Bioretention for Fish Habitat Protection: Treatment Performance and Spatial Prioritization in a Cold Climate

By

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Abstract

Urban stormwater and meltwater mobilize pollutants and transport them into urban streams, where they pose ecotoxicological risks to aquatic life. Bioretention systems (a.k.a. bioretention cells or rain gardens) are a form of green stormwater infrastructure (GSI; a.k.a. low impact development practice or LID) that retain and filter runoff, reducing volumes and improving water quality before it enters receiving watercourses. This dissertation evaluates the potential for bioretention systems to support urban fish habitat protection by reducing runoff volumes and improving water quality before discharging into urban streams. It also examines whether event-scale stormwater and meltwater treatment improvements are detectable through short-term downstream water quality monitoring and identifies where stormwater interventions should be prioritized to support the protection of sensitive fish habitats.

A perspective chapter outlines an integrated approach to urban stream restoration that combines physical habitat improvements with green stormwater infrastructure practices, such as bioretention systems, within urban land use planning and renewal. The chapter emphasizes post-restoration monitoring, incorporating water quality guidelines into GSI performance evaluations, strategically locating restoration and stormwater interventions where they are most likely to support ecological protection, and increasing stormwater management on private property through public engagement.

Field investigations evaluated three bioretention systems discharging into trout-sensitive urban tributaries in Thunder Bay. Water quality was monitored at all three systems, while water quantity was monitored only at Bioretention Systems 1 and 2 because site constraints at Bioretention System 3 prevented reliable inflow and outflow discharge measurements. During rainfall events, Bioretention Systems 1 and 2 fully retained runoff during 43 and 70 of 87 monitored events, respectively. When effluent occurred, suspended solids concentrations decreased by 51-64% across the three bioretention systems and turbidity was reduced at two of the three systems. These reductions in runoff volumes, turbidity, and suspended solids suggest a reduced potential for particulate pollutant delivery and fine sediment inputs into fish-bearing streams.

Additional analyses examined pollutant accumulation in the winter snowpack and bioretention performance during the spring freshet. Roadside snowbanks contained significantly higher concentrations of chloride, suspended solids, and dissolved organic carbon than open field and bioretention sites. During spring melt, peak and total meltwater volumes were reduced at Bioretention Systems 1 and 2, where hydraulic monitoring was feasible, while water quality was evaluated across all three systems. Across the three systems, pH, turbidity, suspended solids, and dissolved organic carbon concentrations decreased in meltwater before discharging to receiving waters.

A rapid assessment framework integrating stormwater impairment data and habitat surveys was developed to identify priority locations for green stormwater infrastructure. Applied to a trout-sensitive tributary in Thunder Bay, Ontario, this framework provides municipalities with a practical tool to prioritize stormwater interventions in locations where they are most likely to support the protection of sensitive fish habitats.

This dissertation makes several novel contributions to stormwater management and fish habitat protection in a cold climate. First, it provides a critical perspectives-based synthesis that identifies why urban stream restoration, stormwater management, and land use planning can fail to protect urban streams when implemented independently. Rather than simply arguing that these practices should be integrated, this chapter

clarifies specific management disconnects between these practices. It is argued that stream habitat restoration focusses on improving physical fish habitat, but does not adequately address impacts from untreated stormwater runoff, that GSI may reduce runoff quantity and improve runoff quality at the site level, without producing detectable ecological recovery, and that land use planning may miss opportunities to reduce future pollutant loading, protect sensitive areas, or support GSI implementation on private land. The contribution of this chapter is a critical synthesis that reframes urban stream revitalization as an integrated planning challenge rather than a separate set of stream restoration, stormwater and land use practices. Stream habitat restoration projects and green stormwater infrastructure (GSI) aim to protect urban streams, but are often implemented independently. This chapter provides a critical perspectives-based synthesis that reframes urban stream revitalization as an integrated planning challenge. It highlights how isolated approaches can limit ecological recovery and identifies strategies to coordinate habitat restoration, GSI performance evaluation, winter snow monitoring, stormwater controls on private land, and riparian protection in cold-climate urban watersheds. Recommendations include post-restoration monitoring, aligning restoration with local degradation, strategic placement of GSI and habitat improvements, and incorporating land-use planning and zoning to protect sensitive areas. These recommendations provide the conceptual framework for the field-based and applied chapters that follow.

Second, it provides empirical evidence from rainfall and snowmelt events demonstrating that bioretention systems can substantially reduce runoff volumes and particulate-associated pollutants before discharging into trout-bearing waters.

Third, it demonstrates that treatment performance differs between particulate and dissolved contaminants, with high reductions in turbidity and suspended solids concentrations, but limited or inconsistent reductions in chloride and nutrients, emphasizing the need to reduce pollutant sources at the source through changes in land use practices to complement bioretention treatment performance.

Fourth, it identifies roadside snowpack as an important seasonal reservoir of sediment, chloride, and organic carbon, highlighting the influence of winter road maintenance, vehicular activity and spring freshet processes on cold climate stormwater quality. Lastly, it develops a spatial prioritization framework that combines stormwater impairment identification with downstream fish habitat sensitivity analysis to guide green stormwater infrastructure placement where the ecological benefit to fish habitats is most likely.

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1 Introduction

1.1 Urban stormwater and fish habitat degradation.

Urban development degrades fish habitats by altering watershed hydrology and increasing pollutant delivery to receiving waterways, resulting in ecological impairments to urban streams. Impervious surfaces such as roadways, rooftops, and parking lots reduce evapotranspiration and infiltration, increasing surface runoff volumes and velocities (Shiferaw et al., 2025; Ongaga et al., 2024), thereby mobilizing both particulate and dissolved pollutants and transporting them into nearby urban streams. As a result, urban waterways often exhibit “urban stream syndrome,” which is characterized by elevated nutrient and contaminant concentrations, increased temperatures, reduced nutrient uptake efficiency, shifts in invertebrate communities from sensitive taxa to taxa that are more tolerant of changes in flow, water quality, and stream morphology, and declines in fish species sensitive to changes in streamflow and water quality (Walsh et al., 2005). The ecotoxicological risks associated with stormwater runoff may increase with more extreme hydrological patterns associated with climate change, such as longer antecedent dry periods and more frequent floods (Cojoc et al., 2024), which increase pollutant buildup during dry periods and lead to the subsequent release of built-up pollutants into fish habitats during rainfall or snowmelt events.

Urban runoff mobilizes both particulate and dissolved pollutants from roadways and surrounding land uses, degrading water quality in urban streams. Weathering of pavement materials and the mobilization of alkaline constituents such as carbonates and pavement dust can increase pH (Popick et al., 2022; Kriech & Osborn, 2022), while emissions from combustion engines can decrease pH (Göbel et al., 2007). Suspended solids originate from tire and brake wear, soil particles (Zhu et al., 2024), and winter sanding (Galfi et al., 2017), which increase turbidity (Huang et al., 2022). Chloride from road deicing practices (Mundahl & Howard, 2025; Hodgins et al., 2024) increase conductivity. Organic debris, sediments, atmospheric deposition (Kalev & Toor, 2020), and plant and soil organic matter (Al-Amin et al., 2024) increase dissolved organic carbon. Fertilizer application, pet waste, and decomposing organic material increase nitrogen and phosphorus loading (Yang & Lusk, 2018). Particles from tire (Sun et al., 2025; Kang et al., 2025) and brake wear (Lopez et al., 2023; Apeagyei et al., 2011), as well as road dust (Candeias et al., 2020), galvanized roofs and gutters (Müller et al., 2020), and vehicle emissions (Nazarpour et al., 2019) increase zinc concentrations. Similarly, tire and brake wear (Fussell et al., 2022; McKenzie et al., 2009) and road dust (Candeias et al., 2020) increase iron concentrations.

Pollutants transported by urban runoff impair fish health and degrade aquatic habitats through multiple physiological and ecological stressors. Elevated turbidity reduces hypoxia tolerance in fishes (Firth et al., 2024), as well as their thermal tolerance (Fortin-Hamel & Chapman, 2024), feeding efficiency, and prey capture by visual predators (Zanghi & Ioannou, 2024). Suspended solids and fine sediments clog gill tissues, reduce feeding success, delay hatching, suppress growth, reduce fish diversity, and lead to mortality (Pilkerton et al., 2025). Elevated dissolved organic carbon impairs gill function (Morris et al., 2024), reduces food availability for benthivorous fish (Tonin et al., 2025), and alters temperature and dissolved oxygen dynamics (Jane et al., 2024). Elevated ammonia alters blood chemistry, induces oxidative damage, suppresses immune responses, and causes neurotoxicity (Xu et al., 2021). Nitrate and nitrite impair growth, oxygen transport, and physiological homeostasis (Monsees et al., 2016; Kroupova et al., 2005; Jie et al., 2026). Elevated phosphorus concentrations can contribute to eutrophication and reduce dissolved oxygen levels, leading to fish mortality (Sedyaaw et al., 2024). Metals such as zinc and iron induce oxidative stress and tissue damage (Kovacik et al., 2025), while chloride disrupts osmoregulation and may reduce growth and salmonid recruitment (Tollefson et al., 2015; Hintz & Relyea, 2017).

Untreated stormwater runoff imposes severe and persistent impacts on aquatic communities by introducing toxic pollutants that degrade water quality and impair biological function. These impacts include reductions in macroinvertebrate diversity and abundance and increased fish mortality due to changes in the chemical composition of surface waters (Peter et al., 2022). In some cases, exposure to untreated stormwater causes irreversible impairments to multiple salmonid species, even after transfer to clean water (French et al., 2022; Chow et al., 2019). Sublethal exposure can also affect fish embryos, leading to high mortality at hatch (McIntyre et al., 2023). As a result, declines in urban fish populations cannot be reversed without first reducing pollutant loading (Feist et al., 2017). Recognition of these impacts and the symptoms of urban stream syndrome have prompted rehabilitation efforts (Booth et al., 2016) and the development of stormwater management strategies (Kasprzyk, 2025) to better protect urban streams.

1.2 Stream habitat restoration.

Stream restoration strategies improve physical habitat conditions in degraded waterways but fail to address the water quality and hydrological stressors driving degradation in urban fish habitats. Common approaches include streambank stabilization, riparian revegetation, and the addition of in-stream habitat features to increase structural complexity in urban streams. Restoration projects frequently incorporate riparian plantings and bank stabilization to reduce sediment loading and improve stream conditions (Christensen et al., 2024). Revegetation of riparian areas can buffer the impacts of surrounding land uses, improve water quality by retaining particulate pollutants, and enhance the ecological integrity of urban stream ecosystems (Gu et al., 2025). In addition, restoring riffles can increase habitat heterogeneity by creating more variable flow conditions and enhancing structural complexity in simplified stream channels (Walsh et al., 2023). Despite these practices, many restoration efforts focus primarily on physical habitat conditions and are not designed to improve water quality (Rubin et al., 2017), nor do they mitigate the hydrological or chemical alterations responsible for degraded water quality and the loss of sensitive taxa (Bernhardt &

Palmer, 2011). Consequently, green stormwater infrastructure, such as bioretention systems, is needed to manage stormwater runoff and protect urban streams.

1.3 Stormwater management and bioretention systems.

Green stormwater infrastructure (GSI) is a decentralized approach that manages runoff at the source to reduce volumes and improve water quality, addressing the hydrological and chemical stressors causing stream degradation. This approach is referred to by various terms globally, including low impact development (LID), low impact urban design and development (LIUDD; New Zealand), water sensitive urban design (WSUD) and stormwater quality improvement devices (Australia), sustainable urban drainage systems (SuDS; United Kingdom), best management practices (BMPs) and source control measures (SCMs; North America), and the Sponge City concept (China) (Fletcher et al., 2015; Chan et al., 2018). GSI uses decentralized practices that promote infiltration, evapotranspiration, and filtration, thereby reducing runoff volumes and limiting pollutant transport (Dietz, 2007). Bioretention systems are among the most widely implemented forms of GSI for reducing runoff volumes and improving water quality before discharging into urban streams.

Bioretention systems (also referred to as bioretention cells, rain gardens, stormwater biofilters or biofiltration systems) (Spraakman et al., 2020b) are a widely implemented form of green stormwater infrastructure that reduce runoff volumes and pollutant loads, thereby mitigating the hydrological and water quality stressors affecting urban streams. They are small, excavated depressions with a mixture of high-permeability soil, organic matter, and native vegetation that rely on ecological interactions to retain and treat stormwater runoff (Roy-Poirier et al., 2010)**.** Using natural physical, chemical, and biological processes, bioretention systems treat stormwater runoff before discharging into receiving streams (Roy-Poirier et al., 2010). Experimental studies have shown that by reducing stormwater toxicity, bioretention systems can reduce or prevent fish mortality and sublethal effects on fish populations (McIntyre et al., 2023). The storage capacity of the soil layers, combined with the surface ponding depth, evapotranspiration, and exfiltration processes in bioretention systems (Nazarpour et al., 2023; Kratky et al., 2017) all act together to reduce peak and total stormwater discharge entering urban waterways (Lammers et al., 2022; Spraakman et al., 2020a).

Bioretention systems mitigate hydrological and water quality impacts on urban streams by attenuating runoff and removing pollutants through filtration, sedimentation, and sorption processes. Attenuation of peak discharge can mitigate flashy hydrologic responses that lead to channel erosion and habitat degradation (Walsh et al., 2005; Paul & Meyer, 2001), while reductions in total discharge lower pollutant loads entering urban streams (Sabbagh et al., 2025). As stormwater infiltrates through the engineered soil media of bioretention systems, pollutants are removed through filtration, sedimentation, and sorption processes (Johansson et al., 2025; Kratky et al., 2021; Spraakman et al., 2020b). These processes effectively remove particulate pollutants such as suspended solids (McPhillips et al., 2023; Kratky et al., 2021) and particulate-bound metals (Furén et al., 2025; Croft et al., 2024; Jafarzadeh et al., 2024) before entering urban streams, even under cold climate conditions (Furén et al., 2025; Kratky et al., 2021). This removal reduces sediment and pollutant transport to urban streams, helping to reduce impacts to fish habitat associated with sediment deposition and stormwater-derived pollutants (Pilkerton et al., 2025).

1.4 Cold climate constraints on bioretention performance.

Cold climate conditions can influence the hydrologic and water quality performance of bioretention systems. In northern climates, freezing temperatures, snow accumulation, and pollutant build-up in the snowpack can affect how bioretention systems perform during the spring melt. Over winter, pollutants from de-icing salts, atmospheric deposition, and other urban activities accumulate in the snowpack and are released into urban streams during the spring freshet (Sofijanic et al., 2021; Nazarenko & Ariya, 2021). Previous studies show that bioretention systems can remain hydrologically active during snowmelt periods and can reduce runoff volumes by promoting infiltration and temporary storage of meltwater (Gougeon et al., 2023; Kratky et al., 2021; Muthanna et al., 2007). Laboratory experiments found that repeated freeze-thaw cycles can create macropores in bioretention media that help maintain high infiltration rates under sub-freezing conditions (Ding et al., 2019). Similarly, while a bioretention system in Calgary, Alberta, remained hydrologically active under cold prairie conditions, the cold-climate experiments were conducted during or following Chinook conditions, which can temporarily raise winter temperatures above freezing and promote intermittent thawing (Khan et al., 2012).

These conditions differ from Thunder Bay, where Chinook-driven thaw events are not characteristic of the winter climate and frost penetration can exceed 2.2 metres (MTO, 2010). Therefore, deep seasonal frost in Northwestern Ontario may delay or limit infiltration until spring thaw, even though cold-climate studies from prairie regions show that preferential flow through cracks, macropores, or partially thawed pathways can allow water movement under some frozen or thawing conditions.

1.5 Bioretention treatment limitations.

Bioretention systems are less effective at reducing dissolved forms of certain pollutants, including metals, nutrients, and chloride, particularly under cold climate conditions. While bioretention systems effectively remove particulate-bound metals through filtration and sedimentation, removal of dissolved metals is less consistent because they remain more mobile in solution, interact less strongly with bioretention media (Croft et al., 2024; Furén et al., 2025), and may leach under certain conditions (Furén et al., 2025). Nutrient removal can also be variable, as nitrogen and phosphorus removal depend on media composition, redox reactions (Hu et al., 2025), and plant community characteristics (Muerdter et al., 2020). Winter road salting can result in elevated chloride concentrations during spring melt (Sofijanic et al., 2021), which may persist in bioretention soils long after the snowmelt period (Shetty et al., 2020). Elevated chloride concentrations can inhibit nutrient retention, influence metal mobility (Hu et al., 2025), and contribute to phosphorus release from bioretention systems (Goor et al., 2021). Additionally, highly soluble pollutants such as chloride are poorly retained in bioretention systems because they remain dissolved and pass through the media during rainfall and snowmelt events (Brown et al., 2024; Shetty et al., 2020).

1.6 Evaluation bioretention performance.

Common metrics used to assess water quality performance may not adequately reflect the ecological effectiveness of bioretention systems. The performance of green stormwater infrastructure practices such as bioretention systems is often evaluated using concentration- and load-based reduction indices (McPhillips et al., 2023; Spraakman et al., 2020b; Liu et al., 2018). These metrics can be misleading because percent reductions depend strongly on inflow concentrations (McNett et al., 2011), meaning that low inflow concentrations may yield low or even negative reductions. Reduction indices may also oversimplify treatment performance because they do not account for the episodic and highly variable nature of stormwater runoff (Aguilar & Dymond, 2019) and assume a consistent relationship between influent and effluent concentrations that may not exist (McNett et al., 2011).

Alternative approaches are recommended to more accurately assess the ecological relevance of bioretention systems at improving water quality to meet requirements of aquatic life. Studies recommend reporting inlet and outlet discharge rates and pollutant concentrations, along with reduction metrics, to better interpret stormwater treatment performance (Spraakman et al., 2020b). Comparing effluent concentrations to established provincial (Provincial Water Quality Objectives (PWQO); Ontario Ministry of the Environment, 1994) and federal (Canadian Water Quality Guidelines for the Protection of Aquatic Life or CWQG; CCME, 2007) standards can help determine whether bioretention systems reduce contaminant concentrations below levels that may pose a risk to aquatic life. In addition to these comparisons, assessing downstream water quality using paired-catchment or before-after-control-impact (BACI) designs can provide further insight into whether bioretention systems produce detectable changes in water quality; however, long-term monitoring is needed to evaluate overall aquatic ecosystem responses.

1.7 Knowledge gaps in cold climate stormwater management for fish habitat protection.

Important knowledge gaps remain regarding the effectiveness of stormwater management strategies for protecting aquatic ecosystems in cold-climate urban watersheds. Although bioretention systems have been widely studied for their ability to reduce runoff volumes and improve stormwater quality (Spraakman et al., 2020b; McPhillips et al., 2023), most studies evaluate performance at the scale of individual stormwater facilities by comparing effluent volumes, contaminant concentrations, and contaminant loads relative to corresponding influent parameters. These approaches provide valuable information about treatment performance but do not necessarily demonstrate whether stormwater controls produce measurable improvements to water quality in downstream reaches.

These knowledge gaps are even more pronounced in cold climate regions, such as Northwestern Ontario, where the effectiveness of bioretention systems remains largely understudied. In northern regions, pollutants from road salts, atmospheric deposition, and other urban activities can accumulate in winter snowpack and become mobilized during spring melt events (Nazarenko & Ariya, 2021). However, the effectiveness of bioretention systems at treating both rainfall and snowmelt runoff under the climatic and geological conditions present in the region remains uncertain. Compared to Southern Ontario and other cold climate areas, bioretention performance in Thunder Bay (Northwestern Ontario) may be limited by shorter growing seasons (110-115 frost-free days in Northwestern Ontario versus 160-170 frost-free days in Toronto) for vegetative establishment (ECCC, 2026), seasonal frost penetration to depths of 2.2 metres (MTO, 2010), and the close proximity to bedrock on the Boreal Shield where the *Ontario Stormwater Management Planning and Design Manual* specifies that at least 1 metre of separation should be maintained between the bottom of an infiltration basin and bedrock (MOE, 2023). Due to these climatic and geological differences, it cannot be assumed that the quantity and quality treatment performance of bioretention systems in Northwestern Ontario is comparable to that reported in other study areas.

Municipalities must decide how to allocate limited resources to stormwater interventions in locations where expected benefits to aquatic life are greatest. Spatial decision-support tools and modelling approaches can help prioritize the placement of green stormwater infrastructure (Lu & Wang, 2021) by identifying locations where urban runoff poses the greatest threat to aquatic ecosystems. Field studies have shown that fish mortality in urban streams is influenced by watershed characteristics associated with urban development, including road density and traffic intensity (Feist et al., 2017). Pollutants generated from traffic activities accumulate on road surfaces and are transported into receiving waters during runoff events (Göbel et al., 2007), where they can cause acute toxicity and physiological impairment in several salmonid species (French et al., 2022; Chow et al., 2019), underscoring the importance of implementing stormwater controls in sensitive areas of urban watersheds. However, despite growing recognition of the impacts of untreated stormwater runoff on fish populations, determining where green stormwater infrastructure retrofits will most likely reduce urban-stormwater runoff-related ecosystem stressors remains a challenge for watershed managers. There is, therefore, a need for practical approaches to identify stormwater hotspots and fish habitats where targeted stormwater interventions are most likely to reduce runoff-related stressors entering sensitive areas.

1.8 Study Area

This section provides the hydrological and stormwater-infrastructure context needed to interpret the site-scale performance of the bioretention systems, including the size and drainage characteristics of the receiving waters, the drainage areas contributing runoff to each bioretention system, neighbouring treated and untreated stormwater inputs, and the internal design features that influence storage, infiltration, detention, and discharge.

McVicar Creek and the Current River in Thunder Bay, Ontario (Figure 1-1), provide an ideal setting to evaluate whether bioretention systems reduce runoff-related stressors to fish habitats in cold-climate urban watersheds. McVicar Creek is a coldwater stream that originates just north of the City of Thunder Bay. It spans approximately 16 km and drains a watershed area of 48 km^2 (Olivier et al., 2016). The upper portion of the watershed remains largely undeveloped and forested, while the lower reaches are urbanized. Land use within the watershed is dominated by rural and rural-residential land uses, with smaller areas of residential, commercial, and industrial development. The creek is 2.5 to 3 metres in width (McGoldrick et al., 2011) with average discharge of 0.3 m^3/s (Olivier et al., 2016) and peak discharge ranging from 4.8 to 39.5 m^3/s at the mouth (McGoldrick et al., 2011). It has been identified as being at extremely high risk of stormwater quality impairments due to its location and carrying capacity (McGoldrick et al., 2011). The creek supports several fish species, including rainbow trout (*Oncorhynchus mykiss*), brook trout (*Salvelinus fontinalis*), blacknose dace (*Rhinichthys atratulus*), nine-spined stickleback (*Pungitius pungitius*), Johnny darter (*Etheostoma nigrum*), and sculpins (*Cottoidea* family; Olivier et al., 2016).

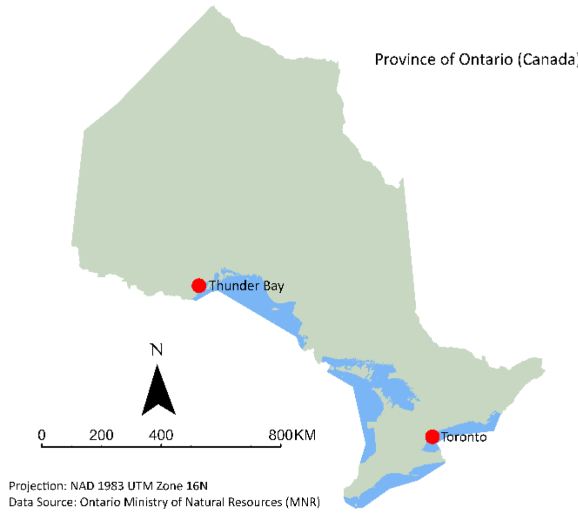


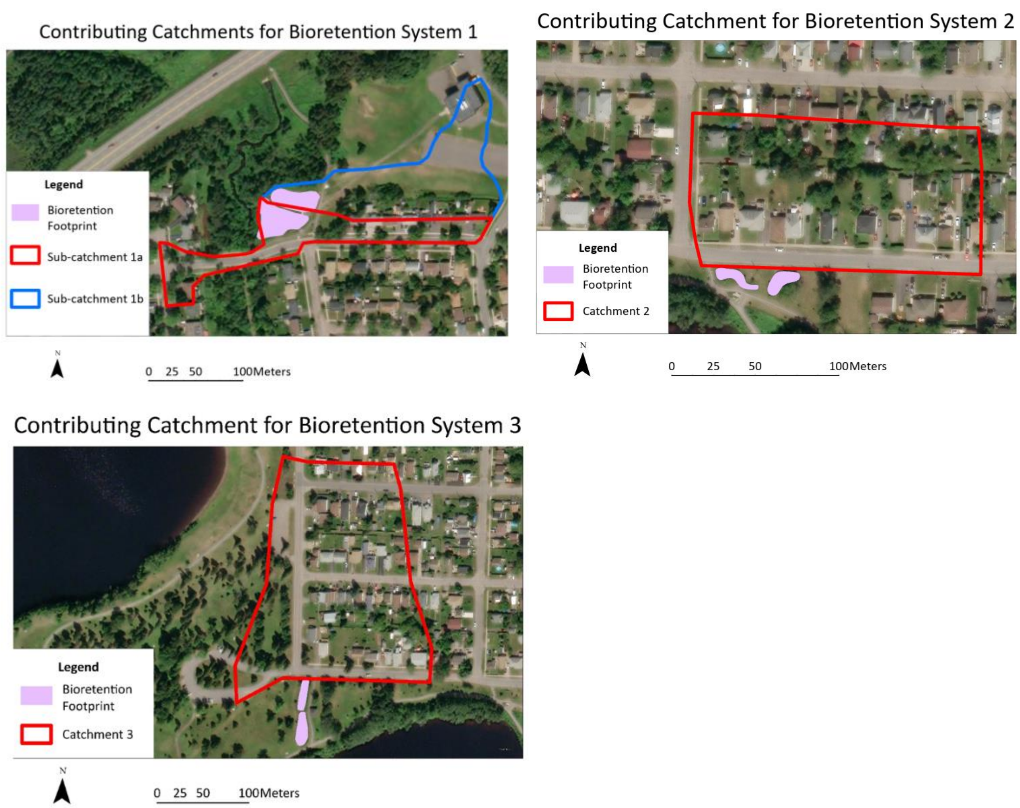
Figure 1-1 Location of Thunder Bay, Northwestern Ontario, Canada.

Bioretention Systems 2 and 3 discharge to Boulevard Lake within the Current River watershed rather than to flowing stream reaches with clearly defined upstream and downstream sampling locations. The Current River is a coldwater river located in the City of Thunder Bay, spanning approximately 64 km and draining a watershed area of 662 km^2 (Olivier et al., 2016). The river has an average mean discharge of 4 m^3/s at the mouth into Lake Superior. The watershed is characterized by extensive bedrock exposure from the Boreal Shield with shallow soils and thin deposits of sandy to silty till overlying bedrock and clay. Much of the watershed remains undeveloped, with approximately 39% of the land classified as rural and large areas designated for open space and environmental protection. Urban development occupies a smaller portion of the lower watershed, with 9% of the area classified as anthropogenic impervious surfaces. The Current River supports several fish species, including rainbow trout (*Oncorhynchus mykiss*) and brook trout (*Salvelinus fontinalis*), rainbow smelt (*Osmerus mordax*), which spawn seasonally, and walleye (*Sander vitreus*) at the mouth of the river.

Differences in the receiving-water settings influenced the monitoring design used in this dissertation. Bioretention System 1 discharged directly into McVicar Creek, where upstream and downstream sampling locations could be established to assess short-term instream water quality responses. In contrast, Bioretention Systems 2 and 3 discharged into Boulevard Lake, a lentic reservoir within the Current River watershed, rather than to flowing stream reaches with clearly defined upstream and downstream sampling locations. Therefore, all three bioretention systems were used to assess site-scale water quality treatment performance, while water quantity performance was assessed only where hydraulic monitoring was feasible. Downstream water quality monitoring was only conducted downstream of Bioretention System 1.

Both McVicar Creek and the Current River have been influenced by historical urbanization and pollution from municipal and industrial effluents. This pollution has led to the degradation of fish habitat in the Thunder Bay Area of Concern, which includes the urbanized portions of both watersheds (Nicholson et al., 2012, p. 6). These impacts have reduced fish habitat quality, species diversity, and fish abundance. Although restoration efforts have improved aquatic habitat conditions through wetland stabilization, restoration of riverine diversity, and the creation of nearshore aquatic habitat, untreated stormwater runoff continues to pose a threat to fish habitat in these watersheds (Nicholson et al., 2012, p. 9). Consequently, improved stormwater management strategies are needed to protect fish habitat in these urban tributaries.

In recognition of the impacts of stormwater runoff on flooding, property damage, and stream ecosystems, the city of Thunder Bay has seen a substantial uptake in green stormwater infrastructure, including bioretention systems. To date, the city operates and maintains 47 city-owned stormwater management facilities, including 37 green infrastructure facilities, 24 of which are bioretention systems. One of these bioretention systems (Bioretention System 1) is in the McVicar Creek watershed, while the other two (Bioretention Systems 2 and 3) are in the Current River watershed. These three bioretention systems are in mature residential neighborhoods with relatively low vehicular traffic (Figure 1-2). Bioretention System 1 is located in the Grandview Gardens neighborhood, a low-traffic residential area composed primarily of single-family detached homes, including bungalows, one-and-a-half storey, and two storey homes. Bioretention Systems 2 and 3 are in the Current River neighbourhood, another established residential area with single family detached homes. The area around Bioretention Systems 2 and 3 also has substantial greenspace, including Boulevard Lake Park and a vegetated corridor along the Current River.



Projection: NAD 1983 UTM Zone 16N. Data sources: City of Thunder Bay, Esri, Vantor, Earthstar Geographics, and the GIS User Community.

Figure 1-2 Catchments draining into the bioretention systems.

Together, the catchment maps, engineering drawings, and maps of neighbouring stormwater infrastructure provide a schematic overview of how runoff moves from contributing drainage areas into each bioretention system, through internal treatment cells, and ultimately to McVicar Creek or Boulevard Lake. These figures also identify nearby untreated stormwater inputs that could influence downstream water quality interpretation at Bioretention System 1 where untreated outfalls enter McVicar Creek near the bioretention system outflow.

The following descriptions summarize the design features most relevant to hydrological and water quality performance, including contributing inflows, cell depth, ponding areas, subdrain configuration, outlet controls, overflow routing, soil media, vegetation, and whether an impermeable liner was identified in the engineering drawings.

Bioretention System 1 has two cells (Figure 1-3). The upper bioretention basin receives runoff from sub-catchment 1b and the lower bioretention basin receives runoff from sub-catchment 1a (Figure 1-2). The upper basin is approximately 0.40 metres deep, with a single smaller pooling area approximately 0.55 metres deep (Figure 1-3). Infiltrated stormwater is collected and conveyed throughout the upper basin via a 150 mm perforated subdrain. The lower bioretention basin is approximately 0.85 metres deep, with two smaller pooling areas, one 1.25 metres deep and the other 1.50 metres deep. Infiltrated stormwater is collected and conveyed throughout much of the basin by a 150 mm perforated subdrain, with the 1.50 metre deep pool being drained by a dedicated 100 mm perforated subdrain. The 150 mm and 100 mm subdrains converge to the outflow into McVicar Creek via a 250 mm PVC pipe. Flow is controlled between the upper and lower basin and between the lower basin and the outflow by two 200 mm diameter knife gate valves. The position of the lower valve adjusts the overall stormwater detention time. The position of the upper valve controls the flow from the upper basin to the lower basin, with the current position only allowing flow from the upper to lower basin during extreme runoff events. The available engineering drawings for Bioretention System 1 did not indicate the presence of an impermeable liner.

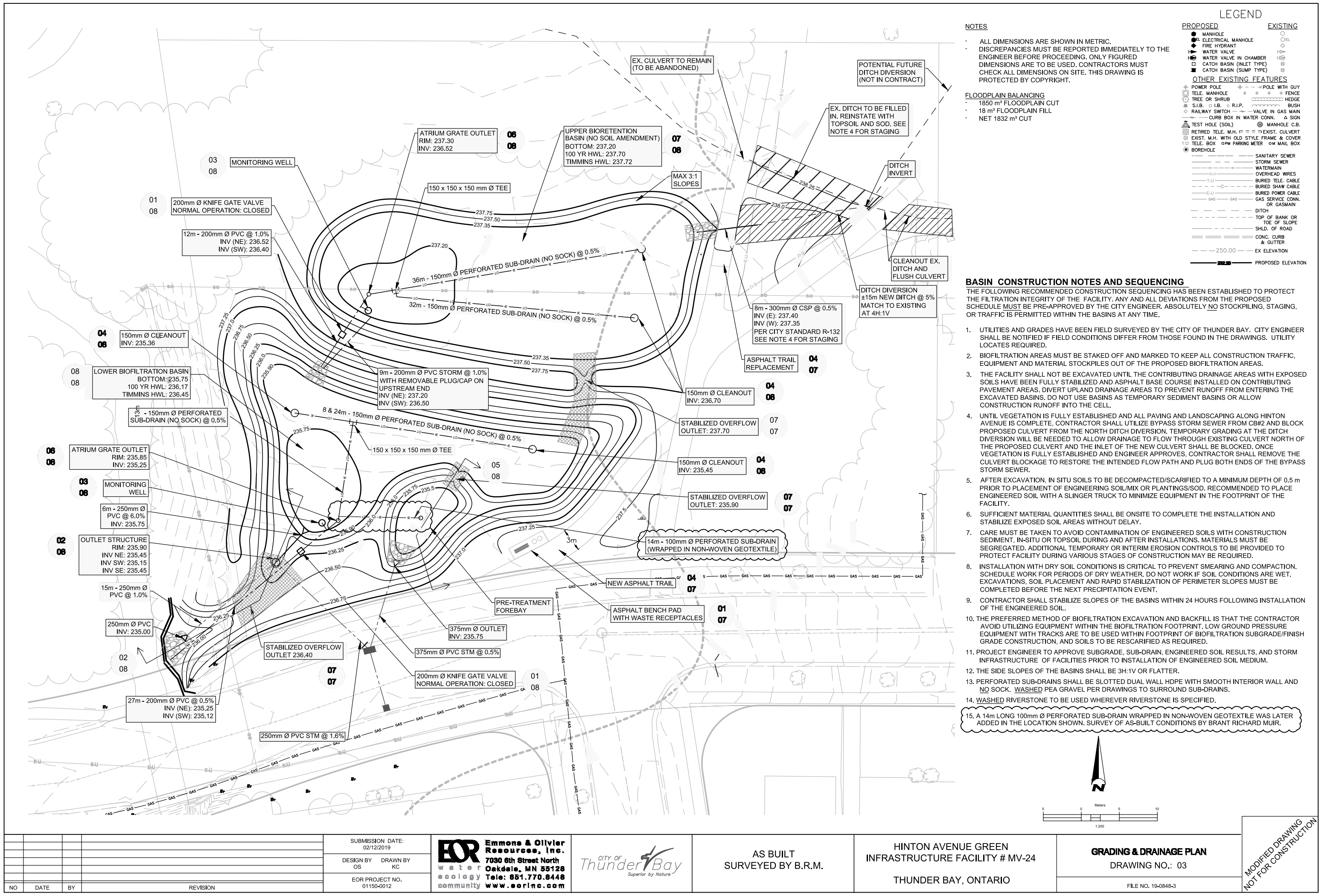


Figure 1-3 Engineering drawing of Bioretention System 1 (City of Thunder Bay).

Vegetation in Bioretention System 1 consists of many herbaceous perennials, emergent wetland plants, grasses, sedges, rushes, trees, and shrubs (Table 1-1).

Table 1-1 Vegetative species in Bioretention System 1

Herbaceous Perennials	**Grasses, Sedges, and Rushes**
Black-eyed Susan (*Rudbeckia hirta*)	American manna grass (*Glyceria grandis*)
Blazing Star (*Liatris spicata*)	Big bluestem (*Andropogon gerardii*)
Blue vervain (*Verbena hastata*)	Blue joint grass (*Calamagrostis canadensis*)
Blue flag iris (*Iris versicolor*)	Canada wild rye (*Elymus canadensis*)
Boneset (*Eupatorium perfoliatum*)	Cord grass (*Spartina pectinata*)
Common milkweed (*Asclepias syriacum*)	Fowl bluegrass (*Poa palustris*)
Common ox-eye (*Heliopsis helianthoides*)	Fox sedge (*Carex vulpinoidea*)
Flat-topped aster (*Doellingeria umbellata*)	Fringed brome (*Bromus ciliatus*)
Golden Alexander (*Zizia aurea*)	Green bulrush (*Scirpus atrovirens*)
Joe-Pye weed (*Eutrochium maculatum*)	Indian grass (*Sorghastrum nutans*)
Large-leaved aster (*Eurybia macrophylla*)	Little bluestem (*Schizachyrium scoparium*)
New England aster (*Symphyotrichum novae-angliae*)	Many-flowered wood rush (*Luzula multiflora*)
	Poverty oat grass (*Danthonia spicata*)
Panicled aster (*Symphyotrichum lanceolatum*)	Rattlesnake manna grass (*Glyceria canadensis*)
	Slender wheat grass (*Elymus trachycaulum*)
Red-stalked aster (*Symphyotrichum puniceum*)	Switchgrass (*Panicum virgatum*)
	Virginia wild rye (*Elymus virginicus*)
Sneezeweed (*Helenium autumnale*)	Wool grass (*Scirpus cyperinus*)
Stiff goldenrod (*Solidago rigida*)	**Trees**
Swamp milkweed (*Asclepias incarnata*)	American tamarack (*Larix laricina*)
Tall meadow rue (*Thalictrum dasycarpum*)	Basswood (*Tilia americana*)
Turtlehead (*Chelone glabra*)	Hackberry (*Celtis occidentalis*)
Wild bergamot (*Monarda fistulosa*)	Red oak (*Quercus rubra*)
Common yarrow (*Achillea millefolium*)	
Emergent Wetland Perennials	**Shrubs**
Sweet flag (*Acorus americanus*)	Huron dogwood (*Cornus racemosa* 'Hurzam')
Water plantain (*Alisma subcordatum*)	Meadow rose (*Rosa blanda*)
Arrowhead (*Sagittaria latifolia*)	

In addition to the treated stormwater discharged from the bioretention system outflow, untreated stormwater from neighbouring catchments is conveyed through traditional storm sewer infrastructure and discharged directly into McVicar Creek without quality or quantity treatment (Figure 1-4). A conventional stormwater outfall located approximately nine metres downstream of the bioretention system outflow discharges untreated stormwater into the creek. A steel culvert approximately six metres downstream of the bioretention system outflow prevented the deployment of t-posts and sampling equipment. Untreated stormwater from this outfall, therefore, could confound any instream water quality benefits from the bioretention system. A second untreated outfall is located approximately 90 metres downstream of the bioretention system.

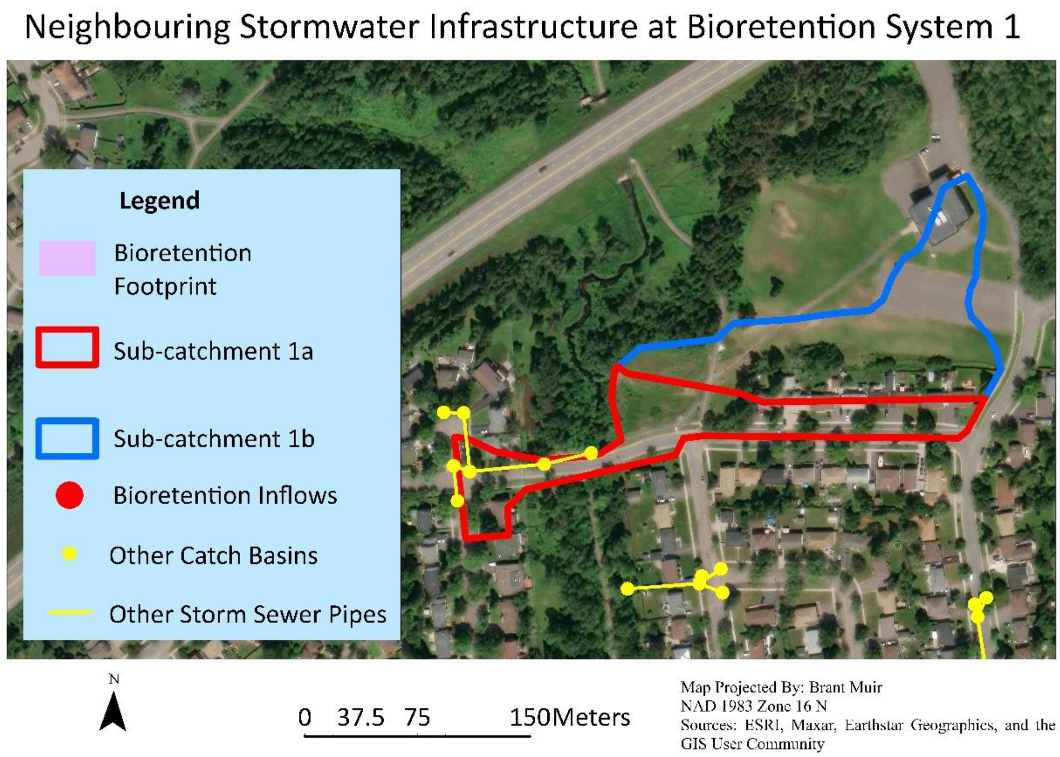


Figure 1-4 Neighbouring stormwater infrastructure at Bioretention System 1.

Bioretention System 2 and Bioretention System 3 also consist of two cells (Figure 1-5; Figure 1-8). However, unlike Bioretention System 1, which receives runoff from two sub-catchments, Bioretention Systems 2 and 3 each receive roadway runoff from a single inflow pipe. In both systems runoff enters the first cell, moves into the second cell, and is discharged through an outflow pipe into Boulevard Lake.

At Bioretention System 2, stormwater from a roadside catch basin is conveyed under the road and into the bioretention system via a 250 mm PVC pipe at a 0.3% grade. Once across the road, the stormwater flows onto a 130 mm concrete runnel, and into a pre-treatment area which was 625 mm deep and consisted of river stone ranging from 50 to 250 mm in diameter. After flowing over the pre-treatment area, stormwater enters the first bioretention cell. This cell has various soil and aggregate layers, including (from bottom to top) a 300-mm deep storage layer consisting of 50 mm diameter clear stone, a 100 mm pea gravel separator with 3-10 mm diameter clear stone, a layer of non-woven geotextile, bioretention soil, which consisted of 85-88% sand, 8-12% fines, 3-5% organic matter, with a p-index (phosphorus) value between 10 to 30 ppm, a pH between 5.5 and 7.5, an infiltration rate between 25 and 203 mm/hr, and is free of stones, stumps, roots, or other objects larger than 50 mm. Above the bioretention soil is a 75-mm deep layer of hardwood mulch.

There is no sub-drain present in the first cell, and overflow from the first cell is conveyed into the second cell via a 200 mm overflow pipe at a 5% grade. This overflow pipe has a round bottom outlet grate, is wrapped with filter fabric and covered with 50-150 mm of washed river stone. The second cell also has various soil and aggregate layers, including (from bottom to top) a 300 mm deep storage layer of 50 mm diameter clear stone, a 100 mm deep choking layer of 3-10 mm diameter pea gravel, a non-woven geotextile, a 500 mm deep layer of bioretention soil with the same composition as the first cell, and topped with a 75 mm deep layer of hardwood mulch. Infiltrated stormwater from the second cell is conveyed into a 450 mm outflow pipe via a 150 mm diameter perforated subdrain, which is at a 0.5% grade and wrapped in a geotextile sock. The engineering drawings for Bioretention System 2 did not indicate the presence of an impermeable liner.

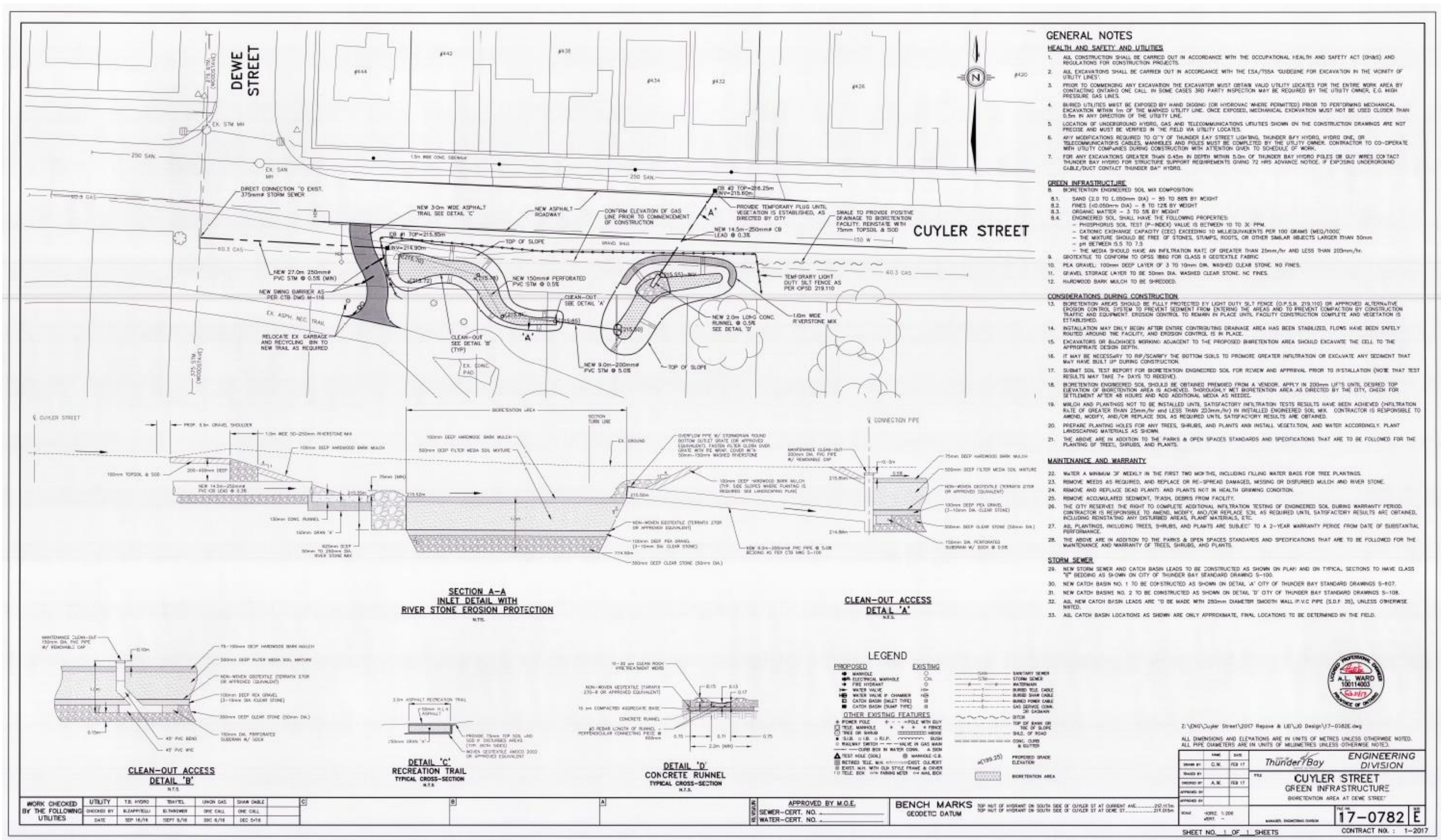


Figure 1-5 Engineering drawing of Bioretention System 2 (City of Thunder Bay).

The vegetation in Bioretention System 2 consists of various herbaceous perennials, grasses, sedges and rushes, trees, and shrubs (Table 1-2).

Table 1-2 Vegetative species in Bioretention System 2

Herbaceous Perennials	Grasses, Sedges, and Rushes
Common yarrow (*Achillea millefolium*) White turtlehead (*Chelone glabra*) Purple coneflower (*Echinacea purpurea*) Prairie blazing star (*Liatris pycnostachya*) Monkey flower (*Mimulus ringens*) Bee balm (*Monarda didyma*) Blue flag iris (*Iris versicolor*) Obedient plant (*Physostegia virginiana*) Black-eyed Susan (*Rudbeckia hirta*)	Side oats grama (*Bouteloua curtipendula*) Fox sedge (*Carex vulpinoides*) Switchgrass (*Panicum virgatum*) Little bluestem (*Schizachyrium scoparium*) **Trees** American tamarack (*Larix laricina*) White birch (*Betula papyrifera*) Showy mountain ash (*Sorbus decora*) **Shrubs** Chokeberry (*Aronia melanocarpa*) Red osier dogwood (*Cornus sericea*) Dwarf bush honeysuckle (*Diervilla lonicera*) Common ninebark (*Physocarpus opulifolius*)

In addition to the stormwater discharged from Bioretention System 2, untreated runoff from nearby catch basins is conveyed through conventional storm sewer infrastructure and discharged directly into Boulevard Lake approximately 16 metres west of the bioretention system without quality or quantity treatment (Figure 1-6).

Neighboring Stormwater Infrastructure at Bioretention System 2

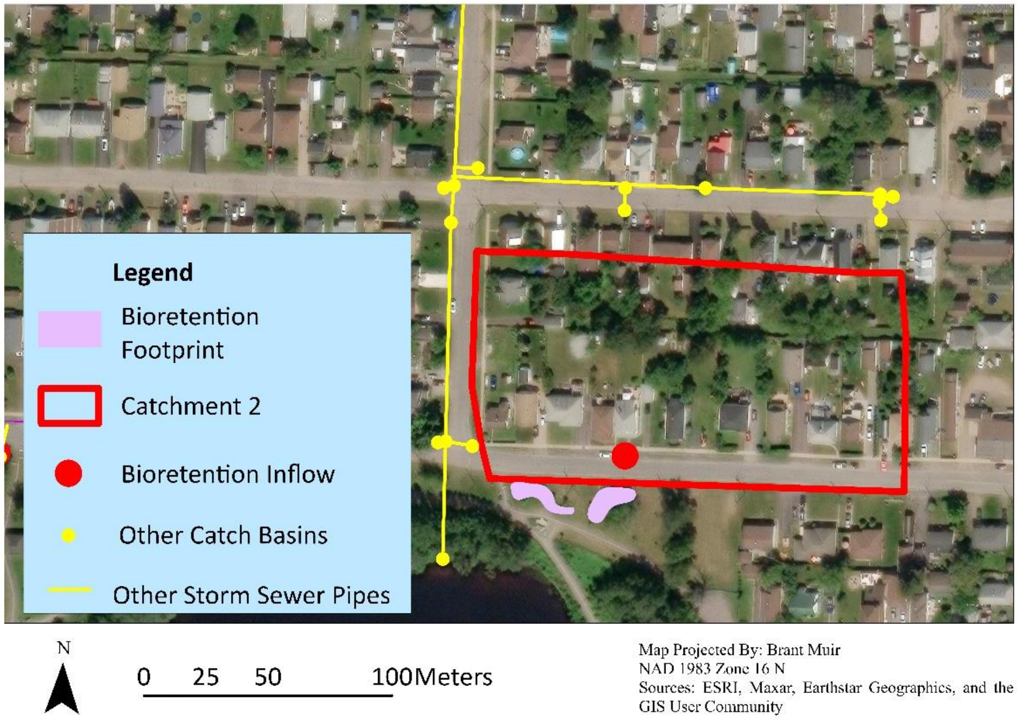


Figure 1-6 Neighbouring stormwater infrastructure at Bioretention System 2.

Like Bioretention System 2, Bioretention System 3 also has two cells, a pre-treatment fore-bay and the main bioretention cell (Figure 1-7). Stormwater from a drainage ditch is conveyed under the road and into a pre-treatment forebay through a 250 mm diameter corrugated culvert with a slope of 0.3%. The pre-treatment forebay is approximately 0.25 metres deep, allowing for a ponding depth of 0.25 metres, before flowing over a 0.5 metre wide separation berm with a 3:1 slope, and into the main cell. The main bioretention cell is approximately 1 metre deep with various layers. From the bottom of the bioretention cell to the surface, the cell has a 300 mm thick layer of clear stone with a 50 mm diameter, a 100 mm thick layer of 3-10 mm diameter pea gravel, non-woven geotextile filter fabric, a 500 mm thick layer of engineered soil mix consisting of 85-88% sand (0.05 - 2 mm diameter), 8-12% fines (<0.05 mm diameter), 3-5% organic matter. The engineered soil mix has a phosphorus soil test (p-index) value between 10 and 30 ppm, a cation exchange capacity exceeding 10 milliequivalents per 100 grams (MEQ\100g), a pH between 5.5 and 7.5, and infiltration rate between 25 and

203 mm/hr and is free of stones, stumps, roots, or other similar objects larger than 50 mm in diameter. Infiltrated stormwater flows through a 150 mm diameter perforated subdrain and out a 450 mm diameter outflow pipe with a slope of 0.5%. The maintenance clean out has a 45^{o} bend approximately halfway down that prevented the deployment of a water level logger in the storage layer of the cell. The engineering drawings for Bioretention System 3 did not indicate the presence of an impermeable liner.

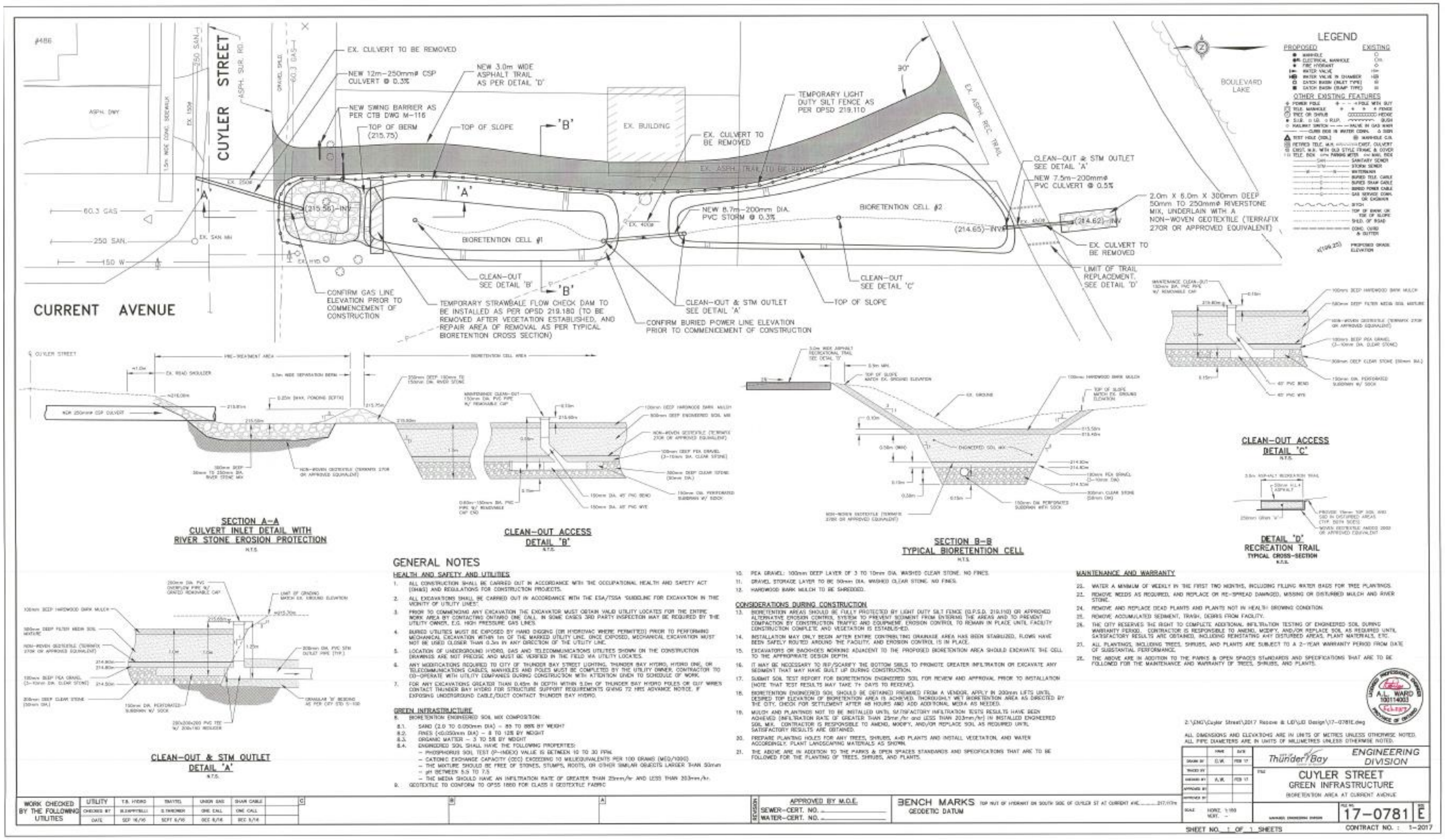


Figure 1-7 Engineering Drawing of Bioretention System 3 (City of Thunder Bay).

The vegetation in Bioretention System 3 includes various vegetative species, including various herbaceous perennials, grasses, sedges, rushes, trees and shrubs (Table 1-3).

Table 1-3 Vegetative species in Bioretention System 3

Herbaceous Perennials	Grasses, Sedges, and Rushes
Common yarrow (*Achillea millefolium*) White turtlehead (*Chelone glabra*) Purple coneflower (*Echinacea purpurea*) Prairie blazing star (*Liatris pycnostachya*) Monkey flower (*Mimulus ringens*) Bee balm (*Monarda didyma*) Blue flag iris (*Iris versicolor*) Black-eyed Susan (*Rudbeckia hirta*)	Sideoats grama (*Bouteloua curtipendula*) Fox sedge (*Carex vulpinoides*) Switch grass (*Panicum virgatum*) **Trees** American tamarack (*Larix laricina*) Silver maple (*Acer saccharinum*) White birch (*Betula papyrifera*) Burr oak (*Quercus macrocarpa*) Red osier dogwood (*Cornus sericea*)
Shrubs Black chokeberry (*Aronia melanocarpa*) Dwarf bush honeysuckle (*Diervilla lonicera*)	

In addition to treated stormwater discharged from Bioretention System 3, untreated runoff from nearby catch basins is conveyed through conventional storm sewer infrastructure and discharged into Boulevard Lake approximately 120 metres east of the bioretention system without any quality or quantity treatment (Figure 1-8).

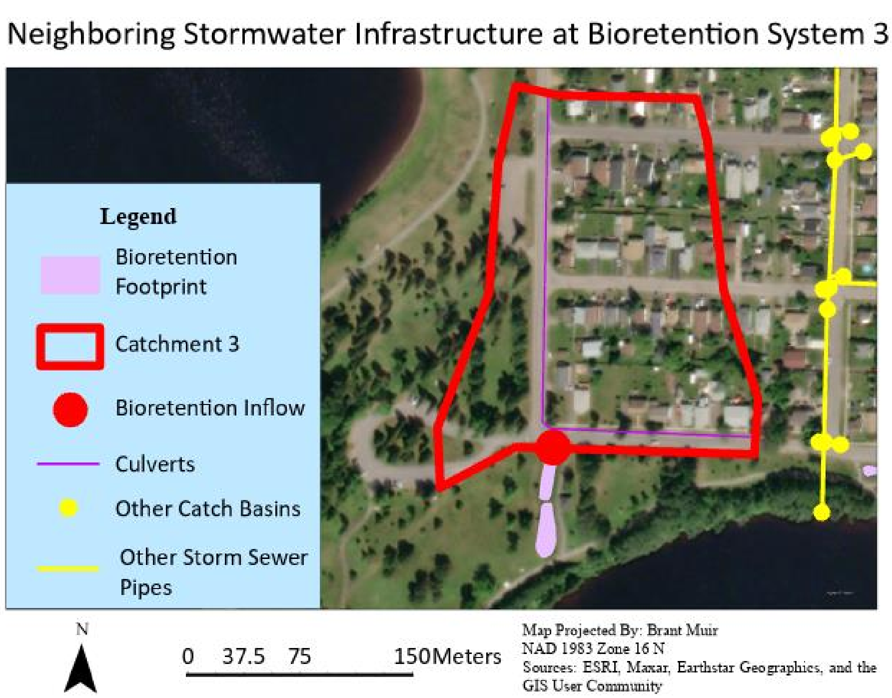


Figure 1-8 Neighbouring stormwater infrastructure at Bioretention System 3.

1.9 Dissertation objectives and structure.

This dissertation evaluates the effectiveness of bioretention systems for improving stormwater and meltwater management in ways that support the protection of sensitive fish habitats in cold-climate urban watersheds. Specifically, it assesses whether bioretention systems reduce runoff volumes and improve stormwater and meltwater quality before discharging into urban streams. It also evaluates whether short-term, event-based monitoring upstream and downstream of treated and untreated stormwater inputs can detect changes in instream water quality. However, because overall fish habitat condition is influenced by multiple watershed-scale stressors, short-term field-based monitoring cannot fully quantify the effects of water quality and quantity improvements from bioretention on downstream habitat condition or isolate the effects of bioretention treatment from other sources of stream quality degradation. Therefore, this dissertation interprets downstream water quality monitoring as a limited assessment of detectable, short-term water quality responses, rather than a complete measure of fish habitat improvement. Lastly, it develops a spatial framework for identifying locations where stormwater management interventions are most likely to support the protection of sensitive aquatic ecosystems.

Chapter 2 examines the role of integrated watershed management in protecting urban stream ecosystems by synthesizing research on stream habitat restoration, stormwater management, and land use planning. This chapter argues that effective urban stream revitalization requires coordinated approaches that address both instream habitat conditions and watershed-scale stressors associated with urban development.

Chapter 3 evaluates the performance of three bioretention systems during rainfall runoff events to determine whether they reduce runoff volumes and improve stormwater quality before discharging into urban tributaries supporting fish populations, and whether short-term downstream water quality responses to bioretention treatment are detectable under geological and climatic conditions unique to Northwestern Ontario (Thunder Bay).

Chapter 4 investigates the performance of three bioretention systems at reducing runoff volumes and improving meltwater quality during spring snowmelt events under climatic conditions unique to Northwestern Ontario (Thunder Bay). To do so, I quantified pollutant accumulation in winter snowpack and evaluated whether bioretention systems reduce meltwater runoff and improve meltwater quality before discharging into urban tributaries and assessed whether short-term downstream water quality responses were detectable during the spring freshet.

Lastly, Chapter 5 develops and applies a spatial framework for prioritizing the placement of stormwater management interventions in urban watersheds by identifying locations where stormwater runoff poses the greatest risk to sensitive fish habitats. This framework can serve as a guide for municipalities to prioritize stormwater management interventions in locations where protection benefits to sensitive fish habitats are expected to be the greatest.

2 Revitalizing Urban Streams through Integrated Habitat Restoration, Stormwater Management and Land Use Planning

Abstract

Stream habitat restoration projects and green stormwater infrastructure (GSI) aim to protect urban streams but are often implemented independently, limiting their ability to address the interacting stressors that degrade urban aquatic ecosystems. This chapter provides a critical perspectives-based synthesis that reframes urban stream revitalization as an integrated watershed planning challenge rather than a set of isolated reach-scale or infrastructure-based interventions. The objective is to identify recurring disconnects among stream habitat restoration, stormwater management, and urban land-use planning, and to develop recommendations that respond directly to those disconnects. The novel contribution of this chapter is the synthesis of these practices into a coordinated framework that links stormwater source control, stream habitat condition, cold-climate pollutant dynamics, and municipal implementation constraints. Using Thunder Bay and McVicar Creek as local cold-climate planning examples, this chapter develops targeted recommendations for long-term event-based monitoring, stormwater controls on private land, spatial prioritization of GSI and restoration projects, and protection of sensitive riparian areas.

AI disclosure

ChatGPT version 5.4 was used to create the figures in this paper.

2.1 Introduction

Impervious surfaces and urban land use practices, such as pesticide, fertilizer, and road salt application, and vehicular traffic, pose a significant threat to urban stream ecosystems. Even relatively low impervious surface cover, often as low as 7% and typically above 10%, has been associated with a significant reduction in biological integrity (Wang et al., 1997) and compromises ecosystem health (Schueler et al., 2009; Lepeška, 2016). Many urban streams exhibit flashier hydrographs, elevated nutrient and contaminant concentrations, altered channel morphology, reduced species diversity, and dominance of pollution-tolerant species (Walsh et al., 2005). These changes, which are collectively referred to as "urban stream syndrome" (Walsh et al., 2005), prompt the need for a more holistic approach to urban stream restoration that considers stream habitat restoration, stormwater management, and land use practices.

Growing awareness of urban stream syndrome has shifted stormwater management from centralized stormwater controls towards decentralized engineered design solutions (Wilfong et al., 2024), such as green stormwater infrastructure (GSI), also known as low-impact development (LID), water-sensitive urban design (WSUD), sustainable urban drainage systems (SUDS), and sponge city (Spraakman et al., 2020a). GSI aims to control runoff at the source by enhancing infiltration, evapotranspiration, and natural storage to improve water quality, restore natural flow regimes, and enhance hydrologic function (Dhakal & Chevalier, 2016). Common practices include bioretention systems (rain gardens), bioswales, and vegetated buffer strips. When properly designed and maintained, these practices can reduce runoff volumes and improve runoff quality, including under cold-climate conditions (Pinneau et al., 2021; Kratky et al., 2021; Shetty et al., 2019). However, despite their potential, current design standards often fail to protect stream ecosystems adequately (Walsh et al., 2016).

In addition, increased flooding and high streamflow events caused by a loss of infiltration and increased runoff from impervious surfaces profoundly impact stream ecology. Floods can affect benthic macroinvertebrates (White et al., 2016), scour fish redds (Carline & McCollough, 2011), exceed the swimming velocity of young fry (Nislow & Armstrong, 2012; Warren et al., 2009), cause catastrophic displacement (Nislow & Armstrong, 2012), and reduce fish (Carline & McCollough, 2011) and macroinvertebrate abundance (Hoopes et al., 1974). Flooding can also increase benthic respiration, lowering dissolved oxygen concentrations (Hutchins et al., 2020). Restoration approaches such as channel rehabilitation, bank stabilization, and enhancing habitat heterogeneity (MacKenzie et al., 2024) can improve instream habitat conditions. However, they often do not consider stormwater management or land use practices. Failure to consider stormwater management and land use practices in restoration approaches can, therefore, limit their ability to restore urban streams effectively.

Despite the well-established link between stormwater runoff and stream habitat degradation, land use planners and engineers often implement stormwater management and stream habitat restoration initiatives in isolation. While both stream habitat restoration measures and stormwater management practices have many strengths, they also have limitations that can reduce their effectiveness when implemented independently. The revitalization of urban streams also requires considerations for land use planning, such as snow and salt management, and the spatial prioritization of interventions. I argue that an integrated approach that combines instream habitat restoration, engineered stormwater controls, and urban land use planning is crucial to revitalizing urban streams and that aligning stream restoration with stormwater management and land use planning provides a more holistic framework for enhancing long-term ecological integrity.

The objective of this chapter is to provide a critical perspectives-based synthesis of stream habitat restoration, stormwater management, and urban land-use planning. Its contribution is to identify recurring disconnects among these practices and develop targeted recommendations that respond directly to those disconnects. Specifically, this chapter examines how habitat restoration may be undermined when stormwater runoff pollutant loading, altered hydrology, sediment transport or habitat fragmentation are not addressed (Rubin et al., 2017; Bernhardt & Palmer, 2011), how GSI performance evaluations may over-emphasize site-scale metrics without linking them to downstream ecological responses (Hopkins et al., 2022; Walsh et al., 2016; Roy et al., 2014), and how land-use planning can constrain or enable restoration through zoning, riparian protection, winter maintenance, spatial prioritization, and private land stormwater controls (Pinto et al,. 2023; Lieberherr & Green, 2018; Ureta et al., 2021). The novel contribution of this chapter is the integration of these components into a coordinated framework that connects stormwater source controls, receiving stream habitat considerations, cold climate pollutant delivery and municipal implementation constraints.

This synthesis is particularly relevant in Thunder Bay, Ontario, a city of the north shore of Lake Superior in Northwestern Ontario, Canada. Thunder Bay is located within the Lakehead Watershed, where several urban watercourses drain through the city towards Lake Superior. The City of Thunder Bay Stormwater Management Plan frames stormwater management as a long-term municipal program that integrates natural systems, infrastructure, monitoring, regulation, green stormwater infrastructure, climate adaptation, and asset management (Olivier et al., 2016). McVicar Creek is one of Thunder Bay's urban watercourses that has been the focus of local stormwater management and rehabilitation planning. The McVicar Creek Protection & Rehabilitation Plan identified stormwater assessment, pollutant monitoring, channel alteration, storm sewer infrastructure, and potential stormwater retrofit opportunities as important considerations for protecting and rehabilitating the creek (Biehn et al., 2014). These local conditions demonstrate why recommendations for urban stream revitalization must be tailored to cold-climate urban watersheds rather than presented as generic best practices.

2.2 Restoring stream habitat

Urban development alters stream morphology and hydrology by increasing impervious surface cover, raising runoff volumes and flow variability, disrupting natural channel form and function (Walsh et al., 2005). Additionally, removing shoreline and riparian vegetation for urban development increases bank erosion and sediment deposition (Walsh et al., 2005). These impacts highlight the need to restore degraded stream systems using streambank stabilization, riparian buffer re-establishment, and instream habitat enhancements (Figure 2-1) to support fish communities and increase habitat heterogeneity (Grabowski et al., 2019).

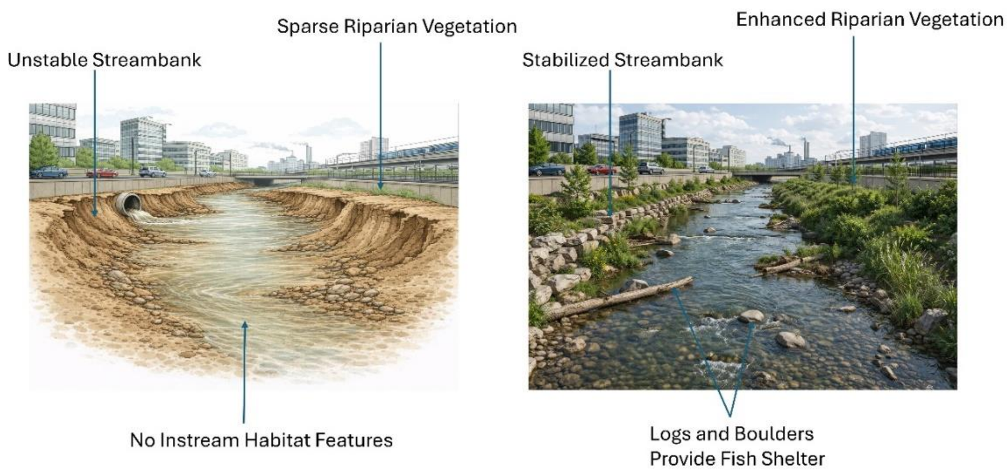


Figure 2-1 Comparison of degraded and restored urban stream conditions.

Streambank rehabilitation can improve physical habitat conditions and reduce erosion but often fails to restore natural hydrology and ecological processes. Streambank rehabilitation involves using hard engineering techniques, vegetation-based stabilization, or combining both to minimize erosion (Bigham, 2020). It may involve reshaping stream channels to create a more sinuous geomorphology (Levi & McIntyre, 2020). These efforts can improve sediment regimes (Griffith & McManus, 2020a), habitat heterogeneity (Champkin et al., 2018), streambed characteristics (Scrimgeour et al., 2013), and overall biodiversity (Palmer et al., 2014; Feld et al., 2011). However, bank stabilization alone often does not return hydrology to reference conditions (Anim et al., 2019b), fully restore ecological processes, or enhance macroinvertebrate communities (Scrimgeour et al., 2013). Consideration of dominant bank erosion mechanisms has also been recommended when evaluating stream restoration outcomes (Bigham, 2020).

Riparian vegetation plays a critical role in regulating stream temperatures, stabilizing streambanks, and reducing nutrient, sediment, and pollutant loads, but is often lost during urban infill (Zaimes et al., 2021; Dodds et al., 2023; Sutter et al., 2021). Vegetated buffers regulate temperatures (Dodds et al., 2023) and mitigate flooding (Eckermann et al., 2022; Varela et al., 2024; Fuller et al., 2022). While revegetation efforts aim to restore riparian function, they have varying success (Mohan et al., 2022). Therefore, in addition to riparian revegetation, restricting further development in riparian zones is essential for long-term habitat protection.

Restoration measures must also address habitat connectivity in fragmented urban streams. While fish habitat enhancements, such as adding meander bends, boulders, and wood structures, can increase habitat complexity and promote salmonid recovery and benthic biomass (Kotalik et al., 2023), and reintroducing riffle-run sequences by constructing rock riffles can increase habitat heterogeneity (Walsh et al., 2023), these interventions do not address challenges associated with highly fragmented urban streams and may not provide meaningful ecological improvements. Fragmentation can constrain fish movement and access to restored habitats, preventing increases in fish populations (Collins & Baxter, 2020) and stream biodiversity (Walsh et al., 2023), despite habitat restoration efforts. As such, in addition to enhancing fish habitat structures, restabilizing streambanks and increasing riparian cover and canopy shading, successful restoration planning must address habitat connectivity (Lavelle et al., 2021) in fragmented urban streams.

Despite the prevalence of stream restoration, there is a lack of post-restoration monitoring to ensure restoration effectiveness. Monitoring is often short-term, overlooking seasonal variability, delayed ecological responses, or geomorphic changes (Sinclair et al., 2023; Reisinger et al., 2019; Spray et al., 2022). Some projects even have unintended consequences, such as increased flood risk from instream structures (Grabowski et al., 2019) and limited benefits in large stream systems (Levi & McIntyre, 2020). Evidence linking hydrologic and topographic restoration to biological outcomes remains limited (Anim et al., 2019a). Therefore, restoration effectiveness cannot be simply by whether physical structures installed, or local habitat features added. Instead, restoration success depends on whether interventions address the dominant watershed processes degrading habitat, including altered hydrology, sediment delivery, pollutant loading and habitat fragmentation. In McVicar Creek, local rehabilitation planning has identified channel alteration, vegetation removal, bank erosion, storm sewer infrastructure, and stormwater retrofit opportunities as important considerations for creek rehabilitation (Biehn et al., 2014). This local example reinforces why reach-scale habitat restoration should be coordinated with stormwater management and land use planning.

2.3 Managing stormwater runoff

As stormwater runoff is one of the primary sources of pollution in urban streams, municipalities must manage this runoff to restore stream ecosystems. In recent decades, there has been a shift from managing stormwater solely for flood protection toward addressing water quality objectives (Bertrand-Krajewski, 2021) in response to a growing concern for ecological sustainability and the effects of urban development on aquatic health. As part of this movement, municipalities are increasingly investing in green stormwater infrastructure (GSI) (Figure 2-2), also referred to as low-impact development (LID), to manage runoff at the source (Rodak et al., 2020). GSI practices can reduce the quantity (Spraakman et al., 2020b; Brodeur-Doucet et al., 2021; Li et al., 2021) and improve the quality (Spraakman et al., 2020b; Kratky et al., 2021) of runoff, even under cold climate conditions. The rise in GSI research from fewer than 30 publications per year in the early 2000s to almost 600 publications in 2021 (Ying et al., 2022; Liu et al., 2023) reflects a shift toward more ecologically oriented stormwater management.

Figure 2-2 Common forms of green stormwater infrastructure for stormwater quantity and quality management.

Despite the increasing adoption of GSI and evidence of its performance in reducing runoff volume, the cumulative effect of small-scale GSI installations on streamflow remains poorly understood. Planners often assume that dispersed local installations will collectively improve urban hydrology (Vogel et al., 2015). Research shows that when implemented at the catchment scale, GSI can reduce peak flow, improve instream hydraulics, and decrease sediment mobility (Zhang et al., 2023; Hopkins et al., 2019, 2022; Anim et al., 2019a). Slower flows can also reduce the entrainment of bed sediments and limit habitat disturbance (Hopkins et al., 2022). However, even with complete capture of runoff from impervious areas, GSI does not completely restore streamflow timing or magnitude to natural levels (Hopkins et al., 2019). These findings suggest that broader land use considerations are needed to support stream recovery at the catchment scale.

The impact of GSI on instream water quality also remains poorly understood. Many studies focus on treatment performance, yet catchment-level effects remain underexplored (Golden & Hoghooghi, 2018; Jalali & Rabotyagov, 2020). Assessing GSI's influence on stream health is complicated by overlapping land use changes, spatial biases in study design, and variability in imperviousness (Jefferson et al., 2017). Some evidence suggests that GSI improves water quality (Roy et al., 2014) by reducing phosphorus (Walsh et al., 2022a), nitrogen, and thermal loads (Walsh et al., 2022a; Scarlett et al., 2018; Roy et al., 2014) and enhances water transparency (Jalali & Rabotyagov, 2020). However, other studies found no reduction in instream nutrients or temperature (Zhang et al., 2023; Simpson & Winston, 2022) and attributed observed water quality improvements to runoff reduction rather than pollutant removal (Jefferson et al., 2017). This inconsistency highlights the need for improved evaluation frameworks and long-term, catchment-scale studies to isolate GSI effects.

While GSI can reduce acute and sublethal toxicity of stormwater (McIntyre et al., 2015), it often does not reverse ecological degradation. Research suggests that GSI alone does not restore periphyton or macroinvertebrate communities and rarely improves downstream biotic indicators (Roy et al., 2014; Walsh et al., 2016). Even increases in biotic integrity scores may not indicate functional recovery, as they seldom reflect a resurgence of sensitive benthic taxa, which are critical markers of ecosystem health (Hopkins et al., 2022). These outcomes suggest that although GSI improves physical and chemical parameters, there is a disconnect between engineering targets and ecological recovery.

While GSI is effective at reducing certain stormwater quantity and quality metrics (Spraakman et al., 2020b; Brodeur-Doucet et al., 2021; Kratky et al., 2021), its ecological impact may be limited without broader integration. The substantial overlap between the goals of GSI and stream restoration suggests that combining these approaches offers greater potential for protecting urban stream ecosystems. Stream revitalization efforts should implement GSI alongside physical habitat improvements, such as streambank stabilization, riparian buffer enhancements, and instream habitat features, to address the complex impacts of urbanization on stream ecosystems (Guimarães et al., 2021). Restoring stream health requires more than engineered infrastructure; it requires aligning hydrologic control, habitat restoration, and catchment-scale planning to enhance ecological outcomes.

2.4 Considering land use

Not only does urban development degrade aquatic habitat by increasing the quantity (Behrouz et al., 2024; Wicke et al., 2021) and degrading the quality (Behrouz et al., 2024; Wicke et al., 2021; Song et al., 2019) of stormwater entering streams, but it also poses significant challenges for restoration efforts. Land use patterns can undermine ecological improvements by constraining the space available for stormwater controls and riparian rehabilitation (Pinto et al., 2023). Consequently, municipalities often implement piecemeal stormwater or habitat interventions where space is available rather than in locations where the ecological return would be most significant. This reactive approach fails to optimize outcomes. To overcome these limitations, watershed managers must account for the specific geomorphic, ecological, and hydrologic characteristics of stream reaches and their surrounding catchments by integrating land use considerations into restoration and stormwater management planning to ensure that interventions align with local stressors and deliver meaningful ecological benefits.

Optimizing restoration efforts requires prioritizing locations where ecological gains will be most significant. Bank stabilization, for instance, yields stronger outcomes in unstable headwater regions than in large, entrenched downstream channels (Levi & McIntyre, 2020). Similarly, revegetating riparian zones should focus on areas most prone to erosion, which can be identified by analyzing slope, proximity to springs, soil erodibility, and surrounding land use (Valente et al., 2021). Enhancing flow heterogeneity is also most beneficial in areas where hydrologic alterations substantially impair stream biology (Irving et al., 2022). A comprehensive approach incorporating stream condition, connectivity, and multiple biological indicators is critical to avoid restoring only those reaches supporting charismatic or easily monitored taxa (Irving et al., 2022). Using tools such as decision support systems (Simperler et al., 2020) alongside GIS and flow-ecology models (Wright, 2021) can help identify sites where restoration investments will have the greatest ecological benefit.

The placement of stormwater controls can be directed based on catchment hydrology and pollutant loading. Prioritizing interventions in hydrologically sensitive areas or pollutant hotspots can maximize their impact (Martin-Mikle et al., 2015; Feist et al., 2017). GIS mapping allows planners to identify roadways, commercial zones, or sub-catchments with disproportionately high contaminant loads (Lu & Wang, 2021; Simpson et al., 2022), enabling spatial targeting of GSI. Even minimal implementation, such as treating runoff from just 1% of a high-impact area, can reduce nutrient and sediment loads by up to 15% (Martin-Mikle et al., 2015). Aligning GSI placement with pollutant hotspots, erosion-prone reaches, and biologically sensitive areas offers a more strategic method of protecting sensitive aquatic habitats.

Land use planning is essential in northern climates, where snow accumulation and melt present distinct challenges. While meltwater runoff typically occurs more gradually than large rainfall events, large volumes released during thaw can overwhelm urban drainage networks and trigger local flooding (Borris et al., 2021). Climate change compounds this risk by increasing the frequency of rain-on-snow events (Seybold et al., 2022). Spring meltwater can deliver concentrated pulses of accumulated contaminants, including metals, sediment, and nutrients, particularly in areas where winter road maintenance and urban snowpack accumulation are important pollutant sources (Vijayan et al., 2024; Popick et al., 2022; Stewart et al., 2013). Winter road maintenance and salt application must also be considered in cold climate areas (Fournier et al., 2022) as part of winter snow quality management.

Stormwater management and stream restoration require careful spatial prioritization due to limited space and competing land demands in urban environments. Existing land use should be accounted for at the outset of planning to ensure that restoration and stormwater controls are feasible, effective, and resilient over time. Land use planning is particularly critical in cold climates, where snowmelt and de-icing compounds exacerbate pollutant loading and runoff timing. It is neither practical nor necessary to restore every reach or treat every catchment equally. Instead, targeting critical zones, such as unstable headwaters, riparian erosion hotspots, and pollutant source areas, offers a strategic path toward ecological recovery. Therefore, considering existing land uses is essential for developing a watershed-scale approach that integrates land use constraints and opportunities to restore the ecological integrity of urban streams.

2.5 Shortcomings of urban stream revitalization

Shortcomings in current stormwater management, stream restoration, and land use planning restrict the successful revitalization of urban streams. In addition to the limitations within each of these areas, stream habitat restoration and stormwater management are often implemented independently, limiting their combined effectiveness.

2.5.1 Shortcomings in stream habitat restoration

Stream restoration projects often do not provide meaningful ecological benefits. For example, structural improvements such as bank stabilization or habitat enhancements often do not restore lost ecological functions or attract target species (Collins & Baxter, 2020). In some cases, they may even support non-native species over native communities (Huntsman et al., 2022). The small scale of their implementation also restricts restoration effectiveness, as small, isolated projects rarely offset the broader impacts of urban development (Kaushal et al., 2023; Brettschneider et al., 2023). Additionally, fluvial geomorphic processes are often overlooked despite their role in shaping stream stability and habitat heterogeneity (Wohl et al., 2024; Piégay et al., 2023). Short-term monitoring compounds these issues, offering little insight into delayed or seasonal ecological responses (Reisinger et al., 2019; Sinclair et al., 2023).

2.5.2 Shortcomings in urban stormwater management

Municipalities widely use green stormwater infrastructure (GSI) to control runoff and reduce pollutant loads. However, researchers typically assess performance through reduction metrics, such as percent removal of contaminants, rather than comparisons to water quality guidelines for aquatic life (Bradford & Gharabaghi, 2004; Walsh et al., 2022b). In northern climates, the influence of contaminated snowpack remains underexplored despite its role as a primary pollutant source during spring melt (Popick et al., 2022; Vijayan et al., 2024). Reporting reduction metrics without raw contaminant concentrations in GSI evaluations also limits our ability to assess ecological effectiveness (Spraakman et al., 2020a). Lastly, even when stormwater controls improve water quality, downstream biological responses are often limited or inconsistent (Roy et al., 2014).

2.5.3 Shortcomings in urban land use planning

Urban land use planning rarely incorporates stream habitat restoration or stormwater management into zoning or redevelopment frameworks, resulting in missed opportunities to co-locate interventions in areas where ecological returns would be most significant (Feist et al., 2017). Limited public space and low uptake on private properties further constrain GSI implementation, notably where public awareness or incentives are lacking (Lieberherr & Green, 2018; Ureta et al., 2021). Riparian zones are also frequently degraded or replaced with non-native vegetation, diminishing their effectiveness at reducing runoff and providing fish habitat (Singh et al., 2021; Foote et al., 2024). Lastly, municipalities often do not implement restoration efforts in locations with the greatest potential ecosystem benefits (Valente et al., 2021).

These shortcomings reveal the need to address urban stream revitalization holistically. Stream health depends on interrelated interactions between hydrology, geomorphology, pollutant loads, and land use factors. As a result, revitalizing urban streams requires integrated planning that aligns revitalization goals with aquatic habitat needs, stormwater management, and land use planning. Integrating these practices is essential to restore the ecological health of urban stream systems.

Since this chapter is a perspectives-based synthesis, the recommendations address disconnects identified across the preceding sections rather than providing generic guidance. Stream habitat restoration may be limited when hydrology, sediment, or connectivity remain unresolved (Walsh et al., 2023; Collins & Baxter, 2020; Anim et al., 2019). GSI can reduce runoff and pollutants, but ecological recovery may be limited if not linked to receiving-stream conditions (Hopkins et al., 2022; Walsh et al., 2016; Roy et al., 2014). Land-use constraints, including limited public space, fragmented ownership, private-property runoff, and degraded riparian zones, restrict implementation (Pinto et al., 2023; Ureta et al., 2021; Lieberherr & Green, 2018). These observations justify recommendations integrating stream restoration, stormwater management, and urban land-use planning in cold-climate urban watersheds.

2.6 Recommendations

Revitalizing urban streams requires integrating stream habitat restoration, green stormwater infrastructure (GSI), and land use planning. However, the recommendations below are not presented as general best practices. Each recommendation responds to a specific limitation identified in the proceeding sections. The stream habitat restoration section showed that physical habitat improvements may provide limited ecological benefit when altered hydrology, sediment delivery, pollution loading, and habitat fragmentation are not resolved. The stormwater management section showed that GSI can reduce runoff volumes and some pollutant loads, but ecological benefits are difficult to demonstrate when studies rely primarily on site-scale reduction metrics rather than receiving stream responses. The land use planning section showed that limited public space, fragmented ownership, runoff from private properties, degraded riparian zones, and winter maintenance practices can constrain stormwater infrastructure and restoration implementation. Therefore, the recommendations below identify specific areas where stream restoration, stormwater management, and urban land use planning need to be coordinated to improve ecological outcomes in cold-climate urban watersheds.

2.6.1 Recommendations for stream habitat restoration

1) Use long-term, event-based monitoring to evaluate whether restoration and GSI produce measurable ecosystem responses.

   Robust, long-term monitoring is essential to evaluate whether restoration and stormwater interventions improve hydrology, morphology, water quality, habitat, and biological responses (Kaushal et al., 2023). In cold-climate streams such as McVicar Creek, seasonal snowpack, road salts, and spring meltwater pulses create dynamics that short-term monitoring would miss (Reisinger et al., 2019; Sinclair et al., 2023). A minimum five-year program should include pre-restoration baselines, one year for initial equilibration, and three years of post-restoration sampling (US EPA, 2003). Longer-term processes may require multi-decadal monitoring: Hubbard Brook studies show that ecological recovery, including streamflow regulation and nutrient cycling, can take 10-20 years (Hornbeck et al., 1997; Bormann & Likens, 1979, p. 153). Chapters 3-4 demonstrate episodic spikes in sediment, chloride, and nutrients that would be missed by short-term monitoring. Multi-year, seasonally tailored monitoring is therefore critical to evaluate ecological effectiveness in cold-climate urban watersheds.

2) Align restoration scale with the degree of degradation.

   Small-scale restoration projects are often ineffective in highly urbanized watersheds, where degradation results from broader, catchment-wide stressors (Rubin et al., 2017). Restoration efforts should be scaled to local hydrologic and ecological conditions and guided by land use, pollution loading, and habitat fragmentation patterns (Griffith & McManus, 2020b; Guimarães et al., 2021). For example, a multi-reach restoration program in Asotin Creek (Washington, USA) implemented approximately 550 large wood structures across 12 km of stream, which increased habitat complexity and corresponded to a 2.5-fold increase in juvenile steelhead density (Bennett et al., 2016). Such increases are unlikely to be achieved through small, single-reach restoration projects.

3) Integrate fluvial geomorphology into restoration design.

   Geomorphic principles such as channel morphology, sediment transport processes, and hydrologic regimes, which are fundamental to restoration outcomes (Hawley et al., 2022), should be integrated into habitat restoration. Implementing features such as riffle-pool sequences and habitat structures should be based on detailed assessments of site-specific geomorphic and hydraulic conditions, including channel gradient, substrate, and flow dynamics, to ensure ecological functionality and physical stability (Chartrand et al., 2023). Incorporating varied streambed configurations to create diverse flow conditions can generate local eddies with pockets of low or negative pressure below and behind boulders and cobbles, increasing opportunities for fish passage through riffle sections, while reversals in horizontal eddies and hydraulic jumps create resting areas in riffle reaches (Newbury & Gaboury, 1993, pp. 87-88).

2.6.2 Recommendations for stormwater management

1) Align GSI performance metrics with water quality guidelines and the CCME WQI.

   Researchers should evaluate GSI performance using ecological thresholds, not just percent reductions. Reporting raw inflow and outflow concentrations (Spraakman et al., 2020a) and comparing them to provincial or federal water quality guidelines allows for more meaningful interpretation. Previous research has shown that relationships between influent and effluent concentrations are often weak or absent, and that reliance on percent removal metrics can therefore be misleading (McNett et al., 2011), reinforcing the importance of comparing effluent concentrations to ecological guidelines. Summarizing parameters with the Water Quality Index (WQI) can also improve the comparability of results between sites and seasons (Bilgin, 2018).

2) Evaluate downstream ecosystem responses to GSI implementation.

Researchers often overlook downstream ecological responses in GSI evaluations. Holistic study designs and statistical approaches like before-after control-impact (BACI) (Liang et al., 2019; Conner et al., 2016; Smokorowski & Randall, 2017) and paired watershed approaches (Smith et al., 2023; Walsh et al., 2022b) can help isolate GSI impacts on biological indicators, although BACI designs still require appropriate baseline data, control sites, and careful interpretation. For example, Liang et al. (2019) used a Bayesian BACI framework to assess GSI effects on stream nutrient concentrations. These tools, which use a comparative approach between reference and treatment sites, are critical for determining whether GSI supports ecological recovery.

3) Incorporate winter snow management in cold climate stormwater monitoring.

Contaminants in winter snowpack pose a significant, often overlooked, threat to receiving waters. Spring melt events release concentrated pulses of metals, nutrients, and other pollutants (Popick et al., 2022). In addition to inputs from vehicular traffic and winter road maintenance (i.e., sanding and salting), airborne particulates, including polycyclic aromatic compounds (PACs), are deposited in the snowpack, accumulating to concentrations that may be toxic to fish embryos, as demonstrated by snowpack contamination and downstream transport associated with oilsands development in the Athabasca River basin (Kelly et al., 2009). Monitoring snowpack contaminant composition across land use types and traffic levels can inform year-round GSI design and placement strategies (Stewart et al., 2013; Fournier et al., 2022).

2.6.3 Recommendations for urban land use planning

1) Integrate GSI with stream restoration initiatives.

Urban land use planning should integrate GSI and stream restoration to provide the greatest ecological benefits. For example, GSI systems with internal water storage zones create anoxic conditions within the bioretention system that enhance nitrate removal during wet weather (Ding et al., 2019), while stream restoration and forested buffers reduce erosion and associated release of nutrients during dry periods (Hejna & Cutright, 2021). Therefore, their integration can further reduce nutrient loading into streams.

2) Use stormwater controls on private land to complement municipal retrofits during watershed-scale planning.

Stormwater controls on private land such as rain barrels, residential rain gardens, downspout disconnection, and lot-level infiltration practices can complement municipal GSI where publicly available space for retrofits is limited. This is particularly relevant in urban watersheds, such as McVicar Creek, where road corridors, buried infrastructure, fragmented land ownership and existing stormwater networks constrain opportunities for large public retrofits. In Thunder Bay, programs already promote private land and community-based stormwater management (Olivier et al., 2016; Biehn et al., 2014). EcoSuperior provides residential rain garden rebates, storm drain education, and an Adopt a Storm Drain program that encourages residences to keep drains clear of leaves trash, and other debris that can enter local streams (EcoSuperior, 2026a, 2026b, 2026c).

However, private stormwater management should not be presented as a simple solution. Voluntary programs may have uneven participation, may be less accessible to renters and lower-income households, and may produce fragmented ecological benefits if uptake does not occur in hydrologically important areas (Lieberheer & Green, 2018; Ureta et al., 2021). For example, a rain barrel or rain garden may on an isolated lot may provide local infiltration or educational benefits, but its ecological value depends on whether it intercepts runoff that would otherwise enter a storms sewer, outfall or sensitive downstream reach of stream. Therefore, stormwater management on private land should be used as a targeted complement to municipal infrastructure, supported by incentives, technical guidance, education, and spatial prioritization, rather than a substitute for watershed-scale stormwater planning.

3) Target projects in ecologically sensitive areas.

Stormwater and restoration efforts should target areas with the greatest ecological benefit. Hydrologically sensitive areas, such as areas near high-traffic roads or dense development, are especially vulnerable to water quality impairments (Martin-Mikle et al., 2015; Feist et al., 2017). GIS tools can identify these areas (Longato et al., 2023; Viezzer et al., 2022; Qiu, 2009) to target project locations. Similarly, restoration measures like riparian planting and bank stabilization should focus on erosion-prone areas identified based on slope, soil conditions, springs, and land use (Valente et al., 2021) to optimize habitat protection.

4) Use municipal zoning to protect riparian areas.

Municipal zoning bylaws should establish protected areas along urban streams. While common in terrestrial and marine systems, established protected areas are rare for freshwater ecosystems, contributing to habitat loss (Valentim et al., 2025). Zoning should restrict development, preserve vegetation, and limit impervious surfaces (Oertel et al., 2024), helping reduce pollutant and sediment loads (Dodds et al., 2023; Sutter et al., 2021), preserve cold-water habitats (Varela et al., 2024), and safeguard sensitive streams from urban sprawl.

2.6.4 Practical limitations and trade-offs of an integrated approach.

Although this chapter argues for greater integration among stream habitat restoration, stormwater management, and land use planning, an integrated approach should not be interpreted as a universal, simple, or low-cost solution to restoring urban streams. Integrated catchment restoration requires coordination across multiple disciplines, spatial scales, monitoring objectives, and implementation partners, which can create governance, monitoring, and project management challenges (Spray et al., 2022). Spatial prioritization of green stormwater infrastructure also requires consideration of sewer infrastructure, implementation potential, land availability, and land ownership (Simperler et al., 2020) not addressed in this perspective. Additionally, because stormwater quality and quantity vary across land uses and because space limitations can constrain stormwater retrofits, the stormwater management strategies discussed should be tailored to local land use conditions and should not be interpreted as a one-size-fits-all solution (Pinto et al., 2023).

Implementing stormwater controls on private land can overcome limitations associated with available public space for stormwater interventions, but they depend on policy instruments such as incentives and outreach to encourage citizen participation (Lieberherr & Green, 2018). Household adoption of GSI can also be influenced by issues surrounding property ownership, previous experiences with flooding, perceptions of the benefits of stormwater management, and perceived barriers to implementation (Ureta et al., 2021).

Furthermore, even when stormwater retrofits are implemented, downstream ecological responses may be limited if major impervious surfaces remain untreated (Roy et al., 2014). Stormwater control measures have often produced minor or inconsistent ecological responses downstream suggesting that management objectives must be explicitly linked to stream protection (Walsh et al., 2016). Long-term watershed monitoring has shown that distributed stormwater management can improve hydrologic outcomes, while water quality benefits are mixed, and sensitive benthic macroinvertebrate species may not fully recover (Hopkins et al., 2022). Integrated planning should, therefore, have realistic expectations, transparent trade-offs, adaptive management, and clearly defined ecological targets.

2.7 Final Considerations

Integrated planning has practical limits. Coordinating restoration, GSI, land-use planning, monitoring, and private-land actions requires collaboration across municipal departments, engineers, ecologists, planners, landowners, and regulators (Lammers et al., 2020; Simperler et al., 2020). In developed watersheds, retrofit space is limited, land ownership is fragmented, and private participation is variable (Pinto et al., 2023; Lieberherr & Green, 2018; Ureta et al., 2021). In Thunder Bay, stormwater management is a long-term municipal program including monitoring, regulation, planning, education, and asset management (Olivier et al., 2016). Even with integration, improvements may not produce immediate detectable changes in water quality or fish habitat because urban streams integrate multiple stressors including altered hydrology, legacy pollutants, channel modification, and degraded riparian zones (Jefferson et al., 2017; Roy et al., 2014; Walsh et al., 2016; Hopkins et al., 2022). Clear prioritization, realistic monitoring, and adaptive management are therefore essential.

There is an urgent need to revitalize urban streams and protect sensitive habitats from ongoing infill and expansion. Habitat restoration practices, such as bank stabilization, riparian planting, and fish habitat enhancements, can improve ecological integrity, while green stormwater infrastructure (GSI) reduces pollutant loads and protects aquatic life. As both approaches aim to improve stream health, they should be integrated into watershed and municipal planning to maximize their collective impact. Municipalities should promote stormwater controls on underutilized private land, target restoration and GSI investments in ecologically sensitive areas, and establish zoning bylaws to protect vulnerable riparian areas from future development. By integrating stream restoration, GSI, and sustainable land use planning, municipalities can improve water quality, enhance ecological resilience, and support the long-term health of urban streams. As development pressures intensify, adopting watershed-scale strategies that align habitat restoration, stormwater management, and land use planning is essential for building resilient urban stream ecosystems.

3 Bioretention Performance at Reducing the Quantity and Improving the Quality of Stormwater Entering Trout-Sensitive Urban Tributaries

Abstract

Bioretention systems are increasingly used to reduce stormwater quantity and improve stormwater quality, but their ability to meet coldwater fish habitat requirements remains uncertain. I assessed water quantity performance at two bioretention systems across 87 rainfall events and water quality performance at three systems during 29 events in two trout-bearing urban watersheds in northwestern Ontario. Water quality was monitored at all three systems, while water quantity was monitored only at Bioretention Systems 1 and 2 because site-specific constraints at Bioretention System 3 prevented reliable hydraulic monitoring. Bioretention Systems 1 and 2 fully retained runoff during 43 and 70 events, respectively, and reduced peak and total discharge by 86-87% and 87-88%. Suspended solids concentrations decreased by 51-64%, and turbidity was reduced at two of the three systems. pH, dissolved organic carbon, dissolved nutrients, metals, and chloride showed limited or inconsistent change, while conductivity increased at all systems. Stream monitoring revealed no measurable water quality differences downstream of either the reference or bioretention outfalls, indicating likely dilution effects. Overall, bioretention systems reduced stormwater discharge and suspended solids delivery, contributing to the protection of coldwater fish habitat.

3.1 Introduction

Urban stormwater runoff entrains pollutants, increasing their delivery into receiving waters (Yazdi et al., 2023) and posing a direct threat to fish habitat in urban streams. Compared to forested and agricultural watersheds, urban catchments can export higher loads of suspended solids and metals into urban streams (Simpson et al., 2023). Impervious surfaces in urban catchments also increase the concentrations of nutrients (Webster et al., 2023), anions (Webster et al., 2023; Rossi et al., 2023), and cations (Webster et al., 2023; Rossi et al., 2023; Smucker et al., 2016). Many of these pollutants enter urban streams during the "first-flush", which refers to the early portion of a rainfall event, when a disproportionate mass of pollutants, particularly particulate-bound pollutants like suspended solids and many metals, are released (Gao et al., 2023; Zakharova et al., 2023). Once released into urban streams, these pollutants pose a significant threat to fish habitats (French et al., 2022; McIntyre et al., 2023), including sensitive salmonid species.

Urban stormwater runoff introduces a range of pollutants that pose various ecological threats to fish health and survival. Changes in pH can reduce sodium uptake and increase ion efflux, lowering salmonid survival (Gonzalez, 2024; Foldvik et al., 2022). Elevated conductivity can impair osmotic regulation and increase the toxicity of contaminants (Fanelli et al., 2024), while elevated chloride concentrations also impair osmoregulation and can be lethal at high concentrations (Dugan & Arnott, 2023; Lai et al., 2022; Erickson et al., 2022). High ammonia concentrations disrupt homeostasis and lead to toxicity in fish (Edwards et al., 2024). Nitrate reduces hypoxia tolerance (Isaza et al., 2021) and nitrite compromises oxygen transport (Jensen et al., 2003). Excess phosphate promotes eutrophication and hypoxia (Dodds & Smith, 2016). High dissolved zinc concentrations disrupt calcium uptake, leading to osmoregulatory stress and sublethal effects, with increased toxicity in soft water (Loro & Wood, 2022; Yu et al., 2023; Stauber et al., 2023). High iron concentrations precipitate as Fe(III) hydroxide, which coats the gills, reducing growth and increasing toxicity at low pH and DOC concentrations (Cardwell et al., 2023; Brix et al., 2023). Consequently, untreated stormwater can lead to fish mortality (French et al., 2022; Peter et al., 2022; Blair et al., 2020), even after exposed fish are transferred to clean water (French et al., 2022).

Due to the toxic effects of these contaminants on fish and other aquatic life, stormwater runoff quality results should be compared to established water quality guidelines, as well as literature-based benchmarks for parameters lacking formal guidelines (Table 3-1). This approach provides a more comprehensive evaluation of stormwater management performance and helps determine whether runoff concentrations may impair fish habitat and health.

Table 3-1 Federal and provincial water quality guidelines used for interpreting water quality results from influent and effluent of the bioretention systems.

Parameter	Federal Water Quality Guideline (CCME)	Provincial Water Quality Guideline (PWQO)	Other Ecological Benchmark	Assessment Criteria Used
Water temperature	Thermal Stratification: Thermal additions to receiving waters should be such that thermal stratification and subsequent turnover dates are not altered from those existing prior to the addition of heat from artificial origins Maximum Weekly Average Temperature: Thermal additions to receiving waters should be such that the maximum weekly average temperature is not exceeded Short-term Exposure to Extreme Temperature: Thermal additions to receiving waters should be such that the short-term exposures to maximum temperatures are not exceeded. Exposures should not be so lengthy or frequent as to adversely affect the important species.	N/A	22°C represents the onset of thermal stress in coldwater fish species, based on established salmonid thermal tolerance literature (Eaton et al., 1995; McCullough, 1999).	22°C
pH	6.5 to 9.0	6.5 to 8.5	N/A	6.5 to 8.5 (PWQO)
Conductivity	N/A	N/A	No widely accepted water quality guideline exists. Conductivity is an aggregate measure of dissolved ions (e.g., chloride, sodium sulphate), which vary in their toxicity to aquatic life and should be interpreted alongside other ion-specific guidelines (e.g., chloride).	None (interpret with chloride)
Chloride	640 mg/L (short-term exposure) 120 mg/L (long-term exposure)	N/A	N/A	120 mg/L (CCME)
Turbidity	**Clear Water:** Maximum increase of 8 NTUs above background levels (short-term); maximum increase of 2 NTUs above background levels (long-term).	No greater than 10% reduction in Secchi depth reading from background conditions.	Approximately 25 NTU has been associated with reduced growth (Sigler et al., 1984; CCME, 2022) and fish density in salmonids under sustained exposure (Sigler et al., 1984). Behavioural effects have been observed at turbidity levels approximately 20-30 NTU (Berg & Northcote, 1985; Gregory & Northcote, 1993).	25 NTU (ecological benchmark)

Parameter	Federal Water Quality Guideline (CCME)	Provincial Water Quality Guideline (PWQO)	Other Ecological Benchmark	Assessment Criteria Used
Suspended solids/sediments	**Clear flow:** Maximum increase of 25 mg/L above background levels for any short-term exposure (e.g., 24-h period). **High flow:** Maximum increase of 25 mg/L above background levels at any time when background levels are between 25 and 250 mg/L, and no more than a 10% of background levels when background is ≥ 250 mg/L.	N/A	Sublethal effects in adult and juvenile fish have been reported at concentrations of 100-300 mg/L (CCME, 2002).	≥100 mg/L (associated with sublethal effects)
Dissolved organic carbon	N/A	N/A	N/A	None (no established water quality guidelines or ecological benchmark).
Ammonia	19 mg/L un-ionized ammonia As un-ionized ammonia concentrations vary with pH and temperature, the corresponding total ammonia guideline also varies with pH and temperature.	20 mg/L unionized ammonia As unionized ammonia concentrations vary with pH and temperature, the corresponding total ammonia guideline also varies with pH and temperature.	N/A	Temperature- and pH-dependent CCME.
Nitrate	550 mg/L (short-term exposure) 13 mg/L (long-term exposure)	N/A	N/A	13 mg/L (CCME)
Nitrite	60 µg/L (0.06 mg/L; long-term exposure) No short-term exposure data	N/A	N/A	0.06 mg/L (CCME)
Phosphate	4-10 µg/L (0.004 - 0.01 mg/L) total phosphorus (oligotrophic lakes and rivers, corresponding to approximately 0.01 - 0.03 mg/L as phosphate (PO_4^{3-})).	30 µg/L (0.03 mg/L) to avoid excessive plant growth 20 µg/L (0.02 mg/L) to prevent nuisance algae 10 µg/L (0.01 mg/L) to maintain aesthetics	N/A	0.03 mg/L (CCME and PWQO)
Zinc	Dependent on DOC concentration, pH, and water hardness.	20 µg/L (0.02 mg/L)	N/A	0.02 mg/L (PWQO)
Iron	110 µg/L (0.11 mg/L) (FEQG)	N/A	N/A	0.11 mg/L (FEQG)

Green stormwater infrastructure (GSI) approaches can reduce the toxicity of stormwater to fish (McIntyre et al., 2015). Bioretention systems are a form of GSI that use vegetated basins with engineered soil layers to retain stormwater, filter contaminants, and promote infiltration (Spraakman et al., 2020a). They consistently reduce peak and total discharge (Spraakman et al., 2020b; Lammers et al., 2022), as well as a range of pollutant concentrations. Even in cold-climate cities, bioretention systems can reduce suspended solids (Spraakman et al., 2020b; Kratky et al., 2021), ammonium and phosphate concentrations (Kratky et al., 2021), and nitrate concentrations when constructed with a submerged saturated zone (Donaghue et al., 2022; Kratky et al., 2021; Søberg et al., 2021). However, while bioretention systems can reduce total zinc concentrations, primarily through particulate filtration and sorption in the filter media (Furén et al., 2025; Croft et al., 2024; Furén et al., 2023), reductions in iron concentrations are often site-specific and variable (Spraakman et al., 2020b). Chloride is generally not retained by bioretention systems, although event-scale load reductions may occur due to reductions in discharged water volume (Spraakman et al., 2020b).

Bioretention systems can reduce pollutant concentrations and loads at the site level, and their widespread implementation can reduce pollutant loads and event-mean concentrations at the watershed scale (Smith et al., 2023). They can also mitigate the impacts of thermal stormwater loads on downstream water temperatures (Martin et al., 2021), thereby protecting cold-water fish species, such as salmonids. In addition to their water quality benefits during wet weather, green stormwater infrastructure practices, such as bioretention systems, can also increase the contribution of cool, filtered groundwater to dry weather flows, thereby reducing instream nutrient concentrations during baseflow conditions (Walsh et al., 2022). However, while GSI practices can improve stream water quality, surrounding land use characteristics (e.g., type, location, and extent of urban development) may also influence the magnitude of these improvements (Bell et al., 2017).

Furthermore, bioretention system performance has been assessed in other cold-climate regions, but there is a lack of region-specific literature on bioretention performance in Northwestern Ontario. Compared to southern Ontario, Northwestern Ontario has a much shorter growing season and fewer frost-free days (Agriculture & Agri-Food Canada, 2023), which may constrain vegetation establishment and warm-season bioretention treatment. The bedrock geology in Northwestern Ontario may also influence reductions in outflow volumes. In karst or fractured bedrock settings, bioretention systems can experience substantial exfiltration through cracks in surrounding soils or bedrock, reducing outflow volumes (DeBusk & Wynn, 2011).

This study was conducted in two trout-sensitive receiving systems with different hydrological settings. McVicar Creek is a smaller coldwater stream that drains a watershed area of approximately 48 km^2 (Olivier et al., 2016), has an average discharge of approximately 0.3 m^3/s (Olivier et al., 2016), and has reported peak discharge ranging from 4.8 to 40 m^3/s near the mouth (McGoldrick et al., 2011). In contrast, the Current River is a larger tributary draining approximately 662 km^2, with an average discharge of approximately 4 m^3/s at its mouth into Lake Superior (Olivier et al., 2016). These differences provide context for interpreting the study design. Inflow and outflow water quantity performance was assessed at Bioretention Systems 1 and 2, while inflow and outflow water quality performance was assessed at Bioretention Systems 1, 2 and 3. Bioretention System 1 discharged into McVicar Creek, whereas Bioretention Systems 2 and 3 discharge into Boulevard Lake, a lentic reservoir on the Current River. Therefore, upstream and downstream water quality responses were assessed only in McVicar Creek where Bioretention System 1 and an untreated reference outfall discharged to defined stream reaches. Additional details on the contributing catchments, neighbouring stormwater infrastructure, storage capacity, vegetation, and inflow and outflow configurations are provided in Section 2.1 to support interpretation of the hydrological and water quality results.

The objectives of this research were to: 1) assess the performance of two large-scale bioretention systems at reducing peak and total stormwater discharge entering two trout-sensitive tributaries in Thunder Bay, Ontario, 2) assess the performance of three large-scale bioretention systems at improving the quality of stormwater entering fish habitats during the first-flush, 3) evaluate whether bioretention quality and quantity performance varies between the bioretention systems, and 4) evaluate downstream water quality in McVicar Creek by comparing instream water quality upstream and downstream of a treated bioretention system outfall and untreated reference outfall during the first-flush.

3.2 Methods

3.2.1 Study area and bioretention systems

Bioretention System 1 receives runoff from two sub-catchments: Inflow 1a, which conveys runoff from the roadway, and Inflow 1b (Figure 3-1), which conveys runoff from a grassed area without the influence of stormwater pollutants related to vehicular traffic. Bioretention Systems 2 and 3 receive roadway runoff from a single catchment (Inflow 2 and Inflow 3, respectively). Each bioretention system discharges to a single outflow (Outflow 1-3). Peak and total stormwater discharge, as well as water quality, were measured at the inflow and outflow of Bioretention Systems 1 and 2. Discharge was not measured at the inflow or outflow of Bioretention System 3, as site-specific constraints (i.e., a submerged inflow pipe and impeded access to the storage layer) prevented the deployment of HOBO U20/U20L water level loggers and the reliable measurement of inflow and outflow discharge. Water quality monitoring was conducted at all three bioretention systems. An untreated storm sewer outfall (Reference Outfall) discharges into the same receiving stream as Bioretention System 1 and serves as a reference site for comparing downstream water quality responses between untreated stormwater inputs and inputs from treated bioretention effluent.

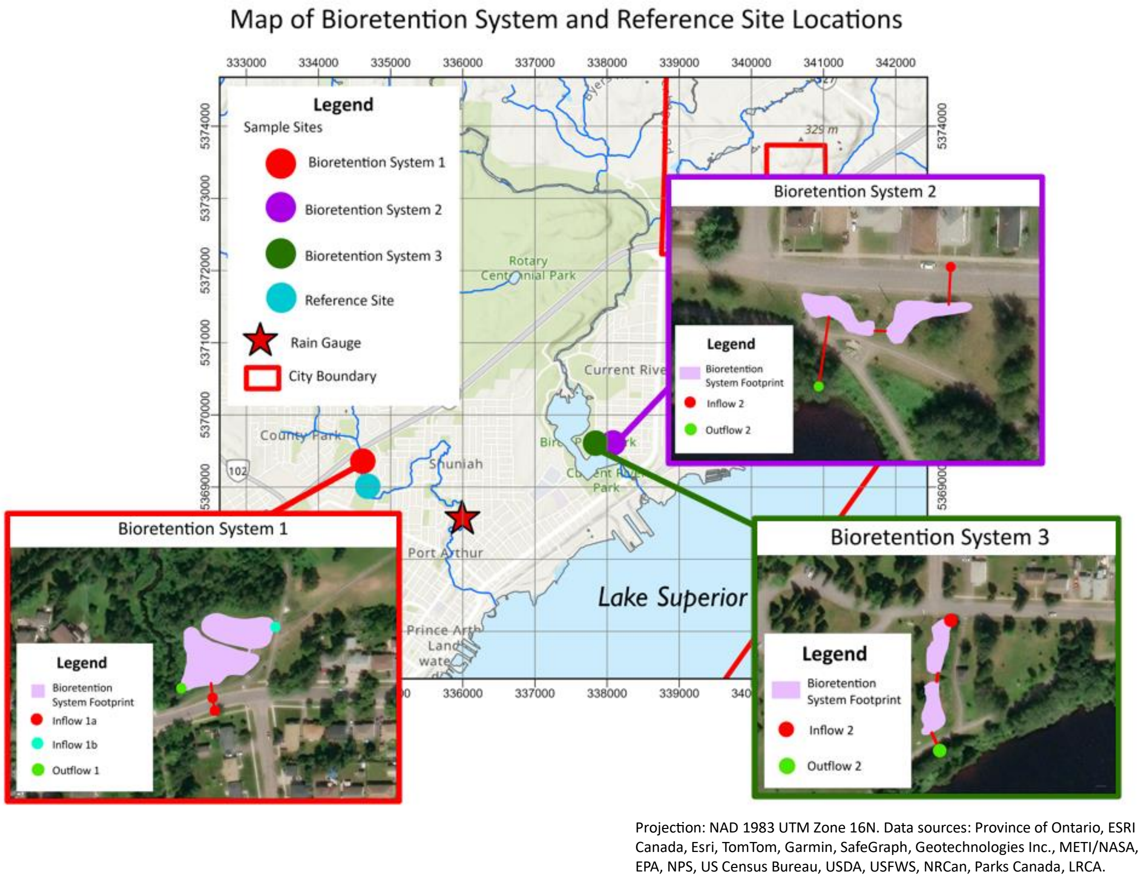
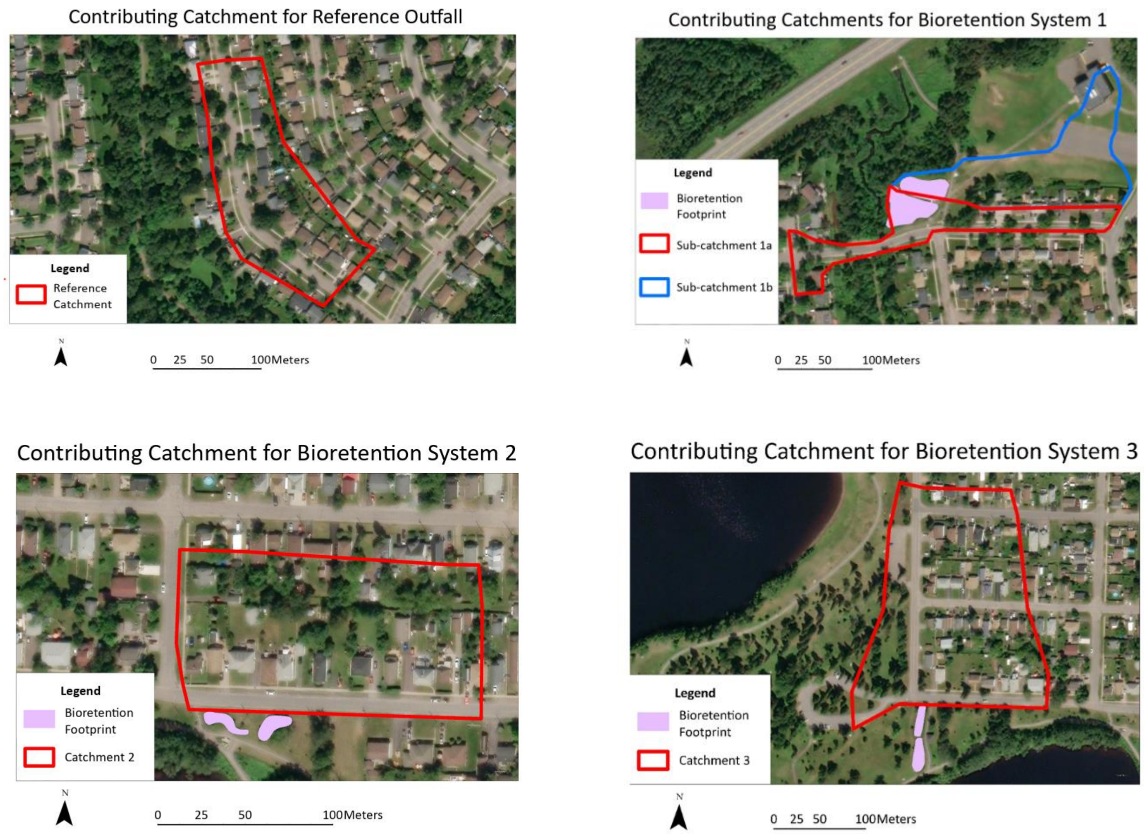


Figure 3-1 Map of bioretention system and reference site locations.

As water quantity and quality performance during rainfall events depends on both catchment drainage characteristics and bioretention system design, the hydrological setting, drainage configuration, storage capacity, and outlet configuration of each bioretention system were considered when interpreting water quantity and water quality results. These features influenced which components of performance could be assessed at each site. Inflow and outflow water quality treatment was assessed at all three bioretention systems, water quantity reductions were assessed at Bioretention Systems 1 and 2 where hydraulic monitoring using water level loggers was feasible, and instream water quality responses upstream and downstream of treated and untreated stormwater effluent was assessed only in McVicar Creek at the reference outfall and Bioretention System 1.

I delineated catchments and quantified impervious surface areas to characterize the contributing areas from each study catchment. Catchment boundaries were delineated in ArcGIS Pro 2.9.0 using city-provided shapefiles. Bioretention System 1 receives runoff from two sub-catchments (Sub-catchment 1a and Sub-catchment 1b), whereas Bioretention Systems 2 (Catchment 2) and 3 (Catchment 3) each receive runoff from a single catchment. All study catchments were in mature residential neighborhoods characterized by primarily single-family, detached housing. The reference site and Bioretention System 1 were in the Grandview Gardens area, a quiet residential neighborhood with relatively low traffic volumes and housing composed mainly of single-detached homes including bungalows, one story, one-and-a-half story and two story houses. Bioretention Systems 2 and 3 are also located in a mature residential neighborhood (Current River), with large areas of greenspace, including Boulevard Lake Park, and a greenspace corridor along the Current River. Housing in Current River is predominantly single family, detached homes.



Projection: NAD 1983 UTM Zone 16N. Data sources: City of Thunder Bay, Esri, Vantor, Earthstar Geographics, and the GIS User Community.

Figure 3-2 Contributing catchments for the reference outfall and Bioretention Systems 1, 2, and 3.

Impervious surfaces were manually digitized using aerial imagery and classified into five categories (roads, buildings, parking areas, sidewalks/paved trails, and other) to determine the main sources of impervious surfaces in each catchment (Table 3-2). The designed storage capacity (m^3) refers to the designed surface and subsurface volume of the bioretention systems. As the reference catchment does not have a bioretention system, nor provide any form of stormwater storage, the designed storage capacity is marked N/A.

Table 3-2 Catchment size, imperviousness, and designed storage capacity of each bioretention system.

	Catchment Size (ha)	% Imperviousness	Designed Storage Capacity (m^3)
Reference Catchment	1.8	49	N/A
Bioretention System 1	2.6	35	616
Sub-catchment 1a	1.1	49	375
Sub-catchment 1b	1.5	30	241
Bioretention System 2	1.6	31	82
Bioretention System 3	4.0	35	157

Impervious Surface Cover	Reference Catchment		Catchment 1a		Catchment 1b		Catchment 2		Catchment 3	
	Area (ha)	% Total Cover	Area (ha)	% Total Cover	Area (ha)	% Total Cover	Area (ha)	% Total Cover	Area (ha)	% Total Cover
Roads	0.23	12.6	0.21	19.4	0.00	0.0	0.12	7.74	0.71	17.8
Buildings	0.29	15.8	0.10	9.3	0.14	9.15	0.21	13.55	0.61	15.29
Parking Areas	0.23	12.6	0.13	12.0	0.17	11.11	0.10	6.45	0.45	11.28
Sidewalks/ Paved Trails	0.09	4.9	0.09	8.25	0.02	0.13	0.04	4.52	0.03	0.75
Other	0.05	2.7	0.0	0.0	0.07	4.58	0.02	1.29	0.01	0.25
Total Impervious Cover	**0.89**	**49**	**0.53**	**49**	**0.4**	**25**	**0.49**	**31**	**1.81**	**45**
Total Pervious Cover	**0.94**	**51**	**0.55**	**51**	**1.1**	**75**	**1.11**	**69**	**2.19**	**55**

The three bioretention systems also differed in internal design features that may influence rainfall-runoff detention, infiltration, and pollutant removal. Bioretention System 1 consists of upper and lower treatment cells, receives runoff from two sub-catchments, and conveys treated water to McVicar Creek through perforated subdrains, gate valves, and a controlled outflow pipe. Bioretention Systems 2 and 3 each receive runoff from a single catchment and discharge to Boulevard Lake through pretreatment and bioretention cells connected to perforated subdrains and outflow pipes. Available engineering drawings did not indicate the presence of impermeable liners in any of the three systems. Vegetation across the systems consisted of herbaceous perennials, grasses, sedges, rushes, shrubs, and trees, with wetland vegetation also present in Bioretention System 1. These design features, together with differences in contributing catchment size, imperviousness, and storage capacity, provide important context for interpreting differences in water quantity and water quality performance among the bioretention systems.

Near the outflow of Bioretention System 1 there is also nearby conventional stormwater infrastructure that discharges untreated stormwater into the creek (Figure 3-3). A conventional stormwater outfall providing no quantity or quality treatment is located approximately 9 metres downstream of the bioretention system outflow and discharges stormwater into the creek without any quantity or quality control. A steel culvert is also present approximately six metres downstream of the bioretention system outflow. A second untreated outfall is located approximately 90 metres from the bioretention outflow and also discharges stormwater into the creek without any quantity or quality control.

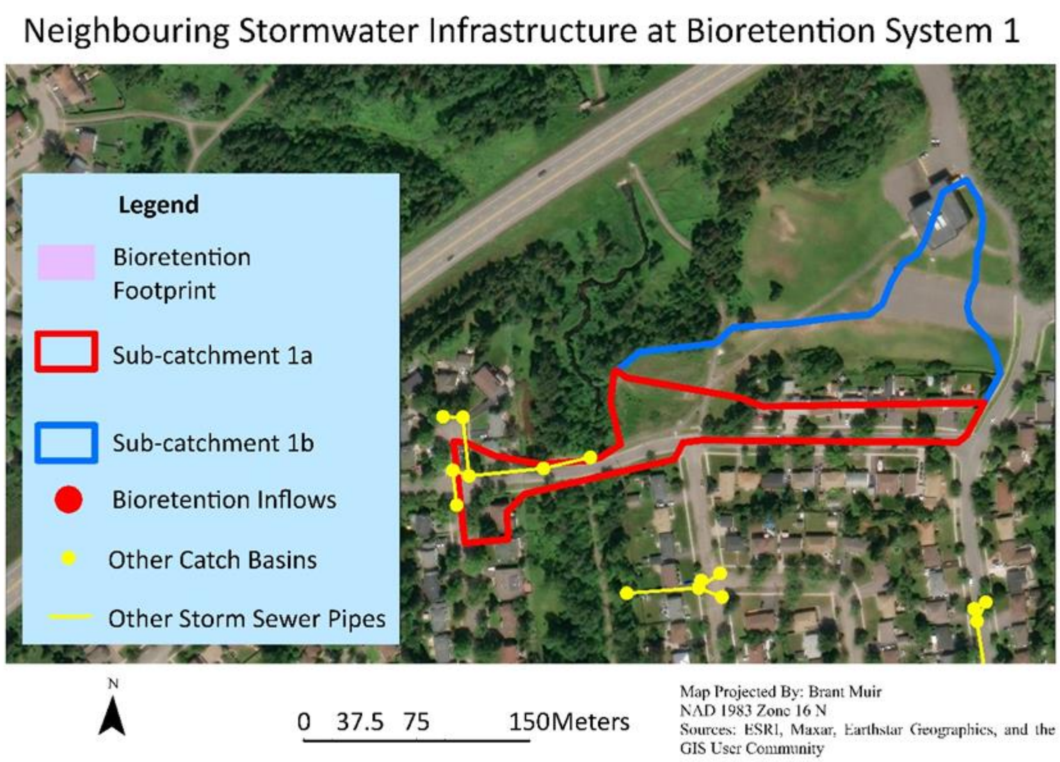


Figure 3-3 Neighbouring stormwater infrastructure at Bioretention System 1.

There is also a traditional stormwater outfall that discharges into Boulevard Lake without any quality or quantity control, approximately 16 metres west of the outflow from Bioretention System 2 (Figure 3-4).

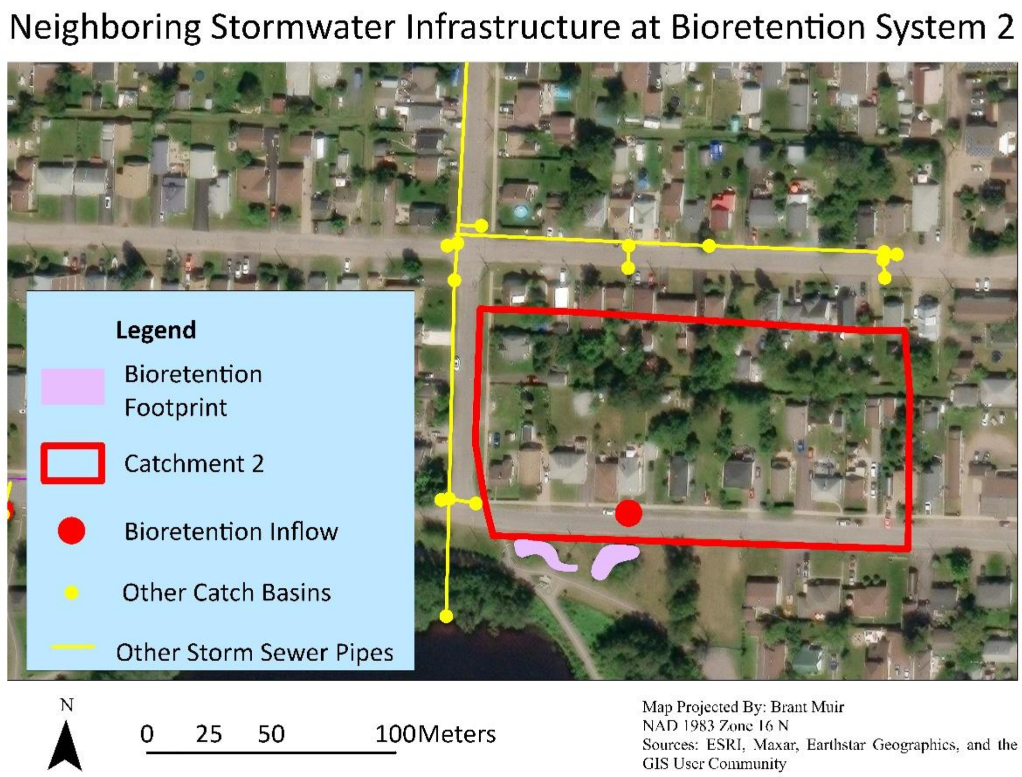


Figure 3-4 Neighbouring stormwater infrastructure at Bioretention System 2.

Similarly, there are several other catch basins and associated stormwater infrastructure to the east of Bioretention System 3 that discharges untreated stormwater into Boulevard Lake without any quality or quantity control (Figure 3-5).

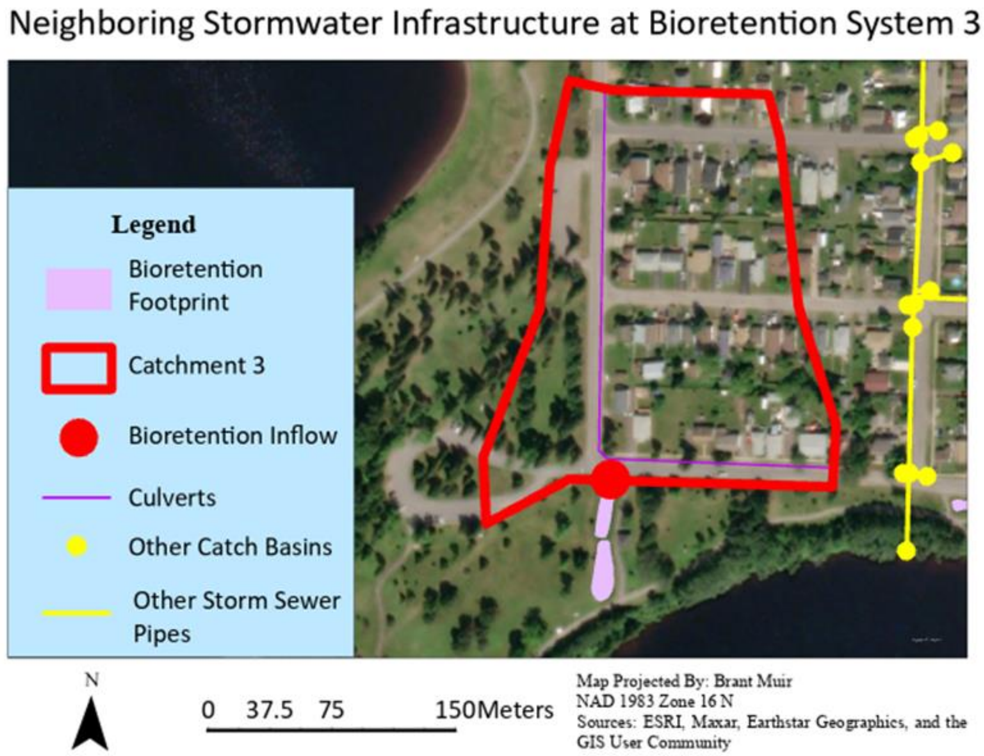


Figure 3-5 Neighbouring stormwater infrastructure at Bioretention System 3.

3.2.2 Study design

I assessed the performance of two bioretention systems at reducing peak and total stormwater discharge, and the performance of three bioretention systems at improving stormwater quality over two snow-free seasons (April through November). To assess water quantity performance, HOBO U20/U20L water level loggers were installed in catch basins (and/or swales) entering the bioretention system inflow(s) and in the storage layer measuring the water level exiting the outflow(s) of Bioretention System 1 and Bioretention System 2. These loggers measured water level and water temperature over 15-minute intervals (See Section 2.3).

Discrete discharge measurements were collected using a bucket and stopwatch and used to create a series of rating curves to determine the relationships between water level and discharge for stormwater entering the inflows (Inflow 1a and Inflow 1b) and the outflow of Bioretention System 1, and for the inflow and outflow of Bioretention System 2. These relationships were used to determine the peak and total discharge that entered and exited Bioretention Systems 1 and 2. The access pipe to the internal water storage layer of Bioretention System 3 had an elbow that prevented a logger from reaching the storage layer. Therefore, discharge was not assessed at this site, and Bioretention System 3 was used to measure water quality only. The performance of bioretention systems was assessed by comparing the peak and total discharge entering and exiting Bioretention Systems 1 and 2.

Within the receiving stream, paired sampling sites were established directly upstream and six metres downstream of both the reference outfall and Bioretention System 1, to evaluate instream responses to untreated and treated stormwater inputs (Figure 3-6). This six-metre distance was chosen to allow mixing within the stream while avoiding confounding effects from downstream infrastructure (e.g., road crossings, other stormwater outfalls, culverts) at the bioretention system location. Upstream and downstream water quality comparisons were not conducted at Bioretention Systems 2 and 3 because both systems discharged directly into Boulevard Lake, a lentic reservoir created by damming the Current River for historical electric power generation, rather than stream reaches with defined upstream and downstream sample locations.

Locations of Bioretention System 1 and the Reference Outfall

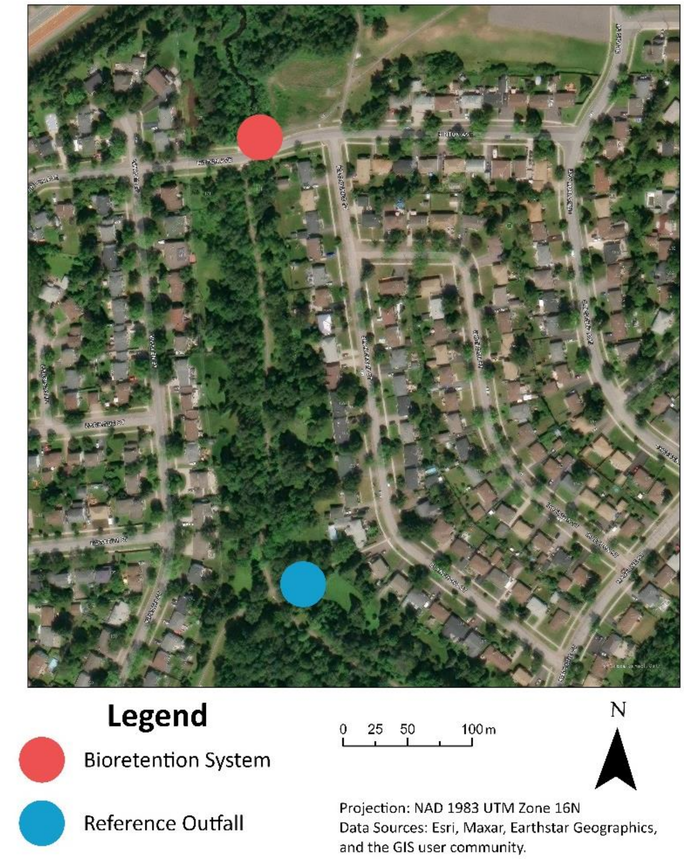


Figure 3-6 Locations of Bioretention System 1 and the Reference Outfall.

3.2.3 Water quantity analysis

I evaluated water quantity performance during 87 rainfall events over two years (August 5 to November 10, 2022; April 28 to November 8, 2023; and April 4 to August 16, 2024). HOBO U20/U20L pressure loggers (Onset Computer Corporation, Bourne, MA, USA) were installed in the catch basins at the Reference Inflow, Inflow 1a, and Inflow 2; in the swales draining to Inflow 1b and Inflow 2; and in the storage layers of Bioretention Systems 1 and 2. Storage-layer water levels were used to quantify discharge from Outflows 1 and 2. I excluded Bioretention System 3 from the water temperature measurements due to the site-specific constraints associated with reliable logger deployment (as discussed in 2.2). All loggers recorded water levels at a 15-minute interval. Discrete discharge measurements were collected using a 10 L bucket and a stopwatch to develop rating-curve relationships. These measurements were then aligned with concurrent logger depths to generate stage-discharge calibration pairs.

Logger data were offloaded approximately every 30 days using HOBOware Pro v3.6.2. Barometric pressure corrections were applied using the Barometric Compensation Assistant, which included an additional HOBO U20/U20L suspended in air near Bioretention System 1. Rating curves were then developed for each monitored location using methods appropriate to the site's hydraulic configuration. The Reference Inflow and Inflow 2 were described using exponential stage-discharge relationships. Inflow 1a was represented using a physically based 14-inch circular-pipe model combining Manning's equation and an orifice equation, while Inflow 1b used a linear regression based on observed bucket measurements. Combined inflow to Bioretention System 1 (Inflow 1c) was computed as the sum of Inflows 1a and 1b. Outflow 1 and Outflow 2 were modelled using exponential relationships between storage-layer water depth and outflow discharge, capped at the corresponding full-pipe discharge capacity.

Continuous discharge time series were generated by applying each rating curve to the full water level record. Rainfall events were delineated using logger timestamps, precipitation records, and a discharge hydrograph, with each event ending when the water level returned to baseline levels for at least 6 hours. For each rainfall event, peak discharge was defined as the maximum discharge recorded within the event window. Total event discharge (L) was computed by numerically integrating the discharge time series across the event duration. Peak discharge was calculated in litres per second (L/s), and total discharge was calculated in litres (L) and cubic metres (m^3) to determine water quantity reductions.

3.2.4 Water quality analysis

Nalgene™ stormwater samplers were suspended within catch basins entering the Reference Inflow, Inflow 1a, and Inflow 2, and placed in the swale entering Inflow 1b and Inflow 3 to capture the first litre of stormwater entering each bioretention system.

Stormwater samplers were also installed below the outflow pipe of each bioretention system. Each sampler contained a floating ball valve that sealed once one litre of water had been collected, ensuring that only the "first-flush" of stormwater entering the inflow and exiting the outflow of each bioretention system was sampled.

Nalgene™ samplers and CT2X water level/conductivity/temperature sensors (Seametrics Inc., Kent, WA, USA) were also affixed to steel t-posts at four instream locations: (1) directly upstream of the reference outfall, (2) approximately six metres downstream of the reference outfall, (3) directly upstream of the bioretention outflow, and (4) approximately six metres downstream of the bioretention outflow (Figure 3-2). The CT2X sensor was used to determine the time when the creek level rose by 2 cm, allowing instream samples to be collected relative to the time when effluent entered the creek at both the reference and bioretention sites (see Supplementary Material). Prior to each rainfall event, the instream samplers were adjusted so that their intakes were positioned two centimetres above the water surface. During these events, when the stream stage increased by two centimetres, these samplers collected a water sample at each instream location. The timing of these increases in stream stage and associated sample collection events is provided in the supplementary material. All samplers were checked during and after each event to confirm successful sample capture.

Immediately after sample collection, pH and conductivity were measured using a multiparameter probe (HI98129; Hanna Instruments, Woonsocket, RI, USA), and turbidity was measured using a portable turbidity meter (2020we; LaMotte Company, Chestertown, MD, USA). Samples were transported on ice and refrigerated within two hours for subsequent laboratory analysis. I used a minimum 72-hour antecedent dry period between qualifying rainfall events to ensure an adequate pollutant load for measurement and that the bioretention system had been drained from the previous event. Subsequent rainfalls that occurred less than 72 hours from the previous rainfall event were excluded from the water quality analysis.

I analyzed the concentrations of ammonia, nitrate, nitrite, orthophosphate, chloride, zinc, and iron using a 9300 photometer (YSI Inc., Yellow Springs, OH, USA; accuracy ± 1% T, where T = percent transmittance) for 2022 samples and a LaMotte SMART3 colorimeter (LaMotte Company, Chestertown, MD, USA; resolution 1% FS; accuracy ± 2% FS, where FS = full scale of the selected measurement range) for 2023 and 2024 samples. Test standards were used to verify the accuracy of both the photometer and colorimeter. Metal samples were neither filtered nor acid-digested prior to analysis; therefore, zinc and iron measurements represent whole-water concentrations that include dissolved, colloidal, and fine particulate-associated fractions.

Colorimetric analyses using the 9300 photometer and SMART3 colorimeter followed the manufacturer-specified reagent procedures and pre-programmed instrument methods for each analyte. For the YSI 9300 photometer, manufacturer methods included the indophenol method for ammonia, Nitratest reduction/diazonium method for nitrate, Nitricol diazonium method for nitrite, Chloridol silver nitrate turbidity method for chloride, low-range phosphate colorimetric method for orthophosphate, Zincon method for zinc, and the manufacturer-specified iron colorimetric method for iron. For the LaMotte SMART3 colorimeter, analyses followed the corresponding LaMotte pre-programmed test methods and reagent systems for each analyte (see Supplementary Materials).

Suspended solids were measured gravimetrically using filtration and drying. Concentrations were measured by filtering collected water through 2 µm pre-combusted and pre-weighed glass fibre filters, which were then dried at 104 °C for 1 hour. The balance used to weigh dried filters (Denver Instruments, readability 0.001 g) was calibrated prior to each use. Dissolved organic carbon (DOC) concentrations were measured by filtering samples through 2 µm glass fibre filters, acidifying the filtrate with 2 mL of 85% phosphoric acid per 30 mL of sample, and analyzing them at the USDA Forest Service Northern Research Station (Grand Rapids, MN). DOC was quantified using a Shimadzu TOC-L CPH total organic carbon analyzer (Shimadzu Corporation, Kyoto, Japan) in 2022 and 2023 and a Shimadzu TOC-L CSH analyzer in 2024. Both DOC analyzers were identical in functionality and used high-temperature catalytic combustion oxidation with non-dispersive infrared detection to quantify organic carbon concentrations. The only difference between the CPH and CSH models was that the CPH model is operated using a PC, whereas the CSH model was a standalone unit. ERA and sucrose checks were used to verify the accuracy of both models.

Water quality measurements from the influent and effluent of each bioretention system were evaluated relative to established federal (Canadian Council of Ministers of the Environment (CCME)) and provincial (Provincial Water Quality Objectives (PWQO)) guidelines for the protection of aquatic life. For parameters with established guidelines (e.g., pH, chloride, phosphate, nitrate, nitrite, and metals), concentrations were compared directly to these thresholds. For ammonia, guideline exceedances were calculated using temperature- and pH-dependent CCME criteria. For parameters without defined guidelines (e.g., turbidity and suspended solids), ecological benchmarks derived from the literature were used. The frequency of exceedances was used to assess the performance of bioretention systems in reducing ecological risk. Pollutant loads were not calculated because the water quality samples represented first-flush concentrations rather than event-mean or flow-weighted composite concentrations. Although total event discharge was measured for Bioretention Systems 1 and 2, multiplying the total event volume by first-flush concentrations would assume that early-event concentrations represented the full runoff event. This assumption was not considered defensible because stormwater pollutant concentrations can vary substantially over the duration of the runoff event. Therefore, water quality treatment performance was evaluated using concentration-based comparisons, while runoff reductions were reported separately.

3.2.5 Statistical analyses

All statistical analyses were conducted in R Studio (version 2026.04.0).

To evaluate hydraulic differences between catchments draining into the bioretention systems, I assessed statistical differences in peak and total discharge entering each inflow. I assessed the normality of residuals using the Anderson-Darling test and the homogeneity of variances using Levene's test. When assumptions of normality of residuals or homogeneity of variances were violated, a transformation sweep was conducted that tested identity, log, log10, Box-Cox, Yeo-Johnson, rank-inverse-normal, square-root, cube-root, and inverse transformations, along with constant offsets and IQR-based outlier removal. When assumptions were met, one-way ANOVA was applied. When variances were unequal, Welch's ANOVA was used. When the assumption of normality of residuals could not be met following transformations, a non-parametric Kruskal-Wallis test was used. Significant ANOVA/Kruskal-Wallis results were followed by Tukey's HSD (for one-way ANOVA), Games-Howell (for Welch's ANOVA), or Dunn's (for Kruskal-Wallis test) post-hoc tests.

To assess bioretention performance, the peak and total event discharge and water quality results at Inflow 1a were paired with the same suite of parameters from Outflow 1. Similarly, peak and total discharge, and water quality results from Inflow 2 were paired with the same results from Outflow 2, and water quality results from Inflow 3 were paired with those from Outflow 3. Analyses used event-matched datasets to preserve the paired structure. For each inflow-outflow pair, transformations were applied to meet the assumption of normality of paired differences. When the assumption of normality of paired differences was met, paired t-tests were used; otherwise, Wilcoxon signed-rank tests, with the test statistic reported as V, were used to assess differences in peak discharge, total discharge, and water quality parameters between the inflow and outflow at each site.

Treatment performance was expressed as a reduction relative to inflow conditions to standardize differences between influent and effluent discharge, concentrations, and temperatures among sites. Standardizing these differences as reductions allowed direct comparison of treatment efficiency between the bioretention systems. Reductions in peak and total discharge were calculated for Bioretention Systems 1 and 2 using Equation 1.

Equation 1)

$$Peak/Total\ Discharge\ Reduction = \frac{Peak/Total\ \ Discharge_{Inflow} -\ Peak/Total\ Discharge_{Outflow}}{Peak/Total\ Discharge_{Inflow}} \times\ 100\%$$

Similarly, reductions in contaminant concentrations for Bioretention Systems 1, 2, and 3 were calculated using Equation 2.

Equation 2)

$$Concentration\ Reduction = \frac{Concentration_{Inflow} -\ Concentration_{Outflow}}{Concentration_{Inflow}} \times\ 100\%$$

Reductions in temperature and pH were calculated as absolute reductions, using Equation 3.

Equation 3)

$$Temperature/pH\ \mathrm{Reduction}\ = Temperature/pH_{Inflow} - Temperature/pH_{Outflow}$$

To assess whether bioretention system performance at reducing peak and total discharge and event maximum water temperatures differed between bioretention systems, differences in the reductions in peak, total discharge, and temperature reductions between Bioretention System 1 and Bioretention System 2 were assessed using a Student’s *t*-test. When assumptions of normality and/or homogeneity of variances were not met, transformations were applied. For parameters in which assumptions were still not met after transformations, a Wilcoxon signed-rank test was used to assess differences in peak and total discharge and in temperature reductions between Bioretention System 1 and Bioretention System 2.

To assess whether the performance of the bioretention systems at reducing contaminant concentrations differed, reductions in contaminant concentrations from Bioretention Systems 1, 2, and 3 were compared using one-way ANOVAs (for parameters that met assumptions of normality of residuals and homogeneity of variances) and Kruskal-Wallis tests (for parameters that did not meet ANOVA assumptions following transformations). Significant omnibus tests (ANOVA or Kruskal-Wallis) were followed by appropriate post hoc tests (Tukey for ANOVA; Dunn for Kruskal-Wallis) to identify which sites showed significantly different reductions.

To assess downstream water quality responses to bioretention system treatment, I first compared peak discharge, total discharge, and contaminant concentrations between 1) the inflow off the roadway (Reference Inflow) at the reference site, which discharges directly into the stream with no quantity or quality treatment; 2) the inflow off the roadway into the bioretention system (Inflow 1a) before receiving quantity or quality treatment; and, 3) the outflow from the bioretention system (Outflow 1) discharging into the stream following quantity and quality treatment. Assumptions of normality (Anderson-Darling) and sphericity (Mauchly's test) were evaluated, and repeated-measures ANOVA was applied with Site (Reference Inflow, Inflow 1a, Outflow 1) specified as a within-subject factor. A Benjamini-Hochberg procedure was applied to correct for multiple comparisons. When the ANOVA was significant, Tukey-adjusted pairwise comparisons identified differences among sites (Reference Inflow vs. Inflow 1a; Reference Inflow vs. Outflow 1; Inflow 1a vs. Outflow 1). The results of these ANOVAs were used to assess differences in peak discharge, total discharge, and contaminant concentrations entering the creek from the untreated reference outfall, entering the bioretention system before treatment, and entering the creek from the treated bioretention system outflow.

To evaluate whether short-term downstream water quality responses were detectable at both the untreated reference outfall and the treated bioretention system, I conducted a series of t-tests. This analysis was intended to test for event-scale differences in water chemistry within the stream and was not designed to quantify overall fish habitat condition or isolate bioretention effects from all other watershed-scale stressors. The assumption of normality of paired differences was tested using Anderson-Darling tests, and transformations were applied to meet this assumption (See Supplementary Material). For each water quality parameter, paired *t*-tests were used when paired differences were normally distributed, and Wilcoxon signed-rank tests (test statistic reported as V) were used to compare the difference in contaminant concentrations upstream and downstream of the untreated reference outfall (i.e., Δ Reference) to the difference in contaminant concentrations upstream and downstream of Bioretention System 1 (i.e., Δ Site1).

3.3 Results

3.3.1 Rainfall characteristics

Rainfall characteristics did not differ significantly between years, indicating consistent environmental conditions across the study period (Table 3-3). The mean event rainfall depth across the entire sampling period was 17 ± 4 mm, with 15 ± 6 mm in 2022, 10 ± 2 mm in 2023, and 34 ± 14 mm in 2024 (p = 0.15). The mean event rainfall intensity was 2.1 ± 0.6 mm/h in 2022, 2.2 ± 0.3 mm/h in 2023, and 1.7 ± 0.2 mm/h in 2024 (p = 0.9), with an overall mean rainfall intensity of 2.1 ± 0.2 mm/h. The mean antecedent dry period (ADP) was 7 ± 1 days in 2022, 6 ± 1 days in 2023, and 7 ± 2 days in 2024 (p = 0.75), with an overall mean ADP of 6.4 ± 0.5 days. The ADP, total rainfall depth, maximum intensity, and mean rainfall intensity for each event are provided in the Supplementary Material.

Table 3-3 Statistical results from comparisons in rainfall characteristics between sample years.

	ANOVA Result
Mean Event Rainfall Depth	$F(2, 43) = 2$, $p = 0.15$
Mean Event Rainfall Intensity	$F(2,43) = 0.1$, $p = 0.9$
Mean Antecedent Dry Period (ADP)	$F(2, 43) = 0.29$, $p = 0.75$

3.3.2 Water quantity performance

Peak and total discharge entering the bioretention systems differed significantly among inflow sites, indicating substantial variability in stormwater inputs between the bioretention systems (Table 3-4; Figure 3-3). Over 87 rainfall events, stormwater was fully retained by Bioretention System 1 over 43 events and by Bioretention System 2 over 70 events. The mean peak discharge from Inflows 1a, 1b, and 2 was 42.4 ± 1.6 L/s, 1.9 ± 0.2 L/s, and 16.8 ± 0.53 L/s, respectively ($p < 0.0001$), with higher peak discharge from 1a than 1b ($p < 0.0001$) and 2 ($p < 0.0001$), and higher peak discharge from Inflow 2 than from Inflow 1b ($p < 0.0001$) (Figure 3-3). The mean total discharge from Inflows 1a, 1b, and 2 was 754 ± 48 m^3, 32 ± 5 m^3, and 268 ± 16 m^3, respectively ($p < 0.0001$), with significantly higher total discharge from Inflow 1a than Inflow 1b ($p < 0.0001$) and Inflow 2 ($p < 0.0001$), and significantly higher total discharge from Inflow 2 than from Inflow 1b ($p < 0.0001$). Due to the bioretention system design (discussed in 2.2) discharge was not measured from Bioretention System 3.

Table 3-4 Statistical results from comparisons in peak and total discharge from bioretention systems inflows during rainfall events.

	Kruskal-Wallis Result
Peak Discharge	$\chi^2(2) = 142$, $p < 0.0001$
Total Discharge	$\chi^2(2) = 102$, $p < 0.0001$

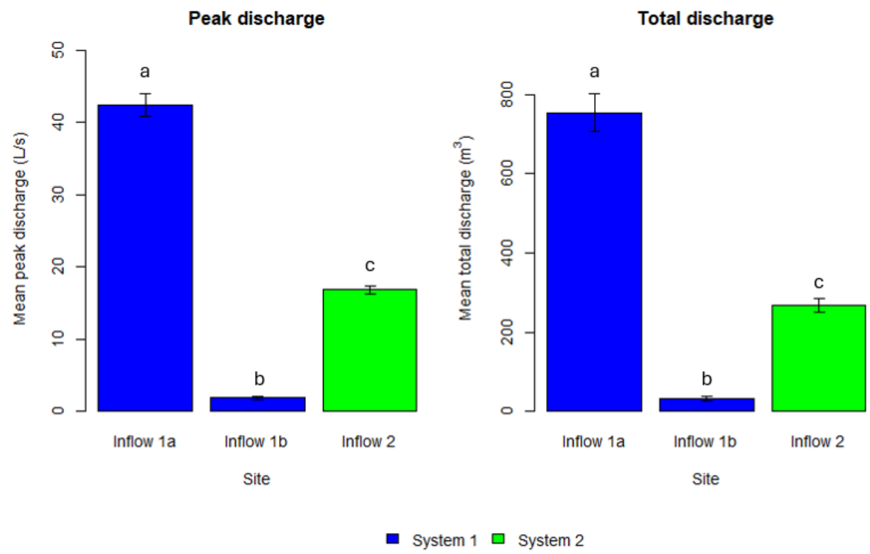


Figure 3-7 Peak (left) and total (right) discharge entering the two inflows (Inflow 1a and Inflow 1b) into Bioretention System 1 and entering the inflow (Inflow 2) into Bioretention System 2.

Bioretention systems 1 and 2 successfully reduced the peak and total discharge, indicating strong hydrologic performance across both systems (Table 3-5; Figure 3-7). Across events, the mean peak discharge decreased from 43 ± 2 L/s entering Bioretention System 1 to 9.4 ± 0.4 L/s exiting Bioretention System 1 ($p < 0.0001$) and decreased from 17 ± 1 L/s entering Bioretention System 2 to 2.7 ± 0.5 L/s exiting Bioretention System 2 ($p < 0.0001$). The total discharge volumes decreased from 771 ± 49 m^3 entering Inflow 1 to 169 ± 13 m^3 exiting Outflow 1 ($p < 0.0001$) and from 268 ± 16 m^3 entering Inflow 2 to 37 ± 10 m^3 exiting Outflow 2 ($p < 0.0001$). Peak discharge was reduced by 87 ± 2% at Site 1 and 86 ± 2% at Site 2, while total discharge volumes were reduced by 88 ± 2% at Site 1 and 87 ± 2% at Site 2 (Figure 3-8). There was no significant difference in peak ($p = 0.84$) or total ($p = 0.52$) discharge reductions between the two bioretention systems. Additional raw-time series figures showing influent and effluent discharge from each sampling date are provided in the Supplementary Materials.

Table 3-5 Statistical results from comparisons in peak and total discharge from bioretention systems inflows during rainfall events.

	Peak Discharge	Total Discharge
Site 1 Comparison	$t(22) = 65$, $p < 0.0001$	$V = 703$, $p < 0.0001$
Site 2 Comparison	$t(8) = 14$, $p < 0.0001$	$t(8) = 5.3$, $p < 0.0001$
Percent Reduction	$V = 174$, $p = 0.84$	$V = 190$, $p = 0.52$

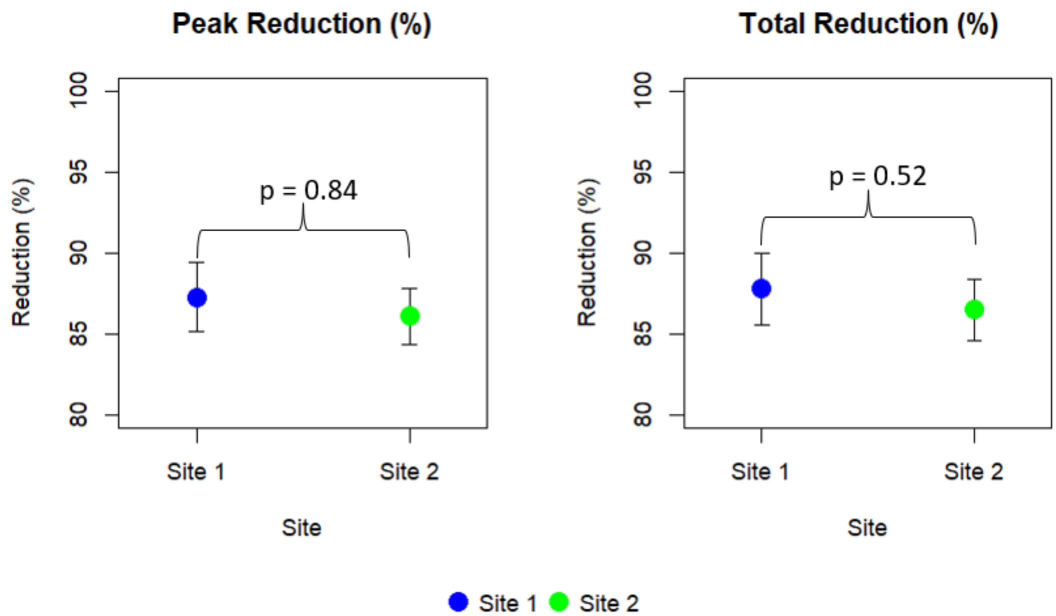


Figure 3-8 Reductions (%) in peak and total discharge from two bioretention systems across 87 rainfall events.

3.3.3 Water quality performance

3.3.3.1 Inflow comparisons

Water quality differed significantly among inflow sites, indicating that variability in influent conditions likely influenced bioretention treatment performance. Event maximum water temperatures ($p = 0.02$), pH ($p = 0.01$), conductivity ($p < 0.0001$), nitrate ($p < 0.0001$), and zinc ($p = 0.004$) concentrations were significantly different between inflow sites (Figure 3-9; Table 3-6). Event maximum water temperatures from Inflow 2 were significantly higher than from Inflow 1a ($p = 0.006$). pH from Inflow 2 was significantly higher than from Inflow 1a ($p = 0.005$) and Inflow 3 ($p = 0.03$). Conductivity from Inflow 3 was significantly lower than from Inflow 1a ($p < 0.0001$), Inflow 1b ($p <$

0.001), and Inflow 2 ($p = 0.01$). Nitrate concentrations from Inflow 1b were significantly higher than from Inflow 1a ($p < 0.0001$), Inflow 2 ($p < 0.0001$), and Inflow 3 ($p < 0.0001$). Zinc concentrations from Inflow 1b were significantly lower than from Inflow 1a ($p = 0.001$), and Inflow 3 ($p = 0.04$), and zinc concentrations from Inflow 1a were significantly higher than from Inflow 2 ($p = 0.04$). All other pairwise comparisons were not significantly different. These differences in influent water quality provide context for the treatment performance of the bioretention systems (3.3.3). As the reference site does not receive treatment, it was excluded from these comparisons. Additionally, due to the bioretention system design (discussed in 2.2), which prevented deployment of a water level/temperature logger, water temperature measurements were not collected at Bioretention System 3.

Table 3-6 Statistical results from comparisons in water quality parameters from bioretention systems inflows during rainfall events.

Parameter	ANOVA Result
Event maximum temperature	$F(2, 189) = 5$, $p = 0.02$
pH	$F(3, 46) = 5$, $p = 0.01$
Conductivity	$F(3, 46) = 11$, $p < 0.0001$
Turbidity	$F(3, 46) = 1.8$, $p = 0.25$
Suspended solids	$F(3, 40) = 1.2$, $p = 0.42$
Dissolved organic carbon	$F(3, 44) = 3$, $p = 0.08$
Ammonia	$F(3, 44) = 0.69$, $p = 0.56$
Nitrate	$F(3, 33) = 18$, $p < 0.0001$
Nitrite	$F(3, 40) = 2$, $p = 0.22$
Phosphate	$F(3, 43) = 0.73$, $p = 0.56$
Zinc	$F(3, 31) = 6.9$, $p = 0.004$
Iron	$F(3, 46) =1.3$, $p = 0.4$
Chloride	$F(3, 40) = 0.89$, $p = 0.53$

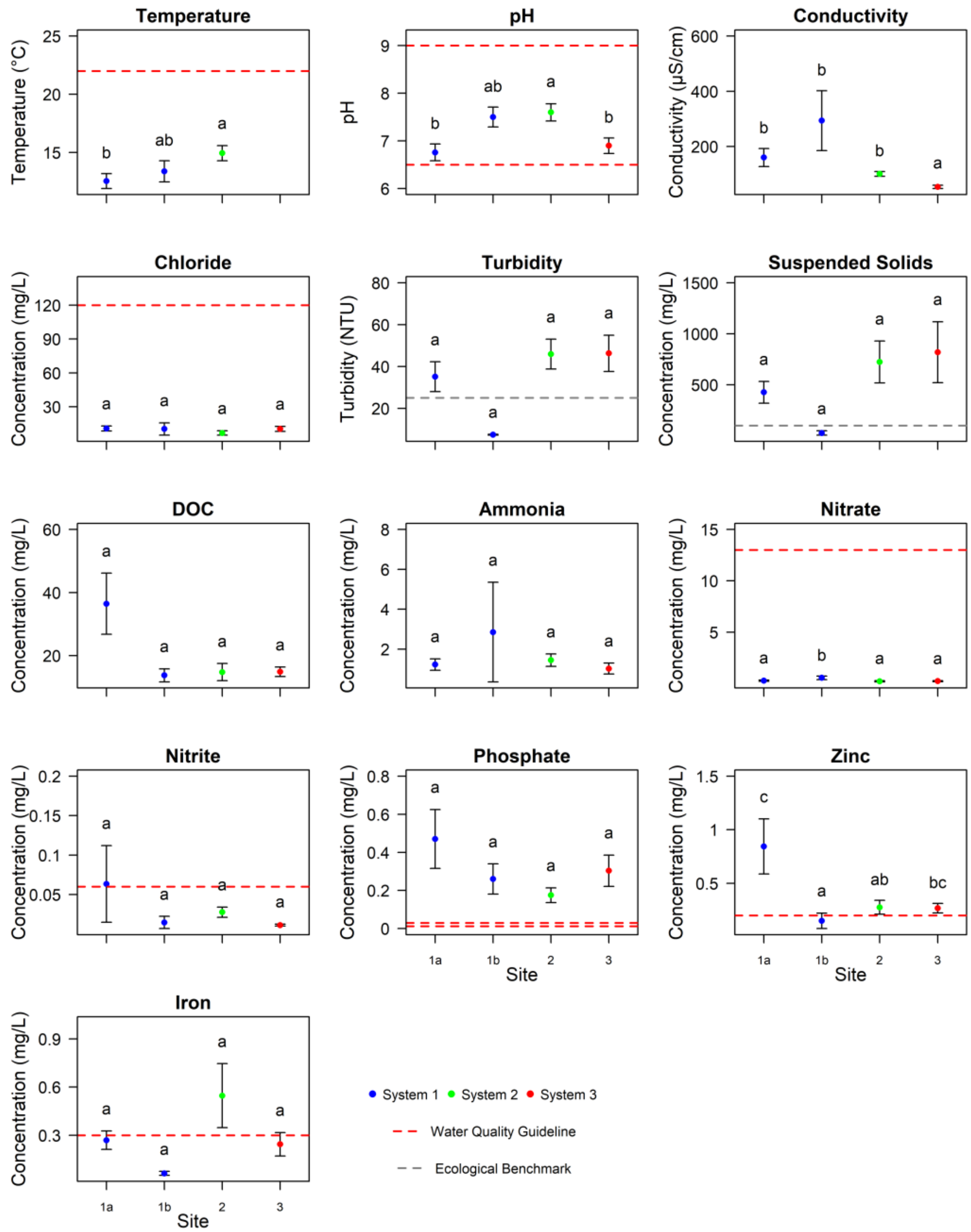


Figure 3-9 Water quality measurements from the inflows of three bioretention systems during rainfall events. Red dashed lines indicate the applicable water quality guidelines, and grey dashed lines indicate literature-derived ecological benchmarks (Table 3-1). Differences in letters indicate significant differences ($\alpha = 0.05$). Temperature data were unavailable for Site 3.

3.3.3.2 Comparison of inflow and outflow water quality

The bioretention systems consistently reduced turbidity and suspended solids concentrations but showed variable performance for other water quality parameters, including site-dependent increases and decreases in effluent concentrations (Table 3-7; Figure 3-10). While all three bioretention systems significantly reduced turbidity (Bioretention System 1: p = 0.005; Bioretention System 2: p = 0.02; Bioretention System 3: p = 0.01) and suspended solids concentrations (Bioretention System 1: p = 0.0001; Bioretention System 2: p = 0.03; Bioretention System 3: p = 0.005, decreases in water temperatures, pH, conductivity, and ammonia and zinc concentrations were site dependent. Water temperatures decreased at Bioretention System 1 (p = 0.001). pH increased at Bioretention System 1 (p = 0.04). Conductivity increased from the effluent of all three bioretention systems, with a significant increase from the effluent of Bioretention System 3 (p = 0.001). DOC concentrations decreased at Bioretention Systems 1 (p = 0.048) and 3 (p = 0.01). Ammonia concentrations decreased at Bioretention Systems 1 (p = 0.03) and 2 (p = 0.03). Zinc concentrations decreased at Bioretention System 1 (p = 0.02). All other outflow measurements were not significantly different from their respective inflows. Additional supporting figures showing raw water quality measurements by sampling date are provided in the Supplementary Materials.

Table 3-7 Statistical results from comparisons in outflow water chemistry relative to inflow water chemistry during rainfall events.

Parameter	Site	Test Result
Temperature	1	$t(24) = -4.4$, $p = 0.001$
	2	$W = 12$, $p = 0.07$
pH	1	$t(26) = 2.4$, $p = 0.048$
	2	$t(12) = -1.4$, $p = 0.32$
	3	$W = 29$, $p = 0.38$
Conductivity	1	$t(26) = 3.8$, $p = 0.004$
	2	$W = 77$, $p = 0.08$
	3	$t(13) = 5.1$, $p = 0.001$
Chloride	1	$W = 212$, $p = 0.46$
	2	$W = 55$, $p = 0.29$
	3	$W = 72$, $p = 0.38$
Turbidity	1	$t(26) = -3.6$, $p = 0.005$
	2	$t(12) = -4.2$, $p = 0.02$
	3	$t(13) = -3.5$, $p = 0.01$
Suspended Solids	1	$W = 0$, $p = 0.0001$
	2	$t(11) = -3.6$, $p = 0.03$
	3	$W = 0$, $p = 0.005$
Dissolved Organic Carbon (DOC)	1	$W = 88$, $p = 0.048$
	2	$t(11) = 0.02$, $p = 1$
	3	$t(13) = -3.4$, $p = 0.01$
Ammonia	1	$W = 99$, $p = 0.049$
	2	$W = 6$, $p = 0.03$
	3	$W = 43$, $p = 0.61$
Nitrate	1	$W = 116$, $p = 0.11$
	2	$t(12) = 1.4$, $p = 0.26$
	3	$t(13) = 1.5$, $p = 0.34$
Nitrite	1	$W = 77$, $p = 0.56$
	2	$W = 6$, $p = 0.11$
	3	$W = 14$, $p = 0.61$
Phosphate	1	$W = 212$, $p = 0.58$
	2	$W = 63$, $p = 0.11$
	3	$W = 46$, $p = 0.61$
Zinc	1	$W = 80$, $p = 0.02$
	2	$W = 45$, $p = 1$
	3	$t(13) = 1.3$, $p = 0.38$
Iron	1	$W = 213$, $p = 0.58$
	2	$W = 24$, $p = 0.21$
	3	$t(13) = -0.98$, $p = 0.45$

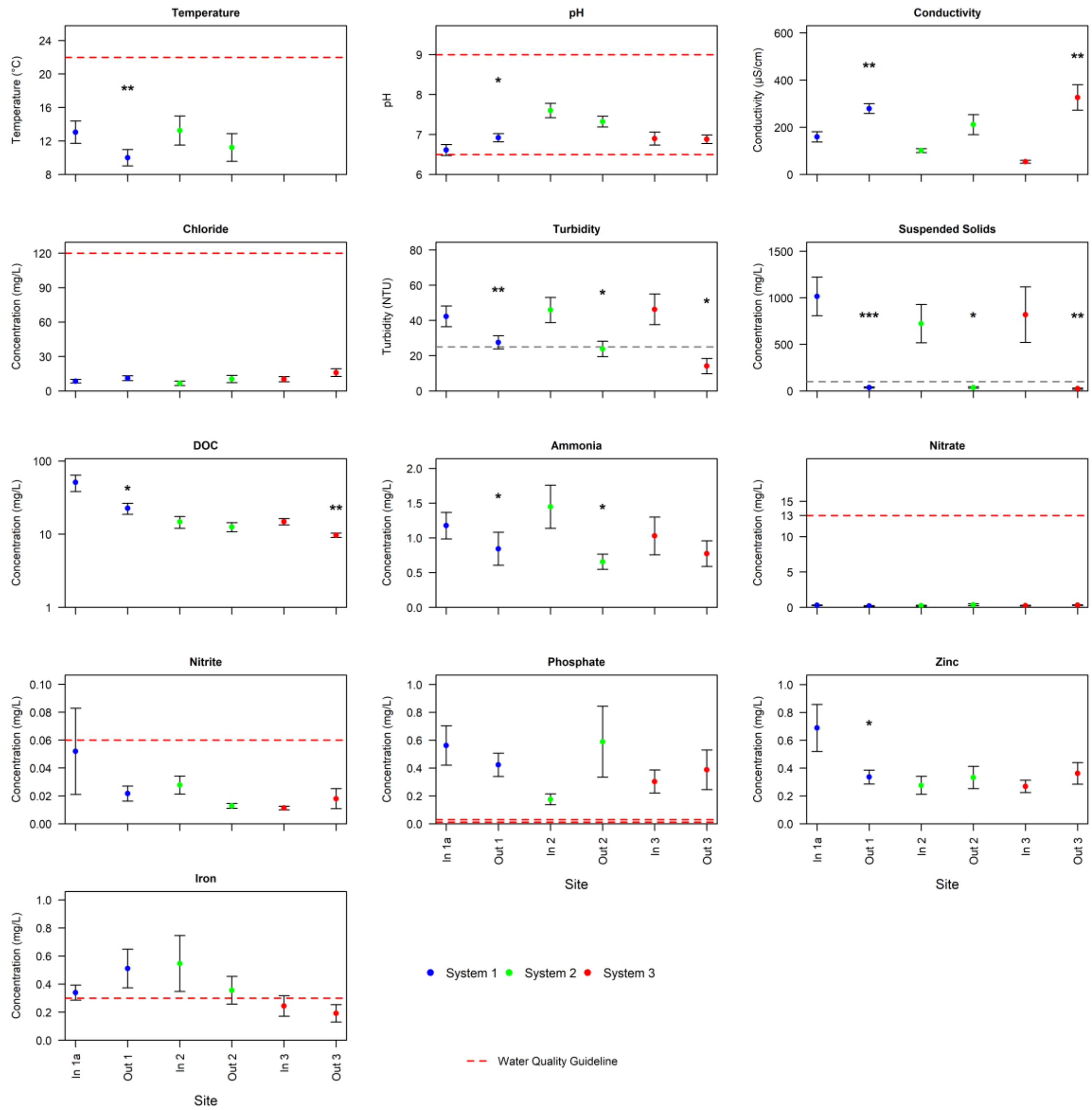


Figure 3-10 Comparison of water quality entering the inflows and exiting the outflows of three bioretention systems during rainfall events.
Red dashed lines indicate the applicable water quality guidelines, and grey dashed lines indicate literature-derived ecological benchmarks (Table 3-1). * indicates significance at α = 0.05 and ** indicates significance at α = 0.01.

The bioretention systems reduced the frequency of water quality guideline and ecological threshold exceedances for pH, turbidity, suspended solids, ammonia and nitrite, and to a lesser extent, water temperature, indicating improved protection of aquatic life (Table 3-8). The pH entering the inflows of Bioretention Systems 1, 2 and 3, respectively, was below 6.5 in 48%, 8% and 21% of events, and the pH exiting the outflows of 1, 2, and 3, respectively, was below 6.5 in 21%, 8%, and 14% of events.

Turbidity entering the inflows of Bioretention Systems 1, 2 and 3, respectively, exceeded ecological thresholds relevant to fish habitat in 70, 85, and 71% of events, while the turbidity exiting the outflows of Bioretention Systems 1, 2 and 3, respectively, exceeded ecological thresholds in 48, 39, and 7% of events. Suspended solids concentrations from the inflows of Bioretention Systems 1, 2 and 3 exceeded ecological benchmarks in 81, 83, and 57% of events, while the suspended solids concentrations from the outflow of Bioretention System 1 exceeded ecological thresholds in 4% of events, and the suspended solids concentrations from the outflows of Bioretention Systems 2 and 3 did not exceed ecological thresholds.

The bioretention systems reduced the frequency of temperature, ammonia, and nitrite exceedances, with particularly strong reductions for ammonia and nitrite (Table 3-8). The water temperatures from the inflow of Bioretention System 1 exceeded ecological benchmarks in 8% of events, while water temperatures from the outflow of Bioretention System 1 exceeded ecological benchmarks in 4% of events. There were no water temperature exceedances from the inflow or outflow of Bioretention System 2, and water temperature was not measured at Bioretention System 3. Ammonia concentrations entering Bioretention Systems 1 and 2 exceeded temperature- and pH-dependent CCME water quality guidelines in 20% and 62% of events, but only ammonia concentrations exiting Bioretention Systems 1 and 2 exceeded the guideline in 4 and 23% of events. Nitrite concentrations from the inflows of Bioretention Systems 1 and 2 exceeded CCME water quality guidelines in 11% and 15% of events, while concentrations from the outflow of Bioretention System 1 only exceeded guidelines in 7% of events, and concentrations from the outflow of Bioretention System 2 did not exceed guidelines for any event.

Despite improvements in some water quality parameters, the bioretention systems did not reduce the frequency of water quality guideline exceedances for phosphate, zinc, iron, and nitrate, indicating limited treatment effectiveness for these pollutants (Table 3-8). Phosphate concentrations from the inflows of Bioretention Systems 1, 2, and 3 exceeded CCME water quality guidelines for 96%, 62%, and 71% of rainfall events, while phosphate concentrations from the outflows exceeded CCME water quality guidelines for 93%, 92%, and 79% of outflow events. Zinc concentrations from the inflow of Bioretention System 1 exceeded CCME water quality guidelines for 96%, while zinc concentrations from the inflows of Bioretention Systems 2 and 3, and from the outflows of Bioretention Systems 1, 2, and 3 exceeded CCME water quality guidelines on every event. Iron concentrations from the inflows of Bioretention Systems 1, 2 and 3 exceeded CCME water quality guidelines for 41%, 39%, and 36% of events, while concentrations from the outflows of Bioretention Systems 1, 2, and 3 exceeded water quality guidelines for 44%, 39% and 21% of rainfall events. There were no exceedances in nitrate concentrations from the inflow or outflow of any of the bioretention systems.

Table 3-8 Frequency of rainfall events in which the inflow and outflow of the bioretention systems exceed water quality guidelines and ecological benchmarks.

Guideline/Threshold	Site	Number of Inflow Events not Met	% of Inflow Events not Met	Number of Outflow Events not Met	% of Outflow Events Not Met
Temperature	1	2	8	1	4
Greater than 22ºC	2	0	0	0	0
pH	1	13	48	6	22
Below 6.5	2	1	7.7	1	7.7
	3	3	21	2	14
Turbidity	1	19	70	13	48
Greater than 25 NTU	2	11	85	5	39
	3	10	71	1	7
Suspended solids	1	21	81	1	4
Greater than 100 mg/L	2	10	83	0	0
	3	8	57	0	0
Ammonia	1	5	20	1	4
Dependent on pH and temperature	2	8	62	3	23
Nitrate	1	0	0	0	0
> 13 mg/L	2	0	0	0	0
	3	0	0	0	0
Nitrite	1	3	11	2	7
	2	2	15	0	0
	3	0	0	1	7
Phosphate	1	26	96	25	93
	2	8	62	12	92
	3	10	71	11	79
Zinc	1	26	96	27	100
	2	13	100	13	100
	3	14	100	14	100
Iron	1	11	41	12	44
	2	5	39	5	39
	3	5	36	3	21

3.3.3.3 Reductions in water quality parameters across sites.

Water temperature, suspended solids concentrations, and to a lesser extent, turbidity, showed the most consistent reductions across sites (Table 3-9; Figure 3-11). While Section 3.3.2 compared differences between inflow and outflow concentrations from each bioretention system, it did not compare performance between systems, and therefore, reductions in water quality parameters from the outflow relative to the inflow were used to compare relative treatment performance between systems. Expressing treatment performance as a percent reduction standardized differences between influent and effluent concentrations, allowing for a more direct comparison of treatment efficiency between the three bioretention systems. The maximum water temperatures from the outflows of both Sites 1a and 2 were lower than their respective inflows but reductions were not significantly different between sites (p = 0.59). Turbidity increased (i.e., negative reduction) slightly at Site 1a and decreased (i.e., positive reduction) at Sites 2 and 3 but following a Benjamini-Hochberg correction for multiple comparisons, these reductions were not significantly different between sites (p = 0.25).

In contrast, other water quality parameters showed variable or inconsistent changes in concentration across sites (Table 3-9; Figure 3-11). Changes in pH were small and not significantly different between sites (p = 0.49). Conductivity increased at all sites, but differences between sites were not significant (p = 0.28). Dissolved organic carbon (DOC) concentrations were reduced at Site 3 but increased at Sites 1a and 2, although these differences were not significant (p = 0.74). Ammonia concentrations increased at Sites 1 and 3 and decreased at Site 2, although differences in reductions between sites were not significant (p = 0.74). Nitrate (p = 0.51), phosphate (p = 0.46), zinc (p = 0.49), and chloride (p = 0.34) concentrations also increased at all sites, but increases were not significantly different between sites. Nitrite concentrations were slightly lower at Site 2 but higher at Sites 1a and 3, but differences were not significant (p = 0.46).

Table 3-9 Statistical results from comparisons in reductions between bioretention systems during rainfall events.

Parameter	Test Result
Temperature	$t(30) = -0.54$, $p = 0.59$
pH	$F(2, 33) = 1$, $p = 0.49$
Conductivity	$F(2, 33) = 3.4$, $p = 0.28$
Chloride	$F(2, 31) = 2.4$, $p = 0.34$
Turbidity	$F(2, 33) = 4.4$, $p = 0.25$
Suspended solids	$F(2, 32) = 2.7$, $p = 0.34$
Dissolved organic carbon (DOC)	$F(2, 32) = 0.3$, $p = 0.74$
Ammonia	$F(2, 33) = 0.31$, $p = 0.74$
Nitrate	$F(2, 32) = 0.87$, $p = 0.51$
Nitrite	$F(2, 33) = 1.6$, $p = 0.46$
Phosphate	$F(2, 33) = 1.5$, $p = 0.46$
Zinc	$F(2, 33) = 1.1$, $p = 0.49$
Iron	$F(2, 33) = 1.1$, $p = 0.49$

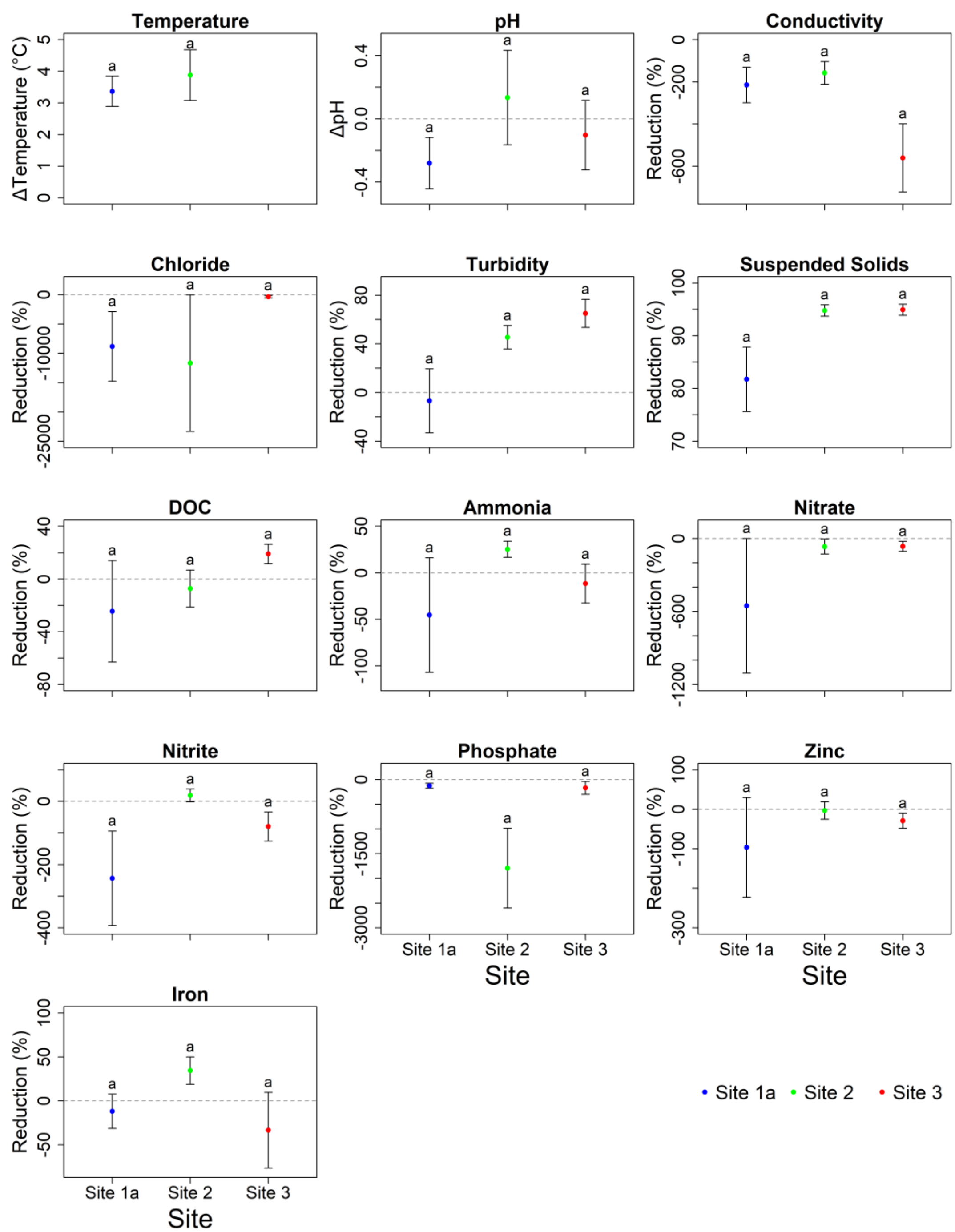


Figure 3-11 Reduction in water quality parameters from the outflow relative to the inflow of three bioretention systems.
Peak water temperature was not measured at Site 3. Letters indicate significance at $\alpha = 0.05$; shared letters indicate that the sites are not significantly different.

3.3.3.4 Peak discharge, total discharge and water quality entering the stream at the untreated reference inflow and the bioretention treatment site.

Peak and total discharge entering the stream from the untreated reference outfall (Reference Inflow) were compared with the discharge entering Bioretention System 1 before quantity treatment (Inflow 1a) and the discharge exiting the system after quantity treatment (Outflow 1) to evaluate differences in stormwater volumes entering the stream under treated and untreated conditions (Table 3-10). Peak discharge from the Reference Inflow, Inflow 1a, and Outflow 1 averaged 10 ± 2 L/s, 47 ± 2 L/s, and 9.2 ± 0.5 L/s, respectively ($p < 0.0001$), with significantly different peak discharge between sample locations. Inflow 1a had significantly higher peak discharge than both the Reference Inflow ($p < 0.0001$) and Outflow 1 ($p < 0.0001$), but the peak discharge from the Reference Inflow and Outflow 1 did not differ significantly ($p = 0.47$). Total discharge from the Reference Inflow, Inflow 1a, and Outflow 1 averaged 166 ± 25 m^3, 963 ± 54 m^3, and 173 ± 14 m^3, respectively, with significantly different total discharge between sample locations ($p < 0.0001$). Inflow 1a had significantly higher total discharge than the Reference Inflow ($p < 0.0001$) and Outflow 1 ($p < 0.0001$), but the total discharge from the Reference Inflow and Outflow 1 was not significantly different ($p = 0.28$).

Table 3-10 Statistical results from comparisons of peak and total discharge entering the stream at the untreated reference inflow and bioretention system.

Parameter	Test Result
Peak Discharge	$\chi^2 = 32$, $p < 0.0001$
Total Discharge	$F(2, 44) = 110$, $p < 0.0001$

Similarly, the water quality measurements from the untreated reference outfall (Reference Inflow), the bioretention system inflow prior to treatment (Inflow 1a), and the bioretention system outflow (Outflow 1) were compared to evaluate differences between untreated stormwater and stormwater treated through bioretention processes entering the same receiving stream (Table 3-11; Figure 3-12). While the bioretention system outflow had increased conductivity and reduced suspended solids concentrations relative to the bioretention inflow and reference inflow, most parameters showed no significant differences. Conductivity ($p < 0.0001$) from the bioretention system outflow was higher than the reference ($p < 0.0001$) and bioretention inflow ($p = 0.008$), while the reference and bioretention inflows were not significantly different ($p = 0.2$). Suspended solids concentrations ($p < 0.001$) from the bioretention system outflow were lower than the bioretention inflow ($p < 0.001$), while neither the reference and bioretention inflows differed ($p = 0.2$) nor did the reference inflow and bioretention outflow ($p = 0.05$). Although ammonia concentrations differed overall ($p = 0.02$), pairwise comparisons were not significant (reference inflow vs. bioretention inflow: $p = 0.98$; bioretention inflow vs. outflow: $p = 0.59$; reference inflow vs. bioretention outflow: $p = 0.07$). The pH ($p = 0.32$), turbidity ($p = 0.32$), and concentrations of dissolved organic carbon (DOC) ($p = 0.32$), nitrate ($p = 0.66$), nitrite ($p = 0.38$), phosphate ($p = 0.32$), zinc ($p = 0.47$), iron ($p = 0.32$), and chloride ($p = 0.47$) were not significantly different.

Table 3-11 Statistical results from comparisons of water quality parameters entering the stream at the untreated reference inflow and bioretention system.

Parameter	Test Result
pH	$F(2, 30) = 2.1$, $p = 0.32$
Conductivity	$F(2, 30) = 13$, $p < 0.0001$
Chloride	$F(2, 30) = 0.85$, $p = 0.47$
Turbidity	$F(2, 30) = 1.7$, $p = 0.32$
Suspended solids	$F(2, 28) = 9$, $p < 0.001$
Dissolved organic carbon (DOC)	$F(2, 21) = 1.7$, $p = 0.32$
Ammonia	$F(2, 21) = 6.8$, $p = 0.02$
Nitrate	$F(2, 22) = 0.43$, $p = 0.66$
Nitrite	$F(2, 23) = 1.3$, $p = 0.38$
Phosphate	$F(2, 20) = 1.6$, $p = 0.32$
Zinc	$F(2, 24) = 0.86$, $p = 0.47$
Iron	$F(2, 25) = 2.2$, $p = 0.32$

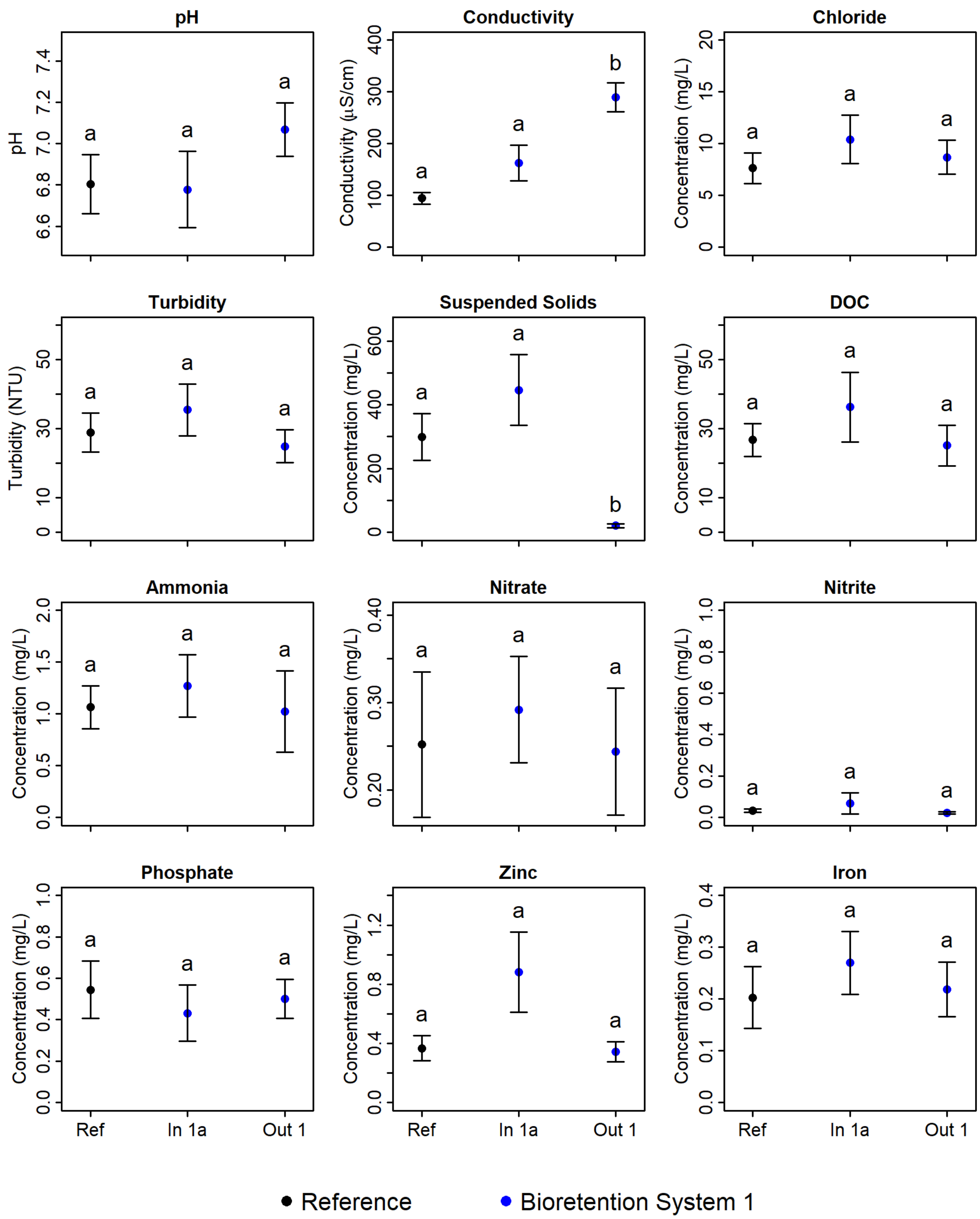


Figure 3-12 Water quality parameters entering the creek from the untreated reference outfall (Ref), entering the inflow into Bioretention System 1 (In 1a) and exiting the outflow (Out 1) from Bioretention System 1 during the spring melt.

3.3.4 Instream water quality response at the untreated reference site.

Water quality measurements taken upstream and downstream of the untreated reference outfall were compared to evaluate whether stormwater inputs altered downstream water quality. No significant differences in water quality measurements were observed between the upstream and downstream sample locations (Table 3-12). The mean time from when water entered the creek through the reference outfall to when the creek rose 0.5 cm to fill the Nalgene™ stormwater sampler up and downstream of the reference outfall was 1 ± 0.4 hours with no significant difference in the collection times up and downstream of the reference outfall (p = 0.28). Further details on the timing of sample collection are available in the Supplementary Material. There was no significant difference in the pH (p = 0.68), conductivity (p = 0.68), turbidity (p = 0.95), or concentrations of suspended solids (p = 0.95), dissolved organic carbon (DOC) (p = 0.95), ammonia (p = 0.68), nitrate (p = 0.84), nitrite (p = 0.83), phosphate (p = 0.95), zinc (p = 0.95), iron (p = 0.95), or chloride (p = 0.95) downstream of the reference outfall relative to upstream of the reference outfall. Additional supporting figures showing the raw water quality measurements by sampling date are provided in the Supplementary Materials.

Table 3-12 Statistical results from comparisons of collection timing and water quality parameters upstream and downstream of the reference outfall.

Parameter	Test Result
Collection timing	V = 3, p = 0.28
pH	$t(23)$ = -1.7, p = 0.68
Conductivity	$t(22)$ = -18.4, p = 0.68
Chloride	V = 122, p = 0.95
Turbidity	$t(23)$ = 144, p = 0.95
Suspended solids	W = 138, p = 0.95
Dissolved organic carbon (DOC)	W = 107, p = 0.95
Ammonia	W = 91, p = 0.68
Nitrate	W = 99, p = 0.84
Nitrite	W = 91, p = 0.83
Phosphate	W = 96, p = 0.95
Zinc	W = 123, p = 0.95
Iron	$t(22)$ = -0.07, p = 0.95

3.3.5 Instream water quality response at the bioretention system.

Water quality measurements taken upstream and downstream of Bioretention System 1 were compared to assess whether stormwater treated through bioretention processes altered downstream water quality relative to upstream conditions. There were no significant differences in the water quality measurements taken upstream and downstream of Bioretention System 1 (Table 3-13). The mean time from when effluent entered the creek from the outflow of Bioretention System 1 and when the creek rose 1 cm to fill the Nalgene^{TM} stormwater sampler upstream of the bioretention system was 1.5 ± 0.5 hours. The mean time from when effluent entered the creek from the outflow of Bioretention System 1 and when the creek rose 1 cm to fill the Nalgene^{TM} stormwater sampler downstream of the bioretention system was 1.1 ± 0.4 hours. There was no significant difference in collection time upstream and downstream of the bioretention system ($p = 0.2$). Further details on the timing of sample collection are available in the Supplementary Material. There were no significant differences in the pH ($p = 0.7$), conductivity ($p = 0.41$), turbidity ($p = 0.31$), or concentrations of suspended solids (p = 0.22), dissolved organic carbon (DOC) ($p = 0.67$), ammonia ($p = 0.66$), nitrate ($p = 0.67$), nitrite ($p = 0.9$), phosphate ($p = 0.19$), zinc ($p = 0.05$), iron ($V = 176$), or chloride ($p = 0.19$) downstream of the bioretention system relative to upstream of the bioretention system.

Table 3-13 Statistical results from comparisons of collection timing and water quality parameters upstream and downstream of Bioretention System 1.

Parameter	Test Result
Collection timing	$V = 9$, $p = 0.2$
pH	$V = 121$, $p = 0.7$
Conductivity	$V = 106$, $p = 0.41$
Chloride	$V = 79$, $p = 0.19$
Turbidity	$V = 218$, $p = 0.31$
Suspended solids	$V = 211$, $p = 0.22$
Dissolved organic carbon (DOC)	$V = 138$, $p = 0.67$
Ammonia	$V = 103$, $p = 0.66$
Nitrate	$V = 138$, $p = 0.67$
Nitrite	$V = 89$, $p = 0.9$
Phosphate	$V = 208$, $p = 0.19$
Zinc	$V = 43$, $p = 0.05$
Iron	$V = 176$, $p = 0.66$

3.4 Discussion

The bioretention systems in this study consistently reduced both peak and total stormwater discharge entering trout-sensitive tributaries, while water quality responses varied across parameters. Treatment was most effective for suspended solids and, to a lesser extent, turbidity and water temperature, with outflow measurements consistently lower and meeting water quality guidelines more frequently, as reflected by fewer outflow exceedances than those from the bioretention system inflows. However, reductions in most nutrients, metals, and dissolved organic carbon were limited or inconsistent across the bioretention systems, and conductivity and chloride concentrations were often higher in the effluent than in the influent. Despite substantial reductions in discharge, turbidity, and suspended solids from the effluent, no detectable changes in instream water chemistry were observed downstream of either the bioretention system or the reference outfall, likely due to strong instream dilution during rainfall events.

Comparisons with water quality guidelines and ecological thresholds showed that bioretention systems reduced the frequency that the pH, turbidity, and concentrations of suspended solids, ammonia (only evaluated at Sites 1 and 2), and, to a lesser extent, nitrite and water temperature exceeded water quality guidelines and ecological thresholds relevant to aquatic life, but generally did not reduce exceedances of phosphate, zinc, or iron. Reductions in ammonia exceedances at Bioretention Systems 1 and 2 were not consistently associated with decreases in ammonia concentrations, as ammonia increased at some sites, but not others. Instead, reductions in exceedance frequency were influenced by the temperature- and pH-dependent nature of the CCME water quality guideline, which governs the proportion of unionized ammonia. Effluent concentrations of phosphate, zinc, and iron frequently exceeded water quality guidelines, and the bioretention systems did not reduce these exceedances. This indicates that while bioretention systems are effective at reducing runoff volume and particulate pollutant concentrations, they are less effective at reducing dissolved pollutant concentrations. As a result, additional measures are required to reduce the frequency of exceedances of dissolved pollutant guidelines and limit the release of phosphorus and metals into sensitive fish habitats during rainfall events.

3.4.1 Bioretention systems reduce stormwater discharge, temperature, turbidity, and suspended solids concentrations.

This research shows that bioretention systems successfully reduced peak and total discharge and peak water temperatures during rainfall events. Reductions in discharge are likely due to surface and subsurface storage, infiltration, and evapotranspiration (Nazarpour et al., 2023; Spraakman et al., 2022). As pollutant load reductions are often greater than concentration reductions due to volume retention (Sabbagh et al., 2025), discharge reductions reduce the loading of particulate pollutants to fish habitats. The bioretention systems also reduced peak water temperatures and the frequency of exceedances of the 22°C thermal threshold associated with the onset of thermal stress on salmonid species (Easton et al., 1995; McCollough, 1999). These reductions in peak water temperatures likely reflect hydrologic detention and heat exchange between infiltrated stormwater and the soil media (Li et al., 2022). However, reductions were significant only at Bioretention System 1, which may reflect a smaller sample size and reduced statistical power at Bioretention System 2 rather than an absence of thermal mitigation.

The bioretention systems also reduced turbidity and suspended solids concentrations, and importantly, the frequency with which these parameters exceeded ecological thresholds relevant to fish habitat. The frequency of turbidity exceedances above the 25 NTU benchmark, associated with reduced growth (Sigler et al., 1984; CCME, 2022) and reduced fish density in salmonids (Sigler et al., 1984), was substantially lower in the effluent of the bioretention systems than in the influent. Exceedances of influent suspended solids concentrations above the 100 mg/L threshold associated with sublethal effects on adult and juvenile fish (CCME, 2002) were also reduced. These reductions indicate effective retention of particulate contaminants (Nazarpour et al., 2023; Spraakman et al., 2020b), likely through sedimentation and filtration within the bioretention media (Sabbagh et al., 2025), thereby reducing the exposure of fish and aquatic organisms to increased turbidity and suspended solids concentrations. Influent solids were predominantly larger-grained sediments, whereas effluent solids were primarily filamentous organic matter, potentially due to the mobilization of plant detritus within the bioretention system. As a result of this organic matter in the effluent, large-grained sediment removals are likely higher than the results indicate.

3.4.2 Reductions in pH, dissolved nutrients (nitrate, nitrite and orthophosphate), and dissolved organic carbon concentrations by bioretention systems were minimal or inconsistent.

Reductions in pH were generally minimal, with little difference between influent and effluent pH. Influent pH values were consistently within the CCME guideline range of 6.5 to 9 (CCME, 2008), and effluent pH remained within this range across all bioretention systems, consistent with other studies that show only minor changes in pH between the influent and effluent (Corbett et al., 2024; Spraakman et al., 2020b). Given the absence of strongly alkaline or acidic influent, minimal changes in pH were expected, as bioretention media primarily buffers pH rather than actively altering it (LeFevre et al., 2014). However, the effluent pH from Bioretention System 1 was within the PWQO water quality guideline of 6.5 to 8 more often than the influent pH, which was often below 6.5, thus reducing the frequency of exposure to acidic conditions.

DOC and dissolved nutrient concentrations were variable, with no consistent reductions or release. Although DOC can enhance metal mobility and export through complexation and desorption processes (Al-Amin et al., 2024), the absence of sustained DOC release suggests that DOC-mediated metal export was unlikely during the monitored rainfall events. Dissolved nutrient concentrations (nitrate, nitrite and orthophosphate) also exhibited limited and inconsistent attenuation, consistent with variable nutrient responses reported for bioretention systems during short-duration rainfall events, in which media composition, mulch layers, and system maturity more strongly influence nutrient behaviour (Spraakman et al., 2020b). In this study, Bioretention System 1 was constructed in 2019, and Bioretention Systems 2 and 3 were constructed in 2017, indicating that all three bioretention systems were established several years prior to monitoring. As such, nutrient treatment performance is unlikely to reflect system maturity. The bioretention systems also did not consistently reduce the frequency that dissolved nutrient concentrations exceeded water quality guidelines, particularly for phosphate, indicating limited effectiveness for dissolved nutrient removal.

3.4.3 Conductivity, chloride, and to a lesser extent metal concentrations often increased after retention in bioretention systems.

Conductivity and chloride concentrations consistently increased in the effluent of all three bioretention systems, indicating a net export of dissolved ions from bioretention cells and limited removal capacity (Shetty et al., 2020). Increased chloride concentrations in the effluent may reflect delayed flushing of road salts retained in the soil media, leading to more elevated levels year-round (Shetty et al., 2020). Despite these increases, effluent chloride concentrations remained below both short-term (640 mg/L) and long-term (120 mg/L) CCME water quality guidelines (CCME, 2011), and below concentrations (3000 mg/L Cl^- for NaCl and 860 mg/L Cl^- for $CaCl_2$) shown to impair trout growth and development (Hintz & Relyea, 2017).

Zinc and iron exhibited occasional increases in effluent concentrations, likely reflecting the persistence of dissolved and colloidal metal fractions that can remain mobile and contribute to variability in total effluent concentrations, despite effective particulate retention (Croft et al., 2024). Changes in event hydraulics and stormwater chemistry can also promote the mobilization or wash-through of particulate and colloidal metal fractions, contributing to episodic increases in total metal concentrations in bioretention effluent relative to influent (Lange et al., 2022). However, guideline exceedances showed that zinc concentrations exceeded water quality guidelines in nearly all inflow and outflow samples, indicating that the bioretention systems did not reduce the frequency of exceedances of thresholds harmful to fish and aquatic life. Exceedances of iron guidelines were more variable, and the frequency of outflow exceedances was not consistently lower than that of inflow exceedances across sites.

3.4.4 No changes in instream water quality were detected downstream of either the untreated reference outfall or the bioretention system.

No changes in any water quality parameters were detected downstream of the reference outfall or the bioretention system, likely due to rapid dilution within the receiving stream and the inherent difficulty in detecting short-duration, event-scale instream changes over short reaches. However, this should not be interpreted as evidence of the absence of stormwater-related impairment to downstream aquatic ecosystems. Rapid dilution and mixing within urban streams can limit the detectability of localized downstream changes, and instream water quality responses may only emerge under specific hydrologic conditions or when evaluated cumulatively across larger spatial scales (Scarlett et al., 2018). Detecting instream responses often requires coordinated networks of stormwater control measures that treat a substantial fraction of connected impervious area, as localized or sparsely distributed controls may not generate measurable downstream effects (Jefferson et al., 2017). Even under widespread implementation, instream responses can remain mixed or parameter-specific due to background variability and persistent urban stormwater inputs (Hopkins et al., 2022). Accordingly, the absence of a detectable instream response reflects methodological and scale limitations rather than a lack of ecological impact from untreated stormwater.

3.4.5 Implications for stormwater management in trout-sensitive watersheds.

Bioretention systems should be prioritized to reduce particulate pollutant loads in urban watersheds, as they effectively reduce peak and total stormwater discharge, turbidity, and suspended solids concentrations. These reductions, in turn, reduce the frequency with which suspended solids concentrations and turbidity exceed ecological thresholds relevant to fish habitat. However, bioretention systems alone do not adequately address dissolved pollutant concentrations, such as chloride and dissolved nutrients, necessitating complementary land-use practices to reduce the release of dissolved pollutants into sensitive fish habitats. For instance, applying fertilizers during dry periods and avoiding wet or high-risk rainfall periods can reduce dissolved nutrient runoff (Murumkar et al., 2025), thereby reducing nutrient inputs not addressed by bioretention alone from entering urban streams.

Reducing dissolved pollutant loads requires targeted land use practices that address contaminant sources throughout the watershed. Reducing winter road salt application may help lower elevated chloride concentrations in effluent, which are attributed to the gradual release of accumulated de-icing salts in subsequent seasons (Brown et al., 2024). As chloride in the bioretention media can increase the mobilization of dissolved metals (Kaushal et al., 2021; Corsi et al., 2010), minimizing road salt application may also help limit the export of dissolved metals from bioretention systems, which were not routinely removed by the bioretention systems and in some cases were higher in the effluent than in the influent. Lastly, promoting residential rain garden adoption can reduce surface runoff exiting private properties (Daniels & Yeakley, 2024; Jennings et al., 2015), thereby limiting nutrient runoff at the source. The combination of bioretention systems to reduce discharge and suspended solids concentrations, along with complementary land use practices to reduce dissolved pollutant concentrations, can therefore help protect trout-sensitive watersheds.

3.4.6 Study limitations and areas of future research.

This research was not without its limitations, which should be considered when interpreting the results. Since continuous water quality sampling was not conducted, event mean concentrations and total pollutant loads could not be quantified. Additionally, the absence of detectable differences in pollutant concentrations upstream and downstream of both the reference outfall and bioretention systems likely reflects downstream dilution and mixing rather than the absence of stormwater impairments, highlighting the inherent challenges of detecting instream water quality responses using grab sampling approaches. Metals were also analyzed as undissolved, unfiltered concentrations rather than dissolved or total recoverable fractions, limiting direct comparison with provincial and federal water quality guidelines. Similarly, while orthophosphate was measured due to its ecological relevance and bioavailability, this approach did not allow direct comparison with Canadian total phosphorus guidelines. Lastly, the organic fraction of suspended solids was not quantified, limiting interpretation of whether effluent turbidity and suspended material reflected inorganic road grit or organic debris mobilized within the bioretention systems.

Due to these limitations, future research should prioritize continuous or automated sampling at bioretention system influent, effluent, and instream locations to quantify event-mean concentrations and pollutant loads entering the bioretention system and receiving streams. Collecting additional samples during storm events and across the stream channel can reduce temporal and cross-sectional variability and associated uncertainty (Harmel et al., 2010), potentially improving the detectability of instream water quality changes. Distinguishing between dissolved, particulate, and total fractions of metals and nutrients, and measuring total phosphorus in addition to orthophosphate, would improve comparability with aquatic life guidelines and strengthen ecological interpretations. Quantifying the organic content of suspended solids would further clarify sediment sources and improve the assessment of bioretention performance for inorganic versus organic material. Longer-term monitoring across a broader range of hydrologic conditions, including low-flow periods with reduced dilution, would enhance understanding of downstream water quality responses. Finally, integrating physical habitat assessments with evaluations of benthic invertebrate and macrophyte communities would help determine whether observed reductions in stormwater pollutants translate into meaningful ecological improvements.

3.5 Conclusion

Bioretention systems consistently reduced peak and total stormwater discharge, turbidity, and suspended solids concentrations entering urban trout tributaries, thereby reducing the potential for sedimentation of spawning gravels and the transport of particulate-bound pollutants into fish habitats. They also reduced the frequency with which water temperatures (Sites 1 and 2), pH, turbidity, and concentrations of suspended solids, ammonia (Sites 1 and 2), and, to a lesser extent, nitrite exceeded established water quality guidelines and literature-derived benchmarks relevant to fish health.

Reductions in ammonia exceedances were primarily due to changes in water temperature and pH, rather than consistent decreases in ammonia concentrations. In contrast, phosphate, zinc, and iron frequently exceeded water quality guidelines in both the inflow and outflow, indicating that these pollutants remain a concern to fish health and are not effectively mitigated by bioretention systems alone. Despite reductions in discharge, water temperature, and suspended solids concentrations, no detectable changes in downstream water quality were observed, likely due to dilution and mixing within the stream.

These findings should be interpreted as concentration-based treatment responses rather than pollutant load reductions because water quality samples represented first-flush concentrations rather than event mean or flow-weighted composite concentrations. Overall, these findings demonstrate that while bioretention systems effectively reduce runoff volumes and particulate pollutant concentrations, integrated stormwater management strategies that combine bioretention systems with targeted land-use practices are needed to reduce dissolved pollutant inputs and better protect urban aquatic ecosystems.

4 From Snowpack to Stream: Winter Pollutant Accumulation, Bioretention Performance and Fish Habitat Implications in a Northern Climate.

Abstract

Bioretention systems can reduce meltwater runoff and improve water quality during the spring freshet, but performance may differ between particulate-associated and dissolved contaminants. This study evaluated pollutant accumulation in winter snowpack and the performance of three bioretention systems in two cold-climate urban watersheds. Water quality was monitored at all three systems, while water quantity was assessed only at Bioretention Systems 1 and 2 because site constraints at Bioretention System 3 prevented reliable discharge monitoring. Roadside snowbanks had higher concentrations of chloride, suspended solids, and dissolved organic carbon (DOC) than open field and bioretention sites. During the spring freshet, Bioretention Systems 1 and 2 reduced peak and total runoff, while all three systems reduced pH, turbidity, suspended solids, and DOC concentrations. However, metal, nutrient, and chloride concentrations were inconsistently reduced, and in some cases increased at the outflow. No significant downstream water quality differences were observed, likely due to dilution and mixing. These findings indicate that bioretention systems can reduce meltwater runoff and particulate pollutants, but improved snow management and enhanced media design are needed to better retain dissolved contaminants and protect fish habitats during spring melt.

4.1 Introduction

The winter snowpack in cold climate urban environments accumulates pollutants that are released during spring melt, posing a serious risk to fish habitat. Roadside snowbanks contain elevated concentrations of suspended solids (Müller et al., 2022), chloride (Lazarcik & Dibb, 2017), heavy metals from vehicles (Vijayan et al., 2024), and combustion-derived pollutants from traffic exhaust (Nazarenko et al., 2017). These pollutants, which are often at higher concentrations in urban areas (Engelhard et al., 2007), are released into aquatic systems at elevated levels during the spring freshet, posing a threat to fish habitat. Snowflakes also have a porous structure whereby metals such as zinc and iron can adhere to them, increasing their atmospheric deposition (Nazarenko & Ariya, 2021), which can enter urban streams and impair gill function and oxygen uptake in fish (Vlasov et al., 2020). Nitrogen and phosphorus from decomposing organic matter (Hobbie et al., 2017) and atmospheric inputs (Bratt et al., 2017) can accumulate in the snowpack, increasing the risk of eutrophication. To compound these threats, winter-accumulated pollutants are often released during spawning and early life stages, when coldwater fish species are most vulnerable to water quality degradation.

During spring freshet, pollutants stored in winter snowpack are rapidly mobilized and discharged into streams over a two to three-week period in concentrated pulses (Vijayan et al., 2024; Chand et al., 2024), resulting in elevated concentrations that pose various risks to fish habitat and health. These pulses contain elevated concentrations of chloride (Lawson & Jackson, 2021), suspended solids (Bilotta & Brazier, 2008), and metals (Hiltibran, 1971; Skidmore, 1970). Chloride and suspended solids associated with winter gritting pose a particular threat to fish habitats during spring runoff. Chloride can impair osmoregulation (Winter et al., 2026), alter behaviour and increase mortality (Hintz & Relyea, 2019), while suspended solids can impair fish growth, respiration, feeding, hatching, and gill condition (Park et al., 2025a). In Northwestern Ontario, the release of these pollutants from the spring snowmelt coincides with rainbow trout (*Oncorhynchus mykiss*) spawning and fry emergence from April to June (DFO, 2013), shortly after snowmelt begins in March (ENRC, 2025). These pollutants can threaten egg viability, juvenile survival, and habitat suitability (Meyer & Wania, 2008), threatening fish habitat.

Bioretention systems (also referred to as bioinfiltration systems or rain gardens) can reduce meltwater volume and pollutant loads; however, their effectiveness under cold-climate conditions with high salt loads and no vegetative growth is uncertain. They use vegetated basins and engineered soils to retain meltwater, filter contaminants, and promote infiltration (Spraakman et al., 2020a). Cold-climate studies have shown that bioretention systems can remain hydrologically active during snowmelt or thawing conditions, although performance may depend on local winter hydrology, frozen media conditions, and preferential flow pathways through cracks or macropores within the filter media (Khan et al., 2012; Kratky et al., 2021).

During winter and spring melt, bioretention systems can reduce suspended solids (McPhillips et al., 2023), phosphate, and ammonium (Kratky et al., 2021), and may remove nitrate under saturated conditions (Kratky et al., 2021). However, chloride can inhibit nutrient and metal retention (Hu et al., 2025), trigger phosphorus release during melt (Goor et al., 2021), and persist in effluent well beyond the snowmelt period (Shetty et al., 2020). These limitations raise concerns about whether bioretention systems can reduce runoff-related pollutants entering downstream aquatic ecosystems during vulnerable life stages in cold-climate urban catchments during the spring freshet.

Although bioretention system performance is widely studied under warm-weather conditions, few studies have evaluated their function during the spring freshet in cold climates, when fish are most vulnerable. Even fewer have evaluated whether bioretention treatment performance produces detectable short-term downstream water quality responses. This disconnect limits our ability to determine whether bioretention systems reduce meltwater-related pollutants before entering urban streams and whether potential changes to meltwater quantity and quality are detectable downstream. This study addresses that gap by evaluating the water quality and quantity performance of three large, double-celled bioretention systems over two spring freshet seasons in two cold-climate urban watersheds and assesses whether short-term downstream water quality changes are detectable in receiving streams. Specifically, this study aims to: 1) quantify pollutant accumulation in winter snowpack, 2) assess whether bioretention systems reduce meltwater volume and pollutant concentrations before discharging into sensitive trout tributaries, 3) evaluate whether bioretention quality and quantity performance varies among bioretention systems, and 4) determine whether short-term downstream water quality differences are detectable upstream and downstream of treated and untreated meltwater discharges during the spring freshet.

4.2 Methods

4.2.1 Study Area

This study was conducted along two coldwater tributaries supporting brook trout (Salvelinus fontinalis) and rainbow trout (Oncorhynchus mykiss). McVicar Creek is a smaller coldwater stream that drains a watershed area of approximately 48 km^2 and has an average discharge of approximately 0.3 m^3/s (Olivier et al., 2016). Peak discharge near the mouth ranges from 4.8 to 40 m^3/s (McGoldrick et al., 2011). The Current River drains a larger watershed (approximately 662 km^2) and has a larger average mean discharge (4 m^3/s). The three bioretention systems evaluated in this chapter receive runoff from mature residential catchments that discharge into either McVicar Creek, or Boulevard Lake within the Current River watershed (Figure 4-1).

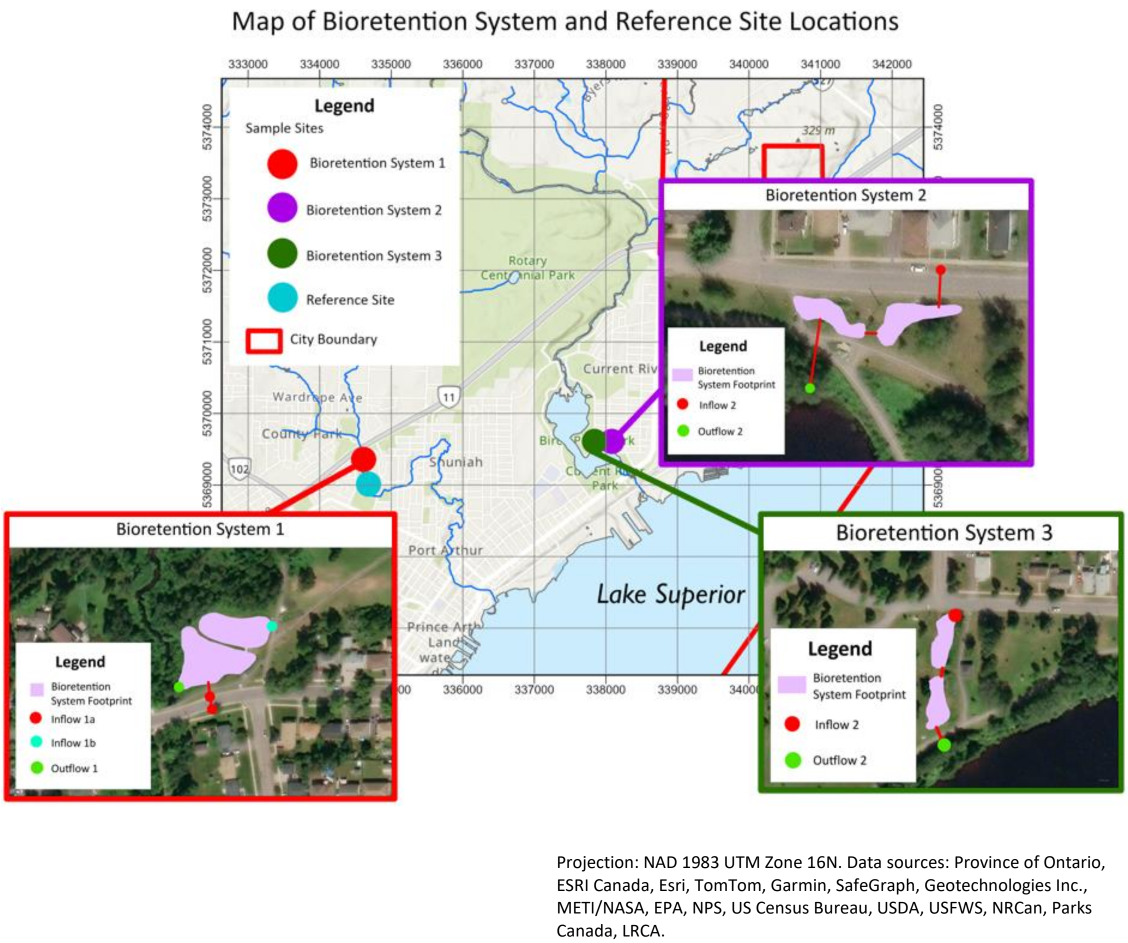


Figure 4-1 Location of the bioretention systems and reference outfall within the study area in Thunder Bay, Ontario.

The region experiences a warm-summer continental climate characterized by prolonged winters and an average annual snowfall of 187.6 cm (Environment and Climate Change Canada, 2025). In three of the catchments, runoff from roadways and adjacent residential areas is directed through bioretention systems before entering urban tributaries. The remaining catchment conveys runoff directly to an urban stream via a conventional catch basin and drainage ditch, without any form of water quality or quantity control. This untreated site was used as a reference condition to compare short-term downstream water quality responses below a conventional stormwater outfall to those below the effluent of a bioretention system providing quantity and quality control.

Bioretention System 1 receives runoff from two inflows. The first inflow, Inflow 1a, drains roadway runoff, and Inflow 1b drains a pervious hillslope with minimal exposure to traffic-related pollutants. Bioretention Systems 2 and 3 each receive roadway runoff from a single inflow (Inflow 2 and Inflow 3, respectively). Each bioretention system discharges to a single outflow (Outflow 1-3, with numbers corresponding to each bioretention system). An untreated storm sewer outfall (Reference Outfall) also discharges into the same receiving stream as Bioretention System 1 and was used as a reference to compare downstream water quality responses from a traditional outfall to those from a bioretention system that provides water quality and quantity treatment.

Detailed design characteristics of the three bioretention systems, including cell depth, ponding areas, subdrain configuration, outlet controls, overflow routing, soil media, vegetation, and liner status, are provided in Section 1.8. Bioretention System 1 receives runoff from two sub-catchments and discharges to McVicar Creek through a controlled outflow system, whereas Bioretention Systems 2 and 3 each receive runoff from a single catchment and discharge to Boulevard Lake through pretreatment and bioretention cells connected to subdrains and outflow pipes. These drainage and design differences were considered when interpreting spring meltwater quantity and water quality performance.

The catchments and associated surface covers were delineated by classifying land use within each study area. Catchment boundaries (Table 4-1) were defined in ArcGIS Pro 2.9.0 using municipal shapefiles from the City of Thunder Bay's engineering division. Bioretention System 1 receives runoff from two sub-catchments (1a and 1b), whereas Bioretention Systems 2 and 3 receive runoff from only a single catchment. Impervious surfaces within each catchment were identified using aerial imagery and manually digitized, then grouped into five categories: roads, buildings, parking areas, sidewalks/paved trails, and other. The designed storage capacity (m^3), which represents the total surface area and subsurface storage volume of each system. Catchment area (ha) and percent impervious surface cover (%) are reported for each site. "N/A" indicates that that a given impervious surface type was not present within a given catchment.

Table 4-1 Catchment size, imperviousness and designed storage capacity for each bioretention system.

	Catchment Size (ha)	% Imperviousness	Designed Storage Capacity (m^3)
Reference Catchment	1.8	49	N/A
Bioretention System 1	2.6	35	616
Sub-catchment 1a	1.1	49	375
Sub-catchment 1b	1.5	30	241
Bioretention System 2	1.6	31	82
Bioretention System 3	4.0	35	157

Impervious Surface Cover	Reference Catchment		Catchment 1a		Catchment 1b		Catchment 2		Catchment 3	
	Area (ha)	% Total Cover	Area (ha)	% Total Cover	Area (ha)	% Total Cover	Area (ha)	% Total Cover	Area (ha)	% Total Cover
Roads	0.23	12.6	0.21	19.4	0.00	0.0	0.12	7.74	0.71	17.8
Buildings	0.29	15.8	0.10	9.3	0.14	9.15	0.21	13.55	0.61	15.29
Parking Areas	0.23	12.6	0.13	12.0	0.17	11.11	0.10	6.45	0.45	11.28
Sidewalks/ Paved Trails	0.09	4.9	0.09	8.25	0.02	0.13	0.04	4.52	0.03	0.75
Other	0.05	2.7	0.0	0.0	0.07	4.58	0.02	1.29	0.01	0.25
Total Impervious Cover	**0.89**	**49**	**0.53**	**49**	**0.4**	**25**	**0.49**	**31**	**1.81**	**45**
Total Pervious Cover	**0.94**	**51**	**0.55**	**51**	**1.1**	**75**	**1.11**	**69**	**2.19**	**55**

4.2.2 Study Design

I collected snow core samples from three different surface types within each study catchment to compare differences in pollutant concentrations in the winter snowpack between roadside snowbanks, which receive traffic-derived pollutants, open fields, and the surface of bioretention systems. Sampling was conducted monthly through the snow-covered season (December 2022 to April 2023; January to March 2024) to capture concentrations throughout the winter. In each catchment, three replicate cores were collected from each surface type to account for within-site variability. This approach allowed for a direct comparison of winter pollutant accumulation across various surface types prior to the spring melt period.

During the spring melt period (April 5-25, 2023, and March 10-April 6, 2024), I assessed the performance of two bioretention systems at reducing peak and total daily meltwater runoff, and the performance of three bioretention systems at improving meltwater quality. Water quantity performance was evaluated by continuously monitoring water levels from the inflows and outflows of Bioretention Systems 1 and 2. HOBO U20/U20L water level loggers were installed within the catch basins/swales conveying meltwater into the bioretention system inflows and within the storage layers at the outflows of both bioretention systems. These loggers recorded water levels over 15-minute intervals (See 2.4).

Discrete discharge measurements were collected from the inflows and outflows of Bioretention Systems 1 and 2 using a bucket and a stopwatch and were used to develop site-specific water level-discharge rating curves for the inflows and outflows of each bioretention system. Discharge could not be assessed at Bioretention System 3 because an elbow in the access pipe prevented deployment of a logger in the internal storage layer. As a result, Bioretention System 3 was used to assess water quality only.

To compare downstream effects between a conventional stormwater outfall (reference outfall) providing no meltwater quality or quantity control and a bioretention system providing meltwater quantity and quality control, paired sampling sites were established within a receiving stream. These sampling locations were 1) immediately upstream of the reference outfall, 2) six metres downstream of the reference outfall, 3) immediately upstream of the bioretention system outflow, and 4) six metres downstream of the bioretention system outflow. This spacing was selected to allow adequate mixing of the discharge within the stream while minimizing interference from downstream infrastructure such as culverts, road crossings, and additional outfalls. Upstream and downstream comparisons were not conducted at Bioretention Systems 2 and 3 because both systems discharged into Boulevard Lake (Figure 4-1), a lentic reservoir created by damming the Current River for historical electric power generation, rather than flowing stream reaches with clearly defined upstream and downstream sampling locations.

4.2.3 Snowpack Sampling

The winter snowpack was sampled and analyzed monthly to quantify pollutant accumulation and characterize snowpack conditions across study catchments. Winter snowpack data were collected from conservation authority staff, including depth measurements, snow volumes, and liquid equivalents. From December 2022 through April 2023 and January through April 2024, I collected snow cores across five catchments (Reference catchment, Catchment 1a, Catchment 1b, Catchment 2, and Catchment 3). Sampling did not occur in December 2023 due to insufficient snowpack accumulation. In catchments 1a, 2, and 3, I collected three replicate cores at each sample location (roadside snowbank, open field, and bioretention system) using a 15 cm-diameter stovepipe. As done by Moghadas et al. (2015), snow samples were collected as vertical snow cores from the top of the snowbank/snowpack to the bottom. In the reference catchment, without a bioretention system, I collected three replicate cores from both the roadside snowbank and an open yard.

Replicates were composited in 68 L bins, homogenized, subsampled into 4 L containers, melted at room temperature, and refrigerated at 4°C before analysis. Snow accumulation at each site was documented photographically (Figure 4-2). Sample locations inclthe snowbank in the reference catchment; (2) an open yard in the reference catchment; (3) the snowbank in sub-catchment 1a; (4) an open field in sub-catchment 1b; (5) the surface of bioretention system 1; (6) the snowbank in catchment 2; (7) an open field in catchment 2; (8) the surface of bioretention system 2; (9) the snowbank in catchment 3; (10) an open field in catchment 3; and (11) the surface of bioretention system 3.

Snowpack depth and volume was quantified to estimate runoff potential during the spring melt period. The monthly snow depth and liquid equivalent were obtained from the Lakehead Region Conservation Authority (LRCA), and water content was calculated for each snow core by dividing the liquid equivalent (mm) by the snow depth (cm) to obtain water content (mm liquid equivalent per cm of snowpack).

(1) Reference snowbank.

(2) Reference field/yard.

(3) Bioretention 1 snowbank.

(4) Bioretention 2 field.

(5) Bioretention system 1.

(6) Bioretention 2 snowbank.

(7) Bioretention 2 field.

(8) Bioretention system 2.

(9) Bioretention 3 snowbank.

(10) Bioretention 3 field.

(11) Bioretention system 3.

Figure 4-2 Accumulation of the snow from open fields, roadside snowbanks and bioretention systems in each of the four study catchments.

4.2.4 Spring Melt Water Quantity Analysis

The daily peak and total discharge from the inflows and outflows were quantified to evaluate the performance of Bioretention Systems 1 and 2 at reducing meltwater discharge during the 2023 and 2024 spring melt periods (April 5 to 25, 2023, and March 10 to April 6, 2024). I deployed HOBO U20/U20L pressure transducers (Onset Computer Corporation, Bourne, MA, USA) at the inflows of the Reference Site (Reference Inflow), Bioretention System 1 (Inflow 1a and Inflow 1b), Bioretention System 2 (Inflow 2), and within the storage layers of Bioretention Systems 1 and 2. The loggers deployed in the storage layers were used to monitor water levels within the bioretention system and associated discharge from Outflows 1 and 2. Water levels were recorded at 15-minute intervals. I used a 10-L bucket and stopwatch to collect discrete discharge measurements from the inflows and outflows of Bioretention Systems 1 and 2. These discrete measurements were paired with the corresponding logger water level readings to generate stage-discharge calibration curves. Due to an elbow in the monitoring pipe for Bioretention System 3, a logger could not be installed, and consequently, no discharge data were collected from this bioretention system.

At the beginning and end of each melt period, I offloaded the water level data to a PC using HOBOware Pro v3.6.2 software. The HOBOware Barometric Compensation Assistant was used to compensate for barometric pressure using barometric pressure collected from a separate HOBO U20/U20L, which was suspended in the air near Bioretention System 1. I developed site-specific rating curves for each monitoring location to relate water level to discharge. Exponential stage-discharge relationships were used to estimate discharge at the Reference Inflow, Inflow 1b, and Inflow 2 using the discrete discharge measurements paired with their corresponding water level measurements. For Inflow 1a, a regression-based stage-discharge relationship could not be established due to a poor model fit. Instead, discharge was estimated using Manning's equation, assuming a 14-inch circular pipe. The discharge entering Inflows 1a and 1b was summed to calculate the total discharge entering Bioretention System 1.

Discharge for Outflow 1 and Outflow 2 were modelled using exponential relationships between storage-layer water depth and discharge, with discharge values limited to full-pipe discharge capacity.

Continuous water level data were converted to discharge using site-specific rating curves (provided in the Supplementary Materials). For each day during the spring melt periods, daily peak discharge was calculated as the maximum value recorded (L/s), and daily total discharge (L and m^3) was calculated by numerically integrating discharge over the course of each day for each inflow and outflow site. Water quantity reductions were expressed as a percent decrease in peak and total discharge from the outflows relative to their respective inflows.

Outflow from the bioretention systems can be delayed by days or even weeks after inflow during spring melt, particularly under varying frost conditions, making it difficult to reliably pair individual inflow and outflow events. Therefore, peak and total discharge reductions were calculated using cumulative inflow and outflow volumes over the full melt period rather than event-based pairing. Reductions in peak and total discharge were calculated using Equations 1 and 2.

Equation 1)

$$Peak\ Discharge\ Reduction\ = \left(\frac{Peak\ Discharge_{Influent} - Peak\ Discharge_{Effluent}}{Peak\ Discharge_{Influent}}\right) \times 100\%$$

Equation 2)

$$Total\ Discharge\ Reduction\ = \left(\frac{Total\ Discharge_{Influent} - Total\ Discharge_{Effluent}}{Total\ Discharge_{Influent}}\right) \times 100\%$$

4.2.5 Spring Melt Sample Collection

Grab samples were collected during the peak afternoon melt (3:00 to 5:00 p.m.) on seven dates in 2023 and four dates in 2024 (See Supplementary Materials) from the influent and effluent of each bioretention system, the reference outfall, and instream locations immediately upstream and 6 m downstream of both the reference outfall and Bioretention System 1. The downstream sampling distance minimized confounding influences from nearby bridges and stormwater outfalls. Field measurements of pH and conductivity were collected using Hanna multiparameter probes (HI98129/HI98130; Hanna Instruments, Woonsocket, RI, USA), and turbidity was determined using a portable turbidity meter (2020we; LaMotte Company, Chestertown, MD, USA). Samples were transported on ice and refrigerated within two hours prior to laboratory analysis. Relationships between stream discharge and velocity at each of the four instream sample locations are provided in the Supplementary Material.

4.2.6 Laboratory Analysis

Ammonia, nitrate, nitrite, orthophosphate, chloride, and whole-water zinc and iron concentrations were determined using a LaMotte SMART3 colorimeter (LaMotte Company, Chestertown, MD, USA; resolution 1% FS; accuracy ± 2% FS, where FS = full scale of the selected measurement range). Instrument performance was verified prior to each use using test standards. Metal samples were analyzed without filtration or acid digestion; therefore, reported zinc and iron concentrations represent whole-water concentrations that include dissolved, colloidal, and fine particulate-associated fractions.

Colorimetric analyses using the SMART3 colorimeter followed the manufacturer-specified reagent procedures and pre-programmed instrument methods for each analyte. Methods included the salicylate method for ammonia (Code 3659-01-SC), cadmium reduction method for nitrate (Code 3649-SC), diazotization method for nitrite (Code 3650-SC), ascorbic acid reduction method for orthophosphate (Code 3653-SC), argentometric method for chloride (Code 3693-SC), Zincon method for zinc (Code 3667-SC), and bipyridyl method for iron (Code 3648-SC). These methods are also summarized in the Supplementary Materials.

Suspended solids were measured gravimetrically using filtration and drying. Concentrations were measured by filtering collected water through 2 µm pre-combusted and pre-weighed glass fibre filters, which were then dried at 104 °C for 1 hour prior to weighing. The analytical balance used to weigh dried filters was calibrated before each use using brass standards.

Dissolved organic carbon (DOC) concentrations were measured by filtering samples through 2 µm glass fibre filters, preserving them with 2 mL of 85% phosphoric acid per 30 mL of sample, and analyzing them at the USDA Forest Service Northern Research Station (Grand Rapids, MN). DOC was quantified using a Shimadzu TOC-L CPH total organic carbon analyzer (Shimadzu Corporation, Kyoto, Japan) for the 2022 and 2023 samples and a Shimadzu TOC-L CSH analyzer for the 2024 samples. Both DOC analyzers were identical in functionality and used high-temperature catalytic combustion oxidation with non-dispersive infrared detection to quantify organic carbon concentrations. The only difference between the CPH and CSH models was that the CPH model was operated using a PC, whereas the CSH model was a standalone unit. ERA standards and sucrose checks were used to verify the accuracy of both models.

Using the measured water quality results, I calculated percent reductions for each parameter and bioretention system as:

$$Percent\ Reduction = \frac{Concentration\ from\ Influent - Concentration\ from\ Effluent}{Concentration\ from\ Influent} \times 100\%$$

Due to the logarithmic scale of pH, reductions in pH were calculated as absolute values (i.e., pH of Influent - pH of Effluent) rather than as percent reductions. For both percent and absolute reductions, negative values indicate that effluent concentrations are higher than influent concentrations.

Pollutant loads were not calculated because meltwater samples were collected as single, daily grab samples, rather than daily mean, event mean, or flow-weighted composite concentrations. Although meltwater discharge was available for Bioretention Systems 1 and 2, multiplying daily runoff volumes by a single grab-sample would assume that the measured concentration represented the full daily melt period. This assumption was not considered defensible because meltwater concentrations can vary throughout the day and melt period. Therefore, water quality treatment performance was evaluated using discharge, and concentration-based comparisons rather than calculating pollutant loads.

Meltwater quality from the inflow and outflow of the bioretention systems was assessed relative to federal (Canadian Council of Ministers of the Environment; CCME) and provincial (Provincial Water Quality Objectives; PWQO) guidelines established for the protection of aquatic life (Table 4-2). Parameters with defined guidelines (e.g., pH, chloride, nutrients, metals) were evaluated by direct comparison with these guidelines. For ammonia, exceedances were determined using the CCME criteria that account for both temperature and pH, as these factors influence the proportion of unionized ammonia and its toxicity. Water temperature was included only for the purpose of calculating ammonia thresholds and was not assessed independently against ecological benchmarks, as temperatures during snowmelt remain below levels associated with thermal stress in coldwater fish species (Easton et al., 1995; McCollough, 1999).

For parameters without established guideline thresholds (e.g., turbidity and suspended solids), literature-derived ecological benchmarks were used as reference values (Table 4-2). The frequency with which water quality guidelines were exceeded was quantified for both the inflow and outflow of each of the three bioretention systems. Snowpack samples were not included in this analysis, as they represent a temporary reservoir of accumulated contaminants rather than the runoff entering receiving streams.

Accordingly, the exceedance analysis was limited to meltwater, which represents the contaminants mobilized and released into fish habitats during the spring freshet.

Table 4-2 Federal and provincial water quality guidelines used for interpreting water quality results from influent and effluent of the bioretention systems.

Parameter	Federal Water Quality Guideline (CCME)	Provincial Water Quality Guideline (PWQO)	Other Ecological Benchmark	Assessment Criteria Used
pH	6.5 to 9.0	6.5 to 8.5	N/A	6.5 to 8.5 (PWQO)
Conductivity	N/A	N/A	No widely accepted water quality guideline exists. Conductivity is an aggregate measure of dissolved ions (e.g., chloride, sodium sulphate), which vary in their toxicity to aquatic life and should be interpreted alongside other ion-specific guidelines (e.g., chloride).	None (interpret with chloride)
Chloride	640 mg/L (short-term exposure) 120 mg/L (long-term exposure)	N/A	N/A	120 mg/L (CCME)
Turbidity	**Clear Water:** Maximum increase of 8 NTUs above background levels (short-term); maximum increase of 2 NTUs above background levels (long-term).	No greater than 10% reduction in Secchi depth reading from background conditions.	Approximately 25 NTU has been associated with reduced growth (Sigler et al., 1984; CCME, 2022) and fish density in salmonids under sustained exposure (Sigler et al., 1984). Behavioural effects have been observed at turbidity levels approximately 20-30 NTU (Berg & Northcote, 1985; Gregory & Northcote, 1993).	25 NTU (ecological benchmark)
Suspended solids/sediments	**Clear flow:** Maximum increase of 25 mg/L above background levels for any short-term exposure (e.g., 24-h period). **High flow:** Maximum increase of 25 mg/L above background levels at any time when background levels are between 25 and 250 mg/L, and no more than a 10% of background levels when background is ≥ 250 mg/L.	N/A	Sublethal effects in adult and juvenile fish have been reported at concentrations of 100-300 mg/L (CCME, 2002).	≥100 mg/L (associated with sublethal effects)
Dissolved organic carbon	N/A	N/A	N/A	None (no established water quality guidelines or ecological benchmark).

Parameter	Federal Water Quality Guideline (CCME)	Provincial Water Quality Guideline (PWQO)	Other Ecological Benchmark	Assessment Criteria Used
Ammonia	19 mg/L un-ionized ammonia As un-ionized ammonia concentrations vary with pH and temperature, the corresponding total ammonia guideline also varies with pH and temperature.	20 mg/L unionized ammonia As unionized ammonia concentrations vary with pH and temperature, the corresponding total ammonia guideline also varies with pH and temperature.	N/A	Temperature- and pH-dependent CCME.
Nitrate	550 mg/L (short-term exposure) 13 mg/L (long-term exposure)	N/A	N/A	13 mg/L (CCME)
Nitrite	60 µg/L (0.06 mg/L; long-term exposure) No short-term exposure data	N/A	N/A	0.06 mg/L (CCME)
Phosphate	4-10 µg/L (0.004 - 0.01 mg/L) total phosphorus (oligotrophic lakes and rivers, corresponding to approximately 0.01 - 0.03 mg/L as phosphate (PO_4^{3-})).	30 µg/L (0.03 mg/L) to avoid excessive plant growth 20 µg/L (0.02 mg/L) to prevent nuisance algae 10 µg/L (0.01 mg/L) to maintain aesthetics	N/A	0.03 mg/L (CCME and PWQO)
Zinc	Dependent on DOC concentration, pH, and water hardness.	20 µg/L (0.02 mg/L)	N/A	0.02 mg/L (PWQO)
Iron	110 µg/L (0.11 mg/L) (FEQG)	N/A	N/A	0.11 mg/L (FEQG)

4.2.7 Statistical Analysis

All statistical analyses were conducted in R (version 4.3.0). Data were screened for transcription errors, and values below analytical detection limits were replaced with one-half the detection limit. Statistical significance was evaluated at $\alpha = 0.05$.

Monthly snowpack depth, liquid equivalent, and calculated water content (mm/cm) were compared between the 2023 and 2024 sample seasons using paired t-tests. The assumption of normality of paired differences for the paired t-tests was assessed with the Anderson-Darling test. Snowpack water quality concentrations were compared among snow core types (field, roadside snowbank, and bioretention surface) using linear mixed-effects models, with core type as a fixed effect and sampling site as a random intercept. Normality of residuals was assessed using the Anderson-Darling test, and homogeneity of variances was assessed using Levene's test. Extreme outliers were evaluated using the interquartile range (IQR) method (>3× IQR) and were removed only when exclusion maintained a balanced design across sampling dates. When necessary, transformations (log, log10, square root, cube root, inverse, Box-Cox, and Yeo-Johnson) and small constants were used to improve normality and homogeneity of variances. When assumptions could not be satisfied following transformations, Kruskal-Wallis tests were used. When significant differences among snow core types were detected, post-hoc comparisons were conducted to identify which snow core types differed from one another. Tukey HSD tests were used following parametric tests (linear mixed-effects model), while Dunn's test with a Benjamini-Hochberg (BH) adjustment were used following non-parametric tests (Kruskal-Wallis).

The daily peak and total discharge from the bioretention system inflows (Inflow 1a, Inflow 1b, and Inflow 2) were compared using Kruskal-Wallis tests, due to persistent deviations from normality. Similarly, due to deviations from normality, peak and total discharge from the inflow of Bioretention Systems 1 and 2 were compared to peak and total discharge exiting the outflow of Bioretention Systems 1 and 2 using Wilcoxon rank-sum tests (Mann-Whitney U tests). Percent reductions in peak and total discharge were calculated over the full melt periods using cumulative inflow and outflow volumes. Uncertainty was quantified using nonparametric bootstrap resampling (5,000 iterations), and reductions are reported as bootstrap means ± standard errors with associated 95% confidence intervals.

As differences in inflow quality can influence bioretention treatment performance, differences in water quality among inflow sites (Inflow 1a, Inflow 1b, Inflow 2, and Inflow 3) were evaluated using blocked one-way ANOVAs with sampling date treated as the blocking factor. Normality of residuals was assessed using the Anderson-Darling test, and homogeneity of variances was assessed using Levene's test. When necessary to meet assumptions, transformations (log, log10, square root, cube root, Box-Cox, and power), small constants, and IQR-based outlier removals were applied to improve normality and/or homogeneity of variances. When parametric assumptions were still not met, Friedman tests were used. When significant differences among inflow sites were detected, post hoc comparisons were conducted to identify which inflow locations differed from one another. Tukey HSD tests were used following parametric analyses and pairwise Wilcoxon signed-rank tests with a Benjamini-Hochberg adjustment were used following non-parametric analyses.

Due to frozen conditions, and limited infiltration, effluent may be released from the bioretention system days or weeks after influent enters the system. As a result, the influent and effluent water quality measurements for each bioretention system were treated as unpaired samples and compared using unpaired two-sample t-tests when assumptions were satisfied, and Wilcoxon rank-sum tests, with the test statistic reported as W, when assumptions were violated. Percent reductions in pollutant concentrations were calculated for each sampling date as (influent − effluent) / influent × 100%. As the pH scale is logarithmic, pH reductions were calculated as absolute differences (influent − effluent). Calculating treatment performance as reductions relative to influent measurements standardized the differences between influent and effluent measurements among bioretention systems and allowed direct comparison of relative treatment performance between systems. Differences in reductions between bioretention systems were evaluated using one-way ANOVAs when assumptions were met, and Kruskal-Wallis (KW) tests when assumptions were violated. When significant differences in treatment performance among bioretention systems were detected, post-hoc comparisons were conducted to identify bioretention systems were significantly different from each other. Tukey HSD tests were used following ANOVA analyses, while Dunn's tests with Benjamini-Hochberg adjustment were used following Kruskal-Wallis analyses.

To compare untreated and treated runoff at Site 1, peak and total discharge from the Reference Inflow, Inflow 1a, and Outflow 1 were evaluated using mixed-effects models with site as a fixed effect and sampling date as a random intercept. Normality of residuals was assessed using the Anderson-Darling test and homogeneity of variances was assessed using Levene's test. When normality assumptions were not met, aligned rank-transformed mixed-effects models were used.

For water quality comparisons among the Reference Inflow, Inflow 1a, and Outflow 1, repeated-measures ANOVAs were conducted using balanced datasets that included only dates on which all three locations were sampled. Sphericity was assessed using Mauchly's test, and Greenhouse-Geisser corrections were applied when violated. When parametric assumptions could not be satisfied, Friedman tests were used. When significant differences among the Reference Inflow, Inflow 1a, and Outflow 1 were detected, pairwise comparisons were conducted to identify which locations differed from each other. Estimated marginal means were used as a post-hoc test for the repeated measures ANOVAs (parametric) and paired Wilcoxon signed-rank tests were used for the Friedman's tests (non-parametric). A Benjamini-Hochberg adjustment was applied to the pairwise contrasts top account for multiple comparisons and control the false discovery rate for each parameter.

Comparisons of upstream and downstream water quality measurements at both the untreated reference outfall and treated bioretention outfall were evaluated using paired t-tests when paired differences were normally distributed and Wilcoxon signed-rank tests when paired differences were not normally distributed, with the test statistic reported as V. P-values were adjusted for multiple comparisons across water quality parameters using the Benjamini-Hochberg adjustment.

4.3 Results

4.3.1 Winter Snow Pack Depth, Volume, and Pollutant Concentrations.

Snow accumulation was greater in 2023 than 2024 (Table 4-3; Figure 4-3). The snow depth at the end of the snowpack sampling periods (April 2023 and March 2024, corresponding to the last pre-melt sampling dates) was 60 cm in 2023 and 15 cm in 2024. The liquid equivalent of the accumulated snow at the end of the snowpack sampling period was 202 mm in 2023 and 36 mm in 2024. The water content of the accumulated snow at the end of the snowpack sampling period was 3.4 mm/cm in 2023 and 2.5 mm/cm in 2024. The mean monthly depth of the winter snowpack was 33 ± 8 cm in 2023 and 14 ± 2 cm in 2024, with a significantly lower monthly snow depth in 2024 than in 2023 (p = 0.003). The mean monthly liquid equivalent of the winter snowpack was 102 ± 24 mm in 2023 and 33 ± 5 mm in 2024, with significantly lower monthly volumes in 2024 than in 2023 (p = 0.01). The mean monthly water content of the snowpack was 3.3 ± 0.3 mm/cm in 2023 and 2.5 ± 0.2 mm/cm in 2024 and was not significantly different between years (p = 0.24).

Table 4-3 Statistical results from comparisons of snowpack depth, volume and water content between years.

Parameter	Test Result
Snowpack depth	$t(3) = 8.8$, $p = 0.003$
Liquid equivalent	$t(3) = 5.8$, $p = 0.01$
Water content	$t(3) = 1.5$, $p = 0.24$

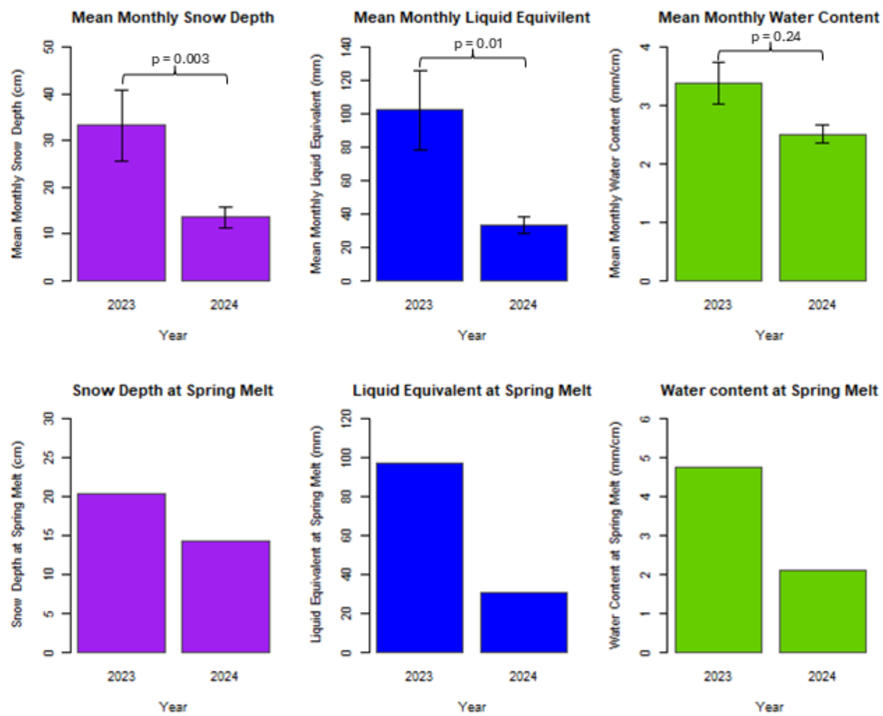


Figure 4-3 Mean monthly snow depth (cm), liquid water equivalent (mm), and water content (mm/cm), and the late winter snow depth (cm), liquid equivalent (mm), and water content (mm/cm) of the snowpack in 2023 and 2024.

Snowpack pollutant concentrations differed among snow core locations, with higher conductivity, turbidity, dissolved organic carbon, and chloride concentrations in roadside snowbanks relative to reference conditions (Table 4-4; Figure 4-4, with letters in Figure 4-4 denoting statistical differences at $\alpha = 0.05$). Within each catchment, snowpack conductivity ($p < 0.0001$), turbidity ($p < 0.0001$), suspended solids ($p < 0.0001$), dissolved organic carbon (DOC; $p = 0.01$), chloride ($p < 0.0001$) concentrations were significantly higher in snowbanks than in open fields and residential yards ($p < 0.0001$; $p < 0.0001$; $p < 0.0001$; $p = 0.02$; $p < 0.0001$) and the snow accumulated on the surface of each bioretention system ($p < 0.0001$; $p = 0.0003$; p < 0.0001; $p = 0.007$; $p < 0.0001$). Ammonia ($p = 0.03$) concentrations were also higher from snowbanks than the surface of bioretention systems ($p = 0.02$). Differences in pH ($p = 0.13$), nitrate ($p = 0.43$), nitrite ($p = 0.13$), phosphate ($p = 0.13$), zinc ($p = 0.8$), and iron ($p = 0.27$) concentrations were not significantly different among snow core locations (i.e., roadside snowbanks, fields, and the surface of bioretention systems). Median concentrations and interquartile ranges used to support the non-parametric analyses presented above are provided in the Supplementary Material.

Table 4-4 Statistical results from comparisons of snowpack pollutant concentrations.

Parameter	Test Result
pH	$F(2, 78) = 2.7$, $p = 0.13$
Conductivity	$\chi^2(2) = 45$, $p < 0.0001$
Chloride	$F(2, 78) = 87$, $p < 0.0001$
Turbidity	$F(2, 78) = 53$, $p < 0.0001$
Suspended solids	$\chi^2 = 54$, $p < 0.0001$
Dissolved organic carbon (DOC)	$\chi^2 = 11$, $p = 0.01$
Ammonia	$\chi^2 = 7$, $p = 0.03$
Nitrate	$F(2, 6) = 1.1$, $p = 0.43$
Nitrite	$\chi^2 = 4.7$, $p = 0.13$
Phosphate	$F(2, 78) = 2.4$, $p = 0.13$
Zinc	$F(2, 6) = 0.23$, $p = 0.8$
Iron	$F(2, 6) = 1.9$, $p = 0.27$
Chloride	$F(2, 78) = 87$, $p < 0.001$

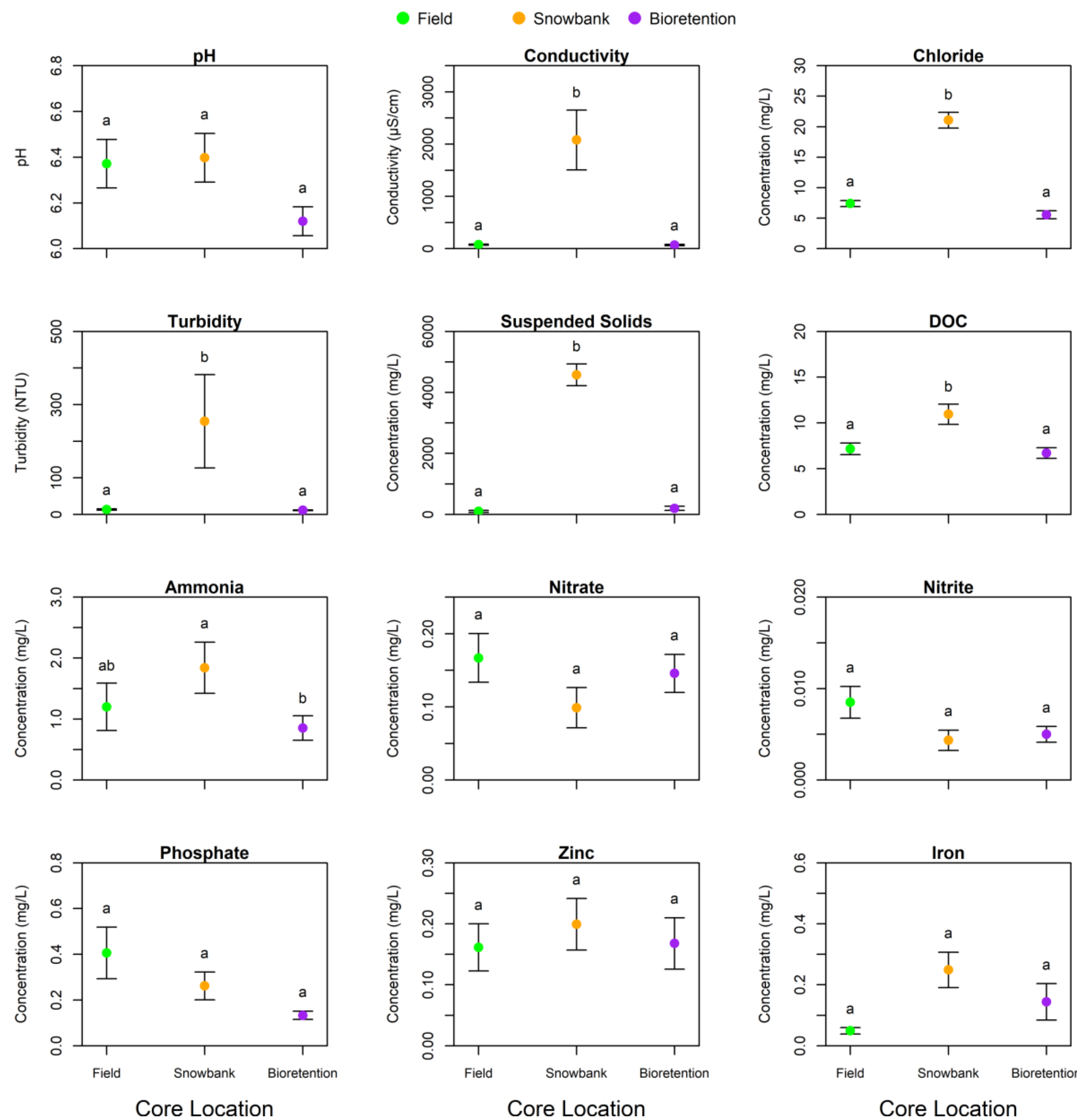


Figure 4-4 pH, conductivity, chloride, turbidity, suspended solids, dissolved organic carbon (DOC), ammonia, nitrate, nitrite, phosphate, zinc and iron concentrations in the winter snowpack.

4.3.2 Water Quantity Performance

As bioretention treatment performance is influenced by inflow discharge, peak and total inflow discharge from the bioretention systems were compared to support the interpretation of their performance (Table 4-5, Figure 4-5). Meltwater entered the first cell of Bioretention System 1 (Inflow 1a), the second cell into Bioretention System 1 (Inflow 1b), and Bioretention System 2 (Inflow 2) on 35, 29, and 45 dates, respectively. The mean peak daily discharge entering Inflows 1a, 1b, and 2 was 12 ± 3, 11 ± 3, and 9.5 ± 1.8 L/s, while the median peak daily discharge entering Inflows 1a, 1b, and 2 was 3.8 (IQR: 2.3-5.1), 0.93 (IQR: 0.87-5.6), and 0.87 (IQR: 0.55-22). The median peak daily discharge entering the three inflows was not significantly different (p = 0.12). The mean daily total discharge entering Inflows 1a, 1b, and 2 was 326 ± 94 m^3/day, 189 ± 68 m^3/day, and 629 ± 122 m^3/day, while the median total daily discharge entering Inflows 1a, 1b, and 2 was 239 (IQR: 72-304) m^3/day, 72 (IQR: 65-164) m^3/day, and 87 (IQR: 27-1696) m^3/day. The median daily total discharge entering the three inflows was not significantly different (p = 0.09). Resultant of difficulties deploying a water level logger in the outflow of Bioretention System 3 (discussed in 2.2), peak and total discharge was not quantified at this site.

Table 4-5 Statistical results from comparisons of peak and total discharge from bioretention system inflows.

Hydrological Parameter	Test Result
Peak daily discharge	$\chi^2(2) = 4.3$, p = 0.12
Total daily discharge	$\chi^2(2) = 4.8$, p = 0.09

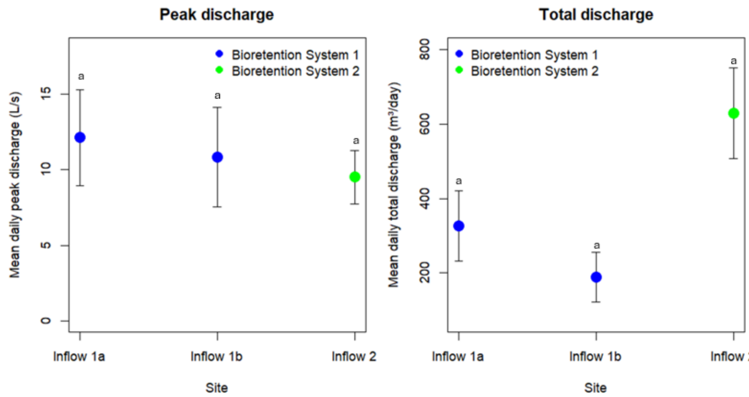


Figure 4-5 Mean daily peak (L/s) and total (m3/day) discharge from inflows into the two cells of Bioretention System 1 (Inflow 1a and Inflow 1b) and the inflow into Bioretention System 2 (Inflow 2) during the 2023 and 2024 spring melt periods.

Peak and total discharge exiting the outflows of Bioretention Systems 1 and 2 were lower than they were entering the bioretention systems, although reductions were only significant for Bioretention System 1 (Table 4-6; Figure 4-6). The mean peak discharge exiting Bioretention System 1 (Outflow 1; 1.8 ± 0.7 L/s) was significantly lower than entering Bioretention System 1 (Inflow 1; 21 ± 4 L/s; $p < 0.0001$). The mean peak discharge exiting Bioretention System 2 (Outflow 2; 2.4 ± 0.3 L/s) was lower than entering Bioretention System 2 (Inflow 2; 9.5 ± 1.8 L/s), but the result was not significant at $\alpha = 0.05$. The mean total discharge entering Bioretention System 1 (Inflow 1) was 482 ± 106 m^3/day, while the mean total discharge exiting Bioretention System 1 (Outflow 1) was 90 ± 42 m^3/day ($p < 0.0001$). The mean total discharge entering Bioretention System 2 (Inflow 2) was 629 ± 122 m^3/day, while the mean total discharge exiting Bioretention System 2 (Outflow 2) was 157 ± 21 m^3/day ($p = 0.16$). In Figure 4-6 *** indicates significance at $\alpha = 0.001$ and n.s. indicates no significance at $\alpha = 0.05$.

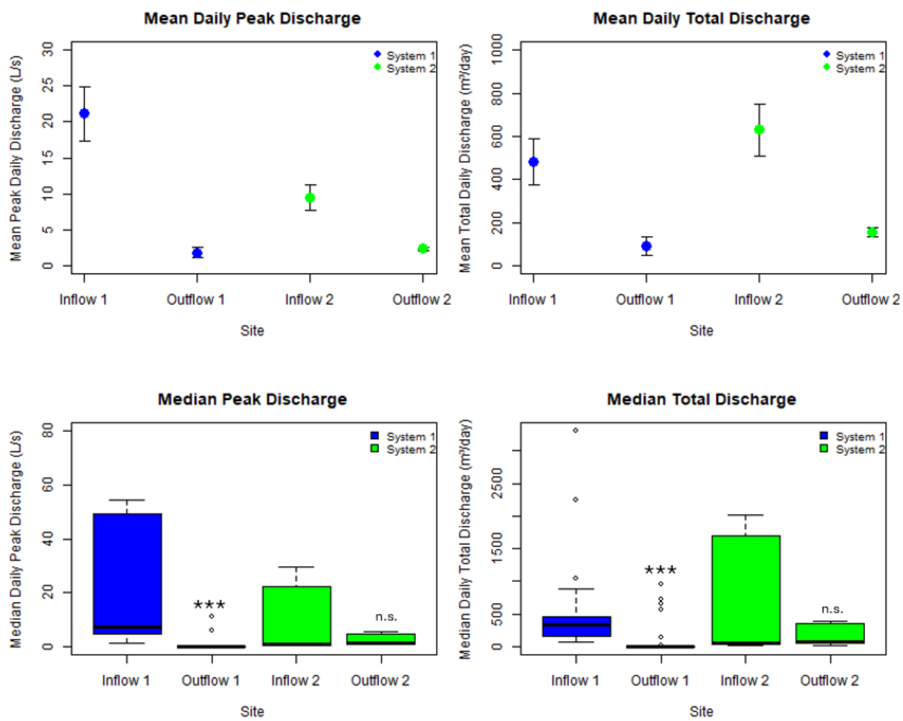


Figure 4-6 Mean daily peak discharge (L/s), mean daily total discharge (m^3/day), median daily peak discharge, and median daily total discharge from the inflows and outflows of Bioretention Systems 1 and 2 during the 2023 and 2024 spring melt period.

Table 4-6 Statistical results from comparisons of peak and total discharge from bioretention system inflows.

Hydrological Parameter	Site	Test Result
Peak daily discharge	Bioretention System 1	W = 1065, $p < 0.0001$
	Bioretention System 2	W = 803, p = 0.17
Total daily discharge	Bioretention System 1	W = 1060, $p < 0.0001$
	Bioretention System 2	W = 789, p = 0.16

Bioretention System 1 reduced the mean daily peak discharge by 91 ± 4% (95% CI: 90-100%), while Bioretention System 2 reduced the mean daily peak discharge by 75 ± 6% (95% CI: 66-77%) (Figure 4-7). Similarly, the mean daily total discharge was reduced by 81 ± 10% in Bioretention System 1 (95% CI: 72-100%) and by 75 ± 7% in Bioretention System 2 (95% CI: 67-77%). Percent reductions are reported without associated p-values because they represent cumulative reductions throughout the spring melt period rather than from independent or event-paired comparisons. Uncertainty was therefore quantified using nonparametric bootstrap resampling to generate 95% confidence intervals. Additional supporting information, including site-specific rating curves and spring melt sampling date information is provided in the Supplementary Materials.

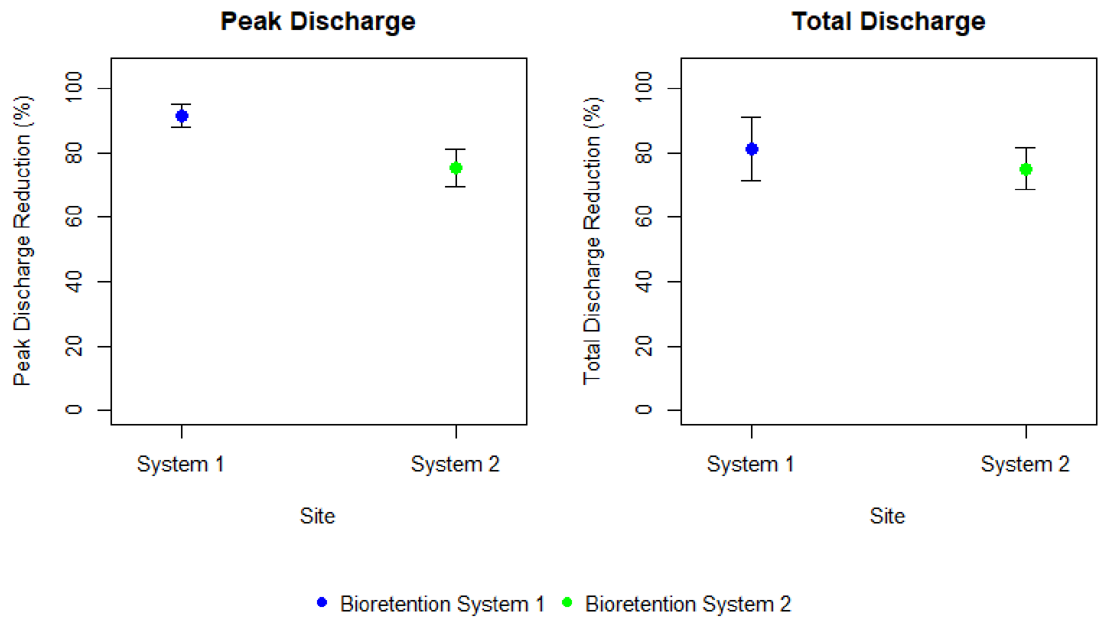


Figure 4-7 Reduction (%) in peak (left) and total (right) discharge from the bioretention systems at Site 1 and Site 2 during the 2023 and 2024 spring melt periods.

4.3.3 Inflow Quality Comparison

As influent concentrations can influence treatment performance, influent water quality was compared among the four monitored inflow sites (Table 4-7, Figure 4-8). The pH (p = 0.02) was significantly lower from Inflow 1b than from Inflows 1a (p = 0.0001) and 2 (p = 0.0001). Turbidity (p = < 0.0001) from Inflow 1b was significantly lower than from Inflows 1a (p = 0.0001), 2 (p = 0.0004) and 3 (p = 0.004). Suspended solids concentrations (p = 0.0001) from Inflow 1b were significantly lower than from Inflows 1a (p = 0.0001), 2 (p = 0.004) and 3 (p = 0.0002). Ammonia concentrations (p = 0.01) from Inflow 1b were significantly lower than from Inflows 1 (p = 0.003) and 3 (p = 0.006). Nitrate concentrations (p = 0.02) from Inflow 1a were significantly lower than from Inflow 2 (p = 0.02). Zinc concentrations ($p < 0.0001$) were significantly higher from

Inflow 2 than from Inflow 1a ($p < 0.0001$), 1b ($p < 0.0001$) and 3 ($p < 0.01$). Chloride concentrations ($p = 0.02$) from Inflows 2 ($p = 0.02$) and 3 ($p = 0.006$) were significantly higher than from Inflow 1b. Figure 4-8 shows the mean water quality measurements entering each bioretention system inflow relative to established water quality guidelines and ecological benchmarks (Table 4-2).

Table 4-7 Statistical results from comparisons of water quality parameters from bioretention system inflows.

Parameter	Test Result
pH	$F(3, 19) = 4.7$, $p = 0.02$
Conductivity	$F(3, 19) = 2.4$, $p = 0.1$
Chloride	$F(3, 19) = 4.7$, $p = 0.02$
Turbidity	$F(3, 19) = 19$, $p < 0.0001$
Suspended Solids	$F(3, 17) = 16$, $p = 0.0001$
DOC	$\chi^2(3) = 8.3$, $p = 0.05$
Ammonia	$F(3, 19) = 6$, $p = 0.01$
Nitrate	$F(3, 19) = 5.3$, $p = 0.02$
Nitrite	$\chi^2(3) = 8.9$, $p = 0.05$
Phosphate	$F(3, 19) = 2.5$, $p = 0.1$
Zinc	$F(3, 19) = 24$, $p < 0.0001$
Iron	$F(3, 19) = 3.3$, $p = 0.05$

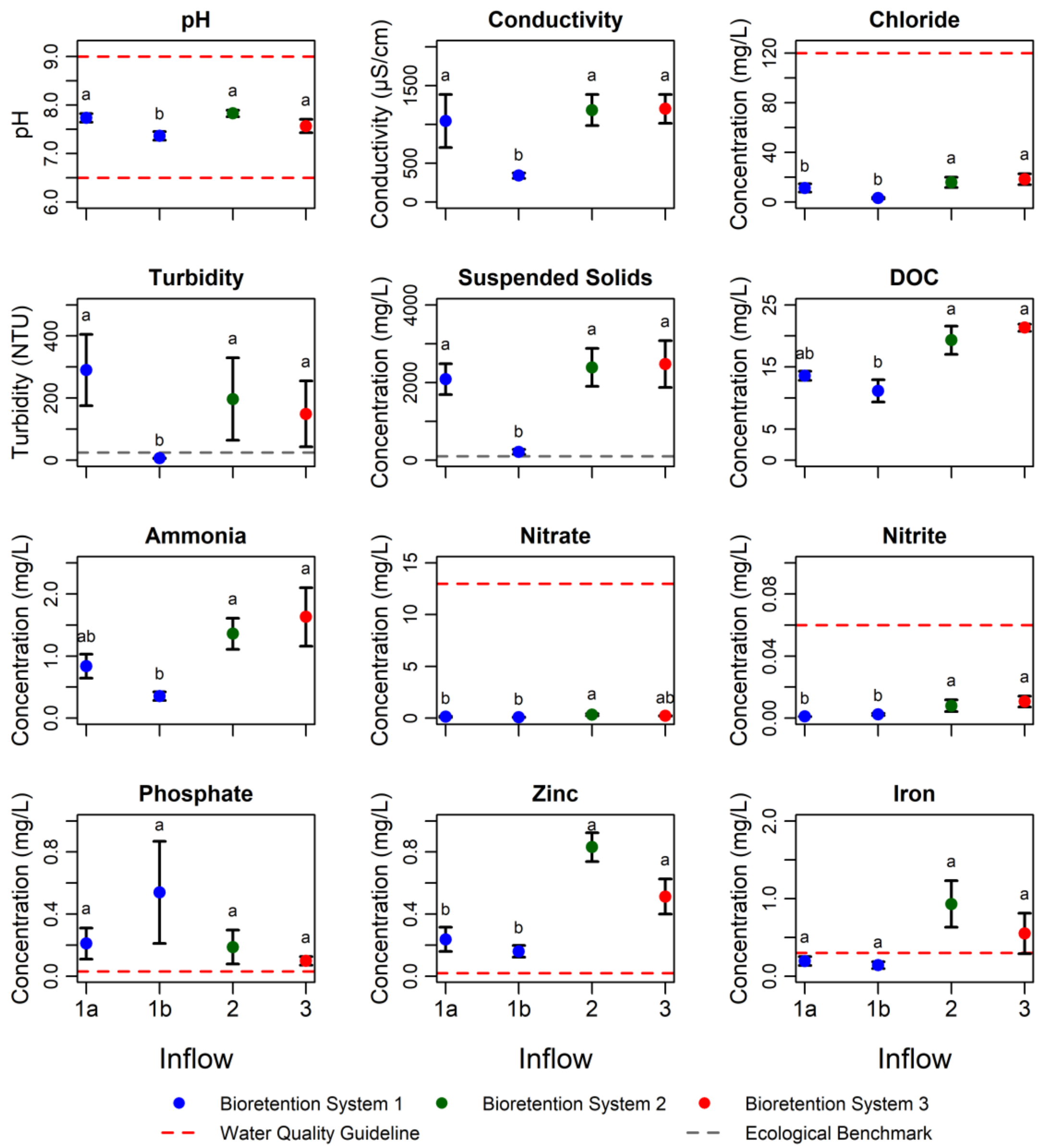


Figure 4-8 pH, conductivity, turbidity, suspended solids, dissolved organic carbon (DOC), ammonia, nitrate, nitrite, phosphate, zinc, iron, and chloride concentrations entering the inflows of the bioretention systems during the spring melt.

4.3.4 Inflow/Outflow Comparison

Relative to the influent, the bioretention systems significantly reduced effluent pH, turbidity, suspended solids, and dissolved organic carbon (DOC) concentrations with site-specific decreases in zinc concentrations (Table 4-8; Figure 4-9). The pH from the effluent was significantly lower in Bioretention Systems 1 (p = 0.002), 2 (p = 0.01), and 3 (p = 0.04). The turbidity from the effluent was significantly lower from Bioretention Systems 1 (p = 0.006) and 2 (p = 0.02). Turbidity from the effluent of Bioretention System 3 was also lower than the turbidity from the influent, although the difference was not significant (p = 0.29). Similarly, suspended solids concentrations from the effluent were significantly lower from Bioretention Systems 1 (p < 0.0001), 2 (p < 0.0001), and 3 (p = 0.0009). DOC concentrations from the effluent of Bioretention Systems 1 (p = 0.02), 2 (p = 0.003), and 3 (p = 0.01) were significantly lower than influent concentrations. Relative to influent concentrations, zinc concentrations from the effluent decreased from Bioretention System 2 (p = 0.04), increased from Bioretention System 1 (p = 0.03), and did not significantly change from Bioretention System 3 (p = 0.88).

Table 4-8 Statistical results from comparisons in outflow water chemistry relative to inflow water chemistry during melt events.

Parameter	Site	Test Result
pH	1	$t(20) = 3.5$, $p = 0.002$
	2	$t(10) = 3$, $p = 0.01$
	3	$t(12) = 2.3$, $p = 0.04$
Conductivity	1	$W = 77$, $p = 0.29$
	2	$t(12) = 0.27$, $p = 0.79$
	3	$t(12) = 3e\text{-}16$, $p = 1$
Chloride	1	$t(20) = -3.8$, $p = 0.001$
	2	$t(12) = -4e\text{-}16$, $p = 1$
	3	$t(12) = -2e\text{-}17$, $p = 0.98$
Turbidity	1	$t(20) = 3.1$, $p = 0.006$
	2	$t(12) = 2.6$, $p = 0.02$
	3	$t(12) = 1.1$, $p = 0.29$
Suspended Solids	1	$t(20) = 6$, $p < 0.0001$
	2	$t(12) = 6.6$, $p < 0.0001$
	3	$t(12) = 4.4$, $p = 0.0009$
Dissolved Organic Carbon (DOC)	1	$t(20) = 2.5$, $p = 0.02$
	2	$t(12) = 3.7$, $p = 0.003$
	3	$t(12) = 2.9$, $p = 0.01$
Ammonia	1	$t(20) = -0.58$, $p = 0.57$
	2	$t(12) = -0.28$, $p = 0.78$
	3	$t(12) = 0.19$, $p = 0.85$
Nitrate	1	$t(20) = -1.6$, $p = 0.13$
	2	$t(12) = 0.35$, $p = 0.73$
	3	$t(12) = -1.7$, $p = 0.11$
Nitrite	1	$W = 38$, $p = 0.06$
	2	$t(12) = -0.98$, $p = 0.35$
	3	$t(12) = -0.7$, $p = 0.5$
Phosphate	1	$t(20) = -2.1$, $p = 0.05$
	2	$t(12) = -1.2$, $p = 0.25$
	3	$t(12) = -1.2$, $p = 0.26$
Zinc	1	$t(20) = -2.5$, $p = 0.03$
	2	$t(12) = 2.4$, $p = 0.04$
	3	$t(12) = -0.16$, $p = 0.88$
Iron	1	$t(20) = -1.4$, $p = 0.17$
	2	$t(12) = 0.24$, $p = 0.81$
	3	$t(12) = 0.07$, $p = 0.95$

In contrast, relative to the influent, effluent chloride concentrations increased from one of the bioretention systems, while conductivity, ammonia, nitrate, nitrite, phosphate, zinc, and iron concentrations were not significantly different (Table 4-8; Figure 4-9). Chloride concentrations increased from Bioretention System 1 ($p = 0.001$) but were not significantly different from Bioretention Systems 2 ($p = 1$) or 3 ($p = 0.98$). Conductivity (System 1: $p = 0.29$, System 2: $p = 0.79$, System 3: $p = 1$), ammonia (System 1: $p = 0.57$, System 2: $p = 0.78$, System 3: $p = 0.85$), nitrate (System 1: $p = 0.13$, System 2: $p = 0.73$, System 3: $p = 0.11$)), nitrite (Systems 1: $p = 0.06$, System 2: $p = 0.35$, System 3: $p = 0.5$), phosphate (System 1: $p = 0.05$, System 2: $p = 0.25$, System 3: $p = 0.26$)), and iron (System 1: $p = 0.17$, System 2: $p = 0.81$, System 3: $p = 0.95$)) concentrations were not significantly different between the influent and effluent of any of the bioretention systems. The water quality results shown below were plotted relative to established water quality guidelines and other ecological benchmarks (Table 4-2). Additional supporting figures and tables for the spring melt sampling dates and water quality comparisons are provided in the Supplementary Materials.

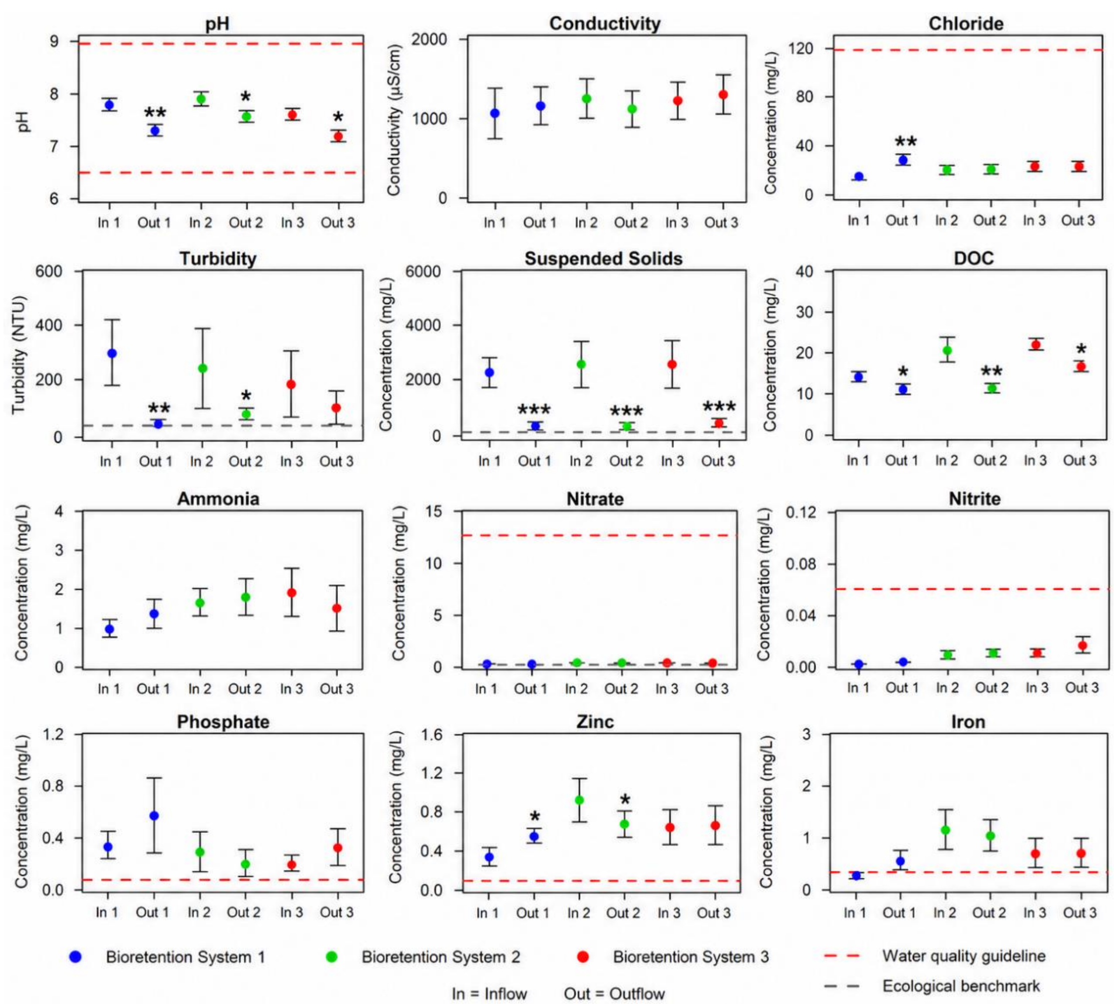


Figure 4-9 pH, conductivity, turbidity, suspended solids, dissolved organic carbon (DOC), ammonia, nitrate, nitrite, phosphate, zinc, iron, and chloride concentrations from the inflows and outflows of the three bioretention systems during the spring melt.

The bioretention systems reduced the number and percentage of melt events in which turbidity and suspended solids concentrations exceeded ecological benchmarks (Table 4-9). The turbidity entering the inflows of Bioretention Systems 1, 2, and 3 exceeded 25 NTU for 91%, 100%, and 86% of melt events, while the turbidity exiting the outflows of Bioretention Systems 1, 2, and 3 exceeded this benchmark for 36%, 71%, and 29% of melt events. Bioretention System 1 also reduced the percentage of events that exceeded the 100 mg/L benchmark for suspended solids. All 11 melt events had inflow concentrations exceeding 100 mg/L, but only 7 had outflow concentrations exceeding 100 mg/L. In contrast, suspended solids concentrations at the inflow and outflow of Bioretention Systems 2 and 3 exceeded this benchmark during all melt events.

The pH, nitrate, and nitrite concentrations in the influent and effluent of all three bioretention systems did not exceed CCME water quality guidelines. Temperature- and pH-based CCME exceedances for ammonia concentrations were unchanged between the influent and effluent, with two exceedances in both the influent and effluent of Bioretention Systems 1 and 2. As the temperature was not measured for Bioretention System 3, ammonia exceedances could not be calculated for this site.

The bioretention systems also did not reduce the number and percentage of melt events that exceeded CCME water quality guidelines for phosphate, zinc, or iron. The phosphate concentrations entering Bioretention Systems 1, 2, and 3 exceeded both PWQO and CCME guidelines during 46%, 100%, and 71% of events, respectively, while the phosphate concentrations exiting Bioretention Systems 1, 2, and 3 exceeded these guidelines during 91%, 71%, and 100% of events, respectively. Zinc concentrations entering Bioretention System 1 exceeded the 110 µg/L (0.11 mg/L) CCME guideline during 82% of melt events, while those entering Bioretention Systems 2 and 3 exceeded this guideline during all melt events. Zinc concentrations from the effluent of Bioretention Systems 1, 2, and 3 exceeded the 110 µg/L guideline during all melt events. The iron concentrations entering Bioretention Systems 1, 2, and 3 exceeded the CCME water quality guideline of 0.11 mg/L during 18%, 57%, and 43% of melt events, respectively. Iron concentrations from the effluent exceeded the guideline during 46%, 71%, and 43% of melt events, respectively. Lastly, none of the chloride concentrations from the influent or effluent of any of the three bioretention systems exceeded the short-term (640 mg/L) or long-term (120 mg/L) CCME water quality guidelines.

Table 4-9 Frequency of spring melt events in which the inflow and outflow of the bioretention systems exceed water quality guidelines and ecological benchmarks.

Guideline	Site	Number of Inflow Deviations	% of Events Deviating	Number of Outflow Deviations	% of Events Deviating
pH (below 6.5)	1	0	0	0	0
	2	0	0	0	0
	3	0	0	0	0
pH (above 9)	1	0	0	0	0
	2	0	0	0	0
	3	0	0	0	0
Turbidity	1	10	91	4	36
	2	7	100	5	71
	3	6	86	2	29
Suspended solids	1	11	100	7	64
	2	7	100	5	100
	3	7	100	7	100
Ammonia	1	2	18	2	18
	2	2	29	2	29
Nitrate	1	0	0	0	0
	2	0	0	0	0
	3	0	0	0	0
Nitrite	1	0	0	0	0
	2	0	0	0	0
	3	0	0	0	0
Phosphate	1	5	46	10	91
	2	7	100	5	71
	3	5	71	7	100
Zinc	1	9	82	11	100
	2	7	100	7	100
	3	7	100	7	100
Iron	1	2	18	5	46
	2	4	57	5	71
	3	3	43	3	43
Chloride	1	0	0	0	0
	2	0	0	0	0
	3	0	0	0	0

3.5 Pollutant Reductions

The pH, turbidity, and concentrations of suspended solids and dissolved organic carbon from the effluent were reduced relative to the influent of all three bioretention systems. While Section 3.4 evaluated whether water quality measurements differed between the influent and effluent of each bioretention system, expressing treatment performance as reductions relative to influent concentrations allowed the magnitude of treatment responses to be compared among bioretention system (Table 4-10; Figure 4-10). The pH was reduced by 0.5 ± 0.1, 0.34 ± 0.08, and 0.43 ± 0.25 at Sites 1, 2, and 3, respectively, with no significant differences in reductions between sites (p = 0.57). The turbidity was reduced by 58 ± 12%, 58 ± 7%, and 4 ± 64% at Sites 1, 2, and 3, respectively, with no significant differences in reductions between sites (p = 0.92). Suspended solids concentrations were reduced by 88 ± 4%, 87 ± 3%, and 85 ± 3% at Sites 1, 2, and 3, respectively, with no significant differences in reductions between sites (p = 0.82), and dissolved organic carbon concentrations were reduced by 25 ± 9%, 39 ± 10%, and 19 ± 5% at Sites 1, 2, and 3, respectively, with no significant differences in reductions between sites (p = 0.54).

Table 4-10 Statistical results from comparisons of pollutant reductions between bioretention systems.

Parameter	Test Result
pH	$F(2,21) = 0.89$, $p = 0.57$
Conductivity	$F(2,15) = 50$, $p < 0.0001$
Chloride	$F(2,21) = 2.2$, $p = 0.32$
Turbidity	$F(2,22) = 0.09$, $p = 0.92$
Suspended Solids	$F(2,21) = 0.2$, $p = 0.82$
Dissolved Organic Carbon (DOC)	$F(2,21) = 1.4$, $p = 0.54$
Ammonia	$F(2,21) = 0.89$, $p = 0.57$
Nitrate	F(2,17) = 4.9, p = 0.06
Nitrite	$F(2,21) = 0.2$, $p = 0.82$
Phosphate	$F(2,21) = 5.4$, $p = 0.05$
Zinc	$F(2,19) = 11$, $p = 0.004$
Iron	$F(2,20) = 1.1$, $p = 0.57$

Zinc, ammonia, nitrate, and phosphate concentrations showed mixed results, with reductions at some sites and increases at others. Zinc concentrations were reduced by 36 ± 8% at Site 2 but increased with negative reductions of -1061 ± 555% and -27 ± 44%

at Sites 1 and 3 (p = 0.004). Reductions were significantly higher at Site 2 than at Site 1 (p = 4e-4), but there were no significant differences between the other sites. Ammonia concentrations were reduced by 3.1 ± 12% at Site 2 and had negative reductions of -79 ± 44% and -8.4 ± 27% at Sites 1 and 3, respectively. However, reductions did not differ significantly (p = 0.57). Nitrate concentrations were reduced by 1 ± 12% at Site 2 and had negative reductions of -3539 ± 2075% and -64 ± 28% at Sites 1 and 3, respectively, but reductions were not significantly different (p = 0.06). Phosphate concentrations were reduced by 19 ± 21% at Site 2 and had negative reductions of -1114 ± 383% and -91 ± 36% at Sites 1 and 3, but reductions were not significantly different (p = 0.05).

Conductivity, iron, nitrite, and chloride concentrations in the effluent were higher than those in the influent at all three bioretention systems, resulting in negative percent reductions. Large negative percent reductions primarily occurred when influent concentrations were very low relative to effluent concentrations. Conductivity increased with reductions of -129 ± 44%, -1.8 ± 13.5%, and -11 ± 14.4%, at Sites 1, 2, and 3, respectively (p < 0.0001), with significantly lower reductions from Site 1 than from Sites 2 (p < 0.0001) and 3 (p < 0.0001), but no significant difference in reductions from Sites 2 and 3 (p = 0.92). Iron concentrations increased with reductions of -899 ± 797%, -44 ± 55%, and -15 ± 37%, at Sites 1, 2, and 3, respectively, with no significant differences in reductions between sites (p = 0.57). Nitrite concentrations also increased by -155 ± 128%, -200 ± 102%, and -49 ± 23% at Sites 1, 2, and 3, respectively, with no significant difference between sites (p = 0.82). Lastly, chloride concentrations increased, with reductions of -633 ± 263%, -256 ± 195%, and -43 ± 33%, respectively, with no significant differences in the reductions between sites (p = 0.32).

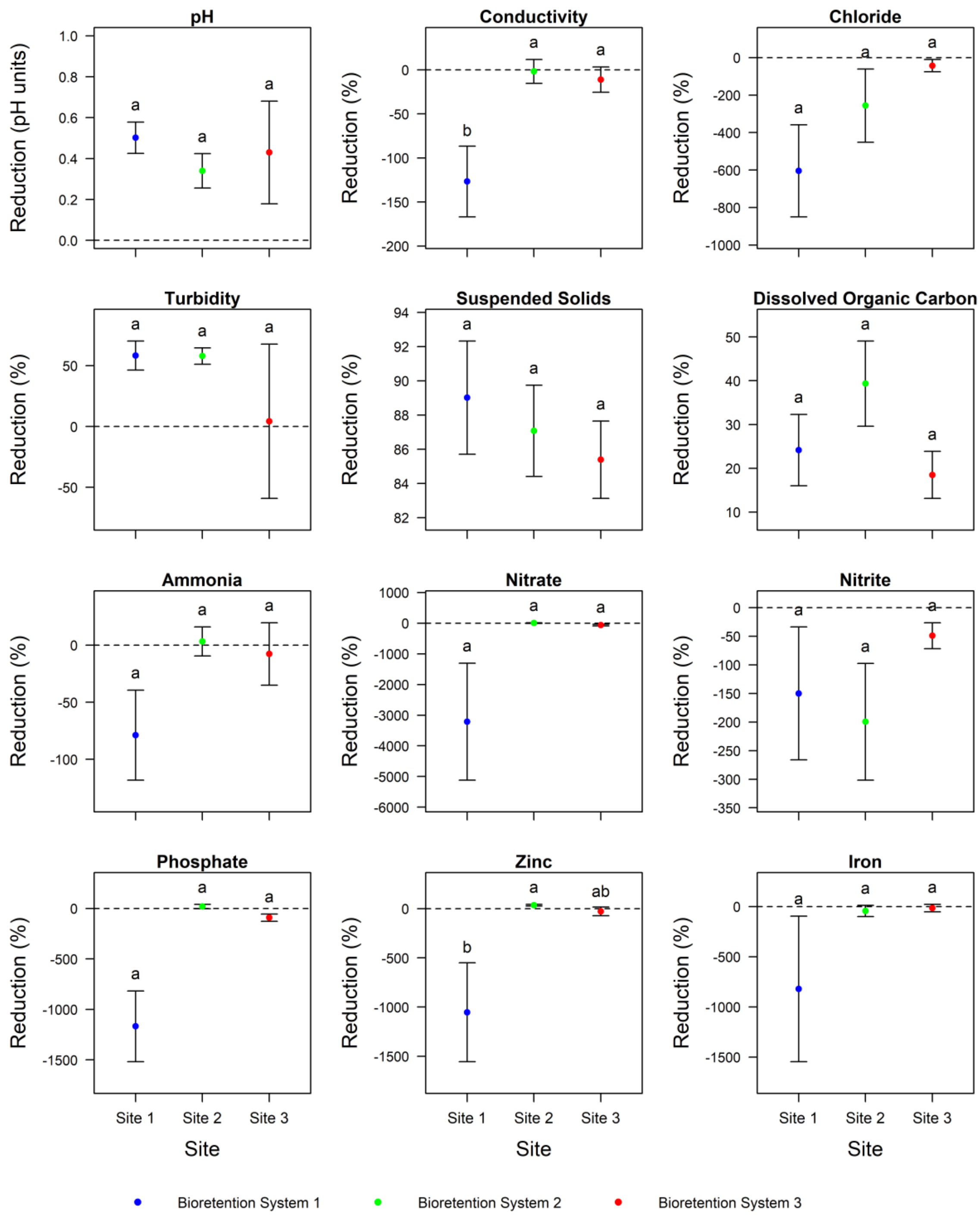


Figure 4-10 Reductions in water quality parameters from the effluent of the bioretention systems relative to the influent.

4.3.5 Water quantity and quality comparison of runoff from the bioretention and reference sites.

The daily peak and total discharge from the untreated reference outfall (Reference Inflow), the inflow into Bioretention System 1 prior to treatment (Inflow 1a), and outflow from Bioretention System 1 (Outflow 1) were compared to identify differences in meltwater discharge entering the same receiving stream under treated and untreated conditions to support the interpretation of potential changes in downstream water quality (Table 4-11; Figure 4-11). Results showed significant differences in peak discharge from the reference inflow, the bioretention system inflow, and the bioretention system outflow ($p < 0.0001$). There was higher peak discharge from the reference inflow ($p < 0.0001$) and from the bioretention system inflow ($p < 0.0001$) than from the bioretention system outflow, and higher peak discharge from the reference inflow than from the bioretention system inflow ($p = 0.002$).

Table 4-11 Statistical results from comparisons of peak and total discharge from the bioretention and reference sites.

Parameter	Test Result
Peak discharge	$F(2, 50.1) = 61$, $p < 0.0001$
Total discharge	$F(2, 77) = 44$, $p < 0.0001$

The results also showed significant differences in the total discharge from the reference inflow, the bioretention system inflow, and the bioretention system outflow ($p < 0.0001$) (Table 4-10). There was higher total discharge from the reference inflow ($p < 0.0001$) and the bioretention system inflow ($p < 0.0001$) than from the bioretention system outflow, and higher total discharge from the reference inflow than from the bioretention system inflow ($p < 0.0001$). Since sampling dates were biased towards higher-flow melt events, the peak and total discharge values in Figure 4-11 differ from those in 3.2, which describe discharge conditions across the entire melt period.

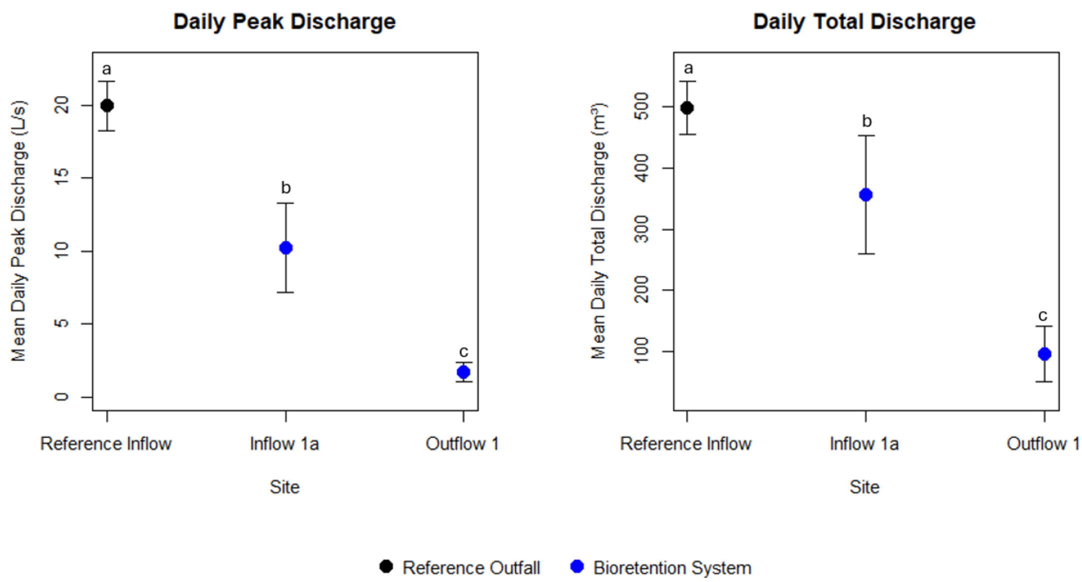


Figure 4-11 Mean daily peak (L/s) and total (m3/day) discharge into the untreated reference inflow (Reference Inflow), into the bioretention system (Inflow 1a) and out of the bioretention system (Outflow 1).

Smilarly, meltwater quality measurements from the untreated reference outfall (Reference Inflow), the inflow into the bioretention system prior to retention (Inflow 1a) and the outflow from the bioretention system post-retention (Outflow 1) were compared to evaluate differences between unretained runoff and runoff retained through bioretention processes entering the same receiving stream. The effluent from the outflow of Bioretention System 1 (Out 1) had significantly lower turbidity (p = 0.006) and suspended solids concentrations (p < 0.0001) than from the reference inflow (Ref) (p = 0.03) and the inflow into the bioretention system (In 1a) (p = 0.004) (Table 4-12; Figure 4-12). The pH (p = 0.03) from the bioretention system outflow was significantly lower than from the bioretention system inflow (p = 0.009). The pH from the reference inflow was not significantly different from the pH from the bioretention system inflow (p = 0.11) or bioretention system outflow (p = 0.11). Similarly, the turbidity (p = 0.006) from the reference inflow and the bioretention system inflow were not significantly different (p = 0.21), but the bioretention system outflow had lower turbidity than from the bioretention system inflow (p = 0.004) and the reference inflow (p = 0.03). The suspended solids concentrations ($p < 0.0001$) from the reference inflow and the inflow into the bioretention system were not significantly different ($p = 0.2$), but the suspended solids concentrations from the bioretention system outflow was significantly lower than the suspended solids concentrations from the bioretention system inflow ($p < 0.0001$) and the reference inflow ($p = 0.0001$).

Table 4-12 Statistical results from comparisons of water quality results from the bioretention and reference sites.

Parameter	Test Result
pH	$F(1.3, 10) = 6.2$, $p = 0.03$
Conductivity	$F(1.3, 11) = 1.2$, $p = 0.32$
Chloride	$F(1.7, 13) = 12$, $p = 0.002$
Turbidity	$F(1.8, 14) = 8$, $p = 0.006$
Suspended Solids	$F(1.7, 13) = 24$, $p < 0.0001$
Dissolved Organic Carbon (DOC)	$F(1.1, 8.7) = 2.6$, $p = 0.15$
Ammonia	$F(1.6, 13) = 1.8$, $p = 0.22$
Nitrate	$F(1.4, 11) = 1.9$, $p = 0.2$
Nitrite	$\chi^2 = 3.1$, $p = 0.21$
Phosphate	$F(1.2, 10) = 6.4$, $p = 0.03$
Zinc	$F(1.5, 12) = 7.6$, $p = 0.01$
Iron	$F(1.6, 13) = 1.3$, $p = 0.31$

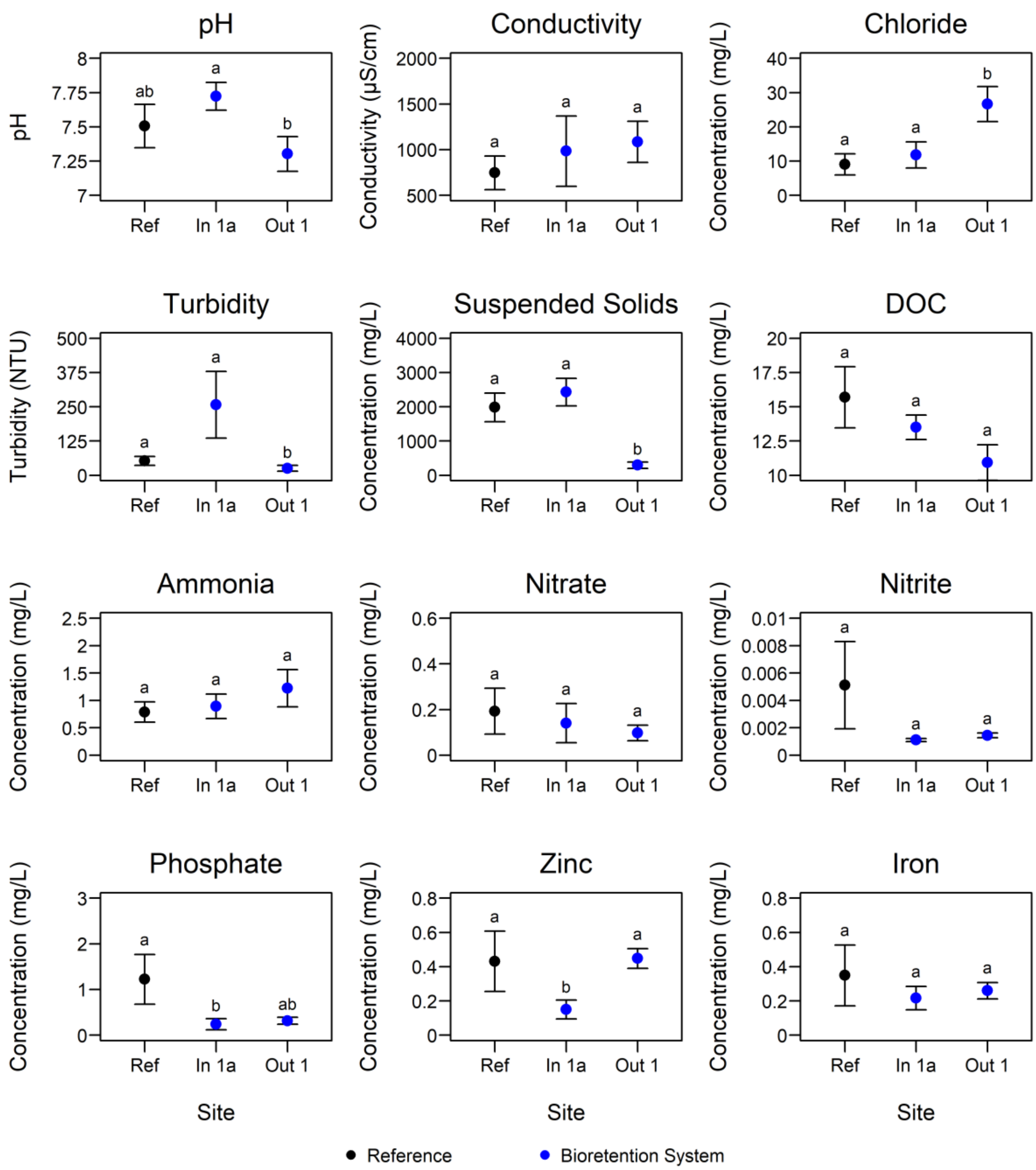


Figure 4-12 Water quality entering the untreated reference inflow (Reference Inflow), the bioretention system inflow (Inflow 1a) and exiting the bioretention system outflow (Outflow 1).

4.3.6 Instream Water Quality Response at Untreated Reference Site

Water quality measurements collected upstream and downstream of the untreated reference outfall were compared to evaluate whether untreated meltwater inputs alter downstream water chemistry (Table 4-13). Across all water quality parameters, there were no significant differences between measurements taken upstream and downstream of the untreated reference outfall. While conductivity increased slightly from 904 ± 129 µS/cm upstream of the outfall to 918 ± 132 µS/cm downstream of the outfall, the increase was not significant after correcting for multiple comparisons (raw p = 0.01; BH-adjusted p = 0.06). The pH (p = 0.76), turbidity (p = 0.46), and concentrations of suspended solids (p = 0.71), dissolved organic carbon, ammonia (p = 0.71), nitrate (p = 0.71), nitrite (p = 0.71), phosphate (p = 0.71), zinc (p = 0.71), iron (p = 0.71), and chloride (p = 0.71) were not significantly different downstream of the reference outfall relative to upstream of the reference outfall.

Table 4-13 Statistical results from water quality parameters upstream and downstream of the reference outfall.

Parameter	Test Result
pH	V = 20, p = 0.76
Conductivity	t(8) = −3.3, p = 0.06
Chloride	t(7) = 0.82, p = 0.71
Turbidity	t(8) = −1.8, p = 0.46
Suspended Solids	t(8) = 1.2, p = 0.71
Dissolved Organic Carbon (DOC)	t(7) = 0.12, p = 0.91
Ammonia	t(8) = −0.54, p = 0.71
Nitrate	t(8) = 0.62, p = 0.71
Nitrite	t(7) = 0.73, p = 0.71
Phosphate	t(8) = −1.1, p = 0.71
Zinc	t(8) = 0.48, p = 0.71
Iron	t(8) = 0.69, p = 0.71

4.3.7 Instream Water Quality Response at Bioretention Treatment Site

Water quality measurements collected upstream and downstream of Bioretention System 1 were also compared to assess whether the effluent from the bioretention system post retention altered downstream water chemistry (Table 4-14). Across all water quality parameters, there was no evidence of a change in water quality downstream of the bioretention system relative to upstream of the bioretention system. Specifically, the pH (p = 0.69), conductivity (p = 0.21), turbidity (p = 0.33), and concentrations of suspended solids (p = 0.76), dissolved organic carbon (p = 0.81), ammonia (p = 0.66), nitrate (p = 0.13), nitrite (p = 0.21), phosphate (p = 0.76), zinc (p =

0.13), iron (p = 0.33), and chloride (p = 0.76) were not significantly downstream relative to upstream of the bioretention system outflow.

Table 4-14 Statistical results from water quality parameters upstream and downstream of Bioretention System 1.

Parameter	Test Result
pH	V = 42, p = 0.69
Conductivity	$t(7)$ = 1.9, p = 0.21
Chloride	V = 39, p = 0.76
Turbidity	$t(8)$ = 1.3, p = 0.33
Suspended Solids	V = 40, p = 0.76
Dissolved Organic Carbon (DOC)	$t(8)$ = -0.25, p = 0.81
Ammonia	$t(10)$ = -0.55, p = 0.66
Nitrate	$t(9)$ = 2.5, p = 0.13
Nitrite	$t(7)$ = -1.9, p = 0.21
Phosphate	V = 18, p = 0.76
Zinc	$t(7)$ = 2.8, p = 0.13
Iron	$t(9)$ = −1.3, p = 0.33

4.4 Discussion

This study assessed pollutant accumulation in roadside snowbanks, the water quantity performance of two bioretention systems and the water quality performance of three bioretention systems during the spring melt period, and the potential for downstream water quality improvements in a northern urban stream. Results show that snowbanks adjacent to roadways have significantly higher conductivity, turbidity, and concentrations of suspended solids, dissolved organic carbon (DOC), and chloride than nearby field and bioretention sites. These pollutants are released into urban streams during the spring freshet, a critical time for coldwater fish species such as brook and rainbow trout, increasing their vulnerability to sublethal and acute toxic effects. During the spring freshet, the bioretention systems demonstrated reductions in runoff volume and concentrations of some water quality parameters (e.g., turbidity, suspended solids, and dissolved organic carbon (DOC)), with pH generally shifting toward more neutral values. However, the bioretention systems were less effective at removing nutrients, chloride, and some metals, as reflected by the persistent frequency of water quality guideline exceedances for phosphate, zinc, and iron. I did not identify any significant changes in water quality upstream and downstream of the untreated reference outfall and the bioretention system, likely due to dilution, short mixing distances, or sampling constraints.

4.4.1 Roadside snowbanks have higher conductivity, turbidity and concentrations of suspended solids, dissolved organic carbon, and chloride compared to the surrounding snowpack.

I found that roadside snowbanks accumulate significantly higher pollutant concentrations, likely due to vehicular traffic and winter road maintenance practices (e.g., salting and sanding). Compared to field and bioretention sites, roadside snowbanks had significantly higher conductivity, turbidity, and concentrations of dissolved organic carbon (DOC), suspended solids, and chloride. Since conductivity (temperature-corrected specific conductance) is frequently used as a proxy for chloride concentrations (Chanat & Custer, 2026; Fanelli et al., 2025), the higher conductivity and chloride concentrations observed in roadside snowbanks are consistent with road salt inputs from winter de-icing practices (Van Meter & Ceisel, 2024; Foley & Steinman, 2023). The increased dissolved organic carbon concentrations observed in roadside snowbanks may reflect vehicle emissions, as snow cover is highly susceptible to soot (black carbon) and associated organic carbon (González-Correa et al., 2022). However, contrary to expectations, metal concentrations were not higher in roadside snowbanks. This may reflect the relatively low traffic volumes in the residential study catchments, as higher concentrations of metals in roadside snow have been reported in areas with higher traffic intensity (Vijayan et al., 2024; Kuoppamäki et al., 2014).

The higher turbidity, conductivity, and concentrations of suspended solids, DOC, and chloride in roadside snowbanks can pose substantial ecological risks when released during the spring freshet. Elevated turbidity can reduce fish activity, increase time spent in refuge areas, and inhibit foraging success (Hood et al., 2025; Leris et al., 2022). Similarly, increased suspended solids concentrations can reduce immune response and feeding rates, impair growth and survival, and disrupt physiological functions (Park et al., 2025b). Elevated DOC concentrations may reduce food availability (Tonin et al., 2025) and promote the formation of metal-DOC complexes, thereby reducing metal toxicity and bioavailability (Gao et al., 2022; Hameed et al., 2019). High conductivity can also cause osmotic stress and may increase the toxicity of certain contaminants (Fanelli et al., 2024). Chloride from road salts can increase salmonid embryo mortality (Winter et al., 2025) and reduce zooplankton survival and reproduction, potentially impacting food web productivity (Woodley et al., 2023). While chloride and associated conductivity pose ecological risks, chloride is not consistently removed by bioretention systems (Goor et al., 2021; Shetty et al., 2020), complicating their management during the spring melt. In Northwestern Ontario, runoff of these contaminants during melt events coincides with rainbow trout (Oncorhynchus mykiss) spawning and fry emergence (DFO, 2013), increasing the vulnerability of early life stages. These impacts, combined with temporal overlap with spawning periods, highlight the importance of managing snowmelt quality through bioretention controls and reduced road salt applications to mitigate pollutant inputs to fish-bearing streams in northern urban settings.

4.4.2 Bioretention systems reduce peak and total discharge volumes during the spring freshet.

Both Bioretention Systems 1 and 2 reduced peak and total discharge volumes during the spring freshet. Bioretention System 1, with a larger catchment, slightly higher impervious cover and greater storage capacity, achieved greater peak and total discharge reductions than Bioretention System 2. These differences highlight the importance of system design features, such as storage volume, ponding depth, and contributing area, in determining runoff-reduction performance (Le Cauchois et al., 2025; Li et al., 2021). High runoff volumes can increase stream flashiness (Johnson et al., 2022; Hopkins et al., 2019) and release fine sediments that can scour spawning substrates, reducing egg-to-fry survival (Johnson et al., 2025). Reducing runoff volumes may, therefore, help protect vulnerable fish habitats and promote more stable hydrologic conditions typical of less-urbanized catchments (Hopkins et al., 2022). Additionally, as pollutant loads depend on discharge volumes, reduced discharge volumes may lower the overall mass loading of particulate-bound contaminants entering receiving waters (Carvalho et al., 2022), protecting sensitive aquatic ecosystems. Overall, these findings show that bioretention systems reduce the quantity of stormwater runoff entering fish habitats during the spring freshet.

4.4.3 Bioretention systems consistently reduce pH, turbidity, suspended solids, and dissolved organic carbon concentrations, but show limited or inconsistent performance for nutrients and chloride during spring freshet.

The bioretention systems consistently reduced turbidity, suspended solids, and dissolved organic carbon (DOC) concentrations during the spring freshet, while pH values generally shifted toward more neutral conditions. Shifts in pH may be attributed to carbon dioxide dissolution in soil porewater, which promotes the formation of carbonic acid during infiltration and can lower effluent pH under certain conditions (Ferdush et al., 2023), contributing to the observed shift toward more neutral pH values. Reductions in turbidity and suspended solids indicate that bioretention systems effectively remove particulate-bound contaminants through sedimentation, infiltration, and filtration processes (Bhurtyal & Ahmari, 2025). Ecological improvements in turbidity and suspended solids concentrations are reflected by reductions in the frequency with which the effluent turbidity exceeded 25 NTU associated with reduced growth (Sigler et al., 1984; CCME, 2022), salmonid density (Sigler et al., 1984), and behavioural effects (Berg & Northcote, 1985; Gregory & Northcote, 1993), and reductions in the frequency with which suspended solids concentrations from Bioretention System 1 exceeded 100 mg/L associated with sublethal effects in adult and juvenile fish (CCME, 2002), while no reductions in suspended solids exceedances were observed at Bioretention Systems 2 and 3.

These reductions in turbidity and suspended solids concentrations are important because elevated turbidity can increase energy expenditure during foraging for visual predators (Zanghi & Ioannou, 2025), reduce prey capture and survival (Illing et al., 2024), and decrease exploration while increasing time spent in shelter (Zöttl et al., 2025). Reductions in DOC may reflect adsorption onto mineral surfaces in the bioretention soil (Amenkhienan et al., 2025; Wang et al., 2024) and co-precipitation during iron oxidation and hydroxide formation (Wang et al., 2024). This is important because DOC can influence metal mobility and bioavailability through complexation, altering dissolved metal transport and exposure risks to aquatic organisms (Li et al., 2026; Broadley et al., 2019).

Despite reductions in pH, turbidity, suspended solids, and DOC concentrations, the bioretention systems demonstrated limited or inconsistent reductions in nutrient (ammonia, nitrate, nitrite, phosphate) and chloride concentrations, as well as in certain metal concentrations (zinc and iron), as reflected by the persistent frequency of water quality guideline exceedances for phosphate, zinc, and iron. Bioretention System 1 showed mixed results for ammonia concentrations, sometimes showing higher effluent concentrations, which may be attributed to cold, saline, or anoxic conditions that suppress nitrification or promote ammonification in the bioretention soil media (Hu et al., 2025). This is consistent with the unchanged frequency of temperature- and pH-dependent ammonia exceedances between the inflow and outflow. Similarly, chloride showed limited removal, with effluent concentrations often comparable to or higher than in the influent at all three bioretention systems, suggesting that accumulated salts from previous melt events were remobilized during the spring melt and released in the effluent of the bioretention systems (Goor et al., 2021; Shetty et al., 2020). These increases in chloride concentrations may also reduce phosphorus removal (Brown et al., 2024). The inconsistent removal of nutrients, chloride, and certain metals underscores the need to optimize bioretention systems for cold-climate conditions, which feature high salt loading and reduced biological activity. Modifications such as deeper or amended media profiles and salt-tolerant vegetation may enhance pollutant retention during winter and early spring (Hu et al., 2025).

4.4.4 Instream Water Quality and Ecological Implications

Although Bioretention System 1 reduced peak and total discharge, turbidity, suspended solids, and dissolved organic carbon concentrations, no significant upstream-downstream differences in stream water quality were observed at either the treated bioretention outflow or the untreated reference outfall. Therefore, the instream component of this study should be interpreted as an event-scale assessment of detectable short-term water quality responses, rather than a complete evaluation of downstream water quality improvement.

The lack of detectable differences in water quality between the instream sampling locations may be due to several factors. Firstly, high stream velocity and discharge can dilute the instream pollutant concentrations, leading to undetectable differences in downstream water quality between the untreated effluent and the bioretention system outflow (Maas et al., 2023; Kaushal et al., 2022). Secondly, stream samples were collected six metres downstream of the untreated stormwater outfall and bioretention effluent. I used this short six-metre distance between the upstream and downstream sampling points to reduce the confounding influence of additional downstream infrastructure. However, it may not be far enough to allow for adequate mixing in the stream to capture localized changes in pollutant concentrations.

The discrete grab sampling method used to assess instream water quality may fail to capture short-duration pulses of pollutants, such as chloride and metals that exhibit rapid, event-driven variability (Popick et al., 2022; Lindfors et al., 2020; Galfi et al., 2017). As a result, the absence of observed differences in downstream water quality, particularly at the untreated reference outfall, does not necessarily indicate the absence of ecological risk. Short-term exposure to elevated pollutant concentrations can still result in acute and sublethal effects on aquatic organisms. Accordingly, relying solely on chemical parameters to assess instream habitat conditions should be approached with caution. Biological assessments (e.g., fish health, macroinvertebrate communities) are essential for determining whether stormwater interventions meaningfully protect downstream aquatic life. In cold-climate urban streams, where snowmelt runoff coincides with sensitive fish life stages, a combination of improved sampling strategies and biological indicators may provide a more complete understanding of treatment effectiveness and ecological outcomes.

4.4.5 Management Implications and Future Directions

While the bioretention systems reduced peak and total discharge volumes and improved some water quality parameters, they did not consistently reduce chloride and nutrient concentrations. These pollutants can pose ecological risks during sensitive life stages of coldwater fish species, particularly during the spring freshet. Although chloride concentrations in this study did not exceed CCME water quality guidelines, the limited capacity of bioretention systems to remove dissolved ions highlights the importance of preventing potential impacts associated with elevated salt and nutrient loading, particularly in areas with higher road salt application. As such, bioretention systems should be complemented by snow management and land-use strategies throughout the catchment to reduce chloride and nutrient inputs at the source. For example, reducing road salt application (McPhillips et al., 2023), using less toxic de-icing agents (Gillis et al., 2021; Fay & Shi, 2012), and relocating snow storage areas away from hydrologically connected surfaces (McClymont et al., 2023) can reduce salt runoff before it enters bioretention systems.

In cold-climate areas, where bioretention systems show limited infiltration during the early thaw, bioretention system design can be improved by incorporating larger storage capacities and deeper soil media to accommodate lower seasonal infiltration and maximize surface storage (Hu et al., 2025; Goor et al., 2021). Soil amendments, such as biochar or zeolite, may further enhance nutrient and metal retention under high salt loads (Hu et al., 2025; Goor et al., 2021). Additionally, adjustable outlet structures can increase residence times and improve pollutant retention (Shetty et al., 2020), particularly during frozen conditions when infiltration is limited. Lastly, fish habitat assessments and benthic macroinvertebrate surveys (Cadmus et al., 2018), longer-term monitoring, and approaches to isolate stormwater inputs from other watershed stressors should be incorporated alongside water quality measurements to further enhance our understanding of bioretention systems' impact on downstream ecosystem health.

4.5 Conclusion

This study evaluated pollutant accumulation in the winter snowpack and the performance of three bioretention systems at reducing the quantity and improving the quality of meltwater entering trout-sensitive urban tributaries during the spring freshet. Roadside snowbanks were a source of elevated conductivity, turbidity, suspended solids, dissolved organic carbon, and chloride, which are released during the spring freshet, coinciding with critical life stages of coldwater fish such as rainbow trout. While the bioretention systems effectively reduced peak and total discharge volumes and improved water quality for several particulate-associated pollutants, they were less effective at reducing chloride and nutrient concentrations, with evidence of increased effluent concentrations.

These findings should be interpreted as concentration-based treatment responses rather than pollutant load reductions. Pollutant loads were not calculated because meltwater samples were collected as single daily grab samples rather than daily mean, event mean, or flow-weighted composite concentrations. Future spring melt monitoring should pair continuous discharge measurements with flow-weighted composite sampling or repeated within-day sampling to quantify pollutant mass removal across the spring freshet.

I could not detect downstream changes in water quality, potentially due to dilution of downstream samples or limitations of conventional grab sampling. In the future, improved snow management and bioretention system design may contribute to reducing pollutant loads and improving fish habitat conditions in northern urban catchments. A more holistic, watershed-scale approach that incorporates bioretention systems to reduce runoff volumes and particulate contaminant loads, along with alternative land-use practices such as reducing road salt and fall fertilizer applications to reduce nutrient and chloride loads, is needed to ensure meaningful ecological outcomes.

5 An Assessment Framework for Prioritizing Stormwater Management in Sensitive Fish Habitats

Abstract

Urban stormwater runoff carries nutrients, metals, and fine sediments into streams, degrading water quality and threatening fish habitat. Constrained by limited resources, managers often struggle to determine where stormwater interventions will have the greatest ecological benefit. Here, I describe an assessment framework to combine stormwater hotspot data, fish occupancy records, and field habitat surveys to identify priority sites for green stormwater infrastructure (GSI) interventions. I applied this framework to McVicar Creek, a trout-sensitive tributary in Thunder Bay, Ontario. Habitat scores downstream of six stormwater outfalls ranged from 3.4 to 4.1 on a five-point scale, showing that some reaches provide favourable physical habitat conditions for fish. By linking stormwater hotspot locations with downstream habitat conditions, the framework directed stormwater management interventions to locations where GSI retrofits are most likely to support fish habitat protection. This approach gives managers a practical, low-cost tool to guide stormwater improvements and support stream conservation in urban watersheds.

AI Disclosure

Illustrations used in the habitat scoring metric (Table 1) were generated with the use of Artificial Intelligence (AI) tools (ChatGPT).

5.1 Introduction

In urban environments, pollutants are entrained during rainfall and spring melt events and released into nearby streams. Once in the stream, these pollutants can lower dissolved oxygen concentrations, alter pH, and introduce toxic substances into sensitive aquatic ecosystems (Mustafa et al., 2024). Untreated urban runoff can also cause acute toxicity and mortality in salmonids, demonstrating the risk that stormwater pollutants pose to sensitive fish habitats (Mustafa et al., 2024; Chow et al., 2019; French et al., 2022). Fine sediment exposure during early development can reduce growth, alter metabolism, and impair survival-related performance in salmonids (Louhi et al., 2023). Urban catchments with dense road networks and high traffic intensity have also been associated with increased coho salmon mortality, while untreated roadway runoff has been shown to cause acute mortality in juvenile coho, steelhead, and Chinook salmon (French et al., 2022; Chow et al., 2019; Feist et al., 2018). Juvenile coho, steelhead, and Chinook salmon exposed to untreated urban stormwater runoff may not fully recover even after transfer to clean water with mortality in steelhead and Chinook delayed by one to two days (French et al., 2022). To protect sensitive fish and their habitats, these findings underscore the urgent need to identify effective stormwater management strategies. Given the association between urban runoff, road density, traffic intensity, and fish mortality risk, green stormwater infrastructure (GSI) practices are important for reducing runoff-related stressors to fish habitats.

GSI practices, particularly bioretention systems, have demonstrated effectiveness in reducing water quality stressors associated with urban runoff. Bioretention systems can reduce several runoff-related pollutants, including suspended solids and nutrients under cold-climate field or controlled experimental conditions (Kratky et al., 2021; Ding et al., 2019), and metals in snowmelt runoff (Muthanna et al., 2007). However, reported performance varies across studies, designs, monitoring approaches and site conditions. At the catchment scale, dispersed stormwater controls have been shown to reduce phosphorus concentrations and summer stream temperatures in areas where effective imperviousness was sufficiently reduced (Walsh et al., 2022). Experimental studies have shown that water quality improvements resulting from GSI implementation can prevent acute mortality in coho salmon (McIntyre et al., 2015; Spromberg et al., 2016; McIntyre et al., 2023) and reduce the sublethal effects of stormwater runoff on fish (McIntyre et al., 2015; McIntyre et al., 2014), including developmental delays and deformities in fish embryos (Young et al., 2018; McIntyre et al., 2014). However, other studies report limited improvements in streamflow or biotic integrity (Roy et al., 2014), with some studies reporting increased nitrate loading downstream (Zhang et al., 2023). As such, bioretention systems should be strategically implemented in locations where they are most likely to reduce runoff-related stressors to fish habitat.

Spatial planning tools have been used to identify priority locations for green stormwater infrastructure (GSI) across a range of objectives. One approach is to prioritize sites based on stormwater-management objectives, including runoff reduction, pollutant-load reduction, and the placement of GSI practices where they are expected to reduce sediment and nutrient loading to receiving waters. For example, GIS-based approaches have been used to identify priority locations for GSI by combining hydrologically sensitive areas with land use and land cover suitability, and by estimating potential reductions in total phosphorus, total nitrogen, and total suspended solids as key decision criteria (Martin-Mikle et al., 2015). Other studies have expanded beyond water quantity and quality benefits by evaluating multiple green infrastructure benefits, including reducing social vulnerability, increasing access to green space, improving air quality, mitigating urban heat island effects, and improving landscape connectivity (Meerow & Newell, 2017; Meerow, 2019). In these approaches, the spatial planning tool is strongly shaped by the objective of the analysis, meaning that priority locations can differ depending on which benefit is emphasized (Meerow, 2019).

Several methods have also been developed to manage, combine, and analyze spatial data for prioritizing GSI locations. Geographic information systems (GIS)-based multi-criteria evaluation can be used to identify trade-offs among green infrastructure priorities, particularly when different criteria are weighted according to stakeholder or planning objectives (Meerow & Newell, 2017). These tools have also been used to determine the spatial suitability of GSI by combining implementation opportunities, including biophysical, socio-economic, planning, and governance criteria, with ecosystem service needs (Kuller et al., 2019). Together, these studies show that GSI prioritization is not a single standardized process, but a flexible decision-support approach in which the selected criteria, weights, and spatial datasets depend on the management objective being prioritized.

In this study, the prioritization objective differs from approaches focused primarily on hydrologic performance, social vulnerability, implementation suitability, or broader co-benefits because this framework is designed to identify stormwater intervention locations where ecological benefits to sensitive fish habitats are expected to be greatest. Prioritizing benefits for fish habitat requires directing stormwater management efforts to pollutant hotspots where urban runoff poses the highest risk and to sensitive habitats where fish populations are most vulnerable to pollutants. Protecting existing high-quality habitats from stormwater-related degradation may be more effective and cost-efficient than attempting to fully restore heavily impaired areas after physical habitat loss has occurred. Even small-scale GSI interventions, covering less than 1% of a sub-catchment, can reduce nutrient and sediment loads by up to 15% (Martin-Mikle et al., 2015), highlighting the benefits of prioritizing infrastructure in critical locations. For example, urban catchments with dense road networks and heavy traffic have higher fish mortality compared with catchments with lower traffic intensity and are therefore priority locations for mitigation (Feist et al., 2021).

As such, prioritizing GSI in hydrologically sensitive areas or pollutant and thermal hotspots, including high-traffic roads, commercial corridors, or sub-catchments with elevated contaminant export, can substantially enhance ecological outcomes (Martin-Mikle et al., 2015; Feist et al., 2017; Lu & Wang, 2021; Simpson et al., 2022). However, comprehensive watershed-scale assessments are often costly and time-consuming, limiting their use by municipalities with constrained budgets and urgent needs. A rapid assessment framework like the one presented here addresses these issues by offering a low-cost approach for identifying stormwater hotspots and sensitive fish habitats where targeted interventions have the potential to yield the greatest expected protection for the restoration and sustainability of urban stream ecosystems.

To identify targeted interventions, a baseline assessment of stormwater pollution sources in McVicar Creek was carried out in 2010 by researchers at Lakehead University, who mapped stormwater outfalls, characterized visible impairments, and conducted targeted water quality sampling (following the Unified Stream Assessment protocol; Stanfield, 2017). This assessment documented widespread stormwater impacts and established a watershed-scale dataset describing stormwater impairments and stormwater outfalls with increased potential to contribute pollutants to the creek. This study builds on that work by combining the 2010 stormwater impairment data with a newly developed field-based fish habitat survey and a spatial decision-making framework to evaluate where GSI retrofits would provide the greatest benefit to fish habitats.

The goal of this study was to develop a rapid assessment framework for prioritizing green stormwater infrastructure (GSI) interventions to protect sensitive fish habitats in urban catchments and apply this framework to an impaired urban stream. This spatial decision-making framework combines three complementary datasets: unpublished stormwater quality data, Ontario Ministry of Natural Resources fish population data (Friday et al., 2023), and novel field-based habitat data collected in 2025. Together, these datasets were used to: (1) identify stormwater outfalls along a trout-sensitive urban tributary exhibiting elevated pollution risk and potential stormwater impacts, (2) evaluate the ecological sensitivity of downstream reaches with respect to fish habitat condition, (3) develop a spatial decision-making framework to rank sub-catchments or outfalls based on ecological risk and the potential benefit of GSI retrofits, and (4) provide planning guidance for targeted stormwater interventions where benefits of fish habitat protection are expected to be greatest. This framework was designed to be adaptable to any urban stream to prioritize stormwater interventions in sensitive fish habitats.

5.2 Methods

To achieve the study objective of prioritizing stormwater management interventions to protect sensitive fish habitats, a four-part methodology was used. First, previously collected stormwater quality data were used to identify outfalls with increased potential to contribute pollutants to the creek, hereafter referred to as stormwater hotspots. Second, 2023 Ministry of Natural Resources (MNR) fish population data were re-analyzed to assess fish community distribution within McVicar Creek. Third, fish distribution data and stormwater hotspot location data were combined to inform a new field-based habitat assessment conducted in 2025 to evaluate the ecological sensitivity of stream reaches downstream of previously identified hotspots. Lastly, fish habitat scores were calculated to rank sites for potential green stormwater infrastructure (GSI) retrofits. This approach was designed as a rapid, low-cost assessment that uses both existing water quality and fish population data and new, targeted fish habitat surveys to guide management decisions.

5.2.1 Stormwater Quality and Impairment Identification

To identify areas with the greatest stormwater-related impairments, I adopted a previously completed baseline inventory (2010-2011) conducted by researchers at Lakehead University. This inventory highlighted the effects of stormwater discharge on the physical, chemical, and biological conditions of the creek. This inventory consisted of three parts: 1) a geographic information systems (GIS)-based analysis of the municipal storm sewer network to identify stormwater outfalls with elevated potential to contribute pollutants to the creek, 2) a Unified Stream Assessment (USA) to assess the physical attributes of the stream and identify problematic outfalls based on field observations, and 3) targeted wet- and dry-weather sampling at stormwater outfalls (unpublished dataset; used with permission from supervisor, Dr. Robert Stewart).

As a starting point, researchers completed a GIS analysis and Unified Stream Assessment to identify problematic stormwater outfalls and narrow the large number of stormwater outfalls discharging into the stream to a smaller subset of candidate sites. GIS analysis included mapping storm sewer infrastructure to determine storm sewer length, pipe diameter, and the number of storm sewer inputs (catch basins) contributing to each outfall location. These variables were used as indicators of stormwater connectivity, which is commonly associated with increased runoff volumes, pollutant delivery, and altered stream hydrology in urban catchments. The Unified Stream Assessment (USA) is an established protocol developed by the Centre for Watershed Protection that provides a standardized qualitative analysis of stream health to identify problem outfalls along the creek based on field observations. Following this protocol, researchers conducted a qualitative inventory of stormwater effluent from 91 stormwater outfalls. For each outfall, researchers documented discharge, colour, odour, downstream impact, and visual observations (e.g., presence of garbage, suds, oils, etc.), and used these observations to classify stormwater outfalls on a five-point scale, with 5 being the most severe.

Based on the results of the GIS analysis and qualitative stream inventory, eight stormwater outfalls were selected for targeted water quality sampling. Targeted water quality sampling was conducted following the U.S. Geological Survey National Field Guide Manual for the Collection of Water-Quality Data (U.S. Geological Survey, 2006). Water quality samples were collected in October during both dry (no precipitation) and wet (precipitation > 5 mm) weather events from the stormwater outfall locations. Physical (pH, conductivity, total dissolved solids, and total suspended solids) and chemical water quality parameters, including total alkalinity, dissolved organic carbon, chloride, total ammonia ($N\text{-}NH_4^+ + NH_3$), nitrite ($NO_2\text{-}N$), nitrate ($NO_3\text{-}N$), phosphate ($PO_4\text{-}P$), sulphate (SO_4), and total metal concentrations (Al, As, Ba, Be, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, V, and Zn), were analyzed to quantify pollutant concentrations at stormwater outfalls. Surface water quality results were evaluated relative to the Ontario Provincial Water Quality Objectives (PWQO) and Canadian Environmental Quality Guidelines (CEQG) for the protection of aquatic life, with U.S. EPA criteria applied where Canadian guidelines were unavailable.

5.2.2 Fish Occupancy Assessment

Fish community distribution in McVicar Creek was assessed using data from the Ontario Ministry of Natural Resources (MNR; Friday et al., 2023). In this study, electrofishing was conducted between August 11-19, 2022, at 13 sites spanning fragmented and unfragmented reaches of the creek using a Smith Root LR24/Apex backpack electrofisher. A single pass was completed at each site, with all fish captured, identified, measured, and released, and species presence and abundance reported by site. For our analysis, the species counts were summed to represent the total fish abundance at each site. Subsites within Site 11 (11a-11i) were aggregated by summing species counts and reach lengths to create a single continuous reach. Total fish counts were then standardized by reach length (fish per metre) by dividing the total fish count by the reach length to enable comparisons across sites of different reach lengths. The standardized values were subsequently mapped in ArcGIS Pro to illustrate spatial patterns of fish occupancy along McVicar Creek. This dataset highlights where fish abundance is concentrated or absent and provides a foundation for evaluating ecological conditions in relation to stormwater hotspots and habitat quality.

5.2.3 Fish Habitat Assessment

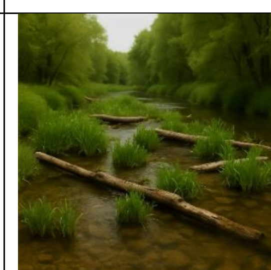
To evaluate the physical fish habitat conditions downstream of stormwater hotspots, field surveys were conducted in McVicar Creek under full-leaf conditions during the late summer of 2025. Each survey reach began at the base of the stormwater outfalls identified as a hotspot in the stormwater quality and impairment assessment (Section 2.1) and extended 40 m downstream. This reach length followed the Ontario Stream Assessment Protocol (OSAP, 2017, Section S1.M1), which recommends a minimum 40 m segment to capture at least one riffle-pool sequence. At each site, fish habitat was characterized using detailed qualitative field observations (Table 1). Descriptions included habitat complexity (e.g., the diversity and frequency of riffle, glide, run, and pool habitats), substrate type and embeddedness (e.g., the dominant substrate and degree of embeddedness by fine sediments), presence and abundance of instream habitat structures (e.g., logs, boulders, macrophytes, etc.), streambank stability, riparian vegetation condition (e.g., buffer width and continuity), sediment deposition, presence and abundance of large woody debris, habitat connectivity (e.g., evidence of barriers to fish passage), and canopy cover and shading. Observations were systematically documented to capture site-specific variation and provide ecological context for interpreting habitat quality (Field data sheet in Supplementary Materials).

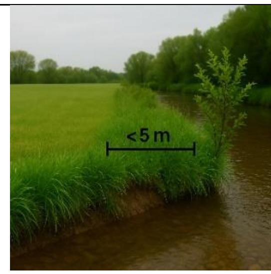
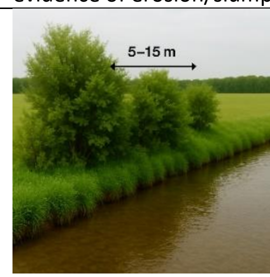
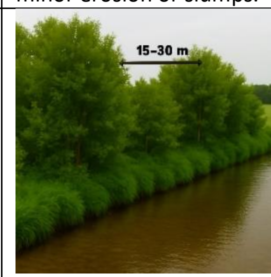
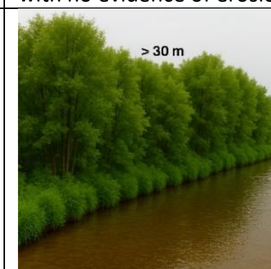
In addition to these field assessments, channel sinuosity was measured as the ratio of channel length to valley length over standardized 100 m reaches (beginning at the base of the stormwater outfall and extending 100 m downstream), using midstream measurements of channel and valley length conducted in Google Earth Pro 7.3.6. Sinuosity is a widely used descriptor of channel morphology, with values < 1.05 typically considered straight, 1.05-1.25 sinuous, and > 1.25 meandering (Leopold & Wolman, 1960; Leopold et al., 1964; Knighton, 1998). Rosgen (1996) also identified 1.2 as a critical threshold distinguishing a meandering alluvial channel. However, because McVicar Creek is a small, confined, wadable stream (2.5-3.0 m bankfull width), strict application of these thresholds would under-represent natural habitat complexity. To align with habitat assessment protocols such as the Ontario Stream Assessment Protocol (Stanfield, 2017), while also capturing site-level variation, sinuosity scores were assigned using adjusted thresholds: < 1.04 = very poor (score 1), 1.04-1.10 = poor (score 2), 1.10-1.20 = moderate (score 3), 1.20-1.30 = good (score 4), and > 1.30 = excellent (score 5).

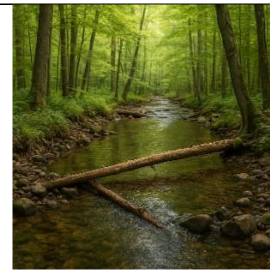
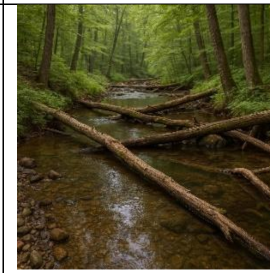
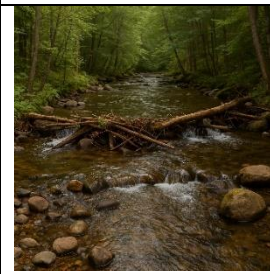
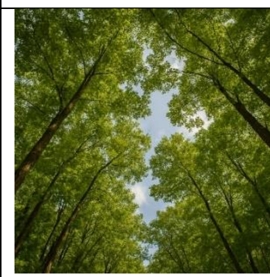
These categories preserve the conventional interpretation that greater sinuosity is generally associated with increased habitat diversity, while scaling appropriately to the morphology of small urban channels present in McVicar Creek.

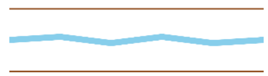
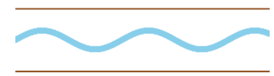
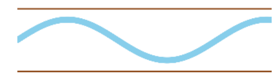
Following this descriptive assessment, each reach was evaluated using a standardized ten-metric scoring framework, adapted from the Ontario Ministry of Natural Resources (MNR) *Aquatic Habitat Inventory Surveys: Manual of Instructions* (MNR, 1987), the Department of Fisheries and Oceans' *Fish Habitat Field Guide* (DFO, 2001), and the *Ontario Stream Assessment Protocol, Version 10* (Stanfield, 2017). These ten metrics were selected to provide a comprehensive, field-based assessment of physical habitat conditions relevant to coldwater fish habitat. The selected metrics were consistent across the three reference protocols and were chosen because they collectively represent key components of stream habitat, including channel form, substrate condition, riparian structure, and habitat connectivity. Metrics were prioritized based on their ecological relevance to salmonid habitat suitability and their applicability to urban stream assessment conditions. The metrics evaluated included: (1) habitat complexity; (2) streambed composition; (3) instream habitat features; (4) bank stability; (5) riparian vegetation and buffer width; (6) sediment deposition; (7) large woody debris; (8) habitat connectivity; (9) canopy cover and shading, and (10) channel sinuosity. Each metric was scored on a five-point scale (1 = very poor to 5 = excellent) according to the thresholds (Table 5-1). Equal weighting was applied across metrics to maintain a simple, transparent and transferable scoring framework.

Table 5-1 Ten-parameter framework used to assess physical fish habitat quality.

	1 - Very Poor	2 - Poor	3 - Moderate	4 - Good	5 - Excellent
1. Habitat Complexity Diversity and frequency of riffle, glide, run, and pool habitats.	 Only a single habitat type is present, with no diversity.	 Only runs and glides are present. Pools and riffles are not present.	 Mostly runs and glides; pools/riffles are occasional or isolated.	 Moderate variety of habitat types, moderate frequency, some uneven distribution.	 Frequent, well-distributed riffles, glides and pools form a complex habitat to support various fish life stages.
2. Streambed Composition Dominant substrate material forming the streambed.	 Silt, clay, or sand, no stable gravels/cobbles present.	 Mostly fine sand/silt with scattered gravels/cobbles.	 Mixed fines and gravels/cobbles.	 Predominantly gravel/cobble substrates with minor fines.	 Gravels/cobbles/boulders provide a stable habitat.
3. Instream Habitat Features Abundance and diversity of instream habitat features (logs, boulders, macrophytes, etc.)	 No instream habitat features present	 < 10%, isolated patches	 10-25%, moderate variety.	 20-50% cover, good variety.	 >50% cover, multiple types.

	1 - Very Poor	2 - Poor	3 - Moderate	4 - Good	5 - Excellent
4. Bank Stability. Degree of erosion, undercutting, and slumping of streambanks.	 Banks are collapsing, bare soil and extensive erosion.	 Frequently, unstable banks and severe erosion occur.	 Moderately unstable, with evidence of erosion/slumps.	 Mostly stable banks, with minor erosion or slumps.	 Stable, well-vegetated banks with no evidence of erosion.
5. Riparian Vegetation and Buffer Width. Width and continuity of the riparian zone (forest and shrubs).	 No natural riparian vegetation.	 < 5 m narrow riparian buffer.	 5-15 m, patchy or disturbed.	 15-30 m, mostly intact.	 >30 m, continuous intact native vegetation (forest or grassland).
6. Sediment Deposition Extent of fine material deposited in the streambed.	 Extensive deposition; deep, continuous deposits smothering substrate (>50% coverage).	 Heavy deposition in most pools/runs (30-50% coverage).	 Mixed fines and gravels/cobbles (gravels/ cobbles 20-30% embedded by fines)	 Limited deposition (10-20% embedded or mostly fine materials).	 Minimal to no deposition. Clean gravels/cobbles with less than 10% embedded by fine sediment.

	1 - Very Poor	2 - Poor	3 - Moderate	4 - Good	5 - Excellent
7. Large Woody Debris (LWD). Abundance of fallen logs or large wood providing habitat structure.	 No LWD present.	 Very little LWD with minimal habitat contribution.	 Sparse LWD with limited function.	 Moderate LWD, contributing significantly to habitat.	 Abundant LWD, creating pools, refuge, and flow variability.
8. Habitat Connectivity. Obstructions prevent fish from moving within the reach.	 Impassible barriers (dams, impassable culverts, etc.)	 Major barriers are present, which severely restrict passage during key periods.	 Partial barriers (e.g. perched culvert or seasonal restriction).	 Minor/seasonal barriers only.	 No barriers to fish passage, fully passable within reach.
9. Canopy Cover and Shading. Percent canopy cover over stream reach.	 <5% cover.	 20-25% cover.	 25-50% cover.	 50-75% cover.	 75-100% cover.

	1 - Very Poor	**2 - Poor**	**3 - Moderate**	**4 - Good**	**5 - Excellent**
10. Channel Sinuosity. Sinuosity of the stream channel.	 Essentially straight with no habitat diversity (sinuosity <1.04).	 Slightly sinuous with little diversity (sinuosity 1.04-1.1).	 Moderately sinuous, some meanders providing habitat variety (sinuousity 1.1-1.2).	 Well-developed meanders provide good habitat variety (sinuosity 1.2-1.3).	 Highly sinuous channel (>1.3), with abundant bends and associated complexity.

The above scores were derived directly from qualitative observations made in the field, ensuring that numerical ratings reflected the physical habitat conditions present at each reach. For example, where field notes indicated riffles, runs, and pools were frequent and well-distributed, the reach was scored as excellent (5) for habitat complexity. Where observations recorded gravels heavily embedded with fines, the substrate metric was scored as poor (2) or very poor (1), depending on the degree of embeddedness. Scores across all ten metrics were averaged to yield a total habitat score out of 5, with higher values indicating greater habitat sensitivity. This combined approach ensured that results retained the richness of descriptive field observations while also providing a transparent, comparable scoring system suitable for integration into the prioritization framework (Section 2.4).

5.2.4 Prioritization Framework

All outfalls included in this study were pre-identified as stormwater hotspots exhibiting elevated pollution risk and stormwater impacts from the 2010 baseline survey. The goal of the prioritization framework was, therefore, not to re-rank outfalls by impairment severity, but to determine where stormwater retrofits would provide the greatest protection of downstream areas that provide high-quality habitat conditions for coldwater fish. This approach was based on the premise that protecting existing high-quality habitats from further degradation may provide greater ecological benefit than implementing stormwater retrofits alone in more physically degraded reaches, where concurrent stream rehabilitation may be first required to restore suitable physical habitat conditions.

Prioritization was based on a combination of quantitative habitat scores and qualitative field observations (as described in Section 2.3). Each hotspot was first assigned a habitat quality score out of 5, derived from the ten-metric framework. Outfalls with downstream reaches averaging ≥4 across metrics (representing good to excellent habitat quality) were designated as high priority retrofit locations, as these reaches were considered to provide the most suitable physical habitat conditions for fish. Sensitive fish species are generally more likely to inhabit, spawn, and successfully recruit in reaches with favourable habitat characteristics (e.g., canopy cover and shading, sinuous channels, low erosion, and clean spawning cobbles); therefore, discharge from stormwater outfalls in these reaches was considered more likely to pose ecological risks to fish communities if left untreated. Reaches with average scores around 3 were assigned moderate priority, while those scoring ≤2 across multiple metrics were considered low priority, as they lack many of the physical characteristics required to support coldwater fish habitat and were therefore less likely to support sensitive fish spawning and habitat use.

These numerical scores were also interpreted considering detailed site observations. For example, where riparian buffers were wide and intact, but substrate quality was impaired by fine sediment, retrofits were considered likely to be especially beneficial for preserving existing high-value features by mitigating localized sediment inputs from stormwater runoff. However, these observations should be interpreted with caution, as fine sediment accumulation may not necessarily be a result of stormwater discharge and could instead reflect sediment transport from impacted areas upstream. Conversely, at sites where both scores and qualitative observations indicated limited habitat quality, retrofits alone were deemed unlikely to yield substantial improvements without concurrent stream habitat restoration. As such, these reaches may represent long-term stream restoration priorities but were considered lower priority for stormwater retrofits alone because they lacked many of the physical characteristics required to support sensitive fish species. These scores (and their subsequent interpretation), therefore, provide a decision-support framework for watershed managers to rank stormwater hotspots by their ecological risk and potential benefit, ensuring that limited resources are directed to locations where runoff mitigation is most likely to have the greatest benefit for protecting sensitive fish habitat.

This framework was evaluated as a screening-level decision-support tool rather than as a predictive hydrological or water quality model. Evaluation focused on whether the prioritization behaved as intended by identifying locations that combined two conditions: previously documented stormwater impairment and downstream physical habitat conditions suitable for sensitive fish species. Since all six candidate sites were pre-identified stormwater hotspots, the ranking step did not attempt to validate pollutant loading or re-rank outfalls by impairment severity. Instead, the framework assessed whether stormwater hotspot locations were associated with downstream reaches where habitat quality suggested higher ecological sensitivity and, therefore, greater potential benefit from runoff mitigation. This provides a practical internal consistency check for the framework, while recognizing that full validation would require post-implementation monitoring of water quality, discharge, pollutant loads, and fish habitat response after GSI retrofits are installed.

5.3 Results

5.3.1 Stormwater Hotspot Identification

A total of 91 stormwater outfalls were identified as draining to McVicar Creek. Initial USA and GIS screening identified eight problematic outfalls exhibiting visible signs of impairment and elevated stormwater impact potential. Water quality sampling at these locations identified elevated pollutant concentrations relative to guidelines for aquatic life at all eight outfalls. Since this assessment, one of the eight outfalls has been retrofitted with a bioretention system and is, therefore, excluded from the present study. Additionally, another hotspot is located on private land and was omitted due to access constraints. Accordingly, the remaining six outfall hotspots were retained for further habitat assessment and prioritization of potential GSI retrofits (Figure 5-1).

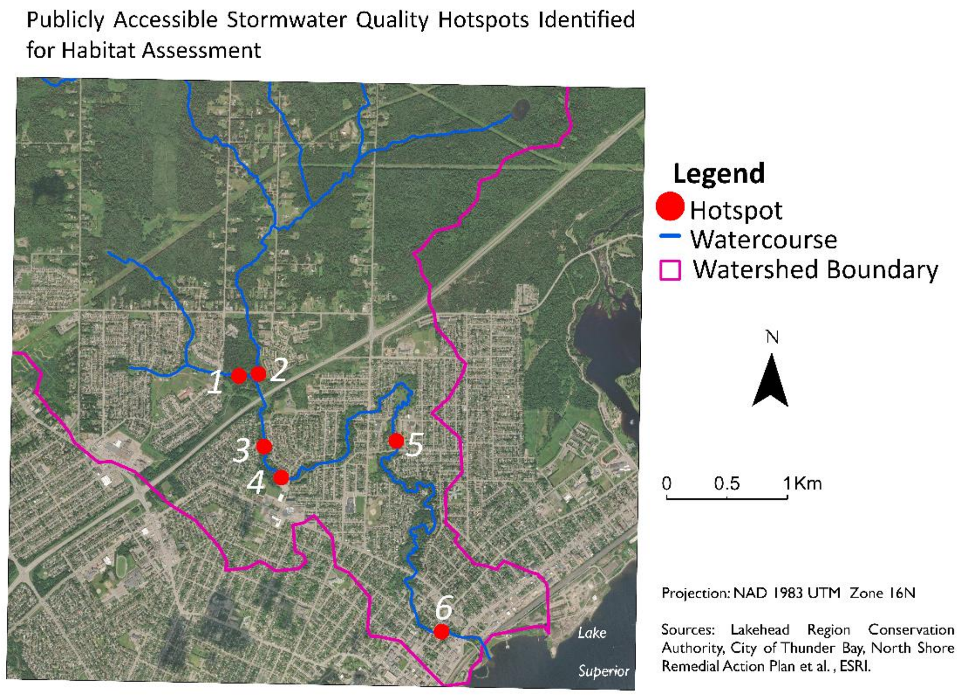


Figure 5-1 Six publicly accessible stormwater hotspots along McVicar Creek, Thunder Bay, Ontario identified for habitat assessment.

5.3.2 Fish Occupancy

Fish were captured at 13 sample locations, ranging from the mouth of Lake Superior to the south, and northward past all identified stormwater hotspots (Figure 5-2). Fish densities in these locations ranged from 0.02 to 0.16 individuals per metre of stream. Higher densities (≥0.10 individuals/m) were present at both less-urbanized headwaters and more urbanized reaches in the lower and mid-catchment. These results indicate that fish are distributed throughout much of the creek, including reaches downstream of stormwater outfalls, underscoring the need to consider habitat quality across the entire stream.

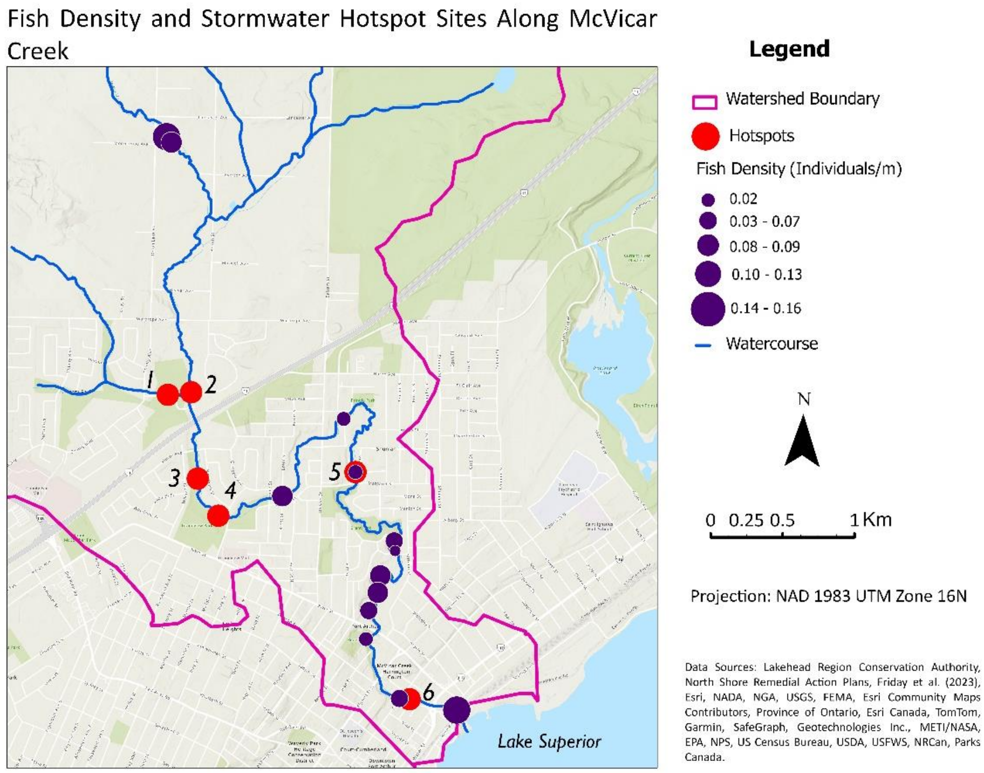


Figure 5-2 Fish density (individuals/m) and stormwater hotspot sites along McVicar Creek.

5.3.3 Physical Habitat Conditions

The stream at Site 1 was extremely narrow (<0.5 m) and shallow (<10 cm), surrounded by a wetland with tall grasses and cattails (Figure 5-3, left panel) that stabilized the streambanks and minimized erosion. Substrates consisted almost entirely of fine-grained sediments, with little gravel or cobble, and only a short, shallow riffle-pool sequence was present. Instream cover was sparse, limited to small branches and minor woody debris. Potential spawning habitat quality was poor due to the fine-dominated substrate. A stormwater outfall discharged directly into the reach, where sediment accumulation, debris, and garbage were evident around the outlet (Figure 5-3; right panel).

Figure 5-3 Stream reach and outfall at Site 1.

At Site 2, the channel widened to 2-3 metres with runs, riffles, and a small pool formed by a fallen tree (Figure 5-4). Substrates included cobbles and gravels and were moderately embedded (20-30%) by fine sediments near the outfall. Instream structure was modest, consisting of scattered boulders and woody debris. Riparian vegetation was dense, with a narrow buffer (3-5 metres) on the east bank and an extensive forest (>70 metres) on the west bank. Canopy cover ranged from sparse (<5%) to moderate (25-50%), and banks remained stable with no signs of erosion. The outflow had a constant flow of water into the stream, even under dry conditions (Figure 5-5; right panel). Overall, the site supported diverse flow conditions and had high riparian buffering and canopy shading suitable for coldwater fish habitat (Figure 5-4; left panel).

Figure 5-4 Stream reach and outfall at Site 2.

The stream reach at Site 3 was approximately 3 metres wide, with well-developed riffle-run sequences and diverse flow conditions (Figure 5-5). Substrates consisted of clean cobbles and gravels with minimal fine sediment, providing suitable spawning habitat. Instream features included boulders and woody debris that created hydraulic diversity, pools, and additional habitat structure. Riparian buffers were substantial: ~6 metres on the west bank and ~20 metres of mature forest on the east bank. Dense canopy shading moderated thermal conditions and maintained habitat quality. Banks were stable and well-vegetated, offering excellent conditions for spawning, rearing, and refuge.

Figure 5-5 Stream reach and outfall at Site 3.

At Site 4, the channel was moderately wide (3-4 m) and featured riffles, runs, and glides, creating varied hydraulic conditions (Figure 5-6). Substrates were gravel-dominated, with fine sediments accumulated near the stormwater outfall and partially embedded riffle gravels. Instream habitat features included boulders, branches, and aquatic vegetation, providing structural diversity. Riparian buffers were wide and continuous (~25 m on the south bank and ~20 m on the north), and canopy cover shaded much of the channel. Overall, the site offered a heterogeneous habitat suitable for multiple life stages, though localized sedimentation around the outfall may impair potential spawning substrates (Figure 5-6).

Figure 5-6 Stream reach and stormwater outfall at Site 4.

The stream at Site 5 was shallow (<10 cm), channelized and uniform, with limited riffle-run development (Figure 7-7; left panel). Substrates consisted mainly of gravels intermixed with fine sediments, and instream cover was scarce, with few boulders or large woody debris. Riparian vegetation was moderately wide (1-10 m) but patchy, resulting in uneven canopy shading. The east bank was stabilized with grasses, while the west bank was stabilized with riprap. Overall, habitat quality appeared constrained by simple morphology, limited canopy shading, and sparse instream structure. A large stormwater outfall discharged constant baseflow into the stream (Figure 5-7; right panel).

Figure 5-7 Stream reach and outfall at Site 5.

At Site 6, the channel widened to ~7 m and was dominated by bedrock substrate, with little gravel or movable cobble available for redd construction. The instream structure was minimal, consisting only of scattered rocks, and no large woody debris or macrophytes were observed (Figure 5-8). Riparian buffers were narrower than at other sites (5 m on the south bank and 8-35 m on the north), providing only partial shading. Canopy cover was patchy, ranging from open areas to 50% coverage. Despite the lack of movable substrate and cover, the bedrock channel supported good flow diversity with pools, runs, and riffles well represented. Overall, the site provided a variety of hydraulic conditions but limited potential spawning and refuge habitat.

Figure 5-8 Stream reach and stormwater outfall at Site 6.

5.3.4 Habitat Sensitivity Assessment

Scores across the ten habitat metrics (habitat complexity, substrate composition and embeddedness, instream habitat features, bank stability, riparian vegetation and buffer width, sediment deposition, large woody debris, habitat connectivity, canopy cover and shading, and channel sinuosity) varied among the six assessed stormwater hotspot reaches, reflecting differences in channel structure, riparian condition, and instream features. Total habitat scores ranged from 35 to 44 out of 50 (Figure 5-9), indicating that while all sites provide important habitat, some reaches provide more favourable physical habitat conditions for fish than others.

Habitat Complexity

Habitat complexity was consistently high across most sites (Figure 5-9). Sites 2, 3, 4, 5, and 6 all scored 5 (excellent), reflecting diverse riffle-run-pool sequences that created hydraulic heterogeneity. Site 1 scored slightly lower (4, good), as heterogeneity was reduced near the stormwater outfall but improved downstream. These results indicate that most of the assessed reaches of McVicar Creek maintained diverse flow and habitat features associated with supporting fish at multiple life stages.

Substrate Composition and Embeddedness

Substrate composition and the degree to which spawning gravels were embedded by fines varied considerably between sites, ranging from 1 (poor) to 5 (excellent) (Figure 5-9). Site 3 scored 5 (excellent), with clean gravels and cobbles providing optimal spawning habitat. Site 1 was most impaired (1, very poor), with substrates dominated by silt and sand, while Site 4 also scored low (2, poor) due to gravels mostly buried by fines. Moderate conditions were observed at Sites 2, 5, and 6 (all 3), where gravels were partially embedded but retained some habitat value.

Instream Habitat Features

The presence and abundance of instream habitat features varied considerably between reaches (Figure 5-9). Site 4 scored 5 (excellent), with a diverse mix of woody debris, boulders, and aquatic vegetation providing structural cover. Sites 1 and 3 each scored 4 (good), with moderate cover from natural obstructions such as logs and rocks. Site 6 also scored 4, reflecting some rocky cover in its bedrock channel but no large woody debris or aquatic vegetation. Site 2 scored 3 (moderate), with a single fallen tree providing habitat structure. Site 5 was the most limited, scoring 2 (poor) due to its uniform channel with little instream habitat features. These differences show that structural cover is unevenly distributed across sites, influencing habitat suitability.

Bank Stability

Bank stability was uniformly excellent across all reaches. Every site scored 5 (Figure 5-9), with dense riparian vegetation and root mats anchoring soils and no visible evidence of slumping or severe erosion. Even at Site 2, adjacent to recreational land use, vegetation maintained bank integrity. At Site 1, the marsh corridor provided strong natural stabilization. These consistent results highlight bank stability as a key strength throughout McVicar Creek.

Riparian Vegetation and Buffer Width

The riparian vegetation varied considerably in width and continuity between sites. Sites 1 and 4 scored 5 (excellent), with wide, continuous buffers exceeding 30 m. Sites 2 and 5 each scored 4, supported by moderately wide (15-30 m) but somewhat constrained or patchy buffers, while Site 3 scored 3 (moderate) due to a narrow (< 5 m) buffer along one bank. Site 6 also scored 3, with patchy cover and reduced continuity (Figure 5-9). Overall, riparian buffers along the reaches remained strong, with narrower or fragmented buffers at some sites due to adjacent urban land uses.

Sediment Deposition

Sediment deposition varied across reaches. Site 1 scored 2 (poor), with widespread fine sediment smothering gravels near the outfall (Figure 5-9). Sites 4 and 5 each scored 3 (moderate), with fines accumulating in pools and glides but leaving riffles relatively free of sediment accumulation. Sites 2 and 3 scored 4 (good), where only localized sediment was observed, and cobbles remained exposed. Site 6 also scored 3, reflecting moderate deposition between bedrock features, while most of the stream channel was sediment-free bedrock.

Large Woody Debris (LWD)

Large woody debris availability showed the widest variation among parameters (Figure 5-9). Site 1 scored 5 (excellent), with abundant logs and branches creating pools, hydraulic variability, and refuge areas. Sites 2, 3, and 4 each scored 3 (moderate), offering some cover but limited to only small areas of the stream channel. Sites 5 and 6 scored the lowest, each scoring 1 (very poor), with no logs or branches present in the reach. These differences indicate that structural cover is a limiting factor in several reaches, particularly where channels are simplified or dominated by bedrock.

Habitat Connectivity

Habitat connectivity was generally high among sites (Figure 5-9). Sites 3, 4, and 5 each scored 5 (excellent), with no barriers to fish passage. Sites 1 and 2 scored 4 (good), reflecting minor seasonal obstructions such as fallen trees, which may impede movement at low flows but did not constitute major barriers. Site 6 also scored 4, with a largely open channel but some reduced connectivity due to its bedrock-dominated form, creating shallow areas that may pose minor seasonal barriers to fish passage. Overall, habitat connectivity remained strong across the study reaches, supporting fish movement within the reach.

Canopy Cover and Shading

Canopy cover and shading varied from moderate (25-50%) to excellent (75-100%) among reaches (Figure 5-9). Sites 1, 3, and 4 each scored *5* (excellent)*,* with dense tree canopies providing 75-100% cover. Sites 2 and 5 scored *3* (moderate)*,* where canopy cover was more limited (25-50%) due to recreation trails, manicured lawns, or patchy vegetation. Site 6 also scored *3*, with variable shading depending on varying canopy cover along the reach. Overall, while canopy cover and shading were high along McVicar Creek, the variability in canopy cover highlights the importance of maintaining tree cover in urban areas to support coldwater habitat for coldwater fish species.

Channel Sinuosity

Channel sinuosity was relatively low at the study sites (Figure 5-9). Site 5 scored 3 (moderate) and had a channel length-to-valley ratio of 1.24, indicating some meander development. Sites 2 and 4 scored 3 (moderate) as well, with ratios around 1.11-1.12. Sites 1 and 3 scored 2 (poor), with only slightly sinuous channels (ratios 1.06-1.07). Site 6 was nearly straight (sinuosity = 1.03), scoring 1 (very poor). Calculations for channel sinuosity are provided in the Supplementary Materials.

Overall Habitat Scores

Mean habitat scores integrated across metrics highlighted differences in fish habitat quality between sites (Figure 5-9). Sites 3 and 4 achieved the highest overall scores (4.1 each), reflecting excellent fish habitat conditions, with strong habitat complexity, high canopy cover, and wide, continuous riparian buffers. Sites 1 and 2 both averaged 3.7, indicating generally good habitat with some localized impairments, particularly from the accumulation of fines on the streambed and limited instream habitat structure for fish shelter. Sites 5 and 6 scored lowest (both 3.4), constrained by poor substrate quality, sparse large woody debris, and reduced canopy cover. Despite these differences, all reaches provided valuable habitat, with Sites 3 and 4 representing the highest quality physical fish habitat where stormwater retrofit considerations may provide the greatest benefit at protecting sensitive fish species.

Habitat Quality Score (1-5)	1	2	3	4	5

	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Habitat Complexity	4	5	5	5	5	5
Substrate Composition & Embeddedness	1	3	5	2	2	3
Instream Habitat Features	4	3	4	5	2	4
Bank Stability	5	5	5	5	5	5
Riparian Buffer	5	4	3	5	4	3
Sediment Deposition	2	4	4	3	3	3
Large Woody Debris	5	3	3	3	1	1
Habitat Connectivity	4	4	5	5	5	4
Canopy Cover	5	3	5	5	3	3
Channel Sinuosity	2	3	2	3	4	1
Mean Score	**3.7**	**3.7**	**4.1**	**4.1**	**3.4**	**3.4**

Figure 5-9 Heat map of habitat scores (1 = very poor, 5 = excellent) across ten metrics at six stormwater hotspots along McVicar Creek.

5.3.5 Discussion

While green stormwater infrastructure (GSI) practices are gaining popularity to reduce adverse impacts on urban streams, prioritizing specific locations for GSI implementation remains a challenge (Garbanzos et al., 2024), and identifying GSI locations during urban planning is important for the success of the project (Gulshad et al., 2024). This study aimed to develop a rapid framework to prioritize GSI interventions to protect sensitive urban fish habitats. I combined three complementary datasets, including unpublished stormwater quality data, fish population data from the Ontario MNR (Friday et al., 2023), and a 2025 field-based fish habitat assessment using repeatable methods to (1) identify stormwater outfalls along a trout-sensitive urban tributary that represent the greatest pollution hotspots and (2) evaluate the ecological sensitivity of downstream reaches with respect to fish habitat conditions. I then (3) ranked outfalls by their ecological risk and potential benefit for GSI retrofits and used these rankings to (4) provide planning guidance for targeted stormwater interventions that maximize fish habitat protection in urban streams. This framework is transferable to other urban streams and was designed to be adopted by municipalities to prioritize limited resources in ways that provide the greatest benefit to fish habitat.

Our first objective was to identify stormwater outfalls along a trout-sensitive urban tributary with elevated potential to contribute pollutants to the stream. The stormwater impairment identification process involved mapping the municipal storm-sewer network, conducting a Unified Stream Assessment (USA), and targeted wet- and dry-weather water quality sampling. Together, the GIS analysis and USA were used to identify problematic stormwater outfalls exhibiting visible signs of impairment and elevated stormwater impact potential, narrowing the large number of stormwater outfalls within the watershed to a smaller subset of candidate sites, including eight stormwater outfalls selected for targeted water quality assessment. Water quality sampling at these locations identified elevated pollutant concentrations relative to aquatic life guidelines, supporting their selection for further habitat assessment and consideration within the present green stormwater infrastructure (GSI) prioritization framework. Using storm-sewer connectivity and drainage characteristics in this way is supported by recent work showing that effective/connected imperviousness and drainage configuration are primary predictors of urban runoff and pollutant export (Walsh et al., 2022b). Given that exposure to roadway runoff can be acutely lethal to juvenile salmonids (French et al., 2022), prioritizing interventions at problematic stormwater outfalls such as these is warranted. Prioritizing GSI at pollutant hotspot locations has been shown to yield greater watershed-scale reductions in runoff and pollutant loads than uniform deployment (Piscopo et al., 2021; Chen et al., 2024).

Our second objective was to assess the ecological sensitivity downstream of stormwater hotspots with respect to fish habitat quality. Sites 3 and 4 had the highest-quality fish habitat, with gravel-cobble substrates and little accumulation of fines, suitable for redd construction (White et al., 2025). Site 2 also had diverse runs and riffles, dense riparian cover, and a pool formed by a fallen log. Large woody debris in this reach increases pool depth and frequency and adds cover and habitat complexity that benefit salmonids (Pess et al., 2022; Dolloff & Warren, 2003). Conversely, Sites 1 and 5 were dominated by fines, were more channelized, had lower canopy cover and riparian vegetation, and had less instream habitat structure. Since salmonid abundance tends to increase where greater instream cover and structural complexity are greater (Foote et al., 2020), Sites 1 and 5 were less suitable for salmonid habitat. The bedrock substrate at Site 6 limits spawning habitat availability since redd construction typically occurs in gravel and cobble substrates (White et al., 2025). Overall, the spawning gravels, instream habitat structure, and canopy cover and shading marked high-quality habitats where stormwater inputs may have the most direct impact on fish populations.

Our third objective was to rank outfalls by their ecological risk and potential benefit from GSI retrofits. Aligned with multi-metric habitat assessment frameworks (Beechie et al., 2023), ecological risk was quantified based on ten habitat parameters. Sites 3 and 4 scored highest due to mobile gravel and cobble for redd construction (White et al., 2025), instream cover and structural complexity linked to higher salmonid abundance (Foote et al., 2020), and riparian shade moderating stream temperature (Fuller et al., 2022) and were thus identified as the highest priority for stormwater retrofits. Sites 1 and 2 had moderate scores and included instream structures associated with higher salmonid abundance (Foote et al., 2020). However, embedded gravel constrains spawning as redds occur in gravel-cobble substrates (White, 2025), and excessive fines reduce the success of incubation fry emergence (Kondolf, 2000). Sites 5 and 6 ranked lower because limited large woody debris and instream features reduce habitat complexity and pool frequency (Pess et al., 2022), and bedrock-dominated substrate offered little gravel and cobbles suitable for redd construction (White et al., 2025). Collectively, these scores convert field observations into a quantitative ranking, indicating where stormwater retrofits may support the protection of sensitive fish habitats.

Our last objective was to provide planning guidance for targeted stormwater interventions in locations where benefits to fish habitat protection are expected to be greatest. To do so, I recommend the following triage. First, prioritize dispersed stormwater controls such as bioretention systems to improve stormwater quality in the most sensitive stream habitats (Sites 3 and 4). Since dispersed stormwater controls can improve downstream water quality parameters, including reducing nutrient concentrations and stream temperatures (Walsh et al., 2022a), these retrofits should be prioritized in areas where stormwater inputs pose the greatest risk to reaches providing high-quality physical fish habitat conditions. This prioritization does not imply that degraded reaches are unimportant but rather reflects the principle that protecting existing high-quality habitats from further degradation may provide greater long-term ecological benefits than attempting to fully restore severely impaired reaches after habitat loss has occurred.

While those stormwater retrofits are implemented, I recommend improving instream habitat conditions to complement stormwater controls at the remaining hotspot sites.

This includes clearing fine sediment in areas with embedded spawning gravels (Wild et al., 2024), restoring riparian canopy cover and shading to moderate summer stream temperatures (Fuller et al., 2022), adding large woody debris to increase habitat cover and pool frequency (Pess et al., 2023), and using boulder placements to increase hydraulic complexity (Golpira et al., 2022). At these more physically degraded reaches, stream rehabilitation may be necessary before stormwater retrofits alone can provide substantial ecological benefits. As the physical habitat quality of these reaches is improved, I recommend integrating stormwater interventions to protect the habitat at the remaining stormwater hotspots, starting with Sites 1 and 2, followed by Sites 5 and 6.

This framework is transferable to other urban watersheds, where hotspots can be screened using USA severity scores, targeted water quality sampling, and GIS-based assessments of storm sewer connectivity and drainage characteristics. When stormwater quality data is unavailable, targeted wet and dry water quality samples can be collected from stormwater outfalls and evaluated against USA severity scores and water quality guidelines to identify stormwater hotspots. Fish occupancy can be confirmed using recent electrofishing survey data (e.g., Friday et al., 2023) or by conducting new surveys to verify the presence of fish in the stream undergoing restoration. In this framework, fish occupancy and density data were used to confirm the presence of fish within stormwater-influenced reaches and guide where downstream habitat assessments were conducted, rather than being directly incorporated into the habitat scoring matrix. Consistent with salmonid habitat-based planning frameworks (Beechie et al., 2023), I scored reach-scale habitat sensitivity using ten equal-weight metrics to support habitat-based stormwater planning. Given evidence that dispersed stormwater controls can improve key water quality parameters in receiving streams (Walsh et al., 2022a), our framework prioritizes these controls, such as bioretention systems, at sites where they are most likely to provide the greatest benefit to urban fish habitats.

This prioritization framework is not without limitations. Firstly, I identified hotspots using GIS connectivity and effective imperviousness as a proxy for stormwater discharge volumes. While effective imperviousness predicts stream response, it does not directly measure outfall discharge volumes (Walsh et al., 2022b). As event dynamics and interannual variability can shift flow, updated monitoring with end-of-pipe discharge measurements and high-frequency water-quality sampling would verify volumes and capture contaminant pulses (Rozemeijer et al., 2025; Bieroza et al., 2023). Secondly, I identified hotspots by guideline exceedances and did not consider total pollutant loads; verification should estimate pollutant loads using paired flow and concentration data (Al Masum et al., 2022; Tiernan et al., 2024).

Although the framework was evaluated using an internal consistency check, it was not independently validated against measured post-implementation ecological outcomes. The framework identifies where GSI retrofits are expected to provide the greatest ecological benefit based on documented stormwater impairment and downstream habitat sensitivity, but it does not demonstrate that retrofits at these locations will reduce pollutant loads, improve water quality, or increase fish use. Full validation would require post-implementation monitoring of end-of-pipe discharge, pollutant loads, downstream water quality, habitat condition, and fish community response.

I also applied equal weights to the ten habitat metrics to maintain a simple, transparent, and transferable scoring framework. Alternative weighting scenarios were not tested in this study, which limits the ability to determine how sensitive site rankings are to different assumptions about habitat importance. For example, a framework designed specifically for salmonid spawning protection might place greater emphasis on substrate composition, sediment deposition, canopy cover, and connectivity than on metrics such as channel sinuosity or bank stability. Future work should evaluate alternative weight sets and report any changes in site rankings, as recommended in multi-criteria decision-making practice (Więckowski & Sałabun, 2023).

I also did not assess practicality or equity (access, maintenance responsibilities, distributional impacts) associated with retrofit placement in these locations, which ultimately needs to be integrated alongside ecological criteria by city planners (LeFevre et al., 2023). While one hotspot on private land was excluded from the framework due to access constraints, broader retrofit feasibility, including underground utilities, storm sewer configurations, land availability, and construction costs, was not formally evaluated. Since these factors may constrain retrofit implementation in some sub-catchments, future applications of this framework should integrate feasibility assessments alongside ecological prioritization to evaluate the practicality of implementing stormwater retrofits in these locations.

Fish occurrence was also inferred from single-season, single-pass electrofishing (Friday et al., 2023). Multi-pass or multi-season data would better target retrofits (Lu et al., 2024) in areas occupied by fish. The framework also treated all fish species equally and did not account for differences in species sensitivity to stormwater runoff. Although fish density data were available downstream of the hotspot locations, these data were not incorporated directly into the prioritization framework. In this framework, fish occupancy and density data were used to confirm that fish were present within stormwater-impacted reaches and guide where downstream habitat assessments were conducted, rather than directly ranking sites based on current fish abundance. Future applications could incorporate the density of sensitive fish species, such as brook trout (French et al., 2022; TRCA, 2009, p. 20), as a biological metric to improve prioritization for sensitive salmonid species.

While more tolerant species, such as creek chub (*Semotilus atromaculatus*) and blacknose dace (*Rhinichthys atratulus*; Friday et al., 2023), were included in this assessment, brook trout (*Salvelinus fontinalis*) are generally considered particularly sensitive to urban stream degradation. For example, in the Don River Watershed, creek chub, blacknose dace, longnose dace, and black sucker were found to dominate the stream community, while brook trout, which are intolerant to heavy siltation, were determined to likely be extirpated from the river system (TRCA, 2009; p. 20), suggesting that brook trout require particular attention in watersheds affected by sediment inputs. In addition to sensitivity to sediment, recent studies suggest that brook trout are highly sensitive to stormwater contaminants, such as 6PPD-quinone, whereas rainbow trout exhibit more intermediate sensitivity (French et al., 2022). Therefore, rather than weighting all fish species equally, further applications of this framework could place greater emphasis on brook trout habitat when prioritizing stormwater interventions.

Lastly, this study was completed by a single researcher, introducing bias in qualitative habitat scoring. Future work should evaluate variability between observers by having multiple trained observers independently apply the same habitat assessment framework at the same sites to quantify scoring consistency and reduce potential observer bias.

5.3.6 Conclusion

I developed and evaluated a rapid prioritization framework that links stormwater hotspots to sensitive fish habitats by combining water quality data, fish-occupancy records, and a 10-metric field habitat assessment. Applied to McVicar Creek, the framework narrowed 91 outfalls to six priority hotspots and identified Sites 3 and 4 as the highest priority locations for GSI. Immediate actions should focus on protecting the most sensitive habitats through dispersed stormwater controls at these sites, while complementary stream rehabilitation measures, such as removing fines from embedded gravels, restoring riparian shade, and adding instream habitat structures, could improve habitat conditions in more degraded reaches. The framework was evaluated as a screening-level decision-support tool by confirming that priority rankings reflected the intended combination of documented stormwater impairment and downstream habitat sensitivity. However, further validation should include end-of-pipe discharge measurements, pollutant load estimates, alternative weighting scenarios, post-implementation monitoring, and periodic re-scoring as new data accumulate. The approach is simple, transferable, and provides managers with a low-cost decision tool to triage limited resources toward locations where GSI retrofits are most likely to reduce runoff-related stressors to high-quality fish habitats in urban streams.

6 Conclusion and Summary of the Chapters

6.1 Overview of Dissertation

In an urban environment, impervious surfaces such as roads, rooftops and parking lots reduce natural infiltration and increase runoff volumes, while the application of pesticides and fertilizers, vehicle corrosion, and winter sanding and salting degrade the water quality of stormwater and meltwater entering urban streams. Changes in stream hydrology and water quality from stormwater runoff have therefore reduced the population sizes of sensitive fish species, leading to a shift toward more tolerant species in urban streams. As a result, green stormwater infrastructure practices, including bioretention systems, have been increasingly implemented to reduce stormwater volumes and improve stormwater quality. However, it is unclear whether bioretention systems provide sufficient treatment in terms of quality and quantity to protect fish habitats, particularly in cold-climate environments with long winters, substantial snowfall, and large runoff volumes during the spring freshet. Since stormwater management interventions, such as bioretention systems, are often implemented independently of stream restoration initiatives and land-use planning decisions, they typically do not include habitat considerations in their planning and implementation. This dissertation evaluated the performance of bioretention systems at managing stormwater and meltwater runoff quality and quantity entering two northern urban trout-sensitive tributaries and developed a framework to prioritize stormwater management interventions that support the protection of sensitive fish habitats.

I organized this thesis into four individual manuscripts, each with the common goal of managing stormwater runoff to reduce stressors to urban fish habitats. The first manuscript examined how municipal planners can integrate stream habitat restoration, green stormwater infrastructure, and land use planning to support the protection of urban streams and fish habitats. The second manuscript evaluated the performance of bioretention systems at reducing the quantity and improving the quality of stormwater before discharging into trout-bearing streams during rainfall events. It showed that bioretention systems reduce peak and total discharge while removing particulate contaminants such as suspended solids and reducing turbidity. The third manuscript examined bioretention system performance during spring snowmelt. It showed that although frozen soil initially impedes infiltration during the early melt, bioretention systems can reduce meltwater volumes and particulate pollutant loads once thawing occurs. The fourth manuscript developed a framework to assess stormwater pollution hotspots and fish habitat conditions and identify priority locations where stormwater management interventions can yield the greatest improvements to fish habitats. Together, these manuscripts demonstrate that bioretention systems are effective at reducing stormwater and meltwater discharge and particulate contaminant concentrations while highlighting the importance of integrating stormwater infrastructure with watershed-scale land management practices to reduce dissolved pollutant inputs and protect sensitive fish habitats.

6.2 Novel Contributions of this Dissertation

This dissertation contributes to the understanding of the use of bioretention systems for improving water quality entering fish habitats in Northwestern Ontario and other cold-climate urban watersheds in five main ways. First, it provides a critical perspectives-based synthesis that reframes urban stream revitalization as an integrated challenge requiring coordination among municipal engineers, ecologists and urban planners, to adequately protect urban coldwater streams. Specifically, Chapter 2 shows that physical habitat restoration may be limited if untreated stormwater runoff continues to degrade receiving waters, that GSI evaluations over-emphasize infrastructure-scale performance without linking outcomes to receiving stream conditions or water quality requirements for aquatic life, and that land use planning can either enable or constrain stormwater management retrofits, implementation of stormwater controls on private land, riparian protection and long-term monitoring.

Second, this dissertation provides field-based evidence that bioretention systems can reduce runoff volumes and particulate-associated pollutants before discharging into fish-bearing waters during warm-weather rainfall events and the spring freshet.

Third, it demonstrates that bioretention performance varies between pollutants, with stronger reductions in particulate-associated pollutants than dissolved pollutants such as chloride and nutrients.

Fourth, it identifies the roadside snowpack as an important seasonal reservoir of sediment, chloride, and dissolved organic carbon, highlighting the importance of winter maintenance and managing spring fresh runoff as part of cold-climate stormwater management.

Fifth, it develops and applies a spatial prioritization framework that links stormwater impairment identification with downstream fish habitat sensitivity, allowing targeted green stormwater infrastructure where the ecological benefit to fish habitats is most likely. Together these contributions address a regional research gap by showing how stormwater treatment, fish habitat protection, and land-use planning can be integrated in a cold-climate urban watershed.

6.3 Integrated Urban Stream Management

Chapter 2 examined stream habitat restoration approaches, green stormwater infrastructure performance, and considerations for urban land use planning to restore degraded urban streams. The effective restoration of urban streams requires an integrated approach that combines physical habitat restoration with watershed-scale stormwater management and land use planning to improve both physical fish habitat and water quality. Restoration projects often aim to improve physical habitat conditions by stabilizing streambanks, incorporating habitat features, and planting riparian vegetation. However, I found that while restoration projects often improve physical habitat conditions, they do not always improve water quality, underscoring the need to address underlying mechanisms of water quality degradation, such as urban runoff. I also found that, due to budget constraints, restoration projects are often implemented at relatively small spatial scales and do not adequately offset the hydrologic and water quality impacts associated with urban development.

This chapter also highlighted several limitations in current stormwater management practices. I found that evaluations of green stormwater infrastructure performance often report percent-reduction metrics, without reporting raw concentrations or comparing effluent concentrations to water quality guidelines relevant to aquatic life. Without comparing effluent quality to water quality guidelines for protecting aquatic life, researchers and watershed managers cannot determine whether green stormwater infrastructure practices reduce pollutants to concentrations safe for aquatic life. Lastly, stream restoration and green stormwater infrastructure both aim to improve urban stream conditions. However, municipalities often plan and implement these separately, limiting opportunities to coordinate management actions and achieve greater ecological benefits.

To overcome these constraints, I recommend establishing long-term monitoring after restoration projects to ensure they achieve the desired outcomes; matching the scale of restoration projects to the scale of stream degradation; integrating fluvial geomorphology into restoration design; and enhancing stormwater performance evaluation by comparing influent and effluent concentrations against ecological thresholds. I also proposed incentives and educational initiatives to encourage stormwater management on private property. I also stressed the importance of winter snow management to reduce pollutants from winter road maintenance, prioritizing the placement of restoration and stormwater interventions in ecologically sensitive areas, and creating municipal planning regulations that safeguard sensitive riparian areas and coldwater ecosystems. Overall, these suggestions offer a framework for integrating land use planning, stormwater management, and stream restoration to enhance the protection of urban fish habitats.

6.4 Rainfall Performance of Bioretention Systems for Protecting Fish Habitats

Chapter 3 evaluated the hydrologic and water quality performance of bioretention systems during rainfall events in a cold climate urban watershed to determine whether these systems improve water quality parameters relevant to fish habitat protection. In urban watersheds, stormwater runoff can rapidly increase stream discharge and transport pollutants into receiving streams. Increases in stream discharges can also erode streambanks, transporting fine-grained sediments downstream where they degrade fish spawning beds in urban streams. To address these problems, municipalities are increasingly using bioretention cells to reduce stormwater quantity and improve stormwater quality before it enters urban streams. Despite their growing use, relatively few studies have examined whether bioretention systems measurably improve runoff conditions in ways that benefit fish habitats in northern urban environments.

Our results show that the bioretention systems were effective in reducing peak discharge and total volume of runoff compared to the influent. These reductions in stormwater volume are particularly important for fish habitat protection because reduced stormwater discharge will help minimize streambank erosion and subsequent sediment mobilization that can interfere with fish spawning habitats. The bioretention systems were also effective at reducing turbidity and suspended solids concentrations. Reductions in turbidity and suspended solids are important for protecting fish habitat because lower suspended concentrations help maintain clean spawning gravels. Suspended sediments can interfere with fish spawning beds by smothering fish redds, reducing egg oxygenation. However, despite effective reductions in suspended contaminants, reductions in dissolved contaminants were less consistent, as bioretention systems were ineffective at reducing nutrients and dissolved metals, while conductivity and chloride concentrations increased following bioretention treatment.

These findings demonstrate that bioretention systems can play an important role in mitigating runoff-related stressors to fish habitat by reducing stormwater discharge and limiting the transport of particulate contaminants that degrade stream substrates and aquatic habitat quality. However, the bioretention systems did not effectively remove dissolved contaminants such as chloride and nutrients. As such, the effective protection of fish habitats requires stormwater controls, such as bioretention systems, to reduce stormwater quantity and particulate contaminants, along with improved land use practices, including reducing road salt and fertilizer application throughout the watershed to reduce dissolved pollutant loads, such as chloride and nutrients, entering fish habitats.

The findings from the rainfall chapter should be interpreted as concentration-based treatment performance and volume reductions rather than pollutant load reductions. Estimates of pollutant loads were not calculated because water quality samples represented first-flush concentrations rather than event mean or flow-weighted composite concentrations. As event mean or flow-weighted composite concentrations were not collected, it was not safe to assume that first-flush samples were representative of the entire runoff event, and therefore, it would be misleading to calculate event-based pollutant loads using only first-flush samples.

6.5 Spring Melt Performance of Bioretention Systems for Protecting Fish Habitats

Chapter 4 examined the accumulation of pollutants in the winter snowpack and the performance of bioretention systems at reducing discharge and removing these pollutants during the spring melt. In northern cities, municipalities regularly apply sand and salt to roadways for winter road maintenance. The application of salt increases vehicle corrosion and contributes to the release of heavy metals. In spring, these contaminants can degrade water quality and harm aquatic ecosystems, including fish spawning habitats. Therefore, this manuscript examined pollutant accumulation in winter snowpack and evaluated whether bioretention systems reduce meltwater volumes and improve meltwater quality before it enters receiving urban streams during sensitive spring spawning periods.

I found that winter sand and salt application and vehicle exhaust increase concentrations of suspended solids, chloride, and dissolved organic carbon along roadside snowbanks, and that spring meltwater transports these pollutants into urban streams and bioretention systems during the spring melt. Our results showed that frozen conditions during the early stages of the spring melt prevented infiltration of meltwater and that meltwater was released through the overflow, bypassing most treatment. Subsequently, as soil temperatures rose, meltwater began infiltrating the bioretention systems, reducing runoff volumes compared to the untreated inflow. The treatment results showed a reduction in turbidity, suspended solids, and dissolved organic carbon concentrations in the effluent compared to the influent of the bioretention systems during the spring melt.

These spring melt findings should be interpreted as concentration-based treatment responses and meltwater volume reductions rather than pollutant load reductions. Load-based estimates were not calculated because meltwater quality samples were collected as single daily grab samples during the peak afternoon melt, rather than daily mean, event mean, or flow-weighted composite samples, and it was not safe to assume that the single sample represented the mean concentration over the full daily melt period.

These findings demonstrate that bioretention systems can reduce particulate pollutant concentrations and organic carbon concentrations during the spring melt, thereby reducing runoff-related stressors to fish habitat. Reductions in turbidity and suspended solids concentrations may improve water clarity, reduce gill irritation, and limit fine sediment inputs that can smother spawning substrates used by salmonids. However, the bioretention systems did not consistently reduce dissolved contaminants such as chloride concentrations. Chloride concentrations are of particular concern in cold water ecosystems because road salt runoff can alter stream chemistry and affect aquatic organisms. The results, therefore, showed that bioretention systems reduce runoff volumes and particulate contaminant transport during spring melt. However, limiting dissolved contaminant inputs into sensitive fish habitats will require reductions in road salt and fall fertilizer application in northern urban watersheds.

6.6 Prioritizing Stormwater Management for Fish Habitat Protection

Chapter 5 developed a framework to prioritize stormwater interventions in locations where benefits to fish habitats are expected to be greatest. Municipalities often have limited financial and logistical resources for implementing green stormwater infrastructure and, therefore, must determine where stormwater controls will be most effective at protecting aquatic ecosystems. Chapter 5, therefore, developed a framework to identify stormwater pollution hotspots and prioritize management interventions in areas where stormwater runoff poses the greatest risk to sensitive fish habitats.

Our prioritization framework used spatial analysis, followed by targeted stormwater sampling and physical fish habitat assessments, to identify stream reaches where stormwater runoff may have the greatest impact on downstream fish habitats. By linking watershed pollutant sources to habitat sensitivity, this framework enables urban planners and watershed managers to identify locations where stormwater controls, such as bioretention systems, can provide the greatest protection for aquatic ecosystems. Using this prioritization framework, I found that several reaches of the stream that were most impacted by stormwater runoff still provide high-quality fish habitat. Prioritizing stormwater interventions at these locations can, thereby, reduce particulate pollutant loading entering these sensitive fish habitats.

Implementing stormwater management practices upstream of fish habitats can be an effective way to manage stormwater discharge, sediment transport, and water quality, thereby supporting fish survival. The framework proposed in the study can serve as a decision-support tool for municipalities to prioritize the implementation of stormwater management practices in areas that are expected to have the greatest benefit to fish habitats.

6.7 Implications for Stormwater Management in Sensitive Fish Habitats

The results of this dissertation show that reducing runoff-related stressors to sensitive fish habitats requires both green stormwater infrastructure and improved land use practices, such as reductions in fertilizer and road salt applications, avoiding application during wet periods, and picking up pet waste. However, the short-term downstream water quality monitoring conducted in this dissertation was not intended to fully quantify changes in overall fish habitat condition or isolate bioretention effects from other watershed-scale stressors. Findings from the rainfall monitoring chapter show that bioretention systems can reduce peak discharge and total runoff volumes during storm events and remove particulate contaminants, such as suspended solids, thereby improving turbidity. Reductions in particulate contaminant concentrations can support fish habitat protection by reducing sediment loads that degrade spawning substrates and by reducing the release of particulate pollutants that can be toxic to fish.

The snowmelt results demonstrate the importance of managing pollutants from winter snowpack in cold climate regions using both bioretention systems to reduce meltwater discharge and particulate contaminants, and reductions in road salt applications and fall fertilizers to reduce dissolved contaminants such as chloride, nutrients and dissolved metals. Pollutants from vehicular activity and winter road maintenance accumulate in roadside snowbanks, resulting in elevated concentrations of chloride, suspended solids, and dissolved organic carbon that enter urban fish habitats during spring melt events.

While the bioretention systems in this study reduced meltwater volumes, turbidity, dissolved organic carbon, and suspended solids concentrations, they did not consistently reduce many dissolved contaminants, including chloride, metals, and dissolved nutrients. The inconsistent treatment of metals, dissolved nutrients, and chloride shows that bioretention systems alone are insufficient to reduce dissolved contaminants and must, therefore, be complemented by reductions in road salt and fall fertilizer applications to limit dissolved pollutant inputs before they enter bioretention systems.

Across the rainfall and spring melt chapters, treatment performance was evaluated using concentration-based water quality comparisons and separate runoff volume analyses rather than pollutant load estimates. Load-based estimates were not calculated because the rainfall water quality samples represented first-flush concentrations, while spring meltwater samples were collected as single daily grab samples rather than daily mean, event mean, or flow-weighted composite samples. Future monitoring should pair continuous discharge measurements with flow-weighted composite sampling or repeated within-event and within-day sampling to quantify reductions in pollutant loads delivered to receiving urban streams.

The prioritization framework identified stormwater hotspots and evaluated the sensitivity of downstream physical fish habitat conditions to determine where green stormwater infrastructure practices, such as bioretention systems, would be expected to reduce stormwater-related stressors on fish habitats. In this framework, fish occupancy data were used to confirm that fish occupy stormwater-impacted reaches and are not confined to headwater reaches, while habitat sensitivity was assessed using a ten-metric field-based habitat assessment. Prioritization, therefore, focused on implementing green stormwater retrofits at hotspot outfalls discharging into reaches with high-quality physical habitat conditions for fish, thereby reducing particulate pollutant loading into sensitive areas. Such prioritization is particularly important in urban watersheds supporting sensitive fish species, such as salmonids, where limited financial and administrative capacity for stormwater retrofits is available, and urban planners must allocate funds to areas that will provide the greatest ecological benefits.

6.8 Future Directions

While this dissertation provides new insights into the performance of bioretention systems in Northwestern Ontario and the prioritization of stormwater management interventions to protect fish habitats, several areas of future research remain. First, long-term monitoring studies are needed to determine whether improvements in stormwater hydrology and water quality translate into measurable ecological responses within receiving streams. Future research should include collecting benthic invertebrate samples and conducting fish surveys to better understand how reductions in stormwater discharge and particulate pollutant concentrations influence habitat quality, species diversity, and abundance. Additional work is needed to improve the treatment of dissolved contaminants, such as chloride and nutrients, as the bioretention systems did not reduce these parameters, which continue to pose a threat to fish health. The application of these frameworks to other watersheds would also aid their generalization and enable municipalities to identify areas where stormwater management is likely to yield the greatest benefits for fish habitat.

6.9 Final Considerations

In urban watersheds, particularly in cold climates, where snowpack accumulation and subsequent snowmelt influence the mobilization and transport of pollutants into aquatic habitats, urban stormwater runoff and spring snowmelt pose a substantial threat to fish habitats. The results of this dissertation demonstrate that bioretention systems can play an important role in protecting fish habitats by reducing stormwater discharge and removing particulate contaminants that degrade aquatic environments. However, bioretention systems alone cannot adequately address dissolved pollutants such as chloride and nutrients. Effective management of urban fish habitats, therefore, requires applying watershed management practices that incorporate green stormwater infrastructure, land management practices, and targeted prioritization of stormwater management practices. Implementing these approaches can help municipalities better manage urban runoff and improve the long-term protection of sensitive fish habitats in urban watersheds.

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Supplementary Materials

Supplementary Materials for Chapter 3

Table S1 Antecedent dry period (days), total rainfall depth (mm), maximum rainfall intensity (mm/hr) and average rainfall intensity of events during the sampling period.

Event Date	Antecedent Dry Period (ADP) (Days)	Total Rainfall Depth (mm)	Maximum Rainfall Intensity (mm/hr)	Average Rainfall Intensity (mm/hr)
2022-08-05 to 2022-08-06	4.6	9.8	11.2	1.9
2022-09-09	6.7	9.4	4.8	1.8
2022-10-12 to 2022-10-13	6.2	8.6	6.4	2.2
2022-11-05 to 2022-11-07	11.3	7.4	2.4	1.1
2023-04-27 to 2023-04-30	2.9	32.6	6.4	1.4
2023-05-19	7.8	16	6.4	2.6
2023-06-24 to 2023-06-25	13.7	18.2	8	1.7
2023-07-09 to 2023-07-10	1.6	3.6	5.6	1.8
2023-07-16 to 2023-07-17	6.3	3.2	3.2	1.2
2023-07-26 to 2023-07-27	3.5	16	21.6	3.1
2023-08-08 to 2023-08-09	1.1	3.6	6.4	1.6
2023-08-16	4.6	3.0	3.2	2.0
2023-10-05 to 2023-10-06	2.3	4.2	6.4	2.1
2023-11-05 to 2023-11-09	9.2	26.6	2.4	1.0
2024-04-08 to 2024-04-09	6.9	6.0	3.2	1.3
2024-04-17 to 2024-04-18	5.3	21.4	5.6	1.8
2024-04-26 to 2024-04-27	3.7	9.4	4.8	1.3
2024-05-15 to 2024-05-19	3.6	12.8	12.8	1.9
2024-05-21 to 2024-05-22	1.2	42.8	16	4.8
2024-06-03	1.6	11.6	6.4	2.0
2024-06-18 to 2024-06-21	2.1	40.2	19.2	2.7
2024-06-28 to 2024-07-01	1.6	39.0	4.0	1.1
2024-08-07 to 2024-08-08	22.0	7.2	3.2	1.4
2024-08-15 to 2024-08-17	5.1	26	5.6	1.7

Table S2 Timing of Instream Sample Collection at Reference Outfall. "Upstream Lag" indicates the lag (in hours) between the time the inflow enters the creek and the time the upstream sampler is filled. "Downstream Lag" indicates the lag (in hours) between the time the inflow enters the creek and the time the downstream sampler is filled. "Difference" is the difference between the collection time of the upstream sample and the collection time of the downstream sample (in hours).

Date	Upstream Lag (hours)	Downstream Lag (hours)	Difference (hours)
2022-02-18	0.73	0.43	0.3
2022-09-25	0.61	0.67	0.07
2023-07-09	0.5	0.5	0
2023-07-26	0.5	0.5	0
2023-08-08	0.5	0.5	0
2023-08-16	0.5	0.5	0
2023-10-06	0.54	0.5	0.04
2023-10-28	0.54	0.5	0.04
2023-11-06	2.1	1.8	0.3
2024-05-18	4.8	4.8	0
2024-06-28	0.5	0.5	0

Table S3 Timing of Instream Sample Collection at Bioretention System Outflow. The "Upstream Lag" indicates the lag (in hours) between the time the outflow enters the creek and the time the upstream sampler is filled. The "Downstream Lag" indicates the lag (in hours) between the time the outflow enters the creek and the time the downstream sampler is filled. "Difference" is the difference between the collection time of the upstream sample and the collection time of the downstream sample (in hours).

Date	Upstream Lag (hours)	Downstream Lag (hours)	Difference (hours)
2022-09-25	5.1	1	4.1
2023-05-19	0.65	0.5	0.15
2023-06-24	5.5	5.25	0.25
2023-06-29	0.5	0.5	0
2023-07-26	0.5	0.61	0.11
2023-08-16	0.5	0.5	0
2023-10-28	0.5	0.5	0
2024-04-17	0.5	0.5	0
2024-05-18	0.5	0.5	0
2024-05-21	0.5	0.5	0
2024-06-28	0.5	0.5	0

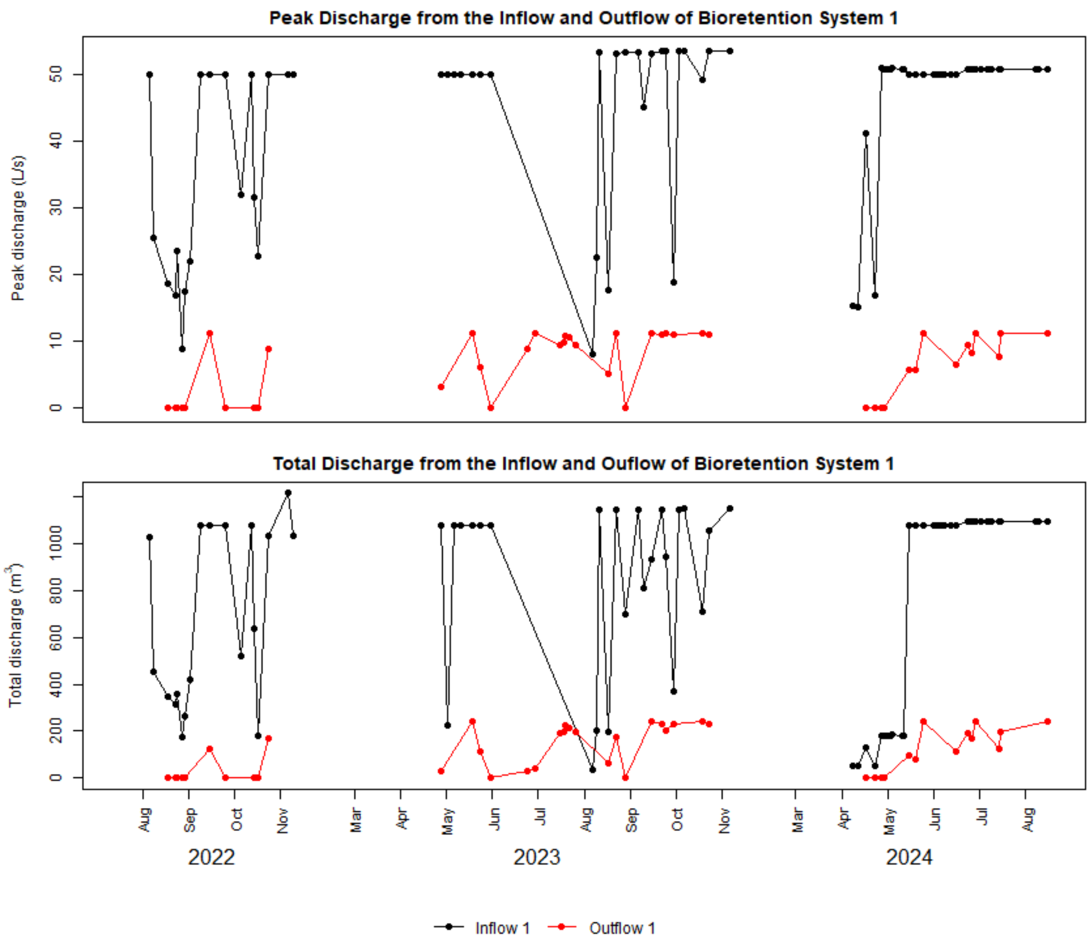

Figure S1 Peak and total discharge from the Inflow (Inflow 1) and Outflow (Outflow 1) during rainfall events between August 2022 and August 2024.

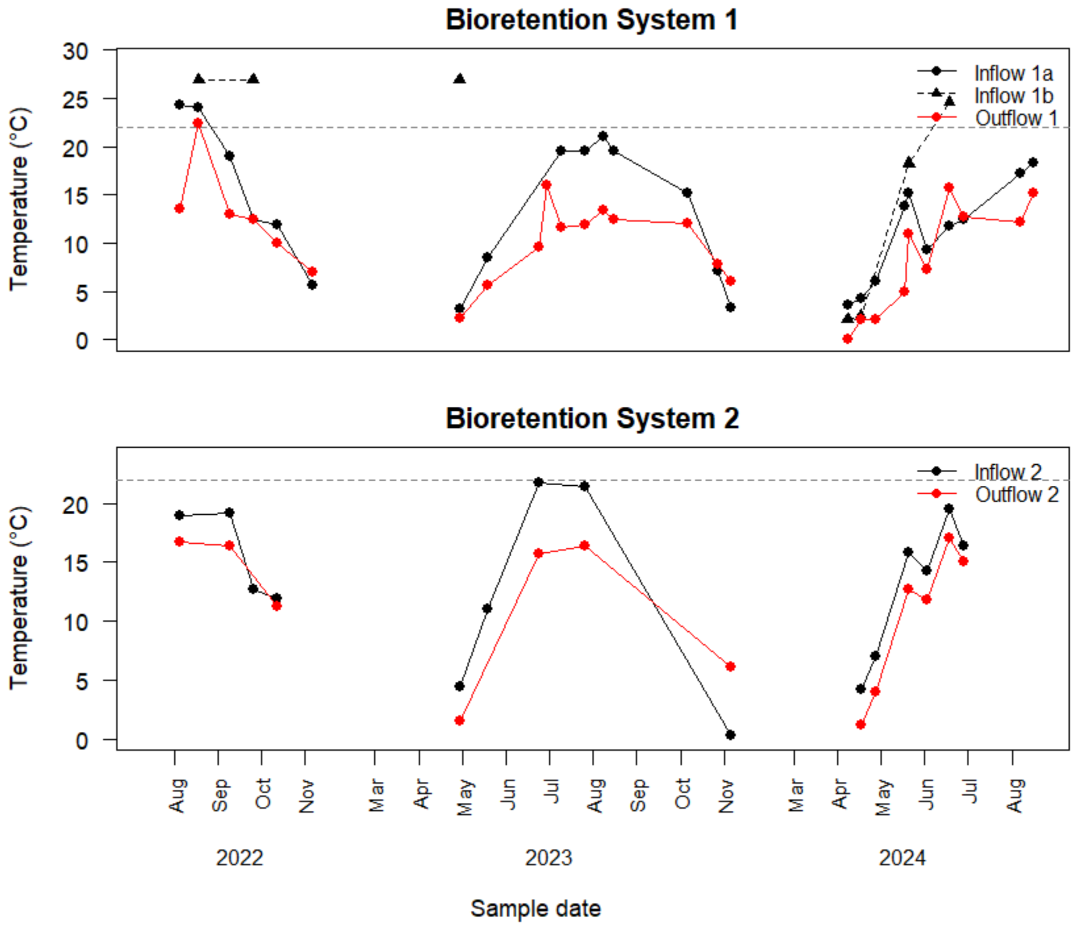

Figure S1 Peak temperature from the inflow (Inflow 1) and outflow (Outflow 1) of Bioretention System 1 and the inflow (Inflow 2) and outflow (Outflow 2) of Bioretention System 2 during rainfall events between August 2022 and August 2024. The dashed line represented the 22°C threshold associated with the onset of thermal stress in coldwater fish.

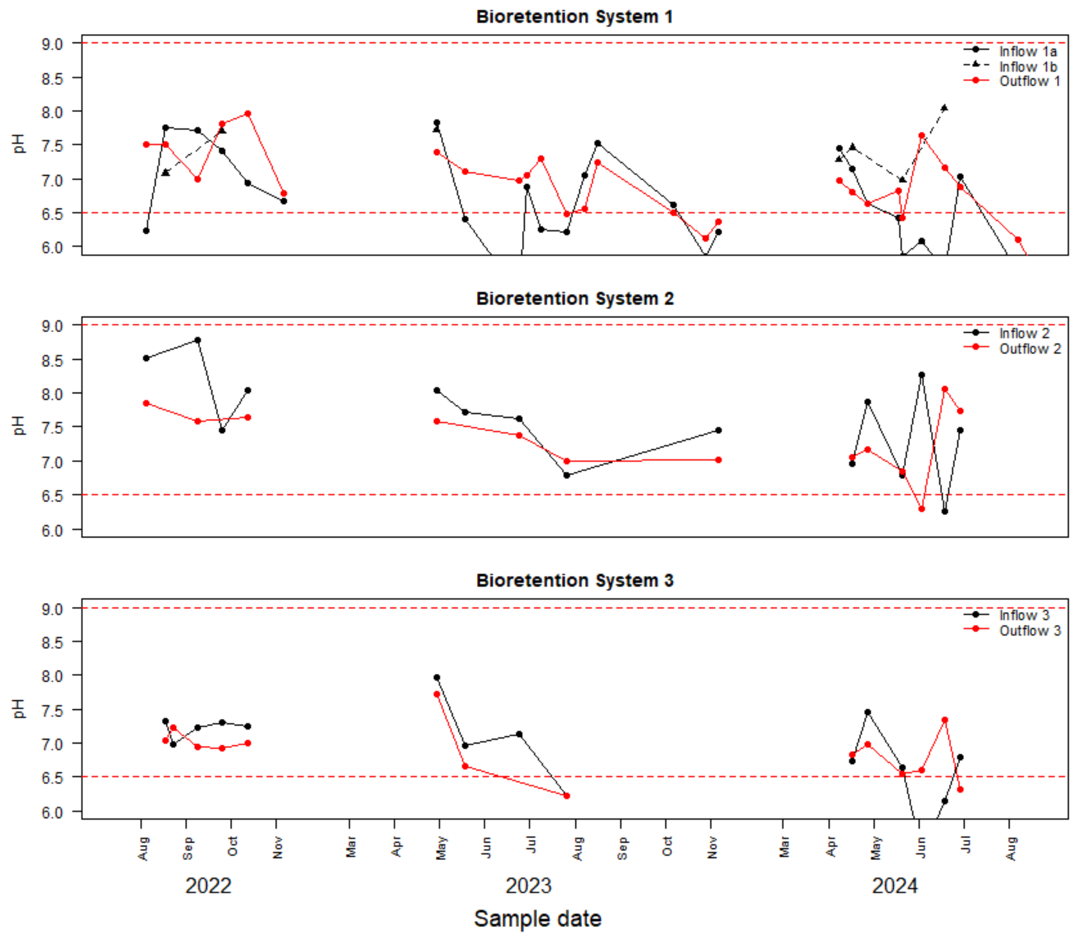

Figure S3 pH from the inflow(s) and outflow of the three bioretention systems during rainfall events. The red-dashed lines show the lower (6.5) and upper (9.0) CCME water quality guideline for the protection of aquatic life.

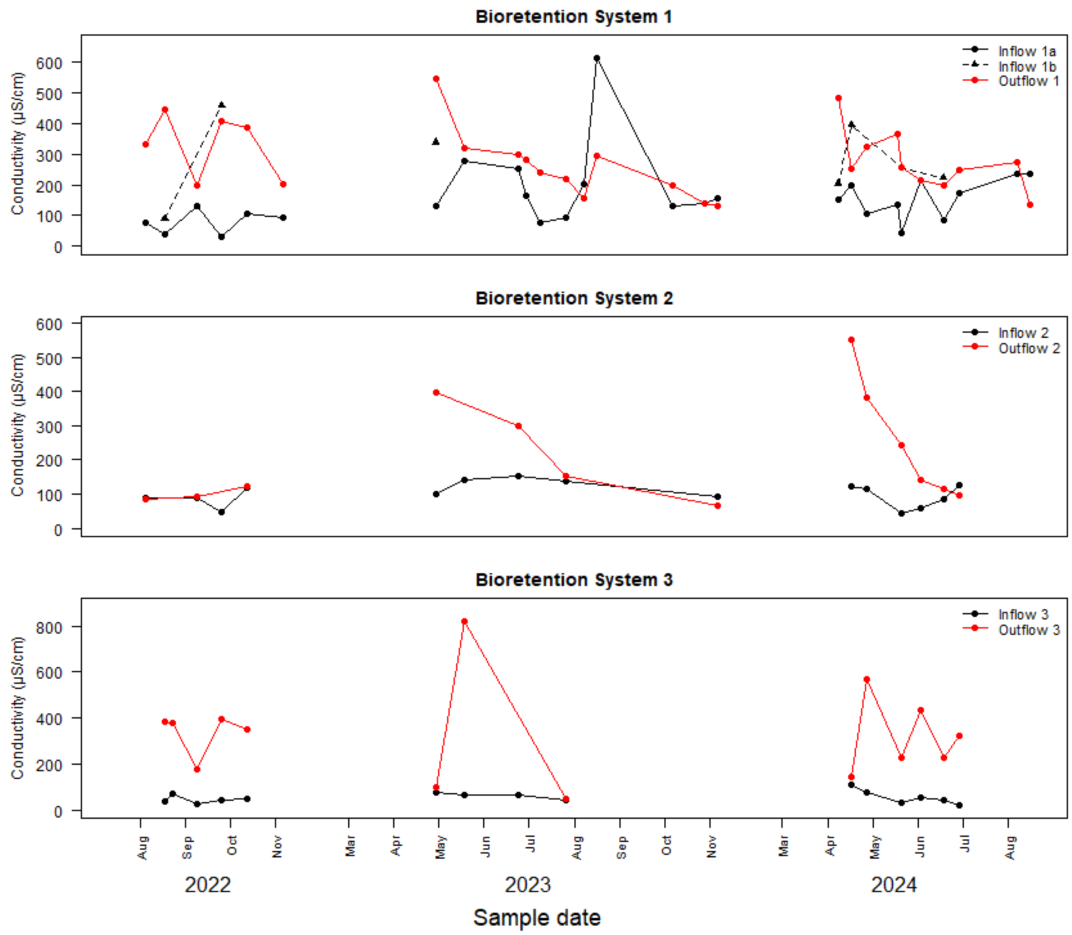

Figure S4 Conductivity from the inflow(s) and outflow of the three bioretention systems during rainfall events.

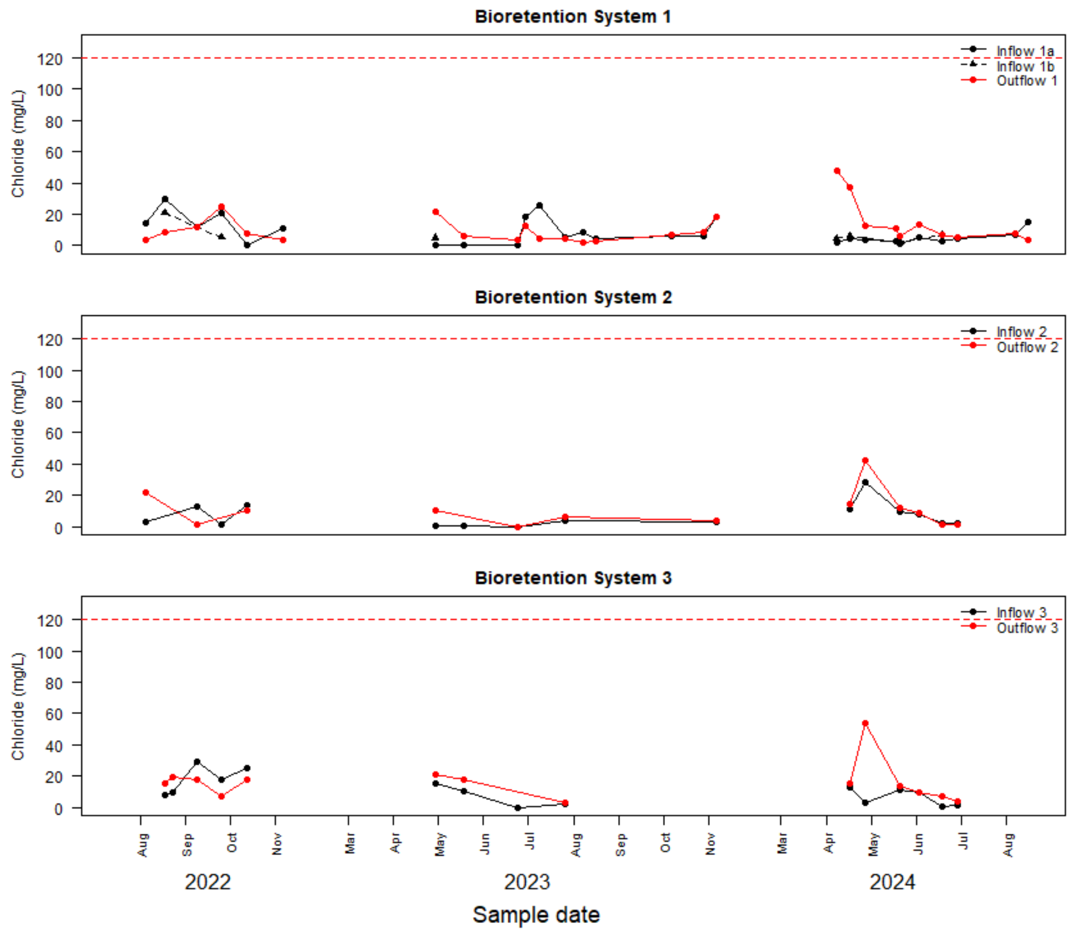

Figure S5 Chloride concentrations from the inflow(s) and outflow of the three bioretention systems during rainfall events. The red-dashed lines show the 120 mg/L CCME water quality guideline for the protection of aquatic life.

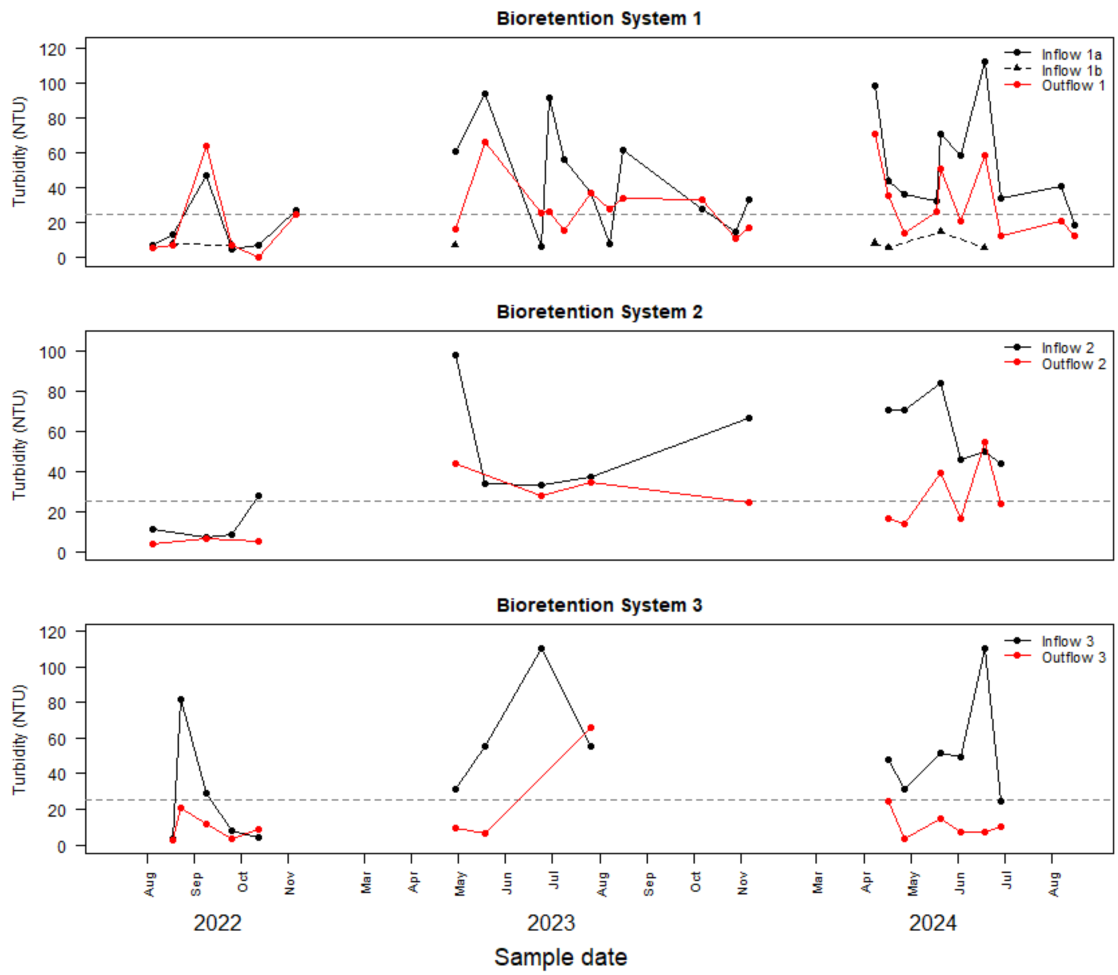

Figure S6 Turbidity from the inflow(s) and outflow of the three bioretention systems during rainfall events. The dashed line shows the 25 NTU literature-derived benchmark associated with reduced growth, fish density and behavioral effects in salmonids.

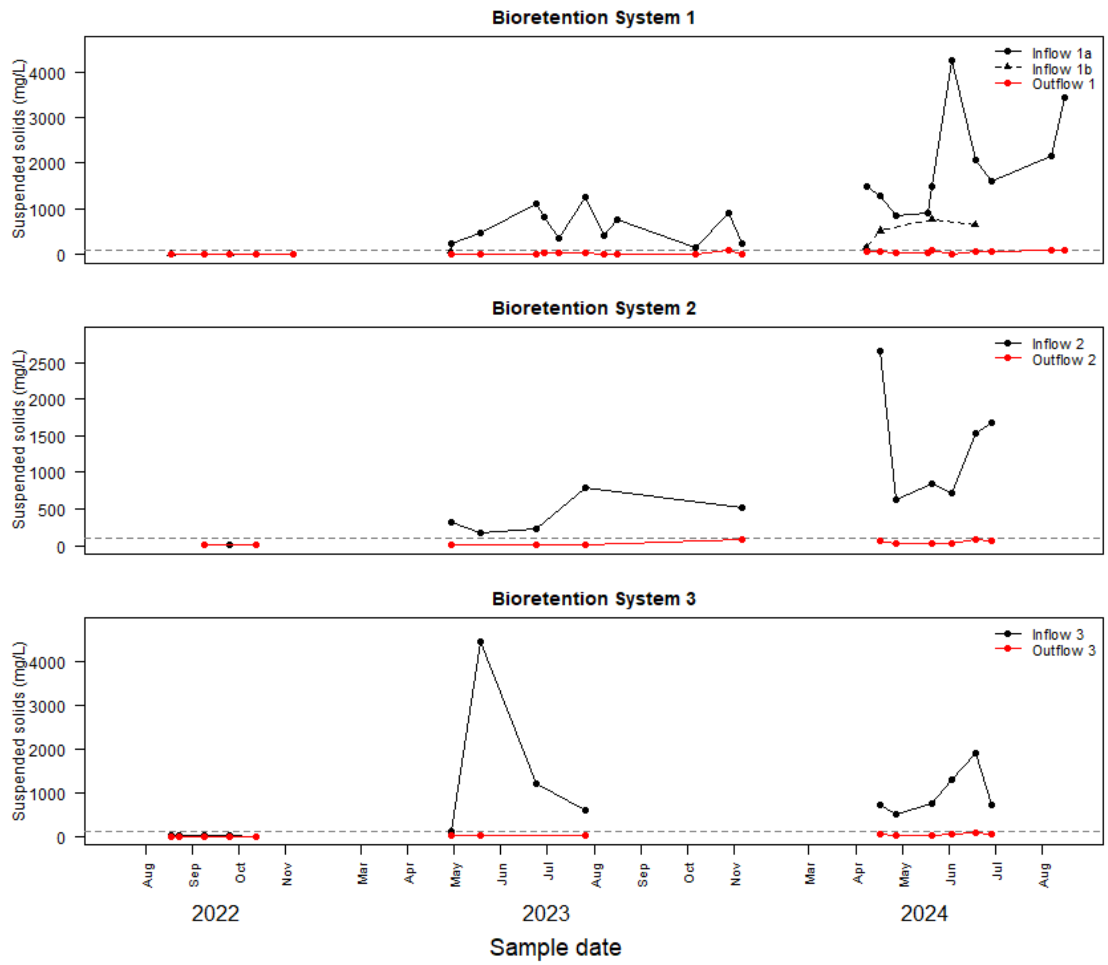

Figure S7 Suspended solids concentrations from the inflow(s) and outflow of the three bioretention systems during rainfall events. The dashed line shows the 100 mg/L literature-derived benchmark associated with sublethal effects in adult and juvenile fish.

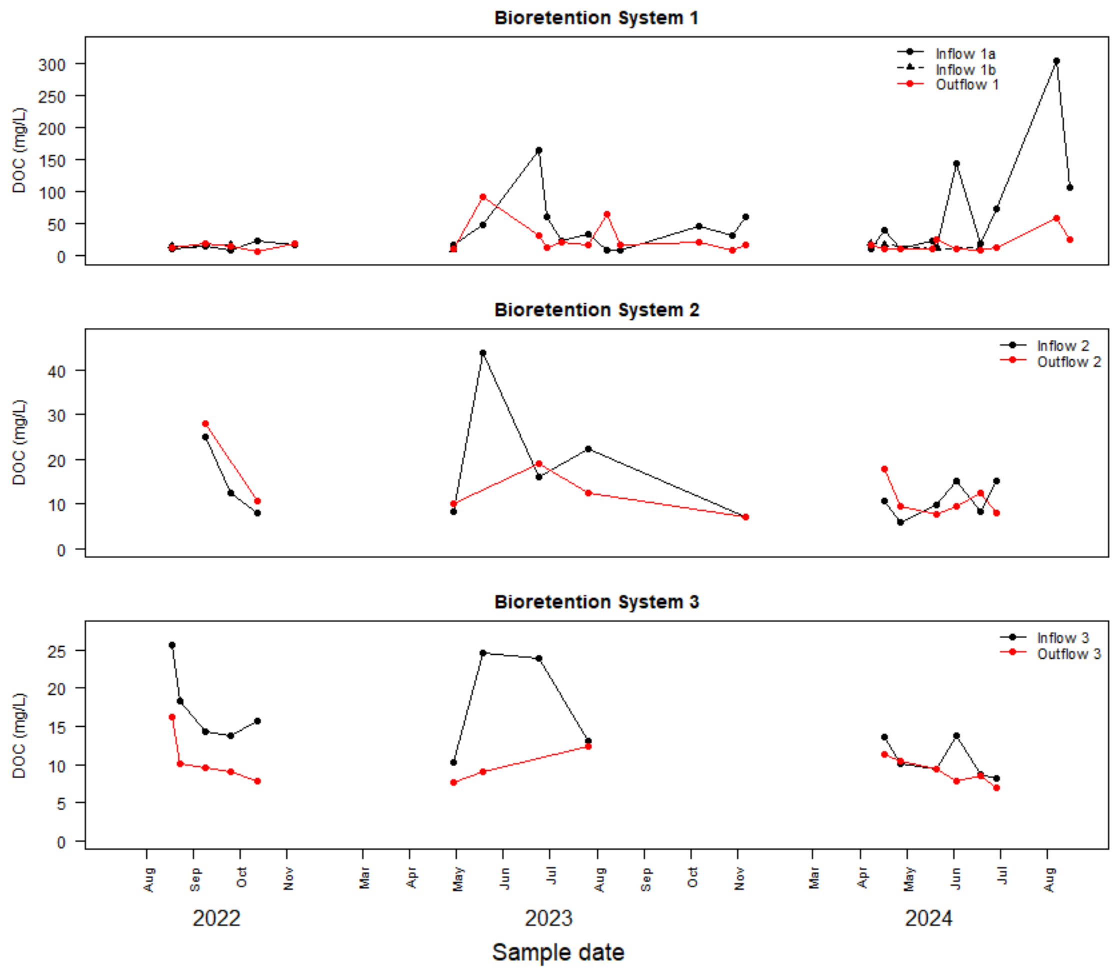

Figure S8 Dissolved organic carbon (DOC) concentrations from the inflow(s) and outflow of the three bioretention systems during rainfall events. No water quality guidelines or ecological benchmark exists for DOC concentrations.

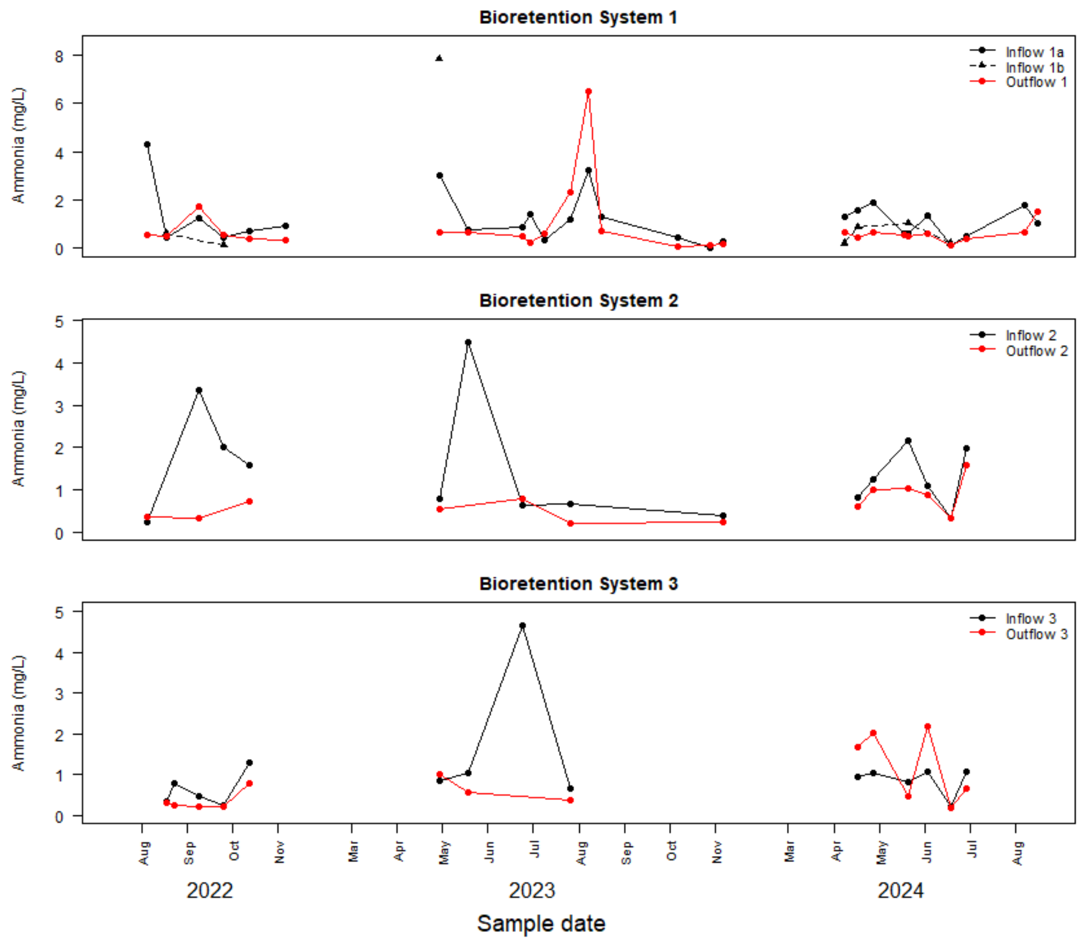

Figure S9 Ammonia concentrations from the inflows and outflows of the three bioretention systems during rainfall events. No fixed ammonia guideline is shown because ammonia toxicity thresholds vary with water temperature and pH. Exceedances where evaluated separately using temperature- and pH-dependent criteria.

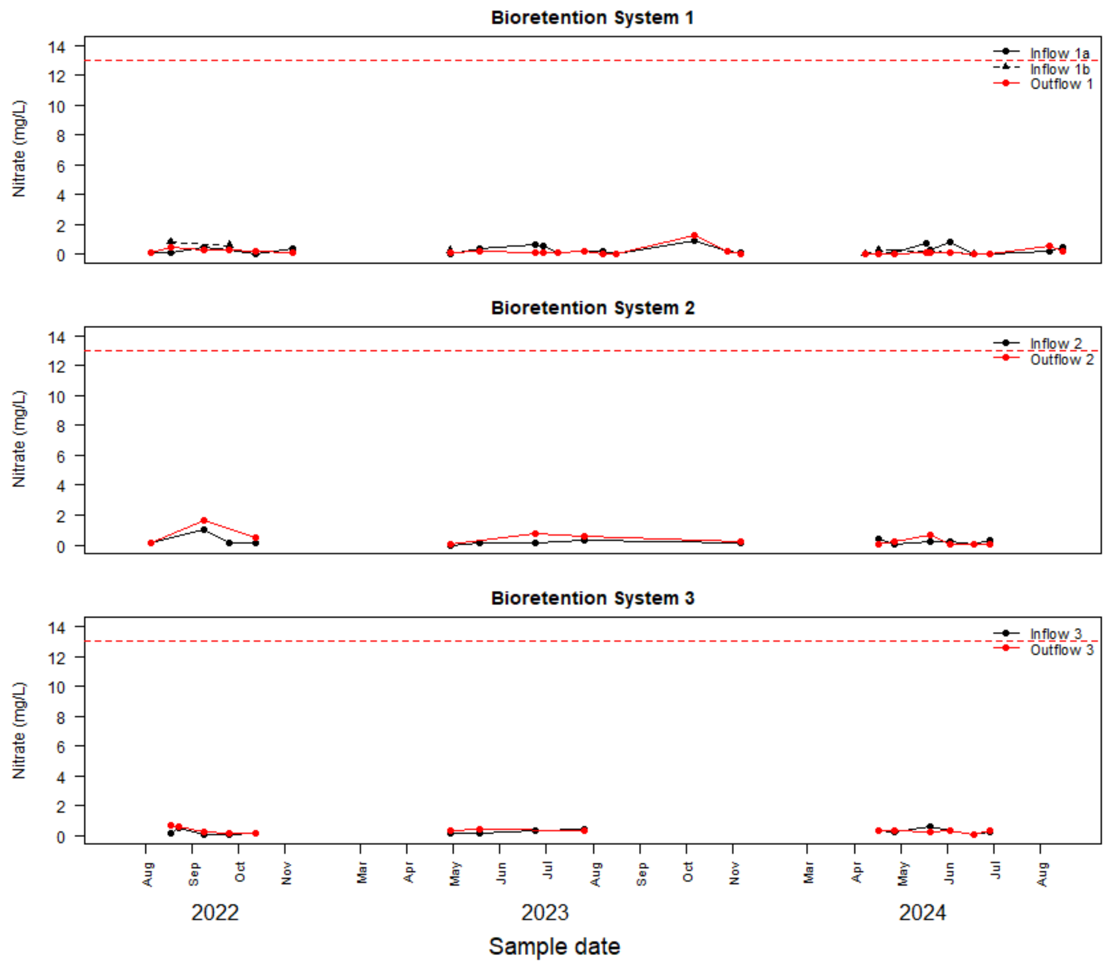

Figure S10 Nitrate concentrations from the inflows and outflows of the bioretention systems during rainfall events. The red dashed line represents the Canadian Council of Minsters of the Environment (CCME) water quality guideline for the protection of aquatic life for long-term nitrate exposure of 13 mg/L.

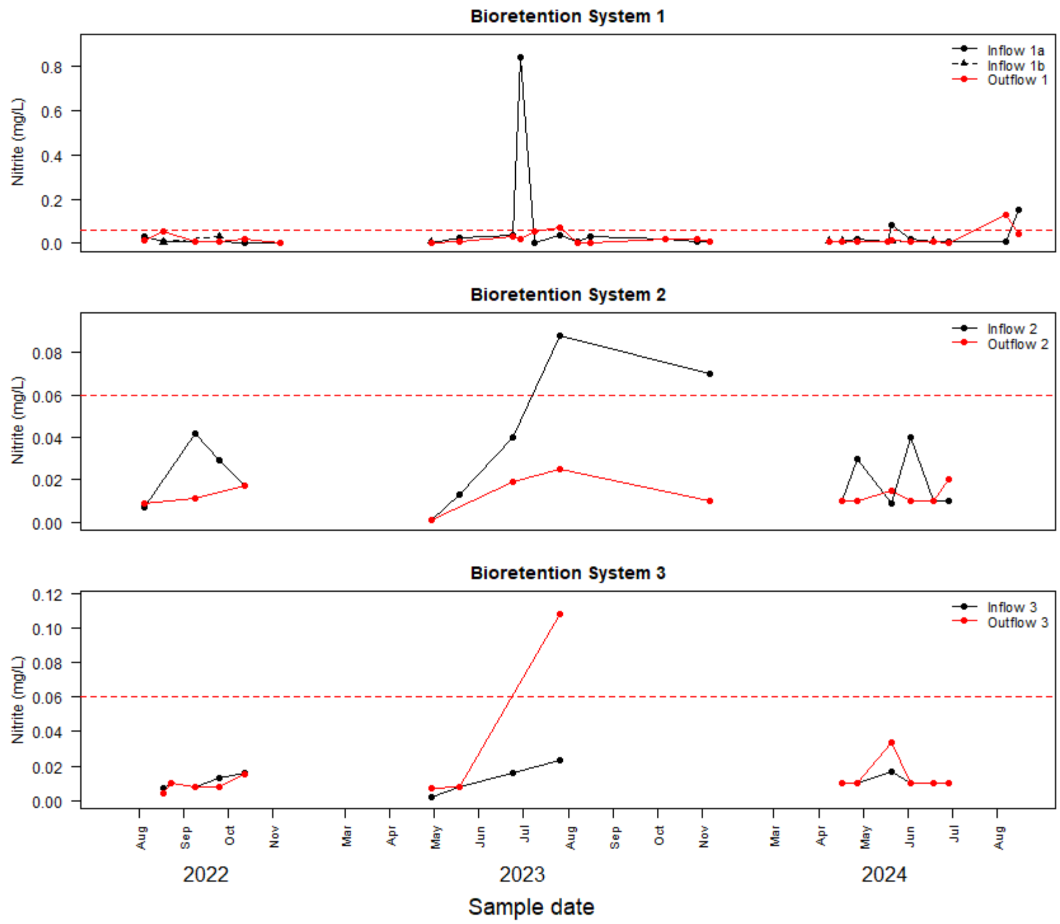

Figure S11 Nitrate concentrations from the inflows and outflows of the bioretention systems during rainfall events. The red dashed line represents the Canadian Council of Ministers of the Environment (CCME) water quality guideline for the protection of aquatic life of exposure of 0.06 mg/L.

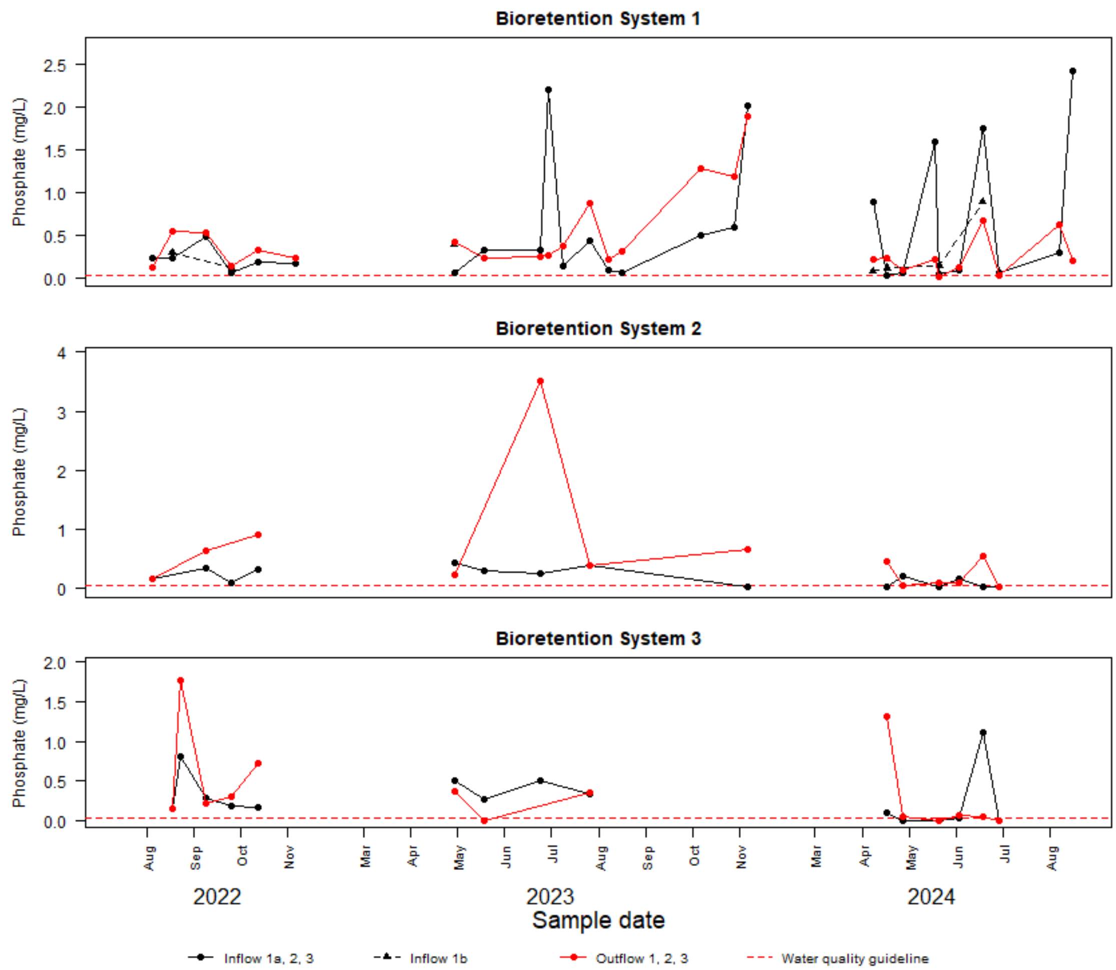

Figure S12 Phosphate concentrations from the inflows and outflows of the bioretention systems during rainfall events. The red dashed line represents the Canadian Council of Ministers for the Environment (CCME) water quality guideline of 0.03 mg/L for oligotrophic systems.

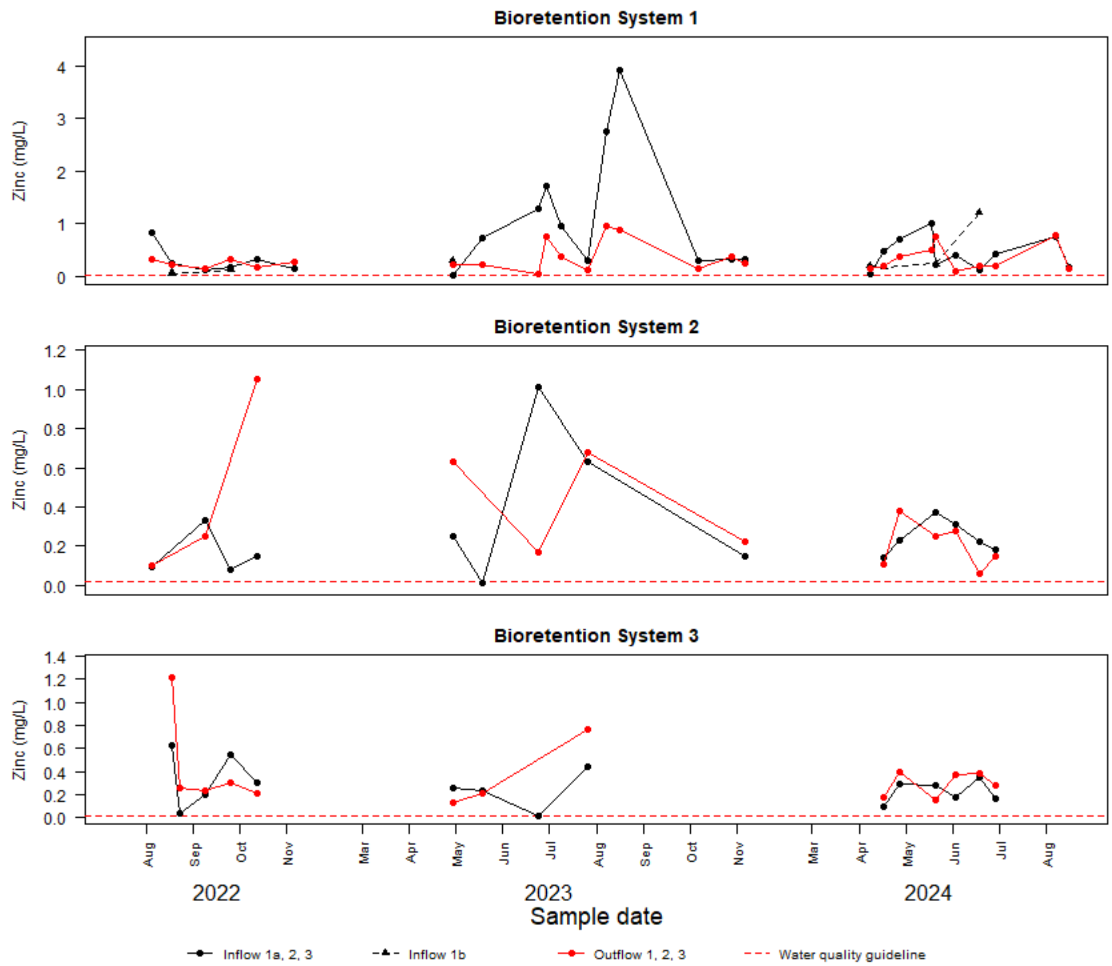

Table S13 Zinc concentrations from the inflows and outflows of the bioretention systems during rainfall events. The red dashed line represents the Provincial Water Quality Objective of 20 µg/L (0.02 mg/L). The CCME guideline was not used here as the guideline is dependent on water hardness which was not measured in this study.

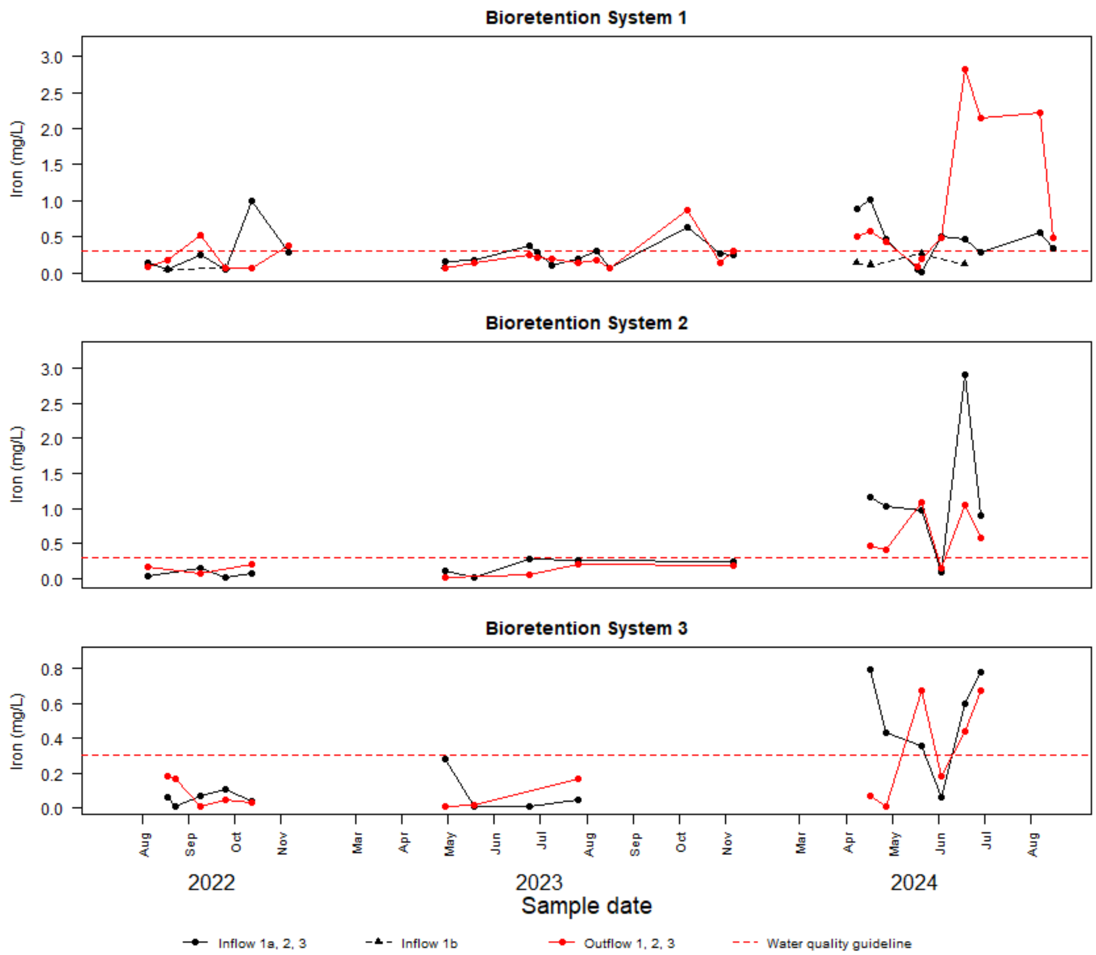

Table S14 Iron concentrations from the inflows and outflows of the bioretention systems. The red dashed line represents the CCME water quality guideline of 300 µg/L (equivalent to 0.3 mg/L).

Analytical methods

Parameter	2022 Instrument	2022 Method	2023-2024 Instrument	2023-2024 Method
Ammonia	YSI 9300 photometer	Indophenol method; Phot 4	LaMotte SMART3 colorimeter	Salicylate method; Code 3659-01-SC
Nitrate	YSI 9300 photometer	Nitratest reduction/ diazonium method; Phot 23	LaMotte SMART3 colorimeter	Cadmium reduction method; Code 3649-SC
Nitrite	YSI 9300 photometer	Nitricol diazonium method; Phot 24	LaMotte SMART3 colorimeter	Diazotization method; Code 3650-SC
Orthophosphate	YSI 9300 photometer	Low range phosphate colorimetric method; Phot 28	LaMotte SMART3 colorimeter	Ascorbic acid reduction method; Code 3653-SC
Chloride	YSI 9300 photometer	Chloridol silver nitrate turbidity method; Phot 46	LaMotte SMART3 colorimeter	Argentometric method; Code 3693-SC
Zinc	YSI 9300 photometer	Zincon method; Phot 35	LaMotte SMART3 colorimeter	Zincon method; Code 3667-SC
Iron	YSI 9300 photometer	Manufacturer-specified colorimetric method	LaMotte SMART3 colorimeter	Bipryidyl method; Code 3648-SC
Suspended solids	Gravimetric analysis	Filtration through 2 µm pre-combusted, pre-weighed glass fibre filters; dried at 104°C for 1 hour	Gravimetric analysis	Filtration through 2 µm pre-combusted, pre-weighed glass fibre filters; dried at 104°C for 1 hour
Dissolved organic carbon	Shimadzu TOC-L CPH total organic carbon analyzer	High-temperature catalytic combustion oxidation with non-dispersive infrared detection.	Shimadzu TOC-L CSH total organic carbon analyzer	High-temperature catalytic combustion oxidation with non-dispersive infrared detection.

Assumption Tests and Transformations

Rainfall characteristics (One-way ANOVA)

Parameter	Levene *p*	AD *p*	Transformation	Constant	IQR Outlier Removal	Transformed Levene *p*	Transformed AD *p*	Test Used
Rainfall Depth	0.07	< 0.0001	Log10	1×10^{-6} mm	N/A	0.64	0.47	ANOVA
Rainfall Intensity	0.52	< 0.0001	Log10	1×10^{-6} mm/hr	N/A	0.29	0.09	ANOVA
Antecedent Dry Period (ADP)	0.9	<0.0001	Log10	1×10^{-6} days	N/A	0.78	0.08	ANOVA

Inflow comparisons (One-way ANOVA)

Parameter	Levene p	AD p	Constant	Transformation	IQR Outlier Removal	Transformed Levene p	Transformed AD p	Test Used
Peak Discharge	<0.0001	0.001	N/A	N/A	N/A	N/A	N/A	Kruskal-Wallis
Total Discharge	<0.0001	<0.0001	N/A	N/A	N/A	N/A	N/A	Kruskal-Wallis
Temperature	0.26	0.007	N/A	Power (exponent 1.5)	N/A	0.48	0.08	ANOVA
pH	0.29	0.86	N/A	N/A	N/A	N/A	N/A	ANOVA
Conductivity	0.01	<0.0001	N/A	Log10	N/A	N/A	N/A	ANOVA
Turbidity	0.7	<0.0001	2 NTU	Inverse	1.5×	0.49	0.4	ANOVA
Suspended Solids	0.62	<0.0001	N/A	N/A	N/A	N/A	N/A	Kruskal-Wallis
Dissolved Organic Carbon	0.03	<0.0001	1e-6 mg/L	Inverse	N/A	0.06	0.13	ANOVA
Ammonia	0.46	<0.0001	0.005	Log10	N/A	0.45	0.06	ANOVA
Nitrate	0.43	<0.0001	N/A	Square Root	N/A	0.31	0.63	ANOVA
Nitrite	0.58	<0.0001	0.0001	Log 10	1.5× (6 outliers removed)	0.51	0.51	ANOVA
Orthophosphate	0.1	<0.0001	N/A	Cube Root	1.0×	0.37	0.13	ANOVA
Zinc	0.12	<0.0001	N/A	Cube Root	N/A	0.19	0.07	ANOVA
Iron	0.06	<0.0001	N/A	Yeo-Johnson	N/A	0.06	0.06	ANOVA
Chloride	0.76	<0.0001	N/A	Square Root	N/A	0.51	0.66	ANOVA

Inflow to outflow comparisons (paired *t*-test)

Parameter	Site	AD or SW *p*	Constant	Transformation	Outlier Removal	Transformed AD or SW *p*	Test Used
Peak discharge	1	0.02 (*AD*)	N/A	N/A	N/A	N/A	Paired Wilcoxon Rank-Sum
	2	0.02 (*AD*)	N/A	N/A	N/A	N/A	Paired Wilcoxon Rank-Sum
Total discharge	1	<0.0001 (*AD*)	N/A	Yeo-Johnson	N/A	0.55 (*AD*)	Paired *t*-test
	2	<0.0001(*AD*)	N/A	N/A	N/A	N/A	Paired Wilcoxon Rank-Sum
pH	1	0.15 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
	1b	0.78 (*SW*)	N/A	N/A	N/A	N/A	Paired *t*-test
	2	0.2 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
	3	0.0001 (*AD*)	N/A	Square root	1.5×	0.44 (*AD*)	Paired *t*-test
Conductivity	1a	0.44 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
	1b	0.41 (*SW*)	N/A	N/A	N/A	N/A	Paired *t*-test
	2	0.03 (*AD*)	5 µS/cm	Inverse	N/A	0.81	Paired *t*-test
	3	0.37 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
Turbidity	1a	<0.0001 (*AD*)	N/A	N/A	1.0×	0.31	Paired *t*-test
	1b	0.39 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
	2	<0.0001 (*AD*)	1 NTU	Log10	N/A	0.13 (*AD*)	Paired *t*-test
	3	0.39 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
Suspended Solids	1a	0.001 (*AD*)	N/A	Log10	N/A	0.3 (*AD*)	Paired *t*-test
	1b	0.02 (*AD*)	1 mg/L	Log10	N/A	0.43	Paired *t*-test
	2	0.02 (*AD*)	N/A	Cube root	N/A	0.31 (*AD*)	Paired *t*-test
	3	0.0004 (*AD*)	N/A	Log10	1.0×	0.51 (AD)	Paired *t*-test
Dissolved organic carbon	1a	0.0001 (*AD*)	N/A	Log10	N/A	0.45 (*AD*)	Paired *t*-test
	1b	0.07 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
	2	0.14 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
	3	0.21 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
Ammonia	1a	0.002 (*AD*)	N/A	Cube-root	N/A	0.97 (*AD*)	Paired *t*-test
	1b	0.0004 (*AD*)	N/A	Signed cube-root	N/A	0.1 (SW)	Paired t-test
	2	0.0003 (*AD*)	N/A	Yeo-Johnson	N/A	0.6 (*AD*)	Paired t-test
	3	0.01 (*AD*)	N/A	Yeo-Johnson	N/A	0.54 (*AD*)	Paired t-test
Nitrate	1a	0.04 (*AD*)	N/A	Yeo-Johnson	N/A	0.1 (*AD*)	Paired *t*-test
	1b	0.3 (*SW*)	N/A	N/A	N/A	N/A	Paired *t*-test
	2	0.91 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
	3	0.07 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
Nitrite	1a	< 0.0001 (*AD*)	N/A	Signed cube-root	N/A	0.15 (*AD*)	Paired *t*-test
	1b	0.04 (*SW*)	N/A	Yeo-Johnson	N/A	0.4 (SW)	Paired *t*-test
	2	0.02 (*AD*)	N/A	Signed square-root	N/A	0.09 (*AD*)	Paired *t*-test
	3	< 0.0001 (*AD*)	N/A	N/A	N/A	N/A	Paired Wilcoxon signed-rank
Orthophosphate	1a	< 0.0001 (*AD*)	N/A	Yeo-Johnson	N/A	0.1 (*AD*)	Paired *t*-test

Parameter	Site	AD or SW *p*	Constant	Transformation	Outlier Removal	Transformed AD or SW *p*	Test Used
	1b	0.85 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
	2	< 0.0001 (*AD*)	N/A	Yeo-Johnson	N/A	0.65 (*AD*)	Paired *t*-test
	3	< 0.0001 (*AD*)	N/A	Signed square-root	N/A	0.96 (*AD*)	Paired *t*-test
Zinc	1a	< 0.0001 (*AD*)	N/A	Signed cube-root	N/A	0.55 (*AD*)	Paired *t*-test
	1b	0.09 (*SW*)	N/A	N/A	N/A	N/A	Paired *t*-test
	2	0.007 (*AD*)	N/A	Signed cube-root	N/A	0.96 (*AD*)	Paired *t*-test
	3	0.62 (*SW*)	N/A	N/A	N/A	N/A	Paired *t*-test
Iron	1a	< 0.0001 (*AD*)	N/A	N/A	1.5×	0.46 (*AD*)	Paired *t*-test
	1b	0.0007 (*SW*)	N/A	Cube-root	N/A	0.11 (*SW*)	Paired *t*-test
	2	0.001 (*AD*)	N/A	Cube-root	N/A	0.55 (*AD*)	Paired *t*-test
	3	0.2 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
Chloride	1a	0.007 (*AD*)	N/A	Yeo-Johnson	N/A	0.34 (*AD*)	Paired *t*-test
	1b	0.98 (*SW*)	N/A	N/A	N/A	N/A	Paired *t*-test
	2	0.08 (*AD*)	N/A	N/A	N/A	N/A	Paired *t*-test
	3	0.0007 (*AD*)	N/A	Box-Cox	N/A	0.05	Paired *t*-test

Discharge and temperature reductions between sites (paired t-test)

Parameter	Inflow AD or SW *p*	Outflow AD or SW *p*	Levene *p*	Constant	Transformation	IQR Outlier Removal	Transformed Levene *p*	Transformed Inflow AD or SW *p*	Transformed Outflow AD or SW *p*	Test Used
Peak Discharge	< 0.0001	0.2	0.78	N/A	N/A	N/A	N/A	N/A	N/A	Wilcoxon Rank-Sum
Total Discharge	< 0.0001	< 0.0001	0.12	N/A	N/A	N/A	N/A	N/A	N/A	Wilcoxon Rank-Sum
Peak Temperature	0.06	0.69	0.97	N/A	N/A	N/A	N/A	N/A	N/A	Paired *t*

Quality percent reduction (one-way ANOVA)

Parameter	AD *p*	Levene *p*	Constant	Transformation	IQR Outlier Removal	Transformed AD *p*	Transformed Levene *p*	Test used
pH	0.07	0.75	N/A	N/A	N/A	N/A	N/A	One-way ANOVA
Conductivity	< 0.0001	0.19	N/A	Signed square root	N/A	0.81	0.98	One-way ANOVA
Turbidity	< 0.0001	0.86	N/A	Yeo-Johnson	N/A	0.51	0.86	One-way ANOVA
Suspended Solids	0.0001	0.006	N/A	Yeo-Johnson	N/A	0.18	0.25	One-way ANOVA
Dissolved Organic Carbon	< 0.0001	0.19	N/A	N/A	0.5×	0.13	0.15	One-way ANOVA
Ammonia	< 0.0001	0.57	N/A	Yeo-Johnson	N/A	0.35	0.31	One-way ANOVA
Nitrate	< 0.0001	0.72	N/A	Arcsinh	N/A	0.05	0.67	One-way ANOVA
Nitrite	< 0.0001	0.24	N/A	N/A	N/A	N/A	N/A	Kruskal-Wallis rank sum
Orthophosphate	< 0.0001	0.03	N/A	N/A	N/A	N/A	N/A	Kruskal-Wallis rank sum
Zinc	< 0.0001	0.42	N/A	Yeo-Johnson	N/A	0.48	0.42	One-way ANOVA
Iron	< 0.0001	0.27	N/A	Yeo-Johnson	N/A	0.43	0.24	One-way ANOVA
Chloride	<0.0001	0.67	N/A	Signed square root	N/A	0.13	0.2	One-way ANOVA

Inflow Reference vs. Inflow 1a vs. Outflow 1 comparisons (Repeated Measures ANOVA)

Parameter	AD *p*	Levene *p*	Constant	Transformation	IQR Outlier Removal	Transformed AD *p*	Transformed Levene *p*	Test used
pH	0.75	0.5	N/A	N/A	N/A	N/A	N/A	RM ANOVA
Conductivity	0.23	0.45	N/A	N/A	N/A	N/A	N/A	RM ANOVA
Turbidity	0.09	0.91	N/A	N/A	N/A	N/A	N/A	RM ANOVA
Suspended solids	0.46	0.11	N/A	N/A	N/A	N/A	N/A	RM ANOVA
DOC	< 0.0001	0.71	N/A	N/A	0.5×	0.88	0.07	RM ANOVA
Ammonia	< 0.0001	0.87	N/A	N/A	0.5×	0.85	0.86	RM ANOVA
Nitrate	0.0003	0.57	N/A	N/A	0.5×	0.41	0.77	RM ANOVA
Nitrite	< 0.0001	0.86	N/A	N/A	0.5×	0.97	0.36	RM ANOVA
Orthophosphate	0.0003	0.98	N/A	N/A	0.5×	0.42	0.69	RM ANOVA
Zinc	0.001	0.38	N/A	N/A	0.5×	0.09	0.63	RM ANOVA
Iron	0.001	0.96	N/A	N/A	0.5×	0.28	0.51	RM ANOVA
Chloride	0.24	0.38	N/A	N/A	N/A	N/A	N/A	RM ANOVA

Instream sample collection timing (paired-t-test).

Site	Ad *p*	Transformation	Constant	Outlier Removal	Transformed AD *p*	Test Used
Reference	< 0.0001	N/A	N/A	N/A	N/A	Wilcoxon signed rank
Bioretention	< 0.0001	N/A	N/A	N/A	N/A	Wilcoxon signed rank

Upstream and downstream comparisons (paired t-test)

Parameter	Site	Ad *p*	Transformation	Constant	Outlier Removal	Transformed AD *p*	Test Used
pH	Reference	0.12	N/A	N/A	N/A	N/A	Paired *t*
	Bioretention	0.006	Signed log10	0.02	1.5×	0.81	Paired *t*
Conductivity	Reference	0.94	N/A	N/A	N/A	N/A	Paired *t*
	Bioretention	0.83	N/A	N/A	N/A	N/A	Paired *t*
Turbidity	Reference	0.002	Signed square root	N/A	N/A	0.97	Paired *t*
	Bioretention	< 0.0001	Signed log10	N/A	N/A	0.74	Paired *t*
Suspended Solids	Reference	< 0.0001	Yeo-Johnson	N/A	N/A	0.35	Paired *t*
	Bioretention	< 0.0001	Signed log10	N/A	N/A	0.92	Paired *t*
DOC	Reference	< 0.0001	N/A	N/A	1.5×	0.52	Paired *t*
	Bioretention	< 0.0001	N/A	N/A	1.5×	0.84	Paired *t*
Ammonia	Reference	0.004	Signed square root	0.01	N/A	0.98	Paired *t*
	Bioretention	< 0.0001	Signed square root	N/A	4.0×	0.9	Paired *t*
Nitrate	Reference	0.002	Signed square root	0.001	N/A	0.99	Paired *t*
	Bioretention	< 0.0001	Signed square root	0.001	1.5×	0.36	Paired *t*
Nitrite	Reference	< 0.0001	Signed square root	N/A	4.0×	0.27	Paired *t*
	Bioretention	0.003	Signed square root	N/A	N/A	0.42	Paired *t*
Phosphate	Reference	< 0.0001	Signed square root	N/A	4.0×	0.27	Paired *t*
	Bioretention	0.003	Signed square root	0.01	1.5×	0.72	Paired *t*
Zinc	Reference	< 0.0001	Signed square root	N/A	3.0×	0.57	Paired *t*
	Bioretention	< 0.0001	Signed square root	N/A	1.5×	0.88	Paired *t*
Iron	Reference	0.83	N/A	N/A	N/A	N/A	Paired *t*
	Bioretention	< 0.0001	Signed square root	N/A	4.0×	0.68	Paired *t*
Chloride	Reference	0.0005	Signed square root	N/A	N/A	0.78	Paired *t*
	Site 1	< 0.0001	Signed square root	N/A	4.0×	0.68	Paired *t*

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Supplementary Materials for Chapter 4

Table S1: Maximum daily air temperature (^{o}C) on dates of melt events.

Date	Maximum Daily Air Temperature (^{o}C)	
April 5, 2023	1.3	
April 12, 2023	15.2	
April 15, 2023	12.0	
April 16, 2023	1.7	* Rain-on-snow event, 16 mm precipitation
April 18, 2023	5.7	
April 21, 2023	4.0	
April 25, 2023	5.1	
March 10, 2024	6.3	
March 15, 2024	9.4	
March 31, 2024	4.0	
April 6, 2024	9.4	

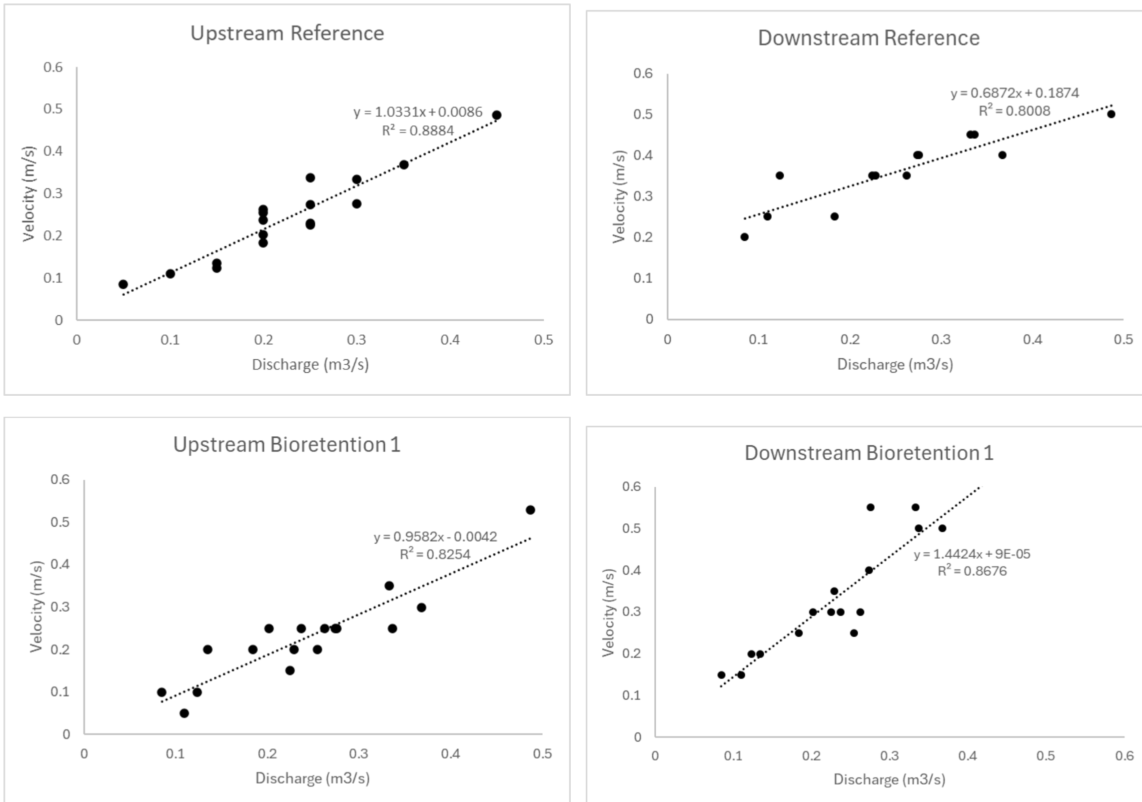


Figure S1: Relationship between stream velocity (m/s) and discharge (m^3/s) at instream sample locations.

Analytical methods

Parameter	Instrument	Method
Ammonia	LaMotte SMART3 colorimeter	Salicylate method; Code 3659-01-SC
Nitrate	LaMotte SMART3 colorimeter	Cadmium reduction method; Code 3649-SC
Nitrite	LaMotte SMART3 colorimeter	Diazotization method; Code 3650-SC
Orthophosphate	LaMotte SMART3 colorimeter	Ascorbic acid reduction method; Code 3653-SC
Chloride	LaMotte SMART3 colorimeter	Argentometric method; Code 3693-SC
Zinc	LaMotte SMART3 colorimeter	Zincon method; Code 3667-SC
Iron	LaMotte SMART3 colorimeter	Bipryidyl method; Code 3648-SC
Suspended solids	Gravimetric analysis	Filtration through 2 µm pre-combusted, pre-weighed glass fibre filters; dried at 104°C for 1 hour
Dissolved organic carbon	Shimadzu TOC-L CSH total organic carbon analyzer	High-temperature catalytic combustion oxidation with non-dispersive infrared detection.

Table S3: Snowpack chemistry by snow core type (field, snowbank, and bioretention systems). Values are reported as mean ± standard error (SE), median, and interquartile range (IQR; Q1-Q3). Sample size (n) represents the number of snow cores analyzed per snow core category.

Parameter	Core Type	n	Mean ± SE	Median	Q1	Q3	IQR
pH	Field	27	6.4 ± 0.1	6.3	5.9	6.5	0.6
pH	Snowbank	27	6.4 ± 0.1	6.5	6.1	6.7	0.7
pH	Bioretention	27	6.1 ± 0.1	6.1	5.9	6.3	0.4
Conductivity (µS/cm)	Field	27	76 ± 10	69	38	84	46
Conductivity (µS/cm)	Snowbank	27	2080 ± 572	1132	317	2581	2264
Conductivity (µS/cm)	Bioretention	27	69 ± 12	58	44	74	30
Turbidity (NTU)	Field	27	13 ± 2	9.4	7.2	19	11
Turbidity (NTU)	Snowbank	27	254 ± 127	48	36	96	60
Turbidity (NTU)	Bioretention	27	11 ± 1	9.9	8.6	13.7	5.2
TSS (mg/L)	Field	27	99 ± 34	46	32	96	64
TSS (mg/L)	Snowbank	27	4574 ± 357	4722	3728	5504	1775
TSS (mg/L)	Bioretention	27	197 ± 68	84	44	153	108
DOC (mg/L)	Field	27	7.2 ± 0.6	7.6	4.9	9.5	4.7
DOC (mg/L)	Snowbank	27	11 ± 1	11	6.3	12	6.5
DOC (mg/L)	Bioretention	27	6.7 ± 0.6	6.5	5.1	8.7	3.5
Ammonia (mg/L)	Field	27	1.2 ± 0.4	0.6	0.37	0.94	0.57
Ammonia (mg/L)	Snowbank	27	1.8 ± 0.4	0.88	0.58	2.1	1.5
Ammonia (mg/L)	Bioretention	27	0.85 ± 0.2	0.56	0.42	0.70	0.29
Nitrate (mg/L)	Field	27	0.17 ± 0.03	0.14	0.07	0.24	0.17
Nitrate (mg/L)	Snowbank	27	0.01 ± 0.03	0.05	0.001	0.15	0.14
Nitrate (mg/L)	Bioretention	27	0.15 ± 0.03	0.11	0.04	0.19	0.15
Nitrite (mg/L)	Field	27	0.009 ± 0.002	0.006	0.001	0.013	0.012
Nitrite (mg/L)	Snowbank	27	0.004 ± 0.001	0.001	0.0005	0.007	0.0065
Nitrite (mg/L)	Bioretention	27	0.005 ± 0.001	0.004	0.0005	0.0075	0.007
Phosphate (mg/L)	Field	27	0.41 ± 0.11	0.18	0.11	0.29	0.18
Phosphate (mg/L)	Snowbank	27	0.26 ± 0.06	0.18	0.09	0.29	0.21
Phosphate (mg/L)	Bioretention	27	0.13 ± 0.02	0.1	0.08	0.18	0.11
Zinc (mg/L)	Field	27	0.16 ± 0.04	0.11	0.03	0.21	0.19
Zinc (mg/L)	Snowbank	27	0.2 ± 0.04	0.12	0.04	0.29	0.25
Zinc (mg/L)	Bioretention	27	0.17 ± 0.04	0.09	0.04	0.23	0.19
Iron (mg/L)	Field	27	0.05 ± 0.01	0.03	0.01	0.08	0.07
Iron (mg/L)	Snowbank	27	0.25 ± 0.06	0.09	0.05	0.35	0.31
Iron (mg/L)	Bioretention	27	0.14 ± 0.06	0.03	0.01	0.11	0.10
Chloride (mg/L)	Field	27	7.4 ± 0.5	7.2	5.8	8.8	3

Parameter	Core Type	n	Mean ± SE	Median	Q1	Q3	IQR
Chloride (mg/L)	Snowbank	27	21 ± 1	21	17	23	5.6
Chloride (mg/L)	Bioretention	27	5.7 ± 0.7	5.1	3.3	8.4	5.2

Table S4: Daily peak and total discharge from the Reference Inflow, Inflow 1a, Inflow 1b, Outflow 1, Inflow 2, and Outflow 2 during the 2023 and 2024 Spring Melt

	Peak Discharge (L/s)		Total Discharge (L)	
	2023	2024	2023	2024
Reference Inflow	13 ± 1	13 ± 1	4 017 ± 264	2896 ± 167
Inflow 1	7.9 ± 0.2	9.9 ± 0.3	3290 ± 141	4860 ± 257
Inflow 1a	7.3 ± 0.1	7.8 ± 0.1	3175 ± 138	4113 ± 172
Inflow 1b	0.6 ± 0.04	2.06 ± 0.22	115 ± 3	747 ± 85
Outflow 1	0.90 ± 0.11	0.75 ± 0.12	145 ± 18	109 ± 17
Inflow 2	0.57 ± 0.02	0.37 ± 0.01	189 ± 8	167 ± 7
Outflow 2	0.090 ± 0.002	0.050 ± 0.002	22.6 ± 0.8	27.8 ± 17.1

Assumption Tests and Transformations

Snow depth and liquid Equivalent (paired t-test)

Parameter	Normality of Paired Differences (SW)	Test Used
Snow depth	0.18	Paired *t*-test
Liquid equivalent	0.29	Paired *t*-test
Water content	0.11	Paired *t*-test

Snow cores (Linear Mixed Effects (LME) Model when assumptions met, Kruskal-Wallis (K-W) when not met)

Parameter	AD *p*	Levene *p*	Constant	Transformation	IQR Outlier Removal	Transformed AD *p*	Transformed Levene *p*	Test used
pH	0.58	0.14	N/A	N/A	N/A	N/A	N/A	LME
Conductivity	< 0.0001	< 0.0001	N/A	N/A	N/A	N/A	N/A	K-W
Turbidity	0.06	< 0.0001	N/A	N/A	N/A	N/A	N/A	K-W
Suspended Solids	< 0.0001	< 0.0001	N/A	N/A	N/A	N/A	N/A	K-W
DOC	0.05	0.03	N/A	N/A	N/A	N/A	N/A	K-W
Ammonia	< 0.0001	0.23	N/A	N/A	N/A	N/A	N/A	K-W
Nitrate	< 0.0001	0.7	N/A	Yeo-Johnson	N/A	0.13	0.72	LME
Nitrite	< 0.0001	0.15	N/A	N/A	N/A	N/A	N/A	K-W
Phosphate	< 0.0001	0.08	N/A	Yeo-Johnson	N/A	0.32	0.08	LME
Zinc	< 0.0001	0.07	N/A	log10	N/A	0.28	0.6	LME
Iron	0.01	<0.0001	N/A	log10	N/A	0.38	0.11	LME
Chloride	0.0006	<0.0001	N/A	Square-root	N/A	0.27	0.17	LME

Inflow comparison (Blocked ANOVA when assumptions met, Friedman's test when assumptions not met)

Parameter	AD *p*	Levene *p*	Constant	Transformation	IQR Outlier Removal	Transformed AD *p*	Transformed Levene *p*	Test used
Peak discharge	< 0.0001	0.93	N/A	N/A	N/A	N/A	N/A	Friedman rank-sum
Total discharge	< 0.0001	0.003	N/A	N/A	N/A	N/A	N/A	Friedman rank-sum
pH	0.19	0.29	N/A	N/A	N/A	N/A	N/A	Blocked ANOVA
Conductivity	0.29	0.12	N/A	N/A	N/A	N/A	N/A	Blocked ANOVA
Suspended solids	0.19	0.1	N/A	N/A	N/A	N/A	N/A	Blocked ANOVA
DOC	0.76	0.03	N/A	N/A	N/A	N/A	N/A	Friedman rank-sum
Ammonia	0.79	0.01	N/A	Log10	N/A	0.23	0.52	Blocked ANOVA
Nitrate	0.27	0.57	N/A	N/A	N/A	N/A	N/A	Blocked ANOVA
Nitrite	0.02	0.07	N/A	N/A	N/A	N/A	N/A	Friedman rank-sum
Phosphate	0.01	0.46	N/A	Log10	N/A	0.32	0.07	Blocked ANOVA
Zinc	0.9	0.52	N/A	N/A	N/A	N/A	N/A	Blocked ANOVA
Iron	0.71	0.01	N/A	Log10	N/A	0.09	0.93	Blocked ANOVA
Chloride	0.81	0.06	N/A	N/A	N/A	N/A	N/A	Blocked ANOVA

Inflow-outflow comparison (t-tests)

Parameter	Site	Inflow AD *p*	Outflow AD *p*	Levene *p*	Transformation	Constant	Outlier Removal	Transformed Inflow AD *p*	Transformed Outflow AD *p*	Transformed Levene *p*	Test Used
Peak discharge	Site 1	< 0.0001	< 0.0001	< 0.0001	N/A	N/A	N/A	N/A	N/A	N/A	Wilcoxon Rank Sum
	Site 2	< 0.0001	< 0.0001	< 0.0001	N/A	N/A	N/A	N/A	N/A	N/A	Wilcoxon Rank Sum
Total discharge	Site 1	< 0.0001	< 0.0001	0.06	N/A	N/A	N/A	N/A	N/A	N/A	Wilcoxon Rank Sum
	Site 2	< 0.0001	< 0.0001	0.0001	N/A	N/A	N/A	N/A	N/A	N/A	Wilcoxon Rank Sum
pH	Site 1	0.3	0.44	0.27	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
	Site 2	0.02	0.48	0.26	N/A	N/A	1.0×	0.1	0.48	0.06	Student's *t*-test
	Site 3	0.04	0.09	0.78	Yeo-Johnson	N/A	N/A	0.06	0.18	0.89	Student's *t*-test
Conductivity	Site 1	0.046	0.22	0.04	N/A	N/A	N/A	N/A	N/A	N/A	Wilcoxon Rank Sum
	Site 2	0.79	0.64	0.96	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
	Site 3	0.08	0.03	0.91	Yeo-Johnson	N/A	N/A	0.15	0.08	0.79	Student's *t*-test
Turbidity	Site 1	0.0002	0.0004	0.04	Inverse	5 NTU	N/A	0.08	0.34	0.16	Student's *t*-test
	Site 2	0.0002	0.0003	0.04	Inverse	5 NTU	N/A	0.18	0.75	0.15	Student's *t*-test
	Site 3	< 0.0001	0.0001	0.63	Log	N/A	N/A	0.11	0.29	0.76	Student's *t*-test
Suspended Solids	Site 1	0.33	0.02	0.01	Square root	N/A	N/A	0.8	0.32	0.17	Student's *t*-test
	Site 2	0.7	0.003	0.02	Log	N/A	N/A	0.35	0.08	0.63	Student's *t*-test
	Site 3	0.58	0.03	0.048	Square root	N/A	N/A	0.91	0.09	0.18	Student's *t*-test
Dissolved Organic Carbon (DOC)	Site 1	0.68	0.25	0.11	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
	Site 2	0.29	0.44	0.03	Square root	N/A	N/A	0.27	0.39	0.09	Student's *t*-test

Parameter	Site	Inflow AD *p*	Outflow AD *p*	Levene *p*	Transformation	Constant	Outlier Removal	Transformed Inflow AD *p*	Transformed Outflow AD *p*	Transformed Levene *p*	Test Used
	Site 3	0.11	0.41	0.16	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
Ammonia	Site 1	0.4	0.52	0.31	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
	Site 2	0.57	0.98	0.35	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
	Site 3	0.4	0.13	0.78	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
Nitrate	Site 1	< 0.0001	0.04	0.54	Log	N/A	N/A	0.22	0.4	0.11	Student's *t*-test
	Site 2	0.02	0.19	0.46	Square root	N/A	N/A	0.22	0.44	0.48	Student's *t*-test
	Site 3	0.24	0.32	0.05	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
Nitrite	Site 1	< 0.0001	<0.0001	0.23	N/A	N/A	N/A	N/A	N/A	N/A	Mann-Whitney *U*-test
	Site 2	0.004	0.14	0.44	Square root	N/A	N/A	0.12	0.4	0.22	Student's *t*-test
	Site 3	0.03	0.001	0.51	Cube root	N/A	N/A	0.35	0.08	0.67	Student's *t*-test
Phosphate	Site 1	0.0001	0.0008	0.61	Log	0.005 mg/L	N/A	0.07	0.05	0.29	Student's *t*-test
	Site 2	< 0.0001	< 0.0001	0.67	Inverse	N/A	N/A	0.17	0.39	0.32	Student's *t*-test
	Site 3	0.25	0.0002	0.47	Log	N/A	N/A	0.37	0.06	0.77	Student's *t*-test
Zinc	Site 1	0.046	0.9	0.38	Square root	N/A	N/A	0.28	0.77	0.06	Student's *t*-test
	Site 2	0.13	0.76	0.9	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
	Site 3	0.33	0.85	0.76	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
Iron	Site 1	0.05	0.0003	0.39	Cube root	N/A	N/A	0.82	0.14	0.88	Student's *t*-test
	Site 2	0.19	0.22	0.46	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test
	Site 3	0.008	0.09	0.8	Square root	N/A	N/A	0.39	0.58	0.63	Student's *t*-test
Chloride	Site 1	0.04	0.14	0.4	Box-Cox	N/A	N/A	0.08	0.06	0.06	Student's *t*-test

Parameter	Site	Inflow AD *p*	Outflow AD *p*	Levene *p*	Transformation	Constant	Outlier Removal	Transformed Inflow AD *p*	Transformed Outflow AD *p*	Transformed Levene *p*	Test Used
	Site 2	0.02	0.24	0.83	Yeo-Johnson	5 mg/L	N/A	0.05	0.31	0.94	Student's *t*-test
	Site 3	0.64	0.63	0.96	N/A	N/A	N/A	N/A	N/A	N/A	Student's *t*-test

Reductions - One-way ANOVA

Parameter	AD *p*	Levene *p*	Transformation	Constant	Outlier Removal	Transformed AD *p*	Transformed Levene *p*	Test Used
pH	0.04	0.37	Yeo-Johnson	N/A	N/A	0.26	0.53	ANOVA
Conductivity	0.07	0.0002	Cube root	N/A	1.5×	0.28	0.65	ANOVA
Turbidity	< 0.0001	0.41	Yeo-Johnson	N/A	N/A	0.14	0.27	ANOVA
Suspended Solids	0.1	0.55	N/A	N/A	N/A	N/A	N/A	ANOVA
DOC	0.16	0.44	N/A	N/A	N/A	N/A	N/A	ANOVA
Ammonia	0.08	0.03	Yeo-Johnson	N/A	N/A	0.06	0.19	ANOVA
Nitrate	< 0.0001	0.2	Yeo Johnson	N/A	1.5×	0.3	0.23	ANOVA
Nitrite	< 0.0001	0.58	Cube root	N/A	N/A	0.15	0.53	ANOVA
Phosphate	0.002	0.0003	Cube root	N/A	N/A	0.12	0.12	ANOVA
Zinc	< 0.0001	0.09	Yeo-Johnson	N/A	1.5×	0.31	0.26	ANOVA
Iron	< 0.0001	0.51	Cube root	N/A	1.5×	0.27	0.96	ANOVA
Chloride	< 0.0001	0.14	Cube root	N/A	N/A	0.84	0.69	ANOVA

Reference vs. Site 1 Inflow vs. Outflow quantity comparisons (Linear Mixed Effects Model)

For all parameters, the linearity of the fixed effects assumption was not applicable because the fixed effect (Site) is categorical. Non-independence among repeated sampling dates was accounted for by including the sampling date as a random intercept in the models. Multicollinearity was not an issue, as the model included only a single fixed effect (Site). Normality of residuals (Anderson-Darling test (AD), homogeneity of variances (Levene test) and the presence of autocorrelation (Durbin-Watson test (DW)) were assessed for each parameter. When assumptions were not met, an aligned-rank transformed linear mixed effects (LME) was used.

Parameter	AD *p*	Levene *p*	DW *p*	Transformation	Constant	Transformed AD *p*	Transformed Levene *p*	Transformed DW *p*	Test Used
Peak Discharge	< 0.0001	0.06	0.57	N/A	N/A	N/A	N/A	N/A	Aligned-rank transformed LME
Total Discharge	< 0.0001	0.27	0.55	N/A	N/A	N/A	N/A	N/A	Aligned-rank transformed LME

Reference vs. Site 1 Inflow vs. Outflow quantity comparisons (Repeated Measures ANOVA)

As repeated measurements from the reference inflow, Inflow 1a and Outflow 1 were all collected on separate dates, and measurements from separate dates were independent of each other. Normality of residuals was assessed using the Anderson-Darling test, and sphericity was assessed using Mauchley's test for each parameter. The presence of extreme outliers was assessed by identifying values greater than 3× the interquartile range (IQR). When extreme outliers were detected, the sensitivity to the outlier was assessed by running the ANOVA both with and without the outliers, if both runs yielded the same p-value, it was determined that the outliers did not affect the model, results and the outlier was retained to maintain a balanced design. If the runs yielded different p-values, the outlier for that date was removed along with the observations from the other two sites to maintain a balanced design.

Parameter	AD *p*	Mauchly *p*	Extreme Outliers	Transformation	Outlier Removal	Constant	Extreme Outliers Transformed	Sensitivity Pass	Transformed AD *p*	Transformed Mauchley *p*	Test Used
pH	0.06	0.06	1	N/A	N/A	N/A	1	Yes, both *p* = 0.026	N/A	N/A	RM ANOVA
Conductivity	0.65	0.07	0	N/A	N/A	N/A	0	N/A	N/A	N/A	RM ANOVA
Turbidity	< 0.0001	< 0.0001	0	Box-Cox	N/A	0.1 NTU	0	N/A	0.39	0.6	RM ANOVA
Suspended Solids	0.4	0.46	0	N/A	N/A	N/A	N/A	N/A	N/A	N/A	RM ANOVA
Dissolved Organic Carbon (DOC)	0.23	0.002	1	N/A	N/A	N/A	1	Yes, both *p* = 0.01	N/A	N/A, GG-correction applied	RM-ANOVA
Ammonia	0.03	0.33	0	N/A	N/A	N/A	N/A	N/A	N/A	N/A	RM ANOVA
Nitrate	< 0.0001	0.19	3	Inverse	N/A	0.05 mg/L	2	Yes, both *p* = 0.2	0.9	0.14	RM-ANOVA
Nitrite	< 0.001	< 0.001	0	N/A	N/A	N/A	N/A	N/A	N/A	N/A	Freidman Test
Phosphate	0.001	< 0.001	0	Cube root	N/A	0.01 mg/L	0	N/A	0.48	0.04, GG-correction applied	RM ANOVA
Zinc	0.001	0.16	1	Log10	N/A	N/A	0	N/A	0.52	0.25	RM ANOVA
Iron	< 0.0001	0.005	1	Log10	N/A	N/A	0	N/A	0.52	0.25	RM ANOVA
Chloride	< 0.0001	0.005	2	Inverse	N/A	0.1 mg/L	2	Yes, both *p* = 0.31	0.23	0.36	RM-ANOVA

Upstream-downstream quality comparison (Paired *t*-tests)

As measurements collected upstream and downstream of the reference outfall were all collected on separate dates, each pair (upstream and downstream) was independent. Normality of paired differences (i.e. upstream measurement - downstream measurement) was assessed using the Anderson-Darling test, and extreme outliers in the paired differences were identified using the interquartile range (IQR) outlier removals. When paired differences met the assumption of normality and contained no extreme outliers, paired t-tests were used. When assumptions were not met following transformations, a Wilcoxon signed-rank test was used to compare water quality parameters upstream and downstream of the reference outfall.

	Site	Normality paired Differences (AD *p*)	Extreme Outliers (> 3× IQR)	Transformation	Outlier Removal	Constant	Transformed Normality of Paired Differences (AD *p*)	Transformed Outliers (> 3× IQR)	Test Used
pH	Reference	< 0.0001	2	N/A	N/A	N/A	N/A	2	Wilcoxon signed-rank
	Site 1	0.02	1	N/A	N/A	N/A	N/A	1	Wilcoxon signed-rank
Conductivity	Reference	0.18	0	N/A	N/A	N/A	N/A	N/A	Paired *t*
	Site 1	< 0.0001	2	Log10	1.5× IQR	1e-12	0.21	0	Paired *t*
Turbidity	Reference	< 0.001	2	Square root	N/A	N/A	0.71	0	Paired *t*
	Site 1	0.003	2	Square root	1× IQR	4.72	0.89	0	Paired *t*
Suspended Solids	Reference	0.27	0	N/A	N/A	N/A	N/A	N/A	Paired *t*
	Site 1	< 0.0001	2	N/A	N/A	N/A	N/A	N/A	Wilcoxon signed-rank
DOC	Reference	0.09	2	Inverse	3× IQR	N/A	0.17	0	Paired *t*
	Site 1	< 0.0001	2	N/A	0.5× IQR	N/A	0.57	0	Paired *t*
Ammonia	Reference	0.0003	3	Yeo-Johnson	3× IQR	N/A	0.35	0	Paired *t*
	Site 1	0.01	2	Log10	N/A	0.028	0.25	0	Paired *t*
Nitrate	Reference	0.42	0	N/A	N/A	N/A	N/A	N/A	Paired *t*
	Site 1	0.03	3	Log10	1× IQR	N/A	0.06	0	Paired *t*
Nitrite	Reference	0.001	4	Box-Cox	3× IQR	N/A	0.24	0	Paired *t*
	Site 1	< 0.0001	2	Log10	1× IQR	0.0002	0.08	0	Paired *t*
Phosphate	Reference	0.0002	4	Inverse	3× IQR	0.021	0.16	0	Paired *t*
	Site 1	0.0001	3	N/A	N/A	N/A	N/A	N/A	Wilcoxon signed-rank
Zinc	Reference	0.0002	1	Inverse	3× IQR	N/A	0.54	0	Paired *t*
	Site 1	< 0.0001	2	Yeo-Johnson	1× IQR	N/A	0.31	0	Paired *t*
Iron	Reference	0.0002	2	Inverse	3× IQR	0.107	0.21	0	Paired *t*
	Site 1	< 0.0001	4	Log10	1.5× IQR	0.016	0.38	0	Paired *t*
Chloride	Reference	0.03	3	Inverse	3× IQR	0.332	0.3	0	Paired *t*
	Site 1	0.002	3	Square root	3× IQR	0.654	0.94	0	Paired *t*

Figures for Snow Core Quality Measurements Over Time

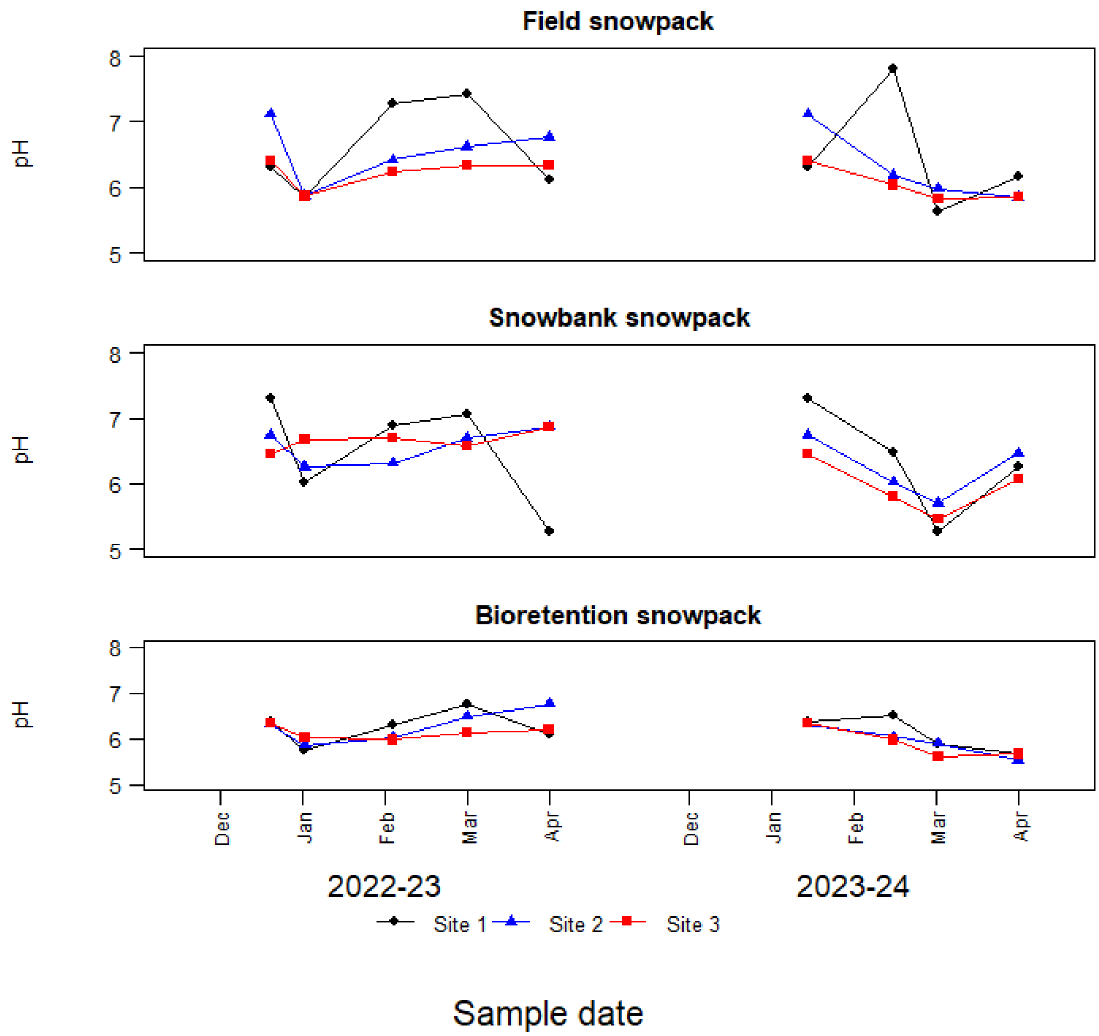


Figure S1: Comparison of pH of snow from open fields, roadside snowbanks and the surface of bioretention systems.

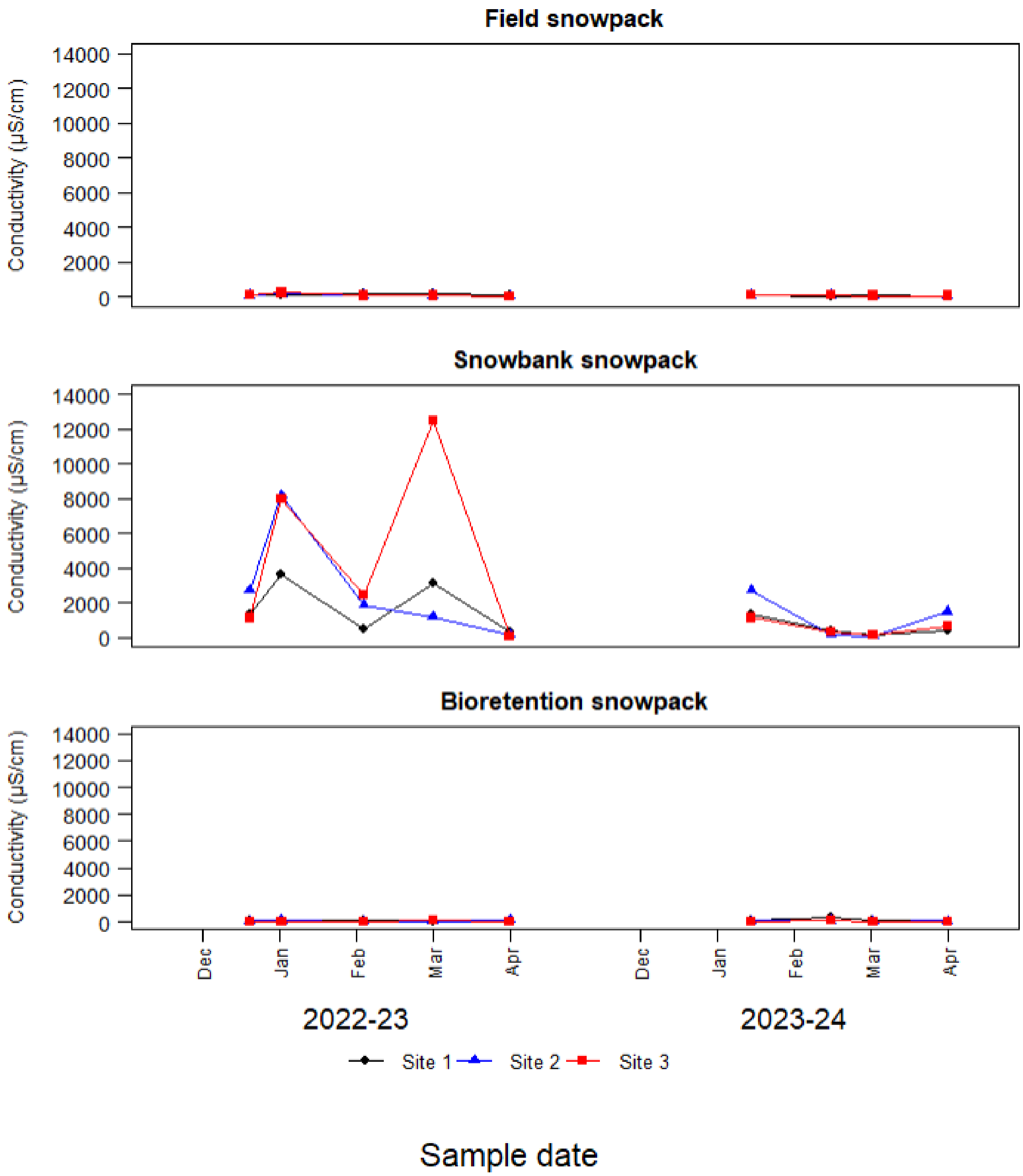


Figure S2: Comparison of conductivity of snow from open fields, roadside snowbanks, and the surface of bioretention systems.

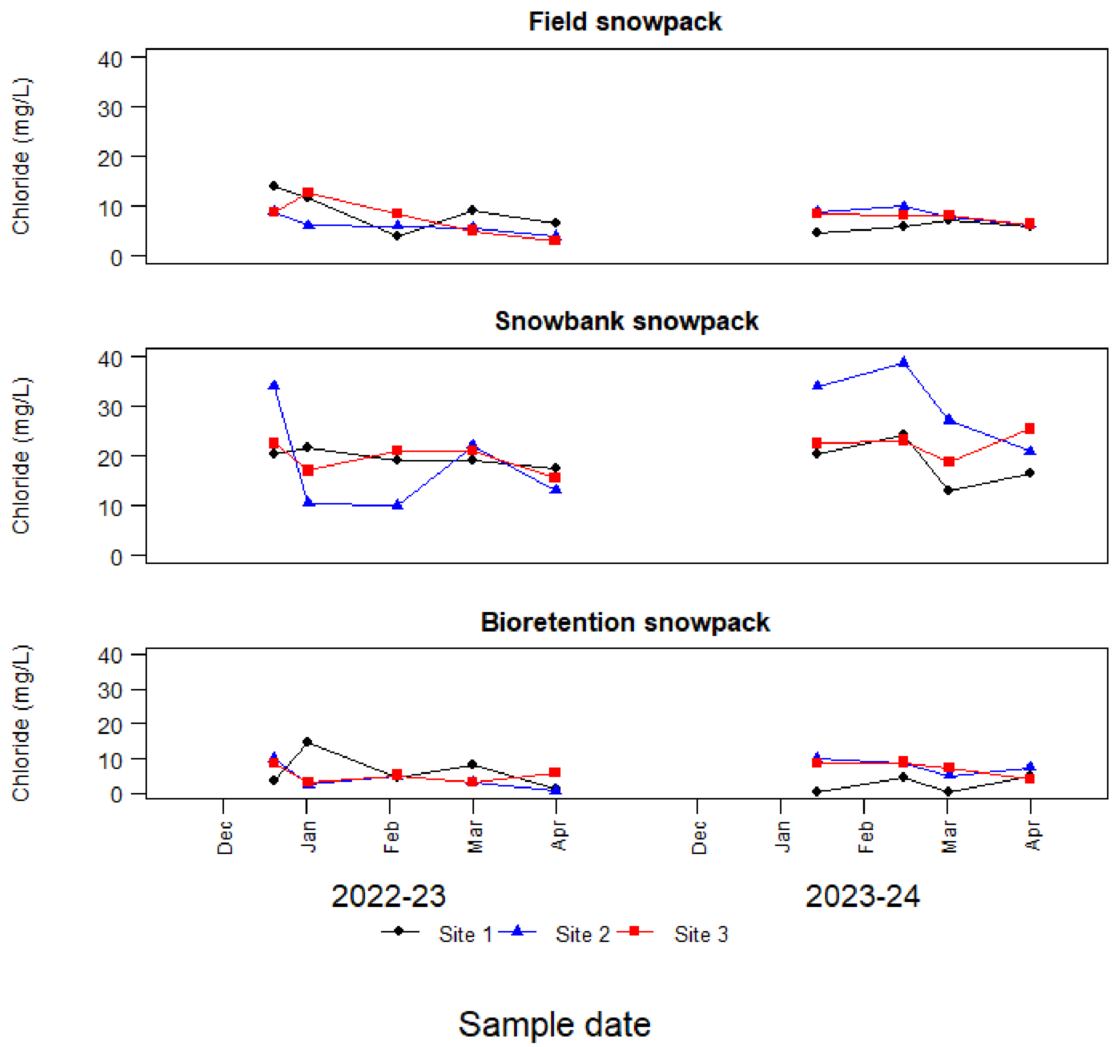


Figure S3: Comparison of chloride concentrations of snow from open fields, roadside snowbanks and the surface of bioretention systems.

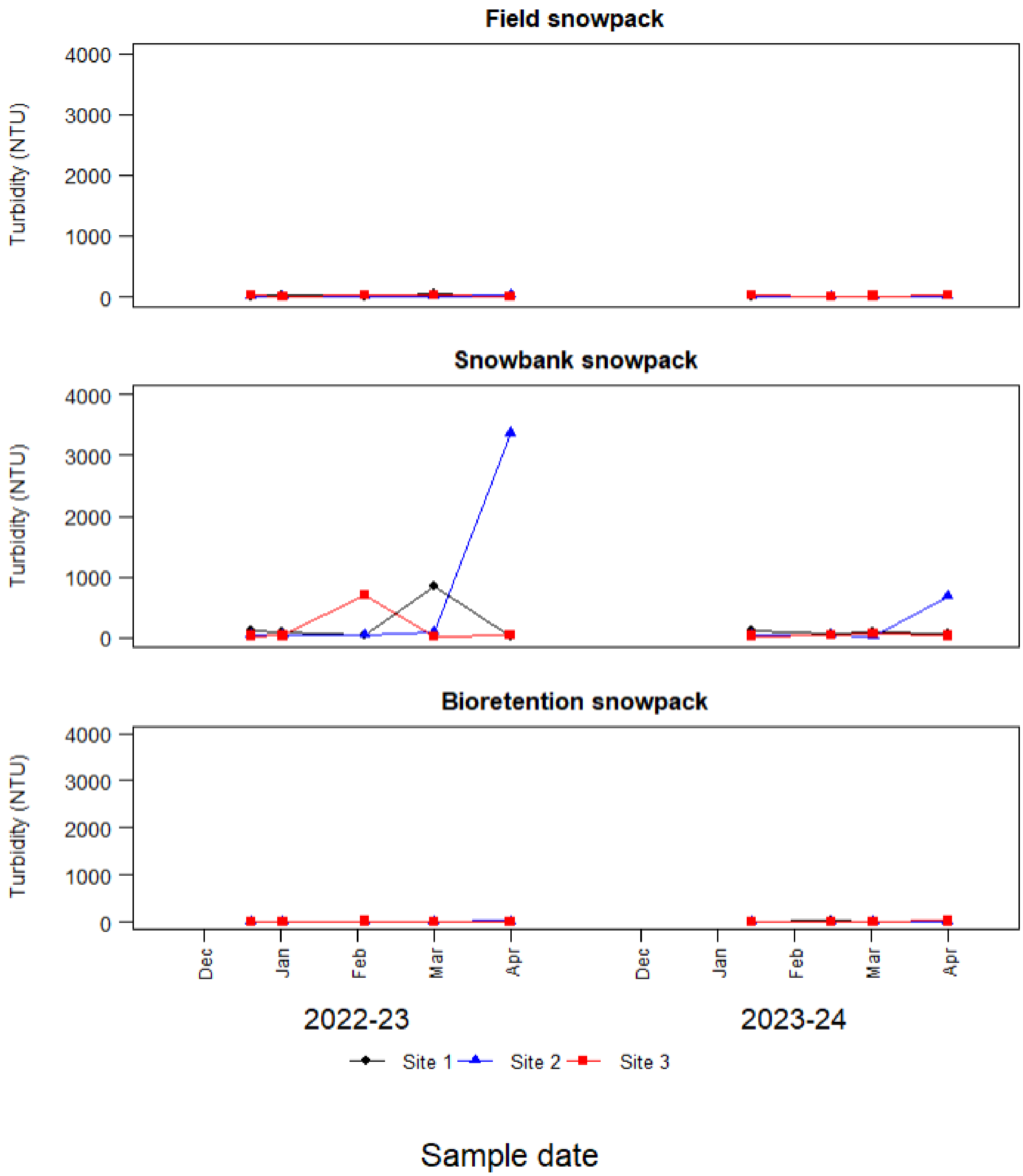


Figure S4: Comparison of turbidity of the snow from open fields, roadside snowbanks and the surface of bioretention systems.

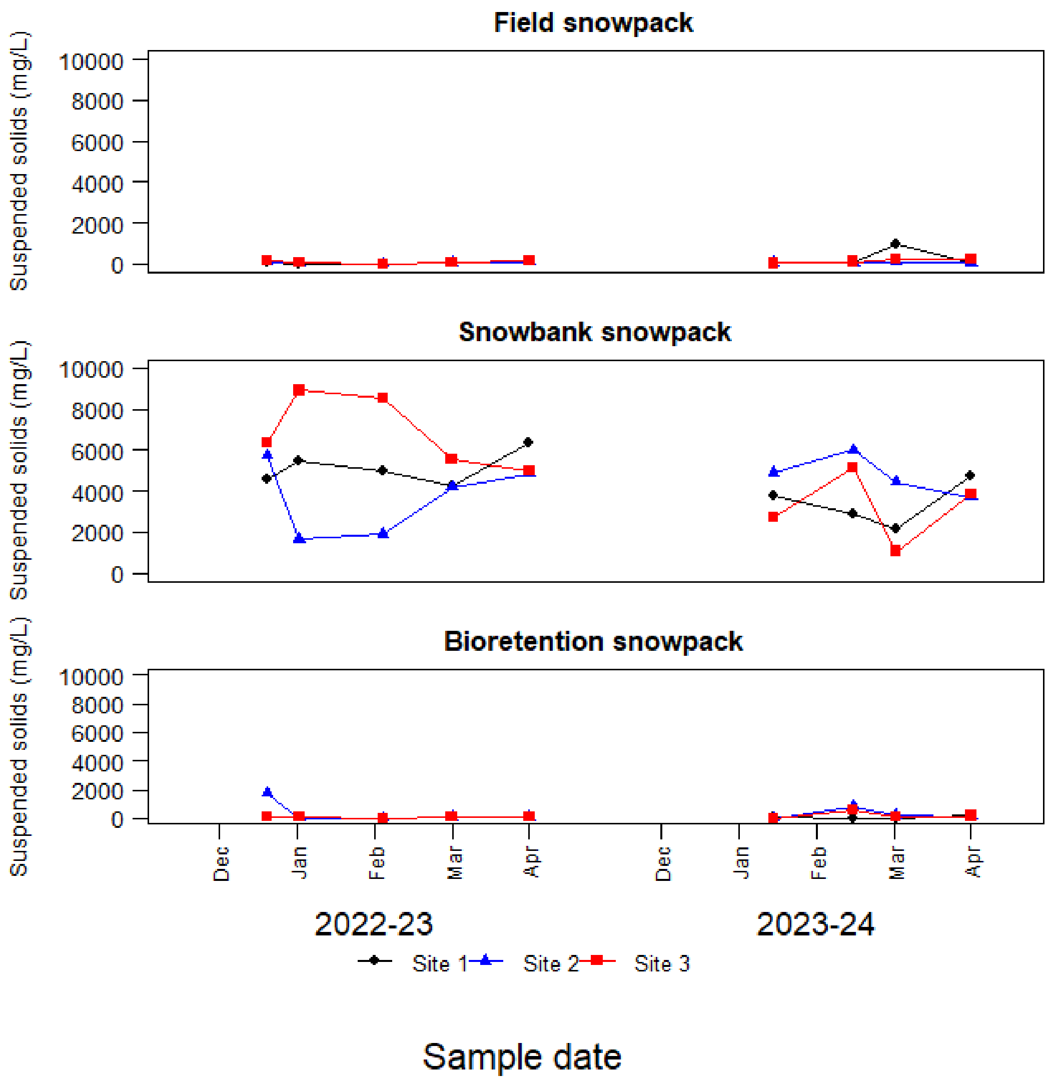


Figure S5: Comparison of suspended solids concentrations of the snow from open fields, roadside snowbanks and the surface of bioretention systems.

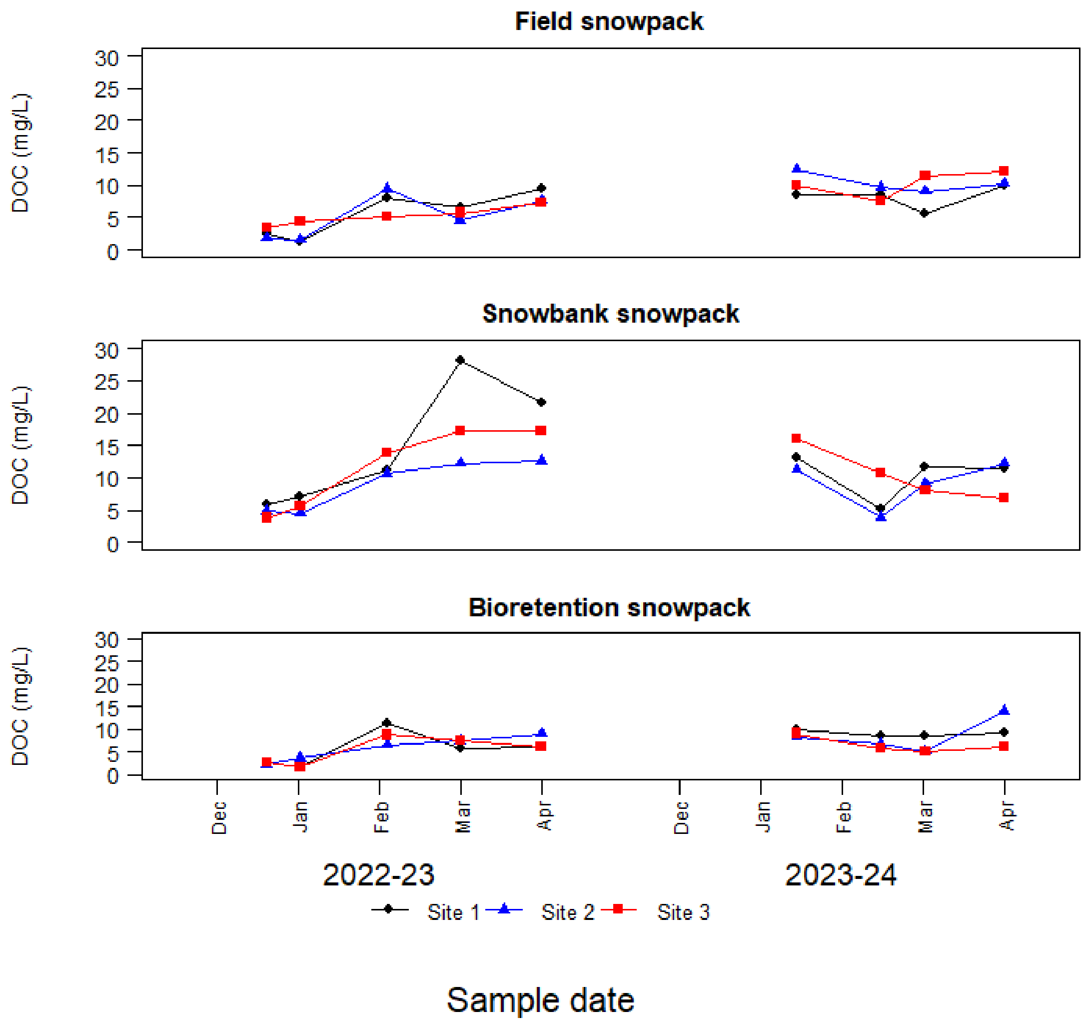


Figure S6: Comparison of dissolved organic carbon (DOC) concentrations of the snow from open fields, roadside snowbanks, and the surface of bioretention systems.

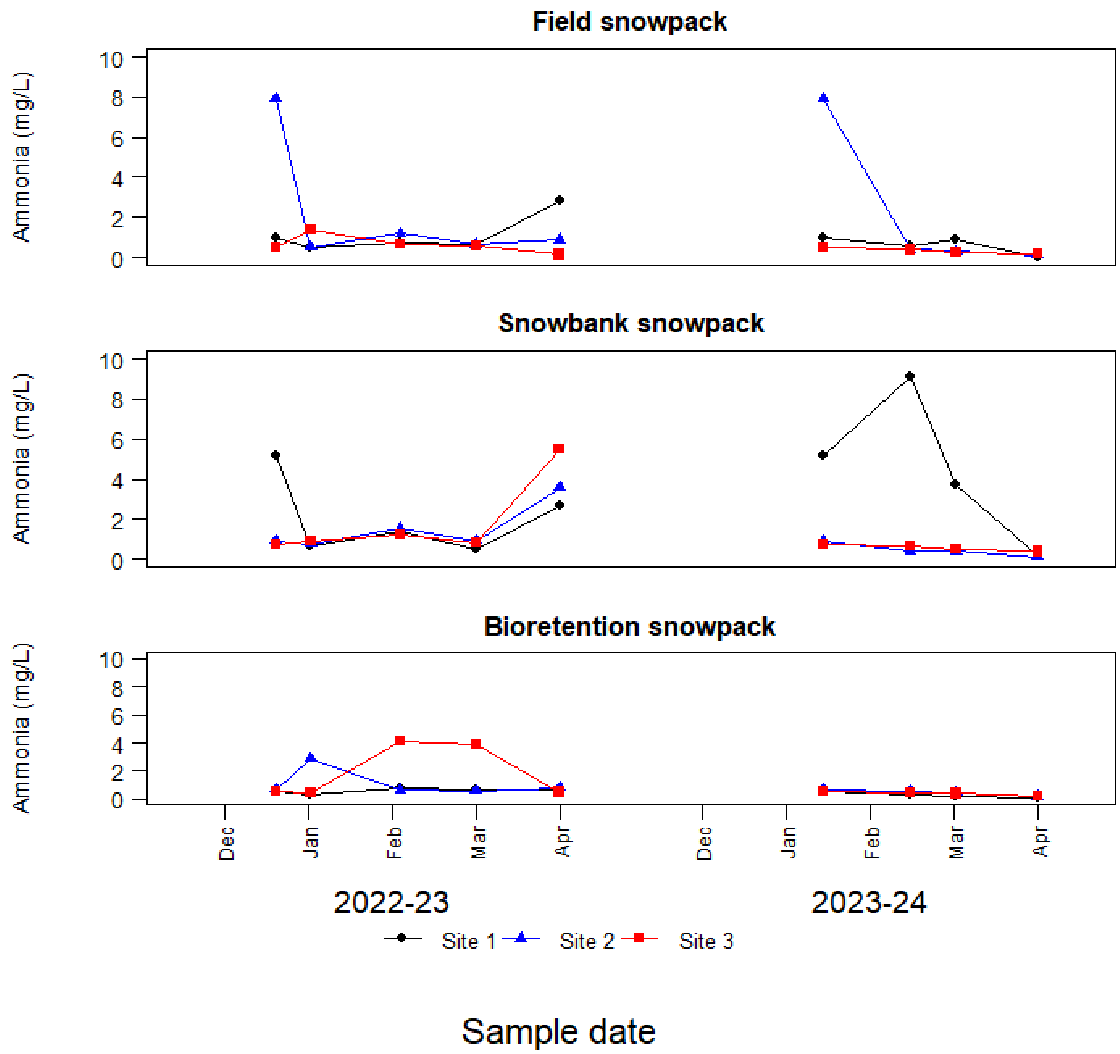


Figure S7: Comparison of ammonia concentrations of the snow from open fields, roadside snowbanks, and the surface of bioretention systems.

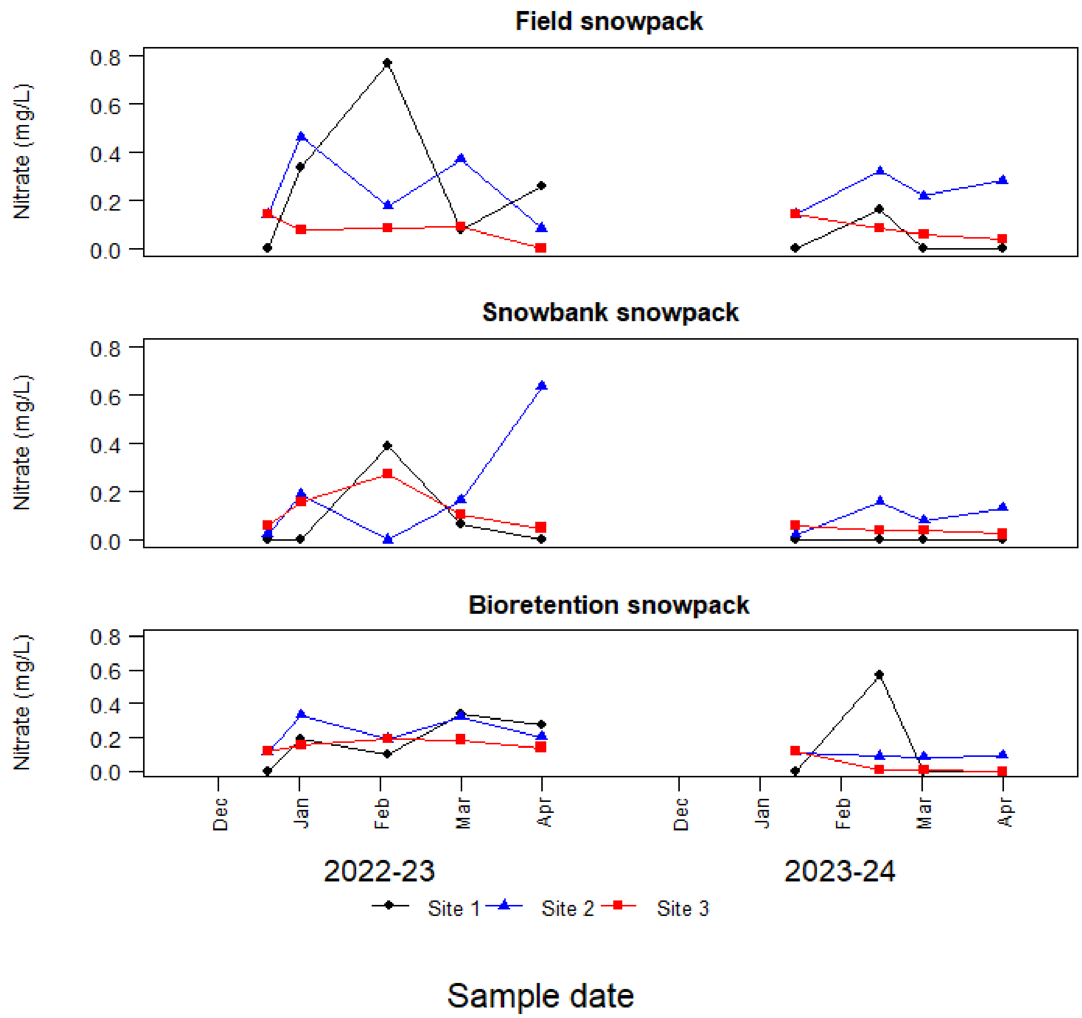


Figure S8: Comparison of nitrate concentrations of the snow from open fields, roadside snowbanks, and the surface of bioretention systems.

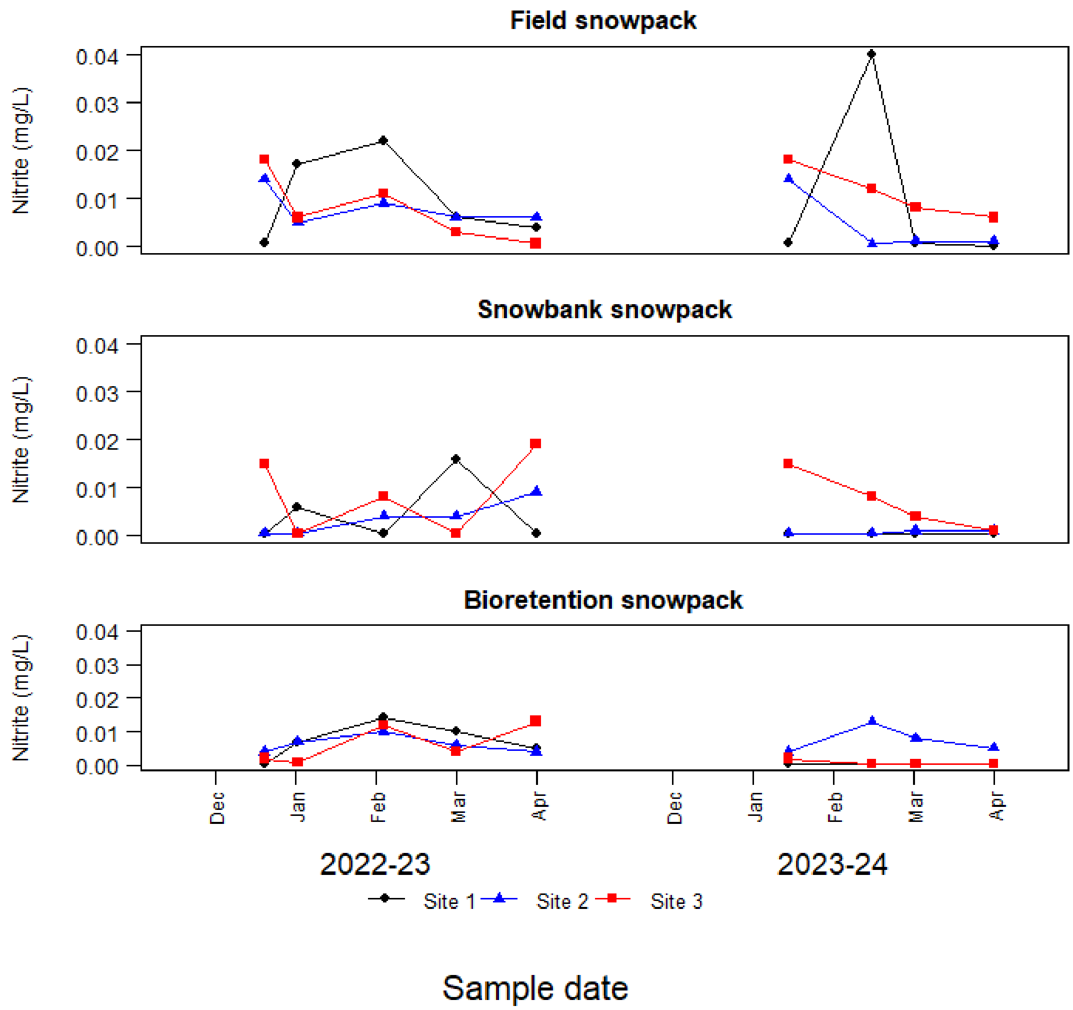


Figure S9: Comparison of nitrite concentrations from open fields, roadside snowbanks, and the surface of bioretention systems.

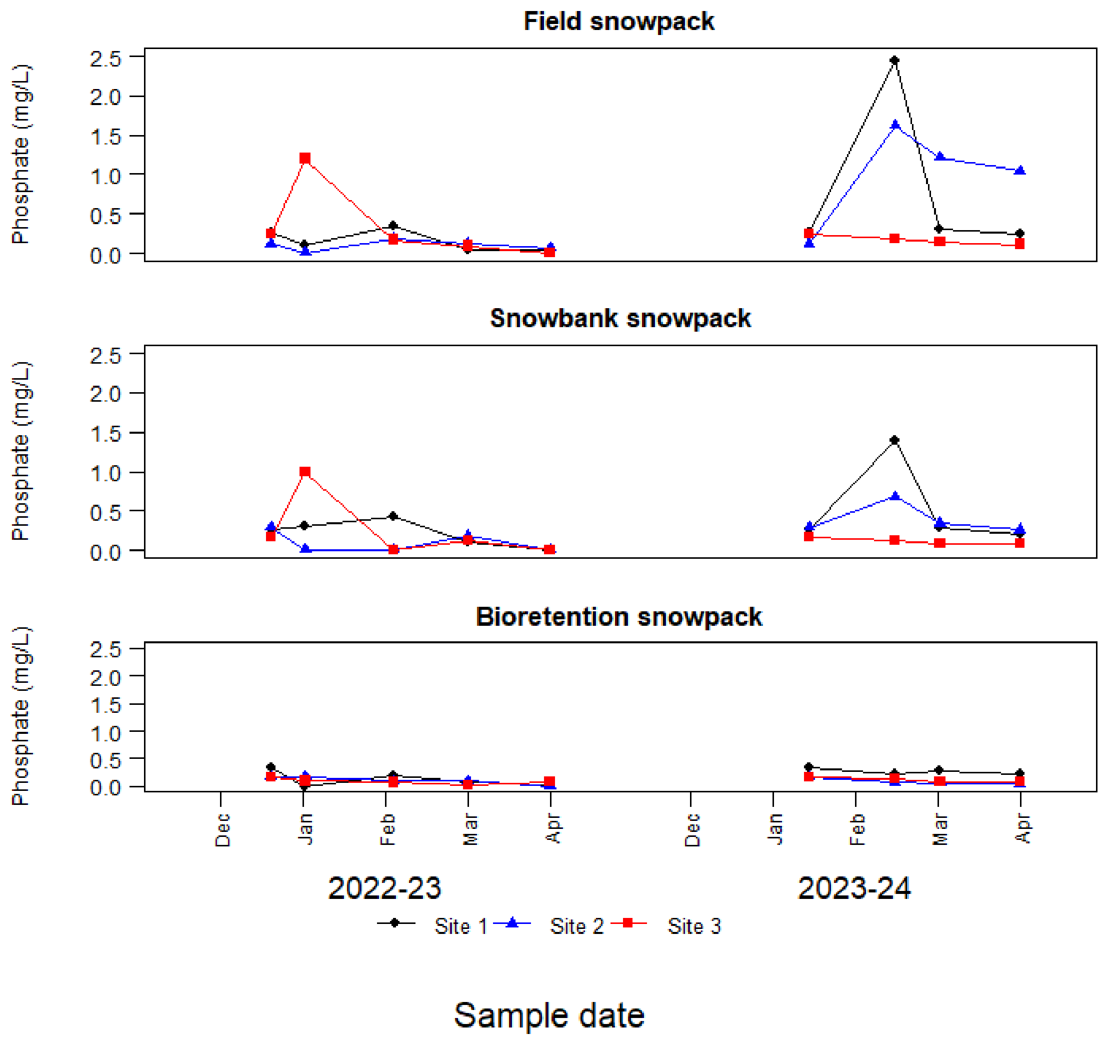


Figure S10: Comparison of phosphate concentrations from open fields, roadside snowbanks, and the surface of bioretention systems.

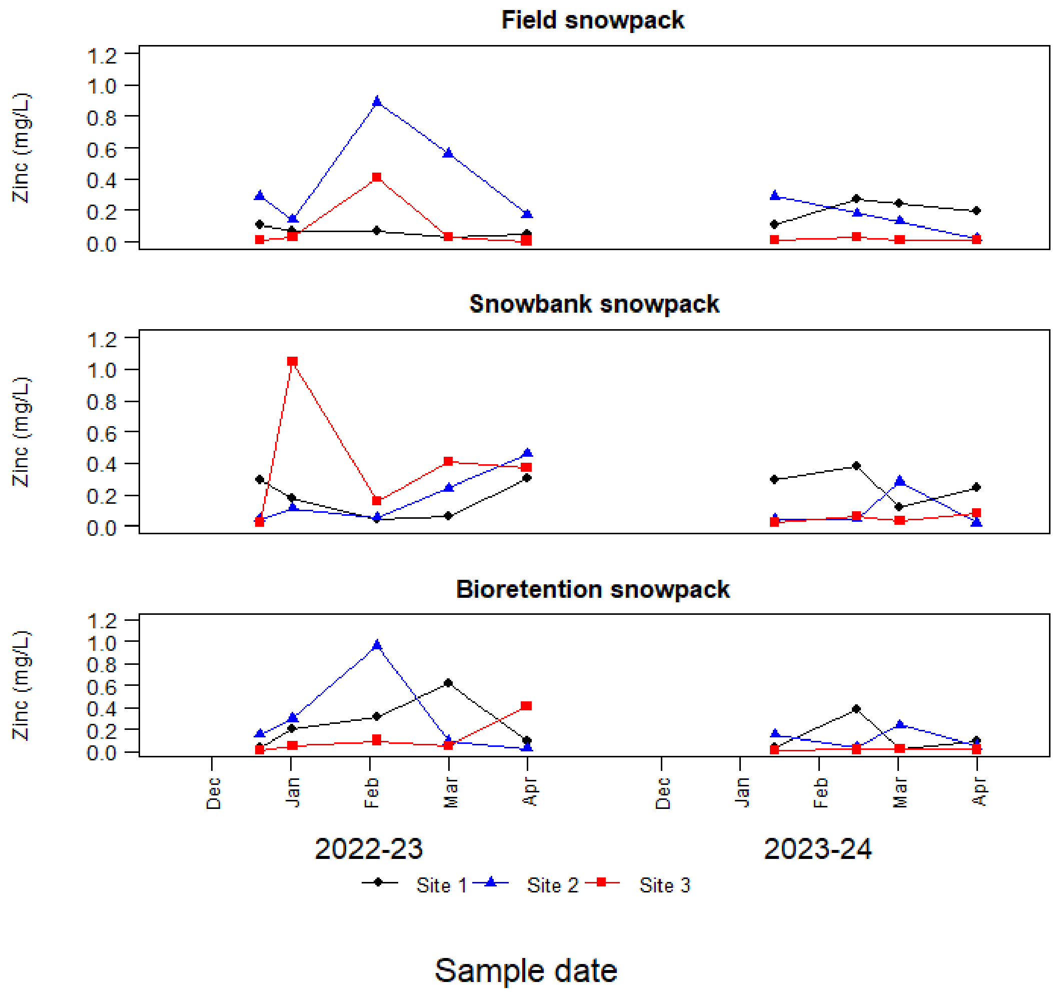


Figure S10: Comparison of zinc concentrations from open fields, roadside snowbanks, and the surface of bioretention systems.

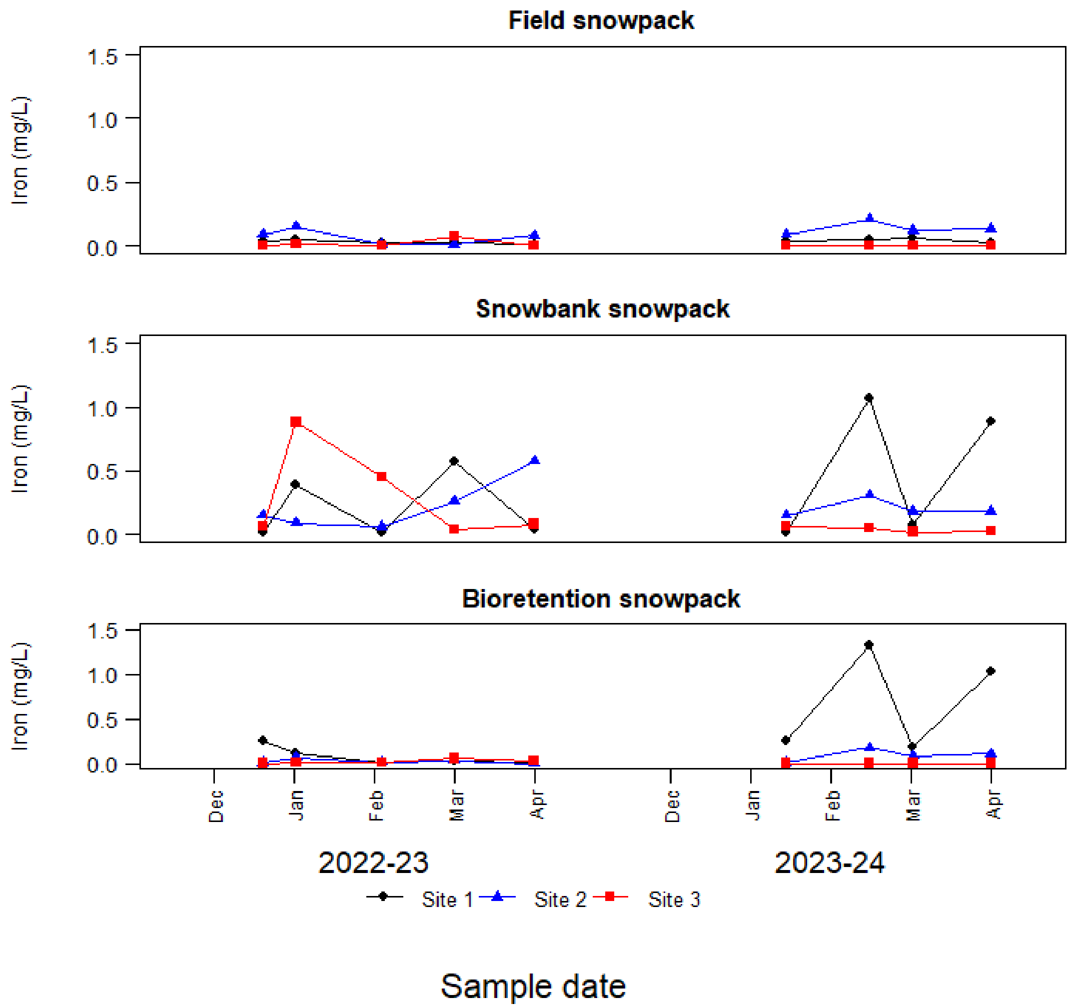


Figure S11: Comparison of iron concentrations from open fields, roadside snowbanks, and the surface of bioretention systems.

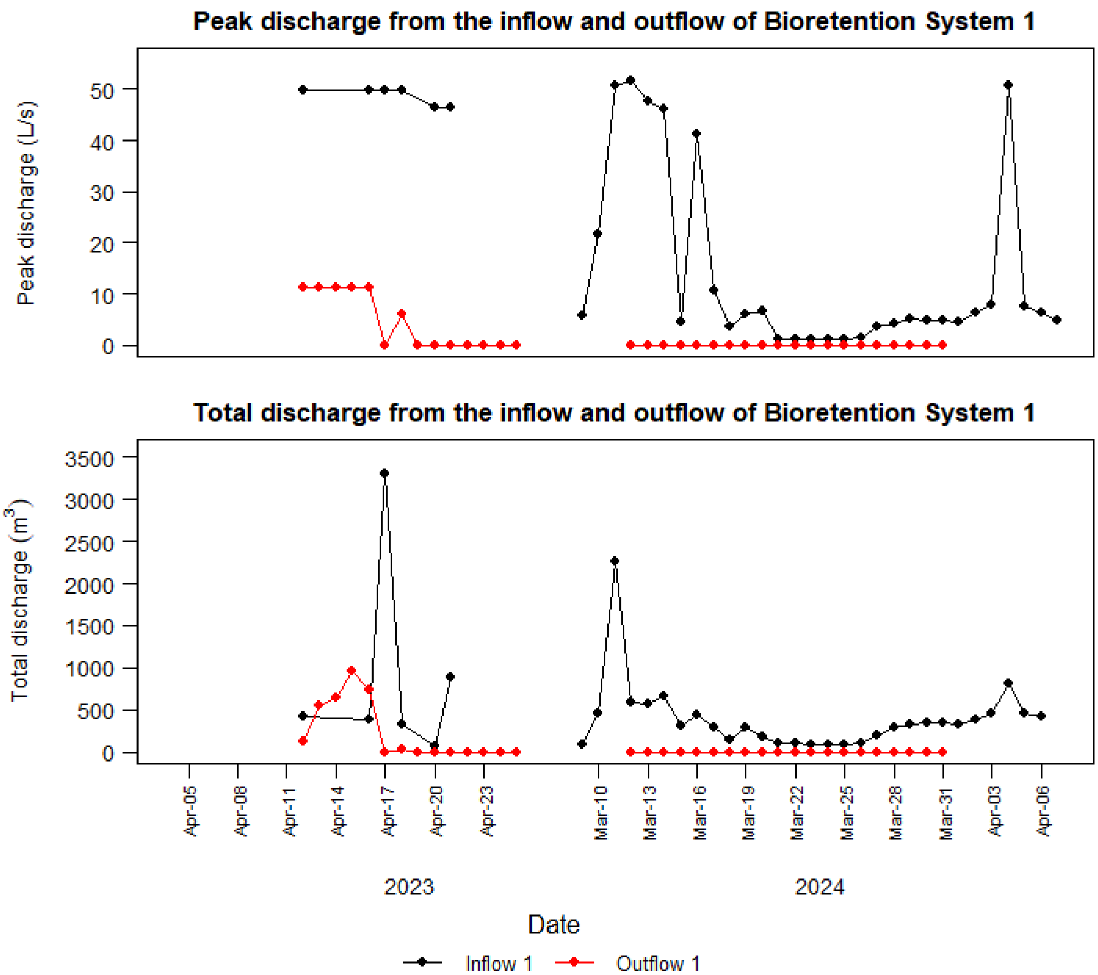


Figure S12 Peak and total discharge from the inflow (Inflow 1) and outflow (Outflow 1) of Bioretention System 1 during spring melt events between March 2023 and April 2024.

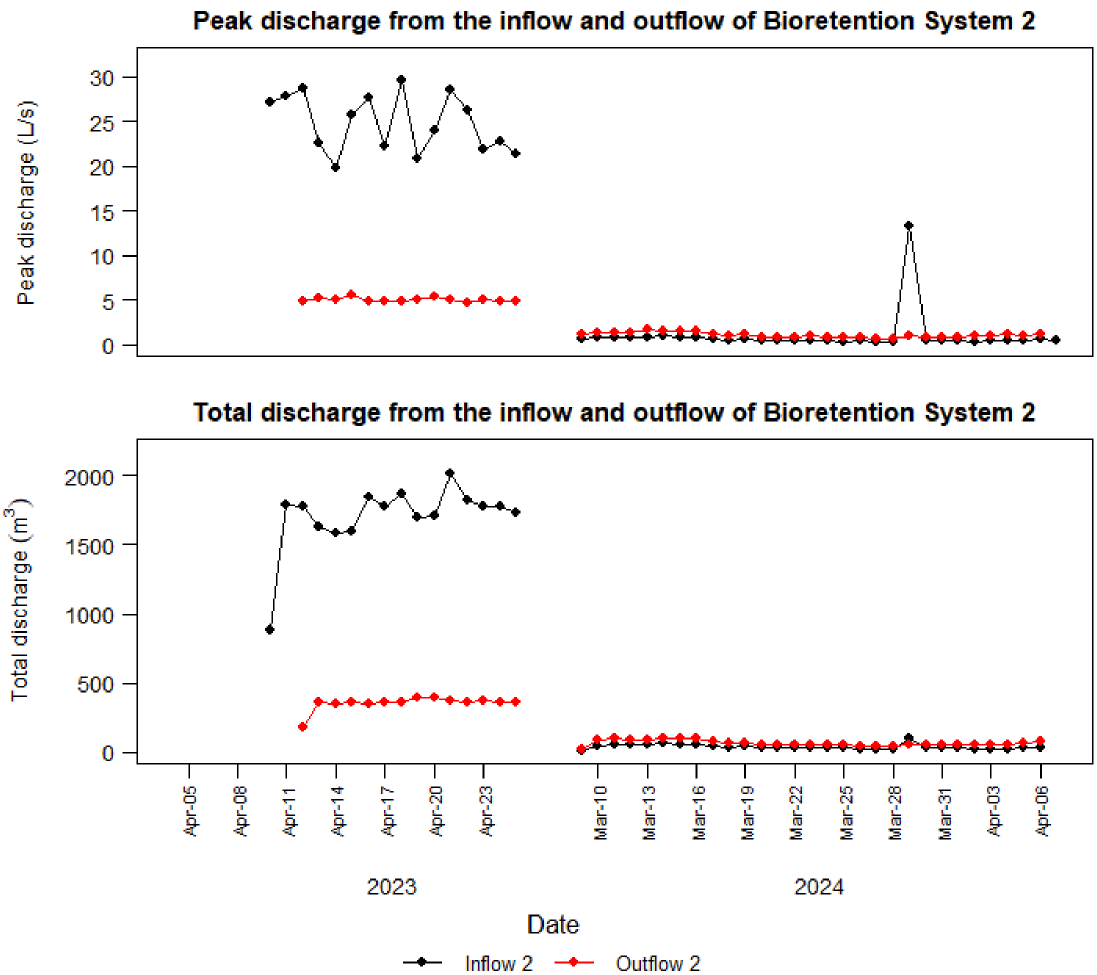


Figure S13 Peak and total discharge from the inflow (Inflow 2) and outflow (Outflow 2) of Bioretention System 2 during spring melt events between March 2023 and April 2024.

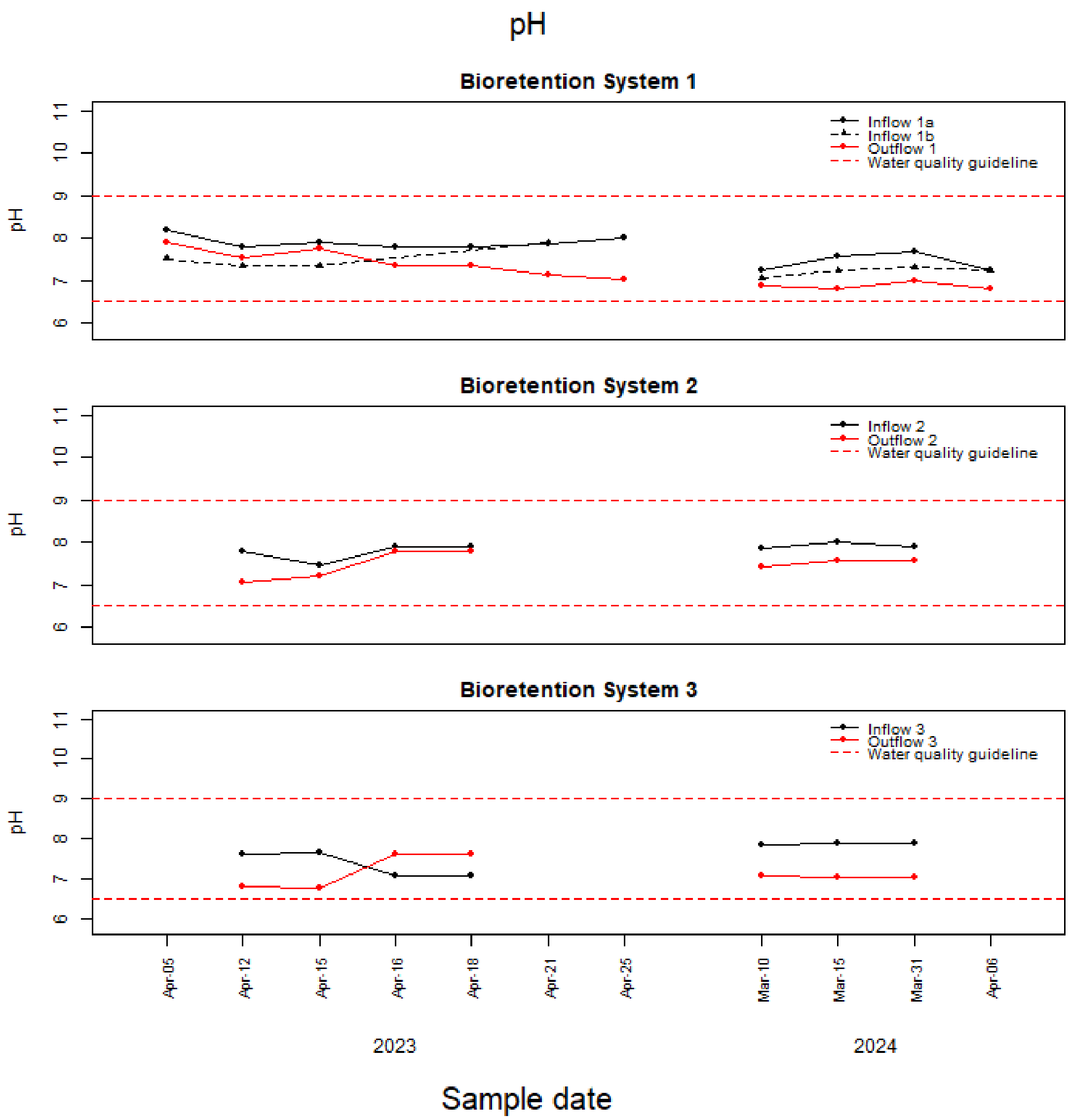


Figure S14 pH from the inflow(s) and outflow of the three bioretention systems during the spring melt. The red-dashed lines show the lower (6.5) and upper (9.0) CCME water quality guideline for the protection of aquatic life.

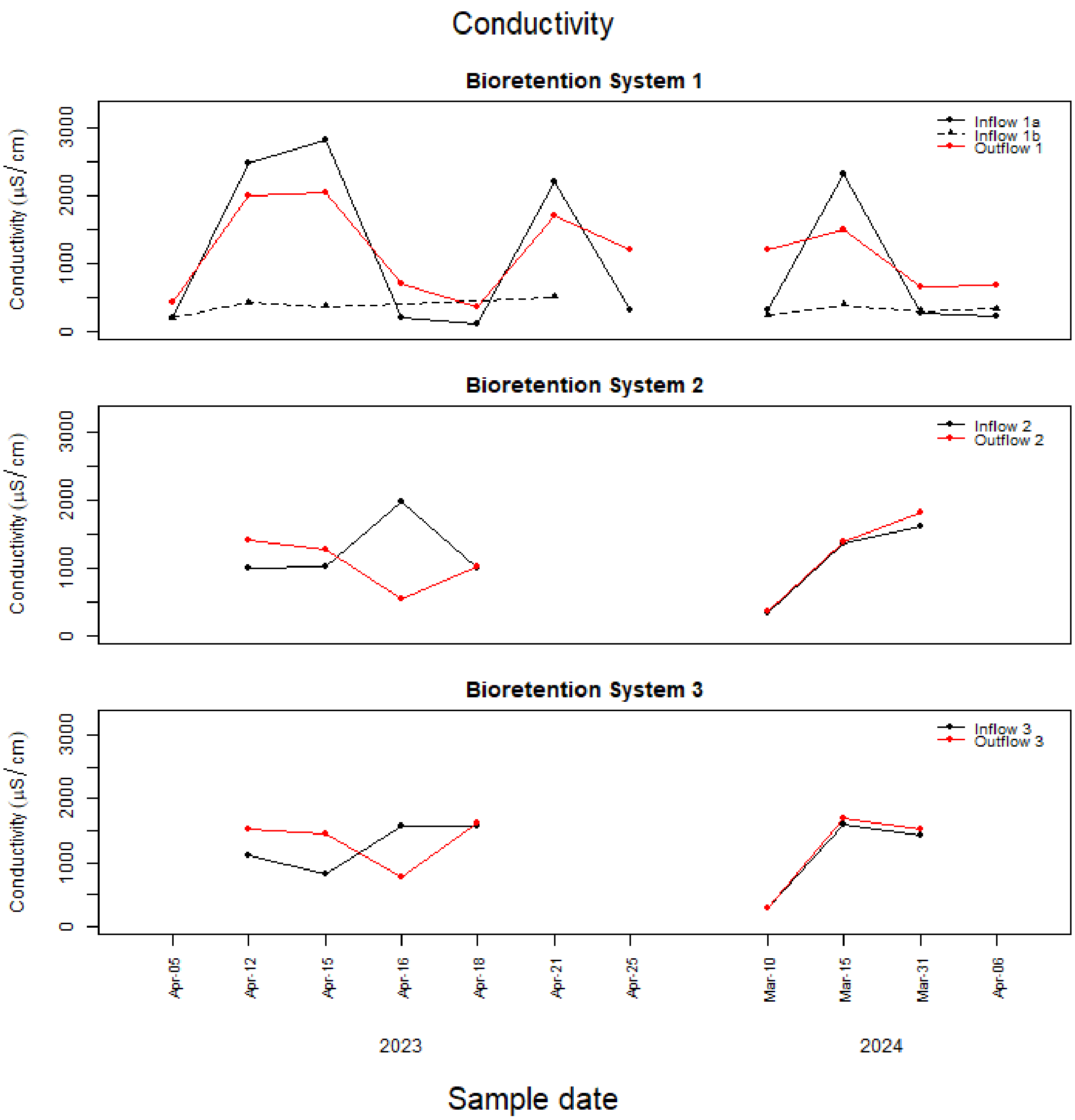


Figure S16 Conductivity from the inflow(s) and outflow of the three bioretention systems during the spring melt.

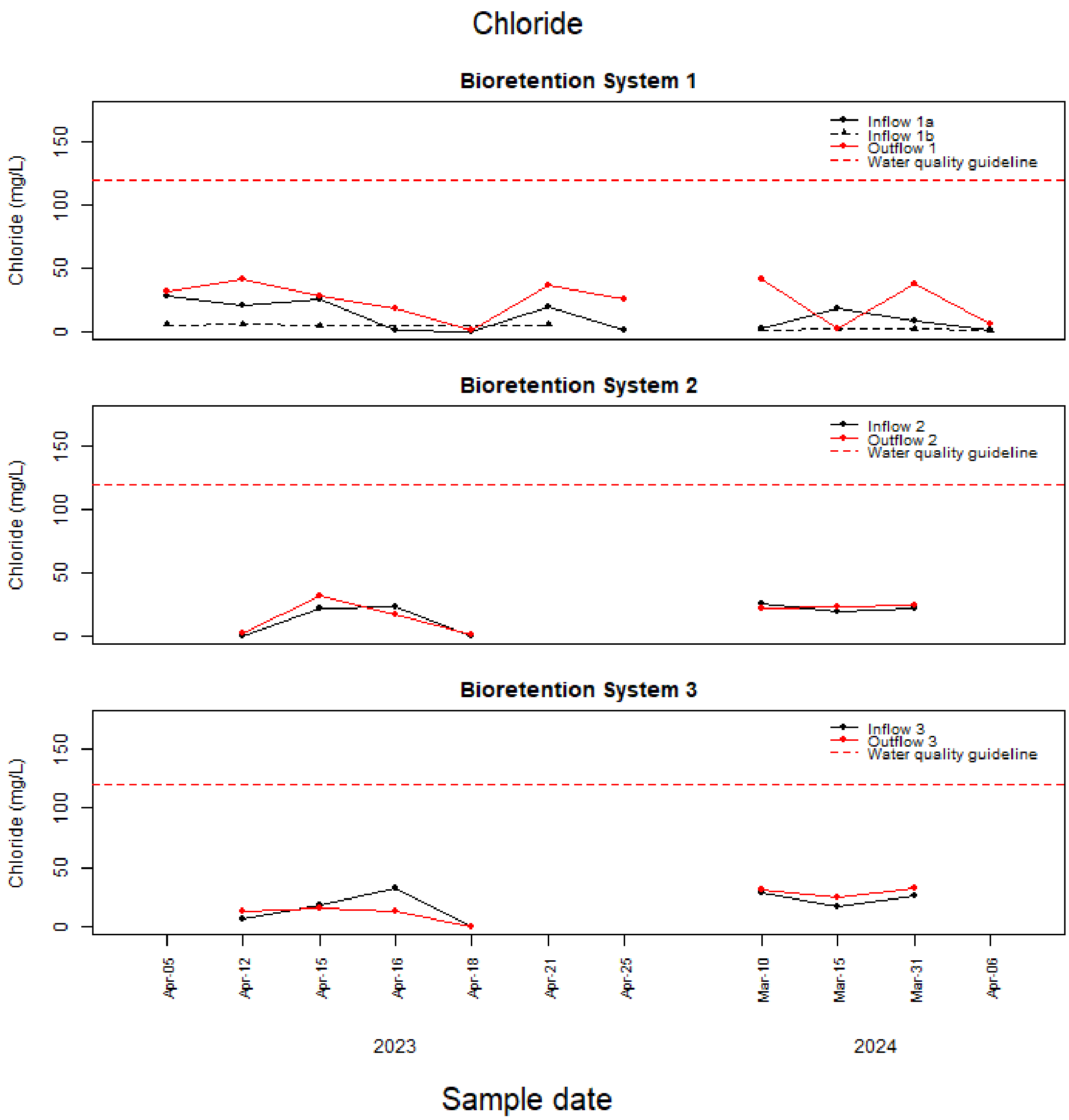


Figure S17 Chloride concentrations from the inflow(s) and outflow of the three bioretention systems during the spring melt. The red-dashed lines show the 120 mg/L CCME water quality guideline for the protection of aquatic life.

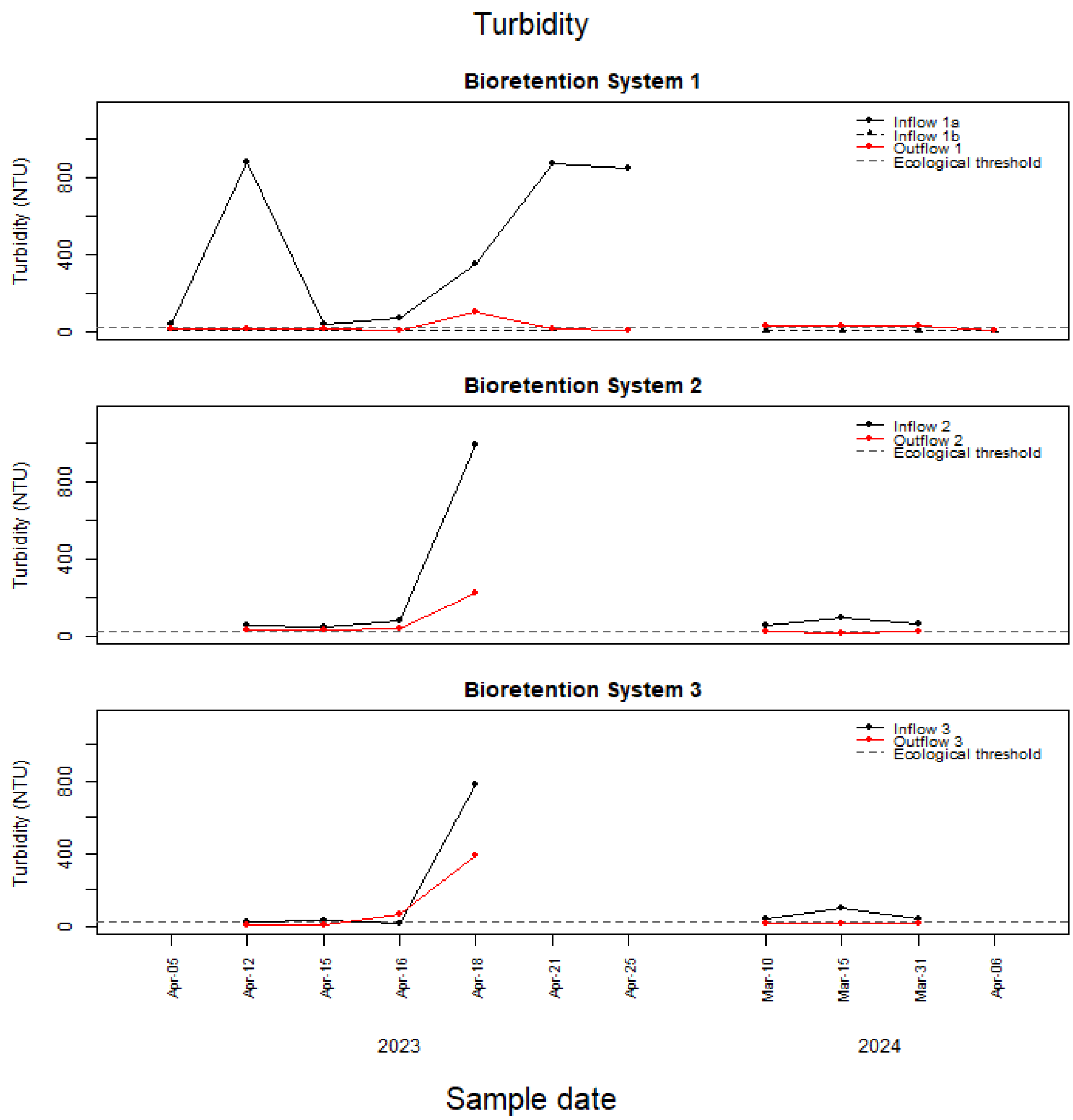


Figure S18 Turbidity from the inflow(s) and outflow of the three bioretention systems during the spring melt. The dashed line shows the 25 NTU literature-derived benchmark associated with reduced growth, fish density and behavioral effects in salmonids.

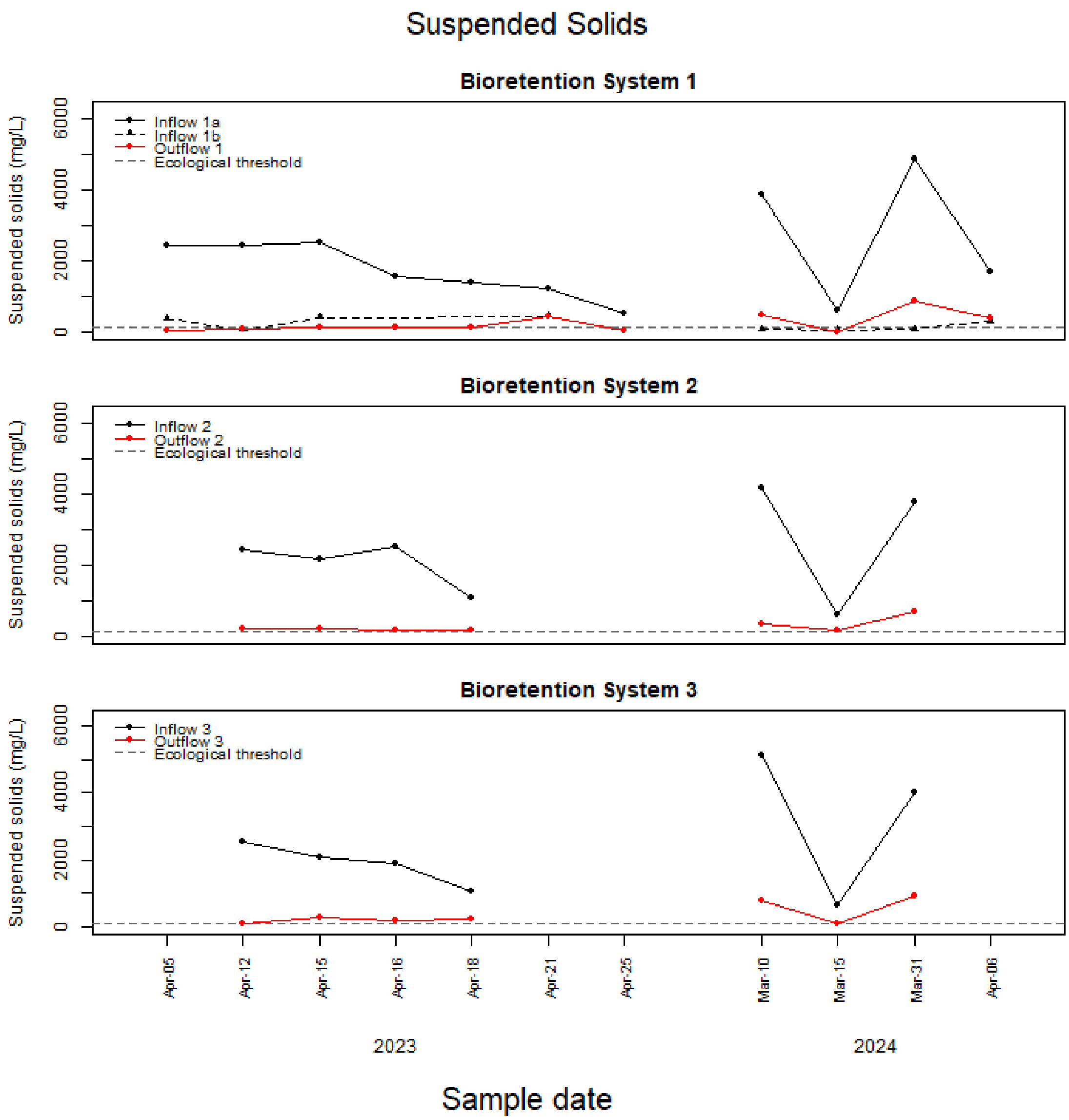


Figure S19 Suspended solids concentrations from the inflow(s) and outflow of the three bioretention systems during the spring melt. The dashed line shows the 100 mg/L literature-derived benchmark associated with sublethal effects in adult and juvenile fish.

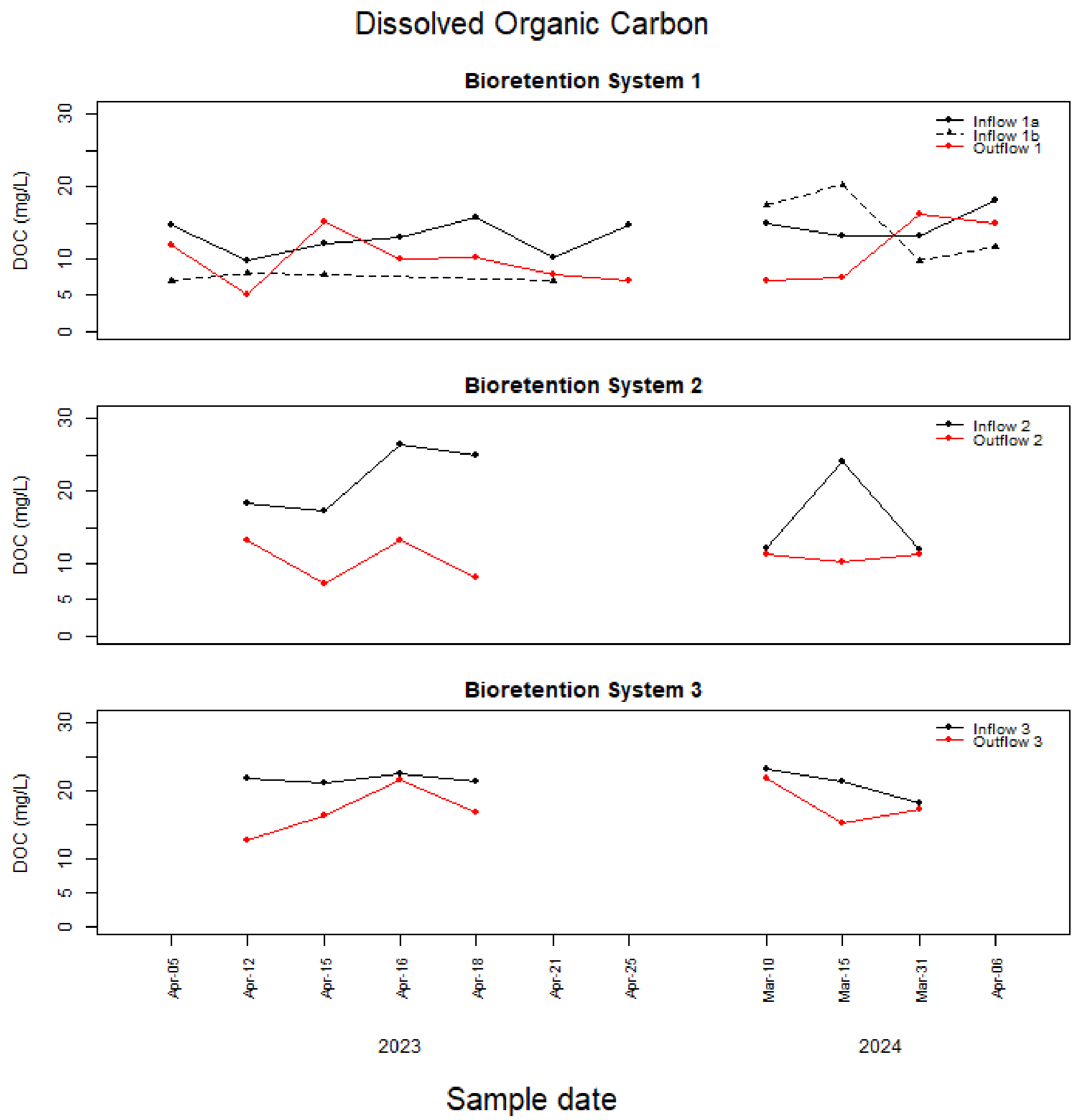


Figure S20 Dissolved organic carbon (DOC) concentrations from the inflow(s) and outflow of the three bioretention systems during the spring melt. No water quality guidelines or ecological benchmark exists for DOC concentrations.

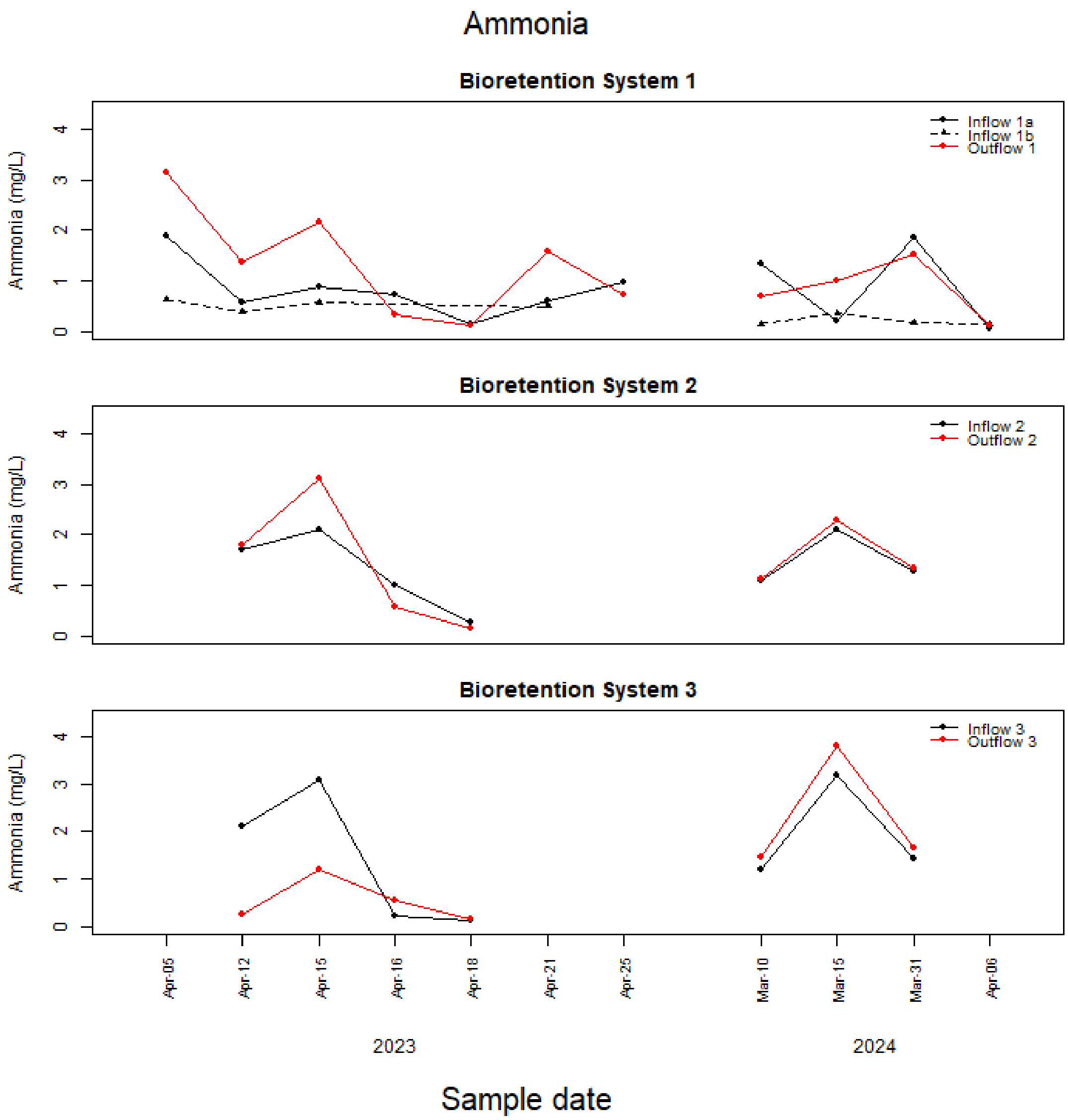


Figure S21 Ammonia concentrations from the inflows and outflows of the three bioretention systems during the spring melt. No fixed ammonia guideline is shown because ammonia toxicity thresholds vary with water temperature and pH. Exceedances where evaluated separately using temperature- and pH-dependent criteria.

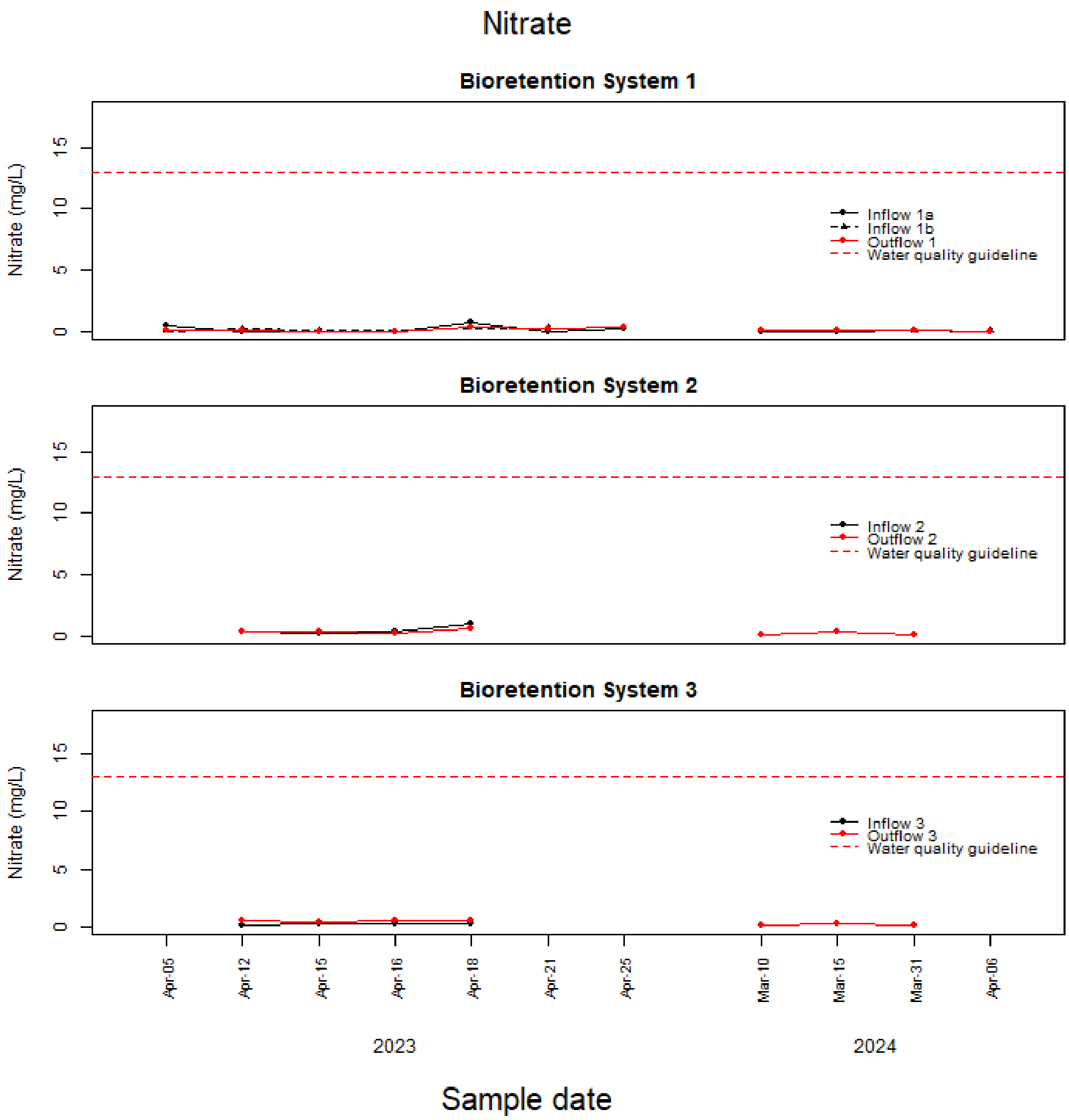


Figure S22 Nitrate concentrations from the inflows and outflows of the bioretention systems during the spring melt. The red dashed line represents the Canadian Council of Minsters of the Environment (CCME) water quality guideline for the protection of aquatic life for long-term nitrate exposure of 13 mg/L.

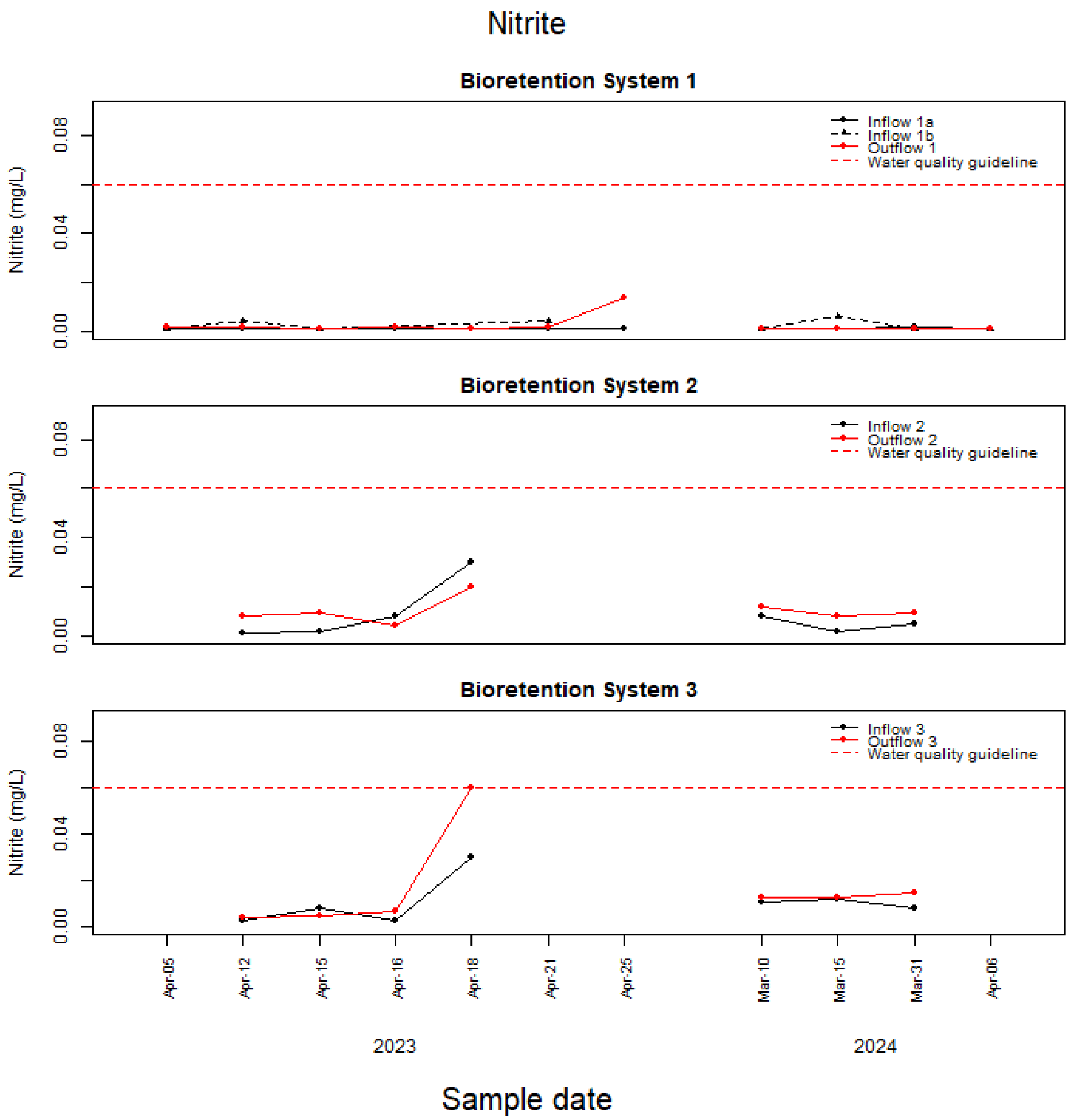


Figure S23 Nitrate concentrations from the inflows and outflows of the bioretention systems during the spring melt. The red dashed line represents the Canadian Council of Ministers of the Environment (CCME) water quality guideline for the protection of aquatic life of exposure of 0.06 mg/L.

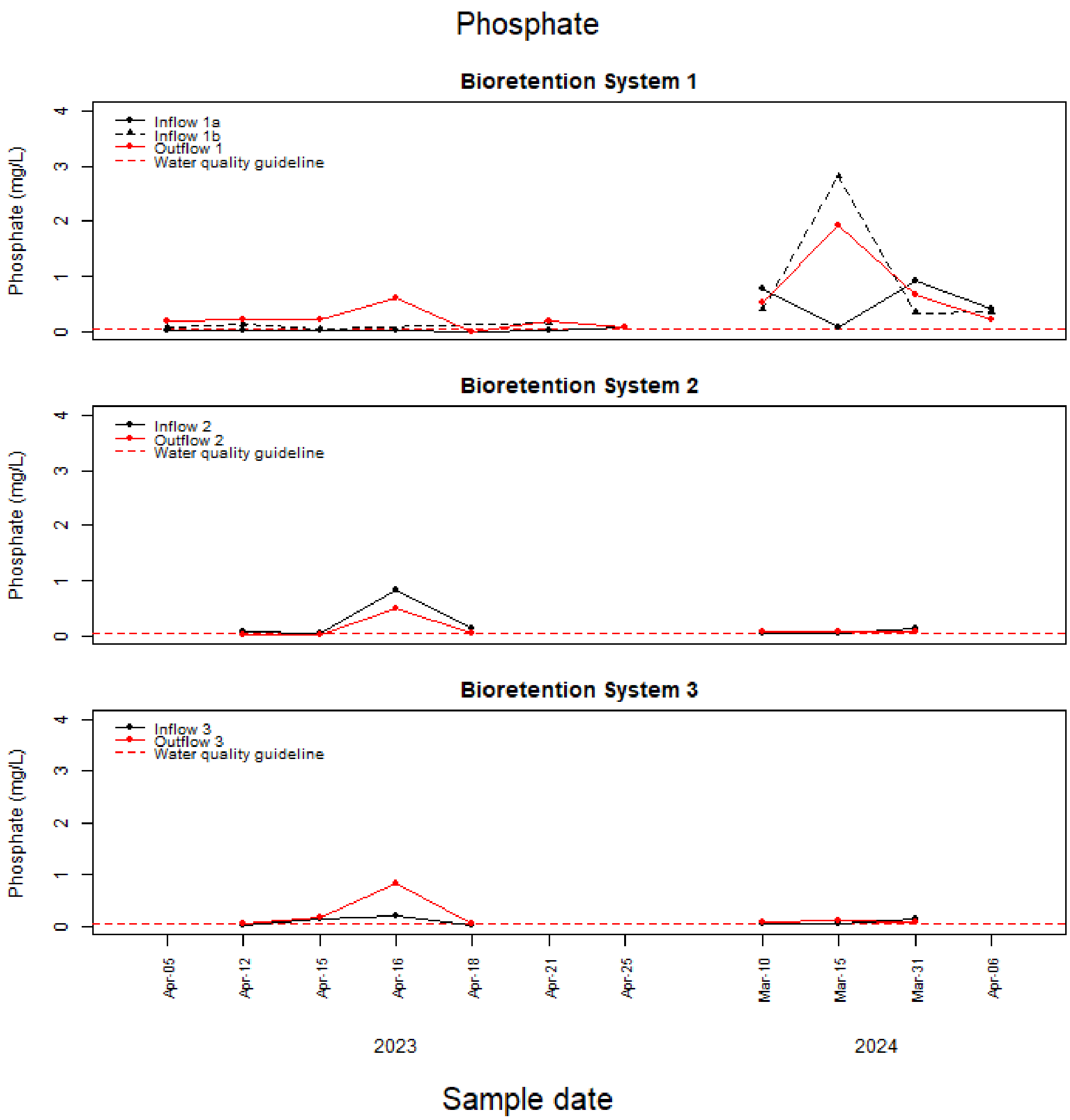


Figure S24 Phosphate concentrations from the inflows and outflows of the bioretention systems during the spring melt. The red dashed line represents the Canadian Council of Ministers for the Environment (CCME) water quality guideline of 0.03 mg/L for oligotrophic systems.

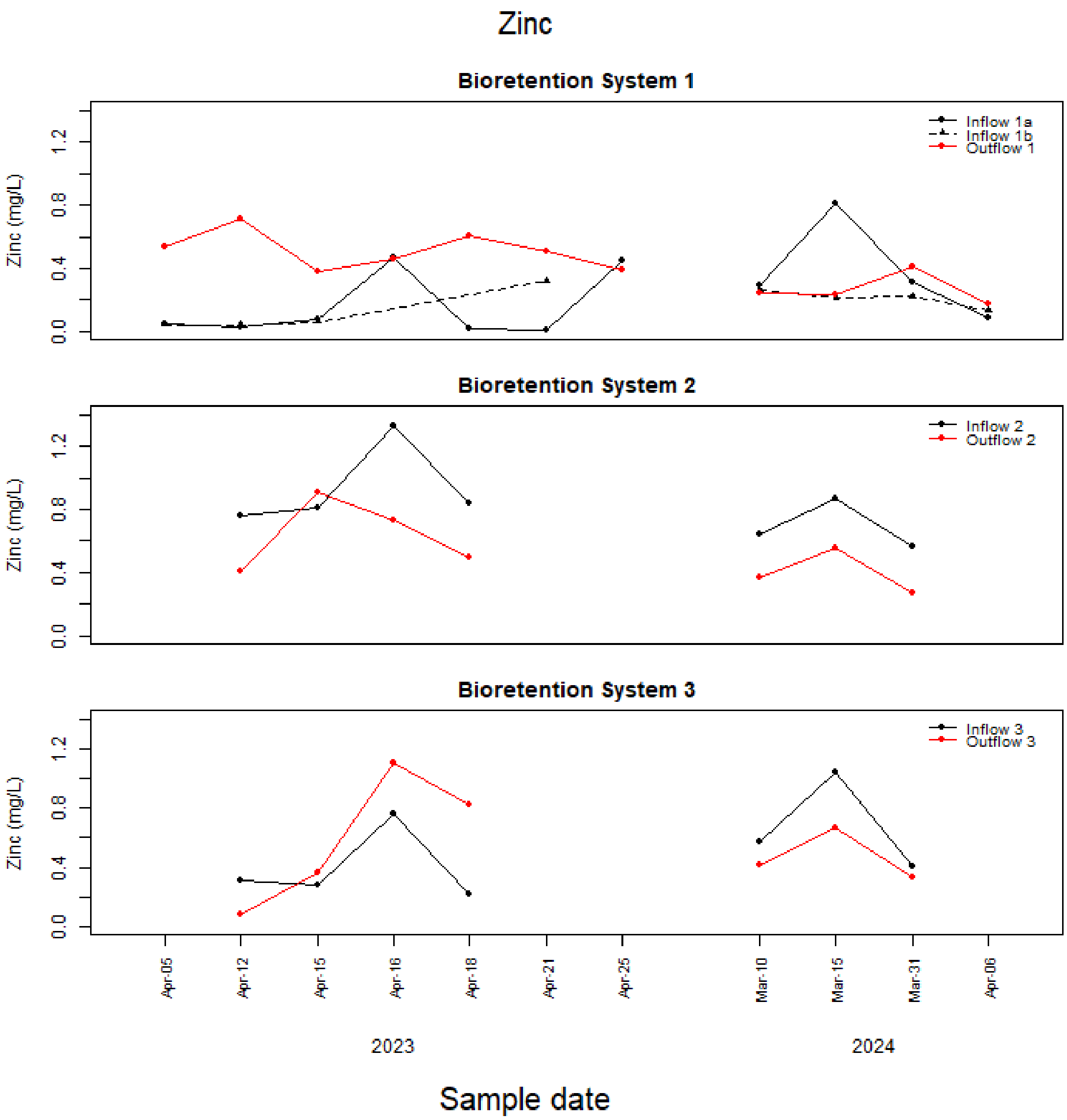


Table S25 Zinc concentrations from the inflows and outflows of the bioretention systems during the spring melt. The red dashed line represents the Provincial Water Quality Objective of 20 µg/L (0.02 mg/L). The CCME guideline was not used here as the guideline is dependent on water hardness which was not measured in this study.

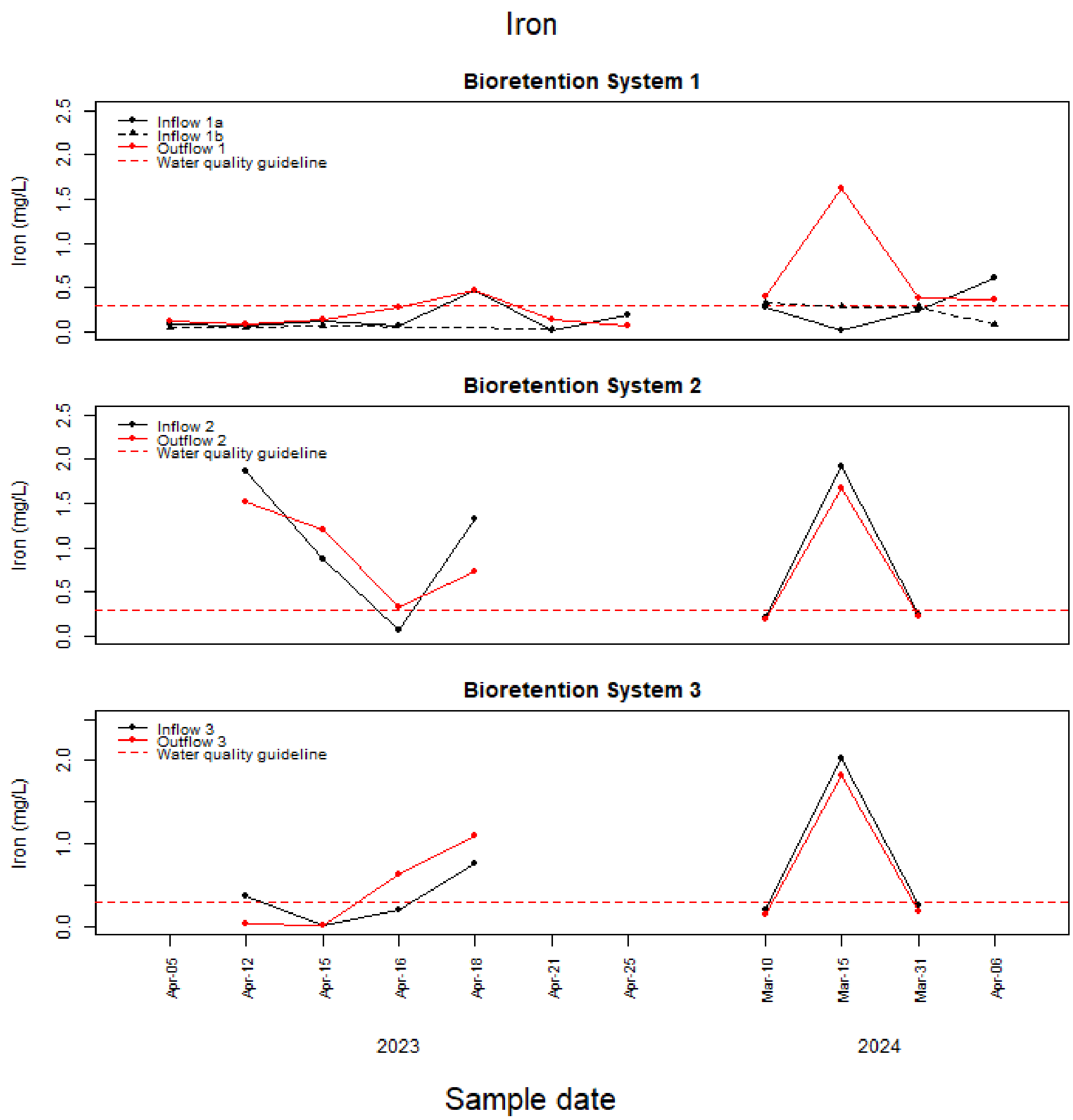


Table S26 Iron concentrations from the inflows and outflows of the bioretention systems during the spring melt. The red dashed line represents the CCME water quality guideline of 300 µg/L (equivalent to 0.3 mg/L).

Supplementary Materials for Chapter 5

Table S5: Field sheet for assessing fish habitat quality.

Site ID:	Date:	Time:
Observer Name:	**Weather Conditions:**	
1. Habitat Complexity Make note of the frequency and diversity of various flow regimes, including riffle, glide, run and pool habitats. Take note of whether there are frequent, well-distributed riffles, glides, and pools to support various fish life stages. If the channel is uniform in shape, with uniform flow and no flow diversity, make note of it.		
2. Streambed Composition Make note of the natural composition of the stream substrate (e.g., boulders, cobbles, shale, bedrock, gravel, sand, silt). Take note of whether the substate is 1) predominantly silt, clay or sand with no stable gravels or cobbles present, 2) mostly find sand or silt with scattered gravels or cobbles, 3) a mix of fines and gravels/cobbles, predominantly gravel/cobble substrates with minor fines, or 4) clean gravels or cobbles with little or no fines present.		
3. Instream Habitat Features Make note of the abundance and diversity of structures that provide fish habitat and shelter (e.g., logs, boulders, macrophytes, etc.). Are there multiple types of habitat structures present or is the habitat confined to a single structure (i.e., just boulders). What proportion of the reach has these structures, i.e., > 50%, 25-50%, 10-25%, <10%, or no instream habitat structures present.		

4. Bank Stability

Make note of the stability of the surrounding stream bank, including the degree of erosion, undercutting or slumping. Make note of whether the banks are well-vegetated with no evidence of erosion, have minor erosion or slumps, moderately unstable streambanks with some erosion, frequently, unstable stream banks with severe erosion, or whether there is major erosion with bare soil and/or collapsing stream banks.

5. Riparian Vegetation

Make note of the condition of the riparian vegetation on both stream banks. Measure the width of the vegetated riparian area from the edge of the streambank. Take note of the density of the vegetation. Is it continuous and intact native vegetation (forest or grassland setting), mostly continuous vegetation with some gaps, or patchy and disturbed? Make note if there is no natural vegetation present along the bank (e.g., a manicured lawn mown right down to the streambank, hardened /impervious surfaces along the streambank, etc.).

6. Sediment Deposition

Make note of the extent of fine sediment deposited in the stream bed. Make note of whether there is extensive deposition from upstream areas resulting in continuous deposits with > 50% of the stream cobbles embedded by fine sediment, if there is heavy deposition constrained mostly in pools and runs covering 30-50% of the stream cobbles, if there is a mix of fines within the cobbles, with cobbles or gravels 20-30% embedded by fine sediment, if there is limited deposition of fine sediment with less cobbles or gravels 10-20% embedded by fine sediment, or if there are clean cobbles or gravels with minimal (< 10% embedded) fine sediment.

Large Woody Debris

Make note of the abundance of fallen logs or large pieces of wood that may provide fish shelter and flow variability. Make note of whether there is abundant large woody debris creating pools, refuge and variable flow conditions; moderate large woody debris contributing significantly to fish habitat; sparse large woody debris with some habitat function; very little, large woody debris providing very little fish habitat; or no large woody debris present.

Habitat Connectivity

Make note of whether the reach is free of obstructions that may prevent fish passage within the reach. Make note of any impassible barriers such as large dams, major barriers that restrict fish passage during key spawning areas, partial barriers (e.g., perched culverts) that partially impede fish passage, whether there are minor or seasonal barriers, or no barriers to fish passage and fish can pass freely throughout the reach.

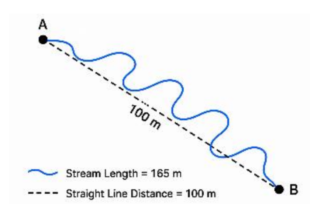
Canopy Cover and Shading

Stand in the middle of the stream and look straight upwards towards the sky. Approximate, as a percent, how much of the stream is covered by canopy and write the estimate below.

Channel Sinuosity

Start at the stormwater outfall. Measure a straight line 100 metres downstream and use this 100 m mark as the end point. Staying in the center of the stream, measure the total stream length between the stormwater outfall and the 100-metre end point. Calculate sinuosity using the equation below, and record the measurement:

Sinuosity = $\frac{Stream\ Length}{Straight\ Line\ Distance} = \frac{\quad}{\quad} =$



Important Note: Sinuosity can never be less than 1.

Calculations for Channel Sinuosity

Site 1 - Valley Length = 100 m, Channel Length = 107 m

Sinuosity = $\frac{Channel\ Length}{Valley\ Length} = \frac{107\ m}{100\ m} = 1.07$

Site 2 - Valley Length = 100 m, Channel Length = 111 m

Sinuosity = $\frac{Channel\ Length}{Valley\ Length} = \frac{111}{100} = 1.11$

Site 3 - Valley Length = 100 m, Channel Length = 106 m

Sinuosity = $\frac{Channel\ Length}{Valley\ Length} = \frac{106}{100} = 1.06$

Site 4 - Valley Length = 100 m, Channel Length = 112 m

Sinuosity = $\frac{Channel\ Length}{Valley\ Length} = \frac{112}{100} = 1.12$

Site 5 - Valley Length = 100 m, channel length = 124 m

Sinuosity = $\frac{Channel\ Length}{Valley\ Length} = \frac{124}{100} = 1.24$

Site 6 - Valley Length = 100 m, channel length = 103 m

Sinuosity = $\frac{Channel\ Length}{Valley\ Length} = \frac{103}{100} = 1.03$