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CATCHMENT AGRONOMY: PROCESSES, MITIGATION MEASURES AND INDICATORS TO PROTECT WATER QUALITY

by

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ABSTRACT.

Rural landscapes are heterogeneous mosaics reflecting interrelations between anthropogenic, technical and environmental processes. Diffuse water pollution due to agriculture is taking place in this mosaic composed of fields and natural areas exchanging water, particles and solutes up to the stream water network. An agricultural catchment is therefore composed of nutrient sources, sinks or bio-reactors. The resulting spatio-temporal patterns contribute to modulate the nutrient export by modifying the buffer capacity of the catchment, i.e. the ratio between nutrient output and input, by diluting and trapping them along the different flow pathways. A new field of knowledge and engineering is emerging, 'catchment agronomy', coupling farm and crop management within the catchment area, firstly as the functional level of the landscape hydrology, and secondly, as a level controlling nutrient emissions to stream water. The catchment can no longer be considered as a sum of nutrient plot and farm outputs, but as a level where nutrients can really be management- and nutrient input and output-regulated. The paper presents the basic concepts related to nutrient attenuation and some of the major mitigation measures at catchment level.

While many scientific studies show the influence of agricultural landscape patterns on the water cycle and water quality, only few of these have proposed scientifically-based and operational methods to improve water management. 'Territ'eau' is a framework developed to adapt agricultural landscapes to water quality protection, using components such as farmers' fields, semi-natural areas or human infrastructures. The nitrogen (N) module is presented here. This enables the estimation of nitrate fate scores: i) at plot level, integrating them on the length of the crop rotation, taking into account environmental drivers (soil, climate) easily available by regional data sets; ii) at catchment level, identifying and qualifying functional semi-natural areas in terms of denitrification and dilution, and assessing their limits and functions, finally calculating nitrate fate scores per headwater catchment. This N module is integrated in a holistic method which allows us to objectivise the functions of the landscape components, and constitutes an established approach for adapting them to new environmental constraints. This framework helps in proposing different approaches for changing agricultural landscape, acting on agricultural practices or systems, and/or conserving or re-building semi-natural areas in controversial landscapes.

CONTENTS

Keywords: Catchment, landscape, water management, nutrient management, water quality, nitrate.

1. INTRODUCTION.

1.1. Rural landscape as a heterogeneous mosaic regulating diffuse pollution.

Diffuse pollution due to agriculture is taking place in a complex matrix of fields and natural areas exchanging water, particles and solutes up to the stream water network. Rural landscapes are heterogeneous mosaics (Forman, 1995) reflecting interrelations between anthropogenic processes (farm management, rural development and land conversion), technical processes (fertiliser and manure applications, crop succession, soil tillage) and natural processes (deposition, soil fixation, transfer and transformation). A chain of inputs, management, storage, transfer and transformation of nutrients acts at short distance (10 to 1 km), in and between, technical systems (fields, farms, groups of farms) and environmental systems (soil, groundwater, atmosphere and ecosystem) (Cellier *et al.* in press, for a review on nitrogen; Wang *et al.* (2004) for a review on phosphorus). The concept of landscape presents a practical dimension for planning, management, conservation and development of territories (Rapport *et al.*, 1998; Deffontaines *et al.*, 1994, 1995).

Nutrient sources, sinks or bio-reactors can be identified within an agricultural catchment (Haycock *et al.*, 1997). Interfaces between fields and streams can soften or short-cut nutrient fluxes. As an example, hotspots of nitrogen (N) deposition near animal housing, or phosphorus (P) accumulation in soil due to manure spreading or intensive grazing close to the farmstead, act as sources. Infiltration of overland flow at the foot of a hedgerow is a sink for water and solutes. A wetland where denitrification and P release occur can be considered as a biogeochemical reactor. These spatial features and their function vary in time due to landscape characteristics (extension of saturated areas), surface characteristics (soil surface conditions, grassland and arable lands) which strongly depends on the cropping systems and the farm-scale allocation of land use as well climate conditions. The temporal dynamics acts on a large range of temporal scales, events, seasons, duration of crop succession. The flow pathways are themselves modified (Aurousseau *et al.*, 2009). The resulting spatio-temporal patterns contribute to modulate the nutrient export by modifying what Viaud *et al.* (2004) have called the buffer capacity of the catchment, i.e. the ratio between nutrient output and input, by diluting and trapping them along the different flow pathways. The catchment can no more be considered as a sum of nutrient plot and farm outputs, but as a level where nutrient can really be management and nutrient input and output regulated.

1.2. Technical and environmental constraints of a rural landscape.

Technical constraints of farms partly determine location of land use, crop and agricultural practices. As an example Thenail (2002) and Thenail *et al.* (2009), studying and the interactions between farm functioning and landscape pattern in north-eastern Brittany (western France), has emphasised the strong spatial pattern of the crop mosaic of dairy farms, which is to a large extent determined by distance to farmstead. Land use is organised into approximate concentric circles around the farmstead: pastures grazed by dairy cows are

located as close to the farmstead as possible, because dairy cows move daily from the farmstead into the fields. Dairy cows density near the farmstead is reinforced when milking robots are used. A second circle consists of fields used for cash crops and forage. The outer circle consists of permanent grasslands grazed by heifers, extensive lands or woodlands, which require little management. This applies to many locations in north-western Europe. By contrast, crop allocation in crop farming systems or intensive breeding farming systems are expected to be less controlled by the distance to farmstead. More generally, the degree of spatial organisation varies according to the farming systems.

Environmental constraints have also such effects. Crops requiring a lot of water for growing, such as maize, are often preferentially located in a bottom slope position. On the other hand crops harvested late in autumn and catch crop sowing cannot always be easily achieved due to a lower soil trafficability. Cerdan *et al.* (2002) have shown that a small proportion of crop-lands contribute effectively to loads of suspended matter and nutriments into the stream, according to the flow pathways. Water and pollutants that flow at the soil surface, generated by low infiltration rates in soil due to soil surface conditions (roughness, soil cover by plant and crusting), are generally re-infiltrated at the bottom of the slope. The heterogeneity of the soil surface conditions determines the total runoff at the outlet of the catchment. Controlling soil surface conditions is efficient in surface flow dominant hydrological systems, where the cropping systems vary in space and determine various soil surface conditions (Papy and Boiffin, 1988).

1.3. 'Catchment agronomy' as a new field to manage activities and water quality.

A new field of knowledge and engineering is emerging, being referred to as 'catchment agronomy', coupling farm and crop management within the catchment area, firstly as the functional level of the landscape hydrology, and secondly, as a level controlling of nutrient emissions into stream water. It concerns:

a) all countries where agricultural mosaic are complex such as in Western Europe, where emission from the landscape can regulate the emission of nutrients;
b) medium size catchments (10-50 km²) where fields and interfaces are already clearly identified, and are the main scale for agri-environmental management;
c) catchments in which the ratio of agricultural lands is higher than a threshold evaluated to be about 50% present highly variable nitrate loads. This variability is explained by the spatial and temporal distribution of N and P inputsover the landscape as well as to the spatio-temporal patterns of the semi-natural and agricultural areas (Strayer *et al.,* 2003; Burt and Pinay, 2007; Kronvang *et al.*, 2005).

Catchment agronomy consists in managing N and P export, taking into account environmental and technical constraints, by:

a) decreasing the nutrient *balance* on catchment;
b) increasing the *heterogeneity* of the landscape, to optimise the relative position of source and sink and attenuate the effect of nutrient hotspots especially during critical periods;
c) introducing *buffer areas* as closer as possible to the sources (P), but also to vulnerable areas which can be streams, lake or wetlands;
d) managing sustainably P stocks in soils to avoid P release from buffer areas (grass buffer strip; see Dorioz *et al.* (2006)) or N stocks in soils and ground waters to avoid N flush during critical periods, or even side effect or interactions between nutrients (C, N, P).

Finally catchment agronomy aims at increasing the nutrient efficiency at catchment level, decreasing as much as possible N and P emission, without side effect or accumulation within the catchment.

Tools are increasingly being developed to simulate and predict the effect of agricultural practices on nutrient emissions in water bodies. Among different models and applications to evaluate the effect of agricultural landscape features, we can cite: SWAT (Neitsch *et al.*, 2002) largely used to different sites, pollutants and landscape features (Ouvang *et al.*, 2008; Wang *et al.*, 2008); TNT (Topographic-based nitrogen transfer and transformation model) used to assess the effect of riparian wetlands (Beaujouan *et al.*, 2001), agricultural practices (Beaujouan *et al.*, 2002) and hedgerows (Viaud *et al.*, 2005) on discharge and/or nitrate emission in water bodies.

However these models cannot predict pollutant emission on any operational catchment:

a) the required data are not generally available;
b) modelling remains a task for specialists;
c) the represented landscape structures and functions are 'archetypal', i.e. over simplified regarding the diversity of situations;
d) environmental and technical constraints, decision rules of the farmers are generally not represented so that the model fails in modelling nitrate emission in real situations.

Indicators have been proposed as an alternative. Several nitrogen and phosphorus indicators at catchment level (de Vries *et al.,* 2005, Heathwaite *et al.*, 2000, 2003, 2005; Bockstaller *et al.,* 2008; Gascuel-Odoux *et al.*, 2009) have been developed to help stakeholders assess the functions of the different features of an agricultural landscape in terms of water quality and to build up major recommendations for catchment management. Such approaches are necessary: various structures act in various conditions, regulating various pollutants, so that these approaches have to objectivise the effect of landscape features within any agricultural catchment.

2. BASICS OF NUTRIENT ATTENUATION AT CATCHMENT SCALE.

Landscape features strongly affect nutrient export and dynamics in agricultural catchments, modulating storage, atmospheric emission, retention and finally regulating the export regime to water bodies (Gascuel-Odoux *et al.,* 2011). Owens *et al.* (2007) cite four main processes affecting buffer interception efficiencies for nutrients, namely by:

a) infiltration according to area, soil permeability, root structure and feature dimensions;
b) adsorption according to contact time, kinetics and soil chemistry (organic matter, Fe oxides etc);
c) sedimentation dependant on flow dynamics, vegetation structure, slope and particle sizes of incoming material, and
d) dilution of runoff.

Once the feature has retained nutrients other processes may act to transform them. An important action for this consists in increasing retention time to allow biogeochemical processing.

2.1. Nitrogen and phosphorus comparison.

Nitrogen and P respond differently to landscape structure. Even if the sources are globally the same, the relative contribution of annual inputs to the nutrients loads in water bodies, the transfer flow paths and the effects of landscape buffers, are highly different in intensity, and associated with varying modes and timings of delivery. This is governed by the nature and location of the source, the hydrological flow pathways and connectivity between the source and water bodies. Nutrient inputs may be continuous or cumulative (due to nitrate (NO_3) enriched ground waters, or P enriched soils), annual/seasonal (associated with fertilisation or biomass recycling), or stochastic (from infrequent erosive storm events). Inspired by the reviews of Haag and Kaupenjohann (2001) and Sharpley *et al.* (2001), Figure 1 (overleaf) compares N and P transfer at catchment level.

As nitrate is very mobile and easily transferred over the landscape, a buffer interface is efficient for nitrate only if it combines a low flow velocity, a high biological activity and a source of carbon for denitrification (Machefer and Dise, 2004). Riparian areas and wetlands are particularly efficient in nitrogen removal. Topographic indices are good indicators to delineate riparian wetlands (Merot *et al.*, 2006), and, therefore, to evaluate their extension and buffering effect at catchment level (Beaujouan *et al.*, 2002). Montreuil *et al.* (2009) have shown a denitrification of about 1 mg/litre/percent of wetlands in catchment area. Long residence time, appropriate soil moisture conditions and stimulation of microbial activity can activate denitrification. Long term or irreversible removal can also achieved by immobilisation in perennial biomass or refractory organic matter.

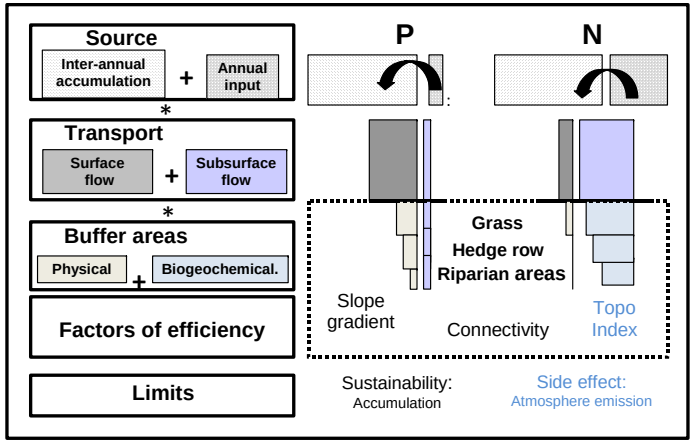


Figure 1 : *Different sources of nutrients can be associated with varying modes and timings of delivery.*

In contrast, phosphorus can be easily trapped even during short and relatively rapid transfer, via sorption of dissolved P and physical retention of particulate-P. In the case of particulate-P, and more generally of particle-bound pollutants such as ammonium and organic nitrogen, two complementary major processes control the efficiency of landscape buffer: infiltration which reduces overland flow, which also results in the retention of dissolved forms of N and P, and filtration due to vegetation cover and surface roughness which reduce flow velocity. These two processes decrease the sediment transport capacity of overland flow, and promote particle deposition. Slope gradients control these two processes.

2.2. Buffer capacity and side effects.

As a consequence, the buffer capacity is significantly different for N and for P. For nitrate, usual values of relative emission (output/input) range from 0.1 to 0.5 with an increasing trend from northern to southern countries which are therefore more efficient at mitigating N export (Billen *et al.* in press), whereas for insoluble phosphorus they are much lower, often below 0.05. For example, in Brittany, western France, in a region strongly impacted by diffuse pollution, the relative emission is about 0.7 for nitrate and less than 0.01 for total phosphorus (Aurousseau, 2002). The buffer capacities of grasslands is discussed by Vertès *et al.* (2010), highlighting the link between biophysical processes and agronomic functioning at farm level. Uncertainty is much higher for P than for N, due to a higher variability of concentrations and the generally lower sampling for P than for N, which contribute to a higher uncertainty of annual fluxes.

Side effects have also to be underlined. It should be remembered that, as no atmospheric loss exists for P, a small delivery ratio actually reflects the increasing saturation of terrestrial buffer features with reactive P and it is

unlikely that this can be infinitely sustained (Dorioz *et al.*, 2006; Stutter *et al.*, 2009; Hoffman *et al.*, 2009). A small proportion of this retention is really long-lived (decades) because P and sediments carrying P accumulated on the soil surface can lead to a possible saturation of the buffer. A major side effect for N is the emission of nitrogen oxides to the atmosphere (Grofman *et al.,* 1998).

3. TYPES OF MITIGATION MEASURES AT LANDSCAPE LEVEL.

Types of mitigation measures at landscape level have been exhaustively presented in Gascuel-Odoux *et al.* (2011) and Dorioz *et al.* (2011) by distinguishing measures in land use and land use patterns, and measures in landscape management.

3.1. Land use and land use patterns.

Four types of mitigation measures in land use and land use patterns can be identified and presented in a gradient of difficulties and a necessary socio-economic dimension:

- *Change agricultural land use patterns* by relocating land use or crop, to increase sink domains between source domains (Beaujouan *et al.*, 2002; Martin *et al.,* 2004; Jouanon *et al.*, 2006).
- *Change cropping systems and agricultural practices*, to improve nutrient storage in soils, by developing for example agroforestry, and to avoid hot spots of nutriment emission (Vinten and Dunn, 2001).
- *Change agricultural land use by 'extensification' options* on land use or crop systems, to decrease nutrient input or increase nutrient storage in soils, considering for P a sustainable storage, mainly by incorporating nutrients in soil organic matter or increasing duration of plant cover.
- *Land use changes from agriculture to non-agriculture use* by afforestation of agricultural lands, to decrease the nutrient budget of the catchment, simply by diluting the nutrient budget of the agricultural lands.

3.2. Landscape management.

Mitigation measures in landscape management can be discriminated regarding the object or landscape component they 'join' together and the fluxes which flow through them. Not all the interfaces of a landscape are buffers. Their state is crucial: they act as buffers if they present a proper structure and are properly managed. If not, they directly connect a source with a water body, and this propagation can include some cascading effects. These interfaces are the following ones:

- *Interfaces between farm water and stream water*. Such interfaces regulate the surface flow connectivity between farm infrastructures that produce waste water and surface water. To attenuate the resulting transfer of nutrients it is desirable: 1) to minimise, the volume of waste water produced at farm level or to regulate the period of losses by water storage (e.g. by tightening slurry storage regulations or separating farm roof water

from runoff); 2) to introduce a buffer (such as a farm pond) which disconnects farm infrastructure, mainly consisting of impervious surfaces, from field drains or surface water.

- *Interfaces between livestock locations and displacements, and stream water*. Direct access of the animals to the stream can damage the river bank, significantly contribute to bank erosion and/or allow a direct flow connection with hillslope erosion (Lefrançois *et al.*, 2007). Direct pollution with organic and inorganic nutrient and faecal contaminants is possible. Mitigation options aim to transform these interfaces into physical barriers between grazing animals and rapid hydrological pathways.
- *Stream water boundaries.* These interfaces, located along permanent streams and ditches, are called riparian buffers and they act differently according to their hydrological conditions: 1) under saturated conditions, there are the riparian wetlands or wet meadows, the objective of water quality management is to maintain, manage or restore them (Gilliam *et al.*, 1997; Fisher and Acreman, 2004), 2) under unsaturated conditions, the objective is to create and manage vegetated buffers strips (Borin *et al.*, 2010; Stutter *et al.*, 2010). In all cases, it is important to maintain and manage a vegetated strip on the river banks.
- *Field boundaries.* Such interfaces control the connectivity from plot to plot regarding surface and subsurface flows. These interfaces are very diverse: simple field boundaries more or less vegetated, hedges and hedgerow, or terraces, level out-fields…., but all are outlets of plots. Properly managed or redesigned these interfaces can become a type of, or be transformed into, classical vegetative filter strips and so contribute to a buffer effect through: 1) attenuation of fluxes from emitting farm fields; an effect which partly depends on the vegetation type and the possible interaction with shallow groundwater; 2) limiting flow concentration and the resulting cascading effects on soil erosion.

The function of all these mitigation options related to N and P emission into water bodies, and dedicated to landscape level, are detailed in Gascuel-Odoux *et al.* (2011), and Dorioz *et al.* (2011).

4. INDICATORS DEVELOPED IN 'TERRIT'EAU' FRAMEWORK FOR LANDSCAPE NITRATE MANAGEMENT.

Despite the well known effect of these structures, operational tools to quantify their effects on operational catchments are scarce. We present below an example of such a tool for nitrate management which is included in a framework 'Territ'eau' (Gascuel-Odoux *et al.*, 2009).

4.1. 'Territ'eau' framework and its nitrogen module.

'Territ'eau' is an operational framework dedicated to medium-size catchments (10 to 100 km²) where stream waters are degraded by diffuse agricultural pollution and where remediation operations are planned, and therefore an

agro-environmental diagnosis is required to identify the origin of stream water pollution and options for mitigation. This framework includes technical and environmental databases and references, and a diagnostic method for the catchment. This diagnosis is carried out based on regional databases and detailed field observations on the agricultural landscape, including fields, field margins, semi-natural and stream-bordering areas. It integrates in time permanent landscape features and agricultural activities on the crop rotations. It takes into account the major agricultural diffuse pollution into water bodies. In carrying out such a functional and holistic diagnosis, this framework promotes the participation of the inhabitants of the territory such as farmers, community staff and the wider public. It is described on a numeric support[1] which is highly visited in France. Finally, it facilitates a comprehensible and functional view of the catchment, using the concept of contributing areas to objectivise the effect of cropping systems and landscape features on stream water quality. This is a basic and important point, since landscape features are generally considered with little understanding, and controversy even remains regarding the water quality issue (Viaud *et al.*, 2004; Merot *et al.*, 2006). This comprehensible and functional view of the catchment is based on scientific results that are expressed in an understandable way so they can be appropriated by the end users.

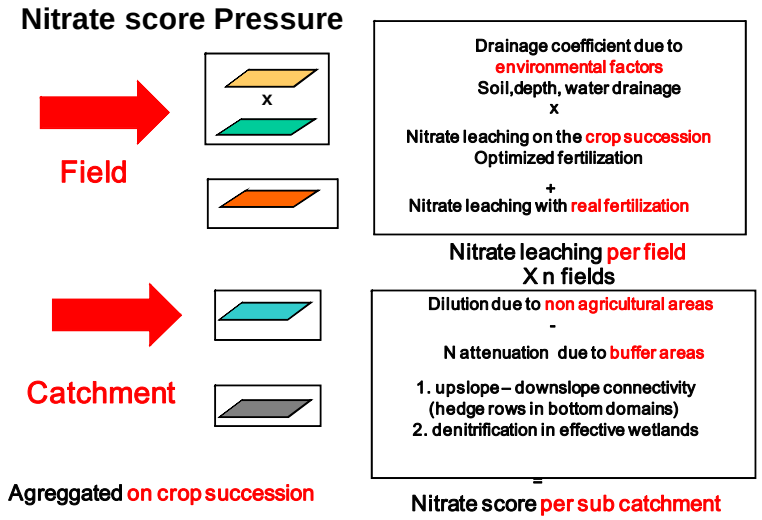


Figure 2: *Estimation of nitrogen leached at catchment scale.*

The nitrogen module aims at estimating nitrate emission into stream water, integrating the length of the crop rotation, and the catchment area (Figure 2). At plot level (Figure 2), nitrate leaching is estimated per plot, by a yearly mean

[1] http://agro-transfert-bretagne.univ-rennes1.fr/Territ_eau/accueil.asp

calculated from the duration of the crop rotation. This estimate combines three factors:

1) an estimate of the nitrate leaching due to the crop system, under balanced fertilisation and assuming a yearly mean value for each crop;
2) a transfer coefficient based on environmental factors (soil, climate); and
3) an estimate of the surplus of fertilisation by the farmer. At landscape level, dilution and denitrification effect of non agricultural lands and wetlands, respectively, are estimated to produce semi-quantitative estimation of nitrate emission in water bodies, called here the nitrate fate score, corresponding to different classes of N emission, each one expressed in kg N/ha.. All these steps are detailed below.

4.2. Plot level.

At plot level, an indicator predicting nitrate leaching integrated with the crop rotation, easy to implement, taking into account the soil and climatic conditions as well agricultural practices, has been established. The three steps of calculation are detailed in Figure 3: firstly, nitrate leaching during winter period is calculated under bare soil (LIXIM model, Mary *et al.*, 1999), agricultural practices being documented from farmers' records, and mineral N in soils from local reference values; secondly, a leaching coefficient allows us to take in account location (drainage maps); thirdly, rotation effects (effects of crop residues, grassland destruction and possible presence of catch crops) are integrated.

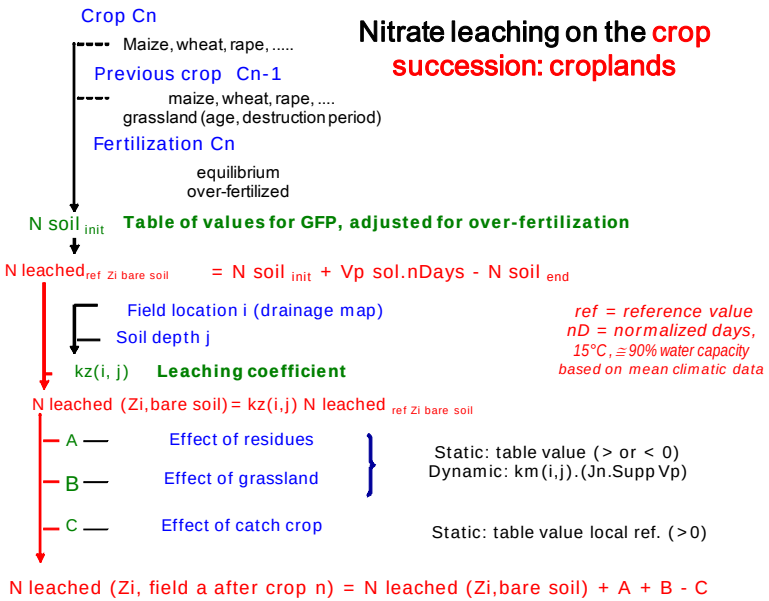


Figure 3: *Details on the estimation of N leached at field level.*

Nitrate leaching is calculated using mineral N balance, from crop harvest to the end of drainage period, as follows:

$$Nleach = SMN_i + D_n.V_p - SMN_f + Effect\ (crop\ residues) - Effect\ (catch\ crop) + Effect\ (grassland)$$

where:

- SMN_i is the amount of mineral N in soil at autumn (1st Oct.).
- SMN_f is the amount of mineral N in soil at end of winter (1st March).
- D_n are normalised days (15°C, soil at water capacity) calculated on N balance duration.
- V_p is the potential mineralisation rate of soil organic matter, expressed in N/ha/D_n.
- *Effect (crop residues)* quantifies the increase (net mineralisation) or decrease (net immobilisation) of nitrogen due to crop residues.
- *Effect (catch crop)* quantifies the decrease of leaching due to catch crop or crop regrowth.
- *Effect (grassland)* quantifies the supplement of N mineralisation due to grassland destruction.

SMN_i values are estimated using regional references. It varies with crops and place in rotations. Considering well adjusted fertilisation, values vary for crop rotations from 55 to 65 kg N/ha after cereals, maize and rape; for ley-arable rotations, they increase after grassland destruction (75 to 150 kg N/ha). After vegetable crops, they vary from 30 to 85 kg N/ha. This indicator is calculated for a high drainage area as a reference, since a complete leaching on the soil profile whatever the soil depth is assumed in this situation.

A leaching coefficient is determined for other situations, to calculate the ratio of N effectively leached. The nitrate remaining after the drainage period is assumed to be taken up by the following crops. This coefficient is based on the annual water balance (Precipitation minus Potential EvapoTranspiration, P-PET) during the recharge period as well as on soil properties. If P-PET is higher than a threshold value, the leaching coefficient is 1. If not, soil hydromorphy and depth are the two criteria used to determine it. Leaching coefficient is determined from simulations with STICS model (Brisson *et al.*, 1996) on 16 climatic years. It varies from 0.72 to 1.

The effect of over-fertilisation is taken in account in SMN_i value, considering that 50 to 80% of N in excess is added to the basal value of SMN_i. In the case of major under-fertilisation, SMN_i value is decreased from 20 kg N/ha. Cumulative N mineralisation is the product of daily mineralisation rate (V_p) by the number of normalised days, on the whole period considered to calculate the N balance. Thanks to moderate variations of these two parameters in the region, mean values are used for calculations, i.e. with 0.65 kg N/ha/D_n and 78 for V_p and D_n, respectively.

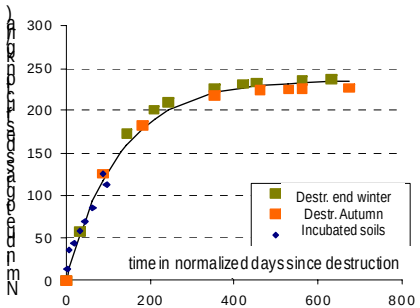
For grasslands, 4 classes of losses were considered, from the relationship between leaching and stocking rates in well managed grasslands (permanent or in ley-arable rotation) (Vertès *et al.*, 2007). Total N input should be a better indicator (Eriksen *et al.*, 2010) but is difficult to document (N fertilisation, N

fixation, N from animal returns), while stocking rate classes can be easily deduced from farmer's registers and local references on fodder systems.

The decrease of leaching due to catch crops is quantified using a regional data set. It depends on its nature (sown catch crop or crop regrowth), sowing date and time of destruction. Values vary from 15 kg N/ha (catch crop sown in mid-October after maize), to 75 kg for well-grown catch crops after a cereal. Short duration catch crops can decrease leaching from about 30-40 kg N/ha.

Incorporation of crop residues can modify N leaching. A decrease is considered for residues with high C:N ratio (>25) and an increase for residues with low C:N ratio. As cereal straw is usually harvested, its effects are not considered. For maize which presents a high C:N ratio, the decrease is 10 kg N/ha. For other crops, harvested in mid-summer, a standard positive value varying from 10 kg N/ha (rape) to 20-30 kg N/ha (fodder peas, potatoes) is applied.

Grassland destruction, in ley-arable rotations including more than 4 years of grassland two effects of grassland are considered: the basal mineralisation rate is slightly increased compared to crop rotations (0.71 vs 0.65 kg N/ha/D_n); the extra mineralisation due to grassland destruction is included in a dynamic way, based on the relationship between N mineralisation and time (expressed in D_n) (Figure 4).



	Vp	Year 1	Year 2
	Destruction		
Grassl. age	February	April	
2-3 yrs	0,22	0,27	0
4-5 yrs	0,33	0,42	0
6-10 yrs	0,45	0,58	0,10
>10 yrs	0,49	0,63	0,15

Figure 4. *Variations of N mineralisation due to grassland destruction versus normalised days (from Besnard et al. (2007), and an expert table used in nitrogen module of Territ'eau framework (Vertès et al., 2007).*

This N module tool allows us to calculate nitrate leaching after every crop for every area according to agricultural practices, then to combine successive crops into rotations. As an example, the main crop rotations with respect to nitrate leaching scores are illustrated in Figure 5.

The leached N calculated for crop rotations managed with very good agricultural practices, in high drainage area vary from 15 to 70 kg N/ha/year. Moderate losses occur with annual crop rotation including efficient catch crops and grasslands with moderate stocking rates. The highest losses relate to rotations including bare soils (no crop nor winter cereal sowing).

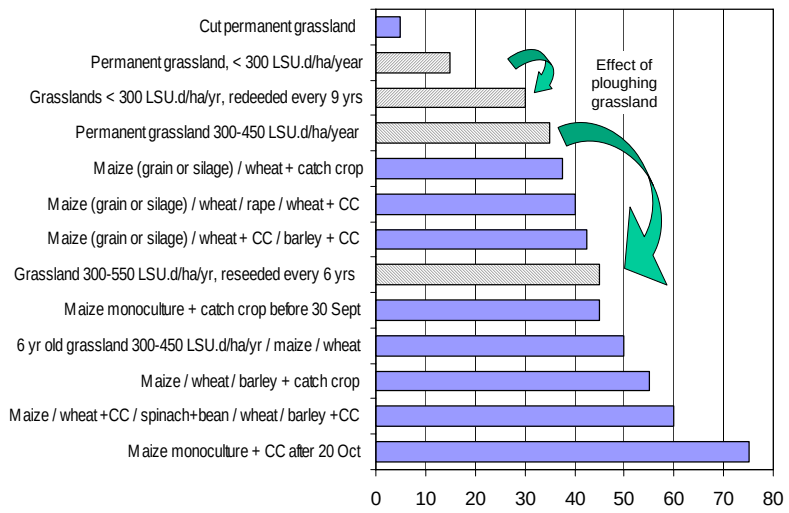


Figure 5: *IN_{leach} for main crop rotations, with optimal fertilisation equilibrium, with deep soil in well-drained areas and good conditions for grassland growth in summer,*
*(CC = catch crop - LSU.d = stocking rates as LiveStock Units*days).*

The agricultural practices of the farmers are identified by a survey. Some data are easily available from the farm fertilisation booklet required since 2006 in the regional directives in France.

4.3. Catchment level.

The nitrate emission at the outlet of a head-water catchment is only calculated if the survey covers more than 60% of whole surface area. The surface area used to compute the mean nitrate leaching per catchment is the whole area including agricultural fields and non-agricultural land (woods, set-aside, roads and housing, etc.) assuming zero leaching under non-agricultural areas.

Losses due to denitrification processes are a function of the existing wetlands (Gascuel-Odoux *et al.*, 2009). This aspect is developed in the Territ'eau framework from an analysis of the literature, as well as experimental and modelling results (Bidois, 1999; Durand *et al.*, 2000; Beaujouan *et al.*, 2001; Montreuil and Merot, 2006). Three criteria allow the user to determine the proportion of well managed wetlands:

a) the type of vegetation, considering its natural and cultural heritage value, i.e. trophic status, use and fertilisation if it is a grassland,

b) the continuity of the upslope field margins, and

c) the bypass flow that deflects denitrification.

Then two criteria assess qualitatively the denitrification per sub-catchment: the proportion of effective wetlands per sub-catchment and the proportion of well managed wetlands. Finally, these criteria indicate if the area of the

efficient wetlands is sufficient to induce any effect on nitrate fate, and if so, which management would optimise it and what recommendations should be formulated. The semi-quantitive score related to the denitrification can up- or down-grade the N fate score by one to two classes, according the conservation and the potential of denitrification of the wetlands. The results are illustrated in Figure 6. The semi-quantification of the denitrification in the wetlands highlights their buffer capacity, but also defines their limit in the case of high N surplus on the hillslope. If the N fate score is too high, the conservation and the management of the wetlands cannot solve the problem of the nitrogen excess due to agricultural practices. Conversely, if the nitrate fate is low or moderate, the conservation and the management of the wetland can contribute to the improvement or deterioration of the N fate.

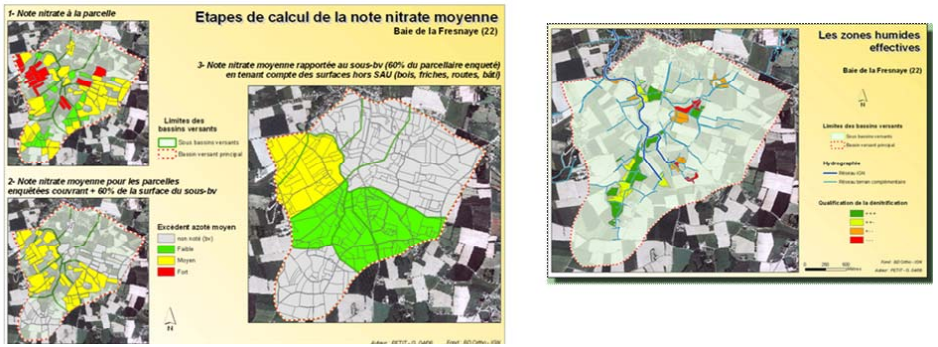


Figure 6: *Nitrate leaching per plot, head-water catchment and denitrification of nitrogen in riparian wetlands.*

Finally, the semi-quantitative assessment is distributed according to 3 classes regarding the nitrate target in the Water Framework Directive (50 mg/litre). The green class corresponds to a correct management regarding this target. The orange class corresponds to a situation for which an improvement in landscape infrastructure could be sufficient to get the target. The red class corresponds to a situation for which any improvement in landscape infrastructure would be insufficient, and consequently, the agricultural system has to change. In the coastal region, where the ecosystem required much lower targets, such an expert table can be easily adapted.

This diagnostic may be used to make land management recommendations: location of the critical plots, and on these plots, the possible solutions which can be related to the crop succession or to the fertilisation; location of the riparian zones where the N fate can be improved by reducing the fertilisation, or protecting the riparian zones from the upslope by re-building hedges or closing by-passes. These recommendations can be combined, and few alternatives reaching a target value can be proposed to the farmers.

5. CONCLUDING REMARKS.

The current approach Territ'eau is original in several aspects. Numerous decision tools have been developed at plot and farm level, yet they are still rare at catchment level. The diagnostic is holistic but not exhaustive: it provides stakeholders with a spatial view and diagnosis of the main hydrological and biogeochemical processes in the catchment. This method improves the dialogue with farmers for the diagnosis and identification of solutions, because a large amount of local and regional data, such as catchment area delineations, field maps, slope distribution, potential wetlands, etc., is collated before going into the field and visiting the farmers. Availability of the data improves the dialogue and ultimately the sense by the end users of belonging to the territory. Scenarios for landscape management offer a wide range of solutions that may concern agricultural practices or even agricultural systems, on the one hand, as well as the location and extent of the semi-natural areas, on the other. These are evaluated not only from a water quality perspective but also considering other environmental impacts such as biodiversity and amenity value. More generally, this approach opens up discussions on ways to find appropriate solutions, which may include increasing natural spaces, improving landscape features and their management, as well as changing agricultural practices or systems. These discussions are open and negotiable (Vidon *et al.*, 2010). No stakes or legal constraints are considered in delimiting areas of interest such as woods or wetlands. Moreover, it allows stakeholders to be guided in agricultural landscape management by arguing the choices on a scientific basis, which is an important point in controversial areas. Generally, according to the final objective of improving water quality, changing agricultural systems or landscape features is viewed here as supplementing other environmental or productive functions as well as human and physical constraints. An intensive use of GIS (Geographic Information System) before and during field work, to implement as early as possible the easy access to available data such as DEM (digital elevation model), the topographic index defining the potential riparian wetlands based on a topographic analysis (Merot *et al.,* 2006).

More generally, catchment agronomy is important because of the noisy response of the water quality due to the climatic variability and the heterogeneity of the landscape in space and time. Methods and tools integrating activities in space and time are therefore of major help in assessing the effect of agricultural activities and of setting objective mitigation options. Co-built operational tools can increase current relation between environmental scientists and society in the framework of the water policy (Mérot, 2008).

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