Nitrous Oxide Emissions from Saturated Riparian Buffers: Are We Trading a Water Quality Problem for an Air Quality Problem?

Morgan P. Davis,* Tyler A. Groh, Dan B. Jaynes, Timothy B. Parkin, and Thomas M. Isenhart

Abstract

Reestablishing perennial vegetation along riparian areas in agricultural landscapes increases ecosystem diversity, provides wildlife corridors, and reduces sediment and nutrient losses from overland flow (Lee et al., 2000, 2003; Berges et al., 2010; McCracken et al., 2012). Multifunctional benefits of vegetated riparian areas have been promoted by the USDA through the establishment of filter strips and riparian forest buffers as a part of the conservation reserve program, with 647,162 ha currently enrolled (FSA, 2017). Whereas riparian buffers reduce nutrient losses from overland flow (Lee et al., 2003), traditional buffers are ineffective at removing nutrients routed through artificial subsurface (tile) drains. Nitrogen flux via tile drainage is a primary source of NO3 in surface waters in the Mississippi River basin (David et al., 2010; Amado et al., 2017). Nitrate concentrations in surface water of the US Corn Belt often exceed USEPA's drinking water standard of 10 mg NO3–N L–1 (Cambardella et al., 1999; Schilling and Zhang, 2004), and N loading affects the size of the hypoxic zone in the Gulf of Mexico (Scaivia et al., 2003; Van Meter et al., 2018).

Nutrient reduction strategies have been developed by many states in the Mississippi River basin to reduce N flux to the Gulf of Mexico through N management and edge-of-field conservation practices (Illinois EPA, 2015; IDALS, IDNR, and ISU, 2017). Edge-of-field practices are designed to remove NO3 primarily through microbial denitrification and include wetlands, bioreactors, and drainage water management. Wetlands are efficient at removing NO3 from large drainage areas (Kovacic et al., 2000; Tömer et al., 2013; Groh et al., 2015) but are expensive to implement and maintain. Woodchip bioreactors can also be a costly practice with maintenance required in the eventual replacement of woodchips (Christianson et al., 2013; Addy et al., 2016). Drainage water management requires regular monitoring and management with conflicting reports on NO3 removal efficiency (Drury et al., 1996; Lavaire et al., 2017). Saturated riparian buffers (SRBs) are a newly developed conservation practice that improve NO3 removal capabilities of traditional conservation buffers to include the treatment of tile drainage (Jaynes and Isenhart, 2014).

Saturated riparian buffers function by intercepting tile water within a distribution box located just inside the buffer near the

Copyright © American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America. 5585 Guilford Rd., Madison, WI 53711 USA. All rights reserved.

J. Environ. Qual.
doi:10.2134/jeq2018.03.0127
Received 6 Apr. 2018.
Accepted 12 July 2018.
*Corresponding author (morgand@iastate.edu).

Abbreviations: BC-1, Bear Creek Site 1; EF5r, emission factor for estuaries; EF5e, emission factor for rivers; GC, gas chromatograph; IA-1, Iowa Site 1; IPCC, Intergovernmental Panel on Climate Change; PVC, polyvinyl chloride; SRB, saturated riparian buffer.

Published online September 6, 2018
field edge. From the distribution box, drainage water is routed into lateral distribution tiles that are installed perpendicular to the field tile and extend along the buffer. Lateral distribution tiles reintroduce drainage water as shallow groundwater into riparian buffer soils (Fig. 1). As drainage water seeps through C-rich alluvial soil, NO\textsubscript{3} is removed through microbial denitrification or stored through microbial immobilization and plant uptake (Jaynes and Isenhart, 2014). Measurements in conjunction with this study indicated denitrification to be the primary mechanism for NO\textsubscript{3} removal in SRBs (Groh et al., 2018). These findings support other studies that found NO\textsubscript{3} removal in traditional riparian buffers with high water tables was primarily through denitrification (Simmons et al., 1992; Ranalli and Macalady, 2010). Complete denitrification is ideal for NO\textsubscript{3} removal, converting NO\textsubscript{3} to N\textsubscript{2} gas. However, incomplete denitrification can result in the production of N\textsubscript{2}O, a powerful greenhouse gas. Nitrous oxide is the third largest greenhouse gas contributor to global radiative forcing and has a global warming potential 282 times (50-yr lifetime adjustment) that of CO\textsubscript{2} (Myhre et al., 2013). Nitrous oxide is 5.5% of the total greenhouse gas inventory in the United States, with agricultural soils representing 75.1% of the total N\textsubscript{2}O inventory (Desai and Harvey, 2017).

Direct N\textsubscript{2}O emissions from the surface of corn (Zea mays L.)–soybean [Glycine max (L.) Merr.] rotations and perennial riparian areas have been well studied (Kim et al., 2009), with the greatest annual N\textsubscript{2}O emissions from fertilized corn (Parkin and Kaspar, 2006; Kim et al., 2009; Fisher et al., 2014; Iqbal et al., 2015). Implementation of edge-of-field conservation practices to remove NO\textsubscript{3} from agricultural drainage could result in increased N\textsubscript{2}O production. Nitrous oxide production from wetlands and bioreactors has been found to be 0.003 to 5.5% of the total N removed (Greenan et al., 2009; Woli et al., 2010; Groh et al., 2015; Bruun et al., 2017). Although some studies have found surface emission rates equal to agricultural fields on a per area basis, these rates are greatly reduced if emissions are considered on a per area of drained land basis (Groh et al., 2015). Furthermore, most studies do not include dissolved N\textsubscript{2}O production or the potential reduction in indirect N\textsubscript{2}O emissions from NO\textsubscript{3} removal.

The Intergovernmental Panel on Climate Change (IPCC) considers indirect N\textsubscript{2}O emissions as N\textsubscript{2}O production in groundwater, rivers, and estuaries from NO\textsubscript{3} that has left agricultural soils (IPCC, 2006). Indirect N\textsubscript{2}O emissions from fertilizer application are difficult to accurately quantify and are often not included in N\textsubscript{2}O budgets from agricultural areas (Reay et al., 2009). However, edge-of-field practices that remove NO\textsubscript{3} from tile drainage have the potential to reduce indirect N\textsubscript{2}O emissions from downstream denitrification.

Research reported here represents the first study of N\textsubscript{2}O emissions from SRBs. We studied two SRBs for 3 yr to address the question: are SRBs trading a water quality problem for an air quality problem? Our objective was to compare N\textsubscript{2}O emissions from SRBs, traditional buffers, and crop fields in corn–soybean rotations. Specific objectives were (i) to quantify annual N\textsubscript{2}O emissions from the soil surface of SRBs, traditional buffers, and crop fields; (ii) to quantify dissolved N\textsubscript{2}O in tile and diverted flow; (iii) to estimate indirect N\textsubscript{2}O emissions from SRBs and drainage tiles; and (iv) to compare total N\textsubscript{2}O emissions from SRBs, traditional buffers, and corn–soybean rotations.
Materials and Methods

Site Descriptions

Nitrous oxide was measured at two SRBs, Bear Creek Site 1 (BC-1) and Iowa Site 1 (IA-1), in Hamilton County in central Iowa. Bear Creek Site 1 was studied in Jaynes and Isenhart (2014), and both site abbreviations are consistent with nomenclature in Jaynes and Isenhart (2018) and Groh et al. (2018). Sites are 12 km apart and receive drainage water from independent fields. Three chambered distribution boxes were installed at both sites with distribution tiles placed at a depth of 70 cm. Pressure transducers (AST4510, American Sensor Technologies) and 45° v-notch weirs were used to calculate field tile flow, diverted flow, and overflow. Details of v-notch calibration and flow calculations can be found in Jaynes and Isenhart (2014, 2018).

Bear Creek Site 1 was installed in October of 2010 into a riparian forest buffer alongside Bear Creek, a second-order stream. The field tile diverting flow into BC-1 drains 5.9 ha of cropland. The distribution tile is 305 m in length, and diverted water seeps through 21 m of buffer soil (0.64 ha) before entering Bear Creek. The riparian forest buffer was established in 1995 and consisted of 6 m of sugar maple (Acer saccharinum L.), 6 m of mixed shrub–grass, and 8 m of switchgrass (Panicum virgatum L.). Additional details on shrub species can be found in Schultz et al. (1995). Soils across the buffer and at the field edge are described as poorly drained Coland series (fine-loamy, mixed, superactive, mesic Cumulic Hapludolls) complex (USDA-NRCS, 2018). The agricultural field adjacent to BC-1 was planted to soybean in 2015, corn in 2016, and soybean in 2017. Anhydrous NH₃ was injected into the crop field at a rate of 120 kg N ha⁻¹ on 19 Apr. 2016.

Iowa Site 1 was installed in June 2013 into a filter strip seeded to switchgrass in 2000. The distribution tile is 308 m in length and 24 m from the stream (0.74 ha). The distribution box at IA-1 receives subsurface drainage water from 4.7 ha of cropland draining into a small tributary of the South Skunk River. Soils at IA-1 are described as a Coland–Terrill (fine-loamy, mixed, superactive, mesic Cumulic Hapludolls) complex (USDA-NRCS, 2018). The agricultural field adjacent to the IA-1 SRB was planted to the same rotation as BC-1. Anhydrous NH₃ was injected into the crop field at a rate of 120 kg N ha⁻¹ on 25 Apr. 2016.

Nitrate Removal

Nitrate samples were collected from the distribution box and at sampling wells (2.3 m deep and fully screened) on the stream edge of the buffer (Fig. 1). Water samples were collected from the distribution box and each well on a weekly basis while each SRB was flowing. Samples were stored at 4°C until analysis for NO₃ using a Lachat 8000 (Zellweger Analytics, Lachat Instrument Division). Diverted flow was calculated from continuous pressure transducer measurements (Jaynes and Isenhart, 2018). Nitrate removal was calculated by subtracting the diverted load from the shallow groundwater load of NO₃ flowing from the SRB. Annual mass NO₃ loads were calculated by multiplying NO₃ concentration by the volume of water between sampling dates and summing over the calendar year.

Nitrous Oxide Emissions from the Soil Surface

Nitrous oxide soil fluxes were measured using static vented chambers equipped with automated sample collection equipment (Davis et al., 2018). Nitrous oxide fluxes from SRBs and traditional buffers were measured from January 2015 through December 2017. Crop field soil N₂O fluxes were collected from January 2016 through December 2017. Circular schedule 40 polyvinyl chloride (PVC) anchors (30-cm diam., 15-cm height) were pushed into the soil leaving 5 cm of exposed anchor above the soil surface. Nine anchors were evenly spaced across both SRBs, and six anchors were installed on traditional buffer counterparts (Fig. 1). Traditional buffers were adjacent to SRBs but not affected by flow diverted into the SRBs. Six anchors were installed ~3 m from the buffer edge within the crop field. Field anchors were placed to reduce the potential of edge effects but remained in a poorly drained soil classification for treatment comparison. Anchors were installed in pairs, with one anchor over a planting row and the other in the interrow. Each pair was spaced evenly over the length of the SRB (Fig. 1). Crop field anchors were only removed for planting, harvest, tillage, and fertilizer application. After fertilizer application, the in-row anchors were placed to include a single fertilizer injection line.

Flux measurements were collected weekly from April through September, twice per month in October, November, and March, and monthly from December through February. Sample were collected between 8:00 AM and 12:00 PM to limit diurnal biases (Parkin, 2008). Weekly sampling intervals have been calculated to have a 90% probability of estimating the average N₂O flux with ±20% accuracy (Parkin, 2008). We estimate from Kim et al. (2009) that 85% of annual N₂O emissions from traditional buffers and cropland in the Bear Creek watershed occur between March and October, with soil thawing events and fertilization producing the largest fluxes.

Chamber tops were constructed from 30-cm schedule 40 PVC pipe to a height of 15 cm, as described in Parkin and Ventera (2010). Chambers were vented and covered with reflective tape to minimize temperature change from solar radiation. Samples were collected from a butyl rubber stopper (Voigt Global) sampling port in the top of each chamber. Automated samplers collected headspace gas at 0, 15, 30, and 45 min after chamber placement through a sampling tube inserted into the sampling port. Samples were stored in the automated sampler in 20-mL syringes equipped with stopcocks until laboratory analysis (Davis et al., 2018).

In the laboratory, 13 mL of sample was injected into evacuated 6-mL glass vials scaled with butyl rubber stoppers (Voigt Global). Gas samples were analyzed on a gas chromatograph (GC) (SRI Instruments, model 8610) equipped with an automated sampler to introduce gas samples into the sample valve of the GC (Arnold et al., 2001). Gas samples traveled through a stainless steel column (0.3175-cm diam. × 74.54-cm length) packed with Haysep D to a 63Ni electron capture detector. Nitrous oxide standards (Air Liquide Specialty gases) were analyzed to calculate sample concentration using linear regression coefficients.

Fluxes were calculated using the “HMR” package in R 3.1.2 (R Development Core Team, 2014). If the HMR model failed to calculate a flux, the software used linear regression or assigned a “no flux” value of zero (Pedersen et al., 2010). Based on linear regression coefficients from Parkin et al. (2012), minimum detectable fluxes were ±18.5 g ha⁻¹ d⁻¹ for the HMR method and ±3.1 g ha⁻² d⁻¹ for linear regression. Annual cumulative fluxes were calculated using linear interpolation between daily
flaxes and summing daily fluxes for the calendar year. Saturated and traditional buffer annual N\textsubscript{2}O emissions from the soil surface were compared within site-years using Welch’s t test. Annual emissions from SRBs and traditional buffers were also compared among site-years using a paired t test.

**Nitrous Oxide Emissions from Groundwater and Tile Drainage**

Nitrous oxide produced in soil may be dissolved in soil water and enter surface waters from tile drainage and shallow groundwater. In this study, we measured dissolved N\textsubscript{2}O load from tile drainage and shallow groundwater within the SRB. Dissolved N\textsubscript{2}O samples were collected monthly (2015–2017) from the distribution box and sampling wells at each SRB (Fig. 1). Samples were collected using a peristaltic pump. Wells were evacuated and allowed to refill before sample water was pumped into a sampling syringe held onto the end of the sampling tube. Triplicate 10-mL samples were collected from the sampling tube and injected into evacuated 20-mL glass vials sealed with rubber butyl stoppers and treated with 0.3 mL of 80% ZnCl\textsubscript{2} solution for preservation. Samples were stored on ice in the field, and vial headspace pressure was adjusted to atmospheric pressure in the laboratory. Vials were overfilled with 7 mL of He to use the automated sampler design to introduce samples to the GC (described above). Vials were shaken for 15 min on a reciprocal shaker to equilibrate dissolved N\textsubscript{2}O with the headspace.

Total dissolved N\textsubscript{2}O was calculated using Henry’s law and Bunsen absorption coefficients (Tiedje, 1994). Nitrous oxide standards were prepared in a similar manner, including atmospheric pressure adjustment and He dilution. Nitrous oxide concentrations (calculations described above) were linearly interpolated between sampling points and multiplied by daily water volumes to calculate dissolved N\textsubscript{2}O loads. The volume of seepage water through the buffer was assumed to be equal to the volume of water diverted into the SRBs. Annual N\textsubscript{2}O loads were calculated for water leaving crop fields and diverted into the SRBs, leached out of the SRBs, and left as overflow discharge. Total indirect N\textsubscript{2}O load from SRB groundwater was calculated by summing overflow loads to seepage water loads.

**Indirect Nitrous Oxide Emissions from Rivers and Estuaries**

Indirect N\textsubscript{2}O emissions within rivers and estuaries from NO\textsubscript{3} flux from shallow groundwater and tile drainage were calculated using the Tier 1 IPCC protocol for estimating indirect N\textsubscript{2}O emissions (IPCC, 2006). The mass of NO\textsubscript{3}−N leached was multiplied by 0.005 kg N\textsubscript{2}O-N kg−1 NO\textsubscript{3}–N, the sum of the default emission factors for rivers (EF\textsubscript{r} = 0.0025 kg N\textsubscript{2}O-N kg−1 NO\textsubscript{3}–N) and estuaries (EF\textsubscript{e} = 0.0025 kg N\textsubscript{2}O-N kg−1 NO\textsubscript{3}–N). Paired t tests were used to examine differences in means of annual dissolved and indirect N\textsubscript{2}O emissions from SRBs compared with the field tiles.

**Nitrous Oxide Loads from Two-Year Rotation**

Cumulative N\textsubscript{2}O loads for traditional buffers and SRBs were calculated for a 2-yr corn–soybean crop rotation, 2016 to 2017. Soil surface N\textsubscript{2}O loads (kg N) were calculated by multiplying 2016 to 2017 cumulative emission rates (kg N ha−1) by the surface area (ha) of the respective SRB. Surface N\textsubscript{2}O loads from traditional buffers and crop fields represent the potential N\textsubscript{2}O loads for the given SRB area. Total N\textsubscript{2}O emissions were calculated by adding direct and indirect loads. Statistical analyses could not be conducted on only two total emission budgets for each treatment. However, we believe that presenting total load data over a corn–soybean rotation best represents N\textsubscript{2}O emissions for treatment comparison.

**Results**

**Tile Flow and Nitrate Removal**

Tile discharge from crop fields ranged from 8279 to 28,772 m\textsuperscript{3} yr\textsuperscript{−1} across SRB sites (Table 1). On average, 40% of total annual flow was diverted into BC-1, whereas an average of 95% of annual flow was diverted into IA-1. The discrepancy in percentage of diverted flow between sites is attributed to a combination of in-field drain tile density (BC-1 > IA-1) and the difference between SRB area to drainage area ratios (0.11 for BC-1 and 0.16 for IA-1). Bear Creek 1 drains a larger area (5.9 ha) into a smaller buffer (0.64 ha) than IA-1 (4.7 ha draining to 0.74 ha of buffer). Diverted NO\textsubscript{3} load ranged from 51 to 85 kg N at BC-1 and from 40 to 70 kg N at IA-1. Flow-weighted mean concentration of diverted NO\textsubscript{3}, ranged from 8.2 to 13.0 mg N L\textsuperscript{−1} at BC-1 and from 3.8 to 7.6 mg N L\textsuperscript{−1} at IA-1. Nitrate removal in the SRBs ranged from 47 to 80 kg N at BC-1 and from 36 to 70 kg N at IA-1 (Table 1). Average removal rates from diverted NO\textsubscript{3} loads were 94% for BC-1 and 95% for IA-1. However, total NO\textsubscript{3} load removal rates were lower at BC-1 (38%) than at IA-1 (88%) due to limited tile water diversion capacity at BC-1.

**Annual Soil Surface Nitrous Oxide Emissions**

Nitrous oxide flux ranged from −1.3 to 533.1 g N ha−1 d−1, and flux was greatest after corn fertilization in late spring and early summer months of 2016 (Fig. 2). Fluxes were <40 g N ha−1 d−1 for 95% of the measured events. Snow accumulation deeper than the anchor height prevented measurements in February 2016. Measurements collected in June 2015 were lost through a malfunction in sample analysis. Missing measurements did not change cumulative emission methodology, and linear interpolation was conducted with remaining flux measurements. Annual N\textsubscript{2}O emissions from soil surfaces ranged from 0.87 to 16.20 kg N ha−1 (Fig. 3). Nitrous oxide emissions from fertilized corn were more than twice the greatest annual emission from SRBs and traditional buffers. Field measurements were excluded from annual N\textsubscript{2}O comparisons to focus attention on the treatment effect of saturating a traditional buffer. Field soil surface emissions were expected to be greater than any perennial system without N fertilizer application (Parkin and Kaspar, 2006; Kim et al., 2009; Fisher et al., 2014). Comparing SRBs to traditional buffers is a more conservative approach for evaluating significant changes in N\textsubscript{2}O emissions from the effect of SRB installation. Annual N\textsubscript{2}O emissions from saturated buffers ranged from 1.12 to 5.83 kg N ha−1. The greatest annual emission at both sites was in 2015. Saturated riparian buffer emissions were significantly greater (P = 0.045) than traditional buffers for only one site-year, BC-1 in 2015 (Fig. 3). Comparing among site-years, annual surface N\textsubscript{2}O emissions from SRBs were not significantly different (P = 0.17) from traditional buffers.
Dissolved and Indirect Nitrous Oxide Emissions

Annual load of dissolved N\textsubscript{2}O leaving crop fields in tile drains ranged from 0.31 to 1.28 kg N. Dissolved N\textsubscript{2}O load diverted into SRBs ranged from 0.19 to 0.67 kg N. Nitrous oxide production was observed at BC-1, where dissolved N\textsubscript{2}O load in seepage water leaving the SRB was greater than the diverted N\textsubscript{2}O load into the SRB. Seepage N\textsubscript{2}O load from IA-1 was less than the N\textsubscript{2}O load diverted into the SRB, indicating apparent consumption or surface release of N\textsubscript{2}O. Total dissolved N\textsubscript{2}O loads from SRB groundwater (0.06–1.69 kg N) were not significantly different ($P = 0.41$) than tile N\textsubscript{2}O loads (0.31–1.28 kg N) among all site-years (Table 2). Estimated indirect emissions of tile NO\textsubscript{3} discharged to rivers and estuaries ranged from 0.205 to 1.165 kg N. Saturated riparian buffers removed NO\textsubscript{3}, significantly reducing ($P = 0.001$) indirect N\textsubscript{2}O emissions from rivers and estuaries by an estimated 0.15 to 0.4 kg N yr\textsuperscript{−1}. Saturated riparian buffer contributions to total indirect N\textsubscript{2}O emissions (groundwater, rivers, and estuaries) were significantly less ($P < 0.02$) than contributions from tile drains (Table 2).

### Table 1. Flow, NO\textsubscript{3} concentration, and NO\textsubscript{3} load from the field tiles and saturated riparian buffers. Days of flow are reported as the number of days that flow occurred.

<table>
<thead>
<tr>
<th>Site†</th>
<th>Year</th>
<th>Flow</th>
<th>Flow-weighted NO\textsubscript{3}</th>
<th>NO\textsubscript{3} load</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Days</td>
<td>Tile (m\textsuperscript{3})</td>
<td>Diverted (mg N L\textsuperscript{−1})</td>
</tr>
<tr>
<td>BC-1</td>
<td>2015</td>
<td>212</td>
<td>28,772</td>
<td>10,339</td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>161</td>
<td>12,111</td>
<td>5,878</td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>97</td>
<td>11,153</td>
<td>3,922</td>
</tr>
<tr>
<td>IA-1</td>
<td>2015</td>
<td>179</td>
<td>12,519</td>
<td>11,453</td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>185</td>
<td>8,279</td>
<td>8,056</td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>133</td>
<td>9,838</td>
<td>9,575</td>
</tr>
</tbody>
</table>

† BC-1, Bear Creek Site 1; IA-1, Iowa Site 1.
Saturated riparian buffers have shown early promise as a conservation practice to remove NO₃ from tile drainage (Jaynes and Isenhart, 2014, 2018). Denitrification is a primary mechanism of NO₃ removal (Groh et al., 2018), and incomplete denitrification can result in the production of N₂O. Concerns of trading water quality problems for air quality problems prompted the measurement of direct and indirect N₂O emissions from two SRBs in central Iowa. Annual direct N₂O emissions from the soil surface were greatest in the crop field in 2016 at both sites. Annual soil surface N₂O fluxes from corn were 16.2 kg N ha⁻¹ for BC-1 and 15.3 kg N ha⁻¹ for IA-1. Observed emissions from fertilized corn in central Iowa average ~10 kg N ha⁻¹ yr⁻¹ (Parkin and Kaspar, 2006; Kim et al., 2009; Iqbal et al., 2015). Greater than average annual N₂O emissions from fertilized corn in our study are likely attributed to measurements focused on poorly drained soils within the crop field over a wet year. Nitrous oxide emissions from poorly drained soils are often greater than from well-drained counterparts (Davidson et al., 2000; Iqbal et al., 2015). Crop field measurements for this study were made on soils that were representative of SRB soils to emphasize the magnitude of N₂O emissions from cultivated riparian areas. Surface N₂O emissions from SRBs and traditional buffers were less than crop fields, and similar to other measurements in riparian areas under perennial vegetation in central Iowa (Kim et al., 2009; Iqbal et al., 2015). Annual surface N₂O emissions from SRBs were significantly greater ($P = 0.045$) than traditional riparian buffers only at BC-1 in 2015 (Fig. 3). In 2015, BC-1 generated the greatest tile flow (28,772 m³), number of tile flow days (212), and diverted NO₃ load (85 kg N) over the study period (Table 1). Increased flow and NO₃ load through BC-1 in 2015 provided the longest period of saturated conditions for the potential to increase denitrification compared with other years. Over all 3 yr and at both sites, N₂O emissions from SRBs were not significantly different ($P = 0.165$) from traditional buffers.

Nitrous oxide emissions from groundwater and tile drainage were measured and emissions from rivers and estuaries were estimated to provide a complete assessment of N₂O losses from SRBs. Few studies have quantified dissolved N₂O in tile drainage or in riparian groundwater (Groffman et al., 1998; Blicher-Mathiesen and Hoffmann, 1999; Sawamoto et al., 2005). Dissolved N₂O concentrations in our study were wide ranging, from 0.1 to 981.1 mg N L⁻¹, with the greatest concentrations from measurements made at the tile outlet (data not shown). A large range of dissolved N₂O concentrations has also been reported in other studies measuring dissolved N₂O in drainage water (Reay et al., 2003; Sawamoto et al., 2003; Parkin et al., 2016). Measurements of dissolved N₂O in this study were made with a greater frequency than in past studies, but greater measurement frequency may provide insights into flow event and seasonal variations. High N₂O concentrations in both tile drainage and SRB seepage water highlight the significance of dissolved N₂O to total N₂O emissions from edge-of-field practices designed to remove NO₃.

**Table 2. Measured dissolved, indirect, and total N₂O emission loads for groundwater (saturated riparian buffer [SRB] and tile) and rivers and estuaries (SRB and tile). Groundwater load SDs are given in parentheses. Indirect river and estuary emissions were estimated using the Intergovernmental Panel on Climate Change Tier 1 protocol.**

<table>
<thead>
<tr>
<th>Site†</th>
<th>Year</th>
<th>Dissolved N₂O emissions</th>
<th>Indirect N₂O emissions</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Groundwater</td>
<td>Tile</td>
<td>SRB</td>
</tr>
<tr>
<td></td>
<td></td>
<td>kg N</td>
<td>kg N</td>
<td>kg N</td>
</tr>
<tr>
<td>BC-1</td>
<td>2015</td>
<td>1.690 (1.556)</td>
<td>1.279 (0.065)</td>
<td>0.765</td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>0.436 (0.537)</td>
<td>0.366 (0.008)</td>
<td>0.300</td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>0.639 (0.757)</td>
<td>0.731 (0.065)</td>
<td>0.540</td>
</tr>
<tr>
<td>IA-1</td>
<td>2015</td>
<td>0.074 (0.047)</td>
<td>0.325 (0.014)</td>
<td>0.030</td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>0.036 (0.020)</td>
<td>0.311 (0.001)</td>
<td>0.045</td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>0.063 (0.017)</td>
<td>0.694 (0.003)</td>
<td>0.055</td>
</tr>
</tbody>
</table>

† BC-1, Bear Creek Site 1; IA-1, Iowa Site 1.
from tile drainage. Future studies would benefit from not being limited to only soil surface measurements, but also including dissolved N$_2$O measurements.

Nitrous oxide in diverted water was consumed or lost through surface emissions as it seeped through soil at IA-1 but was produced at BC-1. Apparent N$_2$O consumption resulted in a reduction of dissolved N$_2$O losses from groundwater at IA-1, whereas N$_2$O production at BC-1 resulted in an increase in dissolved N$_2$O concentrations from tile inlet water to SRB seepage water. Nitrous oxide production was primarily found in one of the five sampling wells at BC-1. Nitrate concentrations were greatest in this sampling well during times of N$_2$O production (data not shown). Nitrous oxide reductase, the enzyme catalyzing N$_2$O reduction to N$_2$, has shown sensitivity to environmental factors including O$_2$ concentration, C/NO$_3$ ratios, and pH (Cavigelli and Robertson, 2001). Preferential flow may have played a role in greater NO$_3$ concentrations and affected environmental factors controlling N reductase activity. Discrepancies in dissolved NO$_3$ and N$_2$O concentrations highlight the potential for improving experimental designs in future SRB studies. Improved designs could include a greater number of sampling wells at the buffer edge to capture a greater spatial variability. Tracer studies could also be used to attribute specific flow proportions to each sampling well (Czapar et al., 1994; Jaynes et al., 2001). However, production at BC-1 was <0.07 kg N yr$^{-1}$, and indirect groundwater emissions from SRBs were not significantly different ($P = 0.41$) than the crop field tile (Table 2).

Indirect N$_2$O emissions from rivers and estuaries were estimated using the IPCC Tier 1 protocol by multiplying NO$_3$ leached in groundwater and lost in tile drainage by default emission factors (EF$_w$ = 0.0025, EF$_e$ = 0.0025). Estimated indirect N$_2$O emissions from rivers and estuaries were significantly less ($P = 0.0007$) from SRBs compared with the crop field due to lower NO$_3$ discharge to streams after diversion into the SRB (Table 2). Nitrate loads from crop field tiles were reduced by 34 to 92%, subsequently reducing indirect N$_2$O emissions from rivers and estuaries. Recent studies have proposed an increase in the default EF$_w$ to 0.0075 kg N$_2$O-N kg$^{-1}$ N (Beaulieu et al., 2011), and a proposed regional EF$_e$ of 0.015 kg N$_2$O-N kg$^{-1}$ N for the Upper US Midwest (Turner et al., 2015). Change in the default value of EF$_w$ would magnify the effect of lower indirect N$_2$O emissions from rivers and streams resulting from NO$_3$ diversion through SRBs. However, these studies do not propose changes to the emission factor for groundwater (EF$_w$) or estuaries (EF$_e$), and the proposed increases in EF$_w$ would

---

Table 3. Total N$_2$O loads from saturated (SRB) and traditional riparian buffers. Direct loads were calculated from surface emissions and measured groundwater and tile loads. Indirect loads were calculated through river and estuary emission estimations. Total emissions were calculated by summing direct and indirect emissions. P values indicated the level of significance for SRBs versus traditional buffers.

<table>
<thead>
<tr>
<th>Site†</th>
<th>Year</th>
<th>Surface N$_2$O emissions</th>
<th>Groundwater and indirect N$_2$O emissions</th>
<th>Total N$_2$O emissions</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>SRB</td>
<td>Traditional</td>
<td>SRB</td>
</tr>
<tr>
<td>BC-1</td>
<td>2015</td>
<td>5.76</td>
<td>1.05</td>
<td>2.455</td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>1.68</td>
<td>2.39</td>
<td>0.736</td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>1.20</td>
<td>0.55</td>
<td>1.179</td>
</tr>
<tr>
<td>IA-1</td>
<td>2015</td>
<td>2.17</td>
<td>0.81</td>
<td>0.104</td>
</tr>
<tr>
<td></td>
<td>2016</td>
<td>1.88</td>
<td>1.38</td>
<td>0.081</td>
</tr>
<tr>
<td></td>
<td>2017</td>
<td>1.61</td>
<td>0.87</td>
<td>0.118</td>
</tr>
<tr>
<td>P</td>
<td></td>
<td>0.17</td>
<td>0.02</td>
<td>0.37</td>
</tr>
</tbody>
</table>

† BC-1, Bear Creek Site 1; IA-1, Iowa Site 1.
be within the combined EF, uncertainty range (0.0005–0.025 kg N₂O-N kg⁻¹ N). Our estimations of EF₅g ranged from 0.0036 to 0.017 kg N₂O-N kg⁻¹ N among site years, within the total EF, uncertainty range. Reduction in indirect emissions from water diverted into SRBs resulted in a significant (P = 0.02) reduction in total dissolved emissions from groundwater, rivers, and estuaries. Although no reduction in measured groundwater N₂O was observed, the estimated reduction of N₂O emissions from rivers and estuaries was significant (P = 0.001) and the driver of the reduction in total indirect N₂O emissions from SRBs.

Total N₂O loads (soil surface + dissolved + indirect) from SRBs were found to be similar to traditional buffers, and less than cropped land for a corn–soybean rotation (Fig. 4). Annual SRB loads were not significantly different (P = 0.37) from traditional buffers (Table 3). Nitrous oxide loads from corn–soybean rotations were ~10 kg N greater than SRB and traditional buffers at both sites. Fertilized corn was the single greatest contributor to N₂O loads across the three treatments. Nitrous oxide fluxes in summer months after corn fertilization (Fig. 2) represented >90% of total emissions from cropland in 2016. Nitrous oxide production per kg N removed in SRBs ranged from 3.2 to 9.3%, greater than other edge of field practices (Kovacic et al., 2000; Christianson et al., 2013; Groh et al., 2015; Bruun et al., 2017). The greatest percentage of N₂O production per kilogram of N removed was at BC-1 in 2015 (9.3%); all other site years were below 4%. Furthermore, comparison studies did not include indirect or dissolved emissions. Surface N₂O production per kilogram of N removed in our study ranged from 2.6 to 7.2%, similar to production from constructed wetlands and woodchip bioreactors (Christianson et al., 2013; Groh et al., 2015). We encourage future studies to include both surface and dissolved N₂O emissions for more accurate estimates.

Conclusions

Soil surface N₂O emissions from fertilized corn were an order of magnitude greater than soil surface N₂O emissions from SRBs and traditional buffers. Saturated riparian buffer N₂O emissions from the soil surface were significantly greater (P = 0.045) than traditional buffers in only one site-year (Fig. 3). Reduction in indirect emissions from water diverted into SRBs resulted in a significant (P = 0.02) reduction in total indirect emissions. Total N₂O loads from direct and indirect sources were greatest from corn–soybean rotations (Fig. 4), and total N₂O loads from SRBs were not significantly greater (P = 0.37) than traditional buffers (Table 3). Our data suggest that installing a SRB into an established traditional buffer will not increase N₂O emissions. Furthermore, replacing cultivated riparian areas with a SRB could reduce N₂O emissions, simultaneously reducing losses of NOₓ to surface water and N₂O to the atmosphere.

Acknowledgments

This project was supported by the Agriculture and Food Research Initiative (AFRI) Competitive Grant no. 2013-67019-21384 from the USDA National Institute of Food and Agriculture and the Iowa Nutrient Research Center. The authors thank the USDA-ARS National Laboratory for Agriculture and the Environment, Ames, IA, staff for their assistance.

References


