Effect of Riparian Forest Cover on Cellulose Decomposition in Agricultural Streams

by

Kristen Nicole Hewitt

A thesis

presented to the University of Waterloo

in fulfillment of the

thesis requirement for the degree of

Master of Science

in

Biology

Waterloo, Ontario, Canada, 2024

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Author’s Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners. I understand that my thesis may be made electronically available to the public.

Abstract

Agricultural practices often result in human impacts to aquatic ecosystems. The presence of riparian forests has been shown to mitigate some of these impacts, however, there is limited knowledge of how riparian forests can mitigate human impacts on microbial community function. This study investigates effects of riparian forest cover on in-stream cellulose decomposition. The study used the cotton strip assay to compare two sets of streams in summer, fall, and winter seasons to assess the association between riparian forest cover and cellulose decomposition in summer in agricultural catchments. Results from summer and autumn show there is greater cellulose decomposition in streams with riparian forest cover than streams without riparian forest cover. Cellulose decomposition also increased linearly with riparian forest in the summer. Decomposition in the assessed streams was also compared to cellulose decomposition in reference streams within the same region. Significant differences were observed between reference and non-forested streams, but not reference and forested streams, in the summer. These differences were not explained by typical environmental parameters (e.g., temperature, nutrients). These results suggest that riparian forest cover influences the decomposition by protecting organic matter processing in streams, though the specific threats against which riparian forests provide protection remain unclear. In future studies, considering agricultural contaminants (e.g., pesticides, pharmaceuticals) and investigating the differences in microbial community composition between groups may help close this knowledge gap.

Acknowledgements

First and foremost, I extend my deepest gratitude to my supervisor, Dr. Adam Yates, for his unwavering support and invaluable guidance throughout the design, implementation, and writing phases of this project. I also wish to thank Dr. Merrin Macrae and Dr. Mark Servos for serving on my committee and providing thoughtful feedback and insights that greatly enhanced this project.

I am also grateful to my lab mates in the StrEAMS Lab for their assistance during fieldwork and their support throughout this journey. Additionally, I would like to express my sincere appreciation to my family and friends for their steadfast encouragement over these past two years.

Finally, I am deeply thankful to the landowners who generously allowed me to conduct research on the streams running through their properties. Without their interest and cooperation, this project would not have been possible.

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Introduction

Organic matter processing in streams involves the breakdown of leaf litter, woody debris, and other organic materials by both microbial and invertebrate communities (Graça et al., 2015) and is a crucial element of healthy ecosystem functioning. Organic matter processing in stream ecosystems provides a major energy source for food webs in streams, even more so in lower order streams (Ferreira et al., 2020). This process is influenced by many environmental factors, including temperature, nutrients, flow regimes, and community composition (Graça et al., 2015). However, anthropogenic land uses, such as agricultural practices, can alter these environmental factors and adversely impact organic matter processing (Graeber et al., 2012, 2015; Price et al., 2019; Williams et al., 2010).

Agricultural practices, such as the application of fertilizers and agrochemicals, the modification of natural landscapes, and the rearing of farm animals, are often associated with impairment of aquatic ecosystems (Gessner & Chauvet, 2002). Agriculture can increase stream temperatures (Bowler et al., 2012) and increase nutrient loading (Piscart et al., 2009) to surrounding streams. These variables are known to influence organic matter processing in streams, with higher temperatures and higher nutrients stimulating organic matter processing. Agrochemicals are also used in various ways, for example pesticides to stop unwanted organisms from harming crops (Bereswill et al., 2013) and antibiotics which are often used in raising farm animals (Matsui et al., 2008). These agrochemicals have also been found to influence organic matter processing (Ding & He, 2010; Rasmussen et al., 2012). Riparian forests are often implemented to mitigate some of these effects on aquatic ecosystems (Liu et al., 2017).

Forested riparian zones are a commonly used best management practice that have the ability to mitigate agricultural stressors on aquatic ecosystems (Tolkkinen et al., 2021; Turunen et al., 2019, 2021). Riparian zones are defined as three-dimensional area surrounding a stream, where terrestrial and aquatic zones interact (Gregory et al., 1991). Forested riparian zones can mitigate the effects of agriculture on organic matter processing by reducing nutrients, agrochemicals, and antibiotics that reach the stream, and moderating temperatures and influencing sediment loads in streams (Broadmeadow & Nisbet, 2004). Though the presence of riparian forest has been shown to mitigate a lot of the impacts of agriculture on the environmental characteristics of streams, limited and conflicting evaluation exists on whether the presence of riparian forests matter for protecting ecosystem functioning (Cornejo et al., 2020; Tanaka et al., 2015). The goal of my study is to investigate whether riparian forests are able to protect microbial-mediated organic matter processing in agricultural streams.

1.1 Literature Review

1.1.2 Organic matter processing in streams

Organic matter processing refers to the decomposition or breakdown of organic material, such as the breakdown of leaf litter and woody debris supplied by streamside vegetation or produced within the stream in natural systems, as well as dissolved organic compounds, feces, and the remains of animals and insects (Richardson et al., 2005; Wantzen et al., 2008). Various groups contribute to the processing of organic matter, and the contributions of microbial and invertebrate communities are often distinguished in the literature (Wantzen et al., 2008). The interplay between these communities and the mechanisms they utilize to decompose organic material highlights the complexity of organic matter processing.

There are three mechanisms of organic matter processing: leaching, conditioning, and fragmentation (reviewed by Abelho, 2001). Leaching refers to the release of soluble organic compounds that primarily occurs when organic matter enters the stream, resulting in a large initial mass loss. Conditioning involves the colonization of organic material by microorganisms, such as bacteria and fungi, which results in the breakdown of the organic matter through metabolization, maceration, and incorporation of the organic matter into secondary production. Conditioning can also enhance breakdown by rendering the material more palatable for macroinvertebrates. Finally, fragmentation is the physical breakdown of organic matter into smaller particles via physical abrasion due to flow, maceration, and shredding by macroinvertebrates. This component of breakdown is especially important during high flow contexts, where fragmentation happens more readily (Abelho, 2001). These three mechanisms are influenced by environmental factors in different ways, which can directly affect organic matter processing (Abelho, 2001).

Microbial communities are particularly important when considering organic matter processing. Bacteria and fungi are typically the primary decomposers of organic matter in streams (Wiegner et al., 2015). The structure of microbial communities are often closely related to environmental factors, such as organic matter quality and quantity (Kominoski et al., 2011), nutrient loading in streams (Qu et al., 2017), and other biogeochemical processes in streams (Boyero et al., 2011). Given that microbial activity is directly influenced by the characteristics of organic matter, the composition of this material plays a crucial role in determining decomposition rates in stream ecosystems.

The composition of organic matter is an important component dictating decomposition in streams, as one study examining the how composition of riparian litter inputs into streams

influences decomposition found that the species of leaves used to measure decomposition significantly impacted microbial-mediated decomposition rates (Lidman et al., 2017). Another study found that riparian land cover and hydrology can influence dissolved organic matter composition in streams (Pisani et al., 2020). Because the composition of riparian forests can influence the composition of organic matter in streams, and due to the fact that the composition of organic matter influences decomposition, it appears that riparian forests are an important component that has been shown to influence organic matter decomposition in streams.

1.1.3 Environmental controls on organic matter processing in streams

The focus of this section is on the predominant controls of organic matter processing specifically in agricultural streams. One of the main controls of organic matter processing is temperature, where higher temperatures result in faster breakdown (Griffiths & Tiegs, 2016; Piggott et al., 2015; Tank et al., 2010; Webster & Benfield, 1986). This is due to increasing metabolic rates of organisms and enhanced chemical decomposition at higher temperatures (Allan & Castillo, 2007). Follstad Shah et al., (2017) synthesized records of leaf litter in streams globally to assess the effects temperature has on litter processing. Their results demonstrate that, though the sensitivity of different leaf species to temperature differed, breakdown rates increased by 5-21% with a 1-4˚C increase in stream temperature. The large magnitude of the relationship reflects the critical influence of temperature on organic matter processing in stream ecosystems.

Nutrient concentrations can also play a vital role in influencing organic matter processing in streams. One study found that as nutrient levels in the stream increase, so does organic matter breakdown, and that there is a decrease in organic matter breakdown at very high nutrient concentration (Woodward et al., 2012). Other studies have also shown that an increase in nutrient concentrations can stimulate organic matter processing in streams (Piggott et al., 2015;

Tank et al., 2010). Nutrients can play an essential role for both growth and metabolisms of microbial communities, such as bacteria and fungi, which increases organic matter processing (Williams et al., 2010; Woodward et al., 2012).

Flow velocity within streams influences organic matter processing, primarily by stimulating physical breakdown as flow velocity increases (Banks et al., 2023; Spänhoff et al., 2007). Flow velocity also significantly influences microbial communities, leading to complex effects on organic matter processing. Ferreira & Graça, (2006) found that low flow conditions favoured more diverse fungal communities, which can stimulate organic matter processing. Contrasting this point, higher flows can also stimulate organic matter processing though increased physical abrasion (Flores et al., 2013). These contrasting relationships highlight the complexity of drivers involved in determining organic matter processing rates.

pH is another physical characteristic that can affect organic matter processing. This is highlighted by Niyogi et al. (2013) who investigated streams affected by acid mine drainage in New Zealand, where there were extreme differences in pH (2.7 – 7). They found that streams with lower pH values had inhibited breakdown rates compared to streams with a neutral pH. This relationship is supported by other studies with a less extreme variation in pH (Webb et al., 2019). The positive relationship between pH and decomposition is likely due to the inhibition of microbial communities, particularly fungi (Ferreira & Guérold, 2017). Therefore, pH levels are a crucial measurement in understanding variations in organic matter processing across different stream environments.

Agrochemicals, pharmaceuticals and personal care products can also influence, and often inhibit, organic matter processes in aquatic ecosystems. Many pesticides, commonly applied to agricultural fields, inhibit organic matter breakdown by impairing microbial communities

(Rasmussen et al., 2012; Schäfer et al., 2012). Conversely, some pesticides have also been shown to stimulate decomposition (Mutshekwa et al., 2022). Antibiotics, which can enter the stream in agricultural areas, can also reduce organic matter breakdown through the inhibition of aquatic communities (Paumelle et al., 2021).

Finally, microbial community composition significantly impacts organic matter processing rates, with more diverse and abundant communities having higher decomposition (Boyero et al., 2021). Research has emphasized that differences in abundance and composition of microbial communities that can be induced by changes in land use can indeed alter organic matter processing rates (Allison et al., 2013). Kominoski et al. (2011) goes further and demonstrates that differences in community composition are indirectly influenced by the quality of inputs from riparian vegetation, which influences the structure of microbial communities. This demonstrates community composition is an important factor to consider when comparing decomposition in different streams, particularly when there is a difference in streamside vegetation.

1.1.4 Impacts of Agriculture on Organic Matter Processing

Agricultural practices impact many of the environmental controls on organic matter processing. Farming often involves the application of fertilizers (i.e., phosphorus and nitrogen) which can be readily carried to streams, resulting in the eutrophication of receiving lakes. This can harm many organisms that rely on freshwater, including humans (Schindler, 2012). The application of pesticides in agriculture can also harm many untargeted organisms, impairing plants, aquatic species, birds, and other animals that make use of the neighboring waterways (Mahmood et al., 2016). The modification of natural landscapes for agriculture can also result in increased temperatures in surrounding streams (Rutherford et al., 2004). Sugimoto et al. (1997)

found that average stream temperatures in Japan increased 6 ^{o}C due to the increase in solar radiation resulting from the reduction of canopy cover along agricultural streams. Subsurface drainage in agricultural areas also influences organic matter processes by altering stream temperature regimes (Poisson & Yates, 2022). These factors can alter stream communities and processes, with cascading effects up the food web (Price et al., 2019).

Several studies have investigated the impact of agricultural land use on organic matter processing. Research has shown that though organic matter invertebrate-mediated decomposition is negatively affected by agricultural intensity, there was no relationship between microbial-mediated decomposition and agricultural intensity (Encalada et al., 2010; Piscart et al., 2009). Contrasting these findings are two other studies. One study found that microbial-mediated decomposition was indeed negatively impacted by agricultural intensity (Voß et al., 2015). Another study, using the cotton strip assay, found that streams with more agricultural land use had higher decomposition than more forested streams (Mancuso et al., 2023). More research is required to understand the dynamic between agriculture and organic matter processing.

1.1.5 Riparian Forest

Riparian forests are one measure used by managers in agricultural landscapes to mitigate the effects that agriculture has on surrounding streams. The presence of riparian forest in agricultural landscapes results in relatively higher infiltration rates in the riparian zone, where water is filtered through the soil rather than entering the stream as overland flow (Celentano et al., 2016), though the strength of the responses of environmental variables to the presence of riparian forest can be altered with differing geology (Richardson & Béraud, 2014). Thus, the presence of riparian forest results in less nutrients and pesticides discharged into the stream

(Stutter et al., 2012). Forested land cover adjacent to streams also provides shade, decreasing water temperatures (Imholt et al., 2012). Mitigation benefits of riparian forests have been shown for physical and chemical characteristics (e.g., temperature, nutrients, sediment; Imholt et al., 2012; Sutter et al., 2012), but less research exists on how riparian forests impact stream functioning.

Previous studies have looked at how the characteristics of riparian forests can affect organic matter processing, with conflicting results. One study in British Columbia, Canada found that riparian forest composition influenced in-stream organic matter decomposition despite similar microbial communities (Kominoski et al., 2011). Conversely, another study in Portugal found no difference in microbial-mediated decomposition with varying riparian forest composition, but did find that microbial community composition differed (Bärlocher & Graça, 2002). Another study in Portugal found that both microbial community richness and litter processing rates were positively correlated with riparian tree species richness (Ferreira et al., 2016). These contradictory results highlight the complexity of the relationship between riparian forest composition and organic matter processing, indicating the need for further research to elucidate other environmental variables that may contribute to these relationships.

A recent study conducted in Uruguay compared decomposition rates between three streams with riparian forest cover and three streams with an open canopy in a catchment that is substantially used for cattle production (Barrios et al., 2022). Though they found no difference in decomposition between stream types, they did find that environmental predictors had differing effects on decomposition depending on the presence of riparian forest cover. Another study found a difference in decomposition between a more pristine stream with riparian forest cover and an impacted stream with little riparian forest cover, where the stream with riparian forest

cover had higher decomposition rates (Casotti et al., 2015). A confounding factor in this study is the lack of consistent watershed characterization, with one being pristine and the other impacted. This discrepancy complicates the attribution of differences in breakdown to either variations in riparian vegetation or inherent watershed differences. The present study aims to address this limitation by ensuring that catchments predominantly consist of agricultural land cover, with variations confined to the riparian zone. Investigating this topic will provide critical insights into the role of riparian forest in moderating the impacts of agriculture on stream functioning. Such knowledge is essential for informing more effective conservation and restoration strategies for streams within agricultural landscapes.

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1.1.6 Cotton Strip Assay for Assessing Organic Matter Breakdown

The cotton strip assay is one way to assess the relative ability of the microbial community to process organic matter. The cotton strip assay involves putting a standardized strip of cotton, made of 95% cellulose (Egglishaw, 1972), in a stream for a set incubation period, typically so the cotton strip reaches approximately 50% tensile strength loss (TSL), after which the strip of cotton is removed, and the tensile strength (defined as the amount of pulling force a material can withstand before it tears apart) lost is determined. This measurement is an indicator of cellulose decomposition, a surrogate measurement for organic matter breakdown (Tiegs et al., 2013)

The cotton strip assay has many advantages over other measures of organic matter breakdown. The cotton strip assay allows for a higher degree of standardization than cannot be achieved using common methods (e.g.,, leaf litter bags) due to the composition of the strip and the highly defined methodology used to make the cotton strips (Tiegs et al., 2013). Cotton strips have a shorter incubation time than many other methods, are ecologically relevant, inexpensive,

and are sensitive to many stressors, including nutrients and temperature (Griffiths & Tiegs, 2016; Piggott et al., 2015; Tiegs et al., 2013). TSL is sensitive to environmental conditions and is therefore used to quantify cellulose decomposition (Tiegs et al., 2007). Since the composition of organic matter significantly influences processing rates (e.g., leaf species, age; Tank et al., 2010), the cotton strip assay offers an ideal substrate for this study by eliminating variation between substrates.

1.2 Research Objectives

The goal of this study was to develop knowledge about the ability of riparian forest cover to protect organic matter processing in streams within agricultural landscapes. The research objectives were as follows:

1. Compare cellulose decomposition in streams with and without riparian forest in summer, fall, and winter seasons in an agricultural landscape and determine the environmental characteristics associated with these differences.
2. Determine the association between cellulose decomposition and the proportion of a stream's riparian area that is forested in an agricultural landscape and establish the environmental characteristics that explain the association.

1.2.1 Predictions

I predicted that agricultural streams without riparian forests would have greater cellulose decomposition than agricultural streams with riparian forest because of higher stream temperatures and higher dissolved nutrient concentrations in agricultural streams without riparian forests. I also predicted that there would be a negative relationship between the proportion of riparian forest and in-stream cellulose decomposition due to the overall positive effects of temperature and nutrients on cellulose decomposition. I predict that a threshold may occur at low

amounts of riparian forest due to the potential negative effects of heavy nutrient enrichment on decomposition.

2.0 Methods

2.1 Study Area

My study was conducted in the Norfolk Sandplain (NSP) of southern Ontario (Figure 1). Land Cover in the NSP is dominated by agriculture (~ 66%). Forest is the second most abundant land cover type (~23%), much of which is located in the riparian areas of streams. Glacial deposits of sand dominate the surface geology, resulting in a water table that is close to the surface due to high infiltration rates (Chapman & Putman, 1984). Delhi, Ontario, the most central weather station for the selected study sites (Figure 2), experiences maximum temperatures in July with a daily maximum of 27.5˚C, and minimum temperatures are experienced in January, with a minimum of -8.9˚C. The greatest monthly average of precipitation is in July, with an average of 88.6 mm, and the lowest monthly average of precipitation is in February at 58 mm (Government of Canada, 2024). Average annual precipitation is approximately 965 mm (Government of Canada, 2024).

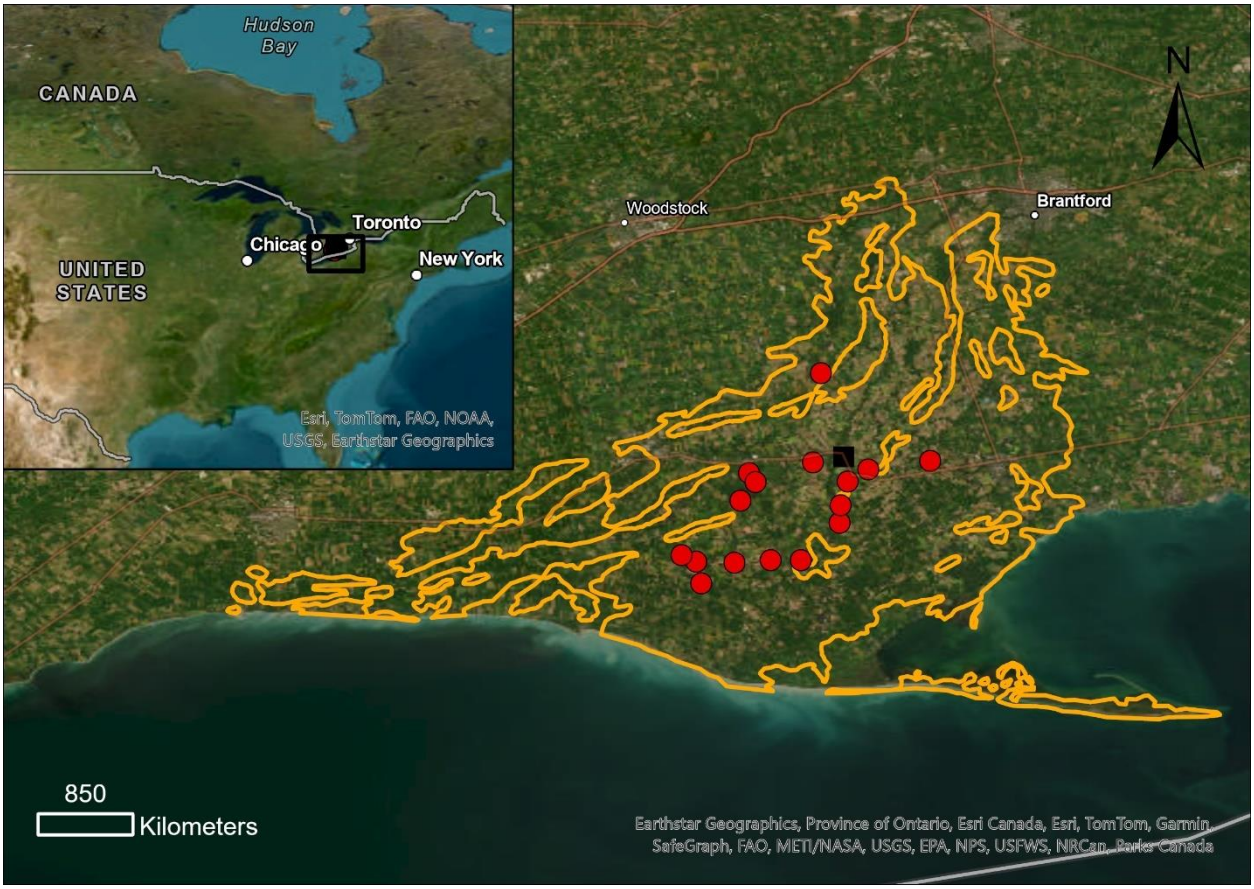


Figure 1. Location of 16 study sites (red circles) and the Norfolk Sand Plain (NSP; orange outline) in southern Ontario, Canada. The Delhi weather station is represented by a black square.

2.2 Study Design and Site Selection

To compare cellulose decomposition between streams with and without riparian forest, two groups of streams (i.e., forested and non-forested) were established. Each group had 5 streams. Forested streams had > 80 % forest in a 20-meter-wide zone on either side of the stream throughout the entire catchment of each stream. Non-forested streams had < 20% forest in a 20-meter buffer surrounding streams throughout the catchment. To minimize confounding sources of variation, all selected streams had catchments less than 10km^2, had similar surface geology and substratum (i.e., sand), and had greater than 70 % agricultural land cover outside of the riparian zone. There was no significant urban infrastructure in any of the catchments. Landowner

permission was obtained, and all 10 streams were sampled over a 3 to 5 week period in three calendar seasons: Summer (July-August 2023), Autumn (October-November 2023), and Winter (January-February 2024).

The relationship between in-stream cellulose decomposition and varying proportions of riparian forest in agricultural catchments was assessed using a gradient design. 16 streams varying from 0% to 100% forest cover in the riparian zone of each catchment (i.e., a 20 meter buffer on either side of each stream) were identified, which include the 10 streams established to compare cellulose decomposition between forested and non-forested stream groups. Selected streams were consistent in catchment and in-stream characteristics as described for sites selected as part of objective 1 to minimize confounding sources of variation. Landowner permission was obtained, and the 16 streams were sampled in the Summer of 2023 (July-August) when streams were expected to experience the greatest cellulose decomposition (greatest microbial activity) and therefore be the most sensitive to differences in cellulose decomposition.

Table 1. Location, area, percent forest cover in the riparian zone, and percent agriculture in the watershed for each study catchment.

Stream ID	Latitude (N)	Longitude (W)	Area (km^2)	Forest in Riparian Zone (%)	Agriculture in Watershed (%)
RFA1	42.8454972	-80.3887583	2.8	3.6	72
RFA2	42.8312417	-80.6058528	8.8	4.2	76
RFA3	42.733308	-80.6874083	3.9	7.6	77
RFA4	42.6991833	-80.6632167	3.9	9.6	74
RFA5	42.8208611	-80.5994361	3.0	18.0	72
RFG1	42.8473167	-80.5278611	3.6	20.0	76
RFG2	42.7984306	-80.6159139	2.3	27.4	72
RFG3	42.9566528	-80.5156083	3.3	36.0	86
RFG4	42.8351778	-80.4637806	5.8	40.8	73
RFG5	42.7260944	-80.6699250	2.3	42.1	71
RFG6	42.8204194	-80.4879583	3.9	60.0	70
RFP1	42.7242833	-80.6237667	2.2	81.5	90
RFP2	42.7256250	-80.5800528	2.4	80.1	81
RFP3	42.7922111	-80.4976361	4.5	82.4	71
RFP4	42.7282667	-80.5462306	1.3	84.0	90
RFP5	42.7716500	-80.4977389	1.6	96.5	71

2.3 Field Sampling and Laboratory Analysis

2.3.1 Cotton Strip Assay

I assessed organic matter breakdown using the cotton strip assay (Tiegs et al., 2013). Following guidelines outlines by Tiegs et al. (2013), cotton strips were cut from Fredix-brand unprimed 12-oz. heavyweight cotton fabric, Style #548 (Fredrix, Lawrenceville, GA, USA). Strips measured of 8 cm in total length, 27 threads in width (approximately 2.5cm) with a 3 mm fray on each side. Cotton strips are composed of 95% cellulose; an organic compound that is biologically relevant as an important component of plant litter (Tiegs et al., 2013). A hole was created 7 mm from the end of the strip to insert cable binders. Cable binders and associated strips were fastened to a chain and secured to the streambed by rebar.

Seven cotton strips were deployed at each site for each sampling period, one for algal biomass (chl-a) and six to determine cellulose decomposition. The cotton strips were placed in runs within each stream to minimize variation associated with habitat-based differences that have been found previously (Webb et al., 2019). In an ideal setting, cotton strips decompose until they reach approximately 50% tensile strength loss (TSL), where strips have been found to be the most sensitive (Tiegs et al., 2013). Cotton strips were incubated for 21 days in the Summer, 28 days in the Fall, and 42 days in the Winter based on other studies in southern Ontario (Poisson & Yates, 2022; Webb et al., 2019),. Incubation times vary for different seasons due to differences in temperatures across the year, which will cause differences in the rates of cellulose decomposition.

After the incubation period, cotton strips were removed from the stream, only being handled from the ends to avoid damaging the strips center where tensile strength is measured. Cotton strips were then cleaned with a small brush to remove excess debris and submerged in 95% ethanol for 1 minute to stop microbial activity and therefore further decomposition of the cotton strip. Cotton strips were then transferred back to the lab and stored in the fridge until processing. Strips were processed within 5 days of retrieval.

In the lab, strips were dried in an oven at 40˚C for 24 hours. Strips were then allowed to cool and TSL determined. TSL is measured using a tensiometer and test stand. Ends of the cotton strip were placed in grips attached to the test stand and the strips were slowly pulled apart at a rate of 2cm/min until it breaks (i.e., peak strength). The recorded peak tensile strength (i.e., when the strip breaks) is subtracted from a reference value (62.28 lb F). Reference values were determined from the tensile strength of cotton strips incubated in distilled water and not exposed to any stressors, having undergone the same process as the strips incubated in stream. By

subtracting the peak tensile strength of a cotton strip from the reference value, we can determine how much the cotton strip has degraded in the stream over time (Tiegs et al., 2013). Percent tensile strength loss per day (TSLD) was calculated using the following equation:

$$\frac{TSL\ \%}{Day} = \frac{\left(\frac{Tensile\ Strength\ (reference) - Tensile\ Strengh\ (treatment)}{Tensile\ Strength\ (reference)}\right) \times 100}{Incubation\ Time\ (days)}$$

To account for variations in temperature among streams, degree-days were used in place of incubation time to calculate percent tensile strength loss per degree-day (TSLDD). Degree-days were calculated by summing the average daily temperatures over each incubation period, following methods described by Benfield, (2007).

2.3.2 Environmental Variables

On the deployment of cotton strips at each site, stream characteristics were measured. Stream width was measured across five transects spaced evenly over 30 meters, 15 meters upstream and 15 meters downstream of the rebar used to secure the cotton strips. Canopy cover was determined at the location of the rebar using a densiometer. Depth was measured at each cotton strip using a meter stick at the end of the incubation period. Flow velocity was also measured at this time at each cotton strip on the stream bed. Water temperature was measured every 30 minutes for the entire incubation period using temperature loggers (U20-001-04, Onset) attached to the rebar with a cable tie.

Water samples were taken from a well mixed run within the stream close to the deployment of the cotton strips. One water sample was collected at each site in each season on strip retrieval, filtered using 0.45um glass fiber filters, refrigerated, and analyzed to quantify soluble reactive phosphorus (SRP), Nitrate-Nitrite, and Ammonium. Water samples were sent to the Biogeochemical Analytical Service Lab at the University of Alberta for analysis using flow

injection analysis (Lachat QuikChem 8500 FIA automated ion analyzer). Information regarding reportable detection limit and accuracy can be found in table 2. Specific conductivity (SpC) and pH was determined using a handheld multi-meter sonde (YSI, Professional Plus) on strip retrieval in the same location.

Table 2. Parameter, reportable detection limit, and accuracy of the Lachat QuikChem 8500 FIA automated ion analyzer used for assessing SRP, nitrate-nitrite, and ammonium concentrations in study water samples.

Parameter	**Reportable Detection Limit (µg/L)**	**Accuracy (%)**
SRP (µg/L)	1	0.5
NO2NO3 (µg/L)	2	0.5
NH4 (µg/L)	3	0.5

A second unfiltered 1 L water sample was collected to measure total suspended solids (TSS), volatile solids (VS), fixed solids (FS) and Turbidity on strip deployment and was refrigerated until processed. Turbidity was measured using a Turner Designs Trilogy Fluorometer and turbidity module. TSS, FS, and VS were analyzed following standard methods for the examination of water (Baird & Bridgewater, 2017). In brief, water from each site was filtered through a grade 934AH filter paper, which was then dried at 105˚C for an hour and the weight of the filter was recorded. Filters were then put into a muffle furnace at 510˚C to ash, taken out of the furnace, and weighed again. TSS, FS, and VS were calculated using the following equations:

$$TSS = \frac{(SW105 - FW) * 1000}{SV}$$

$$FS = \frac{(SW510 - FW) * 1000}{SV}$$

$$VS = \frac{(SW105 - SW510) * 1000}{SV}$$

Where:

SW105 = weight of filter + dried residue (mg)

FW = filter weight (mg)

SV = sample volume (mL)

SW510 = weight of filter + sample weight after ignition (mg)

Chl-a was measured from biofilm growth on an additional cotton strip deployed in each stream every sampling period. When strips for Chl-a analyses were removed from the stream, they were stored in a 15mL falcon tube with 95% ethanol and refrigerated until processed. All Chl-a samples were stored in the dark and processed under green light. Chl-a was determined using standard methods (Baird & Bridgewater, 2017). First, the falcon tube was put into a water bath set to 80˚C for 7 minutes. The ethanol was decanted into a new falcon tube, and strips were squeezed with a glass rod to remove as much liquid from the strip as possible. Chl-a was then measured using a Turner Designs Trilogy Fluorometer. If readings were high (over 200µg/L), samples were diluted by pipetting 1mL of the sample into a new falcon tube, and adding 9mL of 90% ethanol. The falcon tube was inverted 5x and samples were re-measured until readings were under 200µg/L.

2.4 Data Analyses

Two linear mixed effects models (LMEM) were used to assess whether streams with riparian forest cover experienced different rates of decomposition than streams without riparian forest cover. One model was run using TSLD, and a separate model was run using TSLDD.

TSLD and TSLDD were log10 transformed to meet the assumptions of statistical tests based on the Shapiro-wilk normality test and visual analysis of Q-Q plots. Proportion of riparian forest was arcsine transformed to normally distribute the data. Season (levels: summer, fall, winter) and riparian vegetation (levels: forested, non-forested) was included in the model as fixed factors. The interaction between season and riparian vegetation was also included in the models. Stream was included in the model as a random factor to account for natural variation between different streams. Given significant results, post hoc pairwise comparisons were run to investigate differences in cellulose decomposition between groups within each season. GLMMs were run using R (v. 4.3.2) with an alpha of 0.05.

Segmented regression was employed to assess the relationship between the proportion of riparian forest and TSL and identify potential breakpoints the relationship. TSLD and TSLDD values of individual strips were averaged for each stream, and means were log10 transformed based on the Shapiro-wilk normality test and visual analysis of Q-Q plots to meet the assumptions of the regressions. Segmented regression analyses were conducted using SegReg, a statistical program that introduces breakpoints to model a piecewise linear relationship. The program determines the optimal function type (0 to 6; table 3) and breakpoint by maximizing the coefficient of determination. The significance of the association was assessed using an alpha of 0.05.

Table 3. Descriptions of the 7 function types employed by SegReg, a statistical software used to conduct segmented regression analyses.

Type	Description
0	No association
1	Linear association without a breakpoint
2	Two linear sloping lines connected by a breakpoint
3	One horizontal line connected to one sloping line, separated by a breakpoint
4	One linear sloping line connected to a horizontal line, separated by a breakpoint
5	Two horizontal lines, separated by a breakpoint
6	Two lines, at least one sloping, but disconnected

For both objectives, partial least squares (PLS) regressions were used to investigate the relationships between environmental variables and cellulose decomposition. This method is particularly useful for analyzing multivariate datasets, where predictor variables may be highly collinear. Environmental variables (SRP, nitrate nitrite, ammonium, conductivity, pH, flow velocity, depth, turbidity) were log10 transformed and normalized prior to analysis. TSS, FS, and VS were not included as measurements did not vary substantially (variance < 0.05) between streams. Separate PLS regressions were run on TSLD and TSLDD data. In the TSLD analysis, average stream temperature was incorporated (log10 transformed) as an additional predictor variable. PLS regressions were run using TIBCO Statistica 14.0.1.25.

3.0 Results

3.1 Cellulose decomposition in forested and non-forested streams

Average TSLD of cotton strips from all streams across all seasons was 2.2% day^{-1} (standard deviation; ± 0.9% day^{-1}) (Figure 2). The greatest average TSLD was observed in the summer (2.9% ± 1.0% day^{-1}), and the lowest average TSLD occurred in the winter (1.5% ± 0.4% day^{-1}). Average TSLD in the fall was 1.9% ± 0.5 % day^{-1}. The overall average TSLDD within all streams across all seasons was 0.3% degree-day^{-1} (± 0.1% degree-day^{-1}). The greatest seasonal average in TSLDD was observed in Winter (0.3% ± 0.1% degree-day^{-1}), and the lowest seasonal average was observed in Summer (0.2% ± 0.05% degree-day^{-1}). Fall experienced an average of 0.2% ± 0.08% degree-day^{-1}.

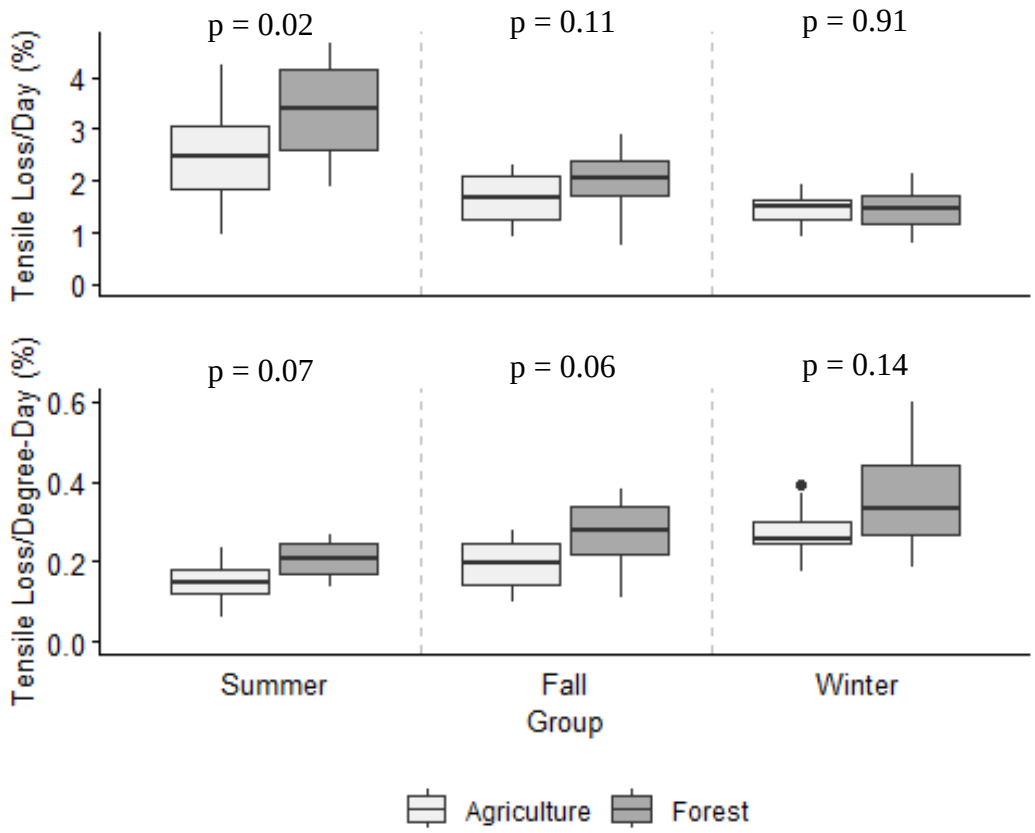


Figure 2. Boxplot of tensile strength loss per day and tensile strength loss per degree day among forested and non-forested riparian agricultural streams in summer, fall, and winter within the Norfolk Sandplain in southern Ontario. p-values represent post hoc pairwise comparisons (Table 5; Table 7).

The fastest average TSLD occurred in forested streams in the summer (3.35% ± 0.92% day^{-1}), and the slowest average tensile loss occurred in agricultural streams in the winter (1.44% ± 0.29% day^{-1}). Among all groups and all seasons, the fastest average TSLDD was measured in forested streams in the winter (0.36% ± 0.12% degree-day^{-1}), and the slowest average TSLDD was in agricultural streams in the summer (0.15% ± 0.05% degree-day^{-1}).

The linear mixed effect model (LMEM) assessing TSLD showed that the interaction between group and season was significant (Table 4). It also revealed that season had a significant effect on TSLD. However, the effect of riparian vegetation alone on TSLD was non-significant. Post hoc analyses (Table 5) revealed in summer, agricultural streams experienced 27% less tensile loss on average than forested streams. There was no difference between forested an non-forested streams in the fall or winter.

Table 4. Linear mixed effects model ANOVA table for tensile loss (percent per day) among seasons and groups.

Source of variation	**Sum of squares**	**Numerator d.f.**	**Mean squares**	**Denominator d.f.**	**F**	***P***
Group	0.038	0.038	1	7.052	2.624	0.149
Season	2.071	1.036	2	136.124	71.949	<0.001
Group x Season	0.124	0.062	2	136.124	4.309	0.015

Table 5. Post hoc pairwise comparison for both groups (Agriculture, Forest) across three seasons (Summer, Fall, Winter) using tensile loss percent per day data.

Season	**Group**	**Estimate**	**Standard error**	**Degrees of freedom**	**t-ratio**	**P**
Summer	Agriculture – Forest	-0.135	0.053	13.9	-2.527	0.024
Fall	Agriculture – Forest	-0.089	0.052	12.9	-1.702	0.113
Winter	Agriculture – Forest	0.006	0.053	13.5	0.114	0.911

The LMEM performed on percent per degree-day tensile loss data showed that the interaction between group and season was not significant (Table 6). Season had a significant effect on tensile loss (percent per day), and the effect of group alone approached significance. Because group was approaching significance, a pairwise comparison was conducted to look at the differences between groups in each season. Though no significant differences were observed, the differences between both groups in summer, and in fall, approached significance (Table 7).

Table 6. Linear mixed effects model ANOVA table for tensile loss (percent per degree-day) among seasons and groups

Source of variation	**Sum of squares**	**Numerator d.f.**	**Mean squares**	**Denominator d.f.**	**F**	***P***
Group	0.039	0.039	1	7.003	4.210	0.079.
Season	1.575	0.788	2	136.024	85.515	<0.001
Group x Season	0.015	0.005	2	136.024	0.568	0.568

Table 7. Post hoc pairwise comparison for both groups (Agriculture, Forest) across three seasons (Summer, Fall, Winter) using tensile loss percent per degree-day data.

Season	**Group**	**Estimate**	**Standard error**	**Degrees of freedom**	**t-ratio**	**P**
Summer	Agriculture – Forest	-0.149	0.071	8.73	-2.089	0.067
Fall	Agriculture – Forest	-0.152	0.071	8.51	-2.141	0.063
Winter	Agriculture – Forest	-0.115	0.071	8.64	-1.612	0.143

3.2 Environmental Variables

Average stream temperature was highest in the summer (16.37˚C) and lowest in the winter (4.81˚C) (Figure 3). GLMs run to compare average temperature between both non-forested and forest groups in each season revealed significant differences between seasons and groups (Table A1). Further post hoc analyses revealed no significant differences between groups within each season.

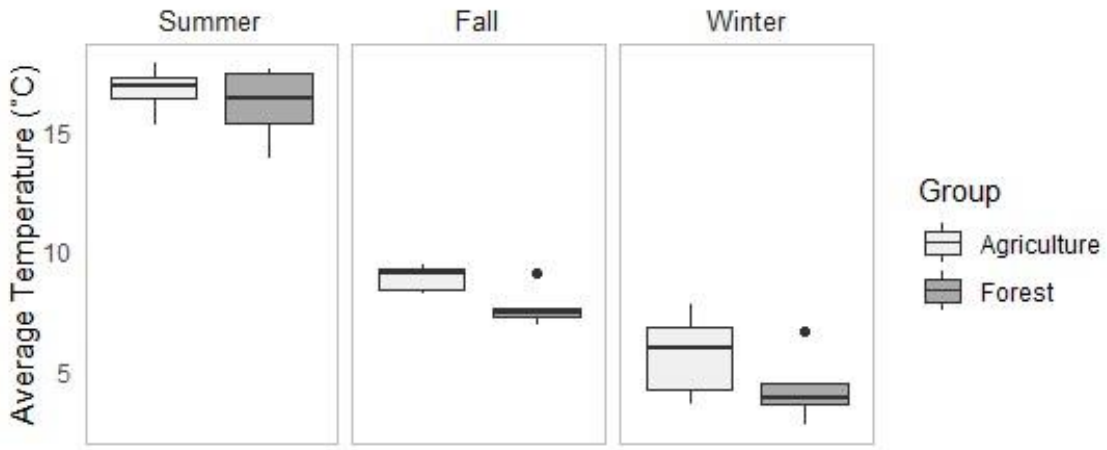


Figure 3. Boxplot of average stream temperature experienced by forested and non-forested riparian agricultural streams in summer, fall, and winter within the Norfolk Sandplain in southern Ontario.

The highest average SRP concentrations were seen in forested streams in the summer (30 µg/L), followed by forested streams in the fall (21 µg/L) and winter (17 µg/L) (Figure 4). The lowest concentrations were seen in the winter in non-forested streams (8 µg/L). Average nitrate-nitrite concentrations ranged from 1201 µg/L in agricultural streams in the summer to 4189 µg/L in forested streams in the winter. Average ammonium concentrations ranged from 27 µg/L in forested streams in the fall to 89 µg/L in forested streams in the winter. GLMs and pairwise post-hoc tests revealed that there was a significant difference between SRP in summer, fall, and winter (Table A1), where forested streams had higher concentrations of SRP in all cases. No differences in nitrate nitrite or ammonium were observed between groups in any of the seasons.

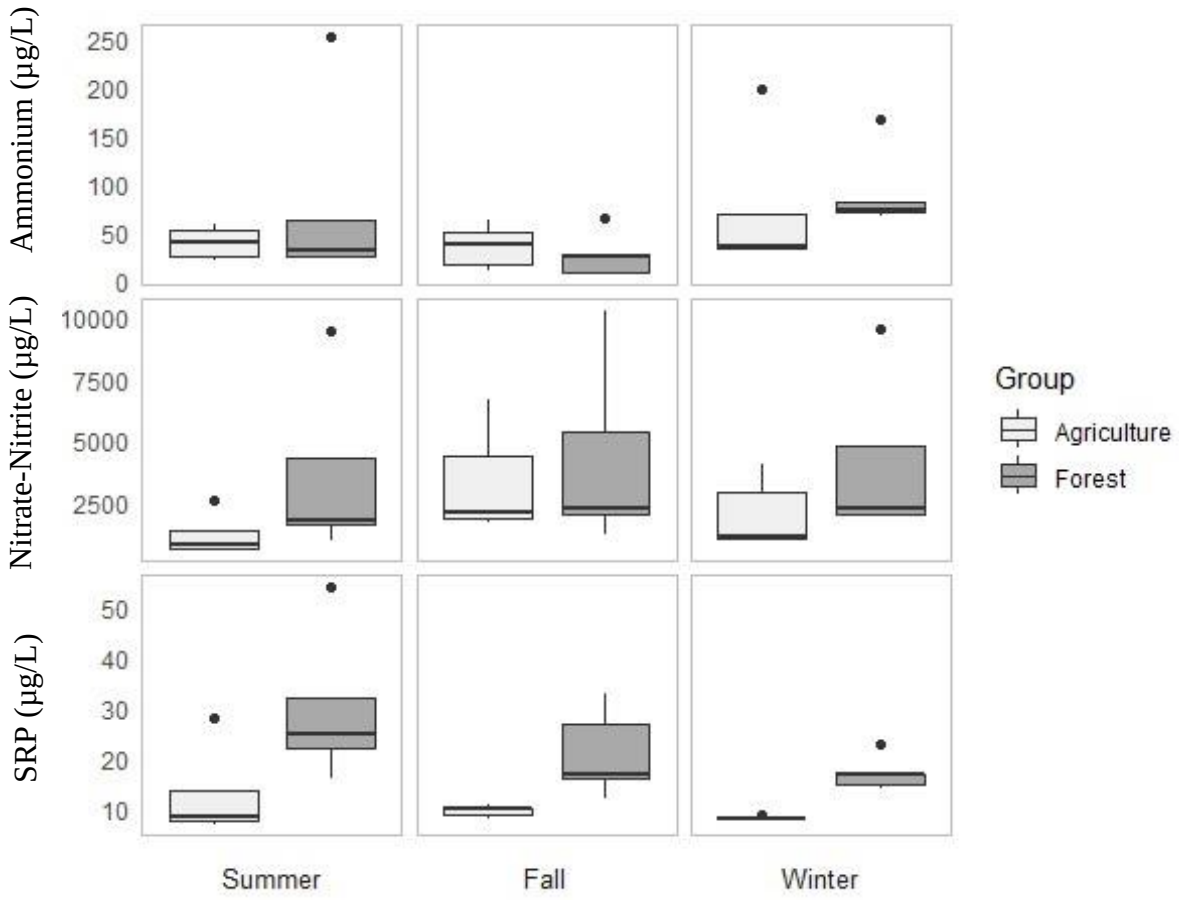


Figure 4. Boxplot showing differences in average ammonium, nitrate nitrite, and SRP experienced by forested and non-forested riparian streams within an agricultural landscape in summer, fall, and winter in the Norfolk Sandplain in southern Ontario.

Average conductivity ranged from 437 mS/m in the summer in forested streams to 602 mS/m in summer in non-forested streams. Average pH ranged from 7.1 to 8.7. Average turbidity was low and did not exceed 22 NTU. Average flow velocity was higher in forested streams compared to non-forested streams in summer, fall and winter. Average chl-a concentrations were highest in summer (0.05 µg/cm^2) and lowest in fall (0.04 µg/cm^2). GLMs revealed that in summer, there was a significant difference in conductivity between agriculture and forest groups

(Table A1) where agricultural sites had higher conductivity (Figure 5). No differences between groups were observed in pH, though there was a significant difference between seasons. No differences were observed in turbidity, flow velocity, or chl-a between groups in each season.

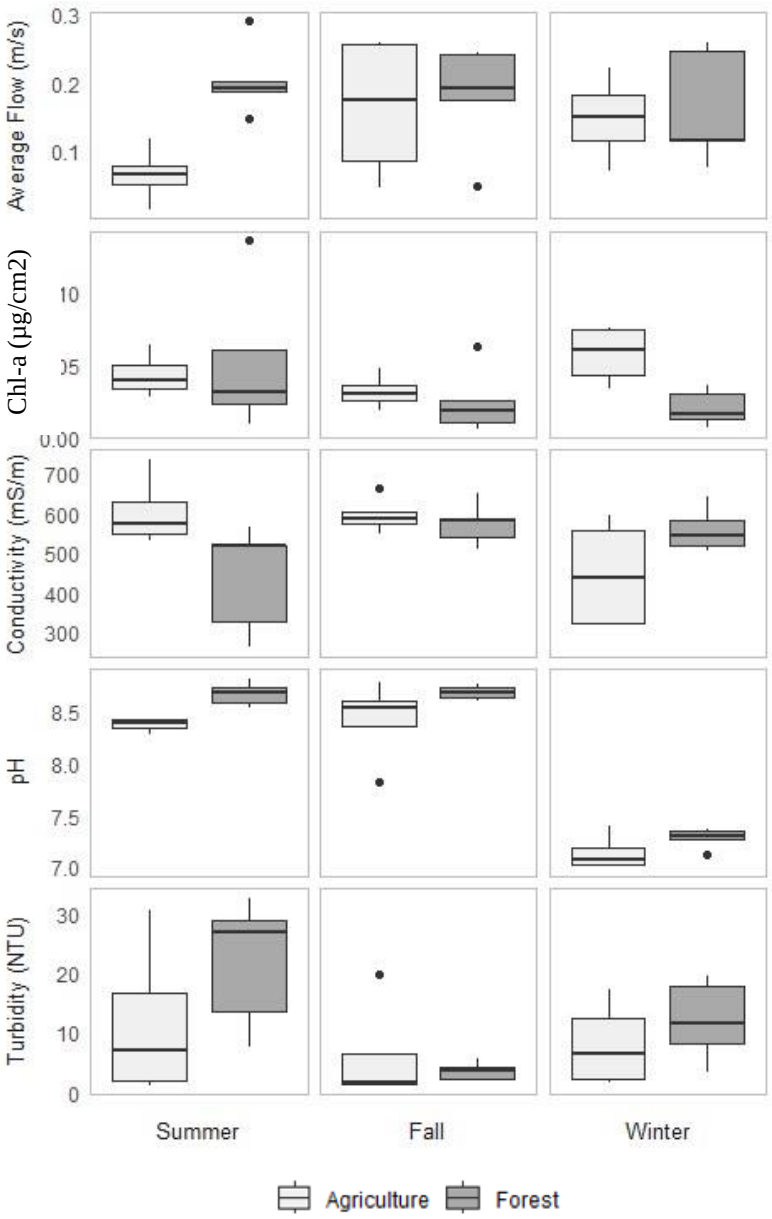
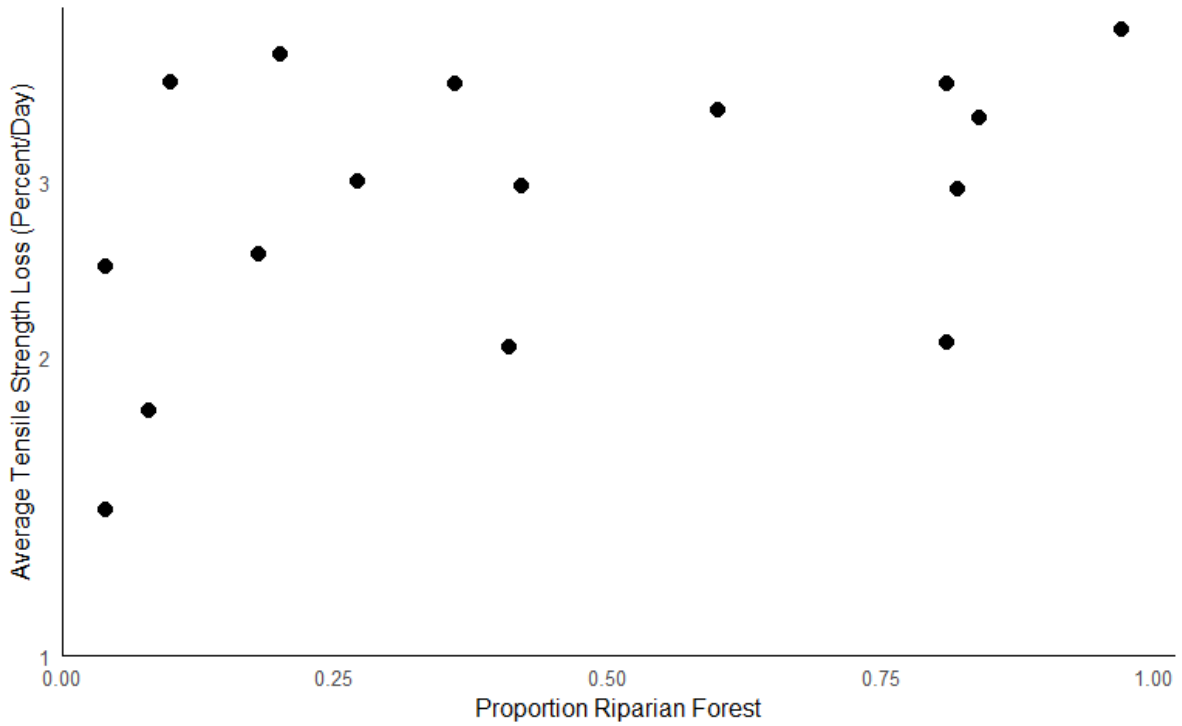


Figure 5. Boxplot showing average flow (m/s), average chl-a (µg/cm2), conductivity (mS/m), pH, and turbidity (NTU) in forested and non-forested riparian streams within an agricultural landscape in summer, fall, and winter in the Norfolk Sandplain in southern Ontario.

PLS regressions run on TSLD data revealed no significant components predicting tensile loss (Q^2 = - 0.288). Similarly, PLS regression run on TSLDD data also revealed no significant components (Q^2 = -7.116).

3.3 Association between cellulose decomposition and riparian forest cover

Average TSLD for all cotton strips in the summer was 2.99 ± 1% day $^{-1}$. The highest average TSLD was recorded at the site with the most riparian forest (4.28 ± 0.26 % day^{-1}; 97% riparian forest; Figure 8). The lowest average TSLD was observed at the stream with the least amount of riparian forest (1.40 ± 0.38% day^{-1}; 4% riparian forest). Average TSLDD among all strips was 0.17% degree-day^{-1} (± 0.05% degree-day^{-1}). The highest TSLDD was at the site with the highest proportion of riparian forest (0.25 ± 0.02 % degree-day^{-1}) and the lowest TSLDD was recorded at the site with the lowest proportion of riparian forest (0.092 ± 0.03 % degree-day^{-1}).



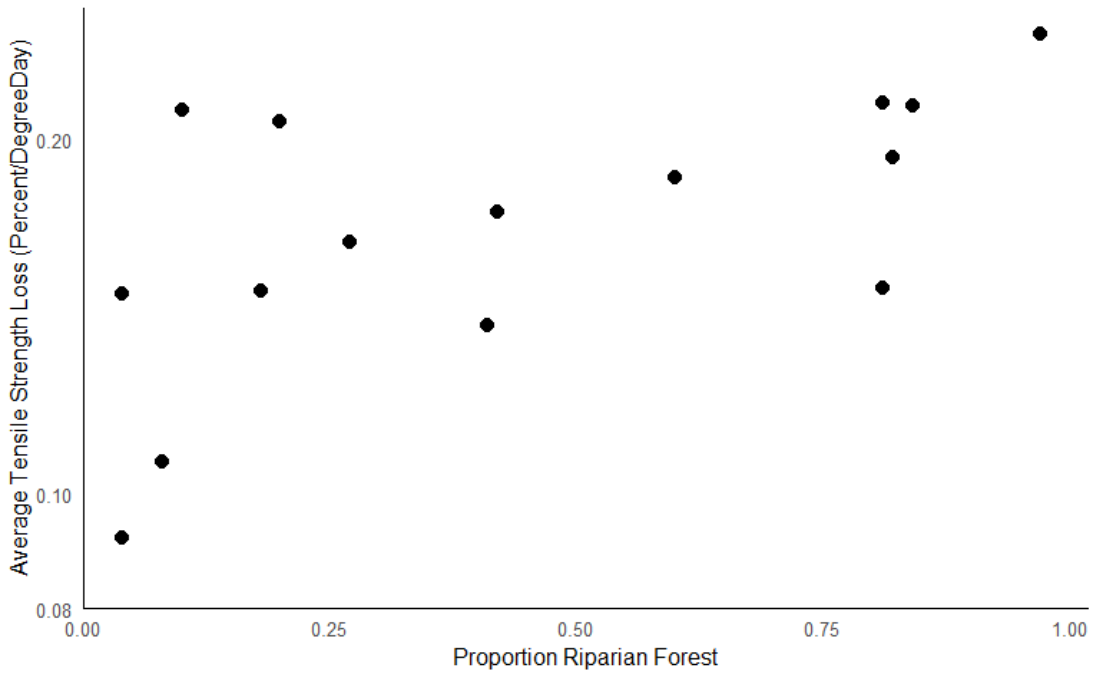


Figure 6. Scatter plots showing average tensile strength loss percent per day (TSLD, log scale) on and average tensile strength loss percent per degree-day (TSLDD, log scale) against

proportion of riparian area that is forested in the Norfolk Sandplain in southern Ontario. One stream is excluded from TSLDD due to missing temperature data.

Segmented regression revealed no significant associations or breakpoints between TSLD and proportion of riparian forest cover ($p = 0.284$). Segmented regression revealed a significant linear association (Type 1) between TSLDD and proportion of riparian forest cover ($R^2 = 0.422$, $p = 0.006$), and no breakpoints were determined.

A spearman correlation matrix revealed a positive correlation between the proportion of riparian forest and SRP ($r = 0.55$), as well as pH ($r = 0.72$). There was also a negative correlation between the proportion of riparian forest and average depth ($r = -0.53$). Notably, there was no correlation found between proportion of riparian forest and other nutrients (nitrate-nitrite, ammonium), temperature (degree-days), or any of the other measured environmental variables (Figure A1; A2; A3).

PLS regressions for TSLD and TSLDD revealed no significant components ($Q^2 = -8.118$, $Q^2 = -2.342$, respectively).

4.0 Discussion

The goal of my study was to investigate how riparian forest cover protects organic matter processing in agricultural streams. Results indicated significant differences in cellulose decomposition between streams with forested and unforested riparian zones with differing amounts of riparian forest cover in the catchment. However, results were opposite to what was predicted, with faster decomposition in streams with forested riparian zones, compared to streams with unforested riparian zones. Moreover, this difference was only observed in summer with TSLD. In contrast, for TSLDD, there were differences in decomposition in fall as well as summer.

4.1 Riparian Forest and Organic Matter Processing

Streams without riparian forests were found to have lower decomposition rates compared to streams that had riparian forests, suggesting that the presence of riparian forest either protects organic matter processing from negative impacts of agriculture, or the presence of riparian forest stimulates organic matter processing in agricultural catchments. To further understand this, I compared my results with those of Webb et al. (2019), who measured decomposition in least disturbed (i.e., reference) streams in the region. When comparing these reference streams my study results (Table A2; Table A3), we found no difference in decomposition between these reference sites and the streams in our study with riparian forest (Figure 10). However, there was a difference between the reference sites and streams in our study without riparian forest. This indicates that forest in the riparian zone is able to protect organic matter processing in streams, and that cellulose decomposition is being inhibited in the agricultural streams.

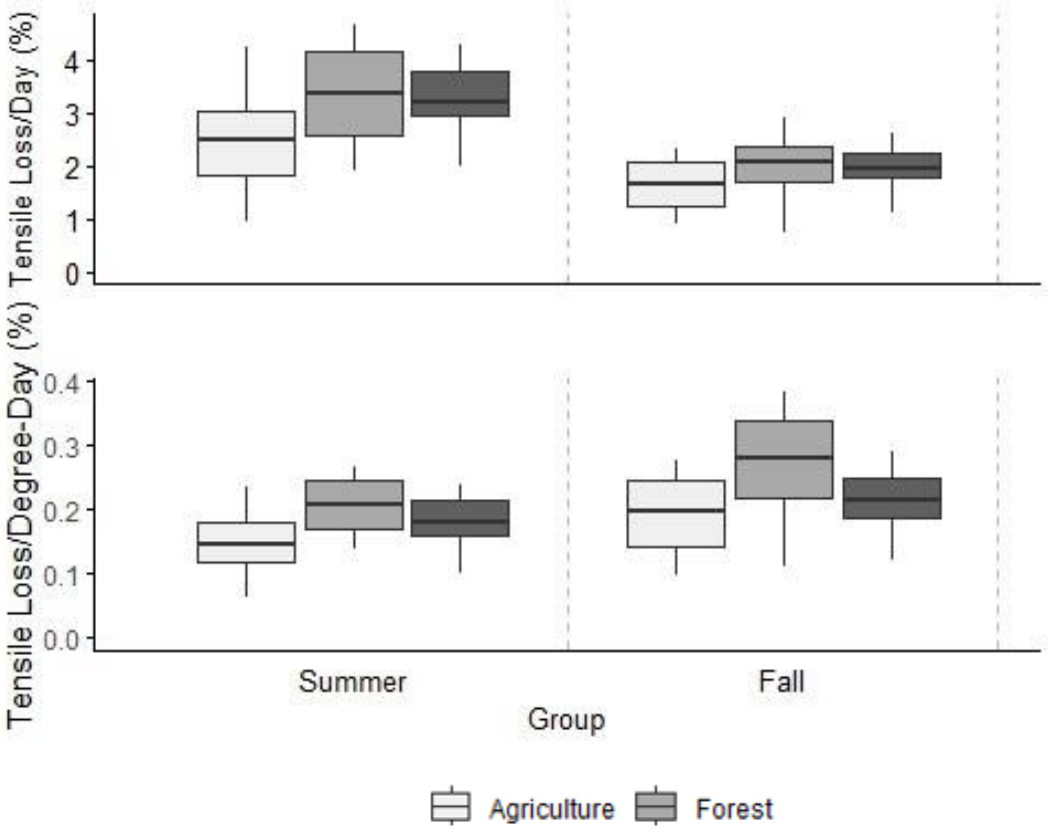


Figure 7. Boxplot of tensile strength loss per day and tensile strength loss per degree day among forested riparian, non-forested riparian, and reference streams in summer and fall within the Norfolk Sandplain in southern Ontario using reference data obtained from Webb et al., 2019.

The protective role of riparian forests in organic matter processing within agricultural streams is not consistently established in the literature. One study found no difference in decomposition rates between streams with open and closed canopy riparian areas in cattle-dominated catchments (Barrios et al., 2022). Similarly, Goss et al. (2014) observed no significant differences in leaf decomposition when analyzing decomposition along two agricultural streams that led into 1km forest fragments. (Goss et al., 2014). The lack of observed differences in these

studies could stem from their limited focus on specific riparian reaches rather than entire catchments. Conversely, another study found higher decomposition rates in agricultural sites with reduced riparian cover and diversity for two of three litter species used to measure decomposition, though they also only evaluated a 100-meter reach and did not account for riparian width or fully characterize the catchments land use (Cornejo et al., 2020). The differing amounts of agriculture across study sites further complicates this comparison. Tanaka et al. (2015) provided contrasting evidence, showing that decomposition rates increased and stabilized within 100 meters of a forest remnant in an agricultural stream. Unlike these studies, my research considered the riparian area across the entire catchment, which may explain why previous studies did not necessarily detect a protective effect of riparian forests.

The finding that agriculture inhibits organic matter processing is consistent with other studies that have found that increasing agricultural land use results in inhibited microbial-mediated organic matter breakdown (Piscart et al., 2009; Voß et al., 2015). Other studies have found contrasting results, with some reporting increased organic matter processing in more agricultural catchments (Arroita et al., 2013; Mancuso et al., 2023), or agriculture having no effect at all (Hagen et al., 2006; Mctammany et al., 2008). Geographic variability in environmental conditions could be a reason for the differences seen in the studies, as geological characteristics can significantly affect organic matter processing (Graça et al., 2015). For example, two studies that found that organic matter processing is inhibited by agricultural land use were located in similar regions, western France (Piscart et al., 2009) and Germany (Voß et al., 2015). Similarly, two studies that found no effect of agriculture on organic matter processing were both conducted in the southern Appalachians (Hagen et al., 2006; Mctammany et al., 2008). While the impact of agricultural land use on organic matter processing is evident, with some

studies showing inhibition and others reporting no effect or even enhancement, the variability in these outcomes highlights the influence of additional environmental factors.

Temperature and nutrients did not appear to contribute to the differences in tensile loss, contrary to previous studies that have found increased temperature and nutrients significantly increasing cellulose decomposition (Ferreira & Chauvet, 2011; Mancuso et al., 2023; Yue et al., 2016). Similarly, flow velocity was not correlated with cellulose decomposition, contrary to other findings that show increased flow velocity can stimulate cellulose decomposition (Bastias et al., 2020). Further, riparian forest did not appear to moderate these environmental variables, contrary to what past research suggests. The fact that these environmental variables were not moderated by riparian forests could be due to tile drainage or groundwater contributions, influencing temperature, nutrients, and flow regimes in streams (King et al., 2015; Loheide & Gorelick, 2006; McMahon et al., 1994; Poisson & Yates, 2022; Siebers et al., 2016) and limiting the beneficial effects of riparian forests. Instead, we look to other variables not measured that could explain differences in decomposition.

A difference in microbial community composition, a variable not measured in this study, could be a reason why decomposition rates were lower in streams with less riparian forest cover. Indeed, streams exposed to agriculture have been found to exhibit reduced microbial community diversity (Laperriere et al., 2020). Likewise, the presence of forest cover in the riparian area has been found to help maintain microbial community diversity (Chavarria et al., 2021). Microbial community diversity has been shown to increase rates of organic matter decomposition (Boyero et al., 2021), which supports the finding that streams with riparian forests have greater decomposition. Further research should investigate how microbial community composition influences decomposition in agricultural streams with varying riparian forest.

The presence of agrochemicals is another factor that could be explaining by finding of inhibited decomposition rates in agricultural streams without riparian forests. Pesticides have been shown to inhibit the microbial community and impair the processing of organic matter (Mack et al., 2024). Similarly, antibiotics that are used in raising livestock and often end up in manure used as fertilizer have also been shown to inhibit the microbial community (Ding & He, 2010; Maul et al., 2009). These substances have also been shown to be successfully buffered by the presence of riparian forest, reducing their impact on stream ecosystems (Bradley et al., 2020). It is possible that the presence of these compounds in the streams that lacked riparian forest cover caused the observed inhibition of cellulose decomposition. This could be particularly apparent in the summer when they are most often applied, explaining the differences observed primarily in the summer, and the relative lack of differences observed in the fall and winter. Further research on the impact that agrochemicals have on decomposition in agricultural streams is required to determine whether pesticides and antibiotics are playing a role in the inhibition of the microbial communities and as a result, the inhibition of decomposition.

4.2 Implications

The findings of this study underscore the importance of riparian forests in protecting organic matter processing in agricultural streams. The observed decrease in organic microbial-mediated decomposition in streams with forested riparian zones suggests that riparian buffers can mitigate the adverse effects of agricultural activities. Notably, this study reveals that a 20-meter riparian forest buffer throughout the catchment is sufficient to protect organic matter processing. This relatively modest width of riparian forest is practical and within the guidelines that many organizations often suggest (10m – 30m) (Broadmeadow & Nisbet, 2004).

This study also suggests that the mechanisms through which agriculture impacts organic matter processing extend beyond the commonly targeted environmental variables: temperature and nutrients. This highlights a need for a broader focus that included other agricultural stressors, such as agrochemicals, altered hydrological regimes, and shifts in microbial community composition.

4.3 Limitations and Future Directions

Despite the valuable insights provided by this study, there are several limitations that need to be considered. This study investigated forest cover in the riparian zone for the entire catchment, which can be impractical from a management perspective as maintaining or restoring that much riparian area can be challenging. Future studies should investigate whether there is a smaller length of riparian forest that can successfully protect organic matter processing in agricultural streams. Studies should also look at whether an intermittently riparian forested system could also protect organic matter processing potentially by providing critical habitat for organisms and a more consistent and natural supply of organic matter. Necessary width should also be researched further, as this study only identified that a 20-meter buffer on either side of the stream appears to protect microbial-mediated organic matter processing. Perhaps there is a smaller width that could still protect organic matter processing that would be more practical to implement throughout the catchment.

Moreover, while this study focused on a specific geographic region, the variability in findings from different regions indicates that broader geographic studies are necessary to generalize these findings. Differences in findings from different regions could be due to a variety of factors, for example different sediment types, differing species of trees in the riparian zone or

perhaps differing compositions of microbial communities that naturally occur across regions. Variations in pesticide use could also be contributing to differences in decomposition, as crop type and size of farm have both been found to influence pesticide patterns, with larger farms having more intensive pesticide use (Jørgensen et al., 2019). Studies should seek to more clearly characterize these variables when conducting research on organic matter dynamics in agricultural settings with forest cover. Furthermore, the impacts of climate change on organic matter processing and riparian forest efficacy warrant further investigation, as changing precipitation patterns and temperatures could alter the dynamics observed in this study. For example, increased frequency and intensity of storm events could lead to greater runoff and erosion, potentially overwhelming riparian forest buffers and limiting their ability to filter pollutants (Welsh et al., 2019).

5.0 Conclusions

This study concludes that a 20-meter riparian forest buffer throughout the catchment effectively protects microbial-mediated organic matter processing in agricultural streams. Contrary to initial predictions, environmental characteristics that were measured did not drive decomposition rates. Despite none of the environmental variables explaining decomposition, the presence of riparian forests consistently maintained near-natural levels of organic matter processing, suggesting that other factors may play significant roles, such as microbial community composition, agrochemicals, and varying agricultural practices. This highlights the critical importance in preserving stream health in human-altered landscapes. These findings emphasize the benefits of integrating riparian forest buffers into broader environmental management practices to minimize impacts of agriculture of organic matter processing.

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Appendix

Table A1. GLMs of environmental variables from the 10 study sites comparing streams with forested and non-forested riparian zones in the Norfolk Sandplain.

Environmental variable	Source of variation	Sum of squares	Degrees of freedom	Mean squares	F	p
Average Daily Temperature	Group	0.047	1	0.047	5.384	0.032
	Season	1.151	2	0.576	65.488	<0.001
	Group x Season	0.007	2	0.384	0.384	0.686
SRP	Group	0.657	1	0.657	21.451	<0.001
	Season	0.118	2	0.059	1.931	0.174
	Group x Season	0.010	2	0.005	0.167	0.847
Nitrate-Nitrite	Group	0.392	1	0.392	3.315	0.085.
	Season	0.299	2	0.150	1.264	0.306
	Group x Season	0.118	2	0.059	0.500	0.615
Ammonium	Group	0.095	1	0.095	0.845	0.370
	Season	0.829	2	0.415	3.684	0.046
	Group x Season	0.293	2	0.147	1.303	0.296
Conductivity	Group	0.008	1	0.008	1.050	0.319
	Season	0.025	2	0.013	1.604	0.229
	Group x Season	0.068	2	0.034	4.338	0.029
pH	Group	0.084	1	0.084	2.133	0.161
	Season	8.026	2	4.013	101.906	<0.001
	Group x Season	0.052	2	0.039	0.653	0.532
Turbidity	Group	0.453	1	0.453	2.367	0.141
	Season	1.410	2	0.705	3.684	0.046
	Group x Season	0.314	2	0.157	0.819	0.457
Flow velocity	Group	0.021	1	0.021	3.715	0.068
	Season	0.004	2	0.002	0.379	0.689
	Group x Season	0.022	2	0.011	2.005	0.160
Chlorophyll-a	Group	0.208	1	0.208	2.238	0.153
	Season	0.177	2	0.088	0.951	0.406

	Group x Season	0.027	2	0.014	0.147	0.865

Environmental variable	Season	Group	Estimate	Standard error	Degrees of freedom	t-ratio	p
Average Daily Temperature	Summer	Agriculture-Forest	0.018	0.062	18	0.286	0.778
	Fall	Agriculture-Forest	0.062	0.063	18	0.979	0.340
	Winter	Agriculture-Forest	0.107	0.081	18	1.319	0.204
Conductivity	Summer	Agriculture-Forest	0.154	0.059	18	2.602	0.018
	Fall	Agriculture-Forest	0.017	0.059	18	0.285	0.779
	Winter	Agriculture-Forest	-0.128	0.077	18	-1.673	0.112

Table A2. Linear mixed effects model ANOVA table and post hoc Tukey for tensile loss (percent per day) among seasons (summer, fall) and groups (forested, non-forested, reference) with data from my study and Webb et al. (2019), all within the Norfolk Sandplain.

Source of variation	Sum of squares	Numerator d.f.	Mean squares	Denominator d.f.	F	*P*
Group	0.06667	0.03333	2	13.119	3.6329	0.05552.
Season	1.48670	1.48670	1	141.720	162.0339	<0.001***
Group x Season	0.02166	0.01083	2	141.628	1.1806	0.31011

Season	Group	Estimate	Standard error	Degrees of freedom	t-ratio	P
Summer	Agriculture – Forest	-0.1582	0.0602	15.7	-2.629	0.0459
Summer	Agriculture – Reference	-0.1572	0.0576	15.7	-2.728	0.0379
Summer	Forest – Reference	0.001	0.0583	16.5	0.017	0.9998
Fall	Agriculture – Forest	-0.1094	0.0608	16.2	-1.800	0.2009
Fall	Agriculture – Reference	-0.1038	0.0589	16.9	-1.763	0.2117
Fall	Forest – Reference	0.006	0.0575	15.6	0.098	0.9948

Table A3. Linear mixed effects model ANOVA table and post hoc Tukey for tensile loss (percent per degree-day) among seasons (summer, fall) and groups (forested, non-forested, reference) with data from my study and Webb et al. (2019), all within the Norfolk Sandplain.

Source of variation	**Sum of squares**	**Numerator d.f.**	**Mean squares**	**Denominator d.f.**	**F**	***P***
Group	0.077	0.038	2	13.101	4.683	0.029
Season	0.386	0.386	1	141.564	46.986	<0.001
Group x Season	0.020	0.010	2	141.483	1.225	0.297

Season	**Group**	**Estimate**	**Standard error**	**Degrees of freedom**	**t-ratio**	**P**
Summer	Agriculture – Forest	-0.174	0.060	15.4	-2.907	0.027
Summer	Agriculture – Reference	-0.128	0.057	15.4	-2.231	0.097
Summer	Forest – Reference	0.046	0.058	16.1	0.796	0.711
Fall	Agriculture – Forest	-0.174	0.060	15.9	-2.885	0.028
Fall	Agriculture – Reference	-0.080	0.058	16.5	-1.372	0.378
Fall	Forest – Reference	0.094	0.057	15.3	1.645	0.258

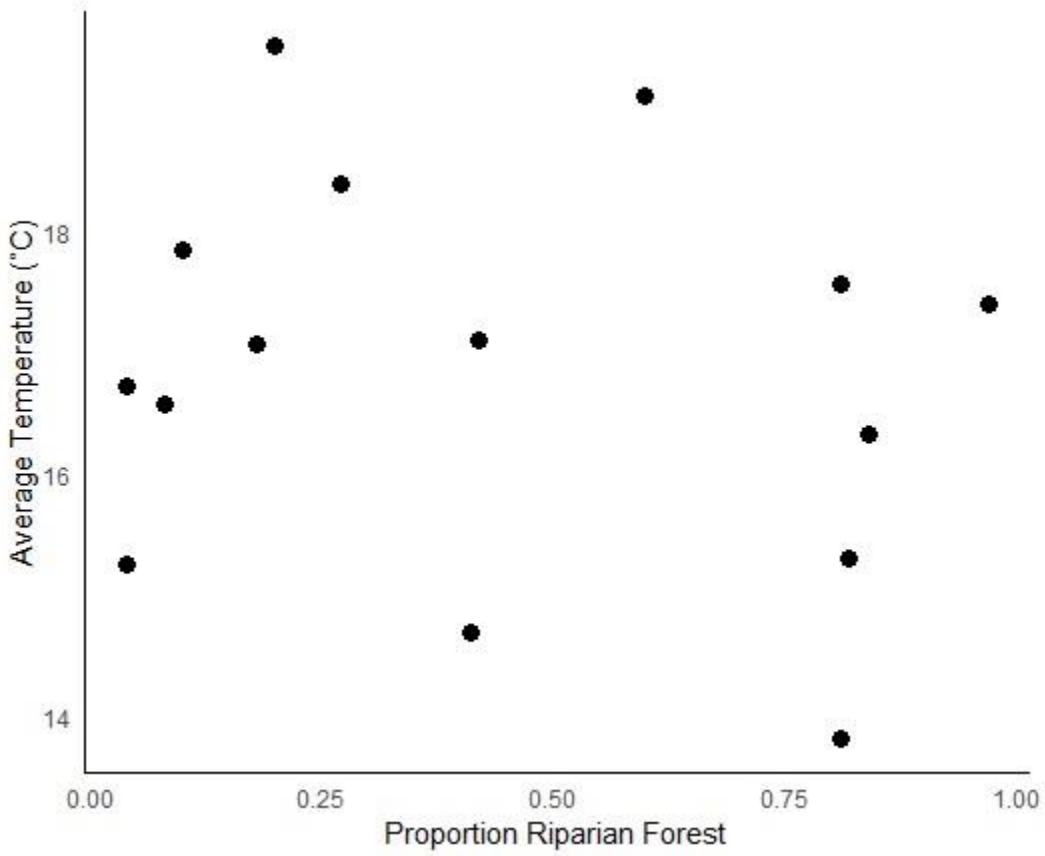

Figure A1. Scatterplot showing average daily stream temperature on the y axis and proportion of riparian forest on the x axis in the summer in the Norfolk sandplain of southern Ontario. One site excluded due to loss of temperature data.

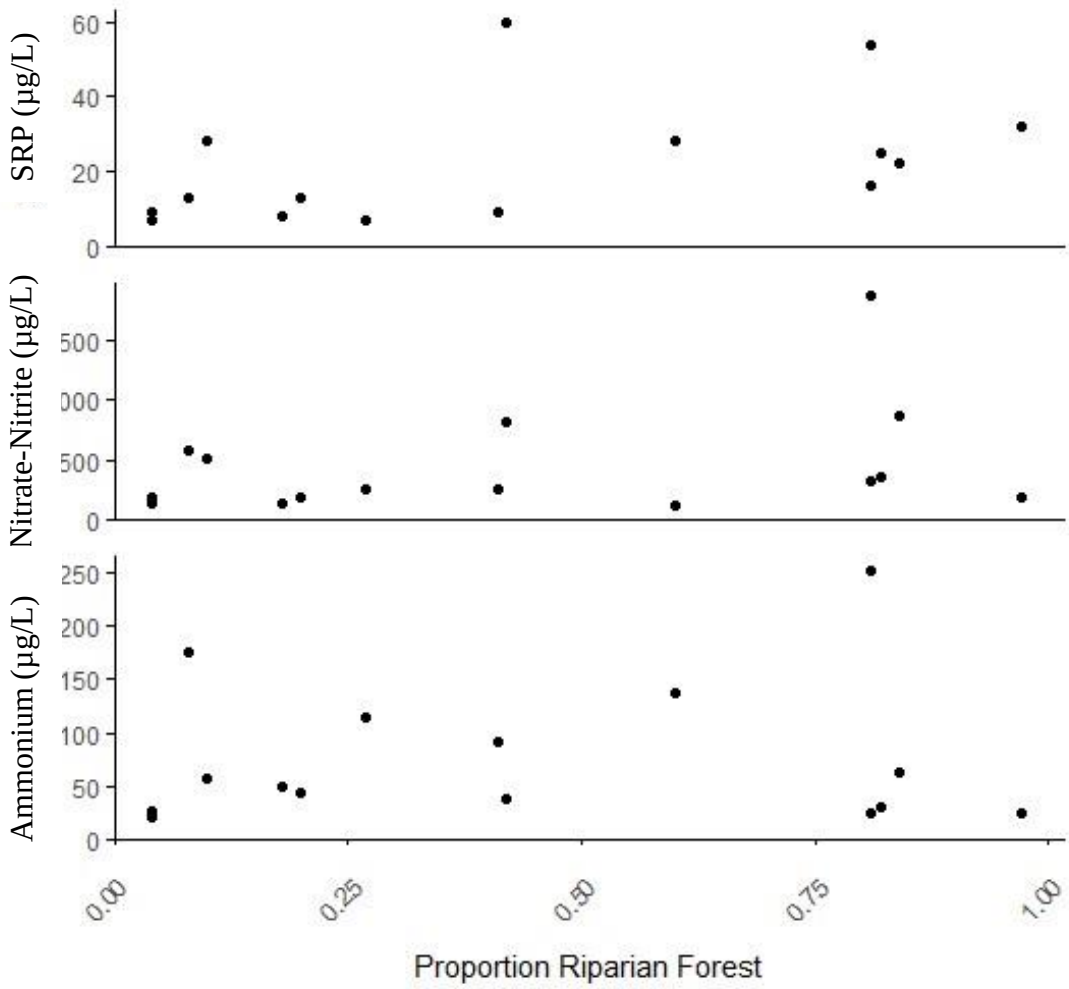

Figure A2. Scatterplots showing SRP (µg/L), Nitrate-Nitrite (µg/L) and Ammonium (µg/L) on the y axis and proportion of riparian forest on the x axis in the summer in the Norfolk sandplain of southern Ontario.

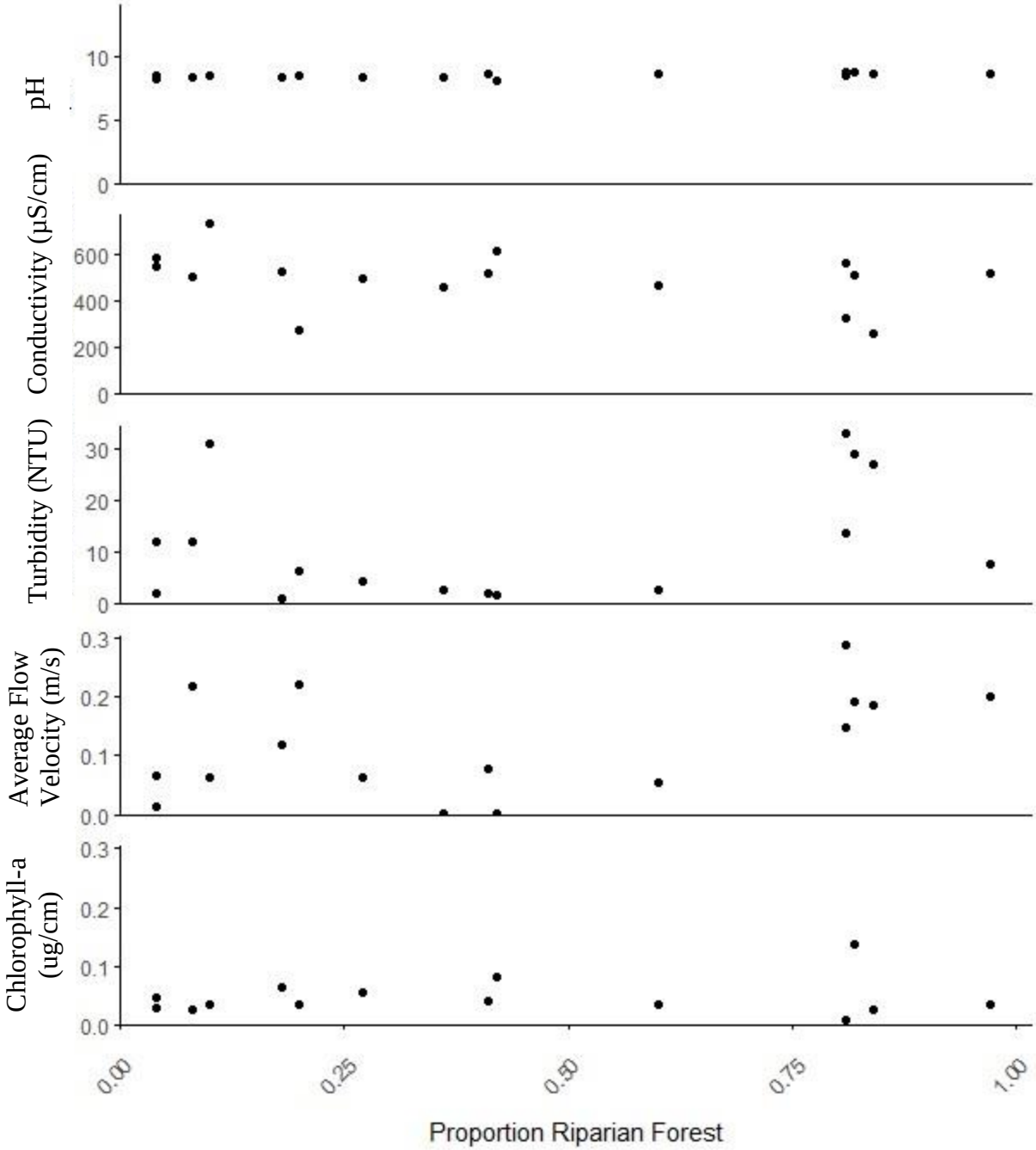

Figure A3. Scatterplots showing pH, Conductivity (µS/m), Turbidity (NTU), Average flow velocity (m/s), and chlorophyll-a (µg/cm) on the y axis and proportion of riparian forest on the x axis in the summer in the Norfolk sandplain of southern Ontario.