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# Assessing nitrogen losses after sewage sludge spreading: A method based on simulation models and spreader technological performances

T. Pacaud<sup>1\*</sup>, M. Pradel<sup>1</sup>, M. Cariolle<sup>2</sup>

<sup>1</sup> Cemagref, UR TSCF, Domaine des Palaquins, F-03150 Montoldre, France

<sup>2</sup> Institut Technique de la Betterave, 45 rue de Naples, F-75008 Paris, France

\* Corresponding author: [thomas.pacaud@cemagref.fr](mailto:thomas.pacaud@cemagref.fr)

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## Abstract

A study is carried out in order to discriminate different spreading technologies and improve them regarding their environmental performances. We define 45 sewage sludge spreading scenarios covering a wide range of situations in France. Several models are used to (i) assess nitrogen losses due to sewage sludge spreading and (ii) calculate additional flow resulting from the technologies performances such as spatial distribution heterogeneity, application rate accuracy and soil compaction of the spreading machine. NH<sub>3</sub> volatilisation due to the spreader performances is generally low and the highest for splash plate technology. Additional emissions are mainly caused by application rate accuracy problems. It is not possible to link NO<sub>3</sub> leaching to the technologies performances: this kind of emissions greatly depends on soil and climate conditions. Denitrification greatly increases in sensitive sites and observed differences between scenarios are only due to the specific compaction impact of each spreader.

Key words: nitrogen emissions, models, spreader performances, sewage sludge.

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## 1. Introduction

Agricultural activities such as nitrogen crop fertilisation are greatly implicated in environmental concerns by contributing to dissipate in the atmosphere ammonia (NH<sub>3</sub>), nitrogen oxides (NO<sub>x</sub>), nitrous oxide (N<sub>2</sub>O) or dinitrogen (N<sub>2</sub>), and to liberate nitrates (NO<sub>3</sub>) in water. As manures, sludges and slurries contain great amounts of nitrogen, there is a growing issue about water and air pollution caused by their application in the field.

Among the factors governing emissions dynamics, many studies have focused on the effluent type (e.g. Akiyama *et al.* 2004) and others have investigated the opportunity to treat them (e.g. Amon *et al.* 2006) or to experiment innovative storage management practises (e.g. Webb *et al.* 2004).

Many authors showed that application technique is also an important lever to reduce ammonia volatilisation, most of the time concerning ammonia emissions after slurry application. Bittman *et al.* (2005) found that band spreading technique associated with immediate incorporation in soil reduces ammonia emissions by half compared with surface spreading. The use of trailing hoses, trailing shoes and injector technologies lead to a reduction ranging from 40 to 60 % in comparison to surface spreading techniques (Smith *et al.* 2000). Other experimental results demonstrated that shallow injection of slurry can contribute to a reduction of ammonia emissions varying between 20 and 75 % compared with a band spreading technique (Hansen *et al.* 2003). This study highlights that using techniques to reduce ammonia emissions can also be associated with an increase of greenhouse gas emissions like N<sub>2</sub>O emissions. The effect of application technique on N<sub>2</sub>O emission is corroborated by Velthof *et al.* (2003). Besides, spatial distribution performances of spreaders seem not to contribute to nitrate leaching but the effect of the application technique has not really been investigated (Thirion *et al.* 2009). As there is an important renewal in spreading techniques assessment methods, it is challenging to develop a generic methodological framework to investigate the link between application techniques performances and related nitrogen emissions in the field. The example of s.s.<sup>1</sup> spreading is taken to cover a wide range of physical properties and thus spreading techniques. This is one of the main issues of

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<sup>1</sup> s.s.: sewage sludge

Ecodefi project which aims at placing environmental challenges in the heart of organic spreading technology design, notably by using Life Cycle Assessment approach. In the methodological framework presented in this paper, biophysical models are used to assess at the same time all nitrogenous emissions (NH<sub>3</sub>, NO<sub>3</sub> and N<sub>2</sub>O) and to link them with three types of spreading machines performances.

## 2. Materials and methods

### 2.1. Scenarios definition and main hypotheses

To assess nitrogenous emissions, 45 scenarios were defined by a panel of experts. They cover a wide range of French pedoclimatic and agronomical conditions. Each scenario (Figure 1) associates (i) one of the nine cropping systems, characterized by a real French site, a specific crop rotation and a spreading period (Table 1) and (ii) one of the five pairs 'spreading machine/s.s.' (Table 2). Recommended s.s. application rate was determined for each of them, according to the plant nutrients balance and regarding French spreading laws, especially those concerning metallic trace elements threshold respect. For most scenarios, application rate depends on phosphate plant requirements. Data related to the cropping systems (climate, soils and practices) and the s.s. (chemical characteristics) were collected and integrated as inputs into the models.

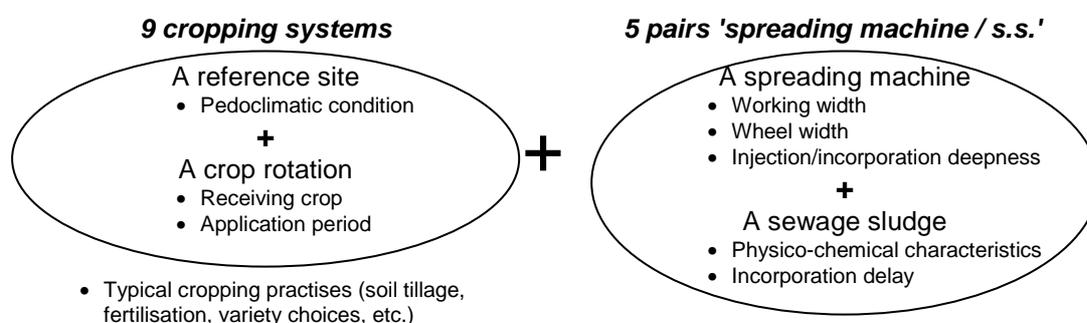


Figure 1. Scenario description

French site	Textural class of the surface layer	pH of the surface layer (0-30 cm)	Reference bulk density of the surface layer	Crop system rotation <sup>2</sup>	Application period	Receiving crop	Main environmental issue
Montoldre (Mt1)	Loamy sand	6.5	1.48	R, WW, WB	End of July	WB	NO <sub>3</sub> leaching
Montoldre (Mt2)	Loamy sand	6.5	1.48	R, WW, WB	August	R	NO <sub>3</sub> leaching
Kerlavic (KI1)	Clay loam	5.8	1.17	WW, CS	End of February	CS	NO <sub>3</sub> leaching
Kerlavic (KI2)	Clay loam	5.8	1.17	WW, CS	March	WW	NO <sub>3</sub> leaching
Kerlavic (KI3)	Clay loam	5.8	1.17	TG, C, WW	End of February	TG followed by C	NO <sub>3</sub> leaching
Mons (Ms1)	Loam	7.8	1.30	SB, WW, WB	End of July	SB	Soil compaction
Mons (Ms2)	Loam	7.8	1.30	NCC, SB, WW, WB	End of July	NCC followed by SB	Soil compaction
Poitou (Pt1)	Argilo-calcareous	7.5	1.25	R, WW, S, WW	August	R	NH <sub>3</sub> volatilisation
Pays Caux (Cx1)	Sandy loam	6.8	1.49	FF, NCC, SB, WW, WB	August	NCC followed by SB	Run off

Table 1. Cropping systems description

<sup>2</sup> R = rape, WW = winter wheat, WB = winter barley, CS = corn silage, TG = temporary grassland, C = corn, SB = sugar beat, S = sunflower, FF = fiber flax, NCC = nitrogen catch crop

Spreading technique	S.s. type	Corg:Norg ratio (g.g <sup>-1</sup> )	Dry matter content (% FM <sup>3</sup> )	Total nitrogen content (g.kg <sup>-1</sup> DM <sup>4</sup> )	P <sub>2</sub> O <sub>5</sub> content (g.kg <sup>-1</sup> DM)	TAN <sup>5</sup> content (g.kg <sup>-1</sup> DM)	pH (d.u.) <sup>6</sup>	Application rate range (DM ton/ha)
Splash plate	Liquid (le)	5.4	6.4	66.0	71.14	8.7	7.1	0.94 - 1.85
Shallow injection	Liquid (ln)	5.4	6.4	66.0	71.14	8.7	7.1	0.94 - 1.85
Vertical moving rotors	Limed solid (ch)	5.8	32.7	29.0	25.04	1.0	11.1	2.74 - 5.26
Vertical moving rotors	Composted with plant waste (co)	11.7	64.0	20.5	18.10	0.5	9.1	5.63 - 7.72
Disks	Pelleted (se)	6.3	89.3	45.1	81.38	1.8	7.2	1.25 - 1.71

Table 2. Description of each 'spreading machine / s.s.' pair

## 2.2. Simulation models to assess nitrogen emissions

Two models are used to assess nitrogen losses: DEAC and STICS. DEAC (Jolivel 2003) aims to calculate nitrate quantities which are likely to leach beyond the roots and the mean nitrogen concentrations in leached water, at the cropping system scale. The formalism takes several fluxes into account: mineralisation / organisation, root absorption, leaching and residual nitrogen (at harvest, at the beginning of the leaching period, at the end of winter and at the end of the leaching period). DEAC has been selected to assess nitrate losses because its formalism is more sensitive to the s.s. application rate than the one of STICS model (results not published).

STICS is a daily time-step crop model (Brisson *et al.* 1998; Brisson *et al.* 2003; Brisson *et al.* 2008). It calculates agricultural (yields, input consumption) and environmental variables (nitrate leaching, volatilisation, denitrification) by using input data describing climate, soil and crop systems. STICS uses generic parameters so that it is able to simulate various crops (wheat, maize, grassland, sugar beet...) (Launay *et al.* 2003). It was tested under various soil-climate conditions without considerable bias (Brisson *et al.* 2002).

STICS is organised into modules dealing with specific mechanisms like yield formation, water balance and nitrogen balance (Figure 2). The crop management module concerns the interactions between the applied techniques (e.g. mineral and organic fertilisation) and the soil-crop system. Each module includes various options so as to be able to simulate various crop systems and to cover a wide range of agro-environmental issues.

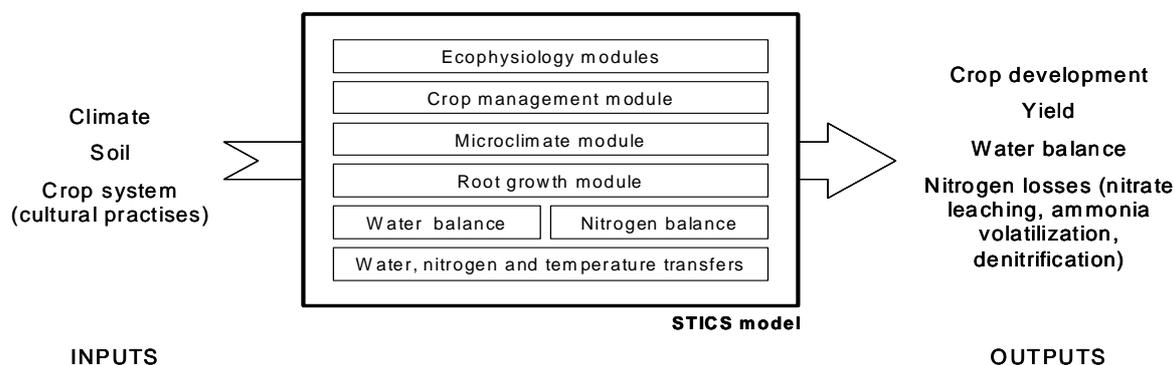


Figure 2. STICS modules

<sup>3</sup> FM: Fresh Matter

<sup>4</sup> DM: Dry Matter

<sup>5</sup> TAN: Total Ammoniacal Nitrogen

<sup>6</sup> d.u.: dimensionless unit

Denitrification model used in STICS is the NEMIS model (Henault *et al.* 2000). It allows an assessment of nitrogen losses from denitrification as nitrogen ( $N_2$ ) or nitrous oxide ( $N_2O$ ). Real denitrification is calculated from potential denitrification corrected by temperature, soil nitrate and water contents. STICS then provides total N losses by denitrification. A  $N_2:N_2O$  ratio is needed to assess  $N_2O$  emissions as they are the only ones to have an environmental effect (global warming). According to several authors (Dambreville *et al.* 2006; Ruser *et al.* 2006; Ciarlo *et al.* 2008), this ratio is quite variable and depends of many factors. As it would be difficult to attribute a ratio to each pedoclimatic situation, a one to one ratio corresponding to the upper value of the study of Ruser *et al.* has been chosen, considering our study forms part of the framework of environmental potential risks assessment.

DEAC and STICS models are used to assess total nitrogen losses in the crop cycle following the s.s. spreading, including a leaching period. For each site, a representative climatic year concerning nitrate leaching is chosen so as to avoid extreme values (the year which is closest to the mean emission value of an eight year period, determined by simulations). We obtain simulated nitrogen losses regarding a wide range of application rates (from zero to double rate) and soil surface layer bulk densities (from 0.8 to 1.7  $g.cm^{-3}$ ).

**2.3. Flows formalisation**

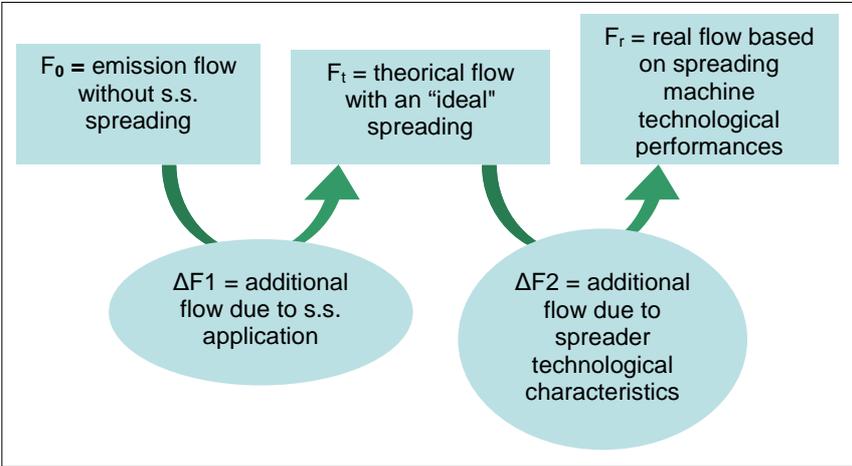


Figure 3. Flows formalisation

Flows are always considered as emissions taking place during the crop cycle following the s.s. spreading. We make the hypothesis that the emissions which occur during the other crop cycles of the rotation may be diluted amongst the emissions due to other fertilisers spreading of the following crops. Besides it may be difficult to discriminate the emissions due to the s.s.

spreading as there is a great uncertainty about the organic matter mineralisation dynamics of the s.s. (Parnaudeau 2009). Hence, this choice leads to allocate all the environmental impacts of the s.s. spreading to the receiving crop cycle, i.e. between the spreading day and the harvest day, integrating most of the time a part of a drainage period (from 9 to 13 months according to the species and the introduction or not of a nitrate catch crop). Even if it is not certain that all nitrogen emissions are taken into account, the objective of scenarios discrimination is achieved.

As shown on figure 3, the simulation results give directly the opportunity to assess (i)  $F_0$ , the total nitrogen flow emitted in the form of  $NO_3$ ,  $NH_3$  and  $N_2O$  with no s.s. spreading (application rate equal to zero), all else being equal for each scenario (e.g. same mineral fertilisation), (ii)  $F_t$ , the total flow induced by the s.s. application, theoretically spread by a machine having an ideal spatial distribution, distributing exactly the recommended application rate at the field scale and having no effect on the soil compaction. The difference between these two types of flows represents  $\Delta F1$  which is the part of the flow resulting from an ideal s.s. application. Furthermore, a method described in the following section is used to come closer to the spreading operation reality and to assess the nitrogen flows  $\Delta F2$ , taking some machine performances defaults into account.

## 2.4. Additional flows $\Delta F_2$ induced by technological performances of spreaders

### 2.4.1. Spatial distribution heterogeneity and application rate accuracy

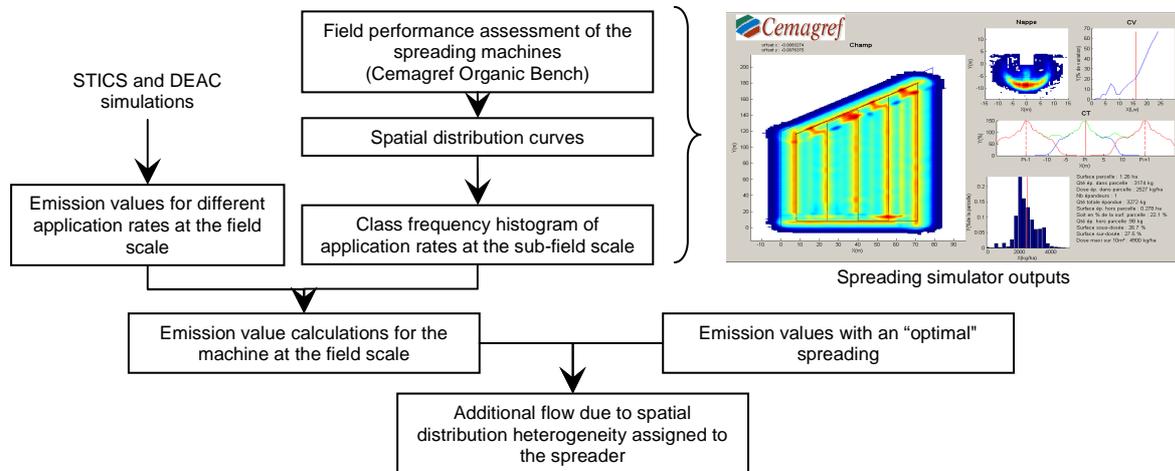


Figure 4. Calculation method of additional flows due to spatial heterogeneity

The spreading simulator (Piron *et al.* 2009) carries out a virtual spreading operation in a real geographical context. This software has been designed in order to allow the operator to analyse a spreading scenario integrating parameters measuring the quality of the spatial distribution of the spreading machine. On the one hand, based on the spreading simulator outputs, the spatial distribution performances of the spreader measured on the Cemob test bench (Rousselet *et al.* 2010) are represented by an application rate map and a class frequency histogram of application rate at the sub-field scale. The field is subdivided into numerous small surfaces characterised by a specific s.s. application rate and a specific nitrogen emission rate. On the other hand, nitrogen losses are assessed for each different application rate thanks to the STICS and DEAC simulation outputs. Balancing these simulated losses with the proportion of field concerned allows us to calculate the nitrogen losses at the field scale. The additional nitrogen emission flows due to spatial distribution heterogeneity assigned to the spreader can be calculated comparing it with the emissions values for homogeneous theoretical application rates  $F_t$  (Figure 4). In the same way, an additional flow due to application rate accuracy at the field scale is determined.

### 2.4.3. Soil compaction under wheels

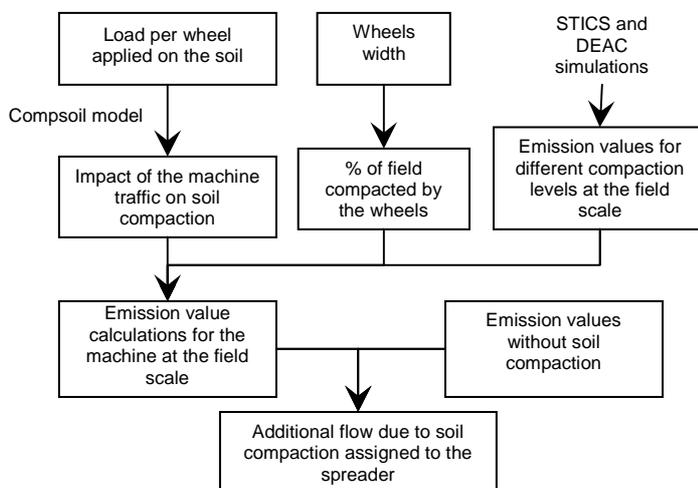


Figure 5. Calculation method of additional flows due to soil compaction

Compsoil model (O'Sullivan *et al.* 1999; Defosse *et al.* 2003) simulates the effect of the running of one or several wheels on the soil compaction which is assumed to be represented by the bulk density value of the soil surface layer. Compsoil outputs provide bulk densities resulting from the running of the spreading convoy for the 45 scenarios. Moreover, we calculate the field area compacted by the machine using the tyres characteristics and the working width of the spreader. We also make the hypothesis that the spreader running

in the field impacts on the soil during the whole crop duration. Emission values for different bulk densities of the soil surface layer have been simulated with STICS model. Considering that emissions are different inside and outside the wheel footprint areas, the additional flow due to soil compaction assigned to the spreader results from the comparison of emissions taking or not compacted areas into account (Figure 5).

### 3. Results and discussion

#### 3.1. Models sensitivity to application rate and soil compaction

##### 3.1.1. Sensitivity to application rate

Curves in Figure 6 represent mean variations for the four different sludges, excepted for the composted sludge which is not included in the denitrification and leaching curves since its pattern is completely opposite (not published). Considering this sludge presents a relatively high C:N ratio (10.6), its mineralisation dynamics may include a nitrogen fixation period after spreading postponing nitrogen mineralisation and potential emissions by leaching and denitrification. That is why the increasing of the application rate leads to greater nitrogen fixation and less important emissions by leaching or denitrification. On the whole, emissions are positively correlated to the application rate, but there are great differences among reference sites. On average, the doubling of the application rate (200 %) nearly generate twice as much emissions by volatilisation (201 %) but not more than 8 % more for denitrification (from 5 % for the liquid sludge to 10 % for the pelleted sludge) and 9 % more for nitrate leaching (from 7 % for the pelleted to 10 % for the limed solid one).

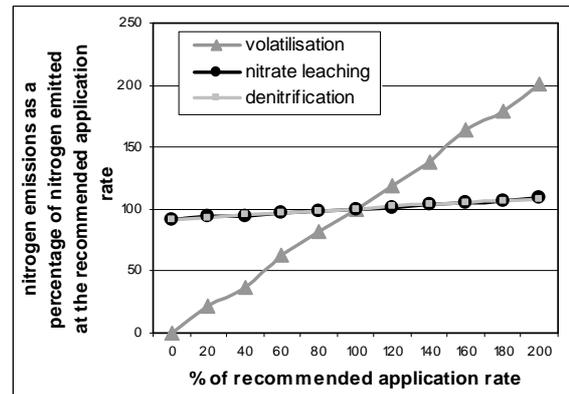
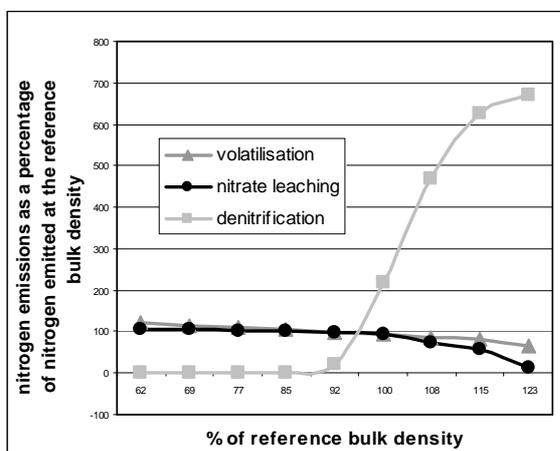


Figure 6. Mean sensitivity of nitrogen emissions to application rate

Ryan et al. studied the application rate effect and found that higher application rate reduces ammonia emission rate (Ryan *et al.* 1975). The models we use do not simulate such a pattern as the volatilisation rate is directly linked with the application rate. More experiments are needed to validate or debate the models results. Fernandes et al. (2005) also showed an application rate effect on nitrous oxide emissions.

Concerning sensibility to the application rate, models predict that a very small part of the nitrogen brought in surplus is likely to be lost by the different processes. This can suppose that the soil plays a buffer role, retaining nitrogen in excess. Moreover, mineral nitrogen is gradually liberated, thus avoiding its accumulation in the soil and favouring its absorption by the crop.

##### 3.1.2. Sensitivity to compaction



At the reference bulk density value, significant denitrification process only occurs in Kerlavic and Poitou sites so that other sites are not included in the calculation of the mean curves. Models show a variable influence of soil compaction (represented by the bulk density) on nitrogen emissions (Figure 7). Soil compaction increase reduces nitrate leaching whose decrease becomes stronger for extreme bulk density values (emissions are reduced by 80% for a 20 % bulk density

Figure 7. Mean sensitivity of nitrogen emissions to soil bulk density

increase). A similar pattern can be observed for volatilisation: it seems that the compaction not only reduces liquid infiltration but also limits air exchanges and thus slightly reduces ammonia volatilisation from soil surface. Concerning denitrification, mean results show a sigmoid relationship between bulk density and denitrification intensity: in sites where significant denitrification process occurs, denitrification intensity appears to be greatly linked to the bulk density in a certain range, corresponding to favourable soil conditions for the denitrifying bacteria flora. This sensitivity patterns will partly explain the results of emissions due to spreading machines performances.

### 3.2. Pedoclimatic and s.s. flows

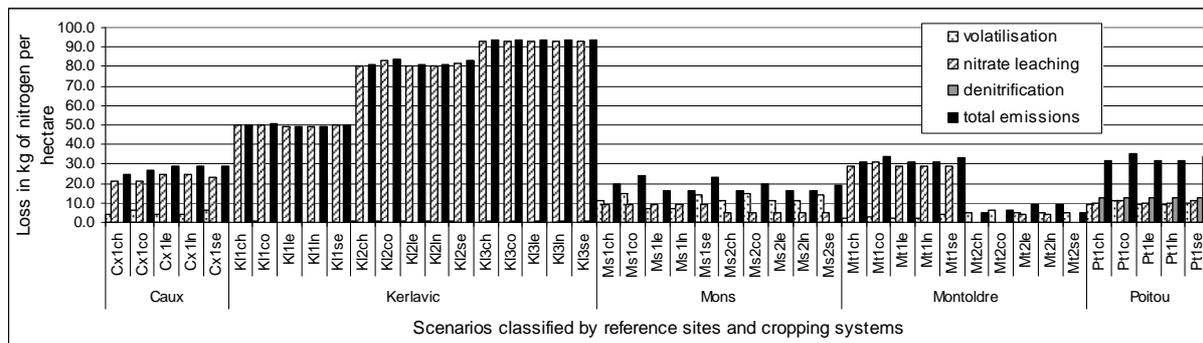


Figure 8. Basal flow without s.s. spreading ( $F_0$ )

Figure 8 presents the basal emissions assessment using biophysical models for the 44 scenarios<sup>7</sup>, with no s.s. spreading. The results confirm that there is an important effect of the reference site for all types of emissions, regardless of the spreading operation and the spreading machine. Models used in this study clearly allow us to distinct reference sites and cropping systems groups of scenarios (especially Montoldre and Kerlavic). There are very much reduced differences between scenarios belonging to the same cropping system as there are not many differences in the technical management. Results also illustrate that there is a great interest to take into account a wide range of pedoclimatic conditions and crops so as to express spreading machines effects in many situations, as these conditions have a predominant influence on emissions.

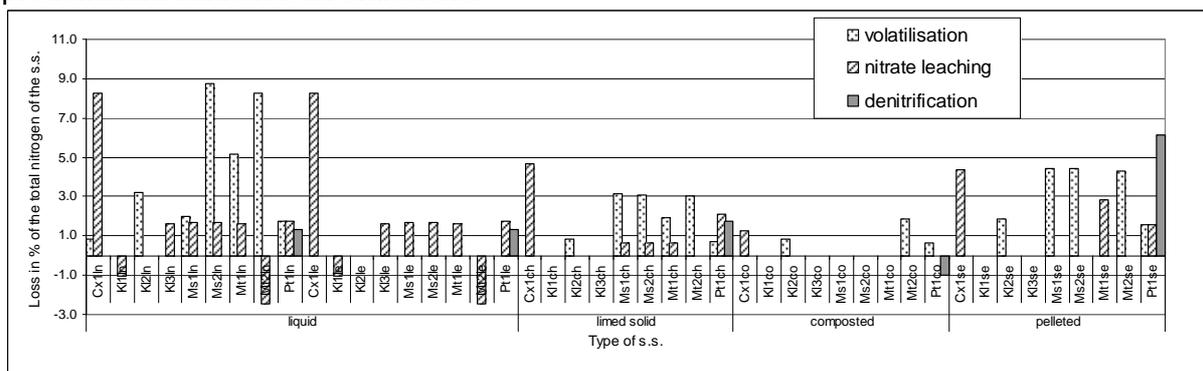


Figure 9. Nitrogen flows due to s.s. spreading ( $\Delta F_1$ )

Even if a relationship can be established between nitrogen emissions and s.s. type, great discrepancies appear among the different scenarios; the interaction effect between s.s. and pedoclimatic context is always predominant. These results represent the additional or avoided nitrogen emitted when the s.s. is applied, considering it is perfectly distributed over the field at the optimal recommended application rate with no compacting effect of the spreading machine. Among the scenarios, positive sign means that applying the sludge leads to additional emissions whereas negative sign means that nitrogen is sequestered into the soil. In percentage of the total nitrogen supplied by the s.s., nitrogen emissions vary from

<sup>7</sup> As one scenario does not represent realistic practises, we only assess 44 scenarios.

0 to 8 % (from 0 to 10 kg of nitrogen per hectare), from -3 to 8 % (from -3 to 10 kg of nitrogen per hectare) and from -3 to 6 % (from -2 to 4 kg of nitrogen per hectare) in the form of ammonia, nitrate and nitrous oxide, respectively. Volatilisation is mainly greater for liquid s.s. spreading surface spreading (first nine scenarios) and is very low for composted and pelleted s.s. because of a high dry matter content. As models are not parameterised for pelleted s.s., experts consider that there is no volatilisation after their spreading (Genermont 2009). Other types of emissions are more linked to the pedoclimatic site than to the s.s. characteristics: the s.s. spreading induces very low nitrate leaching, except for Caux reference site (cropping system Cx1) and additional denitrification only occurs in Poitou (Pt1 scenarios).

Concerning ammonia, our results show that the mean volatilisation rate is equal to 0, 17, 25 and 44 % of the total ammoniacal nitrogen brought by the s.s., for pelleted s.s., composted s.s., liquid s.s. and limed solid s.s., respectively. Some experimental studies present higher ammonia emission rate for liquid s.s. (King 1973; Beauchamp *et al.* 1978; Hall *et al.* 1986; Adamsen *et al.* 1987). Other results are on the whole in accordance with our assessments: for a liquid s.s. spreading, Terry *et al.* (1978) measure a loss of ammonia ranging from 13 % to 25 % of the TAN in clay soils and Ryan *et al.* (1975) assessed it between 11 and 37 % of the TAN in a loamy soil. For a solid municipal sewage sludge (34 g DM / kg<sup>-1</sup>), Parnaudeau *et al.* (2009) found that 45 % of the TAN of the sludge has volatilised. Our results are also in agreement with those of Donovan *et al.* (Donovan *et al.* 1983) who demonstrate that ammonia emissions are greater for a limed solid s.s. than for a liquid one. This may be explained by a higher pH of the limed s.s. and especially in our study, infiltration is less important for the solid s.s. than for the liquid one, the contact with air is then not favoured. No experimental data has been found concerning other type of sludges like composted and pelleted ones.

As seen in Figure 9, nitrous oxide emissions are greatly associated with the denitrification potential of the different sites. Poitou is the only site where nitrous oxide emissions due to the s.s. spreading are observed and the total emissions reach 8 % of the total nitrogen supplied by the s.s.. STICS model does not predict any emission for all other scenarios. This high variability of denitrification potential is confirmed in other experiments: the s.s. spreading can have no effect on N<sub>2</sub>O emissions (Ambus *et al.* 2001), the effect can be very low (Scott *et al.* 2000; Zaman *et al.* 2004; Jezierska-Tys *et al.* 2007; Parnaudeau *et al.* 2009) or quite important (Fernandes *et al.* 2005; Chiaradia *et al.* 2009).

Studies about nitrate leaching following s.s. spreading are not numerous. An experiment reveals that the nitrate loss can reach 17 % of the total nitrogen supplied by a composted s.s. (Esteller *et al.* 2009). Using a simulation model, Parnaudeau *et al.* assessed a greater loss, representing almost 30 % of the total nitrogen contained in the municipal s.s. (Parnaudeau *et al.* 2009). Some authors also underlined that nitrate production potential is more important for liquid s.s. than for dehydrated or dried s.s. (Smith *et al.* 1998) which is in agreement with our results.

### 3.3. Flows induced by spreading machines performances

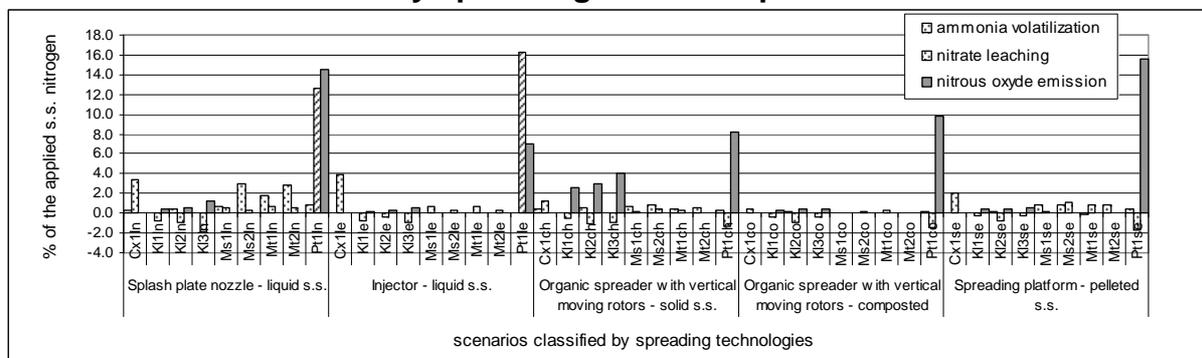


Figure 10. Additional flow to spatial distribution heterogeneity, application rate accuracy and soil compaction of the spreading machine

The additional nitrogen flow due to either the spatial distribution heterogeneity or the application rate accuracy is less than 4% of the total N applied in most of the 44 scenarios, excepted for liquid sludge spreading in one pedoclimatic situation (10 % of the total N applied). Soil compaction seems to be an important emission factor for nitrous oxide, especially in pedoclimatic situations where denitrification potential is high. As there are significant differences between emission values of the spreading scenarios, we are able to discriminate (i) the scenarios and pedoclimatic sites and (ii) the spreading technologies (Figure 10). The NH<sub>3</sub> volatilisation due to the spreader performances is generally low and the highest for splash plate technology (less than 3 % of the total nitrogen supplied by the s.s.). Additional emissions are mainly caused by application rate accuracy problems. It is not possible to link NO<sub>3</sub> leaching to the technologies performances: this kind of emissions greatly depends on soil and climate conditions, confirming Thirion et al. (2009) suggests. Denitrification dramatically increases in sensitive sites and observed differences between scenarios are only due to the specific compaction impact of each spreader.

## 4. Conclusions

This work takes part in the renewal of the framework aiming to assess spreading machines performances related to environmental impacts: there is a need to improve models and emission assessment at the scenario level as it is impossible to validate accurately the results. There is a lack of studies about nitrogen emissions after s.s. spreading and the spreading machines characteristics are most often taken into account only for ammonia emissions, most of the time very partially.

The scenario definition is an important step in this method: the s.s. and the spreading location have a great impact on emission calculations. Indeed, pedoclimatic conditions and s.s. types can hide specific impacts of the studied technologies. Thus, a better discrimination between technologies would have been highlighted if we had chosen more sensitive pedoclimatic conditions or other types of effluents. The biophysical models we used are not well adapted to assess the spreader performances as spreading technologies are not directly taken into account in these models. As conditions predominantly influence nitrogen emissions, it is still important to keep a wide range of pedoclimatic situations so as to express the spreader performances in different contexts.

Although we show that these models can be used to estimate N emissions due to some spreader performances (spatial distribution, application rate and compaction), other spreader performances such as splitting up or injection characteristics still have to be assessed by empirical methods.

This methodological framework paves the way to new approaches (i) integrating technological performances of machines into biophysical and agricultural models, (ii) taking spreading machine performances into account to assess environmental impacts of agricultural practises. These results are finally used to provide data for the emissions inventory of Life Cycle Assessments carried out for these 44 spreading scenarios (Pradel et al. 2010).

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