Vegetative filter strips' performance on sediment erosion and deposition in agricultural drainage ditches of the littoral zone of Eastern Canada

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Abstract

The littoral zone of Lake Saint-Pierre in Québec, Canada, faces significant sediment deposition in agricultural drainage ditches due to snowmelt floods and human activities in the uplands, negatively impacting local fish habitats. This study evaluates the effectiveness of vegetative filter strips (VFS) in mitigating sediment erosion and deposition in these agricultural ditches, contributing to best management practices (BMPs) for water quality. Over a period from November 2019 to June 2023, VFS of various widths (0 m as control, 2 m, and 4 m) were implemented across three experimental sites to assess their performance. Total station surveys and LiDAR-equipped Unmanned Aerial Vehicles (UAV) were used for data collection and analysis, providing detailed measurements of sediment volumes and changes over time. The results indicated that the 2-m VFS reduced sediment deposition by an average of 17% compared to the control, although one site with a 2-m VFS recorded higher sediment deposits. In contrast, the 4-m VFS significantly reduced sediment deposition by 41%, 34%, and 38% at the three sites over the three-year period. Despite variability due to crop rotations and snowmelt events, VFS demonstrated a notable ability to prevent sediment deposition. The study identified several challenges in VFS implementation, including maintenance issues and the growth of in-ditch vegetation affecting efficiency. Furthermore, the comparison between LiDAR and total station surveys showed strong accuracy and consistency for LiDAR, though site-specific factors affected performance. This study underscores the significant role of VFS in sediment control and highlights the need for continuous research and monitoring to overcome practical challenges and enhance BMPs for sustainable agriculture in the littoral zones. The findings suggest that while VFS can effectively reduce sediment deposition, their performance is influenced by site-specific conditions and seasonal variations.

Résumé

La zone littorale du lac Saint-Pierre au Québec, Canada, est confrontée à un dépôt significatif de sédiments dans les fossés de drainage agricoles en raison des inondations dues à la fonte des neiges et des activités humaines en amont, ce qui impacte négativement les habitats des poissons locaux. Cette étude évalue l'efficacité des bandes filtrantes végétales (BFV) pour atténuer l'érosion et le dépôt de sédiments dans ces fossés agricoles, contribuant ainsi aux meilleures pratiques de gestion (MPG) pour la qualité de l'eau. Sur une période allant de novembre 2019 à juin 2023, des BFV de différentes largeurs (0 m comme contrôle, 2 m et 4 m) ont été mises en œuvre à travers trois sites expérimentaux pour évaluer leur performance. Des relevés par stations totales et des véhicules aériens sans pilote (UAV) équipés de LiDAR ont été utilisés pour la collecte et l'analyse des données, fournissant des mesures détaillées des volumes de sédiments et des changements au fil du temps. Les résultats ont indiqué que les BFV de 2 m ont réduit le dépôt de sédiments de 17 % en moyenne par rapport au contrôle, bien qu'un site avec une BFV de 2 m ait enregistré des dépôts de sédiments plus élevés. En revanche, les BFV de 4 m ont réduit de manière significative le dépôt de sédiments de 41 %, 34 % et 38 % sur les trois sites au cours de la période de trois ans. Malgré la variabilité due aux rotations des cultures et aux événements de fonte des neiges, les BFV ont démontré une capacité notable à prévenir le dépôt de sédiments. L'étude a identifié plusieurs défis dans la mise en œuvre des BFV, y compris des problèmes de maintenance et la croissance de la végétation dans les fossés affectant l'efficacité. De plus, la comparaison entre les relevés LiDAR et les relevés par stations totales a montré une grande précision et cohérence pour le LiDAR, bien que des facteurs spécifiques aux sites aient affecté la performance. Cette étude souligne le rôle significatif des BFV dans le contrôle des sédiments et met en évidence la nécessité d'une recherche continue et d'une surveillance pour surmonter les défis pratiques et améliorer les MPG pour une agriculture durable dans les zones littorales. Les résultats suggèrent que bien que les BFV puissent réduire efficacement le dépôt de sédiments, leur performance est influencée par les conditions spécifiques aux sites et les variations saisonnières.

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Contribution of Authors

All the manuscripts included in this thesis were written by Xuechao Chen (first author), Dr. Zhiming Qi (second author and supervisor), Dr. Monique Poulin (third author), and Dr. Shiv Prasher (fourth author). This study was conducted by Mr. Zhang Fan and Mr. Youjia Li, who were responsible for the experiment design and monitoring of ditch sedimentation from 2019 to 2020 and from 2021 to 2022. Starting in 2023, Xuechao Chen is responsible for designing experiments, monitoring ditch sedimentation, doing associated statistical analysis, and writing all parts of publications. Dr. Zhiming Qi played a crucial role in the research by providing financial support and as a prominent supervisor. Dr. Monique Poulin and her research group have implemented vegetative filter strips in all of the drainage ditches they evaluated. Dr. Shiv Prasher oversaw the research and offered definitive input.

Table of Contents

List of Figures and Tables

Figures

Tables

Chapter 1 Introduction

The St. Lawrence River's features are unique among the world's great rivers due to its extraordinary biodiversity, with the second-largest discharge in North America after the Mississippi (Meybeck & Ragu, 1997) and the largest estuary in the world (Dinauer & Mucci, 2017). As the plain along this river, the St. Lawrence Lowlands are characterized by their fertile soil (Moritz et al., 2015), especially areas around Lake St. Pierre (Area et al., 2019; Nicole, 2020). As a fluvial lake created by the Saint Lawrence River, Lake Saint-Pierre lies in the southwestern region of Québec (Hudon and Carignan, 2008). Agriculture is the predominant activity in the lake's surrounding area. This region is home to roughly 25% of all farms in the Province of Québec, Canada, which is geographically defined by the Laurentian Mountains to the north and the Appalachian Mountains to the south (MDDEFP, 2013). The littoral region of Lake Saint-Pierre encompasses a crop area spanning around 50 square kilometres. The favourable climatic conditions and fertile soil provide an environment that benefits the advancement of agriculture (Lévesque et al., 2020). In recent decades, there has been an increase in the level of intensity and specialization of agriculture here (de La Chenelière et al., 2014; MDDEFP, 2013). The alteration resulted in the deterioration of spawning habitats for indigenous fish, causing a decline in their population (Foubert et al., 2020). Yellow perch and northern pike, two iconic fish species of Lake Sainte-Pierre, typically lay their eggs on flooded vegetation (Hudon et al., 2010).

Nevertheless, being a significant agricultural producer in Québec, this region frequently faces the risk of seasonal flooding. The primary cause of flooding events in this region is attributed to the snowmelt in spring and early summer (Farly, 2021; Javelle et al., 2003; Magnan et al., 2002), which is associated with ice jams due to the spring breakup of the ice cover of St. Lawrence River (Buttle et al., 2016). Furthermore, agricultural lands in the littoral region are significantly impacted by sediment accumulation, especially drainage ditches, either through floods or as a result of human activities (Fullen & Catt, 2014; Walling & Fang, 2003). The high sedimentary influx result from periodic snowmelt floods (Beylich et al., 2016), in conjunction with low water flow velocity in the littoral region conducive to sedimentation (Bhattacharya et al., 2016; Van Maren et al., 2015), has resulted in the sediments accumulation within agricultural drainage ditches. Sediments, when acting as pollutants, serve as carriers for transporting various contaminants (Horowitz,1991; Walling, 2006; WASSON et al., 2010). These include, but are not limited to, heavy metalloids, agrichemicals like pesticides, and various nutrient loads (Syvitski et al., 2005), leading to the deterioration of aquatic ecosystems (McCarney-Castle et al., 2017; Syvitski et al., 2005; Van Metre & Mahler, 2004). Hence, measures to reduce sediment accumulation in agricultural drainage ditches should be implemented in the littoral zone of the Lake Saint-Pierre region to prevent the damage mentioned.

1.1 Justification - littoral zone

The littoral zone is the part of a body of water close to the shore. In lakes and rivers, it encompasses the area from the high-water mark to the depth where sunlight no longer penetrates the bottom (Emery & Stevenson, 1957). This zone is rich in aquatic plants and animals due to abundant sunlight, which supports photosynthesis and provides a hospitable environment for various species (Rich & Maier, 2015).

The littoral zone plays a significant role in agriculture, especially in regions adjacent to bodies of water (Wetzel, 1990). Vegetation in this zone helps stabilize shorelines, reducing soil erosion—a crucial factor in maintaining the integrity of agricultural land near water bodies

(Abrahams, 2006). By minimizing erosion, farmers can prevent the loss of fertile soil and reduce sedimentation in waterways, thereby improving water quality. Furthermore, littoral zones are biodiversity hotspots, offering habitats for various species, including beneficial insects, birds, and aquatic organisms that can help control agricultural pests (Cantonati et al., 2020; Schoumans et al., 2014). By supporting a diverse ecosystem, these zones enhance ecological balance and contribute to integrated pest management strategies (Barzman et al., 2015). Additionally, the plants in the littoral zone act as natural buffers, absorbing nutrients from agricultural runoff before they enter the water body, thus reducing nutrient pollution and its associated problems, like algal blooms and eutrophication (Horta et al., 2021; Zamparas & Zacharias, 2010).

1.2 Objectives

This study was intended to offer insights into policy formulation and implementation for soil restoration in Québec. The principal objective of this study was to assess the effectiveness of establishing narrow VFS along the agricultural drainage ditches on sediment deposition within the littoral zone of Lake Saint-Pierre in Québec, Canada. To achieve these objectives, we conducted five total station surveys (2020 - 2023) and four LiDAR surveys (2019 - 2021) with the following specific goals:

i. Undertake a comprehensive assessment of the efficiency of vegetative filter strips as a potential BMP for sediment erosion and accumulation within agricultural drainage ditches in the littoral zone of Lake Saint-Pierre.

ii. Assess the precision of LiDAR technology in relation to traditional field survey-based approaches (Total Station surveys) for monitoring sedimentation in ditches.

1.3 Thesis format

This thesis is structured into chapters as follows:

Chapter 1: This chapter provides the background of the study, including the relevant justification and main objectives.

Chapter 2: This chapter offers a literature review on the current study of vegetative filter strips (VFS) and their sediment trapping efficiency. It also includes a review of LiDAR technology and its application in measuring ditches.

Chapter 3: This chapter details the methodology, covering manual geological surveys (Total Station) and LiDAR applications in measuring agricultural drainage ditches, including relevant ditch volume calculations.

Chapter 4: This chapter analyzes the volume results to assess the efficiency of vegetative filter strips as a potential best management practice (BMP) for sediment erosion and accumulation within agricultural drainage ditches. It also compares ditch volumes measured by Total Station surveys with those measured by LiDAR to determine the accuracy of LiDAR in measuring agricultural drainage ditches.

Chapter 5: This chapter discusses the current results, identifies possible errors in the LiDAR surveys, and suggests improvements for future studies.

Chapter 6: This chapter provides a summary and conclusion of the study.

All references cited in this thesis are presented in the final chapter.

Chapter 2 Literature Review

2.1 Sediment erosion and deposition in agricultural drainage ditches

Agricultural drainage ditches are crucial for managing water flow and maintaining soil productivity in agricultural landscapes (Needelman et al., 2007). However, these ditches often face challenges from sedimentation and erosion, significantly impacting their functionality and environmental health (Dollinger et al., 2015). Erosion in agricultural drainage ditches occurs primarily due to water flow, which can detach and transport soil particles (Herzon & Helenius, 2008). The intensity of erosion depends on several factors, including water velocity, soil type, vegetation cover, and ditch slope. When water flow exceeds the soil's resistance to erosion, particles are detached and carried downstream (Foster et al., 1985). Erosion can be categorized into sheet erosion, rill erosion, and gully erosion, each contributing differently to sediment load in drainage ditches (Morgan & Rickson, 2003).

Sediment deposition occurs when the energy of the transporting water decreases, causing suspended particles to settle (Singh & Hartsch, 2019). This can happen in areas where water flow slows down, such as in broader sections of the ditch or at structural barriers. The deposition rate is influenced by particle size, water flow rate, and the presence of vegetation (Shi et al., 2019; Zong & Nepf, 2010). Fine particles like silt and clay settle more slowly compared to larger sand particles, often leading to sediment accumulation in specific ditch sections (Dollinger et al., 2015). Sediment deposition reduces the hydraulic efficiency of drainage ditches by decreasing their flow capacity (Lecce et al., 2006). This can lead to increased flooding risk and reduced drainage effectiveness, necessitating frequent maintenance to remove accumulated sediments (D'Ambrosio et al., 2022). On the other hand, erosion can lead to ditch deepening and bank instability, compromising the structural integrity of the ditch and requiring costly repairs (Kramer et al., 2019).

Erosion and sediment transport from agricultural fields can carry nutrients, pesticides, and other pollutants into drainage ditches, negatively impacting water quality (Blann et al., 2009; Needelman et al., 2007). Nutrient loading, particularly phosphorus, can lead to eutrophication in downstream water bodies, causing algal blooms and hypoxic conditions detrimental to aquatic life (Rashmi et al., 2022). Managing sediment transport is thus crucial for maintaining water quality in agricultural landscapes (Dollinger et al., 2015). Sediment deposition can also alter the physical habitat of drainage ditches, affecting aquatic and riparian species (Rideout et al., 2022). Accumulated sediment can smother benthic habitats, reduce water depth, and change flow patterns, impacting fish and invertebrate communities (Kemp et al., 2011).

2.2 Impacts of sedimentation and erosion of agricultural drainage ditches

Sedimentation can significantly reduce the hydraulic efficiency of drainage ditches by decreasing their flow capacity and increasing the risk of flooding (Needelman et al., 2007). Conversely, erosion can lead to channel instability and bank collapse, further compromising ditch function (Amarasinghe, 1996). Studies (Dollinger et al., 2017; Streeter & Schilling, 2020) demonstrate the adverse effects of sedimentation and erosion on ditch performance, emphasizing the need for regular maintenance and monitoring.

Sediment-laden runoff from agricultural fields can carry nutrients, pesticides, and other pollutants into drainage ditches, degrading water quality and impacting aquatic ecosystems (Rashmi et al., 2022). The link between sedimentation and nutrient loading in agricultural watersheds suggests that sediment control measures are essential for protecting water quality (Fiener et al., 2005). Sedimentation and erosion can alter the physical habitat of drainage ditches, affecting aquatic and riparian species (Blann et al., 2009). Sediment deposition can smother benthic habitats, while erosion can lead to the loss of riparian vegetation and increased sediment input to downstream water bodies (Kemp et al., 2011; Wilkes et al., 2019; Wood & Armitage, 1997). Research (Kröger et al., 2013) illustrates the ecological consequences of sedimentation and erosion in agricultural drainage systems, advocating for integrated management approaches that consider both hydraulic and ecological factors.

2.3 Vegetative filter strips

As one of the water quality Best Management Practices (BMPs), vegetative filter strip (VFS) plays a critical role in reducing the implications of nonpoint source pollution, with a notable emphasis on sediment erosion and deposition (Daniels & Gilliam, 1996; Lee et al., 2000; Jang et al., 2017), VFS can also be treated as the potential spawning habitat for yellow perch and northern pike (Magnan et al., 2022). Additionally, compared to other BMPs, which can improve sediment erosion and deposition, VFS offers the advantage of less labour and time cost and financial investments, ensuring that farmers' harvest schedules remain unaffected (Anebagilu et al., 2021). Agricultural VFS are bands of continuous vegetation between fields and waterways, which capture field runoff and prevent contaminants from reaching streams (Dosskey et al., 2002). These strips of native or cultivated vegetation act as barriers between water bodies and adjacent lands that could serve as nonpoint pollution sources (Munoz-Carpena et al., 1999). Furthermore, vegetation situated at the lower boundary of impacted zones can decline runoff quantity and its peak speed, primarily due to the hydraulic coarseness of the filter, enhancing infiltration. This deceleration facilitates the sedimentation of suspended particulates, enabling their redeposition within the confines of the VFS (Dosskey, 2001; Lee et al., 2003; Syversen and Borch, 2005).

The effectiveness of VFS in controlling sediment pollution has been frequently shown by the hydrologically regulated leaching plots in recent years (Arora et al., 2003; Bouldin et al., 2004; Duchemin & Hogue, 2009; Patty et al., 1997; Saleh et al., 2018). Patty et al. (1997)

found that 87% to 100% of total suspended solids (TSS) in runoff was retained by a 20-m-wide VFS, similar to the 78% TSS concentration reduction in the runoff by a narrow VFS in northeast Italy (Borin et al., 2005). TSS was assessed through centrifugation using a specified relative centrifugal force after sampling from the storage tanks of the plots. Arora et al. (2003) also found that VFS decreased the mean sediment concentration in the outflow by over 60%, ranging between 60% and 80%, compared to the inflow concentrations. The sediment concentrations were assessed using a gravimetric oven-drying technique to duplicate inflow and outflow water samples. By evaluating three different VFS, sediment concentrations were reduced by 42% to 94% (Saleh et al., 2018). Turbidity levels of water samples in ditches with VFS decreased by approximately 60% compared to ditches without such strips (Bouldin et al., 2004). By modelling in the Tilting-Flume Simulated Rainfall facility, the sediment trapping efficiency of VFS was predicted to be from 45% to 80% under various inflow rates (Hussein et al., 2007). In the agricultural lands of southern Québec, Duchemin and Hogue (2009) found that VFS can reduce around 85% of TSS in runoff water by establishing an integrated VFS system.

However, an extensive body of knowledge exists regarding the limitations to the sediment trapping efficiency of VFS, and findings from uncontrolled field experiments have occasionally indicated restricted effectiveness (Hénault-Ethier et al., 2017). Verstraeten et al. (2006) found that VFS cannot be an effective control measure based on spatially distributed soil erosion and sediment delivery modelling. The degree of runoff convergence upon entering VFS is considered the critical factor in this regard (Hösl, 2014). It has been proven that the convergence of surface runoff into buffer strips leads to a decrease in the VFS's retention capacity of sediment (Arora et al., 2010; Dodd & Sharpley, 2016; Dorioz et al., 2006; Hay et al., 2006). Linear features, like ditches, can reduce the filter effect on sediment in the case of concentrated surface flow (Bach et al., 1994; Hösl et al., 2012). The ability of VFS on sediment erosion and deposition also exhibits significant variability across diverse environments (Hickey and Doran, 2004), and seasonality has been proven to affect VFS effectiveness in northern latitudes (Hénault-Ethier, 2016). In addition to the unnecessaries described above, limited research has been conducted on narrow VFS widths between 1 to 10 meters, commonly observed in agricultural areas (Anebagilu et al., 2021; Hickey & Doran, 2004). There is a significant need for experimental investigations to assess the efficiency of narrow VFS in removing sediment on agricultural farms.

2.4 Sedimentation monitoring approaches in agricultural drainage ditches

Agricultural drainage ditches are essential to farm management systems, as they remove excess water from agricultural fields, thereby improving crop productivity (Strock et al., 2010). However, these ditches also serve as conduits for sediment and nutrient transport, which can significantly impact downstream water bodies. Effective sedimentation monitoring in these ditches is crucial for managing soil erosion and sedimentation (Dollinger et al., 2015; Streeter et al., 2019).

Manual surveying is a traditional method for measuring the volume of a drainage ditch. This approach involves physically measuring the ditch's cross-sectional profiles and longitudinal sections at regular intervals (Krider et al., 2017). Cross-sectional surveying measures the width and depth of the ditch at multiple cross-sections along its length using tools like measuring tapes, leveling rods, and surveying instruments such as total stations or theodolites (Weaver et al., 2005). Profile surveying involves creating a longitudinal profile of the ditch by measuring the elevation along the centerline, helping to understand the slope and overall shape of the ditch (Diplas et al., 1999). The data collected from these surveys can be used to create a three-dimensional model of the ditch, allowing for volume calculations. The use of GPS and total station technology significantly enhances the accuracy and efficiency of ditch volume measurements (Ibitoye, 2017; Vandromme et al., 2017). These tools can precisely capture the three-dimensional coordinates of various points along the ditch (Erickson et al., 2013). GPS surveys involve using GPS receivers to record the precise location of points along the ditch, which can then be used to create detailed maps and cross-sectional profiles (Taboga, 2011). Total stations combine electronic distance and angular measurements, providing high-accuracy data for creating detailed cross-sectional and longitudinal profiles (Shen et al., 2021).

LiDAR is an advanced remote sensing technology that can obtain high-resolution topographic data, making it particularly useful for measuring the volume of larger drainage ditches or networks (Lin et al., 2021; Roelens et al., 2018). Airborne LiDAR, mounted on aircraft or drones, emits laser pulses toward the ground and measures the time for the pulses to return (Kellner et al., 2019). This data creates detailed digital elevation models (DEMs) of the ditch and surrounding area. Terrestrial LiDAR, on the other hand, is used for smaller ditches or specific sections that require high detail, providing highly accurate three-dimensional representations of the ditch topography (Tarolli, 2014). By integrating various methods, from traditional manual surveying to advanced remote sensing and modeling techniques, we can achieve different levels of accuracy and applicability for measuring ditch volume (Lin et al., 2021). Combining these methods can provide comprehensive and accurate results, ensuring better management and maintenance of agricultural drainage systems.

2.5 The role of LiDAR in measuring agricultural drainage ditch volumes

LiDAR (Light Detection and Ranging) is a remote sensing technology that uses laser pulses to measure distances to the Earth's surface, creating high-resolution, three-dimensional maps of the terrain. This technology has proven highly effective in measuring agricultural drainage ditch volumes, as demonstrated in various studies. For instance, Eitel et al. (2016) utilized airborne LiDAR to map agricultural drainage ditches in Iowa, USA, highlighting its capability to accurately capture ditch profiles and calculate volumes. Similarly, Jones and Hobbs (2021) used terrestrial LiDAR to assess erosion and sediment deposition in drainage ditches in the UK, providing insights into sediment transport processes and informing erosion control measures. Smith et al. (2019) further demonstrated the efficiency of drone-mounted LiDAR in surveying complex networks of drainage ditches in the Netherlands, capturing detailed topographic data over large areas.

LiDAR offers several advantages over traditional surveying methods, making it an attractive option for measuring ditch volumes. Its high accuracy captures fine details of the ditch's topography, essential for precise volume calculations and effective drainage system design (Rapinel et al., 2015). Additionally, LiDAR's efficiency allows for rapid surveys over large areas, significantly reducing the time required for data collection compared to manual methods (Ryding et al., 2015). The dense point cloud generated by LiDAR provides a detailed 3D representation of the terrain, useful for various analyses beyond volume measurement, such as erosion assessment and landscape modeling (Eitel et al., 2016). Furthermore, the remote sensing capability of airborne LiDAR enables access to areas that are difficult to reach on foot, making it ideal for surveying extensive or remote ditch networks (Tarolli & Mudd, 2020).

Despite its advantages, LiDAR technology has some limitations. The cost of LiDAR equipment and data processing software can be high, potentially limiting its accessibility for smaller projects or organizations with limited budgets (Chang et al., 2014). The complexity of data processing also requires specialized skills and software, which can be a barrier to its widespread adoption. Additionally, dense vegetation can obstruct laser pulses, leading to data gaps, although advanced processing techniques can mitigate this issue to some extent (Muumbe et al., 2021). Nonetheless, LiDAR's high accuracy, efficiency, and ability to capture detailed topographic data make it an invaluable tool for managing drainage systems and mitigating environmental impacts (O'Neil et al., 2018; Tarolli, 2014; Yen et al., 2011). Ongoing technological advancements are likely to make LiDAR more accessible and user-friendly, enhancing its utility in agricultural drainage management.

Chapter 3. Materials and Methods

3.1 Experimental sites and design

3.1.1 Experimental sites

The study was carried out at three experimental sites located on the Lac Saint-Pierre floodplain, Québec, Canada, near St. Cuthbert (STC - 46°07'48.0" N, 73°07'26.4" W), Yamachiche (YAM - 46°16'04.8" N, 72°51'50.4" W) and Baie-du-Febvre (BDF - 46°08'34.8" N, 72°42'46.8" W) (Figure 1). The experimental sites represent a typical agricultural operation setting for the coastal zone of Lake Saint-Pierre, with an annual corn-soybean rotation, fall tillage, and drained by open ditches directly to the lake. The yearly average temperature in the region is around 7 °C, and average annual precipitation is about 900 mm (ECCC, 2023). All ditches for these sites frequently face the risk of sediment accumulation and seasonal flooding.

Figure 1. Location of the three experiment sites.

3.1.2 Experimental design

This field experiment design follows the completed blocked randomized design, with each site as one block and two treatments: 2- and 4-m wide VFS along each ditch on both sides versus a control (0-m). The experiment started in November 2019 after all nine selected ditches were dredged to a similar initial condition.

In spring 2020, 12 reed canary grass-oat (*Avena sativa*) mixture VFS strips, ranging from 600 to 1000 m long, were planted on both sides of the 2-m and 4-m VFS ditches (Figure 2). They were all seeded with a grain drill equipped with either a mile box or a ridge seeder, at a rate of 29 kg ha^{-1} and 71 kg ha^{-1}, respectively, without application of fertilizer. All strip portions that showed failed or poor RCG establishment (approximately 60% of total strip length) were completely reseeded or overseeded in May 2021. All strips were mowed between August 1 and 15 every year to prevent the regrowth of companion crops and weeds.

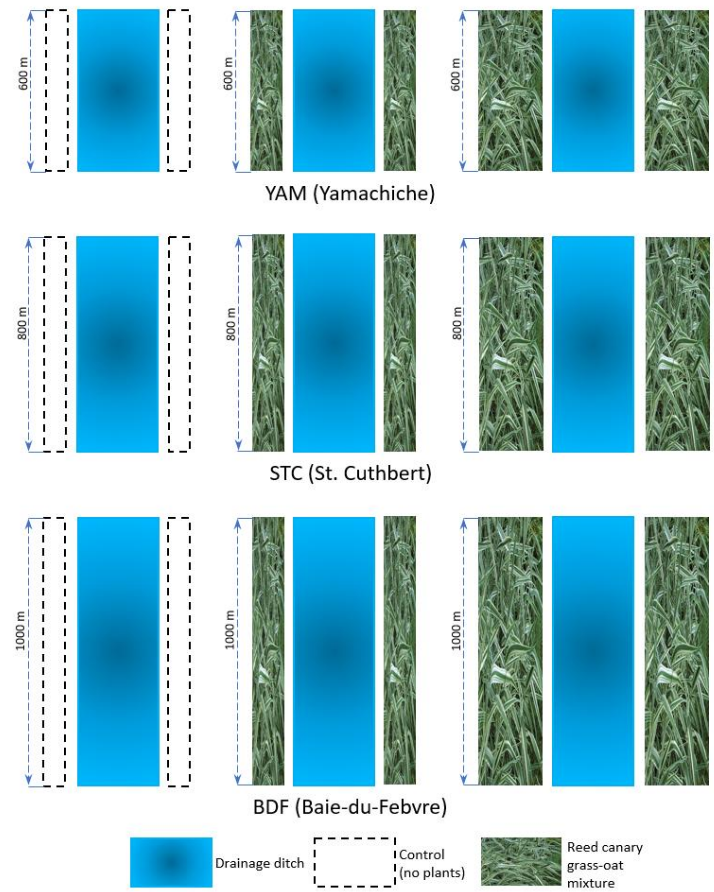

Figure 2. Configuration of the vegetative filter strips (VFS) across three experimental sites, showing the arrangement of 2-meter and 4-meter-wide strips on both sides of the ditches.

To be noted, the YAM 0-m VFS ditch's location was changed to its east side due to the damage of the ditch by stream flow. This location switch was made considering that the original YAM 0-m VFS ditch's dense vegetation could act as a VFS, creating a false representation of non-VFS-equipped ditches. Also, the second half (lengthwise) of the YAM 2-m VFS was planted with pure oat by mistake in 2021.

3.2 Total Station surveys

3.2.1 Ditch cross-section measurements

A series of total station surveys have been conducted in the study area since 2019, and the morphology of the ditches was surveyed by measuring their cross-sections with intervals of 20 meters. In total, seven total station geological surveys were performed to measure the cross-section profiles of the ditches: once after the ditch trenching in 2019, twice per year in 2020 and 2021 (at both the start of the growing and non-growing season), and once per year (in June) in 2022 and 2023 (Figure 3a). However, the first geological survey was unfinished due to harsh weather in the late winter of 2019, and the VFS were replanted due to poor growth in the Spring of 2021.

The reflective stakes were set up at a 20-meter interval on both sides of the ditch to mark the cross-sections to be repeatedly measured. The ditch cross-section profiles' topography was measured using a Leica® TS-06 manual total station and its associate GPR-1 prisms (Figure 3b). The reference points used were all set up on the fixed objectives, like bolts for the base of the streetlight and marking points for drainage pipes. The cross-section topography was constructed via a one-dimension geodetic measurement (elevation only) in a 25 cm interval alone the cross-section profile (20 measurement points per cross-section). To prevent the survey operation's disturbance to the ditch geomorphology, a modified aluminum ladder, graduated every 25 cm, was placed above the ditch as a stepping platform (Figure 3c).

Figure 3. (a) Ditch trenching. (b) the total station. (c) the progress of the measurement with the ladder.

3.2.2 Ditch volume calculation

The definition of sampled ditch cross-section area (A) is based on a slight modification of Roealens et al. (2016). The area between the cross-section profile sides and its highest elevation's horizontal projection line was set to be the cross-section area (Figure 4). The horizontal projection line position was altered from the lowest profile elevation maxima to the profile's highest elevation. This accommodates the interest in monitoring topographic change on both ditch sides.

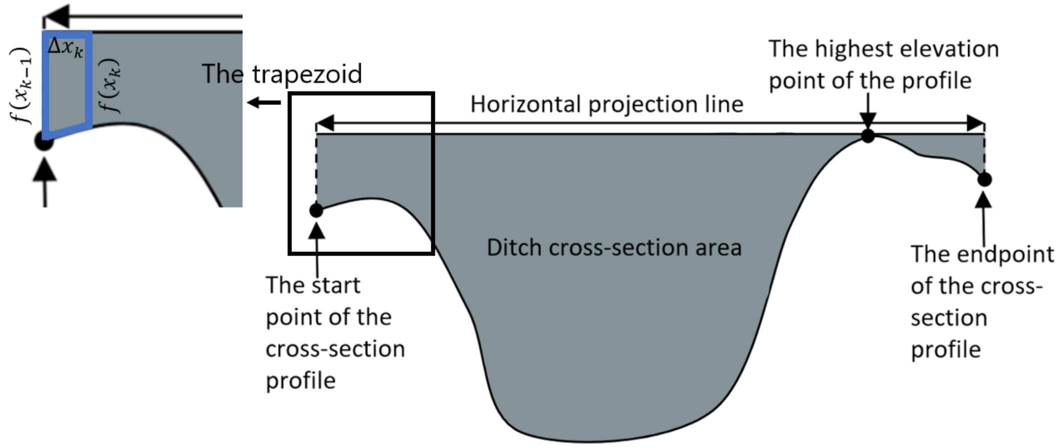

Figure 4. Cross-section area definition.

The ditch's topographic profile was sampled as cross-sections in a 20-meter interval (the number of cross-sections sampled varies by the length of each ditch). Individual ditch's volume (V) was estimated by following Equations 2 and 3:

$$A_K = \frac{(A_i + A_{i+1})}{2} \tag{2}$$

$$V = \sum_{K=1}^{N-1} A_K \times interval \tag{3}$$

where

A_K = average area of two adjacent surveyed cross-section profiles (m^2)

A_i = area of the surveyed cross-section profile (m^2)

K = the K_{th} of the calculated cross-section profile for the ditch

i = the i_{th} of the surveyed cross-section profile for the ditch

V = ditch volume measured (m^3)

N = number of ditch cross-section profiles surveyed for the ditch.

The volume change rates for ditch and sediment deposition ($V\%$) were both calculated by Equation 4:

$$V\% = \frac{(V_{m-1} - V_m)}{V_{m-1}} \times 100\% \tag{4}$$

where

$V\%$ = volume changes rate

V = ditch volume measured (m^3)

m = the m_{th} of total station measurement

3.2.3 Sediment deposition calculation

Sediment deposition was directly shown by the volume of sediment deposits in this study. The sediment deposition volume (V_{SD}) was determined by Equation 5:

$$V_{SD} = V_{m-1} - V_m \tag{5}$$

where

V_{SD} = sediment deposition volume (m^3).

3.3 LiDAR aerial surveys

3.3.1 Data acquisition

All studied ditches have been scanned four times (2019 Nov. (before trenching), 2019 Dec. (after trenching), 2020 Oct., and 2021 Oct.), using the YellowScan^{TM} Surveyor LiDAR system accompanied by a DJITM Matrice 600 UAV (Figure 5a), with the LIDAR system's in-situ GNSS antenna connected to a SpectraTM SP80 GNSS receiver (Figure 5b) in the base-station mode. The UAV was kept within the 2 km radius of the GNSS base station and recorded the LiDAR sensor position using the post-processed kinematic positioning (PPK) method.

Figure 5. (a) LiDAR sensor and its platform UAV; (b) the GNSS-base station setup.

The LiDAR system has a 903 nm laser wavelength and can generate two echoes per shot. The scanning angle was set to 40° (± 20°). The flight height was controlled to be 15 meters above the ground. The collected 3D point cloud data has a point density between 250 – 400 pt/m2. This point cloud data was processed through the LiDAR sensor's associate YellowScan^{TM} Cloud Station software for noise filtering and point cloud classification based on the echo order and signal return strength. The processed point cloud data were used to generate a digital terrain model (DTM) with a spatial resolution of 5cm based on all the bare earth-classified data points.

3.3.2 Data processing algorithms

After collecting raw data (Figure 6) in the field, further processing and analysis were conducted in the laboratory. MATLAB R2024a software (MathWorks, Natick, Massachusetts, USA) was used for data processing and volume calculations, providing detailed analysis and accurate results.

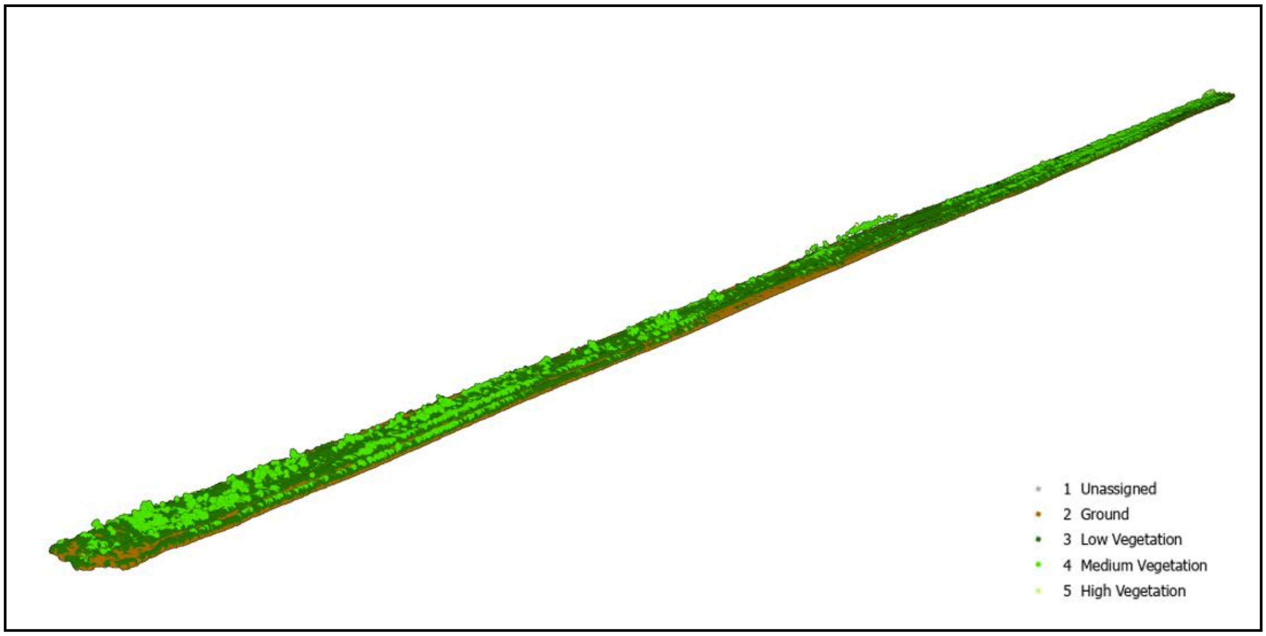

Figure 6. Raw LiDAR data for YAM 4-m VFS ditch in 2020.

All ditches' 3D models were reconstructed based on the bare land points (LiDAR classification number 2) and GPS data to remove other points, mainly low vegetation and water. Figure 7A shows an example of the bare ground point cloud of one drainage ditch. Low vegetation is defined as vegetation with a canopy height of less than 30 centimetres; medium vegetation is defined as vegetation with a canopy height ranging from 30 to 250 centimetres; and those with a canopy height larger than 250 centimetres are referred to as high vegetation (Wang et al., 2017).

Due to the dense low vegetation covering the ditch bank and water in the ditch, the density of laser point clouds cannot be uniform. Especially at the bottom of the ditch, the density is too small, and the points are sparse (Figure 7B). To avoid the above problem, each ditch was cut into 0.5m long segments according to the ditch direction (the ditch sides going opposite to the drainage flow were defined as upstream in this study). A representative cross-section would be selected for each segment.

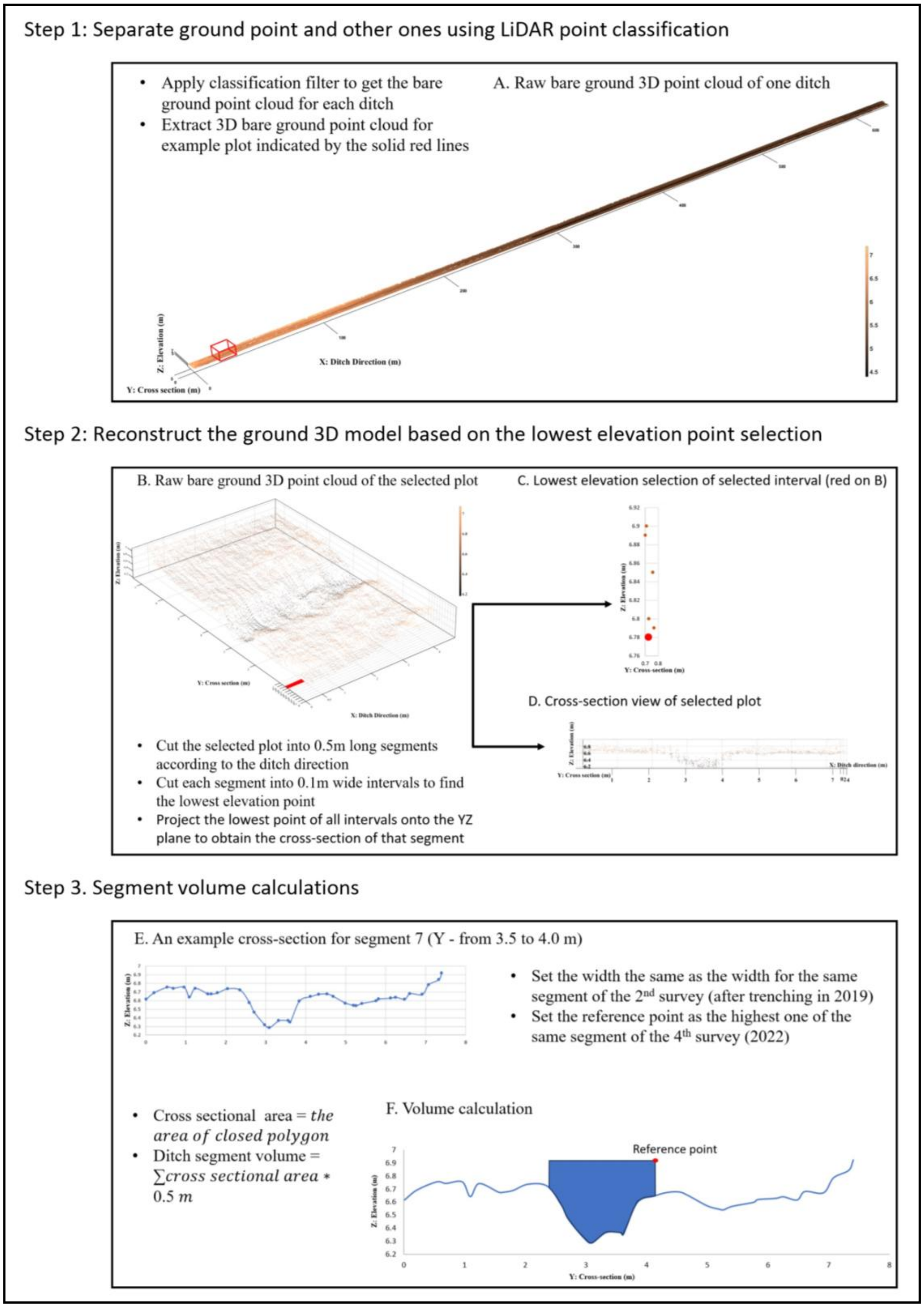

Figure 7. Data processing pipeline. (A) an example of the bare ground point cloud of one ditch; (B) the raw bare ground point cloud of the selected plot; (C) an example of lowest point selection for one interval; (D) the cross-section view after points selection; (E) a cross-section example; (F) volume calculation.

According to the cross-section axis, each segment was cut into 0.1m wide intervals to find the lowest elevation point (Figure 7C, D), which would be used for the volume calculation for the relevant segment. The projection of all intervals' lowest point onto the YZ plane is defined as the cross-section of that segment (Figure 7E). The width for all four times measurements is set the same as the width used for the 2^{nd} survey (after trenching in 2019). The area below the level of reference point, which is the highest elevation point of the same segments of the 4th survey (2022), was the cross-sectional area of the segment (Figure 7F). To calculate the cross-sectional area (A), the trapezoidal rule was used (Figure 4). Since the grid spacing was non-uniform, the cross-sectional area was calculated similarly to Equation 1, shown below (Equation 6):

$$A = \int_a^b f(x)dx = \sum_{k=1}^{n} \frac{f(x_{k-1}) + f(x_k)}{2} \Delta x_k \quad (6)$$

where

A = segment cross-section area (m^2)

a = the start lowest elevation point of each segment cross-section profile

b = the end lowest elevation point of each cross-section profile

x_k = horizontal distance of each ground point from the start point of each profile (m)

$f(x)$ = vertical distance of each ground point from the reference point of each cross-section profile (m)

Based on each segment's cross-section area and the segment length (0.5 m), the individual ditch's volume (V) was estimated by following Equation 7:

$$V = \sum_{i=1}^{N-1} A_i \times interval \tag{7}$$

where

A_i = area of the segment cross-section profiles (m^2)

i = the i_{th} of the segment cross-section profile for the ditch

V = ditch volume calculated (m^3)

N = number of ditch cross-section profiles surveyed for the ditch

$interval$ = 0.5 (m).

3.4 Comparison between Total station results and LiDAR results

The performance of the October 2021 LiDAR data was compared with the October 2021 total station geological data to assess LiDAR accuracy in three aspects:

First, the volumes of all ditches calculated from the Total Station and LiDAR surveys were compared using a paired-sample t-test.

Second, the areas of all cross-sections from the LiDAR surveys were compared with the cross-section areas measured by the Total Station at the same locations.

Third, the elevation data points of the first cross-section profiles (the first 0.1 m) extracted from the LiDAR-generated Digital Terrain Model (DTM) were compared to the geocoordinates of the same profiles obtained from the Total Station survey. The cross-section profile areas were calculated using the same method as that used for the Total Station survey.

The accuracy of area and elevation is assessed using the root mean square error (RMSE) (Equation 8), mean absolute error (MAE) (Equation 9) and correlation coefficient (r) (Equation 10):

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(z_{LiDAR} - z_{total\ station})^2}{n}} \tag{8}$$

$$MAE = \frac{\sum_{i=1}^{n}|z_{LiDAR} - z_{total\ station}|}{n} \tag{9}$$

$$r = \frac{\sum_{i=1}^{n}(z_{LiDAR} - \overline{z_{LiDAR}})(z_{total\ station} - \overline{z_{total\ station}})}{\sqrt{\sum_{i=1}^{n}(z_{LiDAR} - \overline{z_{LiDAR}})^2(z_{total\ station} - \overline{z_{total\ station}})^2}} \tag{10}$$

where

z_{LiDAR} = elevation (m) of data points or cross-section areas (m^2) from the LiDAR survey

$z_{total\ station}$ = elevation (m) of data points or cross-section areas (m^2) from the Total Station survey at the same locations as those from the LiDAR survey

n = the number of data points or cross-sections.

Moreover, all segment areas with the same length along the ditch direction as Total Station surveys in October 2021 were compared using the same factors above (RMSE, MAE, r). All nine ditches' volumes determined by Total Station surveys and LiDAR surveys in October 2021 were compared using the same length and relatively the same width.

3.5 Statistical analysis

The IBM® SPSS Statistics software© (Version: 28.0.0.0) was used during the statistical analysis process. A one-way analysis of variance (ANOVA) was performed using the sediment deposition data collected from June 2020 to June 2023 as the response variable. The factor "VFS width" comprised three levels (0 m, 2 m, and 4 m), with each level including three replicates from the three sites (YAM, STC, and BDF). Considering that initial ditch volume varied by site, we also calculated relative volume-change rates to account for these differences. Additionally, paired-sample t-tests were conducted separately for each site to compare the sediment deposition and volume changes between pairs of VFS widths (e.g., 0 m vs. 2 m) (Kim, 2015). Finally, a two-way repeated-measures ANOVA was employed to examine how VFS efficiency changed over time, treating measurement time as the within-subject factor and VFS width as the between-subject factor (Dien & Santuzzi, 2005). All significance tests were evaluated at $p = 0.10$.

Chapter 4 Results

4.1 Total Station surveys

4.1.1 Ditch volumes under control and different treatments

The volume data measured at different times showed that all ditches at the experimental sites were undergoing sediment deposition during the experimental period (Table 1). The deposition volume of sediment in the ditches ranges from 152.5 to 419.4 m^3 over all years. The maximum sediment deposition happened in the ditch without VFS at STC, and the minimum occurred in the ditch with 4-m VFS at YAM. Based on the one-way ANOVA results ($p > 0.10$), there was no statistically significant difference in ditch sediment deposition among the three VFS widths (0 m, 2 m, and 4 m) from June 2020 to June 2023 within the scope of this experiment. There was also not a statistically significant interaction ($p > 0.10$) between the factor "time" and the factor "VFS width" after analyzing all ditch volumes at different measurements by a two-way repeated measures ANOVA, showing VFS efficiency has not changed significantly over time in the littoral zone of Lake Saint-Pierre.

Table 1. Ditch volume during the experimental period.

Site name	VFS width (m)	Ditch Volume (m^3)					Sediment Deposition (2020.6 - 2023.6) (m^3)
		2020.6	2021.6	2021.10	2022.6	2023.6	
YAM	0	1016.6	911.8	854.2	794.7	749.5	267.1
	2	767.8	680.4	608.6	515.5	486.1	281.7
	4	1162.2	1004.7	956.3	951.7	1009.7	152.5
STC	0	1121.8	1024.3	912.7	818.8	702.3	419.4
	2	1046.7	958.7	870.0	810.3	726.3	320.4
	4	896.0	811.8	747.7	673.8	585.2	310.8
BDF	0	*Insufficient	1339.3	1254.2	1157.6	1020.9	318.4
	2	profile	1129.1	1042.6	999.2	920.3	208.9
	4	numbers	1318.3	1279.6	1238.3	1078.3	240.0

* There are not enough profile numbers to determine the ditch volume. Because only the measurement for the first 580 meters of 1000-meter-long ditches was completed at BDF in June 2020.

However, taking the yearly data of sediment deposition volume in the ditches for all sites in the paired-samples t-test, the sediment accumulation volume in ditches treated with VFS was significantly lower than those with no-VFS ($p < 0.10$). It indicates that VFS can reduce sediment deposition in agricultural drainage ditches in the study area. The average sediment deposition in all of the 2-m and 4-m VFS ditches were 101.7 and 91.5 m^3 per year, corresponding to a reduction of 21.4% and 37.9%, respectively, compared to the control ditches without VFS (Table 2). There were also no significant differences in sediment deposition between 2-m VFS and 4-m VFS for all sites ($p > 0.10$), showing that 2-m and 4-m VFS had similar effects. However, sediment deposition in 4-m VFS ditches was about 10% less than in those of 2-m VFS.

The yearly sediment deposition volumes for each ditch varied (Table 2). The prominent sediment deposition in the ditches occurred from June 2021 to June 2022. The average deposition in June 2021 - 2022 was 135.38 m^3, compared to 103.25 m^3 in June 2020 - 2021 and 75.7 m^3 in June 2022 - 2023. The total precipitation from July 2021 to June 2022 was 899.3 mm, the driest compared to 1068.2 mm and 1082.6 mm in the same months of 2020-2021 and 2022-

2023, respectively, at the Pierreville weather station (46°05' N, 72°50' W). Probably in a drier year, less sediment in the ditches was washed down to the lake. The field sedimentation data showed a high inconsistency and variation. For example, the 2-m VFS showed contradictory effects at three sites – slightly increased sediment deposition at YAM and reduced it at STC and BDF, which may be caused by the poor growth of the 2-m VFS of YAM. The 4-m VFS ditch at the YAM site witnessed the highest sediment deposition among all three ditches in June 2020 - June 2021, while the reason is unknown.

Table 2. Yearly sediment deposition volume during the experimental period.

Site name	VFS width (m)	Sediment deposition volume (m^3)							
		2020.6 - 2021.6		2021.6 - 2022.6		2022.6 - 2023.6		Average	
YAM	0	104.8	a	117.1	a	45.2	a	89.0	a
	2	87.5	a	164.9	a	29.4	a	93.9	a
	4	157.6	a	52.9	a	-58.0*	a	50.8	a
STC	0	97.5	a	205.5	a	116.4	a	139.8	a
	2	88.0	b	148.5	b	83.9	b	106.8	b
	4	84.2	b	138.0	b	88.6	b	103.6	b
BDF	0	Insufficient		181.7	a	136.7	a	159.2	a
	2	profile		129.9	b	79.0	b	104.4	b
	4	numbers		80.0	b	160.0	b	120.0	b
Average	0	101.6		168.1		99.4		129.4	
	2	87.7		147.7		64.1		101.7	
	4	120.9		90.3		124.3		91.5	
	all	103.3		135.4		75.7		107.5	

* Negative value indicates erosion happened during that period.
Yearly sediment deposition volumes and, on average, followed by the same letters are not significantly different at $p = 0.10$.

4.1.2 Sediment accumulation during growing and non-growing season period

There was no significant difference ($p > 0.10$) between sediment deposition volumes during the two periods when the VFS widths were 0 meters, 2 meters and 4 meters. Based on the data collected from June 2021, October 2021 and June 2022, sediment deposition volume during the growing season (2021.6 - 2021.10) and non-growing season period (2021.10 - 2022.6) (Table

3) were taken into comparison. There is little difference ($p > 0.10$) in the sediment deposition volumes between the two periods as no-VFS, ranging from -15.7% to 13.4%. However, as the VFS width increases, the difference gets more prominent, and the deposition situations for the ditches vary. There are three ditches with decreasing deposition volume during the non-growing season, ranging from -90.5% to -32.6%. The deposition volume in the other three ditches showed an increasing trend of 29.9%, 15.3% and 6.7%, respectively.

Table 3. Sediment deposition volume during the growing season and non-growing season.

VFS width (m)	Site name	Sediment deposition volume (m^3)				Volume changes rate between two periods (%ΔV)
		Growing season (2021.6 - 2021.10)		Non-growing season (2021.10 - 2022.6)		
0	YAM	57.7	a	59.4	a	3.1%
	STC	111.5	a	94.0	a	-15.7%
	BDF	85.1	a	96.6	a	13.4%
	average	84.8	a	83.3	a	-1.7%
2	YAM	71.7	a	93.1	a	29.9%
	STC	88.7	a	59.8	a	-32.6%
	BDF	86.5	a	43.4	a	-49.8%
	average	82.3	a	65.4	a	-20.5%
4	YAM	48.3	b	4.6	b	-90.5%
	STC	64.1	b	73.9	b	15.3%
	BDF	38.7	b	41.3	b	6.7%
	average	50.4	b	39.9	b	-20.7%

Sediment deposition volumes followed by the same letters are not significantly different at $p = 0.10$.

4.1.3 Ditch volume change rate vs. ditch length

Based on the flow direction in the ditch during the growing season, the ditch sides going opposite to the flow were defined as the upstream in this study, shown as 0m on the x-axis in Figure 8. The annual ditch volume change rate trends with length vary at all three experimental sites, suggesting sediment deposition at different ditch lengths exhibited spatial variability.

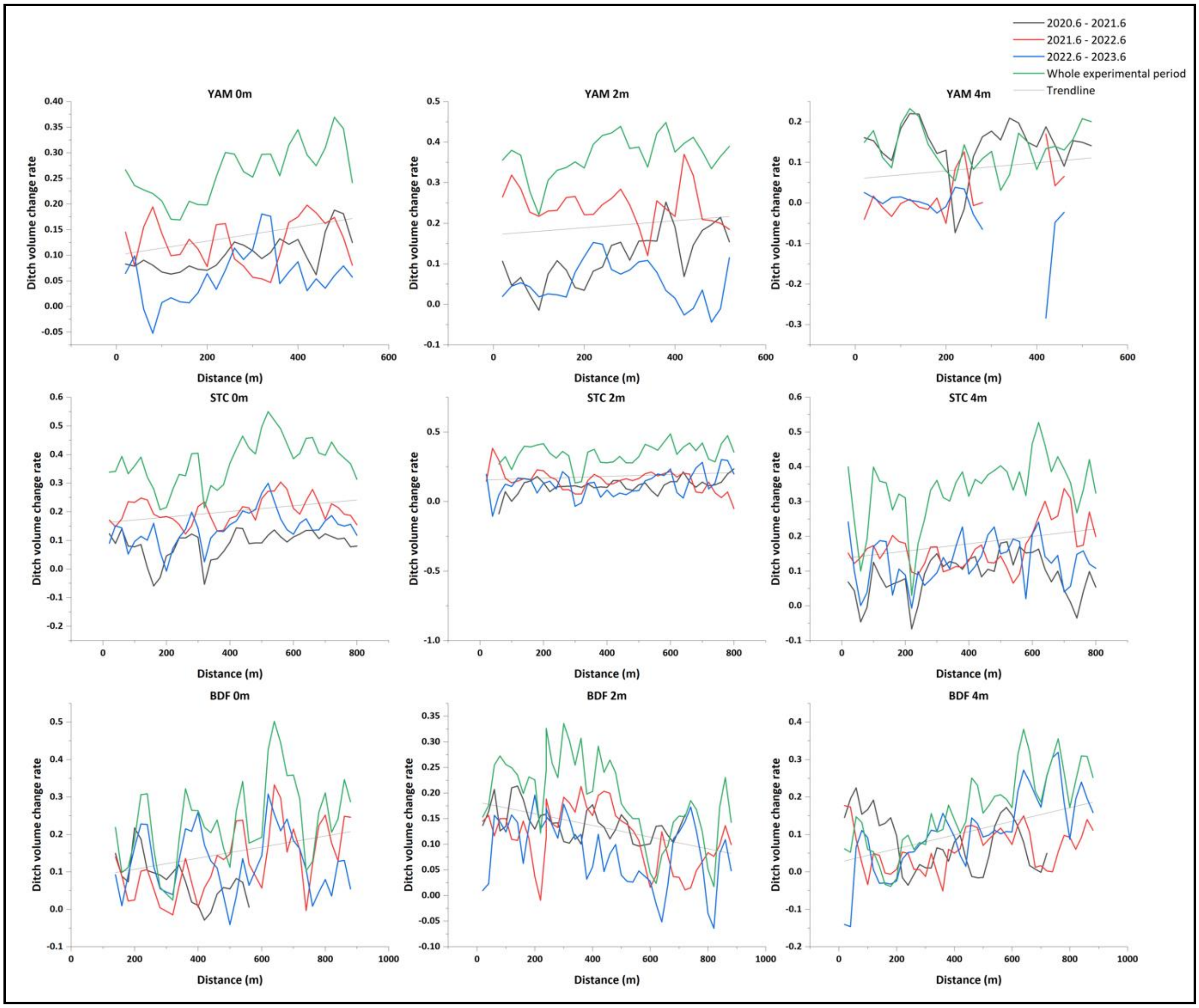

Figure 8. Ditch volume change rate vs. ditch length.

Based on the linear trendlines for the data of all periods, there is a positive association between ditch volume change rate and distance for all ditches but the one with 2-m VFS at BDF. It indicates that at YAM and STC, the ditch volume change rate increases as the ditch length increases. The ditches without VFS and with 4-m VFS both have a positive association between the volume change rate and distance at BDF. However, the volume change rate of the ditch with a 2-m VFS at BDF has a decreasing trend as the distance.

Despite the same depositional trend for all nine ditches, two different kinds of phenomenon were observed: when the change rates were positive (> 0), the ditch profiles showed a trend of getting shallower and narrower (Figure 9a), and when the rates were negative (< 0), they became shallower and broader (Figure 9b). It can be found that the second situation often occurs in the middle and second half of the ditches when the V% is below 0 in Figure 8. Although most parts of the nine drainage ditches are being deposited, erosion has occurred in some parts, and the volume change rates by erosion were within -10%.

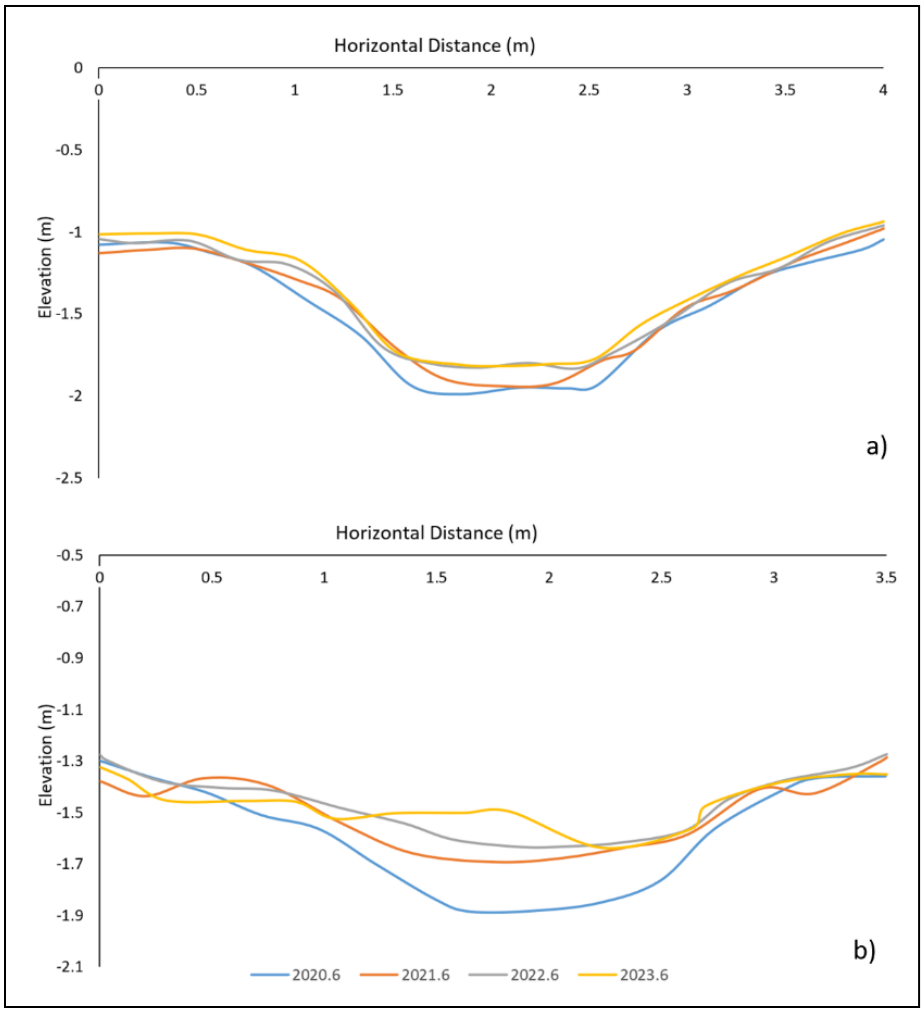

Figure 9. (a) An example of getting shallower and narrower (the cross-section of the control one at 40 meters in YAM); (b) an example of getting shallower and broader (the cross-section of the ditch with a 2-m VFS at 500 meters in YAM).

4.2 LiDAR accuracy

4.2.1 Volume comparison

The volumes of all nine ditches, determined by Total Station and LiDAR surveys conducted in October 2021, were compared using consistent width and length measurements (Table 4). The average width measurements between LiDAR and Total Station surveys are highly consistent across all sites and VFS widths. The overall average widths are 2.29 m for LiDAR surveys and 2.30 m for Total Station surveys, demonstrating a strong agreement between the two methods. However, there is some variability in volume measurements, with differences ranging from -6.64% to 27.26% depending on the site and VFS width. The overall average volumes are 688.61 m^3 for LiDAR surveys and 605.27 m^3 for Total Station surveys, resulting in an average difference of 9.46%. The relatively low average percentage difference of 9.46% and no significant difference between them ($p > 0.10$) indicate that LiDAR is reasonably accurate compared to Total Station measurements, although specific conditions or ditch characteristics might influence the accuracy.

Table 4. Comparison of ditch volumes measured by LiDAR and Total Station surveys (October 2021).

Site name	VFS width (m)	Ditch length (m)	LiDAR surveys		Total Station surveys		different %
			Average Width (m)	Volume (m^3)	Average Width (m)	Volume (m^3)	
YAM	0	540	2.20	591.42	2.22	461.40	21.98%
	2	560	2.25	300.22	2.26	320.16	-6.64%
	4	540	2.54	482.79	2.54	487.56	-0.99%
STC	0	800	1.86	290.53	1.87	277.10	4.62%
	2	800	2.04	427.82	2.01	430.09	-0.53%
	4	800	2.63	745.43	2.61	582.79	21.82%
BDF	0	780	2.53	975.25	2.55	709.37	27.26%
	2	940	2.39	1216.10	2.44	1223.96	-0.65%
	4	980	2.20	1167.92	2.23	955.00	18.23%
Average			2.29	688.61	2.30	605.27	9.46%

Based on the LiDAR data from the last three surveys, ditch volumes were determined for comparison, considering time trends (Table 5). At the YAM site, the ditch volume for a VFS width of 0 meters increased from 1025.869 m³ in 2019 to 1377.816 m³ in 2021. In contrast, the volumes for VFS widths of 2 and 4 meters showed a decreasing trend over the same period. At the STC site, the ditch volume for a VFS width of 0 meters slightly increased, while the volumes for widths of 2 and 4 meters exhibited variability but generally decreased from 2019 to 2021. The BDF site consistently showed the highest initial ditch volumes, which then decreased over time for all VFS widths. Overall, most sites demonstrate a general trend of decreasing ditch volumes from 2019 to 2021, indicating natural sedimentation processes, consistent with Total Station survey results. Due to position changes following the trenching of the ditches, the first LiDAR survey results were not used for ditch volume comparison.

Table 5. Ditch Volume Data by LiDAR.

Site name	VFS width (m)	Ditch length (m)	Ditch volume (m^3)		
			2019.11	2020.10	2021.10
YAM	0	607.5		1025.869	1377.816
	2	619	850.5235	823.1142	391.0219
	4	588	1212.448	846.3251	555.888
STC	0	818	1135.095	301.4201	310.3253
	2	797.5	1068.157	499.731	443.5798
	4	832.5	1799.316	1445.474	817.0651
BDF	0	1027.5	2786.044	1467.355	1481.924
	2	1013.5	2406.043	1708.227	1299.664
	4	1008	2235.382	1504.888	1223.949

4.2.2 Cross-section areas and point elevations comparison

All segment areas with the same length along the ditch direction as the Total Station surveys in October 2021 were compared (Table 6). When considering all sites combined, the RMSE and

MAE values are 0.46207 and 0.33363, respectively. These aggregate metrics suggest moderate accuracy across the dataset.

For the YAM site, the RMSE is 0.31331, and the MAE is 0.24467. These relatively low values indicate that LiDAR performs well, with small deviations between the LiDAR and Total Station values. The correlation coefficient R is 0.58382, suggesting a moderate to a strong positive correlation between the LiDAR and Total Station values. This correlation further supports the reliability of LiDAR for this site.

The STC site exhibits even better performance, with the lowest RMSE and MAE values among the three sites: 0.28343 and 0.21312, respectively. These metrics highlight the high accuracy of LiDAR, indicating minimal errors between the two methods. The correlation coefficient for STC is 0.49314, which, although slightly lower than YAM, still indicates a moderate positive correlation.

In contrast, the BDF site shows the highest RMSE and MAE values, at 0.64086 and 0.50228, respectively. These high values point to larger discrepancies between LiDAR and Total Station values, indicating lower accuracy compared to the other sites. The correlation coefficient for BDF is -0.02492, which is close to zero and slightly negative. This weak correlation implies that the LiDAR values are not closely aligned with the Total Station values, indicating potential challenges in using LiDAR for this specific site.

Table 6. Comparison of LiDAR cross-section calculation accuracy metrics with Total Station.

Site name	n	RMSE	MAE	r
YAM	82	0.31331	0.24467	0.58382
STC	120	0.28343	0.21312	0.49314
BDF	129	0.64086	0.50228	-0.0249
All	331	0.46207	0.33363	0.52992

`The locations of first cross-sections of each ditch are relatively the same for both LiDAR and Total Station surveys. Hence, all the first crosse-sections by the Total Station surveys were selected to compare with the point cloud elevation data of the first 0.1 meters of LiDAR data. The Y-axis of this part of the point cloud data was divided into 0.1m intervals, and the lowest elevation of all data points in the interval represents the elevation of this interval. The RMSE and MAE values of the three sites showed the high accuracy of LiDAR data with the total station geological survey data (Table 7). The used points show correlation coefficients close to 1, indicating a strong linear relationship between predicted and actual values.

Table 7. Evaluation parameters result between the LiDAR and Total Station elevation data.

	n	RMSE	MAE	r
YAM	63	0.116	0.097	0.874
STC	60	0.110	0.085	0.960
BDF	53	0.100	0.072	0.984

4.2.3 Estimated ditch maintenance time by two surveys

Based on the sedimentation results obtained by the two measurement methods, we estimated the time when each ditch would reach 60%, 70%, 80%, 90%, and 100% blockage. This estimation was used to assess potential ditch maintenance needs. The estimated time (in years) by Total Station surveys represents the period from each ditch's earliest measurement until it reaches the specified blockage levels, under the assumption of linear siltation (Table 8). The initial volumes in Table 1 were treated as 100% usable capacity.

Table 8. Estimated years for blockage by Total Station surveys

Site name	VFS width (m)	Blockage Percentage				
		60%	70%	80%	90%	100%
YAM	0	2027.05	2028.06	2029.07	2030.09	2032.01
	2	2025.05	2026.03	2027.01	2028.01	2028.08
	4	2034.03	2036.06	2038.09	2041.02	2043.05
STC	0	2025.04	2026.02	2027.01	2027.08	2028.06
	2	2026.05	2027.05	2028.04	2029.04	2030.04
	4	2025.08	2026.07	2027.05	2028.04	2029.03
BDF	0	2026.07	2027.05	2028.03	2029.02	2030.01
	2	2028.01	2029.02	2030.03	2031.03	2032.04
	4	2028.02	2029.03	2030.04	2031.05	2032.06

From the predicted blockage years for YAM, STC, and BDF under 0 m, 2 m, and 4 m VFS widths at various blockage levels (60%, 70%, 80%, 90%, 100%), increasing VFS width tends to effectively delay the progression of culvert/drainage blockage—especially in postponing complete (100%) blockage. Exceptions (e.g., YAM at 2 m) underscore that in certain locations, smaller VFS widths might trigger unfavorable hydraulic or sediment transport effects, leading to earlier blockage. Table 6 shows the estimated years (starting from November 2019) for each ditch to reach 60%, 70%, 80%, 90%, and 100% blockage as inferred from LiDAR surveys (Table 9). These estimates were based on a linear siltation assumption, calculated from the volume differences between November 2019 and October 2021. However, for the ditch at YAM with a 0-meter treatment, the ditch volume increased from November 2019 to October 2021. As a result, the linear siltation model could not be applied to estimate the blockage time for this case.

Table 9. Estimated years for blockage by LiDAR surveys.

Site name	VFS width (m)	Blockage Percentage				
		60%	70%	80%	90%	100%
YAM	0	/	/	/	/	/
	2	2022.02	2022.06	2022.11	2023.03	2023.07
	4	2022.02	2022.06	2022.11	2023.03	2023.07
STC	0	2021.07	2021.1	2022.01	2022.05	2022.08
	2	2021.12	2022.04	2022.08	2022.12	2023.04
	4	2022.01	2022.06	2022.1	2023.02	2023.07
BDF	0	2022.06	2022.11	2023.04	2023.09	2024.02
	2	2022.06	2022.12	2023.05	2023.1	2024.03
	4	2022.07	2022.12	2023.05	2023.11	2024.04

At STC and BDF, wider vegetative filter strips (2 m or 4 m) consistently help delay the progression from partial (60%, 70%, 80%, 90%) to full (100%) blockage, compared to having no VFS. At YAM, both 2 m and 4 m produce the same timeline, suggesting minimal difference in sediment interception effectiveness under LiDAR-based estimation at these widths.

Chapter 5 Discussion

Although some measurements are missing due to climate constraints and workforce shortages, the existing results can also go a long way toward explaining how well VFS performs in preventing sediment erosion and deposition in the littoral zone of Lake Saint-Pierre, Québec, Canada.

5.1 Efficiency of VFS in preventing sediment deposition and erosion

Crop rotations and the timing and magnitude of snowmelt events make it complicated to explain the relationship between VFS and sediment deposition volumes in the drainage ditches. This study showed that VFS can play a significant role in preventing sediment deposition. Consistent with prior research conducted on similar VFS types (Hussein et al., 2007; Saleh et al., 2018), 4-m VFS of three sites showed a stable and good performance in preventing sediment deposition in the littoral zone of Lake Saint-Pierre. In this study, this kind of narrow VFS can reduce an average of 31.2% of sediment deposition in the drainage ditch during the experimental period.

Nevertheless, another narrower VFS (2-m) only showed its ability to prevent sediment deposition at STC and BDF, reducing an average of 29.0% of sediment deposition. At YAM, there was 4.9% more sediment deposition in the ditch with 2-m VFS than in the control one. VFS efficiency in sediment trapping depends on many factors, like vegetation width, density, flow rates, etc. (Abu-Zreig, 2001; Dillaha III et al., 1986). Based on the flume experiments about VFS by Jin and Römkens (2001), the main factors in determining the sediment trapping efficiency are VFS density, ditch slope gradient and sediment particle size. The total sediment trapping efficiency of VFS tends to decrease significantly as the slope gradient increases (Gall et al., 2018; Jin &

Römkens, 2001; Wu et al., 2023). From the ditch volume with a 2-m VFS and its cross-sections measured in June 2020, the ditch slope gradient (2.3%) is the largest among the three ditches at YAM. Also, notably, the 2-m VFS was destroyed from the cross-section at 220 meters to the end of this ditch, which may affect the efficiency. The above reasons may explain why the ditch with a 2-m VFS acts as more sediment deposits than the control one.

Sediment deposition at individual cross-sections exhibited spatial variability, suggesting that VFS's efficiency in preventing sediment deposition varied in space. Spatial variability may result from the different degrees of runoff convergence upon entering VFS (Arora et al., 2010; Dodd and Sharpley, 2016; Hay et al., 2006). There was no statistically significant interaction between time and the reduced sediment deposition volume, meaning VFS efficiency in trapping sediments is not dependent on time. There was no significant decrease or increase in the efficiency over time during the experimental period. Several papers indicate a similar possibility of degrading VFS sediment trapping efficiency with time (Lambrechts et al., 2014; Xie et al., 2015). Hence, a more extended monitoring period is recommended.

In the areas without snowmelt flooding, net erosion or modest deposition occurs to the drainage ditches throughout the non-growing season. Conversely, the growing season is characterized by the dominance of net deposition (Jin & Römkens, 2001; Mattheus et al., 2009). It is different from the situation in the littoral zone of Lake Saint-Pierre. Snowmelt floods in Spring also contribute to the sediment deposition in the floodplains (Benedetti, 2003; Moody, 2019). The collected data from ditches without VFS showed little difference between sediment deposition during growing and non-growing seasons. However, existing data did not suggest that VFS can effectively prevent sediment deposition during non-growing seasons (there were no significant differences between reduced sediment deposition volumes in the ditches with VFS during growing and non-growing seasons).

5.2 Sediment accumulation in agricultural drainage ditches

There is a widespread belief that sediment accumulation occurs in most agricultural drainage ditches, which leads to the shallowing of the depth of ditches (Pappas & Smith, 2007). It is consistent with the results obtained in this study. During the whole experimental period, all cross-sections measured showed this trend. However, in some experimental years, parts of the cross-sections showed net erosion. This kind of slight erosion can be attributed to bulk/mass erosion (Lecce et al., 2006). The bulk erosion mainly arises from the self-mass failure occurring inside the ditch structure (Tuukkanen et al., 2016). The process of this bulk erosion gradually undermines the structural integrity of the ditch banks, leading to bank sediments' eventual failure into the ditch bed under the force of gravity (Needelman et al., 2007; Stenberg et al., 2015). Based on the mechanism above, it can be assumed that there is a possibility of significant mass failure in the drainage ditch, resulting in the occurrence of ditch bank mass failure and an increase in its width. A flatter ditch bank slope can be caused by mass failure (Huang, 2012). As shown in Figure 6b, that can explain why some parts of the ditches get shallower and broader, and the bank slope decreases.

5.3 LiDAR accuracy and consistency

When examining volume measurements, a notable average difference of 9.46% was observed between LiDAR and Total Station surveys. This difference, while moderate, highlights the inherent variability in volume estimation. The range of differences from -6.64% to 27.26% across different sites and VFS widths indicates that certain conditions or characteristics of the ditches can significantly affect measurement accuracy (Roelens et al., 2018). The overall area accuracy of LiDAR, as indicated by the moderate RMSE and MAE values, supports its use as a reliable method for ditch volume measurements. It is worth noting that the segment length of 0.5 m selected for LiDAR ditch volume calculation in this study is still relatively large. The cross-sectional area difference between two adjacent segments can even exceed 100%, which can have a significant impact on the accuracy of the cross-section area comparison. The elevation comparison in the 0.1 m segment length cross section reflects the alignment of the LiDAR with the Total Station. However, the segment length of 0.1m is not suitable for the volume calculation of the ditch due to the absence of points at the bottom of the ditch for the 0.1 m length segments. Therefore, it is feasible to improve the accuracy by more suitable segmentation length. Also, the performance discrepancies at the BDF site emphasize the importance of considering site-specific factors when selecting a measurement method. The high RMSE and MAE values at BDF indicate that LiDAR may face challenges under certain conditions, which could be due to factors such as vegetation, water presence, or variations in ditch geometry (Lin et al., 2021; Rapinel et al., 2015). The strong correlation between LiDAR and Total Station data at most sites suggests that LiDAR can be a valuable alternative to Total Station surveys, especially where high-resolution and large-area coverage are needed (Jaboyedoff et al., 2012).

However, there are significant differences between estimated years for blockage by two surveys, which can be attributed to multiple factors, including methodological discrepancies, environmental influences, and site-specific conditions. Total Station surveys provide detailed sediment deposition trends but are labor-intensive and prone to human error. In contrast, LiDAR surveys offer efficient, large-scale coverage but may be affected by vegetation, water, and ditch geometry. Total Station estimates showed delayed blockage progression, likely due to higher-resolution data, while LiDAR's faster timelines might result from underestimating sediment accumulation in sparse point cloud areas. Both methods' reliance on linear siltation assumptions may oversimplify complex sediment dynamics.

5.4 External factors affecting VFS performance

Different from the hydrologically regulated leaching plots experiments in the literature review, in the actual application of VFS in farmland, the ditch will encounter artificial damage, like the collapse of the ditch bank caused by the tractors. The artificial damage may cause area errors in the cross-section measured. The collapse of the ditch bank can cause the area of the cross-section to increase, and otters nesting in the ditch can make the area smaller. One of the main difficulties encountered during the experimental period is the maintenance of the VFS. Farmers have insufficient awareness of the protection of VFS and would destroy VFS during sowing. The growth of VFS cannot be guaranteed. For example, the farmers entirely damaged the second half of a 2-m VFS in 2023. Another point worth noting is that in actual agricultural drainage ditches, even the control ones without VFS, there is still a lot of vegetation in the ditches (Figure 10).

Figure 10. In-ditch vegetation in June 2023 (the ditch with 4-m VFS at YAM).

The in-ditch vegetation growth could affect the performance of the VFS in the littoral zones, especially during the agricultural drainage periods (Copper et al., 2004). The seasonal growth of aquatic vegetation in the ditches can increase the hydraulic roughness of the ditches by an order of magnitude, dramatically affecting the sediment trap efficiencies (Watson, 1987). Hence, the sediment trap efficiencies of VFS can be linked to the in-ditch vegetation density in future studies. Moreover, beavers were found nesting upstream of the 2-m VFS ditch at BDF in 2022 and 2023 (Figure 11), which may have influenced the trend of ditch volume change rate vs. ditch length.

Figure 11. Beavers nest the upstream of the ditch with 2-m VFS at BDF.

Chapter 6 Conclusion

In this study, VFS has been shown to reduce sediment deposition of agricultural drainage ditch in the littoral zone of Lake Saint-Pierre. During the experimental period, the 2-m VFS showed sediment removal rates of -5.5%, 23.6%, and 34.4%, respectively, compared to the control ones. The ditch with this kind of VFS at YAM had more deposits (4.9 m^3 per year). For STC and BDF, 2-m VFS reduced sediment deposits by 33.0 and 54.8 m^3 per year ($p < 0.10$). Moreover, the 4-m VFS reduced 42.9%, 25.9% and 24.6% sediment deposition compared to the control ones ($p < 0.10$). The 4-m VFS at three sites showed a similar removal rate, decreasing 38.2, 36.2, and 39.2 m^3 per year. Moreover, as VFS width increased, the average reduction in sediment deposition became more prominent. The 4-m VFS reduced sediment deposits by 10.1% more than the 2-m ones. However, VFS did not perform well in preventing sediment deposition during non-growing seasons ($p > 0.10$). Three ditches with VFS exhibited a decrease in deposition volume during the non-growing seasons compared to the deposition volume during the growing season, at an average of 32.7%. In contrast, three ditches with VFS showed an increasing trend in deposition volume during the non-growing seasons, by 6.7%, 13.4%, and 15.3%, respectively. In future studies, in-ditch vegetation density can be considered a factor affecting VFS performance, especially during growing seasons.

Moreover, both LiDAR and Total Station surveys offer valuable tools for measuring ditch volumes, with LiDAR demonstrating generally strong accuracy and consistency. The variability in performance across different sites highlights the need for careful consideration of site characteristics and methodological limitations. Future work could focus on addressing the factors affecting LiDAR accuracy at challenging sites and exploring ways to enhance its reliability in various conditions. The study also highlights the importance of ongoing monitoring and further research to better understand the long-term effects of VFS. It is also significant to address the practical challenges associated with their implementation in real-world agricultural settings caused by considerable amounts of land and the required vegetation management.

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