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Landscape analysis and assessment for the management of agricultural non point pollution sources

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1. *Introduction*

1.1. The context

It is increasingly recognized that more sustainable approaches are needed for planning and managing landscapes and environment in a general approach.. The spatial dimension of sustainability engages processes and relations between different land uses, ecosystems at different scales, and over time (Botequilha et al., 2002).

The success of landscape planning and environmental management strategies depends largely on the congruence between the operational scales of landscapes and the spatial scope of the planning instruments (Diaz-Varela et al., 2009).

Environmental management is a process of decision-making that considers all the different variables and processes that characterize the environment as a whole. Risk assessments, precautionary principles, adaptive management and scenario approaches are adopted to deal with the uncertainty of nature.

The management of environmental issues is complex. Many optimization models exist but generally there is a knowledge gap in linking all the environmental variables and objectives.

Considering natural variables, landscape features, climate, land use, land cover and all the environmental constraints is a challenge in developing management plans and strategies. Uncertainty and variability are the main feature of this big challenge. As an example, as agricultural systems become more complex, multiple objectives that are in conflict with each other need to be addressed. Competition for scarce resources by different enterprises is a major concern in many agricultural production systems. Competition occurs at farm level e.g. between different crops as well as regional level, where utilization of scarce water resources for agricultural purposes often comes into conflict with the requirement for in stream ecosystem services. For example, in agriculture systems conflict may arise from maximizing economic returns (i.e. net revenue) as opposed to minimizing the use of resources such as water, fertilizer applications. On the other hand, minimizing costs rather than maximizing net revenue may also be important in some water management systems. Under these conditions, multiple criteria decision-making techniques (MCDM) are useful tools to explore different management options.(Xevi and Khan, 2005). The success of multi-objective planning will depend on clear goal definition. If goals and objectives are not articulated at the outset, there will be no criteria for assessing the suitability of specified actions. Having set objectives, it is then necessary to identify, for each one, the elements that are required in a landscape, the quantities or area required, their distribution in the landscape, and the appropriate management regime. If these requirements can be determined, they must be combined in a way that addresses all objectives (Lambeck, 1999).

Furthermore, focusing on water compartment, for example, in most Mediterranean countries groundwater is the primary resource for irrigation and drinking. Nowadays, scarcity of water is a limiting factor for sustainable growth. Therefore, preserving its availability and quality is a key issue. Excessive groundwater withdrawal is a risk factor, because it causes sea-water intrusion phenomena resulting in progressive groundwater salinization, which is particularly damaging for agriculture.

Generally speaking, making decisions in a situation where environmental risk is involved is a political action based on information collected during risk assessment. Therefore, the evaluation of environmental risk due to anthropic activities is an important step in mitigating the impact of such activities on natural resources and in recreating the co-evolutionary process between human and natural components of the environment. A territory is, in fact, the result of a stratification process characterized by the outputs of complex interrelationships between anthropic activities and sites. Among such activities, agriculture is surely one of the most relevant. However, massive soil exploitation, continuous groundwater withdrawal and the use of fertilizers and pesticides have a significant negative impact on the environment. Finding out how compatible a specific anthropic development is with environment conservation by assessing its impact on natural resources is a key step towards understanding the interactions between territory and local activities. In this context, environmental risk assessment and management represent a useful tool for describing local conditions characterized by a high degree of environmental risk. Risk assessment can be defined as the process of estimating the possibility that a particular event may occur under a given set of circumstances. Risk management, on the contrary, is the process in which decisions aimed at reducing risk and preserving public health and environmental resources are made, using information gathered during the assessment phase. Thus, environmental risk assessment could be considered as the basis of a decision making process (i.e. risk management) (Uricchio et al., 2004).

Starting from the concept that, nowadays, many farms are not sustainable in their current form, even if this industry has generated significant wealth, it has also created major problems in the form of land degradation, pollution, loss of biological diversity. Because these processes cannot be managed by individual land-holders acting in isolation, it is essential that there be a shift from traditional farm-based, single-objective land management, to multiple-goal, sustainable management practices implemented at catchment or regional scales.

By the context described above, agricultural activities play a main role in influencing a sustainable development. For example, focusing on farming, the large quantities of slurry and manure that are produced where animals are raised, could be an important source of organic matter and nutrients for agriculture. The use of these wastes via land application, after opportune treatments, could improve physical properties of soil, such as soil porosity, water holding capacity. For this reason, the application manure could be beneficial for soil conservation, especially in degraded soils and soils susceptible to erosion, although the response to soil amendment is soil-site specific. Obviously, the benefits of manure may be partially offset by the risk of water pollution associated with runoff and leaching from fields, especially if rainfall occurs shortly after application.

Ramos et al. (2006) reported that the application of manure to the fields has implications for erosion processes. When manure is applied on the soil surface, the quantity of eroded material decreases but runoff can increase by up to 30%. Hence the pollution of surface water can increase and, if rainfall takes place within a short period after application, faecal coliforms in runoff waters reach high concentrations, exceeding the standards for bathing water quality, especially because they are transported in association with organic matter particles rather than linked to the soil or suspended in water (Ramos et al., 2006).

This thesis study tries to product a general framework of the issues concerning agricultural non point pollution sources, their management and their connection with the landscape. Particularly, the thesis focuses on an area in south Italy where livestock activities are intensive and the consequent manure has to be managed. Further, studies about landscape assessment were developed focusing on Brittany region (France).

The aim of the thesis researches is to have a clear idea of the environmental problems connected to agricultural non point pollution sources in the areas mentioned above and to identify a way to manage and reduce this kind of pollution.

2. Agricultural non point pollution sources

2.1. Nitrate directive

In 1991, the council of environmental ministers of the European Union (EU) adopted the council directive concerning the protection of waters against pollution caused by nitrates from agricultural sources, 91/676/EEC (European Commission, 1991). The directive aims at reducing and preventing water pollution caused by nitrates from agricultural sources. Before the end of 1993, member states were obliged to identify so-called ‘vulnerable zones’ and establish good agricultural practices. These should require that farming be practiced in a way that minimizes the pollution of water by nitrates. These should specifically result in nitrate concentrations in groundwater not exceeding 50 mg $NO3^{-}$/l.

To realize the objectives of the directive, member states established Action Programs, that should include rules for (1) the periods in which the use of certain types of chemical fertilizer are prohibited, (2) the storage capacity for, or the disposal of, livestock manure and (3) limitation of land application of fertilizers. More specifically, account should be taken of local characteristics such as soil conditions (type and slope), climatic conditions and irrigation and balancing N supply with crop N demand, including both chemical fertilizer and manure. Action Programs should also ensure that the amount of livestock manure applied does not exceed the equivalent of 170 kg N/ha, corresponding with 2 livestock units per ha.

Moreover, member states were allowed to derogate from this standard, but only on the basis of objective criteria relating to crop and soil characteristics. Some examples of criteria have already been provided in the directive, such as a long growing season, crops with a high N uptake, high net precipitation and soils with a high denitrification capacity (Sonneveld et al, 2003).

In this context many control and assessment plans were developed. For example to control and prevent nutrient pollution from agricultural non-point sources, the Dutch government introduced the Mineral Accounting System (MINAS) in 1998, a nutrient bookkeeping system which taxes farms with nutrient surpluses exceeding safe threshold values. Moreover, reducing surpluses is assumed to be most effective through an improvement of management. Changing farm management rather than farm structure can therefore reduce nutrient surpluses more effectively, making it a more interesting approach for policy makers. (Ondersteijn et al., 2003).

Even if the directive 91/676/EEC has been in operation since 20 years ago, most of the constraints defined by the directive have been largely not respected by the member states. The main obligations not completely respected are the application of the so called "Action Programmes", the management of the monitoring campaigns to verify water quality in terms of nitrate concentration, the identification of the vulnerable zones.

2.2. Agricultural "wastes"

Organic wastes are typically products of farming, industrial or municipal activities, and are usually called ''wastes'' because they are not the primary product. However the goal is to make the ''waste'' a resource that can be utilized and not just discarded. Possible uses of organic wastes include use as fertilizer and soil amendment, energy production, and production of chemicals (volatile organic acids, ammonium products, alcohols). Agriculture has traditionally used animal manures for fertilizer and improving soil physical and chemical properties. Utilization of various organic wastes in agriculture depends on different factors, including the characteristics of the waste such as nutrient and heavy metal content, energy value, odor generated by the waste, availability and transportation costs, benefits to agriculture, and regulatory considerations. Further considerations about this context regard:

regional imbalances of nutrients (e.g., not enough land on the farm to apply nutrients produced by animals on the farm), imbalance of nutrients in waste compared to crop needs, the alternative use of chemical fertilizers, the variability in nutrient content, satisfying environmental regulations on application amounts, application timing, and application methods, and possible environmental concerns, such as emission of ammonia and other gases, odor, and pathogens. One of the alternative management strategy could be changing animal feeding in a way to convert final outputs (mainly manure) into higher value resources. In addition, potential transport of pathogens via air or water, and environmental air quality concerns with emissions of ammonia, methane, hydrogen sulphide, nitrous oxide, and odorants can result in more regulations, and lead to alternative treatment schemes, which can change the nature of the organic wastes that are applied to land or used in horticulture uses. Concerns for animal welfare and development of innovative energy policies also have potential to affect changes in management of manure and other organic materials. Some of the manure management/treatment systems are discussed in publications such as Burton and Turner (2003).

Several challenges for utilizing organic wastes for soil amendment and fertilizer value exist. These challenges are linked to the possible environmental impacts which may result from land application of animal manures and other organic wastes. Among these impacts the most frequent are the emission of greenhouse gases, emission of bad odors, production of pathogens, the water pollution by nitrogen, phosphorous, organic matter and pathogens, heavy metals soil accumulation.

Suitable equipment and management are thus required. One of the most critical and important aspect is how the application rates are determined. If application rate of animal manure is based on nitrogen, then phosphorus is usually applied in excess of crop uptake, because of the difference in the N:P ratios which exist between the manure content (about 3:1) and the crop requirement (about 8:1). A management option to address excessive N, P, Cu and Zn application to land may include diet manipulation strategies to reduce intake and increase digestibility and absorption of these compounds by the animals, thus producing less nutrients in manure. When the farm produces excessive nutrients for application on-farm, then treatment options can be added to reduce N (e.g., by conversion to dinitrogen gas) or partition nutrients into more concentrated or stable forms that can be transported elsewhere. For reduction in pathogens, lime treatment or high temperature aerobic or anaerobic treatment is required to essentially eliminate pathogens. To reduce odor, aeration treatment offers a viable, but somewhat expensive option. Also, different land application techniques are used to reduce odor and ammonia emission, such as injection.

Composting is a good strategy to move nutrients off farm. By composting, a more valuable resource can be obtained, but markets must be developed and farmers must determine the extra processing to be manageable and economically viable. There is also the possibility of a centralized composting facility that serves several farms. Composting can emit ammonia and other gases of concern to the atmosphere, requiring consideration for collecting and/or treating off-gases.

In some cases, energy recovery from anaerobic or aerobic digestion of organic wastes can be beneficial for certain objectives, but may have minimal benefit for nutrient management (Burton and Turner, 2003). Nutrient content generally remains unchanged, but nutrient availability may be increased and soluble organic matter reduced. Another benefit can be reduction of odor. The energy can be used on-farm or sold. If government energy policies were improved to support more ‘‘green energy’’ production, then more farms might consider anaerobic digestion for manure treatment and energy recovery.

Inherent in considering alternative management schemes for organic wastes are the costs and benefits. If regulations or environmental factors require additional treatment that increases costs of production and operation, then the farmer loses profit unless costs are shared with the government or other agencies. It is not easy to determine environmental costs and benefits of alternative waste management policies. Government can offer incentives such as cost sharing of equipment or guarantee of not changing regulations for a period of time for improved waste treatment.

Economics may also suggest that a cooperative or regional facility is needed for certain waste management schemes. However, farmer and public acceptance of this is important because odor from large treatment plants and transportation of organic wastes on public roads may present more community concerns, and farmers may choose other alternatives if the regional treatment system is not clearly advantageous both economically and for management efficiency. Utilization of organic wastes occurs more easily if there are clear economic incentives. Better organization through farmer cooperatives, organic waste sellers, and government or other agencies could improve the economics. At least initially, more government subsidies may be needed to help distribute nutrients over a larger region by helping with transportation or costs of further processing (Wasterman, 2005)

As a result of relatively more successful control of point sources of nutrients, agricultural nonpoint sources remain the most important contributor of nutrient loads impacting the quality of water resources (USEPA, 2000). The impact of nutrients on the quality of US water resources has been widely documented. Various studies have shown that eutrophication of freshwater depends on the inflow of nutrients, mainly nitrogen (N) and phosphorus (P), from land uses above the body of water. Elevated levels of P have been consistently identified as the most important source of nutrient-induced freshwater quality problems. Due to various biochemical processes, P tends to be the limiting nutrient in the growth of algal forms in rivers and lakes. Thus, excessive P loadings are a significant factor in the eutrophication of some receiving waters. Nonpoint source pollution problems have been attributed in part to conventional tillage practices that lead to increased soil erosion rates and associated sediment-bound nutrient losses, because of the inversion of most or all of the crop residues into the soil. Similarly, fertilizer and manure placement methods have been identified as potential factors impacting water quality. Notwithstanding the impacts of conventional tillage methods, various studies have found that plowing or disking fertilizer into the soil could reduce losses in runoff (Osei et al., 2003).

2.3. Campania region context

The Campania region is situated in the southern Italy, west coast (Fig.2. 1).

The region engendered great interest mainly because it appears as a lab area for the studies about agricultural non point pollution sources and their management.

The regional surface is 13 600 km^2. The territory is characterized by the 50% of hills and 35% of mountains. In 2000 (V Agricultural Census) the SAU was about 590 000 ha representing the 43% of the total surface. In 2005 the SAU decreased until 560 000 ha, first because of agricultural activities abandonment. The main feature of the farms in Campania region is their small dimension (the 40% of the farms is characterized by a surface smaller than 1 hectare, ISTAT 2005). Thus, the decrease of the SAU corresponds to a decrease in number of the smallest farms and, consequently, to a development and increase in surface of the biggest farms.

In 2005 the 37% of the farms' surface was used for arable fields, the 21% for woods and the 14% for grazing.

Data collected in 2005 report that the 11% of the Italian livestock farms are in Campania. The comparison between the number of animal heads, mainly bovine and buffalo, in 2003 and 2005 shows an increase of about 10%. Whereas, in the same period, the number of livestock farms decreased (-20%). In 2007 the number of buffaloes and bovines was 480 000 (National Data Bank BDN, 2008).

In 2000-2004 the value-added by agriculture in Campania was the 4% of the total value added in the region. It was then characterized by an increase in 2003-2005 (+4,5%).

Agricultural activities involve the 5% of the regional workforce that corresponds to the 9% of the national datum (SeSIRCA elaborations 2005).

Several studies were carried out to quantify the non point pollution in Campania. They revealed that there were areas where nitrate concentration in water was great (Fig.2. 2, Onorati et al., 2003). The use of synthetic nitrogen fertilizers (600 kg/ha per year only in the province of Naples) was higher than the national average use. This contributed to the ground water pollution. In Caserta province the great fertilizers' use (about the 100% of the total mineral fertilizers) determined the increase of the pollution of the deeper ground water. Furthermore, the presence of high chloride content in the same samples, where nitrate content was high, involved the pollution by manure. This is perfectly coherent with the agricultural and livestock activities of the region, especially in particular areas such as the Caserta province (Infascelli et al, 2007 and 2009).

Basing on the European and National legislations (mainly Nitrate Directive and Italian Legislative Decree 152/99 and Ministerial Decree 7th April 2006) and on monitoring campaigns in Campania, the Region defined the vulnerable zones to agricultural nitrate (DGR 700/03) and many other regional programs (i.e. Programma d'azione per le zone vulnerabili all'inquinamento da nitrate di origine agricola) in order to obtain good guides lines for monitoring and checking and managing nitrate pollution. The Region identified 115 000 ha of vulnerable zones to agricultural nitrate.

The Region studies and the consequent measures and directive adopted, are all based on deep studies of the territory features. Among these, the soil and hydrogeological studies, conducted by Allocca et al.(2003) and by the Campania Region (www.sito.regione.campania.it/agricoltura/pedologia/home.htm), provide maps and information fundamental to characterize and to determine the connections between the several aspects that take part into the environmental and anthropic processes, such as farming.

a)

b)

Fig.2. 1 Italy (a); Campania region (b)

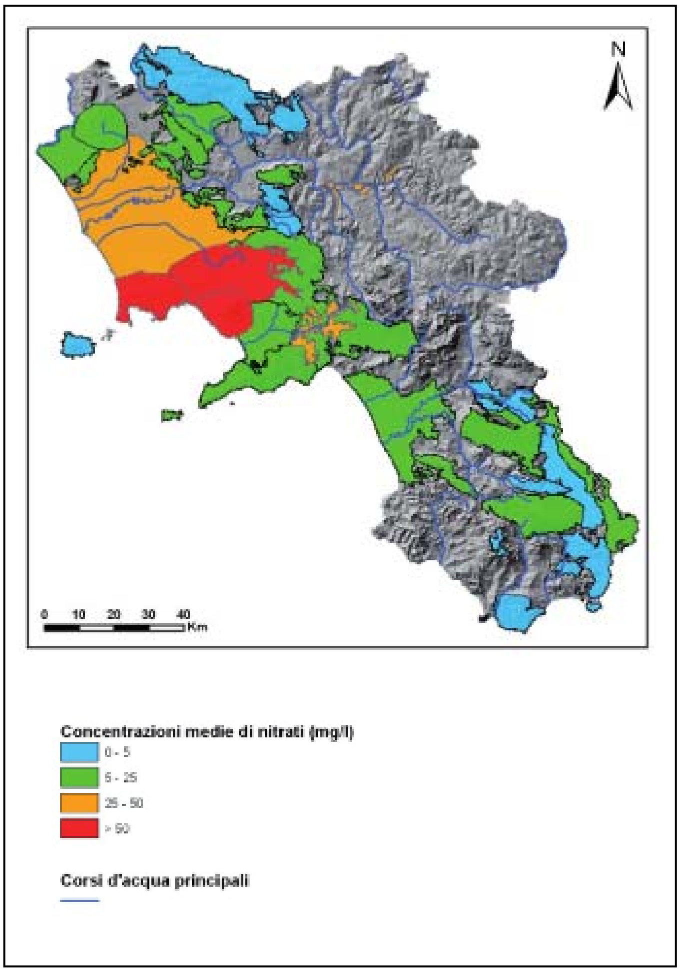


Fig.2. 2 Average nitrate concentration in ground water in 2007, (Adamo et al., 2009)

3. Materials and methods

3.1. GIS

Spatial issues such as the identification of sensitive areas, the determination of the degree of site suitability and the calculation of the waste application rate for various spatial locations can be resolved using a geographic information system (GIS). A GIS is a computer-assisted system for acquisition, storage, analysis and display of geographic data. It is also possible to combine a range of different kind of factors (biophysical, environmental, social, economic) to produce a planning output. The GIS is also capable of combining spatial and non-spatial data to produce a map with site-specific outputs. For instance, the spatial parameters (e.g. crop distribution and site suitability maps) could be combined with the non-spatial parameters (e.g. crop nutrient requirements) using a map algebra. With the availability of manure nutrient contents, the plant nutrient requirement map could also be translated into a manure application rate map using a similar algebraic function within the GIS.

Agricultural, environmental, and/or socio-economic problems may arise where the rate of animal waste application to agricultural fields is determined on a limited range of factors. For instance, using manure to meet the requirements of a single plant nutrient may not necessarily satisfy the requirements for other nutrients, or may lead to run-off and leaching losses to the environment. Moreover, it may direct manure application to inappropriate sites resulting in odour or economic viability problems. For example the rate of animal waste application within agricultural fields is generally based on the crop requirement for either nitrogen (N) or phosphorus (P). However, excess P may be applied to soils where the manure application rate is based solely on the nitrogen content. Excessive application of P is a serious environmental concern because it is the single most important nutrient influencing eutrophication and toxic algae blooms (Leone, 2004).

Therefore, there is a need for site-specific judgement in determining the manure application sites and rates. The site-specific judgements include excluding totally unsuitable areas from animal waste application, varying application rates on suitable areas based on site-specific requirements, and using measures to minimise any environmental impacts of animal waste application on marginally suitable areas (Badri B. Basnet et al., 2002).

GIS, supported by computer development, allows for a complete spatial analysis to solve problems in planning. Each project of environmental assessment has to be supported by data and information that are integrated into the territory context. An Information Geographic System allows the possibility to view, understand, question, interpret, and visualize data in many ways that reveal relationships, patterns, and trends in the form of maps, globes, reports, and charts. This is the reason why GIS technology should be integrated into any farm information system framework: it can be updated and used as support for assessment, review and monitoring actions that aim to pursue environmental and management targets.

In order to plan and monitor the environmental problems, the assessment of hazards and risks becomes the foundation for planning decisions and for mitigation activities. GIS supports activities in environmental assessment, monitoring, and mitigation and can also be used for generating Environmental models.

Enormous variety of environmental data is generated from different sources. GIS is a powerful tool to analyse the present environmental scenario and also to project the future. Some of the examples where GIS can be effectively used are in environmental planning, ground water contamination, water quality, waste management, air and water pollution, natural hazards and their mitigation etc.

GIS is one of the key tools in the environmental management. It can moreover be a communication tool of environmental information between the public and policy makers since it is the technical basis for the multimedia approach in environmental decision-making.

GIS is a powerful tool for environmental data analysis and planning. GIS stores spatial information (data) in a digital mapping environment. A digital map can be overlaid with data or other layers of information onto a map in order to view spatial information and relationships. GIS allows better viewing and understanding physical features and the relationships that influence in a given critical environmental condition. Factors, such as steepness of slopes, aspects, and vegetation, can be viewed and overlaid to determine various environmental parameters and impact analysis.

GIS can also display and analyze aerial photos. Digital information can be overlaid on photographs to provide environmental data analysts with more familiar views of landscapes and associated data.

A geographic information system (GIS) integrates hardware, software, and data for capturing, managing, analyzing, and displaying all forms of geographically referenced information. It allows territorial, but also logic and conceptual analysis.

The only location can not represent a real information, but if numeric, alphanumeric, statistical data are added the geographical datum becomes an information. A GIS stores data and information as thematic layers that can be linked one to each other. A first important distinction to make is between datum and information. the datum is the numeric entity that identify the characteristics of the studied process; the information is the datum reprocessing in order to be able to spatial represent the original datum or to produce new information. For example a DEM can be elaborated to obtain new information such as the slope, the exposure, the altimetry and other (Leone, 2004).

Some of the GIS application fields are: territorial and landscape planning, environmental assessment and monitoring, controlling agricultural production, etc.

3.2. Geostatistics

Natural and environmental phenomena, in general, are typically distributed in space and/or time, characterized by high variability, number and variety. Knowledge of attribute value has little interest unless location and /or time measurement are known for the data analysis. Thus, classical statistics might be too expensive and limiting in dealing with these issues. Geostatistics provides a set of statistical tools for incorporating the spatial and temporal coordinates of observations in data processing.

The development of geostatistics began in 1960s as a methodology for estimating recoverable reserves in mining deposits. Until 1980s it was mainly viewed as a means to describe spatial patterns and interpolate the value of an attribute of interest at unsampled location Today it is extensively used in the mining and petroleum industries, and in recent years has been successfully integrated into remote sensing and GIS. Geostatistic is now increasingly used to model the uncertainty about unknown values through the generation of alternative images that all honour the data and reproduce aspects of the patterns of spatial dependence or other statistics deemed consequential for the problem at hand. A given scenario or transfer function can be applied to the set of realizations, allowing the uncertainty of the response to be evaluated.

There are often only few measurement of the attribute of interest. The resultant predicted maps thus provide poor resolution and the corresponding uncertainty may be very large. In such situations it is critical to deal with information that is more densely sampled, even if further data are achieved by indirect yet exhaustive information or integrated through secondary data in prediction and simulation algorithms.

Geostatistics deals with spatial data, i.e. data for which each value is associated with a location in space. In such analysis it is assumed that there is some connection between location and data value. From known values at sampled points, geostatistical analysis can be used to predict spatial distributions of properties over large areas or volumes.

Statistics generally analyzes and interprets the uncertainty caused by limited sampling. For example, a conventional statistical analysis of core samples from a site investigation program might show that measured cohesion values of a material can be described by a normal distribution. However, this distribution only describes the population of values gathered in the investigation; it does not offer any information on which zones are likely to have high cohesion values and which areas low values. Geostatistical analysis, on the other hand, interprets statistical distributions of data and also examines spatial relationships. For the example given, it is capable of revealing how cohesion values vary over distance, and of predicting areas of high and low cohesion values. The discipline provides tools for capturing maximum information on a phenomenon from sparse, often biased, and often under-sampled data. Ultimately it produces predictions of the probable distribution of properties in space.

Geostatistics, as mentioned above, is a collection of statistical methods which were traditionally used in environment science. These methods describe spatial autocorrelation among sample data and use it in various types of spatial models. Geostatistics changes the entire methodology of sampling. Traditional sampling methods don't work with auto-correlated data and therefore, the main purpose of sampling plans is to avoid spatial correlations. In geostatistics there is no need in avoiding autocorrelations and sampling becomes less restrictive. Also, geostatistics changes the emphasis from estimation of averages to mapping of spatially-distributed populations.

Detailed description of most geostatistical methods can be found in Isaaks and Srivastava (1989) and Goovaertes (1997) .

A first step, in the aim of a good geostatistics response, is to avoid excessive data skewness. Thus, in case of asymmetric distribution of data, a good practice should be handle the extreme values by removing them, if not representative; classifying them in separated classes; using descriptive statistics more robust than the common mean value and the standard deviation (i.e. semivariogram, relative semivariogram, correlation coefficient etc.); transforming data through functions such as logarithmic or square root function.

A tool widely used to measure the degree of correlation between the values of a same attribute in two different locations, at distance h, is the scattergram. Moreover, spatial autocorrelation can be analyzed using covariance functions, correlograms and variograms (or semivariograms), which have the following mathematical equations:

- Covariance $$C(h)=\frac{1}{N(h)}\cdot\sum_{i=1}^{N(h)} z(x_i)\cdot z(x_i+h)-m_{-h}\cdot m_{+h}$$

- Correlogram $$\rho(h)=\frac{C(h)}{\sqrt{\sigma_{-h}^2\cdot\sigma_{+h}^2}}$$

- Semivariogram $$\gamma(h)=\frac{1}{2N(h)}\sum_{i=1}^{N(h)}[z(x_i)-z(x_i+h)]^2$$

where z(x_i) is the value of the attribute z at the point x_i ; h is the measurement distance (so called lag distance); $N(h)$is the number of data pairs at distance h; m is the expected value; s is the standard deviation.

Some base concepts of the geostatistics are:

random functions: in the study area the attributes vary following a specific probability distribution function, whose characteristic parameters are known (i.e. mean value and variance for a normal distribution).

Regionalized variables: they are random variables *z(x)* defined in space. Considering all the possible values that *z(x)* can assume in each point a random function *Z(x)* is obtained. All the random variables that form the random function *Z(x)* have the same cumulative probability distribution *F(z)* independent that does not depend on the location *x*:

$$F(x_1 \ldots x_N; z_1 \ldots z_N) = \text{Prob}\ \{ Z(x_1)<z_1 \ldots . Z(x_N)<z_N \}$$

Stationarity: the random function *Z(x)* is said to be stationary within the study area if the correspondent *F(z)* is invariant under translation:

$$F(x_1 \ldots x_N; z_1 \ldots z_N) = F(x_1+h \ldots x_N+h; z_1 \ldots z_N)$$

Stationarity is not a characteristic of the phenomenon under study, rather a decision made by user for the random function model. A random function is said to be stationary of order two when its expected value *E{Z(x)}* exists and is invariant within the study area and the spatial covariance *C(h)* exists and depends only on the separation vector *h*. Thus with h=0,

$$C(0)= E\{(Z(x)-m)^2\}=s^2$$

where s^2 is the sampling covariance.

The semivariogram (γ(h)) is the tool used to model the degree of spatial dependence between data separated by a vector **h**:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} [z(x_i) - z(x_i + h)]^2$$

where $z(x_i)$ is the variable value at x_i, **h** is the lag distance, *N(h)* is the number of data pairs at distance **h**.

The semivariogram provides information about the spatial continuity and variability of the data. Generally, the semivariogram increases with the distance *h*, until a constant value called *sill* at a distance called *range* (Fig.3. 1).

The sill corresponds to the sampling variance s^2, thus observations out of the *range* are not spatially correlated. The semivariogram value may not tend to zero when *h* tends to zero, although by definition γ(0)=0. Such a discontinuity is called nugget effect.

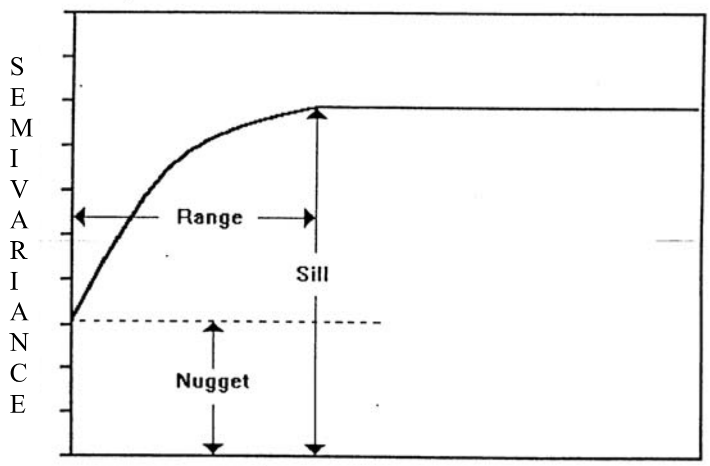


Fig.3. 1 Semivariogram features

It is mainly due to measurement errors and/or spatial sources of variation at distances smaller than the shortest sampling interval.

The semivariogram can be approximated by some mathematical model (Fig.3. 2):

I) Linear model $\gamma(\mathrm{h}) = \mathrm{C}_0 + \mathrm{C} \cdot \mathrm{h} \qquad 0 < h \le a$

II) Spherical model
$$\begin{cases} \gamma(\mathrm{h}) = \mathrm{C}_0 + \mathrm{C}\left[\dfrac{3}{2}\dfrac{\mathrm{h}}{\mathrm{a}} - \dfrac{1}{2}\left(\dfrac{\mathrm{h}}{\mathrm{a}}\right)^3\right] & 0 < h \le a \\ \gamma(\mathrm{h}) = \mathrm{C}_0 + \mathrm{C} & h > a \end{cases}$$

III) Esponential model $\gamma(\mathrm{h}) = C_0 + C[1 - \exp(-\mathrm{h}/a)]$

IV) Gaussian model $\gamma(\mathrm{h}) = \mathrm{C}_0 + \mathrm{C}[1 - \exp(\mathrm{h}^2/\mathrm{a}^2)]$

In these equations *h* is the lag distance, *a* is the *range*, C_0 is the *nugget effect* and C_0+C is the *sill*.

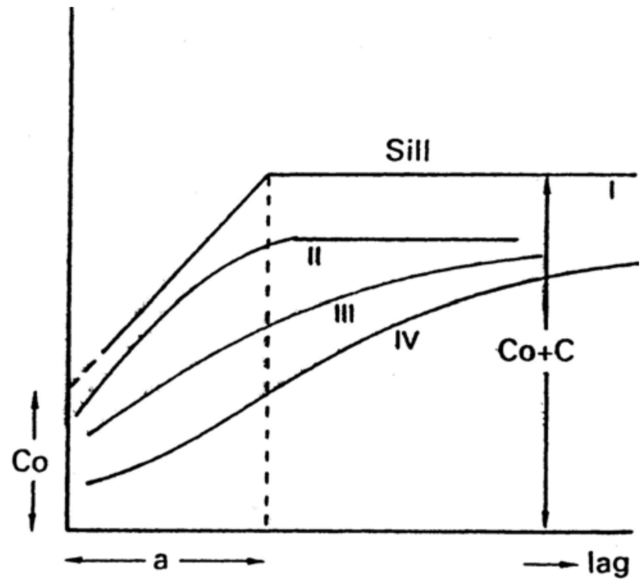


Fig.3. 2 Typical models of semivariogram

If data have intrinsic stationarity, that is the semivariance between two points depends only on the magnitude and the direction of **h** vector and not on positions, and if data have the second order stationarity, the semivariogram can be expressed as:

$$\gamma\,(h)= C(0) - C(h)$$

Thus the semivariance is expressed in terms of spatial covariance, that is autocorrelation C(h) between data and sampling covariance C(0).

Semivariogram/Covariance modeling is a key step between spatial description and spatial prediction. The main application of geostatistics is the prediction of attribute values at unsampled locations (kriging). The empirical semivariogram and covariance provide information on the spatial autocorrelation of datasets. However, they do not provide information for all possible directions and distances.

The kriging interpolator is a least square linear regression algorithms that account for data related solely to the continuous attribute being estimated. It provides optimal and unbiased predictions of the regionalized variable, basing on sampling data and on the structural analysis of the data obtained by the semivariogram.

The basic linear regression estimator is defined as:

$$Z^*(x) - m(x) = \sum_{i=1}^{N} \lambda_i(x)[Z(x_i) - m(x_i)]$$

where $\lambda_i(x)$ is the weight assigned to datum $z(x_i)$; $m(x)$ and $m(x_i)$are the expected values; N is the number of data next to the position x to be estimated.

All flavours of kriging share the same objective of minimizing the estimation or error variance $\sigma^2_E(x)$ under the constraint of unbiasedness of the estimator; that is:

Min! $\quad \sigma^2{}_E(x) = Var[Z^*(x) - Z(x)]$

under $\quad E[Z^*(x) - Z(x)] = 0$

The kriging estimator Z*(x) varies depending on the pattern used to model the random function Z(x).

Usually, a random function can be modelled as:

$$Z(x)= R(x) + m(x)$$

where $R(x)$ is a residual component and $m(x)$ is a trend component.

$R(x)$ is modelled such as

$$E[R(x)] = 0;$$

$$Cov[R(x), R(x+h)]=C_R(h)$$

Thus the expected value of Z at location x is the value of the trend component at that location:

$$E[Z(x)] = m(x)$$

Three main kriging typologies exist:

<u>Simple kriging</u>. It considers the mean m(x) as known and constant within the study area. Thus the estimator is:

$$z^*(x) = \sum_{i=1}^{N} \lambda_i [z(x_i) - m] + m$$

Ordinary kriging. The mean value m(x) is supposed to locally vary and to be stationary within local neighbourhood of the study area:

$$z^*(x) = \sum_{i=1}^{N} \lambda_i \left[z(x_i) - m(x) \right] + m(x)$$

$$z^*(x) = \sum_{i=1}^{N} \lambda_i z(x_i) + \left(1 - \sum_{i=1}^{N} \lambda_i \right) m(x)$$

The weights λ_i have to be determined under the following constraints:

$$E\left[Z^*(x) - Z(x) \right] = 0 \qquad \text{thus} \quad \Sigma\lambda_i=1$$

minimization of $\sigma^2{}_E(x) = Var\left[Z^*(x) - Z(x) \right]$

Thus the estimator becomes:

$$z^*(x) = \sum_{i=1}^{N} \lambda_i z(x_i)$$

Universal kriging. The mean value is considered unknown and variable within the entire study area. The trend component is modelled as linear combination of $f_k(x)$ functions:

$$m(x) = \sum_{k=0}^{K} a_k f_k(x)$$

ak are the coefficients constant but unknown. By convention $f_0(x)=1$, thus the estimator is equivalent to the case of an ordinary kriging.

The general propriety of a kriging estimator are the following:

- The obtained solution is unique;
- The kriging variance is the smallest obtainable;
- The estimator is unbiased;
- The estimator is an exact interpolator
- The kriging is based on the structural analysis of data
- The precision of the prediction can be described by the kriging variance that is determined within all the estimation domain. Hence, it is possible to obtain error maps of the prediction.

3.3. GLEAMS

GLEAMS (Groundwater Loading Effects of Agricultural Management Systems) was developed to simulate edge-of-field and bottom of root zone loadings of water, sediment, pesticides and plant nutrients from the complex climate soil management interactions. Its beginning was in 1984 and it has been updated and evaluated in different climatic region of the world during all this period.

GLEAMS is a continuous simulation, field scale model, which was developed as an extension of the Chemicals, Runoff and Erosion from Agricultural Management Systems (CREAMS) model. GLEAMS assumes that a field has homogeneous land use, soils, and precipitation. It consists of four major components: hydrology, erosion/sediment yield, pesticide transport, and nutrients. GLEAMS was developed to evaluate the impact of management practices on potential pesticide and nutrient leaching within, through, and below the root zone. It also estimates surface runoff and sediment losses from the field. GLEAMS was not developed as an absolute predictor of pollutant loading. It is a tool for comparative analysis of complex pesticide chemistry, soil properties and climate. GLEAMS can be used to assess the effect of farm level management decisions on water quality.

GLEAMS can provide estimates of the impact management systems, such as planting period, cropping systems, irrigation scheduling, and tillage operations, on the potential for chemical movement. Application rates, methods, and timing can be modified to account for these systems and to reduce the possibility of root zone leaching. The model also accounts for varying soils and weather in determining leaching potential.

GLEAMS can be useful in long-term simulations for pesticide screening of soil/management. The model tracks movement of pesticides with percolated water, runoff, and sediment. Upward movement of pesticides and plant uptake are simulated with evaporation and transpiration. Degradation into metabolites is also simulated for compounds that have potentially toxic by-products. Erosion in overland flow areas is estimated using a modified Universal Soil Loss Equation.

GLEAMS Version 3.0 is written mainly in Fortran 77 with some recent modifications in Fortran 99. It can be operated in batch mode which allows specifying data, parameter, and output file names in the batch file rather than entering the filenames as prompted in interactive mode. The batch file named TEST is provided for user information. This file was used to execute GLEAMS30 with the sample data and parameters and generate the sample output files. The order of the files as shown is essential. If only the hydrology and erosion components are to be run, then the pesticide and nutrient parameter files and output files should not be included in the batch file. Likewise, if hydrology, erosion and pesticides are run, the nutrient parameter and output files should not be listed. If selected variables are not specified (in the hydrology parameter file), the selected output file can be eliminated. Inclusion of a selected output file name in the batch file when variables are not selected will not result in a run-time error.

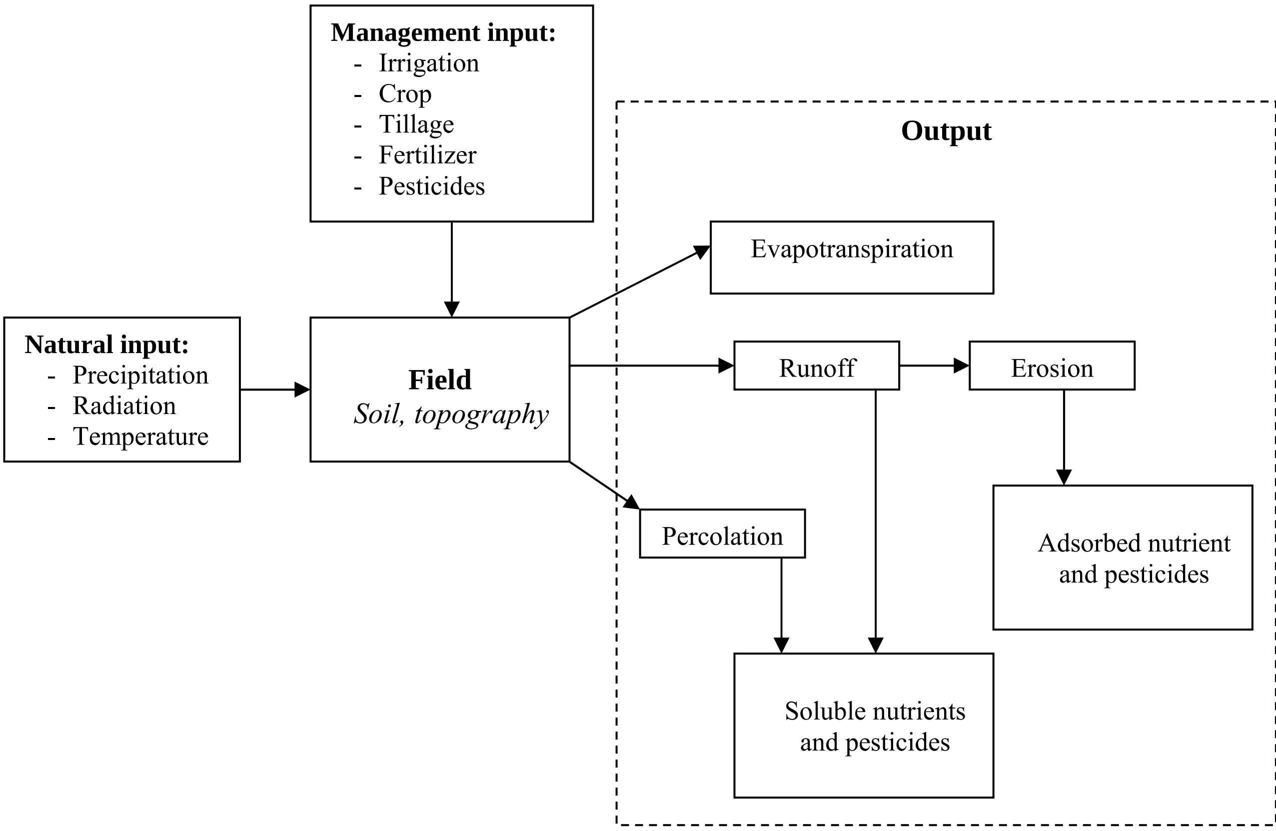


Fig.3. 3 Scheme of GLEAMS function

3.4.Samples analysis

During the last decades the interest in environmental issues and especially in the risk of pollution by high nitrate concentration in water has grown. One of the reason why there was such a develop in research was the risk for human health connected to high nitrate concentration.

The increase of nitrate concentration in water was mainly attributed to intensive livestock activities.

Thus different legislations were developed: the European Nitrate Directive 91/676 provided guide lines to determine the effective or potential pollution in superficial water and groundwater; moreover it gave information to identify nitrate vulnerable zones. In particular, the vulnerable zones in Campania were defined through the regional rule 700/2003.

Solutions for reducing gaseous emissions from storing tanks have been developed since the beginning of ’90. Covers can be used to reduce odour and ammonia emissions from manure stores; indeed, in some arts of Europe, they are required by law. Different materials can be used; the cheapest solution is a natural cover consisting of a floating top layer of fibrous material. This occurs naturally with cattle slurry due to the tendency of some of the suspended matter to float thus forming a crust. Layers can also be created with foam glass or foam clay particles; around 10-15 cm is needed. This solution is quite cheap but they can be destroyed by mixing. As an alternative cover, floating plastics sheets. More expensive solutions include plastic roofs or tent construction. They have the advantage of avoiding wind contact on the liquid surface, thus reducing emissions.

The starting point, by which successive management strategies could be developed to reduce the pollution and, more generally, the negative impacts of manure handling, is manure content characterization. Manure is the output of animal digestive activities, thus nutrients content is strongly influenced by metabolism. Indeed nitrogen, phosphorous and potassium quantity depends on the food conversion ratio. Nowadays there is a drive towards use of feedstuffs that are appropriate to the needs of the animal and which do not represent oversupply of nutrients.

Focusing on the Campania region, the most diffuse and economically important livestock activities are connected to the river buffalo breeding.

Buffalo species is characterized by longer rumination time and longer retention time in rumen, compared to the bovine species (Campanile et al., 1997). Furthermore, buffaloes, generally, are more long live and stronger to diseases; thus the number of therapeutic treatments is low and consequently the traces of elements, that could be dangerous for the agronomic use, are few. The tendency in comparing nutrients content in buffalo manure to nutrients content in bovine manure seems inadequate. This affirmation can be verified only by lab analysis aimed to characterize manure content. Other factors such as livestock management and typology, farm structure, manure management in general (storing, transport etc) can influence manure content.

A study was conducted in Campania region to characterized, by lab analysis, buffalo manure. Before sampling, the storing tank was mixed during 2 hours thus no stratification phenomena and, consequently, no local different nitrogen or phosphorous concentrations occurred.

The tank was virtually divided in 4 sections; the sampling sites were identified avoiding the zones characterized by foam or floating matter.

Sampling was done 40 cm far away from the tank's board by using Coliwasa device. The elementary liquid samples were taken in each of the tank's sector and they were shifted in a 2000ml bottle characterized by a rectangular section, an under plug and a large bottleneck. Thus the global sample was obtained. The rectangular section allows to optimize the loading capacity during the storing phase. To let the gases expand, the bottles were filled until 4cm from the top.

Once mixed, the global sample was divided in 3 rectangular 1000ml bottles with under-plug and large bottle neck. The bottles were filled until 6.5 cm from the top. Thus the subsamples were obtained.

Elementary solid samples were collected and mixed by a shovel. They were sampled only from the storing lanes far away from the litter, from watering troughs, from the feeding sites. The set of the elementary samples constituted the global sample.

The samples' storing period depends on the storing temperature and the substances to detect. Thus at 4°C it is possible to store samples analyzed to determine Dry Matter and Total Kjendal Nitrogen for 7 days and samples analyzed to determine Water-Extractable Phosphorus total (WEPt) for 6 months. After the analysis phase, the samples were stored in freezer at -20°C.

Dry Matter (DM) was determined by using a stove, following the procedures described by Wolf et al. (1997); Total Kjeldahl Nitrogen content (TKN) was determined by following the procedures proposed by Kane (1998), AOAC Official methods n° 978.02. This method is constituted by an acid digestion with catalysts and a next acid distillation process followed by titration. The phosphorous (WEPt) content was quantified by the method reported by Kleinman et al. (2007): samples extraction and dilution; mixing (180-200 epm) for 60 minutes and centrifugation(3.000 rpm) for 10 minutes; finally the method provides for filtration (No. 40 Whatman) and acidification followed by colour test determination and a further PO4 content determination.

The nitrogen and phosphorus content was respectively determined by a Kjeltec device and a spectrophotometer.

The sample analysis provides information about manure content for each farm. Thus changes and improvements in manure and farm management could be promoted.

4. Landscape management

4.1. Rural landscape

The rural landscapes are usual heterogeneous. They consist of a mosaic of agricultural fields, linear structures such as hedges, ditches, watercourses, roads, and non-arable land such as forest, wetlands. In many situation the fields are sources of diffuse pollutants, the other landscape entities can act either as sources, sinks or transfer media, or will simply help to dilute the concentrations of pollutant. When pollutant retention is observed, these structure are often called "buffer zones", and have become an important subject of research and a management tool for reducing pollution.

Experimental studies at the field scale and agriculture practice surveys can provide estimates of pollutant emissions by different cropping or farming systems. Monitoring of water courses can provide estimate of pollutant losses by catchments. The relationship between these two series of estimates is often very complex and variable, partly due to the effect of the different landscape structures, but also due to transfer time and storage in soil, groundwater or vadose zone (Basset-Mens et al., 2005). It is however very important to be able to understand and to predict this relationship, especially to design the most cost-efficient remediation measures, both in term of agricultural practice optimisation and landscape management. This task is fraught with difficulties because experimental studies of specific landscape structures are often difficult to extrapolate (Dillaha and Inamdar, 1997). This is due, first, to the diversity of the structures considered, and second, to the numerous scale-dependent factors at work, such their spatial distribution, their density, their connectivity, etc.

4.2.Landscape structures

It is widely recognized that landscape structures (i.e. non cultivated zone such as hedges, wetlands, embankments) play a main role in influencing the landscape, and more generally, the environment behaviour and processes.

Wetlands have important ecological roles: they represent habitat for different animals and vegetables and mainly they are the interfaces between land and water. They play an important function as buffer against nutrient loads and pollutants that come from land activities. Managing wetlands is a challenge to well balance human activities in a sustainable way.

The interaction between farm activities and wetlands is largely studied (Thenail and Baudry, 2005). It happens that wetlands are located within farm. In this situations the ecological role of wetlands is linked to farm. Specifically agriculture could have a dual influence: both a degradation of the environment and a factor of maintenance of natural resources (soil, water, landscape features, habitats). In this view it is possible to link wetlands features with those of agriculture considering mainly land potentialities and socio-economic conditions or it is possible to deal with the analysis of wetlands role considering their role in technical farm management. Obviously technical functions linked to land use comprehend not only production but also land maintenance and development, so that technical management choices entail farm organization, socio-economic aspects and biophysical systems in farms.

There is a dual interaction between land use and farm: land use is organized in farms following the farmers aims but land use contributes to landscape structuring. Studies conducted by Thenail and Baudry in 2005 suggested that wetlands could be landscape units that could be handled by specific management plans. Due to their interaction with farm management it is possible to consider wetlands as farm management units.

Two main strategies are usually applied to manage wetlands in rural landscape: if the territory is no more used for agricultural activities it can be managed separately; otherwise it is possible to provide the farmer with guidelines to improve management at the field scale. In the first case wetlands are considered as landscape units to manage as not correlated to agricultural production; in the second they are considered independent territories to manage independently from one another and from the overall farm technical management system.

If wetlands are considered as farm units their management is as crucial as the agricultural fields management (Thenail and Baudry, 2005).

The notion of technical management system applied to land use has been developed in agronomy. It is defined as a set of goal-oriented technical means and operations, engaging technical functions, coordinated by a farmer (a manager) according to factors and decision rules. The technical functions associated with land use correspond not only to production elaboration but also to land maintenance and development. Thus the way used to correlate the technical functions with the particular land use of a farm wetland highlights the degree of integration between farm wetland and farm management.

There is a correlation between technical and ecological functions, even is it is not linear. Generally, the level of technical functions decreases with low land use intensity and high ratio of permanent vegetation. The interaction between technical and ecological functions has to be optimize. Generally it would be possible to have high level of technical functions compatible with moderate land uses.

The decrease of land abandonment on farm riparian land would require allocating technical functions back to farm riparian land. In each case a reorganization of the farm is required. Obviously the degree of farm reorganization depend on farming conditions. For instance, management actions are required to reduce land use intensity on farm wetlands when their surface is considerable and when they are near to the farmstead. The correlation between ecological and technical functions in farm wetlands is a key element to develop rules and management guide lines.

Coherently if wetlands are considered as landscape units ecological functions become meaningful at landscape scale. A specific risk associated with advocating management practices on riparian land might be an over-intensification on the rest of the farm territories, due to the transfer of land use and related high-level technical functions.

Managing separately wetlands from farm area could be a way to simplify a the entire farm context. Another point is that wetland use and management practices might become homogeneous if they are entrusted to a fewer number of actors external to farms. Homogeneity might also increase if land use and management are polarized in the landscape, wetland tending to abandonment and other land being subjected to intensification. In both cases, the consequences of a decrease in land use and management diversity of ecological functions at landscape scales need to be evaluated. Habitat homogenization due to a farm homogenization is a central issue to improve a multifactor analysis. The relation between farm diversity and biodiversity, and generally all the different ecological functions, has to be studied at landscape and wetland scale.

There is also the possibility that farms with a specific dimension polarize in zones so that the land use mosaic shows patterns at watershed and wetland scale. That is the reason why different technical and ecological functions combinations exist depending also on the farm’s type and dimension and on the land use mosaic.

Considering wetlands as farm units is a way to pursue the compatibility between technical and ecological functions in these areas.

The main aim of European policies is how to assess agricultural zones from a multifunctional point of view basing on farm diversity and sustainability. Including farms in the management of landscape units is a common shared target. The starting point for agricultural landscape and land use planning is the definition of the landscape units. Often landscape units are defined basing on their ecological features.

A final consideration is the importance to define a landscape unit taking into account the ecological role and the mechanisms that determine it. In this view agronomy is the main discipline that deals with agricultural production considering natural resources and developing methods for analyzing systems for technical land use management. (Thenail and Baudry, 2005).

The potential for nutrient retention of wetland zones is recognized although their variable efficiency especially if used to improve water quality. The efficiency depends strongly on the hydrological setting. The optimal conditions are permeable soils underlain by an impervious substrate at 1-4 m (Hill, 1996), which favour shallow subsurface flow, and a lot of situations depart from this optimum.

The retention processes involved for phosphorus and nitrogen are not entirely compatible. Nitrate will be denitrified in saturated, anoxic conditions, but these conditions can, at the same time, promote the release of soluble mineral and organic P. Saturated conditions will also impede infiltration and support overland flow, which may enhance transport of particulate P. Thus, to assure an efficient removal of both elements, a wetland should include both saturated, anoxic zones and unsaturated, aerated zones. Nevertheless in some cases, nitrogen retention occurs only under a narrow range of hydrological conditions. The maximum delivery of nitrate to the stream occurs generally during high winter flows when the retention is minimum.

The nutrient retention function may also be incompatible with other environmental functions of the wetlands, especially biodiversity: high and uniform levels of nutrients tend to decrease the number of plant species in a wetland, and the regular export of biomass recommended to prevent phosphorus saturation may disturb animal habitats.

One of the main issue that depends on the use of wetlands as denitrify areas is the likely increase of the emission of greenhouse gases, especially CH_4 and N_2O. through a regular supply of NO_3 the CH_4 emission of should remain low keeping the redox potential relatively high. So a good management in buffer zone could ensure controlled methane emissions. On the other side high NO_3 concentrations, low pH and cold temperatures entail the increase of the proportion of N_2O emission in denitrification and these conditions can be met in many riparian zones. Saturated soils promote more likely complete denitrification. The different values of the ratio reported in the litterature can rank from almost 0 to as high as 60 %. This can affect considerably the environmental evaluation of buffer zones.

The IPCC guidelines for N_2O emissions suggest a coefficient of 3% for the indirect emissions due to NO_3 leaching. Assuming that one third of the leached NO_3 flux is denitrified in riparian zones and that 60% of the denitrified NO_3 is emitted as N_2O, this coefficient could reach 20%.

Vegetated filter strips and hedges are landscape structures that can be located anywhere in the catchment, and usually at the limits of the fields. In many cases, they are traditional features of past rural landscape, to delimit property or provide barriers to wind or sun. They were very diverse, comprising earth or stone embankments, and often region-specific trees or shrubs. They have been often removed to increase the field size, facilitate agriculture mechanization and suppress maintenance operations. The new generation of vegetated field limits are often grass strips of defined width, without embankment, located on the downslope limit of the fields only.

Filter strips are mainly designed to reduce overland flow and associated sediments and particle-bound pollutants. Thus, they are best suited for particulate phosphorus, ammonium and organic N. The reduction of pollutant flux is due to higher infiltration rate and reduced flow velocity because of a dense and permanent vegetation cover.

Considering the hedges on embankment, this effect is even more important if the embankment is continuous on the downslope side of the field. The maintenance of these systems has to deal with the creation of preferential pathways within the zone, and the coarse sedimentation that occurs in the first few meters of the zone, affecting vegetation growth. Nevertheless, regular tillage and re-seeding could limit intensive erosive processes. Grazing of grass filter strips is not recommended because it tends to decrease soil infiltrability. As for wetlands, water saturation of the soil obstructs infiltration, therefore it is not recommended to install filter strips in area subjected to frequent waterlogging. This means that locating the filter strips along the watercourses is not necessarily the best option. The location of the filter strips must be carefully chosen according to slopes, drainage areas, slope length, etc. Ideally, GIS-based applications should be used to optimised the distribution of filter strips in the landscape.

The effect of filter strips on dissolved pollutant (nitrate and dissolved phosphorus) is quite limited. The reduction of dissolved pollutant fluxes by filter strips concerns rather the fraction transported by overland flow, and it is mainly due to infiltration. The retention of dissolved pollutant in infiltrated water or subsurface flow can be due to three processes: vegetation uptake (for N and P), adsorption (for P and NH_4), and denitrification. These processes are likely to be less important in filter strips than in the fields, because of the higher infiltrated volumes, enhancing leaching and saturation of the adsorption sites within the soil.

A important exception to this is the case of tree hedges located at the bottom of the slope. These hedges were frequent because they separated the valley bottom from the cultivated slope. This type of hedge play an important role on the subsurface flow of pollutant for different reasons: the water uptake by the trees during the growing season, more intense than the neighbouring crops, creates a dry bulb that tends to slow down the subsurface flow, especially in autumn, enhancing nitrate retention by the riparian zone downslope.

The deep tree roots can reach the groundwater (directly or by capillary rise) and take up water and nutrient from it.

The supply of available carbon by the tree roots and litter fall may enhance bio-transformations in the soil. This results in a significant decrease of nitrate flux where the hedge is conserved.

The effect of landscape structures on nutrient fluxes is still a research topic, although the main processes are relatively well known. The main difficulty is to upscale plot results, which are site specific, to a higher scale (for example small river catchments). Process-oriented distributed models are probably the best tools to do that, although parameter identification problems are often acute with these models.

4.3. The buffer zone concept

The field emissions of pollutant can usually be determined by losses via vertical drainage, lateral subsurface or surface flow, or gaseous fluxes, outside the soil-crop system. These emission can be measured experimentally, estimated by agricultural mass balances and regional references on the rate of soil transformation. Considering the water pollutant, different processes can take place between the emission from the fields and the fluxes at the outlet of a catchment (Basset-Mens et al., 2005; Ruiz et al., 2002) : the pollutant can be stored, without transformation, in the soil, in groundwater and in different landscape elements (hedges, filter strips, wetlands…); it can be stored under another form, for example as organic matter after assimilation by plants or micro-organisms; it can be emitted in gaseous form out of the fields (eg, through denitrification in riparian areas).

Taking into account the export coefficient approach the fluxes at the outlet of a catchment are equal to the sum of the pollutant export from each field; usually the export coefficients are determined experimentally. Whereas basing on the mass balance or directly on measurement, there is often a difference between the sum of field losses and the sum of fluxes at the outlet.

Using a environmental evaluation method such as LCA (Life Cycle Analysis), the ratio between fluxes from environmental compartment and sum of pollutant emissions from different sources is often less than 1 and is called fate factor. For example, for nitrate (soluble pollutant) it ranges between 0.4-0.9, while for the insoluble phosphorous the fate factor is more or less 0.05.

The main driving force for the losses of water pollutant is the climate: nitrate losses, high rainfall rate. Thus, for a same quantity of excess of fertilizers determined by a mass balance at the field, the annual fluxes of pollutant might vary from year to year depending on climate conditions. Consequently the fate factor should be calculated considering a long period of time to account for climate variations. On the other hand if the filed is not in a steady state with crop practices, the storage-destorage processes are able to influence the fate factor. Thus the fate factor strongly depends on the accumulation or realising of nitrate by the groundwater (Fig.4. 1)

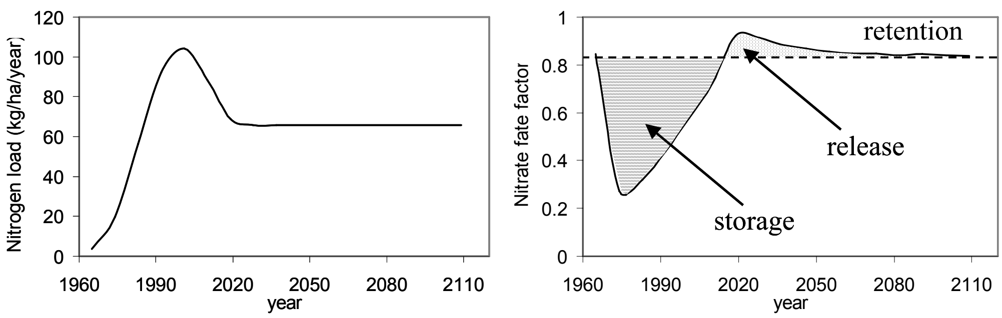


Fig.4. 1 Simulated time series of nitrogen input (left) and nitrogen fate factor (output/input, right) for a high rainfall, high nitrogen input catchment. (Basset-Mens et al., 2005)

On the basis of these considerations, it is quite clear the difficulty to define and assess a “buffering” effect of a landscape structure considering only the comparison between field emission of pollutants and fluxes at the outlet. Studies conducted by Viaud et al. (2004) show a review of the concept of buffer zone. In general, a buffer zone is defined as is a landscape structure set between the source of water pollutant and the receiving water body that may provide a physical and/or biogeochemical barrier against pollution inputs.

This definition encompasses a large variety of processes and of effects (Fig.4. 2).Therefore, the variables considered can be very different one to each (fluxes, concentration..) other as well as the methods used to measured them. Consequently the buffering capacity is strongly linked to the limited zone where it is and so it very difficult to derive from them the overall buffering capacity of a given catchment with a landscape mosaic comprising buffer zones and agricultural fields.

The major interest of these studies is that they provide valuable information on the factors affecting the processes responsible for the pollutant retention in the buffer zones. Different factors influence the pollutant retention processes in the buffer zones: physical, chemical and biological, and depend on the type of pollutant, namely, soluble or particle-bound. Surely, the uniformity of the pollutant input flow is a good condition for the function of a buffer zone.

In the case of insoluble phosphorus and organic nitrogen, the two major processes that affect the retention capacity of the buffer zone are the reduction of overland flow (by promoting infiltration) and the reduction of sediment load in overland flow water (by reducing flow velocity and filtering due to vegetation cover). The sustainability of the buffer zone is influenced by physical adsorption, biogeochemical transformations and maintenance operations such as tillage, re-seeding, mowing.

Considering soluble pollutant (nitrates, soluble phosphorous) biological processes are the main influenced factor for the buffering effect:

plant uptake, microbial assimilation, denitrification. Further parameters are involved in these processes: temperature, pH, redox potential, water residence time, water table depth.

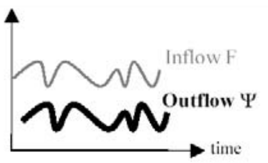
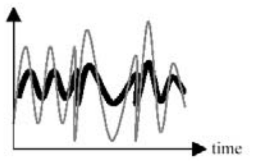
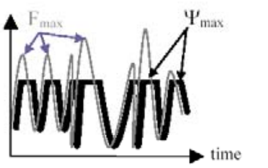
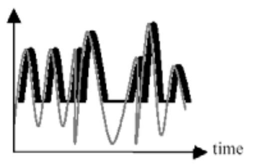
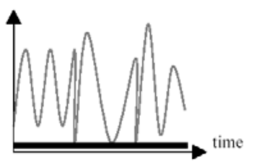
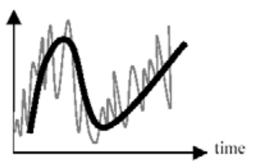
Buffer effect formalism	Input and output patterns	Examples
a) Decrease in signal mean level $B_{decrease} = \frac{F}{\Psi}$ B > 1: sink		Decrease in nitrate concentration through wetlands
b) Decrease in variation range $B_{ecol} = \frac{\Delta F}{\Delta \Psi}$ B > 1: sink		Decrease in pH variation with a chemical buffer
c) Decrease in maximum values $B_{max} = \frac{F_{max}}{\Psi_{max}}$ B > 1: sink		Flood control by reservoirs
Increase in minimum values $B_{min} = \frac{F_{min}}{\Psi_{min}}$ B > 1: source		Low-flow control by reservoirs
d) Physical barrier against signal spreading $B_{decrease} = \frac{F}{\Psi}$ with $\Psi = 0$ and $B = \infty$		Soil particle storage upslope of hedgerows
e) Decrease in variation frequency $B_{filter} = \frac{F\ correlation\ length}{\Psi\ correlation\ length}$ B > 1: sink		Temperature control in river by riparian vegetation

Fig.4. 2 classification of the effects of buffer zones on input (F) and output (Ψ) signals (Viaud et al., 2005).

4.4.Management issues

It is very difficult to assess the nutrient retention by using budget calculations at the catchment scale, due to storage processes and climate variations, or by extrapolating local observations. Another possible method is the statistical analysis of geographical data, including physiography, land use, agriculture and water quality information. The main limitation of this method is the amount and quality of data available. This is in part due to interferences and dependencies between variables, that introduce scatter or ambiguity in the results, e.g., the location and extent of wetlands in a small region depends on the hillslope shape, on the bedrock permeability and on the precipitation which, in turn, affect the conditions of pollutant transfer. Moreover, the variables most adequate to describe the system are not always easy to define. For example, it is often said that the continuity of riparian zones and the length of contact between them and the potential source of pollutants are more important than their extension, but these are not variables easily derived from GIS for large areas.

Wetlands management for diffuse pollution control is an interesting complement to good agricultural practices. First to plant wetland zone in a given system, the two main questions are often about, first, the delimitation of the usable zones and second, the maintenance. Different techniques can be used, including remote sensing, vegetation or soil surveys, in situ measurements, varying in cost and complexity. A methodology has been designed to structure these delimitation problem, based on the distinction between potential, existing and efficient wetlands (Merot et al, 2006). An efficient wetland is identified by considering a specific function, for example its buffer function for water quality or its influence on the hydrological regulation. Through the analysis of remote sensing data or field observation it is possible to observe physical, biological and chemical soil features that let the scientist to define an existing wetland. A potential wetland can be identified by analyzing digital elevation model, maps, climatic data. It is an area that can be potentially a wetland, but it is not an existing wetland.

Although it is clear that the sustainability of a buffer zone depend on the way it is maintained, the problem of the technical and economical feasibility of this maintenance is often very acute in case of wetlands, given the poor value of the vegetation and the reduced accessibility for machines.

A general observation provides that as nitrate is easily transferred in the landscape, the buffer zones really efficient in nitrate retention will be those combining flow velocity decrease and high biological activity, such as wetlands, ponds, and tree hedges. In contrast, phosphorus can be trapped even during relatively rapid transfer via abiotic reactions. A smaller proportion of this retention is really long-lived, and buffer saturation is a greater risk than for nitrate.

The consequence of the great diversity and complexity of landscape buffering processes is that standard regulations on design, location and maintenance of buffer zones are not the best way to optimise them.

The main interest of buffer zones as a complementary solution for diffuse pollution is that the effect on water quality is almost immediate, whereas changes in agricultural practices need time to be applied and time to be effective. Another interest is that it does not affect noticeably the current agriculture economics. However, the complete environmental evaluation of this “end-of-the pipe” solution still need to be done in most cases, taking into account the effect on the other functions of the buffer zones (recreation, biodiversity, hydrology…), the uncertainties on their present and future efficiency, and the risk of trade off with pollution of other environmental compartments (soils and atmosphere).

In Brittany the so called Territ’eau was developed in order to have a diagnostic tool and a reference model to manage the rural landscape, comprising both the non cultivated zones - such as hedges, wetlands etc.- and the cultivated ones. The domain of action is the territory management at collective scale. The project Territ’eau aims to update and share the knowledge about water and associated elements movements in a watershed and about the functional role of landscape structures. Territ’eau is composed by a set of instruments that provide references to connect the agricultural activities to rural landscape management in the aim to improve and preserve superficial water quality. The functional, global and flexible aspect makes Territ'eau a very successful project. Moreover it provides diagnostic elements and suggestions. It is characterized by a modular approach that fits in different situations (pedoclimatic, agricultural..). Agriculture and rural landscape gain, through Territ'eau, a way to make a hierarchy of the pollution risk and a means to organize actions to deal with issues in a given context.

5. Nitrate leakage in a high buffalo district (Caserta province)

5.1. Introduction

Caserta province and especially the medium and lower valley of Volturno Garigliano is considered one of the most relevant livestock area in Italy. As a matter of fact the 2000's census (Istat V, agriculture census) shows a population of 188.000 buffalos and 105.000 cows on a SAT (Total Agronomic Surface) of 107.400 ha, representing all the Caserta district. The breeding of buffalo, typical of the region, had a relevant increase in the last years, boosted by the policy of the milk quota too (the buffalo is not object of the restriction of the milk quotas). This expansion is overlapped to a pre-existent intensive agriculture, in a context of elevate fragmentation of agricultural holdings and poor culture of the methods for environment management. The livestock concentration is very high especially in 21 municipalities situated in the low Volturno basin. In this confined territory (41.000 ha) there are among 200.000 head of cattle, and so it's possible to find density higher than 10 LU/ha SAT (LU= livestock unit defined as a dairy cow of 500 kg body weight).

During the last few decades a significant decrease of water quality has been observed in different parts of the world. Since it has occurred in spite of a considerable increase in depuration of point pollution sources, it has been attributed to an increased contribution of nonpoint pollution sources (NPS), which is, in turn, strongly dependent on land use. The international literature reports many cases with reference to such problems (see for example: Sims and others 1999; Arheimer and others 2004; Haycock and Muscutt, 1995). There is general agreement on considering agriculture the main nonpoint source of nutrients to water (Sims and others 1999; Sharpley and others 1999; Rekolainen and others 1999). According to Community and national rules

(91/676 CEE and L.152/99 It e 258/00It), nitrate leakage is considered a pollution indicator, related to breeding activities. In Netherland (Ondersteijn et al 2003) was evaluated that farm management rather than farm structure can reduce nutrient surpluses more effectively. Furthermore, improving management will also improve financial results.

In this context (Campania region), contrary to in other systems, the new UE instruments (multifunctional agriculture), don't easily succeed in resolving the problems of this compartment with its tools (uncoupling, conditionality, best agricultural practice, etc.), because the buffalo compartment partially puts aside from the public financings.

Aim of this research is to identify the critical aspects of this complex system and evaluate the opportunity to introduce treatment technologies, even if temporary and even if considerable as “end of the pipe technologies”, not ecologically correct in other contexts.

5.2.Materials and methods

In 1999-2000, to quantify the damage, sampling on over than 150 wells, in Volturno Garigliano basin, were collected and nitrate concentration was measured with a portable reflectometer (Boccia et al 2003). The locations of the wells in which samples were collected, were surveyed through a Garmin GPS. To evaluate the results of European Community and Campania government politics (DGR 610, 700, 2382 - 2003) the tests of nitrate concentrations were repeated in the 2004-2005 on 40 wells in the Caserta province.

It was necessary to sample data in suspended and superficial water-breading strata because the underground is interested by a large deep water stream, in which dilutions reduce concentrations.

The typical soil texture of the tested zone, clay-loam, is characterized by a moderated permeability and a good water retention; the pH value is a little bit alkaline and the salt content is great. The organic matter is in high percentage in the soil as the global nitrogen content.

The nitrate concentration tests were carried out in real time with a portable reflectometer Merk and the same samples were analyzed in laboratory with a spectrophotometer (Spectroquant Merk). The difference between the two measuring system was not significant in the aim of the research. Anyway only the laboratory data were considered.

The point concentrations were treated, through kriging techniques (Leone 2004) by Arcmap 9, as variables characterized by intrinsic uncertainty and spatial dependence, in a statistic point of view.

Based on the regional rules the nitrogen enrichments, usable for the typical tested zone cultivars, are: for medic 85 kg/ha; for oats 90 kg/ha; for lilium multiflorum 170 kg/ha; for silo maize 185 kg/ha; for corn maize 235 kg/ha; others culture 100 kg/ha. The total SAT consisted of the 21 critical municipalities is among 41.000 ha and the nitrogen quantity to add through manuring is among 9.000 t/y. The real manure nitrogen quantity to manage is evaluated among 11.500 t/y.

The costs analysis about manure's spreading demonstrates that the cost for spreading is about 1.5 €/t with a distance of 3-5 km (Provolo 2000) but the nitrogen content is only 3 kg/t. So the cost for spreading is the same order or plus, that the cost of chemical fertilization.

It's realistic and reasonable thinking that no more than 5000 t/y of N-manure could be used because in some phenological phases it's better to use synthesis fertilizers. So 6000 t/y of N-manure are not manageable.

5.3.Results and conclusions

The results of the first sampling, executed in 1999, compared with the rules limits (50 mg/l for the "vulnerable" zone and 25 mg/l for the "sensible" zone), give prominence to the existence of a great problem of pollution in the water creek of the studied basin, arising from agricultural management. In the 36% of the samples the nitrate concentration is above 50 mg/l, in 22% of the samples the concentration is between 25 and 50 mg/l and only in 42% of the samples the concentration measured was less then 25 mg/l. Data collected in 2004 and 2005 show that only 7 of 39 monitoring wells have a nitrogen concentration higher than 25 mg/l, and only 4 of these have a concentration higher than 50 mg/l. Thanks to geostatistical analysis (Kriging), the two dataset were processed and the results, with the relative standard error map, are presented in Fig.5. 1.

The kriging prediction standard error maps shows that the kriging was a sufficient realistic prevision of the real nitrate concentration.

At the same time the nitrate concentration and in special mode that measured in the second sampling campaign is unrealistic compared with the breeding activity in act. It's difficult to understand how it was possible to manage a so great breeding activity without exceeding the concentration of 50 mg/l N-NO3 in the wells

Initially the interpretation to this results was done by the help of hydro geologists, explaining the phenomenon through a relevant dilution in the water stream. This was in contrast with the use of synthesis fertilizers at district level (Onorati et al 2003).

It's obvious that the N-manure hasn't an agricultural destiny and so it's possible that the waste are dumped in superficial canals.

In Caserta province the buffalo milk turnover is over than 400 millions €/y and the dairy cows about 200 millions €/y. The cheese factory trade determines turnover among 1 billion €/y.

It's possible to conclude that in the studied region it's necessary to manage 11.500 t/y of N-manure. Among 6000- 7000 t/y of the total nitrogen can be treated through manure plans, also inside the farm structures. The exceeding nitrogen (4.500 – 5.500 t/y) must be managed in a more complex way.

Although the efforts of the regional government, it's our opinion that management of livestock waste in Campania region is not enough controlled and is carried on, almost always, without the necessary equipments and structures. It would be appreciated a more precise, complex and integrate environmental planning.

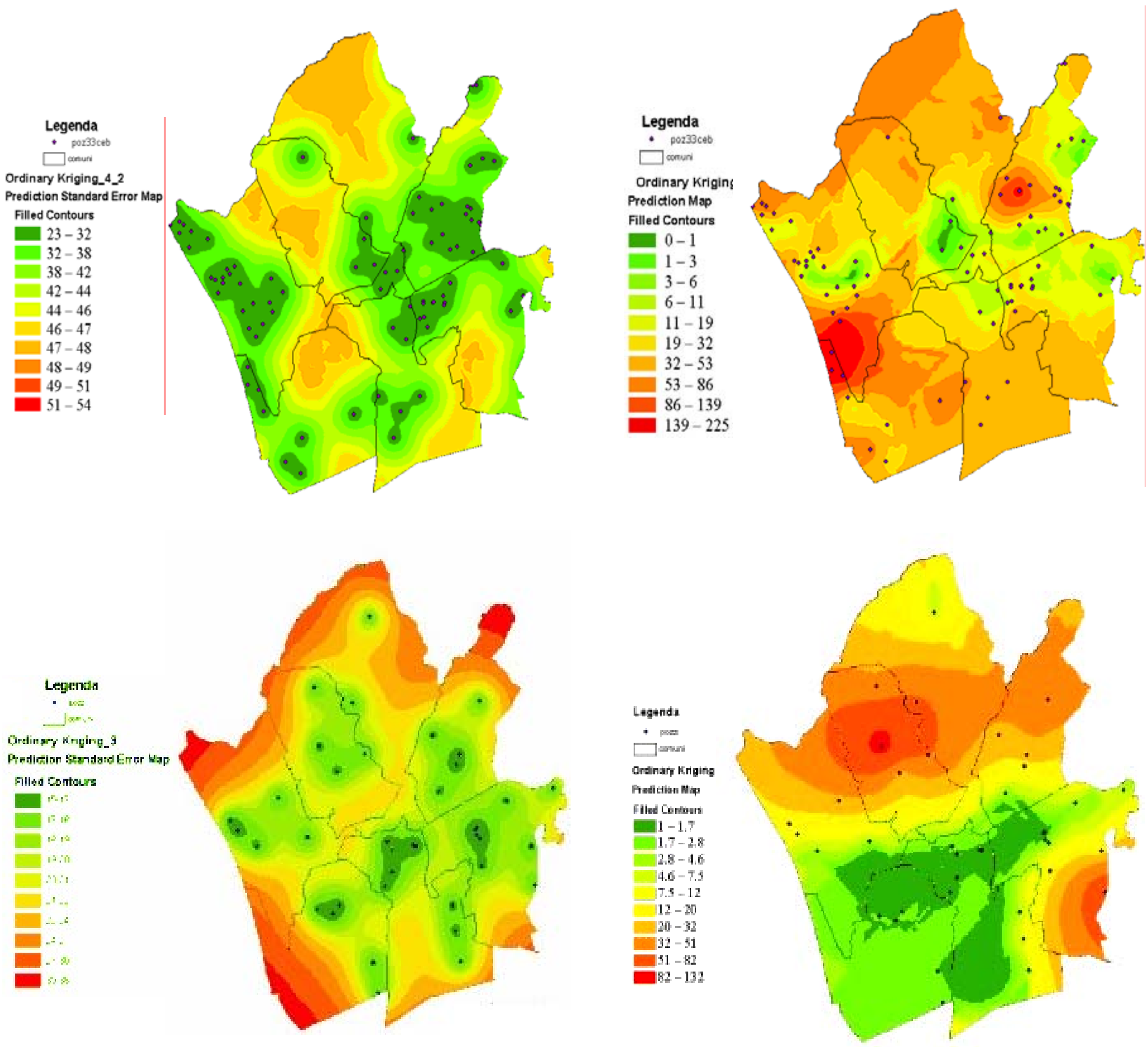


Fig.5. 1 Results of Kriging prediction and their standard error maps

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6. Spatial assessment of animal manure spreading and groundwater nitrate pollution

6.1. Introduction

During the last few decades a significant decrease in water quality has been observed in different parts of the world, despite considerable efforts towards cleaning up point pollution sources. The decrease in water quality has been attributed to an increased contribution of non-point pollution sources (NPS), which are, in turn, strongly dependent on land use. The international literature reports many cases referring to this issue (Arheimer et al., 2004; Viaud at al., 2004; Haycock and Muscutt, 1995; Sims et al., 1994). There is general agreement on considering agriculture the main NPS of nutrients to water (Frost, 1999; Rekolainen et al., 1999; Sharpley et al., 1994; Sims et al., 1999).

Groundwater characterized by high concentration of nitrate is unsuitable for human consumption (Thayalakumaran et al. 2007). Nitrate itself is a compound of low toxicity, but nitrate can be reduced in nitrite, the most toxic form with regard to human health, which can cause infantile methemoglobinemia. In addition, carcinogenic N-nitrosamines can be formed from nitrite (Fine et al., 1977) which is suspected of causing gastric cancer and other malignancies.

Elevated nitrate level in water can also modify the stability and diversity of aquatic ecosystems (Tesoriero et al, 2000). Knowledge and understanding of the processes and conditions that can influence nitrate concentration in water form the base with respect to improving policies and targets aimed at controlling nitrate pollution.

In the European Community and national regulations (EEC 91/676 and Italian Legislative Decrees 152/99 and 258/00), nitrate leakage is considered a pollution indicator which is related, amongst other factors, to livestock rearing activities (Garnier et al., 1999). The European Union (EU) regulation Directive 96/61 EC, Integrated Pollution Prevention and Control (IPPC), requires that an integrated approach to pollution deriving from livestock activities should be implemented. This takes into consideration the possible impacts on the three environmental compartments (water, air and soil) and aims to decrease pollutants at the origin of the production chain.

The Italian Ministry for Agricultural and Forestry Policies has ruled, by means of the Ministerial Decree dated 7th April 2006, that the Regions are required to practice an integrated management of effluents by promoting ways of livestock farming and feed selection aimed at reducing nitrogen content in manure. Some authors (Burton and Turner, 2003; Westerman and Bicudo, 2005) maintain that, in this context, the goal should be to make the ‘‘waste’’ a resource that can be utilized and not just discarded. The large quantities of slurry and manure that are produced every year in areas in which animals are raised, could be an important source of organic matter and nutrients. Recycling this waste via land application could also lead to soil improvement with regard to porosity, structure, and water-holding capacity (Ramos et al, 2006). In order to decide the right balance between reuse and disposal, “best management practices” need to be identified. Moreover, manure management has to involve the identification and evaluation of the criteria necessary for the design and the selection of optimal management systems (Kamarkar et al., 2007; McColl and Aggett, 2007).

Although manure is a valuable source of plant nutrients, it can also be a source of pollution and a threat to the environment unless managed carefully. Studies have revealed that farm management is more effective than farm structure in reducing nutrient surplus (Ondersteijn et al., 2003). Improving management also has the potential to improve financial results (Kuipers and Mandersloot, 1999; Osei et al. 2003). Despite the obvious benefits of optimising nutrient management, practices deemed the best may be unsuitable in some contexts due to social constraints. Therefore, environmental interventions must be flexible and adaptable in the area, where the local traditional farm management practices often have deep roots which may contribute to a strong initial resistance to change.

Income from the sale of milk from dairy cattle and buffalo plays a major role in the Campania regional economy, which is characterised by an undeveloped social fabric. Indeed, the milk income has been estimated at about 600 million €/year to which should be added incomes from the cheese industry. The value of buffalo milk is 1.1 €/l with a production of about 1 900 l/lactation, while the value of cattle milk is 0.4 €/l with a production of 5 000 l/lactation. Hence the income created from dairy production is essential for regional economic improvement.

At present, dairy production from dairy cattle is taxed by the government. This policy of milk quotas has significantly boosted the breeding of buffalo, whose milk is not subject to the tax. This expansion overlaps pre-existing intensive agriculture in a context of high fragmentation of agricultural holdings and a low adherence to environmental management methods.

This study suggests an analytical method for evaluating whether the regulations are being respected, and for evaluating the efficacy of actions undertaken to reduce nitrate concentration in shallow groundwater. Geostatistical tools were used to carry out a spatial analysis of the environmental and the natural variables involved in this issue. The idea was to identify a methodology for the comparison between the nitrate pollution found within the territory and the potential output of pollutant. The lack of correspondence can play the role as a guide line for the next studies aimed at the identification of the real fate of manure.

6.2.Materials and methods

6.2.1. Study site

The province of Caserta, which is part of the Campania region in southern Italy (Fig.6. 1), was chosen as the study area. According to the 5th Italian Agriculture Census in 2000, (Italian National Institute of Statistics (ISTAT)), the middle and lower valleys of the Volturno-Garigliano river system with its 3 141 registered livestock farms belong to the most productive livestock rearing areas in Italy. Focusing on the most economically significant animal species reared in Campania, the 2000 Agricultural Census showed a population of 94 278 river buffalo (*Bubalus Bubalis*) and 52 647 dairy cattle on a Total Agronomic Surface (SAT) of 107 400 ha, representing the entire Caserta district. These values are interesting when compared with the regional buffalo population of 130 732 head, and dairy cattle population of 212 267 head. The number of dairy cattle and river buffalo reared in the study area was assessed, as were the importance of livestock rearing for the local economy and its environmental impact related to nitrate leaching.

The study area’s hydrogeology is characterized by a series of independent, suspended, unconfined aquifers, with poor water circulation; while the deep groundwater has a distinctive flow from east to west (Allocca et al., 2007) (Fig.6. 2).

The soils of the province of Caserta are characterized by clay-loams, whose representative texture is: sand 35%, silt 35%, clay 30%. Their average pH value is 7.7, while they contain 1.5 % organic matter which is related to soil nitrogen forms. Studies have shown the percentage of total nitrogen to be approximately 0.1%.

Fig.6. 1 Location of the Caserta province within Italy

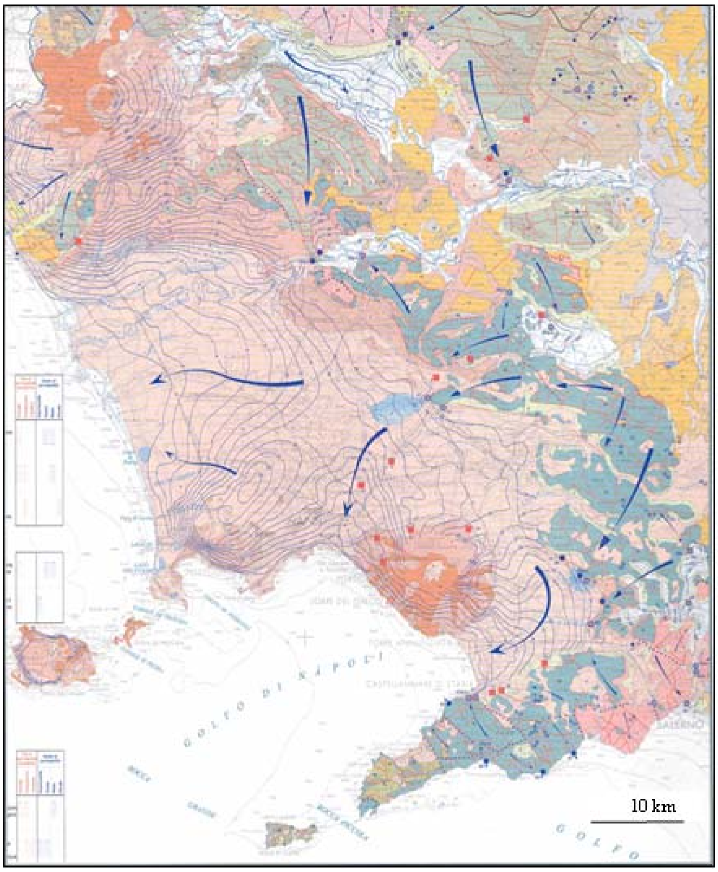

Fig.6. 2 Hydrogeological map of the Campania region extracted from "Carta idrogeologica dell'Italia meridionale" (Allocca et al., 2007).

6.2.2. Nitrogen fertilization

The balance of the average soil nitrogen content includes also the quantities of required to fertilize the crops typical of the study zone. The Campania regional government has developed a “Fertilization Guide” (Ingenito et al., 2003) to estimate nitrogen needs according to soil features and crop type which can be calculated by the following formula:

$$NF = A-B+C+D \qquad (1)$$

where “NF” is nitrogen fertilization, “A” crop needs, “B” benefits of soil fertility, “C” leakage, and “D” immobilization and loss. Crop needs are calculated by multiplying removals and yields (see Table 6. 1) and are related to each specific crop. The benefits of soil fertility are estimated based on the carbon/nitrogen ratio (C/N) for the typical soil. Specifically it is possible to identify two kinds of benefits:

- those from organic matter mineralisation (B1) are 35 kg N/ha and are calculated based on a the C/N ratio between 9 and 12 and an organic matter content equal to 1.5 % ; and
- those from the initial nitrogen content that can assimilated (B2), i.e. about 1% of total nitrogen, are 25 kg N/ha, considering $N_{tot} = 1\ ^{0}/_{00}$.

Hence benefits due to soil fertility are B1+B2=35+25=60 kg N/ha.

Nitrogen leakage (C) is determined by the information in Table 6. 2. The soils typical of the case study area are clay-loam and have moderate drainage, so nitrogen leakage is calculated as 25 kg N/ha, according to these regional rules.

Nitrogen quantities immobilized by biomass by absorption, volatilisation and denitrification are estimated by the equation:

$$D = (B1+B2) * fc \qquad (2)$$

where “fc” is a correction factor (in this case fixed as 0.275, making D= 16.5 kg N/ha.

By means of the above mentioned formula for nitrogen fertilization, nitrogen needs for each selected crop are obtained as follows:

- alfalfa : 40 kg N/ha;
- corn silage: 140 kg N/ha;
- rye grass:125 kg N/ha;
- maize: 190 kg N/ha.
- oats: 45 kg N/ha.

Table 6. 1 Removals and yields of selected study zone’s crops according to regional rules

Crop	**Removals (** kg/10^{-1}t)	**Yields (**10^{-1}t /ha)
Alfalfa	0.30	200
Corn silage	0.20	800
Rye grass	1.60	90
Maize	2.10	100
Oats	1.60	40

Table 6. 2 Amount of nitrogen leakage (kg N/ha)

Drainage \ **Soil texture**	**Sandy**	**Loam**	**Clayey**
Slow	50	40	50
Moderate	40	30	20
Good	50	40	30

6.2.3. Site surveys

To evaluate the impact of livestock activities on the environment, 150 wells were sampled during 1999-2000 in the Volturno-Garigliano basin to determine nitrate concentrations (Boccia et al., 2003).

The study area was analysed by a Geographic Information System (GIS) using Arcview 3.2 ESRI, Redlands CA., USA). GIS has become an increasingly widely used tool to manage, analyse and visualize different datasets related to topography, land use, soil and land cover (Miller et al., 2004; Badri et al., 2002). Thanks to these tools, 2 441 livestock rearing farms were identified by analysing a series of Campania orthophotos dated 1998 (Fig.6. 3). The difference between the farms registered (ISTAT, 2000) and the farms analysed using the orthophotos is probably due to the possible non-correspondence between the farms' registered offices (in Italy the place where an activity is registered does not necessarily coincide with the place where that activity actually takes place) and their geographical location. Based on the quantity of farms identified on orthophotos, municipal ISTAT datasets of buffalo and cattle numbers were allocated as average municipal values to each farm.

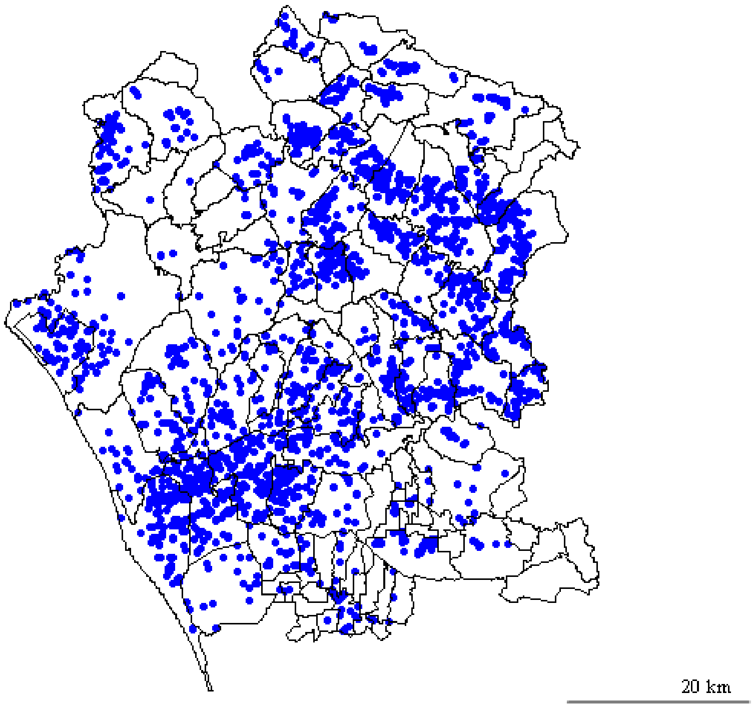

Fig.6. 3 Municipal limits and locations of livestock farms in the Caserta province

Quantities of manure and related nitrate content were calculated following the procedures established by the Italian Ministerial Decree of 7th April 2006, with reference to the Legislative Decree 152/1999. The data obtained were added to the GIS dataset as a specific data layer. Hence a comprehensive assessment of buffalo and cattle spatial locations, in the province of Caserta and, consequently, of the related manure and nitrate quantities produced for each farm location, was obtained.

6.2.4. Spatial analysis of nitrogen production

The spatial distribution of nitrogen quantity within the territory, produced by each farm in a year, was evaluated by a convolution of the farm's spatial distribution with a weighted function characterizing the decrease in significance of the pollutant (nitrogen) presence (in kg/ha) as a function of the increase in distance from the farm. Because we have no information on the way farmers are using manure, we assume that the deposit of manure on field decrease as a function of the distance from the farm location. The weighted function has the following formula in polar coordinates (r, ϑ):

$$\rho(r, \vartheta) = \frac{3}{\pi R^2}\left[1 - \left(\frac{r}{R}\right)^2\right]^2 \tag{3}$$

This function approximates the Gaussian distribution below,

$$\rho(r, \vartheta) = \frac{1}{2\pi\sigma^2} e^{\frac{-r^2}{2\sigma^2}} \tag{4}$$

When $R = 3\sigma$. In these formulas "R" represents the search radius and "σ" the variance of the Gaussian distribution. Hence, through function (3), we assume that the pollutant loses significance for areas outside the search circle (Ter Haar Romeny, 2003).

For the case study we chose a search radius R = 6 000 m because the cost of spreading manure is less than or equal to the doubling of the cost of synthetic nitrogen (0.5 €/kg) within this search radius. These assumptions were made hypothesizing a nitrogen spreading cost of 1.5 €/t considering a distance of 6 km (Provolo, 2000).

6.2.5. Spatial analysis of groundwater nitrate concentration

Due to the intrinsic variability, number and variety of environmental and natural phenomena, the use of classical statistics to study these subjects could prove time-consuming and limiting. Therefore geostatistics tools were chosen to deal with the intrinsic environmental uncertainty. Geostatistical analysis (Goovaerts, 1997) attempts to verify whether or not the variability of a given set of data depends on the location of the measurement point.

Moreover it allows a weighted interpolation of the available data, assuming that the weights are functions of the distance between data.

Samples of nitrate concentration in wells were analysed by means of a semivariogram to model the degree of spatial dependence displaying data semivariance ($\gamma(h)$) related to data distance:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} \left[Z(x_i) - Z(x_i + h) \right]^2 \tag{5}$$

where “$Z(x_i)$“ is a value at a particular location, “h” the measurement distance (the so called “lag distance”) and “$N(h)$” the number of data pairs at distance “h”.

The semivariogram provides information about the continuity and the spatial variability of the process studied. An experimental semivariogram has to be fitted by a semivariogram model. For the case study, an exponential semivariogram model was selected, whose formula is:

$$\gamma(h) = C_0 + C\left(1 - e^{\frac{-h}{a}}\right) \tag{6}$$

where “C_0” is the nugget effect and “($C_0 + C$)” represents the sill and a is the range.

The next elaboration was a prediction of nitrate concentration at those points that were not sampled, by means of kriging interpolation. The existence of a model of spatial dependence allows attribute values to be estimated for unsampled locations (Goovaerts, 1997). Ordinary kriging was used in this study. It entails the hypothesis that the mean may vary in the local neighbourhood within the study area. This assumption is coherent with the features of the study area. The ordinary kriging estimator is described by the equation:

$$z^*(x_0) = \sum_{i=1}^{N} \lambda_i z(x_i) \qquad (7)$$

where "λ_i" are the weights, "$z(x_i)$" the N available data and "$z(x_0)$" the unsampled location where $z^*(x_0)$" is estimated. For the study case, Arcview 3.2 was used to perform this type of interpolation, which was then presented in the form of a map where the attribute spatial distribution over the study area is identified by a graduated colour scale (Fig.6. 4). The next step was to compare the predictions of nitrate pollution in shallow groundwater (Fig.6. 4) with the spatial distribution of the annual nitrogen density produced by each farm (Fig.6. 5). In order to do this, a comparison method was created using Arcview 3.2. A grid of located points, with a mesh of 1 000 meters, was built and the kriging prediction and nitrogen production density values were extracted from the grid. The size of the mesh was chosen to allow a precise analysis of the areas surrounding the farms.

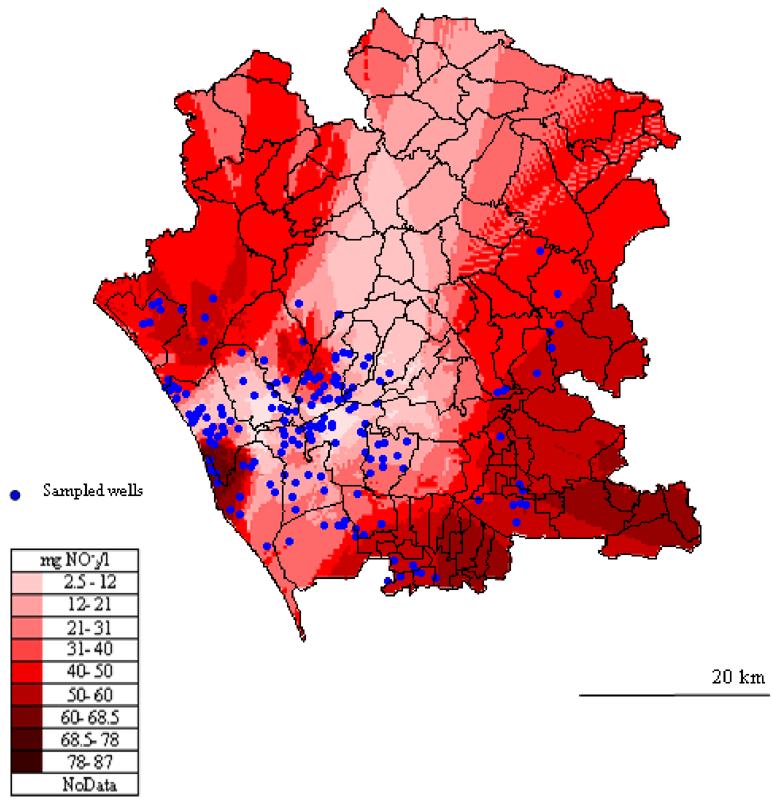

Fig.6. 4 Kriging prediction of nitrate concentration in shallow groundwater in mg/l

To display the possible correlation between nitrate concentration in groundwater and the presence of livestock farms, a scattergram, a tool that is widely used for such elaborations, was developed. Samples were collected in superficial canals in the area of Volturno-Garigliano during September 2007 to assess whether waste was being dumped regularly in superficial streams. Nitrate concentrations were measured by means of a portable reflectometer.

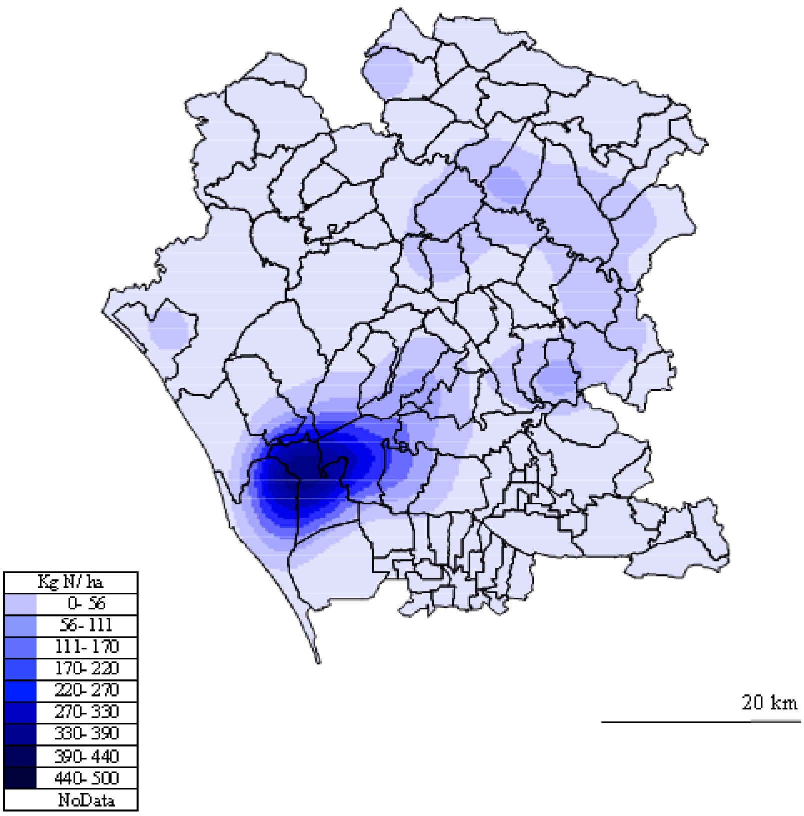

Fig.6. 5 Spatial distribution of nitrogen production by farms, kg/ha

6.3.Results

The data obtained, according to the methods established by the Italian Ministerial Decree of 7th April 2006, show that the province of Caserta (SAT=107 400 ha) has a theoretical total nitrogen production of 14 700 000 kg per year.

The most interesting observation specifically concerns eighteen municipalities (Table 6. 3) where annual nitrogen production per hectare exceeds the regulatory threshold of 170 kg/ha (Dir. 96/61/EC). For example, in "Cancello e Arnone" the concentration of manure nitrogen is about 800 kg/ha per year. These eighteen municipalities have a total agronomic surface equal to 25 663 ha, of which 40% (10 265 ha) is cultivated with corn silage and rye grass with the remaining fraction divided equally between alfalfa, maize, oats and other crops (15% each).

Table 6. 3 The eighteen communes with large excesses in nitrogen production

Municipality	Total N (kg/y)	SAT (ha)	kg N/ha
Alife	405 313	1 714	236.429
Alvignano	462 782	2 577	179.565
Baia e Latina	267 852	1 154	231.986
Cancello e Arnone	252 4091	3 153	800.480
Capua	321 786	1 525	210.975
Castel di Sasso	130 189	651	199.817
Castelvolturno	1 509 041	1 948	774.387
Ciorlano	445 309	1 221	364.413
Grazzanise	1 108 980	2 306	480.871
Marcianise	255 328	869	293.517
Pastorano	321 786	1 083	297.061
Piana di Monte Verna	398 546	1 234	322.779
Pietramelara	241 390	672	359.109
Pignataro Maggiore	339 562	1 394	243.478
Pontelatone	189 476	698	271.432
San Tammaro	229 674	617	371.713
Santa Maria la Fossa	589 840	1 007	585.187
Sant'Angelo d'Alife	364 206	1 831	198.855
Total	10 105 151	25 663	

Table 6. 4 Nitrogen fertilization rate related to the eighteen communes recorded in Table 6. 3

Crop	Kg N/ha	% ha SAT	ha SAT	kg N*ha^{-1}*ha SAT
Alfalfa	40	15%	3 850	154 000
Corn silage + lolium multiflorum	265 (140+125)	40%	10 265	2 720 225
Maize	190	15%	3 850	731 500
Oats	45	15%	3 850	173 250
Other crops	55	15%	3 850	211 750
Total				3 990 725

Table 6. 4 demonstrates that no more than 4 000 t/y of nitrogen is sufficient to fertilize crops, but the total nitrogen production from manure is about 10 000 t/y (Table 6. 3). The data sampled in 1999-2005 were compared with the regulatory thresholds (Italian Legislative Decree 152/1999) for sensitive areas (nitrate concentration of 25mg/l) and vulnerable areas (nitrate concentration of 50 mg/l); the comparison showed that 36% of the samples had a nitrate concentration of approximately 50 mg/l, while in 22% of the samples the concentration was between 25 and 50 mg/l, and only 42% of the samples showed a concentration less then 25 mg/l. The water sampling data were also processed by statistical software programs (Surfer 8 and SPSS), to evaluate their coherence and their statistical distribution. The skewness of the original experimental data was reduced by logarithmic transformation and by removing the extreme values (Fig.6. 6) (Goovaerts, 1997). Hence, the average dissimilarity between data was assessed by an exponential semivariogram model (Fig.6. 7), characterized by a best-fit nugget effect equal to 0.14.

Ordinary kriging predictions were estimated based on the semivariogram (Fig.6. 7).

These predictions of nitrate concentration in shallow groundwater were then compared with the spatial distribution of nitrogen density produced by each farm in a year (Fig.6. 5). For this aim the comparison method, based on a grid with a mesh of 1 000 meters and discussed in the previous section, was used. Data pairs were compared through a scattergram (Fig.6. 8) where model-predicted nitrate concentrations in groundwater were compared with the average nitrogen production by farms. If a correlation between the animal loading and the pollution had existed, the correlation coefficient "ρ" would have been greater than 0.5 and the scattergram would have shown data whose trend could be modelled by an increasing monotonic function. However, the correlation coefficient "ρ" turned out to be -0.38 which demonstrates that the correlation between animal density and nitrate pollution in the groundwater was negative rather than positive.

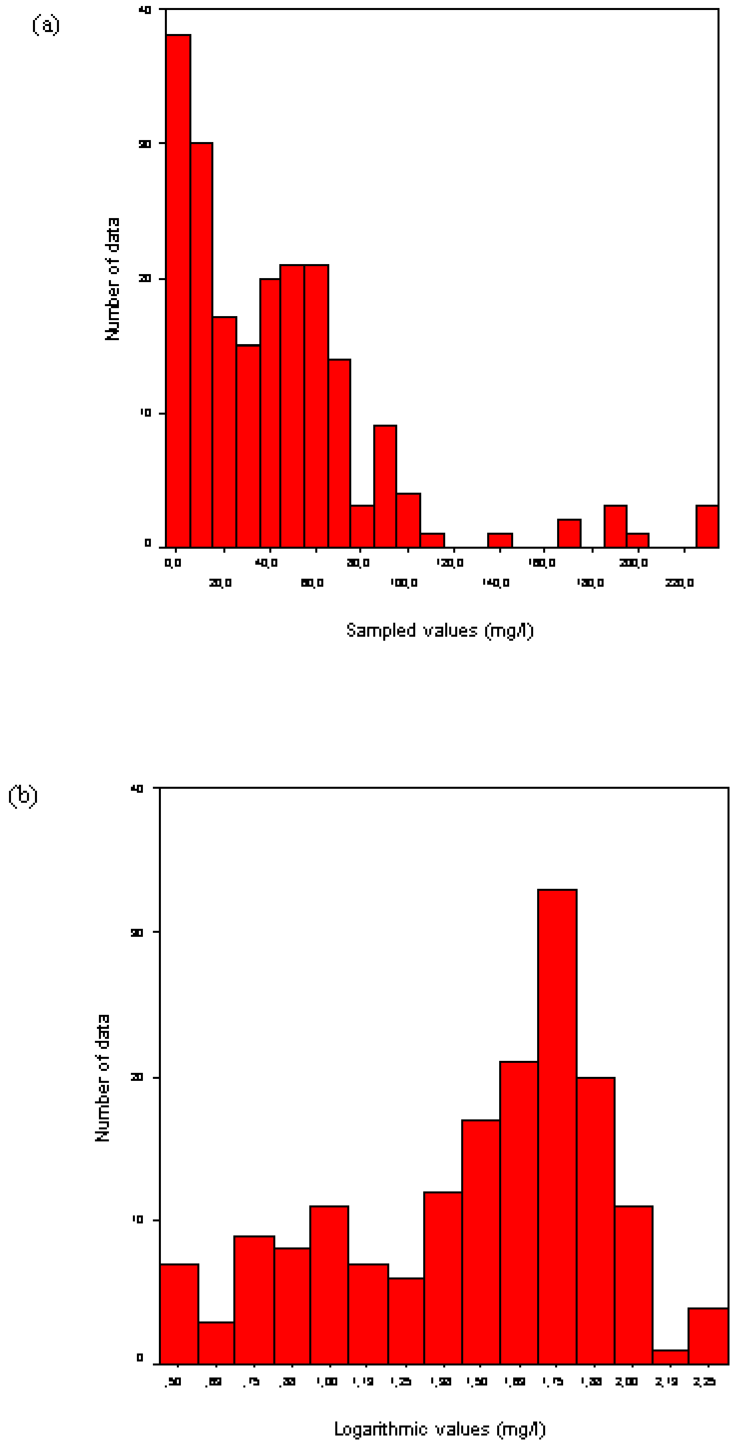

Fig.6. 6(a) Original sampled data distribution. (b) Sampled data distribution after logarithmic transformation (Note: The ratio between nitrate (NO_3^-) and NO_3-N is 4.42).

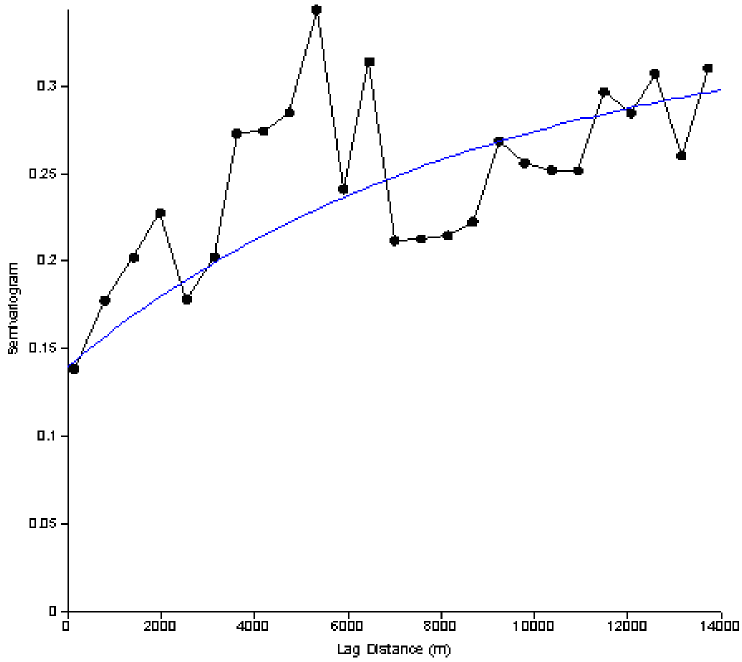

Fig.6. 7 Semivariogram of nitrate concentration in groundwater.

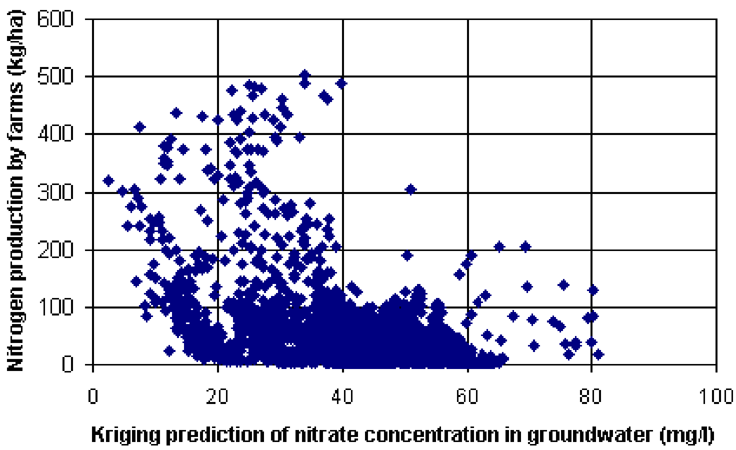

Fig.6. 8 Scattergram: comparison between nitrate concentration forecast in groundwater and nitrogen production.

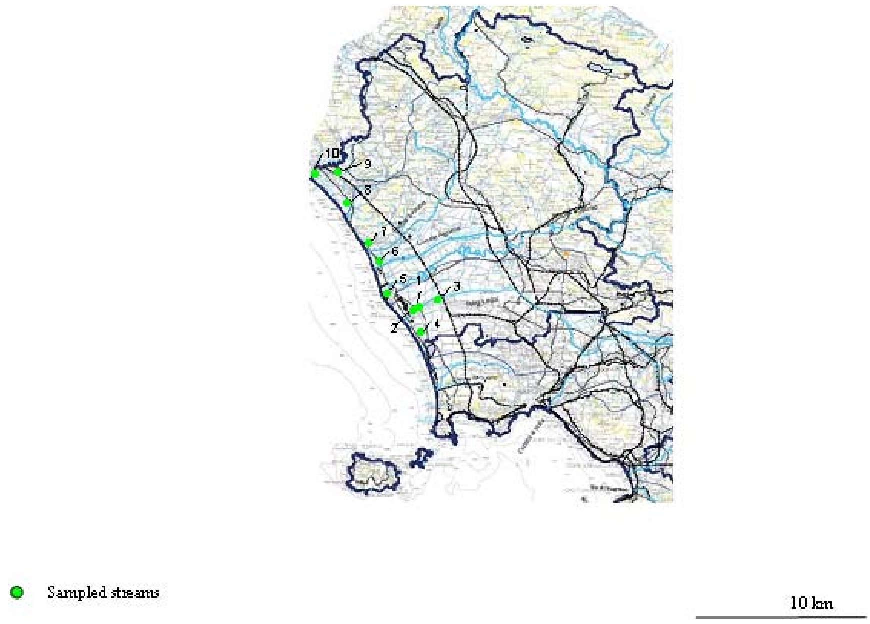

Fig.6. 9 Location of sampled superficial streams

6.4.Discussion

This study has evaluated results obtained by putting EU and regional manure management regulations into effect. The study region followed community directives and thus fertilizer application planning was enforced, at least formally. However, there is a suspicion that, in reality, manure management has been moving in a direction opposite to that intended by the EU and that the discipline of EU nutrient management, both in the study area and in other EU zones, is being circumvented (Neeteson, 2000). The research tries to identify the critical features of this complex system and evaluate whether it would be opportune, or otherwise, to introduce manure treatment technologies, even if they are temporary or considered to be “end of the pipe” technologies and as such would not be considered ecologically correct in other contexts.

The method to identify livestock rearing farms through the analysis of orthophotos, adopted in this study, appears to be well-founded. It allowed, within a margin of error, a representation of livestock farms in the study area which is very close to the true location. The margin of error does not depend on a geographic uncertainty, but on the uncertainty of the actual number of animals in livestock rearing farms. This kind of approximation was handled by distributing the environmental impact of the farm, using a search radius of 6 000 meters. This distance is comparable to the distances that were used for the kriging prediction (the maximum lag distance equal to 12 000 m).

It is interesting to compare in depth the geographic locations of the livestock rearing farm (Fig.6. 3) and the spatial distribution of annual nitrogen density produced by the farms (Fig.6. 5). Through the generation of map layers, it was possible to create from discrete data a map of the probable distribution of nitrogen production. Comparing Fig.6. 3 and Fig.6. 5, it is possible to deduce that the nitrogen production per hectare is not uniformly high throughout the area even where there are many farms, because the average number of animals is low on some farms. Hence proportional information about the possible quantity of nitrogen leaching, that would be obtainable if the manure were managed following the regulations, is given.

Table 6. 3 and Table 6. 4 provide the total nitrogen required for crop production and the total nitrogen in manure at the municipality scale showing 18 municipalities with large nitrogen excesses. The ratio between the total nitrogen required for crop production and the total nitrogen in manure is 1 to 2. Such a nitrogen loading is, in practise, unsustainable in the study area, where there are few or no facilities suited to storing large quantities of manure. Furthermore there are specific areas and periods in which spreading manure is allowed, in reference to regional regulations (DGR 610, 2/14/2003). In view of this, it is necessary to determine the quantity of manure nitrogen that can effectively be used through “good agricultural practice” which would ensure good crops, while also preserving water streams. It should be borne in mind that all the farms in the study area are not

rearing livestock and that it is unlikely that farms which do not would permit regular manure spreading on their soil.

It is clear that considering the 18 critical communes, the amount of effectively manageable manure nitrogen is about 4 000-5 000 t/y, assuming that 80% of the SAT is suitable for spreading, versus a theoretical production of 10 000 t/y of manure nitrogen. The excess nitrogen, 5 000-6 000 t/y, should be handled through a more complete, well-structured system. However, such management systems are in most cases not available in the region. Consequently a great quantity of synthetic fertilizer, almost 100% of nitrogen fertilizer needs, is used in the study area (Onorati et al., 2003). It follows that nitrate pollution of shallow groundwater would be expected, and indeed has been found, in our 1999-2005 monitoring study of nitrate concentration in wells.

The manipulation of the groundwater nitrate data performed to construct the semivariogram, suggests that the spatial correlation of the data also depends on the sampling methods. In this view, Fig.6. 6 shows that the geostatistical approach is essential in order to deal with the uncertainty that is typical of regionalised natural variables. Indeed, the experimental semivariogram shows peaks, probably due to sampling depths, to variation in sampling times and to the influence of different crops on the territory analysed. These sources of uncertainty were handled by kriging.

The Arcview 3.2 generated comparisons between the maps showing the spatial distribution of nitrogen produced by each farm and the predicted spatial distribution of nitrate concentration in groundwater. These comparisons were, however, unsuitable for a precise analysis, which was the reason why a comparison method, using a grid, was introduced.

It is necessary to verify the correlation between the nitrate concentration found in the sample wells and the predicted nitrogen production (Fig.6. 8). Since the correlation coefficient ρ equals -0.38, this suggests that no positive correlation between predicted nitrogen production and groundwater nitrate exists, rather the contrary. A possible explanation is that manure is being disposed as a waste rather than being utilized agronomically, otherwise a closer correlation would be expected between high animal density and high shallow groundwater nitrate concentration.

In this context an evaluation of the economic aspects of manure management takes on great importance. It should be considered that the cost for spreading manure (about 1.5 €/t at a distance of 6 km) may limit compliance with land management plans. Moreover spreading activities entail considerable vehicle movements of and labour, given the low nitrogen concentration in manure (3‰), approximately one hundred days annually available for land application and tanks with a maximum capacity of 10 m^3.

Table 6. 5 Nitrate concentration in superficial streams

#	Long. East	Lat. North	NO_3^- (mg /l)
1	13°59'32"	41°00'12"	11.07
2	13°59'29"	41°00'10"	6.67
3	14°00'54"	41°00'51"	8.53
4	13°59'49"	40°59'28"	56.74
*5	13°56'18"	41°02'04"	5.22
6	13°55'08"	41°04'40"	3.97
7	13°53'34"	41°06'04"	16.40
8	13°51'17"	41°09'52"	10.09
9	13°49'15"	41°14'38"	7.76
**10	13°46'13"	41°14'19"	6.16

* Volturno River, **Garigliano River

The absence of agronomic use of manure is demonstrated by the data collected during the surface water sampling campaign in the Volturno and Garigliano river valleys (Table 6. 5 and Fig.6. 9). Only a single value exceeded the legal threshold for nitrate concentration in surface water (50 mg/l as NO_3). However, this is not surprising if the river flow rates are taken into consideration. For example, rivers such as the Volturno and Garigliano are characterised by average flows of 80 m^3 /s and 120 m^3 /s, respectively, values allowing the dilution of great quantities of nutrients. The Volturno river alone, with its average flow of 80 m^3 /s, can transport about 13 500 *t/y* of nitrate into the sea, a large amount compared to the manure nitrogen production in the province of Caserta.

The continuing growth of the livestock industry in both developed and developing countries coupled with the implementation of rigorous environmental regulations and protocols underline the importance of appropriate manure management systems and practices. Manure management has to be handled by a systematic approach once the criteria for optimal management have been determined.

This study has shown that the shallow groundwater in the study area is clearly polluted. Nevertheless, the concentrations of the pollutant are not proportional to the local livestock densities. Significant nitrate concentrations approaching EU levels of concern have been found in streams, rivers and canals, which suggests that manure is being disposed of, in or near surface water bodies. The absence of a correlation between the nitrogen produced by the farms annually and the nitrate concentration in shallow groundwater in the same area supports this hypothesis. Moreover, studies performed on the subject show that large reductions in nutrient inputs do not necessarily cause an immediate response in terms of water quality improvement, particularly in medium-sized and large areas (Stålnacke et al., 2003). It has also been shown (Frost, 1999) that rather than the present emphasis on soil nutrient content and soil erosion, priority should be given to studying the receiving waters.

It is necessary to set this framework as part of the economic context for manure management. Indeed, livestock farmers today are still reluctant to adopt manure best-management procedures, first of all for economic reasons, i.e. using synthetic fertilizers is easier, faster, more practical, and therefore cheaper, than using manure nutrients even if the cost of buying synthetic fertilizer is 0.5 €/kg, versus 1.5 €/t for the manure nitrogen spreading. Manure management entails storage systems and treatments for its use, plus costs for maintenance, all of which are in short supply in Campania.

It may be necessary to introduce integrated manure management policies. However, this framework is quite unlikely to be enacted in the near future, because it would entail high costs for the transport of manure, given the dilute nature of manure slurries. Hence, farms should be encouraged to adopt liquid-solid separation treatments to reduce costs by 30-40% (the solid fraction is more economically transported than the liquid one). A portion of the liquid fraction could be used in loco and the remaining (approximately 30%) regionally.

A further change might be to relocate livestock farming, even if it does not seem desirable from an occupational and entrepreneurial point of view. Finally, it may be possible to introduce manure treatment technologies, but the energy costs and the equipment required may be prohibitive in the current livestock production system. It is authors' opinion that the ratio between the cost of spreading manure and the benefits due to nitrogen content is critical to drive the planning process. The intervention of specific experts to manage and introduce economic-managerial structures for the entire milk production system would be highly desirable. Moreover, economic interventions to encourage manure utilisation for its nutrient content are likely to be necessary.

6.5. Conclusion

We maintain that management of livestock waste in the Campania region is not well controlled and carried out, in almost all cases, without the necessary facilities and infrastructures. It is possible that other EU areas have manure management issues similar to the area studied in this research.

A more focused and complex multifunctional regional plan, oriented towards a more comprehensive environmental management system, would be highly appropriate.

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7. Impacts of Bubalus Bubalis manure on agronomic management

7.1. Introduction

Buffalo breeding for milk production has a main role in Campania region (Southern Italy). Total heads amount to 250 000 and the annual turnover is higher than 1 billion Euros. In this context livestock activities are characterized by extreme density, meaning number of animals per hectare), but their economical and social importance is crucial.

It is common to liken the buffalo species to the bovine species in terms of manure production and quality. The MWPS Committee (1993) estimated that manure production is 85 kg/die per 1000 kg of animal's weight, considering the liquid and the solid fractions, for both the species. Burton and Turner (2003) reported that for both the species the average total nitrogen content is 4 kg/m^3 for slurry and 5.5 kg/m^3 for solid manure.

One of the consequences of this high animal density is the complex management required for manure, respecting rules, environment and cost optimization.

The costs for spreading manure were analyzed by Provolo (2000). He demonstrated that spreading by tank is not economically convenient beyond 6 km from the farm. at that distance the value of the manure nitrogen is less than his cost for spreading.

In 1999, Boccia et al. (2003) monitored the shallow groundwater in an area, belonging to Campania region, deeply characterized by buffalo breeding, Volturno basin (Caserta province). Their results highlighted the presence of nitrate pollution in groundwater (referring to 1999-2000), but those values were not comparable to nitrate concentrations characteristic of other European agricultural systems, such as Flanders, some areas of Brittany and others.

Starting from this point they used a management model to predict nitrate leaching in groundwater (GLEAMS), considering the daily rainfall data over 40 year. Through this study, it was demonstrated that the risk of nitrate leaching was low enough considering the annual rotation cycle lolium-corn silage, although the buffalo manure was considered comparable to the bovine one.

New sampling campaigns and deeper territorial studies (Infascelli et al., 2007), conducted by geostatistics tools, allowed further considerations about the spatial distribution of groundwater nitrate content in Volturno basin. Results of the geostatistics analysis didn't match the buffalo distribution in the same area. This assumption determined further doubts about the validity of the common comparison between buffalo and bovine manure.

Infascelli et al. (2009) developed a method to assess the effective distribution on the territory of buffaloes, by analyzing orthophotos. Results showed a negative correlation between animal loading and nitrate concentration in groundwater.

Basing on regional funds, analysis were conducted by Infascelli et al (2009) to characterize nutrients content in manure. Nitrogen concentration in buffalo manure resulted the halfway of standard nitrogen content in bovine manure.

These first considerations needed deeply studies on buffalo physiology (Campanile et al., 1997). They allow new perspectives in sustainable livestock, considering the local features in terms of environment, landscape, society and so on.

The aim of this study is to assess which are the livestock areas characterized by nutrients surplus and which is the best way to manage surpluses, basing on the territorial studies, physiological researches and manure samples analysis mentioned above.

7.2.Materials and methods

7.2.1. Assessment of nitrate leaching considering the local context and the agronomic practices

The Caserta district is characterized by a population of 95 000 river buffalo and 53 000 dairy cattle (ISTAT 2000 Italian Agricultural Census) on a UAA (Utilizable Agricultural Area) of 107 400 ha, representing the entire Caserta district with a consequent ratio of 1,4 heads/ha of UAA and a relevant problem of manure management.

Considering that about a half of the total UAA is associated to livestock farms, it can be easily estimated a load of more than 4 heads per ha of UAA of the livestock farms.

Animal feeding is provided with food and fodders of external origin hence it a further increase of heads is foreseen.

The studied watershed is characterised by a very complex underground system constituted by a tectonic ditch (carbonate basement) very wide and deep (more than 2000 m) above which a succession of sea and volcanic alluvial sediments which contains several minor shallow groundwater, enclosed between impermeable and discontinuous horizons, with a meteoric feeding in which the nitrate concentration is strictly correlated to the agricultural practices adopted on the surface (Allocca et al., 2003).

Soils show a permeability varying from high to medium-low without continuous impermeable layers.

Consequently it has been thought that the nitrate content present in the wells not deeper than 40 m is closely correlated to land management.

To assess the risk of non point pollution, especially due to nutrient (nitrogen and phosphorus), a GLEAMS model was used. This allowed the creation of a good simulation model (Garnier 1999).

The model GLEAMS (Groundwater Loading Effects of Agricultural Management Systems) – US Department of Agriculture has been used to simulate the leaching of NO_3 (Leonard 1987, Knisel, 1993).

The model predicts at a field scale, the transformations of nutrients and pesticides, the uptake by the crops and the mobilization of nutrients taking into account climate, soil, hydro-geology crops and agricultural practices (tillage, fertilization, irrigation, pesticides).

To simulate the amount of runoff and percolation, the model needs a daily reconstruction of rainfall; data surveyed in a representative meteorological station of the basin (Caiazzo, S. Maria del Pizzone) have been used. Local data of temperature and solar radiation have also been used.

Table 7. 1 GLEAMS simulation parameter

Crop	Date (Julian day)	Operations
Lolium	263	Ploughing (30 cm) and secondary tilling
	273	Lolium sowing
	349	Plowed down lolium ~2000 Kg/ SS ha
	7 (II year)	Inorganic fertilizer application (urea 92 kg/ha di N)
	60 (II year)	Plowed down lolium ~3000 Kg/ha SS
	65 (II year)	Inorganic fertilizer application (urea 92 kg/ha di N)
	120 (II year)	harvesting lolium ~3000 Kg/ha SS
Corn silage	123 (II year)	Ploughing (40 cm) and secondary tilling
	125 (II year)	Inorganic fertilizer application (18/46 100 kg/ha)
	127 (II year)	Corn silage sowing
	154 (II year)	Weeding (10 cm)
	160 (II year)	Irrigation (5 cm)
	166 (II year)	Inorganic fertilizer application (urea 138 kg/ha di N)
	180 (II year)	Irrigation (5 cm)
	200 (II year)	Organic fertilizer application (1,2 cm of liquid waste 0,3% N)
	200 (II year)	Irrigation (5 cm)
	220 (II year)	Irrigation (5 cm)
	240 (II year)	Irrigation (5 cm)
	260 (II year)	Corn silage harvesting ~30000 Kg/ha SS

7.2.2. Monitoring water from wells

A number of 200 samples have been collected from 150 wells in the years between 1999 and 2005; the position of the wells has been surveyed with a GPS (Fig.7. 1). Nitrate concentration has been measured with a portable reflectometer using a colorimetric test.

The rotation cycle lolium-corn silage, referred to the same years of the sampling and according to the indications obtained by the farmers, has been evaluated as representative of the agronomic management.

Soil analyses, necessary as input for the simulation, have been carried out by a private laboratory.

The study area is characterized by a clay-loam soil texture: 30% clay, 35% sand, 35% silt.

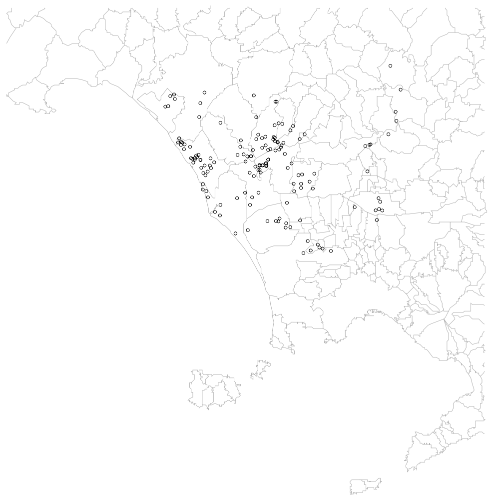

Fig.7. 1 The study area

7.2.3. Manure sampling

Nutrients content in buffalo manure was quantified by lab analysis. In a first step 15 livestock farms were identified in the Caserta area; they represent the typical farm of that area (Fig.7. 2), considering the management and the organization. Each farm had a denomination: from CE1 to CE15. The average surface covered by the farms is between 25 ha and 110 ha.

The sampling and the analysis procedures are based on the use of Coliwasa sampler and are described by Infascelli et al. (2009). Coliwasa was created by the University of Wisconsin. The main parts of the Coliwasa consist of the sampling tube, the stop-cock, and the closure system. The sampling tube consists of a 1.5 m by 4.13 cm inner diameter translucent plastic pipe; usually made of polyvinyl chloride (PVC), PTFE or borosilicate-glass plumbing tube. It is not suitable for sampling in containers over 1.5 m deep.

Fig.7. 2 Storing tanks

The dry matter content was quantified by using a stove for the starting samples, following the method described by Wolf et al. (1997). The total

Kjeldahl nitrogen content (TKN) was determined by following the procedures proposed by Kane (1998), AOAC Official methods n° 978.02. The nitrogen and phosphorus content was respectively determined by Kjeltec tool and spectrophotometer.

7.2.4. Geostatistics study

The study has three main aims:

- the elaboration of the data collected by the wells sampling;
- the prediction of the average manure nitrogen production by livestock farms;
- the identification of the most suitable areas for manure treatment plants in the study zone.

Data collected, that are point data, were elaborated by geostatistics tools through the kriging predictor and the software Arcview 3.2. The aim was to obtain prediction maps and to identify the areas, belonging to Caserta province, where the nitrate content was not measured directly (Infascelli et al. 2007 e 2009), (Fig.7. 3).

The average Nitrogen production in farms was represented by GIS maps. These maps were utilized to calculate the nitrogen surpluses referring to the law threshold (DM 152/2006). Thus it was possible to identify the areas where the nitrogen surpluses could be handled. Nitrogen production was quantify by considering the number of buffaloes and bovines recorded in the Data National Bank (BDN, 2008).

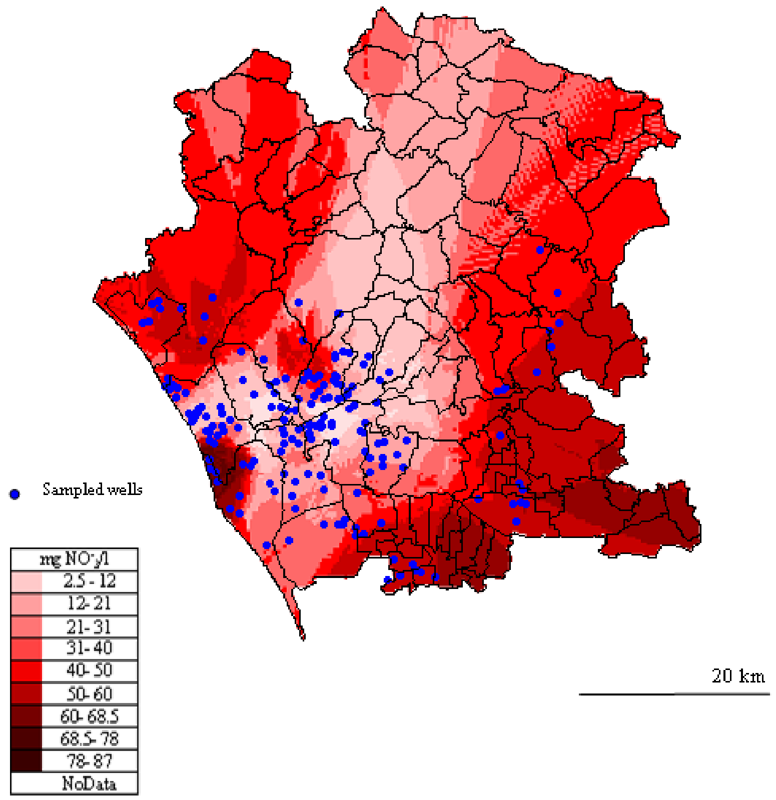

Fig.7. 3 Nitrate content in shallow groundwater obtained by kriging prediction

Two different hypothesis were considered:

1) the buffalo species has physiologic features similar to the bovine species, thus the nitrogen production for each production head is 83 kg N/head/year and for the culling heads is 36 kg N/head/year (as reported in DM 7/4/2006).

2) For the buffalo species the nitrogen production is 57 kg N/head/year for each production head and 32 kg N/head/year for the culling heads (Campanile et al., 2006)

The spatial distribution of the average nitrogen production in each farm was elaborated and visualized by using a convolution function, in GIS environment (the Kernel Density) (Infascelli et al., 2009).

The most rational search radius was hypothesized equal to 6km (Provolo, 2000)

Consequently the hypothesis used to determine the nitrogen load were:

1) nitrogen production equal to 83 kg N/head/year for the adult heads and 36 kg N/head/year for the culling heads (no distinction between buffalo and bovine species). Search radius for the convolution function equal to 6 km (Fig.7. 4)

2) nitrogen production equal to 57 kg N/head/year for the buffalo production heads and 32 kg N/head/year for the buffalo culling heads (it means that the nitrogen production in buffalo manure is about the 65% of the nitrogen production in bovine manure) (Fig.7. 5).

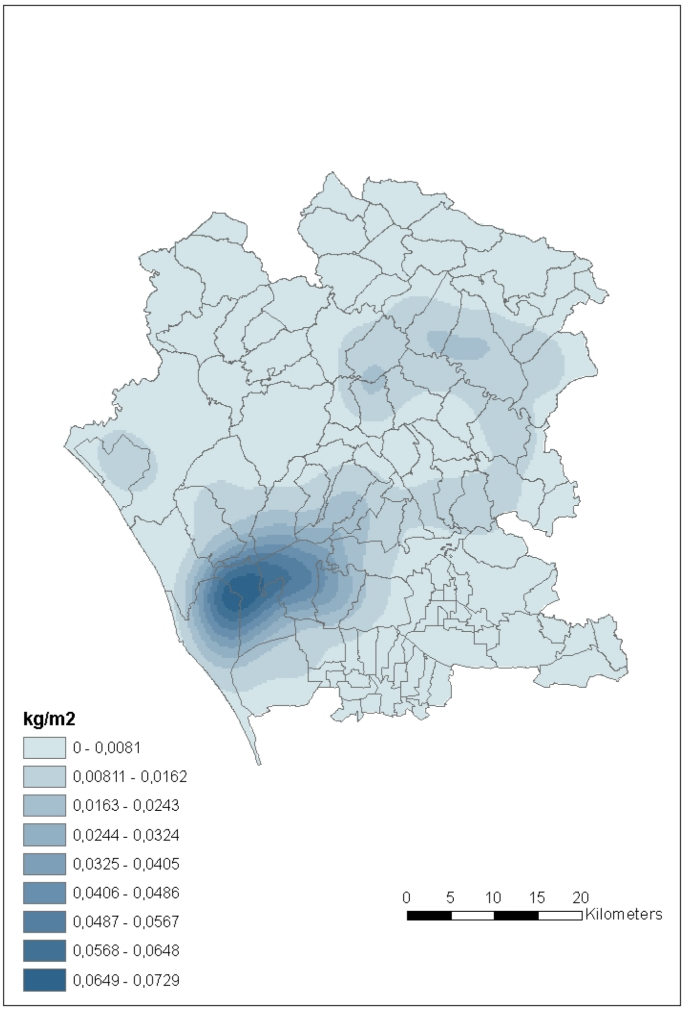

Fig.7. 4 Spatial distribution of nitrogen production, considering the physiology similarity between buffalo and bovine species and a search radius of 6km

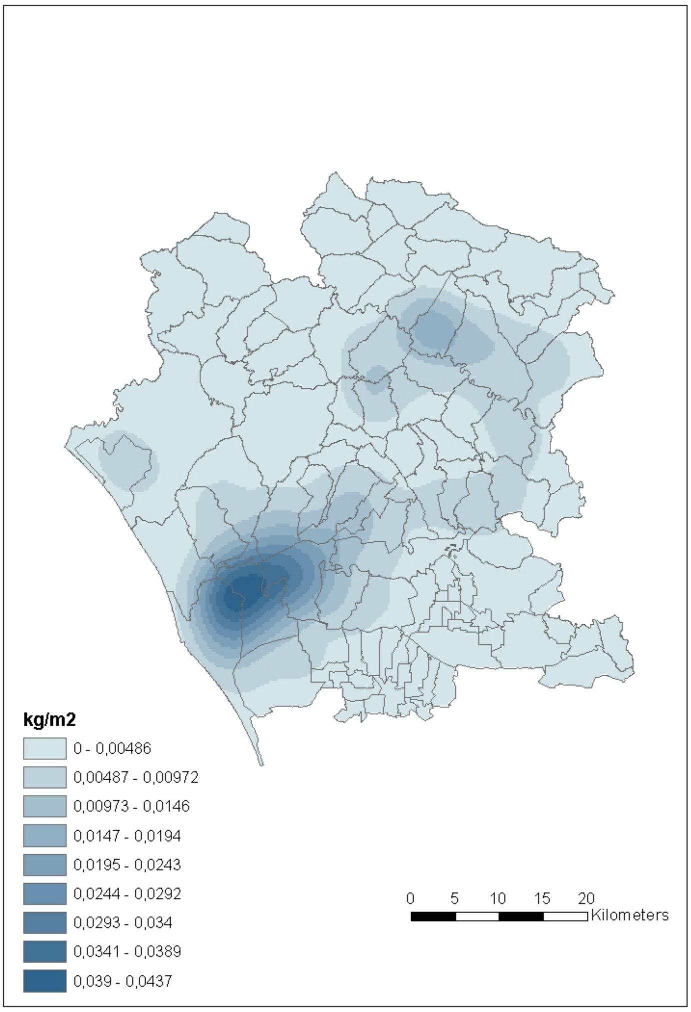

Fig.7. 5 Spatial distribution of nitrogen production, considering that the nitrogen production in buffalo manure is about the 65% of the nitrogen production in bovine manure and a search radius of 6km.

The map of the suitable areas for treatment plant siting was realized by using the Arcmap function “Focal Statistics” with a search radius of 5km and considering the corresponding surface (78 500 000 m^2). Thus, a binary reclassification was conducted to identify the suitable areas. The nitrogen surpluses threshold considered for the reclassification was 150 t/year. This threshold was chosen taking into account that the optimal dimension of a treatment plant should:

- not exceed one MWe, in the aim to obtain simplified authorization procedures;
- be less than 5km far away the collection sites;
- guarantee an agricultural reuse of the post treatment exceeded fraction.

7.3. Results

The results of the NO_3-tests show that the average nitrate concentration in the shallow groundwater in the study area, is very high. The average nitrate concentration in the 200 measures carried out is of 41 mg/l, while the European limit for drinkable water is 50mg/l but the attention limit is 5mg/l.

In order to understand the critical load of the system, the rotation cycle and tillage adopted in the zone has been simulated, imposing only the mineral fertilization usually employed in the conventional practices with a total nitrogen supply of about 340 kg/ha.

Without considering organic nitrogen contribution due to animal waste application, using the daily rainfall data of the last 45 years (average 1340 mm), an average concentration of 24 mg/l has been obtained. Adding a liquid animal waste application in the rate of 1,2 cm with a content of 0,3 % of nitrogen, corresponding to about 4-5 heads/ha, the simulated nitrate concentration reaches 45 mg/l. Such a value is certainly high but on line with the experimental results obtained from the tests of the wells, it confirms the reliability of the model (Leone et al 2001, Garnier et al 1998).

Since the local farmers are not available to avoid chemical fertilization, it was tried to estimate the amount of animal waste added to the chemical fertilization, which lead to a nitrate concentration in the groundwater of 70 mg/l.

The simulation results show that the concentration of 70 mg/l is not exceeded; though increasing the liquid waste supply to 3 cm, that is reaching approximately 1240 kg N/ha year, 900 kg N/ha year coming from animal waste. This is equivalent to approximately 11 heads/ha.

To asses if such a concentrated leakage could be tolerated by the deeper groundwater with a higher rate, hydro-geological and deep territorial studies would be necessary.

The results obtained by the GLEAMS simulation were well-supported by the geostatistical studies that were conducted on the basis of the sampling campaign done in 1999-2005.

The 30 % of the samples collected in 1999-2000 had nitrate concentration higher than 50 mg/l, the threshold fixed by European. Whereas only the 10% of the data collected in 2005 exceeded the same threshold. Thus it is clear the decrease in nitrate concentration in groundwater.

The geostatistical study of the area of Caserta province provided the data to elaborate a map of the predicted nitrate concentration in groundwater and a map of the spatial distribution of the average nitrogen production in farms during a year.

The maps of predicted nitrate concentration in groundwater and of the average nitrogen production in farms during a year were obtained by using kriging interpolation (Infascelli et al., 2009). The comparison between the two maps demonstrated that there is a small negative correlation between the potential nitrogen production and nitrate pollution in groundwater. This lack of an outstanding correlation allows to do some hypothesis: the existence of a perfect agronomic use of manure or a wrong management of livestock activities or some inaccuracy in defining the standard manure composition proposed by legislation.

The analysis, conducted in 2009 to characterized the nutrients content in buffalo manure, provided the data of Table 7. 2. Nutrient content in manure was characterized through the Dry Matter (DM), Total Kjeldahl Nitrogen (TKN) and orthophosphates (PO_4). The values recorded in the table correspond to the mean value and the standard deviation of the characteristic parameters mentioned above.

Table 7. 2 Nutrient content in buffalo manure (data collected and elaborated in 2009)

	Parameter	DM %	TKN mg/l	PO_4 mg/l
Liquid	Mean value	4	1235	610
	Standard Dev.	3	365	285
Semi-solid	Mean value	11	2510	405
	Standard Dev.	-	410	-
Solid	Mean value	21	4860	730
	Standard Dev.	-	-	-

The results of the buffalo manure analysis showed nutrient content variations proportional to the dry matter content, all but PO_4 in semi-solid samples.

The further geostatistics studies showed that nitrogen concentrations, in that areas characterized by high spreading activities, ranges between 0,05-0,07 kg N /m^2 (Fig.7. 4) corresponding to 500-700 kg N /ha /year. Whereas in many other areas the values are about 0,005 kg N/m^2.

In order to identify the best management and control actions, Fig.7. 5 shows an area (about 20 000 ha) where the average nitrogen manure production is higher than 350 kg N/ha.

The law threshold for nitrogen spreading is 170 kg N/ha for the vulnerable areas, but only a 70% of the total agricultural surface is supposed to be available for spreading, thus we hypothesize that the study area can support no more than 100 kg of nitrogen manure per hectare.

The map of the surpluses (considering a threshold of 100 kg N/ha) is represented in Fig.7. 6. The excesses range around 200 kg N/ha.

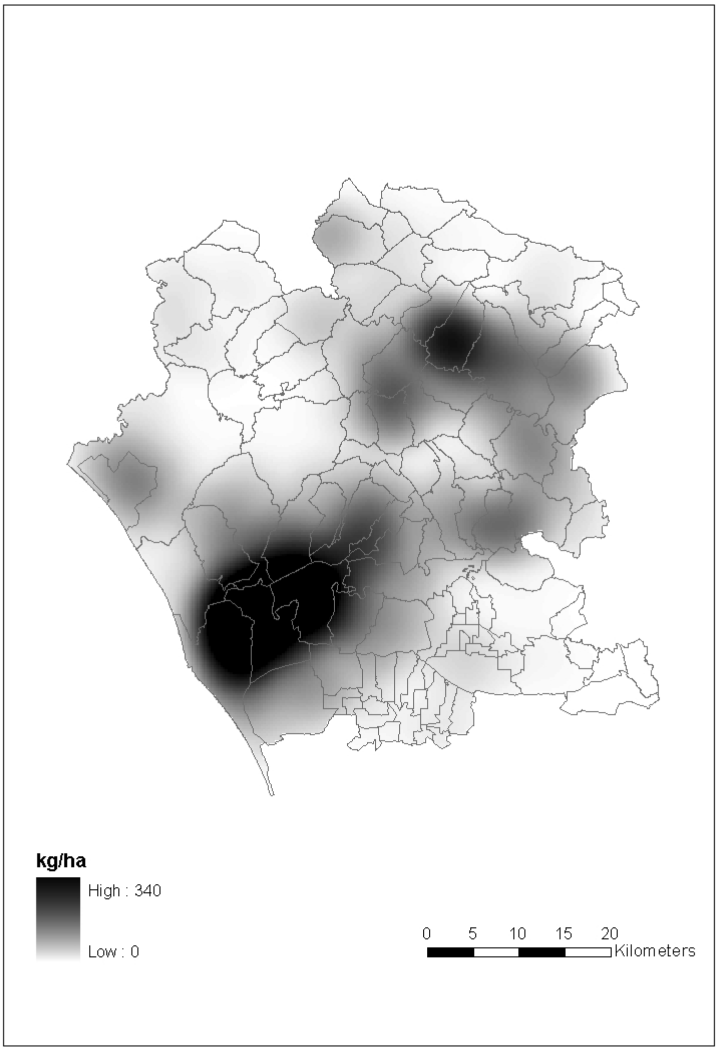

Fig.7. 6 Nitrogen surpluses considering a threshold of 100 kg/ha

Starting from Fig.7. 6, it was possible to identify the most suitable areas for the treatment plant siting.

The chosen technology should comprehend an anaerobic plant, at the beginning, to let the system be economically reasonable.

The optimal dimension of a treatment plant should satisfy three main constraints:

- to not exceed one MWe, in the aim to obtain simplified authorization procedures;
- the distance for the conferment should be less than 5km;
- the plant dimension should guarantee an agricultural reuse

of the post treatment exceeded fraction.

These hypothesis define about 150.000 kg of nitrogen per year that could be used in a treatment plant. Adding organic fractions to increase the carbon- nitrogen ratio, the optimal plant dimension is the one defined above.

Thus, in the case of buffalo manure treatment, the right dimension is about 75000m³ per year.

The suitable areas for a plant siting are showed in Fig.7. 7. These areas were identified considering a conferment radius of 5km and nitrogen surpluses higher than 150 t/year.

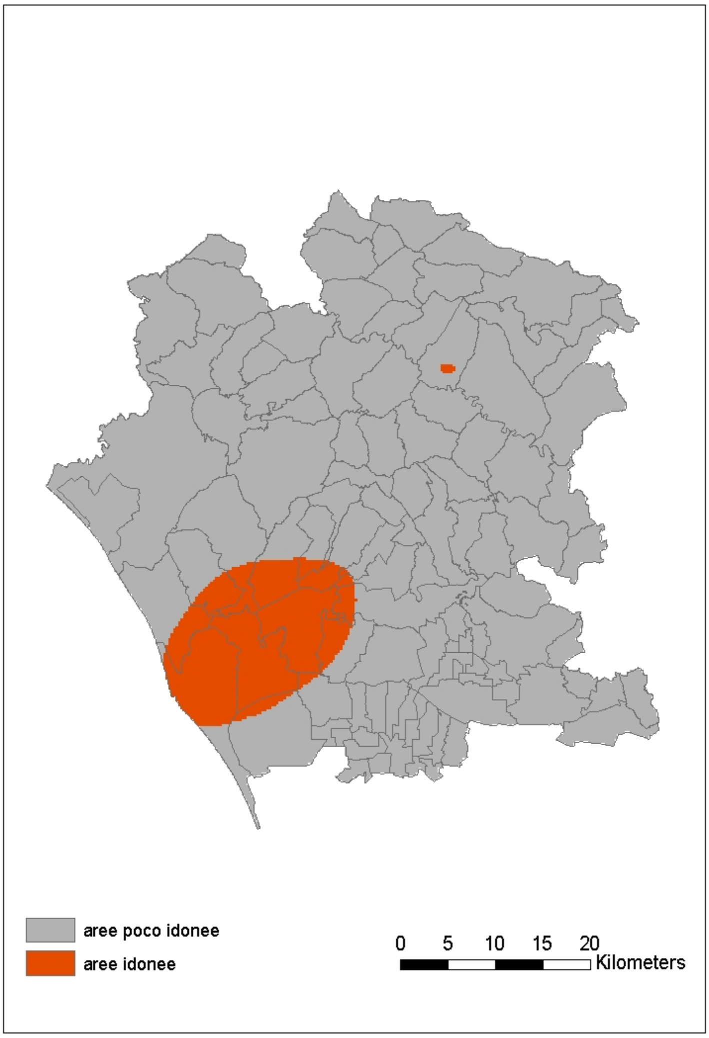

Fig.7. 7 Most suitable areas for the treatment plant siting (orange), not suitable areas (grey)

7.4. Discussion

Notwithstanding the European and local rules and hypothesising to reach the 50 mg/l threshold for nitrate concentration (NO_3), moreover thinking to utilize only nitrogen from manure, the maximum amount that can be managed, according to the GLEAMS model, can be estimated in about 1200 kg/ha.

Such amount corresponds to the load produced by about 15 bovine heads /ha or, basing on the obtained results, by about 30 buffalo heads, but it must be reduced proportionally to the chemical nitrogen supply.

These loads could be managed only where the shallow groundwater isn’t used to drinkable use or where it is sure that it meets a greater aquifer which could tolerate and dilute the consequent load. A deep hydro-geological study could guarantee the feasibility of extreme proposed practices.

Considering the livestock and its increasing trend, a preliminary study performed with a simulation model like GLEAMS seems essential both for a correct arranging of farm fertilization plans, and for a of the management practices to reduce, as far as possible, environmental risk.

Further, it is common opinion that the characteristics of the buffalo manure are similar to the ones of the dairy bovine manure. Accordingly, the total nitrogen content is provide to be equal to 3-4 kg/m^3 in liquid manure and 5.5 kg/m^3 in solid manure (Burton and Turner, 2003). There is a large gap between these values and the ones determined by the lab analysis mentioned above (1.2 kg/m^3 in liquid manure and 2.5 kg/m^3 in solid manure).

This mismatch likely is one of the reason why the inconsistency exists between the founded and predicted pollution, calculated by kriging interpolator, compared to the livestock loading, predicted by the convolution function.

One of the explanations is the different nitrogen metabolism between the buffalo and bovine species. This is mainly due to the tropical provenance of buffalo species (Campanile et al., 1997) and the feeding typology.

Moreover during 2008-2009, in the study area, the average rainfall achieved the 50% more than usually (about 150 mm per month). This can partially explain the low nutrients content in manure by dilution, but the samples measured in this study show too low values of nutrient concentration. Indeed, the standard livestock typology in Campania is characterized by 7 m^2 per head of outdoor surface. Thus, the rain water usually represents about the 25% of the storing tank but during the period 2008-2009 it represented about the 38%. Nevertheless this phenomenon can not explain the low nutrient concentration measured in this study, but only a variation within the 15%.

As mentioned in the results section, the right dimension for a buffalo manure treatment plant is about 75000 m^3 per year. It means that considering a retention time in the digester of 30 days, the digesters volume should be 6500 m^3.

The surpluses in the critical areas are more than 1500 t N/year, thus about 10 plants are needed.

In order to obtain more detailed information, the map represented in fig.8 can be overlay on a land use map and a rural viability map.

7.5. Conclusion

The studies conducted in Campania region showed that the connection between theoretical nitrogen production in farms and nitrate content in groundwater is not so clear.

The lab analysis about nutrient in buffalo manure showed an average content less than 50% than the standard nutrient content adopted for bovine species.

This lower concentrations could be only partially explained by the great rainfall phenomenon that characterized the period 2008-2009. However, on the other hand, they could be the reason why we founded a high nitrate pollution in groundwater only in some particular areas and not so homogeneous in all the Caserta province.

Considering some economic aspects, the low nutrient concentrations that characterize buffalo manure determine high management and transport costs. Consequently, the costs for spreading tend to increase, causing damages to the competitiveness between nitrogen manure and the synthetic one. Basing on the concentrations measured in this study, the spreading cost is about 1 € for each nitrogen kg.

Basing on the simulations, the territory seems not much vulnerable. The annual crop rotation and the climate determine small problems connected to nitrate leaching. Indeed, this is confirmed by the data collected to have a measure of nitrate pollution in groundwater. Theoretically, it should be possible to increase the livestock loading, but it would be out of line with European regulations.

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8. Testing different topographic indexes to predict wetlands distribution

8.1. Introduction

Rural landscapes are characterized by great heterogeneity, thus for any policy of landscape protection and management, the delineation of landscape structures is a prerequisite and the demand for such information at the scale of large catchments or basins. Focusing on wetlands, different studies proposed indexes to predict the their extension, based on topographic and climatic information, mainly within small order catchments. The topic of this study is to determine the validity of different indexes for different orders of catchments and to propose the improved index predicting the delineation of wetland for large order catchments.

The work is based on a 830 km² basin where the actual extension of wetlands is partially known on the base of a soil map. We checked the efficiency of different topographic and hydrological indexes as the climato-topographic index and others, by comparing the map of predicted and actual wetlands, for different stream orders. Results have showed that we improved the prediction of wetland delineation for large catchments when we took into account the importance of the flatness of the bottomland. We proposed the ordinated climato-topographic index that reflects this effect in including the local downhill difference in level to the stream, weighted by the drained volume by the stream as an indicator of the stream order. Such index allows environmental stakeholders a better prediction of wetlands at the management scale.

8.2. The importance of wetlands in the aim of reducing non point pollution

Rural landscapes are characterized by great heterogeneity. Land use mosaic usually shows different patches, with variable size, that cover the land. Some of these patches are non agricultural landscape structures such as hedges, ditches, watercourses, roads. Agricultural fields could be sources of diffuse pollutants, such as nitrogen and phosphorous, while the other landscape structure could be sinks, waterways or could act by diluting the pollutant. Experimental studies at the field scale can provide estimates of pollutant emissions by different cropping system; on the other side it is possible to evaluate the pollutant losses at the catchment scale by monitoring. The relationship between input and output is very complex and variable, due to landscape structures, storage in the soil, transfer time, the occurrence of particular climate conditions (intense precipitations, drought periods) and due to the uncertainty that characterizes these variables and natural phenomena. Even if field emissions could be measured experimentally or estimated by using the agricultural mass balance and regional references on the rate of soil transformation, it is almost impossible to control and predict all the process that can take place between the emission point and the outlet (Basset-Mens at al, 2005). Consequently, it is difficult to assess nutrient retention – when it happens- due to landscape structures simply by comparing field emissions and fluxes at the outlet. A better approach is providing a set of variables to assess the limits and the possibilities for quantifying nutrient retention impacts both at local and at landscape scale, as Viaud et al. studied in 2004.

Different studies have been conducted to define the spatial distribution of those landscape structures that are able to influence the pollutants' load between the source of emission, mainly the field, and the flux at the outlet of the considered catchment. Muscutt et al. (1993) define buffer zone as a landscape structure set between the source of water pollutant and the receiving water body that may provide a physical and/or biogeochemical barrier against pollutions inputs; in extreme situations buffer zone can act as sinks.

Considering mainly the phosphorous and the nitrogen as nutrients to take into account in this context, the insoluble phosphorus and organic nitrogen retention capacity of the buffer zone is influenced by the reduction of overland flow (by promoting infiltration) and by the reduction of sediment load in overland flow water (by reducing flow velocity and filtering due to vegetation cover). The sustainability of the buffer zone will depend on other processes that will limit the mobility of the accumulated pollutants, essentially physical adsorption, biogeochemical transformations and maintenance operations such as tillage, re-seeding, mowing (Dillaha and Inamdar, 1997; Uusi-Kamppa et al., 2000). In the case of nitrates and soluble P, the major processes involved in the buffering effect are essentially biological or microbiological: plant uptake, microbial assimilation, denitrification. The role of different types of buffer zones can be examined according to this general framework, on one hand, the local factors controlling the retention processes and, on the other hand, the landscape context determining the overall buffering capacity of the catchment.

Wetlands have the main role of buffer zones thanks to their capability of retention of the pollutants by denitrification, uptake by vegetation, by dilution in the absence of significant nitrogen inputs (Montreuil and others, 2008). They appear as important landscape structure from a hydrological point of view because they are at the interface of surface flow, river flow and groundwater exfiltration (Naiman and Décamps, 1990).

The wetlands, especially the valley bottom wetlands, are characterized by superficial shallow groundwater and represent an interaction element of the catchment. They reach the saturation seasonally due to the fluxes, more or less superficial, that come from the hillslope. The gradient is able to influence the water's movements towards the deeper aquifer. It has been studied that the hydrographic network influences the wetlands; in particular it has been estimated that until the second or third Strahler order there is a dominant interaction between the wetlands and the hillslope; whereas beyond the third order the river configuration contributes with the versant to influence directly the wetlands function. The wetlands are areas where transfer of flows could occur; they are able to stock the water that comes from the hillslope (groundwater, runoff) as well as the river water after inundation phenomena.

Wetland management for diffuse pollution control could be an interesting support to good agricultural practices. In this aim it is fundamental to define the usable area and the methods to maintain them. The delineation of wetlands is a prerequisite for any policy of wetland protection and management, and there is an increasing demand for such information at the scale of large catchments or basins that is the management scale. The development of GIS and DTM allowed scientists to propose methods aimed to predict in an easy way the delineation of wetlands, when ground-based information is not available. To identify wetlands, Merot et al. (2005) dealt with the distinction between potential, existing and efficient wetlands by using different criteria.

8.3. Indexes to define wetlands' spatial distribution

In this context it is necessary to have a realistic knowledge of the spatial distribution of wetlands within the landscape, first of all for their non-point pollution control and their influence on the ecosystem. One of the first methods to predict their spatial distribution was to consider the geomorphology of catchments, because topography is the driving force of water movements. The wetlands location depends mainly on flowpath convergence, slope and the hydraulic conductivity of the soil. Different indexes have been developed to estimate the location of wetlands. Beven and Kirkby (1979) and Beven (1986) defined a topographic index that considers the local slope and the upslope drainage area. It is based on the hypothesis that the hydraulic gradient of the shallow water table is equal to the local topographic slope angle. In this hypothesis and for a given rainfall depth, the larger is the drainage area and the smaller the slope, the higher is the topographic index. Its equation is:

$$TI = \ln(\frac{a}{\tan\beta})$$

where a is the drainage area and β is the local slope. Considering also the transmissivity T , that usually can be neglected compared to the variations of slope and drainage area, the index becomes a soil-topographic index with the equation:

$$STI = \ln(\frac{a}{T\tan\beta})$$

This kind of indexes required homogeneous rainfall depth to let a comparison between different areas. To obtain a more robust index, it is necessary to introduce the effective rainfall depth, namely the net rainfall that is no evaporated or transpirated. Hence the index becomes a clima-topographic index whose equation is:

$$CTI = \ln(\frac{V_r}{\tan\beta})$$

where Vr is the volume of annual effective rainfall, obtained by multiplying the drainage area with the effective rainfall. In this index the downhill slope (the slope between the point of interest and the river course) is used instead of the local slope (Gascuel-Odoux et al., 1998). In the previous approaches we considered, there are only upslope or local variables that control the wetland indexes and, consequently, the location of the wetlands within the catchment is not taken into account. Some field observations and conceptual approaches conduct us to analyze the influence of the catchment order on the wetland prediction. It is necessary to test the relevance of the influence of the conditions downstream with the river and catchment order (Merot and others, 2003).

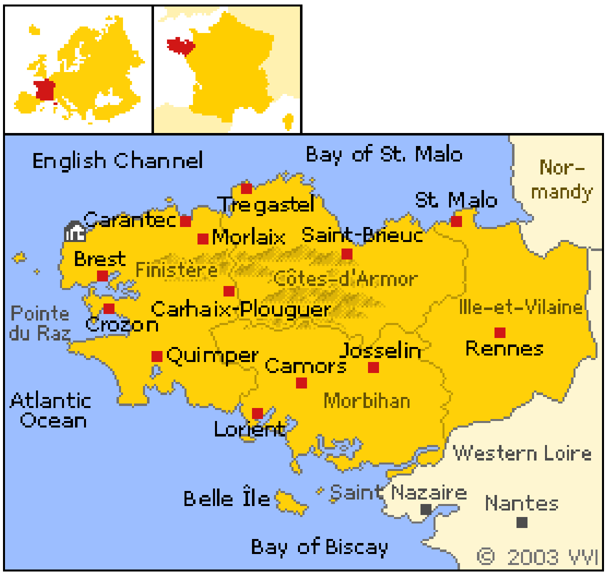
The topographic index, first proposed by (Beven and Kirkby, 1979) in the framework of hydrological modelling, was applied to predict the delineation of wetlands in headwater catchments (Merot et al., 1995; McGlynn and Seibert, 2003). Some improvements were proposed to this topographic index, for instance including the effect on the wetland prediction of the gradient of the mean annual precipitation over the catchment, as it was done in the climato-topographic index (Merot et al., 2003). But some questions rose when people considered large catchments. It is recognised that the assumption on the hydraulic gradient is not always valid close to the stream network, where inverse gradients, from the stream to the bottomland may occur, depending on the water conditions (Burt et al., 2002; Vidon and Hill, 2004). Moreover, one of the conditions for applying the Darcy law is the downslope free water movement in the groundwater. Results at the local scale (Montreuil, 2008; Vidon and Kao, 2005) showed the ability of water exchanges from the watercourse to the groundwater of the riparian wetland.

The aim of the present study is to check the validity of topographic indexes when the connectivity and the interaction between the stream course and the riparian wetland are relevant, as it is recognised when the Strahler order (Strahler, 1957) of the catchment increases (Brinson, 1993; Tabacchi et al., 1998).

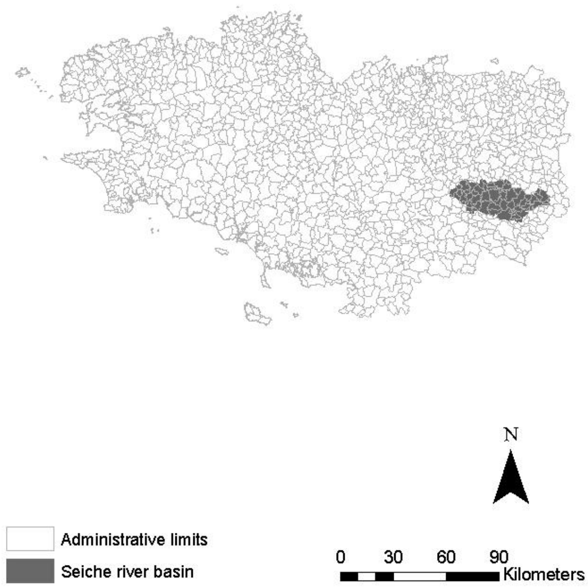
A comparison between the maps of predicted and actual wetlands was made for different stream orders, to test the efficiency of the different topographic and hydrological indexes. It has been showed that the prediction of wetlands for large catchment is more reliable taking into account the flatness of the bottomland.

We considerer necessary also to propose an improved index for the delineation of riparian wetlands in large order catchments considering the local downhill difference in level to the stream, weighted by the drained volume by the stream, as an indicator of the stream order. The efficiency of this new index, called ordinated-climato-topographic, was first tested by Montreuil et al. in the Scorff River Basin (480 km^2), in the Britanny region of

France. In this framework the study area is the Seiche River Catchment (830 km^2), that belongs to Ille-e-Vilaine department, east Brittany (Fig.8. 1).



a)



b)

Fig.8. 1 Brittany region (a); the Seiche river basin (b)

8.4. Material and Methods

8.4.1. Site description

The region has an oceanic climate with effective rainfall roughly estimated as 165 mm per year. The Seiche River Basin has a surface of 830 km^2, the river’s length is 95km. The Seiche’s source has an altitude of 160m while the outlet 15m.

The catchment mainly is schist, so the soil is not permeable. The river flow is not regulated by the groundwater and mainly depends on the rainfall. The average annual flow is $4.60 m^3/s$ but the catchment is characterized by severe drought and floods quite strong depending on the season. The Seiche presents in fact seasonal flows very marked, as so often in the east of Brittany: high waters in winter (December to March) with the average monthly flow between 7.3 and 12.1 m^3/s, and very low water in summer (early June to October) with a decline in the average monthly flow to 0.286 m^3 in August. There are some areas of expansion of floods upstream and in the middle of the Seiche basin. They can reduce the risk of flooding downstream. Land use is characterized by a very open landscape with few hedgerows and embankments. The corresponding SAU (Used Agricultural Surface) is 57 800 ha mainly occupied by traditional agriculture and dairy cow and pig livestock. In 2001, pasture covered about 65% of the SAU, followed by cereal and maize crops (respectively 25% and 10%) (Eaux et Rivières de Bretagne, 2001). The remaining areas (about 250 km2) was characterized mainly by non cultivated soils (25%), woods (20%) and urban areas (15%).

8.4.2. Material

In the aim of this study it was necessary to analyze data layers required to define the different topographic and hydrographic indexes and to test their efficiency and their relation with the catchment features.

A partial 1: 25 000 soil map that covers about 140 km^2 of the Seiche catchment is available. Soils are classified according to four criteria: geologic material, intensity of hydromorphy, soil profile and soil depth. This map identifies an area of about 33km^2 characterized by hydric soil (Fig.8. 2).

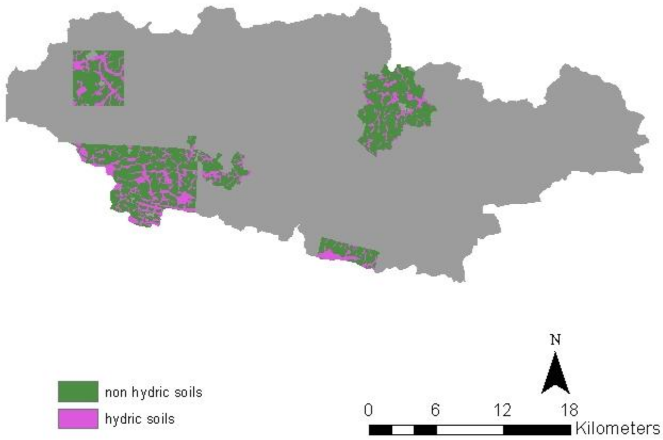


Fig.8. 2 Partial soil map of the Seiche River Basin

The different index calculations were based on the Digital Terrain Model (DTM) of the Seiche catchment characterized by a 50m pixel size. The drainage network was considered as the total set of surface flow pathways in the catchment. In a DTM, the surface flow direction of each pixel was drawn according to the difference of elevation of the contiguous pixels (Depraetere, 1991; Wharton, 1994). Two kinds of models may be used: the multi-directional model and the eight flow direction model (D8) also called mono-directional model. The former assumes that runoff flows from a processing pixel in all downslope directions following a weighting algorithm (Holmgrem, 1994; Squividant, 1994; Quinn et al., 1995); the latter assumes that runoff flows only in the steepest downslope direction. It has been demonstrated (Wolock and McCabe, 1995; Beaujouan et al., 2001) that the use of a multi-directional model is strongly convenient for topographic wetland prediction. On the other side, the mono-directional model is more suitable for modeling the hydrographic network. The hydrologic elaborations based on DTM were developed by the software MNTSurf (Aurousseau and Squividant, 1995).

Discontinuities in the drainage network linked to the generation of DTM create drainage anomalies, i.e. points without outlets. These anomalies are corrected iteratively by virtually filling the points until the connection with a mesh of lower altitude.

The drainage network is, so, divided in 2 parts: the hillslopes with a multidirectional model and the hydrographic network with a mono-directional model. Extracting the hydrographic network from the modelled drainage network shows limitation due to the misfit in the location between the modelled and the observed stream network (Turcotte et al., 2001), specifically on the flat areas. We chose to impose the hydrographic network given by the National Geographic Agency (BD Hydro, from IGN) as a constraint for the DTM. Nevertheless the choice of this network leads to underestimate the length of the headwater functional stream network in this region, where the stream course expands in winter. This underestimation may concern up to 20% of the total length of the stream network. To complete this hydrographic network we adopted the method proposed by (Aurousseau and Squividant 1996; Merot et al. 2003) that defined the hydrographic network as the part of the drainage network that drains the volume of effective rainfall exceeding a threshold volume. This threshold was empirical, defined as the mean value of drained volume observed at the sources of the vectorised network, it was fixed equal to 65 103m3 for all the Brittany region.

The 1: 25 000 vectorial soil map was rasterized considering a cell size of 50m. A raster map, with a 50m pixel size, of the Seiche basin was characterized considering the Strahler order for each sub-catchment; the higher Stralher order recorded is five for this basin.

In summary for this study we have three main map:

- a soil map that covers 140 km^2

- a topographic map
- a stream network with the Stralher classification

8.4.3. Definition of variables and indexes

The topographic variables are showed in Fig.8. 3.

The downslope difference in elevation ($\Delta Z_{downslope}$) was defined as the difference in elevation between the point of interest and the stream network, measured along the hydraulic path (Gascuel-Odoux et al., 1998).

The local slope (Slp_{local}) is used to calculate the ordinated climato-topographic index.

The downslope slope ($Slp_{downslope}$) is the topographic gradient between the point of interest and the stream course, measured along the hydraulic path. It was calculated taking into account the downslope difference in elevation and the downslope drainage distance between the point of interest on the hillslope and the stream network.

The drained volume (Vr) corresponds to the drained area at the point of interest, multiplied by the mean annual affective rainfall. This variable is an indicator of the upslope climatic and topographic condition that could control the wetness of soil.

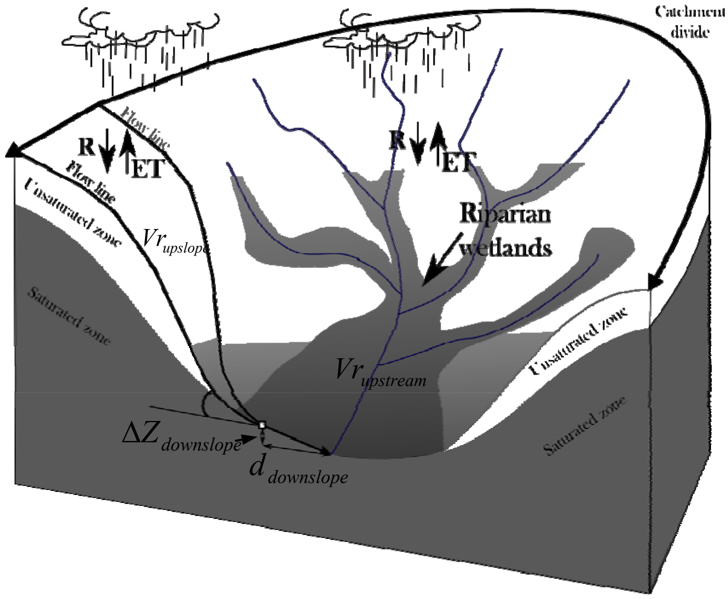


Fig.8. 3 Topographic and hydrological variables and indexes. R: rainfall depth, ET: evapotranspiration depth, $Vr_{upslope}$: upslope drainage volume, Slp_{local}: local slope, $\Delta Z_{downslope}$: downhill difference in elevation, $d_{downslope}$: drainage length to stream, $Vr_{upstream}$: upstream drainage volume.

The topograpchic index (TI) as defined above. A TI map of Brittany is available on Agro-Transfert Bretagne website to identify the potential wetlands.

The climato-topographic-index (CTI), as defined above, that corresponds to the previous TI modified by considering the effective drained volume (Vr) and the local slope (Slp_{local}). It had been validated in studies conducted by Merot et al. (2003), it mainly assumes that saturation is controlled by upslope conditions.

$$CTI = \ln\left(\frac{V_r}{\tan Slp_{local}}\right)$$

In order to test the increasing influence of stream order in the control of the wetland extension from upstream to downstream, the climato-topographic index was modified. The flatness of the bottomland characterised by the downslope difference in elevation ($\Delta Z_{downslope}$) and weighted by the volume drained by the stream (that is an indicator of the upstream-downstream location, $Vr_{upstream}$, and was calculated following the mono-directional modelling criteria) was included in a new index, called the ordinated climato-topographic index, OCTI, in the following way:

$$OCTI = \ln\left(\frac{Vr}{\tan Slp_{local}}\right) + cst \cdot \frac{\ln Vr_{upstream}}{\left(\Delta Z_{downslope} + 0.1\right)}$$

The value 0.1 was added to $\Delta Z_{downslope}$ to avoid a division by zero for sites without sufficient elevation resolution to distinguish riparian wetland elevation and stream elevation. A constant (cst), expressed in the opportune measure unit, was added to weight the mean influence of upslope and downslope variables.

Among this variables and indexes we need to define which of them will be considered indexes able to predict wetlands.

Three elementary variable will be considered as possible predicting indexes:

- $\Delta Z_{downslope}$
- $Slp_{downslope}$
- Vr

The local slope Slp_{local}, used in the OCTI, can not be considered an efficient indicator to identify hydric soils, because it is possible to find flat areas or areas with a low gradient at the top of the hillslope, where there is no shallow groundwater.

Three combined indexes are considered able to predict wetlands:

- TI: Topographic Index
- CTI: Clima-Topographic Index
- OCTI: Ordinated-Clima-Topographic Index

8.4.4. Definition of hydric soils

Hydric zones are generally defined as the areas where soils are affected by redoximorphic features in the topsoil horizon (<20 cm depth). The index's values (and cst for OCTI) were calibrate by giving the optimal delineation of the hydric soils, for the different indexes. For this aim it was hypothesized that soils with hydric class between six and nine are hydric soils. This choice was done by comparing the soil map 1: 25 000 with the TI map. Considering the classes six to nine let a more precise estimation of the hydric solis.

8.4.5. Calibration of the indexes

The calibration was based on the percentage of hydric soils that was forced to be the same for the predicted than the observed rasterized soil map. The quality of the prediction was assessed by the percentage of pixels that were well predicted compared to the total of the observed hydric soils. In this way it was possible to define the threshold of identification of hydric soils for each index. The second step was to estimate the Strahler order influence on the quality of the indexes' prediction.

8.5.Results

Results are first presented considering the distribution of the index for the actual hydric and non hydric soil. Then we define the best threshold of each index that allow to predict the hydric soil with the best ratio between over and under estimation. Fig.8. 4 shows the percentage of hydric soils predicted by each index.

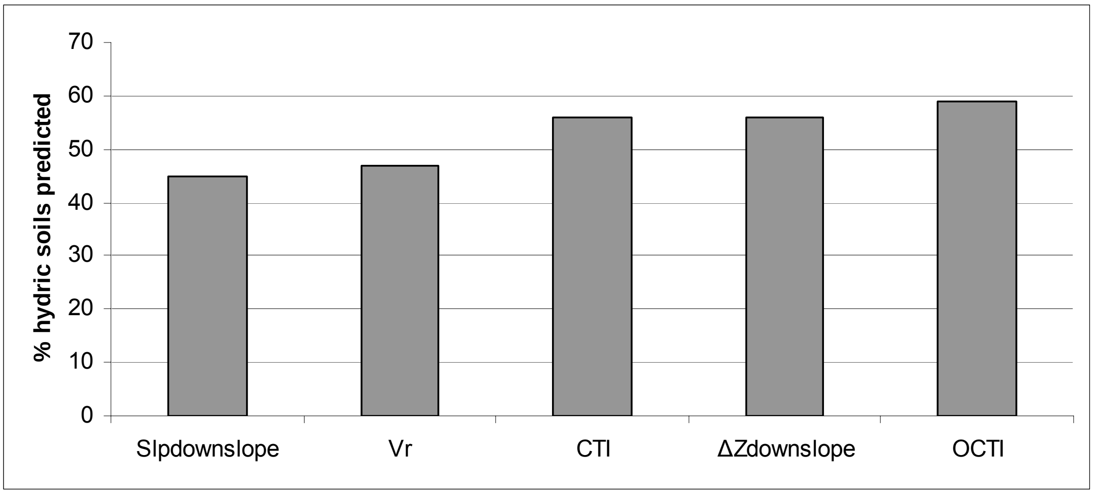


Fig.8. 4 Percentage of predicted hydric soils by topographical and hydrological indexes

In the Seiche basin, the downslope slope ($Slp_{downslope}$) ranges from 0% to 42%. For hydric soils it varies between 0% and 16% while for no-hydric soils it varies from 0% to 28%. The best $Slp_{downslope}$ threshold defined allow us to identify the 46% of the hydric soils is 1.5%. Nevertheless the ratio between overestimation and underestimation is 63%, that shows an underestimation.

The downslope difference in elevation($\Delta Z_{downslope}$) between the point of interest and the stream network, considering the whole catchment, reaches 88m with a mean value of 10m. For the non-hydric soils the $\Delta Z_{downslope}$ value ranges between 0 and 50m, while for hydric soils it ranges between 0 and 45m. The average values are respectively 11m for non-hydric soils and 6m for hydric soils. Fixing a threshold of 3.75 m allowed us to identified the 56% of the hydric soil.

The annual volume of effective rainfall drained by soils, $Vr_{upstream}$, reaches 140 106 m^3 considering the whole catchment, with a mean value equal to 360 103 m^3. Both for the hydric and the non-hydric soils the ranges between 0,25 and 135 106 m^3. The average value are 1117 103 m^3 for hydric soils and 55 103 m^3 for the non-hydric. The threshold that gives the best delineation of hydric soils is 2.8 103 m^3 with 47% predicted.

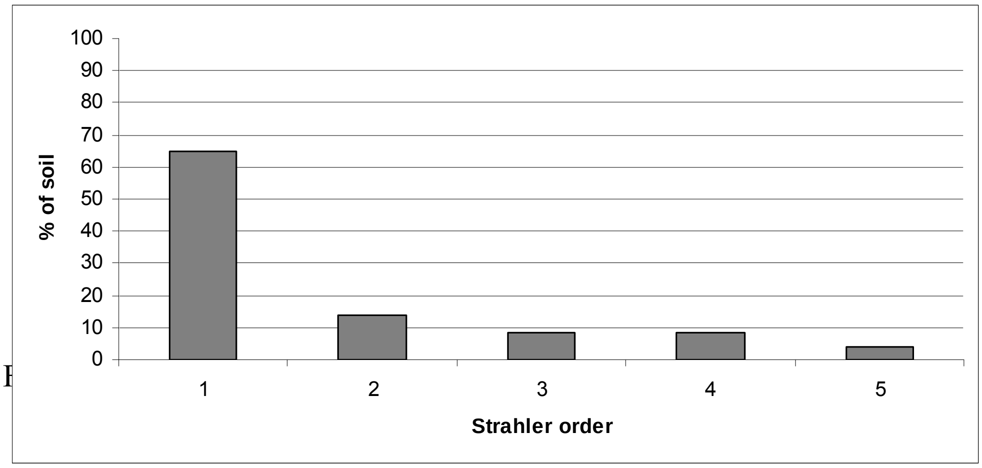
The climato-topographic-index (CTI) lets a prediction of the 56% of the hydric soils if the threshold is fixed at 5.29. The hydric soils have an average CTI equal to 7.5 while for the no-hydric soils it is 4. The respective ranges are 1,25 to 24 and 0,24 to 24. Considering the whole catchment CTI ranges between 0 and 24 with a mean value equal to 5.

For the ordinated-climato-topographic (OCTI) the constant value was chosen by iterative calculation that conducted to a constant (cst) equal to 1.7m. The OCTI values are between 0,88 and 336 in the whole catchment and the mean is 18,5. The hydric soils are characterized by an OCTI that ranges between 2,41 and 336 with a mean of 44, whereas for the other soils the range is 1,27 to 336 and the mean is 9. The threshold fixed to predict the 59% of the hydric soil is 8.28.

8.5.1. The influence of the Strahler order

The Seiche basin is a fifth order catchment. Considering the partial soil map, the 65% of the areas belong to the first-order subcatchments. When the order increases, the corresponding soil map areas decrease, for example only the 4% of the soil map areas are of the fifth order (Fig.8. 5 a and b) shows that considering the whole catchment most of the areas belong to the second Strahler order.

The low contribution of the high order stretches of catchment, if we consider the areas covered by the partial soil map, determines that the calibration of the different indexes is mainly determined by the characteristics of the first-order catchments, when the Strahler order is not taken into account. The percentage of hydric soils is quite stable in relation to the change of stream order: for the order 1, 2, 3, 4 the average percentage of hydric soils is 25%, while for the fifth order this percentage increases and reaches 38% (Fig.8. 6).



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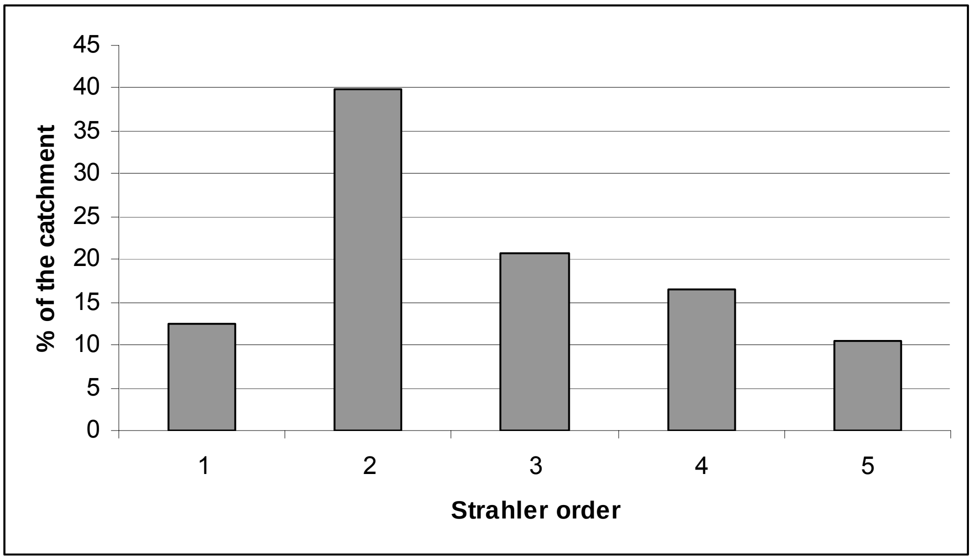
a)



b)

Fig.8. 5 Availability of soil data according to the Strahler order (a); Percentage of subcatchment areas according to the Strahler order (b)

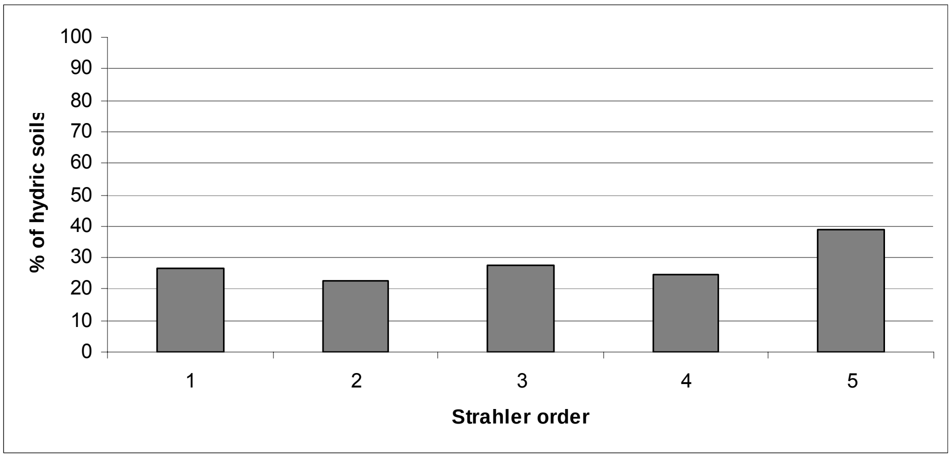


Fig.8. 6 Percentage of hydric soils in relation to the Strahler order

The prediction quality for the hydric soils varies with Strahler order of the catchment stretches. Fig.8. 7 represents the value of the prediction according to the order and Fig.8. 8 the quality and the stability of this prediction.

For the drained volume ($Vr_{upstream}$) the prediction quality decreases with the increase of the order.

The downslope slope ($Slp_{downslope}$) seems to be a good index to predict hydric soils but it is quite unstable if the catchment order changes.

The climato-topographic ICT and the ordinated climato-topographic ICTO indexes have the more stable behaviours through the five order.

The difference in elevation ($\Delta Z_{downslope}$) could not be considered as an reliable index because it overestimates too much.

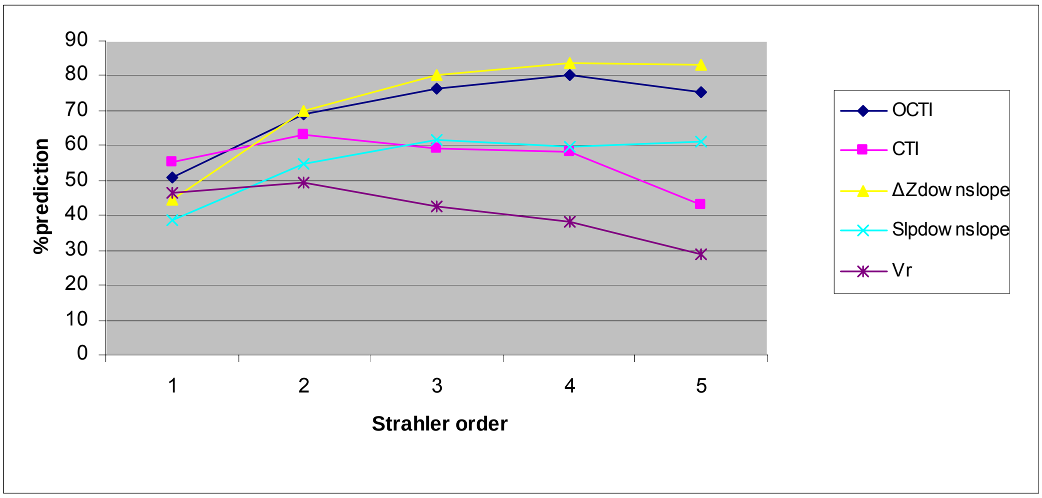


Fig. 8. 7 Prediction percentage of hydric soils for the different indexes in relation to the Strahler order

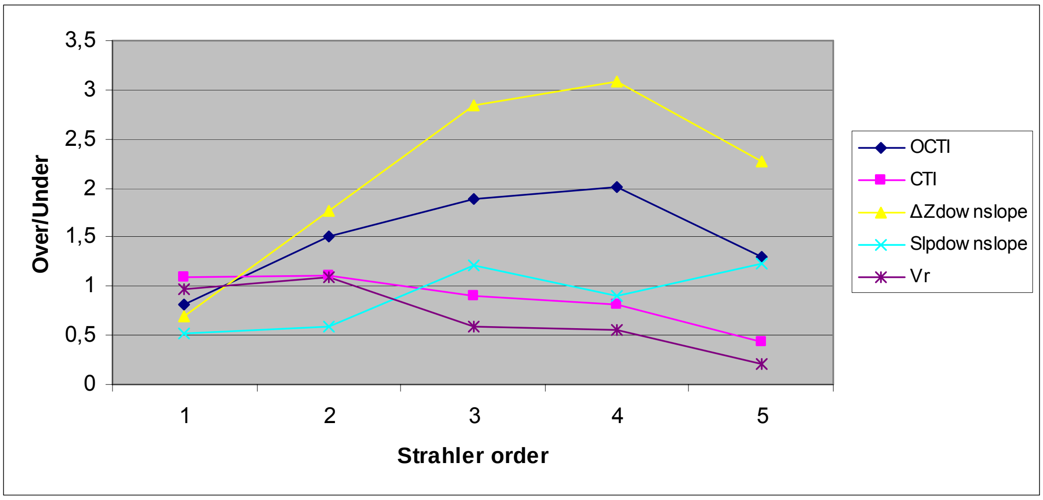


Fig.8. 8 Ratio between percentage of the over and the under estimated hydric soils for each index

A further consideration should be done analyzing the behaviour of the ratio between the thresholds fixed considering each order and the thresholds corresponding to the global catchment (Fig.8. 9). This ratio allows us to verify if the degradation of the hydric soils prediction is due to an underestimation of the soil saturation, which is proportional to the threshold. For each order, a threshold value less than the threshold chosen for the all catchment shows an underestimation of the saturation area.

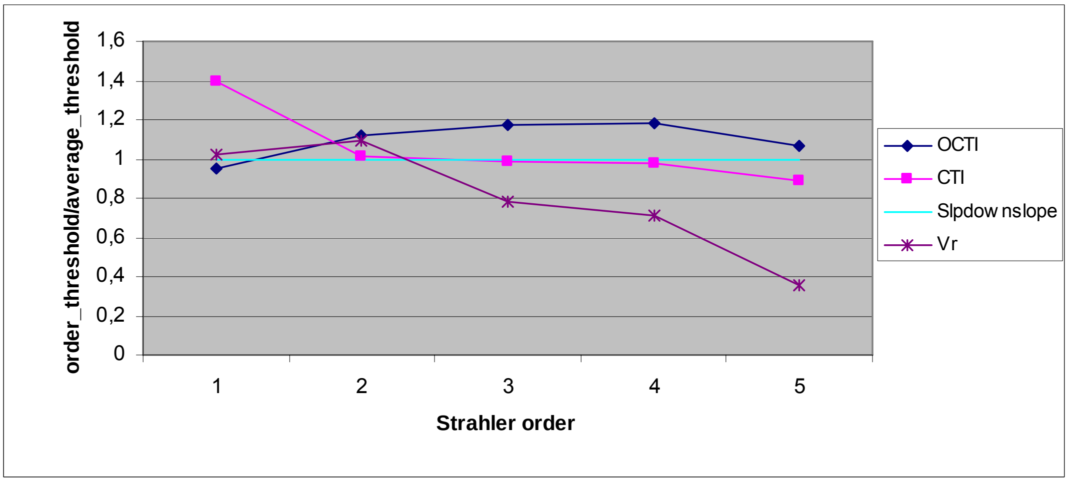


Fig.8. 9 Ratio between the threshold calibrated considering the Strahler order and the mean threshold defined without the Strahler order

Fig.8. 9 shows that the CTI and the OCTI present a stable ratio and it means that their prediction is reliable and stable in changing the Strahler order. Nevertheless the CTI seems to overestimate hydric soils for lower order.

The downslope slope presents a stable behaviour mainly for physical reasons: the lower the downslope is the more saturated the corresponding area usually is (Mourier et al., 2008).

8.6. Discussion

This study attempted to quantify the effect of the changes observed in the organisation and the functioning of the wetlands when the stream order increases. These changes gave a hierarchy for the topographic and hydrologic factors controlling the extension of the wetlands depending on the stream order.

The degradation of the wetland prediction with the climato-topographic index when the stream order increases, corresponds to an under-estimation of the wetlands area. This demonstrates that the convergence of fluxes, estimated by the climato-topographic index is insufficient to explain the extension of the riparian wetlands in high order streams. Two processes, not taken into account in the climato-topographic index, can contribute to the extension of the riparian wetland when the stream order increases: a contribution from the stream to the local saturation of the riparian wetlands or a more important retention of the water fluxes from the local hillslope.

The improvement of the delineation of the riparian wetlands in increasing the saturation probability in a way proportional to the water volume drained by the stream and by the inverse of the downhill difference in level represents these processes weighted by the water volume drained by the stream as an indicator of the stream order. The OCTI seems to be the most precise and reliable index to identify wetlands (Fig.8. 10).

This is consistent with different studies. At a local scale, Burt et al., 2002 and Vidon and Hill, 2004 showed a more frequent occurrence of an inversion of the hydraulic gradient between the stream and the riparian wetland when the topographic downhill gradient is weak. Even without the inversion of hydraulic gradients, the decrease of topographic downhill gradient corresponds to a decrease of the hydraulic gradients and a slower drainage of the riparian wetland, fostering an increase of the wetland area. These studies show the consistency of the downhill difference of elevation as an indicator of interactions between riparian wetlands and the stream.

On another side, the increase of the volume of alluvium and of associated wetlands referred to the width of the hillslope with the order (Schumm, 1977) demonstrates the increasing influence of the stream on the wetland functioning, and the contribution of the stream water to the wetlands during floods. This scheme representing the respective influence of the stream and the hillslope groundwater is consistent with the theoretical framework proposed by Brinson (1993), White (1993) and Tabacchi et al. (1998) and confirms the need to take into account the interactions between the wetlands and the stream to understand their functioning and to model their extension in a large catchment.

The need to identify variables and indexes that allow us to estimate hydric soils in different landscapes is confirmed also by the results obtained by Mourier et al in 2007. They show that linear correlations between topographic index and hydric soils extent are only significant for low Strahler orders (2 or 3). Although in upper catchment settings the extension of wetlands appears linked to variation of the topographic index, downstream topographic modelling that use simple index appears unable to explain hydric soils extent. These studies also show that the correlation between hydric soils width and the hillslope gradient for orders 2 and 3 is $R^2 = 0.30$ and $R^2 = 0.58$ respectively. For higher orders, the morphological variability is too large and no significant correlation can be estimated (R^2 between 0.01 and 0.02). The conclusions obtained by Mourier et al (2007) allowed us to suggest the use of downslope gradient in the topographic indexes to better predict hydric soils.

Although the present study shows that OCTI index better estimates hydric soils and it is the most stable index among those presented in this research according to the change of order, we still are in an initial phase. The calibration of the OCTI parameters is still empirical and based on the particular catchment considered. Studies conducted by Montreuil (2008) on the Scorff river basin (Brittany, France) confirm this lack of reliance in this new index, suggesting us that, probably, new parameters have to be considered.

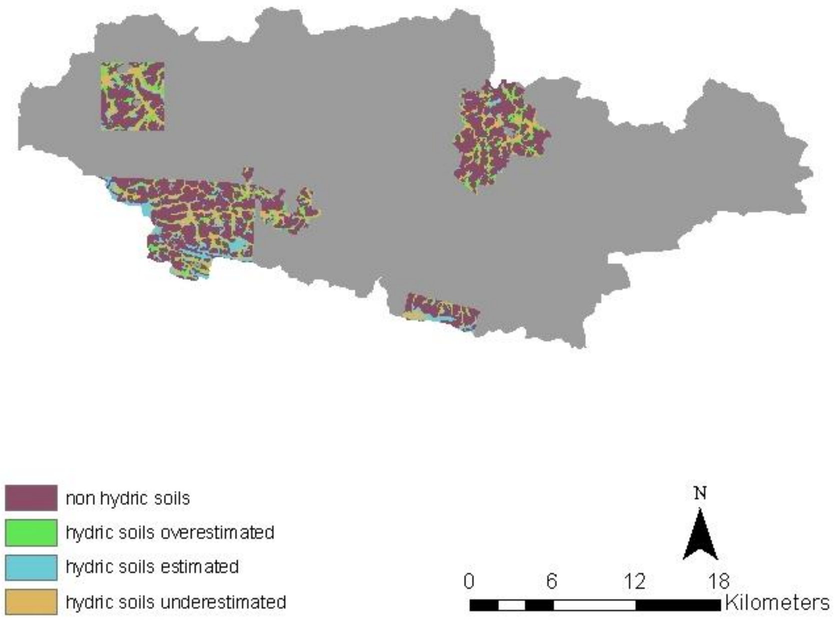


Fig.8. 10 Hydric soils predicted by OCTI

8.7. Conclusion

The objective of this research was to improve the prediction of wetland spatial distribution by using different topographic variables and indexes. Firstly the prediction ability of each index was compared with the actual hydric soils distribution. Secondly, it has been showed that the Strahelr order influences wetlands spatial distribution and the prediction of wetland spatial distribution. Fig.8. 7, Fig.8. 8 and Fig.8. 9 show how the different indexes are influenced by the catchment order. Thus it has been studied and developed the OCTI index that allows us to delineate wetland distribution in relation to the catchment's morphology with more reliability. The OCTI index considers the catchment order by using the water volume drained by the stream as a weighting factor. As showed also through the studies conducted by Montreuil (2008) on the Scorff basin, the OCTI has a better prediction quality than the other topographic indexes and variables have. Nevertheless the need of calibrating the constant value cst and the threshold proves that the OCTI index is still linked to local scale. Further studies have to be conducted to improve the OCTI efficiency and the possibility to use it in larger scale.

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9. Conclusions

It is recognized the need for appropriate tools to apply sustainable and coherent actions to planning and management, especially if the context is the environment or the landscape. Hence, an integrated framework should be developed. This integration should responds to questions such as which is the landscape and environmental context, which are the anthropic activities, which are the economic implications, which is the legislation. Generally, landscape and environmental management targets can best be met through an integrated, rather than independent, strategy.

Knowledge and data about soil, hydrology and land use are basilar to define a spatial phenomena and the agricultural context is largely influenced by human decisions and specific management. The location of the field, the farm buildings, the strategy of the farmer or existing landscape management programs influence land use and land management (fertilization, harvesting, etc.). The environmental context is related to the structure of the environment and natural processes that exist within. Specific properties and dynamics are relevant to assess the environmental impacts of agricultural practices.

Thus, also the main part of the variation in nutrients surpluses can be explained by management characteristics. As a matter of fact the reduction of nutrient surpluses will therefore be more effective if farmers try to optimise nutrient management rather than changing farm structure. Among the management variables, especially fertiliser reduction and improving operational management will significantly reduce the nutrient surpluses.

Operational improvements like more accurate feeding based on the features of animals and improvement of grassland management by better timing of fertilizing, grazing and harvesting and choosing a better way of conservation, are other examples of farm management.

Improving operational management will reduce the nutrient surpluses and at the same time should increase the financial returns. This option should therefore be tried.

Most environmental and farming management systems have to satisfy conflicting demands. Multi criteria decisions support systems provide a potential mechanism for resolving these conflicts. Particularly, considering farm activities, a whole farm approach is a powerful tool for the development of considerations and options to develop and to pursue: it could reveal the interactions between farm components.

Furthermore, a decision support system based on experts' knowledge is a tool to evaluate environmental pollution risk due to the agriculture non point pollution sources. The system has to take into account the uncertainty of the environmental domain resulting from incomplete information.

Fuzzy logic is one of the tools useful to have information about the possibility to reduce the risk of environmental pollution by ranking different anthropic activities according to their degree of hazard. Thus, basing on this kind of considerations, it would be possible to define actions aimed at mitigating the environmental impacts. The system could indicate areas in where efforts, to guide farmers towards more eco-compatible agricultural practices, should be concentrated.

These considerations and the results obtained through this thesis improve the knowledge of farming management systems and highlight new challenges for defining schemes to reduce environmental impact, taking into account also the regulations on environmental safeguard and animal welfare.

The use of analytical systems able to estimate through simulation the nutrient release from agricultural activities taking into account the cropping system, the livestock manure management and the pedoclimatic condition of the area is a challenge for building good farm management. This requires not only the availability of models able to deal with this type of information at a proper scale, but also the preparation of suitable databases to support the processing (as, for example, soil characteristics and the meteorological pattern).

Data availability, currency, and accuracy are critical in developing a management plan. Further refinement of the plan should focus on obtaining more recent, reliable and relevant data, improving the accuracy of data and maps, and refining the information available.

The development of these plans adds a new dimension to farm management by providing guide lines for the management of the interest area, by analyzing social, economic, environmental, and agricultural factors. The plans provide also information about the way to reduce the risk associated with, for example, manure application in areas of lower suitability.

An example of support plan for territory management in general has been developed in Brittany: the so called Territ'eau. It was developed in order to have a diagnostic tool and a reference model to manage the rural landscape. The project Territ'eau aims to update and share the knowledge about water and associated elements movements in a watershed and about the functional role of landscape structures. Territ'eau is characterized by a modular approach that fits in different situations (pedoclimatic, agricultural..). The functional, global and flexible aspect makes Territ'eau a very successful project. Agriculture and rural landscape gain, through Territ'eau, a way to make a hierarchy of the pollution risk and a means to organize actions to deal with issues in a given context.

An important project to develop in areas characterized by intensive agricultural and livestock activities, such as the Campania region (southern Italy), could be the implementation of a tool similar to Territ'eau. In this context a first step should be an action to intensify the relationship between farmers, politicians, researchers, scientists, professional figures. The project needs to be developed in a multifunctional team in order to have a complete and integrated data base and knowledge about the environmental, technical and law framework.

This thesis focused on the Campania region because it is a suitable site to introduce new and effective plans for improving environmental and especially farming management, in order to reduce the impacts of rural activities on the environment, first of all the agricultural non point pollution.

Besides the territory and the productive features of Campania, its administration has done several legislative actions in this context and seems to be ready to implement useful and multifunctional projects (http://www.sito.regione.campania.it/agricoltura/home.htm).

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