

**Impacts of Dredging and Brush Cutting of
Paired Agricultural Drainage Ditches on
GHG Emissions and Nutrient Filtration Capacity**

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Statement of Contributions

This thesis consists of a series of manuscripts prepared for submission to peer-reviewed journals. As the first author of each paper, the majority of the contributions (2019 and 2020 data collection, figure preparation, and writing manuscripts) were completed by myself.

Chapter 1: This chapter was written solely by Andrew Schietzsch with input from David Blowes and Carol Ptacek.

Chapter 2: Andrew Schietzsch wrote the manuscript with feedback from David Blowes, David Lapen, and Carol Ptacek. Prior to Andrew Schietzsch starting the project, Dr. Richard Amos, Oliver Blume, Dr. Ian Clark, and Anne-Martine Doucet worked on collecting the 2018 data used in the study. Emilia Craiovan and Mark Sunohara helped throughout the field project with data collection, field work advice, setting up monitoring equipment.

Chapter 3: Andrew Schietzsch wrote the manuscript with feedback from David Blowes, David Lapen, and Carol Ptacek. Emilia Craiovan and Mark Sunohara helped throughout the field project with data collection, field work advice, setting up monitoring equipment, and water flow analysis.

Chapter 4: This chapter was written solely by Andrew Schietzsch with input from David Blowes and Carol Ptacek.

Abstract

Agriculture provides many beneficial and essential ecosystem services. Along with these beneficial services, the conversion of natural ecosystems into heavily modified agricultural ecosystems is also a source of disservice, including being a major source of global greenhouse gas (GHG) emissions and pollution of downstream waterways due to increased nutrient runoff. Carbon (C), Nitrogen (N) and Phosphorus (P) applied to agricultural fields as fertilizer are a source of carbon dioxide (CO₂), methane (CH₄), and nitrous oxide (N₂O) emissions. Nutrient runoff can lead to excess P in surface water bodies causing algae blooms, and excess N can lead to excess nitrate (NO₃) in rural groundwater (GW) wells. There is a need to establish beneficial management practices (BMPs) to take into account agriculture-derived pollution with all agricultural practices. This thesis informs the development of BMPs by examining the environmental pollution aspects of both: 1) GHG emissions; and 2) nutrient export; resulting from the common practices of brush cutting and dredging of ditches to enhance drainage.

Riparian vegetation in agricultural drainage ditches has been shown to decrease insolation, which decreases soil and water temperatures. This vegetation also hinders drainage by restricting flow, thus raising water levels, which decreases CO₂ emissions and increases CH₄ and N₂O emissions. However, no previous studies have examined in detail the effects of removing drainage ditch vegetation. This study examines the GHG emissions from four ditch microplots in the South Nation Watershed in southern Ontario, Canada following the removal of riparian vegetation from two microplots. The trials took place over three field seasons, and the intervention methods were selected to observe the effects of brush cutting and of dredging on GHG emissions between years. The Control Shrub and Control Tree microplot sites were left unaltered. The Brush Cut Shrub and Brush Cut Tree sites were brushed in Spring 2018 and Dredged in Fall of 2018, with observations at all sites taking place over 2018-2020 growing seasons. Brushing increased CO₂ emissions at the treed site but had little effect on the shrub site. Dredging decreased CH₄ emissions.

Riparian vegetation has also been shown to obstruct the path for water flow, decreasing water velocities and raising water levels, which increases the ability of ditches to filter and retain nutrients. Simultaneously with the GHG research above, this study also examines the N, P, and C

export from two adjacent watersheds within the South Nation Watershed following the removal of riparian vegetation from one of them. The trials took place over two field seasons and the intervention methods were selected to observe the effects of brush cutting and dredging on N and P export over two years. The southern watershed (Brush Cut) was brushed + dredged in 2018 and the northern watershed (Control) was left intact before flow monitoring took place in 2019 and the Fall of 2020. Tile drain discharge containing DOC, N and P, occurred during the Spring and Fall when the water table is higher, but was not observed during the summer. Brush cutting and dredging increased hydraulic outflow and reduced or eliminated NO_3 retention capacity of agricultural drainage ditches by 320% in 2019 and 68% in Fall 2020. This increase in NO_3 export may negatively affect rural water supplies. Lack of O_2 and increased retention of DOC and SO_4 in the Control watershed suggests that significant NO_3 reduction occurred. Differences in P export between Brush Cut and Control in 2019 were small. There are more signs of P transformation in the Control watershed, but brush cutting and dredging may not significantly affect eutrophication.

This thesis will help inform stakeholders about the environmental geochemical costs and benefits of brushing and dredging so that they can develop BMPs that minimize GHG production and maximize nutrient filtration. Future research is needed to determine how many years the effects of these intervention methods remain, and also determine other environmental impacts such as their effects on biodiversity.

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Dedication

To my parents Alan and Millie Schietzsch, my aunt and uncle Ken and Sonia Younger for their endless love and support.

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List of Abbreviations

AAFC Agriculture and Agri-Foods Canada
ANOVA analysis of variance
BMP best management practice
C carbon
CH₄ methane gas
CH₄-C methane gas as carbon
Cl chloride, filtered water
CO₂ carbon dioxide gas
CO₂-C carbon dioxide as carbon
DASB dry above soil biomass
DIC dissolved inorganic carbon, filtered water
DOC dissolved organic carbon, filtered water
Fe iron
ET evapotranspiration
GHG greenhouse gas
GW groundwater
GWP global warming potential
IC inorganic carbon, soil
N nitrogen
N₂ nitrogen gas
NH₄ ammonium
NH₃-N ammonia as nitrogen
NO₂ dissolved nitrite, filtered water
NO₂-N dissolved nitrite as nitrogen, filtered water
NO₃ dissolved nitrate, filtered water
NO₃-N dissolved nitrate as nitrogen
O oxygen
O₂ oxygen gas
OC organic carbon, soil

P phosphorus

PO₄-P phosphate as phosphorus

PVC polyvinyl chloride

S sulphur

SO₄ dissolved sulphate, filtered water

TP total phosphorus, digestion from unfiltered water sample

WEBS Watershed evaluation of beneficial management practices

Chapter 1: Introduction

1.1 Thesis Organization

This thesis is presented in the research paper format as two distinct studies related to separate aspects of agriculture drainage ditches. The first chapter consists of a general introduction. Chapter 2 describes a study that systematically analyzes the effects of ditch brush cutting and dredging on vegetation, soil nutrients, soil moisture, soil temperature, and GHG emissions. Chapter 3 describes a study that analyzes the effects of brush cutting and dredging on water flow and water chemistry. Chapter 4 concludes this thesis with a summary of findings for both studies, their implications to agriculture and the environment, and suggestions for future directions.

1.2 Background

Agriculture provides many beneficial and essential ecosystem services, such as food, fiber, and bioenergy. Simultaneously with these beneficial services, the conversion of natural ecosystems into heavily modified agricultural ecosystems is a source of disservice through lost biodiversity, increased nutrient runoff, sedimentation of waterways, and pesticide poisoning (Power 2010). Further, 8% of greenhouse gas (GHG) emissions in Canada arise from agriculture (Environment and Climate Change Canada, 2020). The challenge is determining a practical balance between these ecosystem services and disservices, to optimally allocate and manage land use (de Groot et al. 2010).

To understand the effects of agricultural conversion on an ecosystem, we must decide how to model ecosystem services in terms of trade-offs between multiple functions, both economic, sociocultural, and ecological (de Groot et al. 2010). While agriculture provides a direct service as food, agricultural nutrient loading causes significant ecological harm to the Great Lakes. Harmful cyanobacterial algal blooms in Lake Erie can be traced to excessive nutrient loading from agricultural landscapes leading to eutrophication (Stumpf et al. 2012). Wetlands and riparian zones act as a regulating service for nutrient treatment (denitrification),

water regulation (infiltration), natural hazard mitigation (dampening extreme floods), climate regulation, and soil formation.

Although much work has been done on the effect of land conversion for agriculture (Fargione et al. 2008), the focus of this study is on the effects of brush cutting vegetation and dredging the bottom of agricultural drainage ditches. Agricultural drainage modification has been a common practice for hundreds of years and is used to reduce soil saturation and standing water during the growing season, decreasing crop damage and enhancing yields (Strock et al. 2010). In southern Ontario, agricultural drainage is modified through the combined effects of extensive ditching between fields and tile drainage (Van Zandvoort et al. 2017). A common practice to sustain the hydraulic performance of these drainage systems is to periodically remove aquatic and riparian vegetation, with additional dredging and drying during the non-growing season (Castaldelli et al. 2015).

This study investigates the interrelationships among soil moisture and nutrient composition, GHG emissions, channel hydraulic discharge, groundwater, and vegetation recovery. This study takes a comprehensive watershed mass balance approach to evaluate nutrient inputs and outputs. The goal is to improve understanding of costs and benefits related to vegetation brushing and dredging of agricultural drainage ditches. Major components of this study focus on GHG emissions, water chemistry, soil chemistry, and vegetation uptake. Prior studies have shown costs and benefits to vegetation brushing and dredging, but none have used as comprehensive of a process-based approach.

1.2.1 Agricultural GHG Emissions

CO₂ emissions in agricultural soils come from soil and root respiration in equal measures (Oertel et al. 2016).



Soil moisture and T are considered dominant controlling factors. One important consideration when evaluating soil GHG emissions is the amount of water-filled pore space (WFPS). Ideal moisture conditions for CO₂ production are between 0.40 (Schaufler et al. 2010) and 0.60 WFPS (Moyano et al. 2013). Oversaturated soils will decrease CO₂ production due to a lack of O₂ and

undersaturated soils will decrease due to a lack of moisture for soil microbes (Moyano et al. 2013). Quantifying the relationship of soil moisture to CO₂ with field observation in low moisture fine grained soils (Van Zandvoort et al. 2017) is challenging. Temperature has a strong positive correlation with soil CO₂ production as shown in laboratory (Fierer et al. 2003) and field experiments (Nangia et al. 2013).

Methanogenesis (eq. 1.2) and methane oxidation (eq. 1.3) is controlled by O₂ availability and microbial processes (Dutaur and Verchot 2007).



(Pihlatie et al. 2004) show that increasing water saturation can increase N₂O production by four orders of magnitude. Nitrous oxide emissions are produced from a series of redox reactions driven by microbial processes. NH₄⁺ and NH₃ from farm applied fertilizers (Duxbury 1994) oxidize to NO₂⁻ (eq. 1.4, 1.5); NO₂⁻ is subsequently oxidized to NO₃⁻ (eq. 1.6) (Stumm and Morgan 2012).



As the soil WFPS increases above 0.7, denitrification (eq. 1.7) becomes dominant (Bateman and Baggs 2005). The dominant N₂O emissions are produced when NO₃⁻ undergoes reduction processes towards N₂ when WFPS increases towards 0.7. Highest levels of N₂O are expected when water saturation is ≥ 0.6 due to saturated soil micropores (Davidson et al. 2000).

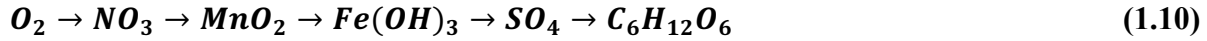
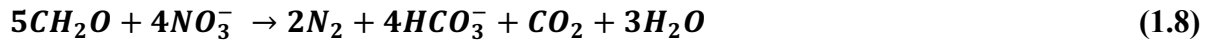


When weighting global warming potential (GWP) of greenhouse gases in terms of CO₂ equivalence, it is important to consider the timescale involved. Both CO₂ and N₂O stay in the atmosphere of Earth for hundreds of years, but CH₄ decays in the atmosphere in a matter of

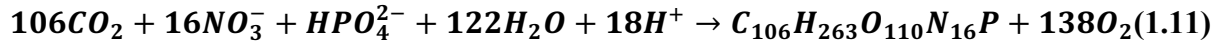
decades (Derwent 2020). In terms of GWP, N₂O is weighted as 273 times more potent than CO₂, and CH₄ is 27-30 times more potent than CO₂ on a 100-year timescale and 83 times more potent than CO₂ on a 20-year timescale (US EPA 2016). Because global warming is currently affecting humans (Lynch et al. 2020), a 20-year CO_{2eq} GWP was used in this study.

1.2.2 Nutrient Discharge

Ammonia in fertilizer not consumed by plant uptake (crops or ditch plants) oxidizes to NO_3^- which is highly mobile. In agriculture NO_3^- , PO_4 , and SO_4 are primarily reduced by the oxidation of organic matter in O_2 -deficient environments (eq. 1.8) (Appelo and Postma 2007). Electron acceptors are reduced from highest to lowest oxidation potential according to eq. 1.10 with O_2 consumption occurring first in a reducing environment and $\text{C}_6\text{H}_{12}\text{O}_6$ is consumed last under highly reducing conditions (Appelo and Postma 2007).



Phosphorus is a limiting nutrient for biological productivity (Spivakov et al. 1999) (eq. 1.11). If more P is available in a waterbody, more primary production biomass is formed. The death of primary producers will reduce the amount of O_2 available to higher order organisms such as fish. The cumulative effect of too much P availability creates large low- O_2 dead zones in downstream waterbodies (Vaccari 2009).



P occurs in a mix of dissolved and particulate components in dynamic equilibrium, with the aqueous phase taking up a smaller fraction. Inorganic P (Orthophosphate), as HPO_4^- and $\text{H}_2\text{PO}_4^{2-}$, is the most soluble form of P in neutral waters. Inorganic P species readily precipitate with Al^{3+} , Fe^{3+} , and Ca^{2+} . Particulate inorganic P is associated with Fe^{3+} complexes and, under reducing conditions, is released as PO_4^{3-} . Organic P in both dissolved and particulate phases undergoes bacterial decomposition to PO_4^{3-} .

1.3 Research Objectives: GHG Emissions

The research presented in this thesis is related to the understanding of GHG emissions in agricultural drainage ditches. Vegetation-rich drainage ditches may be a source of CH_4 emissions

(Dutaur and Verchot 2007), therefore dredging may reduce CH₄ emissions. Vegetation in drainage ditches may act as a CO₂ sink (Nojonen et al. 2013), suggesting that brushing may increase CO₂ emissions. Ditches may be hotspots for CO₂, CH₄ and possibly NO₂.

Specific research questions addressed by this section of the thesis are:

- How are GHG emissions affected by different ditch vegetation microbiomes?
- Do brush cutting and dredging increase ditch GHG emissions in the short and long term?
- What factors such as soil moisture, soil temperature, soil chemistry, influence GHG emissions?

1.4 Research Objectives: Nutrient Discharge

The research presented in this thesis is related to the understanding of the hydraulic flow of nutrients in agricultural drainage ditch systems. Studies indicate N and P discharge from hot spots may be problematic (Pionke et al. 2000; Gitau et al. 2006). The goal of this research on N and P flux is to provide information that will benefit policy makers on the decision to promote enhanced drainage techniques.

Specific research questions addressed by this section of the thesis are:

- How does the timing and composition of the source water flow vary throughout growing seasons?
- How do the practices of brushing and dredging affect hydraulic discharge and cumulative loading of N and P?
- What physical and chemical mechanisms can be attributed to observed changes in hydraulic discharge?

Chapter 2: GHG Emissions Affected by Vegetation Removal in Agricultural Drainage Ditches

2.1 Summary

Vegetation management and dredging of agricultural drainage ditches are practices often necessary to improve field drainage. However, these practices can influence soil greenhouse gas (GHG) emissions in and around the drainage ditches by influencing, for instance, soil/sediment profiles, water/air temperatures, plant nutrient uptake, and hydrology (soil). In this study, surface GHG fluxes (CO_2 , CH_4 , N_2O) were compared between a vegetation brushed + dredged (managed) agricultural drainage ditch and an adjacent ditch that was not brushed or dredged (control), in eastern Ontario, Canada, during three growing seasons (2018-2020). Fluxes were measured on ditch shoulders, midslopes, hyporheic zones, and channel areas. Soil CO_2 emissions increased (15-40%) along the managed ditch after trees were removed, in relation to the control ditch and this increase was likely due to warmer temperatures (3 °C) and increased soil microbial activity because of decreased shading effects. And, moreover, the rapid natural re-establishment of shrubs and grasses after initial woody vegetation brushing did not cause substantial change in fluxes, in relation to time periods immediately following ditch management intervention. In-stream CH_4 emissions after dredging were lower (> 90%). CO_2 and CH_4 were the dominant GHGs fluxes (20-yr $\text{CO}_{2\text{eq}}$) in the riparian areas of the drainage ditches, with N_2O emissions being significantly smaller (1-3%).

2.2 Introduction

Agricultural drainage ditches are typically linear, narrow and incised man-made water courses designed to receive drainage waters from agricultural fields. Drainage ditches are some of the only non-agricultural/semi-natural features in agricultural landscapes (Herzon and Helenius 2008). Thus, they provide a habitat for an array of terrestrial and aquatic flora and fauna. Matter is exchanged readily among adjacent field environments and ditches via runoff and tile drainage (DeSimone et al. 2010; Chen et al. 2018). Water flow in these ditches is usually seasonal, with greater flow occurring in spring and fall and a high probability of flow intermittency in the drier summer months when crop water use is high (Herzon and Helenius 2008). Dosskey et al. (2010) found that the riparian vegetation along ditches can significantly influence stream water quality, through both chemical uptake and by detritus influence on channel soil chemistry. Depending on degree and nature of ditch management, these ditches can in some cases, represent wetland ecosystems in terms of vegetation, hydrology, and organic soil accrual (Dosskey et al. 2010).

Greenhouse gases (GHG) generated from these ditches and along their margins are poorly defined (Brix et al. 2001); as they are often deemed less critical GHG sources than adjacent field environments that receive organic and inorganic fertilizers. Yet, because these areas are subject to wetting and rewetting and are direct receptors of farm drainage and runoff, often laden with agro-chemicals (e.g., chemical and organic fertilizers), these areas should be considered part of the farm-scape in any GHG accounting framework.

N₂O emissions are produced by both denitrification and nitrification processes Hefting et al. (2003), with denitrification found to be responsible for the vast majority of N₂O emissions in riparian zones. Yet DeSimone et al. (2010) found N₂O fluxes were not congruent with higher subsurface N₂O and groundwater nitrate concentrations along riparian zone-field interfaces. Moreover, previous studies suggest that riparian zones can provide trade-offs between nitrate removal from water and increased denitrification. Ollivier et al. (2019) found that in water held behind agriculture dams, N₂O fluxes contributed a small percentage (3.2%) of the total CO₂ equivalent GHG emissions. Outram and Hiscock (2012) found drainage channels in England had 4 times higher N₂O emissions (0.081 kg N₂O-N ha⁻¹ day⁻¹) than a nearby river (0.019 kg N₂O-N ha⁻¹ day⁻¹). A riparian zone may alter the N₂O:N₂ production ratio of the region, because N₂O:N₂ could be affected by factors such as plant, soil, and hydrologic conditions (Groffman et al. 2000).

Converting peat lands into drained landscapes for agriculture increases CO₂ emissions while decreasing CH₄ emissions for a net increase in CO_{2eq} GHGs (Wilson et al. 2016). CH₄ emissions from ditches drained agricultural landscapes with drainage ditches can be comparable to CH₄ emissions from undrained landscapes (Sirin et al. 2012). Peacock et al. (2021a) found that in drained landscapes, ditches dominate the CH₄ budget, outweighing terrestrial CH₄ uptake and that cropland ditches had significantly higher CH₄ emissions than natural land and forest landscapes. Peacock et al. (2021b) found ditch water fluxes of 41 ($\pm$ 17) mg CH₄ m⁻² day⁻¹, and 3640 ($\pm$ 480) mg CO₂ m⁻² day⁻¹ but had insufficient data to disaggregate fluxes by land use or climate zone. Restoration of riparian areas has been linked to increased CH₄ emissions, but without an increased in CO₂ and N₂O emissions (Welsh et al. 2021).

Drivers of CO₂ emissions in riparian zones include soil temperature and possibly plant biomass (Jacinthe et al. 2015). Higher water table depth was found to be a good predictor of CH₄ emissions (Jacinthe et al. 2015). Precipitation events in well drained riparian zones increased CO₂ fluxes and less negative CH₄ fluxes (Vidon et al. 2015). Precipitation events were found to cause short-term bursts of N₂O emissions in riparian zones which faded within 24 hours, but they were not large enough to change the riparian zones from N₂O sinks to N₂O sources (Vidon et al. 2015).

Agricultural drainage ditches and their associated riparian areas can have many costs (farmland value, CH₄, N₂O emissions) and benefits (nutrient removal, carbon sequestration, hydraulic retention, and biodiversity). Therefore, when considering their net value in terms of ecosystem services, it is important to study these kind of systems using a comprehensive approach (de Klein and van der Werf 2014).

The ecosystem services and drainage support for adjacent farmlands are impacted by drainage ditch management, which often includes vegetation removal, vegetation brushing, and channel dredging. The frequency and severity of ditch management depend upon the makeup of the sediment and soils, water-table elevation, vegetation type etc. (Kavanagh et al. 2017). All of these practices have variable impact on the degree and nature of nutrient cycling in and immediately adjacent to the ditch system (Kavanagh et al. 2017).

The goal of this study is to quantify GHG fluxes along agricultural drainage ditches associated with: i) a drainage ditch with >40 yrs of no-dredging management, and ii) a drainage

ditch with vegetation brushing (mowing) and bottom and lower slope dredging. We expect: i.) removal of shading vegetation will increase insolation thereby increasing soil and water temperatures thereby affecting soil microbial activity associated with GHG generation; ii.) dredging of organic sediments/soils will impact redox and nutrient complexing in the ditch and vegetation recovery rates; iii.) improving drainage will increase water outflow, which may lower water levels and soil moisture along banks, thus shifting the drainage ditch from an anaerobic environment to an aerobic environment. We expect that these changes would temporarily decrease CH₄ and N₂O emissions and increase CO₂ emissions until ditch vegetation recover.

2.3 Materials and Methods

2.3.1 Study site

The study site (45.26 N, 75.18 W) is located in eastern Ontario, Canada on two adjacent flat experimental agricultural watersheds (Fig. 2.1) (Wilkes et al. 2014; Sunohara et al. 2015). The experimental watersheds are located in the South Nation river basin (~3900 km²) where there are over 3300 km of drainage ditches (Rideout et al. 2021). The crops are dominated by corn, soybean, forage (hay) field crops and tile drainage overwhelmingly dominates water inputs to the drainage ditches (Sunohara et al. 2015). Both drainage channels were not dredged within 40 years prior to this study and vegetation has since grown inside the ditches, causing them in many places to act as through-flow wetlands (Knox et al. 2008; Sunohara et al. 2015).

From May to September, the 30-year average daily maximum temperature is 22.9 °C, average daily minimum temperature is of 11.0 °C, and total precipitation 490.8 mm (ECCC, 2019). The dominant soil type is classified as Bainsville series (Orthic Humic Gleysol) series (Sunohara et al. 2015).

2.3.2 Watershed Experimental Design

As part of a paired-watershed experiment, the experimental watershed ditches were separated into a reference (northern) watershed of 444 ha and a ditch management treatment (southern) watershed of 225 ha (Fig. 2.1). Sites in the reference watershed are denoted as Control Shrub (Site 10) and Control Tree (Site 11). Sites in the treatment watershed are denoted as Brush Cut Shrub (Site 1) and Brush Cut Tree (Site 5). The denotations refer to dominant vegetation communities along ditch sides at site.

As part of a multi-year before-and-after-control-impact (BACI) experimental design, the ditch bank slope and top of bank of the treatment watershed was brush cut (all vegetation effectively mowed) in winter 2017-18, with the control watershed remaining undisturbed (Fig. 2.1). Site 11 (Control Tree) was left treed, while Site 5 (Brush Cut Tree), was disturbed with the trees and brush cut and mulched. Site 10, (Control Shrub) was left as shrubs and grasses, while Site 1 (Brush Cut Shrub) was brushed of woody vegetation. All sites were monitored in 2018 throughout the growing season, and in November 2018 the treatment ditch underwent dredging, which removed surface soil and sediment and vegetation from the water channel to lower backslope (Fig. A.1).

2.3.3 Greenhouse Gas and Environmental Monitoring

HOBO Weather Station Dataloggers (Onset Computer Corporation, MA) recording relative humidity, air temperature (T), soil T, precipitation, and atmospheric pressure were installed at sites 18, 19, 20, and 21 (Fig. 2.1).

Gas flux measurements followed the non-steady state chamber method outlined in Rochette (2011). Gas flux chamber collars constructed of 41 cm diameter solid PVC pipe were driven into the ground to 20 cm depth. This left 5 cm of head space from the soil surface to the collar top. Lids were constructed of PVC topped with reflective insulation. The lids were vented using 2 mm tubing and sealed to the chambers during deployment with ½” thick memory foam (Dream Solutions USA).

Gas flux measurements were taken at multiple locations at each microplot (Fig. A.2). Two flux chambers were placed on the north shoulder, four chambers on the north bank, and two in stream in 2018. In 2019, flux chambers were placed on the north and south shoulders, as well as northwest, northeast, southeast, and southwest banks of the ditches, and also near the north and south hyporheic zone of each stream. Two chambers were placed in the centre of each ditch: one with holes drilled into it to allow free flow of water, and one with no holes for when water levels are below the height of the holes. In 2020 the Control Tree and Brush Cut Tree microplot sites were not sampled due to COVID-19 constraints to field activities.

To supplement gas flux measurements, CO₂ flux, T, soil moisture measurements, and LI-COR automated flux measurements were used for all bank and shoulder gas flux sampling (Fig. A.2) (LI-8100A, Li-Cor Biosciences, Lincoln, Nebraska, USA). The LI-COR devices were placed upon 8-inch diameter PVC pipe collars. The collars were driven 10 cm into the ground allowing for 5 cm of head space. Each LI-COR station had an ECH2O EC5 5 cm soil moisture sensor and a 8100-203 5 cm Soil Temperature Thermistor Probe, both inserted within 30 cm of the LI-COR collar.

Sample collection was performed over three growing seasons, starting May 15 to October 9, 2018, with a second season from May 23 to September 23, 2019, and a third season July 1 to November 1, 2020. Gas flux, LI-COR, moisture point, and T measurements took place weekly during peak soil respiration times between 10:00 and 14:00. To ensure safety and reduce physical disturbance when traversing the sites during sampling, wooden stairs and bridges were

built and installed. Gas flux measurements involved 25 mL gas samples extracted from each chamber at 5 min intervals (0, 5, 10, 15, 20 min) with 30 mm syringes, then inserted into evacuated 12 mL glass Labco Exetainer® vials with double septa caps. To minimize effects from condensation, 2-3 mg of manganese perchlorate were kept in the vials. LI-COR measurements were automated so that each chamber had two replicates of 4 min CO₂ flux readings with 2 min intervals between replicates. Moisture point and T probe readings were done simultaneously, with T monitored until the instrument stabilized.

Gas samples from the exetainers were analyzed for CO₂, N₂O, and CH₄ at AAFC Ottawa using Agilent method SP1 7890-0467 on Agilent 7890B Gas Chromatograph equipped with special inlet to enable CTC PAL3 autosampler with 2.5mL injection (Carter and Gregorich 2007). Gas data were fit to a linear regression model in MS Excel to calculate gas flux. Gas data was cleaned to remove a small number of concentration measurements that were divergent from acceptable standards (see Appendix A). Gas fluxes that met an acceptable coefficient of determination (R²) standard of >0.8 were retained. Gas flux was calculated using the slope, measured chamber heights, and *in situ* weather station data according to equation 1 derived from Rochette et al. (2008). LI-COR CO₂ fluxes were calculated using a linear model with SoilFluxPro software (LI-COR Biosciences).

$$F_g = \frac{dG}{dt} \times \frac{V}{A} \times M \times \frac{P}{R \cdot T} \quad \text{equation 2.1}$$

F_g is the gas flux

$\frac{dG}{dt}$ is the concentration flux

$\frac{V}{A}$ is the ratio between cylindrical chamber volume and area (chamber depth)

M is the molar mass of the gas

P is the atmospheric pressure during sampling

T is the temperature at the time of sampling

R is the ideal gas constant

Average soil T, soil moisture, and standard error were calculated for each ditch region for each sampling date. To measure effects of soil moisture and soil T on CO₂ flux, multivariate linear regression was calculated using LI-COR data.

We linearly interpolated flux data to estimate cumulative study period mass losses. Cumulative GHG flux 20-year GWP (global warming potential) CO_{2eq} per unit area in each ditch zone was based on 12 hours of flux per day using moving average between two flux values. Cumulative flux per length of ditch was weighted based on a 12 m ditch width: 1 m on each shoulder, 3.5 m on each bank, 0.5 m in each hyporheic zone, 2 m in central stream.

Vegetation samples were collected in paper bags. Bags were weighed and dried at 80 °C in an industrial oven overnight. Soil samples were collected using a JMC Backsaver N-2 Handler sampler which samples 0-30 cm depths. These were divided into shallow 0-15 cm and deeper 15-30 cm portions. Three samples of soil from each depth were frozen. Frozen soil samples were shipped to the University of Guelph Agriculture and Food Laboratory for analysis. Vegetation was sampled throughout the growing seasons, with samples taken in October 2018, June 2019, late July 2019, and in July and November of 2020. Soil was sampled in May, July, August, and October 2019. Soil and vegetation sample locations at each microplot site were on one transect ~25 m upstream and one at ~25 m downstream. For each transect, there were seven sampling locations: north and south shoulders, north and south banks, north and south hyporheic zones, and the center of the ditch. For collecting the vegetation, 1 m by 1 m quadrants were staked off and cut down with gardening shears. Dry vegetation samples were weighed to obtain average dry mass per m² of collection area.

NO₃⁻, NO₂⁻, and NH₄⁺ in water were extracted using 2 M KCL in a 5:1 ratio and analyzed with a Seal AQ2 discrete analyser. Plant available phosphorus was analyzed using the Olsen P method by extracting samples with 0.5 M NaHCO₃ solution and colourimetrically determining P concentration (OMAFRA Soil Fertility Handbook). Total C, total N, and total inorganic C (IC) were analyzed with a CN828 Carbon/Nitrogen (LECO Corp.) by ashing samples at 475 °C for 3 hrs prior to C analysis using catalytic combustion (950 °C) to separate the desired measuring components from foreign gases, then analyzing using thermal conductivity detection and infra-red detection. Organic C was calculated by subtracting IC from the total C result.

ANOVA was used to compare soil M, soil T, and GHG fluxes among treatment sites. When significance (<0.05) was found, a Holm-Sidak paired t-test was used for post hoc analysis to identify significant differences between site pairs. Multivariate linear regression was used to measure correlation between soil moisture and soil temperature, with CO₂ flux being the dependent variable. Correlation of Soil M *versus* CO₂ and Soil T *versus* CO₂ was analyzed by site and ditch region. All statistical tests were computed in SigmaPlot (Systat Software Inc.).

2.4 Results

2.4.1 Vegetation and Soil

The Control Tree site was dominated by Manitoba maple (*Acer negundo*), with a few shrubs (*Rubus idaeus*, *Taraxacum officinale*) and mosses (*Arctium minus*) with dry above soil biomass (DASB) not including trees and larger woody shrubs, were usually below 100 g m^{-2} , peaking at 204 g m^{-2} in July (Fig. 2.2). The Control Shrub site had a variety of vegetation (*Salix* sp., *Typha latifolia*, *Bromus inermis*, *Asclepia syriaca*, *Lythrum salicaria*) which fluctuated in DASB cyclically through multiple growing seasons and had peaks of $\sim 500 \text{ g m}^{-2}$ DASB during summer months and lows of $\sim 200 \text{ g m}^{-2}$ DASB in fall.

By Fall 2018 the bank and shoulder areas of the Brush Cut Tree site had revegetated to 269 and 211 g m^{-2} DASB respectively. The bank and shoulder areas of the Brush Cut Shrub site had revegetated to 491 and 545 g m^{-2} DASB respectively. Subsequently in 2019 and 2020, the brushed sites had peaks of $\sim 500 \text{ g m}^{-2}$ DASB during summer months and lows of $\sim 200 \text{ g m}^{-2}$ in fall.

In October 2018 before dredging, the Brush Cut Shrub site had 1234 and 821 g m^{-2} DASB in the hyporheic zone and stream. In October 2018 the Brush Cut Tree site had 197 and 526 g m^{-2} DASB in the hyporheic zone and stream, compared to the Control Shrub site with 262 and 468 g m^{-2} DASB in the hyporheic zone and stream. In November 2018 the brushed sites were dredged. After dredging the stream and hyporheic zone, vegetative mass dropped to 0 g m^{-2} DASB at the brushed sites. By the Fall of 2020, the instream vegetation DASB at the Brush Cut Shrub (173 g m^{-2}) and Brush Cut Tree site (84 g m^{-2}) had still not recovered to the level of the Control Shrub site (410 g m^{-2}).

The brushed sites had quick DASB recovery by fall after vegetation removal in spring 2018 on the bank and shoulder. By the fall, their bank and shoulder vegetation DASB was very similar to the Control Shrub site. The hyporheic and stream zone DASB of the brushed sites had rapid initial recovery after being scraped bare in fall of 2018. In fall of 2018 the Brush Cut Tree site had a vegetation profile dominated by *Rubus idaeus*, *Solidago canadensis* as well as *Daucus carota*, and *Pastinaca sativa*. The Brush Cut Shrub site had *Bromus inermis* as dominant, as well as *Equisetum arvense*, *Taraxacum officinale*, and *Pastinaca sativa*.

Soil chemistry on the bank and shoulder showed little difference between sites after vegetation brushing, with the site shoulders (places potentially subject to fertilizer drift from

fertilizer activities) having higher N, C, and PO₄-P than the site banks (Fig. A.5 and A.6). NH₄-N values ranged from (0.6-3.3 mg kg⁻¹) with the exception being the bank of the Control Shrub site in spring 2019 (15.7 mg kg⁻¹). Across all sites, similar values of NO₃-N, N, C, IC, and OC were observed. Concentrations were in the range of NO₃-N (1.4-12.8 mg kg⁻¹), N (0.14-0.37 %), C (1.7-5.1 %), IC (0.2-0.4 mg kg⁻¹), OC (1.4-2.6 mg kg⁻¹), and PO₄-P (1.3-3.0 mg kg⁻¹) with obvious seasonal pattern.

Large differences in soil nutrient levels between microplot sites were found in the stream, especially at 0-15 cm depth (Fig. 2.3). The brushed sites were both very similar to each other in regards to nutrient levels. At the brushed sites N and C remained a small fraction (0.1% N, 2% C, and 2 mg kg⁻¹ OC) of levels at both the control sites: the Control Shrub site had a high peak NH₄-N (27.0 mg kg⁻¹) and high peak NO₃-N (11.5 mg kg⁻¹). Plant available phosphorus was much smaller at the brushed sites compared to both the control sites, particularly the Control Tree site with a PO₄-P value of 11.6 mg kg⁻¹. The hyporheic zones of all four microplots showed similar soil chemistry changes to those observed in the stream, but at lesser magnitude (Fig. A.4).

2.4.2 GHG Fluxes

There were clear differences in LI-COR data between sites (Fig. 2.4). An ANOVA analysis on the LI-COR Bank data found significant differences across all four sites in mean CO₂ flux, soil moisture, and soil T. A paired t-test on soil T showed the Control Tree (18±1 °C) site was significantly cooler (P < 0.05) than the other sites by ~3 °C. Mean (SE) soil moisture of the Brush Cut Tree (0.154±0.01) and Control Shrub (0.191±0.01) sites was significantly lower (P<0.01) than the Brush Cut Shrub (0.228±0.01) and Control Tree (0.233±0.01) sites. CO₂-C emissions of the Brush Cut Tree site (65±4 kg ha⁻¹ day⁻¹) were significantly higher (P<0.05) than the Control Tree site (38±2 kg ha⁻¹ day⁻¹), and was consistently higher throughout the growing season except for a small CO₂ spike in the Control Tree site in October and November. CO₂-C emissions of the Brush Cut Shrub site (52±4 kg ha⁻¹ day⁻¹) were significantly higher (P<0.05) than the Control Shrub site (42±2 kg ha⁻¹ day⁻¹). Similar trends were observed in the Shoulder data (Fig. A.7).

CO₂, CH₄, and N₂O fluxes obtained from the vial method varied between sites and between ditch zones (Fig. 2.5). CO₂ Fluxes were higher on the shoulder than the ditch bank and hyporheic zones, with stream bed values having the lowest mean and minimum values. CH₄ fluxes on the shoulder and ditch bank were most often between -0.001 and -0.01 kg ha⁻¹ d⁻¹ CH₄-

C. The hyporheic zone had larger positive fluxes up to $0.1 \text{ kg ha}^{-1} \text{ d}^{-1} \text{ CH}_4\text{-C}$. The stream had significantly higher $\text{CH}_4\text{-C}$ fluxes up to $6 \text{ kg ha}^{-1} \text{ d}^{-1}$ at the Control Shrub site. N_2O flux values usually remained below $0.05 \text{ kg ha}^{-1} \text{ d}^{-1}$, with peaks up to $0.20 \text{ kg ha}^{-1} \text{ d}^{-1}$ at the shoulder of the Control Tree site. 2018 fluxes (Fig. A.8) and 2020 fluxes (Fig. A.9) showed similar GHG fluxes but with large CH_4 fluxes in the Stream Bed observed at different sites. CH_4 flux was very low ($< 0 \text{ kg ha}^{-1} \text{ d}^{-1} \text{ CH}_4\text{-C}$) as was N_2O flux ($< 0.05 \text{ kg ha}^{-1} \text{ d}^{-1} \text{ N}_2\text{O-N}$), except for occasional spikes in the stream and hyporheic zones which created non normally distributed flux distributions (skew 1.60-2.54, kurtosis 1.21-6.30) which were statistically limited.

LI-COR CO_2 flux measurements were more consistent than vial method values because we were able to keep values within $R^2 > 0.95$. Flux values obtained with vial method corresponded well with LI-COR CO_2 flux measurements which were 10-30% higher than those measured with the vial method for all sites, except for the Brush Cut Tree site (Fig. A.12). The Brush Cut Tree site had little correspondence between the LI-COR and Vial Method with the LI-COR fluxes.

In 2018 cumulative CO_2 production within Bank and Shoulder zones was higher at the Bush Cut Shrub and Brush Cut Tree sites than at the Control sites (Fig. 2.6). In 2019 and 2020 the Brush Cut Shrub site cumulative CO_2 production was similar to the Control Shrub site, but the Brush Cut Tree site still had elevated CO_2 production. All sites had greater CO_2 production than the Control Tree site except during October and November of 2020.

ANOVA data showed significant vial CO_2 flux differences between different sites across the Shoulder, Bank, and Hyporheic zones. Paired T-Tests showed that the Brush Cut Tree site on the Bank zone, and Brush Cut Shrub site on the Bank and Shoulder zones had significantly greater CO_2 flux than the Control Tree site. The Brush Cut Shrub site had significantly greater CO_2 flux than Control Shrub site on the Bank zone. Paired T-Tests showed the Control Shrub site had significantly greater CO_2 flux than the Control Tree site on the shoulder, bank, and hyporheic zones.

Multivariate linear regression (Table A.1) shows soil CO_2 production has a strong correlation of with soil T at 5 cm depth ($p < 0.05$) but not Soil M at 5 cm depth ($p = 0.825$) with an R^2 value of 0.178. Soil CO_2 production was positively correlated with soil T on banks and shoulders of all sites except the Control Tree site shoulder (Fig. A.11). CO_2 production was

positively and negatively correlated with soil moisture at different microplot sites and ditch regions with no distinguishable pattern (Fig. A.10).

CO₂ was responsible for the vast majority of GHG emissions in the agricultural drainage ditch sites (Fig. 2.7), but with significant contribution from CH₄ production in the stream bed. N₂O production was a minimal percentage (1-3%) of total CO₂ equivalent GHGs. After brushing vegetation in spring 2018, CO₂ production increased by 15% at the Brush Cut Tree site and 30% at the Brush Cut Shrub site. In 2019 and 2020 the brushed sites had similar CO₂ flux to the Control Shrub site. CO₂ production at the Control Tree site was typically 15-60% lower than at other sites. CH₄ production at the brushed sites decreased after ditch dredging (Fig. 2.7). At the Brush Cut Shrub site CH₄ production dropped from 45 to 0.5 kg km⁻¹ d⁻¹ from 2018 to 2019. At the Brush Cut Tree site CH₄ production dropped from 15 to 0.3 kg km⁻¹ d⁻¹ from 2018 to 2019. In 2020 CH₄ production did not recover at the brushed sites.

2.5 Discussion

2.5.1 Soil and Vegetation

Vegetation was re-established on the Bank and Shoulder zones (October 2018) in one season post-vegetation brushing (May 2018) (Fig. 2.2). The re-established vegetation largely consisted of grasses (*B. inermis*, *E. arvense*), shrubs (*R. idaeus*, *S. canadensis*), and ferns (*E. arvense*). Increased insolation was likely the main factor for the re-establishment of plants at these sites post intervention. This has been shown in other studies (Conner 1995; Noponen et al. 2013).

The slow vegetation recovery observed in the Stream and Hyporheic zones of the brushed sites after dredging (November 2018) can be attributed primarily to physical causes. The more organic rich soils/sediment that were dredged exposed a tight varved silt clay substrate which could make it more difficult for hydrophilic plants to establish. The upper 0-15 cm layers at the brushed sites were relatively lower in key nutrients (NH_4^+ , OC, and PO_4^{3-}), and 15-30 cm layer partially lower compared to the control sites (Fig. A.1), verifying that the dredging process stripped ~20 cm of soil/sediment from the Stream and Hyporheic zones (Fig. 2.3). Mechanical constraints on plant growth coupled with substrates with less nutrients, it is unsurprising that vegetation establishment was constrained during the 2019 and 2020 growing seasons (Vidon et al. 2014).

2.5.2 CO₂

CO₂ production is expected to increase for soils rich with OC and in oxic environments due to soil and root respiration (Oertel et al. 2016). CO₂ production can be mitigated by vegetation coverage that consumes a portion of CO₂ production due to photosynthesis and roots (Berisso et al. 2013). The general trend of increased CO₂ flux going from deeper ditch zones near the water table to higher ditch zones next to farm fields is consistent with the hypothesis that more oxygen rich soils promote greater decomposition activity (Moyano et al. 2013; Oertel et al. 2016). Higher CO₂ fluxes on the shoulder zones compared to bank zones may be due to higher OC content on the shoulder. In terrestrial zones environmental conditions that can lead to increased CO₂ production are increased soil T (Schipper et al. 2014) and increased soil moisture (Moyano et al. 2013). Under high soil moisture conditions (WFPS > 0.6) increased soil moisture is expected to be negatively correlated with CO₂ flux.

Multivariate linear regression shows a strong correlation (R^2 : 0.178, $p < 0.05$) between CO_2 production and soil T which is slightly lower than Jacinthe et al. (2015) (R^2 : 0.24-0.35, $p < 0.05$). All bank and shoulder soil-moisture measurements were below 0.4 WFPS and no significant correlation between soil moisture and CO_2 flux was observed. This observation may be due to the LI-COR ECH2O EC5 soil-moisture probe only probing the first 5 cm of soil, which suggests that soil and root CO_2 production is occurring below 5 cm (Blume et al. 2022).

The LI-COR CO_2 fluxes were 10-30% higher than vial method fluxes, likely because of photosynthesis within the GHG vial chambers. Vegetation was removed by hand from LI-COR chamber collars, but shrubs and smaller vegetation remained inside GHG vial chamber collars. The LI-COR method sampled mainly soil respiration and the gas vial method included CO_2 generated by photosynthesis and root exudates.

Removing shrubs and trees from the bank and shoulders resulted in an initial increase in CO_2 production compared to control sites in 2018. This increase can be attributed to decomposition of dead vegetation remaining from the clearing process, and more insolation increasing microbial activity. Due to revegetation 2019 and 2020, bank and shoulder brushed site CO_2 fluxes mirrored the Control Shrub site. The Control Tree site had lower CO_2 fluxes than other sites. There were few differences in bank and shoulder soil OC content, and soil moisture, so this finding is likely due to Norway Maple trees shading the forest floor, thus lowering soil temperatures, which caused a corresponding reduction in CO_2 flux. When the stream water levels were high, low in-stream CO_2 flux was observed, as O_2 diffusion from the water dominated. When the stream water receded, soil respiration similar to the bank was observed.

2.5.3 CH_4

Small amounts of CH_4 consumption occurred throughout the bank and shoulders, indicating methane oxidation (Dutaur and Verchot 2007). Methanotrophy and CO_2 production often occur together in O_2 rich environments (Maier and Schack-Kirchner 2014) such as the lower soil-moisture bank and shoulder zones compared to higher soil-moisture zones near the centre of the stream. Because of the low CH_4 consumption, this will have limited effect on the total GHG budget. The positive CH_4 fluxes at the stream and hyporheic zones locations can be attributed to methanogenesis (Dutaur and Verchot 2007) due to the presence of an anaerobic environment in the stream area. The large spikes up to $8 \text{ kg ha}^{-1} \text{ d}^{-1} \text{ CH}_4\text{-C}$ in the stream may be due to methane

ebullition rather than diffusion (Whalen 2005). In the years after dredging, CH₄ fluxes dropped significantly at the brushed sites. This change is likely both due to drops in water table (Jacinthe et al. 2015) and soil OC associated with dredging.

2.5.4 N₂O

High N₂O production is expected to occur in partially anaerobic environments. When plants do not consume all available NH₄⁺, the excess NH₄⁺ oxidizes to NO₃⁻ which then denitrifies to N₂ (Stumm and Morgan 2012), with N₂O an immediate by-product. High N₂O production on the shoulder of the Control Tree site can be attributed to smaller berms at the site compared to others and insufficient grassy vegetation to consume excess NH₄⁺ that has run off the adjacent farm fields. The bank and shoulder zones usually had sufficient vegetation to consume fertilizer drift from field applications of fertilizer. Low bank and shoulder N₂O emissions agrees with Vidon et al. (2015). Higher N₂O production also was measured in-stream due to the anaerobic environment, and the ditch is a receptor of N from tile drain inputs (Sunohara et al. 2015). This was not present in all streams since it requires fertilizer input to exceed the N plant uptake capacity. N₂O emissions remaining a low percentage of CO_{2eq} GHGs agrees with Ollivier et al. (2019).

2.6 Conclusions

This study examined vegetation DASB, soil chemistry, and GHG flux over three growing seasons at sites representing extreme cases in terms of agricultural drainage ditch management; one watershed was left unmanaged, and the other with aggressive and complete vegetation brushing along both banks and dredging of bottom and mid slope soils/sediment. Extreme cases that can be used to contrast intermediate conditions as systems equilibrate and rebound from disturbance. As well we studied two contrasting vegetation environments along the ditches: 1. grass/woody shrubs, 2. mature large trees along banks, to determine if one environment was more sensitive to disturbance in terms of GHG than the other which also characterized the fluxes in the different botanically distinct systems along the ditch. The following conclusions can be made:

i) Of the two control watershed sites, the Control Shrub site had higher total GHG emissions ($130 \text{ kg km}^{-1} \text{ d}^{-1}$) than the Control Tree site ($50\text{-}90 \text{ kg km}^{-1} \text{ d}^{-1}$) over the three seasons. This difference can be attributed to the increased shade of large trees lowering soil T and decreasing microbial respiration and CO_2 emissions. The CO_2 emissions from the Control Tree site were 30 kg km^{-1} (15-40%) lower than the CO_2 emissions from Control Shrub.

ii) Impacts of vegetation brushing for the sites dominated by shrubs and grasses had minimal effect on soil nutrient concentrations and GHG emissions. Furthermore, vegetation DASB recovered to pre-intervention values after one season; however the vegetation community was in a different state of succession in relation to pre-intervention conditions.

iii) Dredging the ditches removed aquatic plants and $>20 \text{ cm}$ of organic sediments, which reduced average ditch CH_4 emissions from $15\text{-}50 \text{ kg km}^{-1} \text{ d}^{-1} \text{ CO}_{2\text{eq}}$ to less than $0.5 \text{ kg km}^{-1} \text{ d}^{-1} \text{ CO}_{2\text{eq}}$ over the course of a season. Stream revegetation was still minimal after two years post-dredging likely as a result of the clay rich substrate, which restricted plant establishment. Moreover, CH_4 emission remained low ($<0.5 \text{ kg km}^{-1} \text{ d}^{-1} \text{ CO}_{2\text{eq}}$).

iv) In terms of total 20 year $\text{CO}_{2\text{eq}}$ emissions, CO_2 was the dominant GHG (60-95%) in agricultural drainage ditches with CH_4 also contributing (1-40%). Effects from N_2O were comparatively small (1-3%).

An optimized method to dredging and vegetation clearing should be considered in context of biodiversity and shading potential on adjacent crops. This study helps determine the impacts of

dredging and brushing, so that C sequestration and GHG emission effects can be considered against other resulting environmental impacts including biodiversity, water quality, and wildlife habitat.

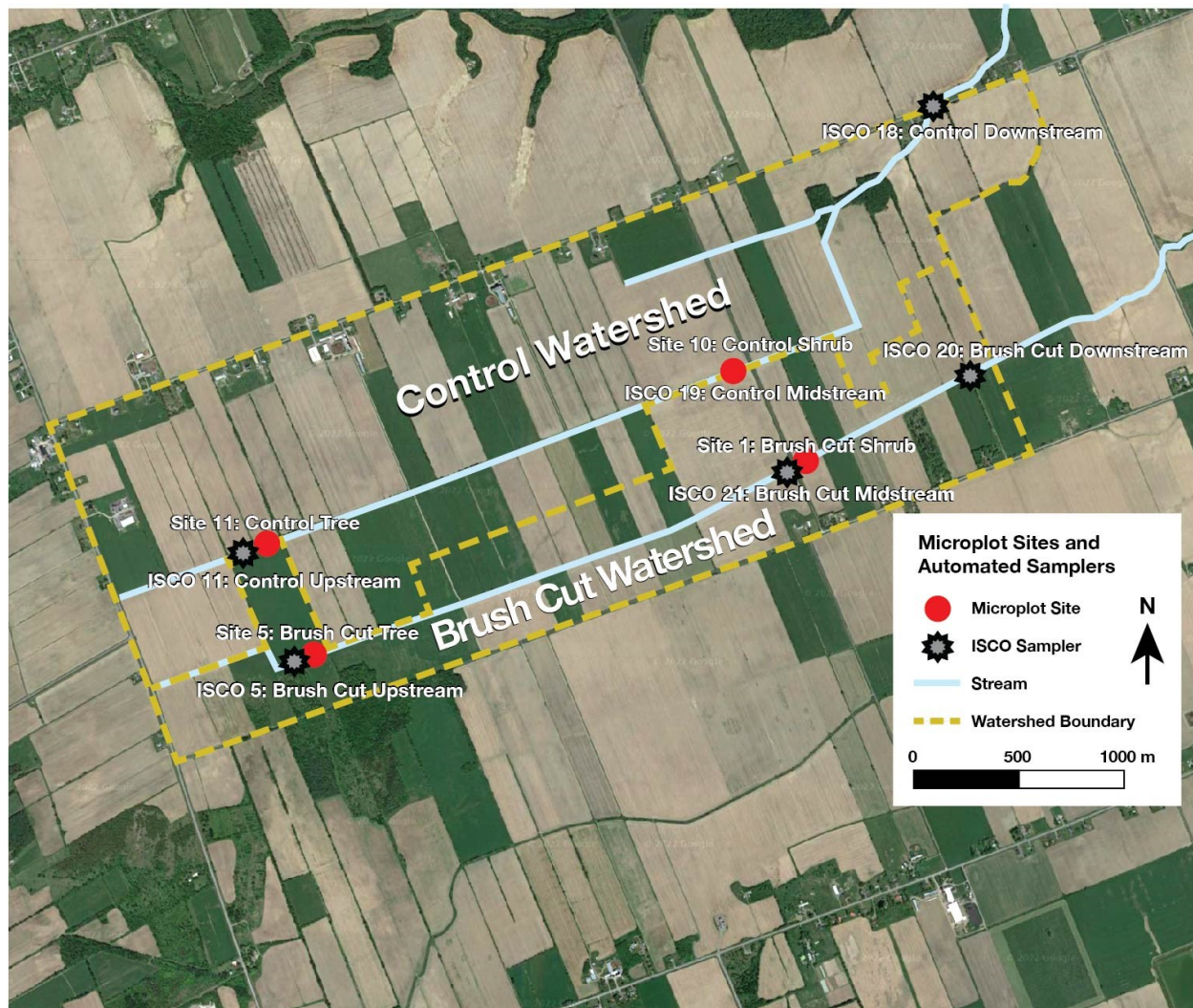


Fig. 5.1 Overhead view of WEBS field site (45.26 N, 75.18 W) showing location of Microplot GHG monitoring sites and flow.

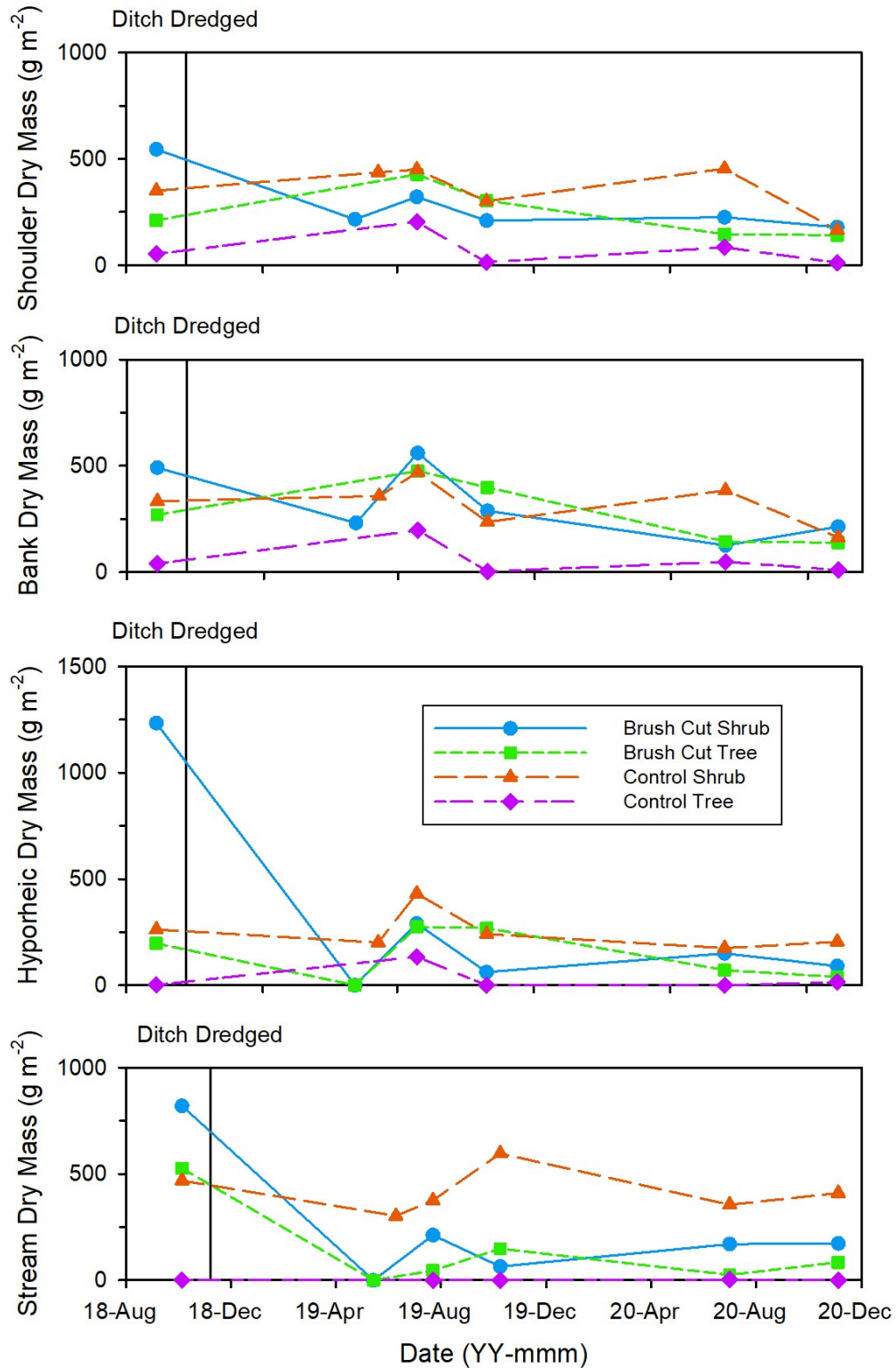


Fig. 5.2 Temporal changes of dry mass of vegetation collected over two years from 1 m² plots at Brush Cut Shrub •, Brush Cut Tree ■, Control Shrub ▲, Control Tree ◆. Vertical line represents ditch dredged event November 2018. Ditch zone from top to bottom Shoulder, Bank, Hyporheic, Instream.

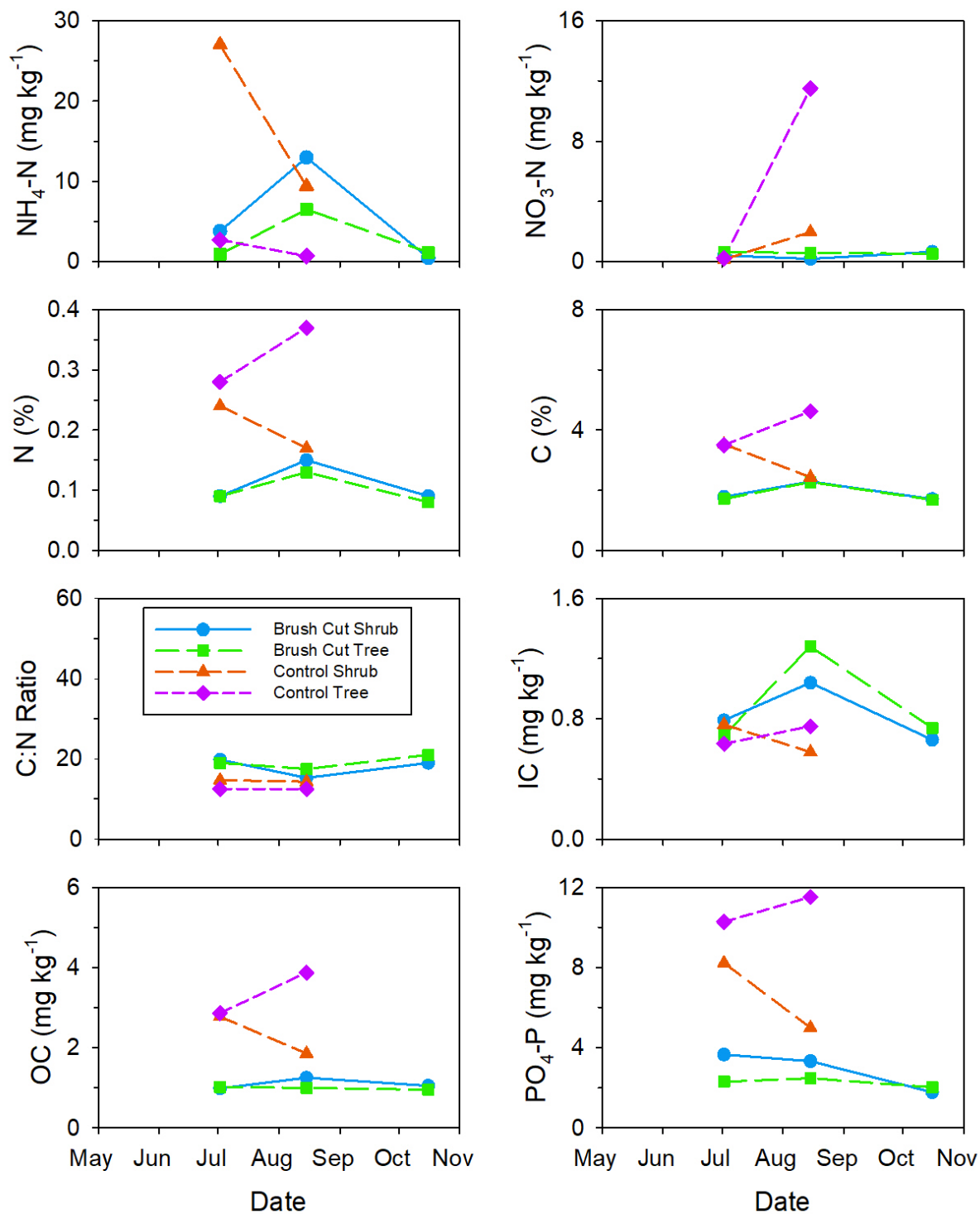


Fig. 5.3 Temporal changes in Instream soil chemical composition for 2019 at 0-15 cm depth for Brush Cut Shrub •, Brush Cut Tree ■, Control Grasses ▲, Control Tree ♦. From top row to bottom row: $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, N, C, C:N ratio, inorganic carbon (IC), organic carbon (OC), plant available phosphorus ($\text{PO}_4\text{-P}$).

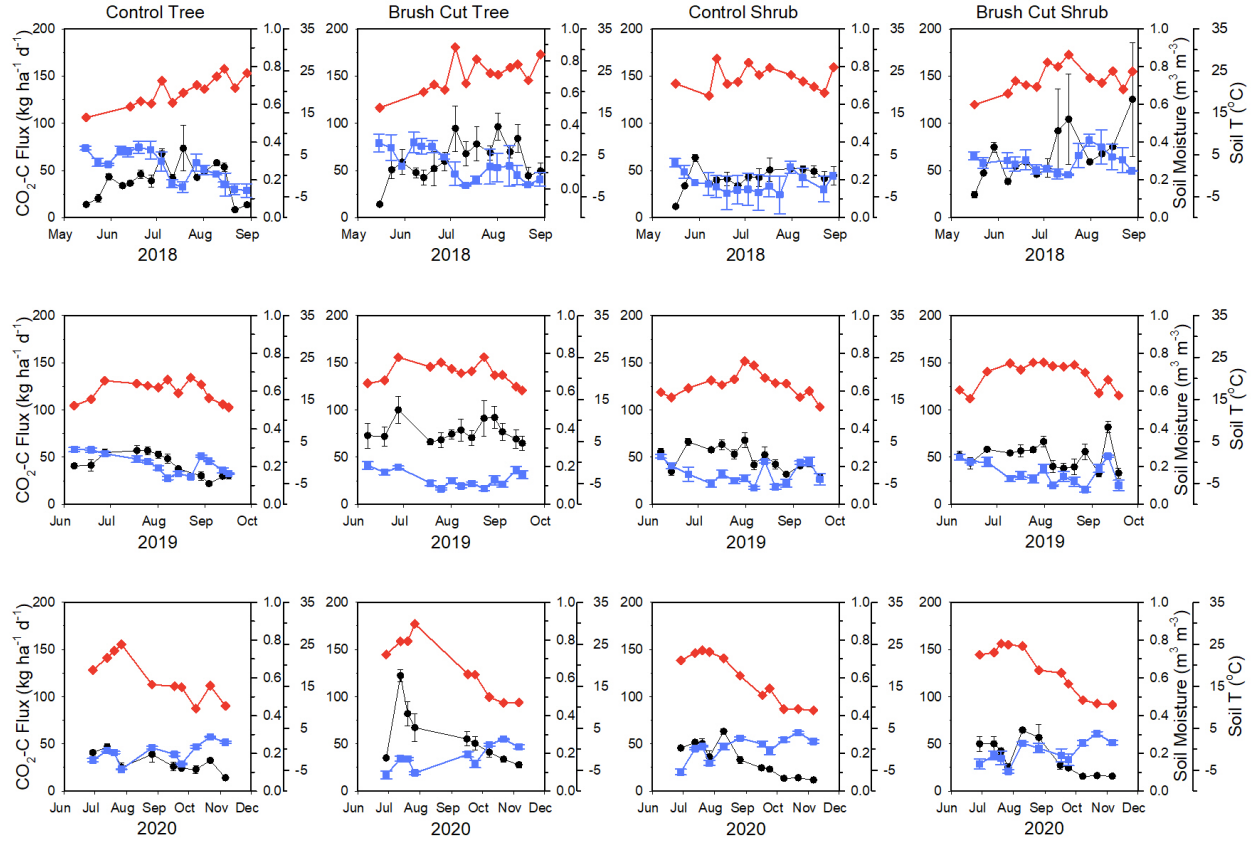


Fig. 5.4 CO₂ Flux •, Soil Moisture ■, and Soil T ♦ time series with calculated standard error bars from LI-COR measurements in agricultural drainage ditch Bank zone. From top to bottom 2018, 2019, and 2020 sampling seasons. From left to right Control Tree, Brush Cut Tree, Control Shrub, Brush Cut Shrub sites.

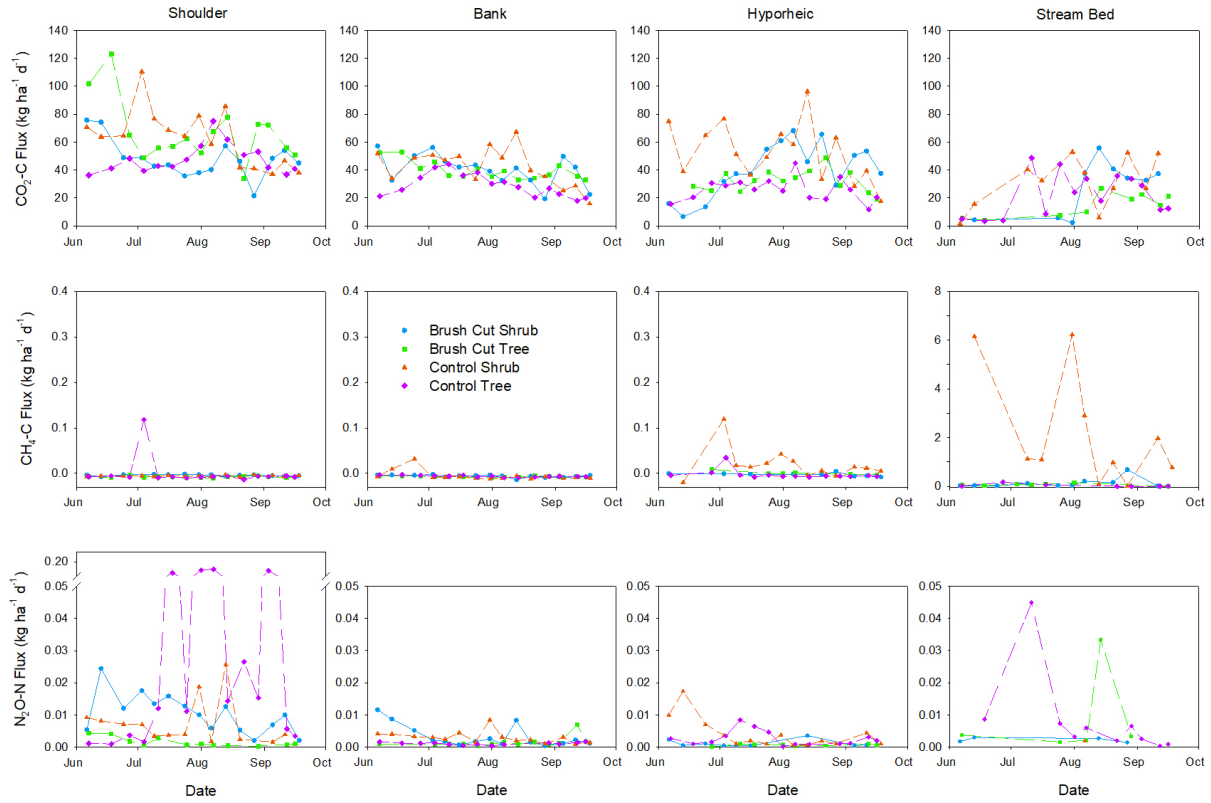


Fig. 5.5 Time Series of CO₂, CH₄, N₂O Fluxes from June until September 2019 for Brush Cut Shrub •, Brush Cut Tree ■, Control Shrub ▲, Control Tree ◆. Only flux with R²>0.8 included. Note different scale for CH₄ flux from stream bed. From top to bottom: CO₂-C, CH₄-C, N₂O-N fluxes. From left to right Shoulder, Bank, Hyporheic, Instream zones.

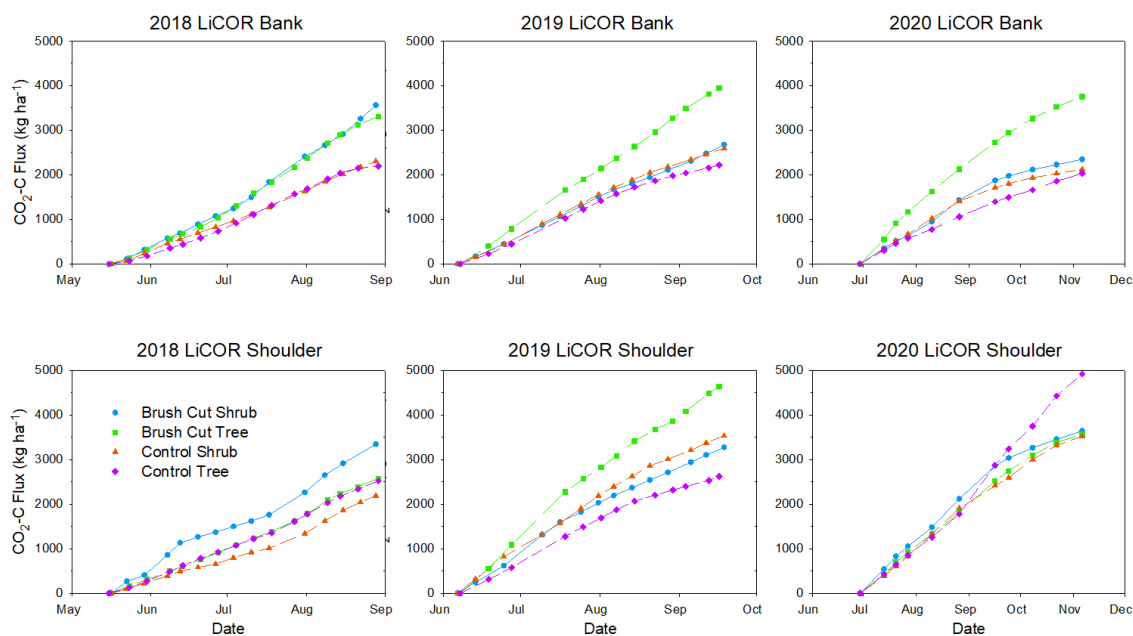


Fig. 5.6 Cumulative CO₂ emissions time series from LI-COR measurements over 2018, 2019, and 2020 sampling seasons for Brush Cut Shrub ●, Brush Cut Tree ■, Control Shrub ▲, Control Tree ◆. From top to bottom Bank and Shoulder zones. From left to right 2018, 2019, and 2020.

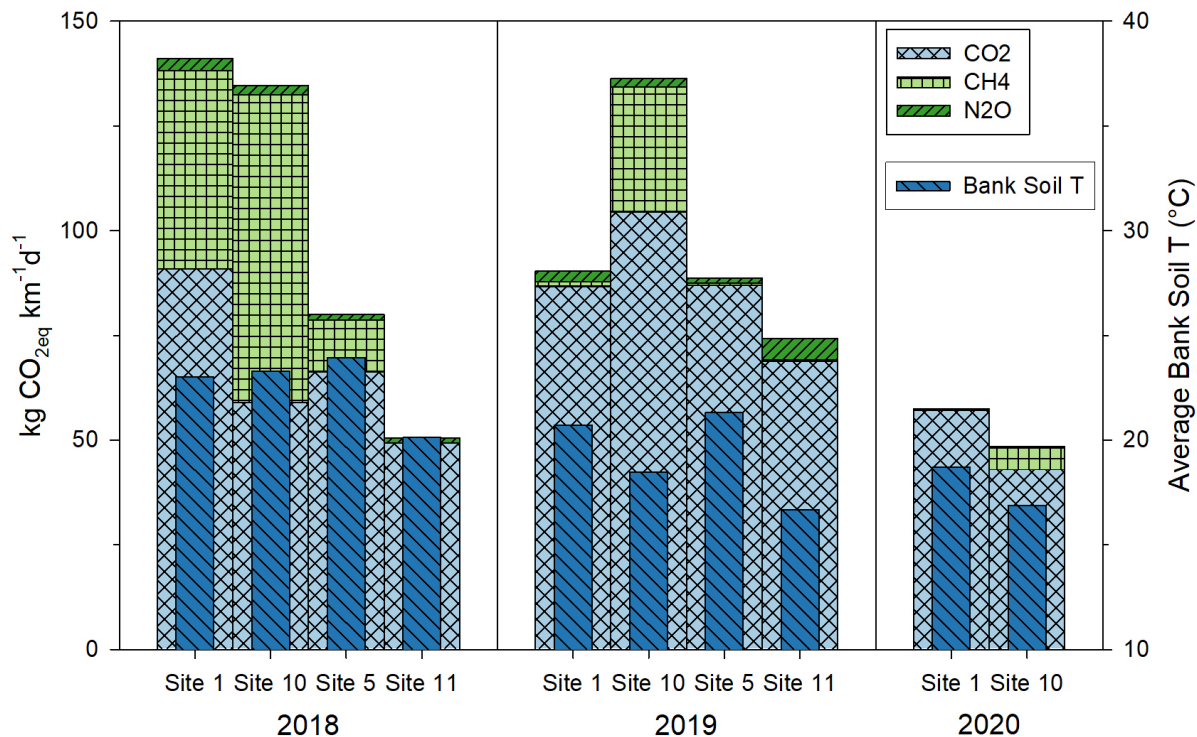


Fig. 5.7 Mean GHG flux CO₂, CH₄, N₂O fluxes reported as 20-year GWP CO₂eq per km of drainage ditch over sampling seasons 2018, 2019, 2020. Weighting based upon ditch cross section area. Site 1 (Brush Cut Shrub), Site 10 (Control Shrub), Site 5 (Brush Cut Tree), Site 11 (Control Tree). Average bank soil T over sampling season.

Chapter 3: Long-Term Observations of Nitrogen and Phosphorus Export in Paired-Tile Drained Agricultural Watersheds under Enhanced Vegetation Removal and Intact Riparian Zone

3.1 Summary

Vegetation management and dredging of agricultural drainage ditches are practices deployed to improve field drainage. However, these practices remove ditch vegetation and soil/sediment nutrients, which can influence N and P export. In this study, export of agricultural nutrients (N, P, C) from a vegetation brushed + dredged drainage ditch in eastern Ontario, Canada during two growing seasons was compared to an adjacent agricultural drainage ditch (Control) which was not brushed or dredged. Water flow and nutrient concentrations were measured at instream sites and in tile drainage throughout the growing season in 2019 and during the Fall of 2020. Tile-drain discharge containing DOC, N, and P occurred during the Spring and Fall, but was not observed during Summer, when the water table was lower. Brush cutting and dredging increased hydraulic outflow and reduced or eliminated the NO_3 retention capacity of agricultural drainage ditches by 320% in 2019 and 68% in Fall 2020. This increase in NO_3 export may negatively affect rural water supplies. Lack of O_2 and increased retention of DOC and SO_4 in the Control watershed suggests that significant NO_3 reduction occurred. There is more PO_4^{3-} the Control watershed indicating more P transformation, however, differences in total P export between Brush Cut and Control in 2019 were small, indicating that brush cutting and dredging do not appear to significantly affect the potential for eutrophication. Measurement of NO_3^- isotope ratios indicate there were likely two distinct sources (8 ‰ and 15 ‰ $\delta^{15}\text{N}$) of NO_3^- flowing from different tile drains which makes it difficult to draw conclusions about denitrification.

3.2 Introduction

Agriculture is a major source of non-point source pollution. Nitrogen (N) and phosphorus (P) mitigation in agriculturally intensive areas is important in terms of critical source area management. Although P and N both discharge to downstream water bodies, P is more often a limiting nutrient in biological productivity (Spivakov et al. 1999) causing large scale eutrophication in lakes and oceans (Rabalais et al. 2002). N discharge is a concern as nitrate (NO_3^-) contaminates rural and municipal drinking water (Kundu and Mandal 2009). A critical source management approach can provide a cost effective strategy to efficiently manage a large-scale pollution problem (Gitau et al. 2006). For example less than 20% of the watershed area can contribute to over 80% of nutrient loads in the Chesapeake Bay watershed (Pionke et al. 2000).

Artificial subsurface (tile) drainage systems are used to improve field drainage for crop production, but these systems also create efficient pathways for transfer of nutrients applied to fields to groundwater and surface water (Lapen et al. 2008). These ditches are long linear features that also effectively transfer nutrients to downstream waterbodies (Rideout et al. 2021). Tile drains effectively transfer excess water, but also inadvertently direct nutrients into agriculture drainage ditches (Lapen et al. 2008). Vegetation on the bank and shoulder of the ditches is commonly brush cut and the stream bed is dredged to enhance drainage-ditch performance (Rideout et al. 2021).

The removal of wetlands and riparian zones limits nutrient trapping, resulting in greater transfer of nutrients to surface waterways. The N removal capacity of high-water retention areas such as wetlands and riparian zones is due both to chemical reactions such as denitrification, and to nutrient retention (soil formation, plant uptake) (Strand and Weisner 2013). Vegetated canals buffer NO_3 pollution via microbial denitrification, and vegetation supports the activity of bacteria (Castaldelli et al. 2015). In many regions of the world, removing wetlands and riparian zones has resulted in large-scale eutrophication downstream of agricultural areas (Vitousek et al. 1997; Howarth et al. 2002). Vymazal and Březinová (2018) reported agricultural drainage ditches to be comparable with constructed wetlands for treating agricultural drainage, with plant uptake responsible for 26.3% of removed N and 14% of removed P.

Some P runoff loss is inevitable, but we can distinguish between acceptable and unacceptable losses of P (Kleinman et al. 2011). Runoff PO_4^{3-} concentration is highly correlated

with water soluble P concentration of surface-applied manure fertilizer. P is released as particulate matter from soil-surface erosion (Brady and Weil 2008) and dissolved P is released from surface runoff (Ryden et al. 1974). Mixing of manure with soil (Kleinman et al. 2002) can assist in mitigation of particulate P release. Concentrations of dissolved P in runoff can be moderated by sorption of P by stream sediments and by biological uptake by aquatic plants and algae (Sims et al. 1998).

Whereas nutrient retention varies significantly within and between agricultural systems, it is dependent on mass loading and plant species of riparian buffer habitat (Strand and Weisner 2013; Fortier et al. 2015). Fortier et al. (2015) found that hybrid poplar woodland buffers retain more C, N, and P storage than comparable herbaceous buffers. Nutrient removal performance in riparian buffers is a function of water retention time, with low steady flow conditions, where retention times exceed 1 to 2 weeks, are optimal for nutrient removal (Woltemade 2000). Ahiablame et al. (2011) found that drainage ditches transported nutrients downstream without significant attenuation of P and NO_3 . Dredging of agricultural ditches decreases surface roughness and removes plant species with high rates of nutrient uptake (Smith and Pappas 2007), disrupting the removal of $\text{NO}_3\text{-N}$, $\text{NH}_4\text{-N}$, and soluble P.

Common ditch intervention practices to improve drainage include brushing of banks and bottom-only cleanout (Kavanagh et al. 2017). Brushing is the practice of cutting and shearing vegetation using large mowers and chainsaws while leaving vegetation root systems intact. Removal of vegetation improves water flow. Bottom-only cleanout is dredging of sediments and vegetation at the bottom of the drain only. This process leaves the bank vegetation intact, without the need for stabilizing the banks and preserves riparian fish habitat along the sides and top of the bank.

Vegetated ditches have lower flow thus longer residence times, which allows for greater microbial growth (Castaldelli et al. 2015; Cui et al. 2020) and plant uptake of nutrients (Vymazal and Březinová 2018). Sites with greater vegetative coverage had lower sediment and nutrient runoff with runoff being greater in winter when crops are not on the field (Martin et al. 2021). Iseyemi et al. (2016) found brushed ditches and non-brushed ditches had similar nutrient filtration capacity at the same residence time but brushing decreased residence time. Intense

storm events (which are becoming more frequent;) hinder ditch nutrient removal due higher flow and shorter residence times (Faust et al. 2018).

In addition to the effects of brush cutting, removing vegetation through dredging removes soil nutrients. Moore et al. (2010) found both vegetated and maintained dredged ditches were effective in removing agricultural nutrient loads, but regularly dredged ditches had better P removal than vegetated ditches, due to better sediment P sorption ability. Fu et al. (2014) found both constructed ecological ditches (gravel + geomembrane) and ditches with natural vegetation had nutrient mitigation capacity, but the constructed ecological ditches improved the nutrient mitigation capacity. In the 12 months after dredging in Indiana US, ditch nutrient retention capacity improved, but increased hydraulic outflow was only temporary, 1 month (Smith and Huang, 2010). Smith and Huang (2010) suggest that rapid regrowth in the disturbed area the year after dredging provides greater opportunity for nutrients to be transformed (i.e., nitrification/denitrification), adsorbed, or biologically removed from the water column than prior to dredging.

The purpose of this research was to determine the influence of dredging and vegetation brushing in drainage ditches imposed on an experimental paired watershed in eastern Ontario, Canada, on stream water flow and mass fluxes of N, and P, during typical growing seasons. We hypothesize that brushing and dredging lowers the surface roughness, providing an unobstructed path for water flow, increasing water velocities and lowering water levels. Residence times of nutrient-rich waters before downstream export will decrease.

Stream water flow, N, and P fluxes as well as mean concentrations over the growing season from two (paired) experimental watersheds were compared. The treatment watershed (Brush Cut) had ditch shoulders and banks brush cut in Spring of 2018 and a bottom-only cleanout in Fall of 2018. The ditch riparian zone has remained intact in the reference watersheds (Control) for over 30 years, with a mixture of treed ecosystems and shrub dominated ecosystems. The treed ecosystems were dominated by Manitoba Maple (*Acer negundo*) and the shrub-dominated areas by (*Salix* sp., *Typha latifolia*, *Bromus inermis*, *Asclepia syriaca*, *Lythrum salicaria*).

3.3 Materials and Methods

3.3.1 Study site

The study site is located on two flat experimental agricultural watersheds (45.26 N, 75.18 W) near the town of St. Albert in Eastern Ontario, Canada (Fig. 2.1) (Sunohara et al. 2015) (Wilkes et al. 2014). The watersheds are part of Watershed Evaluation of Beneficial management practices (WEBs), a long-term environmental monitoring program (Agriculture and Agri-foods Canada, 2013). Water flow and drainage water quality were monitored in paired drainage channels located in the South Nation watershed, where there are roughly 3300 km of drainage ditches in 3900 km² (Rideout et al. 2021). The study area is dominated by field crops (85%). Tile drainage dominates water inputs to the drainage channels. Both drainage channels were not dredged for over 30 years prior to this study. Vegetation had grown inside the ditches, causing them to act as inline wetlands (Sunohara et al. 2015).

From May to September, the 30-year average daily maximum temperature was 22.9 °C, average daily minimum temperature 11.0 °C, and average total annual precipitation 490.8 mm (ECCC, 2019). The dominant soil type is classified as Bainsville (Orthic Humic Gleysol) series (Sunohara et al. 2015). Field crops in the watershed are predominately corn, soybeans, and grass/alfalfa forage crops. The drainage ditches flow predominately in spring and fall, and flow is intermittent in the drier summer months when water uptake by plants/crops is greatest.

3.3.2 Watershed experimental design

As part of a multi-year before-and-after-control-impact (BACI) experimental design, in May 2018, the ditch bank slope and top of bank of the treatment watershed (Brush Cut) were brush cut, with the reference watershed (Control) remaining intact. In November 2018 the treatment stream underwent a bottom only cleanout, which removed surface soil and vegetation from the water channel area. The channel sediments were distributed on the adjacent farm fields. The area was monitored at six flow monitoring stations in 2019 throughout the growing season and at two downstream flow monitoring stations in the fall of 2020.

As part of a paired watershed experiment, the ditches were separated into a Control (northern) watershed and a Brush Cut (southern) watershed. Sites in the control watershed are denoted as Control Upstream, Control Midstream, Control Downstream. Sites in the Brush Cut

watershed are denoted as Brush Cut Upstream, Brush Cut Midstream, and Brush Cut Downstream.

To quantify chemical mass export and retention, two flow cells were calculated for both the Control and Brush Cut Watersheds (Fig. 2.1). Upstream flow cells equaled loading from the midstream sites minus the upstream sites. Downstream flow cells equaled loading from the downstream sites minus the midstream sites.

3.3.3 Water flow and chemical monitoring

HOBO Weather Station Dataloggers (Onset Computer Corporation, MA) recording relative humidity, air T, soil T, precipitation, and atmospheric pressure were installed at sites 18, 19 (Control) and at sites 20, and 21 (Brush Cut) (Fig. 2.1). Two precipitation gauge samplers were set up to collect precipitation from rain events to monitor rainfall water chemistry.

To measure water flow and nutrient concentrations, automated stream samplers (ISCO 6712) with bubble level flow modules (730 Bubble Flow Modules; Teledyne ISCO Inc.) were installed at Sites 18, 19, 20, and 21. Multiparameter sondes (EXO2, YSI Inc.), paired with the samplers, were installed instream at culverts in the Control and Brush Cut ditches. The multiparameter sondes provided in-situ measurements of blue-green algae, chlorophyll, conductivity, $O_{2(g)}$, dissolved organic matter, ORP, pH, salinity, total dissolved solids, and water temperature (Fig. 2.1). Sites 18 and 20 are the outlets of the site boundaries for each stream and were used to monitor cumulative effects. To measure input flow, ISCOs with bubble levels upstream of sites 11 (Control) and 5 (Brush Cut) were added. Meter sticks were set up in the ditch culverts to calibrate water levels. To calibrate flow, manual flow measurements were taken with flow probes (Digital Water Velocity Meter, Global Water). Calibration measurements to compensate for instrument drift were taken throughout the two field seasons.

There are dozens of tile drains within the watershed, making it prohibitive to sample all of them. To measure tile drainage input chemistry, water was sampled from control structures modulating the flow from tile drains at four microplot sites. V-notch weirs with pressure transducers were installed in the control structures to calculate flow rates (Levellogger, Solinst Canada Ltd.).

Groundwater elevations at each microplot site were monitored using previously installed 2.5 cm diameter piezometers (Fig. A.1) and a water level tape (Solinst Canada Ltd.). These include 1.0 m and 1.5 m piezometers in the north banks and 1.0 m and 1.5 m piezometers in the north shoulders.

Samples were collected at WEBs over two growing seasons starting May 23 to September 23, 2019 with a second May to early September field seasonal samples collected in 2020. Sampling during 2020 was impacted by COVID-19 restrictions. Stream water was sampled from all six ISCOs. When flow was consistent, 1 L samples were collected twice per week. When water levels rose 5 cm or more, the ISCO samplers automatically triggered, which collected 24 1L samples at 0, 1, 2, 3, 4, 5, 6, 7, 8, 9, 10, 12, 14, 16, 18, 20, 22, 24, 26, 28, 30, 36, 42, 48, and 54 hr intervals. From those triggers, the first and last sample were collected as well as samples associated with any peaks in turbidity from observed changes in colour between bottles. The remaining water samples were discarded. Tile drain control structures were sampled with a peristaltic pump twice per week when flowing. Groundwater samples were sampled monthly with a peristaltic pump, when piezometers contained sufficient volume. Rainfall water samples were collected after precipitation events that were significant enough to gather a sufficient sample volume (>20 ml) for analysis. Samples were kept in a cooler and returned to the AAFC laboratory on the day of sampling, for preservation and splitting for analysis.

Water samples were analyzed to determine turbidity, pH, ORP, and conductivity with the ProDSS MultiParameter Water Quality Meter (YSI inc.) which was calibrated or checked daily against reference solutions. Samples were split for analysis of total phosphorus (TP), cations, anions, dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), orthophosphate (o-PO₄), ammonia (NH₃-N), NO₃⁻ dual isotopes, and then frozen. All splits were filtered (0.45 µm) except for TP samples. DOC samples were preserved with H₂SO₄ and samples for cation analysis were preserved with HNO₃ to pH < 2. Cation and anion samples were stored in 15 mL HDPE bottles, and DIC and DOC samples were stored in 15 mL glass bottles. Samples for isotope determinations were stored in 60 mL HDPE bottles. Samples for TP analysis were stored in 30 mL snap cap vials with NH₃-N and o-PO₄ in 100 mL snap cap vials.

Concentrations of NH₃-N were determined using a DR1900 HACH spectrophotometer following the salicylate method (HACH 10023/Clin. Chim. Acta, 14, 403, 1966) and o-PO₄

concentrations were determined using the ascorbic acid method (APHA. 2005). DIC and DOC concentrations were determined by wet oxidation with sodium persulfate (Aurora 1030W TOC Analyser). Major anions were analyzed with ion chromatography (IC; Dionex ICS-5000, Thermo Scientific). These analytes included NO_3^- , NO_2^- , SO_4^{2-} , Cl^- , Br^- , F^- . Cation concentrations were determined using inductively-coupled plasma optical emission spectroscopy (ICP-OES) (iCAP 6000, Thermo Fisher). Nitrogen isotope ratios ($\delta^{15}\text{N}_{\text{NO}_3}$ and $\delta^{18}\text{O}_{\text{NO}_3}$) were measured on selected samples that measured a minimum concentration of $0.50 \text{ mg L}^{-1} \text{ NO}_3\text{-N}$. $\delta^2\text{H}$ and $\delta^{18}\text{O}$ water isotopes were analyzed with isotope ratio mass spectrometry (IRMS, IsoPrime Trace Gas, GV Instruments).

Water level and flow data was used to compute calibration curves to calculate volumetric flux according to procedure described in Sunohara et al. (2015). Total seasonal nutrient fluxes for each station were summed. To calculate cumulative loading, gaps in chemistry concentration were linearly interpolated in MATLAB. Volumetric flow was multiplied by concentration to estimate chemical mass outflow. Mass outflow was normalized by ArcGIS calculated tiled area. Time series of all data was plotted in SigmaPlot (Systat Software Inc.).

3.4 Results

3.4.1 Physical Flow

Monitoring in 2019 commenced May 1st and ended November 5th. Rainfall was intermittent throughout the 2019 growing season. The largest events were in May and October; there was little rainfall in August (Fig. 3.1). Stream flow peaked in May, June, and the end of October. In 2019 the Brush Cut watershed (400 L ha^{-1}) had more cumulative stream flow than the Control watershed (300 L ha^{-1}) with most of that coming in Spring. Monitoring in 2020 commenced September 5th due to COVID-19 travel restrictions and ended October 30th. In 2020 rainfall events in the beginning of September, end of September and middle of October were recorded. Stream flow only increased in the middle of October 2020. In the Fall of 2020, the Brush Cut watershed had more cumulative stream flow (150 L ha^{-1}) than the Control watershed (100 L ha^{-1}). Significant tile flow only occurred during spring and late fall in both 2019 and 2020 (Fig. 3.2).

3.4.2 Source Water Signature

Different sources of stream water have distinctive chemical signatures (Table 3.1). Groundwater has higher TP ($0.14 \pm 0.03 \text{ mg L}^{-1}$), higher $\text{PO}_4\text{-P}$ ($0.09 \pm 0.02 \text{ mg L}^{-1}$), higher SO_4 ($107 \pm 44 \text{ mg L}^{-1}$), higher DOC ($9 \pm 4 \text{ mg L}^{-1}$), and higher $\text{NO}_2\text{-N}$ ($0.02 \pm 0.01 \text{ mg L}^{-1}$) than that sampled from tile drains TP ($0.07 \pm 0.02 \text{ mg L}^{-1}$), $\text{PO}_4\text{-P}$ ($0.036 \pm 0.008 \text{ mg L}^{-1}$), SO_4 ($26 \pm 3 \text{ mg L}^{-1}$), DOC ($3.6 \pm 0.3 \text{ mg L}^{-1}$), and $\text{NO}_2\text{-N}$ ($0 \pm 0 \text{ mg L}^{-1}$). Groundwater and Tile water had similar DIC content: groundwater ($53 \pm 13 \text{ mg L}^{-1}$) and tile ($47 \pm 7 \text{ mg L}^{-1}$) and Cl content: groundwater ($14 \pm 3 \text{ mg L}^{-1}$) and tile ($18 \pm 3 \text{ mg L}^{-1}$). $\text{NO}_2\text{-N}$ was only detected in groundwater ($0.02 \pm 0.01 \text{ mg L}^{-1}$), while $\text{NH}_3\text{-N}$ was only detected in tile water ($0.012 \pm 0.004 \text{ mg L}^{-1}$). Tile water had higher $\text{NO}_3\text{-N}$ ($5.1 \pm 0.5 \text{ mg L}^{-1}$) than in groundwater ($0.3 \pm 0.1 \text{ mg L}^{-1}$).

3.4.3 Instream monitoring

In 2019 the Brush Cut watershed had lower mean ditch water levels (0.20 m) and higher mean water velocity (0.16 m s^{-1}) than the Control watershed (0.35 m, 0.06 m s^{-1}) (Table 3.2). The Brush Cut watershed had higher DO (10.7 mg L^{-1}) and pH (7.82) than the Control watershed (8.5 mg L^{-1} DO, 7.62 pH). The Brush Cut watershed had lower DOM (14 RFU), salinity (0.30 PSU),

and lower conductivity (0.51 mS m^{-1}) than the Control watershed (30 RFU, 14 PSU, 0.63 mS m^{-1}).

In 2020, the Brush Cut watershed again had lower mean ditch water levels (0.15 m) than the Control watershed (0.24 m) (Table 3.2), but similar mean water velocity (0.02 m s^{-1} *versus* 0.03 m s^{-1}). The Brush Cut watershed had higher DO (8.8 mg L^{-1} , pH (7.84) than the Control watershed (6.4 mg L^{-1} DO, 7.74 pH). The Brush Cut watershed had considerably lower DOM (14 RFU), salinity (0.30 PSU) and lower conductivity (0.48 mS m^{-1}) than the Control watershed (30 RFU, 14 PSU, 1.27 mS m^{-1}).

3.4.4 Downstream Cumulative Flow

A small mass (0.02 kg ha^{-1}) of $\text{NH}_4\text{-N}$ was exported from each watershed in 2019 (Fig. 3.3). The Brush Cut watershed exported more than four times the mass of $\text{NO}_3\text{-N}$ (3.24 kg ha^{-1}) *versus* only 0.75 kg ha^{-1} in the Control watershed in 2019 (Fig. 3.3). In 2019 the Brush Cut watershed exported less than half the $\text{PO}_4\text{-P}$ than the Control watershed, while TP export numbers were similar. More TP was exported by the Brush Cut watershed in Spring, but in Fall the trend reversed. There were gaps in sample collection from the Control Downstream site for DIC and DOC during times when the ditch was flowing. Cl (2.1 kg ha^{-1}) and total flow (443 L ha^{-1}) were notably higher at the Brush Cut watershed than the Control (1.4 kg ha^{-1} Cl, 287 L ha^{-1} water).

In Fall 2020, a small mass ($0.01\text{-}0.02 \text{ kg ha}^{-1}$) of $\text{NH}_4\text{-N}$ was exported from each watershed (Fig. 3.3). The Brush Cut watershed exported larger masses of $\text{NO}_3\text{-N}$ (0.62 kg ha^{-1}), DIC (13.6 kg ha^{-1}), and DOC (1.6 kg ha^{-1}), and larger water volumes (183 L ha^{-1}) than the Control (0.37 kg ha^{-1} $\text{NO}_3\text{-N}$, 8.3 kg ha^{-1} DIC, 1.0 kg ha^{-1} DOC, 106 L ha^{-1} water). The Brush Cut watershed exported approximately half the $\text{PO}_4\text{-P}$ (0.007 kg ha^{-1}) than the Control (0.015 kg ha^{-1}). The Brush Cut watershed had lower Cl (2.8 kg ha^{-1}) than the Control (9.2 kg ha^{-1} Cl).

3.4.5 2019 Cumulative Flow Cells

The Brush Cut watershed had a large mass of $\text{NO}_3\text{-N}$ export in both upstream (0.338 kg ha^{-1}) and downstream (0.843 kg ha^{-1}) flow cells (Fig. 3.4). The Control watershed had a small mass of $\text{NO}_3\text{-N}$ export in the upstream flow cell (0.074 kg ha^{-1}) and a large mass of retention in the downstream flow cell ($-0.550 \text{ kg ha}^{-1}$). NH_4 export in each flow cell was minimal ($<0.02 \text{ kg ha}^{-1}$).

The Brush Cut watershed had low $\text{PO}_4\text{-P}$ export (0.002 kg ha^{-1}) upstream and low $\text{PO}_4\text{-P}$ retention ($-0.013 \text{ kg ha}^{-1}$) downstream. The Control watershed had higher $\text{PO}_4\text{-P}$ export upstream (0.025 kg ha^{-1}) and higher retention downstream ($-0.129 \text{ kg ha}^{-1}$). The Brush Cut watershed had low TP export (0.011 kg ha^{-1}) upstream and downstream (0.028 kg ha^{-1}). The Control watershed had low TP export upstream (0.010 kg ha^{-1}) and higher export downstream (0.207 kg ha^{-1}). The flow cells did not include the late Fall flow data.

DOC (1.49 kg ha^{-1}) and DIC (2.76 kg ha^{-1}) export were high at the Brush Cut upstream flow cell than the Control upstream flow cell (0.68 kg ha^{-1} DOC, 2.39 kg ha^{-1} DIC). The Brush Cut watershed had limited Cl export in the upstream flow cell (0.17 kg ha^{-1}) and larger Cl export in the downstream flow cell (3.50 kg ha^{-1}). In contrast, the Control watershed retained a large mass of Cl in the upstream flow cell (-2.0 kg ha^{-1}) and exported only a small mass to the downstream flow cell (-0.43 kg ha^{-1}).

The Brush Cut watershed had a large mass of SO_4 export in both upstream (13.0 kg ha^{-1}) and downstream (6.1 kg ha^{-1}) flow cells (Fig. 3.5); in contrast, the Control watershed retained SO_4 in the upstream flow cell (-2.0 kg ha^{-1}) and only limited export in the downstream flow cell (1.1 kg ha^{-1}).

3.4.6 Nitrate Isotopes

Groundwater samples had low concentrations of $\text{NO}_3\text{-N}$ and low $\delta^{15}\text{N}$ isotope ratios in the 5-6 ‰ range (Fig. 3.6). Tile drains had a large range of NO_3 concentrations ($0.5\text{-}8 \text{ mg L}^{-1}$) and all but one sample were clustered around $\delta^{15}\text{N}$ ratios of 8 ‰ and 0 ‰ $\delta^{18}\text{O}$, with one sample at $\delta^{15}\text{N}$ of 15 ‰ and $\delta^{18}\text{O}$ of 0 ‰.

Five of the downstream Brush Cut samples had clustered near 15 ‰ $\delta^{15}\text{N}$ at 6 to 11 ‰ $\delta^{18}\text{O}$. Two samples were near 10 ‰ $\delta^{15}\text{N}$ and 1 to 6 ‰ $\delta^{18}\text{O}$ (Fig. 3.6). The $\delta^{15}\text{N}$ isotope ratios of most of the downstream Control samples ranged from 6-8 ‰ $\delta^{15}\text{N}$ with a wide range of $\delta^{18}\text{O}$ values (0-10 ‰). One sample measured at 13 ‰ $\delta^{15}\text{N}$ and 5 ‰ $\delta^{18}\text{O}$.

3.5 Discussion

3.5.1 Physical Flow

Rainfall events occurred throughout growing seasons of both 2019 and 2020, but increased in tile flow and stream flow in the drainage ditches only occurred in the Spring and Fall seasons. The lack of tile and stream flow during the middle of June to September period is likely due to lower groundwater baseflow, due to increased evapotranspiration (ET) associated with ditch vegetation and crops. This intermittent flow with stagnant summer flow is consistent with previous studies at this site (Sunohara et al. 2015) and other studies on agricultural drainage ditches (Herzon and Helenius 2008).

The Brush Cut watershed ditches had lower mean water levels and higher mean water velocities than the Control watershed which likely was associated with the 2018 dredging to remove ditch plants and reducing surface roughness. Greater cumulative flow in the Brush Cut watershed in both 2019 and 2020 is also likely a result of these changes. Lower water levels and higher flow velocities cause increased turbulence, which is consistent with the higher DO and turbidity in the Brush Cut watershed. Lower mean DOM, salinity, and conductivity in the Brush Cut watershed may be due to increased dilution was associated with greater flow volumes discharged from the Brush Cut watershed.

The Brush Cut watershed had 30-50% more water export than Control in both 2019 and 2020. This observation indicates that in the Brush Cut, less water was retained in the watershed, resulting in lower ET, which was likely caused by fewer ditch plants in the Brush Cut stream consuming water, and a shorter residence time for evaporation.

3.5.2 Chemical Signatures

Groundwater results showed that the water table was at least 1 m below surface of the bank of the drainage ditch. Water from the tile drains, consists of water resident in a drain system throughout the length of each adjacent crop field, mixed with recent precipitation (rain and spring melt) infiltrating from the field surface. Rainwater from the Westport ON weather station (ECCC 2018) shows a mean $0.44 \text{ mg L}^{-1} \text{ NO}_3\text{-N}$, $0.56 \text{ NH}_4\text{-N}$, and no P; therefore elevated concentrations of nutrients in groundwater must originate from adjacent fields in the tiledshed.

The majority (99%) of N in all samples occurs as NO_3^- , indicating that most fertilizer-applied N oxidized to NO_3^- prior to sample collection. This observation is likely because most farmer-applied NH_4^+ was consumed by crops or oxidized before it reached groundwater (Duxbury 1994). Tile water contained higher $\text{NO}_3\text{-N}$ concentrations than groundwater, indicating that NO_3^- is highly mobile in tile system and that the majority of N is directly discharged to the drainage ditch stream, with the tile drains acting as a short circuit to the stream. The presence of NO_2^- in groundwater indicates that denitrification may be taking place in groundwater (Bates and Spalding 1998). The presence of $\text{NH}_3\text{-N}$ in tile drain samples indicates that the fertilizer applied to crops has not fully oxidized before discharging through the tile drains to the drainage ditch (Binkley et al. 2004).

Higher TP and $\text{PO}_4\text{-P}$ concentrations in groundwater than in the tile drainage indicates that significant P has accumulated in groundwater, with some mineralization taking place (Sims et al. 1998; Hussain et al. 2021). We anticipate the potential for SO_4 reduction coupled with DOC oxidation in the groundwater system, which would decrease the mass of SO_4 and DOC (Appelo and Postma 2007). The higher SO_4 and DOC in the groundwater compared to the tile water indicates that although some redox processes are occurring, net DOC and SO_4 accumulation in groundwater continues.

3.5.3 Nitrogen

The small mass of NH_4^+ exported from each watershed is consistent with the small mass of NH_4^+ present in tile water samples.

The Brush Cut watershed exported 320% more NO_3^- downstream in 2019 and 68% more NO_3^- in 2020, than the Control watershed. This corresponds well with the 2019 cumulative flow-cell masses, where more NO_3^- was available for export, both upstream and downstream. The significant increase in NO_3^- export in the Brush Cut watershed is likely due to multiple factors. The higher DO in the Brush Cut stream maintains oxidizing conditions in the stream, hindering denitrification (Oh and Silverstein 1999). Less ditch vegetation and lower soil nutrient concentrations in the Brush Cut stream are available to act as hosts and substrates for denitrifying microbes (Mosier et al. 2002). Higher water velocity results in shorter residence times, providing less time for NO_3^- for denitrification, sorption, or plant uptake (Dettmann 2001; Castaldelli et al. 2015; Vymazal and Březinová 2018; Cui et al. 2020).

As denitrification rates increase, both $\delta^{15}\text{N}_{\text{NO}_3}$ and $\delta^{18}\text{O}_{\text{NO}_3}$ should both increase because lighter NO_3^- molecules are more easily consumed by microbial activity (Poth and Focht 1985). Denitrification increases $\delta^{15}\text{N}$ more than $\delta^{18}\text{O}$, thus the slope of $\delta^{15}\text{N}/\delta^{18}\text{O}$ can indicate rates of denitrification (Clark 2015). The groundwater samples have low $\delta^{15}\text{N}$ values indicating that low NO_3 concentrations in the ditch bank groundwater are not affected by denitrification. The water sourced from tile drain has two distinct $\delta^{15}\text{N}$ profiles of 8 ‰ and 15 ‰, which could indicate that the NO_3 in the tile water could originate from two distinct sources, with the 8 ‰ $\delta^{15}\text{N}_{\text{NO}_3}$ coming from recent crop application and 15 ‰ $\delta^{15}\text{N}_{\text{NO}_3}$ coming from a reservoir of longer residence time water that has undergone denitrification. There are dozens of tile drains flowing into each ditch, so it would be a formidable task to identify which input signature is dominant.

The majority of high $\delta^{15}\text{N}$ values were found in the Brush Cut, with the lower $\delta^{15}\text{N}$ values in the Control watershed, which is contrary to expectations (Mengis et al. 1999; Clark 2015; Griffiths et al. 2016). It was assumed that $\delta^{15}\text{N}$ tile drain values clustered around a mean, but both high (15 ‰) and low (8 ‰) $\delta^{15}\text{N}$ values were found in tile drain samples of the study area. The drainage ditch system is structured in a manner which combines source water from dozens of tile drains. To measure input flow $\delta^{15}\text{N}$ signature, $\delta^{15}\text{N}$ values would have to be analyzed at every tile drain to calculate each contribution to the cumulative $\delta^{15}\text{N}$ input. Without a measured $\delta^{15}\text{N}$ input signature, an assessment of instream denitrification cannot be drawn from the $\delta^{15}\text{N}$ outflow signature.

3.5.4 Phosphorus

Sparse ditch vegetation, more rapid water flow, and lower water levels in the Brush Cut watershed are generally associated with lower rates of P mineralization PO_4^{3-} (Vymazal and Březinová 2018). The effects of these conditions may be offset by more sediment P sorption in the Brush Cut watershed (Moore et al. 2010). The Brush Cut watershed exported more TP in Spring, but less in Fall 2019, making the net effect on TP export inconsequential. The difference between spring and fall data possibly indicates that there was temporary retention of P in the Control watershed ditch in the Spring of 2019 that was later released in Fall of 2019.

The Brush Cut watershed had less PO_4^{3-} export *versus* the Control watershed. The Control watershed had net PO_4^{3-} export in the upstream flow cell, and net retention in the

downstream flow cell. This observation indicates that some P transformation took place in the Control watershed, possibly mineralization of organic P to PO_4^{3-} upstream, which was retained in the downstream flow cell. Dredging decreases plant uptake of P, resulting in lower P retention (Vymazal and Březinová 2018) but this impact is offset by increased P sediment sorption (Moore et al. 2010). The lack of detectable differences in TP export between the Brush Cut and Control watersheds demonstrates that ditch intervention may not affect net P runoff and eutrophication (Vitousek et al. 1997; Rabalais et al. 2002; Howarth et al. 2002).

3.5.5 Carbon

A lack of downstream samples during peak flow prevents calculation of total stream DOC and DIC exports in 2019. Upstream flow cells have over twice the DOC export and over 15% more DIC export in the Brush Cut stream than the Control. In Fall 2020 the Brush Cut watershed exported 60% more DIC and DOC, which is similar to the difference in water export between watersheds. Greater DOC export indicates less OC was consumed by geochemical processes such as OC oxidation where NO_3 and SO_4 were being reduced.

3.5.6 Chloride

In 2019 the Brush Cut watershed had 60% more Cl export than the Control, which is consistent with difference in the water export, but in Fall of 2020 the Control had triple the Cl export than the Brush Cut watershed. This spike in Cl may have been caused by wastewater dumping from a nearby farm.

3.5.7 Sulphur

There was greater SO_4 export in the Brush Cut watershed than the Control watershed in 2019 in both the upstream and downstream flow cells. The Control watershed retained SO_4 in the upstream Control flow cell. SO_4 retention is expected in reducing environments where sufficient O_2 , NO_3 , Mn, and Fe has been consumed. There was no indication of such retention in the Brush Cut watershed.

3.5.8 Synthesis

Brush cutting and dredging led to higher mean DO concentrations, higher mean water velocities, lower mean water levels, and more water export in the Brush Cut watershed. These conditions diminished the retention capacity of the drainage ditches and diminished the function as a vegetated wetland, resulting in more chemical DIC, DOC, NO₃, and SO₄ export. The lack of retention capacity was likely due to multiple factors, including the increased hydraulic outflow, shortened residence time and increased O₂, limiting the biogeochemical reduction capacity of the ditch. The lack of instream plants limited plant nutrient uptake capacity.

Smith and Huang (2010) show that ditch nutrient retention improved during 12 months after dredging, which differs from the findings of the current study. Smith and Huang (2010) observed more rapid regrowth of aquatic plants in the excavated channel (1 month) than was observed in this Ontario study, possibly due to the warmer, wetter climate of Indiana, and differing soil mechanical properties, which inhibited rapid plant regrowth at the WEBs site.

3.6 Conclusions

- i) DOC, N, P are derived from nutrient-rich water originating from tile drainage after precipitation events. The tile flow only occurs in Spring and Fall when the water table is high.
- ii) Brush cutting and dredging in 2018 lowered water levels and increased flow velocities, resulting in a 30-50% increase in water outflow compared to the control ditch in both 2019 and 2020.
- iii) Brush cutting and dredging reduced or eliminated the NO_3 retention capacity of the agricultural drainage ditches studied. NO_3 export *versus* Control was measured at 320% more in 2019, and 68% more in Fall 2020. The increased NO_3 export may negatively affect rural water supplies.
- iv) The differences in P export between Brush Cut and Control in 2019 are small. There are more signs of P transformation in the Control watershed, but brush cutting and dredging may not significantly affect eutrophication derived from excess P loadings.
- v) Analysis of NO_3 isotopes in tile water indicate that NO_3 rich tile water comes from multiple sources, possibly some recently applied and some of longer residence time. NO_3 isotopes were not helpful in determining NO_3 reduction in this complex tile drain system, as this would have required individual measurement of each tile drain.
- vi) Lower concentrations of O_2 and retention of DOC and SO_4 in the Control watershed indicates there is significant potential for NO_3 reduction.

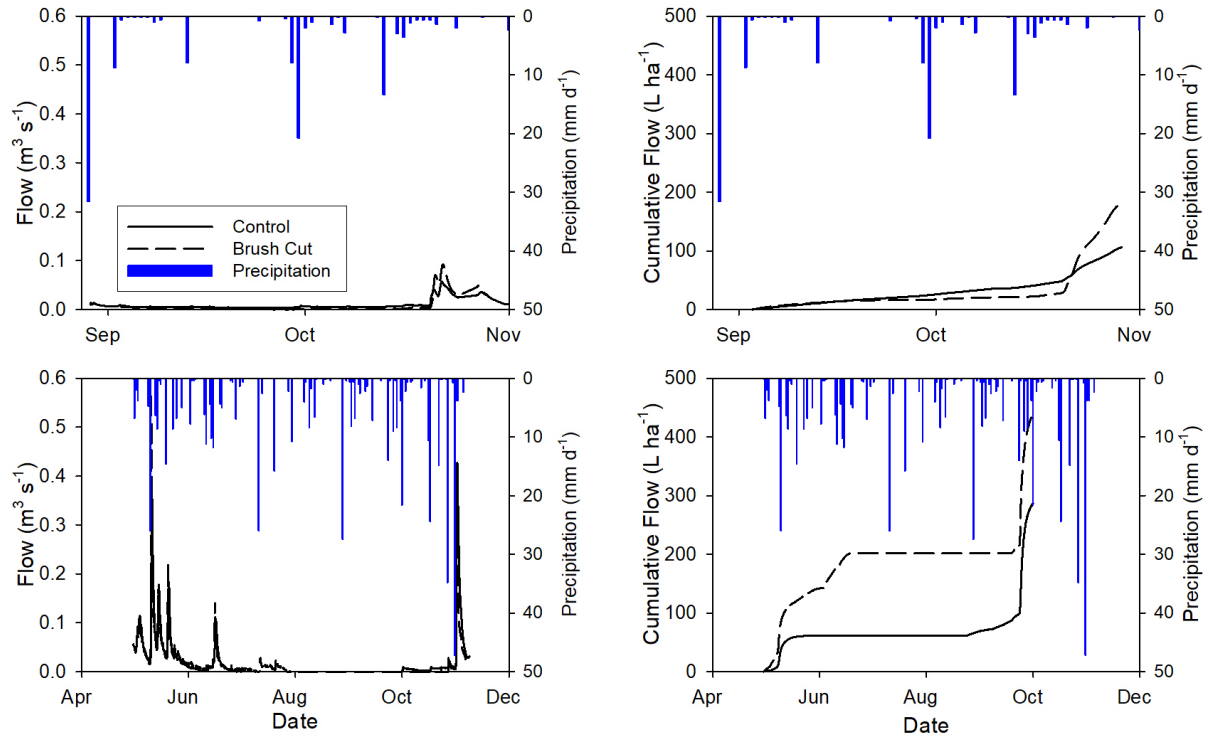


Fig. 6.1 Time series data of watershed flow for Brush Cut in 2019 (Bottom) and 2020 (Top). From left to right (instantaneous flow, cumulative flow) with precipitation in blue.

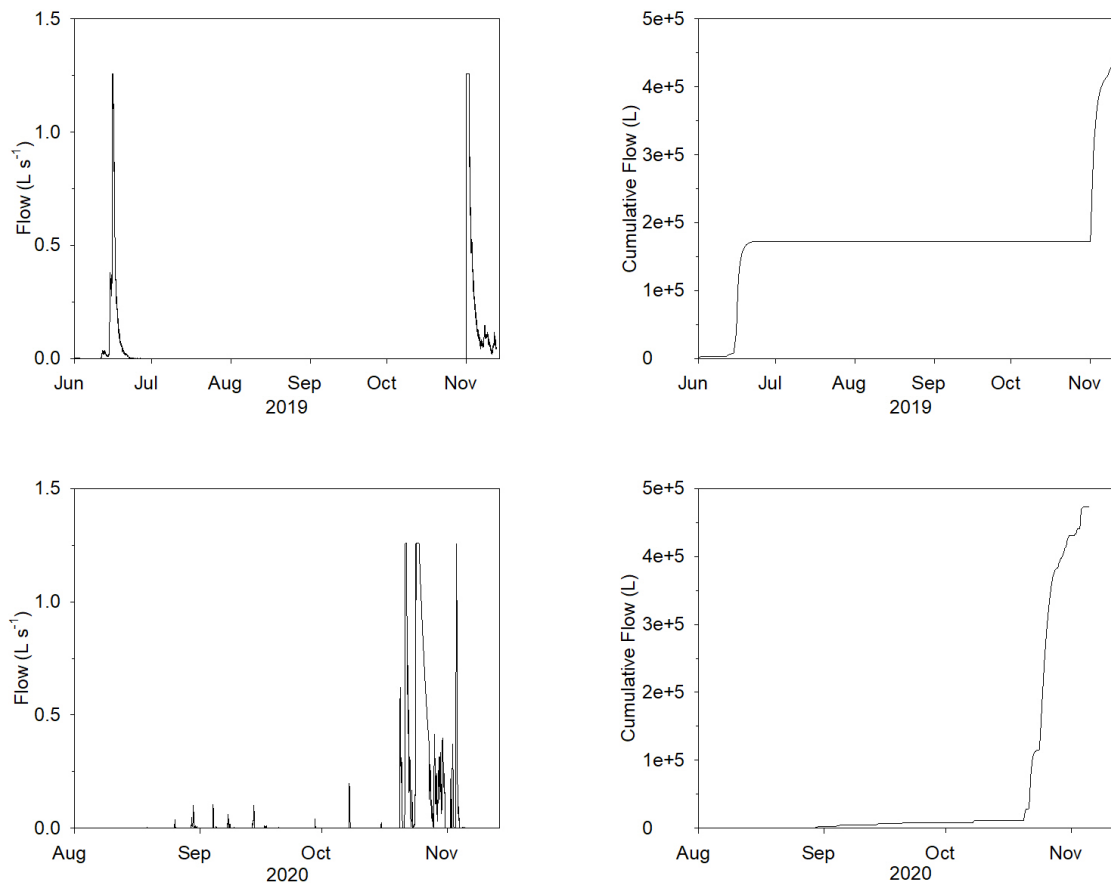


Fig. 6.2 Tile flow from tile 10-NW. From top to bottom, 2019, 2020. From left to right, flow rate (L s^{-1}), cumulative flow (L).

Table 6.1 Source water chemistry concentrations (Mean, Standard Error, Number of measurements) for tile, groundwater (GW) and rainfall sampled in 2019 and 2020.

	Tile			Groundwater			Rain		
	Mean (mg L^{-1})	SE	n	Mean (mg L^{-1})	SE	n	Mean (mg L^{-1})	SE	n
TP	0.07	0.02	14	0.14	0.03	7	-	-	0
$\text{PO}_4\text{-P}$	0.036	0.008	21	0.09	0.02	7	-	-	0
$\text{NH}_3\text{-N}$	0.012	0.004	19	0	-	1	-	-	0
$\text{NO}_2\text{-N}$	0	0	33	0.02	0.01	8	0	0	5
$\text{NO}_3\text{-N}$	5.1	0.5	33	0.3	0.1	8	0.004	0.002	5
SO_4	26	3	33	107	44	8	1.0	0.4	5
DOC	3.6	0.3	15	9	4	5	-	-	0
DIC	47	7	15	53	13	5	-	-	0
Cl	18	3	33	14	3	8	0.4	0.2	5

Table 6.2 Mean Sonde and ISCO measurements sampled at Downstream ISCOs average measurement based on sampling every 15 min from May 3 to July 10 2019 and June 1 to October 30 2020.

	Control 2019	Brush Cut 2019	Control 2020	Brush Cut 2020
Water Level (m)	0.35	0.20	0.24	0.15
Water Velocity (m s ⁻¹)	0.06	0.16	0.03	0.02
Chlorophyll (µg L ⁻¹)	5.9	6.2	14.5	1.1
Conductivity (mS m ⁻¹)	0.63	0.51	1.27	0.48
Dissolved Oxygen (mg L ⁻¹)	8.5	10.7	6.4	8.8
Fluorescent Dissolved Organic Matter (RFU)	30	14	38	21
Oxidation-Reduction Potential (mV)	367	280	293	398
pH	7.62	7.82	7.74	7.84
Salinity (PSU)	0.39	0.30	0.79	0.30

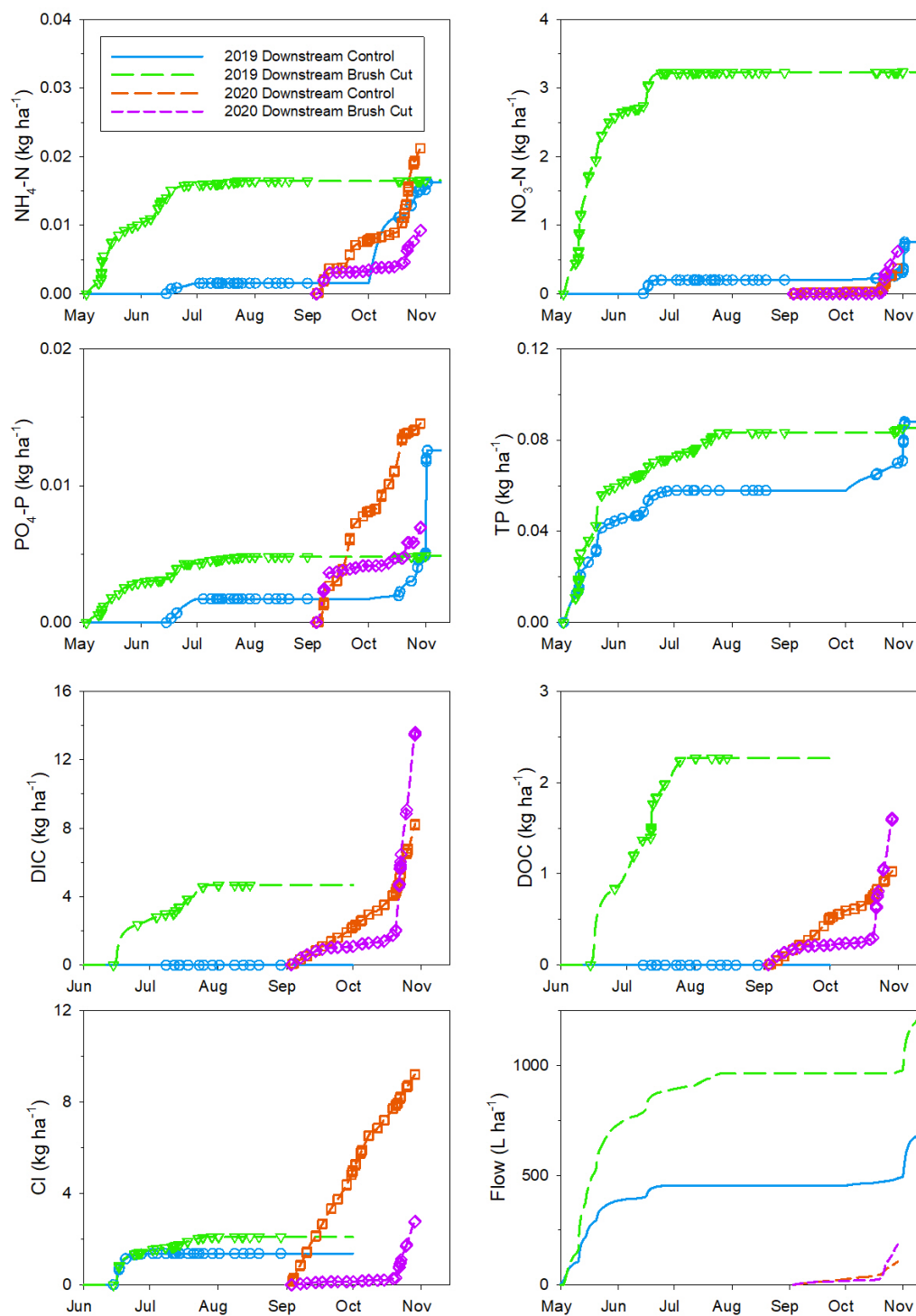


Fig. 6.3 Time series of streamflow cumulative mass per tiled area from Control and Brush Cut ditches over 2019 and 2020 sampling seasons for Downstream Control 2019 •, Downstream Brush Cut 2019 ■, Downstream Control 2020 ▲, Downstream Brush Cut 2020 ◆. Data points indicate times when chemical concentration was analyzed. From top row to bottom row: $\text{NH}_4\text{-N}_{(\text{aq})}$, $\text{NO}_3\text{-N}_{(\text{aq})}$, $\text{PO}_4\text{-P}$, TP (Total Phosphorus), Dissolved Inorganic Carbon (DIC), Dissolved Organic Carbon (DOC), $\text{Cl}_{(\text{aq})}$, Water Flow (L).

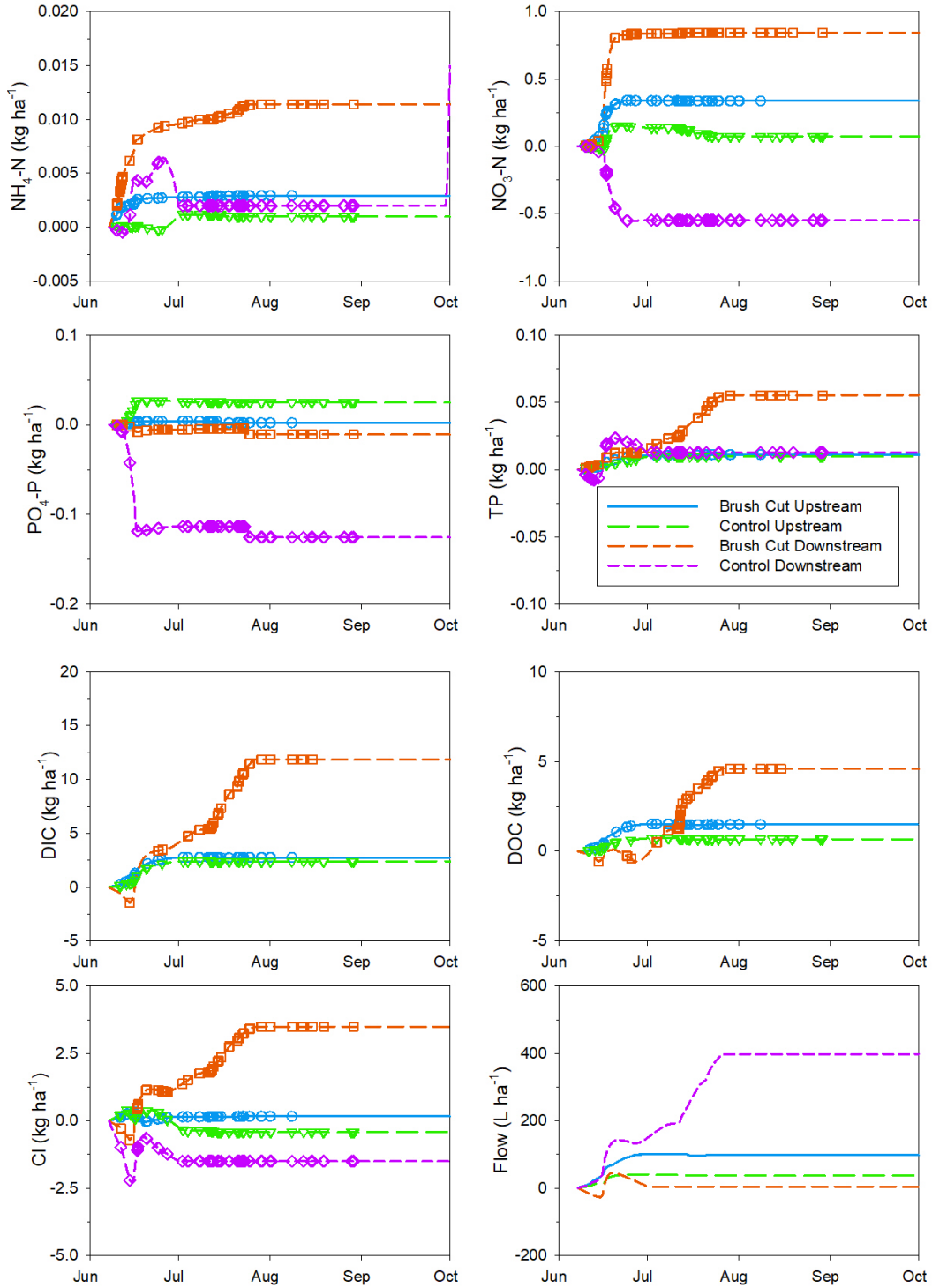


Fig. 6.4 Time series of cumulative mass per tiled area for streamflow accumulation cells from Control and Brush Cut ditches over the 2019 sampling season for Midstream-Upstream Brush Cut •, Midstream-Upstream Control ■, Downstream-Midstream Brush Cut ▲, Downstream-Midstream Control ♦. Data points indicate times when chemical concentration was analyzed. From top row to bottom row: $\text{NH}_4\text{-N}_{(\text{aq})}$, $\text{NO}_3\text{-N}_{(\text{aq})}$, $\text{PO}_4\text{-P}$, TP (Total Phosphorus), Dissolved Inorganic Carbon (DIC), Dissolved Organic Carbon (DOC), $\text{Cl}_{(\text{aq})}$, Water Flow (L).

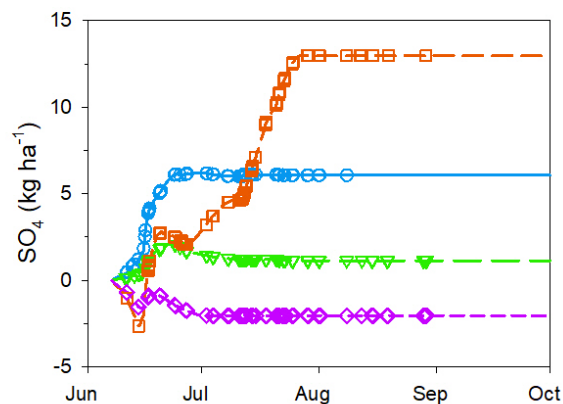


Fig. 6.5 Time series of $\text{SO}_{4(\text{aq})}$ cumulative mass per tilled area for streamflow accumulation cells from Control and Brush Cut ditches over the 2019 sampling season for Midstream-Upstream Brush Cut $\bullet$, Midstream-Upstream Control $\blacksquare$, Downstream-Midstream Brush Cut $\blacktriangle$, Downstream-Midstream Control $\blacklozenge$. Data points indicate times when chemical concentration was analyzed.

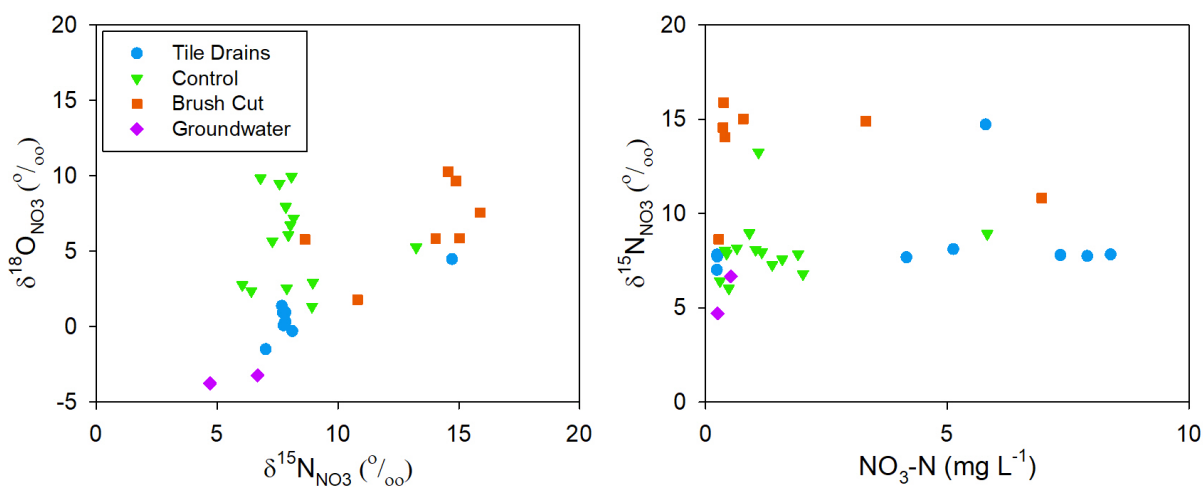


Fig. 6.6 Scatter plots of dual $\text{NO}_{3(\text{aq})}$ isotope data from 2019 sampling season. Left: $\delta^{18}\text{O}_{\text{NO}_3}$ vs $\delta^{15}\text{N}_{\text{NO}_3}$. Right: $\delta^{18}\text{O}_{\text{NO}_3}$ vs $\text{NO}_3\text{-N}$. Sampling locations: $\bullet$ tile drains, $\blacktriangledown$ Downstream Control stream, $\blacksquare$ Downstream Brush Cut, $\blacklozenge$ Groundwater from ditch banks.

Chapter 4: Conclusions

4.1 Summary of Findings

Agricultural drainage ditches have many impacts on the environmental landscape. This project comprehensively monitored GHG emissions over three growing seasons and nutrient hydraulic flow over two growing seasons to evaluate beneficial and detrimental environmental effects of the common practices of clearing vegetation and dredging of agricultural drainage ditches. The current research, as part of the broader agriculture greenhouse gas program, is focused on how drainage ditches affect the fate of agricultural nutrients, applied as fertilizers, and impacts on air quality and water quality downstream. The research also is focused on quantifying how intervention techniques, employed to enhance drainage, such as dredging and brushing, affect the environmental impact drainage ditches, both positively and negatively.

The main findings from Chapter 2, GHG Emissions Affected by Vegetation Removal in Agricultural Drainage Ditches, provides an overview of GHG emissions in agriculture drainage ditches and how those emissions are affected by brushing and dredging. Brushing drainage ditches has the potential to cause a short-term increase in CO₂ emissions in shrubbed ditches and a long-term increase in CO₂ emissions in treed ditches. Dredging has the potential to cause multiple year decreases in CH₄ emissions. To minimize the effect on GHG emissions, it is important to retain trees in drainage ditches to block insolation, lower the soil temperature and reduce soil GHG emissions.

Ditch emissions were between 50 and 130 kg ha⁻¹d⁻¹ CO_{2eq} between April and October (214 days) over the three seasons. A corn-soybean-wheat rotation in the South Nation watershed emits between 4600 and 5030 kg ha⁻¹ CO₂-C and 1.03 kg ha⁻¹ N₂O-N between April and October (Drury et al. 2008). For the South Nation watershed (3900 km²) there are 3300 km of ditches and 60% covered by agricultural field (Rideout et al. 2021). We calculated that drainage ditch emissions are between 0.9% and 2.3% of field emissions of the watershed, depending on the treatment applied to the drainage ditch.

The main findings from Chapter 3, Long-Term Observations of Nitrogen and Phosphorus Export in Paired-Tile Drained Agricultural Watersheds under Enhanced Vegetation Removal and Intact Riparian Zone, are focused on determining how brush cutting and dredging changes the hydraulic flow of nutrients. Brush cutting and dredging increases hydraulic outflow and reduces

or eliminates NO_3 retention capacity of agricultural drainage ditches. This change results in a net increase in NO_3 export which may negatively affect rural water supplies. The differences in P export between the Brush Cut and Control ditches in 2019 were small indicating that brush cutting and dredging may not significantly affect eutrophication. There were indications that the Brush Cut watershed had undergone less denitrification.

4.2 Future Directions

We demonstrate that aggressively removing ditch vegetation via dredging hinders stream regrowth, reducing the wetland function of the ditch. Dredging both hindered the nutrient filtration capacity and decreased CH_4 emissions over two growing seasons. Future research could evaluate the duration required for aquatic ditch vegetation to fully recover from dredging. Subsequent studies could also examine less aggressive intervention methods, or focus on ditch systems with different climate or soil properties,

Our research sampled GHG emissions weekly. As GHG emissions cycle throughout the day and week, future research could sample GHG emissions with increased temporal resolution, especially on CO_2 on the bank or shoulder of the ditch and CH_4 instream to better quantify the exact GHG contributions.

Our current research demonstrates that removing vegetation and dredging agricultural drainage ditches increases NO_3 export through both dilution and denitrification, but we were not able to quantify the contribution from each process. Future research could characterize the microbiome in areas suspected of undergoing higher levels of denitrification to see if more denitrifiers are present.

We quantified how ditch intervention changes geochemical properties, and further research could focus on how it changes insect, or bird populations.

This study builds on an emerging body of research on drainage ditches, and adds mechanistic components of soil, vegetation, groundwater, and surface water fluxes. When considering any BMP, all costs and benefits, not only GHG fluxes, should be considered. Our results and future studies should allow policy makers to make decisions on whether to allocate

funds towards clearing ditches to improve drainage, or even the opposite, subsidizing farmers to preserve intact riparian zones on their properties.

References

- Agriculture and Agrifoods Canada. (2013) Watershed evaluation of beneficial management practices (WEBS): Managing our land and protecting our water through long-term watershed-scale research: Final report (2004-2013). <http://publications.gc.ca/pub?id=9.816845&sl=0>. Accessed 10 March 2022
- Ahiablame LM, Chaubey I, Smith DR, Engel BA (2011) Effect of tile effluent on nutrient concentration and retention efficiency in agricultural drainage ditches. *Agr Water Manage* 98:1271–1279. <https://doi.org/10.1016/j.agwat.2011.03.002>
- Bateman EJ, Baggs EM (2005) Contributions of nitrification and denitrification to N₂O emissions from soils at different water-filled pore space. *Biol Fertil Soils* 41:379–388. <https://doi.org/10.1007/s00374-005-0858-3>
- Bates HK, Spalding RF (1998) Aquifer denitrification as interpreted from *in situ* microcosm experiments. *J Environ Qual* 27:174–182. <https://doi.org/10.2134/jeq1998.00472425002700010025x>
- Berisso FE, Schjønning P, Keller T, et al (2013) Gas transport and subsoil pore characteristics: Anisotropy and long-term effects of compaction. *Geoderma* 195–196:184–191. <https://doi.org/10.1016/j.geoderma.2012.12.002>
- Binkley D, Ice GG, Kaye J, Williams CA (2004) Nitrogen and phosphorus concentrations in forest streams of the United States¹. *JAWRA J Am Water Resour Assoc* 40:1277–1291. <https://doi.org/10.1111/j.1752-1688.2004.tb01586.x>
- Blume O, Guitard E, Crann C, et al Relationships between carbon age and CO₂ efflux in agricultural and drainage ditch soils using the thermonuclear bomb pulse. *Vadose Zone Journal* n/a:e20208. <https://doi.org/10.1002/vzj2.20208>
- Brady NC, Weil RR (2008) The nature and properties of soils, 14th ed. Pearson Prentice Hall
- Carter MR, Gregorich EG (eds) (2007) Soil-surface gas emissions. In: *Soil Sampling and Methods of Analysis*, 2nd edn. CRC Press
- Castaldelli G, Soana E, Racchetti E, et al (2015) Vegetated canals mitigate nitrogen surplus in agricultural watersheds. *Agric Ecosyst Environ* 212:253–262. <https://doi.org/10.1016/j.agee.2015.07.009>
- Chen W, Wilkes G, Khan IUH, et al (2018) Aquatic bacterial communities associated with land use and environmental factors in agricultural landscapes using a metabarcoding approach. *Front Microbiol* 9:. <https://doi.org/10.3389/fmicb.2018.02301>
- Clark I (2015) *Groundwater Geochemistry and Isotopes*. CRC Press

- Conner WH (1995) Woody plant regeneration in three South Carolina *Taxodium/Nyssa* stands following hurricane Hugo. *Ecol Eng* 4:277–287. [https://doi.org/10.1016/0925-8574\(94\)00054-9](https://doi.org/10.1016/0925-8574(94)00054-9)
- Cui N, Zhang X, Cai M, et al (2020) Roles of vegetation in nutrient removal and structuring microbial communities in different types of agricultural drainage ditches for treating farmland runoff. *Ecol Eng* 155:105941. <https://doi.org/10.1016/j.ecoleng.2020.105941>
- Davidson EA, Keller M, Erickson HE, et al (2000) Testing a conceptual model of soil emissions of nitrous and nitric oxides using two functions based on soil nitrogen availability and soil water content, the hole-in-the-pipe model characterizes a large fraction of the observed variation of nitric oxide and nitrous oxide emissions from soils. *BioScience* 50:667–680. [https://doi.org/10.1641/0006-3568\(2000\)050\[0667:TACMOS\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2000)050[0667:TACMOS]2.0.CO;2)
- de Groot RS, Alkemade R, Braat L, et al (2010) Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecol Complexity* 7:260–272. <https://doi.org/10.1016/j.ecocom.2009.10.006>
- de Klein JJM, van der Werf AK (2014) Balancing carbon sequestration and GHG emissions in a constructed wetland. *Ecol Eng* 66:36–42. <https://doi.org/10.1016/j.ecoleng.2013.04.060>
- Derwent RG (2020) Global warming potential (GWP) for methane: Monte Carlo analysis of the uncertainties in global tropospheric model predictions. *Atmosphere* 11:486. <https://doi.org/10.3390/atmos11050486>
- DeSimone J, Macrae ML, Bourbonniere RA (2010) Spatial variability in surface N₂O fluxes across a riparian zone and relationships with soil environmental conditions and nutrient supply. *Agric Ecosyst Environ* 138:1–9. <https://doi.org/10.1016/j.agee.2010.03.007>
- Dettmann EH (2001) Effect of water residence time on annual export and denitrification of nitrogen in estuaries: A model analysis. *Estuaries* 24:481–490. <https://doi.org/10.2307/1353250>
- Dosskey MG, Vidon P, Gurwick NP, et al (2010) The role of riparian vegetation in protecting and improving chemical water quality in streams¹. *JAWRA J Am Water Resour Assoc* 46:261–277. <https://doi.org/10.1111/j.1752-1688.2010.00419.x>
- Drainage Act and Conservation Authorities Act protocol. In: ontario.ca. <http://www.ontario.ca/page/drainage-act-and-conservation-authorities-act-protocol>. Accessed 2 Aug 2022
- Drury CF, Yang XM, Reynolds WD, McLaughlin NB (2008) Nitrous oxide and carbon dioxide emissions from monoculture and rotational cropping of corn, soybean and winter wheat. *Can J Soil Sci* 88:163–174. <https://doi.org/10.4141/CJSS06015>
- Dutaur L, Verchot LV (2007) A global inventory of the soil CH₄ sink. *Global Biogeochem Cycles* 21:. <https://doi.org/10.1029/2006GB002734>

- Duxbury JM (1994) The significance of agricultural sources of greenhouse gases. *Fertil Res* 38:151–163. <https://doi.org/10.1007/BF00748775>
- Environment and Climate Change Canada (2018) Monitoring of atmospheric precipitation chemistry. <https://donnees.ec.gc.ca/data/air/monitor/monitoring-of-atmospheric-precipitation-chemistry/?lang=en>. Accessed 5 Nov 2022
- Fargione J, Hill J, Tilman D, et al (2008) Land clearing and the biofuel carbon debt. *Science* 319:1235–1238. <https://doi.org/10.1126/science.1152747>
- Faust DR, Kröger R, Moore MT, Rush SA (2018) Management practices used in agricultural drainage ditches to reduce Gulf of Mexico hypoxia. *Bull Environ Contam Toxicol* 100:32–40. <https://doi.org/10.1007/s00128-017-2231-2>
- Fierer N, Allen AS, Schimel JP, Holden PA (2003) Controls on microbial CO₂ production: a comparison of surface and subsurface soil horizons. *Glob Chang Biol* 9:1322–1332. <https://doi.org/10.1046/j.1365-2486.2003.00663.x>
- Fortier J, Truax B, Gagnon D, Lambert F (2015) Biomass carbon, nitrogen and phosphorus stocks in hybrid poplar buffers, herbaceous buffers and natural woodlots in the riparian zone on agricultural land. *J Environ Manage* 154:333–345. <https://doi.org/10.1016/j.jenvman.2015.02.039>
- Fu D, Gong W, Xu Y, et al (2014) Nutrient mitigation capacity of agricultural drainage ditches in Tai Lake basin. *Ecol Eng* 71:101–107. <https://doi.org/10.1016/j.ecoleng.2014.07.043>
- Gitau MW, Veith TL, Gburek WJ, Jarrett AR (2006) Watershed level best management practice selection and placement in the Town Brook watershed, New York¹. *JAWRA J Am Water Resour Assoc* 42:1565–1581. <https://doi.org/10.1111/j.1752-1688.2006.tb06021.x>
- Griffiths NA, Jackson CR, McDonnell JJ, et al (2016) Dual nitrate isotopes clarify the role of biological processing and hydrologic flow paths on nitrogen cycling in subtropical low-gradient watersheds. *J Geophys Res: Biogeosci* 121:422–437. <https://doi.org/10.1002/2015JG003189>
- Hefting MM, Bobbink R, de Caluwe H (2003) Nitrous oxide emission and denitrification in chronically nitrate-loaded riparian buffer zones. *J Environ Qual* 32:1194–1203. <https://doi.org/10.2134/jeq2003.1194>
- Herzon I, Helenius J (2008) Agricultural drainage ditches, their biological importance and functioning. *Biol Conserv* 141:1171–1183. <https://doi.org/10.1016/j.biocon.2008.03.005>
- Howarth RW, Sharpley A, Walker D (2002) Sources of nutrient pollution to coastal waters in the United States: implications for achieving coastal water quality goals. *Estuaries* 25:656–676. <https://doi.org/10.1007/BF02804898>

- Hussain SI, Phillips LA, Hu Y, et al (2021) Differences in phosphorus biogeochemistry and mediating microorganisms in the matrix and macropores of an agricultural clay loam soil. *Soil Biol Biochem* 161:108365. <https://doi.org/10.1016/j.soilbio.2021.108365>
- Iseyemi OO, Farris JL, Moore MT, Choi S (2016) Nutrient mitigation efficiency in agricultural drainage ditches: an influence of landscape management. *Bull Environ Contam Toxicol* 96:750–756. <https://doi.org/10.1007/s00128-016-1783-x>
- Jacinthe PA, Vidon P, Fisher K, et al (2015) Soil methane and carbon dioxide fluxes from cropland and riparian buffers in different hydrogeomorphic settings. *J Environ Qual* 44:1080–1090. <https://doi.org/10.2134/jeq2015.01.0014>
- Kavanagh WL, Hoggarth C (2017) Guidance for maintaining and repairing municipal drains in Ontario. <https://www.dsao.net/images/Documents/Dart/General/Guidance-for-Maintaining-and-Repairing-Municipal-Drains-in-Ontario-May-23-2017.pdf>. Accessed 7 June 7 2022
- Kleinman PJA, Sharpley AN, McDowell RW, et al (2011) Managing agricultural phosphorus for water quality protection: principles for progress. *Plant Soil* 349:169–182. <https://doi.org/10.1007/s11104-011-0832-9>
- Kleinman PJA, Sharpley AN, Moyer BG, Elwinger GF (2002) Effect of mineral and manure phosphorus sources on runoff phosphorus. *J Environ Qual* 31:2026–2033. <https://doi.org/10.2134/jeq2002.2026>
- Knox AK, Dahlgren RA, Tate KW, Atwill ER (2008) Efficacy of natural wetlands to retain nutrient, sediment and microbial pollutants. *J Environ Qual* 37:1837–1846. <https://doi.org/10.2134/jeq2007.0067>
- Kundu MC, Mandal B (2009) Agricultural activities influence nitrate and fluoride contamination in drinking groundwater of an intensively cultivated district in India. *Water Air Soil Pollut* 198:243–252. <https://doi.org/10.1007/s11270-008-9842-5>
- Lapen DR, Topp E, Edwards M, et al (2008) Effect of liquid municipal biosolid application method on tile and ground water quality. *J Environ Qual* 37:925–936. <https://doi.org/10.2134/jeq2006.0486>
- Lynch J, Cain M, Pierrehumbert R, Allen M (2020) Demonstrating GWP*: a means of reporting warming-equivalent emissions that captures the contrasting impacts of short- and long-lived climate pollutants. *Environ Res Lett* 15:044023. <https://doi.org/10.1088/1748-9326/ab6d7e>
- Maier M, Schack-Kirchner H (2014) Using the gradient method to determine soil gas flux: a review. *Agric For Meteorol* 192–193:78–95. <https://doi.org/10.1016/j.agrformet.2014.03.006>

- Martin ER, Godwin IA, Cooper RI, et al (2021) Assessing the impact of vegetative cover within northeast Arkansas agricultural ditches on sediment and nutrient loads. *Agric Ecosyst Environ* 320:107613. <https://doi.org/10.1016/j.agee.2021.107613>
- Mengis M, Schiff SL, Harris M, et al (1999) Multiple geochemical and isotopic approaches for assessing ground water NO₃-elimination in a riparian zone. *Groundwater* 37:448–457. <https://doi.org/10.1111/j.1745-6584.1999.tb01124.x>
- Moore MT, Kröger R, Locke MA, et al (2010) Nutrient mitigation capacity in Mississippi Delta, USA drainage ditches. *Environ Pollut* 158:175–184. <https://doi.org/10.1016/j.envpol.2009.07.024>
- Mosier AR, Doran JW, Freney JR (2002) Managing soil denitrification. *J Soil Water Conserv* 57:505–512
- Moyano FE, Manzoni S, Chenu C (2013) Responses of soil heterotrophic respiration to moisture availability: An exploration of processes and models. *Soil Biol Biochem* 59:72–85. <https://doi.org/10.1016/j.soilbio.2013.01.002>
- Nangia V, Sunohara MD, Topp E, et al (2013) Measuring and modeling the effects of drainage water management on soil greenhouse gas fluxes from corn and soybean fields. *J Environ Manage* 129:652–664. <https://doi.org/10.1016/j.jenvman.2013.05.040>
- Noponen MRA, Healey JR, Soto G, Haggard JP (2013) Sink or source—the potential of coffee agroforestry systems to sequester atmospheric CO₂ into soil organic carbon. *Agric Ecosyst Environ* 175:60–68. <https://doi.org/10.1016/j.agee.2013.04.012>
- Oertel C, Matschullat J, Zurba K, et al (2016) Greenhouse gas emissions from soils—a review. *Geochemistry* 76:327–352. <https://doi.org/10.1016/j.chemer.2016.04.002>
- Oh J, Silverstein J (1999) Oxygen inhibition of activated sludge denitrification. *Water Res* 33:1925–1937. [https://doi.org/10.1016/S0043-1354\(98\)00365-0](https://doi.org/10.1016/S0043-1354(98)00365-0)
- Ollivier QR, Maher DT, Pitfield C, Macreadie PI (2019) Winter emissions of CO₂, CH₄, and N₂O from temperate agricultural dams: fluxes, sources, and processes. *Ecosphere* 10:e02914. <https://doi.org/10.1002/ecs2.2914>
- Outram FN, Hiscock KM (2012) Indirect nitrous oxide emissions from surface water bodies in a lowland arable catchment: a significant contribution to agricultural greenhouse gas budgets? *Environ Sci Technol* 46:8156–8163. <https://doi.org/10.1021/es3012244>
- Peacock M, Audet J, Bastviken D, et al (2021a) Global importance of methane emissions from drainage ditches and canals. *Environ Res Lett* 16:044010. <https://doi.org/10.1088/1748-9326/abeb36>
- Peacock M, Audet J, Bastviken D, et al (2021b) Small artificial waterbodies are widespread and persistent emitters of methane and carbon dioxide. *Glob Chang Biol* 27:5109–5123. <https://doi.org/10.1111/gcb.15762>

- Pihlatie M, Syväsallo E, Simojoki A, Esala M, Regina K (2004) Contribution of nitrification and denitrification to N₂O production in peat, clay and loamy sand soils under different soil moisture conditions. *Nutri Cycl Agroecosyst* 70:135–141. <https://doi.org/10.1023/B:FRES.0000048475.81211.3c>
- Pionke HB, Gburek WJ, Sharpley AN (2000) Critical source area controls on water quality in an agricultural watershed located in the Chesapeake Basin. *Ecol Eng* 14:325–335. [https://doi.org/10.1016/S0925-8574\(99\)00059-2](https://doi.org/10.1016/S0925-8574(99)00059-2)
- Poth M, Focht DD (1985) ¹⁵N Kinetic analysis of N₂O production by *Nitrosomonas europaea*: an examination of nitrifier denitrification. *Appl Environ Microbiol* 49:1134–1141
- Power AG (2010) Ecosystem services and agriculture: tradeoffs and synergies. *Philos Trans R Soc B* 365:2959–2971. <https://doi.org/10.1098/rstb.2010.0143>
- Rabalais NN, Turner RE, Wiseman WJ (2002) Gulf of Mexico hypoxia, a.k.a. “The Dead Zone.” *Annu Rev Ecol Syst* 33:235–263
- Rideout NK, Lapen DR, Peters DL, Baird DJ Ditch the low flow: Agricultural impacts on flow regimes and consequences for aquatic ecosystem functions. *Ecohydrology* n/a:e2364. <https://doi.org/10.1002/eco.2364>
- Rochette P (2011) Towards a standard non-steady-state chamber methodology for measuring soil N₂O emissions. *Anim Feed Sci Technol* 166–167:141–146. <https://doi.org/10.1016/j.anifeedsci.2011.04.063>
- Rochette P, Worth DE, Lemke RL, et al (2008) Estimation of N₂O emissions from agricultural soils in Canada. I. Development of a country-specific methodology. *Can J Soil Sci* 88:641–654. <https://doi.org/10.4141/CJSS07025>
- Ryden JC, Syers JK, Harris RF (1974) Phosphorus in runoff and streams. In: Brady NC (ed) *Advances in Agronomy*. Academic Press, pp 1–45
- Schaufler G, Kitzler B, Schindlbacher A, et al (2010) Greenhouse gas emissions from European soils under different land use: effects of soil moisture and temperature. *Eur J Soil Sci.* 61:683–696. <https://doi.org/10.1111/j.1365-2389.2010.01277.x>
- Schipper LA, Hobbs JK, Rutledge S, Arcus VL (2014) Thermodynamic theory explains the temperature optima of soil microbial processes and high Q₁₀ values at low temperatures. *Glob Chang Biol* 20:3578–3586. <https://doi.org/10.1111/gcb.12596>
- Sims JT, Simard RR, Joern BC (1998) Phosphorus loss in agricultural drainage: historical perspective and current research. *J Environ Qual* 27:277–293. <https://doi.org/10.2134/jeq1998.00472425002700020006x>
- Sirin AA, A CA, Suvorov GG, et al (2012) Values of methane emission from drainage ditches. *Environ Dynam Glob Clim Chang* 3:1–10. <https://doi.org/10.17816/edgcc321-10>

- Smith DR, Huang C (2010) Assessing nutrient transport following dredging of agricultural drainage ditches. *Trans ASABE* 53:429–436
- Smith DR, Pappas EA (2007) Effect of ditch dredging on the fate of nutrients in deep drainage ditches of the midwestern United States. *J Soil Water Conserv* 62:252–261
- Spivakov BY, Maryutina TA, Muntau H (1999) Phosphorus speciation in water and sediments. *Pure Appl Chem* 71:2161–2176. <https://doi.org/10.1351/pac199971112161>
- Strand JA, Weisner SEB (2013) Effects of wetland construction on nitrogen transport and species richness in the agricultural landscape—experiences from Sweden. *Ecol Eng* 56:14–25. <https://doi.org/10.1016/j.ecoleng.2012.12.087>
- Strock JS, Kleinman PJA, King KW, Delgado JA (2010) Drainage water management for water quality protection. *J Soil Water Conserv* 65:131A–136A. <https://doi.org/10.2489/jswc.65.6.131A>
- Stumm W, Morgan JJ (2012) *Aquatic Chemistry: Chemical Equilibria and Rates in Natural Waters*. John Wiley & Sons
- Stumpf RP, Wynne TT, Baker DB, Fahnenstiel GL (2012) Interannual variability of cyanobacterial blooms in Lake Erie. *PLOS ONE* 7:e42444. <https://doi.org/10.1371/journal.pone.0042444>
- Sunohara MD, Gottschall N, Wilkes G, et al (2015) Long-term observations of nitrogen and phosphorus export in paired-agricultural watersheds under controlled and conventional tile drainage. *J Environ Qual* 44:1589–1604. <https://doi.org/10.2134/jeq2015.01.0008>
- US EPA (2016) *Understanding Global Warming Potentials*. <https://www.epa.gov/ghgemissions/understanding-global-warming-potentials>. Accessed 7 Dec 2022
- Vaccari DA (2009) Phosphorus: a looming crisis. *Scientific American* 300:54–59
- Van Zandvoort A, Lapen DR, Clark ID, et al (2017) Soil CO₂, CH₄, and N₂O fluxes over and between tile drains on corn, soybean, and forage fields under tile drainage management. *Nutr Cycl Agroecosyst* 109:115–132. <https://doi.org/10.1007/s10705-017-9868-4>
- Vidon P, Jacinthe P-A, Liu X, et al (2014) Hydrobiogeochemical controls on riparian nutrient and greenhouse gas dynamics: 10 years post-restoration. *JAWRA J Am Water Resour Assn* 50:639–652. <https://doi.org/10.1111/jawr.12201>
- Vidon P, Marchese S, Welsh M, McMillan S (2015) Impact of precipitation intensity and riparian geomorphic characteristics on greenhouse gas emissions at the soil-atmosphere interface in a water-limited riparian zone. *Water Air Soil Pollut* 227:8. <https://doi.org/10.1007/s11270-015-2717-7>

- Vitousek PM, Aber JD, Howarth RW, et al (1997) Human alteration of the global nitrogen cycle: sources and consequences. *Ecol Appl* 7:737–750. [https://doi.org/10.1890/1051-0761\(1997\)007\[0737:HAOTGN\]2.0.CO;2](https://doi.org/10.1890/1051-0761(1997)007[0737:HAOTGN]2.0.CO;2)
- Vymazal J, Březinová TD (2018) Removal of nutrients, organics and suspended solids in vegetated agricultural drainage ditch. *Ecol Eng* 118:97–103. <https://doi.org/10.1016/j.ecoleng.2018.04.013>
- Welsh MK, Vidon PG, McMillan SK (2021) Riparian seasonal water quality and greenhouse gas dynamics following stream restoration. *Biogeochemistry* 156:453–474. <https://doi.org/10.1007/s10533-021-00866-9>
- Whalen SC (2005) Biogeochemistry of methane exchange between natural wetlands and the atmosphere. *Environ Eng Sci* 22:73–94. <https://doi.org/10.1089/ees.2005.22.73>
- Wilkes G, Brassard J, Edge TA, et al (2014) Long-term monitoring of waterborne pathogens and microbial source tracking markers in paired agricultural watersheds under controlled and conventional tile drainage management. *Appl Environ Microbiol* 80:3708–3720. <https://doi.org/10.1128/AEM.00254-14>
- Wilson D, Farrell CA, Fallon D, et al (2016) Multiyear greenhouse gas balances at a rewetted temperate peatland. *Glob Chang Biol* 22:4080–4095. <https://doi.org/10.1111/gcb.13325>
- Woltemade CJ (2000) Ability of restored wetlands to reduce nitrogen and phosphorus concentrations in agricultural drainage water. *J Soil Water Conser* 55:303–309

Appendix A: Tables and Figures for Chapter 2

A.1 GHG data cleaning

GHG concentration time series data typically will form a linear trend. Occasionally one data point was quite divergent from the trend indicated by the other four data points in the time series. This could be due to a contaminated Exetainer vial that was not evacuated properly prior to sampling, poor fastening of a GHG chamber lid on the GHG collar, technician error when inserting the gas sample syringe into the Exetainer vial septa cap, or a leaking seal on the septa cap. Technician error can be significant due to the challenge of sampling with sharp syringe needles at specific time intervals while standing in a drainage ditch with sloped uneven and slippery terrain and flowing water. Another issue is that the first sample (0 min) can sometimes be off the trend of the other samples. The first sample is a measure of ambient GHG concentrations and sampled near the soil at the GHG collars before the GHG chamber lids were placed on top of the collars. The deviance from the other samples can be due to this sample being obtained in an ambient environment which may have effects from wind mixing or possibly the technician exhaling near the sample. To ensure slopes are real trends in GHG concentration, all time series trends were manually inspected with slope and R^2 calculation. If four out of five data points showed a strong trend but one data point was subjectively off the trend line, that a single data point was removed to clean the dataset (Fig. A.3).



Fig. A.1 Photographs of ditch and bank brush cutting and ditch dredging.

Top row: a) Control Tree May 2019, b) Brush cutting trees (February 2018), c) Brush Cut Tree (May 2019), d) Brush Cut Tree (Aug 2019). Bottom row: e) Control Shrub (May 2019), f) ditch dredging (November 2018), g) Brush Cut Shrub (May 2019), h) Brush Cut Shrub (August 2019).

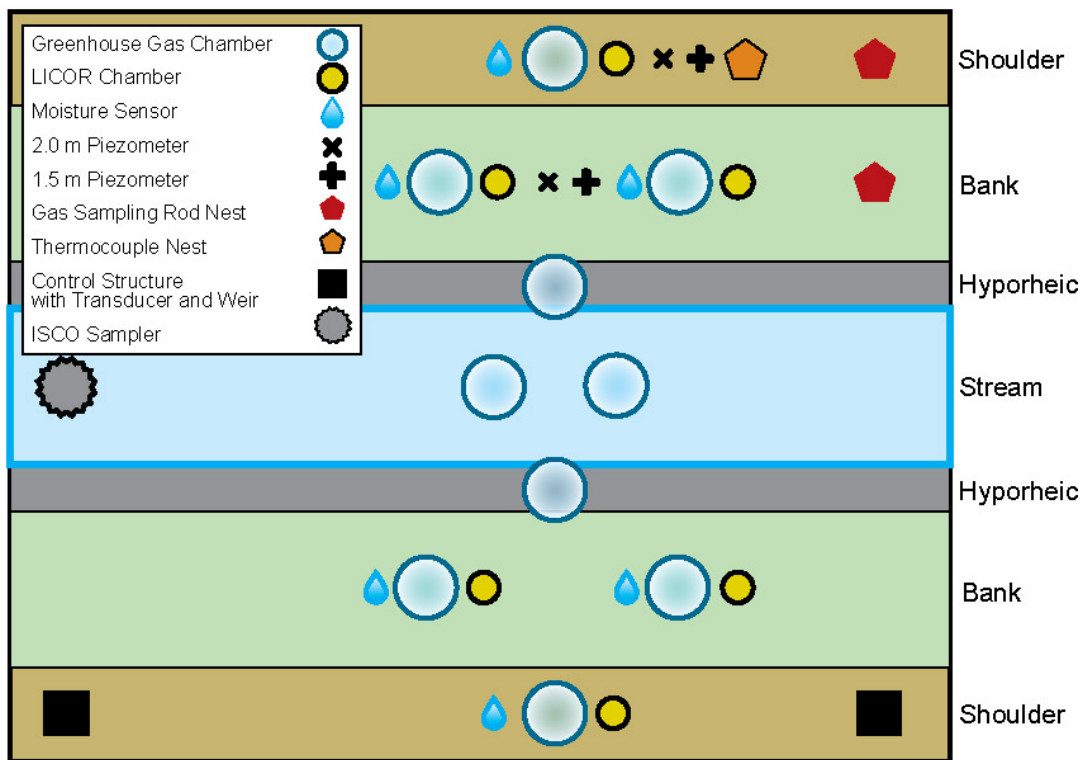
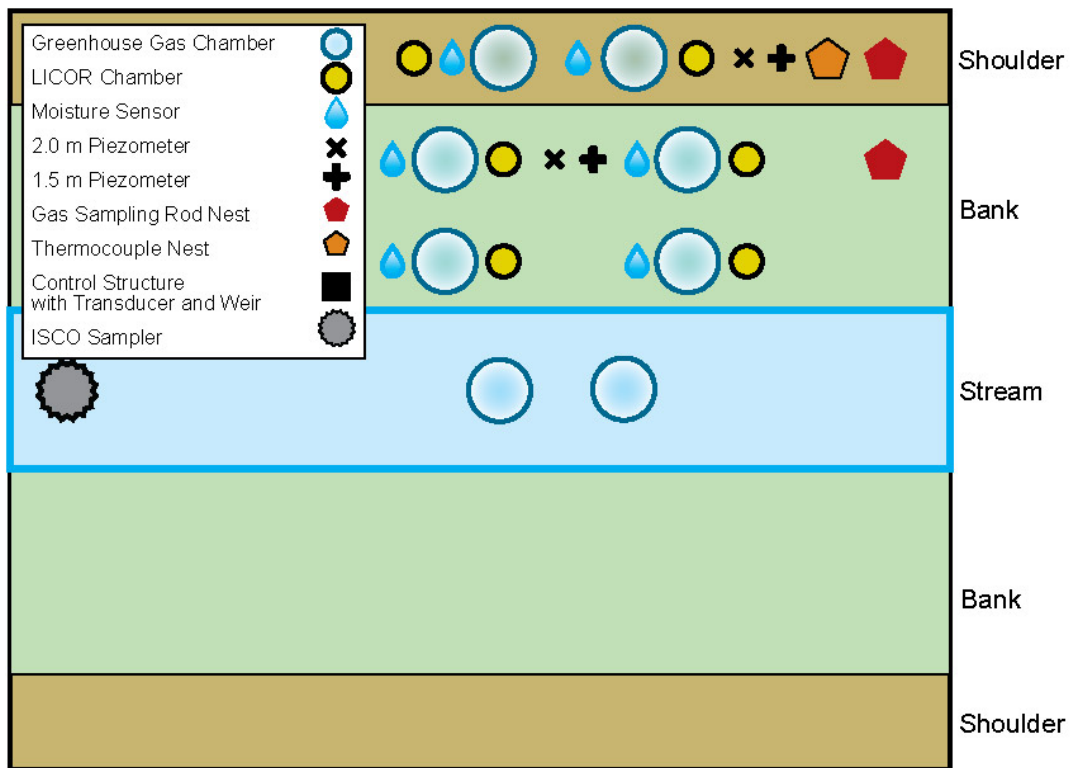


Fig. A.2 Overhead view of microplot site showing layout of instrumentation. Deployed in 2018 above 2019 and 2020 on bottom. Instrumentation type in legend. Ditch zone on right.

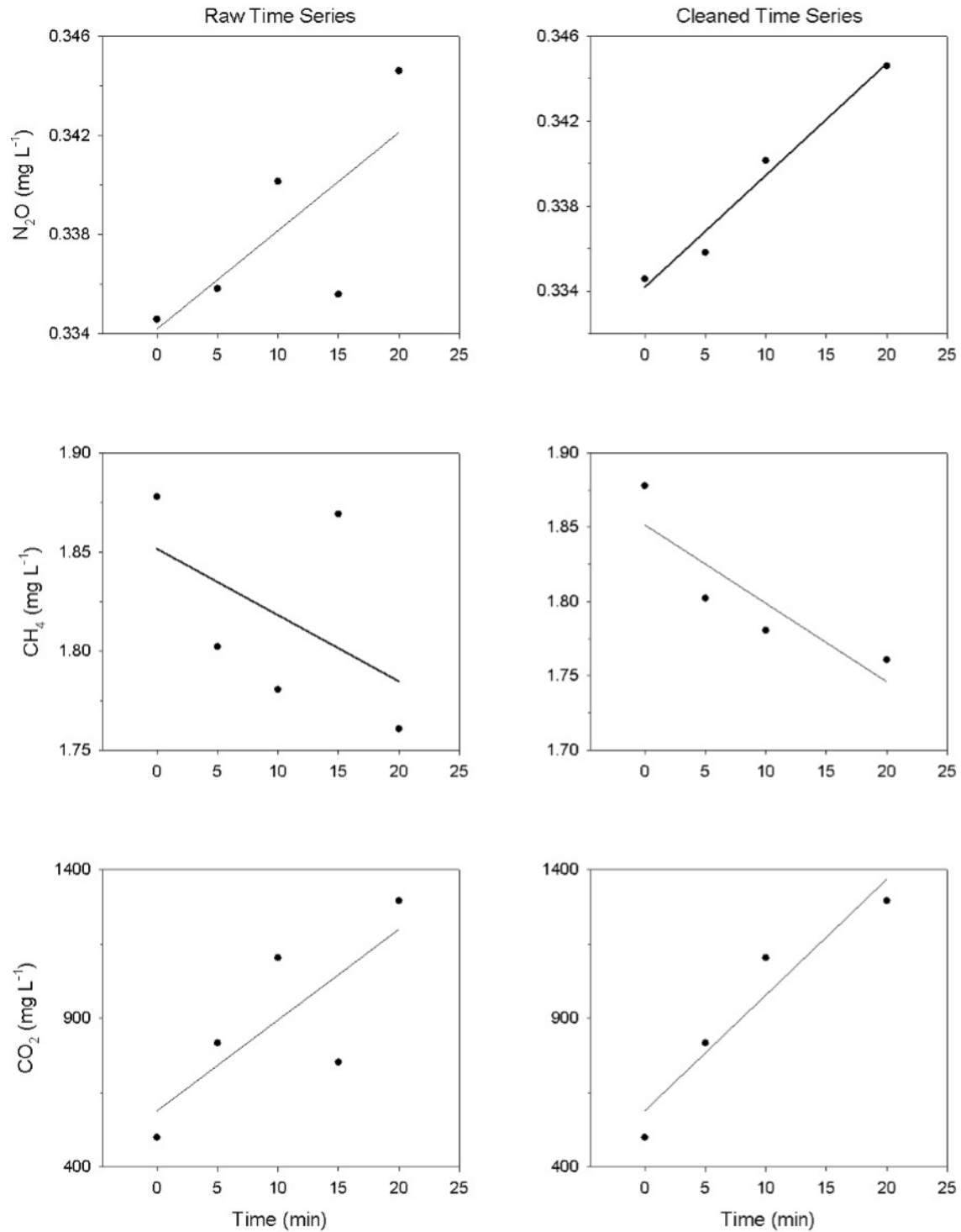


Fig. A.3 Comparison of GHG concentration time series data before and after manual data cleaning. Data is for single GHG chamber location on southwest bank collected on July 9, 2019. Line is linear fit where slope is used in subsequent GHG chamber flux data. From top to bottom: N_2O , CH_4 , CO_2 . From left to right: raw data, manually cleaned data.

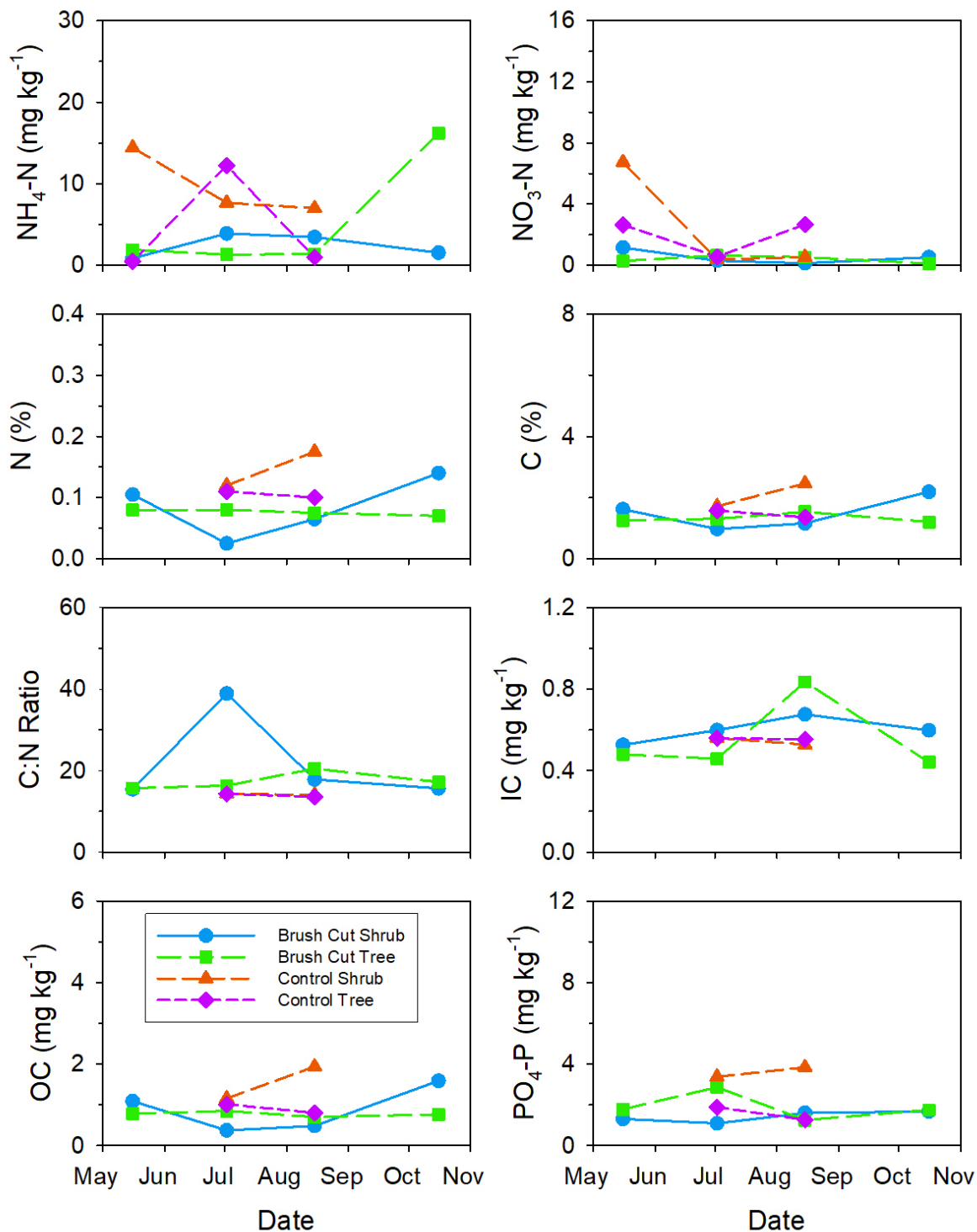


Fig. A.4 Temporal changes in Hyporheic zone soil chemical composition for 2019 at 0-15 cm depth for Brush Cut Shrub •, Brush Cut Tree ■, Control Shrub ▲, Control Tree ◆. From top to bottom: NH₄-N, NO₃-N, N, C, C:N ratio, inorganic carbon, organic carbon, plant available phosphorus.

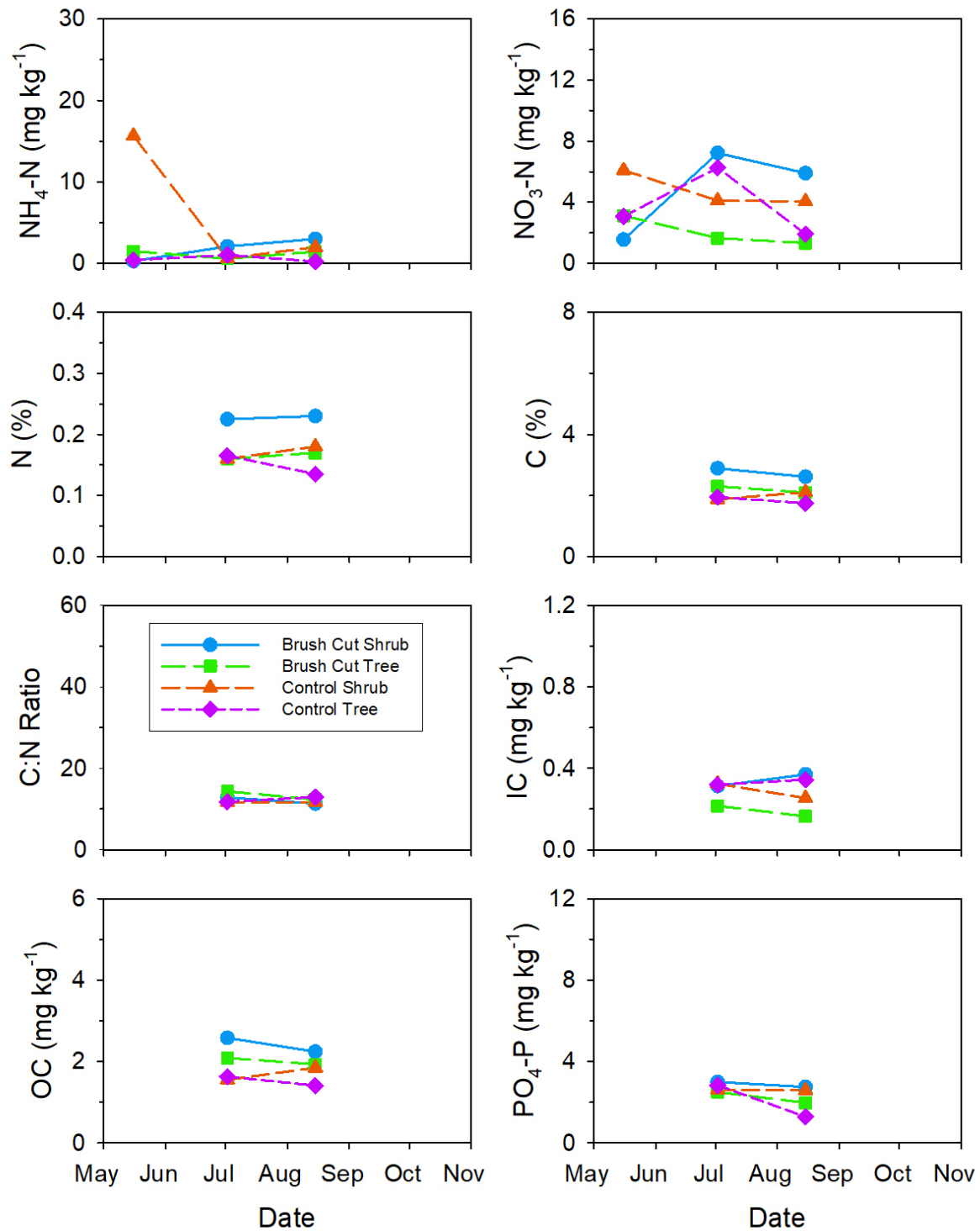


Fig. A.5 Temporal changes in Bank zone soil chemical composition for 2019 at 0-15 cm depth for Brush Cut Shrub •, Brush Cut Tree ■, Control Shrub ▲, Control Tree ◆. From top to bottom: $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, N, C, C:N ratio, inorganic carbon, organic carbon, plant available phosphorus.

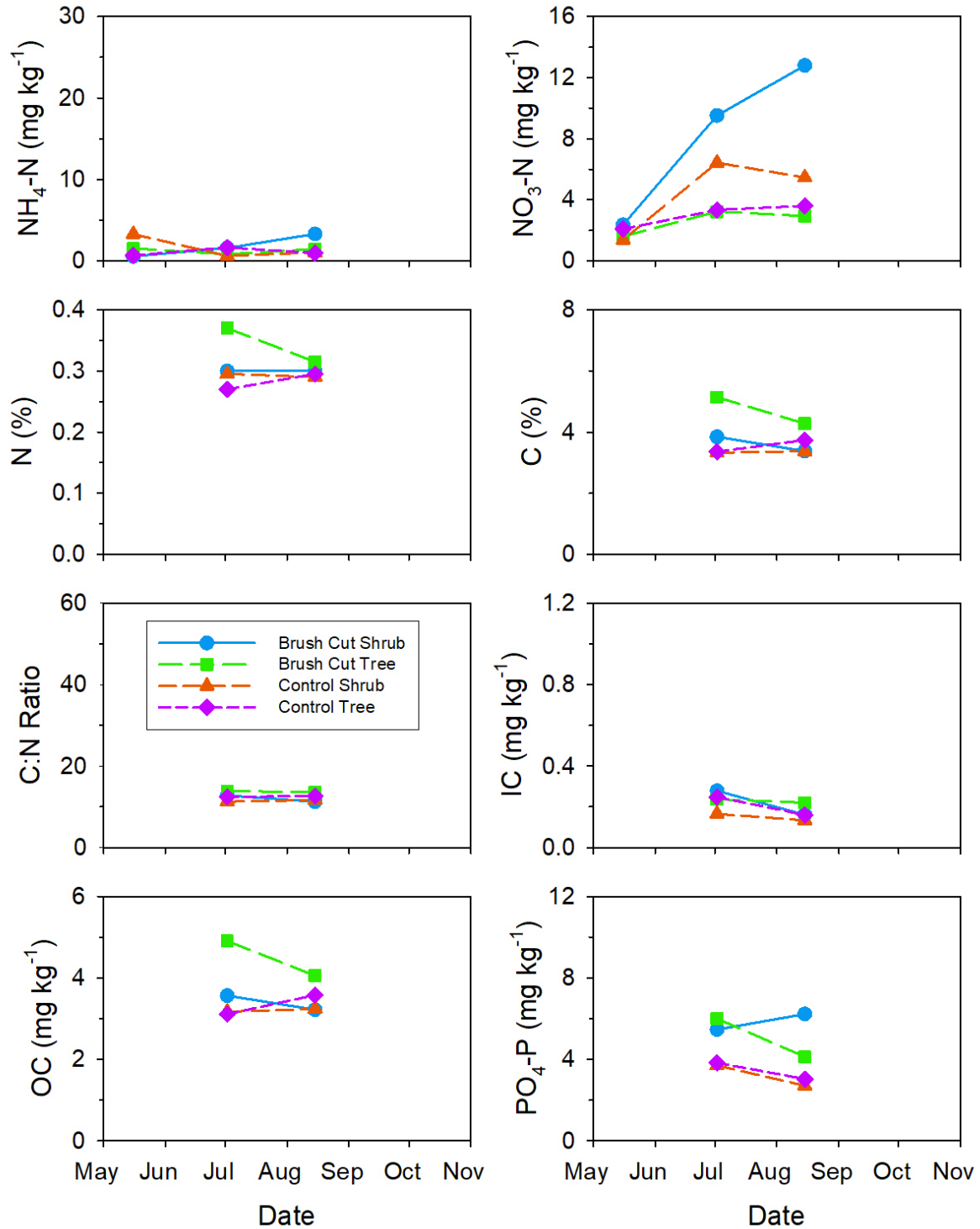


Fig. A.6 Temporal changes in Shoulder zone soil chemical composition for 2019 at 0-15 cm depth for Brush Cut Shrub •, Brush Cut Tree ■, Control Shrub ▲, Control Tree ◆. From top to bottom: $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, N, C, C:N ratio, inorganic carbon, organic carbon, plant available phosphorus.

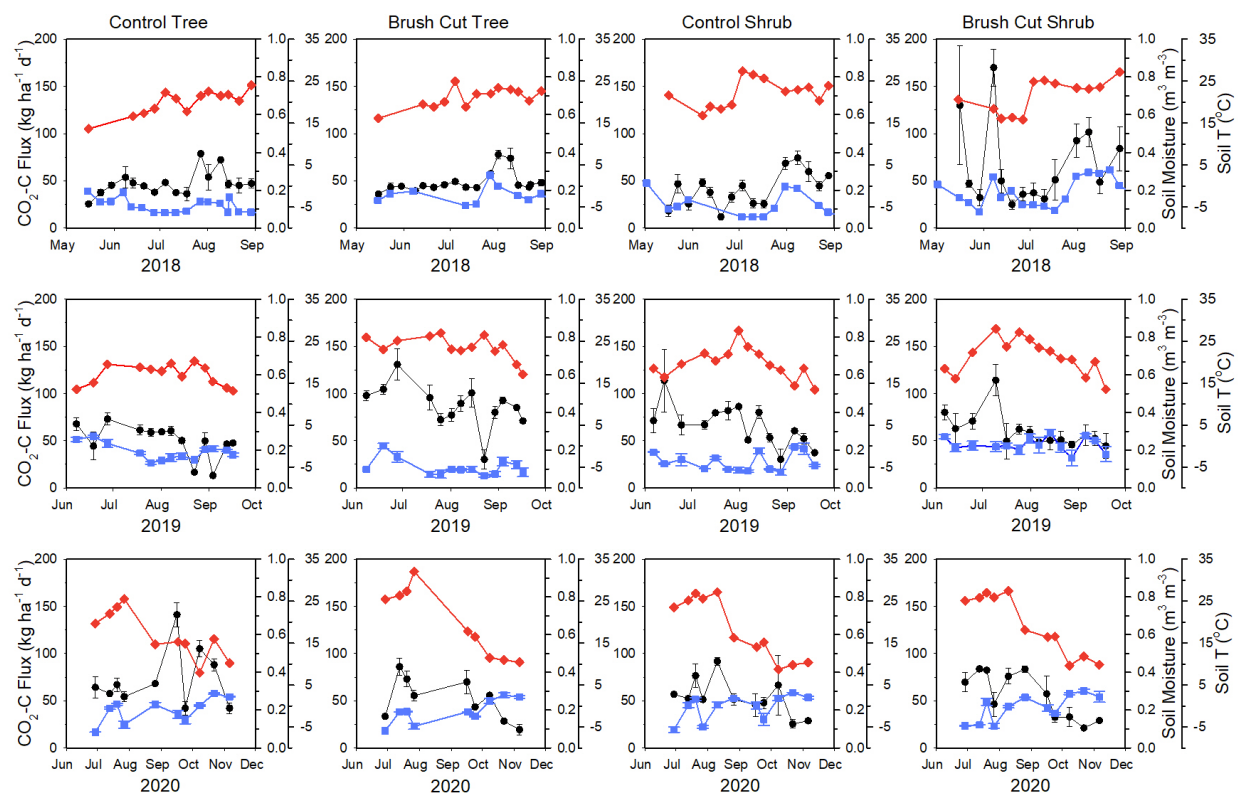


Fig. A.7 CO₂ Flux •, Soil Moisture ■, and Soil T ♦ time series with calculated standard error bars from LI-COR measurements in agricultural drainage ditch Shoulder zone. From top to bottom 2018, 2019, and 2020 sampling seasons. From left to right Control Tree, Brush Cut Tree, Control Shrub, Brush Cut Shrub sites.

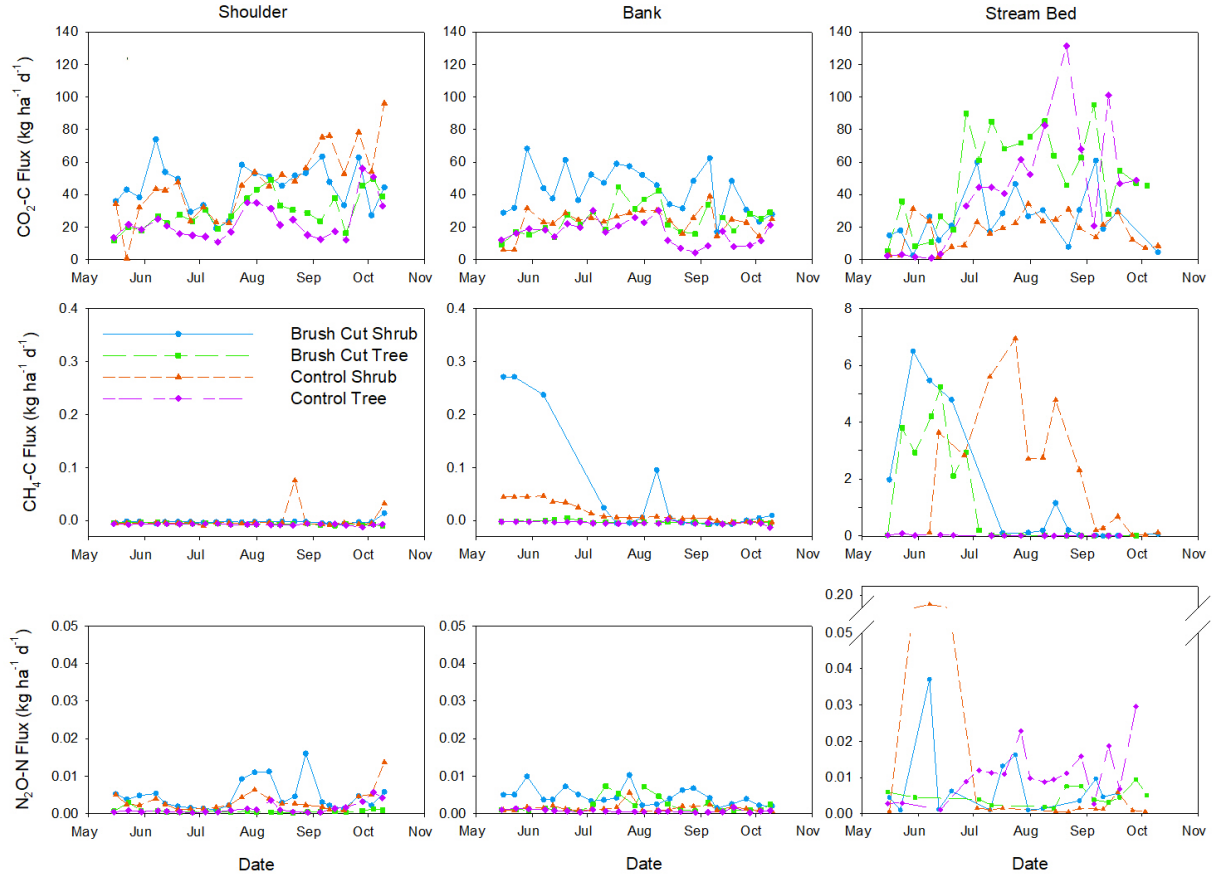


Fig. A.8 Time Series of CO₂, CH₄, N₂O Fluxes from May until November 2018 for Brush Cut Shrub •, Brush Cut Tree ■, Control Shrub ▲, Control Tree ◆. Only flux with $R^2 > 0.8$ included. Note different scale for CH₄ flux from Stream Bed. From top to bottom: CO₂-C, CH₄-C, N₂O-N fluxes. From left to right Shoulder, Bank, Instream zones. 2018 data used different chamber layout than 2019 and 2020. Bank includes Bank and Hyporheic zones.

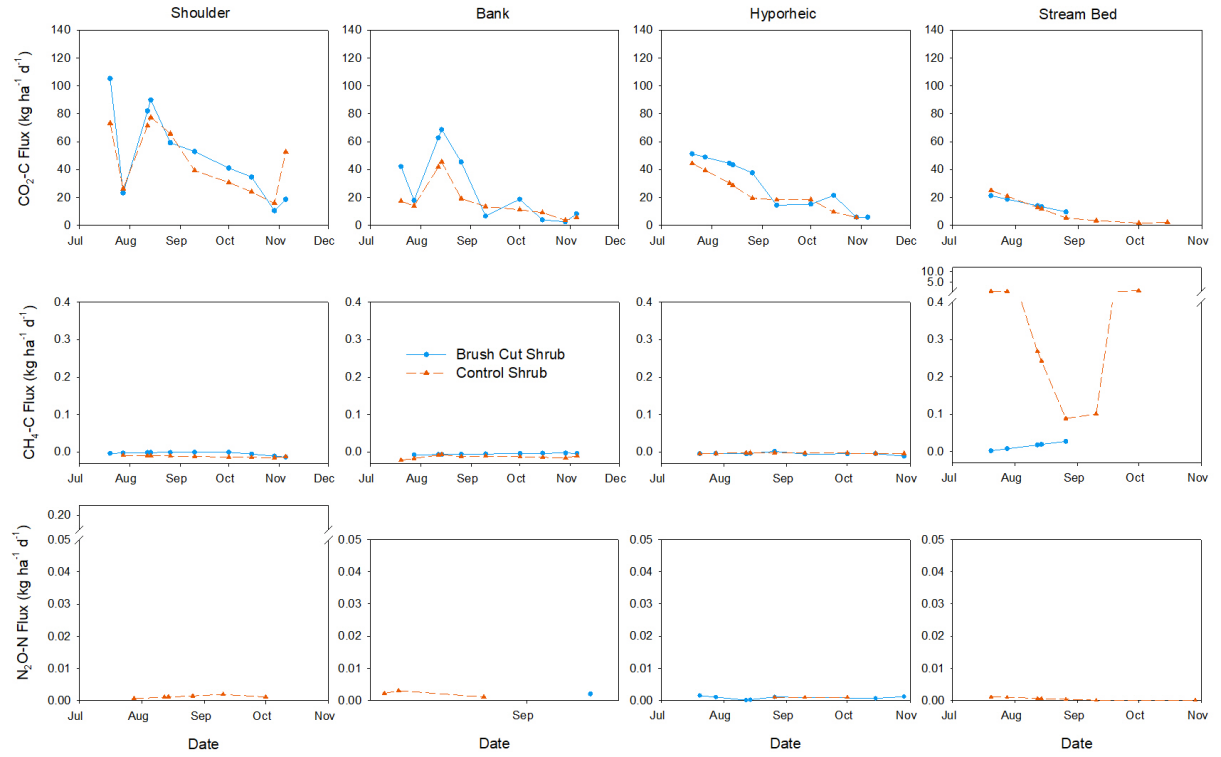


Fig. A.9 Time Series of CO₂, CH₄, N₂O Fluxes from July until November 2020 for Brush Cut Shrub •, Brush Cut Tree ■, Control Shrub ▲, Control Tree ◆. Only flux with R²>0.8 included. From top to bottom: CO₂-C, CH₄-C, N₂O-N fluxes. From left to right Shoulder, Bank, Hyporheic, Instream zones.

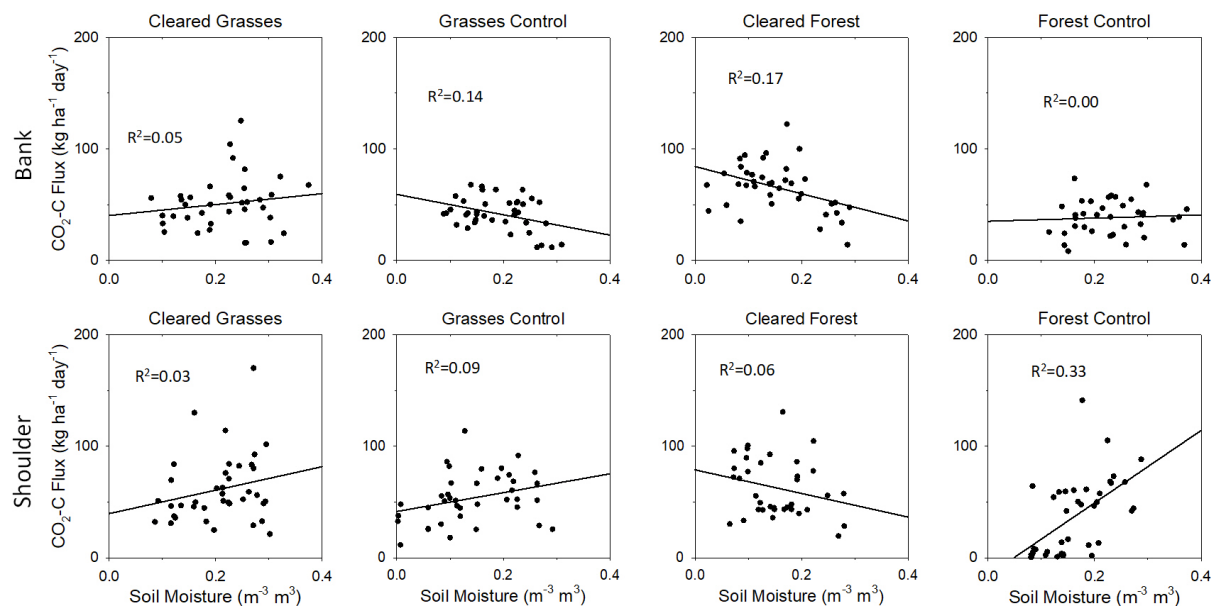


Fig. A.10 LI-COR CO₂ flux versus soil moisture content with calculated R² correlation coefficients. Line is a linear regression to visually indicate correlation. Drainage ditch regions: top row Bank, bottom row Shoulder. Ditch sites from left to right: Brush Cut Shrub, Control Shrub, Brush Cut Tree, Control Tree.

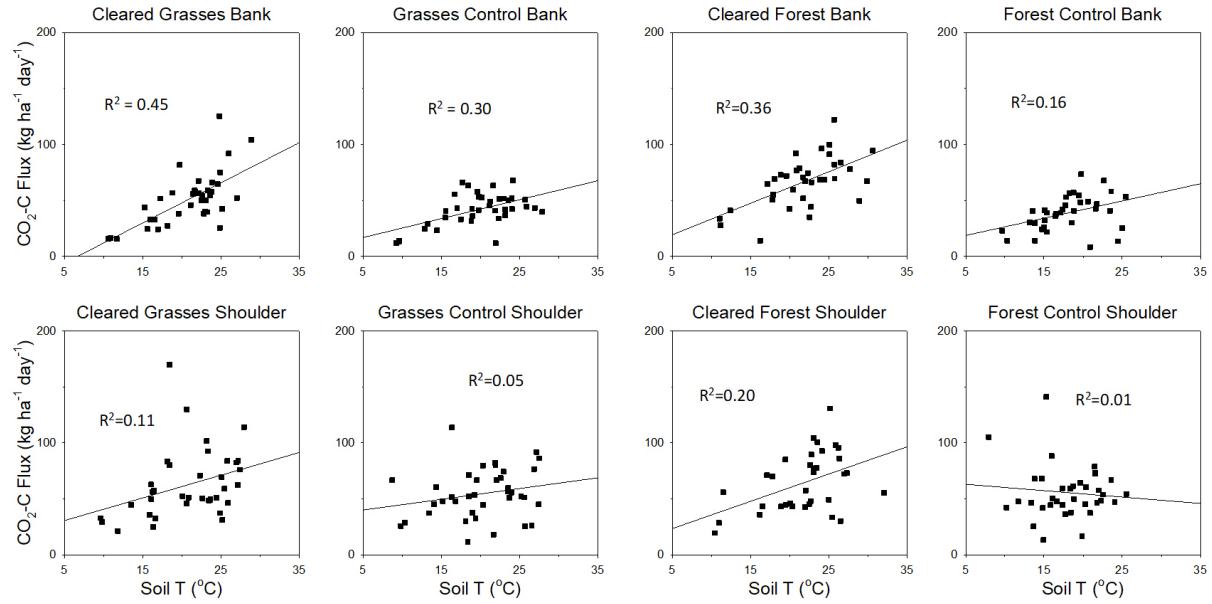


Fig. A.11 LI-COR CO₂ flux versus soil T with calculated R² correlation coefficients. Line is a linear regression to visually indicate correlation. Drainage ditch regions: top row Bank, bottom row Shoulder. Ditch sites from left to right: Brush Cut Shrub, Control Shrub, Brush Cut Tree, Control Tree.

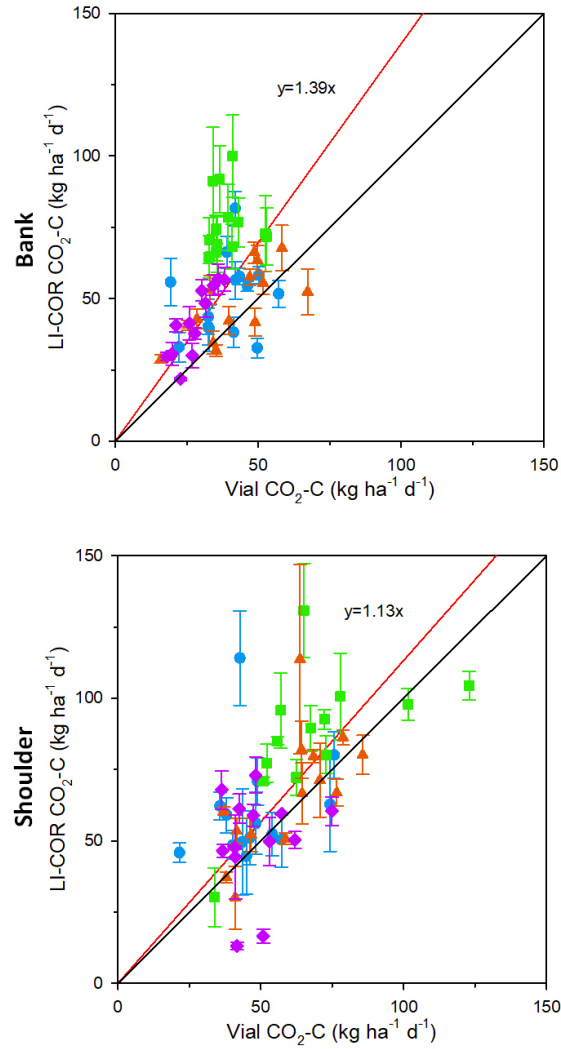


Fig. A.12 LI-COR CO₂ flux with calculated standard error versus Vial method CO₂ flux during 2019 sampling season for Brush Cut Shrub •, Brush Cut Tree ■, Control Shrub ▲, Control Tree ♦. Black line represents theoretical LI-COR=Vial method. Red line represents linear fit with no intercept given by equation above it.

Table A.1 Multiple Linear Regressions LI-COR Data, CO₂ dependent variable, Soil Moisture and Soil T independent variables

Region Name	N	Soil M		Soil T	
		P Value	Significant	P Value	Significant
All sites bank and shoulder	243	0.825	No	<0.001	Yes
Brush Cut Shrub Bank	33	0.060	No	<0.001	Yes
Brush Cut Shrub Shoulder	33	0.466	No	0.062	No
Brush Cut Tree Bank	28	0.690	No	<0.001	Yes
Brush Cut Tree Shoulder	28	0.731	No	0.029	Yes
Control Shrub Bank	32	0.238	No	0.016	Yes
Control Shrub Shoulder	32	0.545	No	0.137	No
Control Tree Bank	28	0.114	No	<0.001	Yes
Control Tree Shoulder	28	0.581	No	0.745	No

Table A.2 Mean and standard error measurements over three years for parameters at different microplot sites

Data Set	Brush Cut Shrub (Site 1)		Brush Cut Tree (Site 5)		Control Shrub (Site 10)		Control Tree (Site 11)	
	Mean	SEM	Mean	SEM	Mean	SEM	Mean	SEM
LI-COR Bank CO ₂	52	4	66	3	42	2	39	2
Bank Soil M	0.23	0.01	0.15	0.01	0.19	0.01	0.23	0.01
Bank Soil T	21	1	22	1	20	1	18	1
Vial Shoulder CO ₂	47	3	45	4	52	3	33	3
Vial Bank CO ₂	38	2	31	2	29	2	22	2
Vial Hyporheic CO ₂	19	4	8	2	24	5	8	2
Vial Instream CO ₂	26	3	41	5	20	2	37	6

Table A.3 One way ANOVA analysis over three years for parameters

Data Set	N	Normality	P	Equal Variance	P	ANOVA significant (Y/N)	P	Holm Sidak Paired T-Test Results
LI-COR Bank CO ₂	41	Failed	< 0.050	Passed	0.078	Y	<0.001	5 significantly higher than 10, 11
Bank Soil M	41	Passed	0.093	Passed	0.656	Y	<0.001	5 significantly lower than 1, 11. 10 significantly lower than 11

Bank Soil T	41	Failed	<0.050	Passed	0.797	Y	0.005	11 significantly lower than 1, 5
Vial Shoulder CO ₂	37	Failed	<0.050	Passed	0.165	Y	<0.001	11 significantly lower than 1, 10
Vial Bank CO ₂	37	Passed	0.884	Failed	<0.050	Y	<0.001	11 significantly lower than 1, 5. 10 significantly lower than 1
Vial Hyporheic CO ₂	22	Passed	0.577	Failed	<0.050	Y	0.041	11 significantly lower than 10
Vial Instream CO ₂	30	Failed	<0.050	Failed	<0.050	Y	0.002	10 significantly lower than 5, 11