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

INDICATORI AGRO-ECOLOGICI PER LA SOSTENIBILITÀ DEL SISTEMA APPEZZAMENTO-AZIENDA.II. NUTRIENTI E FITOFARMACI

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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.

Keywords: Agro-Ecological Indicators, Nitrogen, Phosphorus, Lethal Dose, Lethal Concentration, Toxicity/Exposure Ratio, Predicted Environmental Concentration, Active Ingredient.

Riassunto
La valutazione della sostenibilità dei sistemi aziendali e colturali può essere condotta attraverso l'utilizzo di misure dirette, di modelli di simulazione o di indicatori; questi ultimi hanno il vantaggio di richiedere pochi input, di essere velocemente calcolabili e facilmente interpretabili, permettendo dei confronti nello spazio e nel tempo, rappresentando una sintesi dei processi in sistemi complessi. In un precedente articolo abbiamo proposto una lista di indicatori inerenti l'utilizzo dell'energia fossile, il paesaggio e la gestione del suolo. In questo articolo vengono presentati indicatori riguardanti l'utilizzo dei nutrienti e dei fitofarmaci. Sono stati selezionati indicatori che possono essere applicati a livello di campo ed aziendale, basati su dati ottenibili dagli agricoltori e/o da database esistenti; sono stati esclusi dalla discussione gli indicatori basati su misure dirette.
Il bilancio dei nutrienti è la differenza fra gli input e gli output a livello aziendale o di singolo campo (surplus se positivo, deficit se negativo). I vantaggi di questo indicatore sono la semplicità, la ridotta mole di dati necessari e la sua applicabilità ai diversi elementi. Comunque, il bilancio dei nutrienti non indica quanto surplus può essere effettivamente perso dal

sistema ed in che modo. L'indicatore di rischio per la qualità delle acque integra i surplus calcolati a livello di appezzamento con semplici informazioni pedo–climatiche. Descriviamo inoltre due indicatori relativi alla gestione dell'azoto proposti per i seminativi e per i pascoli al fine di superare i limiti del bilancio dei nutrienti, e l'indicatore relativo alla gestione del fosforo, che confronta la dose di fosforo applicato con quella raccomandata, identificando il rischio riguardante l'esaurimento delle risorse rinnovabili e quello inerente il depauperamento delle riserve del suolo.
Rispetto agli indicatori di rischio per i nutrienti, la definizione di un indicatore di rischio relativo all'utilizzo di fitofarmaci è più problematico. In effetti i fitofarmaci mostrano una gran varietà di effetti potenziali sulla salute umana e sui diversi ecosistemi; di conseguenza, l'analisi dei loro rischi potenziali richiede procedure molto complesse e diversificate in funzione del comparto ambientale considerato (acque sotterranee, acque superficiali, aria e suolo). Ciò ha portato allo sviluppo di diversi indicatori di rischio per i fitofarmaci che differiscono in termini di variabili considerate, area di attività, scala di analisi e metodologie utilizzate (alberi decisionali interattivi, approcci a quoziente di rischio, tavole di punteggio, sistemi fuzzy). Alcuni indicatori usano semplici algoritmi per stimare il rischio, mentre altri fanno ricorso a modelli più complicati. I più semplici e generici indicatori richiedono pochi dati (quali la dose di applicazione), ma generalmente non considerano il destino ambientale e la diffusione dei prodotti chimici. Al contrario, gli indicatori più complessi richiedono l'utilizzo di modelli predittivi per poter valutare l'esposizione potenziale ai diversi principi attivi da parte di organismi non bersaglio. Vengono qui presentati alcuni indicatori di rischio per i fitofarmaci a differenti livelli di complessità i quali possono essere utilizzati a livello aziendale e di campo, al fine di ottenere un quadro dei differenti approcci disponibili in letteratura, evidenziandone i punti di forza ed i limiti.

Parole chiave: *Indicatori Agroecologici, Azoto, Fosforo, Dose letale, Concentrazione Letale, Rapporto Tossicità/Esposizione, Concentrazione Ambientale Prevista, Principio Attivo.*

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; European Environmental Agency, 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.

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.

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.

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 · EW / (WHC + EW), where S is the N surplus obtained with the soil surface balance (kg N ha^{-1}). Therefore the indicator is:

$$I = \frac{S}{WHC + EW} \quad (1)$$

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.

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):

$$I_N = 7 + \sum_{i=1}^{n} kv_i + \sum_{i=1}^{n} kl_i + k_b + k_t \quad (2)$$

Tab. 1 – Volatilisation coefficients (from Bockstaller and Girardin, 2000).
***Tab. 1** – Coefficienti di volatilizzazione (da Bockstaller e Girardin, 2000).*

	No ploughing		Ploughing (by 24 hours)	
	Calcareous soil (>5 %)	Non calcareous soil	Calcareous soil (>5 %)	Non calcareous soil
For fertilization from 1 September to 31 March				
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	

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) (Tab. 1). Then, $kv_i = -V_i / 30$.

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

$$L_i = (X_i - V_i) \cdot t_i \cdot f \cdot b \qquad (3)$$

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:

$$t_i = \frac{D_i - D_{50}}{D_s - D_{50}} \qquad (4)$$

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:

$$N_u + N_o + N_r + N_h + N_f - N_c \qquad (5)$$

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:

$$\left\{ \left(\sum_{i=1}^{n} \left(X_i - V_i - L_i \right) \right) - X_R \right\} / 2 \qquad (6)$$

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

$$\left\{X_R - \left(\sum_{i=1}^{n}\left(X_i - V_i - L_i\right)\right)\right\}/2 \quad (7)$$

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.

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:

$$I_P = 7 + \min(P_{res}, P_{sol}) + kt \quad (8)$$

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):

$$P_{res} = \begin{cases} -\left(P_a - P_r\right)/30 & if \quad P_a > P_r \\ 0 & if \quad P_a \le P_r \end{cases} \quad (9)$$

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.

$$P_{sol} = \begin{cases} \left(P_{aa} - P_r\right)/30 & if \quad P_{aa} < P_r \\ 0 & if \quad P_{aa} \ge P_r \end{cases} \quad (10)$$

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

Tab. 2 – Recommended (+) and non–recommended (–) P applications. Information used in the calculation of the phosphorus indicator I_P (reproduced from Bockstaller and Girardin, 2000).
Tab. 2 *– Applicazioni di P raccomandate (+) e non raccomandate (–). Informazioni usate nel calcolo dell'indicatore fosforo I_P (da Bockstaller e 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.

amount of P applied with organic fertilisers and P applied using the recommended forms of inorganic P fertilisers (Tab. 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 $|P_a - P_r| \le 15$ 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.

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.).

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.):

$$EEP_{air} = AR_{a.i.} \cdot VP/1000 \quad (11)$$

$$EEP_{soil} = AR_{a.i.} \cdot DT_{50soil} \quad (12)$$

$$EEP_{groundwater} = EEP_{soil} \cdot K_{om}^{-1} \quad (13)$$

where $AR_{a.i.}$ is the application rate(kg a.i.ha^{-1}), VP is the vapour pressure at 25°C (mPa), 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.:

$$LI = \sum_{k=1}^{n} \frac{AR_{a.i.k}}{TOX_k} \quad (14)$$

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 kth 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..

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.

Tab. 3 – Example of label precautions and assigned scores (L_{SER}) (reproduced from Lewis et al., 1997b).

Tab. 3 – *Esempi di frasi di rischio in etichette e relativi punteggi (L_{SER}) (da 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

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:

$$P_f = f(L_{SER}) + \sum_{k=1}^{n} f(E_k \cdot AR_k) \qquad (15)$$

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) (Tab. 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}) (Tab. 4)
The S_{leach} value is obtained using the GUS (Groundwater Ubiquity Score: Gustafson, 1989):

$$GUS = \log DT_{50} \cdot (4 - \log K_{oc}) \qquad (16)$$

Tab. 4 – Vapour pressure and corresponding volatilisation risk score (S_{air}) (reproduced from Lewis et al., 1997a).
Tab. 4 – Pressione di vapore e corrispondenti punteggi di rischio volatilizzazione (S_{air}) (da 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

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.

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.

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:

$$NI = \sum_{k=1}^{n} \left[(S_E + S_{Bi} + S_{Be} + S_P + S_B)^3 \cdot A_k \right] \qquad (17)$$

where n is the number of different a.i. applied in one year; S_E is the score attributed to TER for earthworms

Tab. 5a – Scores for the Toxicity/Exposure Ratio related to earthworms (TER_{ew}), used in the calculation of the Norwegian Indicator (NI).
Tab. 5a – *Punteggi per il Rapporto Tossicità/Esposizione per gli organismi terricoli (TER_{ew}), usati nel calcolo dell'Indicatore Norvegese (NI).*

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

Tab. 5b – Scores for the Toxicity/Exposure Ratio related to birds (TER_{Bi}), used in the calculation of the Norwegian Indicator (NI).
Tab. 5b – *Punteggi per il Rapporto Tossicità/Esposizione per gli uccelli (TER_{Bi}), usati nel calcolo dell'Indicatore Norvegese (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).
Tab. 5c – *Punteggi per il Quoziente di Rischio per le api (HQ_{Be}), usati nel calcolo dell'Indicatore Norvegese (NI) (HQ_o = Quoziente di Rischio Orale; HQ_c = Quoziente di Rischio per contatto).*

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

Tab. 5d – Persistence score (S_P) for pesticide in Norwegian Indicator (NI); DT_{50} is half–life of a.i. in soil.
Tab. 5d – *Punteggi per il fattore Persistenza (S_P) dell'Indicatore Norvegese (NI); DT_{50} è l'emivita del principio attivo nel suolo.*

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

Tab. 5e – Bioaccumulation score (S_B) for pesticide in Norwegian Indicator (NI).
Tab. 5e – Punteggi per il fattore Bioaccumulo (S_B) dell' Indicatore Norvegese (NI).

Persistence in soil, DT_{50}	Log Kow < 3	3 ≤ Log Kow ≤ 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

(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.
The score S_E (Tab. 5a) depends on the ratio between $LC_{50\text{-}14days}$ (14 days Lethal Concentration for earthworms of each a.i.; mg a.i. kg $soil^{-1}$) and PEC_{acute_soil} (mg a.i. kg^{-1} soil).

$$PEC_{acute\_soil} = AR_{a.i.} 10^2 \frac{1 - f_{int}}{D_e \cdot BD} \tag{18}$$

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} (Tab. 5b) is based on TER_{bird}, calculated from PIEC (mg kg^{-1}) and dietary toxicity (dietary $LC_{50\text{-}14days}$, mg kg^{-1} food):

$$TER_{bird} = \frac{LC_{50-14days}}{PIEC} \tag{19}$$

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^{-1} body weight) 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. Tab. 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 Tab. 5d and 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.

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:

$$PEC_{H_2O} = \frac{AR_{a.i.} \cdot L}{D_e} \tag{20}$$

with L = $L_{drift} + L_{runoff}$. L_{drift} is calculated according to the Ganzelmeyer tables (Biologische Bundesanstalt, 2000) (Tab. 6), whereas L_{runoff} is calculated using the following equation:

$$L_{runoff} = \frac{Q \cdot f \cdot e^{\frac{3 \cdot \ln 2}{DT_{50soil}}} \cdot 100}{R_a \cdot (1 + K_d)} \tag{21}$$

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:

Tab. 6 – L_{drift} (%) calculated according to Ganzelmeyer tables (from Biologische Bundesanstalt, 2000).
Tab. 6 *– L_{drift} (%) calcolato in base alle tabelle di Ganzelmeyer (da 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

$$K_d = K_{oc} \cdot f_{OC} \quad (22)$$

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

$$f = f_1 \cdot f_2 \cdot f_3 \quad (23)$$

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, 1999). If the slope is < 20%, then

$$f_1 = 0.02153 \cdot slope + 0.001423 \cdot slope^2 \quad (24)$$

or else $f_1 = 1$.

$$f_2 = 1 - f_{int} \quad (25)$$

$$f_3 = 0.83^{wbz}/100 \quad (26)$$

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 (Tab. 7). Finally, the indicator is calculated according to the following equation:

$$SWIPE=(A + D + F) \cdot P1 \cdot P2 \cdot P3 \cdot WI \cdot T \quad (27)$$

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;

$$WI = l_w / (l_{nw} + l_w) \quad (28)$$

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).

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

Tab. 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).
Tab. 7 *– Punteggi e pesi assegnati ad ogni p.a. in base al rapporto PEC/TER ottenuto per i tre organismi non target (da 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

all compartments according to this equation:

$$EPRIP = RS_{groundwater} \cdot RS_{surface\_water} \cdot \cdot RS_{soil} \cdot RS_{air} + 5\ N_4 + 50\ N_5 \quad (29)$$

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.

$$PEC_{groundwater}(\mu g\ l^{-1}) = 2.379 \cdot AF \cdot \frac{AR}{10} \cdot \frac{(1-f_{int})}{P} \quad (30)$$

where 2.739 is a conversion factor to transform mg into µg and days into years, f_{int} (unitless) is crop–interception (Tab. 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:

$$AF = e^{-0.693 \cdot \frac{TR}{DT_{50soil}}} \quad (31)$$

where TR is the average residence time:

$$TR = \frac{L \cdot RF \cdot FC}{Q} \quad (32)$$

where

$$RF = 1 + \frac{BD \cdot f_{OC} \cdot K_{oc}}{FC} + \frac{AC + K_h}{FC} \quad (33)$$

where L (m) is groundwater depth, RF is the retardation factor, Q (m $year^{-1}$) 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.

$$PEC_{acute\_soil} = AR_{a.i.} \cdot 10^2 \frac{(1-f_{int})}{D_e \cdot BD} \quad (34)$$

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:

$$PEC_n = PEC_{acute\_soil} \frac{1-e^{-nki}}{1-e^{-ki}} \quad (35)$$

Tab. 8 – Crop interception factor for different crops (f_{int}) (from Padovani et al., 2004).
Tab. 8 *– Fattore di intercettazione per diverse colture (f_{int}) (da 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 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:

$$C_{air} = \frac{J_0}{V_f} \quad (36)$$

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:

$$J_0 = \frac{D_a \cdot C_{sa}}{d} \quad (37)$$

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:

$$D_a = 0.036\ (76 / MW)^{1/2} \quad (38)$$

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

$$C_{sa} = \frac{PEC_{acute\_soil} \cdot BD \cdot P_a}{FC \cdot P} \quad (39)$$

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

$$P_a = \frac{Z_a \cdot V_a}{Z_a \cdot V_a + Z_w \cdot V_w + Z_s \cdot V_s} \quad (40)$$

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:

$$Z_a = \frac{1}{R \cdot T} \quad (41)$$

$$Z_w = \frac{S}{V_p} \quad (42)$$

$$Z_a = \frac{K_{oc} \cdot BD \cdot Z_w}{1-P} \quad (43)$$

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:

Tab. 9 – Drift percentage (f_{drift}) for different crops used in PEC_{drift} calculation (from Padovani et al., 2004).
Tab. 9 – *Percentuale di deriva (f_{drift}) per differenti colture usata per il calcolo di PEC_{drift} (da Padovani et al., 2004).*

Water distance	Vineyard		Orchards		Vegetables		Field
(m)	Early	Late	Early	Late	<50 cm	>50 cm	
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	

$$PEC_{drift} = \frac{AR_{a.i.} \cdot f_{drift}}{10 \cdot V} \qquad (44)$$

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 (Tab. 9), V is the volume of water in the ditch (per unit of length).

$$V = \frac{h \cdot (b+h)}{b+2h} \qquad (45)$$

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.

$$PEC_{runoff} = \frac{P_r \cdot AR_{3d} \cdot F_{aq}}{D_r} \qquad (46)$$

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:

$$P_r = F_{st} \cdot F_s \cdot F_r \cdot (0.55 \cdot \log K_{oc} + 1.47) \qquad (47)$$

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:

$$F_S = 0.124 \cdot Slope + 0.0082 \cdot Slope^2 \qquad (48)$$

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

$$F_r = 0.028 \cdot RE + 0.00011 \cdot RE^2 \qquad (49)$$

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}).

$$AR_{3d} = PEC_{soil} \cdot \frac{1-e^{-k3}}{k3} \qquad (50)$$

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

$$F_{aq} = 1 / (1 + Q) \qquad (51)$$

$$Q = \frac{2 \cdot K_{oc} \cdot f_{OC}}{100 \cdot R_{off}} \qquad (52)$$

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

$$R_{off} = 3.83 - (0.12\ Rf + 0.00056\ Rf^2) \qquad (53)$$

and in orchard crops

$$R_{off} = -152.4 + (0.4\ Rf) \qquad (54)$$

where Rf (mm) is the annual rainfall.

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:

$$EIP_s = \frac{PEC}{MPC} \cdot AR_{a.i.} \cdot 100 \qquad (55)$$

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.

$$EIP_{groundwater} = \frac{PEC_{groundwater}}{0.1} \cdot 100 \qquad (56)$$

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).

$$EIP_{surface\_water} = \frac{PEC_{surface\_water}}{0.01 \cdot LC_{50water\_organisms}} \cdot 100 \quad (57)$$

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),

$$PEC_{surface\_water} = 0.1 \cdot AR_{a.i.} \frac{D_r}{D_e} \quad (58)$$

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.

$$EIP_{acute\_soil\_risk} = \frac{PEC_{acute\_soil}}{0.1 \cdot LC_{50worm}} \cdot 100 \quad (59)$$

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.

$$EIP_{chronic\_soil\_risk} = \frac{PEC_{2\_years}}{0.1 \cdot NOEC_{earthworm}} \cdot AR_{a.i.} \cdot 100 \quad (60)$$

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

$$PEC_{2\_years}\ (\text{mg kg}^{-1}) = P_2 \cdot PEC_{acute\_soil} \quad (61)$$

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}):

$$PEC_{soil\_moisture} = \frac{PEC_{2\_years}}{K_{om} \cdot f_{OC} + 0.2} \quad (62)$$

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

Discussion and Conclusions

Nutrient indicators

We have selected several indicators of different complexity (Tab. 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 ad-

Tab. 10 – Comparison of the nutrient indicators presented in this paper.
***Tab. 10 –** Confronto fra gli indicatori sui nutrienti presentati in questo articolo.*

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

ET_a = actual evapotranspiration

vantage 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).

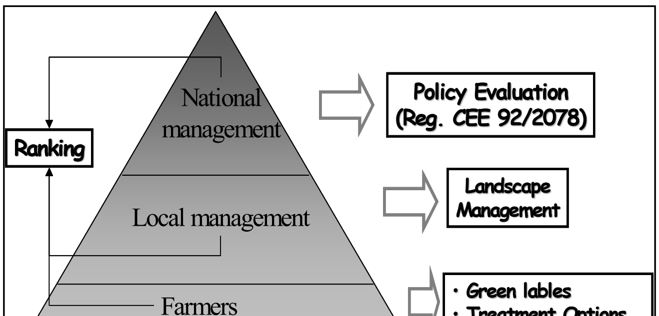

Fig. 1 – Use of pesticide risk indicators at different scale levels. Each level has a particular goal. From farm–level to national management, the ranking of pesticides (classification) in terms of their potential risk can be utilized.

Fig. 1 *– Utilizzo degli indicatori di rischio per i fitofarmaci a differenti livelli. Ogni livello ha un obiettivo particolare. La classificazione in termini di rischio potenziale può essere utilizzata, dal livello dell'organizzazione aziendale a quello nazionale.*

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.

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. 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.. 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.

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