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Predicting soil water and mineral nitrogen contents with the STICS model for estimating nitrate leaching under agricultural fields

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ABSTRACT

The performance of the STICS soil-crop model for the dynamic prediction of soil water content (SWC) and soil mineral nitrogen (SMN) in the root zone (120 cm) of seven agricultural fields was evaluated using field measurements in a coarse-grained alluvial aquifer of the Garonne River floodplain (southwestern France) from 2005 to 2007. The STICS model was used to simulate drainage and nitrate concentration in drainage water in all the agricultural fields of the study area, in order to quantify and assess the temporal and spatial variability of nitrate leaching into groundwater. Simulations of SWC and SMN in the seven monitored fields were found to be satisfactory as indicated by root mean square error (RMSE) and model efficiency being 6.8 and 0.84% for SWC and 22.8 and 0.92% for SMN, respectively. On average, SWC was slightly overestimated by a mean difference of 10 mm (3%) and there was almost no bias in SMN estimations (<0.5%). These satisfactory results demonstrate the potential for using the STICS model to accurately simulate nitrate leaching.

Across the study area, simulated drainage and nitrate concentration were extremely variable from one field to another. For some fields, simulated mean annual nitrate concentration in drainage water exceeded 300 mg NO_3^- L^{-1} and predicted nitrate leaching was close to 100 kg N ha^{-1}, while other fields had very low nitrate losses. About 15% of the farmers' fields were responsible for 60–70% of nitrate leaching. The SMN in late autumn, before winter drainage, was found the main determining factor explaining this variability. This situation may be attributed to unsatisfactory cumulative nitrogen management over the medium term. Ineffective nitrogen management was found to be more detrimental than a single annual incident of overfertilization, particularly in situations of deep soils and in cases of low or highly variable drainage between years.

1. Introduction

The European Water Framework aims to achieve long-term sustainable water management for both surface and groundwater bodies. The first step of this framework is to achieve "good status" for all waters by 2015. One component of good status is the nitrate concentration in both surface water and groundwater. Intensive agriculture has contributed to an increase in nitrate levels in many areas of Europe (Strebel et al., 1989). Alluvial groundwater is particularly vulnerable to nitrate (NO_3^-) leaching due to nitrogen (N) losses from agricultural soils, since agricultural land is characterized by the presence of shallow groundwater and fertile soils suitable for farming. Several studies have shown that nitrate leaching through unsaturated soil can have an important impact on groundwater pollution (Gustafson, 1983; Bijay-Singh et al., 1995; Arrate et al., 1996; Sánchez-Pérez et al., 2003c). However, the relationship between groundwater NO_3^- concentration and N sources used at the soil surface is complex.

Previous studies have proved that crop models present potential for quantifying the impact of agricultural activities on nitrate leaching into groundwater (Wagenet and Hutson, 1996; Loague and Corwin, 1996; Hoffmann and Johnson, 1999). First, they are able to simulate complex processes and calculate variables that are difficult to measure. Several important sources of N in agricultural soils, such as mineralization from organic matter or nitrogen-rich crop residues (*e.g.*, legumes), fertilizer or atmospheric deposition, can be converted to NO_3^- and incorporated into groundwater recharge

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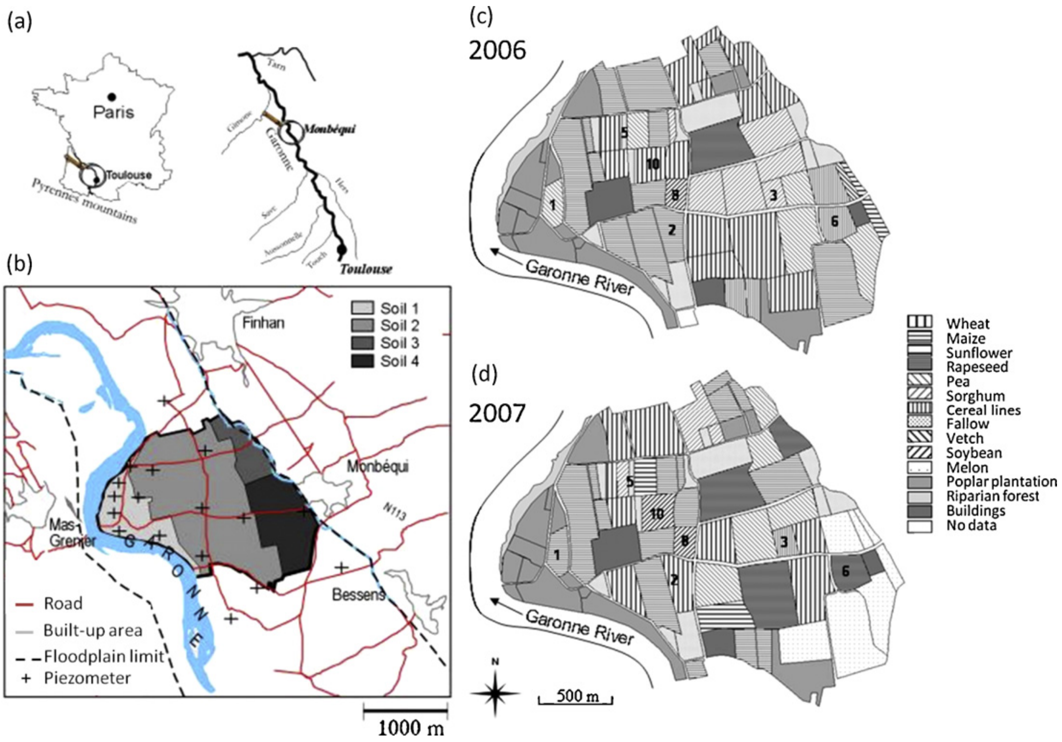

Fig. 1. Map of the study area showing (a) the location of the study area, (b) the floodplain limit, the four soil types defined and the piezometer locations, and (c) 2006 and (d) 2007 land use and locations of the monitored fields (indicated by numbers).

(Böhlke, 2002). That is why soil-crop models, validated using *in situ* measurable variables, such as soil water content (SWC) and soil mineral nitrogen (SMN) content, are useful for quantifying nitrate leaching in agricultural areas.

Another advantage of crop models is their ability to simulate the crop rotation pattern and the fallow period between two main crops over several years. This is critical for studying nitrate leaching because the temporal dynamics of this process are greatly influenced by the status of the crop and climatic variations. This temporal aspect is also useful for predicting the mitigation effects of improved agricultural practices, such as optimized N fertilizer applications and catch crop establishment, on soil and water status. If the SMN level at harvest is high and no new crop is sown immediately, the use of a catch crop is an efficient way to reduce nitrate leaching during fallow periods (see review by Thorup-Kristensen et al., 2003). However, the effects of any catch crop should be evaluated over the long term rather than just the short term (Berntsen et al., 2006).

Many models such as Crop Environment REsource Synthesis (CERES) (Ritchie and Otter, 1984; Jones and Kiniry, 1986), Erosion Productivity Integrated Calculator (EPIC) (Williams et al., 1989) and Simulateur mulTIdisciplinaire pour les Cultures Standard (STICS) (Brisson et al., 1998) are able to simulate crop growth and water and nitrogen balances at field scale. However, the predictive quality of these models has been evaluated mainly on the basis of annual experiments and/or experimental conditions. Their ability to predict water and nitrogen leaching over 2–3 years in “real” farm conditions, which may differ from the agricultural practices applied on experimental sites, needs to be more widely evaluated before they are used to simulate *ex ante* scenarios of cropping systems (Beaudoin et al., 2008). One possibility is to compare temporal simulated and measured soil water and mineral nitrogen in the rooting zone in order to evaluate the ability of the model to simulate the nitrogen cycle and water and nitrate movements in the unsaturated zone of the soil toward the groundwater.

The aim of this study was to analyze and quantify when and where nitrate leaching occurs in an alluvial floodplain, using a dynamic soil-crop model. The objectives of our work were twofold: (1) to evaluate the predictive quality of the dynamic STICS soil-crop model for simulating soil water and mineral-N contents over three successive annual periods in comparison with field measurements; (2) to analyze and quantify the impact of crop sequence and the effect of initial soil mineral content on simulated spatial and temporal nitrate leaching. The work was carried out in farm fields located in the alluvial floodplain, in which conventional agricultural practices are applied.

2. Materials and methods

2.1. Study area

The study site is located in a meander of the Garonne River at Monbéqui in southwestern France (43°53′30″N, 1°13′00″E). The area extends over approximately 12 km^2, with 50 agricultural fields, most of which are used for crops, making up about 75% of the total area (Fig. 1).

The alluvial plain of the Garonne River comprises a succession of terraces. An alluvial aquifer is situated in the first terrace, which is composed of coarse alluvium. The first 50–100 m from the riverbank are covered by riparian forest and poplar plantations, beyond which lies agricultural land. The alluvial aquifer comprises a layer, about 6–7 m thick, overlying impermeable and indurate marl. Previous measurements of nitrate concentration in this aquifer showed considerable spatial variability over a short distance. The measured nitrate concentration in the groundwater (see Fig. 1 for piezometers location) varies widely from

Table 1
Climatic data (rainfall, minimum and maximum temperature and cumulative solar radiation) recorded from November 2004 to October 2007.

Period	Rainfall (mm)	Mean $T_{\min}$ (°C)	Mean $T_{\max}$ (°C)	Cumulative solar radiation (MJ m^{-2})
1/1/04–30/11/05	501	8.7	17.7	4861
1/12/05–31/10/06	644	8.5	20.2	4896
1/11/06–31/10/07	600	8.7	19.0	4971

10 to 90 mg NO_3 L^{-1}, whereas in the river it varies from 10 to 20 mg NO_3 L^{-1} (Sánchez-Pérez et al., 2003b). Interactions between the river and the groundwater could explain part of the spatial distribution of nitrate concentrations. Indeed, dilution and denitrification processes could explain the low groundwater nitrate concentration (~10 mg NO_3 L^{-1}) observed in the alluvial aquifer along the riverbank (Sánchez-Pérez et al., 2003a; Iribar et al., 2008). In fact, there are "hotspots" of denitrification in the aquifer area that are regularly subjected to mixing of river water and aquifer water (McClain et al., 2003; Sánchez-Pérez et al., 2003a; Iribar et al., 2008). In addition, in this area the nitrate concentrations in groundwater are diluted by the river water (Weng et al., 2003; Peyrard et al., 2008). However, there are also large spatial variations in groundwater nitrate concentration inside the alluvial aquifer underlying agricultural land, where the influence of river water is very low.

The main crops in the study area are wheat and maize, with less important crops being peas, sorghum, soybean, rapeseed and sunflower. Melons and vetch (green manure) are also grown occasionally. Some of the maize and wheat fields were being used to evaluate new cultivars in large trials carried out by seed companies. Poplar plantations represented about 15% of the total area and were located near the river. The rest of the area was covered by buildings (5%) and riparian forest (5%).

Mean annual precipitation in the study area is about 660 mm (1994–2007). Meteorological data for the period from December 2005 to October 2007 were collected on the site using an automatic meteorological station (Table 1). From January 2005 to December 2005, precipitation data were obtained from the Monbéqui meteorological station (Meteo France), located 1 km from the study site. Data on temperature, wind, humidity and solar radiation were obtained from Toulouse-Blagnac meteorological station (Meteo France), located 50 km from the site. For the three crop sequences on the monitored fields, the 2004–2005 period was the driest and the 2005–2006 period the wettest. The mean minimum temperature and the mean daily solar radiation were almost the same for all three cropping periods. Total annual rainfall was 501 mm for the 2004–2005 period, 644 mm for the 2005–2006 period and 600 mm for the 2006–2007 period. Seasonal precipitation distribution shows that precipitation tends to be lower in winter than in spring, summer and autumn. In summer, showers and storms can generate short, intense precipitation events (*ca.* 30 mm day^{-1}). During the study period, the groundwater level varied between 2.5 and 5 m below the soil surface and the groundwater did not interact with the root systems of arable crops.

2.2. *Experimental design*

2.2.1. *Monitored fields and field-scale modeling*

Soils cores were collected from 25 fields (including seven monitored fields) in order to determine soil characteristics, *i.e.*, texture, organic matter, pH, total carbonates (Table 2). There was a texture gradient from the riverbank to the end of the first terrace ranging from sandy loam to silty clay loam texture. This gradient was particularly pronounced near the riverbank, where there are riparian forests and poplar plantations. The soil characteristics of the agricultural fields were fairly homogenous. From the soil analysis, four classes of soils were distinguished (Fig. 1). Soil 1 was situated near the Garonne riverbank, its texture was loamy, and it contained a high percentage of limestone. Soil 2, situated a little farther from the riverbed, was a silty loam, and contained less sand and more silt than soil 1. Soils 3 and 4 were silty clay loams, but soil 3 contained less sand and $CaCO_3$ than soil 4, and its pH was lower.

A group of seven fields was monitored from February 2005 (fields 1, 6, and 10) or December 2005 (fields 2, 3, 5, and 8) to October 2007. These fields were a representative sample of all main crops and soil types at the site. They included six agricultural fields (1, 2, 3, 6, 8, and 10) and a fallow field (5), occasionally grazed, with no cropping or mineral-N fertilizer, which was used as the control representing minimum N leaching under the prevailing pedoclimatic conditions (Fig. 1b). The crop sequences and the quantity of N fertilizer and irrigation water applied to each of the monitored agricultural fields are reported in Table 3. Only maize and sorghum were irrigated. Pea and soybean were generally not fertilized. One wheat field was not fertilized in 2006, because the sampling zone was located in an unfertilized area of a wheat trial. Sunflower is usually not fertilized with N because it has low N requirements and its N needs are met by a high level of soil N mineralization in spring and summer.

For each field, soil cores were extracted on 7–13 sampling dates, from February 2005 to November 2007 (see Fig. 1). The soil cores were collected to a depth of 1.2 m using an automatic soil corer. In order to take intra-field variability into account, between 6 and

Table 2
Soil properties of the four soils identified in the study area.

	Soil 1		Soil 2		Soil 3		Soil 4	
Depth (cm)	0–30	30–120	0–30	30–120	0–30	30–120	0–30	30–120
Sand (%)	36	38	30	28	12	10	20	23
Silt (%)	44	44	52	50	59	52	51	48
Clay (%)	20	18	18	22	29	38	29	29
pH	8.2	8.4	8	8.3	7.3	7.5	8.5	8.3
$CaCO_3$ (%)	6.7	7.6	1.7	2.8	0.1	0.1	2	2.6
Organic-C (g kg^{-1})	10	5	9	6	12	8	12	8
Organic-N (g kg^{-1})	0.9	0.4	0.8	0.5	1.0	0.6	1.0	0.6
Field capacity (water in g g^{-1} of soil)	22.6	15.7	21.6	20.9	24.2	23.1	23.0	22.0
Permanent wilting point (water in g g^{-1} of soil)	7.5	6.3	9.6	9.5	9.5	10.2	9.5	9.8

Field	Crop rotation								
	2005			2006			2007		
	Crop	Fertilization (kg N ha^{-1})	Irrigation (mm)	Crop	Fertilization (kg N ha^{-1})	Irrigation (mm)	Crop	Fertilization (kg N ha^{-1})	Irrigation (mm)
1	Soybean	0	0	Winter pea	0	0	Maize	150	175
2				Maize	95	140	Wheat	140	0
3				Sorghum	220	80	Wheat	108	0
5				Fallow	0	0	Fallow	0	0
6	Spring pea	0	0	Wheat	0	0	Rapeseed	178	0
8				Soybean	80	0	Soybean	50	0
10	Sunflower	0	0	Wheat	134	0	Soybean	40	0

10 cores, depending on the size of the field, were taken from each field on each sampling date. Each sample was first divided into four layers of 30 cm and then mixed between the 6 and 10 cores layer by layer before analysis. The first 30 cm corresponded to the plowed horizon; the other layers did not correspond strictly to pedological horizons but were selected to evaluate the capacity of the STICS soil-crop model to simulate water and nitrate movement inside the soil profile. The cores were homogenized and moisture content was measured after drying at 105 °C for 24 h. Samples were extracted with 1 mol L^{-1} KCl solution per 100 g of fresh soil, and nitrate and ammonium contents were measured by continuous flow colorimetry (autoanalyzer, Skalar Analytical).

Soil moisture at field capacity and at wilting point was estimated from gravimetric *in situ* soil measurements. Field capacity of each 30 cm layer was estimated from soil cores sampled 2–3 days after rainfall events during the three winters studied. Wilting point moisture was estimated from measurements made at the end of summer and the beginning of autumn after the crops were harvested. These values are summarized in Table 2. The estimated field capacity values were in good agreement with those estimated using the pedotransfer function developed by Saxton and Rawls (2006), while the wilting points were generally a little lower (1–2%) than those estimated with this function. Moreover, *in situ* soil moisture measurements are more representative than standardized lab experiments (soil homogenized and sieved through 2 mm mesh) carried out on de-structured soil (Mary et al., 1999). For the fields studied, the available soil water for crops varied between 190 and 235 mm to a depth of 1.2 m.

The STICS model was initialized once using soil water and mineral-N contents measured in February 2005 for fields 1, 6 and 10, where spring crops were sown in 2005; and with the corresponding data from December 2005 for fields 2, 3, 5 and 8, where winter crops were sown. The output variables used for model evaluation were the water and mineral-N contents in the whole 1.2 m deep soil profile.

2.2.2. *Sampling in supplementary fields and modeling of the whole study area*

In order to evaluate SMN variability for the whole study area, all fields were sampled in 2007, specifically in July after harvesting of the winter crops (wheat, rapeseed, winter pea) and in early November after harvesting of the spring crops (maize, sorghum, sunflower, melon, and soybean). These fields were sampled and analyzed (water and mineral-N contents) using the same methods as for the monitored fields. While measured SMN and SWC were used as the initialization data for the monitored fields, initialization for the other fields in the alluvial zone was performed in November 2005 using values obtained by inversion of the STICS model in order to minimize differences between predicted and measured SWC and SMN values at harvest 2007. The initial SWC values obtained were close to field capacity as for the monitored fields. Initial SMN values varied between 25 and 300 kg N ha^{-1}, which is the same range of variation as for the monitored fields. Using this method and the estimated initial values, the simulated SMN values at harvest in 2007 were in reasonably good agreement with the measured values. The bias was small (ME = 3.1 kg N-NO_3 ha^{-1}) and RMSE was fairly good (RMSE = 26.9 kg N-NO_3 ha^{-1}). It was then possible to run the model and to calculate nitrate leaching for each field in the study area. As the simulations of the 7 monitored fields showed that drainage was either nil or very low during the 2004–2005 crop sequence, the simulations were only performed on the 2005–2006 and 2006–2007 crop sequences, which are presented in Fig. 1b. Crop management practices were assessed using data on real farm practices collected in surveys of the farmers who manage the monitored fields.

2.3. *Model evaluation*

The statistical evaluation of the model focused on both SMN and SWC measured on the sampling dates. Three statistical criteria were used (Smith et al., 1996):

Model efficiency (*EF*): optimal value = 1

$$EF = 1 - \frac{\sum_{i=1}^{n}(Pi - Oi)^2}{\sum_{i=1}^{n}(Oi - \bar{O})^2}$$

Mean error (*ME*) and its relative value in % (*ME*%): optimal value = 0

$$ME = \frac{1}{n}\sum_{i=1}^{n}(Oi - Pi); \qquad ME\% = \left(\frac{ME}{\bar{O}}\right) \times 100$$

Root mean square error (*RMSE*) and its relative value (*RMSE*%): optimal value = 0

$$RMSE = \sqrt{\frac{1}{n}\sum_{n=1}^{n}(Oi - Pi)^2}; \qquad RMSE\% = \left(\frac{RMSE}{\bar{O}}\right) \times 100$$

where n is the number of observations, Oi the observed value, $\bar{O}$ the mean of the observed values, and Pi the value predicted by the model.

A model efficiency level higher than 0.6 is generally accepted as very efficient. A mean error (%) and a Root mean square error (%) lower than 15% can be considered very efficient considering all the processes simulated and the simplifications used in the model (Smith et al., 1996).

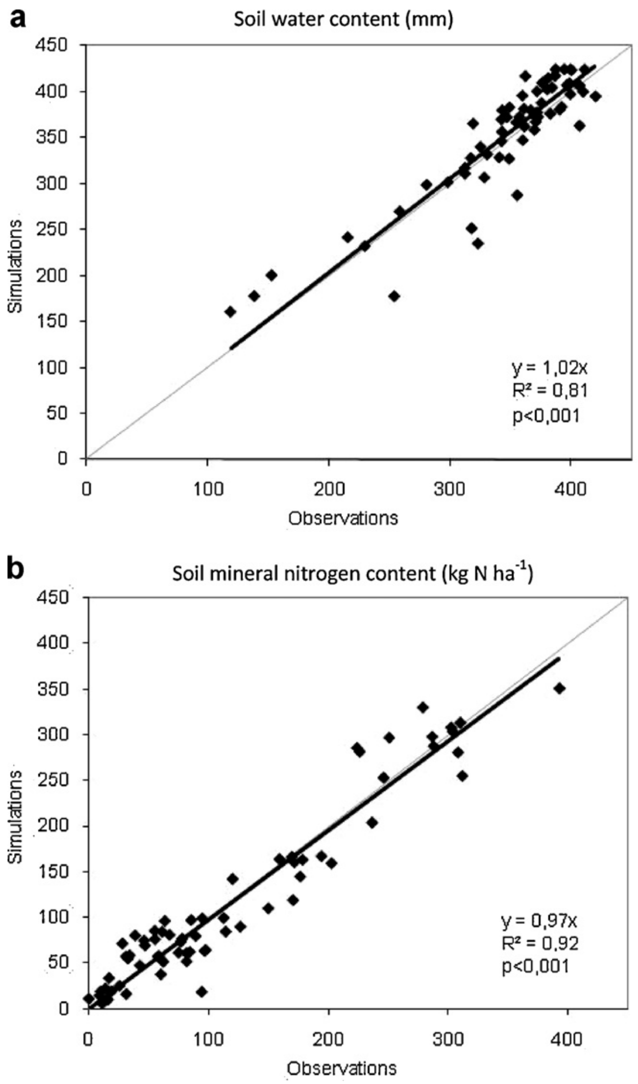

Fig. 2. Observed and simulated values of (a) soil water and (b) soil nitrogen content over a 1.2 m depth in the monitored fields.

2.4. *The STICS model*

This study was carried out using the STICS model, which was mainly developed at the National Institute of Agronomic Research (INRA) in France. STICS is a dynamic soil-crop simulation model functioning at the daytime scale (Brisson et al., 1998, 2002, 2003, 2008). The crop of interest is characterized by its aboveground biomass (carbon and nitrogen), leaf area index, and number and biomass (carbon and nitrogen) of harvested crop organs. The soil description includes four compartments: microporosity (or textural porosity), macroporosity (or structural porosity), fissures (in the case of swelling clay soils) and stones (various types of stones according to their porosity and water storage). The soil is divided into a maximum of 5 horizons but calculations of microporosity are done per 1 cm layer, which is the resolution required to derive nitrate concentration with relevance as shown by Mary et al. (1999). Water transport in soil micropores was calculated for each 1 cm layer using a tipping bucket approach. The daily water budget allows calculation of the water status of the soil, including actual evaporation and crop transpiration, as well as indices of water stress, which reduce leaf growth and net photosynthesis of plants. It is based on estimating the water requirements of the soil–leaf system on the one hand and on the water supply to the soil–root system on the other. The daily nitrogen budget takes into account mineralization from humus and crop residues, denitrification, nitrogen absorption and symbiotic N_2 fixation for leguminous crops.

In the STICS model, the soil is characterized by thickness, bulk density, field capacity and wilting point values for each layer; these properties needed to be specified for each layer whose depth is determined by the user (actual pedological or sampling depth). Other soil property data are required to run the model, such as the organic N, clay, pH and carbonate contents in the plowed layer; these parameters drive the soil N mineralization simulation. The last inputs required are climate data, such as daily minimum and maximum temperatures, solar radiation (global incoming energy), rainfall and calculated potential evapotranspiration. For crop management, the model requires data on sowing (date, depth and density), mineral and organic N fertilization, irrigation and soil tillage with plowing of crop residues and organic products. The model can be used on successive crop sequences without re-initialization every year. Soil water, mineral nitrogen, organic nitrogen and carbon are updated after each crop cycle. Decomposition of crop residues is also taken into account from harvest to the next crop. The STICS model was initially parameterized and validated for bare soil and wheat and maize crops (Brisson et al., 1998), but it has since then been adapted for other crops such as rapeseed, sunflower, soybean, flax, tomato, sorghum, lettuce, white mustard, sugar beet and potato (Brisson et al., 2003). More than 200 output variables can be simulated daily, such as (i) soil water and nitrate contents in each layer, (ii) crop water and nitrogen uptake, and (iii) water drainage, nitrate leaching and nitrate concentration—the output variables highlighted in this study. As the STICS model had previously been calibrated and validated for all the crops studied in the present work (Brisson et al., 1998, 2003), we used the model without any specific calibration of crop parameters. Moreover, no

Table 4
Validation results of simulated soil water content and mineral-N content.

	Field 1	Field 2	Field 3	Field 5	Field 6	Field 8	Field 10	All fields
Soil water content on 0–120 cm (mm)								
Obs number	11	9	7	8	9	9	13	66
ME	−9.7	−11.9	−25.7	27.7	−23.1	2.0	−4.5	−10.3
ME (%)	−3.5	−3.4	−6.7	7.4	−6.3	0.5	−1.4	−3.0
RMSE	16.6	22.2	28.8	46.2	32.8	25.6	30.2	23.7
RMSE (%)	6.0	6.4	7.6	12.3	8.9	6.7	9.1	6.8
EF	0.92	−0.51	−4.70	−1.41	0.82	0.12	0.81	0.84
Soil mineral-N content on 0–120 cm (kg N ha^{-1})								
Obs number	11	9	7	8	9	9	13	66
ME	−6.2	−11.2	9.0	−4.0	18.8	8.0	2.4	0.5
ME (%)	−7.0	−4.8	3.0	−31.2	14.0	9.0	3.6	0.4
RMSE	28.1	31.2	35.5	7.6	33.9	27.7	19.4	27.7
RMSE (%)	31.6	13.5	12.0	58.7	25.2	30.9	29.5	22.8
EF	0.59	0.90	0.51	−0.01	0.68	0.67	0.56	0.92

specific calibration was carried out for any soil or crop processes since the model can simulate a wide range of pedoclimatic and cropping system conditions (Brisson et al., 2003).

3. Results and discussion

3.1. Model validation

For the seven monitored fields, the simulated values were in good agreement with the observed data (Fig. 2). For soil water content (SWC) the simulations were satisfactory (Fig. 2a; $R^2 = 0.81$; $P < 0.001$). However, there was a small tendency for the model to overestimate the lower SWC values. Overall, SWC was overestimated by only 10.3 mm (3.0%) on average (Table 4). The ME values were low and ranged from −25.7 mm to 27.7 mm. There was a slight overestimation for fields 1, 2, 3, and 6 and 10, and a slight underestimation for fields 5 and 8. The prediction error (RMSE) was low and varied between 6.0 and 12.3%. Model efficiency was satisfactory for fields 1, 6, and 10. For field 8, efficiency was slightly higher than zero, while for the other fields (fields 2, 3 and 5) model efficiency was less than zero. This low efficiency could be explained by the small range of variation in the observed data. Thus the model was not able to simulate very small variations in water content (a few mm of water for 1.2 m soil depth), which could have been partly due to measurement precision.

With regard to soil mineral nitrogen (SMN), the model was able to correctly simulate the observations without any bias (Fig. 2b; $R^2 = 0.92$; $P < 0.001$). The range of values of the observed data was large, indicating that the model has sufficient sensitivity to simulate large soil mineral-N variations. The SMN content was overestimated by only 0.5 kg N ha^{-1} on average. This good overall ME was partly due to compensation effects between fields. The soil mineral-N was slightly overestimated in fields 1, 2 and 5, and slightly underestimated in fields 3, 6, 8 and 10. The prediction error (RMSE) varied between 12.0 and 58.7%. The model efficiency was good (>0.6) for fields 1, 2, 6, and 8 and less satisfactory for fields 3 and 10. For field 5 (fallow land used for grazing), the efficiency was less than zero; the high ME and RMSE (31.2% and 58.7% respectively) and the poor efficiency (almost zero) could be explained by the low values and variation in soil mineral-N. Nevertheless, the overall trend for field 5 and the level of concentration were correctly simulated. In general, the values of ME and RMSE were in good agreement with those reported by Schnebelen et al. (2004), Beaudoin et al. (2008) and Jégo et al. (2008) for predicting SMN using the STICS model for various arable crops. Furthermore, it is noteworthy that the ME, RMSE and EF were almost similar in the four soil layers studied, indicating that the STICS model can simulate dynamic SWC and SMN profiles.

Fig. 3 shows the temporal changes in simulated soil moisture and SMN in comparison with observed data for each soil layer in field 1 as an illustration of model performance. These results indicate that the model was able to correctly simulate the temporal changes in water and mineral-N quantities in the different soil layers. The seasonal variations in soil moisture were significant. Summers were characterized by a large decrease in soil moisture, up to values close to wilting point (at least for the three first horizons). However, the crop sequence had an impact on the temporal pattern of these variations. The shorter period characterized by moisture at field capacity during 2005–2006 compared with 2006–2007 could be explained by the longer period of bare soil between pea harvest (2006) and maize sowing (2007) as compared with the period between soybean harvest (2005) and pea sowing (2006). In the deepest soil layer (90–120 cm), soil moisture decreased significantly only during summer 2005.

SMN values in the uppermost layer (0–30 cm) increased in the spring because of soil organic matter and crop residue N mineralization (and N fertilization in 2007). The decreases in SMN observed after each of these three increases were due to crop N uptake and nitrate transfer to deeper soil layers. In the 30–60 cm and 60–90 cm layers, SMN decreased rapidly after the beginning of the simulation due to N absorption by soybean. SMN initially increased at the beginning of 2006 due to nitrate transfer from the upper layer and decreased thereafter because this significant amount of nitrate was transferred to the lower layer. The next increase was also due to nitrate transfer from upper layers. Finally, SMN decreased because of transfer to lower layers and N uptake by the maize crop. In the deepest layer (90–120 cm), SMN increased from March 2006 due to nitrate transfer.

Simulated soil water and nitrogen levels were in good agreement with the measured values despite the wide range of agronomic (crop type, fertilization and irrigation) and environmental conditions encountered during our study. Although for some fields and some sampling times, the simulations were not always completely satisfactory in terms of absolute values, the trends and range of variation were satisfactory for all fields. The good agreement between simulated and measured values provides confidence in the simulations of nitrate leaching and water drainage fluxes. Moreover, it can be postulated that the model correctly simulated (i) the N mineralization dynamics of soil organic matter and the decomposition of crop residues, and (ii) water and nitrate transfer within the soil profile, because no bias was observed in the simulation of SWC and SMN over the entire year for all the monitored fields. This is particularly true during the long bare soil period between two main crops (spring crop sown after winter crop, *e.g.* maize after winter wheat), where no interaction occurred with plant N uptake. The model could then be used to evaluate the relative effects of different input variables on nitrate leaching, as also shown by other authors (*e.g.* Beaudoin et al., 2005).

3.2. Evaluation of spatial and temporal variability in nitrate leaching

3.2.1. Simulated temporal variations in the three cropping years

Temporal changes in simulated drainage, nitrate leaching and nitrate concentration are illustrated in Fig. 4 for fields 1 and 6, which are representative of the seven monitored fields. Due to the low level of precipitation during the previous year, soil moisture was below field capacity during the 2004–2005 winter, which led the model to simulate no drainage or nitrate leaching in field 1 (Fig. 4a and b). In this field, like in other monitored fields, two periods of drainage occurred, as indicated by the measured soil moisture and water balance. The temporal pattern and duration of these periods can vary from field to field according to the cropping sequence. In 2006, the month of March was rainy (107 mm) and the soil micropores were saturated to a depth of 1.2 m during this period. This high rainfall combined with the bare soil was responsible for the first significant simulated drainage event (13 mm) for field 1. From autumn 2006 until spring 2007, all soil layers were close to field capacity, hence every new rainfall event generated drainage, as simulated by the model (Fig. 4a). For field 1, the simulation indicated significant nitrate leaching of 113 kg N ha^{-1} during the study period and a considerable variation in nitrate concentration in drainage water, that is, 50–240 mg NO_3^- L^{-1} (Fig. 4c). The weighted average nitrate concentration over the whole period was 190 mg NO_3^- L^{-1}.

Fig. 4 also shows temporal changes in simulated drainage (Fig. 4d), nitrate leaching (Fig. 4e) and nitrate concentration in drainage water (Fig. 4f) for field 6. In this field, the temporal variation in simulated drainage and nitrate leaching was slightly different from that in field 1. The first significant simulated drainage event occurred in February 2005, earlier than in field 1, and the

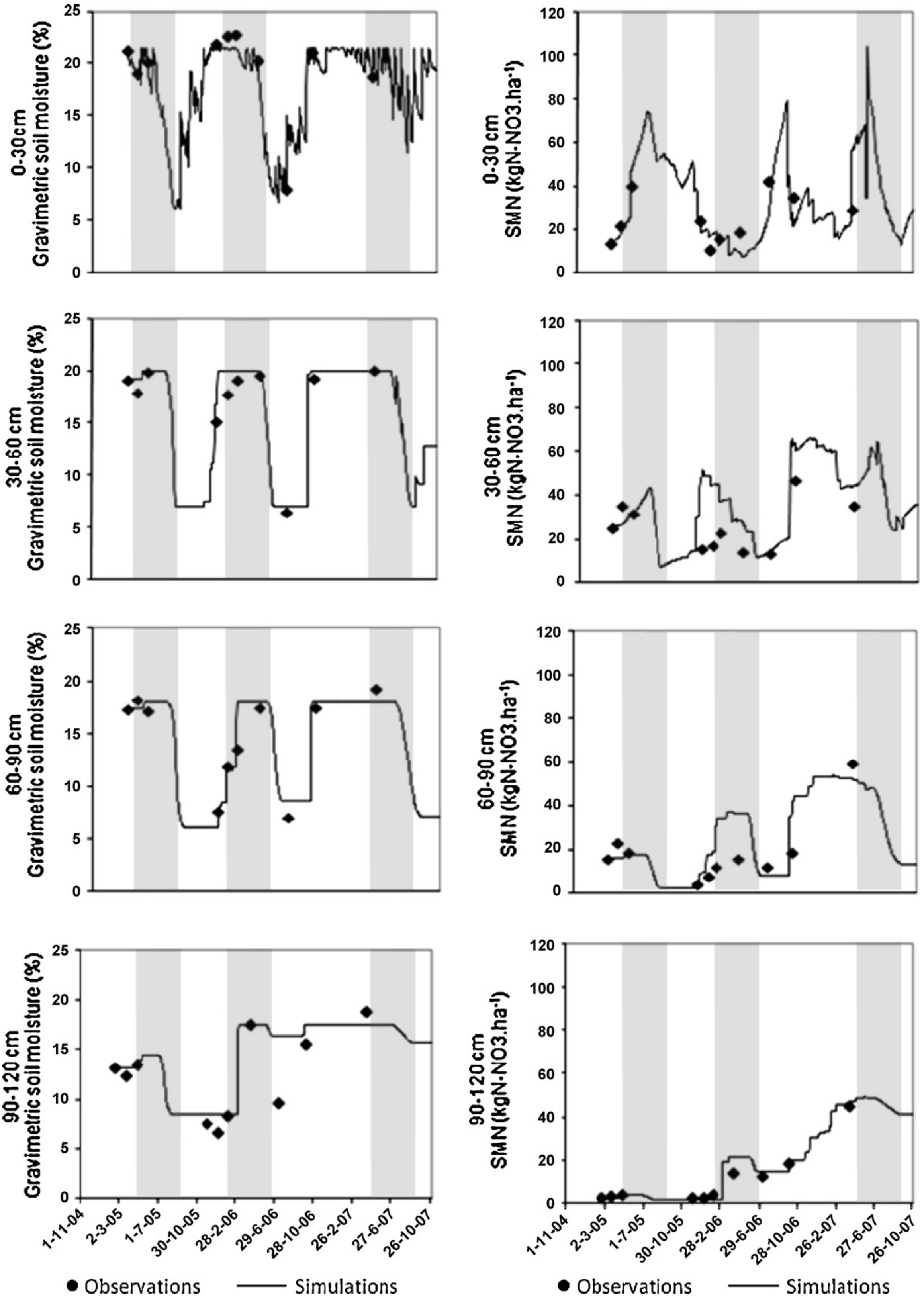

Fig. 3. Temporal changes in observed and simulated gravimetric soil moisture and SMN for field 1 at different soil depths. Cropping periods are indicated in gray.

second drainage flux occurred in April 2005. These drainage events may be explained by the initial SWC, which was higher in field 6 than in field 1 because of the preceding crop. The simulated main drainage period (almost 200 mm) occurred in field 6 in the period from November 2005 to April 2006, in spite of the presence of a winter wheat crop, and the simulated amount of nitrate leaching was 44 kg N ha^{-1} (Fig. 4e), or half that in field 1. Most of this nitrate leaching occurred during the 2006 drainage period with the winter wheat crop being present, when the amount of drainage water was significant and associated with nitrate concentrations

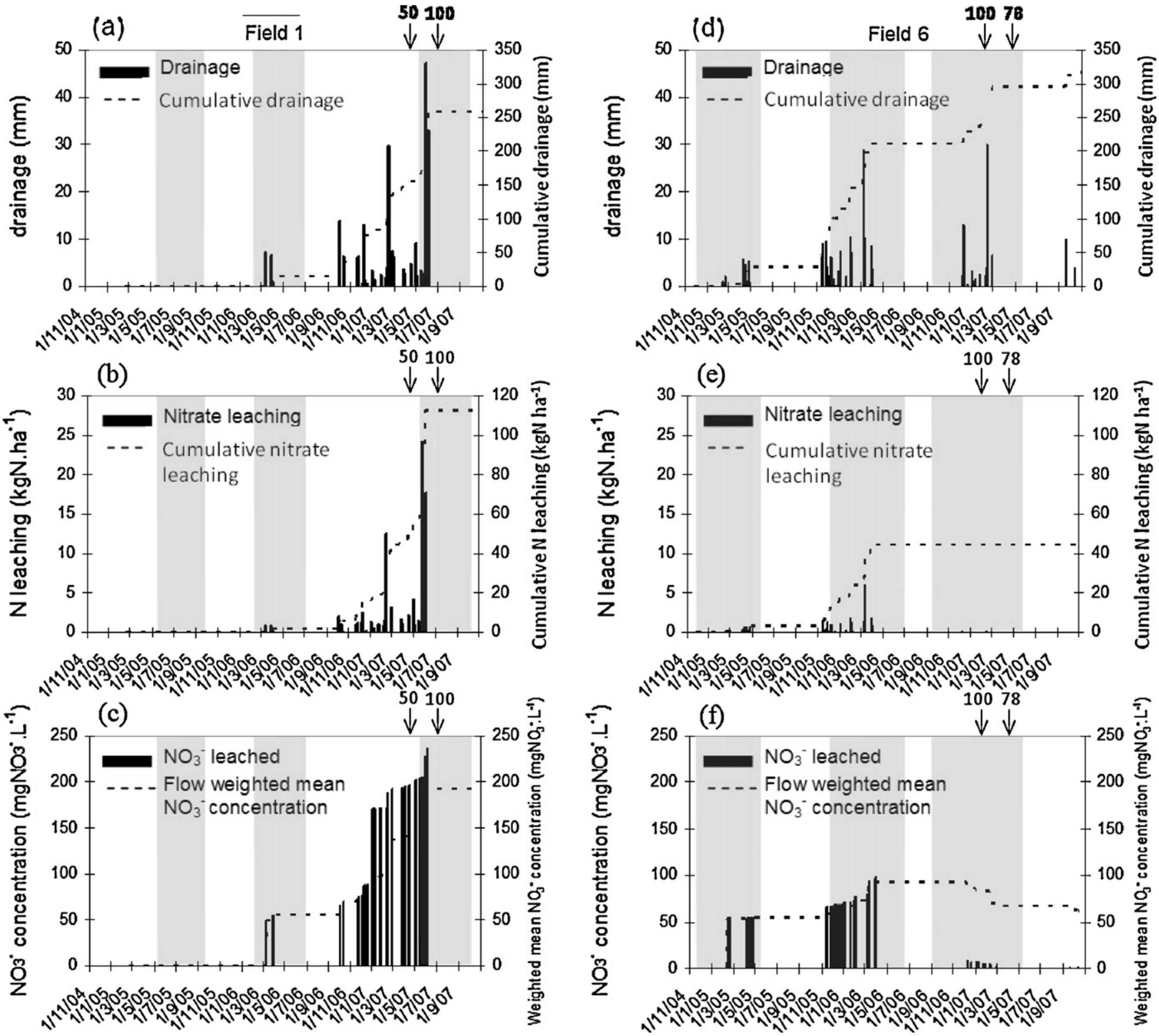

Fig. 4. Simulated temporal changes (a and d) in drainage and cumulative drainage, (b and e) in nitrate-N leaching and cumulative N leaching, and (c and f) in instantaneous nitrate concentration in drainage water and flow weighted mean NO_3^- concentration in fields 1 and 6. Arrows indicate time and amount of N-fertilization. Cropping periods are indicated in gray.

varying between 60 and 100 mg NO_3^- L^{-1} (Fig. 4f). Finally, the simulations indicated there were two main periods of drainage in 2007, January–March and September–October.

Fields 1 and 6 showed two different patterns of temporal distribution of drainage and also of nitrate leaching. Spatial analysis of nitrate leaching in all fields in the study area could help to determine which of these two patterns of distribution is dominant.

3.2.2. *Simulated spatial variation in nitrate leaching*

Fig. 5 shows the cumulative nitrate leaching simulated during two successive crop sequences (2005–2006 and 2006–2007) on all fields in the study area. The 2004–2005 cumulative leaching results are not presented because there was almost no drainage during that period. In the six monitored agricultural fields, nitrate leaching ranged from 5 to 160 kg N ha^{-1} in 2005–2006 and from 5 to 120 kg N ha^{-1} in 2006–2007. In the fallow field (5), nitrate leaching was still predicted to be less than 5 kg N ha^{-1}. In all other agricultural fields, average nitrate leaching was slightly higher in 2006 (38 kg N ha^{-1}) than in 2007 (23 kg N ha^{-1}) ($P < 0.05$). Overall, most of the nitrate leaching occurred during spring 2006, as illustrated for field 1 (Fig. 4b). The level of nitrate leaching was lower during winter 2006–2007 and spring 2007. Nevertheless, for both years, the range of variation in nitrate leaching between the fields was quite large. In 2006 and 2007, just 15% of the fields accounted for 60 and 67% of nitrate leaching, respectively. Nitrate pollution of groundwater is often called “diffuse pollution” in reference to the polluter pays principle. However, in the study areas, the nitrate leaching was associated with point source pollution (at the field scale) and was characterized by considerable spatial variation within a short distance and by temporal variations. There was no significant difference in simulated nitrate leaching between the two main crops in the area (wheat and maize) in either 2006 or 2007. Moreover, there was no significant difference in nitrate leaching among the other crops because of the high spatial variability

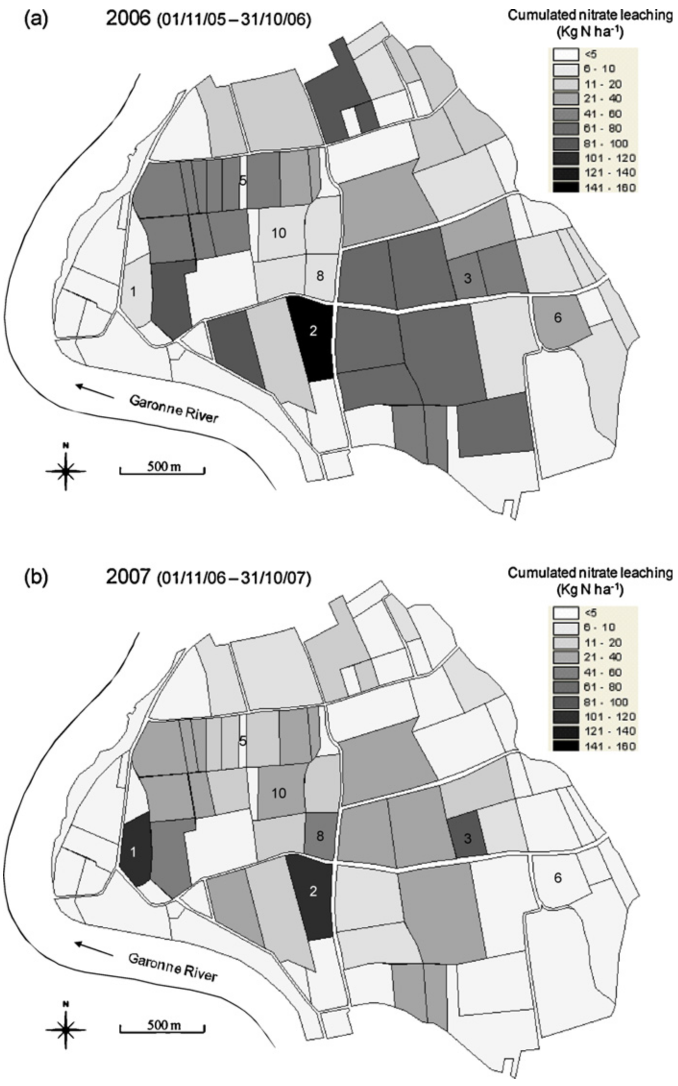

Fig. 5. Spatial distribution of simulated nitrate leaching at 1.2 m depth under all fields in the study area (a) in 2005–2006 and (b) in 2006–2007.

between fields. However, in 2007, simulated nitrate leaching was significantly higher ($P < 0.05$) from the maize and cereal cultivar testing trial fields, whereas in 2006 the difference was not significant ($P = 0.48$). These fields, used for the assessment of new cultivars by breeders, were generally overfertilized in order to avoid the risks of crop nitrogen deficiency.

Soil type did not induce significant differences in simulated nitrate leaching, but initial SMN had a significant impact. In 2006, nitrate leaching was significantly correlated with initial SMN ($y = 0.26x + 8.8$; $R^2 = 0.43$), while in 2007 the correlation was not significant. In 2007, the impact of the new cultivar trial fields of maize and cereal was predominant.

As shown in this work and in several previous studies (Shepherd and Lord, 1996; Beaudoin et al., 2005), SMN at harvest was the key factor explaining the variation in nitrate leaching. SMN measured at harvest (July for winter crops and November for spring crops) in the 40 fields of the area (the seven monitored fields in 2005, 2006 and 2007, and 19 additional fields sampled in 2007) showed that there was no significant difference between SMN at harvest over a depth of 0–1.2 m for the two main crops, despite differences in average values (wheat: 60 ± 13 kg N ha^{-1}; maize: 78 ± 25 kg N ha^{-1}). The mean SMN after wheat to a depth of 1.2 m was higher than the 35–40 kg N ha^{-1} reported by Makowski et al. (1999) and Beaudoin et al. (2005) over a depth of 0–1.2 m in northern France. However, over a depth of 0–60 cm, the measured SMN in a wheat field at harvest (38 kg N ha^{-1}) is somewhat lower than the 43 kg N ha^{-1} over 0–60 cm reported by Webster et al. (2003) in the UK. For the other crops, SMN was not significantly different between wheat and maize but it was lower for rapeseed (25 ± 7 kg N ha^{-1}) and considerably higher for two fields of trial maize lines (215 ± 25 kg N ha^{-1}). These maize lines generally take up less nitrogen than commercial hybrid crops due to their smaller size and slower growth rate. This value indicates that the N fertilizer application rates were not well tailored to these crops, which could explain why the SMN at harvest was so high in these fields.

The SMN contents measured at harvest in the seven monitored fields in 2007 were in agreement with the SMN simulated by STICS (ME = 15.4%; RMSE = 30.5%). Measured SMN contents were also in good agreement with the mean SMN measured in all the fields of the study area at harvest except for fields 2 and 8, where the SMN over a depth of 0–1.2 m was slightly higher than the range of variation in the other fields planted to the same crop. This indicates that, overall, the monitored fields were representative of the fields located in the study area.

Simulated drainage, nitrate leaching and nitrate concentration are detailed in Table 5 for the monitored fields. The drainage and nitrate concentration were extremely variable from one field to another, even within this small study area with its fairly homogenous pedoclimatic conditions and stockless farms. In all cases, the nitrate concentrations in drainage water were considerably higher than those in the Garonne River and, except for a few instantaneous fluxes and the values obtained for the fallow field used for grazing, the simulations indicated that they were greater than 50 mg NO_3^- L^{-1}. The simulated mean weighted nitrate concentrations were extremely high in 2006 for fields 2 and 3 (241 and 334 mg NO_3^- L^{-1} respectively), and in 2007 for fields 1, 2 and 3 (213, 385, and 307 mg NO_3^- L^{-1} respectively). The amount of drainage water and its nitrate concentration below a depth of 1.2 m could explain why the nitrate concentration in the alluvial aquifer could reach values up to 60 mg NO_3^- L^{-1} in some piezometers of the alluvial groundwater. As shown for fields 1 and 6, drainage was more significant in 2006 and 2007 compared to 2005 (almost negligible), which is explained by rainfall variability. The simulated nitrate leaching values were higher than 50 kg N ha^{-1} for field 1 in 2007, for field 2 in 2006 and 2007, for field 3 in 2006 and 2007, and for field 8 in 2007, although plant N uptake and yields fell within the range of variation of the study area (Table 5). The high level of nitrate leaching was associated with high nitrate concentrations in drainage water (>100 mg NO_3^- L^{-1}), which were generally associated with high initial SMN. In the unfertilized fallow land used for grazing, the simulation showed that significant drainage occurred only in 2007 (63 mm) but, owing to the green cover throughout the year, the nitrate concentrations were very low (<10 mg NO_3^- L^{-1}).

3.2.3. *Relationship between simulated nitrate leaching and agricultural practices*

Our work involved analyzing and explaining the high spatial variability of the piezometer measurements in the study area (Sánchez-Pérez et al., 2003b) in connection with simulated drainage, nitrate leaching and nitrate concentration in drainage water at the alluvial floodplain scale. The relationships between nitrate leaching and drainage, and between nitrate leaching and nitrate concentration in drainage water were not significant, indicating that drainage and nitrate concentration in drainage water were not directly linked.

Some studies have shown that previous crop type has an impact on nitrate leaching (Shepherd and Lord, 1996; Hall et al., 2001; Jégo et al., 2008). However, in our study area no significant relationship was found between SMN at harvest and crop type, probably because

Table 5
Input data and simulated output variables of the seven monitored fields.

Input data						Outputs variables simulated					
Year	Simulated period	Field	Crop	Rainfall (mm)	Initial SMN (kg N ha^{-1})	Drainage (mm)	Nitrate leaching (kg N ha^{-1})	Nitrate concentration in drainage water (mg NO_3^- L^{-1})	SMN at harvest (kg N ha^{-1})	Plant N-uptake (kg N ha^{-1})	Dry yield (t ha^{-1})
2005	18/02/05–30/11/05	1	Soybean	350	58	0	0	–	75	179	1.0
	25/11/04–30/11/05	6	Spring pea	425	159	74	10	63	167	231	4.3
	18/02/05–30/11/05	10	Sunflower	350	74	77	7	39	53	139	5.3
2006	01/12/05–31/10/06	1	Winter pea	644	75	37	6	69	162	233	3.0
	01/12/05–31/10/06	2	Maize	644	303	188	142	334	207	177	6.3
	01/12/05–31/10/06	3	Sorghum	644	287	93	51	241	248	230	7.0
	01/12/05–31/10/06	5	Fallow	644	25	0	0	–	14	60	–
	01/12/05–31/10/06	6	Wheat	644	167	137	34	110	65	169	5.8
	01/12/05–31/10/06	8	Soybean	644	58	156	9	24	101	177	4.1
	01/12/05–31/10/06	10	Wheat	644	53	109	6	23	55	163	7.4
2007	01/11/06–31/10/07	1	Maize	600	162	222	107	213	119	240	9.6
	01/11/06–31/10/07	2	Wheat	600	207	120	104	385	125	215	5.1
	01/11/06–31/10/07	3	Wheat	600	248	118	81	307	75	258	10.0
	01/11/06–31/10/07	5	Fallow	600	14	63	<1	<1	10	40	–
	01/11/06–31/10/07	6	Rapeseed	600	63	106	<1	<1	87	277	3.3
	01/11/06–31/10/07	8	Soybean	600	101	235	60	112	81	147	2.9
	01/11/06–31/10/07	10	Soybean	600	54	253	30	53	41	165	4.2

Table 6
Simulation of the impact of initial SMN in fields 2 and 3 on water and nitrate fluxes. Simulated output variables using actual initial SMN are compared with model simulations using an average initial SMN of 80 kg N ha^{-1}.

Input data				Simulated output variables			
Field	Simulation period	Crop	SMN initial (kg N ha^{-1})	Drainage (mm)	Nitrate leaching (kg N ha^{-1})	Nitrate concentration in drainage water (mg NO_3^- L^{-1})	Final SMN (kg N ha^{-1})
2	1/12/05–31/10/06	Maize	303	188	142	334	207
			80	189	36	84	112
	1/11/06–31/10/07	Wheat	207	120	104	385	125
			80	128	15	52	131
3	1/12/05–31/10/06	Sorghum	287	93	51	241	248
			80	93	12	57	141
	1/11/06–31/10/07	Wheat	248	118	81	307	75
			80	117	9	34	83

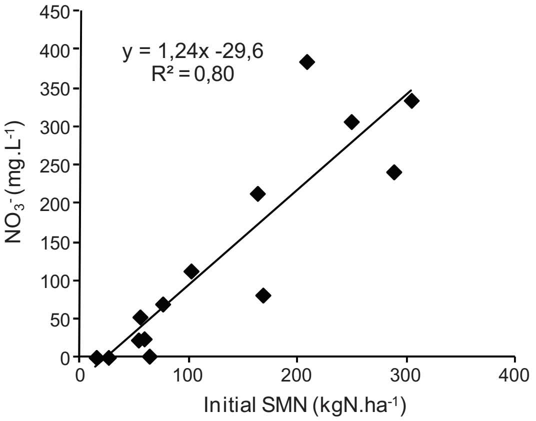

Fig. 6. Simulated nitrate concentration in drainage water as a function of initial SMN in 2006 and 2007.

fertilizer-N is not properly adjusted to meet crop requirements for some crops (*e.g.* new maize cultivar trials) and also because farmers do not analyze SMN as part of their approach for adjusting fertilizer-N. In addition, the three preceding years were very dry and SMN could have accumulated in the soil. Thus, no relationship was found between previous crop and simulated drainage, nitrate leaching, or nitrate concentration (Table 5). There was also no significant correlation between nitrate concentration or nitrate leaching and quantity of N fertilizer applied (Fig. 4a, c, d, and f). This was probably due to the fact that fertilization rates were not adjusted based on the initial SMN level. Consequently, the nitrate concentration could vary widely for the same fertilizer application rate. The adjustment of N-fertilization based on the initial SMN level would have decreased the nitrate concentration in drainage water, as reported in other studies (Hansen and Djurhuus, 1996; Mary et al., 2002; Ferguson et al., 2002) and demonstrated by the scenario with reduced initial SMN.

In the study area, soil type and depth were almost homogenous and although many studies have shown that soil type has an important impact on nitrate leaching (Nieder et al., 1995; Simmelsgaard, 1998; Hoffmann and Johnson, 1999), no significant impact was observed in our study. The small variability of soil properties and the small number of agricultural fields with soil 1 (3 fields), soil 3 (9 fields) or soil 4 (12 fields), compared to soil 2 (26 fields), could explain why no significant relationship was found.

The initial soil nitrogen content, that is, the mineral-N present in the whole 1.2 m depth profile at the beginning of the study and simulation period, was positively and significantly correlated with the nitrate concentration in drainage water (Fig. 6). This is in agreement with other studies (Arregui and Quemada, 2006), which reported that SMN content before planting, together with drainage, was the main factor determining the amount of N leached and thus the nitrate concentration in drainage water. The initial SMN value was particularly high in fields 2 and 3 and, as a consequence, the mean simulated nitrate concentration in drainage water was also very high. In order to examine the importance of initial soil mineral N content, we used scenarios with lower initial SMN values. The simulations of the 2005–2006–2007 crop sequences showed that the nitrate concentration in drainage water from fields 2 and 3 was very high. The use of a catch crop was not possible in these situations because the period of bare soil between main crops was too short. A strong positive correlation was found between these nitrate concentrations and initial SMN. In order to examine this relationship more closely, we carried out simulations for these two fields with an initial SMN of 80 kg N ha^{-1}. For the two fields and for the two successive years simulated, decreasing the initial SMN led to a large decrease in nitrate leaching and nitrate concentration in drainage water, without affecting the main crop yields (Table 6). These results illustrate the importance of reducing high SMN contents during autumn before the winter drainage period in order to reduce nitrate leaching. In such a situation, a catch crop may be a solution for reducing nitrate leaching (Thorup-Kristensen et al., 2003).

The SMN level in late autumn, before winter drainage, was found to be the main contributing factor. This demonstrates that N management was unsatisfactory in the medium term and that cumulative problems associated with unsuitable agricultural practices may be more detrimental for N management than a single annual case of nitrogen overfertilization in cases of deep alluvial soils, particularly in situations of low or highly variable drainage between years.

4. Conclusions

The simulated SWC and SMN values in the dynamic simulations were generally and specifically (temporal and between-layer changes) in good agreement with the measured values. These satisfactory results allowed the model to be used to simulate the temporal and spatial variability in nitrate leaching to provide a diagnostic assessment of the situation. This work could provide the basis for future studies to assess the impact of modifications of agricultural practices aimed at decreasing nitrate leaching and nitrate concentration in drainage water.

There was no significant difference in SMN values at harvest or in nitrate leaching for the different main crops in the study area, although large between-field variations were observed. Nitrogen management in this part of the alluvial floodplain was not effective and hence the nitrate concentrations in drainage water under crops were too high. Drainage and nitrate concentration values varied widely from one field to the next, depending on the previous crop, agricultural practices (with or without irrigation) and annual climate conditions. For some fields, the average annual nitrate concentration in drainage water was greater than 200 mg NO_3^- L^{-1} and nitrate leaching exceeded 100 kg N ha^{-1}. Analyses of temporal and spatial variability in nitrate leaching showed that the pattern of nitrate leaching was extremely specific and irregular (spatially and temporally) and also that the SMN content at the end of autumn, before the winter drainage period, was the most significant factor explaining this variability. For the study area, this means that N management must be aimed at reducing SMN as much as possible in November. This means that N fertilization for the next main crop must be adjusted by taking into account the residual SMN at the beginning of the crop season (soil analysis may be necessary) and by planting catch crops to decrease SMN before the winter.

In order to complement this work and to better assess the impact of the spatial and temporal distribution of nitrate leaching under agricultural fields on the nitrate concentration in groundwater and in the Garonne River, the STICS soil-crop model could be coupled with a hydrogeological model. This would permit simulation of (i) the impacts of agricultural activities on groundwater nitrate concentration and its spatial variability and (ii) the interactions between river water and groundwater. In the case of large rivers such as the Garonne River, groundwater can be influenced by river water several hundred meters from the riverbank. This would make it possible to simulate the impact of agricultural practices on nitrate concentrations in groundwater in a portion of the alluvial plain and better explain the spatial variability of nitrate concentrations in the groundwater. The performance of the coupled model could be evaluated using the groundwater nitrate concentration measured in the piezometers that have been used on this site for several years.

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