Forest management impacts on greenhouse gas fluxes from riparian soils along headwater streams

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Manuscript highlights

- Clear-cutting without a buffer decreased nitrous oxide emission and methane uptake
- Riparian buffers appeared effective in mitigating forest harvest effects
- Groundwater discharge areas had lower methane uptake, especially when harvested

Abstract

Increasing concentrations of atmospheric greenhouse gases (GHGs; CO_2, CH_4, N_2O) causes climate change. Depending on the conditions, soils have the potential to store carbon or to be a source of GHGs to the atmosphere. Riparian soils in particular have high potential to store carbon, but also to be sources of CH_4 and N_2O. Headwater streams make up a large proportion of stream length in a drainage network, and their riparian zones have valuable ecosystem functions. In parallel, the riparian zones of headwater streams are particularly vulnerable to forest harvest. Studies of GHG fluxes from these unique ecosystems remain limited. Our objective was to quantify the effects of forestry practices and groundwater discharge (DIS) areas on GHG emissions from riparian forest soils in coastal British Columbia. We compared nine sites with three different forest management protocols: 1) harvesting with a riparian buffer, 2) no buffer, and 3) reference sites without harvesting. We measured gas fluxes, soil temperature, soil moisture and depth to the groundwater table alongside headwater streams monthly over one growing season. We found that CH_4uptake rates were 65% lower at the no buffer sites, and N_2O emission rates were 52% lower at the no buffer sites, when compared to the reference sites. Additionally, CH_4 uptake was 54% lower at DIS areas than in non-DIS areas. The results of our research help inform forest management by demonstrating that maintaining riparian buffers can be effective in protecting the ecosystem functions contributing to soil GHG fluxes.

Keywords

Carbon dioxide | Headwater streams | Methane | Nitrous oxide | Forest harvest | Riparian buffer | Riparian zone

Introduction

Greenhouse gases (GHGs) such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) can absorb infrared radiation and trap heat in the atmosphere (IPCC 2007). This greenhouse effect is critical for maintaining livable conditions for life on earth, but anthropogenic activities have caused an unprecedented increase GHG concentrations in the atmosphere, leading to global warming and climate change (IPCC 2007). Soils play an important role in climate change, as they have the potential to store carbon or to be a source of greenhouse gases to the atmosphere (Oertel and others 2016).

Riparian zones are the three dimensional zones of direct interaction between terrestrial and aquatic ecosystems (Gregory and others 1991). Riparian ecosystems are a conservation priority due to their disproportionately high value and diversity of ecological functions as well as unique biodiversity combined with high vulnerability to anthropogenic pressures (Ramey and Richardson 2017; Capon and Pettit 2018). Their unique properties, including shallow water tables, high soil organic matter content, and high soil nitrogen concentrations, create the potential for significant carbon sequestration (Gundersen and others 2010), but also to produce significant amounts of anaerobically produced CH_4 and N_2O to the atmosphere (Knoepp and Clinton 2009; Vidon and others 2018).

Forest harvest is a disturbance that can alter biogeochemical processes in soils (Kreutzweiser and others 2008), resulting in a change in the GHG emission rates (Lavoie and others 2013). However, soil ecosystem responses to logging are highly variable and site specific due to microsite level differences in variables such as soil properties, moisture conditions, and biological interactions (Kreutzweiser and others 2008). Therefore, the results from the many upland forest studies are likely to differ from riparian zones.

The most common method of protecting streams and riparian areas from the impacts of forest harvest is the use of riparian buffer zones, consisting of a strip of trees adjacent to the stream that is either left uncut or has limited harvesting (Richardson and others 2012). The riparian zones of headwater streams are particularly vulnerable to forest harvest, as buffer zones are often not required along these small streams (Richardson and Danehy 2007). In British Columbia, 45% of surveyed small streams did not retain a riparian buffer after forest harvest (Kuglerová and others 2020). Headwater streams are small, ecologically significant, tributaries at the most upstream ends of a stream network that make up to 80% of stream length in a given drainage network (Leopold and others 1964). Although riparian buffers are effective in reducing the impacts of forest harvest by intercepting sediments, maintaining bank stability, and providing shading among other benefits (Richardson and others 2012), their usage remains contentious in headwater systems, due to the large volume of timber that buffers remove from commercial use (Richardson and Danehy 2007).

The major drivers of spatio-temporal variation in GHG flux rates include soil temperature and soil moisture (Luo and Zhou 2006). CO_2 emissions consist of the combined emissions from root respiration (autotrophic respiration) and microbial decomposition of organic matter (heterotrophic respiration), and as such, CO_2 emissions are affected by substrate availability (Luo and Zhou 2006). CH_4 is produced by methanogens in anaerobic conditions, and consumed by methanotrophs in aerobic conditions (Oertel and others 2016). Soil pH and substrate availability impact microbial communities contributing to CH_4exchange (Serrano-Silva and others 2014). N_2O is mainly produced by denitrification under anaerobic conditions, in addition to nitrification under aerobic conditions (Oertel and others 2016). Important factors influencing these processes are nitrogen deposition and fertilization, and soil pH (Dalal and Allen 2008).

Forests and trees are critical in regulating water, energy, and carbon cycles (Ellison and others 2017; Ilstedt and others 2016). Thus, deforestation can alter local scale warming, rainfall, water availability, and the emission of GHGs (Ellison and others 2017). Studies have evaluated the effects of clear-cutting on GHG fluxes from forest soils (Kähkönen and others 2002). Clear-cutting has been found to increase soil temperatures (Hashimoto and Suzuki 2004) and increase the elevation of the groundwater table (Bliss and Comerford 2002; Hotta and others 2010). These changes can be attributed to the drastic reduction in leaf area after harvesting, which reduces catchment-wide transpiration rates (Bliss and Comerford 2002) and allows for an increase in incoming sunlight, warming up the soil (Hashimoto and Suzuki 2004). Clear-cutting can increase CH_4 emissions due to greater soil moisture and temperature (Wu and others 2011) and increase N_2O emissions due to greater soil moisture and increased nitrogen availability as a result of increased rates of nitrogen mineralization and/or reduced competition from roots (Kellman and Kavanaugh 2008; Takakai and others 2008). Clear-cutting has inconsistent effects on CO_2 emission rates, with some studies reporting a decline (Striegl and Wickland 1998), an increase (Paul-Limoges and others 2015), or no change (Kähkönen and others 2002) in emissions following harvest. Nevertheless, to our knowledge, no one has yet examined the effects of forest harvest in riparian forests alongside streams on the rates of GHG fluxes.

In this study, we evaluated how forest management practices can alter biogeochemical processes that subsequently affect GHG fluxes from riparian soils along headwater streams. The objectives were: (1) to quantify the effects of forest harvest practices on soil GHG flux rates; (2) to determine if soil temperature, soil moisture, and/or groundwater level were dominant driver(s) of gas fluxes; and (3) to determine the effects of microtopographical variation on GHG flux rates.

Methods

Site description

The study sites were located in the 5,157 ha Malcolm Knapp Research Forest (MKRF), at the foothills of the Coast Mountains, about 40 km east of Vancouver, British Columbia (49° 16’ N, 122° 34’ W) (Figure 1). The biogeoclimatic zone is Coastal Western Hemlock (Klinka and others 2005), and the dominant tree species are Western Hemlock (*Tsuga heterophylla*), Douglas-fir (*Pseudotsuga menziesii*), and Western Red Cedar (*Thuja plicata*) (Klinka and others 2005). The forest is mostly comprised of approximately 90-year old second growth, naturally regenerated following widespread fire in 1925, and again in 1931 (Klinka and others 2005).

The climate is maritime, with slight continental influence due to the mountains and inland location (Klinka and others 2005). The primary climate (Köppen) classification is *Cfb* , temperate oceanic climate (Kottek and others 2006). The climate is characterized by mild temperatures, with wet, mild winters, and cool, relatively dry summers (Klinka and others 2005). Mean annual precipitation and air temperature at the Environment Canada climate station located at the Research Forest (Haney UBC RF Admin, station number 1103332) are 2131 mm and 9.7°C, respectively (data for 1962 to 2006). The annual range in mean monthly minimum and maximum temperature was 5.3°C to 13.8°C over the same period.

Glacial till and colluvium are the predominant parent materials in MKRF (Klinka 1976). In the southern portion, where our sites were located, surficial deposits include glacio-fluvial and glacio-marine deposits from Pleistocene era glaciation, overlaying compacted till or bedrock (Klinka and others 2005; Klinka 1976). The soils formed on these materials are shallow and can be expected to be coarse, acidic, and low in basic cations (Klinka and others 2005). The soils at all sites were designated a Humo-Ferric Podzol, except one site (E10B) had soil of the Organic Order (likely a Hydric Mesisol). Previously reported properties of mineral soil in undisturbed forest stands in Malcolm Knapp Research Forest included 39 Mg ha^{-1} total carbon content, 2.5 kg ha^{-1} nitrate content, and 11.5 kg ha^{-1} available phosphorous content (Turk and others 2008).

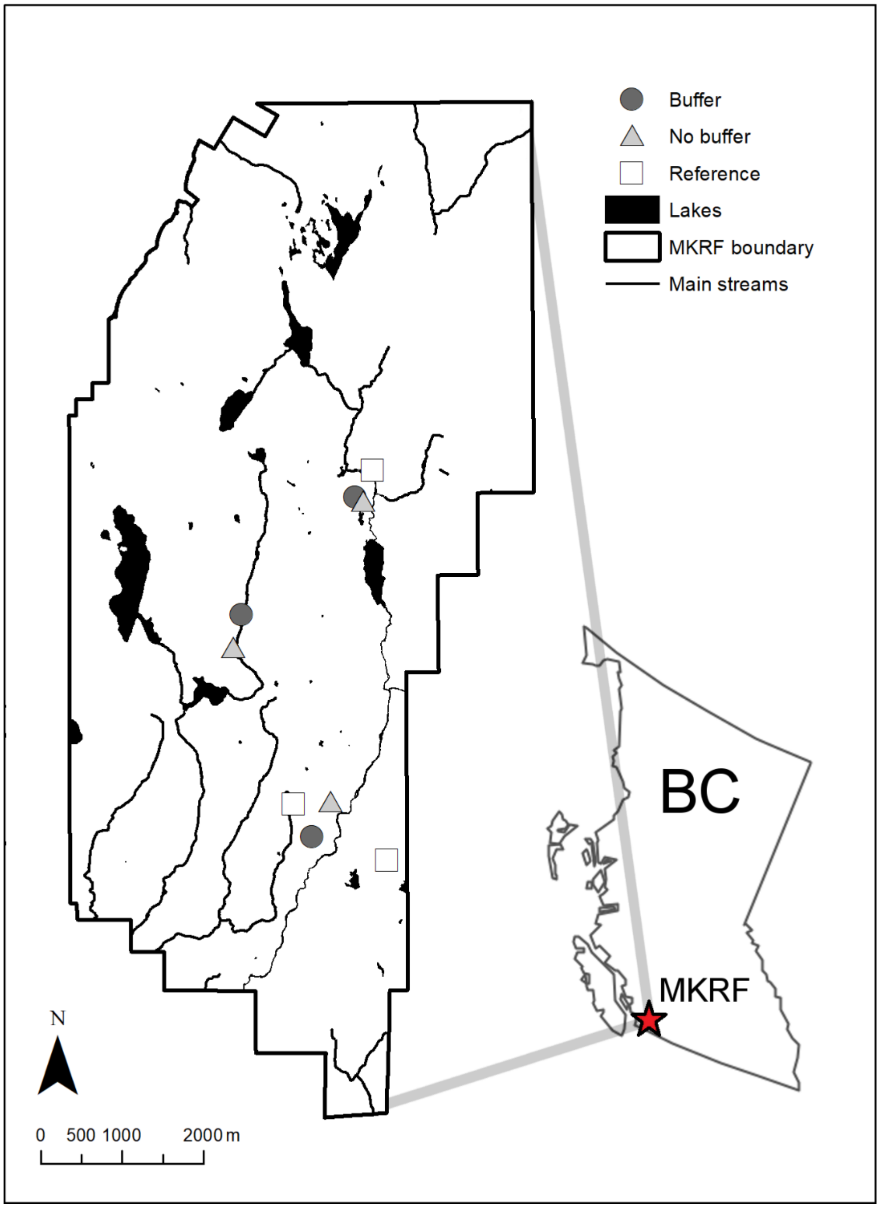

Figure 1. Location of study sites alongside headwater streams at Malcolm Knapp Research Forest (MKRF), British Columbia, Canada.

Study design and criteria

This study compared three different forest management practices in and near the riparian zone of headwater streams. A total of nine (n = 9) headwater streams were chosen as field sites; three sites were not recently logged (˜ 90 year-old stands), representing relatively undisturbed reference conditions ("reference"); three sites had a riparian buffer (reserve) zone of trees left standing along each side of the stream ("buffer"); and three sites were clear-cut to the stream margin without a riparian buffer zone ("no buffer").

Field site selection criteria were chosen *a priori* to identify comparable streams and riparian zones (Table 1). Using geographic information system data provided by MKRF, we identified clear-cut cut-blocks that were harvested no more than five years earlier, in Arcmap 10.6.1 (ESRI, Redlands, CA, USA). The selected buffer and no buffer sites were logged three to five years earlier. At the buffer sites, the average width of the buffer zone was 13.0 ± 2.8 m, and ranged from 11.0 to 16.2 m. The study sites met Richardson and Danehy's (2007) criteria for a headwater stream of a mean width <3 m and catchment area <100 ha. The riparian zones at the study reaches had a gentle slope gradient below 30°, with an average bank slope of 13.2° ± 8.4 (Table 1). This ensured comparability of groundwater table dynamics between sites, since stream water levels have more control on riparian water table levels when the stream bank slope has a lower gradient (Burt and others 2002).

Table 1. Characteristics of riparian study sites alongside headwater streams under differing forestry management conditions in Malcolm Knapp Research Forest, British Columbia, Canada. Values represent mean ± *SD* of three sites for each treatment. See Table A1 in Appendix A for the values for each site.

Reference	No buffer	Buffer
Stream bankfull width (m)	2.3 ± 1.7	1.4 ± 0.7
Catchment area (ha)	27.3 ± 19.0	6.63 ± 6.3
Elevation (m)	305.5 ± 36.5	329 ± 37.4
Reach length (m)	64.0 ± 9.2	46.3 ± 11.2
Bank slope (°)	37.7 ± 22.8	18.8 ± 6.6
Stream slope (°)	8.7 ± 5.2	7.9 ± 3.7

Greenhouse gas and soil sampling

The net terrestrial biosphere-atmospheric exchange of CO_2, CH_4, and N_2O was measured using closed, static chambers (Figure A1 in Supplemental Materials). Gas samples were collected on approximately a monthly basis from June to September, 2019. At each site, six chambers were placed along the study reach, on level ground, about 1 to 2 m from the stream bank-full margin, in order to capture an area within the zone of influence of the stream (Gregory and others 1991). Chambers (diameter of 30.5 cm, height of 23 cm) made of grey PVC pipe were permanently inserted about 10 cm below the soil surface at least 10 days (on average 23 days) prior to the first gas sampling to reduce the effects of soil and root disturbances.

At each site, the chambers were stratified according to local groundwater discharge conditions, in groundwater discharge (DIS) and non-groundwater discharge (ND) microsites, in order to account for some of the high spatial variability associated with GHG fluxes from soils (Vidon and others 2015). These DIS areas, or discrete riparian inflow points, occur when upland-originating groundwater converges and discharges in a depression in the topography of the riparian zone (Kuglerová and others 2014). The DIS areas were identified in Arcmap 10.6.1 using a 1 m digital elevation model (DEM) and flow accumulation modelling using a channelization threshold of 1 ha. This modelling process assumes that topography and gravity control water movement, and that the groundwater flow path follows the ground surface (Kuglerová and others 2014). The DIS areas were then confirmed with field observations of topography, wetness, and hydrophilic vegetation.

During gas sampling, a PVC lid was placed on top of the chamber and headspace air samples were taken at 0, 15, 30, and 45 minutes after closure. Headspace air samples of 20 mL were taken from a rubber septa sampling port in the middle of the lid using a 23 gauge needle and a 50 mL syringe after pumping 20 mL of the headspace gas twice to facilitate mixing. The gas sample was then injected into a pre-evacuated 12 mL exetainer (LabCo Ltd., Lampeter, Wales) until over-pressurized. After gas sampling, air temperature of the headspace was recorded and two ambient air samples were taken for reference.

Sampling was performed between 9:15 and 16:30 h to capture peak fluxes and reduce the effects of diurnal variation (Parkin and Venterea 2010). Given the difficulty of sampling 54 chambers in one day, three sites were sampled per day over three days, in randomized order within treatment, each month. Gas samples were analysed on a 7890A gas chromatograph (Agilent Technologies Inc., CA, USA) equipped with a flame ionization detector and an electron capture detector (Agilent Technologies Inc., CA, USA).

On each gas sampling date, volumetric soil moisture (± 3%) was recorded at each chamber using a GS3 ProCheck portable probe (Decagon Devices, Inc., Washington, USA) by using the mean of three readings, each no more than 0.5 m from each chamber, at a depth of 5.5 cm from the LFH layer. The depth to the water table was also measured on each sampling date by blowing into a thin tube attached to a meter stick lowered into a well until the bubbling noise of the groundwater was heard, and corresponding depth on the meter stick was noted. The well was made of perforated PVC pipe wrapped in landscape fabric installed at least 40 cm deep into the soil. Based on the groundwater table level data from the wells installed at each chamber, eight DIS areas were re-classified after the fact in cases where the well was dry for at least 80% of the sampling occasions. Over the entire sampling period, continuous soil temperature readings (± 0.5°C) were taken at 1 hr intervals using iButton® dataloggers (DS1992L- Thermochron and DS1923- Hygrochron, Maxim Integrated Products, USA) buried about 10 cm below the soil surface. At each site, one ibutton was buried adjacent to a chamber in a DIS and ND area, respectively, and an additional ibutton was buried at the most upstream chamber, 0.5 m and 1.5 from stream bank-full width, respectively. Air temperature (± 0.04°C) was measured at each site using two HOBO U23 Pro v2 data loggers (Onset Computer Corporation, MA, USA) located 0.5 m and 1.5 m from the stream bank-full width at each study reach.

Statistical analysis

Gas flux rates were calculated by linear regression of gas concentrations over time. Each time series was evaluated for goodness of fit by visual inspection (Collier and others 2014). Additional quality control measures included visual inspection for abnormally high and low values outside the range of reported riparian emissions, as well as Cook's Distance statistical test (Zuur and others 2007). In sum, these quality control measures resulted in the removal of 11% of flux rate data points for CO_2, 26% for CH_4, and 34% for N_2O. Using the ideal gas law, the flux rate was converted to µmol, and then the molecular mass was used to translate this value into µg or mg. These equations are described by Collier and others (2014). For statistical analysis, we used the software *R*3.6.1 (R Core Team 2020). We used linear mixed effects (LME) models to evaluate the forest management effects on GHG fluxes using the "glmmTMB" package (Brooks and others 2017). The models included the AR(1) autoregressive covariance structure to account for temporal autocorrelation and repeated measures (Kravchenko and Robertson 2015). The model residuals were tested for normality and homogeneity of variance. When the model residuals violated the assumption of normality, they were transformed and tested again. In the case of models with N_2O flux rate as the response variable, the flux data were log-transformed. We used a *post hoc* Tukey's HSD test to compare pairwise differences in treatment levels. For all statistical analyses, significance was accepted at $p < 0.05$.

Results

Environmental variables

Mean soil temperature (June to September) was, on average, the lowest at the reference sites (13.6 ± 1.5 °C; mean ± *SD*), intermediate at the buffer sites (14.4 ± 1.3 °C), and highest at the no buffer sites (14.9 ± 1.3 °C) (Figure 2). Maximum and mean daily air temperature followed the same trend, with a mean temperature of 14.5 ± 2.0 °C, 14.8 ±2.2 °C, and 15.3 ± 2.6 °C, at the reference, buffer, and no buffer sites, respectively. Mean soil temperature was similar at the groundwater discharge (DIS) areas (14.1 ± 1.6 °C) compared to the ND areas (14.5 ± 1.5 °C).

Mean soil moisture (measured as volumetric soil water content) was highest at no buffer sites (Figure 3A). The mean soil moisture from June to September was 43.3 ± 14.4%, 45.5 ±16.7%, 50.4 ± 14.3% at the reference, buffer, and no buffer sites, respectively. In terms of local groundwater conditions, mean soil moisture was significantly higher in the DIS areas at 55.4 ± 11.4% compared to 41.6 ± 15.1% in the ND areas, across all treatments.

Mean depth to the groundwater table was the lowest at the no buffer sites, and was significantly lower than the buffer and reference sites (Figure 3B). The mean depth to the groundwater table (June to September) was 28.7 ± 9.6 cm, 26.2 ± 12.7 cm, and 15.9 ± 8.9 cm at the reference, buffer, and no buffer sites, respectively. In terms of local groundwater conditions, mean depth to the groundwater table was significantly lower at the DIS areas (20.0 ± 12.6 cm) compared to the ND areas (27.4 ± 10.1 cm), across all treatments. There was no statistically significant difference in daily average soil temperature and soil moisture between treatments.

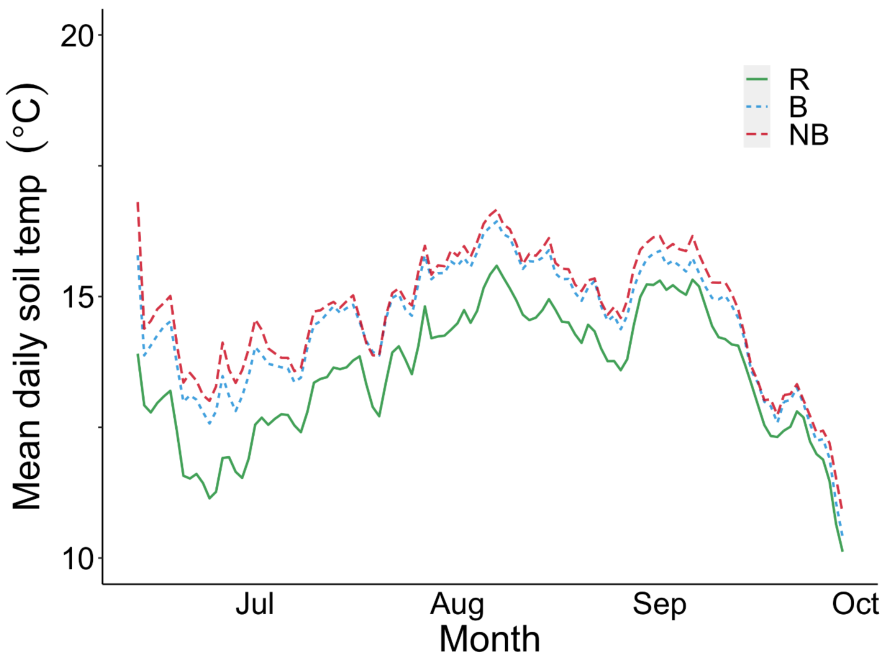

Figure 2. Mean daily soil temperature ~10 cm below the soil surface in the riparian zone of headwater streams in the Pacific coastal rainforest of British Columbia across reference (R, n = 3), buffer (B, n = 3), and no buffer (NB, n = 3) treatments from June to October 2019.

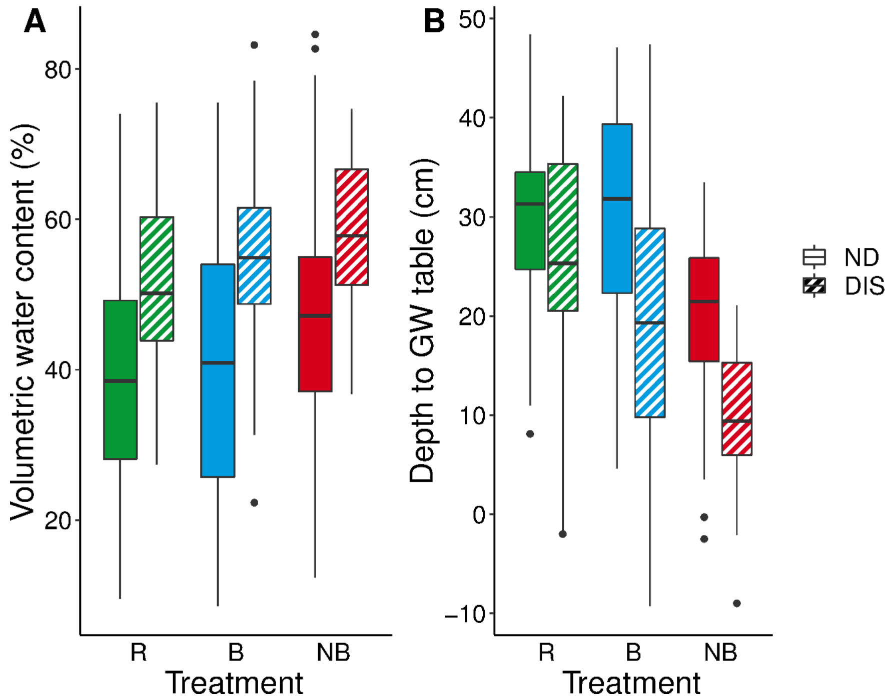

Figure 3. Mean volumetric water content (A) and depth to the groundwater table (B) in the riparian zone of headwater streams in the Pacific coastal rainforest of British Columbia across reference (R, n = 3), buffer (B, n = 3), and no buffer (NB, n = 3) treatments, as well as in groundwater discharge (DIS) areas and in non-groundwater discharge (ND) areas from June to October 2019. Boxplots display the median, 25th and 75th percentiles, whiskers (1.5 times the IQR), and individual outliers (dots), for this and subsequent boxplots.

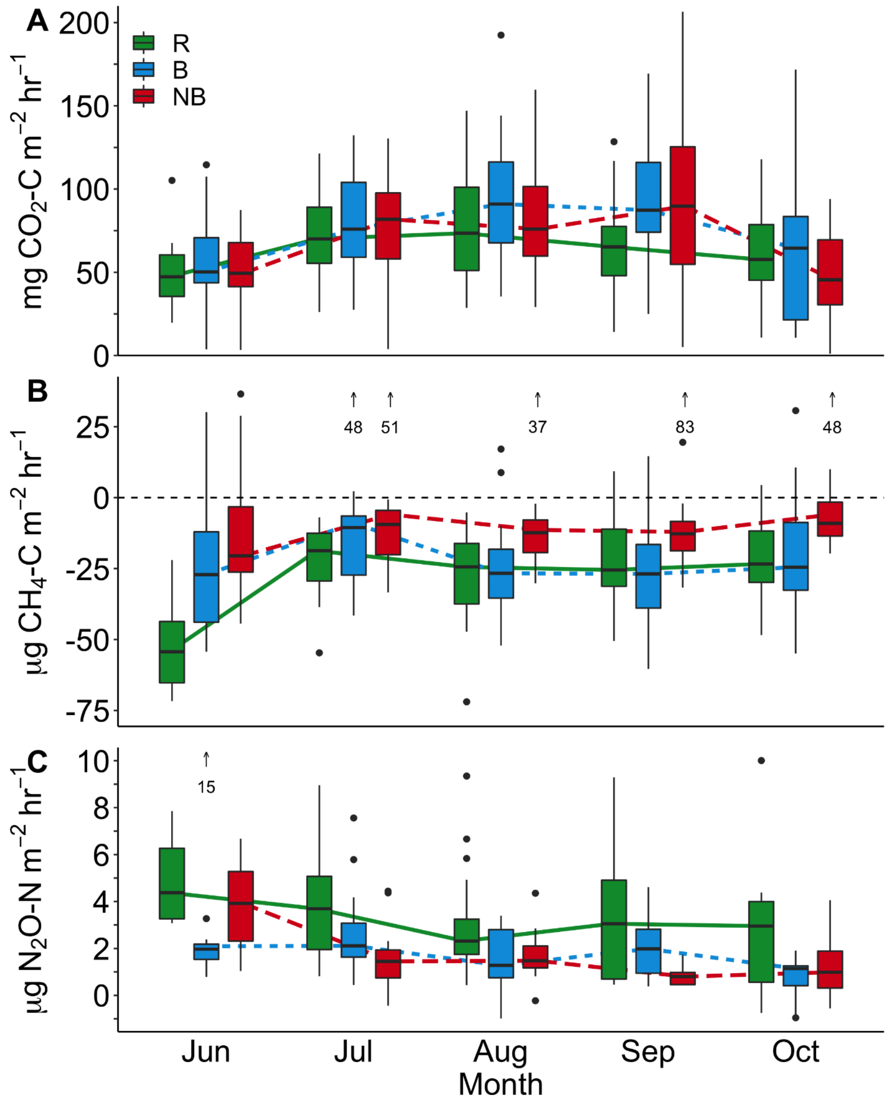

Figure 4. Riparian forest soil carbon dioxide (A), methane (B), and nitrous oxide (C) flux rates from reference (R; n = 3), buffer (B; n = 3), and no buffer (NB; n = 3) treatments alongside headwater streams in the Pacific coastal temperate rainforest of British Columbia across five sampling periods from June to October, 2019. Arrows indicate outliers along with their values.

Carbon dioxide

Mean soil CO_2 efflux (mg CO_2-C m^{-2} h^{-1}) for the reference, buffer, and no buffer sites for the entire sampling period (June to September) was 65.7 ± 29.9, 76.0 ± 39.1, 71.5 ± 40.9, respectively (Figure 4A). Based on the linear mixed effects (LME) model, CO_2emissions were not significantly different between treatments (Table 2). However, over the sampling period, CO_2 emissions were, on average, 16% higher at the buffer sites and 9% higher at the no buffer sites compared to the reference sites, respectively. Mean soil CO_2 emissions peaked in the middle of the growing season, with 85.9 ± 38.5 mg CO_2 m^{-2}h^{-1} in July and 83.6 ± 42.0 mg CO_2m^{-2} h^{-1} in August. Mean soil CO_2 emissions were the lowest in the spring followed by the autumn, with 53.1 ± 24.78 mg CO_2m^{-2} h^{-1} in June and 56.9 ± 33.9 mg CO_2 m^{-2} h^{-1} in September. CO_2 fluxes were not significantly different between local groundwater conditions. On average, over the study period CO_2 efflux was 64.9 ± 37.6 at DIS sites and 74.9 ± 36.3 mg CO_2 m^{-2} h^{-1} at ND areas. Soil temperature, soil moisture, and depth to the groundwater table were significant predictors of CO_2 fluxes according to the LME models, with a marginal r^2 of 0.16, 0.11, and 0.06, respectively. For every degree increase in soil temperature, the mean CO_2 flux rate increased by 8.13 mg. For every percent increase in soil moisture, the mean CO_2 flux rate decreased by 0.58 mg. For every cm increase in depth to the groundwater table, the mean CO_2flux rate increased by 0.48 mg.

Table 2. Pairwise difference between Treatment levels (reference, R; buffer, B; and no buffer, NB) using Tukey's HSD *post hoc* test for the linear mixed effects models explaining the dynamics of carbon dioxide, methane, and nitrous oxide fluxes, respectively. All models included the autocorrelation term "AR1(Week + 0 | Site/Chamber)". Bolded comparison indicates a significant effect at $p < 0.05$.

Model	**Comparison**	**Est.**	SE	p
CO_2 ~ Treatment	R - B	-8.65	11.2	0.72
R - NB	-1.86	11.4	0.99	
B - NB	6.79	11.4	0.82	
CH_4 ~ Treatment	R - B	-6.65	6.82	0.59
R - NB	**-22.35**	**6.96**	**< 0.01**	
B - NB	-15.71	6.97	0.07	
N_2O ~ Treatment	R - B	0.33	0.14	0.06
R - NB	**0.39**	**0.14**	**0.02**	
B - NB	0.06	0.14	0.90	

Methane

Mean soil CH_4 fluxes (µg CH_4-C m^{-2} h^{-1}) for the reference, buffer, and no buffer sites for the entire sampling period (June to September) were -26.3 ± 17.7, -19.7 ± 21.6, and -9.1 ± 22.2, respectively (Figure 4B). Therefore, on average, all of the treatments were a net CH_4 sink over the study period. Based on the LME model, CH_4 uptake was significantly different between treatments (R - NB: $p < 0.01$, Table 2); CH_4uptake was on average 1.26 times lower at the buffer sites than the reference sites, and 3.71 times lower at the no buffer sites than the reference sites. CH_4 fluxes were significantly ($p < 0.001$) different between local groundwater conditions, with CH_4 uptake being significantly lower in the DIS areas (Figure 5). CH_4 gas flux was -11.2 ± 22.5 µg CH_4 m^{-2} h^{-1} at the DIS areas while -24.4 ± 19.2 µg $CH_4$$m^{-2}$ h^{-1} at the ND sites, on average, for the entire study period. Soil moisture and depth to groundwater were significant predictors of CH_4 fluxes according to the LME models, with a marginal r^2 of 0.37 and 0.46, respectively. For every percent increase in soil moisture, mean CH_4 flux rate increased by 0.50 µg. For every cm increase in depth to the groundwater table, mean CH_4flux rate decreased by 0.67 µg.

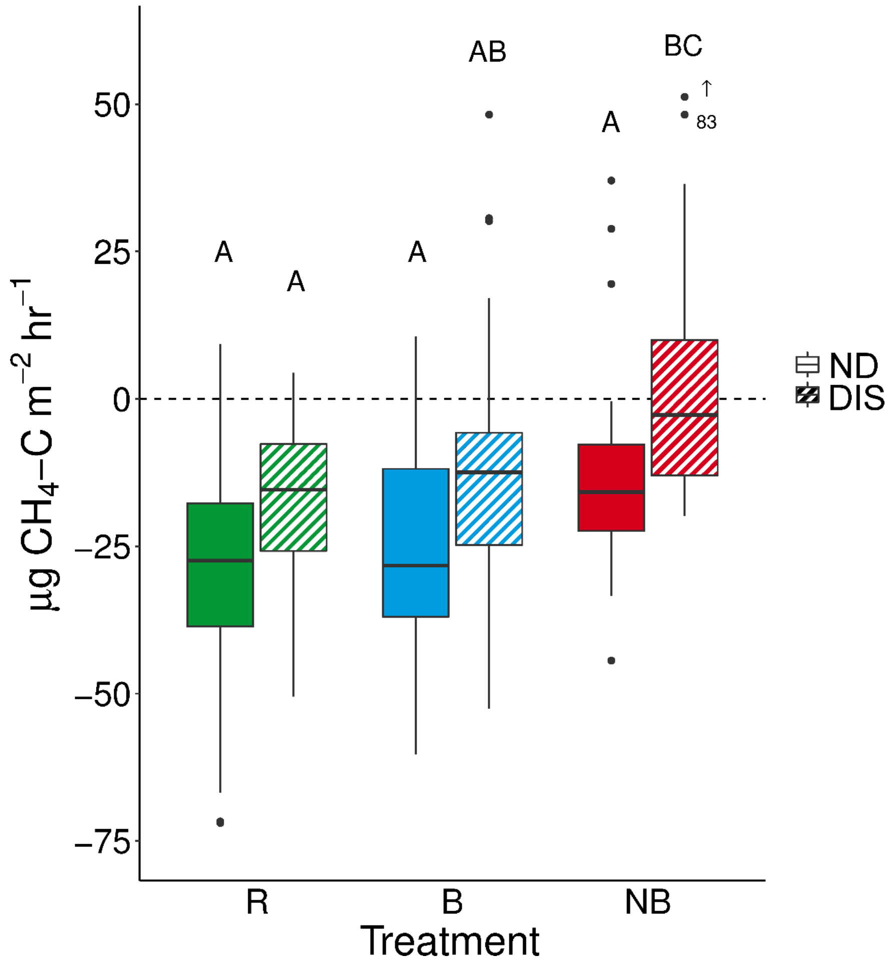

Figure 5. Riparian forest soil methane flux rates from reference (R; n = 3), buffer (B; n = 3), and no buffer (NB; n = 3) treatments, and from groundwater discharge (DIS) and non-groundwater discharge (ND) areas along headwater streams in the Pacific coastal temperate rainforest of British Columbia in 2019 (June - September). Boxplots that do not share the same uppercase letter show significant differences based on LME models ($p < 0.05$).

Nitrous oxide

Mean soil N_2O fluxes (µg N_2O-N m^{-2} h^{-1}) for the reference, buffer, and no buffer sites for the entire sampling period (June to September) were 3.6 ± 2.6, 2.1 ± 2.4, and 1.7 ± 1.6, respectively (Figure 4C). According to the LME model, there was a treatment effect on N_2O fluxes, with significantly (R - NB: p = 0.02, Table 2) higher fluxes at the no buffer sites compared to the reference sites. Over the sampling period, N_2O fluxes were, on average 1.70 times lower at the buffer sites and 2.05 times lower at the no buffer sites, compared to the reference sites, respectively. Additionally, mean soil N_2O fluxes were the highest in the spring, and gradually declined throughout the summer to the fall, with the highest monthly mean measured in June (4.1 ± 2.9 µg N_2O m^{-2} h^{-1}) and lowest monthly mean in September (1.6 ± 2.2 µg N_2O m^{-2} h^{-1}). N_2O fluxes were not significantly different between local groundwater conditions. On average, over the study period N_2O efflux was 2.5 ± 2.2 at the DIS areas and 2.4 ±2.5 µg N_2O m^{-2} h^{-1}at the ND areas. None of the environmental variables (i.e. soil moisture, soil temperature, and depth to the groundwater table) were significant predictors of N_2O fluxes.

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Discussion

For the sampling period, the riparian forest soils examined in this study were a net sink of CH_4, and a net source of CO_2 and N_2O. The reference sites had significantly higher N_2O emissions and significantly higher CH_4 uptake than the sites with no buffer. CH_4 uptake was significantly higher in the ND areas compared to the DIS areas, reflecting the difference in soil moisture and resulting aerobic conditions required for methanotrophy and anaerobic conditions required for methanogenesis (Oertel and others 2016). Soil temperature, soil moisture, and depth to the groundwater table were significant predictors of CO_2 emissions. Soil moisture and depth to the groundwater table were significant predictors of CH_4 fluxes.

Effects of forest harvest

Although riparian zones are widespread across the global landscape, we found very few studies examining GHG fluxes from the riparian zones of small streams (Goodrick and others 2016; Soosaar and others 2011), and even fewer for headwater streams (Chang and others 2014; De Carlo and others 2019a). Due to the distinct environmental conditions of riparian zones (Vidon and others, 2018), the effects of forest harvest on GHG fluxes are likely to differ here from upland soils. Riparian zones of headwater streams in particular are unique due to their steep geomorphology, high groundwater influence, and susceptibility to low flows (Richardson and Danehy 2007). Therefore, this study fills the knowledge gap of the effects of forest harvest on GHG fluxes from riparian zones of headwater streams.

Clear-cutting has inconsistent effects on CO_2 emission rates (Striegl and Wickland 1998; Ullah and others 2009), with studies reporting a decline (Striegl and Wickland 1998), an increase (Lavoie and others 2013; Paul-Limoges and others 2015), or no change (Kähkönen and others 2002) in CO_2 emissions following harvest. In this study, there was no significant difference in CO_2emission rates between treatments, although CO_2emissions were highest at the buffer sites. This lack of a significant treatment difference may have been because of the distinct conditions in riparian forests compared to upland forests. Riparian zone soils have high moisture levels due to shallow water tables and a strong groundwater influence (Goodrick and others 2016). Therefore, the rise in temperature following forest harvest that can explain the rise in CO_2 following harvest in upland forests (Lavoie and others 2013), may be buffered by the generally moister and cooler soils of riparian forests (Clinton and others 2010), resulting in a less distinct rise in CO_2 emissions following harvest in riparian forests compared to upland forests. Moreover, the mild mean annual temperature and narrow range in annual temperature in our study region may have meant limited temperature differences, resulting in a small effect size of the treatments.

Soil temperature, soil moisture, and depth to the groundwater table were significant predictors of CO_2 fluxes.

This is consistent with the large body of literature on the drivers of soil respiration (Luo and Zhou 2006). Soil respiration usually increases exponentially with temperature, reaches a maximum, and then declines (Luo and Zhou 2006). Temperature controls many aspects of soil respiration from the activity of cellular enzymes, to root growth and microbial activity (Luo and Zhou 2006). Soil moisture is another well-established driver of soil CO_2 emissions, with the common conceptual relationship where soil respiration is low under dry conditions, reaches a maximum at intermediate soil levels, and decreases at high soil moisture content where anaerobic conditions depress aerobic microbial activity (Luo and Zhou 2006). The interactive effects of these two variables on soil respiration is a key knowledge gap (Meyer and others 2018). Our research provides valuable information to help understand the effects of these factors on soil respiration in the unique riparian environment.

The results of our study were in line with other studies, which have also found that clear-cutting increased CH_4 efflux from the soil (Kähkönen and others 2002; Wu and others 2011). This rise can be attributed to higher average summer soil temperatures, greater soil moisture, and higher dissolved organic carbon concentrations in clear-cuts (Ullah and others 2009; Wu and others 2011). Forest harvest can result in soil compaction, reduced transpiration, and a rise in the groundwater table, all of which promote waterlogging and anaerobic conditions (Christiansen and others 2017; Gundersen and others 2010). In our study, soil moisture and depth to the groundwater table were significant predictors of CH_4 fluxes, while temperature was not. In agreement with our results, CH_4 uptake was three times lower following clear-cutting in a temperate spruce forest in southern Germany (Wu and others 2011). In another study, clear-cutting turned a spruce forest soil in Finland from a sink to a source of CH_4, with a 40% decrease in CH_4 consumption rates (Kähkönen and others 2002). The lower CH_4 uptake following forest harvest may be explained by the harmful impacts of soil disturbance on methanotrophic bacteria, resulting in the inhibition of CH_4-oxidation (Le Mer and Roger 2001; Wu and others 2011). Alternatively, the anaerobic conditions created by higher soil moisture concentrations following forest harvest can promote the production of CH_4 (Wu and others 2011). A clear-cut wetland in Québec, Canada produced 131 times more CH_4 than the undisturbed wetland soil, likely due to higher soil temperature and soil moisture in the clear-cut (Ullah and others 2009). In our study, the no buffer sites still were a net sink, albeit a weak sink, of CH_4 over the growing season. This is likely because unlike many wetlands, most riparian soils do not have consistently anoxic soils, which promote methanogenesis (Dalal and Allen 2008). Given that there was no significant difference in CH_4 flux rates between the buffer and reference sites, it appears that riparian buffers may be effective in preserving soil ecosystem conditions contributing to CH_4 fluxes. Consequently, riparian buffer zones may be an effective strategy for forest managers interested in maintaining CH_4 balance in riparian zone soils.

The lower N_2O emissions in the no buffer sites compared to undisturbed riparian zones was a surprising result because many other studies have reported an increase in N_2O emissions following forest harvest. Typically, forest harvest can increase soil moisture and mobilize soil nitrogen, promoting N_2O emissions from logged forest sites (Kreutzweiser and others 2008). Higher N_2O emissions were seen following forest harvest in the taiga region of eastern Siberia, Russia (Takakai and others 2008). Additionally, N_2O emissions were 2.7 times higher in clear-cut than in mature black spruce forest soil in Québec, Canada (Ullah and others 2009). However, these studies were not conducted in riparian forests, which have unique conditions such as shallow water tables, high soil organic matter quality and quantity, and high soil nitrogen availability, unlike most upland forests (Knoepp and Clinton 2009; Vidon and others 2018). Given that forests are typically sources of N_2O (Dalal and Allen 2008), the lower N_2O fluxes in the no buffer compared to the reference sites is a departure from undisturbed ecosystem function. The unexpectedly low N_2O fluxes at the no buffer sites could be explained by the mechanical soil disturbance in the riparian zone caused by forest harvest. The disruption of the structure and function of microbial communities responsible for nitrification and denitrification could contribute to the comparatively low N_2O fluxes at the no buffer sites (Tan and others 2005). Meanwhile, N_2O fluxes at the buffer sites were not significantly different from the reference sites, thus, riparian buffers may be effective in preserving soil ecosystem conditions contributing to N_2O fluxes. A decline in soil CO_2emissions was attributed to the disruption of the soil surface and death of tree roots following the clear-cutting of a jack pine stand in Saskatchewan (Striegl and Wickland 1998). Moreover, soil compaction as a result of forest harvest has been found to reduce net nitrification rates in the forest floor and mineral soil as well as reduce the soil microbial biomass nitrogen (Tan and others 2005).

None of our measured environmental variables (i.e. soil temperature, soil moisture, and depth to the groundwater table) were significant drivers of N_2O fluxes. Thus, perhaps some environmental variables not measured in this study could explain some of the unexplained treatment differences. For instance, soil nitrogen concentrations are an important driver of N_2O fluxes (Christiansen and others 2012). A large proportion of nitrogen in upland soils is transferred to riparian ecosystems, where it can be retained via biological assimilation or removed via denitrification (Pinay and others 2018). Denitrification rates were positively correlated with soil nitrate content in a riparian forest along the Louge River in south-west France (Pinay and others 1993). Moreover, streamwater nitrate concentrations have been found to increase in the short term following forest harvest in western North America due to enhanced soil nitrification rates (Feller 2005). This increased flow of nitrate from the soil into streams in the short term, subsequently returns to pre-harvest levels due to uptake by rapidly growing biomass (Feller 2005). This phenomenon could explain the lower N_2O emissions we observed at the sites with rapidly re-growing vegetation, harvested over three years earlier, compared to the reference sites.

In addition to the effects of external nutrient inputs, the potential for nitrogen retention and removal can vary between climatic regions, based on differences in soil moisture conditions and nitrogen supply. In temperate regions with high humidity, such as our study region, high soil moisture promotes nitrification and denitrification (Christiansen and others 2012). In contrast, Mediterranean and arid regions have lower denitrification rates due to low water availability and short residence time of water and solutes (Pinay and others 2018).

In the context of other studies, the soils in our study area have a moderate nutrient content and the GHG flux rates in this study were comparable to those reported in other temperate riparian zones. The soils in undisturbed stands at Malcolm Knapp Research Forest have previously been categorized as medium-nutrient (Collins and others 2001), with reported mineral soil nitrate content of 2.5 kg ha^{-1} and a C:N ratio of 23 (Turk and others 2008). Mean gas flux rates at the reference sites (June to September) were 65.7 ± 29.9 mg CO_2 $m^{-2}h^{-1}$, -26.3 ± 17.7 µg CH_4-C m^{-2} h^{-1}, and 3.6 ± 2.6 µg N_2O-N m^{-2} h^{-1}. Since we measured gas fluxes from June to September, our average flux rates should be slightly higher than annual flux rates reported from temperate studies that include winter months when microbial activity is lower, and hence flux rates are lower. The average annual CO_2 flux rate in flood-prone riparian forest in an agricultural landscape along the White River in Indiana, USA measured 48.8 mg CO_2m^{-2} h^{-1} and was significantly related to the C:N ratio (16.4) of soil organic matter (Jacinthe and others 2015). At a temperate riparian forest in an agriculturally dominated landscape in Ontario, Canada, the mean annual CO_2 emissions were 18 mg CO_2m^{-2} h^{-1}, with a soil organic carbon content of 84.1 g C kg^{-1} and a total nitrogen content of 12.5 g N kg^{-1} (De Carlo and others 2019a). At the same location, mean annual N_2O emissions were 5.93 µg m^{-2} h^{-1}, and were significantly correlated with soil total nitrogen (12.5 g N kg^{-1}) (De Carlo and others 2019b). A riparian forest in New Jersey had a C:N ratio of 21 and a N_2O emission rate of 0.9 µg N m^{-2} h^{-1} (Ullah and Zinati 2006). Nevertheless, there is a lack of studies about GHG emissions from riparian zones of headwaters streams, moreover most of the literature is in agriculture-dominated landscapes and not in forested riparian areas impacted by forest harvest practices.

Effects of local groundwater conditions

Landscape features that dictate soil characteristics, such as local microtopography, can be important for predicting riparian GHG emissions as they may affect the spatial distribution of soil moisture, nutrients, and organic matter, thus consequently affecting the intensity of GHG emissions (Jacinthe and Vidon 2017; Soosaar and others 2011). Local groundwater discharge conditions may create particularly important microsite variation in the riparian zones of streams. Soil conditions in DIS areas have been found to have higher base cations, soil moisture, pH levels, and nitrogen concentrations when compared to surrounding soils (Giesler and others 1998). These soil conditions may influence the processes controlling soil GHG fluxes. However, it is unknown how forest harvest in the riparian zone might influence conditions in DIS areas, and how GHG fluxes may subsequently be influenced. We hypothesized that due to the higher soil moisture at DIS areas, the emission of anaerobically produced CH_4 and N_2O would be higher than at ND areas. There were no significant differences between the DIS and ND areas for CO_2 and N_2O emissions, although ND areas generally had higher CO_2 emissions on average and DIS areas generally had higher N_2O emissions. However, CH_4uptake was significantly lower in the DIS sites compared to the ND sites. This means that DIS sites were more likely to be CH_4 sources, while ND sites were more likely to be CH_4 sinks. Similar results were found in riparian zones in central Indiana, where a topographic depression in the riparian forest accounted for 78% of annual CH_4 emissions, despite only covering <8% of the total land area (Jacinthe and others 2015). Additionally, GHG fluxes from flowing stream waters have been found to peak downstream of DIS areas, due to their lateral gas inputs from riparian soils (Lupon and others 2019). Given that CH_4 fluxes were highest in DIS areas at no buffer sites, the results of our study provide additional support for the use of hydrologically adapted buffers, which provide more protection for wet areas, such as DIS areas, in the riparian zone (Tiwari and others 2016). The variable buffer width adapted to site-specific hydrological conditions can protect biogeochemical and ecological functions as well as provide economic savings when compared to fixed-width buffers (Tiwari and others 2016).

Conclusions

In conclusion, our work shows that forest harvest and local groundwater conditions influence GHG emissions from riparian forest soils alongside headwater streams. Our results demonstrate that riparian buffers may be effective in protecting soil ecosystem functions contributing to CH_4 and N_2O fluxes. Nevertheless, our findings should not diminish the importance of maintaining intact riparian forests for other benefits, such as the unique biodiversity riparian areas sustain. Given our finding that local groundwater conditions play an important role in driving CH_4 fluxes, forest managers may choose to consider hydrologically adapted buffers to reduce CH_4 emissions. This new information about the effects of forest harvest in the riparian zone on GHG flux rates should be considered in local policy discussions regarding how forest harvest in the riparian zone can contribute to climate change, and what can be done to mitigate or diminish the impacts. Considering that many small streams are left unprotected in British Columbia (Kuglerová and others 2020) and elsewhere, perhaps the leverage of climate change mitigation may influence policy changes leading to increased retention of riparian buffer zones in the region. In sum, this research may be useful to forest managers interested in managing riparian buffer zones for GHG balance and climate change mitigation.

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