Litter input, soil quality and soil carbon dioxide production rates in varying riparian land uses along a first order stream in Southern Ontario, Canada.

by

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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

Forested riparian zones, which function as a buffer between agricultural fields and streams, filter out contaminants and sediment from the fields thereby improving water quality, cool the water with shade from trees, stabilize the stream bank and provide habitat for wildlife. However, in many agricultural areas, riparian vegetation has been removed for crop production or pasture purposes. Riparian restoration or rehabilitation is a way of restoring riparian ecosystem functions. This study examines the effect of riparian rehabilitation via tree planting along a first-order creek in Southern Ontario, 25 years after rehabilitation. Litter input, soil quality parameters and soil CO_2 production rates were determined for the rehabilitated riparian zone, a grass-forb riparian zone and a natural forest riparian zone. Total litter input was 480, 580 and 295 g m^{-2} y^{-1} for the rehabilitated riparian zone, grass riparian zone and forest riparian zone, respectively. Soil bulk density was higher and hydraulic conductivity was lower for the rehabilitated riparian zone compared to the grass riparian zone and forest riparian zone. The concentration and soil stock of organic carbon and total nitrogen was lowest for the rehabilitated riparian zone compared to the grass riparian zone and forest riparian zone which were similar. The effect of riparian zone on soil CO_2 production rates varied over the season. From spring to mid-summer, rates were 167, 224 and 104 mg C m^{-2} h^{-1} for the rehabilitated riparian zone, grass riparian zone and forest riparian zone, respectively. Soil CO_2 production rates did not differ significantly ($p < 0.05$) between riparian zones for late summer and fall sampling dates. Soil CO_2 production rates were significantly negatively correlated with soil C/N and positively correlated with soil pH and litter input. Soil CO_2 production rates were positively correlated with soil temperature ($r = 0.32$) and negatively correlated with soil moisture ($r = -0.48$). Of the three riparian zones, the natural forest riparian zone exhibited the least amount of seasonal fluctuation for soil CO_2 production rates, soil moisture and temperature. Results from this research indicated that more time

is needed before soil quality and soil CO_2 production rates of the rehabilitated riparian zone reach values similar to the natural forest riparian zone.

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Introduction

Climate change, poor water quality, food insecurity and loss of biodiversity are among the serious global environmental issues that the world is facing (Diamond, 2005). Pressure placed on the world's natural resources such as soil and water by a growing population has led to problems such as reduced water quality, soil degradation and an increase in atmospheric green house gases (GHG) (Foley et al., 2005). The magnitude of negative impacts will likely increase if no mitigation actions are implemented.

The terrestrial part of our earth represents a resource that, if managed correctly, could mitigate some of the negative impacts of anthropogenic activities. Soil is the second largest carbon (C) pool and represents an opportunity to affect climate change through C sequestration (Lal, 2004b). Land management techniques which reduce sediment runoff and planting of permanent vegetation along riparian zones could improve water quality (Lowrance et al., 1984; Osborne and Kovacic, 1993; Schultz et al., 2004). Maintaining or improving soil organic matter content in the soil could increase crop yields thereby reducing food insecurity (Lal, 2004b). Protecting terrestrial natural ecosystems or restoring degraded land would prevent further loss of biodiversity.

Land use change, driven by the need for shelter and food of an increasing human population, has resulted in large portions of natural ecosystems being converted to agricultural production (Foley et al., 2005). Fields used for crop production or pasture often extend to banks of rivers or creeks affecting water quality through field runoff and erosion. With extensive research showing the benefits of permanent vegetation in the riparian zone pollutants (Clausen et al., 2000; Hill, 1996; Zaimes et al., 2008), interest in restoration or rehabilitation of riparian zones has increased. Although numerous projects have been implemented, few have been followed up with further research to determine whether the objectives of the restoration work were met (Bernhardt et al., 2005). In addition, the potential of rehabilitated riparian zones to reduce soil GHG emissions and sequester C has not been thoroughly investigated. Research is needed to determine the environmental impacts of projects so that future restoration and rehabilitation will be promoted and guided by successes of previous work.

Research Goals and Objectives

The following research was undertaken to determine the effect of rehabilitation of a riparian zone on soil CO_2 production rates and related soil quality indicators.

The objectives of the study were:

1) Quantify and compare litter fall in a rehabilitated forest riparian zone, a grass-forb riparian zone and a natural forest riparian zone.

2) Quantify and compare soil quality characteristics in a rehabilitated forest riparian zone, a grass-forb riparian zone, and a natural forest riparian zone.

3) Quantify and compare soil CO_2 production rates in a rehabilitated forest riparian zone, a grass-forb riparian zone, and a natural forest riparian zone.

1. Literature Review

1.1 Anthropogenic Climate Change and Green House Gases

Atmospheric concentrations of the GHGs carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) have increased measurably since 1750. Of these, CO_2 is the most important making up 77% of anthropogenic GHG emissions in 2004 (IPCC, 2007a). Atmospheric CO_2 increased from 280 ppm during the pre-industrial era to 379 ppm in 2005 and is attributed primarily to anthropogenic activities such as burning of fossil fuel and land use change (IPCC, 2007a). Although lower in atmospheric concentration than CO_2, CH_4 and N_2O are stronger absorbers of infrared radiation and have 23 and 296 times, respectively, the global warming potential of CO_2. From pre-industrial times to 2005, atmospheric concentrations of CH_4 and N_2O have increased from 715 to 1774 ppb and 270 to 319 ppb, respectively. Increases are predominantly attributed to agriculture and fossil fuel burning (IPCC, 2007a) .

Increases in atmospheric GHGs impact the climate of the earth by trapping infrared radiation (Rastogi et al., 2002). This has resulted in an increase in average global air and ocean temperatures, an increase in atmospheric water vapour content, a decrease in the area of glaciers and icecaps, a rise in global sea levels, and changes in precipitation resulting in longer droughts and more frequent extreme weather events (IPCC, 2007a). Although policies and practices have been implemented to reduce GHG emissions, they have not been able to counteract the growth of emissions globally (IPCC, 2007b). However, there is a large potential for reduction of GHG emissions using mitigation technologies and practices in many sectors of society. For example, more fuel efficient vehicles, alternate energy sources such as solar or wind, heat and power recovery, improved cropping and grazing practices, and reforestation could potentially reduce GHG emissions (IPCC, 2007b).

1.2 Land-use Change

The earth is required to support an ever increasing human population. World population grew from 2.5 billion to 6.8 billion from 1950 to 2009 and is estimated to reach 9.1 billion in 2050 (United Nations, Department of Economic and Social Affairs, Population Division, 2009). This is placing ever increasing demands on the natural resources such as water, arable land and oil. There is evidence that human resource use has already passed the carrying capacity of the earth (Kitzes et al., 2008). According to Wackernagel (2002) the ecological impact of humans may be calculated by determining the area of productive land and water necessary to produce the resources consumed and to mitigate the waste produced by people. They determined that since the 1980s, human use of natural resources has exceeded the capacity of the earth to replenish these resources.

The increasing demand for natural resources by a growing world population has resulted in large amounts of natural forest and grassland being used for agricultural production, development for shelter or industry and resource extraction such as mining or logging. Environmental impacts of the extensive land use change that has occurred over the past 150 years include a change in the hydrologic cycle, reduction in biodiversity, water quality degradation and a change in the global carbon cycle (Foley et al., 2005). For example, agricultural production has increased by 475 million hectares since 1960 (FAOSTAT, 2009), and now occupies about 40% of the land surface (Foley et al., 2005). Commercial agricultural production removes natural vegetation thereby reducing biodiversity, affects water quality through the runoff of fertilizers, pesticides and sediment into the watershed, may affect water availability through irrigation, and results in a loss of C storage in the soil.

The loss of soil C into the atmosphere through land use change adds to other increasing anthropogenic C emissions. Between 1950 and 1998, approximately 176 Pg of C were emitted as a result of land use change (IPCC, 2000). Houghton (2008) estimates that from 1850 to 2000, the global net flux from changes in land use was 148.6 Pg C and the average annual flux was 1.47 Pg C yr^{-1} from 2000 to 2005. In addition to increasing C emissions, loss of soil C as a result of land use change affects soil quality through the loss of soil organic matter.

1.2.1 Soil Organic Matter and Carbon Sequestration

Soil organic matter (SOM) is a vital component of a well structured soil and contributes to soil quality and productivity. SOM and soil structure are dynamically interrelated where SOM enhances soil structure through aiding in development of aggregates and a well structured soil will prevent rapid decomposition of SOM (Blanco-Canqui and Lal, 2004).

Lal (2004b) lists the principle functions of SOM: a source and sink of plant nutrients; increases cation exchange capacity; absorption of water; promotion of soil aggregation leading to increased water infiltration; substrate for energy for soil organisms; formation of soil aggregates; and a buffer against fluctuations in pH and soil temperature. Soil organic matter also indirectly affects water quality since improved soil aggregation and water infiltration reduces the sediment load and pollutants entering nearby streams. In addition, SOM contains approximately 55% C and therefore represents the largest terrestrial reservoir of C (Blanco-Canqui and Lal, 2004).

Carbon is sequestered through complex soil-atmosphere interactions controlling plant growth, CO_2 fluxes and microbial decomposition. Soil organic matter is composed of active and stable factions and long term C sequestration is determined by the plant residue quality and its proximity to microorganisms (Blanco-Canqui and Lal, 2004). According to Tisdall and Oades (1982), aggregate binding agents can be classified as transient, temporary and persistent. Transient agents are organic materials such as polysaccharides from roots and microbial biomass which are produced quickly and are easily decomposed. Temporary binding agents are roots and hyphae which accumulate in the soil and may remain for several months or last for several years. Organic matter such as humic compounds and polymers associated with microaggregates are referred to as persistent because they are resistant to microbial decomposition (Ashman and Puri, 2002).

Jastrow et al. (2007) describe two mechanisms for sequestering soil organic carbon (SOC): 1) biochemical alternation involves transformation of SOM via both biotic and abiotic processes to humic materials which are recalcitrant to microbial attack and 2) physiochemical protection can significantly increase the residence time of C in the soil by chemical or physical sorption to an existing surface and through occlusion of C in macro and micro aggregates, preventing access to microorganisms. To ensure optimal conditions for biochemical alteration, Jastrow et al. (2007) suggest managing soil to prevent extended wet or dry conditions and avoiding disturbance of the soil. Physiochemical protection may be optimized through practices that stabilize soil aggregates such as minimum tillage or perennial vegetation. In addition, preventing soil erosion is important for the protection of soil structure and therefore SOC.

In contrast, practices which often accompany land use change from natural ecosystems to agricultural production such as burning of vegetation, cultivation and drainage enhance mineralization and the release of CO_2 into the atmosphere (Lal, 2004b). The loss of SOC may cause soil degradation because of loss of soil structure, a decrease in infiltration and soil moisture, and a reduction in available nutrients all of which lead to a decline in potential for biomass production. The result is poorer plant growth and less organic matter returning to the soil resulting in yet more soil degradation. If practices maintaining this positive feedback loop continue, the soil will become incapable of sustaining soil flora or fauna (Lal, 2004b).

1.2.2 Land-use change and Carbon Sequestration

The five principal global C pools from largest to smallest are: oceanic (38,000 Pg C), geologic (5000 Pg C), pedologic (2500 Pg organic C and 1526 Pg inorganic C), atmospheric (760 Pg C) and biotic (560 Pg C). (Lal, 2004b). Major climatic factors that affect the amount of organic carbon (OC) in the soil are moisture and temperature. In general cool and wet climates tend to have soils higher in OC than warm, dry climates (Lal, 2004b). In addition, management of the soil has a large impact on whether OC is sequestered in the soil or emitted into the atmosphere. For example, tillage breaks down soil aggregates where C is sequestered, mixing microbes and organic particles. It also introduces oxygen into the soil, increases soil temperature and influences soil moisture. All of these factors increase the rate of SOC decomposition resulting in the release CO_2 into the atmosphere (Lal, 2004a).

In general, land use change from natural ecosystems to agricultural production decreases the SOM (Tufekcioglu et al., 2001). Higher rates of decomposition and oxidation or mineralization of SOC leads to a release of CO_2 to the atmosphere (Lal, 2004b). Grandy and Robertson (2006) reported losses of 85 g CO_2-C m^{-2} from the soil during the first 30 days after initial cultivation of a temperate midsuccessional grassland community. Carbon dioxide losses continued into the second and third year of cultivation. A review of literature examining C loss following cultivation of natural ecosystems reported the mean loss of soil C inventory in the A horizon to be between 24 to 43% of the original C of uncultivated soil (Davidson and Ackerman, 1993).

According to Lal (2004b) degraded agro-ecosystems represent an opportunity for C sequestration through improved management of agricultural soils by using practices which return more organic matter to the soil and reduce the loss of CO_2 through mineralization or erosion. Rastogi et al. (2002) suggested three strategies for C sequestration in soils: using soil management techniques to maintain or increase existing C levels such as reduced tillage; improving soil fertility; and implementing practices such as reforestation that have high potential for accumulating organic matter in vegetation which then enters the soil. For example, twelve years after afforestration of heavily grazed land, a Siberian larch (*Larix sibirica* Ledeb.) plantation was a sink for C with average annual net sequestration of 108 g C m^{-2} over a three year period (Bjarnadottir et al. 2007). Conversion of marginal land, which is of low agricultural value, to permanent vegetation would likely have only a minor economic impact. Riparian zones where agricultural production is difficult due to factors such as frequent flooding, high moisture contents or sloping topography could be considered as having low agricultural value, but if reforested would have a high potential for C sequestration in vegetation and soil.

1.2.3 Land-use Change in Ontario

An estimated 90% of Southern Ontario was forested before European settlement (Larson, 1999). With the arrival of European immigrants in the 18th and 19th centuries, large tracts of forested land were cleared for agricultural production and logging (Head, 1975). By 1920, approximately 90% of forests in Southern Ontario had been removed and replaced by human settlements and agriculture (Larson, 1999). Considering that temperate forests are one of the largest sinks of green house gases (Dalal and Allen, 2008), the removal of these forests in Ontario represents a significant loss of C from land to the atmosphere.

Presently many Southern Ontario regions have only 5 to 20 percent forest cover (Riley and Mohr, 1994) and a large proportion of forested riparian zones adjacent to freshwater ecosystems have been lost due to logging, agriculture and urban settlement. Barton et al. (1985) examined 38 streams in Southern Ontario for factors affecting trout population. They found that 35% of the streams had less than 20% forested riparian vegetation whereas only 15% had greater than 70% riparian forest. A report commissioned by the town of Milton in Southern Ontario stated that riparian vegetation was minimal or completely lacking along tributaries of a local creek due to agriculture and urban encroachment (Philips Engineering Limited, 2010). According to Naiman et al. (1993), more than 80% of riparian corridors in North America and Europe have been removed.

Restoring land to natural ecosystems has the potential of sequestering C and reducing atmospheric GHGs. Riparian zones, which may be difficult to cultivate because of their high moisture content or their uneven topography, provide an ideal landscape for restoration. In addition, forested riparian zones have been found to contain greater biomass of total plant material than nonforested riparian zones (Gregory et al., 1991) and therefore reforestation of riparian zones should be a first choice where possible tree planting opportunities exist.

1.3 Riparian Zones

A riparian zone is defined as land that borders water bodies such as rivers, lakes and oceans (Hall, 1988; Malanson, 1993). It is comprised of the stream bank from the low to high water marks and extends further into areas that are influenced by high water tables and flooding (Naiman et al., 1993). Due to the close proximity of water, the riparian zone tends to be significantly different from areas more distant from the river or lake (Malanson, 1993).

Riparian zones are among the most diverse and complex ecosystems on earth (Naiman et al., 1993). They encompass a wide range of landforms some of which are subjected to flooding events of varying intensity and frequency. Other riparian zones, lie in constrained reaches where flooding seldom occurs (Malanson, 1993). Riparian zones are also affected by upland runoff which may contain organic debris, sediments or nutrients. Soils in riparian zones range from continuously wet areas to drier well drained soils. In addition, soils may be affected by water erosion and alluvial deposition (Gregory et al., 1991). Riparian plant communities also exhibit diversity in their structure, composition and ecological processes. The variation in riparian topography, redistribution of sediment and high productivity potential of plant communities due to the proximity of water, collectively contribute towards creating a species rich environment (Naiman et al., 1993). For example, Gregory (1991) found that plant species richness along streams in Oregon and California, was significantly greater in the riparian zone than an upland area. Along the Adour River riparian corridor in France, a total of 900 species were found (Tabacchi et al., 1990).

1.3.1 Riparian Function

The riparian zone plays an important role in the aquatic ecosystem and helps to maintain water quality. Roots from riparian vegetation hold soil and sediments in place thereby preventing erosion of the river bank (Zaimes et al. , 2008). Shading from the trees maintains water at a temperature suitable for aquatic wildlife and the woody debris and leaf litter provide an energy source for aquatic organisms (Schultz et al., 2004). The riparian vegetation also acts as a buffer protecting the water system from runoff of adjacent land such as agricultural fields which may contain pollutants (Clausen et al., 2000; Hill, 1996). In addition, riparian corridors provide important habitat for wildlife (Pedroli et al., 2002; Ward, 1998).

Using several parameters to quantify soil erosion, Zaimes et al. (2008) determined that riparian forests stabilized the stream bank and reduced erosion compared to pasture and row cropping systems. A grass riparian zone was also effective in reducing stream bank erosion. Another study showed that while plant roots help to hold soil in place, above ground stems also contribute to reducing erosion during flooding events by slowing waters and retaining material transported in the water (Gregory et al., 1991). Beeson and Doyle (1995) investigated four meandering rivers in British Colombia following a flood event. They specifically looked at bends in the rivers and found that areas without riparian vegetation were five times more likely to experience bank erosion compared to vegetated bends.

Riparian zones serve as a filter for sediments and sediment-borne nutrients and pollutants carried in surface runoff (Lowrance et al., 1997). Shultz et al. (1995) found that a riparian buffer was effective in reducing the amount of nitrate and atrazine from a corn field entering the adjacent stream. Carolina poplar (*Populus x canadensis* Moench.) was effective in filtering nitrate from a saturated soil solution (O'Neill and Gordon, 1994). Cey et al. (1999) observed that nitrate rich surface water from an agricultural field flowing over the riparian zone was diverted downward, infiltrating the soil. Forested riparian zones and grass riparian zones also reduced the amount of nitrate in shallow ground water resulting in a lower amount entering the stream compared to row crops (Osborne and Kovacic, 1993). Waterborne inputs of nitrogen (N), Calcium (Ca), Phosphorus (P) and Magnesium (Mg) in shallow groundwater from agricultural fields were significantly greater before passing through a forested riparian zone (Lowrance et al., 1984). Peterjohn and Correll (1984) recorded decreases in nitrogen, particulate phosphorus and particulate organic carbon in runoff from a corn field. Their calculations showed that approximately 89% of N was retained by the riparian forest compared to 8% retention by the cornfield. The calculated P retention by the riparian forest zone was 80% compared to 41% for the cropland. Retention of N and P occurred for both surface runoff and ground water. In contrast, cutting of a forested riparian zone increased runoff by 41% and the output of dissolved inorganic materials increased 14 fold (Fisher and Likens, 1973).

Solar radiation is absorbed and reflected by vegetation, and therefore the riparian zone influences the amount of radiation reaching the water. The plant community in the riparian zone has a direct effect on the degree of shading of the water especially for smaller streams (Barton et al., 1985). A dense forest riparian zone may completely shade the stream. Without shade, solar radiation striking the water transforms into heat, changing the temperature of the water and often negatively affecting the aquatic life in the stream (Barton et al., 1985; Gregory et al., 1991; Schultz et al., 2004). For example, Kiffney et al. (2003) found that as the forested riparian zone width decreased, mean and maximum water temperature increased. They also noted that an increase in water temperature significantly changed the benthic communities, periphyton biomass and insect colonization in the stream.

Woody debris from a forested riparian zone creates debris dams which hold soil in place, trap other debris and sediments, and redirect water currents thereby reducing the energy and erosive potential of the water (Gregory et al., 1991; Naiman and Decamps, 1997). Wood debris as well as other organic matter from riparian zones also serve as allochthonous food sources and habitat for aquatic organisms (Fisher and Likens, 1973; Gregory et al., 1991). Ninety-nine percent of the annual energy input of Bear Brook, a second order stream in Northeastern United States, came from allochthonous sources. Of this, 34% was used by consumers in the brook with the remaining 66% throughput available for ecosystems downstream. Litter and throughfall from adjacent riparian forests accounted for 44% of this energy input (Fisher and Likens, 1973).

1.3.2 Riparian Restoration and Rehabilitation

Most of riparian vegetation has been removed where intensive agriculture is dominant (Schultz et al., 2004). In the United States, Hair (1988) estimates that up to 90% of riparian forests have been replaced by agricultural activities. Early “river improvement” efforts in Australia consisted of removal of tree trunks from channels and clearing the river banks of trees to improve the flow of water. In spite of erosion concerns and a government edit in 1803 prohibiting clearing of riparian vegetation, removal of riparian vegetation continued (Warner and Bird, 1988). A 2010 study in Southern Ontario reported that riparian vegetation was minimal or completely lacking along tributaries of a creek due to agriculture and urban encroachment (Philips Engineering Limited, 2010).

In light of the benefits that riparian zones provide, restoration or rehabilitation of degraded riparian zones will have a significant positive impact on the environment. The Society for Ecological Restoration, defines ecological restoration as “the process of assisting the recovery of an ecosystem that has been degraded, damaged or destroyed” (Clewell et al., 2004). Rehabilitation and restoration are similar in that they share a focus on pre-existing ecosystems as a reference with the goal of repairing the ecosystem’s productivity and service. Restoration differs in that it also includes the goal of re-establishing the species and community structure of the reference ecosystem (Clewell et al., 2004).

Historically, ecological restoration has focused on restoring the ecosystem to its pre-disturbance state. However, Choi (2004) states that if the goal is an ecosystem that will persist into the future, restoration should focus on recovering ecosystem functions rather than restoring the exact pre-disturbance community and structure, especially if climatic conditions have changed or the ecosystem is so degraded that it will no longer support pre-disturbance species. For many riparian zones there is little data on pre-impact conditions or human activity has severely altered the area making it difficult or impossible to replicate the original ecosystem (Clewell et al., 2004). Therefore a more practical approach to riparian restoration would be restoring ecosystem services such as C sequestration, erosion control, aggregate formation and stabilization, and retention of sediments, nutrients and other pollutants from runoff. This would result in improved water quality through the treatment, conversion or reduction in the amount of substances entering the aquatic ecosystem (Osborne and Kovacic, 1993).

The importance of the riparian zone and its functions has been widely recognized and therefore restoring a permanent vegetative community along stream banks is encouraged in many agricultural areas (Schultz et al., 2004) including Ontario (Ontario Soil and Crop Improvement Association, 2011). Schultz et al. (1995) suggested a combination of fast growing trees such as silver maple and popular, native bushes, and grasses which together function as a sink for nutrient and pesticide runoff from agricultural fields. The large root system of trees planted next to the stream bed is effective in stabilizing the stream bank, shrubs provide diversity in both above and below ground plant structure and the rapid turnover of roots and large amounts of organic matter from grasses help to redevelop aggregate structure and microbial activity thereby improving soil quality (Schultz et al., 2004). In Australia, indigenous species such as water gum (*Tristaniopsis laurina* Peter G.Wilson & J.T.Waterh) and crimson bottle brush (*Callistemon citrinus* (Curtis) Skeels) that can tolerant flood disturbance and sediment deposition are presently used for riparian rehabilitation. Imported European willows (*Salix spp*.) which were recommended for riparian rehabilitation for many years because of their fast establishment became invasive weeds clogging river channels and trapping sediment and other debris (Warner and Bird, 1988; Webb and Erskine, 2003).

Reforestation of riparian ecosystems has resulted in improvement of soil conditions. For example, soil organic matter in riparian zones along the San Antonio River, Texas, USA increased as natural succession occurred following abandonment of farmland. Soil organic matter peaked after 20 years in mid-successional plant communities dominated by leguminous trees communities (Bush, 2008). A multispecies riparian zone was established along Bear Creek in Iowa, USA, using hybrid poplar (*Populus X euramericana* 'Eugenei', red osier dogwood (*Cornus stolonifera* Michx.), ninebark (*Physocarpus opulifolius* L.) and a 7.3 m strip of switchgrass (*Panicumm virgatum* L.). Seven years after establishment, soil organic matter had increased by 8.5% (Marquez et al., 1998). Schultz et al. (2004) reviewed literature of the riparian rehabilitation research in Iowa, USA and concluded that the riparian vegetation positively influenced both terrestrial and aquatic ecosystems. They noted that a 9-metre wide woody riparian zone reduced sediment loss and removed N and P from surface runoff. Additionally, soil water infiltration rates increased by five times and stream bank erosion decreased by 80%. Soil microbial activity and soil microbial biomass increased in the riparian zone which led to enhanced soil denitrifaciton rates. Within the stream, they observed an increase in substrate availability and fish diversity. Similarly, restoration of a first-order stream in Connecticut decreased concentrations of N, P, and sediment in surface flow from an upland cornfield (Clausen et al., 2000).

Although larger streams such as the Sacramento River are often the focus of restoration or rehabilitation because of their visibility (Alpert et al., 1999), it is the smaller first and second order streams that make up the majority of channel length in a watershed (Moore and Richardson, 2003). Most of the water that enters water courses comes from these streams, and therefore improving water quality should focus on restoration or rehabilitation of smaller streams when resources are limited (Correll, 2005). Naiman et al. (1993) recommends that restoration should include the numerous head water streams as well as larger rivers downstream.

1.4 Research Objectives

Land use change has resulted in a significant reduction in the global SOC stock accompanied by emissions of CO_2 into the atmosphere. However, according to Lal (2004b), restoration of degraded soils such as marginal agricultural land or riparian zones can greatly enhance the sink capacity of these soils. Research has shown that well-established riparian zones provide numerous benefits (Zaimes et al, 2008; Shultz et al. 2004; Naiman & Decamps, 1997). However, it should not be assumed that similar benefits will result from all riparian restoration or rehabilitation projects. An estimated 15 billion dollars has been spent on river restoration projects in the United States (Bernhardt et al., 2005). Approximately 80% of these projects included riparian restoration or rehabilitation. However, only 10% of the projects reported any form of follow up monitoring or assessment (Bernhardt et al., 2005). In addition, research that has been conducted on rehabilitated riparian zones has generally focused on soil quality characteristics. Little work has been done to determine how soil CO_2 production rates are affected by rehabilitation of riparian zones.

Research is necessary to determine if rehabilitation of degraded riparian zones results in lower soil CO_2 production rates and increased C sequestration. Better understanding of soil CO_2 production rates and soil quality indicators will build on the present body of knowledge concerning GHG emissions. This knowledge will also guide restoration ecologists as they work towards creating riparian zones that will withstand and recover from natural and human disturbances. In addition, evidence of C sequestration and reduced soil CO_2 production rates in rehabilitated riparian zones will provide additional motivation to replant degraded riparian zones.

The proposed research will focus on assessing the impact of riparian rehabilitation on litter input, soil quality and soil CO_2 production rates.

Research question: What is the effect of rehabilitation of a riparian zone on litter input, soil quality and soil CO_2 production rates?

The objectives of the study therefore were:

1) Quantify and compare herbaceous and leaf litter input in a rehabilitated forest riparian zone, a grass-forb riparian zone, and a natural forest riparian zone.

2) Quantify and compare soil quality characteristics of soil organic carbon, total nitrogen, bulk density, water infiltration rate, and microbial community structure in a rehabilitated forest riparian zone, a grass-forb riparian zone, and a natural forest riparian zone.

3) Quantify and compare the spatial dynamics and the magnitude of soil CO_2 production rates in a rehabilitated forest riparian zone, a grass-forb riparian zone, and a natural forest riparian zone.

1.5 Hypothesis

1. There will be significant differences between the rehabilitated forest riparian zone and the natural forest riparian zone for characteristics of litter input, soil quality and CO_2 production rates.

H_0: The rehabilitated forest riparian zone will exhibit similar characteristics of litter input, soil quality and CO_2 production rates as the natural forest riparian zone.

2. The rehabilitated forest riparian zone will exhibit enhanced litter input, enhanced soil quality and lower CO_2 production rates compared to the grass-forb riparian zone.

H_0: There will be no difference in litter input, soil quality or soil CO_2 production rates between the rehabilitated forest riparian zone and the grass-forb riparian zone.

2. Methods and Materials

2.1 Research Site

This study was conducted along Washington Creek, a 9 km long, first order stream located in Oxford County, Ontario, Canada within the Grand River watershed (Figure 2.1). The climate of Oxford County is temperate with cold winters, hot humid summers and 204 frost free days annually. The mean annual temperature is 7.2 °C and the mean annual precipitation is 912 mm (Environment Canada, 2011).

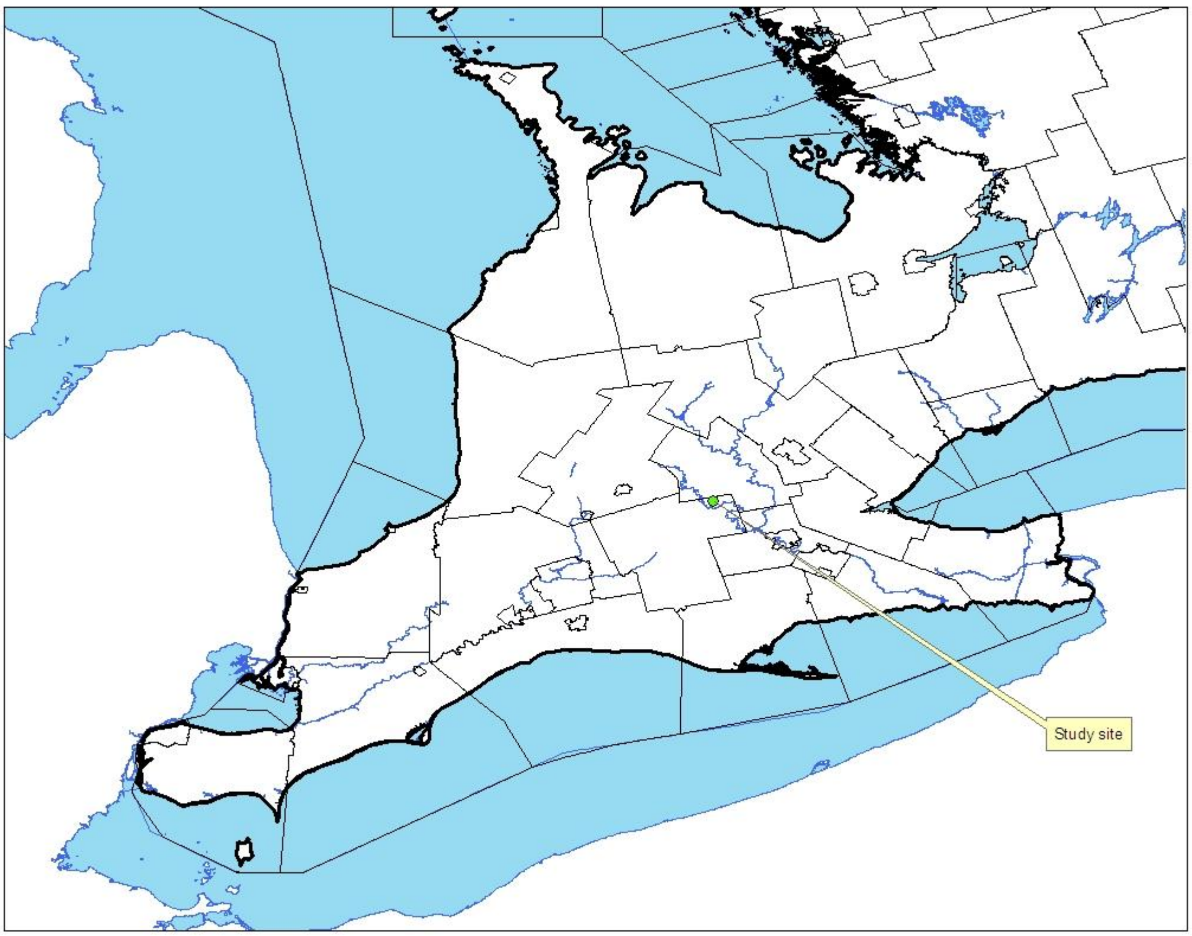

Figure 2.1. Map of Southern Ontario showing location of study site (DMTI Spatial, 2010).

Washington Creek enters the Nith River south of Plattsville, Ontario (43° 18' N 80°33' W) and is predominantly surrounded by agricultural land cropped to corn (*Zea mays* L.) and soybeans (*Glycine max* (L.) Merr.) or used for pasture. The area for this study was located north of Township Road 12 and east of Oxford Road 3 in Oxford County.

Soils close to the study site are grey-brown Podzolic and classified as Guelph loam, Brisbane sandy loam and Fox loamy sand (Figure 2.2). Soils adjacent to Washington Creek itself are classified as Bottomland.

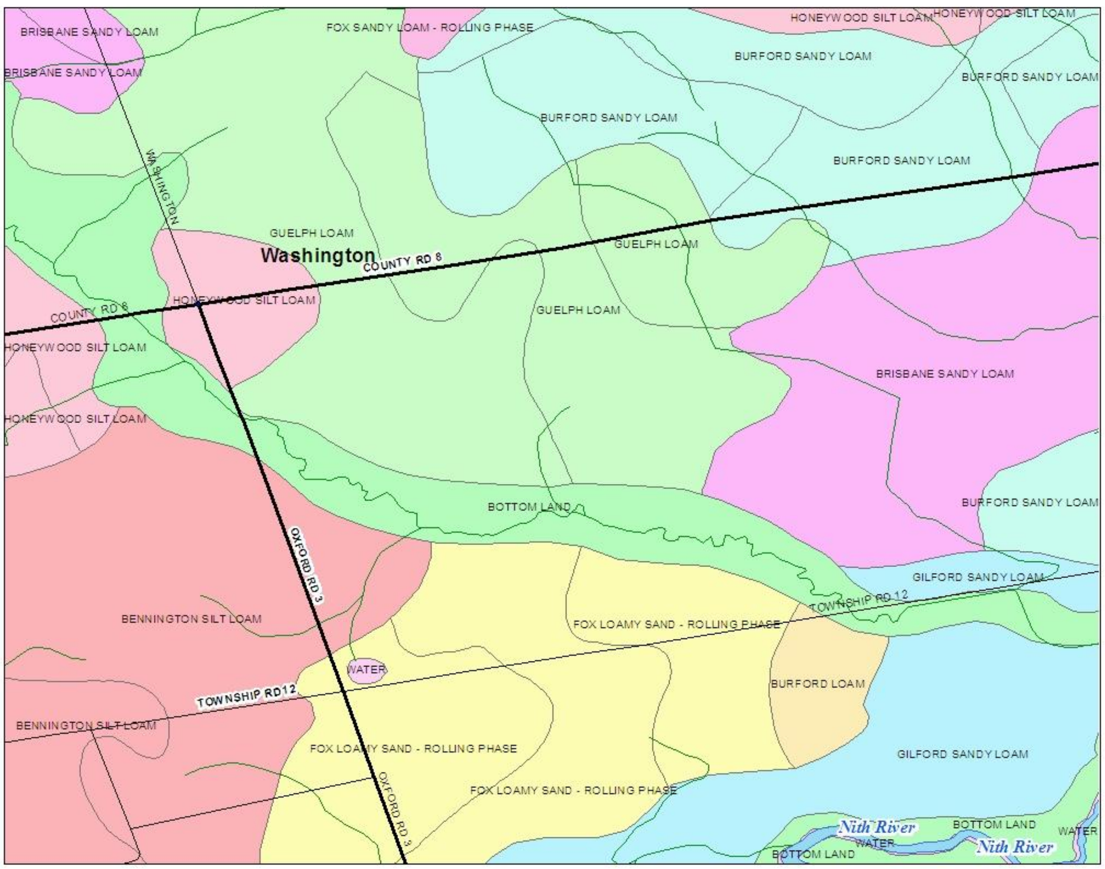

Figure 2.2. Soil classification along Washington Creek, Southern Ontario (OMAFRA, 2008, used with permission)

Bottomland soils consist of alluvial deposits of sands, silts and clays laid down in the creek bed and bank. The size of the area varies depending on the meander of the creek. The parent material of Guelph loam is a glacial till derived from soft brown limestone. They are well drained soils with a texture of loam or silt loam making them excellent for agriculture. The parent material of the Fox soils is calcareous sand. However, the amount of sand in the soils varies considerably. In the study area, the addition of loam till gives the surface soil a loamy sand texture. The Brisbane soils are imperfectly drained soils that developed from gravel deposits. The surface soil at the study site is a sandy loam (Wicklund and Richards, 1961).

In 1985, rehabilitation of a 1.6 km section of the creek was initiated. Prior to rehabilitation, land was cultivated to the edge of the creek and planted to corn or soybeans (Figure 2.3). With little vegetation to hold the soil in place, erosion and land degradation was a major problem (Gordon et al., 1996).

Figure 2.3. Washington Creek in 1985 prior to rehabilitation (Source: A. Gordon, 1985).

Rehabilitation took place over a 6-year period and included planting a riparian zone 30 to 50 m wide with trees and shrubs. Hybrid poplar clones (*Populus spp.)*, several alder species *(Alnus incana* (Du Roi) R.T. Clausen., *A. glutinosa* (L.) Gaertn. and *A. rubra* Bong) and silver maple *(Acer saccharinum* L.)

were planted in blocks or rows depending on land availability resulting in 3 x 3 m spacings. In 1986, chemical and mechanical weed control was conducted to aid in the establishment of newly planted trees. Further plantings of multifloral rose vines *(Rosa multiflora* Thunb.*)* silver maple, Russian olive *(Elaegnus angustifolia* L.), red osier dogwood *(Cornus stolonifera* L.) occurred in 1990 and 1991 (Table 2.1).

Table 2.1. Washington Creek Rehabilitation History from 1985 to 1991 (Adapted from Gordon et al., 1996).

Year	Activity
1985	Planting of poplar along stream in blocks and rows
1986	Planting along stream - poplar - alder - silver maple
	Chemical and mechanical weed control
1988	Addition of gravel to parts of stream
1989	Thinning of poplar trees
1990	Thinning of poplar trees
	Planting of multifloral rose vines and silver maple
1991	Planting of Russian olive, red osier dogwood and silver maple

Washington Creek provides an interesting research area because it allows for comparison of the effect of different land uses within a short distance. Different studies have been conducted at Washington Creek since rehabilitation in 1985 including evaluation of fauna and flora, solar radiation, water quality and nitrogen cycling (Gordon et al., 1996; Mallory, 1993; Oelbermann and Gordon, 2000; Oelbermann and Gordon, 2001; Oelbermann et al., 2008; Plascencia-Escalante, 2008).

2.2 Research Design

For this study, three sites along the creek representing riparian zones of different widths and vegetation were selected (Figure 2.4).

Figure 2.4. Aerial view of Washington Creek showing three riparian research sites (Source: Google Earth, January 2010).

Study sites:

1. **Rehabilitated riparian zone:** A rehabilitated buffer of 30 to 50 m planted to silver maple, poplar, alder and shrubs (Figure 2.5).
2. **Grass riparian zone**: Upstream from the rehabilitated area, a grass-forb buffer of 50 to 80 m separates the creek from a pasture. Up until five years ago, when a fence was constructed, cattle were allowed access to the creek in this area (Figure 2.6).

3. **Forest riparian zone**: A natural forest, located upstream from the grass riparian zone served as a reference site (Figure 2.7). The principle tree species are sugar maple (*Acer. saccharum* Marshall) and basswood (*Tilia sp*. L.).

Figure 2.5. Rehabilitated riparian zone along Washington Creek, Ontario, 2010.

Figure 2.6. Grass riparian zone along Washington Creek, Ontario, 2010.

Figure 2.7. Forest riparian zone along Washington Creek, Ontario, 2010.

Within each of the three riparian sites, eight sampling areas were selected, 20 metres apart, located along a transect two metres from the creek bank. Half of the sampling areas were on each side of the creek resulting in a nested block design with three treatments (riparian zones), two blocks (side of creek) and four reps within each block for a total of 24 sampling areas. In each sampling area, chambers for measurement of soil CO_2 production were placed in the soil, soil samples were taken and vegetation and leaf fall were removed. The chambers were placed along the transect and collection of soil samples, vegetation and litter was conducted along the transect.

Since it was not possible to find another first order stream in Southern Ontario with similar land uses, this was a pseudo-replicated study which limits the universality of the results.

2.3 Methods

2.3.1 Litter Collection and Analysis

At each of the 24 sampling areas (eight per riparian zone), one m^2 quadrats were placed within 2 m of the chamber so that half of the quadrat was on either side of the transect. Above ground herbaceous vegetation was cut as close to the ground as possible (approximately 1 cm of vegetation remained) on August 4 and 6, 2010. Vegetation was placed in bags and transported to the Soil Ecosystem Dynamics Laboratory, University of Waterloo, Waterloo, Ontario, Canada where it was dried at 60°C for 30 hours and weighed. A representative sample was stored in paper bags for further analysis.

Leaf fall from each of the eight sampling areas in the rehabilitated riparian zone and forest riparian zone was collected biweekly in September and October, 2010 in litter traps measuring 2 m^2 that were tied to trees or stakes and suspended 1.0 m off the ground. Leaves were transported to the Soil Ecosystem Dynamics Laboratory dried at 60°C for 30 hours, weighed and stored in paper bags until further analysis. Dried leaf weights of all dates were added together to determine total leaf litter. Total litter input was determined by adding the dried weight of the herbaceous vegetation and total leaf litter adjusted for area and recorded as g m^{-2} y^{-1}.

A sample of the dried herbaceous vegetation and leaf litter was ground using a Kinematica Polymix plant grinder (Px-MFC 90D, Switzerland) followed by grinding in a Retsch ball mill (Model MM200; Haan, Germany) resulting in a finely ground vegetation. Approximately 5 µg were placed in tin capsules for OC and total nitrogen (TN) analysis on a Costech Elemental Analyzer (Model 4010; Cernusco, Italy).

2.3.2 Soil Sampling and Analysis

Soil sampling was conducted on June 10 and 11, 2010 using a split tube sampler 40 cm in length with a cutting head inside diameter of 48 mm (Eijkeldamp, Netherlands). Two soil cores were taken at each of the 24 sampling areas. Soil cores were taken along the transect on each side of the chamber and approximately 1 m from the chamber.

The soil cores were separated into depths of 0-10 cm, 10-20 cm, 20-30 cm and 30-40 cm. For each depth, a 3-cm portion of the core was placed in a bag for determination of bulk density. For the remaining soil, the two cores of each sampling area were combined in a plastic bag, placed in a cooler and transported to the Soil Ecosystem Dynamics Laboratory. Approximately 10 g of the soil were refrigerated until January 18, 2011, when analysis for microbial community structure was performed. The remaining soil was air dried and passed through a 2 mm sieve and used for determining soil pH, particle size, soil TN and SOC.

2.3.2.1 Soil Physical Properties

Bulk density cores were weighed, dried at 105°C for 48 hours and then weighed again. Bulk density was determined by dividing the mass of the oven dried soil by the volume of the soil (Hao et al., 2008).

Soil particle analysis was determined for soil depths of 0-10 and 10-20 cm. Fifty grams of soil were treated with 125 ml of $NaPO_3$. This was stirred and left for 20 hours. The mixture was then mixed vigorously, put into a 1 litre graduated cylinder which was filled to 1 litre with deionised water and mixed again. A hydrometer was inserted in the cylinder and readings taken every hour for 8 hours. The diameter of particles in suspension was calculated with the equation:

$$D = K \sqrt{L/T} \tag{2.1}$$

where *D* is the diameter of particles in mm, *K* is a constant dependant on the temperature of the suspension (A.S.T.M., 1964) (K values ranged from 0.01286 at 25°C to 0.01365 at 20°C), *L* is the effective depth of the hydrometer (A.S.T.M., 1964) and *T* is the time interval in minutes from the beginning of sedimentation.

Percent clay was determined when the particle diameter in suspension was 0.002 mm with the equation:

$$P = Ra/W(100) \tag{2.2}$$

where *P* is the percentage of soil particles in suspension, *R* is the corrected hydrometer reading, *a* is a correction factor for soil specific gravity (in this case specific gravity was assumed to be 2.65 and therefore *a* = 1) and *W* is the oven dry weight of the soil.

The soil and water mixture was then passed though a #200 mesh screen leaving only sand particles (0.063 – 2mm) on the screen. The sand was dried and weighed. Silt was determined by subtracting the amount of clay and sand from the dried weight of the soil sample (A.S.T.M., 1964; Lambe, 1951). Results were recorded as percent clay, silt and sand. Clay percentage was entered into the calculations for determination of soil CO_2 production rates.

Soil hydraulic conductivity was measured at each of the 24 sampling areas in July, 2010 using a Guelph pressure infiltrometer (Figure 2.8). The infiltrometer foot ring was inserted 4 cm into the soil. The reservoir was then attached, the air tube pushed down to prevent water loss and the reservoir filled with water. The air tube was then lifted to 5 or 10 cm. The height of the tube was recorded as head 1 (H1). Water level on the reservoir was recorded at 1 minute intervals until the rate of water fall was constant . The air tube was then lifted another 10 cm (H2) and water levels recorded at 1 minute intervals until rate of water fall was constant. If the water moved rapidly out of the reservoir, water levels were recorded at 15 to 30 second intervals.

Figure 2.8. Determining hydraulic conductivity in the rehabilitated riparian zone with a Guelph pressure infiltrometer.

Hydraulic conductivity (K) was calculated using the two-head analysis and recorded as cm s^{-1} (Reynolds and Elrick, 2005).

$$K = T(q_2 - q_1)/(H_2 - H_1) \tag{2.3}$$

and $T = C_1 d + C_2 a$, (2.4)

where d is the depth of insertion of the foot ring (cm), a is the inside radius of the insertion ring (cm), and $C_1 = 0.316\pi$ and $C_2 = 0.184\pi$ are constants used when $d \geq 3$ cm and $H \geq 5$ cm (Reynolds, 1993; Youngs et al., 1993).

and $q_i = Q_i/\pi a^2$ (2.5)

where $Q_i = XR_i$, and X is the cross sectional area of the reservoir area (cm^2) and R_i is steady state rate of fall of water in the reservoir (cm s^{-1}) at H_i.

2.3.2.2 Soil Chemical Properties

Soil pH was determined using an Accumet AB 15 bench-top meter (Fisher Scientific, USA) which was first calibrated using standard pH solutions of 4, 7 and 10. Ten grams of soil and 20 ml of deionised water were put into a 30 ml beaker. The suspension was stirred intermittently for 30 minutes and then let stand for one hour. The electrode from the pH meter was immersed in the liquid and the reading recorded (Hendershot et al., 2008).

For all four soil depths, approximately 5 g of soil were finely ground (<250µm) using a Retsch ball mill (Model MM200; Haan, Germany). Carbonates were then removed by adding 60 ml of 0.5 M hydrochloric acid to 2 g of soil. The mixture was initially stirred for 10 minutes and stirred twice more during the next 24 hours. After 24 hours, the acid was removed using a 10 ml pipette and replaced with distilled water. This procedure was repeated every 24 hours for the next 4 days (Midwood and Boutton, 1998). Soil was dried at 60°C for 24 hours and ground again in a mortar and pestle. Approximately 5 µg of soil were place into tin capsules and analyzed for SOC and TN on a Costech Elemental Analyzer (Model 4010; Cernusco, Italy). The elemental analyzer was calibrated with three different weights of the standard atropine. SOC and TN were quantified by multiplying the mass (g kg^{-1}) by bulk density (g cm^{-3}) and depth (cm) and recorded as g m^{-2}.

2.3.2.3 Soil Biological Properties

Microbial community structure for soil depth of 0-10 cm was assessed using Biolog Ecoplates ™ (Biolog Inc., CA). Soil samples from the four sampling areas within each block were combined and a soil suspension for Ecoplate™ inoculation was prepared by adding 2 g of soil to a 10 ml solution of 0.85% sodium chloride. The solution was shaken vigorously for 15 minutes to dislodge the microbes from the soil and then serial diluted to 1:10,000. Using a multichannel millipipete, a 150 µl sample of the diluted solution was added to each well in the Ecoplate™ and incubated at 25°C for 7 days. At 24-hour intervals, color development, based on C substrate utilization, was recorded as optical density (OD) at 590 nm with a Biotek absorbance microplate reader (Model ELx800™, Vernmont, USA) (Garland and Mills, 1991).

Three variables were determined to characterize microbial community structure: average well colour development (AWCD), microbial community richness and the Shannon Index. For all calculations the OD on the seventh day was used and was first corrected by subtracting the value of the control well also on the seventh day. Respiration by the microbial community was determined by calculating average well colour development (AWCD) with the equation:

$$AWCD = \sum ODi/n \tag{2.6}$$

where ODi is the corrected OD of each individual well and n is the number of carbon sources (31 for Biolog Ecoplates ™). Community richness (R) was represented by the number of C sources utilized by the microbial community where the corrected OD was greater than 0.25. The Shannon Index (H), a measure of diversity and evenness, was calculated with the equation:

$$H = -\sum pi\ (ln\ pi) \tag{2.7}$$

where *pi* is the ratio of the corrected *ODi* to the sum of the total OD of all wells.

2.3.3 Soil CO_2 Sampling and Analysis

At each sampling area, a non-flow-through, non-steady-state vented chamber was placed in the soil to measure soil CO_2 production rates following recommendations in the literature (Livingston and Hutchinson, 1995; Rochette and Eriksen-Hamel, 2008; Venterea et al., 2009). The chamber consisted of two parts: a permanent anchor made of a PVC pipe, 30 cm in diameter and 30 cm in height of which 10 cm was driven into the soil, and a cap, also made PVC pipe. The cap was fitted with a septa (2 cm diameter) which was used for extraction of air and a 10 cm long vent tube (9 mm diameter) and covered with reflective insulation. Insulating foam was placed on the underside of the cap to ensure a tight fit on the anchor. The cap was placed on the anchor at time of sampling and left on for the duration of the sampling time (Figure 2.9). Between sampling days, chambers were left open to the air.

Chambers were placed in position April 26, 2010, one week before the first air sampling date and remained in place until sampling was completed in October. Vegetation within the chambers was cut at ground level and removed at time of chamber placement. Any vegetation within the chambers was cut prior to each air sampling.

Air samples were taken from the chambers every two weeks from May until October, 2010. Caps were placed on the chambers for 30 minutes and air samples were removed at time 0, 15 and 30 minutes by inserting a 5 ml air-tight syringe (Luer-Lok Tip. BD, Frankin Lakes, NJ, USA) into the chamber septa and removing a 5 ml air sample. The air sample was transferred into a 3 ml evacuated glass vial (Labco

Ltd., High Wycombe, UK) for transport to the Soil Ecosystem Dynamics Laboratory, University of Waterloo, for analysis. Time required for sampling of all chambers was five hours. To minimize diurnal differences, chambers were sampled between 10 am and 3 pm.

At the time of air sampling, soil moisture and temperature were measured beside the chambers at a depth of 5 cm using a HH2 Moisture Meter with a WET-2 sensor (Delta-T Devices, Cambridge, UK). Air temperature was also recorded at a height of 1 m above the ground.

Figure 2.9. Chamber for measuring soil CO_2 with cap on located in the rehabilitated riparian zone along Washington Creek, Ontario.

The CO_2 concentration of the air samples was determined using a gas chromatograph (GC) (Aglient 6890N, California, USA) equipped with a thermal conductivity detector. Samples were run in sequence (time 0, time 15, time 30 minutes) to minimize errors associated with GC drift. The GC was calibrated using standards with CO_2 concentrations of 100 ppm, 500 ppm, 1000 ppm and 50,000 ppm. Using the values obtained from these standards, a regression line was calculated and used to determine CO_2 ppm for air samples.

Venturea (2010) reviewed problems associated with using chambers to determine soil CO_2 production rates and concluded that they can result in errors due to the suppression of the gas concentration gradient at the soil surface. A method was therefore developed for calculating soil CO_2 production rates taking into account factors which could result in an underestimation of production rates. The method is based on the gas transport theory, but also takes into account soil properties of clay content, bulk density, soil water content, soil temperature and pH as well as chamber characteristics including the ratio of the chamber internal volume to surface area and deployment time. A calculation spreadsheet developed based on this method was used for calculating soil CO_2 production rates for this study. Results were recorded as mg C m^{-2} h^{-1}. All equations are described in detail by Venturea (2010).

2.3.4 Statistical Analysis

An analysis of variance (ANOVA) was used to determine significant differences between treatments. All data was first tested for homogeneity of variance with Levene's test and normal distribution using a Shapiro-Wilks test. Where assumptions were violated, a natural log transformation of the data was performed and data re-analyzed. If the F statistic was significant, Tukey's test was used to determine which treatment means were significantly different from each other. If ANOVA assumptions were still violated following the natural log transformation, data were analyzed using non-parametric tests of Kruskal Wallis, followed by Wilcoxon signed-rank test for pair comparisons if the Kruskal-Wallis test was significant.

A repeated measures ANOVA was run for CO_2 production rates, soil moisture and temperature since the data from the 13 sampling dates were obtained from the measurements of the same sampling area and therefore are not independent. For the repeated measures ANOVA, there is an additional assumption, the assumption of sphericity. This assumption was tested with Mauchly's test and when violated, corrections were applied to produce a valid F statistic.

Pearson's correlation assumes normal distribution. Where distributions were not normal, a natural log transformation resulted in normal distribution. Correlations were used to determine the relationship between soil quality indicators, litter input and soil CO_2 production rates. Correlations of soil CO_2 production rates with soil moisture, soil temperature and air temperature were run on individual dates and a one-sample t-test was performed on the mean correlation of all dates to determine significance. In addition, Principal Component Analysis, was used for microbial community analysis.

All statistical analysis was run using SPSS (SPSS Science Inc., v. 19.0, 2010).The threshold probability level for determining significant differences was $p<0.05$.

3. Results

3.1 Litter input

The grass riparian zone had significantly greater annual herbaceous input for total litter (580 g m^{-2} y^{-1}), litter C and litter N (260 and 11.5 g m^{-2} y^{-}1, respectively) than the rehabilitated riparian zone or forest riparian zone which did not differ significantly (Table 3.1). Carbon concentration (g kg^{-1}) of the herbaceous litter was significantly lower in the forest riparian zone, whereas N concentration (g kg^{-1}) was lowest for the grass riparian zone compared to the other two sites. The C/N ratio for the three riparian zones was highest for grass, followed by the forest riparian zone and the rehabilitated riparian zone.

Total leaf litter (g m^{-2} y^{-1}) and leaf litter C and N (g m^{-2} y^{-1}) were significantly greater in the rehabilitated riparian zone compared to the forest riparian zone (Table 3.1). However, C and N concentrations (g kg^{-1}) and the C/N ratio did not differ significantly between the two treatments.

Total above ground litter as well as litter C and N (g m^{-2} y^{-1}) was lowest for the forest riparian zone and did not differ significantly between the rehabilitated riparian zone and grass riparian zone. The overall C/N ratio did not differ significantly for the three treatments (Table 3.1).

Table 3.1. Annual herbaceous and leaf litter data for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. Standard errors are given in parentheses.

		Riparian Zone		
Litter type	**Parameter**	**Rehabilitated**	**Grass**	**Forest**
Herbaceous vegetation	**Total (g m^{-2} y^{-1})**	189 (22)^{A}	580 (71)^{B}	158 (38)^{A}
	C (g kg^{-1})	436 (5)^{A}	443 (7)^{A}	418 (2)^{B}
	N (g kg^{-1})	26.2 (1.6)^{A}	19.6 (1.1)^{B}	22.1 (1.1)^{A}
	Litter C (g m^{-2} y^{-1})	82 (9)^{A}	260 (34)^{B}	66 (16)^{A}
	Litter N (g m^{-2} y^{-1})	5.1 (0.8)^{A}	11.5 (1.8)^{B}	3.5 (0.8)^{A}
	C/N	17 (1)^{A}	23 (1)^{B}	19 (1)^{AB}
Leaf	**Total (g m^{-2} y^{-1})**	291 (25)^{A}		137 (20)^{B}
	C (g kg^{-1})	474 (3)^{A}		483 (5)^{A}
	N (g kg^{-1})	12.5 (0.6)^{A}		13.1 (0.3)^{A}
	Litter C (g m^{-2} y^{-1})	137 (11)^{A}		66 (10)^{B}
	Litter N (g m^{-2} y^{-1})	3.6 (0.3)^{A}		1.8 (0.3)^{B}
	C/N	38 (2)^{A}		37 (0.8)^{A}
Total litter (herbaceous + leaf)	**Total (g m^{-2} y^{-1})**	480 (25)^{A}	580 (71)^{A}	295 (38)^{B}
	Litter C (g m^{-2} y^{-1})	220 (11)^{A}	259 (35)^{A}	132 (17)^{B}
	Litter N (g m^{-2} y^{-1})	8.6 (0.8)^{A}	11.5 (1.8)^{A}	5.3 (0.8)^{B}
	C/N	26 (2)^{A}	23 (1)^{A}	26 (1)^{A}

Means followed by different uppercase letters comparing litter between riparian zones are significantly different ($p<0.05$) between treatments.

3.2 Soil properties

3.2.1 Soil Physical Characteristics

Bulk density (g cm^{-3}) was significantly higher in the rehabilitated riparian zone for all depths compared to the grass riparian zone and forest riparian zone (Table 3.2). Although there was a trend for increased bulk density with increasing depth with all treatments, the differences were not significant.

Soil particle analysis showed no significant differences in composition of percent sand, silt and clay between the three riparian zones (Table 3.2). According to the soil textural triangle, the composition indicated that the soil was a loam soil.

Differences among treatments for hydraulic conductivity (K) were not significant, although the lowest hydraulic conductivity which was recorded for the rehabilitated riparian zone corresponded with the highest bulk density (Table 3.2).

Table 3.2. Soil physical properties of bulk density, particle size and hydraulic conductivity (K) for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. Standard errors are given in parentheses.

		Riparian Zone		
Soil parameter	**Depth**	**Rehabilitated**	**Grass**	**Forest**
Bulk density	0-10	0.92 (0.02)^{Aa}	0.73 (0.05)^{Ba}	0.74 (0.04)^{Ba}
(g cm^{-3})	10-20	0.94 (0.05)^{Aa}	0.79 (0.05)^{ABa}	0.75 (0.51)^{Ba}
	20-30	1.04 (0.04)^{Aa}	0.83 (0.06)^{Ba}	0.72 (0.03)^{Ba}
	30-40	1.05 (0.04)^{Aa}	0.87 (0.05)^{Ba}	0.80 (0.05)^{Ba}
	Mean	**0.99 (0.02)^{A}**	**0.80 (0.04)^{B}**	**0.75 (0.04)^{B}**
Particle size	0-10 cm			
- **Sand (%)**		43.6 (8.5)^{A}	43.9 (6.4)^{A}	53.7 (2.8)^{A}
- **Silt (%)**		48.8 (7.4)^{A}	48.2 (5.9)^{A}	39.0 (2.3)^{A}
- **Clay (%)**		8.4 (1.5)^{A}	7.9 (0.5)^{A}	7.9 (0.6)^{A}
	10-20 cm			
- **Sand (%)**		46.0 (7.6)^{A}	53.1 (3.3)^{A}	46.2 (2.3)^{A}
- **Silt (%)**		47.1 (6.5)^{A}	39.2 (3.2)^{A}	45.3 (1.9)^{A}
- **Clay (%)**		6.9 (1.1)^{A}	7.9 (0.3)^{A}	8.5 (0.5)^{A}
K (cm s^{-1})		0.0066 (0.0024)^{A}	0.0087 (0.0029)^{A}	0.0084 (0.0020)^{A}

Soil property means followed by different uppercase letters are significantly different ($p<0.05$) between riparian zones, within depths.
Soil property means followed by different lowercase letters are significantly different ($p<0.05$) between depths, within riparian zones

3.2.2 Soil Chemical Characteristics

The soil pH did not differ significantly between treatments at soil depths of 0-10 cm and 10-20 cm (Table 3.3). For the 20-30 cm and 30-40 cm depths and the mean of the four depths, the pH for the forest riparian zone was lower than the rehabilitated riparian zone or grass riparian zone, although differences between the forest riparian zone and grass riparian zone were only significant for the 20-30 cm depth.

The SOC and TN concentrations (g kg^{-1}) were significantly lower in the rehabilitated riparian zone compared to the grass riparian zone or forest riparian zone which did not differ significantly (Table 3.3). Results for SOC and TN stock (g m^{-2}) showed similar patterns as the concentration. Stock values were significantly lower in the rehabilitated riparian zone for most depths and the overall mean compared to the grass riparian zone and forest riparian zone. The soil C/N ratio was significantly higher in the forest riparian zone compared to the rehabilitated riparian zone for most depths and the mean C/N ratio. The soil C/N for the grass riparian zone was between the two other riparian zones.

For all three riparian zones, SOC and TN concentrations tended to decrease with decreasing depth, although differences were significant only for the rehabilitated riparian zone (Table 3.3). Similarly, both SOC and TN stock decreased with increasing depths for the rehabilitated riparian zone. The C/N ratio increased with decreasing depth in the grass riparian zone and forest riparian zone, but differences among depths were not significant in the rehabilitated riparian zone.

Table 3.3. Chemical soil properties at 4 depths for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. Standard errors are given in parentheses.

		Riparian Zone		
Soil property	**Depth**	**Rehabilitated**	**Grass**	**Forest**
pH	0-10	7.84 (.09)^{Aa}	7.67 (.09)^{Aa}	7.74 (.18)^{Aa}
	10-20	8.01 (.08)^{Aa}	7.56 (.18)^{Aa}	7.56 (.26)^{Aa}
	20-30	8.03 (.09)^{Aa}	7.80 (.13)^{Aa}	6.99 (.38)^{Ba}
	30-40	8.15 (.14)^{Aa}	7.38 (.23)^{ABa}	6.83 (.37)^{Ba}
	Mean	**8.01 (.04)^{A}**	**7.60 (.12)^{AB}**	**7.28 (.22)^{B}**
Soil organic C (SOC) (g kg^{-1})	0-10	55.0 (4.2)^{Aa}	89.1 (8.3)^{Ba}	89.4 (7.3)^{Ba}
	10-20	51.0 (5.1)^{Aab}	78.0 (12.0)^{Aa}	80.8 (9.9)^{Aa}
	20-30	37.2 (3.4)^{Abc}	85.0 (19.3)^{Ba}	79.4 (8.1) ^{Ba}
	30-40	27.4 (3.0)^{Ac}	70.7 (14.0)^{Ba}	76.6 (16.0)^{Ba}
	Mean	**42.6 (3.0)^{A}**	**80.7 (12.4)^{B}**	**81.5 (9.2)^{B}**
Soil total N (TN) (g kg^{-1})	0-10	4.3 (0.3)^{Aa}	7.4 (0.8)^{Ba}	6.6 (0.5)^{Ba}
	10-20	4.2 (0.3)^{Aa}	6.2 (1.0)^{Aa}	5.6 (0.6)^{Aa}
	20-30	3.0 (0.2)^{Ab}	6.0 (1.3)^{Ba}	5.0 (0.5)^{Ba}
	30-40	2.3 (0.2)^{Ab}	4.8 (0.8)^{Ba}	4.7 (0.9)^{Ba}
	Mean	**3.5 (0.2)^{A}**	**6.1 (0.9)^{B}**	**5.5 (0.5)^{AB}**
Soil C/N	0-10	13.7 (0.4)^{ABa}	12.1 (0.4)^{Ba}	13.6 (0.4)^{Aa}
	10-20	12.0 (0.4)^{Aa}	12.6 (0.3)^{Aab}	14.2 (0.5)^{Bab}
	20-30	12.3 (0.6)^{Aa}	14.0 (0.5)^{ABb}	15.9 (0.7)^{Bb}
	30-40	11.7 (0.6)^{Aa}	14.2 (0.6)^{ABb}	16.0 (0.8)^{Bb}
	Mean	**12.2 (0.4)^{A}**	**13.2 (0.3)^{A}**	**14.9 (0.5)^{B}**
SOC stock (g m^{-2})	0-10	5035 (373)^{Aa}	6268 (412)^{ABa}	6495 (400)^{Ba}
	10-20	4833 (516)^{Aa}	5772 (491)^{Aa}	5874 (692)^{Aa}
	20-30	3860 (273)^{Aab}	6430 (911)^{Ba}	5561 (448)^{ABa}
	30-40	2788 (243)^{Ab}	5932 (1050)^{Ba}	5840 (1076)^{Ba}
	Mean	**4129 (243)^{A}**	**6100 (614)^{B}**	**5942 (531)^{B}**
Soil TN stock (g m^{-2})	0-10	398(30)^{Aa}	520 (33)^{Ba}	478 (27)^{ABa}
	10-20	403(43)^{Aa}	461 (41)^{Aa}	413 (48)^{Aa}
	20-30	315(15)^{Aab}	460 (61)^{Ba}	350 (26)^{ABa}
	30-40	237(17)^{Ab}	407 (58)^{Ba}	360 (59)^{ABa}
	Mean	**338(21)^{A}**	**462 (41)^{B}**	**400 (32)^{AB}**

Soil property means followed by different uppercase letters are significantly different ($p<0.05$) between riparian zones, within depths.
Soil property means followed by different lowercase letters are significantly different ($p<0.05$) between depths, within riparian zones.

3.2.3 Soil Biological Characteristics

Results from Biolog Ecoplates™ showed no significant differences between treatments for microbial community structure based on C substrate utilization. Values for average well colour development, richness and Shannon Index did not differ significantly (Table 3.4). Principal component analysis (PCA) resulted in 11 components with Eigen values of greater than one which explained 89% of the variation among ecoplate results. However, there were no significant differences between treatments for these 11components. When the PCA was run using Eigen values of greater than two in order to reduce the number of components, six components explained 65% of the variation, but again there were no significant differences between treatments for these components.

Table 3.4. Soil biological properties: Microbial community structure as measured by average well colour development (AWCD), Richness and Shannon Index based on carbon substrate use in Biolog Ecoplate™ for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. Standard errors are given in parentheses.

	Riparian Zone		
Parameter	**Rehabilitated**	**Grass**	**Forest**
AWCD	0.20 (0.03)^{A}	0.22 (0.05)^{A}	0.26 (.06)^{A}
Richness	6.0 (0.6)^{A}	6.7 (1.3)^{A}	8.0 (1.9)^{A}
Shannon Index	2.00 (0.22)^{A}	2.23 (0.11)^{A}	2.33 (0.12)^{A}

Microbial community structure means followed by different uppercase letters are significantly different ($p<0.05$) between riparian zones.

3.2.4 Correlations: Soil and Litter Input

Soil bulk density and pH were negatively correlated with SOC and TN concentrations, the C/N ratio and SOC and TN stock. Correlation coefficients between litter input and amount of C and N in the soil ranged from 0.21 and 0.29, but were not significant. In addition, there was a significant relationship between C and N in the soil (Table 3.5).

Table 3.5. Pearson correlation matrix of soil characteristics and litter input (n=24) for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. All characteristics excluding hydraulic conductivity (K) and litter are an average of 0 - 40 cm depths.

	Bulk density	pH	K	SOC	Soil TN	Soil C/N	SOC Stock	TN Stock
Bulk density	1.00							
pH	0.81**							
K	-0.07	-0.13						
Soil organic C (SOC)	-0.86**	-0.81**	0.11					
Soil total N (TN)	-0.79**	-0.70**	0.13	0.97**				
Soil C/N	-0.73**	-0.72**	-0.07	0.59**	0.39			
SOC stock	-0.76**	-0.75*	0.12	0.97**	0.96**	0.52**		
TN stock	-0.57**	-0.54**	0.17	0.86**	0.94**	0.17	0.93**	
Total litter	0.07	0.18	-0.19	0.18	0.27	-0.18	0.19	0.29
Litter C	0.07	0.16	-0.18	0.21	0.30	-0.18	0.23	0.33
Litter N	0.09	0.15	-0.01	0.20	0.29	-0.19	0.22	0.33
Litter C/N	-0.16	-0.35	-0.11	0.09	0.04	0.19	0.09	0.00

Values followed by * and ** are significantly correlated at $p < 0.05$ and $p < 0.01$ respectively.

3.3 Soil CO_2 Production Rates and Climatic Factors

3.3.1 Soil CO_2 Production Rates

Statistical analysis of soil CO_2 production rates showed a sampling date by treatment interaction. Since several different patterns emerged when plotting soil CO_2 production rate over sampling dates (Figure 3.1), results were re-analyzed based on these patterns resulting in non-significant sampling date by treatment interactions.

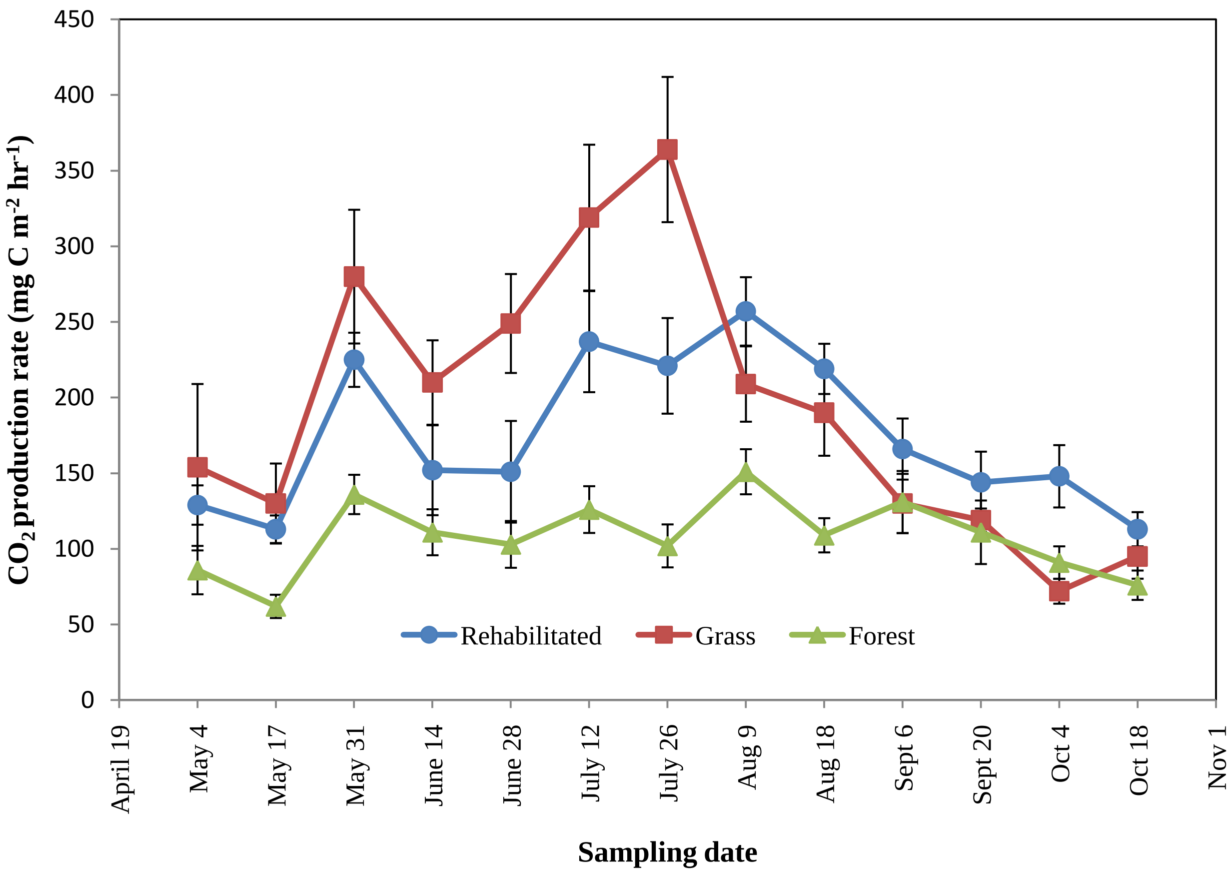

Figure 3.1. Soil CO_2 production rates from May 4 to October 18, 2010 for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. Vertical bars represent standard errors.

From May 4 until July 12, CO_2 production rates for the forest soil were significantly lower than both the rehabilitated riparian zone and grass riparian zone (Table 3.6). Although CO_2 production rates during this time period were consistently higher for the grass riparian zone compared to the rehabilitated riparian zone, differences were not significant. On July 26, production rates peaked for the grass riparian zone resulting in significantly higher rates than both the forest riparian zone and rehabilitated riparian zone. Soil CO_2 production rates decreased sharply for the grass riparian zone from Aug. 9 onwards and were lower than the rehabilitated site for the remainder of the sampling period. For the last four sampling dates (Sept. 6 to Oct. 18), differences were not significant between the three riparian zones (Table 3.6). Seasonal fluctuations over the sampling period were smallest in the forest riparian zone (Figure 3.1).

Table 3.6. Soil CO_2 production rates (mg C m^{-2} h^{-1}) averaged over time periods for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. Standard errors are given in parentheses.

	Riparian Zone		
Dates	**Rehabilitated**	**Grass**	**Forest**
May 4 – July 12	167 (11.7)^{A}	224 (18.4)^{A}	104 (6.5)^{B}
July 26	221 (31.6)^{A}	364 (48.0)^{C}	102 (14.2)^{B}
Aug. 9, 18	238 (14.4)^{A}	199 (18.4)^{A}	130 (10.6)^{B}
Sept. 6 - Oct. 18	140 (9.30)^{A}	104 (7.6)^{A}	133 (8.6)^{A}
All dates	175 (14.8)^{A}	194 (15.5)^{A}	107 (21.1)^{B}

Means followed by different uppercase letters are significantly different ($p<0.05$) between riparian zones within time period.

3.3.2 Soil Moisture

Statistical analysis of soil moisture showed a sampling date by treatment interaction. However, the interaction appeared to be primarily due to the degree of difference between treatments since the relationship between the three treatments was the same most of the 13 sampling dates. The forest riparian zone had the highest soil moisture at each sampling date (Figure 3.2). The rehabilitated riparian zone was significantly lower than the forest riparian zone for all but one sampling date, but did not differ significantly from the grass riparian zone. Averaged over all sampling dates, volumetric soil moisture was 28%, 33% and 42 % for the rehabilitated riparian zone, the grass riparian zone and the natural forest riparian zone, respectively. Seasonal fluctuations in soil moisture were more extreme for the rehabilitated riparian zone and grass riparian zone (17% and 12%, respectively) compared to the forest riparian zone (7%).

Rainfall during the sampling period was fairly regular with the exception of a dry period from May 14 to 30 and from July 26 to August 8. This resulted in lower soil moisture values for the May 31 and August 9 sampling dates.

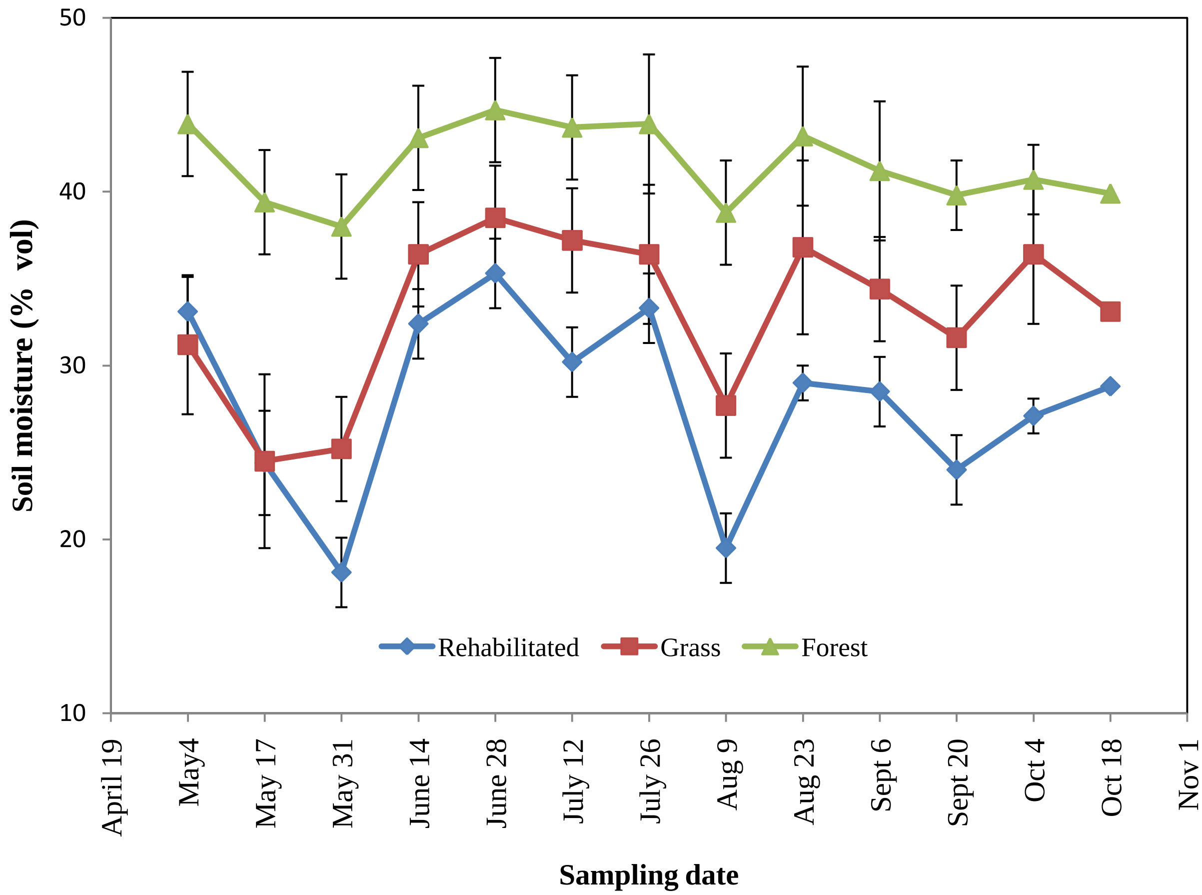

Figure 3.2. Soil moisture from May 4 to October 18, 2010 for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. Vertical bars represent standard errors.

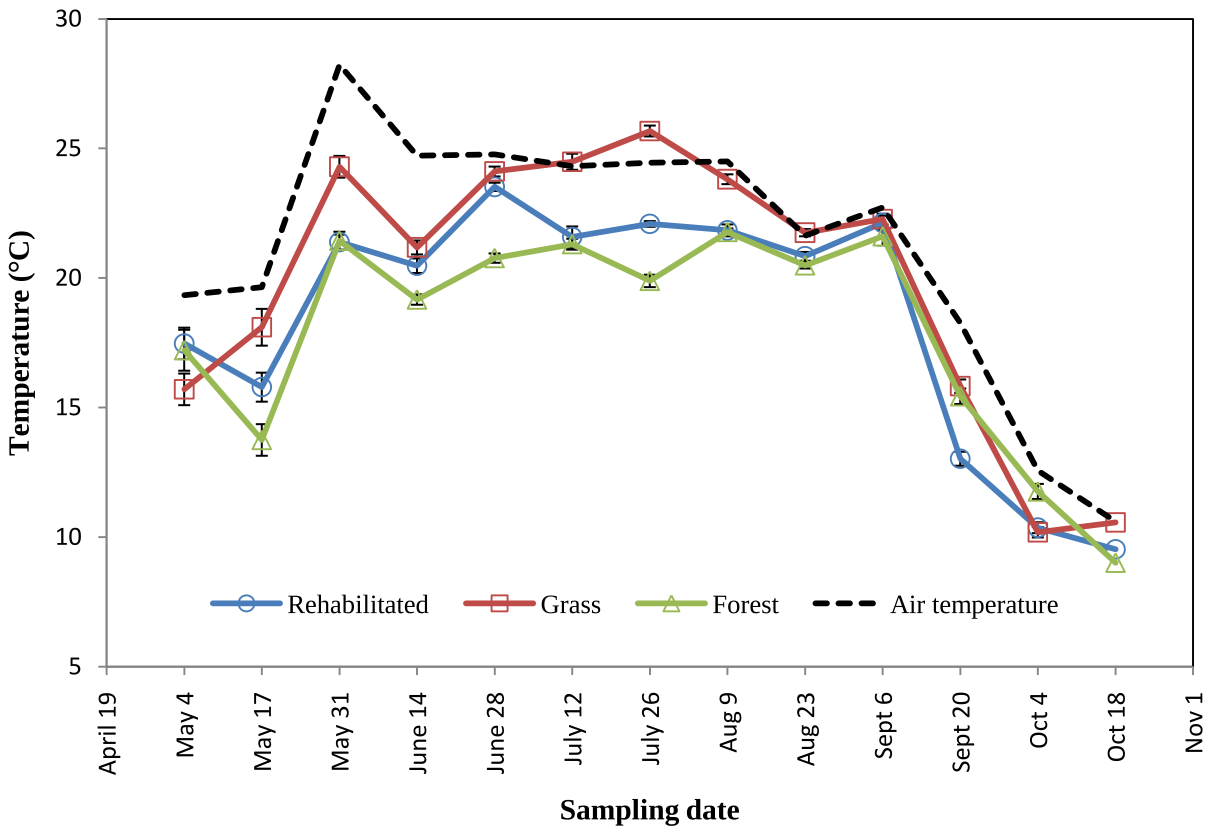

Figure 3.3. Soil temperature and mean air temperature during sampling from May 4 to October 18, 2010 for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. Vertical bars represent standard errors.

3.3.3 Soil Temperature

Statistical analysis of soil temperature showed a sampling date by treatment interaction. Soil temperatures ranged from a high of almost 26 °C in July in the grass riparian zone to a low of 9 °C in late fall in the forest riparian zone. For most dates from the time period of May 17 to August 23, the soil temperature in the grass riparian zone was significantly higher than the rehabilitated riparian zone and forest riparian zone. The grass riparian zone also exhibited the greatest fluctuation in temperature during this time period with a difference of 4.6 °C between the highest and lowest soil temperatures compared to 3 and 2.7 °C for the rehabilitated riparian zone and forest riparian zone, respectively (Figure 3.3). Patterns for soil and air temperature were similar, indicating that soil temperature was strongly influenced by air temperature (Figure 3.3). In most cases, air temperature was higher than soil temperature.

Correlations

3.3.4 Soil CO_2 Production Rates with Soil Properties and Litter

Mean soil CO_2 production rates for periods of May 4 to July12, August 9 to 18, and September 6 to October 18 time periods as well as July 28 were correlated with soil properties and litter input. Soil pH was significantly correlated with soil CO_2 production rate for late summer and fall dates. Soil C/N was negatively correlated with soil CO_2 production rate for all dates (significant at $p<0.10$ for July 28 and Sept. 6 to Oct. 18) explaining from 37 to 46% of the variation (Table 3.7).

Total litter, litter C and litter N were significantly and positively correlated with soil CO_2 production rate for the spring and summer sampling dates (Table 3.7).

Table 3.7. Pearson's coefficients of soil CO_2 production rates (mean for each time period) correlated with soil properties and total litter for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada. All data except hydraulic conductivity and litter are averages of 0 – 40 cm depths.

	Sampling dates			
Parameter	**May 4 – July 12**	**July 28**	**Aug. 9, 18**	**Sept. 6 – Oct. 18**
Bulk density	0.09	0.07	0.34	0.20
Hydraulic conductivity	-0.12	-0.10	-0.12	0.01
pH	0.40	0.33	0.59**	0.49*
Soil organic C (SOC)	-0.02	-0.02	-0.36	-0.26
Soil total N (TN)	0.12	0.10	-0.28	-0.19
Soil C/N	-0.42*	-0.37	-0.46*	-0.38
SOC Stock	0.01	-0.01	-0.36	-0.27
TN Stock	0.21	0.17	-0.22	-0.14
Total litter	0.48*	0.52*	0.54**	-0.03
Litter C	0.48*	0.51*	0.55**	-0.02
Litter N	0.44*	0.48*	0.46*	-0.07
Litter C/N	-0.07	-0.10	0.00	0.16

Values followed by * and ** are significantly correlated at $p < 0.05$ and $p < 0.01$ respectively.
Natural log transformation performed on all data to obtain normal distribution.

3.3.5 Soil CO_2 Production Rates and Climatic Factors

Correlations of soil CO_2 production rates and climatic factors differed depending on the riparian zone. Although the correlation with soil moisture was negative for all three riparian zones, it was significant only for the rehabilitated riparian zone and forest riparian zone. Soil temperature however, was significant for all three riparian zones (Table 3.8).

Table 3.8. Pearson's coefficients of soil CO_2 production correlated with soil moisture, soil temperature and air temperature for a rehabilitated, grass and forest riparian zone along Washington Creek in Southern Ontario, Canada.

	Riparian Zone			
Parameter	**Rehabilitated**	**Grass**	**Forest**	**All treatments**
Soil moisture	-0.33*	-0.27	-0.53**	-0.48**
Soil temperature	0.31**	0.24*	0.42**	0.32**
Air temperature	-0.18	0.12	0.15	0.27**

Values followed by * and ** are significantly correlated at $p < 0.05$ and $p < 0.01$ respectively.
Natural log transformation performed on all data to obtain normal distribution.

4. Discussion

4.1 Litter Input

Herbaceous litter contributions of 11.5 g N m^{-2} y^{-1} and 259 g C m^{-2} y^{-1} from the grass riparian zone at Washington Creek correspond well with organic matter input of 9 to 10 g N m^{-2} y^{-1} and 250 g C m^{-2} y^{-1} found on a cool season grass riparian zone in Iowa (Tufekcioglu et al., 2003). The understory of a hybrid poplar (*Populus* × *euroamericana* ' Eugenei') rehabilitated riparian zone in Iowa was substantially higher than the rehabilitated site at Washington creek. However, the Iowa study was conducted only seven years after establishment of the riparian zone (compared to 25 years since establishment at Washington Creek) and the amount understory would likely decrease as the poplars matured due to increased shading.

At Washington creek, leaf litter at the rehabilitated site contributed 291 g m^{-2} total biomass during the autumn season which falls within the range of values reported elsewhere for mature temperate riparian forests (225 to 440 g leaf litter m^{-2} yr^{-1}) (Bell et al., 1978; Gosz et al., 1972; Hughes, 1971; Lang and Forman, 1978) and was higher than the values of a 7-year-old rehabilitated popular riparian zone in Iowa (Tufekcioglu et al., 2003). In addition, leaf litter at the rehabilitated site at Washington Creek was substantially higher than amounts recorded in an earlier study at the same site (141 and 152 g m^{-2} yr^{-1} in 1996 and 1997, respectively) (Oelbermann and Gordon, 2000). The increase in leaf litter in 2010 provides evidence of continued maturation of trees in the rehabilitated site.

The lower amount of leaf litter in the forest riparian zone compared to the rehabilitated riparian zone was unexpected. Leaf litter of a mature riparian forest along Laurel Creek, a second order stream located 30 km from Washington Creek with similar tree species, including sugar maple (*Acer saccharum*), bitternut hickory (*Carya Cordiformis* (Wangenh.) K.Koch), American basswood (*Tilia Americana*), American beech (*Fagus grandifolia* Ehrh.) and eastern white cedar (*Thuja occidentalis* L.), was 318 g m^{-2} y^{-1} (Oelbermann and Gordon, 2000); substantially higher than the value of 137 g m^{-2} y^{-1} obtained at the forest riparian zone at Washington Creek. At some sampling sites at Washington Creek, the close proximity of cedars trees could have reduced the amount of litter collected. However, leaf litter excluding sites with cedar present was 190 g m^{-2} y^{-1} which is still lower than most other reported values for a mature temperate riparian forests which range from 225 to 440 g m^{-2} yr^{-1} (Bell et al., 1978; Gosz et al., 1972; Hughes, 1971; Lang and Forman, 1978). An in-depth study of leaf litter including more sampling sites and determination of the type of trees contributing to the leaf litter may help to explain the low values found in this research.

When leaf litter and understory were added together to determine total litter input, the grass riparian zone and rehabilitated riparian zone did not differ significantly. However, total litter input for the forest riparian zone was lower than the rehabilitated riparian zone mainly due to the lower amount of leaf fall, although herbaceous litter input was also lower.

4.2 Soil Properties

4.2.1 Soil Physical Properties

At Washington Creek, bulk density of the rehabilitated riparian zone was 0.24 g cm^{-3} higher than the forest riparian zone. It appears that the impact of heavy machinery used for yearly cultivation, planting and spraying of the row crops had negative long term effects on soil bulk density. To the knowledge of this author, there have been no reported studies in a temperate region that have compared bulk density of a rehabilitated forest riparian zone with a natural forest riparian zone. However, in a boreal forest, bulk density was higher for an establishing forest 25 years after clear cutting where heavy tree harvesting machinery would have compacted the soil, compared to a mature forest (125 years after cutting) and an old growth forest (250 years old) (Schwendenmann, 2000).

At Washington Creek, bulk density of the rehabilitated riparian zone was 0.19 g cm^{-3} higher than the grass riparian zone. The rehabilitated area was planted to row crops previous to rehabilitation whereas the grass site was pasture. Even though trampling by cattle would have had some impact on soil bulk density, the presence of permanent vegetation may have reduced the impact. In addition, it is likely that the high number of fibrous roots in the grass riparian zone loosened the soil resulting in lower bulk density values in a shorter time period compared to the rehabilitated riparian zone. Similarly, Bharati et al. (2002) found higher bulk density under a rehabilitated silver maple and black walnut (*Juglans nigra* L.) riparian zone compared to the grass riparian zone. They also found that six years after removal of cattle, the grass riparian zone had a lower bulk density compared to a grazed pasture and cultivated fields (Bharati et al., 2002). The authors suggested that significant reductions of bulk density can occur within six years of removal of cattle and establishment of perennial vegetation. At Washington Creek, a fence was constructed five years previous to this study and the undisturbed growth of grasses and forbs since that time likely resulted in a significant reduction in bulk density.

Hydraulic conductivity, which is a measure of the rate of water infiltration into the soil, may be affected by soil texture, vegetation, activity of micro-organisms or compaction zone and therefore may be highly variable within an ecosystem (Radke and Berry, 1993). At Washington Creek, the coefficient of variation for hydraulic conductivity was 100%, 95% and 66% for the rehabilitated riparian zone, grass riparian zone and forest riparian zone, respectively which may explain the lack of significant differences between treatments. However, the hydraulic conductivity was 23% lower at the rehabilitated site compared to the grass riparian zone and forest riparian zone. In contrast, Bharati et al. (2002) found that a silver maple riparian zone had a higher infiltration rate than a grass riparian zone. They attributed the higher infiltration to macropores which they associated with an observation of a large number of earthworm pores. They also suggested that infiltration rate was an indication of soil quality. The lower infiltration rate and higher bulk density found in the rehabilitated riparian zone at Washington Creek indicated that the negative effects of row cropping on soil quality parameters of bulk density and hydraulic conductivity may require more time for recovery compared to a pasture riparian zone.

4.2.2 Soil Chemical Properties

Differences of pH at Washington Creek were only significant for the 20-30 cm, 30-40 cm depths and the mean of the four depths with the forest riparian zone having lowest values. Other studies, although not in riparian zones, have noted a decline in pH over time in temperate hardwood forests (Drohan and Sharpe, 1997; Knoepp and Swank, 1994; Miller and Watmough, 2009) and boreal forests (Brais et al., 1995; Schwendenmann, 2000). For example, Drohan and Sharpe (1997) reported decreases in pH in the A horizon of 11 different hardwood forests of -0.1 and -0.23 after 14 and 36 years, respectively. They suggested that the decline in pH was due to nutrient uptake by vegetation as the forest matures and the leaching of base cations. In addition to nutrient uptake, Miller and Watmough (2009) suggested that leaching losses due to anthrogenic inputs of SO_4 and NO_3 may have caused acidification of forest soils in Southern Ontario. Schwendenmann et al. (2000) recorded a pH of 7.31 for an establishing boreal forest compared to 6.38 for a mature forest. They suggested that soil weathering and the accumulation of organic matter causes carbonates to dissolve resulting in a decrease in base saturation.

At Washington Creek, the mean SOC and soil TN concentrations for the four soil depths were 39 and 2.0 g kg^{-1} lower, respectively, for the rehabilitated riparian zone compared to the forest riparian zone. Other studies comparing younger forests and mature stands in various climatic conditions report similar results. In a boreal forest, Schwendenmann et al. (2000) recorded SOC and soil TN concentrations of 13 g kg^{-1} and 0.8 g kg^{-1}, respectively, in the top 20 cm of 25-year-old forests compared to 40 g kg^{-1} and 2.6 g kg^{-1} in mature forests. In a temperature region, Bush (2008) found increases in SOC and TN with increasing years of natural succession following cultivation. He reported SOC and TN values of 40.5 and 3.2 g kg^{-1}, respectively, in a 25-year-old forest community compared to 62 and 5.8 g kg^{-1} in 45-year-old communities. This data indicated that maximum C was reached at 49 years and then decreased likely due to C sequestration in vegetation. Bush (2008) also predicted that soil N would reach a maximum at 82 years and then decline due to N sequestration in the vegetation. Using a model based on data obtained from numerous sources and ecosystems, Houghton et al. (1983) predicted that it would take 40 years for a temperate forest to reach its potential SOC level which would be slightly below the SOC content of the original forest. Considering research in the literature and the higher level of litter input at the rehabilitated riparian zone along Washington Creek compared to earlier sampling dates, it is probable that the rehabilitated riparian zone will continue to accumulate SOC and soil TN, but as suggested by Houghton et al (1983), it may not reach the same levels as the natural forest.

Soil OC and TN concentrations were 38.1 and 2.6 g kg^{-1} lower for the rehabilitated riparian zone compared to the grass riparian zone at Washington Creek. This is additional evidence that cultivation of row crops before rehabilitation may have caused greater soil degradation than the riparian zone under pasture. In addition, a significant period of time would have been required for the establishment and maturation of trees planted in the rehabilitated riparian zone before they would recycle substantial amounts of nutrients deeper into the soil. For example, Vesterdal et al. (2002) found a 10 year lag period before development of forest floor following afforestration of arable land and suggested that there would also be a lag period for SOC accumulation. However, the permanent vegetation in the grass riparian zone along Washington Creek would have continued to accumulate OC at depths measured in this study even during cattle grazing. Another factor affecting C accumulation in the soil is how vegetation allocates its C. For example, Tufekcioglu et al. (2003) found that 10-year-old riparian stands of poplar accumulated more of their C and N in above ground biomass whereas grasses accumulated more C belowground. Sharrow and Ismail (2004) reported that 90% of C in a pasture was stored underground whereas approximately 60% of C in a Douglas fir forest was stored in ground biomass.

At Washington Creek, soil OC and TN concentrations for the grass riparian zone and forest riparian zone were not significantly different. There are conflicting results in the literature where SOC in forests and grassland are compared. Tufekcioglu et al. (2001) reported higher SOC concentration for a poplar riparian zone compared to a grass riparian zone six years after rehabilitation of cropped land for the poplar riparian zone and the end of intensive grazing for the grass riparian zone. In contrast, a review of numerous studies showed that the amount OC in soils of temperate grasslands tends to be higher than temperate forests (Houghton et al., 1983). Research also showed that SOC and TN content in grassland versus soils where woody plant invasion had occurred were dependent on soil moisture. At higher moisture levels, both SOC and TN were higher in grassland compared to the woodlands whereas the opposite was true in dry soils (Jackson et al., 2002). Results at Washington Creek indicated that for riparian zones where plants would not experience moisture stress most years, a grass riparian zone would have similar amounts of SOC and TN as a natural forest zone in the top 40 cm.

Only the rehabilitated riparian zone showed a significant decrease in SOC and TN as depth increased, however the same trend was observed for the grass riparian zone and the forest riparian zone. Others have also noted decreases in SOC and soil TN with an increase in soil depth (Jackson et al., 2002; Sharrow and Ismail, 2004; Vesterdal et al., 2002).

The higher C/N ratio of 15 in the forest riparian zone compared to the other two riparian treatments was likely tied to the litter input. In the forest, 53% of the litter going into the soil came from leaf material which had a higher C/N ratio than the herbaceous litter. The proportion of leaf litter for rehabilitated riparian zone was 39% and the grass riparian zone had only herbaceous litter. The many years accumulation of litter with a high C/N ratio would have resulted in a high soil C/N ratio. Schwendenmann et al. (2000) found a similar C/N ratio of 15 for a mature boreal forest, but this did not differ from the C/N ratio for a 25-year-old establishing forest.

4.2.3 Soil Biological Properties

There are several limitations associated with the use of Biolog Ecoplates^{TM} . These include 1) selection for microorganisms which oxidize the 31 substrates included in the ecoplates, 2) selection for microorganisms which are active at the incubation temperature of 25° C and 3) since more than one microorganism may oxidize the same substrate, there may be an underestimation of species of organisms Martin 1991; Garland & Mill, 1991). However, the relative ease of use and low cost of this method make it useful in preliminary studies of microbial community structure and effective for detecting differences between ecosystems (Martin et al., 1999). In addition, the use of Biolog Ecoplates^{TM} for the study at Washington Creek was only one among a number of different soil tests used to evaluate soil quality.

At Washington Creek, no difference in soil microbial community structure was found between the three different riparian zones. Martin et al. (1999) used Biolog GN microplates, which are similar to Biolog Ecoplates^{TM}, except that they contain 95 different substrates, to investigate differences between a grass riparian zone and forest riparian zone in a temperate region. They found no differences in the top 10 cm of soil sampled in June and suggested that the high level of spatial variability in ecosystems may make it difficult to detect differences. However, they did find significant differences in soil sampled in August. At Washington Creek, testing for soil microbial community structure at more than one sampling date may have shown significant differences between riparian zones.

4.2.4 Relationship among Soil Properties

The significant negative correlation between soil bulk density and SOC, soil TN, SOC stock and TN stock is an indication of the importance of SOC on soil structure. Bulk density is a measure of the volume of solid soil particles and pore space between soil particles or aggregates (Hao et al., 2008). Soil organic matter promotes and stabilizes soil aggregates and pore space (Tisdall and Oades, 1982) and therefore the increases observed in SOC (indicating an increase in SOM) would also increase the granular structure of the soil resulting in a decrease in bulk density.

4.3 Soil CO_2 Production Rates and Climatic Factors

4.3.1 Soil CO_2 Production Rates

In this study, soil CO_2 production rates ranged from 60 to 360 mg C m^{-2} hr^{-1}. These values are similar to those reported elsewhere for temperature forested riparian zones (Hopfensperger et al., 2009; Teiter and Mander, 2005) and temperate forests (Groffman et al., 2006).

Mean soil CO_2 production rates for the grass riparian zone were 68 mg C m^{-2} h^{-1} higher than the rehabilitated riparian zone from spring to mid-summer, although the differences were not significant for most of the sampling dates. Similarly, Tufekcioglu et al. (2001) found that seven years after rehabilitation, soil CO_2 production rates were not significantly different between a poplar riparian zone and grass riparian zone in Iowa, although higher rates for the grass riparian zone were recorded for June and July measurements.

At Washington Creek, the soil CO_2 production rates for the forest riparian zone were significantly lower than the rehabilitated riparian zone for spring and summer sampling periods. There are no reports in the literature comparing soil CO_2 production rates in a temperature rehabilitated riparian zone with a natural forest riparian zone. However, in Ohio, USA, a reclaimed mine site was rehabilitated and planted to white pine (*P. strobus* L.), white ash (*Fraxinus americana* L.), tulip poplar (*Liriodendron tulipifera* L.), sycamore (*Platanus occidentalis* L.) and autumn olive (*Elaeagnus umbellata* L.) (Shrestha and Lal, 2008). Thirty years after planting the mine site, the rehabilitated forest had similar soil CO_2 production rates compared with an adjacent natural forest. However, the natural forest had been planted at the same time as the rehabilitated mine site whereas at Washington Creek the natural forest was older than the rehabilitated forest. The similar soil CO_2 production rates found for the rehabilitated mine site forest compared to a the natural forest of the same age in Ohio suggested that given more time, the soil CO_2 production rates at the Washington Creek rehabilitated site may be similar to those of the natural forest riparian zone.

Vegetation type has been shown to have an effect on soil CO_2 production rates. At Washington Creek, the grass riparian zone had significantly higher soil CO_2 production rates than the forest riparian zone. In a review comparing soil CO_2 production research on different ecosystems, Raich and Tufekcioglu (2000) found that grasslands averaged 20% higher soil CO_2 production rates compared to forests. They explained that higher CO_2 production rates were likely a result of grasses allocating more photosynthate to their roots compared to trees which would also allocate C to woody material. On a rehabilitated mine site, the pasture produced more soil CO_2 than either a rehabilitated forest or a natural forest (Shrestha et al., 2009). Shrestha et al. (2008) suggested that the continuous addition of grass roots and leaf or shoot organic matter to the soil led to high rates of decomposition and soil CO_2 production rates.

At Washington Creek, soil CO_2 production rates showed strong seasonal variation with rates increasing from early spring until mid summer and then decreasing during late summer and fall. Others have noted a similar seasonal pattern from a variety of ecosystems such as temperate and boreal forests (Hasselquist et al., 2010; Russell and Voroney, 1998; Vargas and Allen, 2008), grasslands (Vargas and Allen, 2008), a pine plantation (Carlyle and Than, 1988), a reclaimed mine site (Shrestha and Lal, 2008) and riparian forests and grassland (Hopfensperger et al., 2009; Pacific et al., 2008; Tufekcioglu et al., 2001). In these studies, the seasonal variation in soil CO_2 production rates was mainly attributed to differences in air temperature. Root respiration and microbial respiration, two processes which result in soil CO_2 emission, have been shown to increase with higher temperatures (Rastogi et al., 2002).

The statistical analysis of soil CO_2 production rates at Washington Creek revealed a sampling date by treatment interaction. Differences among riparian zones were significant from early spring to late summer, but during September and October, there were no significant differences. Likewise, Shrestha and Lal (2008) found that differences between grass and forests in a rehabilitated mine site were more prominent during the spring and summer compared to autumn. At Washington Creek, lower autumn temperatures accompanied by the decline in plant growth rates resulted in lower soil CO_2 production rates which did not differ between riparian zones.

At Washington Creek, the soil of the forest riparian zone had a greater ability to buffer seasonal fluctuations in soil CO_2 production rates since rates were lower in the forest riparian zone compared to the rehabilitated riparian zone and grass riparian zone. The difference between the lowest and highest soil CO_2 production rates were 89 mg C m^{-2} hr^{-1} for the forest riparian zone compared to 144 and 292 mg C m^{-2} $hr^{-1}$for the rehabilitated riparian zone and grass riparian zone, respectively.

4.3.2 Soil Moisture

Volumetric soil moisture contents at Washington Creek averaged for all sampling dates were 42%, 33% and 28% for the forest riparian zone, grass riparian zone and rehabilitated riparian zone, respectively. The higher bulk density and lower hydraulic conductivity of the rehabilitated soil may have reduced water infiltration into the soil resulting in lower soil moisture. Similar to Washington Creek, Bharati et al. (2002) found greater soil moisture content in a grass riparian zone compared to a rehabilitated silver maple/ash riparian zone. They suggested that absorption of water through well developed tree roots and higher evapotranspiration by the trees may explain the lower soil moisture.

Age of the forest may also affect soil moisture since higher levels of SOM in older forests leads to increased water holding capacity and therefore higher soil moisture (Kern, 1995) . Schwendenmann et al. (2000) reported higher soil moisture content in a 125-year-old boreal forest compared to a 25-year-old forest. At Washington Creek, the forest riparian zone which had a higher SOC content also had higher soil moisture compared to the 25-year-old rehabilitated site. The forest riparian zone also exhibited less seasonal fluctuation of soil moisture content than either the rehabilitated riparian zone or the grass riparian zone.

4.3.3 Soil Temperature

The grass riparian zone had the highest soil temperature for most sampling dates from May 31until the August 23. During this time period, soil temperatures averaged 24, 22 and 21°C for the grass riparian zone, the rehabilitated riparian zone and the forest riparian zone, respectively. In addition, the greatest variation in soil temperature during these months occurred with the grass riparian zone. Similarly, Olsen and Van Miegroet (2010) reported higher soil temperature and greater variability under a forb-grass rangeland compared to a forest of aspen or conifers.

At Washington Creek, although differences were small, the rehabilitated riparian soil temperatures were consistently higher than the forest riparian soil from May 4 until August 9. Similarly, in a boreal forest, Schwendenmann et al. (2000) found that soil temperature was higher in a 25-year-old forest compared to mature (125 years) and old growth (250 years) forests. Research has shown that soil organic matter serves as a buffer to increases in soil temperature (Lal, 2004b). The higher SOC in the forest riparian zone at Washington Creek compared to the rehabilitated riparian zone may explain the difference in soil temperature.

4.4 Relationships with Soil CO_2 Production Rates

4.4.1 Soil Properties

The relationship between soil CO_2 production rates and soil properties is complex and is influenced by ecosystem characteristics. For example, Shrestha and Lal (2008) found significant correlations between CO_2 production rates and soil physical characteristics of bulk density, infiltration and pH. Guner et al. (2010) found that soil CO_2 production rates were positively correlated with sand content and negatively correlated with clay content. Tufekcioglu et al (2001) reported that soil CO_2 production rates were significantly correlated with SOC. A review of soil CO_2 emissions listed soil texture, pH, salinity, vegetation, manure and fertilizer application and tillage as factors affecting soil CO_2 production rates (Rastogi et al., 2002). For soil properties measured at Washington Creek, only pH (for mean of all sampling dates r =.47) and soil C/N ratio (for mean of all sampling dates r = -0.44) were significantly correlated with soil CO_2 production rates.

The negative correlation between soil C/N ratio and soil CO_2 production rates suggested that the availability of easily decomposable organic matter (a low C/N ratio) increases soil CO_2 production. Even though OC stock was high in the forest riparian zone, the high C/N ratio may have been responsible for the lower soil CO_2 production rates compared to grass riparian zone where OC stock was also high, but soil C/N ratio was lower.

In addition, the total litter, litter C and litter N were significantly correlated with soil CO_2 production rates for spring and summer sampling dates. Similar results have been reported elsewhere. A review of published data comparing above ground litter production and soil CO_2 production rates revealed a positive correlation between the two factors in mature forests and grasslands suggesting that soil respiration is greater with greater detritus availability (Raich and Tufekcioglu, 2000). Research carried out on numerous forest ecosystems throughout the world also showed a positive linear relationship between soil CO_2 production rates and above ground litterfall (Raich and Nadelhoffer, 1989). Hopfensperger et al. (2009) found that understory plant cover in different forested riparian zones was important in predicting CO_2 soil emissions. They suggested that areas with a higher percentage of plant cover would have higher root and microbial activity leading to an increase in CO_2 emissions. Results from a study of forests under different climatic conditions in Japan suggested that soil CO_2 production rates are caused by decomposition of SOM which is influenced by litter fall, C stock and soil temperature (Shinjo et al., 2006). However, Hopfensperger et al. (2009) concluded that climatic factors such as soil temperature and moisture are more important factors in determining soil respiration than soil physical and chemical properties.

4.4.2 Climatic Factors

At Washington Creek, soil CO_2 production rates were significantly correlated with soil temperature for all three riparian zones, but soil moisture was significantly correlated with CO_2 production only in the rehabilitated riparian zone and forest riparian zone. When combining all three riparian zones, soil moisture, soil temperature and air temperature were significantly correlated with soil CO_2 production rates. Similarly Tufekcioglu et al. (2001) found that among riparian sites of poplar, cool season grass, switch grass, corn and soybean, soil CO_2 production rates correlated most often with soil temperature within individual sites and with both soil temperature and moisture when all sites were considered together. Vargas and Allen (2008) found that soil CO_2 production rates in a mixed conifer and oak forest were correlated with both soil temperature and soil moisture. However, under herbaceous vegetation, only soil temperature was significantly correlated with CO_2 production. They suggested that processes regulating soil respiration were more complex at the forest site compared to the herbaceous site.

At Washington Creek, soil CO_2 production rates were positively correlated with soil temperature ($r = 0.32$) and air temperature ($r = 0.27$) and negatively correlated with soil moisture ($r = -0.48$). Research conducted in temperate and more northern zones consistently records increases in soil CO_2 production rates associated with increases in air and soil temperatures (Hasselquist et al., 2010; Hopfensperger et al., 2009; Pacific et al., 2008; Tufekcioglu et al., 2001; Vargas and Allen, 2008). However, the response of CO_2 production rates to soil moisture has been more variable. Negative correlations have been reported for a mixed conifer and oak forest (Vargas and Allen, 2008), fallow land (Kowalenko et al., 1978), a rehabilitated mine site (Shrestha and Lal, 2008) and a pine (*Pinus radiata*) forest (Carlyle and Than, 1988). In contrast, Hasselquist et al. (2010) found sharp increases in soil CO_2 production rates after rain events in a semi-arid soil and recorded a positive correlation between soil moisture and soil CO_2 production rates. Likewise in a semi-arid forested ecosystem in India, soil CO_2 production rates were influenced by soil moisture rather than soil temperature (Jha and Mohapatra, 2011). In a boreal forest, soil moisture explained very little of the variation in soil CO_2 production rates (Russell and Voroney, 1998). Tufekcioglu et al. (2009) suggested that both temperature and moisture are important factors in soil CO_2 production rates, however, when one becomes limiting, it then becomes the controlling factor and the other may have little or no influence.

Research has shown that where soil moisture is limiting, there is a positive relationship between soil CO_2 production rates, but where there is no moisture stress, the relationship tends to be negative. For example, in arctic soils, soil CO_2 production rates increased as soil moisture increased, but decreased under anaerobic conditions (Sjögersten et al., 2006). The authors suggested that low carbon input and moisture stress limited respiration under dry soils. In addition, where a negative relationship between soil moisture and soil CO_2 production rates has occurred, it may have been due to a reduction in soil air-filled porosity leading to lower CO_2 emissions (Carlyle and Than, 1988). Pacific et al. (Pacific et al., 2008), concluded that within a riparian zone, high soil water content limited soil gas transport resulting in lower soil CO_2 emissions with increasing soil moisture. In contrast, they found that upland soils with lower soil moisture exhibited higher soil CO_2 production rates as soil moisture increased.

Soils along riparian zones are generally not under moisture stress due to the proximity of water. This factor could partially explain the negative correlation between soil moisture and soil CO_2 production rates at Washington Creek. However, a negative correlation between soil moisture and soil temperature may also play a role in lowering CO_2 production (Carlyle and Than, 1988). At Washington Creek, there was also a negative correlation between soil moisture and soil temperature for the rehabilitated riparian zone and forest riparian zone. As suggested by Vargas and Allen (2008), it may be that processes influencing soil CO_2 production rates at Washington Creek were more complex at the two sites under forest compared to the grass riparian zone.

5. Conclusions and Recommendations

5.1 Summary and Conclusions

Riparian vegetation has been removed in agricultural communities in many regions including Southern Ontario resulting in soil degradation, erosion and reduced water quality due to runoff containing nutrients and pesticides from surrounding fields (Barton et al., 1985; Naiman and Decamps, 1997; Schultz et al., 2004). The important functions of riparian vegetation have been well documented (Lowrance et al., 1997; Schultz et al., 2004; Zaimes et al., 2008). Rehabilitation of degraded riparian zones has been shown to re-establish ecosystem functions such as filtering runoff from fields, improving soil quality, cooling stream water through shading and preventing erosion of the stream bank (Schultz et al., 2004). In addition, rehabilitation of riparian zones may have the potential of providing ecosystem services through the sequestration of C and the reduction of GHGs. Degraded agro-ecosystems such as riparian zones present a large opportunity for sequestering C thereby reducing atmospheric GHGs (Lal, 2004b).

Washington Creek is an interesting research site because of the different riparian land uses within a short distance including an area that was rehabilitated 25 years ago. The rehabilitated section provides a unique opportunity to investigate the effects of rehabilitation of a degraded riparian agro-ecosystem and compare the results to a natural forest riparian ecosystem which would have existed previous to cultivation. The purpose of this research was to determine the effect of the riparian rehabilitation on litter input, soil properties and soil CO_2 production rates through comparison to a grass-forb riparian zone (formerly a pasture) and a natural forest riparian zone.

Most soil quality parameters measured indicated that soil quality of the rehabilitated site was not as good as in the forest riparian zone. Bulk density was higher, hydraulic conductivity was lower and soil C and N were lower in the rehabilitated riparian zone compared to the forest riparian zone. However, there were no significant differences in microbial community structure. Soil CO_2 production rates were higher in the rehabilitated riparian zone compared to the forest riparian zone for most dates measured. In addition, the forest riparian soil appeared to have better buffering capacity compared to the rehabilitated riparian zone since seasonal fluctuations in soil moisture, soil temperature and soil CO_2 production rates were lower in the forest riparian zone. Therefore the first hypothesis stating that there would be differences between the rehabilitated riparian zone and the forest riparian zone was accepted.

The rehabilitated riparian zone and the grass riparian zone had similar levels of litter input. However, the rehabilitated riparian zone exhibited poorer soil quality than the grass riparian zone due to a higher bulk density, lower hydraulic conductivity and lower soil C and N. For most sampling dates, soil CO_2 production rates were not significantly different between the two riparian zones. Therefore the second hypothesis stating that the rehabilitated forest riparian zone will exhibit enhanced litter input, enhanced soil quality and lower CO_2 production rates compared to the grass-forb riparian zone was rejected.

It has been 25 years since the rehabilitation of Washington Creek. Researchers have suggested that that time periods of 40 to 80 years are necessary for a forest to return to its pre-disturbance state (Bush, 2008; Houghton, 2008). The results from this study showed that a time period of greater than 25 years is required for the rehabilitated riparian ecosystem at Washington Creek to fully recover following soil degradation due to conventional row cropping. Plascencia-Escalante (2008) who conducted research at Washington Creek in 2003 to 2005 also concluded that there had not been sufficient time since rehabilitation for ecosystem processes such as N cycling to recover to pre-disturbance levels. However, the present study also indicated that rehabilitation of riparian zones can have a positive environmental impact. Litter input in the rehabilitated area was higher than the natural forest suggesting that in time, the rehabilitated riparian zone will sequester more C in the soil and may have comparable soil quality and soil CO_2 production rates to those of the natural forest

This research also provides additional motivation for the protection of natural riparian ecosystems where they still exist. The evidence from this research and others showing that long time periods are required for riparian zones to recover to their pre-disturbance levels, provides strong argument for the protection and preservation of natural riparian zones.

At Washington Creek there was a positive relationship between CO_2 production rates and soil temperature, and an inverse relationship between CO_2 production rates and soil moisture. In addition, soil CO_2 production rates were positively correlated with soil pH and litter input and negatively correlated with soil C/N.

5.2 Recommendations

Previous studies conducted at Washington Creek since rehabilitation in 1985 include evaluation of fauna and flora, solar radiation, water quality and nitrogen cycling (Gordon et al., 1996; Oelbermann and Gordon, 2000; Oelbermann and Gordon, 2001; Oelbermann et al., 2008; Plascencia-Escalante, 2008). Considering the amount of time required to establish a mature forest, data collection at Washington Creek should continue so that changes in the rehabilitated area could be assessed over a longer time period to determine if soil quality continues to improve. A significant limiting factor in this study was the lack of data from an earlier time period for parameters measured, making it difficult to determine changes in soil quality over time. Repeated measurements of soil quality parameters and soil CO_2 production rates in future years would add to the understanding of how rehabilitation affects soil over time.

Other areas of research could include an in-depth examination of the soil microbial structure and a closer analysis of plant communities including measurements of the quantity and size of root growth. Future studies could also look more closely at water quality. Analysis of the creek at its source, and after each land use would help understand how the different riparian zones are influencing water quality.

Riparian zones offer important ecosystem functions and better understanding of the effects of rehabilitation of degraded agricultural riparian zones will help to guide restorationists in the future and provide additional evidence of the benefits of restoration in terms of improving soil quality, sequestering C, reducing soil GHG emissions and improving water quality.

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