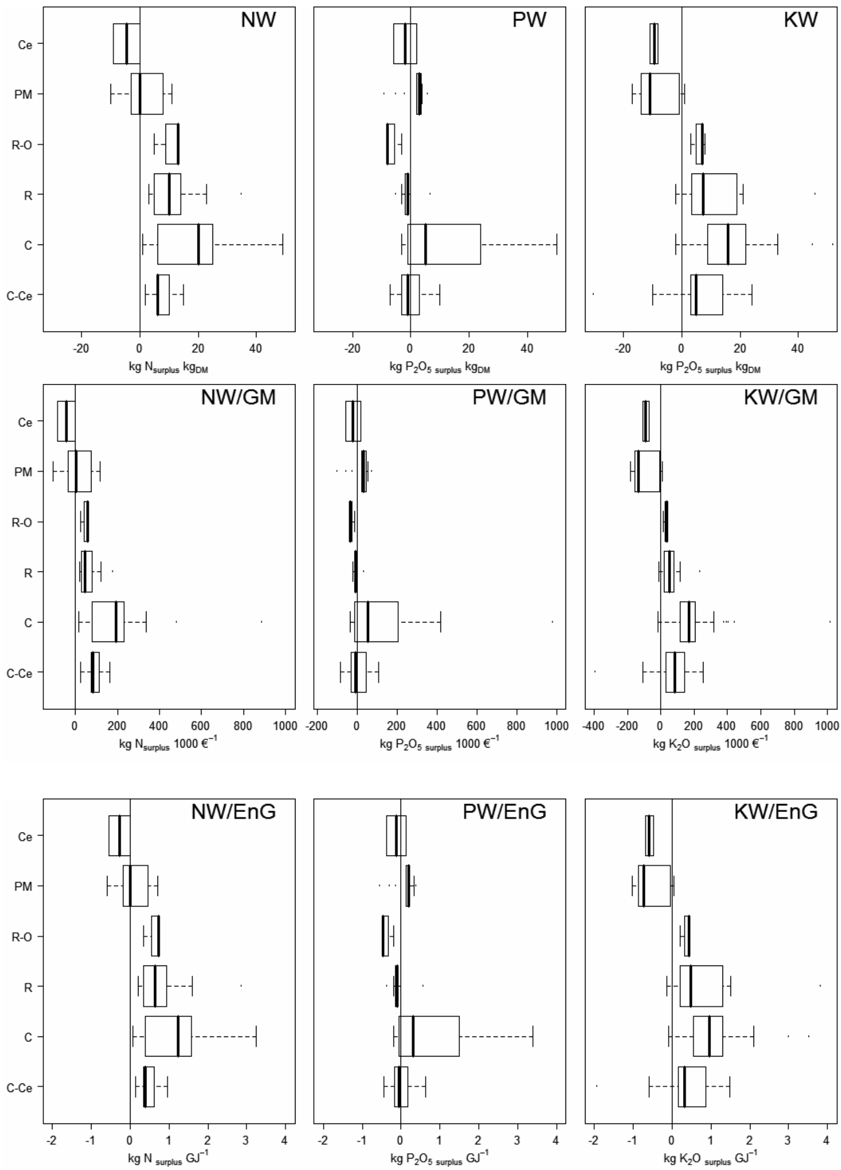
Distribution of indicators calculated at field level

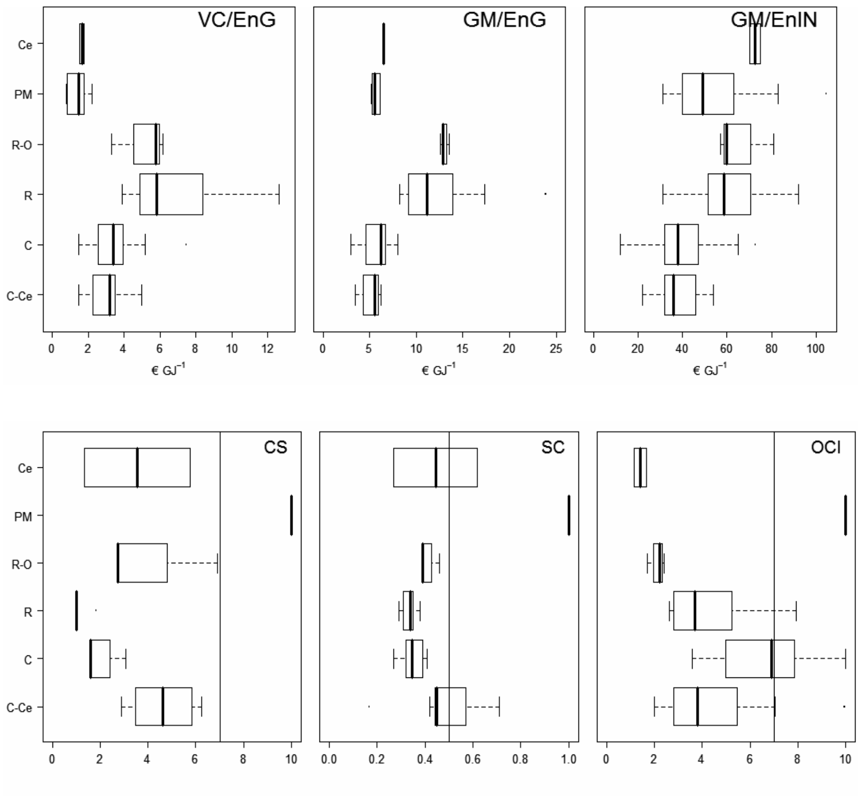



NS
PS
KS
Ce
PM
R-O
R
C
C-Ce
kg N ha^{-1}
kg P_2O_4 ha^{-1}
kg K_2O ha^{-1}
EnIN
EnOUT
GJ ha^{-1}
GJ ha^{-1}
EnG
EnE
GJ ha^{-1}


LIa
Ce
PM
R-O
R
C
C-Ce
0 100 200 300 400 500 600
TOX unit ha^{-1}
LIc
0 20 40 60 80 100 120
TOX unit ha^{-1}
LIe
0 20 40 60 80
TOX unit 10^{-3} ha^{-1}
LIf
0 5 10 15 20 25
TOX unit ha^{-1}
LIbe
0 2 4 6 8 10
TOX unit ha^{-1}
LIr
0 5 10 15
TOX unit 10^{-3} ha^{-1}

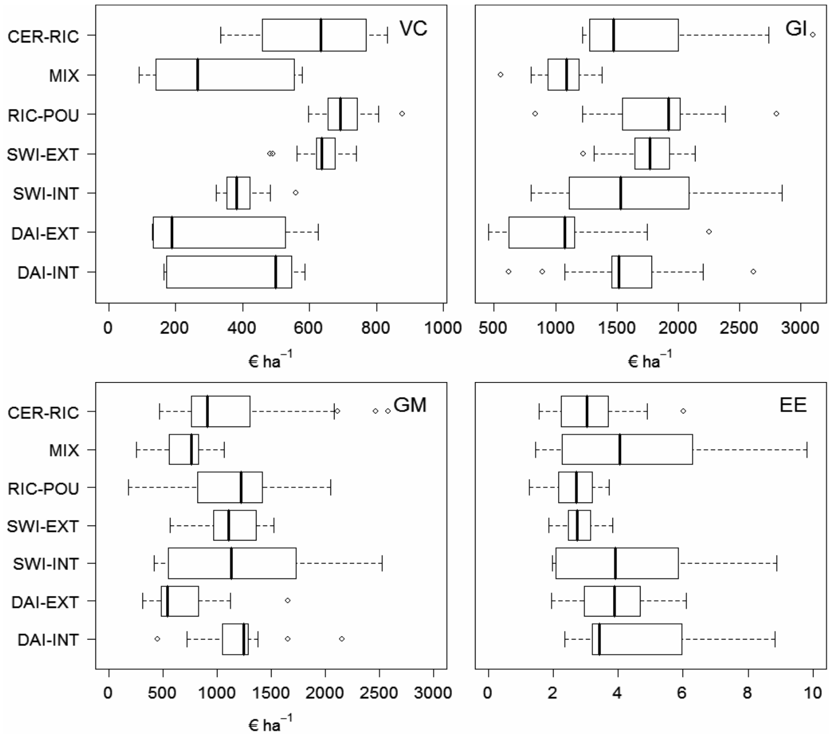

LIb
EEPa
EEPs
EEPw
NE
PE
KE
Ce
PM
R-O
R
C
C-Ce
0 10 20 30 40 50 60 70
TOX unit 10^{-3} ha^{-1}
0 2 4 6 8
kg a.i. ha^{-1} mPa
0 50 100 150
kg a.i. ha^{-1} d
0.0 0.1 0.2 0.3 0.4 0.5 0.6 0.7
kg a.i. ha^{-1} d
0 20 40 60 80 100
kg_{DM} kg^{-1} N_{input}
0 100 200 300 400
kg_{DM} kg^{-1} $P_2O_{5\ input}$
0 50 100 150
kg_{DM} kg^{-1} $K_2O_{\ input}$

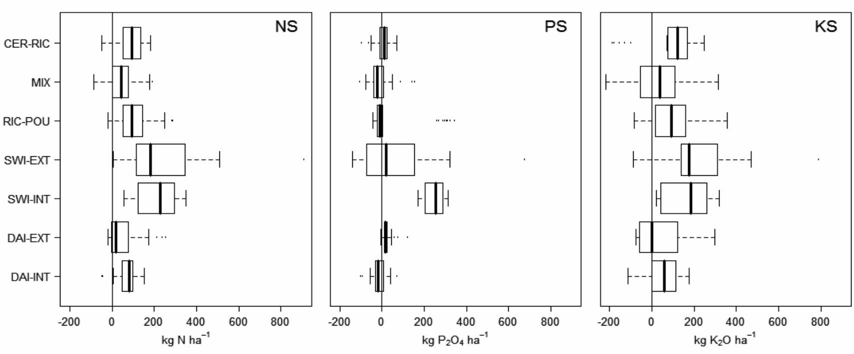

NW
PW
KW
NW/GM
PW/GM
KW/GM
NW/EnG
PW/EnG
KW/EnG
Ce
PM
R-O
R
C
C-Ce
kg N surplus kgDM
kg P2O5 surplus kgDM
kg P2O5 surplus kgDM
kg N surplus 1000 €-1
kg P2O5 surplus 1000 €-1
kg K2O surplus 1000 €-1
kg N surplus GJ-1
kg P2O5 surplus GJ-1
kg K2O surplus GJ-1

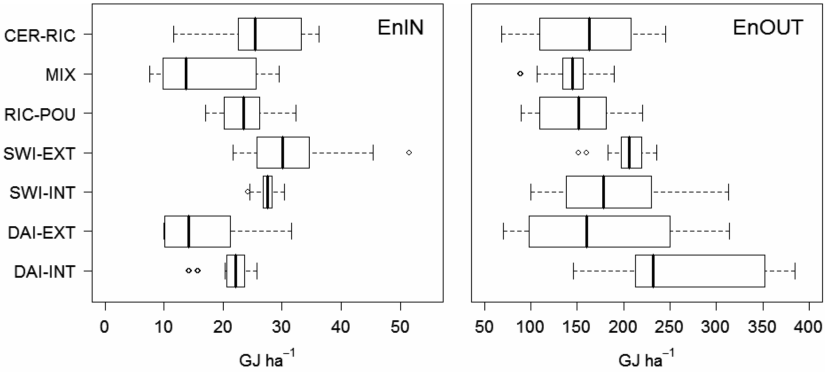

VC/EnG
GM/EnG
GM/EnIN
Ce
PM
R-O
R
C
C-Ce
€ GJ⁻¹
CS
SC
OCI

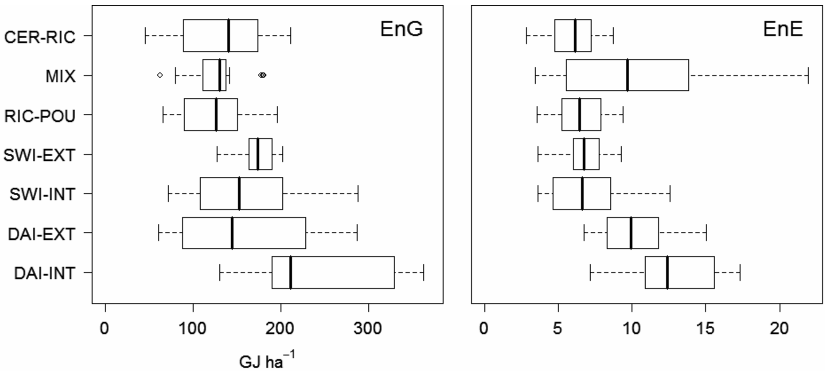
Appendix C.

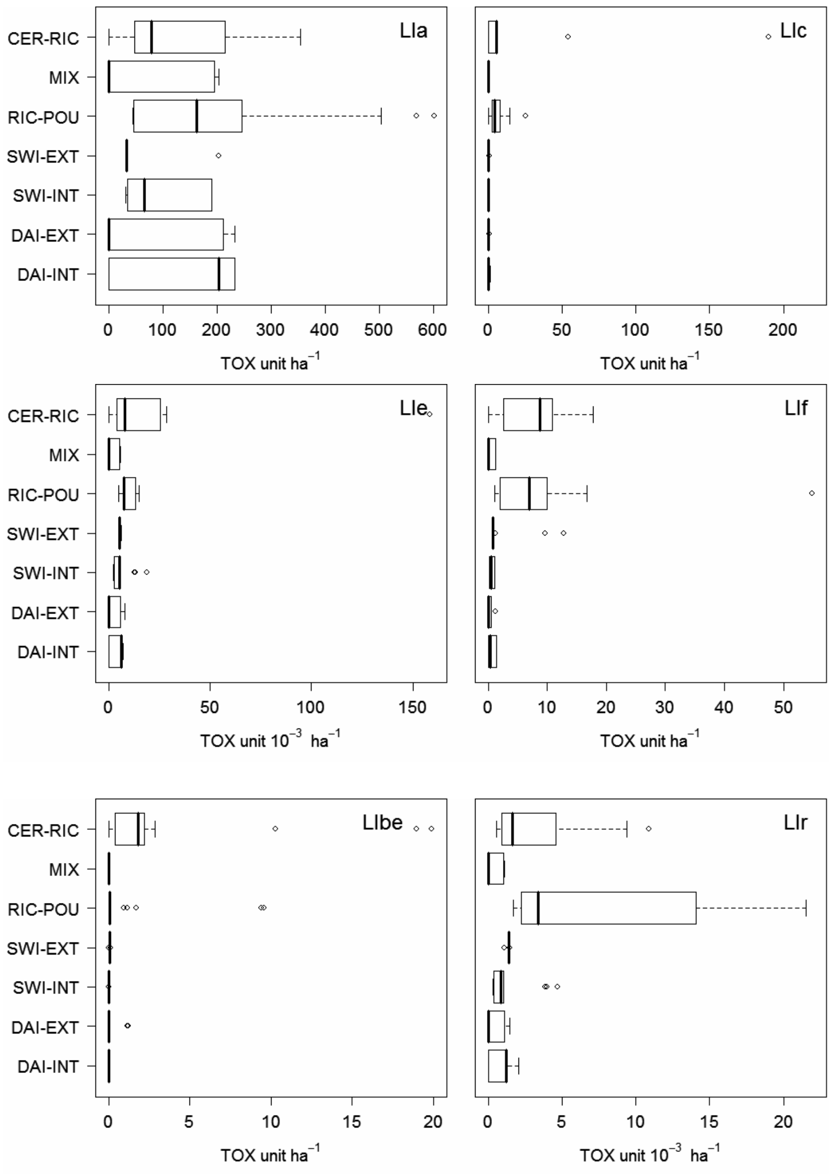
Distribution of indicators calculated at crop level and grouped by farm.

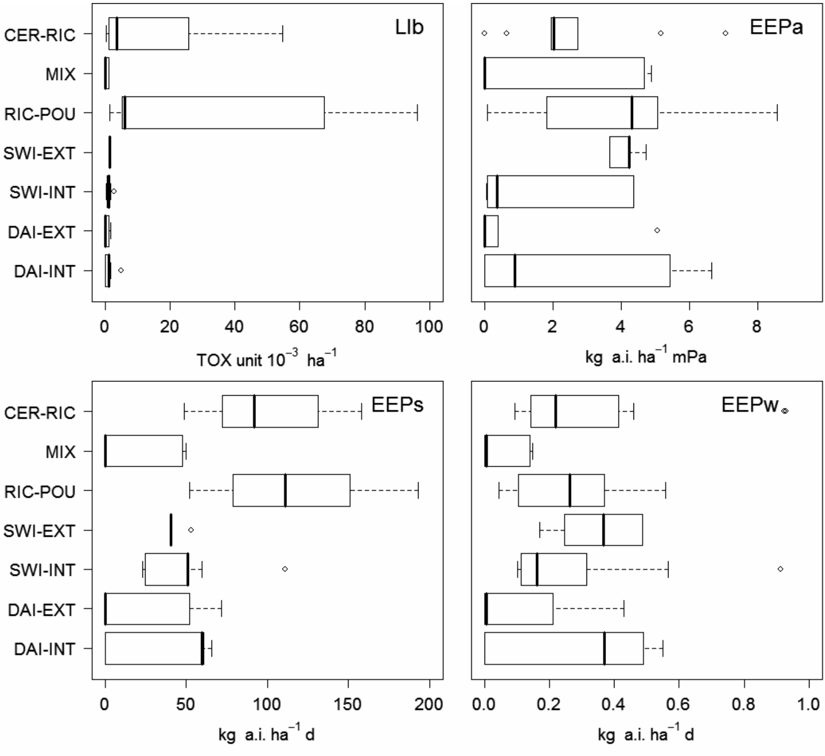


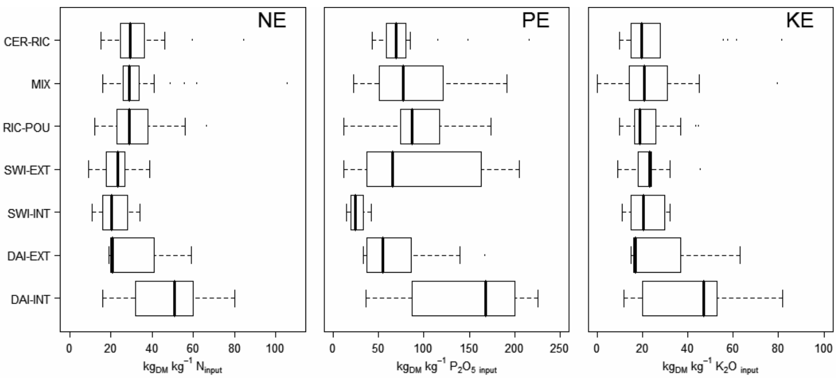

NS
PS
KS
CER-RIC
MIX
RIC-POU
SWI-EXT
SWI-INT
DAI-EXT
DAI-INT
kg N ha^{-1}
kg P_2O_4 ha^{-1}
kg K_2O ha^{-1}

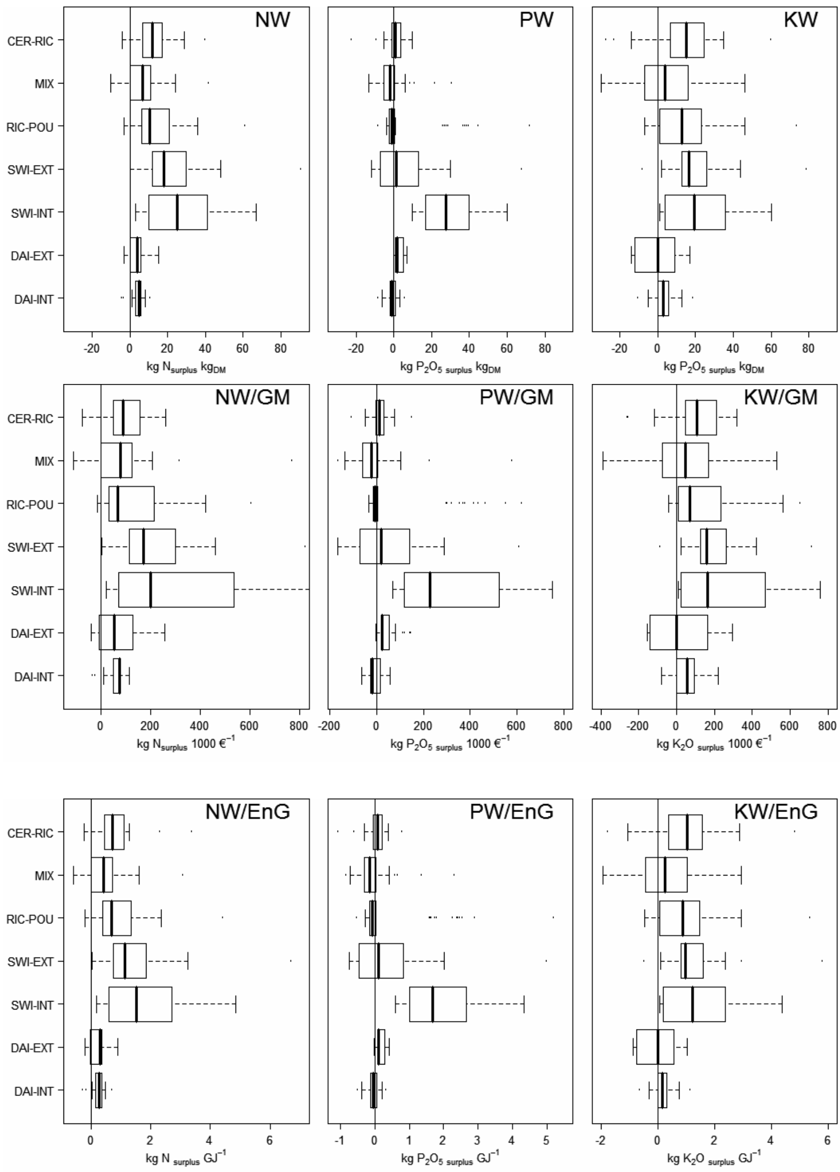

EnIN
EnOUT
CER-RIC
MIX
RIC-POU
SWI-EXT
SWI-INT
DAI-EXT
DAI-INT
GJ ha^{-1}
GJ ha^{-1}

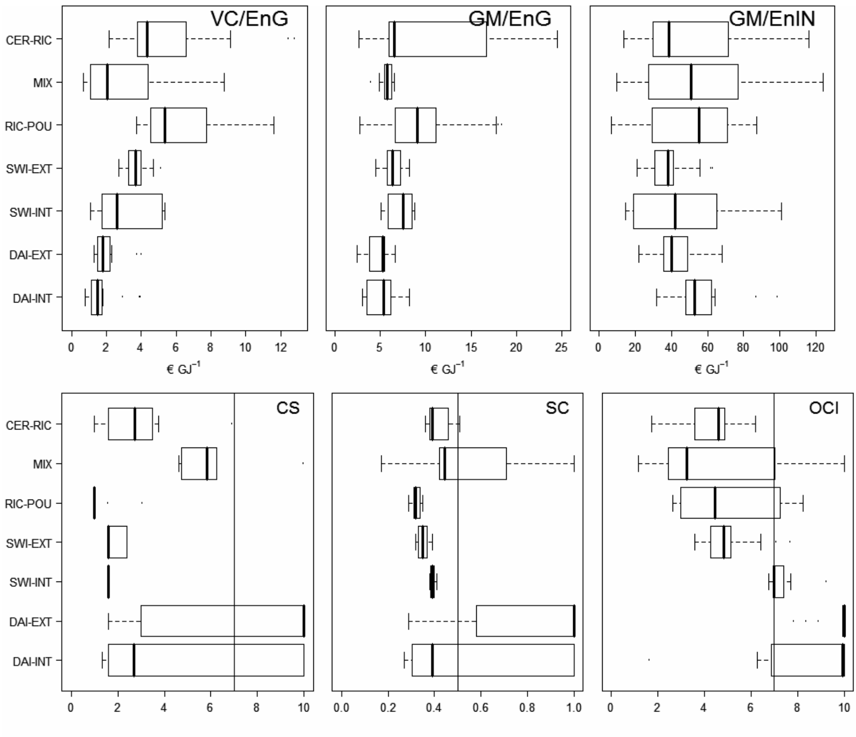

EnG
EnE
CER-RIC
MIX
RIC-POU
SWI-EXT
SWI-INT
DAI-EXT
DAI-INT
GJ ha^{-1}


LIa
LIc
LIe
LIf
LIbe
LIr
CER-RIC
MIX
RIC-POU
SWI-EXT
SWI-INT
DAI-EXT
DAI-INT
TOX unit ha^{-1}
TOX unit 10^{-3} ha^{-1}


LIb
EEPa
EEPs
EEPw
CER-RIC
MIX
RIC-POU
SWI-EXT
SWI-INT
DAI-EXT
DAI-INT
0 20 40 60 80 100
TOX unit 10^{-3} ha^{-1}
0 2 4 6 8
kg a.i. ha^{-1} mPa
0 50 100 150 200
kg a.i. ha^{-1} d
0.0 0.2 0.4 0.6 0.8 1.0
kg a.i. ha^{-1} d


NE
PE
KE
CER-RIC
MIX
RIC-POU
SWI-EXT
SWI-INT
DAI-EXT
DAI-INT
0 20 40 60 80 100
kg_{DM} kg^{-1} N_{input}
0 50 100 150 200 250
kg_{DM} kg^{-1} $P_2O_{5\ input}$
0 20 40 60 80 100
kg_{DM} kg^{-1} $K_2O_{\ input}$


NW
PW
KW
NW/GM
PW/GM
KW/GM
NW/EnG
PW/EnG
KW/EnG
CER-RIC
MIX
RIC-POU
SWI-EXT
SWI-INT
DAI-EXT
DAI-INT
kg N surplus kgDM
kg P2O5 surplus kgDM
kg P2O5 surplus kgDM
kg N surplus 1000 €-1
kg P2O5 surplus 1000 €-1
kg K2O surplus 1000 €-1
kg N surplus GJ-1
kg P2O5 surplus GJ-1
kg K2O surplus GJ-1



VC/EnG
GM/EnG
GM/EnIN
CER-RIC
MIX
RIC-POU
SWI-EXT
SWI-INT
DAI-EXT
DAI-INT
€ GJ^{-1}
€ GJ^{-1}
€ GJ^{-1}
CS
SC
OCI

Appendix D.

VALUTAZIONE DI ALTERNATIVE GESTIONALI DELLA CONCIMAZIONE AZOTATA DEL MAIS ATTRAVERSO MODELLISTICA DI SIMULAZIONE

Evaluation of alternative nitrogen management in corn using simulation model

Luca Bechini, and Nicola Castoldi;
Riv. Ital. Agrometeorol. /Ital. J. Agrometeorol. 1:71–72.

Abstract
Allo scopo di migliorare la comprensione della dinamica dell'azoto nel sistema suolo-coltura di un'azienda zootecnica suinicola, abbiamo eseguito simulazioni di una monosuccessione di mais da granella, coltivato secondo la gestione praticata dall'agricoltore o secondo itinerari tecnici alternativi. Questi prevedevano la riduzione della concimazione azotata, in modo fisso tutti gli anni, o in modo variabile a seconda del contenuto di nitrati nel terreno allo stadio di sesta foglia (V6). I risultati mostrano che, rispetto alla gestione praticata attualmente dall'agricoltore, è possibile ridurre la concimazione azotata e la lisciviazione di nitrati; tale riduzione può essere modulata a seconda dell'andamento stagionale (ma sopportando il costo aggiuntivo di campionamento e analisi), oppure può essere fissata a circa 75 kg N ha^{-1}. L'utilizzo di colture di copertura non riduce ulteriormente le perdite di nitrati per lisciviazione.

Introduzione
Nell'ambito di un progetto di ricerca sull'agricoltura nelle aree protette, stiamo mettendo a punto un metodo di valutazione della sostenibilità ambientale ed economica di diversi sistemi agricoli, e lo stiamo applicando a otto aziende del Parco Agricolo Sud Milano. Le aziende vengono da noi monitorate tramite visite periodiche, nel corso delle quali registriamo tutti gli interventi eseguiti in ciascun appezzamento

(lavorazioni, fertilizzazioni, trattamenti fitosanitari, irrigazioni e raccolte). La peculiarità del nostro approccio risiede nel fatto che ci basiamo quanto più possibile su dati esistenti (banche dati della pubblica amministrazione, letteratura scientifica) o su dati facilmente ottenibili tramite colloqui tecnici con gli agricoltori, cercando di escludere il ricorso a misure dirette. Di conseguenza privilegiamo gli indicatori agro-ecologici come strumento di sintesi. Nel corso dell'indagine abbiamo riscontrato elevati surplus per i bilanci azotati calcolati a scala di singolo appezzamento per la coltura del mais (*Zea mays* L.), soprattutto nelle aziende zootecniche. Allo scopo di valutare le strategie per ottimizzare la gestione dell'azoto (N) in queste aziende, stiamo quindi conducendo un'analisi modellistica della dinamica dell'azoto nel sistema suolo-coltura; coerentemente con lo spirito del progetto, stiamo valutando l'applicabilità dei modelli senza basarci su misure apposite, conducendo la parametrizzazione con dati già disponibili. In questa comunicazione riferiamo i risultati preliminari relativi ad un'azienda zootecnica.

Materiali e metodi

L'azienda ha una superficie di 81,5 ha, un carico zootecnico (allevamento di suini pesanti) di 1,21 t peso vivo ha^{-1}, ed è interamente coltivata a mais da granella; la distribuzione dei reflui avviene nel periodo autunno-invernale, l'aratura e la preparazione del letto di semina nel mese di marzo, e la semina del mais tra l'ultima decade di marzo e la prima di aprile. La concimazione minerale viene effettuata con urea in presemina ed in copertura (70 e 140 kg N ha^{-1}). I suoli (Unità Cartografica 18: ERSAL, 2000) sono Aquultic Haplustalfs coarse loamy, mixed, mesic, con profondità della falda in estate di 1,6 m. Il modello CropSyst (Stöckle et al., 2003) è stato utilizzato per simulare una monosuccessione di mais da granella, utilizzando dati meteorologici (Sant'Angelo Lodigiano, 45°14'N 9°24'E) generati con ClimGen (Stöckle et al., 2003) per 50 anni, escludendo i risultati dei primi 10 e fornendo media e deviazione standard dei rimanenti 40. Il suolo è stato parametrizzato utilizzando le informazioni della carta pedologica: profondità della falda, e, per ogni strato del profilo, spessore e dati di tessitura e carbonio organico. Le proprietà idrologiche sono state stimate con le equazioni proposte da Saxton et al. (1986). L'infiltrazione e la redistribuzione dell'acqua nel profilo sono state simulate con l'equazione di Richards. Il mais è stato seminato automaticamente dal modello a partire dal 15 marzo di ogni anno quando la temperatura dell'aria era di almeno 9°C per 5 giorni consecutivi. A causa della limitata disponibilità idrica nella zona, l'irrigazione consiste in due soli interventi per stagione, uno in prossimità della fioritura, l'altro circa 20 giorni dopo. Sono stati confrontati questi scenari: ORD: gestione agronomica dichiarata dall'agricoltore; LPR: come ORD, ma con liquami distribuiti in primavera una settimana prima della semina; PSNT190, PSNT160, PSNT130, PSNT100: questi scenari simulano l'esecuzione del Pre Sidedress Nitrate Test (PSNT), che consiste nella misura del contenuto dei nitrati nei primi 30 cm di terreno allo stadio di sesta foglia (SMNV6); il modello distribuisce automaticamente, con concime di sintesi, la quantità di azoto (NAP) necessaria per raggiungere un'asportazione totale definita dall'utente (NASPT); NAP = (NASPT – SMNV6 – NASPV6) / 0,7, dove NASPV6 è l'azoto già asportato dalla coltura al V6. Tenendo conto della mineralizzazione che avviene tra il momento in cui si esegue il test e la maturazione fisiologica (ca. 60 kg N ha^{-1}), abbiamo fissato NASPT a quattro diversi livelli (190, 160, 130 e 100 kg N ha^{-1}), per valutare la possibilità di ridurre gli apporti di azoto di sintesi. Gli scenari PSNT, al contrario degli altri, risultano ogni anno in una diversa dose di azoto distribuita, in base alle dinamiche del terreno; negli scenari PSNT i liquami vengono distribuiti in primavera. Altri scenari studiati sono stati: FERT100 e FERT75: come LPR, ma senza concimazione in presemina, e riducendo la concimazione in copertura a 100 e 75 kg N ha^{-1}; CC-PSNT130 e CC-PSNT80, che prevedono la coltivazione di una coltura di copertura (loglio italico, *Lolium multiflorum* Lam.) nel periodo autunno-vernino, interrata in marzo prima della semina del mais (anche in questo caso la fertilizzazione del mais viene gestita con il PSNT); CC-F75, che prevede la coltura di copertura e la concimazione del mais con una dose fissa di azoto di 75 kg N ha^{-1}.

Risultati

Lo scenario ORD (Tab. 1) è caratterizzato da elevate perdite di nitrati, che avvengono principalmente nel periodo agosto - marzo (83% del

Tab. 1 - Media e deviazione standard (ds) delle principali variabili simulate per i diversi scenari posti a confronto.

Scenario	**Rese (granella mais)**		**N di sintesi distribuito**		**N lisciviato**		**Variazioni di ricavo**	**Variaz. C org. suolo (30 cm)**
	media	*ds*	*media*	*ds*	*media*	*ds*	*media*	*media*
	t ss ha^{-1}		kg N ha^{-1}				€ ha^{-1}	kg C ha^{-1} a^{-1}
ORD	**10.9**	**1.1**	**210**	**0**	**120**	**58**	**0**	**9**
LPR	10.9	1.1	210	0	120	59	-2	26
PSNT190	10.9	1.1	126	15	37	19	55	26
PSNT160	10.9	1.1	86	14	3	5	80	26
PSNT130	**10.8**	**1.0**	**52**	**14**	**1**	**1**	**96**	**18**
PSNT100	10.0	0.8	17	12	0	0	-10	-17
FERT100	10.9	1.1	100	0	9	9	71	26
FERT75	**10.9**	**1.1**	**75**	**0**	**1**	**1**	**88**	**24**
CCI-P130	10.7	1.0	67	13	0	0	-37	105
CCI-P80	8.5	0.6	9	8	0	0	-341	0
CCI-F75	10.8	1.0	75	0	0	0	-30	111

totale); il mese in cui le perdite sono maggiori è agosto, a causa degli alti residui di azoto minerale presenti nel profilo alla maturazione della coltura. La distribuzione primaverile dei reflui (LPR) non consente di conseguire vantaggi produttivi o ambientali. Gli scenari PSNT

dimostrano che è possibile conseguire rese simili a quelle ORD diminuendo la concimazione fino a mediamente 52 kg N ha^{-1} (PSNT130); questo corrisponde a un risparmio di circa 96 € ha^{-1}. Il costo economico da sostenere per l'applicazione del PSNT (campionamento + analisi + spedizione del campione, stimabili in circa 20 € ha^{-1} per un appezzamento di 5 ha) e l'onere organizzativo, tuttavia, ci hanno spinto a valutare anche l'efficacia di una concimazione uguale tutti gli anni, applicabile senza esecuzione del PSNT (scenari FERT100 e FERT75): in tali casi l'azoto mediamente applicato e le perdite sono lievemente maggiori, ma gli scenari sono economicamente remunerativi (risparmio di 71 e 88 € ha^{-1}). Gli scenari con le colture di copertura consentono di ridurre la lisciviazione rispetto a ORD ma non rispetto a PSNT e quindi non appaiono interessanti per i maggiori costi (semente e gasolio), anche se consentono un incremento della dotazione di sostanza organica del suolo.

Conclusioni

La riduzione delle concimazioni minerali azotate, indagata attraverso le simulazioni effettuate con un modello dinamico dei sistemi colturali, è possibile nell'azienda studiata; il risparmio di concime conseguibile non è costante ma varia a seconda dell'annata e andrebbe quantificato attraverso un apposito test da effettuarsi sul terreno immediatamente prima dell'esecuzione della concimazione di copertura. Diversi fattori influiscono sulla bontà delle previsioni presentate in questo lavoro: i) l'incertezza nella simulazionedella mineralizzazione dei reflui zootecnici: l'immobilizzazione di N, frequentemente documentata in letteratura nei casi di utilizzo di reflui zootecnici, non è simulabile con la versione di CropSyst qui utilizzata; sarebbe per questo necessario l'utilizzo di un modello a più pool (es. Probert et al., 2005); ii) l'incertezza nella simulazione della mineralizzazione della sostanza organica stabile nel lungo periodo; iii) l'importanza delle dinamiche di falda locali, difficili da conoscere nel contesto del nostro progetto, ma di fatto evidenti dall'osservazione dei profili pedologici in molte aree del Parco. Allo scopo di migliorare la conoscenza delle dinamiche del sistema stiamo eseguendo misure dirette di alcuni indicatori utili per una gestione operativa: contenuto in azoto minerale del profilo all'emergenza e alla raccolta; Pre Sidedress Nitrate Test; late-stalk nitrate test.

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Appendix E.

CROPPING SYSTEMS SUSTAINABILITY EVALUATION WITH AGRO-ECOLOGICAL AND ECONOMIC INDICATORS IN NORTHERN ITALY

Nicola Castoldi, and Luca Bechini
Farming Systems Design 2007, Int. Symposium on Methodologies on Integrated Analysis on Farm Production Systems, M. Donatelli, J. Hatfield, A. Rizzoli Eds., Catania (Italy), 10–12 Sep. 2007, book 2 - Field-farm scale design and improvement, pp. 147–148.

Introduction

The objective of this work is to evaluate the environmental and economic sustainability of crop management in seven farms of the Sud Milano Agricultural Park (Italy; 45°N, 9°E), one of the most intensive and lucrative agricultural areas in Italy, with environmental concerns derived by the intensive use of resources (nutrients, energy, and pesticides). One of the most suitable tools which can be applied in this context are agro-ecological indicators based on farmers' interviews (Castoldi and Bechini, 2006; Castoldi et al., 2007). We present the preliminary results of two years (2005-2006).

Methodology

We used information derived by interviewing the managers of seven farms of different types (DAI-INT = dairy intensive; DAI-EXT = dairy extensive; SWI-INT = swine intensive; SWI-EXT = swine extensive; RIC-POU = rice and poultry; MIX = mixed; CER-RIC = cereals with rice; Table 1). A set of indicators was been selected and calculated at field level (for a total of 131 fields), by aggregating the observations of a 2-year period. The indicators are divided in five classes: i) economic indicators: gross income (GI), variable costs (VC: sum of the costs for gasoline, lubricants, pesticides, fertilizers, and seeds), economic balance (GI-VC) and economic efficiency (GI/VC); ii) nutrient indicators: NPK soil surface balances; iii) energy indicators: energy input (EI: sum of energy in the gasoline, lubricants, pesticides, fertilizers, seeds, and

Table 1 – Characteristics of the seven farms monitored

Farm	DAI-INT	DAI-EXT	SWI-INT	SWI-EXT	RIC-POU	MIX	CER-RIC
Total area (ha)	58	134	35	81	115	48	55
Crop type (%)							
Corn	58	19	75	90	13	41	31
Rice					85		42
Wheat	4					18	17
Barley	4	3				23	
Meadows	29	79				9	
Others crops	9					9	4
Trees			25				
Set-Aside (%)	5	2		10	3	9	5
Livestock (Mg l.w. ha^{-1})	1.88	0.86	5.69	1.22	0.16	0.31	0.00

machinery), energy output (EO: energy content of yield), energy balance (EO-EI), and energy efficiency (EO/EI); iv) soil indicators: crop sequence indicator (it evaluates the average goodness of each previous-successive crop combination), soil cover index (the percentage of soil cover by crops or residues in one year), and soil organic matter indicator (it evaluates if the management on a specific soil tends to accumulate or deplete soil organic matter); v) pesticide indicators: load index (the ratio between the application rate and the toxicity of active ingredient, a.i.), calculated separately for rats, birds, earthworms, bees, fishes, crustaceans, and algae, environmental exposure–based pesticide

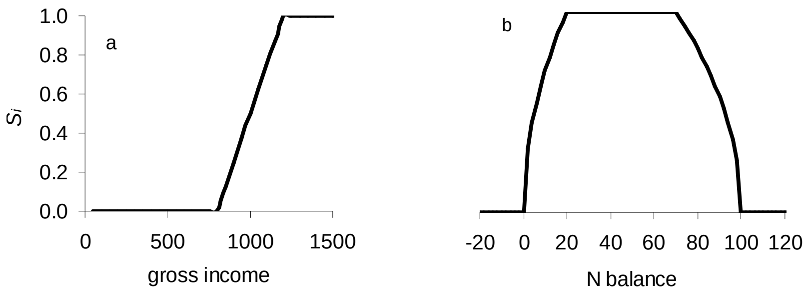

Figure 1 – Sustainability function for the gross income (€ ha^{-1}) and the soil surface N balance (kg N ha^{-1}).

indicators (calculated using physical–chemical properties of each a.i. characterizing its fate in air, soil, and groundwater).

Optimum and unsustainable ranges for each indicator have been taken from literature, expert knowledge, or from simulation models (meta-models); a sustainability function provides a sustainability index (Si), which equals 1 if the indicator value is in an optimum range and 0 if it is in an unsustainable range. In order to avoid a sharp boundary, values between 0 and 1 are assumed in between these ranges, with a user-defined linear (Fig. 1a), or non-linear (Fig. 1b) function. The Sis were averaged by indicator class (Sc), field (Sf) and farm.

Results

Table 2 – Farm-level sustainability indexes: average values (S_c) for each indicator class, and statistics of field-level indexes (S_f)

		DAI-INT	DAI-EXT	SWI-INT	SWI-EXT	RIC-POU	MIX	CER-RIC
Farm-average S_c	Economic	1.00	0.58	1.00	0.97	0.90	0.81	0.90
	Nutrients	0.51	0.77	0.33	0.36	0.51	0.51	0.70
	Energy	0.98	0.73	1.00	0.88	0.84	0.95	0.64
	Soil	0.62	0.89	0.52	0.56	0.32	0.69	0.51
	Pesticides	0.94	0.97	0.98	0.84	0.38	0.92	0.48
S_f	Farm-average	0.81	0.79	0.77	0.72	0.59	0.78	0.65
	Minimum	0.66	0.61	0.75	0.62	0.47	0.44	0.34
	Maximum	0.95	0.83	0.80	0.77	0.69	0.92	0.78
	St. dev.	0.10	0.08	0.02	0.03	0.07	0.12	0.11

The aggregated indexes at farm level are shown in Table 2. The S_c for the economic indicators shows a complete sustainability (1.00) for the intensive livestock farms, and is lower (0.58) for the DAI-EXT. One reason is that, by partially accounting the nutrient content of manure, intensive farms save mineral fertilizers and therefore reduce VC; furthermore their yields are usually high, increasing the GI. On the other hand the hay has a low price, penalizing particularly the DAI-EXT farm. Sustainability of nutrient management is generally poor, especially for the swine farms. The S_c for energy is normally high, due to the good energy performance of corn, present in all farms. The lowest value

(0.64) is in the CER-RIC, due to the relatively low yields of rice and wheat. Soil management is correct (0.89) in DAI-EXT (use of animal manure and crop residues to maintain soil organic matter; continuous soil cover) and unsatisfactory in the other farms that do not have enough manure for all fields and where in some cases the straw is harvested. The intensive use of pesticides in rice cropping induces a very low S_c for the rice farms (0.38 and 0.48); the opposite situation occurs in the meadows, where no pesticides are used and in the barley and wheat, where herbicides are occasionally used. Overall, farm-average S_f is satisfactory for the dairy farms (0.81-0.79). Moderate average S_fs are obtained by swine (0.77-0.72) and MIX (0.78) farms. The minimum S_f is not very low in the dairy and swine farms (>0.60), while in the others there are fields with low (0.47-0.44) or very low (0.34) S_fs. None of the fields is completely sustainable (S_f=1), and in many cases the maximum S_fs are lower than 0.90. The variability of S_f among farm types is relatively limited, as, due to the large number of indicators used, a bad rate for a given indicator may be compensated by the good performance of another.

Conclusions

According to this framework, dairy farms are among the best farming systems of the area. This is partly due to the role of permanent meadows, which appear to be very sustainable. In fact, meadows have low inputs of nutrients, pesticides and energy, and good soil management, even if they have limited economic value. The mixed and the intensive swine farms are also good. The sustainability of rice cropping systems is much lower, due to intensive use of herbicides and fungicides. Maize cultivation is also critical, due to intensive use of nutrients, particularly in animal farms. A critical step in this approach is the definition of the sustainability functions that provide the S_i index based on the value of the indicators. Another limitation is that we have given the same weight to each indicator class and to each indicator within each class. This choice, apparently simplistic, is partly justified by less restrictive ranges assigned to less important indicators. The calculation can be improved by differentiating the weights assigned to indicators or to classes, basing the choice on different stakeholders' interests. Finally, this approach takes into account only crop cultivation;

further work is needed to evaluate the economic and ecological sustainability of animal production systems. This might change substantially the overall farm sustainability: for example, it is expected that the inclusion of the income due to milk production will improve the economic balance for the dairy farms.

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Appendix F.

SUSTAINABILITY SOLUTION SPACE USING AGRO-ECOLOGICAL INDICATORS AT FIELD LEVEL

Nicola Castoldi[1], Luca Bechini, Julia Steinberger, Claudia Rebecca Binder
[1] Dep. of Crop Science, Univ. of Milano, Italy.
[2] Dep. of Geography, Univ. of Zurich, Zurich, Switzerland.
Farming Systems Design 2007, Int. Symposium on Methodologies on Integrated Analysis on Farm Production Systems, M. Donatelli, J. Hatfield, A. Rizzoli Eds., Catania (Italy), 10–12 Sep. 2007, book 2 - Field-farm scale design and improvement, pp. 99–100.

Introduction

The objective of this work is to evaluate the economic and environmental sustainability of cropping and farming systems in the Sud Milano Agricultural Park (Italy), applying the Sustainability Solution Space for Decision-Making (SSP; Wiek and Binder, 2005). We base our analysis on the results of 2-year interviews carried out in seven farms of different types (Castoldi and Bechini, 2007). With this methodology, we are evaluating which kinds of cropping systems are inside or outside the sustainability space.

Methodology

We have selected a sub-set (14) of the 24 indicators calculated at field level (131 fields analyzed) (Castoldi and Bechini, 2007). They describe: i) economic performance (gross income, variable cost [VC]), economic efficiency [EcE]), ii) soil management (crop sequence indicator [CSI], soil cover index, soil organic matter indicator [SOM]), iii) nutrient management (N and P balances [NB, PB]), iv) energy use (energy input [EI], output and efficiency), and v) pesticide application (load index for crustaceans [LIc], fish [LIf] and rats).

To consider the interactions and trade-offs among these indicators, we applied the SSP method, where a solution space for sustainability is

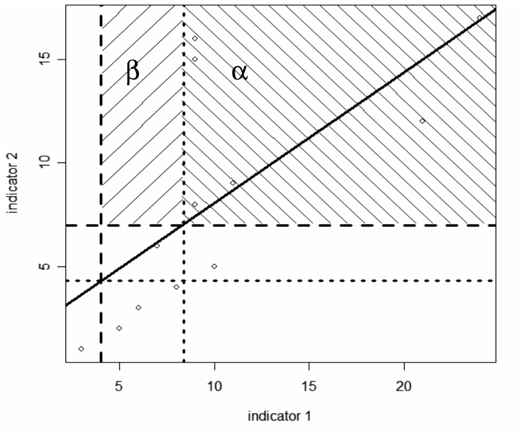

Fig. 1 – Example of SSP method.
Solid line: regression line between indicators; dashed line: initial range (sustainability thresholds); dotted line: intersect range; area $\alpha + \beta$: initial sustainability range; area α: final sustainability range.

calculated based on the sustainability ranges of the selected indicators and on their functional relationships. This solution space guides decision-making by providing a realistic set of possible system states.

After the selection of crucial system indicators, upper and/or lower bounds for sustainability are then defined for each indicator (dashed line, Fig. 1). The initial sustainability space ($\alpha + \beta$, Fig. 1) thus resembles an N-dimensional rectangle. However, the functional relationships between the indicators in fact limit this space. The next step is thus to take into account the functional relationship between the indicators, which could be a linear correlation, as shown in Fig. 2a, or a more complex relationship as in Fig. 2b and 2c.

Based on these relationships (correlations and trade-offs), a sustainability solution space is found (α, Fig. 1). A linear correlation can serve to determine new sustainability upper or lower bounds, as in Fig. 1, whereas a more complex relationship will result in a more complex, non-rectangular space. This space can then be used to find optimum system states, taking into account indicator trade-offs and correlations.

We have developed an optimization software that uses the functional relationships between indicators, providing the SSP space. For each indicator, we defined thresholds identifying complete sustainability and unsustainability values. It is not possible to define an intermediate range with partial sustainability, because the actual version of the software is not able to use this information. To study the functional relationships within the cropping systems, we have calculated the linear regression equation between 91 pairs of indicators. For each pair, if the R^2 was bigger than 0.7, the relation between the two indicators was the regression equation (Fig. 2a). If the R^2 was between 0.5 and 0.7, the relationship was not defined with a line but with the area delimited by the parallels of the regression equation. The distance between the parallels was equal to the maximum difference between the regression function and the confidential interval (p=99.9999%; Fig. 2b). In the cases with R^2<0.5, we delimited a squared area defined by the tangent to the ellipse that included 90% of the values (Fig. 2c).

Table 1 – Regression coefficients between pairs of indicators

Indicator 1	Indicator 2	R^2
VC	EI	0.76
VC	CSI	0.78
EcE	SOM	0.75
NB	PB	0.72
LIc	LIf	0.70

Results

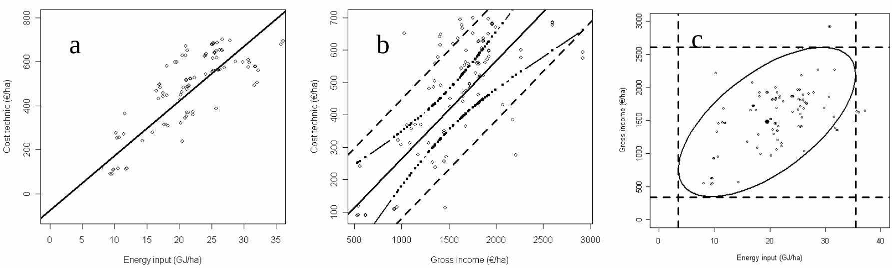

Fig. 2 – Examples of relationships between indicators, with R^2>0.7 (a), from 0.5 to 0.7 (b), and <0.5 (c). Solid line: regression (a and b), or ellipse that include the 90% of the indicator value pairs (c). Dashed line: empirical probable area between pairs of values. Bullet line: confidential interval.

Only five indicator pairs have a R^2 > 0.7 (Table 1) and four have R^2 from 0.5 to 0.7. The reason is that in general most of the selected indicators are independent from the others. Indeed, they are able to describe different aspects of field sustainability in different compartments. The information provided by different indicators has a small replication of information, because the behavior of each indicator is generally independent from the others.

Conclusions

We are currently working on the integration of these relationships with the ranges of sustainability, to identify sustainable combinations of indicators for cropping systems management, i.e. when indicator values are inside the sustainability area (α, Fig. 1). In addition, it will be possible to understand in which direction it is necessary to drive the farmers' management in order to improve sustainability. This will be done while respecting the functional relationships described by the realistic range between pairs of indicators.

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Curriculum vitae

Nicola Castoldi was born February 18th, 1979 in Cassano d'Adda (Milano), Italy.

He started his higher study on University of Milano in 1998, and graduated in April 2004, in Environmental Sciences, agro-forestry specialization; mark 110/110 cum laude.

In January 2005 he started his PhD research at the Department of Crop Science of University of Milano. During this period he spent three months of internship at ITC (International Institute for Geo-Information Science and Earth Observation) in the Department of Earth Observation Science, under the guide of Prof. Alfred Stein, and other three months at Department of Geography, Division of Social and Industrial Ecology at the University of Zürich Irchel, under the guide of Prof. Claudia Rebecca Binder. During the PhD he has participate to the PRIN 2004 project "Un agricoltura per le aree protette" ("Agriculture for protected areas"), funded by the Italian Ministry of University and Research, and coordinated by Prof. M. Borin.

He is member of Italian Society for Agronomi (SIA) and of European Society for Agronomy (ESA).

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