EFFECT OF VEGETATIVE FILTER STRIP ON SEDIMENT DEPOSITION IN AGRICULTURAL DRAINAGE DITCHES IN THE LITTORAL ZONE OF LAC SAINT- PIERRE

Birkhoff Li
Department of Bioresource Engineering,
McGill University,
Montreal

August 2022

A thesis submitted to McGill University in partial fulfillment of the requirements of the degree of **Master of Science** in **Bioresource Engineering**

Abstract

Sedimentation in agricultural drainage ditches has created a complex environmental, economic, and ecological issue in the littoral zone of floodplains, especially in the Lac Saint-Pierre region of Quebec, Canada. The vegetative filter strip (VFS), a best management practice (BMP) for agricultural drainage ditch, could be one of the prominent solutions for this nexus of issues in the Lac Sainte-Pierre region. The focus of this study is to analyze the role of the vegetative filter strip's performance on sediment deposition prevention in the Lac Saint-Pierre area through field monitoring of three experiment sites across the Lac Saint-Pierre shoreline using total stations and Light Detection and Ranging (LiDAR). Three different filter strip widths (0, 2, and 4m) are set as treatments and planted across all experiment sites, and the ditch volume change is monitored from three years onward from ditch dredging. Our results suggest that there is no significant difference in sediment deposition across different widths of VFS, indicating a limited impact of vegetative filter strip on sediment deposition in the littoral zone of the Lac Saint-Pierre region in the first 3 years. Also, the performance of the LiDAR system in the densely vegetated drainage ditch condition is tested by comparing the elevation, and drainage ditch cross-section profile area measured using the LiDAR survey. To those using the total station survey. Both LiDAR elevation and profile cross-section area measured by the LiDAR scan show relatively large errors compared to the total station data.

Résumé

La sédimentation dans les fossés de drainage agricole a créé un problème environnemental, économique et écologique complexe dans la zone côtière des plaines inondables, en particulier dans la région du Lac Saint-Pierre au Québec, Canada. La bande filtrante végétative (VFS), une meilleure pratique de gestion (BMP) pour les fossés de drainage agricole, pourrait être l'une des principales solutions à ce problème dans la région du Lac Saint-Pierre. L'objectif de cette étude est d'analyser le rôle de la performance de la bande filtrante végétale sur la prévention du dépôt de sédiments dans la région du lac Saint-Pierre par le biais d'un suivi sur le terrain de trois sites expérimentaux sur la rive du lac Saint-Pierre à l'aide de stations totales et de systèmes de détection et de télémétrie par la lumière (Lidar). Trois bandes filtrantes de différentes largeurs (0, 2 et 4 m) sont utilisées comme traitements et plantées sur tous les sites d'expérimentation, et le changement de volume du fossé est surveillé à partir de trois ans après le dragage du fossé. Nos résultats suggèrent qu'il n'y a pas de différence significative dans le dépôt de sédiments à travers les différentes largeurs de VFS, indiquant un impact limité de la bande filtrante végétale sur le dépôt de sédiments dans la zone côtière de la région du Lac Saint-Pierre au cours des 3 premières années. De plus, la performance du système Lidar dans les conditions d'un fossé de drainage à végétation dense est testée en comparant l'élévation et la superficie du profil de la section transversale du fossé de drainage mesurées à l'aide du levé Lidar. À celles mesurées par la station totale. L'élévation et l'aire de la section transversale du profil mesurées par le balayage Lidar montrent des erreurs relativement importantes par rapport aux données de la station totale.

Acknowledgement

This study was made possible by the financial support of MAPAQ, which is part of MAPAQ's sustainable management of the Lac Saint-Pierre shoreline. I would first like to extend my deep appreciation to my supervisor Dr. Zhiming Qi. I appreciate his patience, passion and vast knowledge of scientific and Chinese literature. Although this project was in japery for most of its life span, he always supported me during the project and beyond.

I am grateful to the researcher at the University of Laval under Dr. Monique Poulin’s management. Their passion and tremendous effort in planting and managing the vegetative filter strip make this research possible.

I want to offer a special thanks to Dr. Shiv Prasher, my master’s study committee member, for providing guidance even during his sabbatical.

I would also like to acknowledge Mr. Zhang Fan's contribution to this project since I inherited his data and project from 2021 onward. I wish him the best on his new journey back home.

A special thanks go to the summer students and lab colleagues who spent days in the field helping me conduct the total station survey.

Finally, I want to thank Mr. Ziwei Li and Ms. Yirong Zhang for their encouragement and emotional support.

Contribution of Authors

All manuscripts present in this thesis were authored by Youjia Li (first author), Dr. Zhiming Qi (second author, supervisor), Dr. Monique Poulin (third author) and Dr. Shiv Prasher (fourth author). This study was inherited from Mr. Zhang Fan, which conducted the experiment design and ditch sedimentation monitoring from 2019 – 2020. From 2021 onward, experiment design, ditch sedimentation monitoring, related statistical analysis, and all part of manuscripts are written by Youjia Li. Dr. Zhiming Qi provided financial support and a significant supervising role in the research. Dr. Monique Poulin and her research group have established vegetative filter strips for all studied drainage ditches. Dr. Shiv Prasher supervised the research and provided conclusive feedback to the research.

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Chapter 1 Introduction

The Problem

The sediment-related problem in agricultural drainage ditch has developed into a severe problem economically, environmentally, and ecologically in the Lac Saint-Pierre region (LSP), Quebec (TCRLSP, 2017). The high sediment influx from the snowmelt-originated seasonal inundation (Cahoon and Reed, 1995; Leonard, 1997; Leonard and Luther, 1995; Palinkas and Engelhardt, 2019)and intensive agricultural practices, combined with sedimentation-encouraged hydrometric features (low-velocity water flow) (Bowmer et al., 1994; Jiang et al., 2007; Le Nguyen and Vo Luong, 2019), have created sediment buildups in agricultural drainage ditches.

The sediment buildups have led the agricultural drainage ditch prone to being clogged, where frequent ditch dredging is required to maintain sufficient drainage for agricultural production (Smith and Pappas, 2007). Dredging is costly and burdens farmers economically (Rein, 1999). On the environmental side, the St. Lawrence River, a major receiving water body for the LSP drainage ditch, is reported to have severe nonpoint source pollution, resulting in its eutrophication and phytoplankton bloom events (Blann et al., 2009). These cataphoric events have been proven to correlate with both the suspended and the dissolved form of ditch sediments at LSP(Hudon and Carignan, 2008; Rondeau et al., 2000). Ecologically, the clogged ditch has also weakened the local floodplain river fishes' reproduction ability by deteriorating the hydrologic connectivity between the floodplain and permanent water bodies, leading to a multi-decade-long population decay of its local flagship species (i.e., yellow perch) (Foubert et al., 2020; TCRLSP, 2017).

Justification

The concept of the littoral zone (figure 1.1) is referred to as the shallow-water, near-shore region of the lake, interfacing between the land and water(Ostendorp et al., 2004; Wetzel, 2001). Such a concept differentiates the periodic inundation area in the shore zone from the rest of the land-river interface (i.e., the Riparian zone) (Dąbrowska et al., 2016).

In agro-environmental literature, researchers rarely differentiate the littoral zone from the rest of the shoreline (Geng et al., 2017). Commonly, the littoral zone VFSs (Vegetative filter strip) are treated and referred to as riparian zone VFSs (Lauvernet and Muñoz-Carpena, 2018; Muñoz-Carpena et al., 2018)This mixed-up between the littoral and riparian zones could create a false expectation of VFS's efficiency in the littoral zone due to the significant hydrogeomorphic mechanisms difference between itself and the riparian zone (Shellberg et al., 2013). The frequent backwater occurrences and the overbank flow usually shifted the major sedimentation inducement from incremental rainfall-infiltration erosion to subtle gully mass failures (Jin et al., 2016; Shellberg et al., 2013), which could cause VFSs to have a limited interference on the sedimentation.

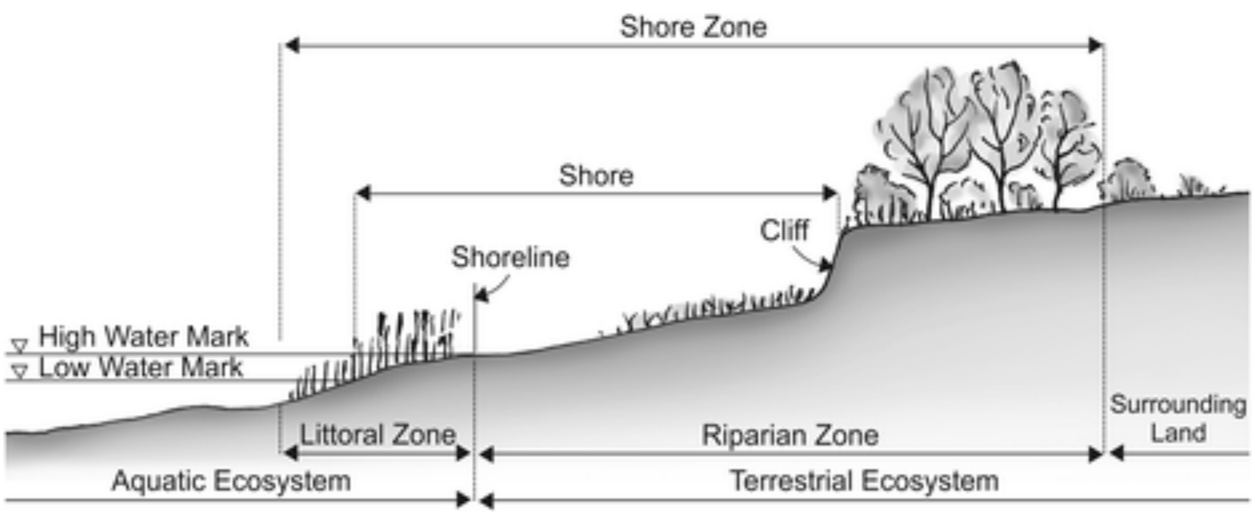

Figure 1.1 the definition of zones and boundaries in a shore zone cross-section (Dąbrowska et al., 2016)

To determine whether VFS's sedimentation interference capability is still adequate in the littoral zone, a field study monitors gully (i.e., agricultural drainage ditch) 's sedimentation over multiple turns of the littoral zone's hydrological cycle is needed. Also, solving this problem will have substantial environmental, social and economic benefits for the agricultural production in the littoral zone of the floodplain area.

A great example is the agricultural drainage ditch in the Lac Saint-Pierre region. Primarily located in the littoral zone, these ditches are the major receiver of crop field-originated sediments (Blann et al., 2009). These sediments clogged the drainage ditch, causing agricultural producers to dredge the drainage channel to maintain an adequate drainage capability (Rein, 1999).

Sediment, a pollutant, also deteriorates the living environment for the in-situ aquatic animal (Foubert et al., 2020; Hudon and Carignan, 2008; Rondeau et al., 2000; TCRLSP, 2017).. On top of that, it binds with field contaminantcontaminant(e.g., nonpoint source pollutant, pesticide, etc.) and travels to the received waterbodies (i.e., St. Laurance River) through the hydrological cycle.

The bounded field contaminantcontaminant can be released into the receiving water bodies, leading to eutrophication and other water quality issues (Hudon and Carignan, 2008; Rondeau et al., 2000). Therefore, actions to limit the sediment deposition in drainage ditches must take place in the littoral zone of the Lac Saint-Pierre region before further damage is done.

Proving that VFSs could minimize the sedimentation deposition in the Lac Saint-Pierre region's littoral zone could alleviate the Lac Saint-Pierre region's sediment-related problems. Hence, this thesis aims to assess the vegetative filter strip's effectiveness under the typical LSP littoral zone agricultural production environment for drainage ditch sedimentation prevention by conducting a field experiment. This field experiment has three different VFS width settings (4m, 2m, and 0m (control)), though monitoring the sediment deposition volume by calculating the change in ditch volume through a series of total station and LiDAR topographic surveys. Also, we explore and examine the capability of the UAV-based LiDAR technology under densely vegetative conditions to investigate whether UAV-based LiDAR could be an adequate method for ditch sedimentation monitoring.

EXPERIMENT SETTING

A total of three sites in the coastal region of Lac Sainte-Pierre region near St. Cuthbert (46.130°N, 73.124°W), Yamachiche (46.268°N, 72.864°W), and Baie-du-Febvre (46.143° N, 72.713° W) have been used for this study (figure 1.2.a). These three sites will be abbreviated STC (Site near St. Cuthbert), YAM (Site near Yamachiche), and BDF (Site near Baie-du-Febvre) in this thesis. The above experiment sites (STC, YAM, BDF) represent a typical LSP agricultural operation setting, with an annual soybean-corn rotation and fall tillage. And all three sites have an average annual precipitation of around 1000mm.

All monitoring ditches conducted a dredging process by local contractors in 2019 Nov. (figure 1.2.b), with different widths of VFS planted alongside the adjacent area between the ditch and cropland in spring 2020 (figure 1.2.c).

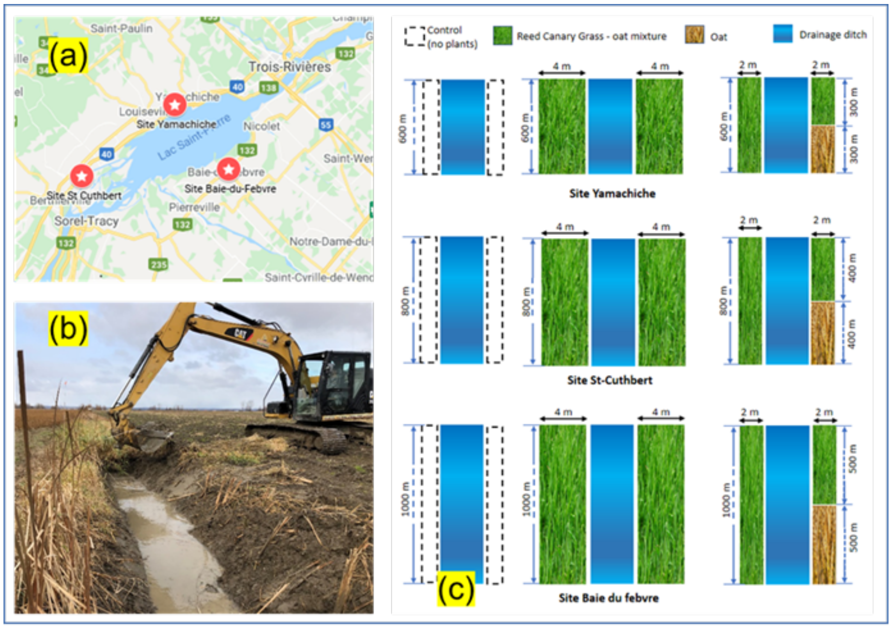

Figure 1.2 the location of three experiment sites (a), the dredging process in 2019 (b), and the layout of the VFS (c).

OBJECTIVE

This work aimed to assess the possibility of using the VFS as a best management practice and its related monitoring method to encounter the sediment deposition problem in the Lac Saint-Pierre region in Quebec, providing viable information for policymakers.

The primary objectives were:

1. Evaluate whether VFS significantly affects sedimentation deposition prevention in the coastal area of the Lac Saint-Pierre region.
2. Evaluate LiDAR's accuracy compared to existing field survey-based ditch sedimentation monitor methods.

THESIS FORMAT

Chapter two is a literature review of studies addressing: a) the issue caused by sedimentation, b) how vegetative filter strips could ease the sedimentation process, c) UAV-based LiDAR technology, and d) the potential error source of UAV-based LiDAR under the densely vegetated conditions.

The first objective is presented in Chapter three, entitled “Effect of vegetative filter strip on sediment deposition in drainage ditches in the littoral zone of Lac Saint-Pierre”. This paper discusses the sedimentation prevention performance of the vegetative filter strip and whether different vegetative filter strip width configurations could affect the sedimentation of the drainage ditch.

The second objective is presented in Chapter four, entitled 'Utilizing unmanned aerial vehicle-based light detection ranging technology in agricultural drainage ditch sedimentation monitoring.' This paper presents the LiDAR survey result of drainage ditch sedimentation over three experiment fields, comparing the LiDAR survey result to the total station and identifying the possible error source of the LiDAR technology.

Chapter 2 REVIEW OF THE RELEVANT LITERATURE

AGRICULTURAL DRAINAGE DITCH AND SEDIMENTATION PROCESS

In humid and sub-humid regions of North America (i.e., Eastern Canada, Eastern United States, and most Midwestern United States), agricultural drainage ditch is considered a vital agricultural infrastructure to fulfill the need for field moisture control. Besides preventing excessive moisture from damaging the field crop, the appearance of these ditches also enhances the field vehicle mobility during the spring, enabling earlier tillage and seedbed preparation (Madramootoo et al., 2007)

The drainage ditch requires dredging to maintain its drainage ability as sediment buildup from the erosion events (Smith and Pappas, 2007). The maintenance event is performed and paid for by farmers or local government agencies who perceive that drainage water is not efficiently removed from the field. The dredging frequency can be varied from every 5 years to every 50 years (Smith and Pappas, 2007). The dredging process is usually considered expensive, becoming a financial burden for the agricultural producer (Rein, 1999).

The process of sediment accumulation in the agricultural drainage ditch is commonly referred to as sedimentation (Ostrovsky et al., 2014). As erosion agents (water, wind, ice and waves) have detached materials from the topsoil or ditch banks during the soil erosion. These detached materials (i.e., sediment) are eventually deposited into a new environment where the erosion agent cannot carry the detached material's weight.

Despite sedimentation as a natural process, it is common knowledge that anthropogenic activities (i.e., agricultural operations) could accelerate soil erosion and intensify sedimentation (Pimentel et al., 1995), resulting in more frequent dredging. On top of that, sediment is usually considered a pollutant due to its carrier capability of field pollutants (e.g., nonpoint source pollutants, chemicals, pathogens, etc.). In considering these factors and under the general background of global warming and intensification of agricultural practices, significant effort has been invested into the study of moderating the sedimentation process, aiming to create a more sustainable food and fiber production environment.

AGRICULTURAL DRAINAGE-RELATED SEDIMENTATION ORIENTATED ISSUES

The sediment-orientated issues can bring a nexus of impact on the agroecosystem's economic, ecological, and environmental wellbeing (Bang et al., 2013). The sediment's physical presence could deteriorate the hydrological connectivity between the drainage ditch and its received waterbodies. Such deterioration of hydrological connectivity will eventually be developed to a stage where the ditch's drainage capability cannot fulfil the agricultural production requirement. Eventually, the ditch will be clogged and requires dredging (Smith and Pappas, 2007) (Smith and Pappas, 2007); due to the sediment-originated deterioration activities.

The deterioration of hydrological connectivity can also create a negative cascade effect on floodplain river fish. This type of migrated fish travels between the permanent water bodies (i.e., river and lake) and seasonal water bodies (i.e., agricultural drainage ditch) following the aquatic-terrestrial environment dynamics in the floodplain (Foubert et al., 2020; Hudon and Carignan, 2008). As the hydrological connectivity aggravates, the aquatic-terrestrial shift process of the agricultural drainage ditch accelerates. Correspondingly, this acceleration narrows the time window of aquatic-terrestrial transition in ditches, repressing the success rate for aquatic life seasonal-permanent waterbodies migration. The low success rate is reflected via high occurrences of stranded and dead fish larvae reported from field surveys (Foubert et al., 2020). Such low larval survival rate influences fish's reproduction capability, eventually destabilizing the population and causing low-scale specie's extinction. In extreme cases, it can create a chain reaction across the food web, especially among the wetland bird species (Keppeler et al., 2016).

Sediments from agricultural drainage ditches also affect the environment of their received waterbodies. It can create both direct and indirect toxicity to the related system through suspended and dissolved forms.

Phosphorus (P), the key pollutant for surface water eutrophication, is heavily associated with sediment. P from organic or inorganic fertilizer can be easily bonded to sediment by the selective adsorption mechanism through its P-favorable geochemical and physicochemical characteristics (i.e., metal (Fe and Al) oxide formation, high organic matter concentration) (Asomaning, 2020).. The P-bonded fine particles (particulate P (PP)) are then transported into water bodies as sediment through erosion. These particles (PP) will serve as bioavailable P to alga (the major contributor to eutrophication) either through its original form or dissolved form (dissolved P (DP)) by an oxic-anoxic interface transformation.

Besides P, nitrogen (N), another major pollutant in nonpoint source pollution (NSP), is also associated with the erosion-originated sediment input. In heavily eroded areas, research has shown that the erosion-related particles N (PN) contribute a substantial portion of the total N export during the storm, alone side with dissolved N (consensus believed primary N source) (Inamdar et al., 2015). The PN from erosion was transported into water bodies through the hydrological process, then undergoes physical (desorption) or biogeochemical (e.g., nitrification and denitrification) process into the biologically available forms. The abundant influx of biologically available N prompts algae growth, a major water quality concern.

THE VEGETATIVE FILTER STRIPS (VFS)

Vegetative filter strip (VFS) or vegetative buffer strip (VBS) is often established as a conservation practice or best management practice (BMP) to reduce sediment deposition in waterways (Munoz-Carpena et al., 1999). Through planting with dense vegetation in the adjacent area between the crop field and drainage ditch, these dense vegetations were proven to significantly moderate both the magnitude and impact of sedimentation (Baltensweiler et al., 2017).

The mechanism of VFS is well studied and can be summarized into two major aspects: a) VFS's root structure-related mechanism and b) VFS's above-ground structure-related mechanism.

The densely grown VFS root structure improves the infiltration capacity of the topsoil by altering the soil structure (Hatchett et al., 2006). Evidence (Sheng et al., 2021) has shown that as the root structure of the VFS develops, the porosity of the topsoil layer increases. Also, because of the absence of pesticides and tillage in VFS compared to the cultivated areas, animals (i.e., rodents, moles and earthworms) tends to have a higher presence in the VFS. Animal support macrofauna activity, thus forming rapid flow paths in the soil structures (García-Serrana et al., 2017). Together, the increased porosity and macrofauna increase the hydrologic conductivity of the soil, leading moisture to infiltrate into the deeper soil layer rather than becoming the overland flow (a.k.a. runoff) (the determining force for causing sediment formation).

The VFS above-ground structure intercepts and absorbs the overland flow. When the grass foliage intercepts the overland flow, it creates drags. The drag lowers the speed of the runoff (Jin and J. M. Römkens, 2001 (Jin and Römkens, 2001). Such a speed-reducing effect decreases the turbulent force in the runoff, the major carrying force for sediment transportation. The declining force sediment to be deposited into the VFS rather than the receiving waterbodies. More than that, the above-ground structure of VFS and debris have been indicated as an excellent absorbent for sediment. On top of it, the organic matter increased from VFS's development is proven to have a stabilizing capability for particle aggregate (a major source of sediment formation) (Veum et al., 2012).

Besides reducing the sediment deposition in waterways, VFS also remediate sediment's impact on its connected ecosystems. It is proven to uptake liable nutrients from the field, with evidence of VFS's significant biomass increase compared to the control experiment (i.e., similar vegetation grown in identical climate condition in a non-VFS setting) (Parmeland, 1995; Dorioz et al., 2006).These uptake nutrients contribute to VFS's growing, and eventually return to the field in the form of litter, providing bio-available nutrient for crops (Kieta et al., 2018). In brief, the VFS modifies both the timing and the form of the nutrient out flux from the cropland to the associate waterway. This result less field nutrient being bound to sediment, minimize its impact.

To this day, VFS is a common practice for sediment retention best management practices. With its low labour, time and capital costs compared with similar functioned BMPs, VFS does not disturb the harvest schedule for farmers. These advantages have led VFS to be recommended in the USA and other regions, with certain parts of Europe making it mandatory alone reivers.

VFS WIDTH EFFECT ON SEDIMENTATION PREVENTION

VFS width is one of the most important design factors for sediment trapping (Anebagilu et al., 2021). Furthermore, some studies (Stutter et al., 2021) show a positive correlation between the VFS width and sediment trapping efficiency within a certain width range. The addressed correlation is attributed to VFS's greater opportunity for infiltration or deposition (or both) at a wider width.

As the VFS width increases, more drag is generated for sediment transportation. An increase in drag will offset the gravity-countered turbulent-originated force, forcing more sediment to deposit into the VFS, especially the sediment with smaller dimensions (Jin and Römkens, 2001). However, as VFS width increases, a diminishing improvement in the sediment trapping performance is found due to the non-linear response of drag coefficient to the VFS width.

However, under-sized buffers will provide inadequate protection for water bodies, especially to smaller particles (i.e., clay), which significantly influence the water quality based on their high absorption capabilities.

Finding an optimal width is critical for VFS design, where an undersized buffer strip width will provide insufficient protection for sedimentation. Still, a larger buffer strip width removes land from production, resulting in economic loss (Anebagilu et al., 2021). Therefore, finding the optional VFS width is becoming a critical task for any VFS deployment.

Unfortunately, a universal optimum width for buffers does not exist due to the highly dependent nature of VFS performance on soil, vegetation and climate conditions (Lacas et al., 2005). Numerous attempts to model the relationship (Hayes et al., 1979; Muñoz-Carpena et al., 2018; Munoz-Carpena and Parsons, 2004; Munoz-Carpena et al., 1999) between the VFS width and sediment transportation have been made.

However, high variation of surface roughness across different vegetation conditions. Such variance is due to the highly complex nature of turbulence flow and the involvement of infiltration in sediment transportation. VFS models usually implement empirical equations, resulting in an insufficient estimation for real-life usage. For those models which integrate physical-based processes, the required data inputs are usually unrealistic to obtain in the field condition. These model limitations make most research rely on real-life experiments, making pilot-scale VFS field experiments a popular method for identifying the optimum VFS width.

CONCERNS ABOUT VFS USAGE ON SEDIMENTATION MODERATION UNDER THE LITTORAL CONDITION

VFS sedimentation moderation performance (abbreviated as VFSs performance) can vary highly over different deployment environments (Fox et al., 2016). Namely, the environment can influence VFSs infiltration enhancement ability and impact moderation effect towards the soil erosion medium.

For this reason, in drainage studies, researchers have focused on investigating different VFSs deployment environments' hydrologic characteristics and their effect on sedimentation moderation(Cooper et al., 2004; Dabney et al., 2006; Helmers et al., 2008). However, little research has been conducted on littoral condition VFSs due to the neglect of the difference between the riparian and littoral zones in agri-environmental studies.

In parallel to the littoral zone, the VFSs in the riparian zone gain much attention. Besides the reason for being a popular BMPs in such conditions, the additional impact of the soil surface dry-wet alternation cycle has drawn curiosity. The dry-wet alternation cycles (Song et al., 2022) caused by the groundwater seepage or field moisture change can destabilize the gully structure and result in bulk erosion fueling the sedimentation. As the result of bulk erosion intensification, reports show that VFSs performance could be retarred; but still create noticeable sedimentation prevention results in most experiment cases (Greenwood et al., 2018).

Despite a similar nature in the soil dry-wet alteration cycle and low groundwater levels between the littoral and riparian conditions, the frequent and more prolonged inundation leads to hydrogeomorphic related process (i.e., river backwater and overbank flood) in the littoral condition is more intense than the riparian zones. Concluding littoral VFSs performance based on the current studies on riparian VFSs is insufficient to close the knowledge gaps.

Due to the complex nature of river backwater (Jin et al., 2016) and overbank flood and the temporal and spatial variance in the hydrologic condition in littoral settings, both models and lab experiment also shows insufficiency in mincing the littoral hydrologic conditions and creating a strong need for long-time field VFS experiments in the littoral zone.

NEEDS OF SEDIMENTATION MONITORING IN LITTORAL AGRICULTURAL DRAINAGE DITCH ENVIRONMENT AND CURRENT PROGRESS IN SEDIMENTATION MONITORING

Sedimentation, the process of sediment mobilization and deposition, plays a vital role in many ecosystems. Especially in the floodplain ecosystem, sediment transports nutrients between the aquatic and terrestrial environments (Felipe-Lucia and Comín, 2015; Posthumus et al., 2010; Tockner and Stanford, 2002). As a result of this source-sink transportation, a large amount of organic matter has accumulated in the littoral zone of the floodplain (Battin et al., 2008; Roach et al., 2014).

The sediment dynamic in the littoral zone has recently attracted some researchers' attention since anthropogenic alternation of the littoral zone has intensified, especially in the agricultural production sector (Taylor et al., 2019). Quantifying sedimentation (i.e., sediment deposition quantity) in the agricultural drainage ditch, a major sediment sink in the agroecosystem, is essential for understanding any sedimentation-related cycle in the agricultural floodplain setting (Repasch et al., 2020). Sediment-related cycles, such as the soil geo-weathering cycle and soil biogeochemical cycle, are vital to our food security (Lal, 2007). Therefore, this quantification could also provide essential information for agricultural watershed management.

Quantifying sedimentation in the littoral agricultural drainage environment is challenging; the standard sedimentation monitoring method (i.e., sedimentary budget) (Warrick and Milliman, 2003) and its inlet and outlet focus estimation process have fundamentally defected under the periodic aquatic-terrestrial alternation.

Most of the sedimentary budget relies on monitoring data on sediment concentration difference and the water volume difference between the hydrological inlet and outlet of the quantifying environment (Rosati, 2005). The concentration is usually measured through the total suspended solids (TSS) values of the water influx and outflux. This method requires water samples to be drawn in high frequency for TSS measurement or to deploy water turbidity sensors. The latter approach requires an accurate correlation function between the water turbidity and TSS content (Rügner et al., 2013), which can be highly varied between different watershed or environment conditions.

Monitoring the TSS in ultra-low water conditions during the dry season and monitoring the water volume during the soil inundation period is extremely hard in a littoral zone drainage ditch setting. As the hydrologic inlet and outlet cannot be confined into a known geometry, the artificial flow control (gated flumes) and runoff estimation usually occur in the littoral ditch settings. The gated flumes affect the hydrological cycle, a sediment determination aspect in the littoral environment. Runoff estimation usually relies on either computational simulation or empirical estimation equation in the combination of small-scale runoff monitoring in an environment-mimicking control condition (Song et al., 2022).

The amount of sensor and estimation involved in the sedimentary budget quickly escalates the complexity and data uncertainty (Cappucci et al., 2020)in the littoral agricultural conditions, making it unviable for many littoral agricultural drainage ditch settings. Needless to say, the data usually lack spatial resolution since only one data point corresponds to one set of hydrological inlet-outlet.

Geomorphology budget is another method for estimating sedimentation; by comparing the topographic difference between the drainage ditch environment over a period, the amount of sediment deposition can be accurately calculated. Geological surveying, either remotely LiDAR (Notebaert et al., 2009), SAR-satellites (Argyilan et al., 2005) radio, photogrammetric(Marzolff and Poesen, 2009); or physically (GNSS positing or total station survey), must be directly conducted in the monitoring environments (Capoane et al., 2015). However, the periodic inundation limits both the survey time and accessibility of the physical survey, forcing unprecedented labour requirements for the geomorphology budget method, limiting its usage and spital coverage.

UAV-LIDAR: THE PROMINENT METHOD FOR LITTORAL DITCH ENVIRONMENT SEDIMENTATION MONITORING

In recent years, remote sensing and bathymetric survey development have made remote sensing an attractive solution for topographic mapping of the shallow-water environment (i.e., littoral zone) (Kearns and Breman, 2010). However, considering the centimetre or even millimetre magnitude of topographic alternation created by the sedimentation, most bathymetric capable remote sensing solutions are incapable due to their low accuracy (Gao, 2009). The high accuracy feature of LiDAR (Light detection and ranging technology) leads itself to become one of the few prominent solutions for littoral environment topographic mapping.

By measuring the time difference between the emitting and receiving laser beam and recording the orientation and sensor positing location, LiDAR can reach a centimetre or even millimetre level of accuracy (Legleiter, 2012). The millimetre and micrometre level of light foot print and high sampling density enables lights to get the ground structure between the coverage of the foliage, hence having a vegetation penetration effect. Under a particular wavelength, LiDAR could acquire bathymetric survey capability with its laser can penetrate shallow water and reach the beds of the waterbodies (Legleiter, 2012). Compared to other remote-sensing technologies such as sonar, it is a versatile, cost-effective and detailed method.

However, using LiDAR to quantify sediment deposition in the littoral agricultural drainage ditch setting is still a relatively new area for most researchers due to the high investment requirement on the aircraft vehicle and LiDAR sensor. Also, most littoral agricultural drainage ditches are densely vegetated, requiring a higher LiDAR data point density to represent the topographic accurately; achieving such data point density will receive a flight height that is impossible for manned aircraft vehicles to achieve (Mandlburger et al., 2015).

In recent years, the availability and the load capability of a commercially unmanned aircraft vehicle (UAV) have greatly improved, making it possible to serve as an aerial platform for LiDAR sensors. Due to its low speed and low flight attitude nature, higher LiDAR data point density can be achieved (Lin et al., 2019), posting an opportune for highly spatial detailed sediment deposition quantification in the littoral ditch setting, raising the need for an accurate assessment of the UAV-based LiDAR sediment monitorization, and filling the knowledge gap of how higher density LiDAR point data will affect the vegetation penetration capability in the littoral area. Further exploration of these topics could point to the research direction of the LiDAR sensor development and data processing procedure for UAV-based LiDAR in littoral settings.

Chapter 3 EFFECT OF VEGETATIVE FILTER STRIP ON SEDIMENT DEPOSITION IN DRAINAGE DITCHES IN THE LITTORAL ZONE OF LAC SAINT-PIERRE

INTRODUCTION

The intensification of modern agricultural practices was a result of meeting the increasing demand of global food consumption (Godfray et al., 2010; Foley et al., 2011). This intensification accelerates soil erosion, where topsoil materials being detached, and deposits into the connected environment. Sediment, the detached materials, causes both direct (i.e., high affinity to field contaminants) and indirect disturbance (i.e., deterioration on hydrologic connectivity) to the ecosystem. The sediment-related contamination is considered as a major obstruction to the agricultural sustainability (Pimentel et al., 1995; Wijetunge and Sleath, 1998).

The agricultural drainage ditch, a primary sediment sink, is the frontline for sediment to interact with the agroecosystem. Any sediment dynamic change in these drainage ditch could cause rippling effect to the ecosystem. An effect that can impact its environment, economy and ecology (Blann et al., 2009). Such a broad impact on the ecosystem is attributed the hydrological junction role of the drainage ditch, where it connects the upland crop field and waterbodies. One example of this impact can be found at the littoral zone at Lac Saint-Pierre region (LSP), Quebec.

Undergoes an intensified agricultural practice and a sedimentation favourable hydrological condition (i.e., periodic land inundation (Geissen et al., 2006), low-velocity ditch water flow (Dai and Boll, 2006), and shallow groundwater table (Le Nguyen and Vo Luong, 2019)), a large sediment influx has taken place in the local drainage ditch (Bowmer et al., 1994). Consequently, this sediment influx lead drainage ditch prone to be clogged; leading constant dredging required (Smith and Pappas, 2007). Conceivably, these dredging pose a significant economic burden to the producer in the agricultural sector.

Concurrently, the enormous sediment influx affects its receiving water bodies (i.e., St. Lawrence River, North America). Compelling evidence has linked sediment to the nonpoint source pollution in the river via its suspended and dissolved forms (Blann et al., 2019). Literature has suggested that LSP littoral zone's sediment is a major contributor of this pollution, and responsible for the recent year's eutrophication and phytoplankton bloom in the waterbodies

(Hudon and Carignan, 2008; Rondeau et al., 2000; TCRLSP, 2017).

The large sediment influx also deteriorates the ecology of the environment. Hydrological connectivity between the ditch and its connected floodplain is retarded. The retardation causes an accelerated aquatic-terrestrial shift in the floodplain fish reproduction season. Researcher (Foubert et al., 2020) has linked this shift to the multi-decade population decay on Yellow Perch (a local flagship species). Such decrease in population leads local fishery sector suffers from a perilous economic loss (TCRLSP, 2017).

To encounter the sediment-related issues, remediation process is needed. Subsequently, a solution to moderate sedimentation is required. Through heavy research in the past decade, series of best management practices (BMPs) have been designed. Along those BMPs, vegetative filter strips (VFSs) could be a prominent solution for the LSP sediment problem.

VFSs are planted with dense vegetation in the adjacent area between the cropland and drainage ditch (Munoz-Carpena et al., 1999). These dense vegetation filter pollutants (include sediments) from the runoff, preventing it reaching the waterbodies. VFSs are proven to ease the sediment deposition by minimizing soil erosion. In addition, VFSs are considered a time, labour and capital efficient BMP. (Compared to its counterparts (e.g., no-till system, intercropping cover crops, etc.) in the agricultural production settings). With the addition of a non-disturbance nature on the crop harvest schedule, VFSs is recommended or even mandatory in the United States (Helmers et al., 2008; Poletika et al., 2009) and certain parts of Europe (Dabney et al., 2006).

Despite the abundant usage of VFS in inundation settings at riparian zone, VFS's performance in the littoral zone remains a mystery. Considering the different hydrological dynamic for both zones (Shellberg et al., 2013), VFS's performance could result significant difference. To elaborate, the cause of sedimentation could be highly differentiated between two environments (i.e., littoral zone and riparian zone).

Littoral zone encounters a much drastic gully mass failure than riparian zone. This sudden mass failure contributes a significant portion of sediment influx. In some instance, mass failure erosion has surpassed rainfall-infiltration erosion, becoming the sedimentation-driven force (Shellberg et al., 2013). The reason is related the higher frequency occurrence of the backwater (Jin et al., 2016) and drastic overbank flow (Shellberg et al., 2013) via the littoral zone's unique inundation pattern (Dąbrowska et al., 2016).

It is undeniable, the VFSs are proven efficient towards rainfall-infiltration erosion. In addition, with evidence that VFSs have certain resiliency on the shallow ground water condition (where its infiltration enhancement effect could be minimum). However, with the addition gully mass failure, VFS effectiveness in littoral settings is uncertain. In this study, we hope to investigate VFS's influence in the littoral zone at LSP with different width configuration in the agricultural drainage ditch.

To achieve our goal, we monitor VFS performance under two width configurations (4 m, 2 m, versus a control (0 m)) at three different sites in the LSP region. Sediment quantity changes are observed using the geomorphic budget method through deformation survey by total station. Our research could provide valuable information on using VFS for sedimentation moderation in littoral drainage ditch setting. Also, this could provide valuable information for policymakers in the LSP region. Material and methods

Study sites

Three experiment fields were selected in the littoral zone of the Lac Saint-Pierre floodplain in Quebec Canada; the experiment fields are referred to as STC (46.130°N, 73.124°W), YAM (46.268°N, 72.864°W) and BDF (46.143° N, 72.713° W) in this study. All three sites follow an annual soybean-corn rotation and fall tillage operation. All three sites have an average precipitation of around 1000mm.

Two different VFS width configurations, 2 m and 4 m, versus a control (0 m), were investigated in this field experiment. To be noted, YAM 0m VFS ditch's location was changed to its east side. This switch in location was in the consideration that the original YAM 0m VFS ditch's dense vegetation could act as a VFS, creating a false representation of non-VFS-equipped ditches.

The experiment started in November 2019 when all the selected ditches were dredged to a similar initial condition. A reed canary grass (Phalaris arundiacea)-oat mixture VFS was planted on both sides of the 2m and 4m VFS ditches in spring 2020; expect the second half (lengthwise) of the VFS on one side of the 2m ditch was planted with pure oat. In the 2021 spring, the VFS was replanted due to poor growth.

Sedimentation quantification

To quantify the sedimentation, a geomorphology approach was used. This method (equation 3.1) utilized the topographic/geomorphic change over time in the studied ditch. In our case, this change was estimated through ditch volume difference. Hence, the result will be presented in the percentage of sedimentation volume change ($\%\ \Delta V$), where positive was interpreted as sediment loss and vice versa for the deposition.

$$\%\ \Delta V = \frac{V_t - V_{t-1}}{V_{t-1}} * 100\%$$

Equation 3.1

Where:

$\%\Delta V$ = precent of ditch percentage volume change over an investigation phase (%), where the positive value means sediment deposition and the negative value means sediment loss.

V_t = ditch volume (m^3) measured at the time t.

V_{t-1} = ditch volume (m^3) measured at the previous time t-1.

Ditch's topographic profile is sampled as cross-sections in a 20-metre interval (an average of 40 ditch cross-sections sampled per volume estimation for one ditch, number of ditch cross-section sampled varies by ditch length). Individual ditch's volume (V_t) is estimated (equation 3.2.a.) based on the ditch length (L) and its average of the estimated cross-section area ($\overline{A_t}$). The definition of sampled ditch cross-section area (A_i) is based on a slight modification on Roealens et al. (2016). The area between the cross-section profile and its highest elevation's horizontal projection line was set to be the cross-section area (figure 3.1). We alter the horizontal projection line position from the lowest profile elevation maxima (Roealens et al. (2016)'s approach) to the profile's highest elevation. This accommodates the interest in monitoring topographic change on both site of the ditch.

$$V_t = \overline{A_t} * L$$

Equation 3.2.a

$$\overline{A_t} = \frac{\sum_{i=n} A_i}{n}\Big|_t$$

Equation 3.2.b

Where:

V_t= ditch volume (m3) measured at the time t.

$\overline{A_t}$= average ditch cross-section area at the time t.

L= the length (m) of the ditch (table 3.1).

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

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

Table 3.1 Ditch length used for calculation.

Site	BDF			STC			YAM		
VFS width	4m	2m	0m	4m	2m	0m	4m	2m	0m
Length (m)	1020	1040	1025	845	825	830	600	635	610

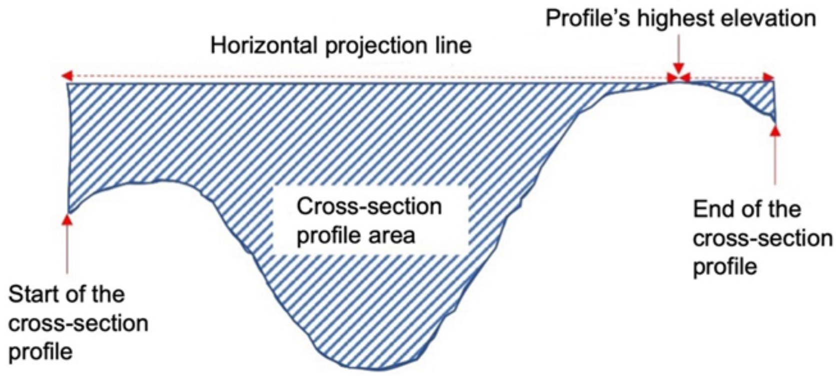

Figure 3.1 cross-section area definition

Ditch cross-section profile's topography is measured using a Leica® TS-07 manual total station and its associate GPR-1 prism. The cross-section topography is constructed via a one-dimension geodetic measurement (i.e., elevation only) in a 25 cm interval alone the cross-section profile (20 data points per cross-section).

EXPERIMENT SETTING

All studied ditch's volume is estimated nine times from Nov. 2019 to May 2022. With first two volume estimation on before (Nov. 2019) and after (Dec. 2019) the ditch dredging operations. The other five estimations were conducted yearly in May and October from May 2020 to May 2022.

During the Nov. 2019 – Oct. 2020, the start and the end of the sampled cross-section is not strictly controlled across the measurement. The cross-section length and coverage are largely depended on the surveyor's personal judgment. Such inconstancy in ditch cross-section creates noticeable uncertainty to the result. Also, to be noted that in this period (Nov. 2019 – Oct. 2020), no precautionary actions were implemented to prevent the survey operation's disturbance to the ditch geomorphology. As a result, surveyors stomped into the ditch when conducting their task (figure 3.2.a), causing bed sediment deformation.

To minimize the above problem, we modified the measurement procedure (figure 3.2.b) from May 2021 onward. In this period, reflective driveway markers (NuVue™ Fibreglass Driver Marker, 208.28 cm height (82 inches)) were set up at an 80-meter interval on both sides of the ditch. The reflective driveway marker provides a control on the sampling interval (20-meter interval) for ditch topographic profile. Also, a modified aluminum ladder is used as a set platform during the elevation measurement, avoiding further measurement-caused bed sediment deformation. This modified aluminum ladder also was marked every 25 cm of its length, providing a control on the elevation data interval (25 cm interval) for the cross-section topography.

Perceiving the possible ambiguity from comparing ditch volume estimated from different measure protocols, the experiment was divided into two different phases: a) Experiment Phase 1: Nov. 2019 – Oct. 2020, and b) Experiment Phase 2: May 2021 – May 2022. Consequently, the two experiment phases are treated as separated datasets; ditch topographic change and the sedimentation quantity were only compared and analyzed within its experiment phase.

Figure 3.2 example survey operation in Experiment Phase 1: 2019 Nov. – 2021 May (a), example survey operation in Experiment Phase 2: 2021 May – 2022 May (b), driveway reflector and aluminum ladder used in Experiment Phase 2 survey (c).

STATISTICAL ANALYSIS AND SOFTWARE

The sedimentation volume change ($\% \, \Delta V$) for one studied ditch between two of its volume measurements is treated as the independent sample for all statistical analysis. The independent sample T-test was implemented. It is intended to test the sediment quantity change at various VFS width configurations (4 m, 2 m, and control (0 m)). Each VFS width configuration is set as sample groups (4 m, 2 m, and control (0 m)), and three experiment sites act as replicas. The IBM® SPSS Statistics software© (Version: 28.0.0.0) was used during this process, and the one-sided significance p-value is reported in the result section.

The ditch cross-section area and its volume estimation were processed through similarity measures (Version: 0.4.4) developed by Jekel et al. (2019), a computation library specialized in calculating the interaction between hysteresis loops and other geometry based on Python programming language© (Python Software Foundation).

RESULTS

Experiment Phase 1 (2019 Nov. – 2020 Oct.) and Experiment Phase 2 (2021 May – 2022 May) have shown a different trend for ditch volume change (table 3.2). When ditches are within one year after the dredging process (Experiment Phase 1), a decreased volume trend can be spotted (figure 3.3), representing sediment deposition's occurrence.

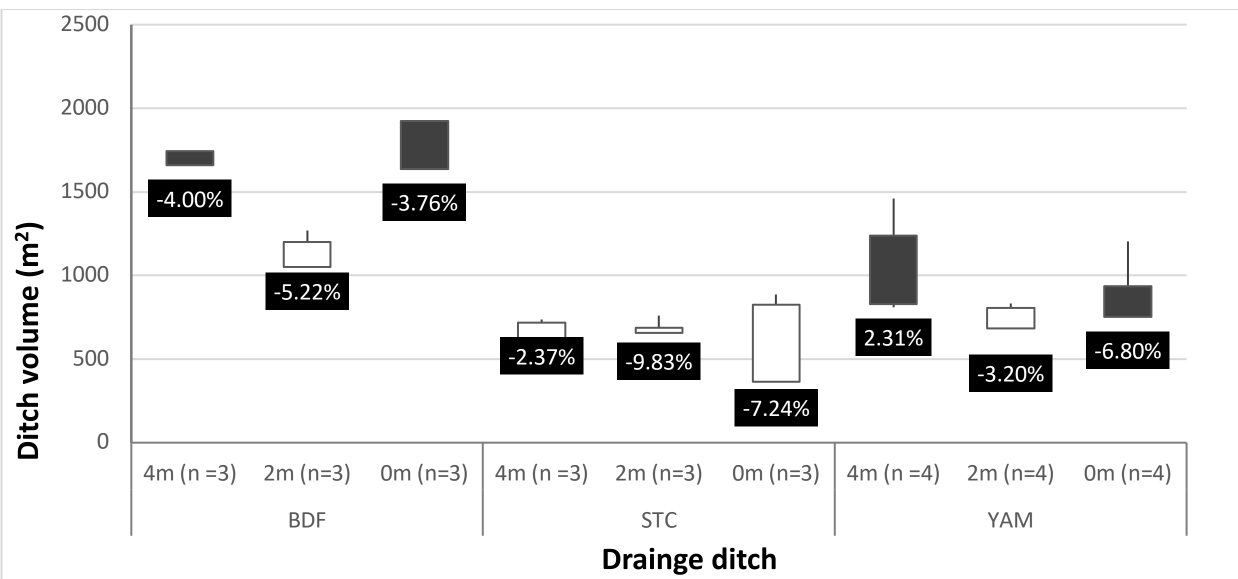

Figure 3.3 high-low-close chart [1] for ditch volume changes for Experiment Phase 1 (Percentage change from 2020 May - 2020

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[1] Upper/Lower bar boundary in chart: Ditch volume during the start/end of the Experiment Phase (higher volume as upper and vice

Oct. shown in black data label)

Despite decreasing volume as we expected, the magnitude of this drastic change is surprising. For instance, most ditch volumes have declined back (or lower) to their pre-dredging volume (figure 3.3). However, our eye observation shows that the ditch profile in 2020 Oct. (one year after dredging) is wider and deeper compared to its pre-dredging state (figure 3.4). On the other hand, it is unprecedented to see a dredged ditch filled with sediment within only a year. Therefore, we suspect the survey procedure in Experiment Phase 1 cannot represent the ditch's sediment deposition condition's true condition.

Table 3.2 Ditch volume during the experiment period

Site	**VFS width (m)**	**Ditch volume (m^3)**								
		Experiment Phase 1: 2019 Nov. – 2020 Oct.					Experiment Phase 2: 2021 May – 2022 May			
		2019 Nov.	2019 Dec.[2]	2020 May	2020 Oct.	% ΔV (2020 May. – 2020 Oct.)	2021 May	2021 Oct.	2022 May	% ΔV (2021 May. – 2022 May.)
BDF	4	1742.13		1729.26	1660.09	-4.00%	1891.67	1980.82	1909.559	1.16%
	2	1050.92		1267.31	1201.16	-5.22%	1695.07	1697.75	1736.761	3.70%
	0	1923.79		1700.67	1636.72	-3.76%	1761.60	1852.46	1910.657	7.45%
STC	4	416.15		735.63	718.20	-2.37%	973.08	971.68	901.4205	0.96%
	2	657.04		761.63	686.76	-9.83%	896.57	936.63	1152.026	29.45%
	0	364.34		887.77	823.50	-7.24%	1042.14	1174.51	1332.329	27.33%
YAM	4	1237.09	1458.79	811.49	830.23	2.31%	1201.26	1244.58	1275.739	2.61%
	2	685.26	736.02	832.00	805.37	-3.20%	830.82	801.48	719.6257	-8.54%
	0	935.31	1203.45	*806.22*[3]	*751.39*[2]	-6.80%	*993.87*[2]	*993.87*[2]	*1103.199*[2]	*2.93%*

Part of the reason for this suspicion was that the ditch profile's length and starting position were inconsistent during the measurement. This inconsistency results from ambiguous profile definition in Experiment Phase 1; profiles were defined base on the surveyor's visual recognition. Surveyors usually chose both ends of the profile where a large slope change occurs. Such behavior tends to cause a longer profile width in the pre-dredging phase, where visually

versa)

Upper/lower wick in chart: Highest/lowest recorded ditch volume during the time frame of Experiment Phase 1

[2] The total station survey in 2019 was only conducted at YAM due to the occurrence of snow and thick ice in ditches.

[3] The ditch volumes that calculated from the new YAM 0m location (eastside to its original location)

noticeable slope changes are further from the ditch center. In contrast, in the post-dredging phase, the noticeable slop changes are closer to the ditch center, causing a shorter profile width. This closer range was due to dredging practices where only the bottom part of ditches was exacted to maintain a more intact ditch bank structure. This width identification method caused a larger volume bias (figure 3.4) towards the pre-dredging ditch, where a wider profile cross-section was used; and eventually reflected on a larger volume in most of the ditch pre-dredged phase compared to its 2020 Oct state.

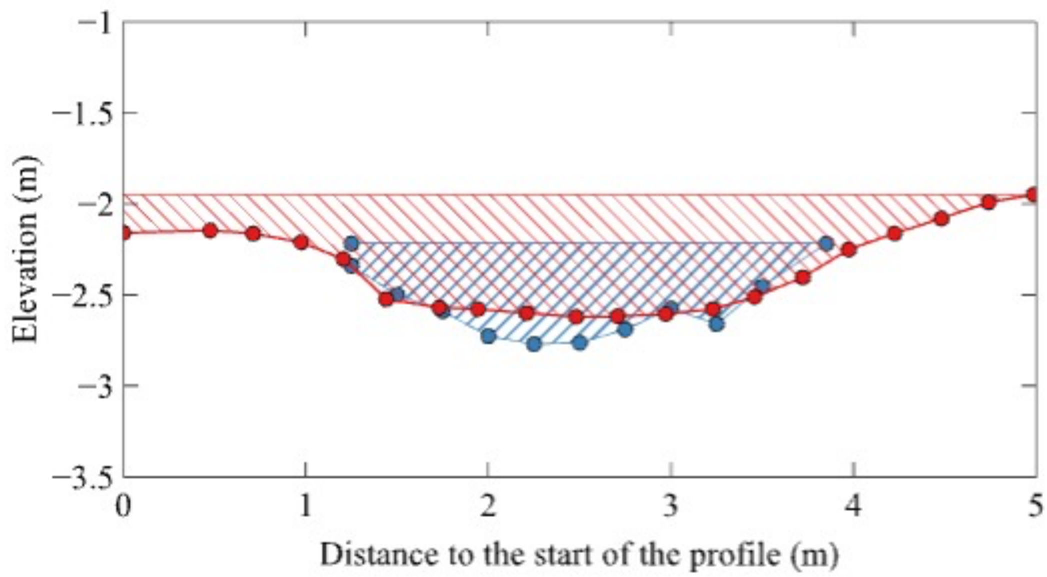

Figure 3.4 Illustration of the volume bias based on an example profile (YAM 4m VFS 60m) between 2019 Nov. (shaded in red) – 2020 Oct. (shaded in blue)

Contrasting with experiment phase 1, all ditches expect YAM 2m has its ditch volume increase in experiment phase 2 (2021 May – 2022 May), with the only expectation on YAM 2m ditch. The decrease in ditch volumes indicates sediment loss has happened. This volume increase can be largely explained by the width expansion of the ditch, which has a larger magnitude than the sediment deposition in the ditch bottom. (figure 3.5)

For the effect of VFS on sediment deposition, no statistically significant difference ($p<0.05$) was discovered from different VFS width configurations (including the 0m control) in any period of the experiment (Table 3.3). Hence, this finding suggests that VFS's deployment in the agricultural drainage ditch at Lac Saint-Pierre littoral zone does not create a noticeable effect on sediment deposition (or sediment loss).

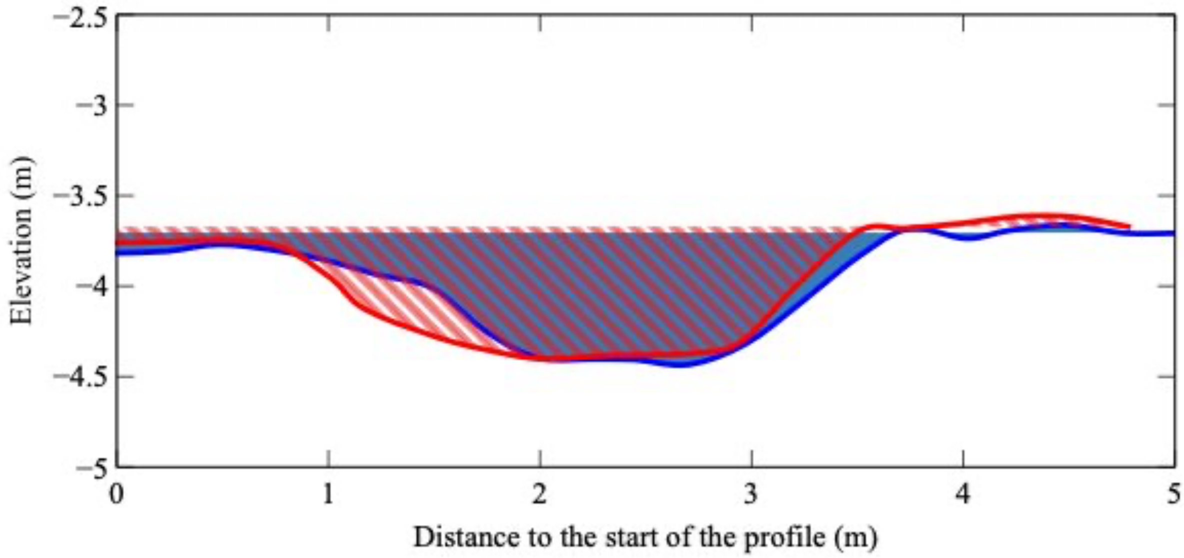

Figure 3.5 Illustration of volume chanege on an expample profile (BDF 2m VFS 80m) between 2021 May (shaded in red) to 2022 Oct. (shaded in blue)

Table 3.3 Ditch volume change in percentage for 2021 May – 2022 May (Experiment Phase 2).

VFS width (meter)	Experiment periods		Cumulative volume change % ΔV
	2021 May 2021 Oct.	2021 Oct. 2022 May	
4	1.70% ± 2.42% a (n=3)	7.54% ± 11.32% a (n=3)	4.62% ± 7.99% a
2	2.86% ± 7.79% a (n=3)	4.66% ± 10.68% a (n=3)	3.76%± 8.42% a
0	2.74% ± 5.17% a (n=3)	1.80% ±0.67 % a (n=3)	2.27 % ±5.17 % a

* The identical superscripts for each column indicate no statistical significance ($p > 0.05$) is found between the VFS width at each individual experiment period and cumulative volume change.

DISCUSSION

The enlarging ditch volumes in Experiment Phase 2 contradict the volume decrease expectation as sediment deposition is commonly believed to be the main reason for ditch dredging to maintain ditches' drainage performance. Since common perception believes that the deposited material from the sedimentation shallowing the ditch's depth (Pappas and Smith, 2007), effectively reducing the water storage volume in the ditch. This raises a question: how come the ditch still needs dredging in LSP littoral zone, despite an increasing ditch volume?

Research in the erosion mechanism of alluvial gully (Shellberg et al., 2013) might provide insight for this explanation. The alluvial gully erosion can be categorized into surface erosion (Brooks et al., 2009) and bulk erosion(a.k.a., mass erosion) (Mostafa et al., 2008; von Burkersroda et al., 2002), these two types of erosion results. Compared to the runoff bringing detached sediment from upland towards the alluvial gully (Meyer et al., 1999), bulk erosion usually comes from the self-mass failure from the gully structure (Rose et al., 2015). This mass failure commonly happens on the banks of the gully, where noticeable slopes are present. As erosion happens, it weakens the banks material's structure, eventually failing due to its gravity, depositing the bulk material into the ditch bed (Brooks et al., 2009; Shellberg et al., 2013).

This mechanism leads us to hypothesize that predominant mass failure could occur in our drainage ditch, causing ditch bank mass failure, and widen the width (Vanmaercke et al., 2021) of our ditch. This extended ditch width increases the total ditch cross-section volumes (Oparaku and Iwar, 2018) and offsets the sediment deposition from the bulk erosion.

The consideration that the littoral area undergoes a more intensive and more frequent groundwater seepage (Fox et al., 2007) and overbank flood inundation (Shellberg et al., 2013) could all weaken the soil structure from its alternating soil wetting-drying field (Wang et al., 2015), near-surface pore-water pressure (Natter et al., 2012). Also, the addition of subaerial erosion forces could encourage this mass failure process with the freeze-thaw cycle Field (Oztas and Fayetorbay, 2003), which has proven to break the soil aggregate and destabilize the soil structure (Leuther and Schlüter, 2021). Unfortunately, we cannot provide any decisive evidence on the ditch width change nor quantification of bulk mass erosion and its contribution to the total sedimentation process. As our experiment design did not focus on monitoring the width change. On top of that, a high robust ditch width definition needs to be researched, which can be highly expressive to latitudinal geomorphology change.

This hypothesis on the domination of bulk mass erosion in littoral ditch sedimentation could also explain our result of VFS's limited effect on sedimentation moderation. From the VFS's mechanism standpoint, its sediment filtration mechanism was primarily targeted toward the runoff-carried sediments (surface runoff). This mechanism has primarily relied on the slowing effect on the water (i.e., erosion medium). Whereas in the bulk mass failure erosion, VFS's above-ground structure (i.e., the major contributor to slowing water effect) has a limited effect on mass failure prevention.

Despite the dense root development from the VFS could improve the infiltration of the locating soil and create a more resilient soil structure through its root development. However, literature (Lauvernet and Muñoz-Carpena, 2018; Muñoz-Carpena et al., 2018) shows that under high water table conditions, VFSs could result in a significant decrease in its infiltration capacity, where the shallow groundwater table affects the surface hydrological response of a VFS by generating soil water saturation during rainfall (Li and Kuo, 2021). This could heavily retar the VFSs efficiency in our experiment.

On the other head, we could not ignore that VFS is not well developed in a certain position around our experiment ditches, leaving the adjacent land fallow. This could create uncertainty about the quantity of sediment deposition/loss. Also, we could not ignore the potential of the uncertainties and bias generated by the topographic survey and our cross-section area definition.

CONCLUSION

Considering the analysis from our three-year investigation over two experiment periods, VFSs and their width configurations show no significant effect in sedimentation moderation. The insignificant of VFS effect might be caused by limited influence of VFS on bulk erosion under littoral condition, or because of the poorly developed vegetation. Still, we cannot put a firm confirmation on our results, since the bias and uncertainty from our ditch volume estimation.

Despite the geomorphology budget approach is a robust method that doesn't rely on imperial equations (i.e., turbidity to total suspended solid concentration) or a strong dependent on hydrometric data. Our implementation of this approach still cannot fulfill the accuracy and precession needs for the sediment deposition study. The reason can be concluded to the high uncertainty of data caused by the lack of geodetic controls on data points.

Geomorphology budget approach's accuracy is heavily relying on the ability to track the alteration in the shape of the object. Tracking changes in any objects needs a store control on measurement's position. In our case, the absence of horizontal control network leads a high variability on the accuracy of measurement position. Consequently, different locations of the cross-section are compared to calculate the dimension difference of the ditch across time.

Including a high order horizontal control network could lead either a significant increase in the experiment cost (i.e., setting up long-term survey pegs in every monitor cross-section) or increasing the labour cost for the experiment (i.e., running three-dimension geodetic measurement). However, in a realistic standpoint neither will be a sustainable option.

For further studies, author recommends exploring the options of using remote-sensing options in geomorphology budget approach for ditch sedimentation quantification. LiDAR (Light detection and ranging technology), a remote-sensing technology, can provide high accuracy three dimensional geodetic measurements and wide data coverage for ditch topology. However, LiDAR's performance under dense vegetation still highly varied. Where in VFS studies, this variability could be detrimental.

Chapter 4 EVALUATION OF UAV- LIDAR'S ACCURACY IN THE QUANTIFICATION OF SEDIMENTATION IN DITCHES UNDER THE LITTORAL SETTINGS

INTRODUCTION

Agricultural drainage ditches are vital food production infrastructures due to the high portion (25%) of North American croplands' drainage requirements. Behind its supportive role in agricultural production, the drainage ditches have also been considered the main entrance for field contaminant to enter the whole agroecosystem (Blann et al., 2009; Hudon and Carignan, 2008; Rondeau et al., 2000). The transportation of these field contaminant is heavily coupled with the sedimentation dynamics, where surface soil detaches from the upland, absorbs the dissolved or particulate field contaminants, and carries them to the associate waterbodies (Gregoire et al., 2009).

Monitoring the sedimentation is necessary before any remediation process takes place in the agricultural drainage ditch. This is especially true in the littoral zone, where sedimentation is heavily related to the soil geo-weathering cycle and soil biogeochemical cycle (Repasch et al., 2020). These two cycles can heavily influence North American food production and its associated waterbodies' sustainability (Lal, 2007).

Currently, the quantification process of sedimentation can deviate into two major categories: sedimentary budget (e.g., Warrick and Milliman, 2003) and geomorphology budget method (e.g., Argyilan et al., 2005; Capoane et al., 2015; Marzolff and Poesen, 2009; Notebaert et al., 2009). In most sedimentation quantification scenarios (i.e., large waterbody), the sedimentary budget method is used. This quantifying method usually relies on the hydrometric data and data and water quality data over the monitored environment. It uses the TSS (total suspended solids) and water flow volume between the monitor inlet and outlet to estimate the amount of sedimentation (Rosati, 2005).

However, in the littoral setting, the flow volume during the periodic inundation area is hard to be monitored due to the nonpoint source of water dynamic by the river backwater and groundwater seepage. In addition, the difficulties of estimating the TSS concentration in field runoff further decrease the budget method's potential under the littoral vegetated drainage ditch environment (Cappucci et al., 2020). On top of that, for small-scale research where hydrometric and water quality data is unavailable, numerous scientific equipment must be installed in situ during the investigation period, limiting its accessibility.

On the other side, the geomorphology budget method provides a reliable estimation in the littoral setting (Cappucci et al., 2020) due to its direct measurement of the changing sediment quantity at the target sedimentation sink (i.e., agricultural drainage ditch). In contrast, empirical and conceptual estimation and calculation are made on the sedimentary budget method. However, intense labour is needed during the topographic surveys behind its robust method. This is mainly due to the limited land accessibility of the littoral zone, where it turns into an aquatic environment from periodic flood inundation. Despite remote sensing can be implied to drastically decrease the labour needs compared to the physical survey (e.g., total station survey, GNSS positioning, erosion pin, etc.,); however, the existence of heavy vegetation and the accuracy requirement still limited its usage in monitoring sedimentation in agricultural littoral drainage ditch condition.

Light detection and ranging technology (LiDAR) is a highly potential remote sensing solution for sedimentation monitoring. LiDAR uses light as a pulsed laser to measure the range of subjects, coupling with sophisticated GNSS posting schemes and high-precision IMUs (inertial measurement units). Together, this sensor complex (LiDAR) has made LiDAR a highly accurate remote sensing method with a potential accuracy in centimetres (Legleiter, 2012). Most importantly, its millimetre and micrometre-level laser footprint allows ranging lights to reach the ground structure through gaps in-between the vegetation (Millard et al., 2009; Pinton et al., 2020). However, research showed that only decimeter-level of accuracy can be achieved in manned aircraft platform LiDAR, due to its limited capability in penetrating the dense vegetation in the agricultural drainage ditch settings. Many researchers believe that this is due to the low sampling density during the survey, which leads to low-number or none ranging lasers being able to reach the ground structure (Aguilar et al., 2010). This limited vegetation penetration capability is probably caused by the decreased possibility for ranging laser to aim in-between the gaps of vegetation foliage, as dense vegetation is presented in the littoral condition.

In recent years, the availability and the load capability of commercially unmanned aerial vehicles (UAV) have greatly improved, making it possible to serve as an aerial platform for the LiDAR sensors (Lin et al., 2019). The low fly height and low airspeed nature enable it to have a denser data point density than the traditional platforms (i.e., manned aircraft). The increase in sampling density might finally overcome LiDAR's vegetation penetration issues in dense vegetation littoral drainage ditch conditions.

To test whether the UAV-based LiDAR can penetrate vegetation in densely vegetated littoral conditions, we use an out-of-box UAV-based LiDAR system and its manufacture-provided data-points processing software to survey nine heavily vegetated drainage ditch in the littoral zone of Lac Saint-Pierre. The performance of UAV-based LiDAR will be evaluated based on its vertical accuracy and ditch cross-section area with the ground control data measured by the physical survey method (total station survey). Also, a linear regression-based LiDAR correction model is developed to see whether a higher accuracy could be accomplished. Finally, the hypothesis of error sources and the current limitation for UAV-based LiDAR system in the densely vegetated littoral condition is presented. This research intends to provide valuable information for the viability assessment of using LiDAR for sedimentation monitoring of the densely vegetated drained ditch in the littoral area.

MATERIAL AND METHOD

STUDY AREA

Nine agricultural drainage ditches from three experiment sties in the littoral zone of Lac Saint-Pierre region are selected in this study. These experiment sites are located near town St. Cuthbert, (46.130°N, 73.124°W), Yamachiche (46.268°N, 72.864°W), and Baie-du-Febvre (46.143° N, 72.713° W) in Quebec Canada. Three drainage ditches are selected per experiment sites, with dense vegetation occurrence in the drainage ditch. In addition, reed cannery grass-oat mixture vegetation filter strips (VFS) were planted alongside the adjacent area in spring 2020 between the ditch and cropland to create more vegetation in the littoral condition.

Each ditch will be abbreviated as a combination of its site location and the width of the VFS planted (4m, 2m, and 0m for no VFS ditches). The site location will be abbreviated as STC (Site near St. Cuthbert), YAM (Site near Yamachiche), and BDF (Site near Baie-du-Febvre).

GROUND CONTROL POINTS

The ground control point was measured in October 2021. Ground control points were grouped into sets of ditch cross-sections in an 80-meter interval alongside the longitudinal direction of the ditch. For each control point set, ground control points were established in a 25-cm interval alongside the direction of the cross-section. In addition, reflective driveway markers (NuVueTM Fiberglass Drive Marker, 208.28 cm height) were placed as survey pegs on both banks of the ditches (figure 4.1.a).

An Emlid™ RS GNSS receiver is used to measure the geolocation of the reflective driveway maker (figure 4.2.b) into the NAD83/UTM zone 18N spatial reference network and CGVD2013 vertical control network.

To assure the maximum accuracy of the vertical height of the ground point, a Lecia™ TS-06 plus manual electronic total station with its associate GPR-1 prism is used to survey its geolocation and elevation into a temporary non-georeferenced grid. Moreover, that temporary grid was later transferred into the NAD83/UTM zone 18 N and CGVD 2013 reference network, using the geolocation of the GNSS measured reflective driveway marker.

During the ground control point measurement, a 21-ft Mastercraft Grade 2 Aluminum extension ladder was placed across the ditches, serving as a stepping platform during the survey, minimizing the bank deformation caused by anthropogenic interaction (i.e., surveyor stepping into the ditch) during the survey (figure 4.2.c.). In addition, the ladder was marked every 25cm of its length to ensure the sampling interval was strictly followed.

Figure 4.1 the placed reflective driveway (a), the GNSS survey operation for the reflective driveway marker's location (b), and the illustration of the ladder stepping platform (c).

UAV-based LiDAR survey

All studied ditches have been scanned four times (2019 Nov. (before dredging), 2019 Dec. (after dredging), 2020 Oct. and 2021 Oct.), using the YellowScan™ Surveyor LiDAR system accompanied by a DJI™ Matrice 600 UAV (figure 4.3.a), with the LIDAR system's in-situ GNSS antenna connected to a Spectra™ SP80 GNSS receiver (figure 4.3.b) in the base-station mode. The UAV was kept within the 2 km radius from the GNSS base station and recorded the LiDAR sensor position using the post-processed kinematic positioning (PPK) method.

The LiDAR system has a 903 nm laser wavelength and the capability of generating 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/m^2. This point cloud data was processed through LiDAR sensor's associate YellowScan™ 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.

Elevation and cross-section area extraction

The LiDAR data from 2021 Oct. were used to evaluate the accuracy of LiDAR for sedimentation quantification for two aspects: a) Bare-earth elevation and b) ditch cross-section area estimation. These two aspects represent the common information needed for sedimentation quantification under the topography difference-related method.

The elevation data is extracted from the generated DTM based on the ground control points' geolocation. The elevation data is directly compared to the ground control point data for bare-earth elevation accuracy evaluation. After the extraction of bare-earth elevation from DTM, the bare-earth elevation points were grouped into sets of ditch cross-sections for the ditch cross-section. The cross-section area is the area between the cross-section profiles and the horizontal projection line of that profile's highest elevation (figure 4.3). Both cross-section profiles from ground control points and cross-section profiles from LiDAR strictly followed the above definition, and both cross-section areas were calculated.

Figure 4.2 LiDAR sensor and its platform UAV (a), the GNSS-base station setup (b).

Ditch volume calculation for LiDAR

The ditch volumes are calculated using LiDAR generated DTM. First, all the non-ground classified LiDAR points (i.e., vegetation or artificial entities) are being removed. Next, the remaining points are processed into a digital terrain model (DTM) with a special resolution of 5 cm, using the average elevation value of the LiDAR points. This DTM is then used to calculate the ditch volume using the “volume calculation” toolbox from ArcGIS Pro™ by ESRI®.

Linear regression-based LiDAR error correction model

A linear regression-based LiDAR error correction model is implemented in LiDAR's elevation database on the following (equation 4.2):

$$Z_{correction} = a * Z_{LiDAR} + b$$

(Equation 4.2)

Where:

$Z_{correction}$ = *The elevation data after the error correction model (m)*

Z_{LiDAR} = *The LiDAR elevation data (m)*

a, b = *site-specific correction parameter*

The correction parameter value ($\boldsymbol{a, b}$) is determined by fitting a linear regression model (equation 4.2) between the LiDAR bare-earth model and the ground control point (GCP) from each studied site. The interception parameter ($\boldsymbol{\beta_0}$) of this linear regression model is set as the correction parameter ($\boldsymbol{a}$) in the linear regression-based LiDAR correction algorithm. The interception parameter ($\boldsymbol{\beta_1}$) is set to be the correction parameter ($\boldsymbol{b}$) in the LiDAR correction algorithm (equation 4.3).

$$Z_{GCP} = \beta_0 + \beta_1 * Z_{LiDAR}$$

(Equation 4.3)

Where:

Z_{GCP} = *The elevation data from the ground control point (m)*

Z_{LiDAR} = *The LiDAR elevation data (m)*

β_0= *interception parameter for the linear regression model*

β_1= *coefficient parameter for the linear regression model*

This linear regression-based LiDAR error correction model (equation 4.2) will be evaluated through the same evaluation aspects (both bare-earth elevation and cross-section area) of the non-corrected LiDAR data using the same data extraction method.

EVALUATION PARAMETER

Both linear regression-based model processed LiDAR accuracy and native (no linear regression-based correction model involved) are assessed using these statistical indexes: root mean square error (***RMSE***) (equation 4.3.1), mean absolute error (***MAE***) (equation 4.3.2), relatively error (equation 4.3.3), and coefficient of determination (***R^2***) (equation 4.3.4).

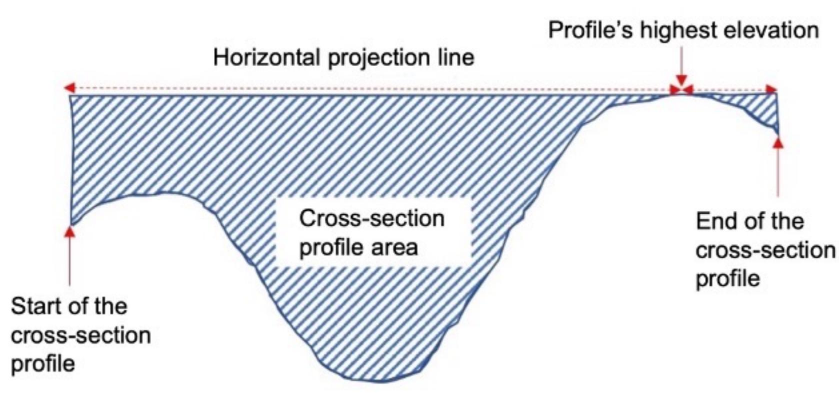

Figure 4.3 cross-section area definition

$$RMSE = \sqrt{\frac{\left(\sum X_{lidar} - X_{reference}\right)^2}{n}}$$

(Equation 4.3.1)

$$MAE = \frac{\sum \left| X_{lidar} - X_{reference} \right|}{n}$$

(Equation 4.3.2)

$$Relative\ error = \frac{1}{n}\left(\frac{\left| X_{lidar} - X_{reference} \right|}{X_{reference}}\right)$$

(Equation 4.3.3)

$$R^2 = \left(\frac{\sum\left(X_{lidar} - \overline{X}_{lidar}\right)\left(X_{reference} - \overline{X}_{reference}\right)}{\sqrt{\sum\left(X_{lidar} - \overline{X}_{lidar}\right)^2\left(X_{reference} - \overline{X}_{reference}\right)^2}}\right)^2$$

(Equation 4.3.4)

Where:

$RMSE$ *= Root mean squared error*

MAE *= mean average error*

R^2 *= coefficient of determination*

X_{lidar} *= elevation (m) or the cross-section area (m²) of the LiDAR or error-corrected LiDAR data from the linear regression-based model*

$X_{reference}$ *= elevation (m) or the cross-section area (m²) of the reference data*

n *= the number of the data points*

RESULT AND DISCUSSION

DITCH VOLUME TREND

The LiDAR-derived ditch volume data (figure 4.4, figure 4.5) shows that drainage ditch volumes decrease across the two-year periods after the 2019 Nov. (ditch dredging event). Underlying the average 10% ditch volume increase between the 2019 Nov. to 2019 Dec. from the ditch dredging process, it shows that two years after dredging, six out of nine ditches have returned to their pre-dredging volume. These drastically accumulated sediments need to mediate the sedimentation process since a typical North American drainage ditch requires dredging in a 5

– 50 years range.

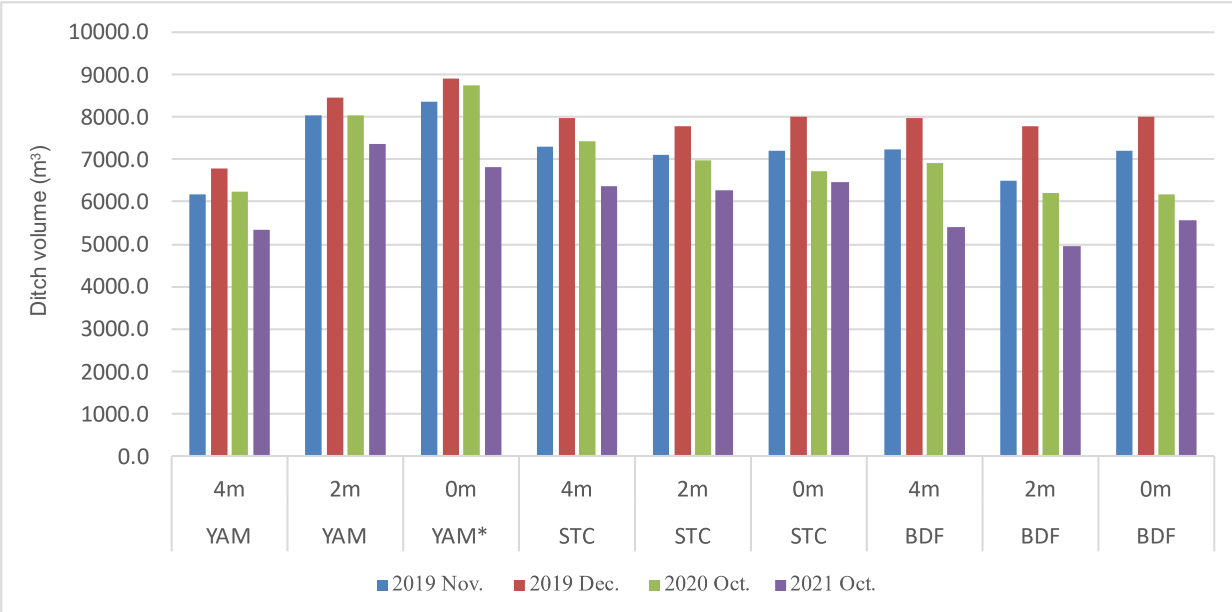

Figure 4.4 Ditch volume in 2019 Nov. – 2021 Oct. by LiDAR survey.

ELEVATION AND CROSS-SECTION ACCURACY

The evaluation parameter (table 4-1) for both UAV-based LiDAR measured bare-earth elevation and ditch cross-section area shows a 10cm – 20cm error (in both ***MAE*** and ***RMSE***) when compared with the ground control data measured by the total station. Notably, this decimeter level of accuracy is drastically away from the manufacture's claimed 5cm ***RMSE*** accuracy.

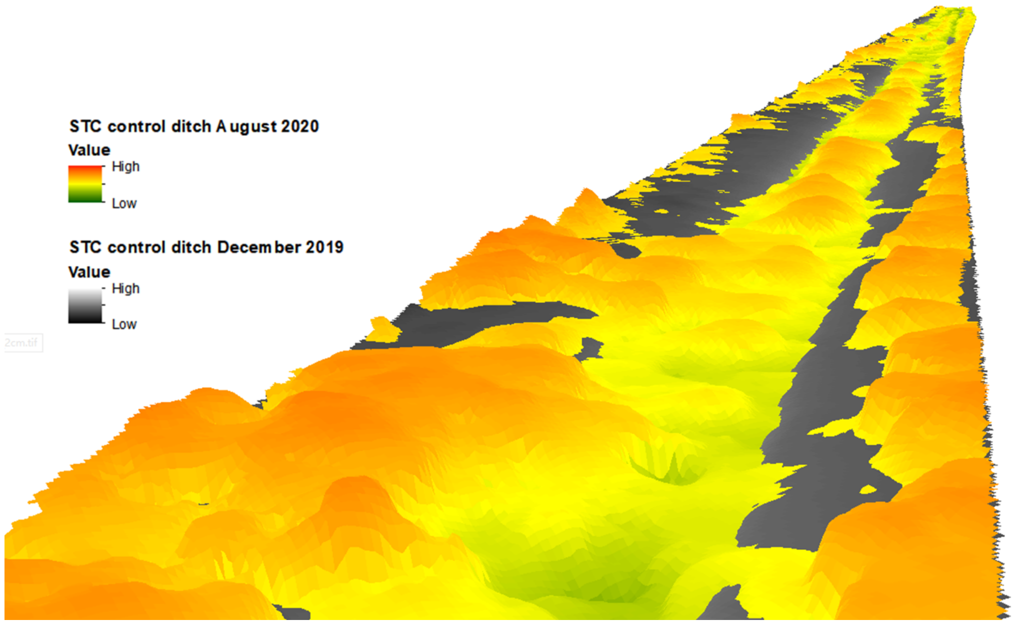

Figure 4.5 Example of LiDAR scan (STC 0m ditch) in 2019.

Table 4.1 Evaluation parameter for elevation and cross-section data between the LiDAR and ground control points.

Site	Correlation parameter between the LiDAR and ground control points									
	Elevation data					Cross-section area				
	n	RMSE (m)	MAE (m)	R^2	Relativity error	n	RMSE (m^2)	MAE (m^2)	R^2	Relativity error
BDF	827	0.092	0.209	0.551	0.036	37	0.077	0.206	0.195	0.035
STC	640	0.054	0.177	0.884	0.029	32	0.033	0.130	0.624	0.022
YAM	460	0.038	0.166	0.951	0.029	23	0.011	0.082	0.564	0.015

Neglecting the possibility of manufacturing, this magnitude of difference suggests that the LiDAR system's performance is hindered in our survey condition. Since our survey was conducted when no water was presented in the drainage ditch, this led us to suspect it may relate to the densely vegetated condition in drainage ditch. Evidence can be seen from both the height bias distribution of LiDAR elevation data (figure 4.6). Our visual inspection further on the cloud point data further assure our susception (figure 4.7.a). Ditch cross-section area data exhibit a similar result, where a relatively 18% - 24% error is present between the LiDAR data and ground control points.

Our attempts to utilize a linear regression-based model for LiDAR data correction receives mixed bag of results (table 4-2). Accuracy improvement can be found in the corrected elevation, where the biggest accuracy increase is shown at site YAM, where both the ***RMSE*** and ***MAE*** have decreased by half; site BDF receives minimum improvement where only its RMSE has decreased by 0.3 cm. However, with the linear correlation, bare-earth elevation error still remains at a decimeter level, far off the LiDAR sensor's theoretical performance.

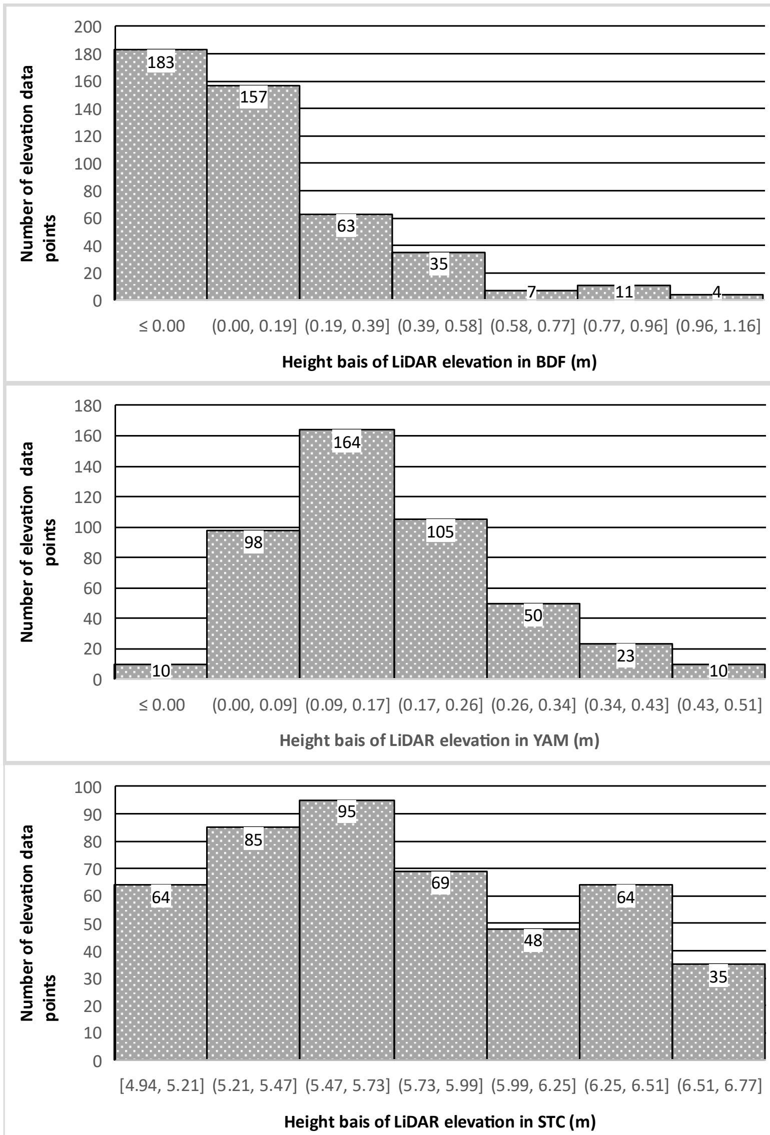

Figure 4.6 Distribution of height bias of LiDAR elevation data

On top of that, we receive a decrease in accuracy with the cross-section area estimation when using the correction model; besides site BDF receiving a small overall improvement for its evaluation indexes, all other sites undergo a minor error increase (0.6% - 1% increase). Such results suggest that our statistical correction method has a limited effect on LiDAR error correction, indicating that post-processing of categorized LiDAR point cloud received the minimum return or even negative returns, physical modification of the LIDAR system, or algorithmic improvement of the LiDAR raw data processing software (i.e., noise filtering and data point categorization) is needed for increasing the LiDAR system accuracy.

Figure 4.7 Limited Vegetation detected in ditch shown in white circle (highlighted in green) (a), adequate vegetation detection in nearby field crop (b).

In this study, we had set our UAV-LiDAR system to produce the maximum point density (250 – 400 pts/m^2) that LiDAR manufacture recommend. Comparing to the manned-aircraft platform LiDAR survey (around 4 – 10 pts/m^2), we have a much denser point data, which theoretically speaking could provide better vegetation penetration capabilities since more lasers could pass through the dense vegetation and reach the bare surface.

Table 4.2 Evaluation parameter for elevation and cross-section data after the linear-regression base model correction.

Site	Correlation parameter between the LiDAR and ground control points after the linear regression-based model correction							
	Elevation data				Cross-section area			
	n	RMSE (m)	MAE (m)	Relativity error	n	RMSE (m^2)	MAE (m^2)	Relativity error
BDF	827	0.077	0.206	0.035	37	0.221	0.381	0.193
STC	640	0.033	0.130	0.022	32	0.091	0.241	0.254
YAM	460	0.011	0.082	0.015	23	0.241	0.254	0.184

However, our UAV-LiDAR elevation error is at the same level as Conte et al. (2013)'s manned-aircraft LiDAR elevation error (decimeter level error). In combine with evidence (figure 4.6 and figure 4.7.a) that LiDAR has fails to identify majority of the vegetation, this indicate that error might be caused by LiDAR system failing to identify the dense herbs structure in littoral area.

Especially, consider the fact that LiDAR's successful identification on nearby crop field (figure 4.7.b), makes us believes that lacking bare-earth structure entry caused by insufficient point cloud density is not the major issues for UAV-LiDAR's error in densely vegetative littoral condition. In fact, the error is caused by the unsuccessful recognition of vegetation by the filtering algorithm (provide by LiDAR manufacture), which consider the herb structure as bare earth.

CONCLUSION AND RECOMMENDATION

Our UAV-based LiDAR system resulted in a decimeter-level error in the densely vegetated drainage ditch environment compared to the total station measured ground control data, which led the LiDAR system to provide an inaccurate estimation of ditch volume. Post-processing method, such as the proposed linear-regression-based LiDAR correlation model, has a minimum effect on improving the LiDAR performance in sedimentation monitoring. We suspect the error is mainly due to the failure of the LiDAR system to distinguish the vegetation from bare-earth material. Hence, the author suggests that future studies should focus on improving the non-terrain object filter algorithm, eliminating the false negative (type II error) scenario in the non-terrain object detection.

For current agricultural ditch sedimentation quantification, LiDAR-based technology is far away from its maturity. The physical survey, such as the total station and RTK survey, still is the best option for the topographic difference-related method for calculating the quantity of sediment deposition. Even though LiDAR could prove an invaluable spatial detail by generating thousands of data points per hectare, the toll on data accuracy cannot be ignored.

Chapter 5 Conclusion and summary

The major goal of this thesis was to assess the vegetative filter strip's sedimentation moderation performance under the littoral agricultural drainage ditch condition in Lac Saint-Pierre region, Quebec. Our sediment quantity change data shows that neither VFS nor its width configuration has a statistically significant influence on sedimentation moderation in the agricultural drainage ditch.

In Chapter 2, a comprehensive literature review was performed to outline the needs of testing the vegetative filter strip in the littoral condition in Lac Saint-Pierre for agricultural drainage ditch sedimentation moderation. A general debrief was given to show the necessity to solve the agricultural drainage ditch-originated sedimentation problems in Lac Saint-Pierre. Also, the reason of choosing and testing the vegetative filter strip in the littoral condition was clearly identified from both an economical and academical point. On top of that, a comprehensive technical review on the current sedimentation monitoring method was made, and the reason of selecting UAV-based LiDAR was also given.

In Chapter 3, sediment quantity change was monitored using the geomorphic budget method through varies total station survey. Nine studied ditch volumes changes were calculated, through ditch cross-section area estimation and ditch length, to achieve this a specific definition of ditch cross-section area was proposed. With the volume changes data, statistical analysis was preformed and shows a result of neither VFS, nor its width configuration has a significant influence towards the sedimentation moderation in littoral ditch environment. In the end, the domination of bulk erosion was attribute to the reason for LSP's limited performance, and potential advice for further investigation was given.

In Chapter 4, both the result of ditch sediment changes over 3 years using UAV based-LiDAR and an accuracy analysis of using UAV based-LiDAR system in the densely vegetative at littoral drainage ditch condition was conducted. With LiDAR survey data compared with total station-measured topographic data, an assessment on both LiDAR system's elevation and cross-section area accuracy was given. In addition to that, a linear-regression based LiDAR data correction algorithm was proposed and implemented to further explore the potential of the UAV based-LiDAR data. The linear-regression correction algorithm shows limited effect towards the improvement of LiDAR system in the densely vegetated environment. The findings shows that UAV based-LiDAR system has a decimeter level of accuracy, and its cross-section area estimation is not sufficient for the sedimentation monitoring usage. Improvement direction and errors source speculation was also addressed in the later part of the chapter.

FUTHURE WORK

The project result indicating the Vegetative filter strip has a limited effect on the littoral conditions in the Lac Saint-Pierre region. However, the detail reason has not been investigated; also, current ditch-volume related sedimentation change quantification method was find limited representation of sedimentation due to the possible ditch width-caused volume increase. Studies that use both ditch width and volume needs to be conducted in a longer time frame to assure an accurate quantification on the sedimentation dynamic. As for the UAV-based LiDAR for the sedimentation monitoring, the top priority is to propose a robust and efficient LiDAR datapoint filtering and bare-earth identification system needs to be developed; bring the vegetation filtering performance to a sufficient level for the densely vegetative condition in littoral area.

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