



Cultivating *Salix Viminalis* in Agricultural-Riparian Transition Areas to Mitigate Agriculturally Derived N₂O Emissions from Potato Cropping Systems on Prince Edward Island

Holly D. M. Wilts · David L. Burton ·
Aitazaz A. Farooque

Received: 5 April 2022 / Accepted: 2 November 2022 / Published online: 21 November 2022
© The Author(s) 2022

Abstract Cultivating shrub willow (*Salix viminalis*) in agricultural-riparian transition areas has been proposed as a strategy for mitigating elevated riparian nitrous oxide (N₂O) emissions in agricultural regions. Nitrogen-based fertilizers are water soluble, enter riparian areas through surface runoff and subsurface lateral flow, and are converted to N₂O by incomplete anaerobic denitrification. *Salix* buffer strips can intercept and recycle fertilizer nitrate (NO₃⁻) into their biomass and/or promote complete denitrification, reducing N₂O emissions. We investigated the impact of *Salix viminalis* buffers on N₂O emissions relative to grassed buffers and upslope cultivated fields in potato rotations at 5 research sites across Prince Edward Island (PEI), Canada. Greenhouse gas (N₂O, CO₂, CH₄) flux at the soil-atmosphere interface was measured using non-steady-state static chambers in

2018 and 2019. NO₃⁻ exposure, soil temperature, and soil moisture content were quantified. Agricultural-riparian *Salix* significantly reduced N₂O emissions even when high NO₃⁻ inputs occurred and following precipitation events. Mean cumulative seasonal reductions of 1.32 kg N₂O–N ha⁻¹ (–0.02 to 6.16 kg N₂O–N ha⁻¹) were observed in *Salix* relative to cultivated fields; however, they were not significantly different than grass. The mean cumulative average global warming potential of *Salix* was 613 kg CO₂e ha⁻¹ lower than cultivated fields, with reductions of up to 2918 kg CO₂e ha⁻¹. Differences in N₂O flux between vegetation types were the greatest influencing factor. No hot moments of N₂O emission were observed in *Salix* following high rainfall events, which coincided with up to 95% decreases in N₂O emissions in *Salix* relative to cultivated fields.

Supplementary Information The online version contains supplementary material available at <https://doi.org/10.1007/s11270-022-05945-7>.

H. D. M. Wilts
Faculty of Science, University of Prince Edward Island,
Charlottetown, PE, Canada

D. L. Burton (✉)
Department of Plant, Food, and Environmental Sciences,
Dalhousie University, Truro, NS, Canada
e-mail: dburton@dal.ca

A. A. Farooque
Department of Sustainable Design Engineering, University
of Prince Edward Island, Charlottetown, PE, Canada

Keywords *Salix* · Shrub willow · N₂O · Nitrous oxide · Greenhouse gas emissions

1 Introduction

Riparian zones are vegetated areas between terrestrial and aquatic environments that play an important role in protecting and improving surface water quality in rivers and streams through pollution attenuation (Audet et al., 2013; Borjesson, 1999; Henault-Ethier et al., 2017; Jacinthe & Vidon, 2017; Schultz et al., 1995; Volk et al., 2006; Young & Briggs, 2005).

This includes the retention of agriculturally derived nutrients, agro-chemicals, and sediments as well as removal through processes such as denitrification. As such, management of riparian areas is considered a beneficial management practice (BMP) for water quality protection (Schultz et al., 1995) and they are increasingly mandated in many regions, including Prince Edward Island (PEI), Canada.

Riparian areas are frequently sites of elevated greenhouse gas (GHG) emission (Audet et al., 2013) within the landscape—a direct result of wet conditions and large amounts of slowly decomposing organic matter in riparian soils—and particular concerns exist in agricultural regions. While riparian areas are effective at reducing shallow groundwater nitrate (NO_3^-) concentrations and preventing it from entering waterways (Young & Briggs, 2005), incomplete denitrification can result in elevated nitrous oxide (N_2O) emissions (Audet et al., 2013; Drewer et al., 2012; Hellebrand et al., 2008; Jacinthe & Vidon, 2017; Lutes et al., 2016). This is concerning, since N_2O is a potent greenhouse gas with a global warming potential 298 times that of carbon dioxide (IPCC, 2013). Wet, anaerobic conditions, often prevalent in riparian soils, promote denitrification as a major pathway by which groundwater NO_3^- levels are reduced. Incomplete denitrification results in the release of elevated levels of N_2O into the atmosphere (Audet et al., 2013; Fisher et al., 2014; Jacinthe & Vidon, 2017). Thus, there is concern that efforts to protect riparian water quality may be at the expense of N_2O production when NO_3^- is present.

Agriculture was responsible for approximately 8.2% of anthropocentric GHG emissions in Canada in 2020 and 75% of national N_2O emissions (Environment and Climate Change Canada, 2022). Nitrogen fertilizer application, which has increased by 89% since 2005, was responsible for 21% of total agricultural emissions in 2020 and, since 2005, annual GHG emissions from Canadian agricultural soils have increased by 8 Mt CO_2 eq (Environment and Climate Change Canada, 2022). Nitrogen is an essential plant macronutrient that is frequently limiting to plant growth since most soils contain insufficient plant-available N (Havlin et al., 2013; Zebarth & Rosen, 2007). Fertilization of agricultural soils with synthetic N that is transformed to NO_3^- by the soil biota has contributed to dramatically increased crop yields; however, this has been accompanied by high losses of

NO_3^- to the environment via leaching and denitrification (Erisman et al., 2008). In humid and sub-humid climates such as PEI, soil N_2O emissions in agricultural systems occur mainly due to microbial denitrification (Aulakh et al., 1984; Gauder et al., 2012; Granli & Beckman, 1994; Kavdir et al., 2008). Fertilizer-derived NO_3^- serves as an alternate electron acceptor in microbial respiration under conditions of limited oxygen supply and is a major contributor of agricultural N_2O emissions (Hellebrand et al., 2008). Accumulation of NO_3^- in the soil also inhibits the reduction of N_2O to N_2 , increasing the ratio of $\text{N}_2\text{O}/\text{N}_2$ resulting from denitrification (Cho et al., 1997a, b; Gillam et al., 2008; Weier et al., 1993; Zebarth et al., 2012). Since water-filled pore space (WFPS) and NO_3^- availability are major controllers of N_2O emissions (Gillam et al., 2008; Ruser et al., 2001), and NO_3^- is highly mobile within the soil (Havlin et al., 2013), leakage of fertilizer-derived NO_3^- into wetter riparian areas can increase the likelihood of it entering the denitrification pathway and situations favoring incomplete denitrification, resulting in the release of N_2O . This explains the observance of high N_2O emissions in riparian areas in some agricultural regions.

PEI, Canada, is the province with the largest potato production area in Canada (33,720 ha in 2016). This represents 20.8% of total PEI cropland and 24.2% of total cropland dedicated to potatoes in Canada (Statistics Canada, 2017). Potatoes require high fertilizer N inputs to produce the tuber yield and size profile desired by the processing industry, in part due to their shallow root systems and corresponding low N use efficiency (Zebarth & Rosen, 2007). Fertilizer rates for potatoes in Atlantic Canada can exceed 200 kg N ha^{-1} , with up to 55% of this being lost to the environment (Belanger et al., 2001) and leaching losses of NO_3^- from potato fields have been observed to sometimes exceed 171 kg N ha^{-1} (Zebarth & Rosen, 2007). NO_3^- leaching from potato fields and contamination of both groundwater and surface waterways are known to be problematic on PEI (Jiang et al., 2015; Liang et al., 2019; Zebarth et al., 2015). Management requirements of potatoes, such as the high degree of soil disturbance, as well as soil and climatic conditions specific to PEI (highly permeable sandy surface soils with low natural fertility and organic matter contents), contribute to high fertilization requirements. High annual rainfall, particularly in the non-growing season, also increases

the potential for NO_3^- leaching losses (Zebarth et al., 2015). As a result, the risk of elevated riparian N_2O emissions on PEI is great.

Cultivating *Salix* spp. (*Salix*) bioenergy buffer strips in riparian areas or downslope field edges in agricultural-riparian transition zones has been proposed as a potential beneficial management practice (BMP) for protecting surface water quality while simultaneously reducing agriculturally derived N_2O emissions (Borjesson, 1999). It is thought that by intercepting and recycling NO_3^- from upland fertilizer application as it is lost through leaching or surface runoff, downslope riparian *Salix* buffer strips will remove NO_3^- via plant uptake for biomass accumulation and by complete denitrification, thereby reducing the amount at risk of incomplete denitrification and N_2O loss in riparian areas. Bioenergy *Salix* are managed for maximum nutrient uptake and biomass accumulation through regular coppicing on a 3-year cycle (Amichev et al., 2014), which can maximize NO_3^- uptake and consequent N_2O reductions. *Salix* have long growing seasons, rapid growth, extensive root systems, continuous soil cover, and a 25-year lifespan, which make them good candidates for limiting soil NO_3^- accumulation by maximizing and sustaining plant uptake (Borjesson, 1999; Bressler et al., 2017; Dimitriou et al., 2012; Kavdir et al., 2008).

The effectiveness of *Salix* at reducing ground-water NO_3^- levels is well documented, even when large inputs are occurring (Aronsson et al., 2000; Borjesson, 1999; Bressler et al., 2017; Dimitriou et al., 2012; Ferrarini et al., 2017; Heller et al., 2003; Young & Briggs, 2005), and *Salix* biomass production is directly enhanced by soil NO_3^- supply, indicating that plant uptake is a major removal pathway (Volk et al., 2006). Harvested biomass N removal in *Salix* averages around $25 \text{ kg N ha}^{-1} \text{ year}^{-1}$ (Amichev et al., 2014). Borjesson (1999) estimated that *Salix* buffer strips planted between annual cropped fields and streams could prevent $70 \text{ kg N ha}^{-1} \text{ year}^{-1}$ from entering waterways when nitrate inputs were greater than $15 \text{ kg N ha}^{-1} \text{ year}^{-1}$; 2/3 of this was due to plant uptake with the remaining 1/3 lost to denitrification (Borjesson, 1999).

The impact of *Salix* on GHG emissions at the soil-atmosphere interface is less well studied, in part because the spatial and temporal distribution of GHG emissions varies greatly across the

landscape (Audet et al., 2013; Bressler et al., 2017; Lutes et al., 2016; Pacaldo et al., 2014a, b; Whitaker et al., 2018). Soil GHG emissions are affected by factors such as soil texture, hydrology, land use and management, climate, soil microbes, and vegetation (Borjesson, 1999; Hellebrand et al., 2008; Lutes et al., 2016). Most studies have found that *Salix* reduce N_2O and increase CO_2 emissions at the soil-atmosphere interface compared to other vegetation and/or land use types, while their impact on CH_4 is mixed (Borjesson, 1999; Bressler et al., 2017; Drewer et al., 2012; Gauder et al., 2012; Kavdir et al., 2008; Lutes et al., 2016; Pacaldo et al., 2014b; Smialek et al., 2006; Whitaker et al., 2018). However, most studies have focused on *Salix* bioenergy crops grown as a field scale replacement for other crops. N_2O emissions at the soil-atmosphere interface from *Salix* have been found to vary greatly (Palmer et al., 2014; Whitaker et al., 2018). To date there have been very few studies related to the impact of *Salix* on shallow soil GHG emissions when grown as buffer strips in riparian areas, and none specifically on their impact in riparian areas downslope of potato cropping systems on PEI. Since wet conditions and large quantities of decomposing organic matter in riparian soils also promote elevated CO_2 and CH_4 emissions, while these gases are not the primary focus of this study, there is a need to measure the impact of *Salix* on these other GHG gasses in riparian settings to allow for a more complete GHG accounting.

Previous field studies conducted as part of the Bioeconomy Crop Initiative with the PEI Department of Agriculture on PEI found that *Salix viminalis* was effective at carbon and nutrient sequestration on field edges bordering riparian zones on PEI, producing higher aboveground biomass yield with greater N concentration than other *Salix* varieties studied (Gooijer et al., 2011; Government of PEI, 2010). Coppiced *Salix viminalis* produced annual aboveground biomass yields of up to $23.4 \text{ tDM ha}^{-1} \text{ year}^{-1}$ and accumulated N at a rate of $160 \text{ kg ha}^{-1} \text{ year}^{-1}$ when both above and belowground biomasses were considered (Pharo, 2012, as cited in Lantz et al., 2014; Schroeder, 2019; Wright, 2012, as cited in Lantz et al., 2014). For that reason, *Salix viminalis* was selected for this GHG study. Since riparian areas within 15 m of waterways are protected on PEI, this study occurred on downslope field edges in agricultural-riparian

transition zones, just outside the protected 15-m riparian buffer, as a proxy for the riparian area.

The goal of this study was to determine the effect of *Salix viminalis* buffers on GHG emissions at the soil-atmosphere interface in agricultural-riparian transition zones downslope of potato cropping systems on PEI. The main objective was to compare soil N₂O, CH₄, and CO₂ emissions between agricultural-riparian *Salix viminalis* buffers, agricultural-riparian grass buffers, and upslope agricultural fields in 3-year potato rotations. This included quantifying the effects of increased distance from the agricultural field and wetting events on buffer N₂O emissions. A secondary objective was to measure ancillary environmental parameters such as soil moisture, temperature, and NO₃⁻ availability to help identify controllers of N₂O emissions. Our hypothesis was that *Salix viminalis* buffers would reduce N₂O emissions and elevate CO₂ and CH₄ emissions relative to cultivated fields in potato rotation and grass buffers. Further, we hypothesized the main drivers of N₂O flux would be soil NO₃⁻ availability and soil moisture content.

2 Materials and Methods

2.1 Description of the Study Sites

The experiment was conducted in 2018 and 2019 on Prince Edward Island (PEI), Canada (lat. 46°25'N, long. 63°00'W) at five study sites distributed across the island in agricultural-riparian transition zones and their upslope agricultural fields. PEI is located in the Gulf of St. Lawrence and has a humid-continental climate with a mean annual temperature of 5.7 °C and annual rainfall of 887 mm (Government of Canada, 2019a). Total precipitation over the 2018 and 2019 study intervals from early May to mid-November was 815 mm and 835 mm, with average spring (May/June), summer (July/August), and fall (September/mid-November) rainfalls of 275, 109, and 431 mm in 2018 and 233, 146, and 456 mm in 2019 (Government of Canada, 2019b). Spring and fall precipitations were higher than average, while summer precipitation was below average in both years. Spring air temperatures were lower than average (10.8 and 11.2 °C), while summer (19.7 and 19.5 °C) and fall were above average in both study years (9.9 and 10.3 °C).

Across all sites, mean soil temperature was 15.7 °C, with higher recorded temperatures in the cultivated fields (mean 16.3 °C) than the *Salix* (mean 15.5 °C) and grass (mean 15.6 °C) buffers. Mean soil moisture content was 28.4% with higher overall values recorded in the *Salix* (mean 30.2%) and grass buffers (mean 32.2%) than the cultivated fields (mean 20.7%).

PEI surface soils tend to be coarse textured (sandy-loam), well drained, acidic, low in organic matter with low natural fertility. Soil types at the study sites were mapped to Charlottetown or Aldberry Series with orthic humo-ferric podzols in the Canadian System of Classification being the dominant subgroup, moderately coarse to coarse texture on gently undulating to rolling relief (2–15%). Soils are mapped as well to moderately well drained, although the water table varied from deep to near the soil surface for part of the growing season at the various sites (MacDougall et al., 1988). Underlying this, at a depth of approximately 1–15 m, is a less permeable basal till layer that can result in a perched water table during periods of high precipitation (Zebarth et al., 2015). The bedrock consists of fractured sandstone. All sites were located adjacent to fields cultivated to potato crop rotations.

The experiment used a randomized complete block design with three treatments: *Salix* buffer (with 2 location sub-treatments: row 1 and row 3), grass buffer, and upslope cultivated agricultural field. Treatments were replicated three times at each of the five experimental sites. The study sites were established in 2016 (site A and site B), 2017 (site E and site C), and 2018 (site D); therefore, this experiment occurred during years 3 and 4, 2 and 3, and 2 post-establishment (online resource 1, Table I). Each site consisted of 10-m-wide *Salix viminalis* and grass buffer plots planted on the agricultural field edge adjacent to the riparian zone. *Salix viminalis* buffers were planted in 4 rows with 2-m row spacing and 0.5-m or 0.75-m in-row plant spacing.

This study occurred prior to any coppicing activity. The grass buffer and grass between the *Salix* rows were managed by mowing approximately biweekly; however, no fertilizer or irrigation was applied. The agricultural field was managed by the producer as per normal cropping practices for the crop being grown as part of a 3-year rotation including 1 year of potatoes (online resource 1, Table II).

2.2 Monitoring of Gas Fluxes

Soil gas fluxes were measured approximately biweekly using the static chamber method as described in Burton et al. (2008) and Burton (2013) from May to November in 2018 and 2019 in order to evaluate treatment effects on GHG emissions. Site A underwent weekly measurements of soil gas fluxes to better evaluate temporal variation in GHG emissions and was selected based on its location which was favorable for more frequent sampling. Measurements captured the entire *Salix* growth season, from budding in the spring to leaf drop in the fall, over 2 calendar years. Each non-steady-state static chamber consisted of a collar made of polyvinyl chloride (PVC) pipe (diameter 20.3 cm; height 10 cm) with a removable PVC chamber (diameter 20.3 cm; height 15 cm) that fitted over each collar. Chambers were covered with reflective foil-faced insulating bubble wrap, had closed cell foam gaskets consisting of closed-cell foam weather-stripping tape installed around the bottom to ensure an airtight seal with the collar, and contained a butyl rubber suba seal sampling port and plastic air vent tube (diameter 0.4 cm; length 7.5 cm). Total volume of the chamber was 5.0 L, while volume of the protruding collar was approximately 2.0 L but varied throughout the season, covering a soil area of 315 cm².

Twelve collars were installed at each study site with the following distribution: three in the upslope agricultural field perpendicular to the buffer area, three in the grass buffers distributed to reflect the width, three in row 1 of the *Salix* buffer, and three in row 3 of the *Salix* buffer. Additionally, in 2019, three extra collars were installed at each study site between the *Salix* buffers and the agricultural field. Collars were installed in early spring, a minimum of 24 h before initial GHG flux measurements were taken and remained permanently installed at the sites throughout the season. Collars that were damaged or otherwise required replacement throughout the season were also installed a minimum of 24 h before GHG flux measurements were obtained. Field collars were removed during planting and harvest and subsequently replaced. Collars in fields in the potato crop phase of the rotation were placed in the ridge row location. Collars were sunk into the ground with roughly 5 cm protruding and collar

height was measured on a monthly basis in order to calculate individual collar headspace volume.

Twenty-milliliter disposable nylon syringes (Becton–Dickinson 14–823-2B) fitted with 25 gauge, 5/8" length Luer-Lok tip needles (Becton–Dickinson 14-826AA) were used to obtain 20-mL air samples from the headspace of each chamber at 0, 10, 20, and 30 min after deployment. Samples were subsequently injected and stored in 12-mL pre-evacuated exetainers (Labco Limited 739 W) fitted with butyl rubber septa. Air samples were stored in a cool dark location until they were shipped to the Greenhouse Gas Lab (Dalhousie University, Faculty of Agriculture) in Truro N.S. for analysis.

Gas samples were analyzed for N₂O, CO₂, and CH₄ using a Varian Star 3800 Gas Chromatograph (Varian, Walnut Creek, CA) fitted with an electron capture detector (N₂O), thermal conductivity detector (CO₂), and a flame ionization detector (CH₄). Samples were transferred from the exetainers to the GC using a Combi-PAL Autosampler (CTC Analytics, Zwingen, Switzerland). As described in Snowdon et al. (2013), the electron capture detector was operated at 350 °C, 90% Ar, 10% CH₄ carrier gas at 20 mL min⁻¹, Haysep N 80/100 pre-column (0.32 cm diameter × 50 cm length), and Porapak QS 80/100 mesh analytical columns (0.32 cm diameter × 200 cm length) in a column oven operated at 80 °C. The pre-column was used in combination with a four-port valve to remove water from samples. The thermal conductivity and flame ionization detectors were operated in series at 130 °C and 200 °C, respectively, pre-purified helium (He) carrier gas at 20 psi, Haysep N 80/100 mesh (0.32 cm diameter × 50 cm length) pre-column, followed by a Porapak QS 80/100 mesh (0.32 cm diameter × 200 cm length) analytical column maintained at 80 °C.

Individual gas fluxes (F_c) for N₂O, CO₂, and CH₄ (g N₂O–N ha⁻¹ day⁻¹, kg CO₂–C ha⁻¹ day⁻¹, and g CH₄–C ha⁻¹ day⁻¹) were calculated from changes in each gas concentration within the static chamber according to Hutchinson and Livingston (1993):

$$F_c = (dC/dt)((V_c * M_{mol})/(A * V_{mol}))$$

with, as described by Snowdon et al. (2013), dC/dt being the rate of change in N₂O, CO₂, or CH₄ concentration (mol mol⁻¹ h⁻¹), A being the surface area (m²) of the collar, V_c being the total volume (L) of the

enclosure (chamber volume plus collar volume), M_{mol} being the molar mass of N_2O -N, CO_2 -C, or CH_4 -C (g mol^{-1}), and V_{mol} being the volume of 1 mol of N_2O , CO_2 , or CH_4 (L mol^{-1}) inside the chamber corrected for temperature using the ideal gas law. The rate of change in concentration in the headspace (dC/dt) was calculated using a simple linear regression of the gas concentrations versus time over the deployment period and all flux values were expressed as $\text{g N}_2\text{O-N ha}^{-1} \text{ day}^{-1}$, $\text{kg CO}_2\text{-C ha}^{-1} \text{ day}^{-1}$, and $\text{g CH}_4\text{-C ha}^{-1} \text{ day}^{-1}$. Positive F_c values indicated the soil was emitting the gas, while negative F_c values indicated the soil was acting as a sink for the gas. Cumulative growing season gas fluxes were calculated for each individual chamber by linear interpolation between sampling dates (Burton et al., 2008). This assumed that gas fluxes measured represented average daily flux and that growing season emissions are a continuous function of daily flux estimates. Cumulative chamber emission values were averaged across chambers within a treatment to determine mean cumulative emission values for each treatment and sub-treatment. N_2O and CH_4 were converted to CO_2 equivalents (CO_2e) using conversion factors of 298 and 25 to determine the global warming potential (GWP) of greenhouse gas emissions at the soil-atmosphere interface. CO_2 was not included in the GWP calculations since CO_2 contributions to global warming potential in soils where active photosynthesis is occurring are assessed as a change in carbon stocks rather attempting to measure both soil CO_2 emissions and photosynthesis. Changes in C stocks for *Salix* growth in these plots is dealt with in a subsequent paper.

2.3 Soil and Atmospheric Characteristics

Each sampling event included measurements next to each collar of soil surface temperature (depth 6 cm) using a portable soil temperature probe (Cole-Parmer 90,090–06) and surface moisture content (depth 6 cm) using a HH2 moisture meter and attached ML3 ThetaProbe Soil Moisture Sensor (Delta-T devices, Cambridge England). Air temperature ($^{\circ}\text{C}$) and humidity (%) were also recorded within 10 cm of each collar using an Amprobe Digital Thermo-hygrometer (model TH-2).

Nitrate exposure (NE) was calculated by obtaining measurements of shallow soil NO_3^- acquired

by inserting a single Anion Plant Root Simulator (PRS) probe (Western Ag Innovations, Saskatoon) vertically into the soil within 12 in of each collar. Each anion PRS probe was buried to a depth of 6 cm and had an absorbing surface area of 17.5 cm^2 . Probes were exchanged approximately biweekly over the field season to obtain a continuous assessment of nutrient supply rate, with the three replications being combined for each treatment (field, grass, *Salix* row 1, *Salix* row 3) for each of the sites and deployments, and sent to Western Ag Innovations for analysis of NO_3^- , determined colorimetrically by automated flow injection analysis system. Each determination results in a flux or NO_3^- supply rate to the surface of the probe ($\mu\text{g nutrient}/10 \text{ cm}^2 \text{ ion-exchange membrane surface area/time of burial}$). Nitrate exposure (NE) was calculated as the sum of sequential burials at a location over the monitoring period and expressed as $\mu\text{g nutrient}/10 \text{ cm}^2 \text{ ion-exchange membrane surface area}$.

Daily precipitation data for 2018 and 2019 was retrieved from the New Glasgow PEI weather station, the nearest Government of Canada maintained weather station to site A, located 16 km from the site (Government of Canada, 2019b).

2.4 Data Analysis

Statistical analysis was performed using JMP software for Windows version 14 (JMP®, Version 14. SAS Institute Inc., Cary, NC, 1989–2019). Descriptive statistics, one-way analysis of variance (ANOVA), and regression models were used to analyze the data. A one-way ANOVA was done to assess the effect of treatment (*Salix* buffer, grass buffer, cultivated field) on soil N_2O , CO_2 , and CH_4 fluxes. Where ANOVA demonstrated significant treatment effects, the Student *t* test was used to compare mean GHG emissions between treatments. Statistical significance was determined at $p > 0.05$. Linear regression was used to determine the relationship between soil GHG emissions and environmental parameters. Linear regression was used to determine the relationship between NE and cumulative N_2O emissions over the monitoring period. Descriptive statistics were used to qualitatively observe differences between crop type on GHG emissions.

3 Results

Soils at all research sites were net sources of N_2O over the monitoring period with observed daily fluxes ranging from -10.2 g to 400 g $\text{N}_2\text{O-N ha}^{-1} \text{ day}^{-1}$ (mean 5.65 g $\text{N}_2\text{O-N ha}^{-1} \text{ day}^{-1}$). Observed N_2O flux was often low, with very few emissions occurring below 12°C . Greatest N_2O emissions occurred in the cultivated fields, particularly those cultivated to potatoes, sudangrass, and black peas, with most emission events occurring above 12°C . There was a positive relationship between soil moisture content and N_2O emissions, which was most pronounced in fields cropped to potato (r^2 : 0.122; $p=0.0001$), which also had the highest N_2O emissions (Fig. 1a). Available soil NO_3^- , as measured by NE, explained the greatest amount of the variation in N_2O emissions (r^2 : 0.582; $p<0.0001$) (Fig. 1b).

Cumulative N_2O flux exhibited considerable between-site and within-site variability. Cumulative seasonal emissions within treatments across all sites and years ranged from -0.039 to 6.31 kg $\text{N}_2\text{O-N ha}^{-1}$ (mean 0.46 kg $\text{N}_2\text{O-N ha}^{-1}$) with higher emissions occurring in the cultivated agricultural fields relative to the grass and Salix buffers (Fig. 2(a)).

In seven of nine site years, there was a significant effect of treatment on N_2O flux, with emissions in the cultivated fields being significantly higher than the Salix or grass buffers ($p=0.0001$), but no significant difference between the two buffer treatments. Mean

cumulative seasonal reductions in N_2O emissions of 1.3 kg $\text{N}_2\text{O-N ha}^{-1}$ (with maximum reductions of 6.2 kg $\text{N}_2\text{O-N ha}^{-1}$ observed at site A in 2019) were observed in the Salix buffers relative to the cultivated fields, with mean cumulative seasonal reductions in N_2O emissions of 1.4 kg $\text{N}_2\text{O-N ha}^{-1}$ in the grass relative to the cultivated fields (with maximum reductions of 6.1 kg $\text{N}_2\text{O-N ha}^{-1}$ observed at site A 2019).

There was also a site impact on N_2O emissions (Fig. 2(b)). Between sites, crop type impacted N_2O emissions, with considerable variation between crop types: N_2O flux was lower in the Salix buffers than the cultivated fields planted to potatoes, sudangrass, and black peas, but not than the fields planted to clover (Fig. 3). N_2O fluxes in the Salix buffers were similar to fields planted to hay, except at site E in 2018, when N_2O flux in the Salix was higher than the hay. The cultivated fields planted to potatoes, sudangrass, and black peas were the greatest sources of N_2O (Fig. 3). Over all sites, mean cumulative seasonal N_2O emissions were 1.3 kg N ha^{-1} lower in the Salix buffers than the adjoining cultivated field upslope. When considering only sites with cultivate fields planted to potatoes, sudangrass, or black peas (crops receiving 30 kg N ha^{-1} or more), mean cumulative seasonal N_2O emissions were 2.1 kg $\text{N}_2\text{O-N ha}^{-1}$ lower in the Salix buffers than the upslope cultivated field.

CO_2 flux was positively related to soil temperature (r^2 : 0.209, $p=0.0001$), with higher emissions occurring in the warmer months of July and August

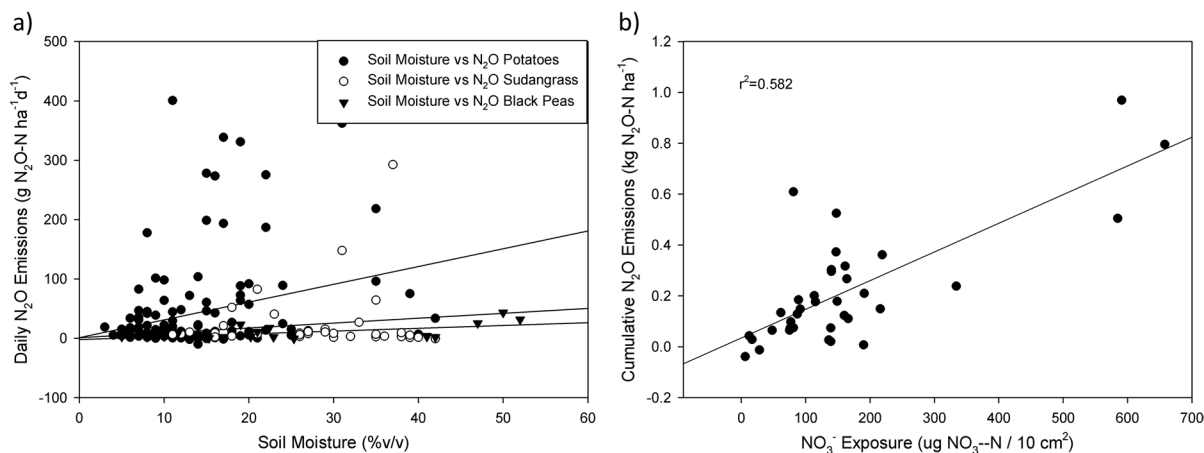


Fig. 1 Relationship between **a** soil moisture (% v/v) and daily N_2O flux (kg $\text{N}_2\text{O-N ha}^{-1} \text{ day}^{-1}$) and **b** NO_3^- exposure (ug $\text{NO}_3^- \text{--N} / 10 \text{ cm}^2$) and cumulative N_2O flux (kg $\text{N}_2\text{O-N ha}^{-1}$)

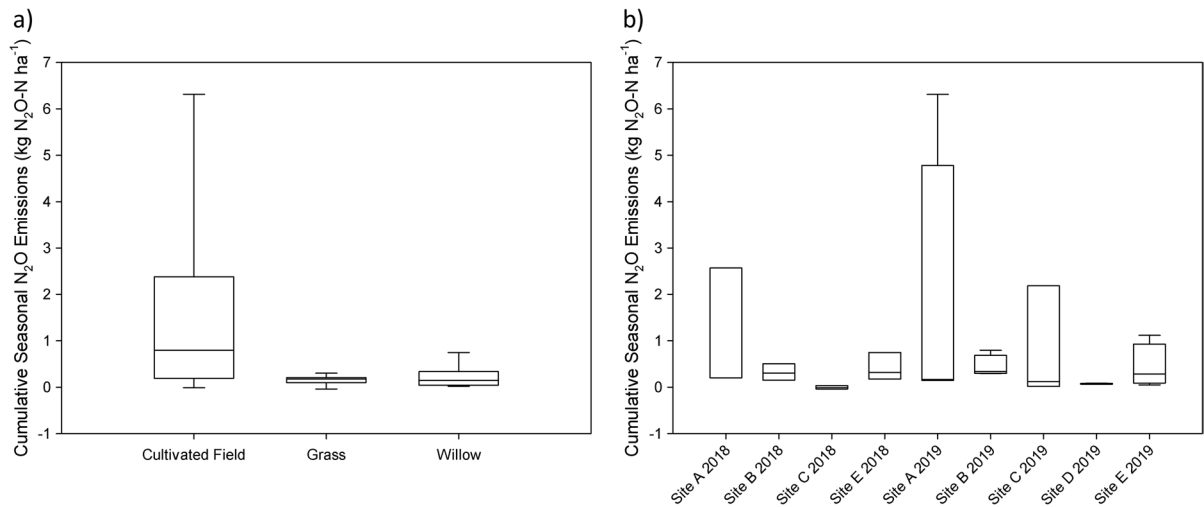


Fig. 2 Median, 25th quartile, 75th quartile, maximum, and minimum cumulative seasonal N_2O emissions for each **a** treatment at the 5 research sites over 2 years and **b** research site in each year of sampling

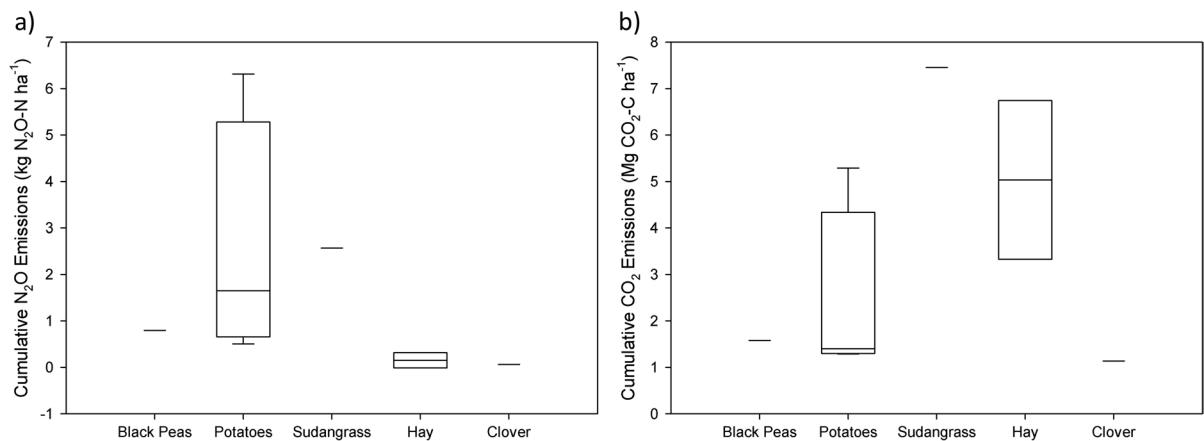


Fig. 3 Median, 25th quartile, 75th quartile, maximum, and minimum cumulative field **a** N_2O and **b** CO_2 emissions for each crop type

(online resource 1, Fig. 1). No relationship was found between soil moisture content and CO_2 flux. Cumulative CO_2 emissions within the treatments ranged from 1.13 to 12.8 $Mg\ CO_2-C\ ha^{-1}$ (mean 4.40 $Mg\ CO_2-C\ ha^{-1}$ across all treatments, sites, and years), with significantly ($p=0.0001$) higher emissions occurring in the grass and Salix buffers (4.8 and 5.5 $Mg\ CO_2-C\ ha^{-1}$) than the in cultivated fields (3.3 $Mg\ CO_2-C\ ha^{-1}$). Within the buffer treatments themselves, CO_2 emissions were significantly greater in the Salix treatments than the grass treatments ($p=0.0271$),

although the effect was less than the difference between either buffer treatment or the cultivated field ($p=0.0001$) (Fig. 4(a)). Measured gas fluxes in the cultivated fields revealed that fields planted to sorghum sudangrass, hay, and potatoes had higher CO_2 flux than black peas and clover (Fig. 3(b)). When broken down by crop type within the potato rotation, CO_2 emissions in the Salix buffers were greater than those in the black peas (4.4 $Mg\ CO_2-C\ ha^{-1}$ Salix vs. 1.6 $Mg\ CO_2-C\ ha^{-1}$ black peas), clover (2.1 $Mg\ CO_2-C\ ha^{-1}$ Salix vs. 1.1 $Mg\ CO_2-C\ ha^{-1}$

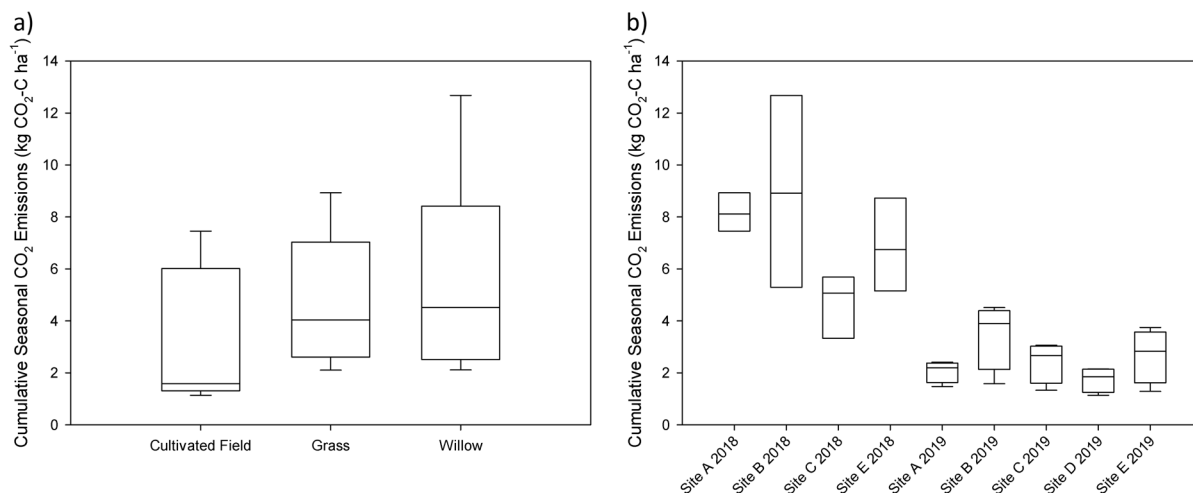


Fig. 4 Median, 25th quartile, 75th quartile, maximum, and minimum cumulative seasonal CO₂ emissions for each **a** treatment at the 5 research sites over 2 years and **b** research site in each year of sampling

clover), hay (7.2 Mg CO₂-C ha⁻¹ Salix vs. 5.0 Mg CO₂-C ha⁻¹ hay), and potatoes (5.2 Mg CO₂-C ha⁻¹ Salix vs. 2.3 Mg CO₂-C ha⁻¹ potatoes), but similar to those from sudangrass (8.1 Mg CO₂-C ha⁻¹ Salix vs. 7.5 Mg CO₂-C ha⁻¹ sudangrass). Measured gas fluxes also exhibited considerable between-site variability (Fig. 4(b)). ANOVA revealed a significant effect of site on measured CO₂ flux, with emissions at site B being significantly higher than all other sites (in both 2018 and 2019; $p=0.0001$ to $p=0.0015$) and emissions at site D being significantly lower than all other sites ($p=0.0025$ to $p=0.0246$). 2019 emissions at both site B and site D were higher than 2018 emissions, which could be explained by higher precipitation in 2019. Additionally, site A had significantly higher emissions than site E in 2018 ($p=0.0495$), while site E and site C had significantly higher emissions than site D in 2019 ($p=0.0002$; $p=0.0025$), and site E had significantly higher emissions than site A in 2019 ($p=0.0151$). However, relative to the crop types, CO₂ emissions in the Salix buffers were significantly higher than those in the black peas ($p=0.0031$), clover ($p=0.0108$), hay ($p=0.0001$), and the potatoes ($p=0.0001$), but not significantly different from the sudangrass.

While CH₄ emissions were detected at all sites, they were extremely low with no obvious trends or drivers. Methane fluxes ranged from -142 to 174 g CH₄-C ha⁻¹ day⁻¹ with mean emissions of -1.2 g

CH₄-C ha⁻¹ day⁻¹ across all sites and years. Soils at all research sites and years were slight net CH₄ sinks except site E and site C in 2019 which were slight net sources.

Nitrate exposure (NE), the cumulative seasonal soil NO₃⁻ flux, averaged 340 µg/10 cm² across all sites and years, with higher NE observed in the cultivated field (mean 948 µg/10 cm²) than the Salix and grass buffers (151 and 109 µg/10 cm², respectively) (Table 1). The highest observed NE was 2909 µg/10 cm², which occurred in the cultivated field at site A in 2019.

Cumulative N₂O flux and NE were not observed to change across the Salix buffer zone. The mean NE

Table 1 Nitrate exposure (µg/10 cm²) by treatment

Site	Year	Salix row 1 (R1)	Salix row 3 (R3)	Grass	Cultivated field
A	2018	61	164	114	204
	2019	149	166	89	2909
B	2018	147	334	92	585
	2019	81	87	140	658
C	2018	17	12	6	28
	2019	190	136	160	1967
D	2018	81	77	76	75
	2019	81	77	76	75
E	2018	591	148	115	161
	2019	139	139	191	1949

across the Salix buffer zones ranged from 162 ug 10 cm² in row 1 to 140 ug 10 cm² in row 3; however, there was not a consistent decrease between sites. Similarly, while the mean cumulative N₂O emissions were greater in row 1 of the Salix buffer than row 3 (0.27 kg N₂O-N ha⁻¹ vs. 0.16 kg N₂O-N ha⁻¹) across all sites, this difference was not significant, and in some instances N₂O flux was trended to higher in row 3 than row 1 although there was no significant difference.

Site A underwent more frequent, weekly, monitoring of GHG emissions to evaluate temporal variations in GHG flux. At that site no hot moments of N₂O emission were observed in the Salix or grass buffers after wetting events. This is in contrast to the cultivated fields which demonstrated hot moments of N₂O emission during this same period. Over 2018 and 2019 there were four notable N₂O emission events occurring on August 21, 2018, June 26 and July 3, 2019, September 12, 2019, and October 10, 2019, all shortly after large precipitation events (Fig. 5). There were no corresponding high N₂O emission events observed in the Salix and grass buffers on these dates.

When the cumulative N₂O and CH₄ emissions for each site and treatment were converted to CO₂e, the overall GHG balances of the cultivated fields were significantly higher than the Salix and grass buffer treatments ($p=0.0076$ and $p=0.0052$), but there was no significant difference between the Salix and grass treatments (Fig. 6). Despite significantly

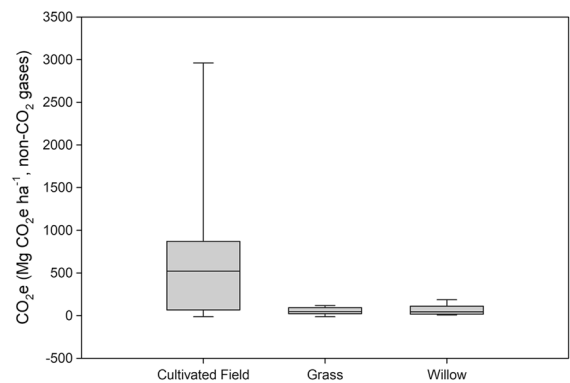


Fig. 6 Median, 25th quartile, 75th quartile, maximum, and minimum cumulative seasonal GHG emissions in CO₂ equivalents for each treatment at the 5 research sites over 2 years (Mg CO₂e ha⁻¹)

higher CO₂ emissions in the willows relative to the other treatments, even when CO₂ was included in the CO₂e calculation, the overall GHG balance of the Salix remained significantly lower than the cultivated fields. However, this fails to consider the increased photosynthesis in the Salix treatment. The overall GWP of the Salix buffers (64.8 Mg CO₂e/ha) was 9.5% of the cultivated fields (678 Mg CO₂e/ha), but 119% of the grass (54.6 Mg CO₂e/ha). While the mean average GWP of emissions at the soil-atmosphere interface in the Salix buffers was 409 Mg CO₂e/ha lower than the cultivated fields across all sites, reductions of up to 1849 Mg CO₂e/ha were observed (at site A in 2019).

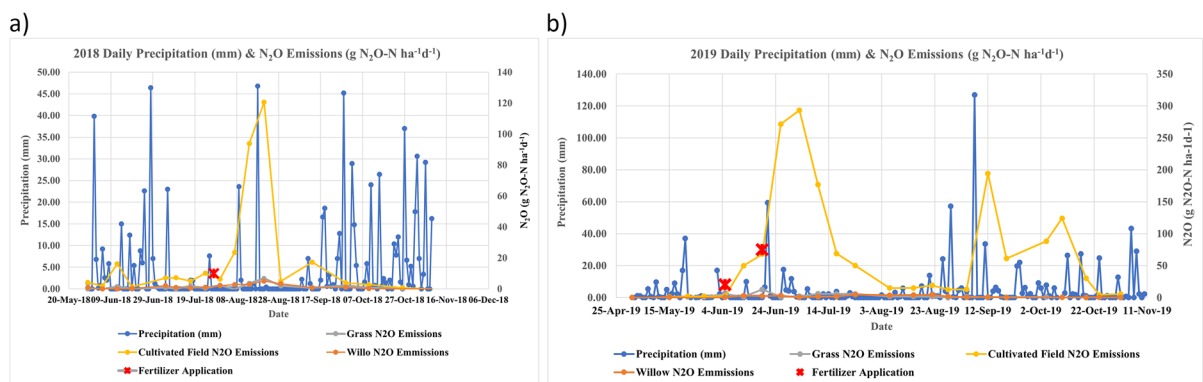


Fig. 5 Daily precipitation and N₂O emissions at the Emerald research site in **a** 2018 and **b** 2019. In August 2018 the previous year's wheat crop was plowed under and the field fertilized

and planted to sorghum sudangrass. In June 2019 the field was cultivated to potatoes and fertilized preplant and 4 weeks after hilling

4 Discussion

4.1 Nitrous Oxide

This study found that N_2O emissions in Salix and grass agricultural-riparian buffers were low, in general, and significantly lower than in upslope cultivated fields in potato rotation, which often had larger N_2O emissions. Both Salix and grass treatments were effective in reducing N_2O emissions relative to the cultivated field.

Denitrification is known to be influenced by soil temperature and water filled pore space (Smith et al., 2018), which was evident in our study by the correlations, albeit weak, between N_2O (a product of incomplete denitrification) and soil temperature and moisture content. However, soil NO_3^- can out compete with N_2O for terminal electron acceptors resulting in incomplete denitrification increasing N_2O emissions (Gauder et al., 2012; Gillam et al., 2008; Hellebrand et al., 2008). Thus, the accumulation of NO_3^- favors NO_3^- reduction preferentially to N_2O reduction, the final step of the denitrification reaction, even in the presence of the *nosZ* enzyme (Audet et al., 2013; Cho et al., 1997a, b; Smith et al., 2018). This results in N_2O accumulating in the soil and being released to the atmosphere. This relationship is demonstrated in our study by the relatively strong positive relationship observed between soil NE and cumulative N_2O emissions.

The magnitude of cumulative N_2O emission can be explained by differences in NE and soil moisture content, which resulted in N_2O emissions in the agricultural-riparian buffer treatments being significantly lower than the upslope cultivated fields. N_2O emissions in the cultivated fields were large following fertilizer application and when wetting events occurred, particularly in mid-season when the soils were warm and had high NO_3^- concentration. During these times, buffer N_2O emissions remained low. This was a result of low soil NO_3^- availability in the buffer zones, as indicated by differences in NE, and higher soil moisture content in the buffer zones which favor the reduction of N_2O to N_2 (complete denitrification). This points to the efficacy of Salix at absorbing fertilizer-derived NO_3^- transported from agricultural fields, accruing in the Salix biomass and/or creating an environment conducive to complete denitrification. In either case, this limits shallow soil

NO_3^- movement to the adjoining waterway. It is also possible that some of the NO_3^- from the cultivated fields leached to groundwater and/or was leached from the riparian zone before denitrification could occur, but this is unlikely as groundwater is discharging to the adjacent stream through much of the year. Riparian areas, and especially those in agricultural regions, are frequently observed to have elevated N_2O emissions relative to upslope fields, as a result of N fertilizers accumulating in riparian soils (Billen et al., 2020; Fisher et al., 2014; Gold et al., 2001; Jacinthe et al., 2015). This did not occur in our study, indicating that uptake or denitrification of agriculturally derived NO_3^- in cultivated Salix and grass buffer treatments managed to prevent the buildup of NO_3^- in the soil as indicated by lower NE. The one exception, where elevated N_2O emissions were observed in the Salix buffers relative to the cultivated field, occurred in 2018 at site E. At this site, N_2O flux in the Salix was significantly greater than the cultivated field (in hay) and grass buffer for the first few months of the season before declining to statistically similar levels to the grass and field. This difference is related to both the low N_2O emissions associated with cropped field and increased emissions associated with the relative immature Salix at this site in early 2018, which would impact its ability to take up soil NO_3^- . The site E Salix was planted in late July 2017; therefore, they were less than a year old at the time that data collection commenced for this study in May 2018. At all other sites, Salix were planted as early in the previous season as possible, so the shrubs were in their second (or third) year of growth when data collection for this study began. Previous studies have found that there is an initial delay of 1–2 years before the environmental effects of Salix on water quality and soil GHG emissions take effect (Aronsson et al., 2000; Palmer et al., 2014; Schultz et al., 1995; Smialek et al., 2006). This is thought to be due to immature developing root systems which have limited capacity for NO_3^- uptake (Aronsson et al., 2000). For example, Palmer et al. (2014) observed that willow increased N_2O emissions in the initial 2 years post-establishment, with observed emissions of 0.07 to 18.74 $\text{Mg CO}_2\text{eq ha}^{-1} \text{ year}^{-1}$ (0.23 to 62.8 $\text{kg N}_2\text{O ha}^{-1} \text{ year}^{-1}$). This, along with off-field leakage of the 2017 field fertilizer application during the wetter fall 2017/spring 2018 seasons, could explain the elevated Salix N_2O emissions observed in May and June 2018.

Within the cultivated fields themselves, there was great variability in N_2O flux depending on what crop within the potato rotation was present and its management. Potatoes had high N_2O emissions (mean $2.5 \text{ kg N}_2\text{O-N ha}^{-1}$), which was expected due to their large N fertilizer requirements. The mean rate of fertilization for potato crops in our study was 184 kg N ha^{-1} (online resource, Table 2). Estimated emissions for potato fields in our study using the Canadian Tier II regional emission coefficient (0.014) as used in the Canada's National Inventory Report predict cumulative seasonal N_2O emissions of $2.6 \text{ kg N}_2\text{O-N ha}^{-1}$, similar those observed in our study. This can be explained by the wet rainy climate of PEI, which could favor denitrification and incomplete denitrification processes from occurring. Given that potatoes require extensive soil disturbance during both planting and harvest, which also coincide with the wettest times of year on PEI, the potential for NO_3^- losses off-field is particularly high during the potato phase of the rotation; however, no correspondingly high N_2O emissions or soil NO_3^- were observed in the Salix or grass buffers. The potato phase can be seen as the least "environmentally friendly" stage of the rotation in terms of fertilizer losses off-field, and therefore the ultimate test in evaluating the ability of Salix buffers to mitigate fertilizer-derived N_2O emissions. Fields planted to sudangrass (mean $2.6 \text{ kg N}_2\text{O-N ha}^{-1}$) and black peas (mean $0.8 \text{ kg N}_2\text{O-N ha}^{-1}$) also had elevated N_2O emissions relative to Salix/grass buffers, with daily emissions sometimes exceeding those of potatoes. These results were unexpected as the purpose of the non-potato crops in the rotation is to mitigate some of the environmental degradation caused by potato cultivation, including keeping nitrogen in the soil and mitigating N_2O emissions. These high emissions can be explained by crop fertilization and other management factors that resulted in increased soil available NO_3^- , and comparable N_2O emissions to the potato crop. The sudangrass was planted and fertilized with nitrogen (30 kg N/ha) in mid-August 2018 (warm, moist soil) following field tillage (soil disturbance) and incorporation of the previous hay crop (carbon addition). This resulted in ideal conditions for N_2O emissions and N_2O emissions, previously low in 2018, were associated with this event. Most of the N_2O emissions from the black pea crop occurred early in the 2019 growing season. There are several explanations for this. In 2018 this site

was planted to potatoes, and the extremely wet fall in 2018 on PEI resulted in the grower being unable to harvest the potatoes in this field. As a result, the 2018 potato crop was plowed into the ground prior to planting the black peas in spring 2019. The black pea crop was also fertilized at planting (30 kg N/ha). The increased soil organic residues from the potato crop as well as the fertilizer application could have both contributed to elevated early-season N_2O emissions at this site. These results underscore the importance of management in determining N_2O emissions. Apparent NO_3^- capturing cover crops such as sudangrass or black pea can result in increased N_2O emissions if accompanied by N fertilization, especially if fertilization occurs when the soil is warm and wet.

Other field management considerations may explain differences in observed N_2O emissions between crops within the potato rotation relative to Salix/grass. N_2O emissions in the salix/grass buffers were similar to fields planted to hay even when crop fertilization occurred. The site D field was planted to hay (timothy/clover) in 2019 and was fertilized (15 kg N/ha). Despite receiving N fertilizer inputs, N_2O emissions even within the cultivated field were low at this site (mean daily $0.4 \text{ g N}_2\text{O-N/ha day}$). The NE for the cultivated field was low, similar to the Salix and grass treatments indicating that NO_3^- was not accumulating in this cropped field. The N_2O emissions were lower than other fields that received higher-rate fertilization and had higher NE values. In addition to the nitrogen status of this field, the low emissions can be explained by the perennial crop and the lack of tillage in this field, which has been found to reduce N_2O emissions relative to tilled fields (Hellebrand et al., 2008). The site D field was previously tilled in fall 2017 when the last potato crop was harvested and had been in constant hay production since (barley in 2017, timothy/clover in 2018). A similar situation presented itself at site C, which was planted to timothy/alfalfa in both 2017 and 2018 but received no fertilization. Field N_2O emissions and NE were low in 2018 ($-0.01 \text{ kg N}_2\text{O-N ha}^{-1}$)—which was similar to the agricultural-riparian grass and Salix buffers (mean daily -0.04 and $0.04 \text{ kg N}_2\text{O-N ha}^{-1}$).

Our results are comparable to most other studies which have also observed Salix N_2O emissions to be low, even when receiving fertilizer inputs, and that N_2O flux in Salix is reduced compared to most other vegetation and/or land use types (Borjesson, 1999;

Bressler et al., 2017; Drewer et al., 2012; Gauder et al., 2012; Kavdir et al., 2008; Lutes et al., 2016; Smialek et al., 2006; Whitaker et al., 2018). Some exceptions exist. Similar to our study, Bressler et al. (2017) did not observe significant differences in N_2O flux between hay and Salix; however, other agricultural crops had elevated N_2O relative to Salix. Likewise, our results corroborate previous findings that N_2O fluxes in Salix are thought to be primarily linked to nitrogen fertilizer inputs (Borjesson, 1999; Drewer et al., 2012; Gauder et al., 2012; Heller et al., 2003; Lutes et al., 2016).

However, other studies have found that denitrification is a major process occurring in Salix (Aronson et al., 2000; Dimitriou et al., 2012) including in 2- to 3-year actively growing Salix riparian buffers in agricultural regions (Ferrarini et al., 2017). As such, the potential for incomplete denitrification and the release of N_2O exists if NO_3^- accumulates. Since riparian areas tend to be sites of elevated denitrification regardless of vegetation type due to wetter conditions and accumulations of soil organic carbon (Fisher et al., 2014), the need for further study of the effect of Salix on GHG emissions in these settings exists. Preventing significant amounts of NO_3^- from accumulating, such as when NO_3^- influx is lower than the uptake capacity of Salix, can control incomplete denitrification. While our study indicates that Salix agricultural-riparian buffers can be effective at mitigating agriculturally derived riparian N_2O emissions on PEI, further study of Salix in a variety of hydrogeomorphically distinct riparian settings is needed to better understand the potential and limitations of this Salix application.

4.2 Soil Respiration (CO_2)

CO_2 emissions at the soil-atmosphere interface are primarily caused by aerobic microbial respiration and root respiration, and are affected by soil moisture content, temperature, and soil organic matter content (Jacinthe & Vidon, 2017; Lutes et al., 2016; Pacaldo et al., 2014a; Smith et al., 2018). Temperature is usually the main determinant of CO_2 flux since microbial activity and root respiration increase at higher temperatures and observed CO_2 flux in our study followed this seasonal trend at all sites and treatments, with highest emissions occurring in July and August. This is consistent with other studies that have found

seasonally determined CO_2 flux under Salix in a variety of settings (agricultural, riparian, forest) (Bressler et al., 2017; Gauder et al., 2012; Jacinthe et al., 2015; Lutes et al., 2016; Vidon et al., 2014).

In our study, larger Salix CO_2 emissions relative to other treatments may be explained by leaf decay and the high proportion of fine root hairs in Salix that turnover and decompose approximately once per season (Pacaldo et al., 2013; Rytter, 2001). Increased fine root biomass in soils is positively correlated with soil CO_2 emissions (Hu et al., 2016), and Salix are known to contribute a large amount of decomposing organic carbon to soils (approximately 4.5–10 tonnes of dry matter/ha) through leaf litter and root decomposition (Borjesson, 1999). Annual fine root production of *Salix Viminalis* in sandy soils has been observed to be 28–50% of the net primary productivity, with most decomposition occurring within a year (Rytter, 2001) although smaller fine roots can turn over up to 4 times in a single year (Borjesson, 1999). Studies have also found that Salix do not appear to cause soil organic matter levels to increase long term, indicating that this carbon is released back to the atmosphere (Pacaldo et al., 2013).

In our study, differences in CO_2 emissions by crop type are consistent with other studies that observed elevated CO_2 fluxes in Salix relative to most, but not all, annual cropping systems (Bressler et al., 2017; Gauder et al., 2012; Pacaldo et al., 2014b). However, the situation is complex: other studies have found that transitioning from annual cropping systems to Salix has decreased soil CO_2 emissions (Borjesson, 1999) and that Salix variety and previous land use can impact observed CO_2 flux (Lutes et al., 2016; Nikiema et al. 2012).

CO_2 flux under Salix was similar to sudangrass, and the larger emissions observed in sudangrass can be explained by a variety of management factors. The sudangrass was cultivated and fertilized in-season, and soil CO_2 emissions can intensify with soil disturbance and N application as a result of enhanced microbial activity and C availability due to increased plant growth (Lutes et al., 2016 via Gauder et al., 2012). In our study, cumulative seasonal CO_2 flux was positively correlated with cumulative seasonal available NO_3^- ($r^2=0.119$; $p=0.053$). Soil disturbance aerated the soil and mixed in decomposing organic matter from the previous harvested crop, during the warmest months of the year and stimulated microbial activity—enhancing CO_2 emissions.

Higher CO₂ emissions at site B can be explained by high rates of fertilization in both study years (online resource, Table II) in conjunction with wetter soil conditions. Site B had the second highest overall mean soil moisture content of 31.9%. Conversely, lower CO₂ emissions at site D can be explained by lower rates of fertilization and dry soil conditions: site D was the driest site overall, with a mean soil moisture content of 20.3% in 2019. CO₂ emissions can intensify with N fertilizer application as a result of enhanced microbial activity and C availability from increased plant growth (Gauder et al., 2012). Additionally, CO₂ flux can be elevated due to more frequent soil tillage that promotes oxidation of soil carbon and decomposition of organic matter (Borjesson, 1999). Site B was in potatoes in 2018, and therefore experienced soil disturbance (tillage) during planting in the spring. This site was not harvested in 2018 due to a particularly wet fall, leaving the potatoes to be tilled into the soil in spring 2019 when the 2019 black pea crop was planted. Soil tillage 2 consecutive years, heavy fertilization, and the increased decomposing organic matter from the discarded potato crop could all have contributed to elevated CO₂ emissions at this site. In contrast, site D did not experience any soil tillage in 2019 or in 2018 (the year prior to monitoring this site). This site was in barley in 2018 and timothy/clover in 2019 when the site was monitored for GHG flux. Lower fertilization rates, lack of soil tillage, and dryer soil conditions could all have contributed to lower CO₂ emissions observed at site D.

4.3 Methane

This study showed that across all sites and years, overall CH₄ fluxes at the soil-atmosphere interface were not significantly different in Salix agricultural-riparian buffers than in grass agricultural-riparian buffers or from upslope cultivated agricultural fields in potato rotation, except at site A. Overall, observed CH₄ fluxes were low at all sites and in all treatments, and each site exhibited spatial variability with both sinks and sources of CH₄ occurring within each site on most sampling dates, which is consistent with observations in other Salix studies (Bressler et al., 2017; Jacinthe et al., 2015). However, cumulative seasonal CH₄ fluxes revealed most sites to be slight net sinks except at site E and site C in 2019.

4.4 The Impact of Increasing Distance from the Cultivated Field on N₂O Fluxes and Shallow Soil NO₃⁻ Exposure Across the Width of the Salix BufferZone

In our study, there was no impact of increasing distance from the cultivated field on N₂O fluxes due to NE being consistently low (<200 ug N/10 cm²) across the entire width of the buffer zone and much lower than the NE values for the cultivated fields receiving >30 kg N/ha which ranged 500–3000 ug N/10 cm².

It is well established that Salix is extremely effective at improving soil water quality due to its capacity to significantly reduce groundwater NO₃⁻ levels even when large amounts of NO₃⁻ are present or being applied (Aronsson et al., 2000; Borjesson, 1999; Bressler et al., 2017; Dimitriou et al., 2012; Dimitriou et al., 2009; Ferrarini et al., 2017; Heller et al., 2003; Young & Briggs, 2005). Ferrarini et al. (2017) and Young and Briggs (2005) both observed significant reductions in soil NO₃⁻ concentration across Salix riparian buffers relative to upland agricultural fields. It follows that corresponding decreases in N₂O emissions could be expected, since NE (NO₃⁻ accumulation) is an important predictor of N₂O emissions (Burton et al., 2008). In general, it has been estimated that Salix buffer strips planted between annual cropped fields and streams can retain N at a rate of 70 kg N ha⁻¹ year⁻¹ when nitrate inputs are greater than 15 kg N ha⁻¹ year⁻¹, and that 2/3 of this is due to plant uptake, with the remaining 1/3 lost to denitrification (Borjesson, 1999). Of this 1/3 denitrified N, approximately 3% is thought to be released as N₂O, representing 1% of nitrate retained by Salix buffers (Styles et al., 2016). In contrast, NO₃⁻ that is transported to waterways has a 75% risk of being lost as N₂O (Styles et al., 2016).

N₂O emissions observed at the soil-atmosphere interface reflect N₂O produced throughout the entire depth of the soil profile. As N₂O produced at greater depths diffuses upward, it is more likely to be further reduced to N₂ if wet conditions are present. The fact that N₂O emissions in the Salix buffer were generally low and there was no observed reduction across the buffer zone indicates that Salix were effective creating conditions that prevented incomplete denitrification from occurring.

4.5 The Impact of Wetting Events on N_2O Fluxes

Brief increases of N_2O production following wetting events are common and often responsible for the majority of soil N_2O emissions (Audet et al., 2014; Fisher et al., 2014; Hellebrand et al., 2008). Riparian zones in agricultural regions have been found to be particularly at risk for high N_2O emissions due to the presence and accumulation of high influxes of water-soluble NO_3^- from upland agricultural fertilization following precipitation events (Fisher et al., 2014; Kaushal et al., 2014; Vidon et al., 2010). Denitrification is a major mechanism by which riparian areas act as ecosystem control points for mitigating the transfer of N from upland agricultural land to waterways (Bernhardt et al., 2017; Young & Briggs, 2005). Incomplete denitrification, which releases N_2O , can occur as a result of the *nosZ* enzyme (responsible for the final conversion of N_2O to N_2) becoming dormant when soil conditions are dry for an extended period (Robertson & Groffman, 2015; Zaady et al., 2013), but riparian areas seldom experience extended dry periods. In soils where the *nosZ* enzyme is present, N_2O reduction has been shown to be controlled by nitrate content and water-filled pore space (Dandie et al., 2008). As a result of high water contents, riparian areas (and by implication, agricultural-riparian buffers) may be more likely to experience elevated instances of N_2O emissions than upland areas following precipitation events (Audet et al., 2013; Aronsson et al., 2000; Fisher et al., 2014; Jacinthe & Vidon, 2017). Maximum rates of denitrification are often triggered when water-filled pore space exceeds 60–70% and sufficient soil NO_3^- and organic carbon is present (Dandie et al., 2008; Jacinthe & Vidon, 2017).

In our study, there were several instances of elevated N_2O emissions in the cultivated fields following high rainfall occurrences that coincided with high field NO_3^- availability (observed N_2O fluxes of 120, 293, 194, and 124 g $N_2O-N\ ha^{-1}\ day^{-1}$). These were the sites with high NE and cumulative N_2O emissions across all sites were correlated with NE. However, there were no corresponding high N_2O emissions observed in the Salix or grass agricultural-riparian buffers (observed grass 6.7, 2.1, 0.5, $-0.7\ g\ N_2O-N\ ha^{-1}\ day^{-1}$; observed Salix 4.0, 0.4, 1.4, $0.4\ g\ N_2O-N\ ha^{-1}\ day^{-1}$) which had low NE values. This represents a 94–99.9% decrease in N_2O emissions in the

agricultural-riparian buffers during wetting events and suggests this is not a major factor stimulating N_2O emissions in these systems. The lack of N_2O spikes in the buffer areas also indicates the effectiveness of this BMP at mitigating elevated riparian N_2O emissions. Salix would intercept and absorb fertilizer-derived NO_3^- for plant uptake as it was being transported from the agricultural field, and the lack of buffer N_2O emissions following these precipitation events suggests soil moisture conditions were consistently high enough in the buffers to maintain high-enough concentrations of activated *nosZ* enzyme and maintain soil NO_3^- sufficiently low to allow complete denitrification of any remaining NO_3^- .

While some other studies have observed elevated N_2O in Salix following precipitation (Kavdir et al., 2008; Ley et al., 2018; Lutes et al., 2019; Nikiema et al., 2012; Palmer et al., 2014), this seems to be more common in initial and more immature cultivations and is linked to NO_3^- availability. Our findings are in alignment with Bressler et al. (2017) who also failed to observe elevated N_2O flux in fertilized Salix, and Ley et al. (2018) who observed much lower N_2O emissions in Salix than an unaltered control in floodplain soils.

4.6 Overall GWP of Soil-Atmosphere Interface N_2O and CH_4 Emissions

Our study found that non- CO_2 GHG emissions at the soil-atmosphere interface in Salix and grass buffers had a significantly lower GWP than upslope agricultural fields and that differences in N_2O flux between treatments were the main reason for these differences in overall GWP. These findings differ from Bressler et al. (2017), who found that transitioning to Salix from conventional crops on marginal (excessive moisture) cropland had no overall effect on GHG emissions at the soil-atmosphere interface. Their study found that CO_2 equivalents were similar between Salix, hay, and corn, since increased CO_2 emissions in Salix were roughly balanced by increased N_2O emissions in conventional crops, suggesting that “... in spite of differences for individual gasses, Salix does not provide the ecosystem service of greenhouse gas emission reduction compared to corn or hay” (p137). However, unlike our study, Bressler et al. (2017) incorporated CO_2 flux into GWP calculations. Even when CO_2 emissions are considered, the overall GWP

of *Salix* in our study is still significantly lower than upslope agricultural fields. Other studies looking at the GWP of *Salix* on both a life cycle and field scale soil GHG emissions have opted to not include CO_2 flux in GWP calculations since *Salix* are considered a carbon-neutral energy source (Volk et al., 2006) or carbon balance in these situations is generally assessed by measuring carbon stocks (Gauder et al., 2012). Similarly, in their life cycle assessment Volk et al. (2006) found that *Salix* bioenergy crops have a small GWP (39–52 kg CO_2 equivalents per MWh electricity produced), mainly due to N_2O and CH_4 emissions. In their field study Gauder et al. (2012) did not include soil CO_2 flux when calculating GWP of different bioenergy crops (including *Salix*) based on trace gas flux measured at the soil-atmosphere interface. N_2O was found to have a bigger impact on GWP than CH_4 , and fertilized *Salix* was found to have a lower (but negative) global warming potential than fertilized miscanthus or maize, while unfertilized *Salix* had a slightly higher global warming potential (although still negative) than unfertilized miscanthus, but lower than unfertilized maize.

5 Conclusion

This study found that *Salix viminalis* and grass grown in agricultural-riparian buffer strips downslope of cultivated fields in potato rotation did not exhibit elevated N_2O emissions and were successful at preventing near surface NO_3^- transport beyond the buffer zone. *Salix* buffers had N_2O fluxes that were an average of 86% lower than upslope cultivated fields (up to 98% lower), representing mean cumulative seasonal reductions of 1.3 kg $\text{N}_2\text{O}-\text{N ha}^{-1}$ (up to a maximum observed reduction of 6.16 kg $\text{N}_2\text{O}-\text{N ha}^{-1}$). *Salix* buffers also displayed significant cumulative seasonal GHG reductions compared to upslope fields. The mean cumulative average GWP of the *Salix* buffers was 409 Mg $\text{CO}_2\text{e ha}^{-1}$ lower than the cultivated fields, with reductions of up to 1813 Mg $\text{CO}_2\text{e ha}^{-1}$ observed. There was a large amount of between and within-site variation in soil moisture content observed across all the research sites, and soil water content was positively correlated with N_2O emissions. Our study did not observe a change in N_2O emissions across the width of the buffer zone and did not observe elevated N_2O emissions in buffers following

precipitation events; however, these events did result in elevated N_2O emissions in cultivated fields. While CH_4 emissions were generally low in our study, with no significant differences between treatments, site A was the only one to experience occasional flooding, and these events were marked by elevated CH_4 emissions.

While high N_2O emissions were anticipated in fields cultivated to potatoes, this study also observed unexpectedly elevated N_2O flux outside of the potato phase of the rotation, in fields planted to sorghum sudangrass and black peas. These emissions were linked to nitrogen fertilization events occurring later in the growing season when the soil was warm and wet as well as decomposition of the previous years unharvested potato crop at one of the sites. Differences in cumulative N_2O emissions were primarily related to NO_3^- accumulation as reflected in NE.

Our research confirms that cultivating *Salix* buffer strips on downslope field edges bordering riparian zones is an effective strategy for mitigating agriculturally derived nitrous oxide (N_2O) emissions in riparian areas on Prince Edward Island. While *Salix* were no more effective than a grass agricultural-riparian buffer at reducing N_2O and overall GHG emissions at the soil-atmosphere interface, *Salix* have the additional benefit of high biomass accrual with associated carbon and nutrient sequestration, topics which will be addressed in a subsequent paper.

Funding This study was funded by the East Prince Agri-Environmental Association through funding provided by Agriculture and Agri-food Canada's Agricultural Greenhouse Gas Program. Funding was also provided by an NSERC Discovery Grant (Burton).

Data Availability The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

Declarations

Conflict of Interest The authors declare no competing interests.

Open Access This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated

otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit <http://creativecommons.org/licenses/by/4.0/>.

References

- Amichev, B. Y., Hangs, R. D., Konecsni, S. M., Stadnyk, C. N., Volk, T. A., Belanger, N., Vujanovic, V., Schoenau, J. J., Moukoui, J., & Rees, K. C. J. V. (2014). Willow short-rotation production systems in Canada and Northern United States: A review. *Soil Science Society of America Journal*, 78(6), S168.
- Aronsson, P. G., Bergstrom, L. F., & Elowson, S. N. E. (2000). Long-term influence of intensively cultured short-rotation Salix coppice on nitrogen concentrations in groundwater. *Journal of Environmental Management*, 58(2), 135–145.
- Audet, J., Elsgaard, L., Kjaergaard, C., Larsen, S., & Hoffmann, C. C. (2013). Greenhouse gas emissions from a danish riparian wetland before and after restoration. *Ecological Engineering*, 57, 170–182. <https://doi.org/10.1016/j.ecoleng.2013.04.021>
- Audet, J., Hoffmann, C. C., Andersen, P. M., Baattrup-Pedersen, A., Johansen, J. R., Larsen, S. E., ... Elsgaard, L. (2014). Nitrous oxide fluxes in undisturbed riparian wetlands located in agricultural catchments: Emission, uptake and controlling factors. *Soil Biology and Biochemistry*, 68, 291–299. <https://doi.org/10.1016/j.soilbio.2013.10.011>
- Aulakh, M. S., Rennie, D. A., & Paul, E. A. (1984). Gaseous nitrogen losses from soils under zero-till as compared with conventional-till management systems. *Journal of Environmental Quality*, 13(1), 130–136.
- Belanger, G., Walsh, J. R., Richards, J. E., Milburn, P. H., & Ziadi, N. (2001). Predicting nitrogen fertilizer requirements of potatoes in Atlantic Canada with soil nitrate determinations. *Canadian Journal of Soil Science*, 81(5), 535–544.
- Bernhardt, E. S., Blaszcak, J. R., Ficken, C. D., Fork, M. L., Kaiser, K. E., & Seybold, E. C. (2017). Control points in ecosystems: Moving beyond the hot spot hot moment concept. *Ecosystems*, 20(4), 665–682. <https://doi.org/10.1007/s10021-016-0103-y>
- Billen, G., Garnier, J., Gossel, A., Thieu, V., Théry, S., & Hénault, C. (2020). Modeling indirect N₂O emissions along the N cascade from cropland soils to rivers. *Biogeochemistry*, 148(2), 207–221. <https://doi.org/10.1007/s10533-020-00654-x>
- Borjesson, P. (1999). Environmental effects of energy crop cultivation in Sweden - I: Identification and quantification. *Biomass & Bioenergy*, 16(2), 137–154.
- Bressler, A. S., Vidon, P. G., & Volk, T. A. (2017). Impact of shrub willow (*Salix* spp.) as a potential bioenergy feedstock on water quality and greenhouse gas emissions. *Water, Air, and Soil Pollution*, 228(4), 170.
- Burton, D. L. (2013). *Standard operating procedure greenhouse gas (CO₂, CH₄ and N₂O) flux measurement*. Dalhousie University.
- Burton, D. L., Zebbarth, B. J., Gillam, K. M., & MacLeod, J. A. (2008). Effect of split application of fertilizer nitrogen on N₂O emissions from potatoes. *Canadian Journal of Soil Science*, 88, 229–239.
- Cho, C. M., Burton, D. L., & Chang, C. (1997a). Kinetic formulation of oxygen consumption and denitrification processes in soil. *Canadian Journal of Soil Science*, 77, 253–260.
- Cho, C. M., Burton, D. L., & Chang, C. (1997b). Denitrification and fluxes of nitrogenous gases from soil under steady state oxygen distribution. *Canadian Journal of Soil Science*, 77, 261–269.
- Dandie, C., Burton, D. L., Zebbarth, B. J., Henderson, S., Trevors, J. T., & Goyer, C. (2008). Changes in bacterial denitrifier community abundance over time in an agricultural field and their relationship with denitrification activity. *Appl. Environ. Micro.*, 74, 5997–6005.
- Dimitriou, I., Mola-Yudego, B., & Aronsson, P. (2012). *Impact of Salix short rotation coppice on water quality*. Springer Science + Business Media.
- Dimitriou, I., Busch, G., Jacobs, S., Schmidt-Walter, P., & Lamersdorf, N. (2009). A review of the impacts of short rotation coppice cultivation on water issues. *Landbau-forschung Volkenrode.*, 59, 197–206.
- Drewer, J., Finch, J. W., Lloyd, C. R., Baggs, E. M., & Skiba, U. (2012). How do soil emissions of N₂O, CH₄ and CO₂ from perennial bioenergy crops differ from arable annual crops? *GCB Bioenergy*, 4(4), 408–419.
- Environment and Climate Change Canada (2022). National inventory report 1990–2020: Greenhouse gas sources and sinks in Canada 2022. Canada's submission to the United Nations Framework Convention on Climate Change. Environmental and Climate Change Canada, Ottawa Ontario. Cat. No. En81–4E-PDF.
- Erisman, J. W., Sutton, M. A., Galloway, J., Klimont, Z., & Winiwarter, W. (2008). How a century of ammonia synthesis changed the world. *Nature Geoscience*, 1, 636–639.
- Ferrarini, A., Fornasier, F., Serra, P., Ferrari, F., Trevisan, M., & Amaducci, S. (2017). Impacts of Salix and miscanthus bioenergy buffers on biogeochemical N removal processes along the soil-groundwater continuum. *GCB Bioenergy*, 9(1), 246–261. <https://doi.org/10.1111/gcbb.12340>
- Fisher, K., Jacinthe, P. A., Vidon, P., Liu, X., & Baker, M. E. (2014). Nitrous oxide emission from cropland and adjacent riparian buffers in contrasting hydrogeomorphic settings. *Journal of Environmental Quality*, 43(1), 338–348.
- Gauder, M., Butterbach-Bahl, K., Graeff-Hnninger, S., Claupein, W., & Wiegel, R. (2012). *Soil-derived trace gas fluxes from different energy crops - Results from a field experiment in southwest Germany*. Blackwell Publishing LTD.
- Gillam, K. M., Zebbarth, B. J., & Burton, D. L. (2008). Nitrous oxide emissions from denitrification and the partitioning of gaseous losses as affected by nitrate and carbon addition and soil aeration. *Canadian Journal of Soil Science*, 88, 133–143.

- Gold, A. J., Groffman, P. M., Addy, K., Kellogg, D. Q., Stolt, M., & Rosenblatt, A. E. (2001). Landscape attributes as controls on ground water nitrate removal capacity of riparian zones. *Journal of the American Water Resources Association*, 37(6; 6), 1457–1464.
- Gooijer, H., Pharo, C. & Schroeder, W. (2011). Agroforestry: A profitable land use. *Proceedings of the 12th North American Agroforestry Conference, Athens, Georgia, USA, 4–9 June 2011*.
- Government of Canada (2019a). Canadian Climate Normals 1981–2010 Station Data.
- Government of Canada (2019b). Station results - Historical data. New Glasgow PE Daily Data Reports.
- Government of PEI (2010). Bioeconomy crop initiative. PEI Department of Agriculture and Forestry. Accessed June 2019 from: http://www.gov.pe.ca/photos/original/af_flex_bioprogram.pdf#:~:text=Bioeconomy%20Crop%20Initiative%20The%20Bioeconomy%20Crop%20Initiative%20is,Canada%20and%20the%20Province%20of%20Prince%20Edward%20Island
- Granli, T., & Beckman, O. C. (1994). Nitrous oxide from agriculture. *Norwegian Journal of Agricultural Sciences*, 12, 128.
- Havlin, J. L., Beaton, J. D., Tisdale, S. L., & Nelson, W. L. (2013). *Soil fertility and fertilizers: an introduction to nutrient management*, (Ed.8). Prentice-Hall Inc.
- Hellebrand, H. J., Scholz, V., & Kern, J. (2008). Fertiliser induced nitrous oxide emissions during energy crop cultivation on loamy sand soils. *Atmospheric Environment*, 36, 8403.
- Heller, M. C., Keoleian, G. A., & Volk, T. A. (2003). Life cycle assessment of a Salix bioenergy cropping system. *Biomass and Bioenergy*, 25(2), 147–165.
- Henault-Ethier, L., Larocque, M., Perron, R., Wiseman, N., & Labrecque, M. (2017). Hydrological heterogeneity in agricultural riparian buffer strips. *Journal of Hydrology*, 546, 276–288. <https://doi.org/10.1016/j.jhydrol.2017.01.001>
- Hu, X., Liu, L., Zhu, B., Du, E., Hu, X., Li, P., Zhou, Z., Ji, C., Zhu, J., Shen, H., & Fang, J. (2016). Asynchronous responses of soil carbon dioxide, nitrous oxide emissions and net nitrogen mineralization to enhanced fine root input. *Soil Biology and Biochemistry*, 92, 67–78. <https://doi.org/10.1016/j.soilbio.2015.09.019>
- Hutchinson, G. and Livingston, G. (1993). Use of chamber systems to measure trace gas fluxes. Pages 63–78 in L. A. Harper, A. R. Mosier, J. M. Duxbury, and D. E. Rolston, eds. *Agricultural ecosystem effects on trace gases and global climate change*. Special Publication No 55. ASA, CSSA, SSSA, Madison, WI.
- IPCC (2013). *Climate change 2013: The physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge University Press.
- Jacinthe, P. A., Vidon, P., Fisher, K., Liu, X., & Baker, M. E. (2015). Soil methane and carbon dioxide fluxes from cropland and riparian buffers in different hydrogeomorphic settings. *Journal of Environmental Quality*, 44(4), 1080–1090.
- Jacinthe, P. A., & Vidon, P. (2017). Hydro-geomorphic controls of greenhouse gas fluxes in riparian buffers of the white river watershed, IN (USA). *Geoderma*, 301, 30–41. <https://doi.org/10.1016/j.geoderma.2017.04.007>
- Jiang, Y., Jamieson, T., Nyiraneza, J., Somers, G., Thompson, B., Murray, B., & Geng, X. (2015). Effects of fall vs. spring plowing forages on nitrate leaching losses to groundwater. *Ground Water Monitoring and Remediation*, 35(1), 43–54.
- Kaushal, S. S., Mayer, P. M., Vidon, P. G., Smith, R. M., Pennino, M. J., Newcomer, T. A., Duan, S., Welty, C., & Belt, K. T. (2014). Land use and climate variability amplify carbon, nutrient, and contaminant pulses: A review with management implications. *Journal of the American Water Resources Association*, 3, 585. <https://doi.org/10.1111/jawr.12204>
- Kavdir, Y., Hellebrand, H. J., & Kern, J. (2008). Seasonal variations of nitrous oxide emission in relation to nitrogen fertilization and energy crop types in sandy soil. *Soil & Tillage Research*, 98, 175–186. <https://doi.org/10.1016/j.still.2007.11.002>
- Lantz, et al. (2014). Benefit-cost analysis of hybrid Salix crop production on agricultural land in eastern Canada: Assessing opportunities for on-farm and off-farm bioenergy use. *Biomass and Bioenergy*, 63, 257–267. <https://doi.org/10.1016/j.biombioe.2014.01.027>
- Ley, M., Lehmann, M. F., Niklaus, P. A., & Luster, J. (2018). Alteration of nitrous oxide emissions from floodplain soils by aggregate size, litter accumulation and plant-soil interactions. *Biogeosciences*, 15(22), 7043–7057.
- Liang, K., Jiang, Y., Nyiraneza, J., Fuller, K., Murnaghan, D., & Meng, F.-R. (2019). Nitrogen dynamics and leaching potential under conventional and alternative potato rotations in Atlantic Canada. *Field Crops Research*, 242, 107603. <https://doi.org/10.1016/j.fcr.2019.107603>
- Lutes, K., Oelbermann, M., Thevathasan, N. V., & Gordon, A. M. (2019). Assessing the impact of fertilizer application on net soil-derived emission budgets from a temperate Salix (*Salix miyabeana*) short rotation coppice system. *Biomass and Bioenergy*, 120, 135–143.
- Lutes, K., Oelbermann, M., Thevathasan, N. V., & Gordon, A. M. (2016). Effect of nitrogen fertilizer on greenhouse gas emissions in two Salix clones (*salix miyabeana* and *S. dasyclados*) in southern ontario, canada. *Agroforestry Systems*, (5), 785. <https://doi.org/10.1007/s10457-016-9897-z>
- MacDougall, J. I., Veer, C., & Wilson, F. (1988). *Soils of Prince Edward Island: Prince Edward Island soil survey*. Research Branch, Agriculture Canada.
- Nikiema, P., Rothstein, D. E., & Miller, R. O. (2012). Initial greenhouse gas emissions and nitrogen leaching losses associated with converting pastureland to short-rotation woody bioenergy crops in northern michigan, USA. *Biomass and Bioenergy*, 39, 413–426. <https://doi.org/10.1016/j.biombioe.2012.01.037>
- Pacaldo, R. S., Volk, T. A., & Briggs, R. D. (2014a). Carbon sequestration in fine roots and foliage biomass offsets soil CO₂ effluxes along a 19-year chronosequence of shrub willow (*salix x dasyclados*) biomass crops. *BioEnergy Research*, 7(3), 769–776. <https://doi.org/10.1007/s12155-014-9416-x>
- Pacaldo, R. S., Volk, T. A., Briggs, R. D., Abrahamson, L. P., Bevilacqua, E., & Fabio, E. S. (2014b). Soil CO₂ effluxes, temporal and spatial variations, and root respiration in shrub willow biomass crops fields along a 19-year

- chronosequence as affected by regrowth and removal treatments. *GCB Bioenergy*, 6(5), 488–498. <https://doi.org/10.1111/gcbb.12108>
- Pacaldo, R., Volk, T., & Briggs, R. (2013). Greenhouse gas potentials of shrub willow biomass crops based on below- and aboveground biomass inventory along a 19-year chronosequence. *BioEnergy Research*, 6(1), 252–262. <https://doi.org/10.1007/s12155-012-9250-y>
- Palmer, M. M., Forrester, J. A., Rothstein, D. E., & Mladenoff, D. J. (2014). Conversion of open lands to short-rotation woody biomass crops: Site variability affects nitrogen cycling and N₂O fluxes in the US northern lake states. *GCB Bioenergy*, 6(4), 450–464.
- Pharo, C. (2012). Personal communication. Regional Manager, sustainable Agri-production systems, Agriculture and Agri-food Canada, Charlottetown, PEI.
- Robertson, G. P. & Groffman, P.M. (2015). Nitrogen transformations. Pages 421–446 in E. A. Paul, editor. *Soil Microbiology, Ecology, and Biochemistry*. 4th ed. Academic Press, Burlington, Massachusetts, USA. <https://doi.org/10.1146/annurev.environ.032108.105046>
- Ruser, R., Flessa, H., Schilling, R., Beese, F., & Munch, J. C. (2001). Effect of crop-specific field management and N fertilization on N₂O emissions from a fine-loamy soil. *Nutrient Cycling in Agroecosystems*, 59, 177–191.
- Rytter, R. M. (2001). Biomass production and allocation, including fine-root turnover, and annual N uptake in lysimeter-grown basket willow. *Forest Ecology and Management*, 2–3, 177.
- Schroeder, W. (2019). Personal communication. Agroforestry research Advisor, Agriculture and Agri-food Canada, Indian Head, Saskatchewan.
- Schultz, R. C., Colletti, J. P., Isenhardt, T. M., Simpkins, W. W., Mize, C. W., & Thompson, M. L. (1995). Design and placement of a multispecies riparian buffer strip system. *Agroforestry Systems*, 29(3), 201–226.
- Smialek, J., Bouchard, V., Lippmann, B., Quigley, M., Granata, T., Martin, J., & Brown, L. (2006). *Effect of a woody (salix nigra) and an herbaceous (juncus effusus) macrophyte species on methane dynamics and denitrification*. Society of Wetlands Scientists.
- Smith, K. A., Ball, T., Conen, F., Dobbie, K. E., Massheder, J., & Rey, A. (2018). Exchange of greenhouse gases between soil and atmosphere: Interactions of soil physical factors and biological processes. *European Journal of Soil Science*, 1, 10. <https://doi.org/10.1111/ejss.12539>
- Snowdon, E., Zebarth, B. J., Burton, D. L., Goyer, C., & Rochette, P. (2013). Growing season N₂O emissions from two-year potato rotations in a humid environment in New Brunswick, Canada. *Canadian Journal of Soil Science*, 93(3), 279–294.
- Statistics Canada (2017). Farm and farm operator data : Prince Edward Island has the largest potato crop in Canada.
- Styles, D., Börjesson, P., D'Hertefeldt, T., Birkhofer, K., Dauber, J., Adams, P., ... Rosenqvist, H. (2016). Climate regulation, energy provisioning and water purification: Quantifying ecosystem service delivery of bioenergy Salix grown on riparian buffer zones using life cycle assessment. *Ambio*, 45(8), 872–884.
- Vidon, P., Jacinthe, P.-A., Liu, X., Fisher, K., & Baker, M. (2014). Hydrobiogeochemical controls on riparian nutrient and greenhouse gas dynamics: 10 years post-restoration. *Journal of the American Water Resources Association*, 3, 639. <https://doi.org/10.1111/jawr.12201>
- Vidon, P., Allan, C., Burns, D., Duval, T. P., Gurwick, N., Inamdar, S., Lowrance, R., Okay, J., Scott, D., & Sebestyen, S. (2010). Hot spots and hot moments in riparian zones: Potential for improved water quality management. *Journal-American Water Resources Association*, 2, 278.
- Volk, T. A., Abrahamson, L. P., Nowak, C. A., Smart, L. B., Tharakan, P. J., & White, E. H. (2006). The development of short-rotation Salix in the northeastern United States for bioenergy and bioproducts, agroforestry and phytoremediation. *Biomass and Bioenergy*, No 8–9, 715.
- Weier, K. L., Doran, J. W., Power, J. F., & Walters, D. T. (1993). Denitrification and the dinitrogen nitrous-oxide ratio as affected by soil-water, available carbon, and nitrate. *Soil Science Society of America Journal*, 57(1), 66–72.
- Whitaker, J., Field, J. L., Bernacchi, C. J., Cerri, C. E. P., Ceulemans, R., Davies, C. A., ... McNamara, N. P. (2018). Consensus, uncertainties and challenges for perennial bioenergy crops and land use. *GCB Bioenergy*, (3), 150. <https://doi.org/10.1111/gcbb.12488>
- Wright, T. (2012). Personal Communication. Soil and Water Engineer, PEI Department of Agriculture and Forestry, Charlottetown, PEI.
- Young, E. O., & Briggs, R. D. (2005). Shallow ground water nitrate-N and ammonium-N in cropland and riparian buffers. *Agriculture, Ecosystems and Environment*, 109, 297–309. <https://doi.org/10.1016/j.agee.2005.02.026>
- Zaady, E., Groffman, P. M., Standing, D., & Shachak, M. (2013). High N₂O emissions in dry ecosystems. *European Journal of Soil Biology*, 59, 1–7. <https://doi.org/10.1016/j.ejsobi.2013.08.004>
- Zebarth, B. J., Danieleescu, S., Nyiraneza, J., Ryan, M. C., Jiang, Y., Grimmett, M., & Burton, D. L. (2015). Controls on Nitrate Loading and Implications for BMPs under intensive potato production systems in Prince Edward Island, Canada. *Ground Water Monitoring and Remediation*, 35(1), 30–42.
- Zebarth, B. J., & Rosen, C. J. (2007). Research perspective on nitrogen bmp development for potato. *American Journal of Potato Research*, 84(1), 3–18.
- Zebarth, B. J., Snowdon, E., Burton, D. L., Goyer, C., & Dowbenko, D. (2012). Controlled release fertilizer product effects on potato crop response and nitrous oxide emissions under rain-fed production on a medium-textured soil. *Canadian Journal of Soil Science*, 92, 759–769.