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Assessing nitrate leaching during the three-first years of *Miscanthus × giganteus* from on-farm measurements and modeling

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Abstract

Miscanthus × giganteus is often regarded as one of the most promising crops to produce sustainable bioenergy. This perennial crop, renowned for its high productivity associated with low input requirements, in particular regarding fertilizers, is thought to have low environmental impacts, but few data are available to confirm this. Our study aimed at assessing nitrate leaching from *Miscanthus × giganteus* crops in farmers' fields, thus including a wide range of soil and cropping system conditions. We focused on the first years of growth after planting as experimental studies have suggested that *Miscanthus × giganteus*, once established, results in low nitrate leaching. We combined on-farm measurements and modeling to estimate drainage, leached nitrogen, and nitrate concentration in drainage water in 38 fields located in Center-East France during two winters (November 2010 to March 2011, November 2011 to March 2012).

Nitrate leaching and nitrate concentration in drainage water were on average very low. Nitrate leaching averaged 6 kg N ha⁻¹ whereas nitrate concentration averaged 12 mg l⁻¹. These low values are attributable to the low estimates of drainage water (mean = 166 mm) but also to the low soil mineral nitrogen contents measured at the beginning of winter (mean = 37 kg N ha⁻¹). Our results were, however, very variable, mainly due to the crop age: nitrate leaching and nitrate concentration were critically higher during the winter following the first growth year of *Miscanthus × giganteus*, reflecting the low development of the crop. This variability was also explained by the range of soil and cropping conditions explored in the on-farm design: shallow and/or sandy soils as well as fields where establishment failed had a higher risk of nitrate leaching.

Keywords: bioenergy crop, environmental impacts, *Miscanthus × giganteus*, nitrate losses, on-farm research, soil mineral nitrogen

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Introduction

Miscanthus × giganteus (hereafter referred to as *M. giganteus*) is often regarded as one of the most promising crops to produce biomass for bioenergy (Lewandowski *et al.*, 2000, 2003; Heaton *et al.*, 2004, 2008a,b, 2010; Hastings *et al.*, 2008, 2009). This tall C4 perennial rhizomatous grass from Asia is well known for producing high yields with low fertilizer and pesticide inputs and its long life span (Lewandowski *et al.*, 2000; Heaton *et al.*, 2004; Miguez *et al.*, 2008; Dohleman & Long, 2009). It is therefore expected to have lower environmental impacts than annual crops (Powlson *et al.*, 2005; Rowe *et al.*, 2009; Heaton *et al.*, 2010), particularly

regarding nitrogen (N) losses in the environment. However, studies assessing the exact amount of nitrogen lost in the form of N₂O, NH₃, and NO_x emissions or nitrate leaching in groundwater are scarce. N losses are still an important consideration when appraising the overall sustainability of producing bioenergy from *M. giganteus*.

Decreasing the risk of nitrate leaching involves better management of the nitrogen cycle, increasing the use of available nitrogen by crops, and reducing soil nitrogen content at the beginning of winter. Fertilizer requirements of *M. giganteus* are less than other crops (Beale & Long, 1997; Cadoux *et al.*, 2012). Compared with annual crops (most of which are harvested during summer) *M. giganteus* can still take up mineral nitrogen during autumn, before the period of heavy rainfall and/or low evapotranspiration (in the ecological conditions

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prevailing in Europe), i.e., when the risk of leaching increases. This late N uptake (associated with the uptake of water over a longer period than for annual crops) as well as the extensive rooting system of the crop (Neukirchen *et al.*, 1999; Monti & Zatta, 2009) could then limit the risk of nitrate leaching during the winter (Powlson *et al.*, 2005; Rowe *et al.*, 2009).

However, few references are available on this aspect. Beale & Long (1997) examined nitrate leaching over a year with deep drainage lysimeters under a 3-year-old *M. giganteus* crop and measured nitrate concentrations averaging 17.7 mg l^{-1} . Christian & Riche (1998) studied nitrogen leaching with porous cups during the three-first growth years and found that leaching was low when *M. giganteus* was unfertilized, except during the first winter following planting, when N losses were almost ten times greater than those measured in the following years. In the same experiment, Christian *et al.* (2008) observed a 10-year mean leaching of 22.4, 26.7, and $62.9 \text{ kg N ha}^{-1}$ when *M. giganteus* received 0, 60, and 120 kg N ha^{-1} , respectively. This experiment also suggested a higher risk of N leaching during the year following crop establishment. McIsaac *et al.* (2010) observed with lysimeters that annual nitrate losses from unfertilized *M. giganteus* were similar to those observed in unfertilized switchgrass (*Panicum virgatum*) and far smaller than under maize (fertilized with 168 or 202 kg N ha^{-1}) rotated with unfertilized soybean. Smith *et al.* (2013) highlighted with tile drains and resin lysimeters that nitrate leaching in *M. giganteus* decreased with crop age, but decreased more slowly in case of establishment problem.

All these experiments provide information on N leaching, but do not cover the range of environmental conditions and cropping systems where *M. giganteus* might be grown. In particular, they do not account for the variable soil, climatic, and growing situations existing in agricultural conditions. They are based on lysimeters, drained perimeters, or ceramic cups, which provide direct measurements of nitrate fluxes and/or water fluxes, but cannot be installed on a wide range of sites. Soil cores provide information on soil nitrogen and water contents but not on fluxes. Standard crop models can predict water and nitrogen fluxes below the rooting zone, but they require more data to be parameterized with good accuracy, and their predictive capacity is often poorly characterized. However, nitrate leaching simulation models such as LIXIM (Mary *et al.*, 1999) allow water and mineral N measurements to be converted into water and nitrate fluxes and are well adapted to account for variable soil, climatic, and growing situations. Our study aimed therefore at assessing the risk of winter nitrate leaching during the first

growth years of *M. giganteus* in a wide range of agricultural conditions, combining soil samplings on 38 farmers' fields and the use of LIXIM.

Materials and methods

Description of the farmers' field network

We investigated a set of 38 farmers' fields located in Burgundy (Center-East France) in a 3000 km^2 area ranging from $46^{\circ}54'$ to $47^{\circ}37'N$ and from $4^{\circ}22'$ to $5^{\circ}46'E$ (Table 1). The field survey was carried out during two growing seasons and two winters: winter 2010–2011 (hereafter referred to as Period 1) and winter 2011–2012 (Period 2). A total of 19 fields were planted with *M. giganteus* in spring 2009 and 17 fields in spring 2010, after two kinds of preceding crops: annual crops (wheat, corn, or sunflower) and set-aside (meadows established for more than 5 years). The fields covered five soil types, depending on depth and texture.

Fields were planted mechanically with rhizomes at densities ranging from 1.53 to 2.51 rhizomes per m^2 (mean = 1.93 rhizomes per m^2). The crops were chemically protected against weeds during the year of establishment and in the second year, before the first regrowth. When necessary, fields were also weeded before the second regrowth. Thirty-three of the 36 fields were unfertilized. The three remaining fields were fertilized with about 30 kg N ha^{-1} in March or April. Shoots were not harvested at the end of the establishment year, but were crushed at the end of December. During the following years, fields were mechanically harvested in late March or early April.

Measurements

Soil mineral nitrogen (SMN), i.e., soil nitrate (SN) and soil ammonium (SA), along with soil water content (SWC), was measured five times from 2010 to 2012 at the end of winter, after harvest (in March 2010, early April 2011, and late March 2012) and at the beginning of winter (in mid-November 2010 and 2011). Samplings were concentrated within a short time period: the delay between the first and the last sampled fields did not exceed 9 days and averaged 5 days. Soil cores were collected down to 120 cm maximum with a hydraulic coring device (auger diameter = 2 cm). Each core was split into four layers (0–30, 30–60, 60–90, and 90–120 cm). Composite soil samples were made by mixing 10 cores collected across the field. The samples were frozen until extraction and subsequent analysis. Nitrate and ammonium were extracted using a KCl solution (1 M) and analyzed by continuous flow colorimetry. A soil subsample was weighed and dried for 72 h at 105°C and gravimetric water content was estimated by measuring the weight loss after drying.

On 20 of the fields, shoot density was measured at the end of the growing season on two 25 m^2 plots including six *M. giganteus* rows. These plots were randomly sited, but precautions were taken to avoid field borders and extreme field areas that were not representative for the field.

Table 1 Characteristics of the farmers' field network

Planting year	Preceding crop	Soil type*	% clay†	% sand†	Max sampling depth (cm)	Number of fields
2009	Set-aside (grassland)	C	46	12	120	2
		CC	37	25	40	1
		A	24	37	40	1
		LC	32	14	120	1
		L	21	22	120	3
	Annual crop	C	48	8	120	4
		CC	34	4	70	1
		A	–	–	–	–
		LC	35	24	120	3
		L	31	12	120	3
2010	Set-aside (grassland)	C	42	11	120	1
		CC	47	12	80	1
		A	14	60	110	1
		LC	34	15	120	2
		L	25	18	120	4
	Annual crop	C	51	11	120	3
		CC	30	31	50	1
		A	28	28	65	2
		LC	–	–	–	–
		L	27	18	120	2

*C, clay soil; CC, calcareous clayey soil; A, alluvial soil; LC, loamy clay soil; L, hydromorphic loamy soil.

†Mean value in the 0–120 cm layer.

LIXIM model

LIXIM (Mary *et al.*, 1999) simulates both water and nitrate fluxes in soils and allows calculation of nitrogen mineralization and leaching in bare soils, assuming that these are the dominant processes affecting nitrogen. The model has been successfully assessed in various field experiments with bare soils (Justes *et al.*, 1999; Mary *et al.*, 1999). As *M. giganteus* does not take up nitrogen or transpire during winter, it can be likened to a bare soil during the period considered here (from late autumn to early spring). LIXIM is a layered, functional model with a daily time step. Input data are SWC, SN, and SA measured in soil cores, standard meteorological data, and simple soil characteristics: bulk density (D_b), water content at field capacity (θ_{fc}), and water content at wilting point (θ_{wp}). Two parameters can be estimated by fitting the observed and simulated SWC and SN in each soil layer at the end of the time interval: the ratio of actual to potential evapotranspiration (k) and the potential rate of mineralization (V_p). In this study, we set k at 0.60, which is a common value for a bare soil (Mary *et al.*, 1999) and fitted V_p for each field.

Step 1: characterizing and explaining the variability in soil mineral nitrogen across fields during the first growth year of *M. giganteus*

We focused our analysis on the soil mineral nitrogen measured at the beginning of winter (SMN_{BW}) as it is a key variable to assess the risk of nitrate leaching during winter. The analysis of variance with a mixed model (R Development Core Team, 2008, version 2.14.2, package lme4) was used to study the effect of soil type and crop characteristics (age, preceding crop) on

SMN_{BW} . A mixed model was used with soil type (SOIL), crop age (AGE), and preceding crop (PREC) as fixed effects, whereas field and year were defined as random effects:

$$SMN_{BW} = \mu + \alpha_1 AGE + \alpha_2 PREC + \alpha_3 SOIL + \beta PREC:SOIL + \gamma_f + \gamma_y + \varepsilon$$

Where μ is the intercept; $\alpha_1, \alpha_2, \alpha_3, \beta$ are unknown parameters; γ_f and γ_y stand for the random effects associated with the field and year, respectively, and follow a normal distribution $\gamma_f = N(0, \sigma_f)$ and $\gamma_y = N(0, \sigma_y)$; and ε is the error term, $\varepsilon = N(0, \sigma)$. Significance of the fixed effects was assessed and partial r^2 , i.e., r^2 estimated including only the fixed effects, was computed.

For fields where shoot density was measured, a second analysis was carried out using shoot density instead of crop age.

Step 2: nitrate leaching assessment with LIXIM

SMN and SWC measured at the beginning of winter were used as initial values to run LIXIM. Weather data (daily rainfall, potential evapotranspiration, and air temperature) were collected from two Meteo France weather stations (Ouges, 5°04'41'E – 47°15'38"N, and Chamblanc, 5°04'41'E–47°15'38"N) and two stations set up for the study in Lucenay-le-Duc (4°29'53'E–47°36'22"N) and Chissey sur Loue (5°44'12'E–47°01'37"N). The data used for each field were taken from the closest weather station, which was always within 20 km. Input data for soil characteristics were estimated for each soil layer through direct measurements (soil moisture at field capacity θ_{fc}) or using pedotransfer functions (soil moisture at the wilting point θ_{wp} and bulk density D_b). θ_{fc} was estimated as the mean value of SWC measured at the end of winter, assuming that soils were at field

capacity at that period of time. We considered this assumption as acceptable as the mean SWC value was closed to the maximum SWC value observed during the study. θ_{wp} was calculated using information about soil texture and a pedotransfer function defined by Bruand *et al.* (2004). D_a was estimated similarly and was then corrected by the proportion of pebbles. The depth below which drainage and leaching occurred was set as a function of crop age: 90 cm for 1-year-old crops and 120 cm for 2-year-old crops. This depth can be compared with the maximum rooting depth, which was measured on soil trenches in nine fields covering the range of soil types and crop ages of the on-farm network: we found that rooting depth did not exceed 75 cm for 1-year-old crops. In fields where the sampling soil depth was less than 120 cm due to mechanical constraints, the rooting depth was set as the mean sampling depth estimated from all the measurements on that field. The other model parameters, i.e., the maximum soil depth contributing to water evaporation Z_e and N mineralization Z_m were determined using references from Mary *et al.* (1999).

We tested the ability of LIXIM to simulate the SWC and SN measured in late winter for all fields by comparing fitted values to observed values. We also computed the root mean square error and the mean relative error. A sensitivity analysis was made on the effect of varying the ratio of actual to potential evapotranspiration (k) on drained water, leached N, and nitrate concentration in drained water, by varying k by $\pm 20\%$ ($0.48 \leq k \leq 0.72$) and $\pm 50\%$ ($0.3 \leq k \leq 0.9$).

For each field and each winter, the amounts of drained water (DRAIN) and leached nitrate (QLN) and the mean concentration of nitrate in the drained water (CLN) were calculated using LIXIM. The influence of soil type and cropping system characteristics (crop age, preceding crop) was studied through analysis of variance using a mixed model with field and year as random effects (R Development Core Team, 2008, version 2.14.2, package lme4).

Results

SMN measured at the beginning of winter

For all fields and all soil samplings, SMN averaged 39 kg N ha⁻¹ (Fig. 1a). Between sampling dates, mean SMN varied little, from 31 kg N ha⁻¹ (beginning of winter 2011) to 43 kg N ha⁻¹ (end of winter 2011) on average. SMN varied greatly between fields, ranging from 9 to 100 kg N ha⁻¹ at the beginning of winter 2010 and from 12 to 83 kg N ha⁻¹ at the beginning of winter 2011. However, at each date more than two-thirds of the fields had SMN below 50 kg N ha⁻¹. On average, SMN consisted of half nitrate and half ammonium.

SMN measured at the beginning of winter (SMN_{BW}) is an indicator of nitrate leaching risk, and was strongly

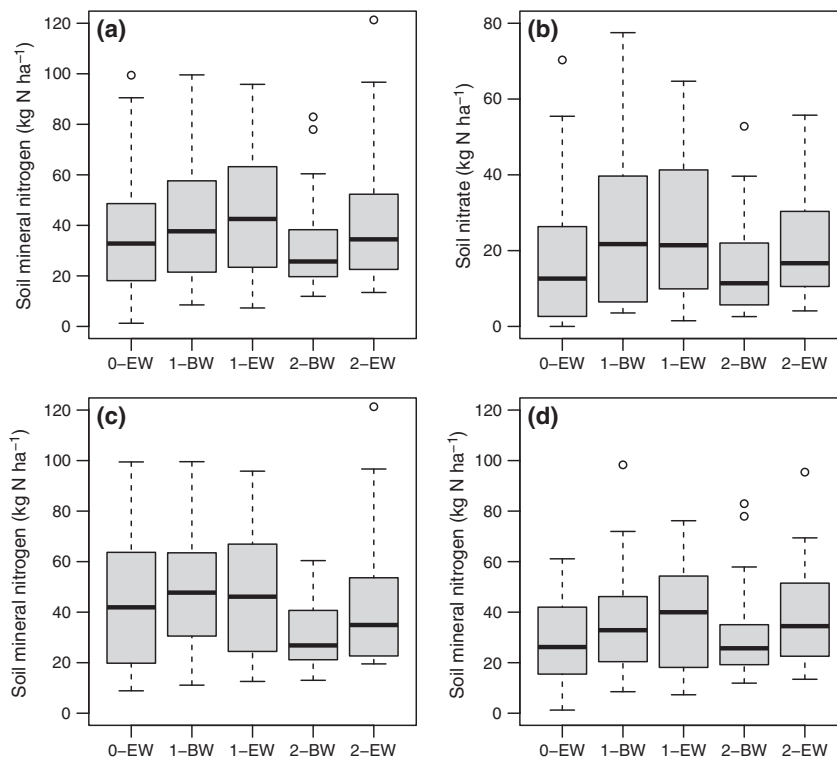


Fig. 1 (a) Total soil mineral nitrogen (kg N ha⁻¹), (b) soil nitrate (kg N ha⁻¹), (c) total soil mineral nitrogen for fields which were set-aside before the plantation of *M. giganteus* (kg N ha⁻¹), and (d) Total soil mineral nitrogen for fields where arable crops were grown before the plantation of *M. giganteus* (kg N ha⁻¹). 0_EW: end of winter 2009–2010, 1_BW: beginning of winter 2010–2011, 1_EW: end of winter 2010–2011, 2_BW: beginning of winter 2011–2012, and 2_EW: end of winter 2011–2012.

dependent on soil type (P -value = 1.2×10^{-4} , partial $r^2 = 0.45$). It was also significantly linked to crop age (P -value = 0.0043, partial $r^2 = 0.14$), and weakly to preceding crop (P -value = 0.071, partial $r^2 = 0.06$) and the interaction between soil type and preceding crop (P -value = 0.062, partial $r^2 = 0.16$). In both years, calcareous clayey soils (CC) and alluvial soils (A) had the highest SMN_{BW} (on average more than 40 kg N ha^{-1}), whereas clay soils (C), loamy clay soils (LC), and hydromorphic loamy soils (L) had SMN_{BW} values ranging from 10 to 60 kg N ha^{-1} (Fig. 2). L soils had the lowest soil mineral N contents at the beginning of winter. On the first two sampling dates (end of winter 2010 and beginning of winter 2011), fields that were set aside before the planting of *M. giganteus* tended to have higher SMN, but this effect disappeared at the following samplings (Fig. 1c and d). SMN decreased when crop age increased, particularly between the first and the second growth year. The decrease was more variable between the second and the third growth year (Fig. 3). In the 20 fields where shoot density was measured,

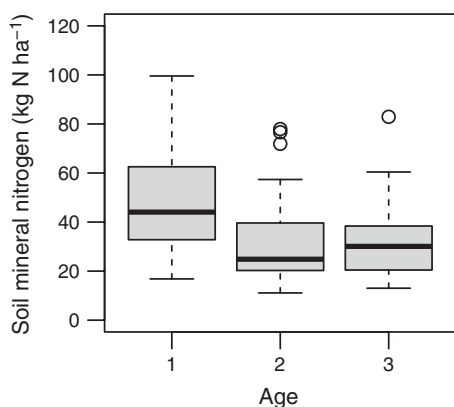


Fig. 2 Total soil mineral nitrogen (kg N ha^{-1}) measured at the beginning of winter (a) 2010 and (b) 2011 as a function of soil type. A, alluvial soil; C, clay soil; CC, calcareous clayey soil; L, loamy soil; LC, hydromorphic loamy soil.

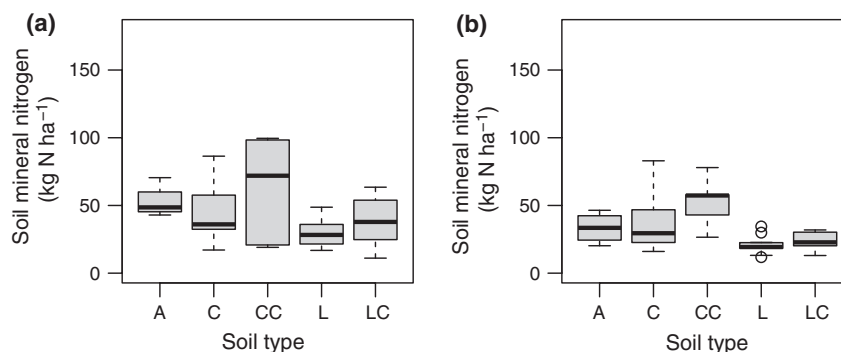


Fig. 3 Total soil mineral nitrogen (kg N ha^{-1}) measured at the beginning of winter as a function of crop age.

SMN_{BW} was negatively correlated with shoot density (P -value = 0.035, partial $r^2 = 0.06$).

Assessment of the ability of LIXIM to simulate soil water content (SWC) and soil nitrate (SN) in late winter

LIXIM was able to reproduce the soil water content (SWC) and the soil nitrate content (SN) measured in late winter (Fig. 4). The root mean square error was 35 mm for SWC and 8.3 kg N ha^{-1} for SN. The goodness of fit as a function of soil depth was good for SWC (data not shown). For SN, the goodness of fit was good for the 0–30 cm (mean observed SN = 9.9 kg N ha^{-1} , mean simulated SN = 9.3 kg N ha^{-1}) and the 30–60 cm layers (mean observed SN = 7.0 kg N ha^{-1} , mean simulated SN = 7.4 kg N ha^{-1}), and decreased for the deeper layers, 60–90 cm and 90–120 cm. This result was consistent with the modeling approach as simulated SN was highly dependent on the fitted potential rate of N mineralization, whereas the maximum soil depth contributing to N mineralization did not exceed 36 cm. SN tended to be overestimated in the 60–90 (mean observed SN = 3.0 kg N ha^{-1} , mean simulated SN = 5.5 kg N ha^{-1}) and 90–120 cm layers (mean observed SN = 1.1 kg N ha^{-1} , mean simulated SN = 1.9 kg N ha^{-1}). However, as the amounts of nitrate in those layers were small, the overestimation over the whole soil profile was also small (mean observed SN = $23.6 \text{ kg N ha}^{-1}$, mean simulated SN = $27.8 \text{ kg N ha}^{-1}$).

The analysis of sensitivity to k (the ratio of actual to potential evapotranspiration) showed that it had little influence (Table 2). The amounts of drained water and leached N and the mean nitrate concentration decreased when k increased and vice versa. When k decreased by 20%, drainage, leached N, and nitrate concentration increased by 8%, 7%, and 7%, respectively. The biggest variation in these three output variables was obtained for a 50% decrease in k : drained water and nitrate concentration increased by about 22% and leached N increased by 33%. However, this low value of k (0.30) is

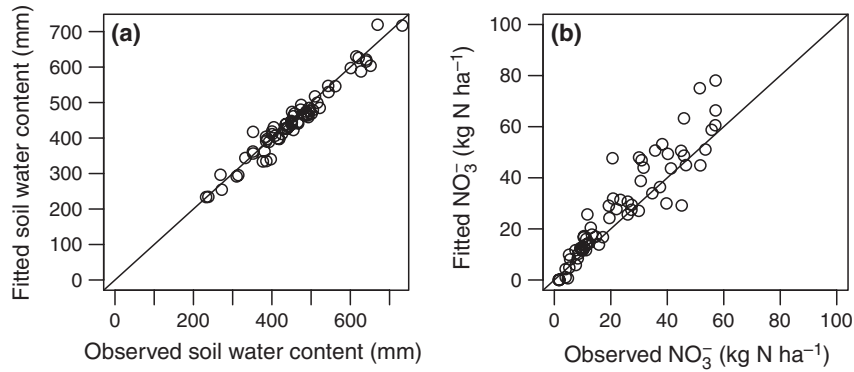


Fig. 4 Comparison between observed and simulated values of: (a) soil water content (mm) and (b) soil nitrate (kg N ha^{-1}) in late winter for both years. Simulated values were obtained with LIXIM. The continuous lines are the 1 : 1 lines.

Table 2 Sensitivity analysis of water drainage and N leaching to variation in k ($k = 0.60$). Variations in drained water, N leached, and $[\text{NO}_3^-]$ as a function of k are shown

k	Drained water (mm)	N Leached (kg N ha^{-1})	$[\text{NO}_3^-]$ (mg l^{-1})
0.30 (-50%)	21%	33%	23%
0.48 (-20%)	8%	7%	7%
0.72 (+20%)	-7%	-10%	-5%
0.90 (50%)	-13%	-20%	-9%

unlikely. We concluded that the nitrate concentrations simulated by the model were not very dependent on model parameterization and can be treated with confidence.

Water drainage

The amount of water drained below the maximum sampling depth was variable between years (Fig. 5). The

mean value was 216 mm during the first winter 2010–2011 and 115 mm during the second winter 2011–2012. This is mainly due to the weather, as the first winter was wetter than the second (for instance, 295 mm vs. 257 mm at the Ouges weather station, compared to 263 mm for the long-term average estimated over the past 20 years). Between-field variability in drainage was high during winter 2010–2011 (range: 145–355 mm, $\text{SD} = 48$). Only two fields had drainage above 300 mm. They had a soils and the high drainage was due to the high proportion of sand in these soils. Drainage was even more variable during the second winter 2011–2012 (range: 0–356 mm, $\text{SD} = 83$). Four fields stood out: the highest values, above 300 mm, were found in the two previously mentioned fields; two fields had no drainage. Drainage fluctuated around 50 mm in eight fields.

Analysis of variance, with year as a random effect, showed that drained water was influenced by soil type, crop age, and the interaction between soil type and preceding crop (Table 3). Drainage decreased with older

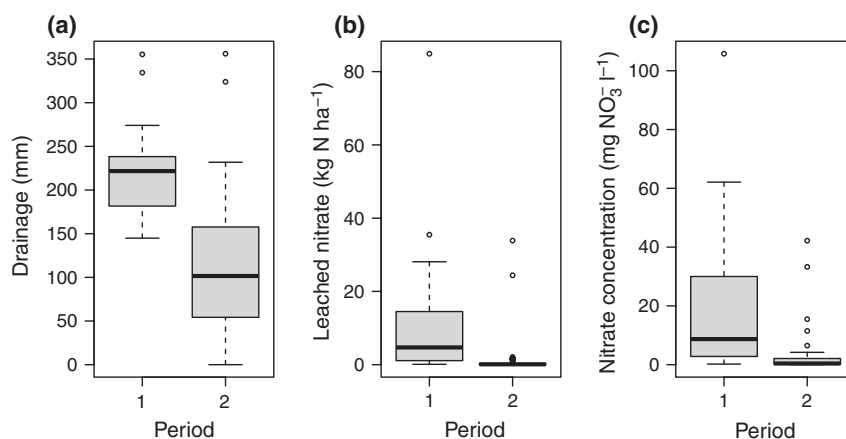


Fig. 5 LIXIM simulation results: (a) drained water (mm), (b) leached nitrate (kg N ha^{-1}), and (c) mean nitrate concentration in drained water ($\text{mg NO}_3^- \text{l}^{-1}$). Period 1: winter 2010–2011; Period 2: winter 2011–2012.

Table 3 Factors influencing leached nitrate, drainage, and nitrate concentration in drained water (***: significance with $P < 0.001$; **: significance with $P < 0.01$; *: significance with $P < 0.05$)

Fixed effects	Drained water (mm)		N Leached (kg N ha^{-1})		[NO ₃ ⁻] (mg l^{-1})	
	<i>P</i> -value	Partial <i>r</i> ²	<i>P</i> -value	Partial <i>r</i> ²	<i>P</i> -value	Partial <i>r</i> ²
Age	$7.7 \times 10^{-8***}$	0.36	$1.2 \times 10^{-6***}$	0.24	$1.3 \times 10^{-6***}$	0.37
Soil	$6.6 \times 10^{-5***}$	0.28	$6.6 \times 10^{-8***}$	0.45	$1.5 \times 10^{-4***}$	0.35
Preceding crop	0.11	–	0.023*	0.05	0.12	–
Preceding crop: Soil	0.0026**	0.20	$5.8 \times 10^{-5***}$	0.25	0.0050**	0.24

crops, which can be related to an increase in crop transpiration due to the higher biomass produced when crop age increased.

N leaching

N leaching varied greatly between sites and years (Fig. 5). During the first winter, the amount of leached N averaged 11 kg N ha^{-1} , but ranged between 0.1 and 85 kg N ha^{-1} ; 80% of the fields had N leaching of less than 20 kg N ha^{-1} . During the second winter, the average amount of leached N was much lower (average 2 kg N ha^{-1}); 95% of the fields lost less than 5 kg N ha^{-1} . The amount of leached N was found to be

influenced by weather, soil type (Table 3; Fig. 6), and crop age (Table 3; Fig. 7). Two soil types exhibited higher nitrate leaching: A soils and, to a lesser extent, CC soils. They represent the shallowest soils and, according to the model outputs, should have the greatest potential mineralization rate (Fig. 6). Leached N decreased when crop age increased (Fig. 7). The lower rainfall observed during the second winter 2011–2012 influenced the decrease in leached N. However, variance analysis with year as a random effect demonstrated the significant effect of age (Table 3). Unlike drainage, the preceding crop had a significant effect on N leaching, in addition to soil type and the interaction between soil type and preceding crop (Table 3).

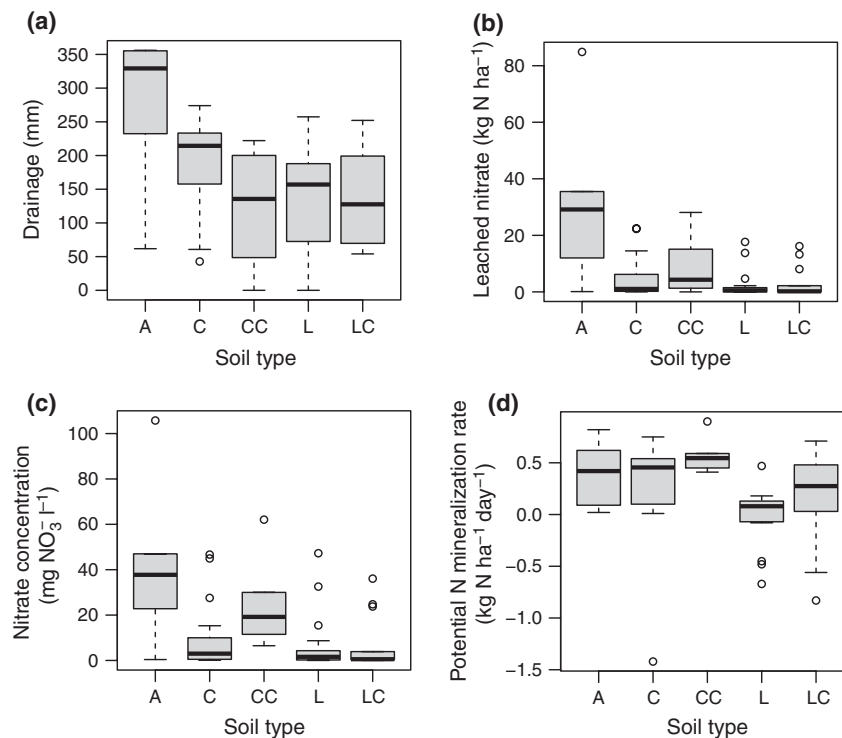


Fig. 6 LIXIM simulation results as a function of soil type: (a) drained water (mm), (b) leached nitrate (kg N ha^{-1}), and (c) mean nitrate concentration in drained water ($\text{mg NO}_3^- \text{ l}^{-1}$). A, alluvial soil; C, clay soil; CC, calcareous clayey soil; L, loamy soil; and LC, hydromorphic loamy soil.

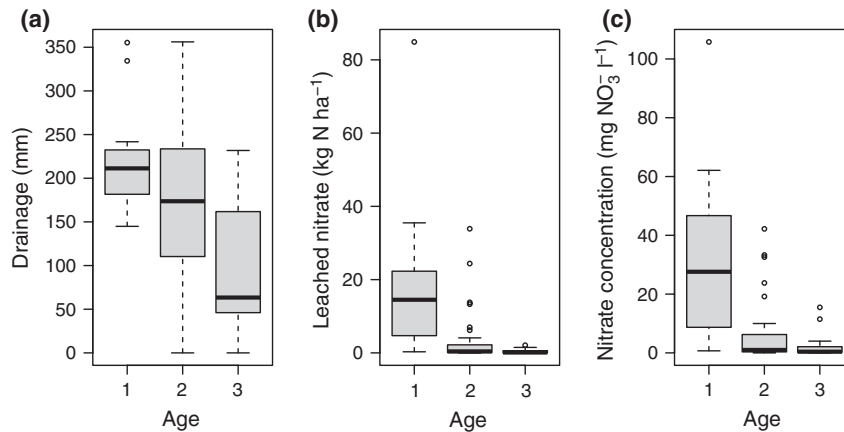


Fig. 7 LIXIM simulation results as a function of crop age: (a) drained water (mm), (b) leached nitrate (kg N ha⁻¹), and (c) mean nitrate concentration in drained water (mg NO₃⁻ l⁻¹).

Nitrate concentration of drained water

Nitrate concentration in drained water averaged 20 mg NO₃⁻ l⁻¹ during the first winter and ranged from 0.2 to 116 mg NO₃⁻ l⁻¹ (SD = 24 mg NO₃⁻ l⁻¹) (Fig. 5). The concentrations decreased during the second winter, ranging from 0 to 42 mg NO₃⁻ l⁻¹ with an average of 4 mg NO₃⁻ l⁻¹ (SD = 10 mg NO₃⁻ l⁻¹). They were all below the European threshold of 50 mg NO₃⁻ l⁻¹. 15% of the fields were close to or exceeded the threshold during the previous winter. Nitrate concentration was significantly influenced by soil type and an interaction between soil type and preceding crop (Table 3). It was also markedly dependent on crop age (partial $r^2 = 0.37$): the concentration was much lower in older crops (2 or 3 years old) than in first year crops (Fig. 7).

Discussion

The aim of this study was to assess nitrate leaching during the first establishment years of *M. giganteus*. Our study was based both on a farmers' field network to measure soil mineral nitrogen and soil water content and on a model to simulate nitrate leaching during winter from the measured data before and after winter. Our approach relied on a large number of actual farmed sites rather than on a limited number of experimental sites with numerous replicate measurements, our aim being to give an account of the between-field variability, which is scarce in the literature on *M. giganteus*. Direct measurements of water and nitrate fluxes through lysimeters or drained perimeters could obviously not be used in our situation. Ceramic cups would have provided information on nitrate concentration but not on water fluxes and could not have been installed on all the fields. Combining soil core samplings and the use of

a model to convert measurements into fluxes was therefore a suitable approach which allowed us to quantify not only the amount of N leached but also the nitrate concentration of the drained water. The amount of N leached is an important criterion because it is both a potential pollutant and a valuable resource. However, in the context of the EU limit for drinking water of 50 mg l⁻¹, it is also relevant to consider nitrate concentration. As mentioned by Goulding *et al.* (2000), although it is likely that nitrate in drained water leaving a field will be diluted or denitrified 'between drain and stream or soil and aquifer', it is appropriate to take the EU limit as a target.

Nitrate leaching assessed in our study was on average very low. It was much greater during the first winter 2010–2011 (average = 11; min = 0.1, max = 85 kg N ha⁻¹) than during the second 2011–2012 (average = 2; min = 0; max = 34 kg N ha⁻¹). The reduction between the 2 years could be mainly attributed to the age of the crop but also to the effect of weather. The nitrate concentration calculated in drained water averaged 12 mg l⁻¹ in our study, but varied from 0 to 106 mg l⁻¹ over sites and years. Christian *et al.* (2008) found that the 10-year mean N winter losses increased from 22 kg N ha⁻¹ in unfertilized crops to 63 kg N ha⁻¹ in crops receiving 120 kg fertilizer-N ha⁻¹. These average values are strongly influenced by the peak value of 154 kg N ha⁻¹ measured during the first winter after establishment, which was attributed to previous agricultural practices and heavy winter drainage (Christian & Riche, 1998). The results of McIsaac *et al.* (2010) are much more comparable with ours as they measured annual nitrate losses (from spring to spring) averaging 3 kg N ha⁻¹ y⁻¹. Christian & Riche (1998) observed nitrate concentrations averaging 12 mg l⁻¹ during the first three growth years, which is very close to our findings,

whereas Beale & Long (1997) measured a slightly higher concentration averaging 18 mg l^{-1} . On a watershed scale, Ng *et al.* (2010) used a *M. giganteus* crop model combined with a hydrological model to show that introducing the crop into a watershed can decrease the nitrate load. When 10, 25, or 50% of the farmed area was converted into *M. giganteus* (with 90 kg N ha^{-1} as fertilizer) instead of a corn/soybean rotation (with 190 kg N ha^{-1} applied on corn), nitrate load decreased by 6.5, 16.5, and 29.5%, respectively. In contrast, using the same model as Ng *et al.* (2010), Wu & Liu (2012) found that converting 10% of cornfields into *M. giganteus* in a watershed did not affect the nitrate load, whereas converting all the native grassland of the same watershed into *M. giganteus* increased the nitrate load by 5%.

All these results are clearly lower than the average losses usually measured for conventionally managed arable crops. For instance, Beaudoin *et al.* (2005) used LIXIM in northern France on cropping sequences based on winter rapeseed, winter cereals, spring pea, and sugar beet. They found that a mean amount of leached nitrate of 27 kg N ha^{-1} (with a range $16\text{--}50 \text{ kg N ha}^{-1}$ according to the soil, and $11\text{--}42 \text{ kg N ha}^{-1}$ according to the crop). Those losses lead to a mean nitrate concentration of 49 mg l^{-1} with a range $31\text{--}92 \text{ mg l}^{-1}$ according to the soil type and $32\text{--}80 \text{ mg l}^{-1}$ according to the crop. Constantin *et al.* (2010) found that conventional farming systems in northern France monitored over 13–17 years lead to a mean nitrate concentration varying from 53 to 109 mg l^{-1} . Likewise, on 256 fields located in Germany, Nieder *et al.* (1995) estimated with a model that leached N ranged from 16 kg N ha^{-1} for sugar beet to 88 kg N ha^{-1} for maize, and $20\text{--}40 \text{ kg N ha}^{-1}$ for cereals. Stopes *et al.* (2002), comparing organic and conventional farming, found leaching losses of 46 kg N ha^{-1} for an organic clover-based ley–arable system, 58 kg N ha^{-1} for a conventional long-term arable system, and 57 kg N ha^{-1} for conventional long-term grass. On the long-term Broadbalk experiment, Rothamsted, UK, mean amounts of N leached from continuous winter wheat fertilized with an optimum amount of $150\text{--}200 \text{ kg N ha}^{-1}$ were about 30 kg N ha^{-1} , but ranged from 10 to 60 kg N ha^{-1} under the influence of the weather (Goulding *et al.*, 2000).

Basically, low nitrate losses may be due to low soil nitrogen content at the beginning of winter and low drainage during winter. Nitrate concentrations are reduced by low soil nitrogen content, but are also reduced by high drainage due to a dilution effect. In our study, drainage averaged 166 mm, but differed severely between Period 1 where it was about 216 mm and Period 2 with an average of 115 mm. It was also much more variable during Period 2 (0–356 mm) than

during Period 1 (145–355 mm). Beaudoin *et al.* (2005) estimated with LIXIM higher drainage values that averaged 231 mm (219–263 mm according to soil type), whereas Constantin *et al.* (2010) measured with a lysimeter drainage of about 200 mm with catch crop (94–563 mm according to the site) and 215 mm without catch crop (120–593 mm). Goulding *et al.* (2000) observed on the Broadbalk long-term experiment a 10-year mean of 245 mm (range: 111–474 mm). Drainage as estimated in our study was thus rather low. Nevertheless, besides drainage, nitrate losses estimated in our study can also be related to the small amount of soil mineral nitrogen measured (SMN) at the beginning of winter. Total soil mineral nitrogen (SMN) that we measured in late autumn averaged 42 kg N ha^{-1} for the first period (winter 2010–2011) and 31 kg N ha^{-1} for the second one (winter 2011–2012). These average values were lower to those found by Beaudoin *et al.* (2005) in northern France where total SMN for different arable crops averaged 55 kg N ha^{-1} . Furthermore, SMN in our study consisted of half nitrates and half ammonium, whereas SMN contained only 17% of ammonium in theirs.

The low average values observed for SMN at the beginning of winter were associated with high variability: from 9 to 100 kg N ha^{-1} for the first period and 12 to 83 kg N ha^{-1} for the second. The same high variability was observed for leached nitrate (from 0 to 85 kg N ha^{-1}) and nitrate concentration (from 0 to 106 mg l^{-1}). Values reported in the literature are commonly variable but only to a smaller extent. For instance, Beaudoin *et al.* (2005) observed SMN ranging from 40 to 64 kg N ha^{-1} according to the soil type and from 40 to 95 kg N ha^{-1} according to the crop, 95 kg N ha^{-1} being observed after a pea crop. Likewise, Nieder *et al.* (1995) simulated N leaching ranging from 16 to 88 kg N ha^{-1} according to the crop. The presence of very low SMN in our study could be due to the studied crop and the soils included in the on-farm design. *M. giganteus* was mostly unfertilized, whereas it might still absorb nitrogen until late in autumn through an extensive rooting system (Neukirchen *et al.*, 1999; Monti & Zatta, 2009). Moreover, as it is a perennial crop, it is likely that the lack of cultivation in the second and third years reduced mineralization of soil organic matter, as suggested by Christian & Riche (1998). Besides, our on-farm design included some with very loamy hydromorphic soils where N mineralization is presumably low. On the other hand, high SMN was found on deep clay soils and calcareous clayey soils deeper than 80 cm, where mineralization rates were higher. The on-farm design also included fields where the crop failed to establish, resulting in very low shoot densities of less than 15 shoots per m^2 , compared to an average of 35

shoots per m² and a maximum of 60 shoots per m² on the densest crops. Those fields with low shoot densities also had higher SMN, which may have been due to low nutrient uptake due to the poor growth of the crop. Smith *et al.* (2013) as well highlighted that establishment problems in *M. giganteus* caused a lag in nitrate leaching decrease.

Our results show a strong influence of soil type. Besides the effect of the mineralization rate on SMN mentioned above, soil type also influenced the amount of drained water: the soil with the highest sand contents had the highest amount of drained water. As these soils also exhibited the highest mean amount of leached N, they had the highest mean nitrate concentration. The influence of soil type was similar to the observations made by Nieder *et al.* (1995), Boniface (1996), Simmelsgaard (1998), and Beaudoin *et al.* (2005) on annual crops: deep clayey or loamy soils experience lower nitrate leaching than shallow soils, in particular, when the latter have a high sand content. Nieder *et al.* (1995) estimated in Germany, N leaching averaging 16 kg N ha⁻¹ for coarse soils (i.e., sandy soils) and 63 kg N ha⁻¹ for fine-textured soils (i.e., silty, loamy, and clay soils). The relative differences between soils were said to be smaller for concentration than for leaching, which may result from dilution by water in shallow and/or sandy soils (Simmelsgaard, 1998; Beaudoin *et al.*, 2005). We observed that phenomenon, but the difference in nitrate concentrations between the shallowest soils of our study (CC and A) and the others was still significant. It is interesting to note that the soils with the lowest N leaching, i.e., deep clayey or loamy soils, were also the ones with the highest shoot densities, i.e., the ones where we can expect the highest yields.

Besides soil type, we also noticed an effect of crop age, which can be related to the crop development: during the first year of growth, shoot density and plant height remain low, leading to a smaller nutrient and water absorption. Christian & Riche (1998) also found that N losses decreased when crop age increased. During the first winter following planting of *M. giganteus* without fertilization, they observed losses of 154 kg N ha⁻¹. That amount decreased to 8 and 3 kg N ha⁻¹ during the two following winters, which is similar to the dynamic observed in our study. However, in our study, the amount of nitrate leached during the winter after establishment averaged only 17 kg N ha⁻¹ (min = 0.1 kg N ha⁻¹, max = 85 kg N ha⁻¹). However, Christian & Riche (1998) mentioned that previous agricultural practices (i.e., long-term grass 4 years earlier and incorporation of bean residues) may have induced a high rate of N mineralization. Besides, drainage amounted to 478 mm during the first winter of their experiment, whereas it averaged 200 mm in our study.

In the following winters drainage was closer to what we estimated: 262 and 150 mm against an average of 166 mm in our study. Christian & Riche (1998) observed a mean nitrate concentration of 32 mg l⁻¹ for the first winter and 3 and 2 mg l⁻¹ for the following two winters. Due to the difference in drainage, the mean nitrate concentration we estimated for the first winter was very similar to theirs (31.3 mg l⁻¹). In the winters after the second and third years of growth, these concentrations were about 7 and 3 mg l⁻¹, respectively, which is also similar to their values.

Conclusion

Combining nitrate and water content measurements on 36 fields with the use of the LIXIM model to estimate nitrate and water fluxes, our study confirmed that growing *M. giganteus* results in low N leaching during the first years of growth, strengthening the idea that the crop is associated with a low risk of groundwater pollution by nitrates. Estimated mean nitrate concentration averaged 12 mg l⁻¹, far below the European limit of 50 mg l⁻¹. We also confirmed that the highest risk is in the first year of growth, when crop development is at its lowest.

Although we showed that nitrate leaching was low on average, we also found that it was very variable. The variability was not only associated with crop age but also with soil type and crop development. Unlike earlier experimental studies on *M. giganteus*, our on-farm design included a wide range of soil types and growing conditions. Deep loamy or clayey soils exhibited the least N leaching. Conversely, we found that the risk of N leaching was the highest in two situations: (i) in fields with shallow and/or sandy soils and (ii) when crop establishment fails. As fields with shallow and/or sandy soils have low available soil water, they have also a higher probability of establishment failure.

Except for those risky situations, our study confirmed that regarding nitrate losses, *M. giganteus* has a better environmental profile than annual crops. Available comparisons with other perennial candidate bioenergy crops such as switchgrass or short rotation coppice suggest that those crops present the same advantage in terms of N leaching as *M. giganteus* when they are unfertilized (Makeschin, 1994; Aronsson *et al.*, 2000; Aronsson & Bergstrom, 2001; McIsaac *et al.*, 2010). However the nutrient requirements of those crops are commonly said to be higher than those of *M. giganteus* (Lewandowski *et al.*, 2003; Powlson *et al.*, 2005; Heaton *et al.*, 2010). Furthermore, recent findings suggest that N₂O emissions from *M. giganteus* are also low (Drewer *et al.* 2012; Gauder *et al.* 2012), confirming that the crop is associated with low N losses into the environment,

which strengthens its potential as a bioenergy crop. Besides, Gopalakrishnan *et al.* (2012) suggested that bioenergy crops such as *M. giganteus* could also be grown in buffer strips adjacent to current agricultural crops. With such a spatial configuration, the bioenergy crops could reuse nutrients present in runoff and leachate from the conventional row-crops, thus allowing energy production while providing environmental services.

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