Soil nitrogen dynamics in switchgrass seeded to a marginal cropland in South Dakota

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Abstract

The potential ecological impacts of switchgrass (Panicum virgatum L.), as a biofuel feedstock, have been assessed under different environmental conditions. However, limited information is available in understanding the integrated analysis of nitrogen (N) dynamics including soil nitrate (NO3\(^-\)) , nitrous oxide (N2O) emissions, and NO3\(^-\) leaching under switchgrass land management. The specific objective was to explore N dynamics for 2009 through 2015 in switchgrass seeded to a marginally yielding cropland based on treatments of N fertilization rate (N rate; low, 0; medium, 56; high, 112 kg N ha\(^{-1}\)) and landscape position (shoulder, backslope, and footslope). Our findings indicated that N rate impacted soil NO3\(^-\) (0−5 cm depth) and surface N2O fluxes but did not impact NO3\(^-\) leaching during the observed years. Medium N (56 kg N ha\(^{-1}\)) was the optimal rate for increasing biomass yield with reduced environmental problems. Landscape position impacted the N dynamics. At the footslope position, soil NO3\(^-\), soil NO3\(^-\) leaching, and N2O fluxes were higher than the other landscape positions. Soil N2O fluxes and NO3\(^-\) leaching had downward trends over the observed years. Growing switchgrass on marginally yielding croplands can store soil N, reduce N losses via leaching, and mitigate N2O emissions from soils to the atmosphere over the years. Switchgrass seeded on marginally yielding croplands can be beneficial in reducing N losses and can be grown as a sustainable bioenergy crop on these marginal lands.

Keywords: landscape position, nitrate (NO3\(^-\)) leaching, nitrogen (N) dynamics, nitrogen (N) fertilization rate, nitrous oxide (N2O), soil nitrate (NO3\(^-\)), switchgrass (Panicum virgatum L.)

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Introduction

Switchgrass (Panicum virgatum L.) is a native perennial warm-season (C\(_4\)) grass in North America and a widely adapted endemic species (Lewandowski et al., 2003). It was identified as a renewable energy source by the US Department of Energy in 1985 (Lee et al., 2012) and selected as a model potential bioenergy crop in 1991 (Wright, 2007). In addition to its economic benefits as a biofuel feedstock, the potential ecological impacts of switchgrass need to be assessed under the local environmental conditions (Hartman et al., 2011). Nitrogen (N) fertilization is a key factor in switchgrass management because N is a limiting nutrient for switchgrass (Owens et al., 2013; Hong et al., 2014). However, the amount of N inputs depends on various soil conditions in switchgrass fields (Kering et al., 2012). The N dynamics in soil-plant system involve many complex and interacting processes that transform and transport N in and out of the system, and the soil N supply, crop N uptake, and N losses are the main processes to understand the N dynamics (Stockdale et al., 1997). The sources of soil N include organic [e.g., soil organic matter (SOM), crop residues, and manure] and inorganic [e.g., ammonium (NH\(_4\)) fertilizer] forms of N. However, the organic forms are hardly available for crop use (Schimel & Bennett, 2004). They are first converted to inorganic forms through mineralization in which bacteria digest organic materials and release NH\(_4\)\(^+\) (Schimel & Bennett, 2004). Then the NH\(_4\)\(^+\) is rapidly converted to nitrate (NO3\(^-\)) under optimum climate conditions through nitrification in which an aerobic process is performed by autotrophic bacteria and archaea (Whalen & Sampedro, 2010). The NO3\(^-\) and NH\(_4\)\(^+\) are the major inorganic forms of N taken up by plants. However, if not managed properly, excessive N losses occur. Most N losses occur in two ways including NO3\(^-\) leaching and N gas
losses. As a negatively charged ion, NO$_3^-$ in soils is repelled by the negatively charged clay mineral surfaces in soil rather than attracted to them. It is totally soluble, and moves freely through the most soils (Follett, 1995). Nitrogen gas losses occur as bacteria convert NO$_3^-$ to N gases such as nitrous dioxide (N$_2$O) which is then lost to the atmosphere through denitrification in which denitrifying bacteria use NO$_3^-$ instead of oxygen in the metabolic processes (Luce et al., 2011). Therefore, management practices are needed to reduce the N losses.

The N dynamics in a switchgrass ecosystem are crucial in the assessment because increased N inputs are likely to result in significant N losses via surface runoff of ammonium (NH$_4^+$), nitrate (NO$_3^-$) leaching into surface and groundwater, and gaseous emissions of ammonia (NH$_3$) and nitrous oxide (N$_2$O) into the atmosphere. Among these N losses, N$_2$O fluxes, and NO$_3^-$ leaching contribute major concern because of their severe impacts on the environment. The excessive N$_2$O in the atmosphere not only contributes to global warming but currently is the single most important ozone-depleting gas (Dai et al., 2014). Nitrate leaching can lead to excess N in rivers, lakes, and groundwater that can cause water quality problems in natural water systems (Ma et al., 2001; Madakadze et al., 2003). Therefore, in this study, N fertilization, soil NO$_3^-$, N$_2$O leaching, and soil N$_2$O fluxes were selected to explore N dynamics in switchgrass fields.

Many studies reported the N dynamics in crop fields based on different treatments (e.g., Luce et al., 2011; Clough et al., 2013). However, the main factors in N dynamics in most of these studies were investigated separately, and very few studies included N fertilizer, soil N$_2$O fluxes, NO$_3^-$ leaching, soil properties, and yield for some crops (e.g., Maharjan et al., 2014; Zhou et al., 2014). However, the studies used the short-term data and did not evaluate the impacts over time. A few studies have reported some results of N dynamics in switchgrass fields such as N use efficiency across the United States (e.g., Owens et al., 2013), gross N mineralization and nitrification rates in surface soils after N fertilizer application (Garten et al., 2010), N translocation from aboveground to belowground biomass during senescence (Pedroso et al., 2014), N$_2$O emissions (Nikiema et al., 2011; Schmer et al., 2012), and inorganic N leaching (McIsaac et al., 2010). However, these studies investigated various parameters separately to study the N dynamics. Little is known about the information of integrated analysis of N fertilizer, soil NO$_3^-$, soil N$_2$O emissions, and NO$_3^-$ leaching, particularly over time frames exceeding 2 years.

Landscape position is another key factor influencing soils (Wang et al., 2001; Majaliwa et al., 2015). Landscape position or slope position reflects spatial variabilities in N dynamics. A few studies have been related to landscape spatial effects on soil N dynamics in switchgrass fields (Schmer et al., 2011; Ontl et al., 2015). However, these studies focused primarily on the differences of the selected parameters among different sites. Little is known about the impacts of landscape position on N dynamics in a switchgrass field. Therefore, the objectives of this study were to: (i) evaluate the effects of N rate, landscape position, and time on soil NO$_3^-$, N$_2$O leaching, and N$_2$O fluxes, and (ii) investigate the optimum N rate for increasing the switchgrass biomass yield while reducing environmental problems.

**Materials and methods**

**Description of study site and treatments**

The study site is located at 45°16′24.55″ N, 97°50′13.34″ W, near Bristol, South Dakota, United States. It was arranged into 12 plots (each plot is approximately 21.3 m wide and 365.8 m long) with 2–20% slope. The plots were laid out in a split-plot design comprised of three N rate treatments (low, 0 kg N ha$^{-1}$; medium, 56 kg N ha$^{-1}$; high, 112 kg N ha$^{-1}$) as whole plots and three landscape positions (shoulder, backslope, and footslope) as subplots with four replications. The N fertilizer was applied in late May or early June for each year. Switchgrass [cultivar: Sunburst; planting rate: 10 kg pure live seed (PLS) ha$^{-1}$] was planted on May 17, 2008. The previous crop grown on these plots was soybean (*Glycine max.* L.). Switchgrass was harvested once annually around a killing frost. The soils at the site are dominated by loamy soils (Mbonimpa et al., 2015). The average daily temperature and the average annual precipitation for 30 years from 1986 to 2015 were 6.42 °C and 619 mm, respectively.

**Sampling and analysis**

Soil samples were collected from each plot during June of 2009, 2010, 2011, 2012, and 2013 before yearly N fertilizer application for measuring the NO$_3^-$ contents at the 0- to 5-, 5- to 15-, 15- to 30-, 30- to 60-, and 60- to 100-cm depths. Soil NO$_3^-$ was determined using the KCl Extraction/d-Reduction Method (Bremner & Edwards, 1965; Keeney & Nelson, 1982). Water samples from switchgrass fields were collected using suction lysimeters installed and fixed at 1.0 m depth and 36 positions at this study site from 2009 to 2015 (Lai et al., 2016). The samples were collected 3–6 times per year from April to November. The water samples were analyzed for NO$_3^-$ leaching concentrations by a NO$_3^-$ analyzer Dionex DX500 using the US Environmental Protection Agency (EPA) 300.1 method (Hautman & Munch, 1997). There were some missing samples because of no leachate at some positions resulting from dry weather.

Soil N$_2$O emissions were monitored using the vented polyvinyl chloride static flux chambers (25 cm diameter × 15 cm height; Hutchinson & Mosier, 1981), which were installed and
fixed in the fields from 2010 to 2015, according to the guidance of Parkin & Venterea (2010). Soil N\textsubscript{2}O fluxes were calculated as the change in headspace gas concentration over time within the enclosed chamber volume (Parkin & Venterea, 2010), and the average of two chambers was used to represent each plot for further analysis (Mbonimpa et al., 2015). To further analyze the variabilities of N\textsubscript{2}O fluxes during the growing season, each growing season was divided into the three periods: April–June, July–August, and September–October, which are approximately in response to the germination or green-up and vegetative stages, elongation stage, and reproductive stage of switchgrass growth, respectively (Moore et al., 1991).

Daily minimum and maximum temperature and precipitation measurements and calculation were described in our previous publication (Lai et al., 2016). The method of water-filled pore space (WFPS) calculation was described in a previous study (Mbonimpa et al., 2015). Nitrogen use efficiency (NUE; mass of biomass per mass of N) for each harvest was calculated as (Novoa & Loomis, 1981; Zemenchik & Albrecht, 2002): NUE = (yield at N\textsubscript{9} – yield at N\textsubscript{0})/mass of N applied where N\textsubscript{9} = N level > 0, and N\textsubscript{0} = no N applied (Owens et al., 2013).

**Data analysis**

Statistical comparisons of soil NO\textsubscript{3} concentrations among the three N rates and three positions were obtained using all pairwise differences method (adjusted by Tukey) by a mixed model, where the N rate, position, and position \times N rate were considered as fixed effects and replication and replication \times N rate as random effects using the GLIMMIX procedure in SAS 9.4 (SAS, 2013). The repeated measures analysis for comparing the soil N\textsubscript{2}O fluxes and NO\textsubscript{3} leaching contents under different N rates and positions was conducted using PROC MIXED in SAS 9.4 (SAS, 2013). Data were transformed when necessary using SAS 9.4 (SAS, 2013). The transformation was determined using the Box-Cox method (Box & Cox, 1964, 1981) using SAS 9.4 (SAS, 2013). The data trend test was conducted using the Mann–Kendall method (Mann, 1945; Kendall, 1975) with slopes estimated by the Sen Estimator (Sen, 1968) using the package “mblm” in \texttt{R} (Komsta, 2013; R Core Team, 2016). Significance was determined at \(\alpha = 0.05\) level for all statistical analysis except for the trend test where \(\alpha = 0.10\) due to limited degrees of freedom (Robert et al., 1997; Royer et al., 2007).

**Results**

**Soil NO\textsubscript{3}**

Nitrogen fertilization rate, position, and depth significantly impacted soil NO\textsubscript{3} concentrations in different years (Table 1). Interaction effects of N rate \times position, N rate \times depth, and position \times depth on soil NO\textsubscript{3} were significant for some years (Table 1); therefore, the data were analyzed separately for each depth. At 0- to 5-cm depth, annual mean soil NO\textsubscript{3} concentrations under high N rate (0.72 mg kg\textsuperscript{-1}) were significantly lower than those of medium (2.28 mg kg\textsuperscript{-1}) and low N (3.16 mg kg\textsuperscript{-1}) rates in 2010. Soil NO\textsubscript{3} concentrations under the low N rate were significantly higher than under the medium (80% higher) and high (104% higher) N rates in 2012. Soil NO\textsubscript{3} concentrations under low N rate were significantly 68% higher than that of high N rate in 2013. At other depths, N rate did not impact soil NO\textsubscript{3} concentrations (Table S1).

Position significantly impacted soil NO\textsubscript{3} concentration at 0- to 5-cm depth in 2011 and 2012, at 5- to 15-cm depth in 2011 and 2012, at 15- to 30-cm depth from 2011 to 2013, and at 60- to 100-cm depth in 2009 and 2012. The annual mean soil NO\textsubscript{3} concentration at the footslope position was significantly higher than the backslope and shoulder positions up to 30-cm depth in these years. For example, at the 0- to 5-cm depth, the soil NO\textsubscript{3} concentration at the footslope was 90% and 111% higher in 2011 and 2012, respectively, compared with the shoulder position, and 94 and 99% higher than the backslope position. For the 30- to 60- and 60- to 100-cm depth, the soil NO\textsubscript{3} concentrations were variable across all the years (Table S1).

The interactions of N rate \times position on soil NO\textsubscript{3} concentrations were significant at 5- to 15-cm depth in 2012 and 30- to 60-cm depth in 2010 (Table S1a,b). At 5- to 15-cm depth in 2012, under high N rate, the NO\textsubscript{3} concentration at the footslope (2.88 mg kg\textsuperscript{-1}) was significantly higher than backslope (1.28 mg kg\textsuperscript{-1}) and shoulder (0.96 mg kg\textsuperscript{-1}) positions. At footslope position, soil NO\textsubscript{3} concentration under high N rate (2.88 mg L\textsuperscript{-1}) was significantly higher than the medium (1.33 mg kg\textsuperscript{-1}) N rate (Table S2). Soil NO\textsubscript{3} concentrations generally decreased with the increase in soil depth. For each depth, the annual mean soil NO\textsubscript{3} concentrations followed an upward parabola curve over the five observed years for each N rate and position (Fig. S1).

**Soil NO\textsubscript{3} leaching**

The N rate did not impact soil NO\textsubscript{3} leaching for each year (Table 2). However, landscape position significantly influenced the NO\textsubscript{3} leaching. Annual leaching at the footslope position (5.55 mg L\textsuperscript{-1}) was significantly higher than shoulder position (2.08 mg L\textsuperscript{-1}) in 2009. Nitrate leaching at the footslope position (1.17 mg L\textsuperscript{-1}) was significantly higher than the backslope (0.52 mg L\textsuperscript{-1}) and shoulder (0.48 mg L\textsuperscript{-1}) positions in 2011 (Table 2). The interactions of position \times time on NO\textsubscript{3} leaching were significant in 2010 and 2011 (Table 2). Nitrogen application rate and landscape position did not impact the leaching in 2010 and 2011, but there was a significant time effect on the leaching for each N rate and position. However, the effects of all interactions (position \times time...
and N rate \times \text{time}) on the contents were not significant (Table S3).

Trend tests showed that the annual mean soil NO$_3^-$ leaching for each rate and position was in a downward trend over the observed years (slopes were negative), and the downward trends under high and low N rates and at backslope and footslope positions were significant ($P < 0.1$; Table S4). These downward trends and their functions (quadratic functions) are presented in Fig. 1. Furthermore, daily mean NO$_3^-$ leaching presented a downward trend curve with a quadratic function: $y = 0.013x^2 - 0.48x + 4.55$ ($R^2 = 0.88$) over the observed days in 2009–2015. The peak observations were 5.05, 2.36, 1.58, 0.89, 0.41, and 1.06 mg L$^{-1}$ in 2009, 2010, 2011, 2013, 2014, and 2015, respectively (Fig. S2a).

Soil N$_2$O fluxes

Annual mean soil N$_2$O fluxes under the high N rate were significantly higher than those under low N rate for 2014 (30% higher). In 2015, soil N$_2$O fluxes under the high N rate were significantly higher than fluxes under the medium N rate (24% higher), which were significantly higher than fluxes under the low N rate (55% higher). Soil N$_2$O fluxes at the footslope were significantly higher than the shoulder in 2014 (61% higher) and 2015 (38% higher). Among N rates, the highest and the lowest fluxes were observed with the high N rate in 2009 (4.17 g ha$^{-1}$ day$^{-1}$) and with the low N rate in 2014 (1.09 g ha$^{-1}$ day$^{-1}$), respectively. Among positions, the highest and the lowest fluxes were observed at the footslope in 2009 (3.86 g ha$^{-1}$ day$^{-1}$) and shoulder in 2014 (0.94 g ha$^{-1}$ day$^{-1}$). Time effects on the fluxes were significant in all the five observed years (Table 3). Moreover, the trend test showed that the fluxes for each N rate and position followed a downward trend over the observed years (slopes were negative), and these downward trends were significant ($P < 0.1$; Table S4). The downward trends and their functions (quadratic functions) are presented in Fig. S3.

The daily mean soil N$_2$O fluxes under different N rates and positions followed a downward trend quadratic curve over the observed days from 2009 through 2015, and the flux peaks generally coincided with that of temperature and precipitation in each growing season (Fig. 2). The fluxes under the high N rate were generally higher than medium and low N rates. The footslope generally had higher N$_2$O fluxes compared with the shoulder position. Under the high N rate, the observed peak fluxes were 9.70, 4.46, 4.54, 2.37, and 3.35 g ha$^{-1}$ day$^{-1}$ in 2010, 2011, 2012, 2014, and 2015, respectively. At the footslope, the peaks were 5.94, 5.22, 4.73, 2.59, and 3.09 g ha$^{-1}$ day$^{-1}$ in 2010, 2011, 2012, 2014, and 2015, respectively (Fig. 2). Furthermore, the seasonal analysis showed that the N rate had a significant influence on the N$_2$O flux in September–October in 2014 and May–June, July–August, and September–October in 2015 (fluxes under high > medium > low N rate).

Under the high N rate, the highest average N$_2$O flux

| Table 1 | Type III tests of fixed effects results ($P > F$) from the mixed model analysis of variance (ANOVA) for soil NO$_3^-$ content under different N rates, positions, and depths for each year |
|--------|-------------------|-----------------|-----------------|-----------------|-----------------|
|        | Soil NO$_3^-$ Leaching |
|        | 2009 | 2010 | 2011 | 2012 | 2013 |
| $N$    | 0.40 | 0.02 | 0.29 | 0.03 | 0.47 |
| $P$    | 0.37 | 0.06 | <0.001 | <0.001 | 0.086 |
| $N \times P$ | 0.09 | 0.65 | 0.27 | <0.001 | 0.28 |
| $D$    | 0.02 | <0.001 | <0.001 | <0.001 | <0.001 |
| $N \times D$ | 0.72 | <0.001 | 0.31 | 0.57 | 0.77 |
| $P \times D$ | 0.16 | 0.17 | 0.03 | 0.60 | 0.003 |
| $N \times P \times D$ | 0.80 | 0.88 | 0.91 | 0.99 | 0.17 |

*Significance level $z = 0.05$. 

*Means within the same column followed by different small letters are significantly different at $P < 0.05$ for landscape position and N rate.

Table 2 | Annual mean soil nitrate (NO$_3^-$) leaching content for the 100 cm depth from 2009 to 2015 under high, medium, and low N rates applied to switchgrass seeded at the shoulder, backslope, and footslope positions |
|--------|-------------------|-----------------|-----------------|-----------------|-----------------|
|        | Soil NO$_3^-$ Leaching |
|        | 2009 | 2010 | 2011 | 2014 | 2015 |
| $N$ rate | 3.49* | 1.96* | 0.80* | 0.29* | 0.47a |
| Medium   | 0.78* | 0.51* | 0.17* | 0.36* |
| Low      | 4.12* | 2.85* | 0.79* | 0.22* | 0.35a |
| Position ($P$) | 2.08b | 2.22a | 0.48b | 0.23a | 0.53a |
| Shoulder | 3.43ab | 1.48a | 0.20a | 0.30a |
| Backslope | 5.55* | 2.83a | 1.17a | 0.23* | 0.26a |

*Means within the same column followed by different small letters are significantly different at $P < 0.05$ for landscape position and N rate.

†The effects of $P \times$ Time were significant, thereby, the data were analyzed separately for each N rate and position (Table S3).
was observed in May–June of 2010 (5.33 g ha⁻¹ day⁻¹),
which was 1.6, 2.8, 3.0, and 3.1 times higher compared
to those in May–June of 2011, 2012, 2014, and 2015,
respectively. Position significantly impacted N₂O fluxes
in July–August and September–October in 2014, and
May–June, July–August, and September–October in
2015 (fluxes at footslope > shoulder position). At the
footslope, the highest flux was observed in May–June
of 2010 (4.51 g ha⁻¹ day⁻¹), which was 1.3, 2.5, 2.2, and
2.9 times higher compared to those in May–June of
2011, 2012, 2014, and 2015, respectively (Table S5).

### Discussion

**Soil NO₃⁻, NO₃⁻ leaching, and N₂O fluxes**

This study showed that N rate under switchgrass field
significantly impacted the soil NO₃⁻ in the 0- to 5- and
5- to 15-cm depths (Table 1, S1, and S2) and N₂O
fluxes in 2014 and 2015 (Table 3), but did not impact
soil NO₃⁻ leaching for all the observed years (Table 2).
These findings provide a unique insight into the
mechanism of N dynamics in this switchgrass field. At
this study site, the major N input was N fertilizer and
output mainly included switchgrass yield, NO₃⁻ leach-
ing, and N₂O fluxes. At the 0- to 5- and 5- to 15-cm
depths, soil NO₃⁻ concentrations under the low N rate
were generally higher than the medium rate, which
was greater than those under high N rate (Table S1a).

### Table 3 Annual mean soil N₂O fluxes from 2010 to 2015
under high, medium, and low N rates applied to switchgrass
seeded at the shoulder, backslope, and footslope positions

| Treatment | Soil N₂O Fluxes | 2010  | 2011  | 2012  | 2014  | 2015  |
|-----------|-----------------|-------|-------|-------|-------|-------|
| N rate    |                 |       |       |       |       |       |
| High      |                 | 4.17a | 2.65a | 2.18a | 1.42a | 2.15a |
| Medium    |                 | 3.21a | 2.53a | 2.04a | 1.16b | 1.74b |
| Low       |                 | 2.89a | 2.16a | 1.74a | 1.09b | 1.12c |
| Position (P) |              |       |       |       |       |       |
| Shoulder  |                 | 3.09a | 2.27a | 1.90a | 0.94b | 1.40b |
| Footslope |                 | 3.86a | 2.58a | 2.04a | 1.51a | 1.93a |

*Means within the same column followed by different small letters are significantly different at *P* < 0.05 for landscape position and N rate.

This was mainly related to two processes. The first
process was switchgrass uptake of soil NO₃⁻ at the 0-
to 15-cm depth. The most soil NO₃⁻ at this depth was

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Fig. 1 Trends of annual mean soil NO₃⁻ leaching contents (mg L⁻¹) under (a) the three N rates applied to switchgrass seeded at (b) the three positions over the observed years.
Switchgrass yield under high and medium N rates (high N rate > medium N rate) was significantly higher than the low N rate (Fike et al., 2017). Therefore, more soil NO$_3^-$ under high and medium N rates can be taken up by switchgrass compared with low N rate. In this process, the mean nitrogen use efficiency (NUE) under the medium N rate significantly higher than that of the high rate (Table S6), indicating the field under the high N rate had more N losses. The second process was the conversion of soil NO$_3^-$ into gaseous emissions in the 0- to 15-cm depth [the emissions through denitrification process is generally limited to topsoil (Lamb et al., 2014)]. More soil NO$_3^-$ can be consumed through denitrification at the higher N rates as evidenced by the higher N$_2$O fluxes, a phenomenon also observed by Schmer et al. (2012). The two processes could lead to higher soil NO$_3^-$ content under low N rate than higher N rates (Table S1a). These can also elucidate the observation that the N rate did not impact NO$_3^-$ leaching at 100-cm depth (Table 2), which differs from the previous studies for other crops (e.g., Ludwick et al., 1976; Squires, 2013). Moreover, the first process (higher biomass yield with N application leading to greater utilization of soil NO$_3^-$ under the high and medium N rates compared with the low N rate) could reduce the N gaseous emissions under the high and medium N rates in the second process. Meanwhile, the second process (consumed more soil NO$_3^-$ under higher N rates than low N rate) could also limit the switchgrass growth under higher N rates in the first process. The two processes impacted one another. This was different from the relationships between the NO$_3^-$ leaching process and “N$_2$O emissions and switchgrass uptake” because leaching is a physical process (Lamb et al., 2014) and cannot strongly influence the N$_2$O fluxes and switchgrass uptake, even though more N$_2$O emissions and switchgrass uptake can reduce the NO$_3^-$ leaching.

Furthermore, the N rate impacted soil N$_2$O fluxes in 2014 and 2015 but not in 2010–2012 (Table 3 and Table S5). This is primarily due to the impacts of N rate on the parameters that determine soil N$_2$O fluxes. These parameters were reported separately in previous studies. Based on the results of the previous studies (Lin et al., 2000; Tian et al., 2010), the equation for calculating soil N$_2$O fluxes was given:

$$N_2O\ flux = \frac{(0.001 \times N_{nut} + N_{denit}) \times (10^{(0/\Phi \times 0.026 - 1.66)})}{(1 + 10^{(0/\Phi \times 0.026 - 1.66)})},$$

where $N_{nut} \text{ and } N_{denit}$ are the nitrification and denitrification rate, respectively, $\Phi$ is the soil porosity, and $\theta$ is the soil volumetric water content. The $N_{nut}$ and $N_{denit}$ were given by Parton et al. (2001) after synthesizing the results of previous studies.

Fig. 2 Trends of daily mean soil N$_2$O fluxes (g ha$^{-1}$ day$^{-1}$) from switchgrass fields under (a) the three N rates at (b) the two landscape positions and (c) daily maximum ($T_{Max}$) and minimum ($T_{Min}$) air temperature and precipitation over the observed days in 2010–2015. The arrows indicate the fertilization dates. The N fertilizer was applied in late May or early June for each year.
Nitrate fluxes among different N rates were not significantly different among different N rates (Mbonimpa et al.)

1

\[ N_{\text{nit}} = Net_{\text{min}} \times K_1 + K_{\text{max}} \times NH_4 \times F(t) \times F(WFPS) \times F(pH), \]

2

\[ N_{\text{denit}} = 0.5 \times \tan \left( 0.6 \times (0.1 \times WFPS - a) / \pi \right), \]

3

where Net_{\text{min}} is the daily net N mineralization from the soil organic matter (SOM) decomposition, \( K_1 \) is the fraction of Net_{\text{min}} (\( K_1 = 0.20 \)), NH4 is the soil ammonium concentration, \( K_{\text{max}} \) is the maximum fraction of NH4\(_4\) nitrified (\( K_{\text{max}} = 0.10 \)), \( t \) is the soil temperature, WFPS is water-filled pore space (WFPS = \( \theta \times BD/(1-1/BD, 0.65) \); BD, soil bulk density), and \( F \) is function that indicates the effect of the parameter on nitrification rate, \( a \) is a function of soil gas diffusivity (D/D0) and heterotrophic respiration (CO2-het). In this study site, \( t \) and \( \theta \) were not related to N rate. The N rate could increase Net_{\text{min}} through increasing SOM (Brown et al., 2000; Frank et al., 2004), resulting in the increase of N_{net} based on Eq. 2 (Parton et al., 2001). However, the increase in N_{net} is relative small compared to N_{denit} because the proportion of nitrification and denitrification product released as N2O fluxes are 0.001 and 1, respectively, based on Eq. 1. N rate did not impact BD and pH in 2010–2013 (data not shown). WFPS is calculated using \( \theta \) and BD, thereby, it is also not significantly different among different N rates (Fig. S5b). D/D0 is a function of WFPS, BD, and \( \theta \) at field capacity. Combining the equations above, the N2O flux is a function of \( t \), \( \theta \), \( \phi \), Net_{\text{min}}, NH4, WFPS, pH, BD, D/D0, and CO2-het. In this study site, \( t \) and \( \theta \) were not related to N rate. The N rate could increase Net_{\text{min}} through increasing SOM (Brown et al., 2000; Frank et al., 2004), resulting in the increase of N_{net} based on Eq. 2 (Parton et al., 2001). However, the increase in N_{net} is relative small compared to N_{denit} because the proportion of nitrification and denitrification product released as N2O fluxes are 0.001 and 1, respectively, based on Eq. 1. N rate did not impact BD and pH in 2010–2013 (data not shown). WFPS is calculated using \( \theta \) and BD, thereby, it is also not significantly different among different N rates (Fig. S5b). D/D0 is a function of WFPS, BD, and \( \theta \) at field capacity. NH4 was increased with N rates, but it usually does not accumulate in the soil (Woodruff & Ruger, 1948). CO2-het was not significantly different among different N rates (Mbonimpa et al., 2015). Therefore, this could be the reason that N2O fluxes among different N rates were not significantly different based on Eq. 1–3. In 2014 and 2015, however, the increase in SOC and total N at 0- to 5-cm and 5- to 15-cm depths after 5 years of continuous N application could enhance higher denitrification rate under high and medium N rates than the low N rate (Helgason et al., 2005; Luce et al., 2011), along with various temperature and precipitation over the observed years (Fig. S4), resulting in the significant difference of the N2O fluxes between high and/or medium N rates and low N rate (Table 3).

**Topography impacts on soil NO\(_3^n\), NO\(_3^+\) leaching, and N\(_2\)O fluxes**

Topography (landscape position) significantly impacted soil NO\(_3^n\) (Table S1), NO\(_3^+\) leaching (Table 2), and N\(_2\)O fluxes (Table 3) in some years. Topography can strongly impact soils through soil erosion and changes in SOM distribution (Guzman & Al-Kaisi, 2011). Shoulder and backslope positions are generally eroded, and footslope positions are usually deposited (McCarty & Ritchie, 2002); thus, the soil nutrients and SOM are normally accumulated at the footslope (Papiernik et al., 2007). Subsequently, soil NO\(_3^n\) concentrations at the footslope were significantly higher than upper positions (Table S1).

Soil NO\(_3^n\) leaching is a physical event (Lamb et al., 2014). It occurs through the movement of soluble NO\(_3^n\) with water in the soil that must be permeable for water movement (IPNI, 2015). At this study site, the landscape slope ranges from 2% to 20% slope, therefore, water moves through the soil from shoulder and backslope to footslope (downslope) position because of the higher kinetic energy of water, resulting in higher NO\(_3^n\) leaching contents at the footslope. In contrast, water at the backslope position is unable to remain for a longer duration because of the higher slope, leading to lower NO\(_3^n\) leaching content at the backslope compared with footslope and shoulder positions. Therefore, NO\(_3^n\) leaching at the footslope was significantly higher than the other positions.

N\(_2\)O fluxes at the shoulder and footslope positions were not significantly different in 2010–2012 but were significantly different in 2014 and 2015 (Table 3). Soil BD and pH at footslope positions were lower than at the shoulder, but soil moisture content (\( \theta \)) at the footslope was higher in 2010–2013 (data not shown). Therefore, the WFPS at the shoulder position was not significantly different from that of the footslope (Fig. S6), leading to similar \( N_{\text{denit}} \) between the shoulder and footslope based on Eq. (3). Because denitrification is the principal pathway by which the N\(_2\)O emissions enter the atmosphere (Weier et al., 1993), the similar \( N_{\text{denit}} \) between the shoulder and footslope positions could result in the similar N\(_2\)O fluxes observed at these landscape positions in 2010–2012. However, in 2014 and 2015, the perennial switchgrass had been planted for 5 and 6 years, respectively, along with the impacts of climate (especially, the precipitation) over the observed years (Fig. S4), and thereby, the difference of BD, pH, and WFPS between the footslope and shoulder at the 0- to 5- and 5- to 15-cm depths could be higher than in 2010–2012, resulting in a higher difference of \( N_{\text{denit}} \) between footslope and shoulder. This could lead to N\(_2\)O fluxes at the footslope that was significantly higher than the shoulder in 2014 and 2015.

**Temporal variations of soil NO\(_3^n\), NO\(_3^+\) leaching, and N\(_2\)O fluxes**

Time significantly impacted soil NO\(_3^n\), NO\(_3^+\) leaching, and N\(_2\)O fluxes. Soil NO\(_3^n\) concentrations generally had an upward parabola curve from 2009 to 2013 (Fig. S1).
Soil NO$_3^-$ concentrations in 2009 were higher than those in 2010 (Table S1 and Fig. S1). These soil NO$_3^-$ dynamics were primarily attributed to the fact that switchgrass was planted in 2008, and it had very poor stand during the establishment (2008) year, thereby, soil NO$_3^-$ was hardly taken up by switchgrass. Furthermore, due to the poor stand and in order to improve establishment, switchgrass remained unharvested in the field in 2008. Meanwhile, soil NO$_3^-$ was continually supplied through the natural processes of mineralization of SOM and nitrification (Randall & Mulla, 2001). During fall of 2009, switchgrass was harvested, indicating that the higher soil NO$_3^-$ was removed in 2009, resulting in lower soil NO$_3^-$ content in 2010. Similarly, the continuous harvest of switchgrass in 2010 and 2011 led to the lower soil NO$_3^-$ in 2011 and 2012. However, soil NO$_3^-$ in 2013 was higher compared to 2010, 2011, and 2012 (Fig. S1). The possible reasons for this observation in 2013 were two-fold: (i) the deep rooting system of switchgrass had developed an abundant and dense network of arbuscular mycorrhizal (AM) fungi (Hooker et al., 1992; Wang et al., 2011) allowing for greater uptake of nutrients, and (ii) temperature and precipitation in 2013 were more favorable for switchgrass growth compared with 2010–2012. The AM fungi can enhance N mineralization, increasing the availability of N to the plant, particularly, significantly in soils with low nutrient status (Jackson et al., 2008). This indicated that planting switchgrass can store N nutrient to improve soil fertility.

Soil NO$_3^-$ leaching contents followed a downward trend over time (2009–2015; Table S4, Fig. 1, and Fig. S2). This is in accord with previous results (Christian & Riche, 1998) but differs from others (Liu et al., 2003) that reported that the leaching fluctuated over time. However, the reasons for the observation in this study were different from the previous studies that showed that the previous practices and lack of cultivation would have reduced the leaching (Christian & Riche, 1998) and that irrigation and rainfall mainly drove the leaching (Liu et al., 2003). In this study, the downward trend in NO$_3^-$ leaching is several factors. First, an abundance of AM fungi associated with the switchgrass enlarges the nutrient interception zone, prevents nutrient loss after rain-induced leaching events, reduces the volume of soil leachate (Asghari et al., 2005; van der Heijden, 2010), enhances rates of N immobilization and stores more N in roots and rhizomes, thus reducing the risk of N loss via leaching (Cavagnaro et al., 2015). This indicated that the growing switchgrass can reduce the N loss via leaching. Second, noncultivation in this field, since the switchgrass was established in 2008, could reduce SOM mineralization, and thereby can have a favorable effect in reducing leaching (Christian & Riche, 1998). Third, annual precipitation followed a slight downward trend from 2009 to 2015 (Fig. S4). Lower precipitation could result in less soil moisture, which can reduce denitrification rate, reducing the nitrate available for leaching (McIsaac et al., 2010). The positive correlation between the daily mean NO$_3^-$ leaching contents and monthly precipitation (they had a time downward trend; $r = 0.22$; Fig. S2) also supported the result. Fourth, the continuous percolation of precipitation through the soils presumably leached basic cations (e.g., Ca$^{2+}$ and Mg$^{2+}$) and replaced them with acid-forming cations such as H$^+$, Al$^{3+}$, and Fe$^{2+}$ (Tesgaye et al., 2006), which could absorb more soil NO$_3^-$, which was taken up by switchgrass, over the observed years, resulting in the downward trend in the leaching.

Soil N$_2$O fluxes had a downward trend quadratic curve over the years in this switchgrass field (Fig. 2, Fig. S3 and Table S4). This could primarily be attributed to the fact that switchgrass is a perennial with a deep rooting system (Clark et al., 1998; Blanco-Canqui, 2010), environmental conditions such as climate changes, and to nondisturbed soils (the soil has not been cultivated since the switchgrass was established in 2008). Root systems in the soils without disturbance can increase SOM concentration (Thomas et al., 1996) and lower BD (Clark et al., 1998) over the years. The increasing SOM can increase $N_{\text{min}}$, resulting in the increase of $N_{\text{nut}}$ based on Eq 2 (Parton et al., 2001), while the decreasing BD over time can reduce WFPS, decreasing $N_{\text{nut}}$ and $N_{\text{denut}}$ over the years based on Eq 2 and 3, respectively (Parton et al., 2001). The negative effect of BD on $N_{\text{nut}}$ could offset partial position effect of SOM on $N_{\text{nut}}$ which is not the main pathway to produce soil N$_2$O emissions [0.001 and 1 are the proportion of nitrification and denitrification product released as gaseous nitrogen, respectively, based on Eq 1 (Lin et al., 2000; Tian et al., 2010)]. Denitrification is the primary pathway that produces N$_2$O emissions to enter the atmosphere (Weier et al., 1993). Therefore, the decreasing $N_{\text{denut}}$ resulting from the decreasing BD is the main factor in impacting soil N$_2$O fluxes over the years, leading to a downward trend of soil N$_2$O fluxes over the years. The deep root system of switchgrass can enhance translocation of SOC to deeper layers, reducing decomposition of SOC and promoting long-term SOC sequestration (Frank et al., 2004), thereby reducing the microbial processes over years (Liebig et al., 2005; Blanco-Canqui, 2010). Thus, the N$_2$O fluxes could follow a downward trend over time. These findings indicated that growing switchgrass can reduce the N$_2$O fluxes over time. Deep roots can also increase water infiltration rate over time (Katsvairo et al., 2007), resulting in more leaching of NO$_3^-$ in topsoil over time. This reduces the source of N for denitrifying bacteria over time (Hofstra & Bouwman, 2005), resulting in a downward trend of the soil N$_2$O fluxes over the years. However, the function of deep
roots on mitigation of soil N$_2$O emission over time cannot be overestimated. The deep roots cannot continually and infinitely reduce N$_2$O emission. It is more likely that N$_2$O fluxes could be reduced over several years after switchgrass establishment, then keep a stable flux with slight fluctuations in succeeding years (Fig. 2 and Fig. S3). Furthermore, the annual precipitation in 2010 was highest from 2010 to 2015 (Fig. S4). This could result in a downward trend for WFPS over the years, leading to decreasing N$_2$O fluxes. The Figs S5 and S6 showed that the WFPS had a downward trend from 2010 to 2012, but a slight upward trend from 2012 to 2015. The WFPS trends were similar to the N$_2$O curves (Figs S5 and S6) over the years. Therefore, the WFPS may be an important factor in impacting the N$_2$O fluxes in the switchgrass field.

These findings can provide government organizations with demonstrated references to make policy that supports switchgrass as a viable alternative bioenergy feedstock. This is primarily due to many governments that are creating policies to encourage the development of renewable sources of energy (Miller & Kumar, 2013) because (i) the conventional petroleum fuels are one of the primary sources of GHG emissions (Li & Mupondwa, 2014), and in the United States, the largest source of anthropogenic GHG emissions is burning fossil fuels for electricity, heat, and transportation (EPA, 2017), (ii) First generation biofuel crops, including corn, soybean (Glycine max L.), canola (Brassica napus L.), sunflower (Helianthus annuus L.), rapeseed (Brassica napus L.), and palm (Elaeis sp.), have caused several issues: cost increases, negative effects of land use changes (e.g., decrease in forest and grasslands and increasing GHG emissions), and relative decrease in food supply (Miller & Kumar, 2013; Drenth et al., 2014). Also, these findings provide producers with essential information on the optimum application of N fertilizer and its interactions with different positions on a marginal land to improve both their economic incomes and the soil and air quality in South Dakota.

Conclusions

The present study was conducted to evaluate the impacts of N fertilizer, landscape position, and time on the N dynamics in switchgrass seeded to a marginally yielding cropland. The data from this study indicate that switchgrass has the potential for improving the soil N dynamics, and environmental sustainability of marginally yielding croplands. It was concluded that (i) N rate impacted soil NO$_3^-$ only at the top soil and N$_2$O fluxes but did not impact soil NO$_3^-$ leaching under switchgrass field in South Dakota, United States, (ii) medium N (56 kg N ha$^{-1}$) rate is the optimal rate for the switchgrass to increase the biomass yield, and reduce soil N$_2$O emissions and NO$_3^-$ leaching, (iii) topography impacted the N dynamics; soil NO$_3^-$, NO$_2^-$ leaching, and soil N$_2$O fluxes were, in general, higher at the footslope compared to the other positions, (iv) switchgrass seeded on marginally yielding cropland can store N nutrient to improve soil fertility, reduce N loss via leaching, and mitigate N$_2$O emissions from soils to the atmosphere over the observed years. Switchgrass can help in reducing N losses over time by improving the soil properties of the marginally yielding fields and can be grown as a sustainable bioenergy crop on these fields.

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Supporting Information

Additional Supporting Information may be found online in the supporting information tab for this article:

Table S1a. Annual mean soil nitrate (NO₃⁻) content for the 0- to 5- and 5- to 15-cm depths from 2009 to 2013 under high, medium, and low N rates applied to switchgrass seeded at the shoulder, backslope, and footslope positions.

Table S1b. Annual mean soil nitrate (NO₃⁻) content for the 15- to 30- and 30- to 60-cm depths from 2009 to 2013 under high, medium, and low N rates applied to switchgrass seeded at the shoulder, backslope, and footslope positions.

Table S1c. Annual mean soil nitrate (NO₃⁻) content for the 0- to 5- and 5- to 15-cm depths from 2009 to 2013 under high, medium, and low N rates applied to switchgrass seeded at the shoulder, backslope, and footslope positions.

Table S2. Mean soil NO₃⁻ content at the 30-60-cm depth in 2010 and 5-15-cm depth in 2012 for each N rate and position.

Table S3. Annual mean soil NO₃⁻ leaching content at the 100 cm depth in 2010 and 2011 for each N rate and position.

Table S4. P-values of trend test and estimated slopes for annual mean soil NO₃⁻ leaching content (mg L⁻¹) and N₂O fluxes (g ha⁻¹ d⁻¹) from switchgrass fields under different N rates and landscape positions over the observed years.

Table S5. Seasonal mean soil N₂O fluxes from 2010 to 2015 under high, medium, and low N rates applied to switchgrass seeded at the shoulder, backslope, and footslope positions.

Table S6. Nitrogen level effect on Nitrogen-use efficiency (NUE) in switchgrass biomass harvested at this study site from 2011 to 2015.

Figure S1. Distributions of annual mean soil nitrate (NO₃⁻) contents (mg kg⁻¹) under (a) high, (b) medium, and (c) low N rates applied to switchgrass seeded at (d) shoulder, (e) backslope, and (f) footslope positions for the 0- to 5- (D1), 5- to 15- (D2), 15- to 30- (D3), 30- to 60- (D4), and 60- to 100-cm (D5) depths from 2009 to 2013.

Figure S2. Trends of (a) daily mean NO₃⁻ leaching content and (b) monthly mean maximum (T max) and minimum (T min) air temperature and monthly precipitation at the sampling days from 2009 to 2015.

Figure S3. Trends of annual average soil NO₃⁻ fluxes (g ha⁻¹ d⁻¹) under (a) the three N rates applied to switchgrass seeded at (b) the two positions over the observed years.

Figure S4. Trends of annual precipitation and mean air temperature from 2009 to 2015.

Figure S5. Trends of (a) daily mean N₂O fluxes (b) daily water-filled pore space (WFPS) under high, medium, and low N rates and (c) daily mean maximum (T max) and minimum (T min) air temperature and daily precipitation from 2009 to 2015.

Figure S6. Trends of (a) daily mean N₂O fluxes (b) daily water-filled pore space (WFPS) at shoulder and footslope positions and (c) daily mean maximum (T max) and minimum (T min) air temperature and daily precipitation from 2009 to 2015.