Sedimentary processes in large, regulated river systems in the Canadian subarctic

by

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Abstract

The goal of this thesis is to develop an improved understanding of sedimentary processes along the Nelson River system in the subarctic region of Canada. Sediment sources and transport dynamics in Lake Winnipeg, the Upper Nelson River (UNR) – between Lake Winnipeg and Split Lake – as well as the Burntwood River (BR) – the major tributary of the Nelson River in the subarctic region – were investigated. The properties of Lake Winnipeg's bottom sediment and the lake's total sediment budget were used to provide a better understanding of: a) the sedimentation dynamics in Lake Winnipeg; and b) its role in sediment transport in the Nelson River system.

The sediment source fingerprinting technique and long-term record of sediment load data on the BR and the UNR were used for separating the importance of climate change from human-induced environmental changes on these two regulated rivers. Moreover, the influence of Split Lake on the downstream delivery of sediment to the Lower Nelson River and Hudson Bay was investigated by developing the sediment budget for this riverine lake.

In addition, the collection of a representative sample of ambient suspended sediment using a well-established time-integrated sampler and two adapted discrete samplers was investigated. The performance of these samplers was examined in a controlled laboratory and under field conditions. Assessing these samplers was conducted to determine the most suitable device to collect representative bulk samples from the Nelson River system.

The results show that the sediment load derived from the prairies area is sequestered in Lake Winnipeg, along with nutrients and contaminants bound to them. Another key finding was that sediment derived from bluff erosion on the northern shore of the lake is

the major source for sediment being exported in the lake's outflow. This thesis also found that the UNR is characterized by increases in sediment loading as a result of climate change. However, in the BR, cross-watershed water diversion caused a seven-fold increase in sediment discharge and since diversion flow regulation near the licenced limit has muted the response to variability in local precipitation and runoff. This thesis also provides several recommendations for further research.

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Declaration academic achivement

The outline of this thesis follows the “grouped manuscript style” whose guidelines are appointed by the Faculty of Graduate Studies, University of Manitoba. It contains four individual papers published in peer-reviewed journals. Chapter 1 contains an introduction about the study area, research objectives, and thesis contributions; Chapters 2 to 5 are manuscripts, each containing an abstract, introduction, methods, results, discussion, and conclusions; Chapter 6 provides the final synthesis and general conclusions of this research and recommendations for future research.

Chapters 2 to 5 have been published as four journal papers. M. Goharrokhi exercised control in developing the research idea, designing and administering projects, supervising research assistants, implementing the field works and experiments, interpreting the data, drawing the figures, writing all the manuscripts, submitting the manuscripts and responding to reviewer’s comments, and submitting the revisions. The co-authors contributed to developing the research idea, analyzing data, and editing the manuscripts for publication.

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CHAPTER 1: Introduction

1.1 Background and rationale

Rivers can account for approximately 95% of the total sediment load in the oceans (Owens, 2008). While fluvial sediments are transported in both bed load and suspended load mode, suspended sediment is of greater interest since the majority of sediment transport to the ocean is performed by this form of transport (Walling, 2013). Syvitski et al. (2003) report that 90-95% of total annual fluvial sediment flux to the coastal zone is suspended sediment. Another reason for paying particular attention to the suspended sediment transport mode is the behaviour of the fine sediment particles (i.e., <63 µm) due to their size, shape, specific surface area, and electrical charges (He and Walling, 1996).

Excessive fine sediment may, however, result in severe issues for aquatic systems, including increasing turbidity, reducing light penetration, reducing reservoir capacity, impairment of navigation, clogging of fish gills, and impacts on recreational activities (Davis and Fox 2009; Haddadchi et al., 2013; Krishnappan et al. 2009; Walling, 2005, 2013).

In addition, recent decades have seen a growing awareness of the strong relationship between inorganic suspended sediment load and organic matter, nutrients, and contaminants transported through fluvial systems. For example, Walling (2005), indicated that more than 75% of the total phosphorus (P) load in catchments of UK rivers is carried in particulate form. Electrical charges of fine suspended sediment are, at least in part, responsible for binding and transporting nutrients and contaminants in the fluvial system (He and Walling, 1996).

Effective watershed management practices to limit the negative influences of fine sediment need information on sediment mobilization and delivery processes (Walling, 1983) and the impacts of natural and human-induced environmental changes operating within watersheds on these processes (Walling and Collins, 2008). Collecting such data is not straightforward since this source of pollution comes from different parts of the watershed and heavily depends on spatiotemporal variability of hydrological components (Walling, 2013).

A direct approach has been developed to assign sources of erosion to transported materials based on the use of environmental tracers which began in the early 1960s (Corcoran and Kelley, 2006). Sediment tracing technology was, originally, developed to measure erosion, estimate sediment age, and calculate rates of sediment deposition and accumulation in fluvial, lacustrine, and marine environments (Corcoran and Kelley, 2006). A recent development of this approach is determining the sources of problematic fine-grained sediment and is termed the sediment source fingerprinting technique (Walling, 2005; 2013).

This thesis is a part of the Hudson Bay System Study (BaySys) project, which is a collaborative effort of six academic institutions and Manitoba Hydro and addresses several research objectives, including the understanding of the role of climate change and hydroelectric regulations in sediment and chemical (including carbon and mercury) transport dynamics in Hudson Bay and its river system (Barber, 2015).

The research described in this thesis was also funded by the Natural Sciences and Engineering Research Council (NSERC) of Canada under the CREATE H2O Program. This program was the first science-engineering research training program in Canada that generally focused on First Nations water and sanitation security (Theroux, 2017) as many First Nations communities are affected by poor water quality (Health Canada, 2017; Theroux, 2017). Therefore, access to safe drinking water resources (i.e., as a human right) has long been an issue for First Nations communities in Canada. Previous studies reveal that lack of financial and technical resources, infrastructure problems, inequitable resource distribution, sociopolitical and governance challenges have, at least in part, framed this issue (Arsenault et al., 2018; Borrows, 1997; Doerfler et al., 2013). Therefore, lacking clean water access has usually forced many First Nations communities to: a) live with multi-year boil water advisories; b) treat water sources with inefficient treatment facilities; and c) address local water resources contamination with little to no policy commitments (Arsenault et al., 2018).

First Nations communities comprise approximately 70% of northern Manitoba's population and the residents of these communities (e.g., Norway House, Cross Lake, Split Lake, and Fox Lake) are heavily dependent on the Nelson River system as a source of livelihood and well-being (Hackett et al., 2018). It is worth mentioning that there are strong and meaningful spiritual connections between water and the First Nations communities. As an example, in their cultures and values, water is not only required to be treated as an animate being (Arsenault et al., 2018) but also should be treated as a relative (Craft and Blakley, 2022). With regard to the importance of the Nelson River, the economy of the First Nations in northern Manitoba also relies on commercial fishery and food services (Statistics Canada, 2013).

The subarctic region of the Nelson River has been influenced by energy resource projects. Anthropogenic activities, and in particular hydroelectric dams along this system, fracture the river and landscape, alter the flow regime of the Nelson River system throughout a year, and change habitats (Craft and Blakley, 2022). These human activities consequently affect sediment generation, transport, and deposition processes and may lead to environmental problems such as excessive sediment production and transport.

Given spiritual connections among water and the First Nations communities (Arsenault et al., 2018), hydroelectric dams along this system deeply affect communities' culture, values, social heritage and their overall health and well-being (Hackett et al., 2018). One of the most significant contributions of this study is to assess the role of climate change and hydro-electric regulation on sediment sources and transport in the Nelson River (i.e., as an important resource for the First Nations communities in northern Manitoba) before entering Hudson Bay. Sediment transport dynamics and sources data for the Nelson River provide a valuable means of assessing the quality of water resources required by for First Nations communities.

Water, soils, and different forms of sediment samples (e.g., suspended sediment) were collected from different compartments of the study area. In the next step, several analytical techniques were performed to characterise these three different types of samples. Finally, along with using sediment properties and long-term records (>40 years) of suspended sediment loads, the sediment source fingerprinting approach was utilized to determine: a) inorganic suspended sediment sources; b) the influence of lakes and reservoirs on sediment transport; and c) the impacts of natural and human-induced environmental changes on the functioning of sediment mobilization and delivery processes in this regulated system.

1.2 Overview of the study area

The study area is the Nelson River system upstream of the outlet of Split Lake (Figure 1-1). This area includes Lake Winnipeg (23,750 km^2); Split Lake (~274 km^2); the Burntwood River (watershed area = 25,500 km^2; average annual discharge (Q_{ave}) = 917 m^3/s), the Grass River (watershed area = 15,400 km^2; Q_{ave} = 66 m^3/s), and the Upper Nelson River (between Lake Winnipeg and Split Lake; watershed area = 30,800 km^2 excluding the Grass River watershed (discussed below); Q_{ave} = 2,390 m^3/s).

Lake Winnipeg is located in the center of Manitoba, Canada (Figure 1-2). It is the eleventh largest lake by surface area in the world (Newbury et al., 1984), the seventh largest lake in North America (Kimiaghalam and Clark, 2017), and the sixth-largest freshwater lake in Canada (Manitoba Water Stewardship, 2011). The Lake Winnipeg watershed includes portions of four Canadian provinces (Alberta, Saskatchewan, Manitoba, and Ontario) and parts of four US states (Montana, North Dakota, South Dakota, and Minnesota) which covers ~1,000,000 km^2 with approximately 650,000 km^2 of farmland (65%) (Environment and Climate Change Canada and Manitoba Agriculture and Resource Development, 2020; Manitoba Water Stewardship, 2011).

Lake Winnipeg is comprised of three main regions: the South Basin (~2,780 km^2; 10% of the total lake volume; average depth = 9 m; water residence time = 1.3 yr (i.e., 1996-2006); average total suspended solids (TSS) = 14.4 g/m^3), the Narrows (~3,450 km^2; 9% of the total lake volume; maximum depth = 60 m), and the North Basin (~17,520 km^2;

81% of the total volume; average depth = 13.3 m; water residence time = 4.3 yr (i.e., 1996-2006), average TSS = 5.4 g/m^3) (Manitoba Water Stewardship, 2011; Matisoff et al., 2017).

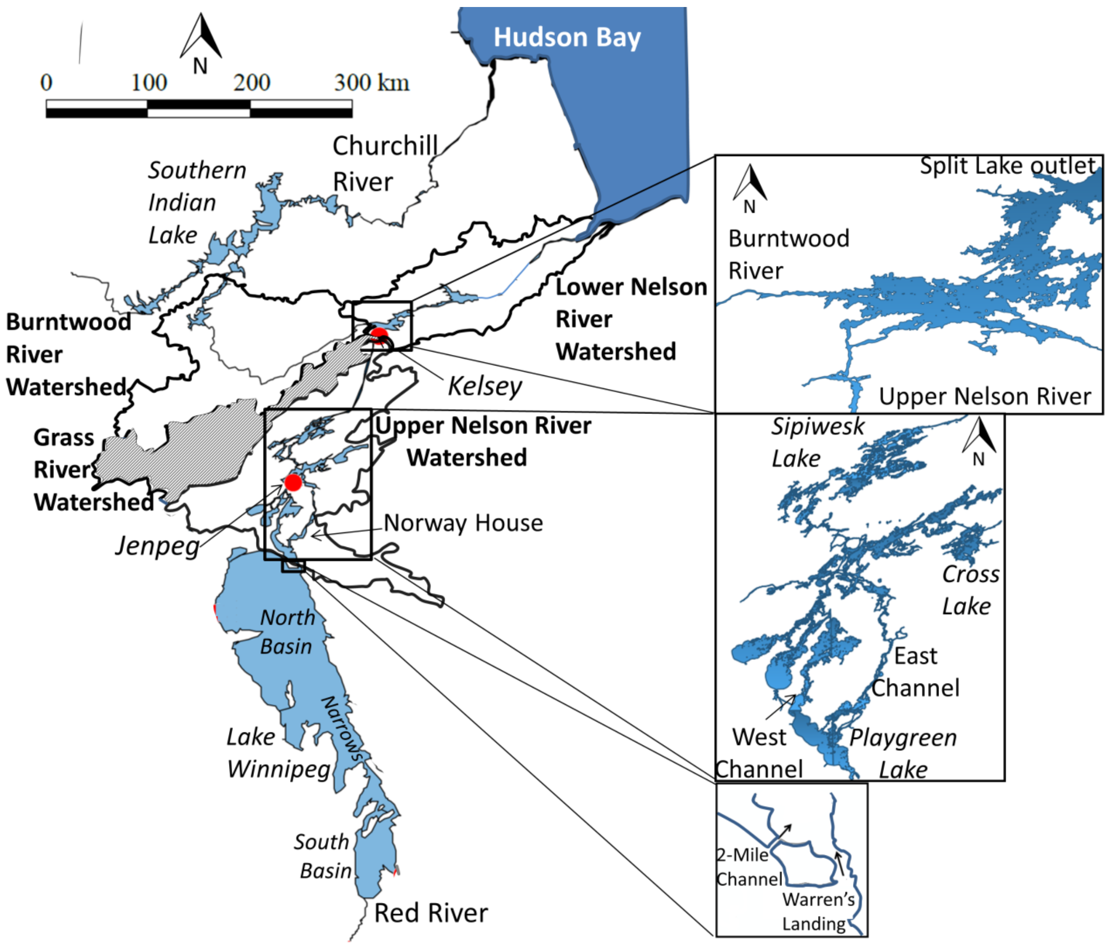

Figure 1-1 Left: study area including Lake Winnipeg and its three distinct regions (South Basin, Narrows, and North Basin), Upper Nelson River, Grass River, Burntwood River, and Lower Nelson River watersheds (red dots represent the Jenpeg and Kelsey generating stations); right (up): Split Lake; right (middle): the Upper Nelson River between Lake Winnipeg and Sipiwesk Lake; right (below): Lake Winnipeg outlets (2-Mile Channel and Warren's Landing). The scale bar refers to the left side of the figure.

The lake receives discharges from several large river watersheds and many smaller tributaries. The Saskatchewan, Red-Assiniboine, and Winnipeg River watersheds are the largest in area and contribute the greatest discharges to the lake (Matisoff et al., 2017). The Winnipeg, Saskatchewan, and Red-Assiniboine Rivers accounted for 49%, 25%, and 16% of water discharges, respectively. The remaining 10% of the water discharges are received from small rivers, including the Dauphin River. The average annual water discharge for the Winnipeg, Red, and Saskatchewan Rivers (i.e., 1996-2006) are 1,064, 346, and 556 m^3/s, respectively (Manitoba Water Stewardship, 2011).

The lake is also the third largest hydroelectric reservoir in the world and has been regulated since 1976 due to the construction of: a) 2-Mile and 8-Mile Channels at the outlet and ~60 km downstream from the lake outlets, respectively; b) the Jenpeg hydroelectric generating station ~100 km downstream of Lake Winnipeg; and c) a control structure at Cross Lake ~130 km downstream of Lake Winnipeg (Figure 1-1).

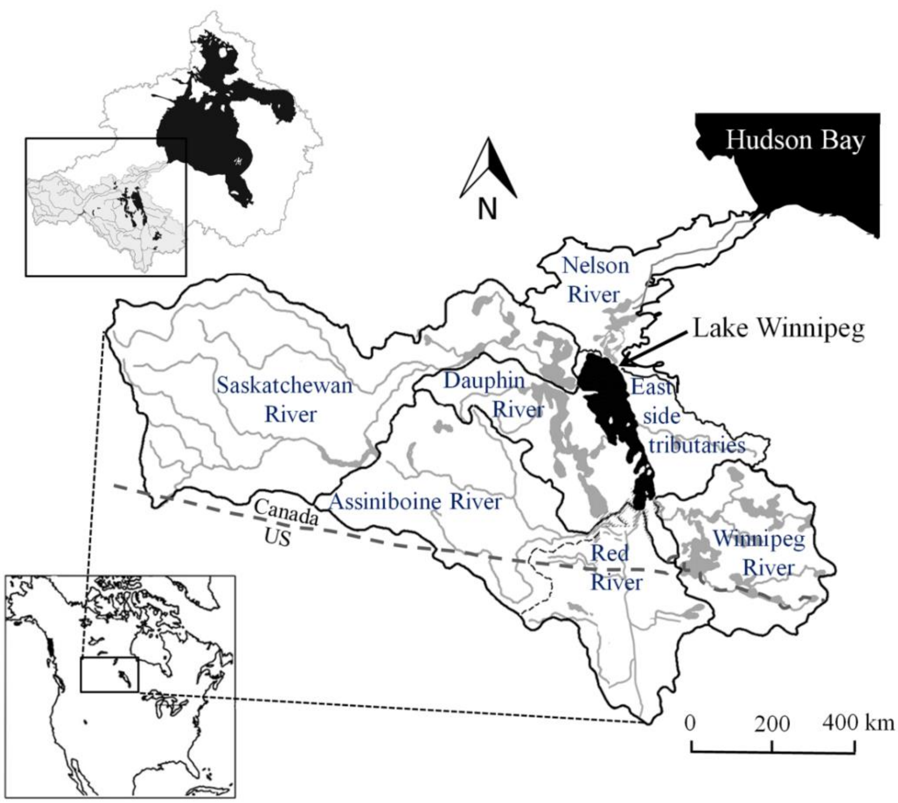

Figure 1-2 the Lake Winnipeg and Nelson River watersheds.

The Nelson River continental-scale watershed (1,125,520 km^2) is a complex and regulated fluvial environment (Figure 1-1 and Figure 1-2). The Nelson River is the third largest in terms of water discharge in Canada and provides 16.3% of freshwater inflow to Hudson Bay (Duboc et al., 2017). It originates from Lake Winnipeg which is at an elevation of 218 m above sea level (Newbury et al., 1984) at 2-Mile channel and Warren's Landing outlet. This 680 km river with an average discharge of 3,540 m^3/s flows into Hudson Bay through relatively steep granitic and gneissic bedrock of the Precambrian Canadian Shield (Newbury et al., 1984).

Since 1976, 75% of the Churchill River discharge (Q_{ave} ~700 m^3/s) has been diverted to the Burntwood River to increase flows through the Nelson River. The merging point of the Nelson River and Burntwood River (i.e., Split Lake) divided the Nelson River into the Upper and Lower Sections. A number of dams have been constructed in the Upper and Lower sections of the Nelson River taking advantage of the Precambrian Shield bedrock river channel and high flow rate. The first dam on the Nelson River (i.e., Kelsey) was completed in 1961 on the Upper Nelson River (Manitoba Hydro, 2015). Manitoba Hydro later constructed a cascade of five dams on the Upper and Lower Nelson River to take advantage of a 218 m drop over a 680 km river section namely Kettle, Jenpeg, Long Spruce, Limestone, and Keeyask which were completed in 1974, 1979, 1979, 1990, 2021 respectively. It should be noted that the Lower Nelson River is not part of the study area of this thesis, but is the subject of on-going research.

1.3 Goals and objectives

The overall goal of this thesis is to develop an improved understanding of sedimentary processes along the Nelson River System. Given that the Nelson River is heavily regulated and includes several lakes and reservoirs, it is essential to: a) investigate the influences of these features on sediment sources and dynamics; and b) explore the impacts of environmental changes on the functioning of these features (the main objective of the BaySys research project).

To achieve this main goal, it is also important to collect representative suspended sediment samples from different components of this river system. Because of the broad scope of this topic, this thesis focused on a number of objectives to meet the overall goal:

1. Assess the sedimentation dynamics within Lake Winnipeg and thereby determine the role of this large lake in regulating sediment transport from the upstream watershed to the downstream river system.
2. Investigate the sediment sources and transport dynamics in the Upper Nelson River (i.e., downstream of Lake Winnipeg) and the Burntwood River system and assess the impacts of natural and human-induced environmental changes on the functioning of these sources and sinks.
3. Explore the options for collecting representative suspended sediment samples of sufficient mass to assess their quality and to determine their sources across different compartments of the study area (e.g., multiple scales of lakes and reservoirs, as well as large, regulated rivers).

1.3.1 Sedimentation dynamics within Lake Winnipeg and its role in sediment transport in the downstream river system

This research objective endeavors to achieve at least three purposes. Firstly, determine the properties of lake bottom sediment and patterns of sediment accumulation rates so as to assess sediment storage within the lake. Secondly, construct a total (i.e., organic and inorganic) sediment budget to explore the extent of dis-connectivity of the riverine suspended sediment sources in the lake. Finally, quantify the contribution of the main suspended sediment sources at the Lake Winnipeg outlet and identify the role of this lake in modifying sediment transport to the downstream river system (the Upper Nelson River system).

1.3.2 Sediment sources and transport in the Burntwood River and the Upper Nelson River

Sediment source fingerprinting and long-term records (>40 years) of suspended sediment loads in these two rivers were used to identify the dominant sediment sources and their significance in sediment transport dynamics. In addition, they were used to investigate the importance of natural and human-induced environmental changes in altering sediment loads and budgets. Moreover, the role of Split Lake (which represents the outlet of the Burntwood River and Upper Nelson River watersheds) on downstream delivery of sediment was assessed by constructing a total (i.e., organic and inorganic) suspended sediment budget for the lake.

1.3.3 Collecting representative suspended sediment samples within the study area

Quantifying the physical and biochemical properties of suspended sediment particles in fluvial systems is an important part of understanding environmental, geomorphological, and hydrological processes operating within watersheds. A comprehensive quantification of sediment properties (e.g., particle size, organic matter/carbon content, and contaminants) usually requires large, bulk samples of sediment given the mass of material required for multiple laboratory analyses. Suspended sediment sampling devices, therefore, play a key role in the understanding of such processes if they are capable of collecting a representative sample of the ambient fluvial fine suspended sediment. There are two main types of suspended sediment sampling equipment: discrete and time-integrated devices. While the former ones collect sediment samples at specific times within a hydrological event, the latter samplers, which are also passive samplers, obtain samples from a given location over an extended period. This part of the project: a) provides detailed assessment of a commonly used time-integrated sampler; and b) examines the performance of two adapted discrete devices to collect a sample of large mass of ambient suspended sediment.

1.3.3.1 Assessing a time-integrated fine suspended sediment sampler

Assessment of the source, transport, and fate of suspended sediment and sediment-associated nutrients and contaminants is heavily dependent on the accuracy of the suspended sediment sampling equipment used to provide truly representative samples of ambient sediment. It has been well documented that the source composition of the sediment can vary within and at the outlet of watersheds through time. Spatiotemporal variability in precipitation and sediment mobilization processes; reworking of deposited sediments within watersheds; and anthropogenic activities are likely some of the factors responsible for possible variations in sediment sources over time. Passive sampler devices have the potential to provide representative suspended sediment samples which are able to overcome these constraints as they collect sediment over a prolonged period.

An affordable and easily fabricated time-integrated fine sediment sampler for use in small (first and second order) rivers was developed by Phillips et al. (2000). The original aim of this sampler was to obtain substantial quantities (e.g., ≥10 g) of fine sediment over either a single (e.g., freshet) or a combination (e.g., rainfall) of events, in the absence of a power source. In this study, the hydrodynamic behaviour of the sampler developed by Phillips et al. (2000) and its efficiency in terms of mass collection and particle size distribution, and its capability for assessing sediment flux within river systems was assessed.

1.3.3.2 Evaluation of continuous-flow centrifuge and continuous-flow filtration system techniques for sampling suspended sediment

The reliable interpretation of any sampler-derived data regarding sediment transport dynamics in the Nelson River system is significantly linked to the accuracy of sampling devices. Although, the time-integrated sediment sampler (described in Section 1.3.3.1) has been shown to collect representative suspended sediment in transit in first and second order streams, it may not be the best option for sampling in large rivers with strong currents and significant variations in water level. In addition, this sampling device is not suitable for lakes and reservoirs as these aquatic environments may not have the required water current conditions for the time-integrated sediment sampler.

One of the approaches for collecting enough representative suspended sediment samples to achieve detailed geochemical analysis is discrete or point-sampling samplers. Assessing the application of centrifuge and filtration techniques for collecting suspended sediment in lakes, reservoirs, and rivers is, therefore, part of this study. The sampling devices include using high-volume continuous-flow filtration (PENTEK filter bag with 1 μm nominal pore size) and high-volume continuous-flow centrifuge systems (US M-512 manual centrifuge). The sampling efficiency of each device on the collection of suspended sediment in different environments was examined.

1.4 Thesis structure

Following the general introduction in this chapter, each objective will be described and expanded in "grouped manuscript" style in its own chapter (Chapters 2 to 5). Therefore, each Chapter includes sections on introduction, methods, results, discussion, and conclusions. Chapters 2-5 were previously published in peer reviewed journals. The order of the Chapters in this document is based on the chronological order of the published papers and not based on the order of the objectives (Figure 1-3).

Chapter 2 contributes to objective 3 by: a) characterising the hydrodynamic behaviour of a commonly used time-integrated fine sediment sampler using an acoustic Doppler velocimeter in controlled laboratory conditions; and b) measuring the mass collection efficiency of the sampler by an acoustic Doppler current profiler under field conditions.

Chapter 3 also contributes to objective 3 by collecting representative suspended sediment samples using two different discrete samplers. A field study was conducted in three different freshwater systems in Manitoba, Canada to examine and compare the performance of two high-flow rate systems as alternative approaches: the M512 continuous-flow centrifuge (M512); and continuous filtration using PENTEK 1 μm filtration bags (filtration system).

Chapter 4 presents a comprehensive study to achieve objective 1 by explaining the relative, and sometimes conflicting, influence of the key natural controls on sedimentation dynamics within Lake Winnipeg (e.g., sediment properties, local morphology, and lake hydrodynamics). This chapter also includes a total (i.e., organic and inorganic) sediment budget for the lake. This study will also compare the sediment budget of Lake Winnipeg to those of the Laurentian Great Lakes (i.e., Lakes Erie, Huron, Michigan, Ontario, and Superior) and will highlight the dominant source of the sediment loading to these large lakes.

Chapter 5 presents a study on sediment sources and transport in the Upper Nelson River and the Burntwood River and the effects of natural and human-induced environmental changes (e.g., climate forcing of hydrological changes, construction of dams, and cross-watershed water diversion) on sediment transport dynamics. This chapter will address objective 2 by analysing the spatiotemporal variability of sediment loads over >40 years and using the sediment source fingerprinting technique in these two large, regulated rivers. The impacts of different scales of natural lakes and hydroelectric reservoirs on sediment sources and transport in the Upper Nelson River and the Burntwood River will also be examined. In addition, the influence of Split Lake on the downstream delivery of sediment will be assessed by constructing a suspended sediment budget for the lake.

Chapter 6 summarizes the previous chapters by reviewing the significant findings. In addition, it also provides detailed recommendations for future work in the Nelson River system and its estuary and presents the implications of this research for local First Nations.

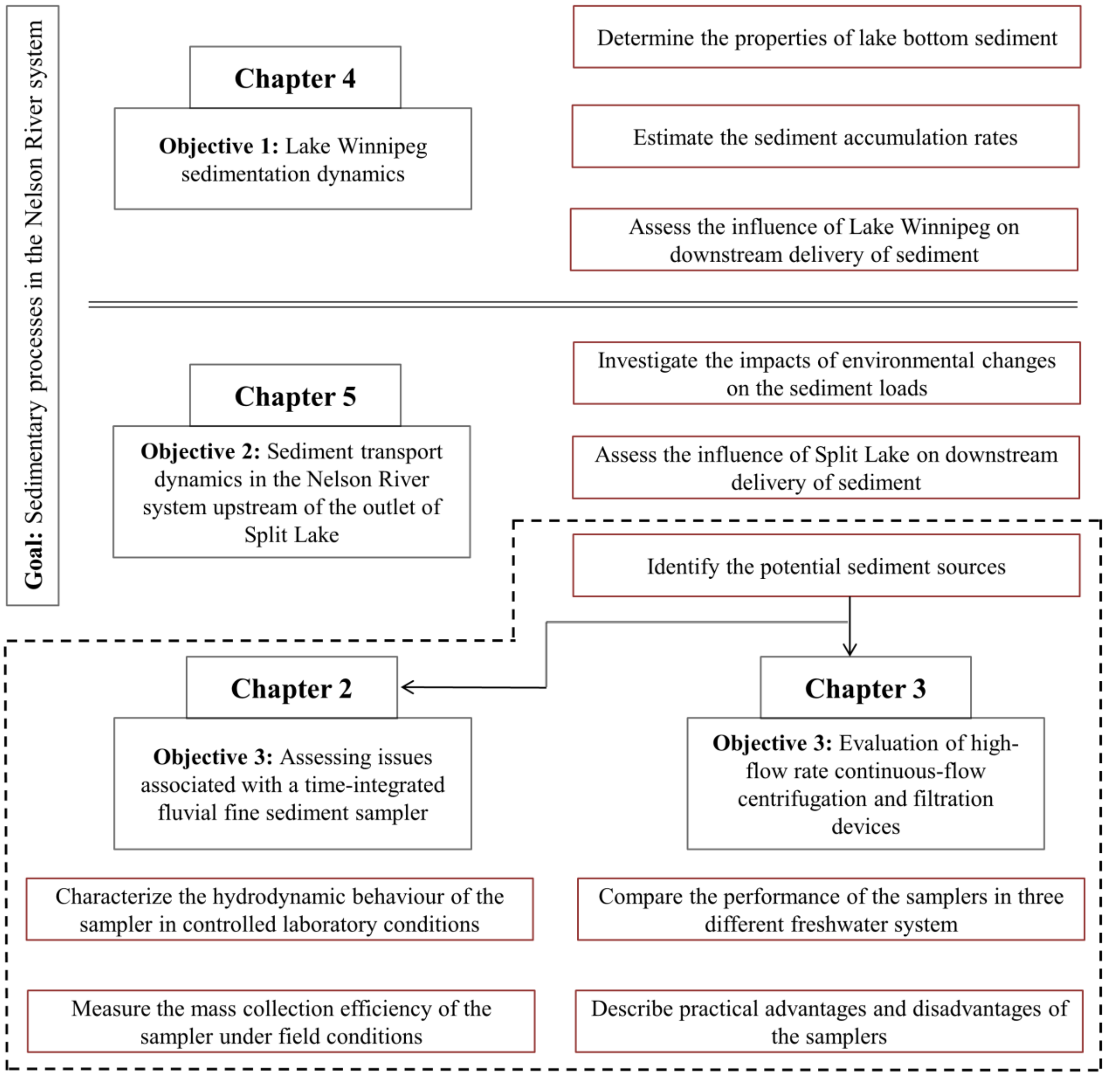

Figure 1-3 A conceptual diagram of research goal, objectives, and approaches divided by chapter. Red boxes denote simplified approaches.

1.5 References

Arsenault, R., Diver, S., McGregor, D., Witham, A. and Bourassa, C., 2018. Shifting the framework of Canadian water governance through Indigenous research methods: Acknowledging the past with an eye on the future. Water, 10,49, w10010049.

Barber, D., 2015. BaySys - Contributions of climate change and hydroelectric regulation to the variability and change of freshwater-marine coupling in the Hudson Bay System (Research Proposal). Centre for Earth Observation Science, University of Manitoba.

Borrows, J., 1997. Living between Water and Rocks. First Nations, Environmental Planning and Democracy. University of Toronto. 47, 417–468.

Corcoran, M.K., and Kelley, J.R., 2006. Sediment-Tracing Technology: An Overview. ERDC TN-SWWRP-06-10. Vicksburg, MS: US. Army Engineer Research and Development Center.

Craft, A. and Blakley, J. (Eds.). 2022. In Our Backyard: Keeyask and the Legacy of Hydroelectric Development. University of Manitoba Press.

Davis, C.M., Fox, J.F., 2009. Sediment fingerprinting: review of the method and future improvements for allocating nonpoint source pollution. Journal of Environmental Engineering. 135, 490–504.

Doerfler, J.; Sinclair, N.J.; Stark, H.K. (Eds.). 2013. Centering Anishinaabeg Studies: Understanding the World through Stories, Michigan State University Press: East Lansing, MI, USA, United States & University of Manitoba Press: Winnipeg, MB, Canada, ISBN 978-1-60917-353-1.

Duboc, Q., St-Onge, G., Lajeunesse, P., 2017. Sediment records of the influence of river damming on the dynamics of the Nelson and Churchill Rivers, western Hudson Bay, Canada, during the last centuries. The Holocene. 27(5), 712-725.

Environment Canada, Manitoba Water Stewardship., 2011. State of Lake Winnipeg: 1999 to 2007. Manitoba Stewardship Board & Environment Canada, Winnipeg, Manitoba. 209 pp.

Environment and Climate Change Canada, Manitoba Agriculture and Resource Development., 2020. State of Lake Winnipeg-2nd Edition-Highlights. Manitoba Agriculture and Resource Development & Environment and Climate Change Canada, Winnipeg. 175 pp.

Hackett, P., Liu, J. and Noble, B., 2018. Human health, development legacies, and cumulative effects: environmental assessments of hydroelectric projects in the Nelson River watershed, Canada. Impact Assessment and Project Appraisal, 36.413-424.

Haddadchi, A., Ryder, D.S., Evrard, O. and Olley, J., 2013. Sediment fingerprinting in fluvial systems: review of tracers, sediment sources and mixing models. International Journal of Sediment Research. 28(4), 560-578.

He, Q., Walling, D.E., 1996. Interpreting particle size effects in the adsorption of 137Cs and unsupported 210Pb by mineral soils and sediments. Journal of Environmental Radioactivity. 30(2), 117-137.

Health Canada., 2017. Drinking water advisories: First Nations south of 60. Ottawa, ON: Health Canada.

Kimiaghalam, N., Clark, S.P., 2017. Morphodynamics of diversion channels in Northern Manitoba, Canada. Canadian Water Resources Journal / Revue canadienne des ressources hydriques. 42:2, 149-162, DOI: 10.1080/07011784.2016.1249961.

Krishnappan, B.G., Chambers, P.A., Benoy, G., Culp, J., 2009. Sediment source identification: a review and a case study in some Canadian streams. Canadian Journal of Civil Engineering. 36, 1622-1633.

Manitoba Hydro, 2015. Regional Cumulative Effects Assessment for Hydroelectric Developments on the Churchill, Burntwood and Nelson River systems: Phase II Report. Winnipeg, Manitoba.

Matisoff, G., Watson, S.B., Guo, J., Duewiger, A., Steely, R., 2017. Sediment and nutrient distribution and resuspension in Lake Winnipeg. Science of the Total Environment. 575, 173-186.

Newbury, R.W., McCullough, G.K., and Hecky, R.E., 1984. The Southern Indian Lake impoundment and Churchill River diversion. Canadian Journal of Fisheries and Aquatic Sciences. 41, 548-557.

Owens, P.N., 2008. Sediment behaviour, functions and management in river basins. In: Owens, P.N. (Ed.) Sustainable Management of Sediment Resources: Sediment Management at the River Basin Scale. Elsevier, pp 1-29.

Phillips, J.M., Russell, M.A., Walling, D.E., 2000. Time-integrated sampling of fluvial suspended sediment: a simple methodology for small catchments. Hydrological Processes. 14(14), 2589-2602.

Statistics Canada., 2013. Norway House 17 IRI. Manitoba (Code 4622058). National Household Survey (NHS) Profile. National Household Survey. Statistics Canada Catalogue no. 99-004-XWE. Ottawa, ON: Statistics Canada.

Syvitski, J.P., Peckham, S.D., Hilberman, R. and Mulder, T., 2003. Predicting the terrestrial flux of sediment to the global ocean: a planetary perspective. Sedimentary Geology. 162, 5-24.

Theroux, J., 2017. Characterizing turbidity and identifying sediment sources in Norway House Cree Nation drinking water using sediment fingerprinting. Unpublished MSc Thesis, University of Manitoba, Winnipeg, Canada.

Walling, D.E., 1983. The sediment delivery problem. Journal of Hydrology. 65(1-3), 209-237.

Walling, D.E., 2005. Tracing suspended sediment sources in catchments and river systems. Science of the Total Environment. 344, 159-184.

Walling, D.E., 2013. The evolution of sediment source fingerprinting investigations in fluvial systems. Journal of Soils and Sediments. 13, 1658-1675.

Walling, D.E., and Collins, A.L., 2008. The catchment sediment budget as a management tool. Environmental Science and Policy. 11, 136-143.

CHAPTER 2: Assessing issues associated with a time-integrated fluvial fine sediment sampler

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2.1 Abstract

Collecting a representative time-integrated sample of fluvial fine-grained suspended sediment (<63 µm) is an important requirement for the understanding of environmental, geomorphological, and hydrological processes operating within watersheds. This study: a) characterised the hydrodynamic behaviour of a commonly used time-integrated fine sediment sampler (TIFSS) using an acoustic Doppler velocimeter (ADV) in controlled laboratory conditions; and b) measured the mass collection efficiency (MCE) of the sampler by an acoustic Doppler current profiler (ADCP) under field conditions. The laboratory results indicated that the hydrodynamic evaluations associated with the original development of the TIFSS involved an underestimation of the inlet flow velocity of the sampler which results in a significant overestimation of the theoretical MCE. The ADV data illustrated that the ratio of the inlet flow velocity of the sampler to the ambient

velocity was 87% and consequently it can be assumed that a representative sample of the ambient fine suspended particles entered into the sampler. The field results showed that the particle size distribution of the sediment collected by the TIFSS was statistically similar to that for the ambient sediment in the Red River, Manitoba, Canada. The MCE of the TIFSS in the field trials appeared to be as low as 10%. Collecting a representative sample in the field was consistent with the previous findings that the TIFSS is a suitable sampler for the collection of a representative sample of sufficient mass (e.g., >1 g) for the investigation of the properties of fluvial fine-grained suspended sediment. Hydrodynamic evaluation of the TIFSS under a wider range of hydraulic conditions is suggested to assess the performance of the sampler during high runoff events.

2.2 Introduction

Studying fluvial fine-grained suspended sediment (<63 µm) provides an understanding of the environmental, geomorphological, and hydrological processes operating within watersheds. Fine inorganic sediment behaves differently than coarser sediment mostly due to its characteristics such as weight, density, specific surface area, and electrical charge distribution. Lighter fine-grained sediment requires less flow energy for transportation in suspension, and thus is transported further downstream, often reaching the coastal zone, whereas heavy, coarse-grained inorganic sediment moves more slowly, typically near the bottom of rivers (Walling, 2013). The high surface area (up to >100 m^2/g) and electrical charge distribution of fine inorganic suspended sediment are key factors for both flocculation (Droppo, 2001) and the binding of trace elements, organic matter, nutrients, and contaminants which are subsequently transported within rivers (Walling et al., 2000). The properties of fine-grained sediment can also provide information on: a) the relative contribution of the source of sediment (i.e., sediment source fingerprinting: Walling, 2013; Owens et al., 2016); and b) the impact of climate change and anthropogenic activities (e.g., changing land use, gravel mining, dam construction) on watershed processes (e.g., Foster and Lees, 1999; Kondolf et al., 2018). Fine suspended sediments are, therefore, suitable gauges of physical and biogeochemical processes in watersheds (e.g., erosion), as well as useful indicators of nutrient and contaminant conditions in aquatic ecosystems (Owens et al., 2005).

Given the highly sporadic nature of fine sediment transport dynamics in rivers, an ideal sampling method is presumably capable of collecting 100% of the sediment that enters the sampler continuously over a suitable time period (i.e., high-flow event, season, or year), and from the entire river cross-section (i.e., depth- and width-integrated). Since this ideal sampling method is not practical, different samplers have been developed to collect a representative sample of the ambient fluvial fine suspended sediment. Phillips et al. (2000) developed an affordable and easily fabricated time-integrated fine sediment sampler (TIFSS) for use in small (first and second order) streams of lowland areas where velocities are expected to be low (i.e., <1 m/s) with little to no maintenance requirements. This device collects temporally integrated samples from one point in a cross-section. The original aim of the TIFSS was to obtain substantial quantities (e.g., ≥10 g) of fine sediment over either a single (e.g., freshet) or a combination (e.g., rainfall) of events, in the absence of a power source. Typically, this apparatus is composed of an inlet tube, (4 mm (ID) × 15 cm), an outlet tube (4 mm (ID)), and a polyvinylchloride (PVC) chamber (98 mm (ID) × 100 cm) (Figure 2-1).

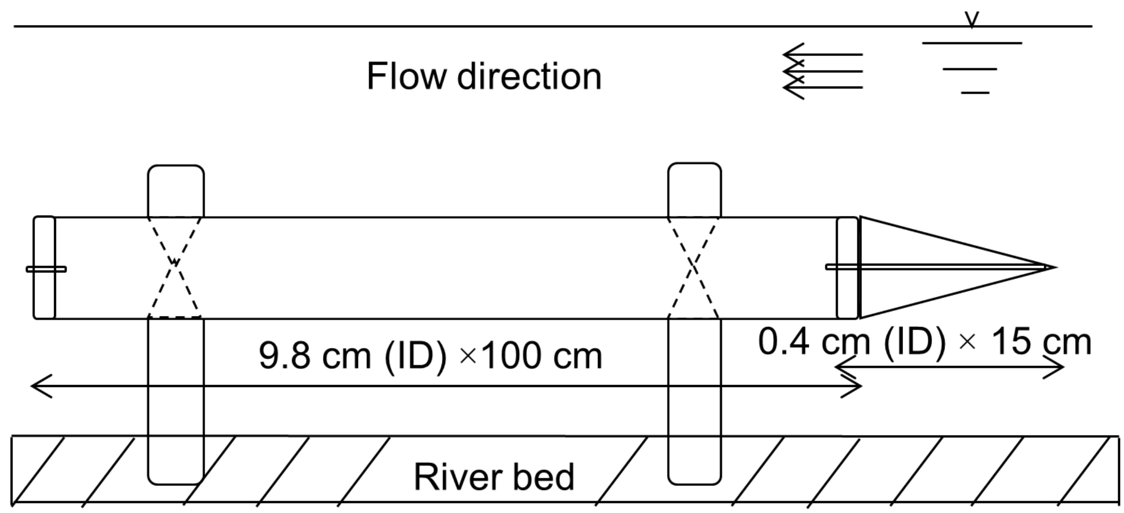

Figure 2-1 Side of time-integrated fine sediment sampler (figure not to scale).

The TIFSS is installed horizontally in the river at ~60% of average depth with the inlet tube facing into the flow. Once the TIFSS is installed in the stream, water passes through the inlet tube at a velocity similar to the ambient flow velocity. The main chamber of the TIFSS decreases the water flow velocity by a factor in excess of 600 relative to the inlet tube flow velocity. Sedimentation of the fine sediment particles occurs as the water passes through the chamber and exits the outlet tube. Given the above characteristics, the TIFSS has the potential to provide an assessment of spatiotemporal sediment delivery processes as this cost-effective sampler can be installed at multiple locations within a hydrological network (e.g., Perks et al., 2017), including in remote settings and estuarine environments (Elliott et al., 2017).

The TIFSS has been extensively used to collect bulk samples of fine suspended sediment from a range of countries and environments, including Australia (Garzon-Garcia et al., 2017), Brazil (Le Gall et al., 2017), Canada (Smith and Owens, 2014a), Finland

(Gonzales-Inca et al., 2018), Japan (Yoshimura et al., 2015) and the UK (Perks et al., 2017). Most published studies tend to focus on the utilization of this device for: a) the assessment of continuous sediment fluxes; b) the properties of the transported sediment (e.g., particle size composition, contaminant, and nutrient content); and c) for identifying sediment sources in river studies. Researchers have also altered the original design of the Phillips et al. (2000) sampler (Table 2-1) to better suit their needs by changing the diameter of the inlet and/or outlet tube, and the length and/or diameter of the main body (McDonald et al., 2010; McDowell and Wilcock, 2007; Perks et al., 2014). Almost all of these modifications follow the basic principles of sedimentation and flow continuity within the sampler.

Table 2-1 Various modified versions of the time-integrated fine sediment sampler (TIFSS).

Study	Inlet tube diameter (mm)	Main chamber		Outlet diameter (mm)
		Diameter (mm)	Length (cm)	
McDowell and Wilcock (2007)	2	48	100	2
McDonald et al. (2010)	2	65	22.8	3
Perks et al. (2014)	8	90	100	8

The hydrodynamic performance of the sampler and its mass collection efficiency (MCE) and particle size distribution (PSD) collection efficiency were first examined by Phillips et al. (2000) through a series of laboratory experiments. The influence of the existence of large composite particles on the PSD collection efficiency of the TIFSS under natural conditions, for rivers in the UK, was also discussed in the same paper. The effectiveness of the sampler, or its modified versions, to collect a representative sample of ambient fine sediment (regarding mass, PSD, and geochemical properties) has subsequently been investigated through several assessments. For example, Russell et al. (2000) evaluated the use of the sampler for characterizing a number of geochemical properties of the fine sediment load for several rivers in the UK. These authors concluded that the sampler statistically represented the ambient fine sediment in terms of its geochemical properties. McDonald et al. (2010) reported a study undertaken in Nunavut in the Canadian High Arctic with the river water velocity as high as 2.5 m/s, aimed at assessing a modified sampler (i.e., Table 2-1) through another field evaluation. They: a) increased the ratio between the diameter of the main chamber and that of the inlet tube to enhance the MCE; b) reduced the length of the main chamber from 1000 mm to 228 mm; c) and placed a 3 mm (ID) outlet tube at the top of the modified sampler, instead of the end, to suit the environment. In this case, and without further evaluation of the hydraulic performance of the modified sampler, application of the TIFSS did not accurately represent the ambient fine sediment concentration and its particle size composition. They argued that higher water velocities (>1 m/s), shorter main chamber, and smaller inlet/outlet tube diameters may have significant effects on the poor performance of the modified version of TIFSS both in terms of MCE and PSD collection efficiency.

Subsequently, Perks et al. (2014) further validated the feasibility of using another modified version of the sampler to assess the fine sediment load of a river in the UK. The TIFSS was found to be inefficient in estimating the absolute fine sediment load, and the sampler underestimated the sediment mass flux between 66% and 99%. However, a significant and constant relationship between samples collected over 2 years and the reference sediment load suggested that the mass of sediment collected by the TIFSS provided a relative measure of the sediment load. This supports the earlier finding of Schindler Wildhaber et al. (2012) for rivers in Switzerland. Perks et al. (2014) also reported statistically similar sediment properties between a number of samplers which were installed at the same river location within their research area. They used indirect approaches, such as the scaling factor and reverse stage–discharge relation methods, to convert the mass collected by the TIFSS samplers to the total sediment mass flux through river cross-sections.

The most recent evaluation of the TIFSS was conducted by Smith and Owens (2014b). They assessed the maximum sediment MCE in a series of flume-based experiments. For two different kinds of sediment (d_{50} = 6.8 ± 0.2 µm and d_{50} = 99.5 ± 0.2 µm) (average ± one standard error) and under an initial ambient suspended sediment concentration of 161 g/m^3, the MCEs of the TIFSS were determined to be 34% and 87%, respectively. They argued that different MCEs for two different medium particle sizes reveals the fact that the MCE is dependent on the PSD of the ambient sediment. They also reported that the sampler collected sediment with representative PSDs during these experiments. However, field testing showed that sediment samples collected by the TIFSS had different concentrations for some metals when compared to nearby samples of fine-grained channel bed sediment in the Quesnel River, British Columbia, Canada, which could, in part, be due to the different time period over which samples were collected, in addition to differences in the particle size composition of the two types of sediment (i.e., bed and suspended).

To date, there has been little direct work to assess the mass and particle size collection efficiencies of the original TIFSS. In addition, the hydrodynamic behaviour of this device has not been rigorously characterized and validated since its inception. In recognition of the need to re-assess the sampler, given its growing use within watersheds, the objective of this study was to re-examine the hydraulic characteristics, as well as the mass and particle size collection efficiencies, of the TIFSS.

2.3 Seminal paper review

Flume tests were conducted in the laboratory by Phillips et al. (2000) to characterize the hydrodynamic behaviour of the sampler by establishing the relation between the flow velocities of the inlet tube and the ambient flow in the flume. Inlet tube flow velocities were estimated under different flume flow velocities over the range of 15.4 to 58.5 cm/s. The inlet tube flow velocity during each flume flow velocity was estimated by measuring water velocity within a 7-mm (ID) glass tube that was placed over the inlet tube and by introducing air bubbles into the open end of the glass tube. The velocity within the glass tube was measured by observing how much time it took for the air bubble to pass along a 500 mm section of the glass tube. The average flow velocity in the smaller inlet tube (i.e., 4 mm (ID)) was then estimated by:

$$y = 1.75\ (x) \qquad \text{(Equation 2.1)}$$

where: y is the average flow velocity in the 4-mm (ID) inlet tube (cm/s); x is the average flow velocity in the 7-mm (ID) glass tube (cm/s); and 1.75 was assumed as the cross-sectional area reduction ratio (or coefficient factor, CF). The logarithmic regression fitting method was then used to demonstrate the close relation (i.e., coefficient of determination (R^2) = 0.99) between measured flume flow velocities and estimated inlet tube velocities:

$$k = 2.074\ (z) - 2.182 \qquad \text{(Equation 2.2)}$$

where: k is the logarithmic inlet tube flow velocity; and z is the logarithmic flume flow velocity.

Using Equation 2.2, Phillips et al. (2000) reported that the inlet tube flow velocities were in the range of 47% to 90% of the flume flow velocity. Therefore, the authors assumed that: a) turbulent flow in the flume, and frictional and inertial forces within the inlet tube were the primary factors causing the decrease in the inlet tube flow velocity relative to the flume flow velocity; and b) the TIFSS was not an isokinetic sampler, meaning that there was a risk of concentrating coarser particles (>63 µm) in the sampler. However, they suggested that the oversampling for coarse sediments was less of an issue since fine suspended sediments were the dominant component of the sediment load in small first and second order lowland streams, and because the <63 µm fraction is often the desired component for most applications (e.g., physical and geochemical properties, contaminant transport, sediment fingerprinting).

The TIFSS efficiency in terms of the mass and PSD collection were assessed in the laboratory by introducing 1,000 g/m^3 of water–sediment mixture into the sampler. In this step, two different chemically and ultrasonically dispersed PSDs of sediment (d_{50} = 3.3 and 10.2 µm) were emptied into buckets, and the sampler was connected to the base of the buckets. The inflowing discharges to the TIFSS were estimated as 0.025 and 0.242 L/min for each bucket, so as to represent ambient flow velocities of 30 and 60 cm/s and inlet tube flow velocities of 3.3 and 32.1 cm/s (i.e., using Equation 2.2). The authors concluded that the sampler was effective in collecting fine suspended sediment across a range of flume flow velocities and across the whole particle size range evaluated, with MCEs between 31 and 71%. The Kolmogorov-Smirnov test (K-S test) demonstrated that only under the low flume flow velocity (30 cm/s) and coarser sediment conditions (d_{50} = 10.2 μm), was the PSD of the collected sample statistically representative of the continuous inflowing sediment. However, they proposed that the mass collection and particle size efficiency of the TIFSS would be expected to increase under natural conditions probably due to the existence of aggregated particles or flocs.

The influence of sediment aggregation on the TIFSS efficiency was examined through field deployment. The sampler was installed in natural rivers in the UK, and suspended sediments were characterized by regularly collecting point samples of ambient sediment during the period of deployment. The authors highlighted two limitations to achieving a robust assessment of the TIFSS: a) a lack of information to quantify the mass of continuous inflowing sediment (i.e., ambient and inlet tube flow velocity); and b) difficulties in comparing the PSD of the TIFSS with the point samples of ambient suspended sediment. They were not able to overcome the first constraint; however, the concentration-weighted average PSD of the point suspended sediment samples was calculated to determine the PSD collection efficiency of the sampler (for details, see Phillips et al. (2000)). The authors stated that both the concentration- and inlet tube flow velocity-weighted average PSD calculations were required to obtain a fully representative sample entering the TIFSS. Nevertheless, as described previously, the lack of information on point ambient velocity was a major constraint to performing a thorough comparison between sediment collected by the TIFSS and the truly weighted average PSD of continuous inflowing sediment (i.e., with reference to the ambient concentration and the inlet tube flow velocity).

Phillips et al. (2000) reported that time-integrated samples were statistically representative of the PSD under natural field conditions. They also suggested that the samples may be representative of the physical and geochemical properties of the ambient fine sediment. It is worth noting that the MCE of the TIFSS is limited by the effective PSD of ambient fine sediments and the residence time of such sediments within the main chamber. Therefore, Phillips et al. (2000) suggested that the TIFSS is suitable for collecting bulk representative samples rather than for estimating the time-integrated mass flux.

The determination of the MCE and PSD collection efficiency of the TIFSS in the laboratory experiments, described above, was dependent on the estimation of inlet tube flow velocity, which must accurately represent ambient flow velocity. There is still uncertainty in knowing the relationship between ambient and inlet tube flow velocities due to some issues. First, the simple air bubble method could be a source of error for estimating the inlet tube flow velocity, and this is discussed further later. Second, based on the mass continuity equation (i.e., $\dot{\mathrm{m}}_{\mathrm{glass\ tube}} = \dot{\mathrm{m}}_{\mathrm{inlet\ tube}}$), the correct CF of the reduced cross-sectional area (A) of the inlet tube should be 3.06 (i.e., $\mathrm{CF} = \frac{\mathrm{A}_{\mathrm{glass\ tube}}}{\mathrm{A}_{\mathrm{inlet\ tube}}} = (\frac{7\ \mathrm{mm}}{4\ \mathrm{mm}})^2$) and not 1.75 (Equation 2.1); the latter value is the ratio of the diameter of the glass tube over the inlet tube ($\frac{7\ \mathrm{mm}}{4\ \mathrm{mm}}$). Third, by using the Phillips et al. (2000) logarithmic equation (i.e., Equation 2.2, considering 1.75 as the reduced cross-sectional area coefficient), the inlet tube flow velocity at a flume flow velocity of 30 cm/s should be 7.6 cm/s and not 3.3 cm/s.

These underestimations in the inlet tube flow velocities provide a longer residence time within the sampler and subsequently increase the MCE of the TIFSS. Therefore, the sediment mass and the measured PSD collection efficiencies (i.e., associated with the flume flow velocities of 30 cm/s and 60 cm/s) in the original laboratory research are likely to be incorrect. Consequently, the maximum mass efficiencies presented in Smith and Owens (2014b) and other similar studies are uncertain because they also used Equation 2.2 to estimate inlet tube flow velocity. Further evaluation should, therefore, be conducted to address the issues just described.

Although the original laboratory assessments of the TIFSS acknowledged that there were some uncertainties (e.g., overestimating the mass and actual PSD collection efficiencies associated with the flume flow velocities of 30 cm/s and 60 cm/s), it has been documented that the sampler provides useful spatiotemporal information of fine sediment transport and associated sediment properties, within watersheds. It is worth noting that the primary purpose of the original TIFSS was to obtain detailed information on the properties of the fine sediment transported in fluvial system by collecting a representative sample of ambient sediment rather than estimating the fine sediment mass flux within a river over time.

2.4 Sample evaluation

In the present study, the relation between the ambient and the inlet flow velocities were investigated in a laboratory flume using an acoustic Doppler velocimeter (ADV). In addition, the efficiency of the sampler in terms of the mass and PSD of the fine suspended sediment collected were assessed in a natural river (Red River, Manitoba, Canada). To obtain the MCE of the TIFSS in the field: a) the ambient river flow velocity at the TIFSS location was measured using an acoustic Doppler current profiler (ADCP) during the period of deployment; b) the average inlet tube flow velocity was then estimated based on continuous river velocity measurement (i.e., using ADCP) and using the empirical relation between ambient and inlet tube flow velocities obtained in the controlled laboratory conditions (i.e., using ADV); and c) the mass of continuous inflowing sediment was calculated using point ambient ADCP velocity measurements and by collecting point river samples during the field evaluation using the following equation:

$$M' = A \times \sum_{j=1}^{n} (T_j \times V_j \times TSS_j) \qquad \text{(Equation 2.3)}$$

$$j = 1, 2, \ldots, n$$

where: M' is the mass of continuous inflowing sediment (g); A is the internal inlet tube cross-section area (0.126 cm^2); T_j is the time interval between two subsequent point samples (s); V_j and TSS_j are the inlet tube flow velocity (cm/s) and total suspended solids concentration (g/m^3) associated with each time interval (i.e., $\frac{\partial TSS}{\partial T}$ or $\frac{\partial V}{\partial T} \neq 0$); j is the sample number; and n is the total number of samples. The TSS concentration for each point sample (i.e., water–sediment bottle sample collected from the river) was obtained by filtering a well-mixed, measured volume of that sample through Whatman GF/F pre-weighed glass fibre filters (0.7 µm pore size) in the laboratory according to the ASTM standard D3977-97 (ASTM, 2013).

Evaluation of the PSD collection efficiency of the sampler was based on the absolute PSD using a laser diffraction particle sizer (Malvern Mastersizer 2000, Malvern, UK). The absolute PSD for each point sample and sediment collected by the sampler were obtained after performing standard pre-treatment procedure including using hydrogen peroxide to remove organic matter and adding a 0.4% solution of sodium hexametaphosphate to disperse and homogenize inorganic sediment samples (see Phillips et al. (2000) for more details). Given the ADCP velocity measurements and the TSS concentration values at each time interval throughout the study period, the velocity-weighted and concentration-weighted average PSD of the point suspended sediment samples was calculated using:

$$D_i = \frac{\sum_{j=1}^{n}((d_i)_j \times V_j \times TSS_j)}{\sum_{j=1}^{n}(V_j \times TSS_j)} \qquad \text{(Equation 2.4)}$$

$$j = 1, 2, \ldots, n$$

where: D_i and d_i are the diameter that i% of particles are smaller in weighted average PSD and point samples, respectively.

2.4.1 Hydrodynamic characteristics: laboratory evaluation

2.4.1.1 Experimental setup and measurement procedure

The laboratory experiments were conducted in a rectangular flume in the Hydraulics Research & Testing Facility at the University of Manitoba, Canada. The flume was 13.1 m long, 0.95 m wide, 0.71 m deep, and had a bed slope of 0.0056. The flume bed was sheet metal and the walls were glass. A flow straightener consisting of several rows of PVC pipes (0.14 m diameter, 0.25 m long) surrounded by two wire mesh screens was located in the upstream head tank to filter the flow. A louvered tailgate manually controlled water level at the downstream end of the flume and discharge was measured by an ultrasonic flow meter. In this study, the flume discharges were controlled in a way that average flume flow velocities are similar to those of Phillips et al. (2000) (i.e., five velocities in the range 15 to 60 cm/s).

A down-looking Vectrino II profiling ADV (Nortek, Rud, Norway) was used to measure inlet tube flow velocity. The ADV was mounted on a frame over the flume (Figure 2-2), so it could be moved in the vertical direction and along the centerline of the flume. Prior to conducting the experiments, the ADV probe was aligned in the streamwise and vertical directions using a small level, and an initial experiment was undertaken to determine the length of flume required for the flow to be fully developed. Under the minimum flow velocity of the flume (i.e., 15 cm/s), the flow became fully developed at 7 m downstream of the flume inlet; therefore, the TIFSS was located at 9 m from the flume inlet. For all tests, the water level was maintained constant at 30 cm, and the TIFSS was installed at the centerline of the flume and in this condition the distance between the inlet tube of the TIFSS and the flume bed was 14.5 cm (i.e., ~50% of the water depth). As designed by the manufacturer the center of the sampling volume of the ADV was located 5 cm below the sensors to avoid any disturbance of the flow structure at the focus point of the transmitter. Preliminarily tests were conducted to determine an appropriate closest horizontal distance of the ADV from the TIFSS inlet tube. Hence, the ADV collected velocity data for 20 minutes at different horizontal distances of 0, 5, 10, 15, and 20 mm to the sampler inlet tube. Analysis of the data indicated that 5 mm was the smallest distance that the ADV could record long-term accurate velocity measurements. As all of the flows associated with the five velocities were subcritical (i.e., Froude number less than unity), it is reasonable to assume that flow conditions were controlled from downstream. In other words, flow conditions including the frictional drag forces associated with the inlet tube walls or internal controls on velocity within the inlet tube of the TIFSS were transmitted upstream. Thus, it can be assumed that the flume flow velocity measurements acquired 5 mm upstream of the inlet tube were representative of the velocities within the inlet tube.

Figure 2-2 The time-integrated fine sediment sampler and acoustic Doppler velocimeter used to determine the inlet tube flow velocity in the laboratory-based evaluation

To measure the flow velocity profile in front of the sampler, the ADV sampling volume was placed at three locations: one directly in front of the TIFSS inlet tube; one 7 mm vertically above; and one 7 mm below the sampler. For each of the measurement locations, the flow velocity was recorded for 20 minutes at a frequency of 100 Hz. The ADV sampling volume was divided into seven separate cells of 1 mm height where water velocity was simultaneously measured, which resulted in a total of 21 water velocity measurement points in front of the sampler inlet. After the measurements in the laboratory, a post-processing code in MATLAB was utilized to filter the data set based on the correlation and signal to noise ratio (SNR) values. In each cell, if more than 10% of measured data had a correlation below 30% or had a SNR value less than 15 dB, the cell was eliminated.

2.4.1.2 Laboratory data analysis

The velocity profile distributions in front of the TIFSS show that for each run (i.e., different flume flow velocities) the minimum velocity occurred at the inlet tube and increased away from the centerline of the sampler (Figure 2-3). The results of the relation between the measured inlet tube flow velocities (i.e., ADV velocity profile measurements) and flume flow velocities in the range 15 – 60 cm/s are presented in Figure 2-4, which demonstrates a strong, statistically significant linear relationship. Figure 2-4 also illustrates the relation between the measured flume flow velocities and inlet tube flow velocity estimations associated with: a) the original development by Phillips et al. (2000) (i.e., using the diameter of the glass tube over the inlet tube ($\frac{7\text{ mm}}{4\text{ mm}}$) as the reduced cross-sectional area coefficient; Equation 2.2); and b) the air bubble method (i.e., Phillips et al. (2000) procedure) with the correct cross-sectional area reduction ratio (i.e., $(\frac{7\text{ mm}}{4\text{ mm}})^2$). Figure 2-4 demonstrates that estimated inlet tube flow velocities with the correct CF for the reduction ratio (i.e., 3.06) at 60 cm/s of flume flow velocity is in agreement with the ADV inlet tube flow velocity measurements. It can also be seen that the percentage differences between the flume flow and estimated inlet tube flow velocities with the correct CF at five flume flow velocities are not consistent and this percentage increases as the flume flow velocity decreased (e.g., 78% and 6% at flume flow velocities of 15 and 60 cm/s, respectively). Possible explanation for this inconsistency is that at lower flume flow velocities, the frictional and inertial forces within the 600 mm glass tube exerted additional limitation on the inlet tube flow velocity. Moreover, Phillips et al. (2000) observed (i.e., by introducing an air bubble) that developing turbulent flow at higher flume flow velocities (i.e., >60 cm/s) caused significant decreases in the inlet flow velocity. Therefore, it is most likely that the air bubble method for estimating inlet tube flow velocity may not be a precise method for whole desired range of flume velocities and could be a source of error. It can also be seen that for developing a regression, logarithmic transformation of the flume flow velocities and inlet tube flow velocities is not required.

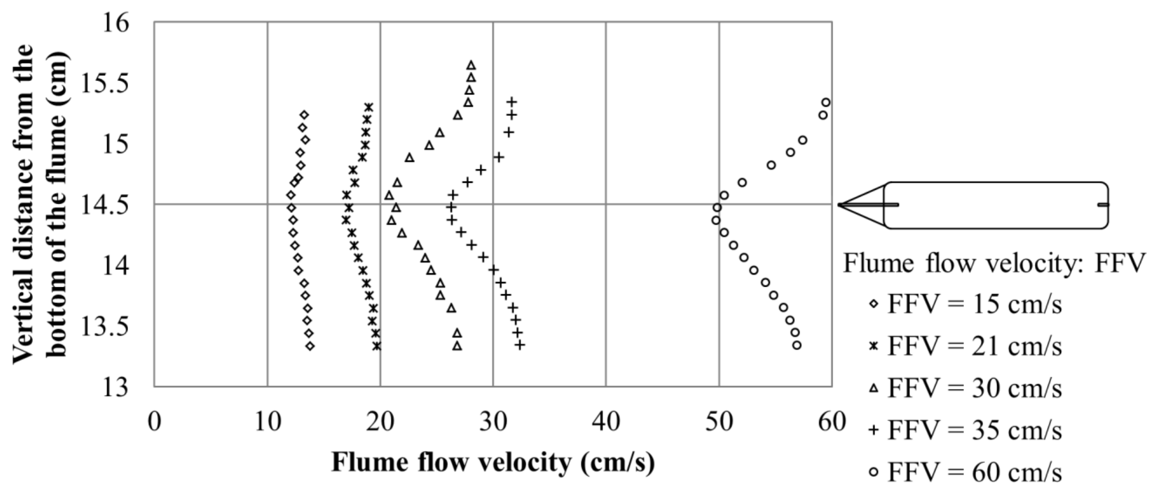

Figure 2-3 Velocity profiles in front of the time-integrated fine sediment sampler for various flume discharges

The results of the laboratory test of the hydrodynamic characteristics of the sampler (Figure 2-4) highlight the marked increases in the measured inlet tube flow velocities compared to the original estimated inlet flow velocities (i.e., air bubble method; CF = 1.75). The inlet tube flow velocities at flume flow velocities of 60 and 30 cm/s, for example, were measured (i.e., using the ADV) to be 52 and 24 cm/s, respectively, and were estimated (i.e., by Phillips et al. (2000)) to be 32.1 and 3.3 cm/s, respectively. Therefore, the MCEs of the TIFSS in the original laboratory tests by Phillips et al. (2000) were influenced by underestimations of the inlet tube flow velocities.

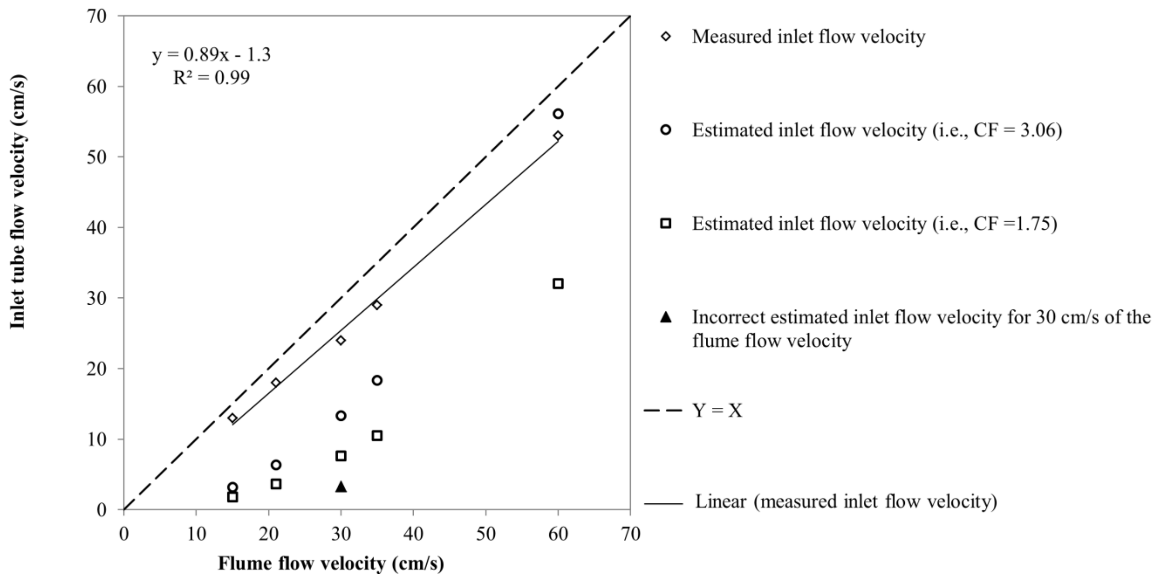

Figure 2-4 Linear relationship between ambient flume flow, measured inlet, and estimated inlet tube flow velocities. CF, coefficient factor

Despite these uncertainties associated with the MCEs in the original laboratory assessment, three points are worthy of attention regarding the performance of the TIFSS. First, the original purpose of the design and development of the TIFSS was to collect a representative sample of substantial mass (e.g., >10 g) rather than estimating the continuous fine sediment mass flux. Second, and closely related to the first point, the two previous studies which assessed the use of the original TIFSS (i.e., not its modified versions) for collecting fine sediments in natural river systems (i.e., Phillips et al., 2000; Smith & Owens, 2014b), show that the sampler collects a representative sample of ambient fine suspended sediment. Third, the ADV velocity measurements (i.e., Figure 2-4) confirmed that the ratio of the inlet tube flow velocity to the flume flow velocity (i.e., inflow efficiency) within the range 15 – 60 cm/s was 87% ± 2% (average ± one standard error), which means that the TIFSS can be considered as an isokinetic sampler since this ratio is in the acceptable range of 1.00 ± 0.15 (Szalona, 1982). In relating the TIFSS inflow efficiency (i.e., 87%), it can be also assumed that a representative sample of fine suspended sediment in ambient water (in terms of concentration and PSD) enters into the sampler (Garcia, 2008). Given the fact that a major, if not the dominant, part of the fine-grained sediment load transported in many rivers is composed of aggregates and flocculated particles (Droppo, 2001), the TIFSS collects a sufficient mass of such particles according to the basic principles of sedimentation (i.e., settling velocity). As a result, it is unlikely that the issues raised above will influence the functionality of the TIFSS for the collection of a representative sample of suspended sediment under most field conditions.

2.4.2 Field evaluation

After obtaining the empirical relation between ambient and inlet tube flow velocities (i.e., Figure 2-4), the TIFSS was attached to a frame and installed approximately 5 m from a riverbank in the Red River in Winnipeg, Manitoba, Canada (14 U, 634498E, and 5518722N) for three days in November 2016. This short duration and one-time field sampling was performed to provide additional insight into the ability of the sampler to collect representative samples as well as to evaluate a novel approach for measuring its MCE in the field. The Red River is a wide, open channel (Goharrokhi, 2015) and its suspended sediment load is mainly composed of clay- and silt-sized particles

(Kimiaghalam et al., 2016). Some of the primary characteristics of the Red River during the ice-free period are reported in Table 2-2 (Goharrokhi, 2015; Kimiaghalam et al., 2016).

Table 2-2 Primary properties of the Red River, Manitoba, Canada, during the ice-free period

Property	Maximum	Minimum	Average
Bed gradient (m/km)	-	-	0.04
Water surface elevation (m)	229.0	222.5	223.6
Top width (m)*	159	115	130
Thalweg depth (m)*	10.5	4.5	5.5
Hydraulic radius (m)*	7.0	2.5	3.9
Velocity (m/s)*	1.08	0.10	0.64
Discharge (m^3/s)	1300	50	176
Suspended sediment concentration (g/m^3)	1500	10	121

*: at a monitoring site located 4 km upstream of the TIFSS location

A local water level recorder was installed upstream of the sampler prior to the beginning of the project which measured water level every 15 minutes. Continuous water level monitoring was used to observe potential hourly and daily stage variations. Continuous point velocity measurements were obtained for 1 hour at a frequency of 1 Hz using an ADCP RiverSurveyor M9 (SonTek, San Diego, USA) which was mounted on the TIFSS frame. In addition, 1 l bottle and 7 l bucket point samples were collected every 6 hours at the TIFSS location to: a) capture the variation of TSS concentrations using the water bottle samples; b) perform PSD analysis after collecting sediments from the 7 l water–sediment mixture bucket samples by allowing sediments to settle for 7 days (described later); c) calculate the weighted average PSD for the point samples; and d) assess the PSD collection efficiency of the sampler. The TIFSS was retrieved after 3 days and the collected sample was emptied into a 20 l bucket. After 7 days the clear supernatant water of the 20 l and the 7 l buckets was carefully siphoned and the settled sediments were then air dried and retained for subsequent analyses (for details, see Perks et al. (2014)).

The total mass of sediment collected by the TIFSS after 72 hours was 4.7 g which is a sufficient mass for a broad spectrum of analyses including geochemical content, particle size composition, organic matter/carbon content, and colour properties for sediment fingerprinting. The ADCP velocity measurements indicates that the average river velocity for the three-day period and at the TIFSS location was 15 ± 2.1 cm/s and accordingly the average inlet tube flow velocity was calculated as 13 cm/s (see Figure 2-4; inlet tube flow velocity (cm/s) = (0.89 × ambient flow velocity (cm/s) – 1.3). In the Red River, the water surface elevation between bank-full and low flow conditions typically varies by up to 7 m (Table 2-2); however, the variation of the water surface elevation for the entire period (i.e., 3 days) was less than 10 cm. Therefore, the flow regime during the study period was approximately steady state, the velocity distribution at the TIFSS location did not change significantly and, thus, the average ADCP velocity measurement over 1 hour and the average inlet tube flow velocity associated with that (i.e., 15 and 13 cm/s, respectively) can be considered representative of the period of field deployment. The average TSS concentration value was 143 ± 13.5 g/m^3. Given the internal inlet tube cross-section area (i.e., 0.126 cm^2) of the TIFSS, the ADCP velocity measurements, and the TSS concentration values at each time interval throughout the study period, the inflowing sediment mass to the sampler was calculated as 60 g (i.e., using Equation 2.3), and subsequently, the MCE of the sampler was estimated to be 8%. This low measured MCE is consistent with estimations by Perks et al. (2014) which indicates that this device may not be suitable for estimating absolute sediment load; a conclusion that is broadly supportive of the findings of other evaluations (e.g., Phillips et al., 2000; Schindler Wildhaber et al., 2012).

The weighted average PSD for point suspended sediment samples (i.e., concentration- and inlet tube flow velocity-weighted) was obtained using Equation 2.4. The K-S test was applied on the PSD of the time-integrated sample and the weighted average PSD (see Phillips et al. (2000) for more details). The similarity of the PSD shown in Figure 2-5 and the K-S test result (0.025 at 95% confidence level) indicate that the PSD of the suspended sediment collected by the TIFSS is statistically representative of the suspended sediment load of the Red River at the time of deployment.

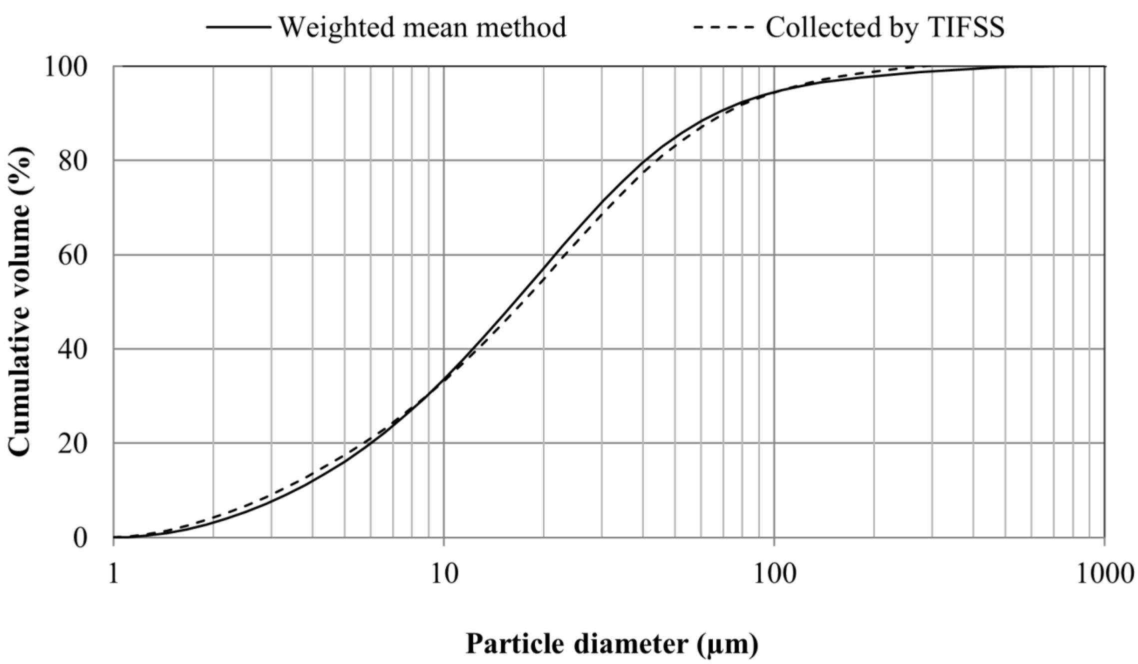

Figure 2-5 concentration-weighted average method and the sediment collected by the time-integrated fine sediment sampler (TIFSS)

2.5 Conclusions

The TIFSS is a well-established and reliable sampler for collecting a representative sample (i.e., in terms of PSD and geochemical properties) from small, first and second order lowland streams over extended periods. However, there are two issues of concern associated with the laboratory evaluations of the TIFSS in the original paper by Phillips et al. (2000). First, is the incorrect cross-sectional area reduction coefficient (i.e., 1.75) which results in an underestimation of the inlet tube flow velocity. Second, and consequently, under the original development of the TIFSS (i.e., using chemically and ultrasonically dispersed fine sediment) the MCEs of the sampler were overestimated. This paper re-examined the performance of the TIFSS using different measurement methods and characterized the potential influences of the issues raised above on the functionality of the sampler. Results from ADV measurements in controlled laboratory conditions demonstrated that the original equation between the ambient and inlet tube flow velocities should be modified. In addition, an ADCP was used to determine the mass of continuous inflowing sediment under field conditions and to calculate the concentration- and velocity-weighted average PSD of the time-integrated sediment entering the sampler. The findings of laboratory experiments indicated that the real inlet tube flow velocity of the TIFSS is significantly higher than previously reported (up to 7.6 times), and in turn, the sampler provides less travel time for the composite particles inside the main chamber, resulting in lower MCE than previously assumed. In contrast with the laboratory findings in Phillips et al. (2000), the ADV measurements illustrated that the TIFSS inflow efficiency was 87%. Therefore, it can be assumed that throughout the sampler's operating velocity, a representative sample of the fine particles of the ambient water enters into the sampler. Field deployment of the TIFSS indicated that the sampler collected a representative sample of fine suspended sediment in terms of PSD which may reflect both the isokinetic behaviour of the sampler and the existence of aggregated particles or flocs as a dominant natural state of fine cohesive materials in river systems. The MCE of the sampler in the field was measured to be as low as ~10%. The results of the field trails support previous studies that found that while the TIFSS collects a statistically representative sample of ambient suspended sediment, care must be exercised when considering the mass of the sediment collected by the sampler as an indicator for estimating the absolute time-integrated mass flux of sediment during the period of field deployment. Given the fact that a large proportion of the suspended sediment flux probably occurs during high runoff events and flood flows, which in natural rivers may be greater than 60 cm/s, it is recommended that future work should examine the effect of higher flow velocities on the performance of the TIFSS.

2.6 References

ASTM standard D3977-97., 2013. Standard Test Methods for Determining Sediment Concentration in Water Samples. ASTM International, West Conshohocken, PA, USA.

Droppo, I.G., 2001. Rethinking what constitutes suspended sediment. Hydrological Processes, 15, 1551-1564.

Elliott, E.A., Monbureau, E., Walters, G.W., Elliott, M.A., McKee, B.A., Rodriguez, A.B., 2017. A novel method for sampling the suspended sediment load in the tidal environment using bi-directional time-integrated mass-flux sediment (TIMS) samplers. Estuarine, Coastal and Shelf Science, 199, 14-24.

Foster, I.D. L., Lees, J.A., 1999. Changes in the physical and geochemical properties of suspended sediment delivered to the headwaters of LOIS river basins over the last 100 years: a preliminary analysis of lake and reservoir bottom sediment. Hydrological Processes, 13, 1067-1086.

Garcia, M.H. 2008., ASCE Manual of Practice 110—Sedimentation Engineering: Processes, Measurements, Modeling and Practice. In World Environmental and Water Resource Congress 2006: Examining the Confluence of Environmental and Water Concerns. pp. 1050

Garzon-Garcia, A., Laceby, J.P., Olley, J.M., Bunn, S.E., 2017. Differentiating the sources of fine sediment, organic matter and nitrogen in a subtropical Australian catchment. Science of the Total Environment, 575, 1384-1394.

Goharrokhi, M., 2015. Effect of hydraulic shear stress on the banks of the Red River. Unpublished MSC Thesis, University of Manitoba, Winnipeg, Canada.

Gonzales-Inca, C., Valkama, P., Lill, J.O., Slotte, J., Hietaharju, E., Uusitalo, R., 2018. Spatial modeling of sediment transfer and identification of sediment sources during snowmelt in an agricultural watershed in boreal climate. Science of the Total Environment, 612, 303-312.

Kimiaghalam, N., Goharrokhi, M., Clark, S.P., 2016. Assessment of wide river characteristics using an Acoustic Doppler Current Profiler. Journal of Hydrologic Engineering, 21, 06016012.

Kondolf, G.M., Schmitt, R.J., Carling, P., Darby, S., Arias, M., Bizzi, S., ..., Oeurng, C., 2018. Changing sediment budget of the Mekong: Cumulative threats and management strategies for a large river basin. Science of the Total Environment, 625, 114-134.

Le Gall, M., Evrard, O., Dapoigny, A., Tiecher, T., Zafar, M., Minella, J.P.G., ... Ayrault, S., 2017. Tracing sediment sources in a subtropical agricultural catchment of Southern Brazil cultivated with conventional and conservation farming practices. Land Degradation & Development, 28, 1426-1436.

McDowell, R.W., Wilcock, R.J., 2007. Sources of sediment and phosphorus in stream flow of a highly productive dairy farmed catchment. Journal of Environmental Quality, 36, 540-548.

McDonald, D.M., Lamoureux, S.F., Warburton, J., 2010. Assessment of a time-integrated fluvial suspended sediment sampler in a high arctic setting. Geografiska Annaler: Series A, Physical Geography, 92, 225-235.

Owens, P.N., Batalla, R.J., Collins, A.J., Gomez, B., Hicks, D.M., Horowitz, A.J., ... Trustrum, N.A., 2005. Fine-grained sediment in river systems: environmental significance and management issues. River Research and Applications, 21, 693-717.

Owens, P.N., Blake, W.H., Gaspar, L., Gateuille, D., Koiter, A.J., Lobb, D.A., ... Woodward, J.C., 2016. Fingerprinting and tracing the sources of soils and sediments: Earth and ocean science, geoarchaeological, forensic, and human health applications. Earth-Science Reviews, 162, 1-23.

Perks, M.T., Warburton, J., Bracken, L., 2014. Critical assessment and validation of a time-integrating fluvial suspended sediment sampler. Hydrological Processes, 28, 4795-4807.

Perks, M.T., Warburton, J., Bracken, L.J., Reaney, S.M., Emery, S.B., Hirst, S., 2017. Use of spatially distributed time-integrated sediment sampling networks and distributed fine sediment modelling to inform catchment management. Journal of Environmental Management, *202*, 469-478.

Phillips, J.M., Russell, M.A., Walling, D.E., 2000. Time-integrated sampling of fluvial suspended sediment: a simple methodology for small catchments. Hydrological Processes, 14, 2589-2602.

Russell, M.A., Walling, D.E., Hodgkinson, R.A., 2000. Appraisal of a simple sampling device for collecting time-integrated fluvial suspended sediment samples. In The Role of Erosion and Sediment Transport in Nutrient and Contaminant Transfer, IAHS Publication No. 263. IAHS Press, Wallingford, 119-127.

Schindler Wildhaber, Y., Michel, C., Burkhardt-Holm, P., Bänninger, D., Alewell, C., 2012. Measurement of spatial and temporal fine sediment dynamics in a small river. Hydrology and Earth System Sciences, 16, 1501-1515.

Smith, T.B., Owens, P.N., 2014a. Individual and cumulative effects of agriculture, forestry and metal mining activities on the metal and phosphorus content of fine-grained sediment; Quesnel River basin, British Columbia, Canada. Science of the Total Environment, 496, 435-442.

Smith, T.B., Owens, P.N., 2014b. Flume-and field-based evaluation of a time-integrated suspended sediment sampler for the analysis of sediment properties. Earth Surface Processes and Landforms, 39, 1197-1207.

Szalona, J.J., 1982. Development of a bag-type suspended-sediment sampler/ A study of methods used in measurement and analysis of sediment loads in streams, Report

Y: Minneapolis, Minnesota, Federal Inter-Agency Sedimentation Project, St. Anthony Falls Hydraulic Laboratory, 32 p

Walling, D.E., Owens, P.N., Waterfall, B.D., Leeks, G.J., Wass, P.D., 2000. The particle size characteristics of fluvial suspended sediment in the Humber and Tweed catchments, UK. Science of the Total Environment, 251, 205-222.

Walling, D.E., 2013. The evolution of sediment source fingerprinting investigations in fluvial systems. Journal of Soils and Sediments, 13, 1658-1675.

Yoshimura, K., Onda, Y., Sakaguchi, A., Yamamoto, M., Matsuura, Y., 2015. An extensive study of the concentrations of particulate/dissolved radiocaesium derived from the Fukushima Dai-ichi Nuclear Power Plant accident in various river systems and their relationship with catchment inventory. Journal of Environmental Radioactivity, 139, 370-378.

CHAPTER 3: Evaluation of high-flow rate continuous-flow centrifugation and filtration devices for sampling and concentrating fine-grained suspended sediment

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3.1 Abstract

Fine-grained (<63 µm) suspended sediment is an important vector for transporting contaminants in aquatic systems. Characterization of physical and biogeochemical properties of suspended sediment usually requires bulk samples to assess its quality and to determine its source. Low-flow rate (~2 to 4 L/min) continuous-flow centrifugation (CFC) systems may need a time period from several hours to one day to collect such samples, and thus, due to their low inflow rate, limits application of these devices. A field study was conducted in three different freshwater systems in Manitoba, Canada to examine and compare the performance of two high-flow rate systems as alternative approaches: the M512 continuous-flow centrifuge (M512); and continuous filtration using PENTEK 1 µm filtration bags (filtration system). It was determined that the mass collection efficiency (MCE) for the M512 (in absolute terms) was similar to low-flow

rate CFC systems. As with low-flow rate CFC systems, the M512 preferentially collected particles of a certain size range (i.e., in the case of M512, particles ≥0.83 μm) and accordingly this may affect the collection of a truly geochemically and physically representative sample in waters containing a high proportion of finer particles (≤1 μm). Several filtration systems in series improved its MCE performance and, in terms of total collected mass, this configuration appears to be as efficient as the low-flow rate CFC systems in an equivalent sampling time. The results of this study confirmed that, in nearly all cases, the filtration system collected a representative sample of ambient suspended sediment, in terms of particle size composition, and geochemical and colour properties. In practice, it is suggested that the filtration systems in series have advantages over M512 as the filtration system is more portable and cost-effective with a lower power demand.

3.2 Introduction

There are an increasing number of studies that are concerned with determining the fluxes and properties of suspended particulate material (inorganic and organic) in aquatic systems such as rivers and lakes. This interest mainly stems from the fact that suspended sediment fluxes are indicative of landscape erosion processes (e.g., Syvitski et al., 2005; Vercruysse et al., 2017) and also because fine suspended sediment is an important vector for transporting chemicals such as carbon, phosphorus and many contaminants (e.g., metals, radionuclides and persistent organic pollutants) (Bilotta and Brazier, 2008; Horowitz, 1991). Consequently, studies have collected suspended sediment samples and determined its physical and biogeochemical properties so as to assess its quality and to determine its source (i.e., sediment fingerprinting; Owens et al., 2016). While many of these properties can be determined on sediment samples of low mass (i.e., <1 g), often bulk samples of between 1 and 10 g are required for a more comprehensive assessment of sediment properties including particle size, organic matter/carbon content, and contaminants.

There are two main approaches to collect large samples of suspended sediment from aquatic systems, namely, active and passive. The passive approach typically involves the use of unattended equipment over a long period of time (i.e., weeks to months) and by employing the principles of sedimentation (Goharrokhi et al., 2019; Phillips et al., 2000). Some practical limitations include strong flows or currents, as well as interference from animals, debris, and boat and ship traffic, which may compromise the use of the passive approach in certain situations. The active approach – which is known as discrete or point-in-time sampling – relies on a power source and the collection of large volumes of ambient water–sediment mixture in much less time (i.e., hours) than the passive approach. With the active approach, the collected water–sediment mixture can be processed in the field (i.e., at site) or by bringing the sample to the laboratory for subsequent processing. The latter procedure, however, suffers from a number of drawbacks such as the need to handle large sample volumes and long processing times. Moreover, the transformation of a number of elements from the dissolved phase to the particulate phase and vice versa during handling and processing in the laboratory has been documented (Horowitz et al., 1989). Considering these potential alterations, the properties of sediments collected by using this approach might not be representative of the ambient suspended sediment properties at the time of collection. In situ point-in-time sampling, whereby a concentrated sediment sample with low water content is obtained, therefore, eliminates the constraints outlined above.

The collection and processing of a large volume of ambient water–sediment mixture in the field is carried out by using pumps to draw such a mixture from aquatic systems and then passing this through additional equipment such as centrifuge and filtration devices. The principle for particle retention by centrifugation techniques depends on both the particle density, which influences settling, and inflow rate, which influences the water–sediment residence time within centrifuge systems (Bates et al., 1983). In other words, heavy individual/composite particles and particles which feed into the spinning bowl under a low inflow rate may be collected more efficiently by the centrifuge devices. Packham et al. (1961) were, perhaps, the first researchers that adapted centrifugation for collecting bulk suspended sediment samples in the field. They used a continuous-flow centrifugation (CFC) device with an inflow rate of 0.125 L/min over approximately one month for a number of rivers in England. Since then, three different commercially available types of this system (single-bowl, multiple-bowl, and bowl with internal vanes and cups) have been employed in many countries, including Canada (e.g., Reid et al., 2020), France (e.g., Abuhelou et al., 2017), Germany (e.g., Keßler et al., 2019), Japan (e.g., Mbabazi et al., 2019), Switzerland (e.g., Rossé et al, 2006), and the UK (e.g., Owens and Walling, 2002).

Previous studies on the assessment of CFC devices have shown that these provide relatively high mass collection efficiencies (MCE) in the collection of suspended sediments (>65%) (for details, see Rees at al. (1991)). However, the inflow rate for such devices typically does not exceed 6 L/min. Moreover, Horowitz et al. (1989) and Ongley and Thomas (1989) recommended that for collecting bulk suspended sediment samples in aquatic systems with low suspended sediment concentrations (SSC; <30 g/m^3), very fine particles (<10 µm), and high concentrations of organic matter, the inflow rate of the low-flow rate CFC devices should be as low as 2 L/min. This low inflow rate increases the time of CFC deployment in the field, and thus limits its application for collecting large masses of suspended sediment in most studies.

In this present study, a low speed (3,000 rpm), heavy (~200 kg), and high-flow rate CFC system (the M512, U.S. Centrifuge Systems, New York, USA; Figure 3.1a and Appendix A-A1) – which has not previously been assessed in aquatic sediment studies – is evaluated in terms of MCE and particle size distribution (PSD). The ability of the M512 centrifuge (hereafter referred as the M512) to collect a representative sample from freshwater systems was also examined by comparing the colour and geochemical properties of the sediment collected by the device with those of the ambient suspended sediment. The recommended inflow rate by the manufacturer for this type of CFC device is 37 L/min, which was used in this study. The M512 is a single bowl centrifuge in which the ambient water–sediment mixture continuously and directly feeds into the spinning bowl, and then travels vertically upward through the bowl. Theoretically, sediments heavier than ambient water are separated and deposited on the inside wall of the bowl. The effluent drains out freely through the outlet fitting and hose. The M512 device, similar to the low-flow CFC, suffers from inherent limitations such as: a) high purchasing price (discussed below) and maintenance costs; b) heavy weight and inconvenience of portability; and c) considerable electric power demand.

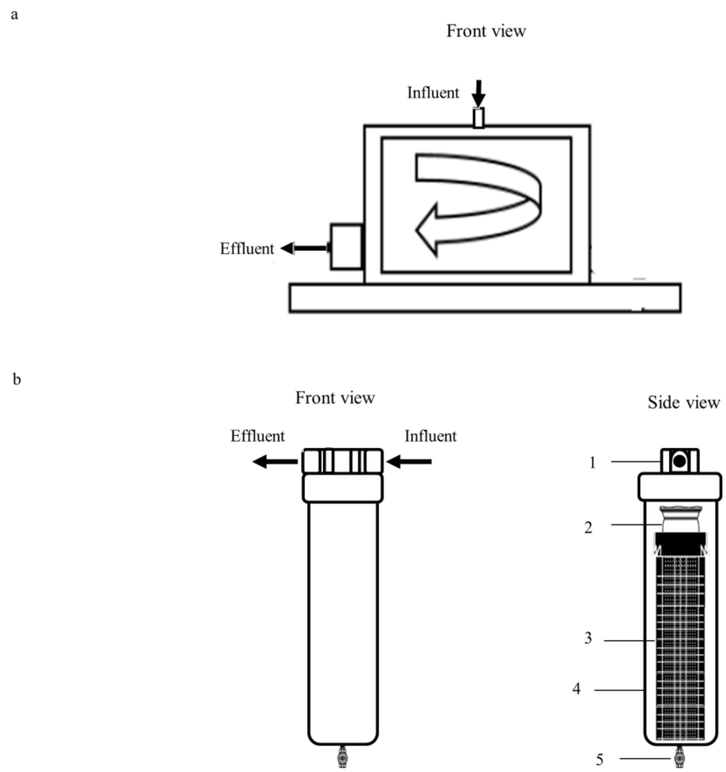

Figure 3-1 Schematic diagram of the two devices. (a) M512 continuous centrifuge (length: 101.6 cm; width: 49.5 cm; height: 50 cm), (b) PENTEK filtration. 1: Filter cap; 2: Replacement filter bag; 3: Basket; 4: Filter sump (Height: 25 or 50 cm); 5: Ball valve

The use of filtration techniques in water quality and sediment transport investigations is amongst the oldest methods for the collection of suspended sediment (Droppo, 2006; Wotton, 1994). However, this method is mainly focused on using membrane filters and provides a low mass of suspended sediment that may prohibit a broad range of physical and biochemical sediment analysis (Bates et al., 1983; Ongley and Blachford, 1982). Potentially, *in situ* high-flow rate continuous-flow filtration (CFF) devices can collect a sufficiently large amount of such material for subsequent property analysis. This approach, however, has not received as much attention as either CFC or micro-analytic filtration techniques.

The PENTEK filtration device (Figure 3.1b and Appendix A-A1) with a 1 μm nominal pore size filter bag (i.e., PBH Series; PENTAIR, Brookfield, USA; hereafter referred to as the filtration system) was evaluated in this study under two different inflow rates: 26.5 and 53 L/min. The original applications of this device include residential and industrial filtration. The filter bags are made of lightweight, corrosion resistant, polypropylene washable felt. They are an ideal filtration media to separate bulk suspended sediment samples from water due to their capacity for collecting particles with a wide range of inflow rates (up to ~150 L/min), with the manufacturer's recommended inflow rate of approximately 130 L/min. Application of this approach and an assessment of the representativeness of samples obtained by using an adapted filtration system have not been investigated in aquatic sediment studies. This manageable filtration system is simple, light (~5 kg) and economically feasible (discussed below) and may provide an alternative approach for the collection of bulk suspended sediment samples.

This paper: (a) evaluates the sampling performance of the M512 and filtration systems; and (b) assesses their drawbacks and practical applications in the field. A field study in three contrasting water bodies was performed to evaluate the influences of different physical factors (i.e., ambient SSC and PSD, sampling time, and inflow rate) on their sediment mass collection efficiency. The ability of the M512 and filtration system to provide a truly representative sample of freshwater sediment are also evaluated by comparing the PSD and geochemical and colour properties of sediments collected by these devices with those of ambient suspended sediments in the same water bodies.

3.3 Study area

The suspended sediment samples were collected at 12 sites from three different freshwater systems in Manitoba, Canada, during spring and summer of 2016 and 2017: Nelson River (NR), Red River (RR), and Lake Winnipeg (LW). An overview of the study area is presented in Figure 3-2. The collection of large quantities of suspended sediment using these two devices and subsequent analyses of physical properties and elemental composition are part of a major study (i.e., Hudson Bay System Study; BaySys) that addresses several research objectives, including the understanding of the sediment and chemical (including carbon and mercury) transport dynamics of the Nelson River system (e.g., Déry et al., 2016).

The Nelson River system is the third largest river in terms of water discharge in Canada (Qave = 2,480 m^3/s), and eventually flows into Hudson Bay through relatively steep granitic and gneissic bedrock of the Precambrian Canadian Shield (Duboc et al., 2017; Newbury et al., 1984). This 680 km river originates from the outflow of Lake Winnipeg and has average concentrations of total suspended solids (TSS) between 9 and 16 mg/L and 15 and 18 mg/L during the ice-free and ice-covered (winter) periods, respectively (Weiss, 2012). Lake Winnipeg is the eleventh largest lake by surface area (23,750 km^2) and the third largest hydro-electric reservoir in the world (Manitoba Water Stewardship, 2011; Newbury et al., 1984). This water body experiences frequent excessive algal blooms (Matisoff et al., 2017) and is divided into three parts: shallow South Basin (10% volume; average depth = 9.7 m; average TSS concentration = 11.8 mg/L), Narrows (9% volume; average depth = 7.2 m; average TSS concentration = 11.9 mg/L), and North Basin (81% volume; average depth = 13.3 m; average TSS concentration = 5.2 mg/L)

(Brunskill et al., 1980; Manitoba Water Stewardship, 2011; Matisoff et al., 2017). The lake receives discharges from several large rivers including the Red River. The Red River accounts for ~16% of the discharge into the lake, with an average monthly discharge of 436 m^3/s and a peak flow rate of approximately 1,300 m^3/s (Kimiaghalam et al., 2015; Manitoba Water Stewardship, 2011). Almost one third of the Red River Basin is located in Canada, the rest lying to the south in the USA. The average TSS concentration of the Red River is 121 mg/L and 10 mg/L during the ice-free and ice-covered periods, respectively, with a maximum TSS of 1,500 mg/L under high flow conditions. Most of the Red River suspended sediments are fine silt and clay (Kimiaghalam et al., 2015).

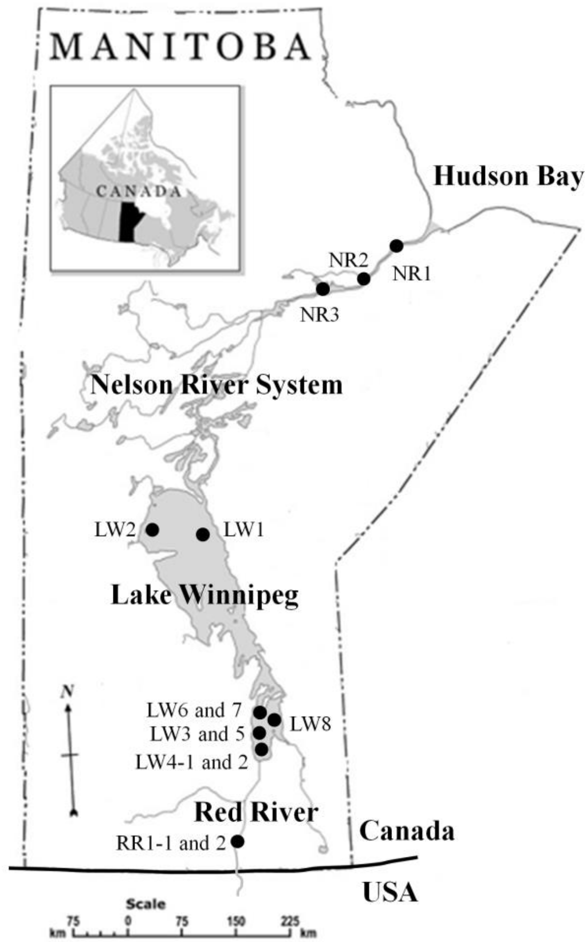

Figure 3-2 Map showing the sites located in the Red River (RR), Lake Winnipeg (LW), and Nelson River (NR) in Manitoba, Canada (note: most of the Red River is located in the USA)

3.4 Field and laboratory methods

River and lake ambient water–sediment mixture was pumped with a Dayton stainless steel submersible pump (Dayton Electric Manufacturing Co., Chicago, USA) and then processed through these two unpressurized devices. The unpressurized flow was measured by using Sotera 825 and 850 flow meters (Tuthill Transfer Systems, Fort Wayne, USA) and drained from the devices under gravity. At most sampling sites, the M512 and filtration system were employed concurrently to evaluate and compare the effectiveness of these devices under identical input sediment concentrations. In the Red River, the water–sediment mixture was taken with the submersible pump approximately 20 m from the river bank at a bridge from ~50% of the water depth. In the Nelson River, the submersible pump was placed approximately 40 cm above the river bed (i.e., for sampling and concentrating fine-grained suspended sediment) close to the river bank considering site accessibility and personal safety. The Lake Winnipeg water–sediment mixture was collected from the *MV Namao* research vessel (Lake Winnipeg Research Consortium (LWRC)) at a depth of approximately 2 m below the surface of the water. The research vessel held position at the sampling locations, and a large volume of lake ambient water–sediment mixture was processed through both the M512 and filtration system (Figure 3-2). The potential effects of using a sequence of filtration systems in series, on both increasing the MCE and the temporal variations of the MCE over the sampling period, were also examined at a number of sites. The field activities in these three freshwater systems are summarized in Table 3-1.

Table 3-1 Details of sampling sites and methods employed for suspended sediment collection from the Red River (RR), Lake Winnipeg (LW), and Nelson River (NR)

No.	Site ID	Date	Sampling station	UTM Zone	Easting	Northing	Methods employed †
1	RR1-1	April- 24-2016	Red River at South Perimeter Bridge	14 U	634238	5516368	CFC (37), IE
2	RR1-2	July- 19-2017					CFF (26.5), CFF (53), IE
3	LW1	July-22-2016	Lake Winnipeg - North Basin	14 U	571909	5897037	CFC (37), CFF (26.5), CFF (53), IE
4	LW2	July-25-2016	Lake Winnipeg - North Basin	14 U	483005	5892620	CFC (37), CFF (26.5), CFF (53), IE
5	LW3	June-1-2017	Lake Winnipeg - South Basin	14 U	642827	5610679	CFC (37), CFF (26.5), CFF (53), IE
6	LW4-1	June-2-2017	Lake Winnipeg - South Basin	14 U	653716	5589137	CFC (37), CFF (26.5), CFF (53), IE
7	LW4-2	Aug-7-2017					CFC (37), CFF (53), CFF*4 (53), IE

8	LW5	July-24-2017	Lake Winnipeg - South Basin	14 U	653493	5598282	CFF (53), CFF*2 (53), IE
9	LW6	July-24-2017	Lake Winnipeg - South Basin	14 U	644885	5611108	CFF (53), CFF*2 (53), IE
10	LW7	Aug-4- 2017	Lake Winnipeg - South Basin	14 U	656047	5618392	CFC (37)
11	LW8	Aug-8-2017	Lake Winnipeg - South Basin	14 U	685333	5614134	CFC (37), CFF (53), IE
12	NR1	July-28-2017	Nelson River at Conawapa	15 V	451088	6282905	CFF (26.5), CFF*2 (53), IE
13	NR2	July-29-2017	Nelson River at Long Spruce	15 V	416536	6250951	CFF (26.5)
14	NR3		Stephens Lake	15 V	398408	6249283	CFF (53)

†: Not all methods were employed at each sampling location. Sampling methods employed: CFC (37): Using M512 continuous centrifuge with the inflow rate of 37 L/min; CFF (26.5) and CFF (53): Bulk suspended sediment collection using filtration system at an inflow rate of 26.5 L/min and 53 L/min, respectively; CFF*2 (53) and CFF*4 (53): Collection of suspended sediment using 2 and 4 filtration systems in series at 53 L/min, respectively; IE: Collection of influent/effluent suspended sediment periodically.

At selected sites, 1-L bottles and 7-L buckets of influent suspended sediment (i.e., ambient water–sediment mixture) and 1-L bottles of effluent suspended sediment were collected periodically. The 1-L bottle samples were used to determine TSS concentrations and to examine the effects of sampling time and inflow rate on the MCE during sampling. The 7-L buckets of influent suspended sediment samples were processed for concentrating the influent (ambient) water–sediment mixture and performing PSD tests. Also, at three sites (i.e., RR1-1, RR1-2, LW4-1) between 100 and 200-L of ambient water–sediment mixture were collected in 20-L buckets so as to compare the colour and geochemical properties of sediment samples collected by the M512 and filtration system with those of ambient sediment.

At the end of sampling, the sediment retained in the bowl of the M512 were collected using a stainless steel spatula and were transferred to plastic Ziploc bags. For the filtration system, dirty filter bags were also transferred to large Ziploc bags. All plastic bags were stored in a refrigerator (at $4^{o}C$) on board the ship (i.e., LW) or in coolers (i.e., NR and RR) to preserve the collected sediment for subsequence analysis. It is worth noting that the times required for sampling and concentrating ambient water–sediment mixture may not be suitable for assessing microbial community dynamics (behaviour) and genomics; however, for the purposes of this study the method described would be adequate. After collection, the filter bags were washed in 20-L buckets and the suspended materials were allowed to settle for 3 to 7 days following procedures described in Perks et al. (2014). The clear supernatant water in the buckets was siphoned off and the sediments were dried at room temperature. The same procedures were followed for the 7-L influent samples and 20-L buckets of ambient water–sediment mixture in order to concentrate suspended sediments. Similarly, all sediment samples collected by the M512 were dried at room temperature.

Subsamples of collected sediments, as well as all influent and ambient samples, were subject to disaggregation prior to analysis to obtain primary particle size characteristics. The procedure includes removing all organic substances from the samples by adding hydrogen peroxide and dispersing the inorganic fraction using sodium hexametaphosphate. Details of this analytical procedure have been published elsewhere (e.g., Phillips et al., 2000). All suspended sediments collected by the filter bags were wet sieved through 1 mm stainless steel mesh to remove any possible filter fabric in the samples. A Malvern Mastersizer 2000 (Malvern, UK) laser diffraction particle size analyser was used to quantify the primary (chemically dispersed) size fraction of influent sediment samples and the suspended sediments collected by these devices; each sample was analysed in triplicate and typically precision was better than ± 10 per cent for any particle size. In this study, the particles were grouped as very fine sand (i.e., 120-63 µm), large silt (i.e., 63-32 µm), medium silt (i.e., 32-16 µm), fine silt (16-8 µm), very fine silt (8-2 µm), and clay (< 2 µm). The Kolmogrov-Smirnov statistical test (K-S test) was also used for comparisons of the primary PSD of the influent sediments and the sediments collected by the M512 and filtration system.

The TSS concentration was determined in duplicate by measuring the dry weight of a subsample of the 1-L influent/effluent water–sediment samples. The subsample of the influent/effluent samples were filtered in the laboratory through pre-weighed and pre-ashed Whatman GF/F glass fibre filters based on the ASTM standard D3977-97 (ASTM, 2013). Precision was better than ± 9 per cent. The MCE for both devices and at each sampling time was calculated according to influent/effluent TSS concentration values:

$$\mathrm{MCE}\ (\%) = \frac{\sum_1^n X_i}{n} = \frac{\sum_1^n \left(1 - \frac{Y_i}{Z}\right) \times 100}{n} \qquad \text{Equation 3.1}$$

where X_i is the instantaneous MCE (%), Yi is the instantaneous effluent TSS concentration (g/m^3), Z is the influent TSS concentration (g/m^3), and n is the total number of effluent suspended sediment samples.

Horowitz et al. (1989) provided a comparison of the instantaneous MCE and the true sediment MCE (the dry sediment mass collected by the sampling device versus the total dry suspended sediment influent mass) for four low-flow CFC devices at five sampling sites in the USA. They found that, on average, the true MCE was just slightly lower than the instantaneous MCE (i.e., 92% and 95%, respectively). Therefore, the average of a time series of instantaneous MCEs can be considered reasonably accurate for estimating the true MCE.

Based on previous studies of sediment source fingerprinting in the Tobacco Creek, a sub-watershed of Lake Winnipeg (Barthod et al., 2015; Koiter et al., 2013), ambient and collected suspended sediment at sites RR1-1, RR1-2 and LW4-1 were analysed for a suite of diagnostic sediment properties: three geochemical/radionuclide properties (i.e., As, U, ^{137}Cs) and the colour coefficients of Z, L, X, and Y. The aim of these comparisons was to assess the ability of the M512 and filtration system to collect a representative sample of the ambient suspended sediment in terms of sediment properties commonly used for sediment source fingerprinting (see also Russell, Walling, and Hodgkinson (2000) for more details). The concentrations of As and U were measured using ICP-MS following a microwave-assisted digestion with nitric acid at the Northern Analytical Laboratory

Services, University of Northern British Columbia, Canada (for details, see Owens et al., (2019)). The colour coefficients were obtained using a spectroradiometer (ASD FieldSpec Pro, Analytical Spectral Device Inc., Boulder, CO, USA) at the University of Manitoba, Canada, for spectral readings over a 360–830 nm wavelength range. The X, Y, Z, and L colour coefficients were determined using the Commission Internationale de l'Eclairage (CIE) method following the procedures and methods used by Barthod et al. (2015). Caesium-137 activity concentrations were determined using gamma spectrometers at the University of Manitoba, Canada (for details, see Koiter et al. (2013)). The precision of the laboratory measurements for each property (i.e., geochemical concentration and colour coefficients) was usually better than ± 15 percent (and will be discussed further below) and was determined by performing duplicate measurements on the ambient suspended sediment samples collected at these three sites.

3.5 Results and discussions

3.5.1 Mass collection efficiency (MCE) of the M512 and filtration system

The variation of instantaneous MCE for the filtration system (i.e., CFF (26.5) and CFF (53)) and M512 (i.e., CFC (37)) at sites RR1-1 and LW4-1 (Table 3-1) are shown as a function of time in Figure 3-3 (see Appendix A-A2 for additional examples). There is limited variation in the instantaneous MCE of the M512 during the sampling period (average ± 1 standard error = 78% ± 0.7% and 20% ± 6.4% at RR1-1and LW4-1 sites, respectively) and this parameter is not a function of the amount of water centrifuged. The consistency of the values of instantaneous MCE over time and at a fixed inflow rate for the M512 (i.e., 37 L/min) is in contrast with the finding of Horowitz et al. (1989) and Ongley and Thomas (1989). They hypothesized that in low-flow rate CFC systems, the enhancement in MCE with increasing sampling time was due to the possible build-up of a cohesive layer of sediment in the bowl. The TSS concentration of the ambient water, processed water–sediment volume, the MCE of the devices, and dry mass of sediment collected by each device at sites RR1-1 and LW4-1 are presented in Appendix A-A3.

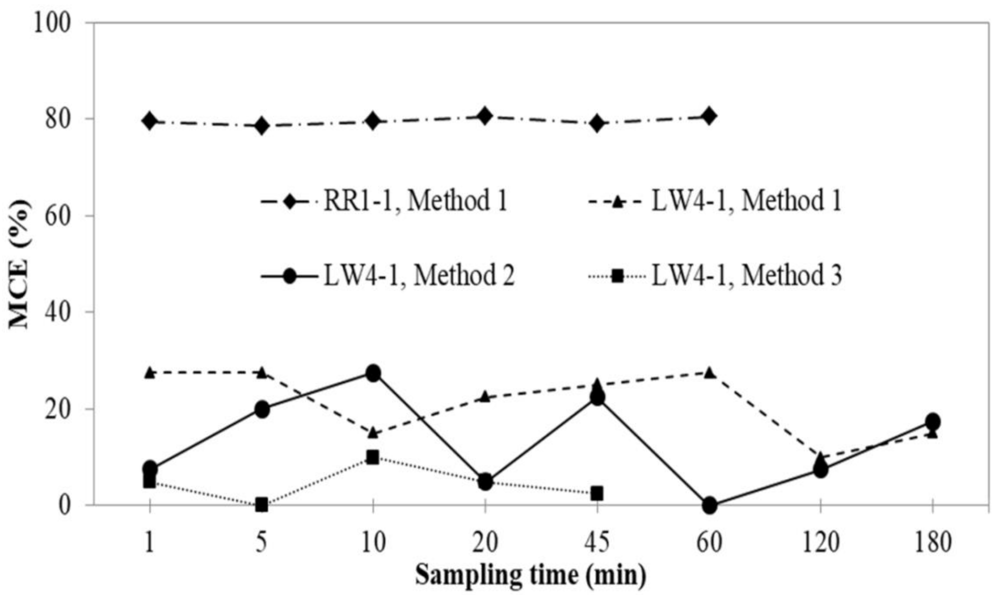

Figure 3-3 the instantaneous mass collection efficiency (MCE) of M512 with the inflowing rate of 37 L/min (CFC (37)) at site RR1-1 (on April 24th, 2016); and M512 (CFC (37)), filtration system with the inflowing rates of 26.5 L/min and 53 L/min (CFF (26.5) and CFF (53), respectively) at site LW4-1 (on June 2nd, 2017). See Table 3-1 for details on each method. RR is Red River and LW is Lake Winnipeg

Figure 3-3, Appendix A-A2, and Appendix A-A3 indicate that under a low TSS concentration (i.e., 35 g/m^3), the filtration system with slower inflow rate (CFF (26.5)) has a higher MCE. This finding may be attributed to the decreasing residence time of the ambient water–sediment mixture within the filtration system during greater inflow rates, which reduce the opportunity for sediment retention. The influence of increasing the inflow rate on the MCE of the low-flow rate CFC and other similar sampling and concentrating systems has also been documented previously (e.g., Horowitz et al., 1989; Ongley & Blachford, 1982). However, from a practical perspective and given the need to provide representative samples, the trade-off between slower inflow rate and shorter sampling time is still a matter of debate.

The potential effects of using a sequence of filtration systems in series on the MCE at LW4, LW5, LW6, and NR1 are presented in Appendix A-A4. As expected, the potential improvement of the MCE by using several filtration systems in series is confirmed and the maximum improvement of the average MCE for filtration systems in series (using Equation 1) was 14% (at LW5). A combination of low ambient TSS concentrations and very fine particles may be responsible for the relatively low improvement of the MCE of filtration systems in series. This hypothesis is consistent with the previous findings that these factors negatively control the devices' performance. Given the similarity of MCEs of the CFF (26.5) and CFF*2 (53) at site NR1 (i.e., averages of 21% and 20%, respectively), the latter filtration system may collect a similar quantity of suspended sediment yet, in half, or less, of the CFF (26.5) sampling time.

In general, assessment of the performance of these devices supports previous studies by Ongley and Blachford (1982), Burrus et al. (1989), Horowitz et al. (1989), Rees et al. (1991) that as the input sediment concentration increases, the MCE value also increases. Considering Figure 3-3, Appendix A-A2, and the typical value of the MCE for the low-rate CFC (>65%) reported in the literature, it may be concluded that: a) the M512 appears to not always be as efficient (in terms of percentage MCE) as the low-flow rate CFC systems; b) the M512 has a higher MCE compared to the single filtration system; and c) the MCE values of the filtration systems in series, in all cases, were greater than 10%. Nevertheless, the lower MCE of the M512 and filtration systems in series are compensated by their higher flow rate. Therefore, in practice, in an equal sampling period and under the same ambient conditions, both the M512 and filtration system in the series configuration may provide an equivalent quantity of suspended sediment sample mass (in absolute terms) compared to low-flow rate CFC devices. Appendix A-A.5 presents the required processing time to collect a 10 g sample of suspended sediment at different input TSS concentrations for a hypothetical low-flow rate CFC system (CFC (6)) with 65% and 90% MCE as well as the M512 and filtration systems in series with an assumption of 15% and 10% MCE, respectively.

3.5.2 Particle size distribution (PSD) collection efficiency of the M512 and filtration system

Using the K-S test with a 95% confidence interval, in nearly all cases, there are no significant differences in the primary PSDs between the sediment collected by the filtration system and those of influent suspended sediment (Table 3-2 and Figure 3-4). Therefore, in most cases the filtration system collected a statistically representative sample of inflowing sediment. There are two cases where there was a consistent difference. Site LW3 is a shallow harbor in Lake Winnipeg, and the samples collected by both devices may have been affected by resuspension of bottom sediments due to ship and boat traffic and/or wave actions. In addition, at the end of the tests at NR1 the submersible pump fell to the bottom of the Nelson River, and it is likely that the filtration system also collected some of the coarser materials on the bottom of the river, which subsequently influenced the PSD of collected sediments.

Table 3-2 The Kolmogorov-Smirnov statistical test (K-S test) results for the particle size distribution (PSD) of sediment samples collected with the M512 and filtration system compared to the PSD of influent material

Site ID	Influent d_{10} (μm)	Influent d_{50} (μm)	Influent d_{90} (μm)	Method employed	Collected d_{50} (μm)	K-S test value*
RR1-1	3.7	11.7	15.9	CFC (37)	12.8	0.095
RR1-2	1.8	4.6	16.1	CFF (26.5)	6.4	0.041
				CFF (53)	7.8	0.147
LW1	5.7	18.3	43.9	CFC (37)	20.7	0.059
				CFF (26.5)	18.2	0.085
				CFF (53)	18.3	0.026
LW2	2.5	8.0	28.1	CFC (37)	7.7	0.021
				CFF (26.5)	8.2	0.053
				CFF (53)	8.1	0.0204
LW3	1.5	4.1	20.5	CFC (37)	8.4	**0.180**
				CFF (26.5)	23.0	**0.186**
				CFF (53)	23.2	**0.190**
LW4-1	0.1	1.3	7.9	CFC (37)	2.4	**0.463**
				CFF (26.5)	0.4	0.065
				CFF (53)	1.9	0.130
LW4-2	0.1	1.7	11.2	CFC (37)	3.7	**0.516**
				CFF (53)	1.8	0.140
				CFF* 4 (53) Filter 1	3.2	0.170
				Filter 2	2.7	0.136
				Filter 3	1.4	0.039
				Filter 4	0.4	0.108
LW7	0.1	0.2	2.2	CFC (37)	2.1	**0.730**
LW8	2.4	8.7	24.8	CFC (37)	8.3	0.013
				CFF (53)	11.7	0.127
NR1	0.1	3.1	24.7	CFF (26.5)	38.7	**0.404**
				CFF* Filter 1	37.9	**0.462**
				Filter 2	46.9	**0.450**
NR2	2.5	6.2	26.1	CFF (26.5)	8.1	0.131
NR3	1.7	4.5	27.0	CFF (53)	6.3	0.015

* = the PSD differences between two distributions are not significant at 95 % (α = 0.5) for test values less than the critical value of 0.17; bold values indicate significant differences.

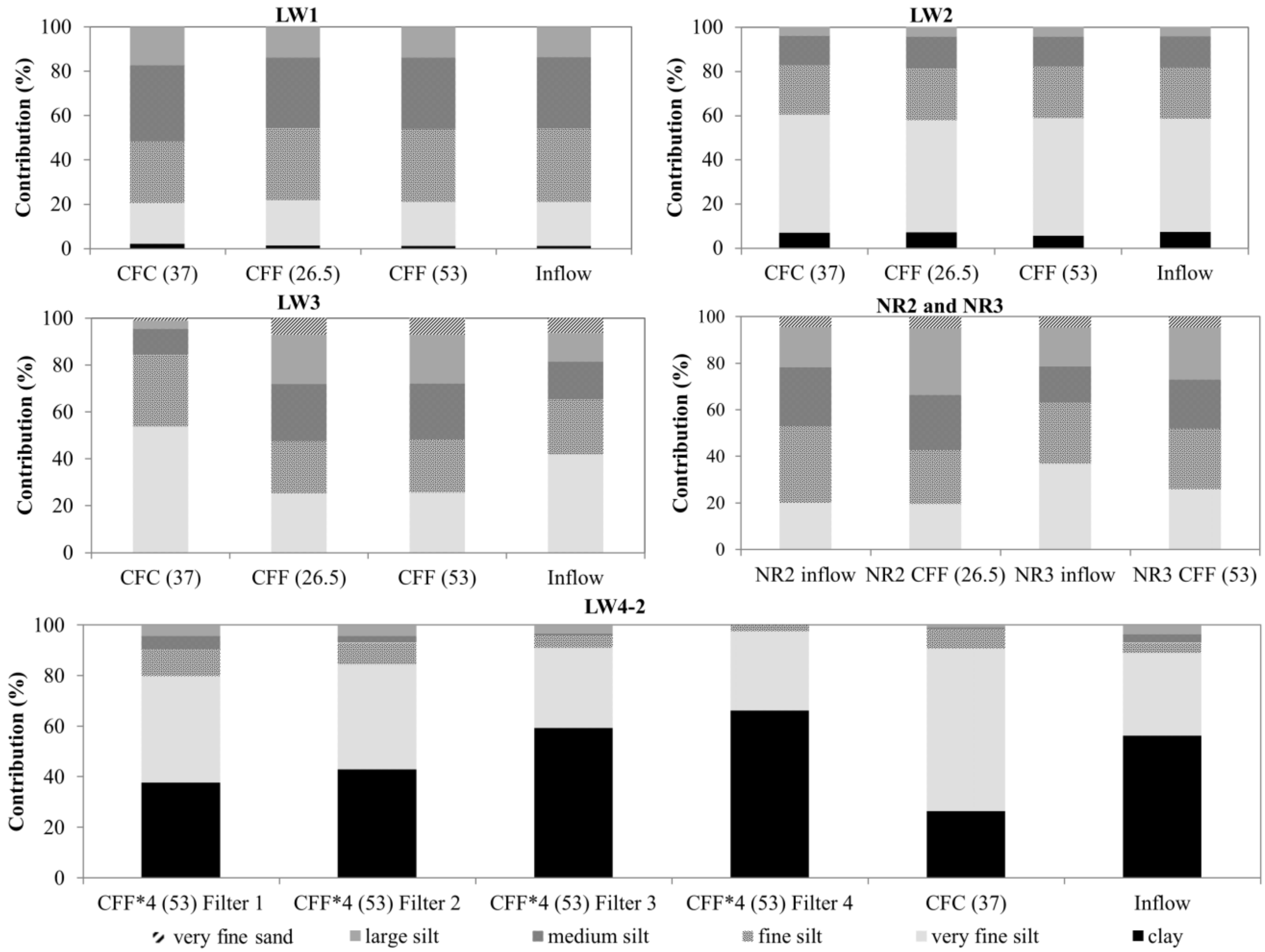

Figure 3-4 Volumetric proportion of different particle size classes in influent and collected sediment by both devices at sites LW1, LW2, LW3, LW4-2, NR2, and NR3

Figure 3-4 also shows the volumetric proportion of different suspended sediment class sizes collected by each filtration system in a four-series configuration at LW4-2. Figure 3-4 and Table 3-2 illustrate that the contribution of clay-sized sediments in the third and fourth filtration systems, as expected, were considerably higher than in the first and second filters. The proportions of all size classes in the sediments collected by each filtration system are statistically similar to those of influent sediment (Table 3-2). In addition, Appendix A-A.6 indicates that (using the K-S test) no significant PSD variations were observed in sediment samples collected by each two subsequent filtration systems (i.e., first and second, second and third, or third and fourth). Hence, in the filtration systems in series configuration, it would be possible to mix sediments collected by all filtration systems into a single composite sample with insignificant bias in PSD. As a result, the filtration system may be considered as an effective device for collecting suspended sediment with a representative particle size composition.

Figure 3-4 also demonstrates that the M512 and filtration system are directly comparable devices for the collection of a representative sample when the suspended sediments in the influent are greater than clay size (e.g., sites LW1 and LW2). Figure 3-4 and Table 3-2, on the other hand, show that the PSD of sediment collected by the M512 is statistically different from that of the inflowing sediment when the ambient sediment contains a high proportion of finer particles (<2 μm) (e.g., site LW4-2). This poor efficiency in collecting finer particles may be linked to the minimum particle diameter that can be collected by the M512. The PSD results of all samples collected by the M512 indicate that the cut-off value for the M512 is approximately 0.83 μm. This value is coarser than the lower threshold value for the low-flow rate CFC system that was estimated by both Rees et al.

(1991) and Moody and Meade (1994) (i.e., 0.37 and 0.1 μm, respectively, for Sharples–Pennwalt Model AS–12 low-flow CFC), but is similar to those assessed by Rossé et al. (2006) and Santiago et al. (1990) (i.e., 1 μm). This restriction may lead to over-sampling of materials coarser than the limiting size (Figure 3-4).

3.5.3 Assessing geochemical and colour properties of suspended sediment collected by the M512 and filtration system

Table 3-3 presents the magnitude of percentage difference between several property values associated with the sediment collected by the M512 and filtration system and those of ambient suspended sediment at three different sites (i.e., RR1-1, RR1-2, LW4-1). In most cases, the differences between the property concentration and coefficient values for the sediment collected by the filtration system and the ambient suspended sediment values are within the analytical precision. Thus, these results suggest that the filtration system collects a sample that is truly representative of the geochemical and colour properties of suspended sediment. These results support previous findings (described above) that the sediment collected by the filtration system provides a representative PSD of the ambient suspended sediment.

Under high TSS concentration conditions (i.e., RR1-1; 250 g/m^3), the differences between the values of geochemical concentrations and colour coefficients of samples collected by the M512 and those of ambient suspended sediment are within the bounds of analytical precision. However, at site LW4-1 the differences between the sediment collected by the M512 and ambient sediment are beyond the analytical precision for all properties. Table 3-3 also indicates that the percentage differences of geochemical properties of sediment collected by the filtration system are beyond the analytical

Table 3-3 A comparison of the differences between the geochemical concentrations and colour coefficients of the ambient suspended sediment and sediment collected by the M512 and filtration system

Site ID: RR1-1			
Property†	Ambient suspended sediment value	Absolute % of analytical precision	% differences†† CFC (37)
As	4.96	5.1	-1.0* (5.01)
U	2.07	5.3	2.4 (2.02)
^{137}Cs	0	0	0
X	32.20	7.6	-5.6 (34.11)
Y	27.85	8.1	-5.3 (29.36)
Z	7.17	6.5	-3.9 (7.45)
L	61.57	5.3	-2.2 (62.95)

Site ID: RR1-2				
Property	Ambient suspended sediment value	Absolute % of analytical precision	% differences	
			CFF (26.5)	CFF (53)
As	5.55	6.9	1.0(5.60)	3.5 (5.75)
U	2.46	11.3	**26.7** (1.88)**	**20.6 (2.00)**
^{137}Cs	0	0	0	0
X	34.46	13.5	3.9 (33.15)	8.6 (31.63)
Y	29.90	13.7	4.8 (28.50)	9.5 (27.20)
Z	7.87	15.2	10.3 (7.10)	15.8 (6.72)
L	61.57	5.7	2.0 (60.35)	4.0 (59.15)

Site ID: LW4-1					
Property	Ambient suspended sediment value	Absolute % of analytical precision	% differences		
			CFC (37)	CFF (26.5)	CFF (53)
As	6.45	6.9	**14.5 (5.58)**	**8.4 (5.93)**	**19.8 (5.29)**
U	2.03	11.3	**19.5 (1.67)**	**18.3 (1.69)**	**17.1(1.71)**
^{137}Cs	0	0	0	0	0
X	34.56	10.6	**-28.9 (46.22)**	2.2 (33.80)	-1.0 (34.90)
Y	30.00	10.5	**-30.3 (40.69)**	1.5 (29.54)	-1.0 (30.29)
Z	7.81	9.6	**-33.4 (10.94)**	3.0 (7.58)	3.1 (7.57)
L	61.65	8.4	**-12.6 (69.96)**	1.0 (61.05)	-1.0 (62.25)

precision at this site. The high percentage differences may be attributed to the low TSS concentration (35 g/m^3) and very fine PSD (median diameter (d_{50}) = 1.3 µm) of the lake suspended sediment at LW4-1 site. These results are also in agreement with the low-flow CFC assessment studies (Horowitz et al., 1989; Ongley and Thomas, 1989) which concluded that care must be exercised when CFC devices are used for collecting bulk suspended sediment samples in freshwater systems with low suspended sediment concentrations (SSC; < 30 g/m^3) and/or very fine particles (< 10 µm).

Particle comparison between the M15 and filtration systemTable 3-4 describes a number of advantages and disadvantages of the M512 and filtration system. The filtration system demands less power generating equipment. Therefore, two couplings of pump-filtration (single or in series) systems can be concurrently operated from a single portable generator to draw more ambient water–sediment mixture, reduce the sampling time, and thus enable more sites to be covered in one day. In addition, in the authors' work on ships, the filtration system was operated in the absence of a power supply by using the ship's inline water system (e.g., fire hydrant). On the other hand, the use of the M512 on the ship needed electrical modification to accommodate this device on the ship's electrical power system. Moreover, other studies report that using low-flow rate CFC systems on a ship or small craft suffers from potential difficulties including bearing failure, contact between the rotational bowl and the stationary parts due to the ship's movements, and, therefore, sample contamination from the CFC components (Ongley and Thomas, 1989). Although the effect of the composition of the different components of both the M512 and filtration system (under different collecting conditions) on the potential contamination of the collected samples is beyond the scope

Table 3-4 Comparison of the M512 and filtration system

Device	Advantages	Disadvantages
M512	• easy for cleaning the bowl • high absolute mass collection efficiency • large sediment holding capacity (<200 g)	• heavy (200 kg) • large size (Length: 101.6 cm; Width: 49.5 cm; Height: 50 cm) • expensive (purchasing cost: ~$30, 000 US) • limited sampling site accessibility (i.e., truck or wheeled cart mounted) • point-in-time- and space-sampling • potential for fragmentation of particles due to centrifugal shear stress effects • cut-off particle diameter • power source dependency for start-up and operation
Filtration system	• high flow rate (i.e. up to 180 L/min) • manageable size and weight (18.5 cm diameter × 59 cm height; ~5 kg) • easy and straightforward deployment including estuarine and marine conditions • acceptable absolute mass collection efficiency (i.e., in series configuration) • covering several sites in one day • river/stream site accessibility in difficult terrain • inexpensive (purchasing cost: ~$120 US) • sufficient sediment holding capacity (<10 g) without clogging	• point-in-time- and space-sampling • potential for fragmentation of particles due to torturous flow through filter media • extra steps (i.e., wash filter bags) in laboratory and increasing processing time • possibility of losing some samples in post-processing steps • possible transformation of elements from the dissolved phase to the particulate phase and vice versa during processing

of this study, the purpose in presenting this potential concern is to draw attention to possible future research.

The filtration system, with the total estimated cost of ~$8 US per filter bag and less than $120 US for the whole system (at the time of writing), is more cost-effective compared with the M512 (i.e., apart from maintenance costs, the M512 typically costs ~$30,000 US). Therefore, from practical and cost standpoints, the filtration system is a feasible method to investigate the spatiotemporal variability of fine-grained (<63 µm) suspended sediment characteristics, especially in large-scale studies. However, rinsing the dirty filter bags (i.e., those used in the filtration system) in a laboratory as an extra sample processing step, longer processing time, and the potential for losing some sediment during sample processing are factors that may restrict the utility of this approach in some situations.

3.6 Conclusions and Recommendations

The use of high-flow rate CFC (i.e., M512) and CFF (i.e., filtration system) devices for the collection of fine-grained (<63 µm) suspended sediment from freshwater systems was evaluated. The mass collection efficiency (MCE) for both sampling devices was a function of influent suspended sediment concentration, and this finding is comparable to the findings of low-flow rate CFC system evaluations. The instantaneous MCE of the M512 during different sampling periods exhibited limited variation, and the MCE was not a function of the volume of ambient water–sediment mixture centrifuged. The results also showed that the single filtration system had lower MCE compared to the M512; however, using a number of the filtration systems in series improved the MCE. In terms of the MCE, the M512 and several filtration systems in series are as efficient as low-flow rate CFC systems in collecting a bulk sample of suspended sediment in a similar sampling time.

The filtration system in nearly all cases collected a representative sample of inflowing suspended sediment in terms of PSD. However, the M512 device collected a representative PSD sample of inflowing sediments only when the influent particles were ~ ≥2 μm. The lower cut-off particle diameter for sediment collection using the M512 was determined to be ~0.83 μm, and this value was not a function of influent PSD or TSS concentration. The size threshold may affect the collection of sediment with a representative particle size composition.

Similar to the PSD efficiency, for most geochemical and colour properties, the filtration system was able to collect a representative sample of ambient suspended sediment. Under high suspended sediment concentrations, the sample collected by the M512 was also representative of the fluvial suspended sediment in terms of both geochemical concentrations and colour coefficients. However, the percentage differences between the colour coefficient and geochemical concentration values for the samples collected by the M512 and the ambient suspended sediment containing a high proportion of finer particles (≤0.83 μm) and lower sediment concentration are outside of the analytical precision. This suggests that the M512 may not collect a truly representative sediment sample in such conditions.

This study also suggested that, from a practical perspective, the filtration system (particularly in series) provides advantages over the M512 device including ease of portability, lower purchasing price, and lower power demand. Thus, this system can be considered as a viable alternative for sampling and concentrating fine-grained (<63 µm) suspended sediment in lake and river environments.

3.7 References

Abuhelou, F., Mansuy-Huault, L., Lorgeoux, C., Catteloin, D., Collin, V., Bauer, A., ... Montargès-Pelletier, E., 2017. Suspended particulate matter collection methods influence the quantification of polycyclic aromatic compounds in the river system. Environmental Science and Pollution Research, 24, 22717-22729.

ASTM standard D3977-97., 2013. Standard test methods for determining sediment concentration in water samples. West Conshohocken, PA, USA: ASTM International.

Barthod, R.M., Liu, K., Lobb, D.A., Owens, P.N., Martinez-Carreras, N., Koiter, A.J., Petticrew, E.L., McCullough, G.K., Lui, C., Gaspar, L., 2015. Selecting color-based tracers and classifying sediment sources in the assessment of sediment dynamics using sediment source fingerprinting. Journal of Environmental Quality, 44(5), 1605-1616.

Bates, T.S., Hamilton, S.E., Cline, J.D., 1983. Collection of suspended particulate matter for hydrocarbon analyses: continuous flow centrifugation vs. filtration. Estuarine, Coastal and Shelf Science, 16, 107-112.

Bilotta, G.S., Brazier, R.E., 2008. Understanding the influence of suspended solids on water quality and aquatic biota. Water Research, 42, 2849-2861.

Brunskill, G.J., Elliott, S.E.M., Campbell, P., 1980. Morphometry, hydrology, and watershed data pertinent to the limnology of Lake Winnipeg. Department of Fisheries and Oceans (DFO), Winnipeg, Canada.

Burrus, D., Thomas, R.L., Dominik, J., Vernet, J.P., 1989. Recovery and concentration of suspended solids in the upper Rhone River by continuous flow centrifugation. Hydrological Processes, 3, 65-74.

Déry, S.J., Stadnyk, T.A., MacDonald, M.K., Gauli-Sharma, B., 2016. Recent trends and variability in river discharge across northern Canada. Hydrology and Earth System Sciences, 20, 4801-4818.

Droppo, I.G. 2006. Filtration in particle size analysis. Encyclopedia of Analytical Chemistry, Wiley, Chichester, UK.

Duboc, Q., St-Onge, G., Lajeunesse, P., 2017. Sediment records of the influence of river damming on the dynamics of the Nelson and Churchill Rivers, western Hudson Bay, Canada, during the last centuries. The Holocene, 27, 712-725.

Goharrokhi, M., Pahlavan, H., Lobb, D.A., Owens, P.N., Clark, S.P., 2019. Assessing issues associated with a time-integrated fluvial fine sediment sampler. Hydrological Processes, 33(15), 2048-2056. https://doi.org/10.1002/hyp.13451

Horowitz, A.J., 1991. A primer of sediment-trace element chemistry. Lewis Publishers, USA.

Horowitz, A.J., Elrick, K.A., Hooper, R.C., 1989. A comparison of instrumental dewatering methods for the separation and concentration of suspended sediment for subsequent trace element analysis. Hydrological Processes, 3, 163-184.

Keßler, S., Pohlert, T., Breitung, V., Wilcsek, K., Bierl, R., 2019. Comparative evaluation of four suspended particulate matter (SPM) sampling devices and their use for monitoring SPM quality. Environmental Science and Pollution Research, 6, 5993-6008.

Kimiaghalam, N., Goharrokhi, M., Clark, S.P., Ahmari, H., 2015. A comprehensive fluvial geomorphology study of riverbank erosion on the Red River in Winnipeg, Manitoba, Canada. Journal of Hydrology, 529, 1488-1498.

Koiter, A.J., Lobb, D.A., Owens, P.N., Petticrew, E.L., Tiessen, K.H., Li, S., 2013. Investigating the role of connectivity and scale in assessing the sources of sediment in an agricultural watershed in the Canadian prairies using sediment source fingerprinting. Journal of Soils and Sediments, 13(10), 1676-1691.

Matisoff, G., Watson, S.B., Guo, J., Duewiger, A., Steely, R., 2017. Sediment and nutrient distribution and resuspension in Lake Winnipeg. Science of the Total Environment, 575, 173-186.

Manitoba Water Stewardship. 2011., State of Lake Winnipeg: 1999 to 2007. Winnipeg, Canada: Environment Canada.

Mbabazi, J., Inoue, T., Yokota, K., Saga, M., 2019. Phosphorus bioavailability in rivers flowing through contrasting land uses. Journal of Environmental Chemical Engineering, 7(2), 102960.

Moody, J.A., Meade, R.H., 1994. Evaluation of the method of collecting suspended sediment from large rivers by discharge-weighted pumping and separation by continuous-flow centrifugation. Hydrological Processes, 8, 513-530.

Newbury, R.W., McCullough, G.K., Hecky, R.E., 1984. The Southern Indian Lake impoundment and Churchill River diversion. Canadian Journal of Fisheries and Aquatic Sciences, 41, 548-557.

Ongley, E.D., Blachford, D.P., 1982. Application of continuous flow centrifugation to contaminant analysis of suspended sediment in fluvial systems. Environmental Technology, 3, 219-228.

Ongley, E.D., Thomas, R.L., 1989. Dewatering suspended solids by continuous-flow centrifugation: Practical considerations. Hydrological Processes, 3, 255-260.

Owens, P.N., Walling, D.E., 2002. The phosphorus content of fluvial sediment in rural and industrialized river basins. Water Research, 36, 685-701.

Owens, P.N., Blake, W.H., Gaspar, L., Gateuille, D., Koiter, A.J., Lobb, D.A., ... Woodward, J.C., 2016. Fingerprinting and tracing the sources of soils and sediments: Earth and ocean science, geoarchaeological, forensic, and human health applications. Earth-Science Reviews, 162, 1-23.

Owens, P.N., Gateuille, D., Petticrew, E.L., Booth, B., French, T.A., 2019. Sediment-associated organopollutants, metals and nutrients in the Nechako River, British Columbia: a current study with synthesis of historical data. Canadian Water Resources Journal, 44, 42−64.

Packham, R.F., Rosaman, D., Midgley, H.G., 1961. A mineralogical examination of suspended solids from nine English rivers. Clay Minerals Bulletin, 4, 239-242.

Perks, M.T., Warburton, J., Bracken, L. 2014. Critical assessment and validation of a time-integrating fluvial suspended sediment sampler. Hydrological Processes, 28, 4795– 4807.

Phillips, J.M., Russell, M.A., Walling, D.E., 2000. Time-integrated sampling of fluvial suspended sediment: a simple methodology for small catchments. Hydrological Processes, 14, 2589-2602.

Reid, T., Droppo, I.G., Weisener, C.G., 2020. Tracking functional bacterial biomarkers in response to a gradient of contaminant exposure within a river continuum. Water Research, 168, 115167.

Rees, T.F., Leenheer, J.A., Ranville, J.F., 1991. Use of a single-bowl continuous-flow centrifuge for dewatering suspended sediments: Effect on sediment physical and chemical characteristics. Hydrological Processes, 5, 201-214.

Rossé, P., Vignati, D., Dominik, J., 2006. Effects of continuous flow centrifugation on measurements of trace elements in river water: intrinsic contamination and particle fragmentation. Hydrological Processes, 20, 2745-2754.

Russell, M.A., Walling, D.E., Hodgkinson, R.A., 2000. Appraisal of a simple sampling device for collecting time-integrated fluvial suspended sediment samples. In The role of erosion and sediment transport in nutrient and contaminant transfer. IAHS Publication No. 263. (pp. 119–127). Wallingford, UK: IAHS Press.

Santiago, S., Thomas, R.L., Loizeau, J.L., Favarger, P.Y., Vernet, J.P., 1990. Further discussion on the intercomparison of the trace metal concentrations and particle size of fluvial sediment recovered from two centrifuge systems. Hydrological Processes, 4, 283-287.

Syvitski, J.P., Vörösmarty, C.J., Kettner, A.J., Green, P., 2005. Impact of humans on the flux of terrestrial sediment to the global coastal ocean. Science, 308, 376-380.

Vercruysse, K., Grabowski, R.C., Rickson, R.J., 2017. Suspended sediment transport dynamics in rivers: multi-scale drivers of temporal variation. Earth-Science Reviews, 166, 38-52.

Weiss, A., 2012. The use of acoustic Doppler current profiler technology to quantify total suspended solids under ice. Unpublished MSc Thesis, University of Manitoba, Winnipeg, Canada.

Wotton, R.S., 1994. Methods for capturing particles in benthic animals. In: Wotton, R.S. (Ed.), The biology of particles in aquatic systems, pp. 183-204. Lewis Publishers, USA.

CHAPTER 4: Sedimentation dynamics within a large shallow lake and its role in sediment transport in a continental-scale watershed

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4.1 Abstract

A comprehensive understanding of the sedimentation dynamics within Lake Winnipeg (surface area is 23,750 km^2) and its role in sediment transport in the downstream river system was achieved by determining the properties of lake bottom sediment and patterns of sediment accumulation rates, and by constructing a total (i.e., organic and inorganic) sediment budget. Net deposition was the governing process in the South and North Basins, whereas transportation dominated in the Narrows. The largest fluvial source of sediments to the lake, the Red River, supplies 35% of the total sediment load. Although sediment accumulation rates in profundal zones progressively decreased northward from this source at the south end of the lake, high sediment accumulation rates with low inventories of fallout radionuclides in the northern margin of the North Basin indicate a second sediment source, which was determined to be erosion of north shore banks, which accounts for up to 50% of the total sediment load to the lake. The nearshore-offshore

gradient in bottom sediment properties in the North Basin confirmed that the signature of this new source can reach at least 20 km southward into the lake. However, the properties of bottom sediments, sedimentation dynamics, and sediment budget suggested that some of the materials eroded from the north shore are exported without interaction with the lake bottom and this local sediment source is the dominant source for the downstream river system. It was concluded that Lake Winnipeg effectively disconnects the downstream Nelson River from sedimentary processes in its upstream watershed (953,250 km^2).

4.2 Introduction

Lake Winnipeg is the 11th largest freshwater lake in the world by surface area and is a key feature within the Nelson River watershed (1,125,520 km^2) (Duboc et al., 2017; Environment and Climate Change Canada and Manitoba Agriculture and Resource Development, 2020). Figure 4-1 includes a map of this continental-scale watershed which stretches from the Rocky Mountains to near Lake Superior, and straddles the Canada–USA border. It discharges into southwestern Hudson Bay. The water quality and ecological status of Lake Winnipeg have been adversely affected by nutrient enrichment due to both high external loading and internal release of nutrients (Matisoff et al., 2017; McCullough et al., 2012). While a combination of climate forcing of hydrological changes in surrounding sub-watersheds and anthropogenic activities (e.g., agricultural practices and sewage release) have resulted in increased loading of nutrients from the watershed into Lake Winnipeg (Bunting et al., 2016; Kling et al., 2011; McCullough et al., 2012), the release of redox–regulated nutrients (particularly phosphorus) from anoxic surficial bottom sediment to overlying water during summer stratification is the main source of internal loading of nutrients (Manitoba Water Stewardship, 2006; Nürnberg and LaZerte, 2016). As a consequence of this external and internal nutrient loading, the lake is regularly fouled by severe blooms of cyanobacteria and diatoms (Kling et al., 2011).

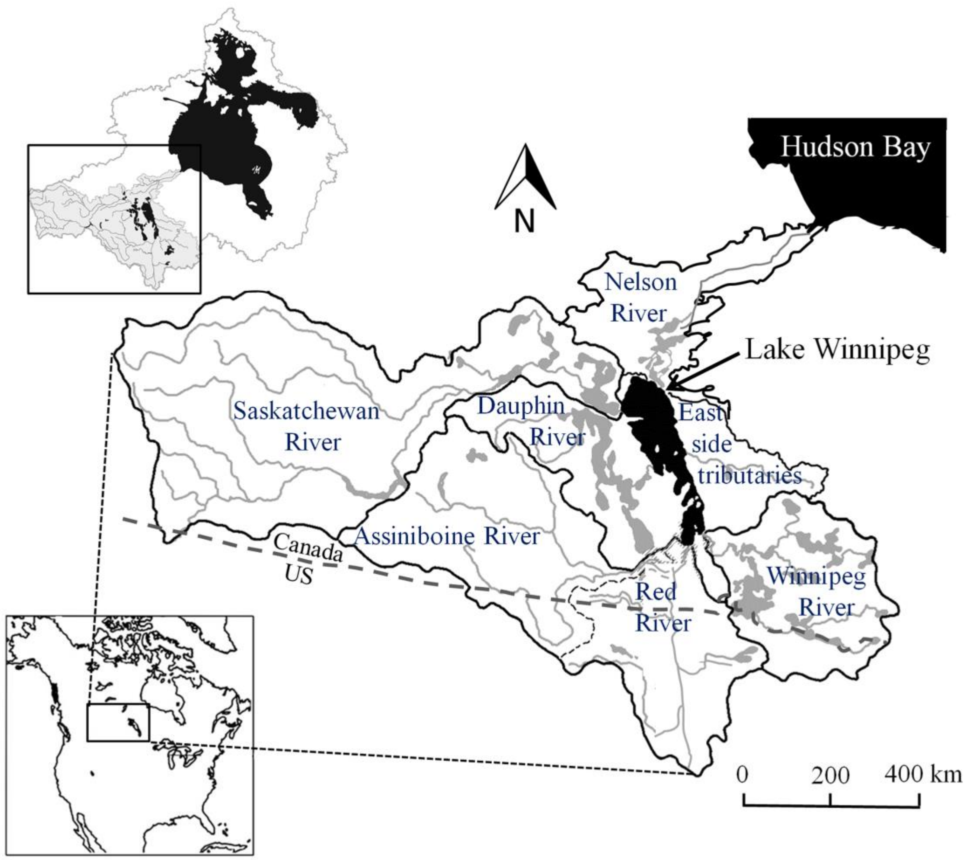

Figure 4-1 Hudson Bay and the Nelson River watersheds. The scale bar refers to the Nelson River watershed

Given the high capacity of fine suspended sediment (i.e., <63 µm) in binding and transporting nutrients (such as phosphorus) and contaminants (such as some fallout radionuclides), due to, at least in part, its large specific surface area (SSA) (He and Walling, 1996; Owens and Walling, 2002), sedimentation dynamics may exert a large control on the fate and dynamics of nutrients, especially phosphorus, in Lake Winnipeg and its water quality and ecological status. Furthermore, sedimentation dynamics may have a significant influence on both the quality and quantity of sediment being exported to downstream systems (i.e., the Nelson River and potentially Hudson Bay). Thus, an improved understanding of sedimentation dynamics in this large water body is of fundamental importance.

Determining the properties of the bottom sediment in Lake Winnipeg and the pattern of average annual sediment dry mass accumulation rate (hereafter referred to as the DMAR), as well as interpretation of the spatial variability of these parameters in relation to controlling factors, are important prerequisites to understand the sedimentation dynamics in the lake. In recent years, a number of researchers have studied the spatial variation of DMARs by dating sediment cores that were either few in number (i.e., <5) or short in length (<10 cm) (e.g., Bunting et al., 2016; Lockhart et al., 1998; Lockhart et al., 2000; Matisoff et al., 2017; Wilkinson and Simpson, 2003). The results of these studies were interpreted based on the assumption that DMAR diminishes down-lake due to a progressive loss of suspended sediment from the mouth of the Red River, a major source of sediment in the south, to the outlet in the north (Matisoff et al., 2017; McKindles et al., 2019). There has been little detailed information published on the relative, and sometimes conflicting, influence of the key natural controls on sedimentation dynamics within the lake (e.g., sediment properties, local morphology, lake hydrodynamics, and wind energy and patterns).

The Red River is not the only major source of sediment to the lake. Brunskill and Graham (1979) estimated that each year substantial quantities of bank sediments along the north shore contribute to the sedimentary processes in Lake Winnipeg. The banks of the north shore consist of 8–12 m high glacio-lacustrine sediments overlain by peat. However, there is little information available on the amount and patterns of sediment accumulation from this northern source (Scott et al., 2011).

In order to address these knowledge gaps, in this work, 50 sediment cores were collected throughout the lake with an emphasis on the northernmost region of the lake. Spatial and vertical variations of key sediment properties, including water content (i.e., the ratio of water weight to total wet weight), porosity, particle size composition and SSA, and organic matter (OM) content were measured. In addition, the chronology of sediment accumulation was determined using concentrations of natural and anthropogenic fallout radionuclides (i.e., excess ^{210}Pb (hereafter referred to as $^{210}Pb_{xs}$) and ^{137}Cs) in the sediment cores. DMARs were then used along with published information on the major sedimentary basins within the lake, the sediment loadings from the largest tributaries and from erosion of north shore bluff deposits, and the lake outflow flux to: a) quantify the total annual dry mass accumulation; and b) establish a total (i.e., organic and inorganic) sediment budget for Lake Winnipeg.

This lake-wide study is part of a major collaborative study of the Hudson Bay System and its watershed (i.e., BaySys; Figure 4-1) that addresses a number of research objectives such as the understanding of the potential impacts of hydroelectric generating stations on sediment and chemical transport dynamics in the Nelson River system (Capelle et al., 2020; Goharrokhi et al., 2020; Guéguen et al., 2016). As the Nelson River is the largest river (by watershed area and freshwater discharge) contributing to Hudson Bay (Déry et al., 2018), this work will contribute to the goals of the BaySys project by providing information on: a) the role of Lake Winnipeg in reducing fluxes of sediment from its contributing watershed; and b) the potential contribution of the north shore eroded materials to the lake and the Nelson River system.

4.3 Study area

Lake Winnipeg, in Manitoba, Canada, has a surface area of 23,750 km^2 (Figure 4-2), with a contributing watershed of 953,250 km^2 (Figure 4-1) dominated by agricultural land use (i.e., ~657,740 km^2) (Bunting et al., 2016; Environment and Climate Change Canada and Manitoba Agriculture and Resource Development, 2020; Manitoba Water Stewardship, 2011). This watershed includes portions of four US states (Montana, North Dakota, South Dakota, and Minnesota) and parts of four Canadian provinces (Alberta, Saskatchewan, Manitoba, and Ontario) (Mayer et al., 2006). Lake Winnipeg is comprised of a South and a North Basin (~2,780 km^2 and ~17,520 km^2, respectively), separated by a long Narrows region (~3,450 km^2) (Brunskill et al., 1980; Scott et al., 2011). The latter is restricted to channels as little as 2.2 km in width (Figure 4-2). The average water depths of the South Basin, Narrows, and North Basin (i.e., 10 m, 7 m, and 13 m, respectively) mean that Lake Winnipeg is a shallow and vertically well-mixed lake with substantial resuspension of surficial bottom sediments (Brunskill et al., 1980; Matisoff et al., 2017). The lake has been regulated by control structures and constructed channels at, and downstream from, its outlet since 1976. Provincial licencing allows that when the lake level is within the natural historical range of 216.7–217.9 meter above sea level (i.e., m asl), the outflow be managed to support hydroelectric generation; when it exceeds this range, the outflow must be managed so as to return it within its natural range (Manitoba Department of Mines and Natural Resources, 1970). Consequently, this regulation program may have resulted in the modification of several of Lake Winnipeg's key characteristics, including seasonal: a) water and sediment outflow; b) sediment accumulation rates and patterns; and c) biomass productivity (Liu et al., 2007; Manitoba Water Stewardship, 2011; McCullough et al., 2001).

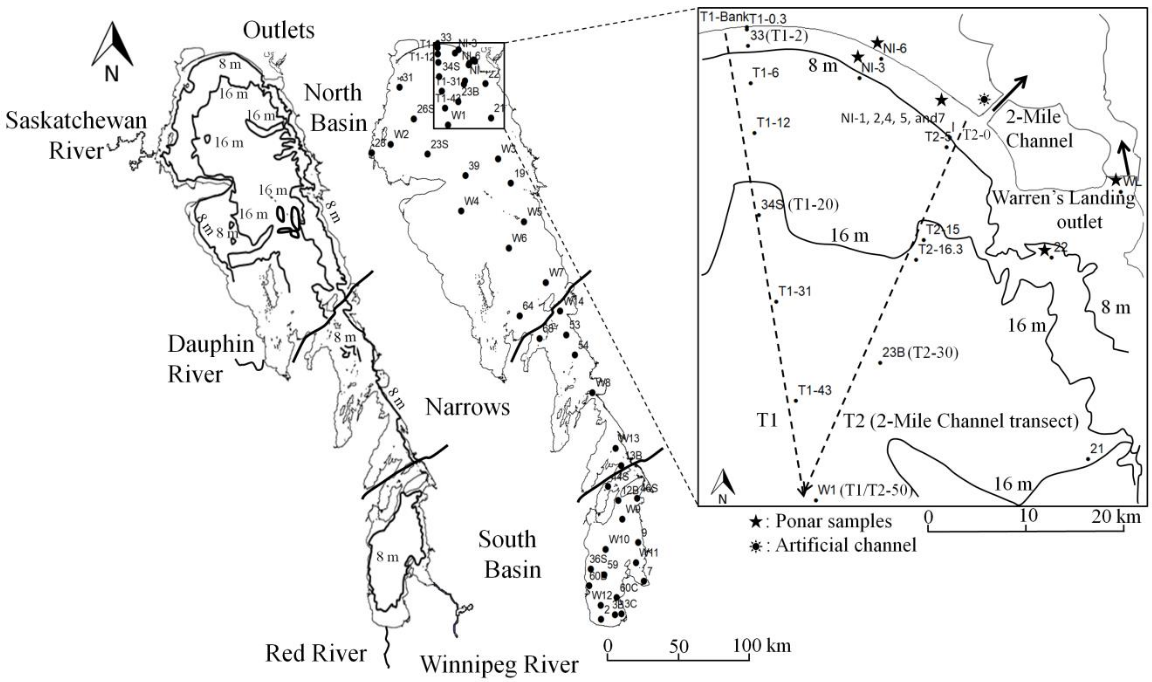

Figure 4-2 Left: Lake Winnipeg, its main riverine inputs, bathymetry (8 m and 16 m depth contours; adapted from Brunskill et al. (1980)), three main areas of the lake (i.e., South Basin, Narrows, and North Basin), and sampling sites; right: sampling sites along two transects, 8 m and 16 m depth contours, and Lake Winnipeg outlets. The outlets flow into the Nelson River system which eventually discharges into Hudson Bay

During the 2008 to 2016 period, the lake received 96% of its total water inflow from the Red, Winnipeg, Dauphin, and Saskatchewan Rivers, which supplied 16%, 43%, 8%, and 29% of the inflow, respectively (Environment and Climate Change Canada and Manitoba Agriculture and Resource Development, 2020). The Red and Winnipeg Rivers flow into the South Basin, whereas the Dauphin and Saskatchewan River flow into the North Basin (Figure 4-2). The Red River is a Great Plains river and dominated by agricultural land use, while the Winnipeg River drains a predominantly boreal watershed in the Precambrian Shield. The Dauphin River drains alpine, prairie, and Boreal Plains landscapes via Winnipegosis and Manitoba Lakes before flowing into Lake Winnipeg. The Saskatchewan River is also a Great Plains river that originates in the Canadian Rockies and flows eastward through agricultural lands (Binding et al., 2018; Brunskill et al., 1980).

Although the Winnipeg and Saskatchewan Rivers supply most of the water inflow to Lake Winnipeg, natural lakes and anthropogenic storage reservoirs intercept most of their sediment loads. Brunskill et al. (1980) estimated that over 99% of the Saskatchewan River's natural sediment load is retained in the reservoirs upstream of Lake Winnipeg. They also stated that the Red River suspended sediment load may commonly exceed the sediment load of the Winnipeg River by an order of magnitude. Similarly, the Dauphin River originates from Lake Manitoba and thus it can be assumed that the contribution to Lake Winnipeg's sediment load from this river is relatively insignificant (discussed further below). The Red River, therefore, is the major source of riverine sediment and associated nutrients to Lake Winnipeg (Lévesque and Page, 2011; McCullough et al., 2012). The sediments in this river are predominantly silt-size (i.e., between 2 and 63 μm) with high SSAs (e.g., 1.3 m^2/g; estimated using a Malvern Mastersizer 2000) relative to larger particles such as fine and medium sands (Goharrokhi, 2015).

Net water flow from the sources in the South Basin is northward towards the outlet at the northern end of the lake. However, this general net movement is frequently interrupted by strong, short-duration, bi-directional flow through the Narrows region, caused by lake level setup forced by strong northerly or southerly winds, or by subsequent seiching in both basins (Brunskill et al., 1980). High current velocities are attained when the large volumes of water from the wide North or South Basins are forced through the much narrower passage between them. Kenney (1979) found that in 1976 and at ~10 km south of site W8 (Figure 4-2) such fluxes exceeded the average and maximum net annual northward water flow (i.e., the sum of Red and Winnipeg Rivers discharges) by up to two and one order(s) of magnitude, respectively. The maximum point (i.e., at 5.5 m below the surface water) and depth–averaged current measurement corresponding to the wind–induced water exchange in that year (i.e., between May 14 and October 15) was 90.2 and 85.2 cm/s, respectively towards the south (i.e., opposite to the net hydraulic gradient of the lake). This may exceed the threshold for motion of particles through most of the silt range (i.e., the Red River suspended sediment) even if deposited during quiescent periods (Hjulström, 1935). Kenney (1979) claimed that 1976 was a relatively calm year with light winds and he expected greater wind–induced water exchanges could occur in other years. Therefore, Brunskill et al. (1980) suggested that the inter–basin water exchanges have considerable effects on: a) Lake Winnipeg water residence time (i.e., 4.3, 1.3, and 3.5 years over the period of 1999 to 2007 for the lake as a whole, South Basin, and North Basin, respectively); b) inter–basin transfer of nutrients and contaminants; and c) the distribution of phytoplankton and zooplankton. The water residence time for the South Basin, for example, was estimated to be 0.17 years due to inter–basin exchanges of water (Brunskill et al., 1980).

While the Red River in the south is the largest source of fluvial sediments to the lake, it is not the only major source. As described above, Brunskill and Graham (1979) identified erosion of high glacio-lacustrine silt banks along the north shore as a second major sediment source. They inferred that most of this additional sediment source was deposited in the North Basin. However, in the current study, we hypothesize that some of this material is transported downstream through an artificial channel, the 2-Mile Channel, which was opened in 1976.

The natural outlet of Lake Winnipeg at Warren's Landing at the northeast corner of the North Basin crosses a broad, shallow sill (~1.5 km wide and <3 m deep) into the Nelson River (Figure 4-2). Since 1976, however, ~66% of water leaving the lake passes through this natural outlet and the rest flows through the narrow (surface width = ~207 m) and deeper (average water depth = 8.6 m) 2-Mile Channel, ~14 km west of the natural outlet (Kimiaghalam and Clark, 2017).

The 2-Mile Channel increased the lake's outflow capacity by up to 50% without commensurate increases in lake level, hence allowing higher outflow while meeting licencing requirements to maintain lake level within the pre-regulation range (Manitoba Hydro, 2014a). Bank and bottom erosion near and along the channel has been monitored since 1978 by bathymetric surveys over a number of cross sections (Kimiaghalam and Clark, 2017). The bank near the inlet of the 2-Mile Channel receded by ~1.5 m/yr between 1978 and 2011 (Manitoba Hydro, 2014b), and of the order of 1 m over the same 33-year time period at a location within the channel (Kimiaghalam and Clark, 2017).

4.4 Methods

A UWITEC ID corer (i.e., inside diameter of 8.6 cm) (UWITEC, Mondsee, Austria) was used to collect 50 bottom sediment cores in spring and summer 2016 (Figure 4-2). The sediment cores were collected from the *MV Namao* research vessel from established sampling and monitoring locations (Manitoba Water Stewardship, 2011). The core sampling program also included two additional 50 km transects southward from the north shore (i.e., transects T1 and T2 (2-Mile Channel transect); Figure 4-2) so as to capture the influence of sediment eroded from the north shore. The *Namao's* tender (small external boat) was used for core sampling along these transects in shallow water depths. In addition, four grab samples of the bank materials forming the north shore were collected at the starting point of these two transects to represent the full bank height (i.e., ~8–12 m of glacio-lacustrine sediments overlain by peat) using a small trowel to a depth of ~5 cm. The tender was also used to collect bottom sediment samples using a Ponar grab sampler at a number of sampling sites (n = 10) near the north shore (including site 22), at the Warren's Landing outlet, and sampling site 7 (Figure 4-2). These latter samples (excluding site 7) were collected to assess the extent to which the signature of north shore materials can be detected into the North Basin.

Overall, this sampling program yielded a total of 15, six, and 29 sediment cores from the South Basin, Narrows, and North Basin, respectively. The depths of the sediment cores varied between 5 and 45 cm due to spatial variations of different properties of the bottom sediment including texture and density. The sediment cores were extruded with a piston immediately after collection on the ship. As the piston pushed the sediment out at the set intervals (discussed below; using 5 mm acrylic sheets attached to the piston frame), an extension was placed on top of tubes to collect and cut the cores. The sediment cores were also maintained vertically during extrusion to avoid disturbing the sediment–water interface. Sediment cores at 19 sampling sites were sliced at 5 cm intervals to obtain bottom sediment properties (e.g., particle size compositions and OM content). The remaining 31 cores were sectioned into 1 cm intervals for both fallout radionuclide activity concentrations (i.e., $^{210}Pb_{xs}$ and ^{137}Cs) and the same set of analyses as described above. Sampling site coordinate, lake water depth at the time of sampling, and section interval (i.e., 5 or 1 cm) associated with each sediment core is reported in Appendix B-B1. All the core, Ponar, and north shore material samples were placed in prelabeled plastic bags and stored in an onboard refrigerator (at 4^{o}C) to prevent degradation of materials and transformation of properties.

In the laboratory, the water content of each sediment layer was measured by weighing the sample before and after drying at 85^{o}C for 24 hr (Baskaran et al., 2015). Considering the water content for each layer, porosity (φ_j; non-dimensional) was calculated using the approach described in Baskaran et al. (2015) (methods are explained in Appendix B- B2). Further analyses were performed after homogenization and pulverization of the samples. The OM content was determined by combustion of ~3 g subsample at 550^{o}C for 16 hr after drying the subsample at 105^{o}C for 24 hr (for details, see Appendix B-B2 and Siev et al. (2018)). The organic matter (OM) content information is presented in Appendix B-B13.

The volumetric primary particle size distribution (PSD) of the samples was measured in triplicate using a Malvern Mastersizer 2000 laser diffraction particle size analyser (Malvern, UK) at the University of Manitoba, Canada. This device reports a spherical equivalent of a particle rather than the true size. To obtain a fully disaggregate and dispersed sample (i.e., primary, or absolute PSD), the organic fraction of approximately 1 g subsample was removed using hydrogen peroxide. Then the subsample was chemically dispersed with sodium hexametaphosphate (see Goharrokhi et al. (2019) for details). In this study, the particles were grouped as clay (<2 µm), silt (2-63 µm), and sand (63-2000 µm) particles.

To obtain a chronology for each core, gamma spectrometry was used to simultaneously measure ^{137}Cs, ^{226}Ra, and total ^{210}Pb activities (Bq/kg) at the University of Manitoba. While ^{137}Cs (half-life 30.2 years) is a anthropogenic fallout radionuclide, which was produced as a result of the atmospheric testing of thermonuclear weapons in the 1950s and 1960s, unsupported (excess) lead-210 ($^{210}Pb_{xs}$ = total ^{210}Pb – ^{226}Ra) is a natural fallout radionuclide which is deposited continuously from year to year and can be used to determine the DMAR over approximately the past 100 years. The vertical activity concentration distribution of fallout radionuclides for each core sample were measured by: a) placing each sediment sample into a Petri dish or scintillation vile; b) sealing those containers and waiting to achieve equilibrium between ^{226}Ra and its daughter ^{222}Rn (i.e., approximately 21 days) (Owens et al., 1999); c) putting the containers on top of detectors and counting for between 25,000–86,400 seconds; and d) measuring the unsupported ^{210}Pb, ^{226}Ra, and ^{137}Cs activities from the 46.5, 186.2, and 661.6 keV peaks, respectively (He and Walling, 1996).

To estimate the DMAR at each site using the vertical distribution of ^{137}Cs, the depth or cumulative mass depth (M) of the peak ^{137}Cs concentration was divided by the time between 1963 and 2016 (i.e., the time of the core collection). In this approach, the location of the peak ^{137}Cs concentration in the core profile was assumed to be associated with the period of peak fallout of this anthropogenic radionuclide in 1963 (Matisoff et al., 2017). The cumulative mass depth for each core was calculated using Baskaran et al. (2015) (the method is explained in Appendix B-B2).

As the Red River is the largest source of fluvial sediments to the lake, it would be expected that among commonly used $^{210}Pb_{xs}$ models, the Constant Rate of Supply (CRS) model may be more applicable for the dating of the collected sediment cores (i.e., at least for the South Basin). The DMAR using the distribution of $^{210}Pb_{xs}$ was, however, obtained using the Constant-$^{210}Pb_{xs}$ Flux: Constant Sedimentation model (CFCS-model) as Matisoff et al. (2017) indicated that this approach yields reliable ages for sediment cores collected from the depositional basins of Lake Winnipeg. In addition, by using the $^{210}Pb_{xs}$-CFCS model, the consistency between the CFCS model ages and ^{137}Cs-peak horizon-based ages can be assessed (Mabit et al., 2014); however, we do recognize that other $^{210}Pb_{xs}$ dating model exist (e.g., CRS). The fallout radionuclide inventories (Bq/cm^2) for each core sample were obtained by summing the product of the ^{137}Cs or $^{210}Pb_{xs}$ activity concentration (Bq/g) of each sediment layer by the mass depth associated with that layer (g/cm^2; Baskaran et al., 2015).

In order to establish a sediment budget for Lake Winnipeg, fluvial discharges reported herein were calculated from the Water Survey of Canada (WSC) hydrometric records (https://wateroffice.ec.gc.ca/). Fluvial sediment fluxes were calculated using these fluvial discharge records and total suspended solids (TSS) data (measured by filtration at 1.2 µm) supplied by Manitoba Department of Agriculture and Resources Development (MB DARD) on request. The Thiessen polygon spatial interpolation method (Yamada, 2016) was used and all statistical analyses were undertaken using ArcGIS and R Statistical Software through RStudio v1.2.5033, respectively. The R package ggplot2 was used to create most of the plots.

4.5 Results and discussion

4.5.1 Spatial analysis of bottom sediment water content

The expected trend of decreasing water content with increasing sediment depth, due to compaction and dewatering, was observed in all sediment cores (see Appendix B-B3 for profiles of the longer cores). The range of water content (W_w) values for the selected sediment cores collected in each region are listed in Table 4-1. Lake Winnipeg regions in this study are referred to as: a) the South Basin; b) Narrows; c) offshore of the North Basin (hereafter referred to as the NB offshore); and d) northern margin of the North Basin (i.e., ~< 4 km) (hereafter referred to as the NB nearshore).

Water content in the top 5 cm varies between some regions (Appendix B-B4 and B5). The average W_w along the general longitudinal flow path (i.e., from the mouth of the Red River) in the South Basin and NB offshore are higher in both regions (79% and 80%, respectively) compared to the Narrows and NB nearshore (66% and 32%, respectively). While the Narrows is located between the South Basin and NB offshore, the lower W_w of the Narrows' sediment cores indicates that this region is characterized by more compact sediments. Given the spatial variations of W_w (Appendix B-B4 and B5) and the relation among several controls (e.g., bottom sediment properties, wind speed and dominant direction, fetch length, and lake morphometry) and the potential for sediment resuspension and focusing in lakes, as documented by Håkanson (1977) and Blais and Kalff (1995), the lake bed can be classified into three zones. These are: a) the area with

$W_w < 50\%$ as the sediment erosion zone (i.e., the NB nearshore); b) the area with W_w between 50% and 75% as the sediment transport zone (the Narrows); and c) the area with $W_w > 75\%$ as the sediment deposition zone (the South Basin and NB offshore) (Appendix B-B5).

Bathymetry, to some extent, mirrors the sediment transport and sedimentation dynamics in Lake Winnipeg (Matisoff et al., 2017). Brunskill and Graham (1979), for example, estimated that sedimentation occurs between the area delineated by the 8 m and 16 m depth contour of the South Basin and northern part of the North Basin, respectively (Figure 4-2). Appendix B-B6 shows the contours of Lake Winnipeg at 2 meter intervals related to the percentage surface area of each main region (i.e., the South and North Basins and Narrows). Bathymetry data supports the classification of the lake bottom regarding sedimentation processes based on the water content values. For instance, it can be seen that the Narrows which was recognised as a largely non-depositional region is considerably shallower than both South and North Basins. On the other hand, 1,736 km^2 (~66%) of the South Basin is below the 8 m depth contour and, therefore, is one of the major sedimentary basins in the lake (Figure 4-2; discussed below; Brunskill and Graham, 1979).

4.5.2 Average annual dry mass accumulation rate (DMAR)

For sediment cores which yield reliable ^{137}Cs dates, DMARs estimated based on ^{137}Cs profiles are within ± 25% of the DMARs using the $^{210}Pb_{xs}$-CFCS model, and these two independent chronological methods are reasonably consistent (Table 4-1). In sediment cores for which the complete profiles of ^{137}Cs activity concentration were not recovered or the ^{137}Cs profiles have no objectively discernible peak, the vertical profiles of $^{210}Pb_{xs}$

activity were used to estimate DMARs. The ^{137}Cs and natural logarithm (Ln) of $^{210}Pb_{xs}$ activity profiles versus the cumulative mass depth for the sediment cores are presented in Appendix B-B7, which shows that at most sites, the trend of exponential decay in the vertical profile of $^{210}Pb_{xs}$ were preserved (Baskaran et al., 2015).

For the sediment cores sectioned at 1 cm interval (i.e., n = 31), DMAR ranged from 670 to 3,000 g/m^2/yr in the South Basin, 0 to 1,630 g/m^2/yr in the Narrows, and 230 to 1,820 g/m^2/yr in the NB offshore. The highest recorded DMAR (3,000 g/m^2/yr) was at the closest site to the Red River (W12) and the decline in DMAR from the Red River to the Narrows is consistent with an inverse relationship between DMAR and distance from this largest sediment source in the South Basin (Table 4-1; Figure 4-3).

The DMAR trend is, however, interrupted by zero accumulation rate at site W8 (which had a rocky bottom and no sediment was retrieved at this site, hence the assumption of no net sedimentation) and high DMARs at sites 68 and 64 (i.e., the northernmost and southernmost site in the Narrows and NB offshore, respectively) (Table 4-1; Figure 4-3). This spatial trend suggests that: a) the distance–DMAR inverse relation approach may not be suitable for explaining the sedimentation dynamics in the Narrows; and b) some areas in the Narrows (e.g., the middle area with very low or zero DMARs) are mostly characterized as sediment transport zones, which is consistent with the spatial classification of Lake Winnipeg based on the lake's bathymetry and W_w .

Table 4-1 Core length and water content, average annual sediment dry mass accumulation rates (DMARs) using the $^{210}Pb_{xs}$-CFCS model and peak ^{137}Cs values, and inventories of ^{137}Cs and $^{210}Pb_{xs}$ for the individual sediment core in the South Basin, Narrows, offshore of the North Basin (NB offshore); and northern margin of the North Basin (i.e., ~ < 4 km; NB nearshore).

Region	Sampling site	Core length (cm)	Water content (%)	DMAR g/m²/yr $^{210}Pb_{xs}$	DMAR g/m²/yr ^{137}Cs	Inventory Bq/m² $^{210}Pb_{xs}$	Inventory Bq/m² ^{137}Cs
South Basin	W12	30	70-81	3000 (0.82)[b]	ND[c]	6630	> 1280[d]
	59	25	57-81	1330 (0.92)	ND	8430	> 3030[d]
	36S	23	67-76	1100 (0.63)	ND	4960	> 1810[d]
	W10	24	64-85	1430 (0.88)	ND	6290	> 2410[d]
	9	15	54-85	670 (0.95)	ND	5060	590
	46S	19	55-91	1200 (0.85)	ND	5400	1570
	44S	24	64-77	830 (0.94)	ND	15000	4300
Narrows	W13	19	61-63	700 (0.81)	ND	4080	2130
	W8		No sediment retrieved and lake bottom was rocky				
	54	21	65-68	180 (0.92)	ND	1760	120
	53	16	64-71	800 (0.95)	ND	3590	200
	W14	17	61-63	800 (0.96)	ND	4570	400
	68	15	56-77	1630 (0.94)	ND	5910	540
NB offshore	64	27	70-81	1820 (0.89)	ND	10040	> 1660[d]
	W6	28	72-84	700 (0.86)	600	4470	3160
	W4	32	73-86	420 (0.89)	400	5440	1390
	W3	32	73-85	800(0.91)	850	5660	4690
	39	36	70-85	500 (0.90)	650	5060	2140
	23S	38	73-84	260 (0.82)	400	4100	2050
	W2	30	62-89	1000 (0.90)	ND	9300	3000
	W1 (T1-50)	45	75-88	700 (0.93)	590	6200	4100
	21	23	57-78	240 (0.98)	ND	2470	270
	26S	38	72-88	350 (0.93)	430	3720	1900
	T1-43	30	75-85	940 (0.88)	ND	7840	1760
	T1-31	30	72-89	340 (0.93)	230	4250	1530
	34S (T1-20)	25	70-90	500 (0.88)	ND	4470	680
	T1-12	20	57-80	820 (0.97)	900	2700	620
	T1-6	16	38-61	1370 (0.86)	1450	3750	680
	23B (T2-30)	31	73-90	460 (0.91)	570	5200	2240
	T2-16.3	20	67-79	970 (0.89)	ND	4910	> 1430[d]
	T2-15	23	61-84	300 (0.89)	410	1960	1450
NB nearshore	T2-5	5[a]	41	0	0	0	0
	33 (T1-2)	20	17-67	ND	0	0	0
	T1-0.3	5[a]	25	0	0	0	0
	T1-0	5[a]	22	0	0	0	0
	T2-0.3	8[a]	34	0	0	0	0

T2-0	5[a]	23	0	0	0	0
T1-shore	-	-	0	0	0	0
T2-shore	-	-	0	0	0	0

a: Cores were sectioned at 5 cm interval.
b: Number in the parenthesis is R value in regression line between Ln($^{210}Pb_{xs}$) and cumulative mass depth.
c: ND = Not dateable as there was not a definitive ^{137}Cs activity peak.
d: Cores not long enough to capture the maximum activity of ^{137}Cs.

Given the threshold for motion of silt-sized particles (the dominant size of the Red River sediment) (Hjulström, 1935), the zero DMAR at site W8 conforms to the available hydrodynamic information for Lake Winnipeg. This finding may add further confidence to the overall conclusions presented by Kenny (1979) suggesting that wind–induced currents are responsible for the considerable bi–directional water exchange (e.g., 90.2 cm/s). Kenny (1979) and Brunskill et al. (1980) hypothesised that the reasons for the southward injected wind–induced water exchanges are mainly the shallow water depth along both sides of the shore in the northern Narrows region, the long fetch in the North Basin, and abundant wind energy towards the south. Also, it is well known, from studies on large shallow lakes with long fetches (e.g., Lake Erie), that wave–induced currents may: a) reach the surficial bottom sediment; b) increase the applied shear stress on the bed; and c) lead to sediment resuspension due to exceedance of the bottom sediment threshold for motion (Lou et al., 2000; Valipour et al., 2017). In addition, low DMARs in most of the Narrows support results from the hydrodynamic model of Lake Winnipeg developed by Zhao et al. (2012), who showed strong, coherent northward currents in the Narrows.

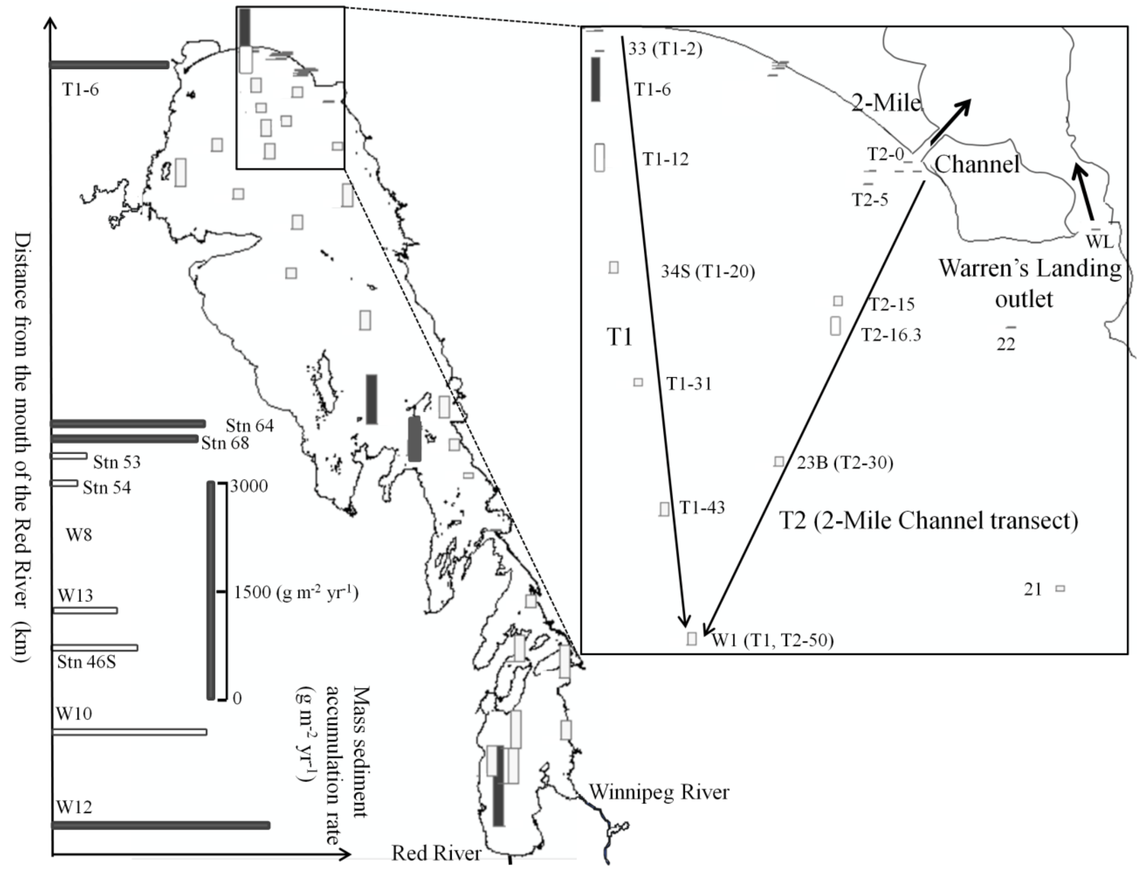

Figure 4-3 Left: schematic of longitudinal trend of average annual sediment dry mass accumulation rates (DMAR) from the Red River to the north shoreline (T1); middle and right: DMAR for sampling sites in Lake Winnipeg (bars in black represents sites with highest DMAR in each region (i.e., South Basin, Narrows, North Basin (NB) offshore; and NB nearshore); the scale for the bars is associated with the DMAR on the left diagram).

High DMARs and the large $^{210}Pb_{xs}$ inventories at sites 68 and 64 (1,820 and 1,630 g/m^2/yr, respectively) suggest a large deposition influx of fine-grained sediment into that region. It can be interpreted that sediments are transported laterally out of the main path of flow in the Narrows (i.e., close to the east shore) into that region with chaotic flow and weaker currents (Zhao et al., 2012) and then deposited. Also, an increase in the DMAR with increasing distance from the middle point in the Narrows (i.e., W8) may be connected with the settling of riverine materials out of the suspension in the moderate hydrodynamic conditions, as both sites are located in the first sedimentary basin in the North Basin (discussed below).

The low ^{137}Cs and $^{210}Pb_{xs}$ inventories in the sediment cores collected from the Narrows (i.e., 120 to 2,130 Bq/m^2 and 1,760 to 5,910 Bq/m^2, respectively, excluding site W8; Table1) support the argument that there are several processes affect DMAR in this part of the lake. Furthermore, the reported average TSS concentrations in the South Basin between 1999 to 2007 (i.e., 11.8 g/m^3; Manitoba Water Stewardship, 2011) and the Narrows (i.e., 11.9 g/m^3; Matisoff et al., 2017) suggest that the Narrows may not exert a significant influence on the reduction of the suspended sediment load, given that there are no major fluvial and shore erosion inputs in this region (Brunskill and Graham, 1979), and that it is mainly a transportation zone.

The DMARs progressively decrease in the North Basin toward the two outlets; i.e., 2-Mile Channel and Warren's Landing (Table 4-1; Figure 4-3). Low DMARs in the sediment cores along the 2-Mile Channel transect (i.e., T2-15 and T2-5; Table 4-1 and Appendix B-B7) as well as the lack of fallout radionuclide concentrations and inventories in the samples at the 2-Mile Channel and Warren's Landing outlets (i.e., sites T2-0, 22, and Warren's Landing) could be attributed to: a) the Coriolis force (i.e., increased currents along the east side of the lake); b) the dynamic nature of the lake at the outlets (i.e., increased flow velocity at the outlet); and c) the effect of strong winds and bi–directional water exchanges between Lake Winnipeg and the downstream channel system (Bijeljanin, 2013; Kimiaghalam and Clark, 2017) (Table 4-1; Figure 4-3).

The DMARs, however, vary markedly among sediment cores in transect T1 (range from zero to 1,370 g/m^2/yr; Figure 4.3). The DMARs at sites T1-6 and T1-12 are almost twice as high as most of the other DMARs in that area and similar to the DMARs in the South Basin. This suggests the contribution of an additional sediment source into the North Basin. These high DMARs are in good agreement with sedimentation rate estimated by Brunskill and Graham (1979) in the area of the 16 m depth contour close to the northernmost region of the lake (i.e., 140–2,200 g/m^2/yr). Despite the high DMARs, the low concentrations and inventories of both ^{137}Cs and $^{210}Pb_{xs}$ in the sediment cores at sites T1-6 and T1-12 (Table 4-1 and Appendix B-B7) may reflect the supply of considerable quantities of a land-based sub-surface sediment source (i.e., north shore eroded materials) to these sites, which are low in fallout radionuclide activity concentrations.

4.5.3 Particle size selectivity and sedimentation dynamics

The limited vertical variation in particle size compositions for each sediment core (Appendix B-B8) gives confidence in the interpretation of the DMAR based on the location of the maximum activity of ^{137}Cs concentration and natural logarithm (Ln) of $^{210}Pb_{xs}$ activity profiles versus the cumulative mass depth. In other words, down-core variations in the radionuclide concentrations are not appear to be due to changes in particle size (Mabit et al., 2014; Owens et al., 1999).

Given the emphasis of this study on the North Basin, the primary particle size composition of bottom sediment in the NB offshore is presented in Figure 4-4. The composition of the sediment in cores T1-6 and T1-12 is noticeable different than the other cores reflecting their closeness to the north shore. Considering the other cores, the limited spatial variability in particle size composition within the NB offshore region (Table 4-2; Figure 4-4) may reflect: a) the fact that Lake Winnipeg is a shallow and well-mixed lake; and b) limited variability in hydro- and sedimentation dynamics within this region. In the South Basin (Table 4-2; Appendix B-B9), however, some measures of particle size composition (e.g., d_{50} and SSA) for the sediments located in the east side of the South Basin (sites 7 and W11) are different from those in sediment cores in the centre and west side of the South Basin (Table 4-2; sites W10, 59, and 36S). These differences in particle size composition of sediment in sites close to the Winnipeg River input may reflect differences in the riverine sediment sources in the South Basin.

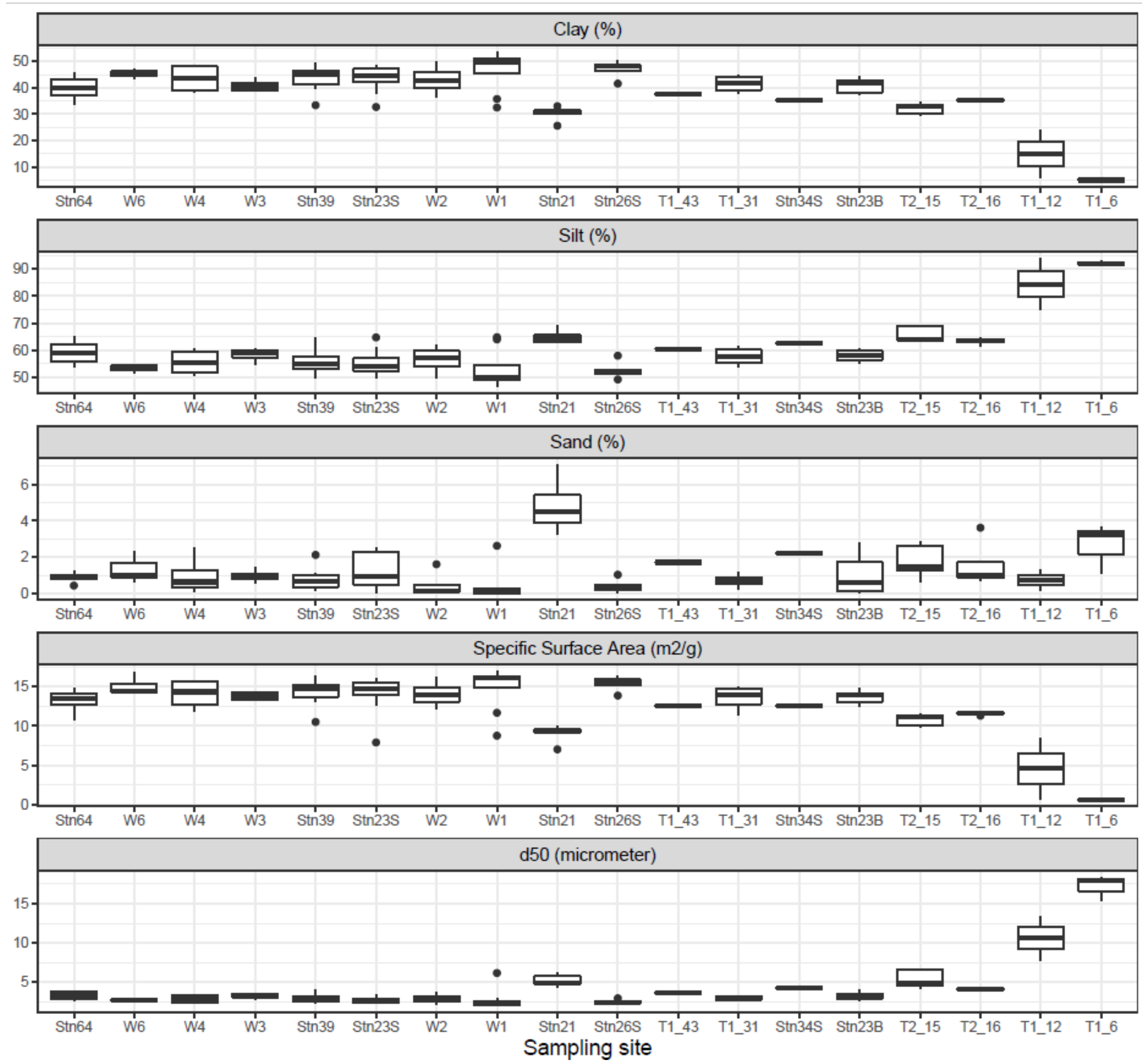

Figure 4-4 Box and Whisker plot showing the variability of the average values of selected measures of primary particle size composition of the sediment cores for the North Basin (NB) offshore region

There was considerable variability in the primary particle size compositions of bottom sediment between the regions (Table 4-2). The average values of different particle size classes for the top 5 cm of the sediment cores along the general longitudinal flow path, and for the NB nearshore are presented in Figure 4-5. Considering Table 4-2, Appendix B-B9, and Figures 4-4 and 4-5, it can be seen that: a) as expected, coarser suspended sediments (i.e., silt) are selectively deposited in the South Basin; and b) the NB nearshore bottom sediment particles are quite distinct (i.e., coarser) than those in the NB offshore.

Table 4-2 Range, average, and coefficient of variation of select measures of particle size composition of representative sediment cores for the sampling sites within the four regions of Lake Winnipeg.

Region	Sampling site	Sample size[a]	Clay (%)	Silt (%)	d_{50} (μm)		SSA (m^2/g)	
					average	CV	avera	CV
South Basin	2	1	10	88	6	0.04	1.28	0.03
	59	25	20-26	74-80	4	0.07	1.80	0.26
	36S	5	25-27	75-80	3	0.02	1.94	0.01
	W10	20	16-19	81-83	5	0.11	1.54	0.08
	W9	1	18-20	81	5	0.02	1.55	0.02
	7[b]	1	5	77	20	0.06	0.64	0.04
	W11	1	6	79	16	0.06	0.75	0.04
	9	15	13-16	72-77	6	0.11	1.32	0.07
	44S	20[c]	14-16	82-86	5	0.04	1.46	0.04
Narrows	W13	10	8	82-83	11	0.12	0.92	0.06
	54	20	13-16	84-85	5	0.08	1.39	0.05
	53	15	11-14	82-87	8	0.14	1.17	0.09
	W14	10	8-10	84-85	10	0.12	0.99	0.06
	68	10	13-15	70-84	6	0.12	1.33	0.06
NB offshore	64	30[c]	34-46	54-65	3	0.25	13.76	0.20
	W6	30	43-47	51-55	3	0.04	14.71	0.05
	W4	20	38-48	51-61	3	0.22	13.68	0.22
	W3	20	38-44	55-60	3	0.17	14.02	0.11
	39	35[c]	33-49	50-64	3	0.25	14.20	0.18
	23S	40[c]	38-48	50-61	3	0.25	12.44	0.42
	W2	20[c]	36-50	50-62	3	0.23	14.49	0.12
	W1 (T1-50)	40	32-53	46-65	4	0.64	14.32	0.21
	21	25	26-33	63-69	5	0.38	9.17	0.52
	26S	25	42-50	49-58	2	0.11	15.45	0.06
	T1-43	5	38-40	61-63	4	0.02	12.52	0.02
	T1-31	20	38-45	55-62	3	0.14	13.61	0.24
	34S (T1-20)	5	35-37	62-64	4	0.08	12.57	0.07
	T1-12	10	6-24	75-94	11	0.28	4.59	0.86
	T1-6	15	5-6	92-93	17	0.16	0.66	0.20
	23B (T2-30)	30[c]	37-45	55-61	3	0.25	13.27	0.15
	T2-16.3	20[c]	35	61-64	4	0.08	11.60	0.06
	T2-15	25	29-36	64-69	5	0.23	10.84	0.09
	T2-5	1	62	38	1	0.24	21.41	0.07

a: Each sample was measured in triplicate.
b: Ponar samples.
c: Composite subsample for each 5 cm increments.

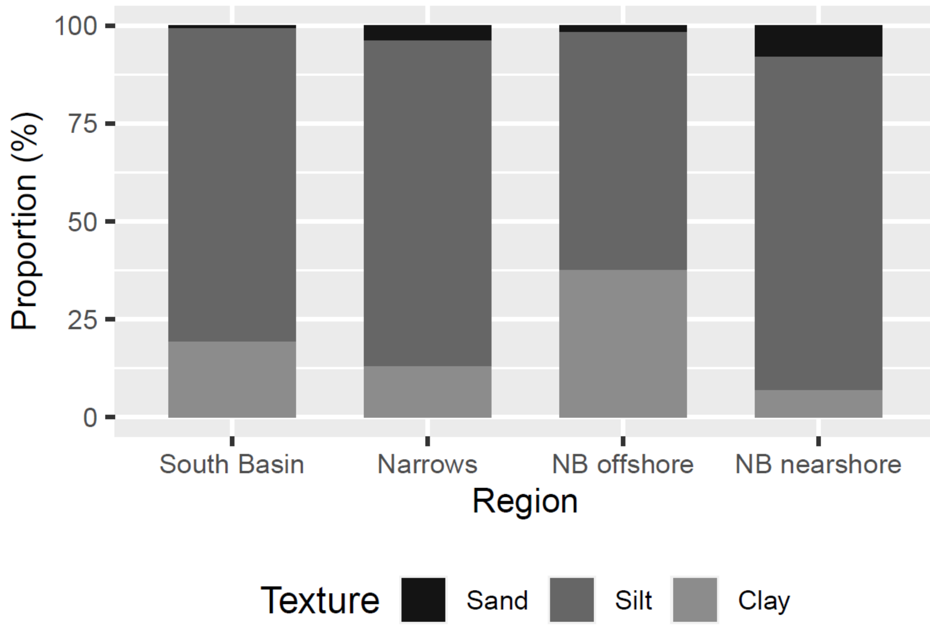

Figure 4-5 Average values of volumetric proportion of different particle size classes of the top 5 cm of sediment cores along the general longitudinal flow path in each region (the sample size for the South Basin, Narrows, North Basin (NB) offshore, and NB nearshore is 16, 20, 50, and 11, respectively).

Fallout radionuclide concentrations ($^{210}Pb_{xs}$ and ^{137}Cs) for the top 1 cm of sediment cores collected in this study and sediment cores taken in 2012-2013 by Matisoff et al. (2017) are presented in Figure 4-6. This figure illustrates that typically the surficial bottom sediments in the NB offshore region contain higher values of $^{210}Pb_{xs}$ and ^{137}Cs than those in the South Basin and Narrows. This mainly stems from the selective transportation of fines to the NB offshore and the fact that the finer sediments contain larger SSA to sorb and transport nutrients (e.g., phosphorus) and contaminants (e.g., some fallout radionuclides) (He and Walling, 1996; Owens and Walling, 2002). Therefore, it can be concluded that the particle size selectivity of sediment transport and deposition processes may, at least in part, be responsible for: a) the spatial patterns of nutrient and contaminant contents of lake bottom sediment (Brunskill et al., 1980); b) the existence of the nutrient-rich particles in the NB offshore (Matisoff et al., 2017; Nürnberg and LaZerte, 2016); and c) large summer blooms in the North Basin due to internal loading (Kling et al., 2011).

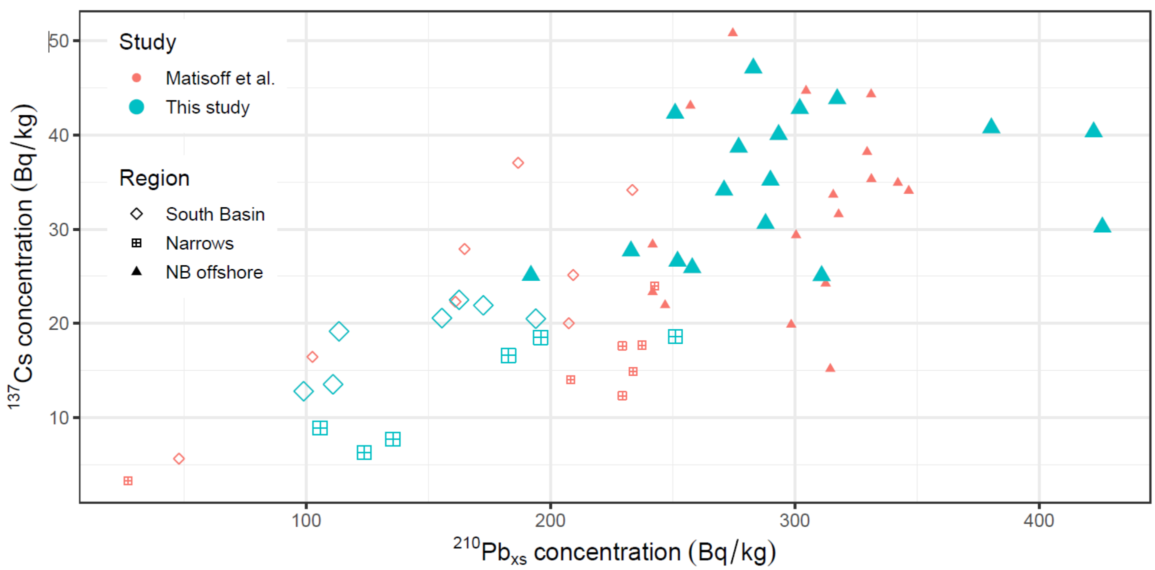

Figure 4-6 Relation between ^{137}Cs and ^{210}Pb$_{xs}$ activities of the top 1 cm of sediment cores collected in this study and by Matisoff et al. (2017) in the South Basin, Narrows, and North Basin (NB) offshore.

The particle size compositions of the north shore materials and sediment cores along two ~50 km transects (i.e., T1 and 2-Mile Channel transects; Figure 4-2) were also used to document the effect of the north shore erosion on lake sedimentation processes, as well as sediment transport to the downstream channel system (i.e., Nelson River). Figure 4-7 shows the different particle size fractions for the top 5 cm of sediment cores along the transects and at Warren's Landing outlet. Along transect T1, the bottom sediments near the north shore (e.g., T1-2) are predominantly silt-sized similar to the bank materials forming the north shore (Kolmogorov-Smirnov test, $p < 0.05$; for more details, see Goharrokhi et al. (2019) and Phillips et al. (2000); data not shown). However, the silt to clay ratio declines with increasing distance from shore until the particle size distribution is indistinguishable from that of sediments in the NB offshore region (i.e., T1-50). Given the information contained in Figures 4-4 and 4-7, and considering the sediment OM content and sediment budget, which are discussed below (Appendix B-B2 and section 4.4.4, respectively), the signature of the north shore materials can be distinguished at least ~20 km southward into the lake.

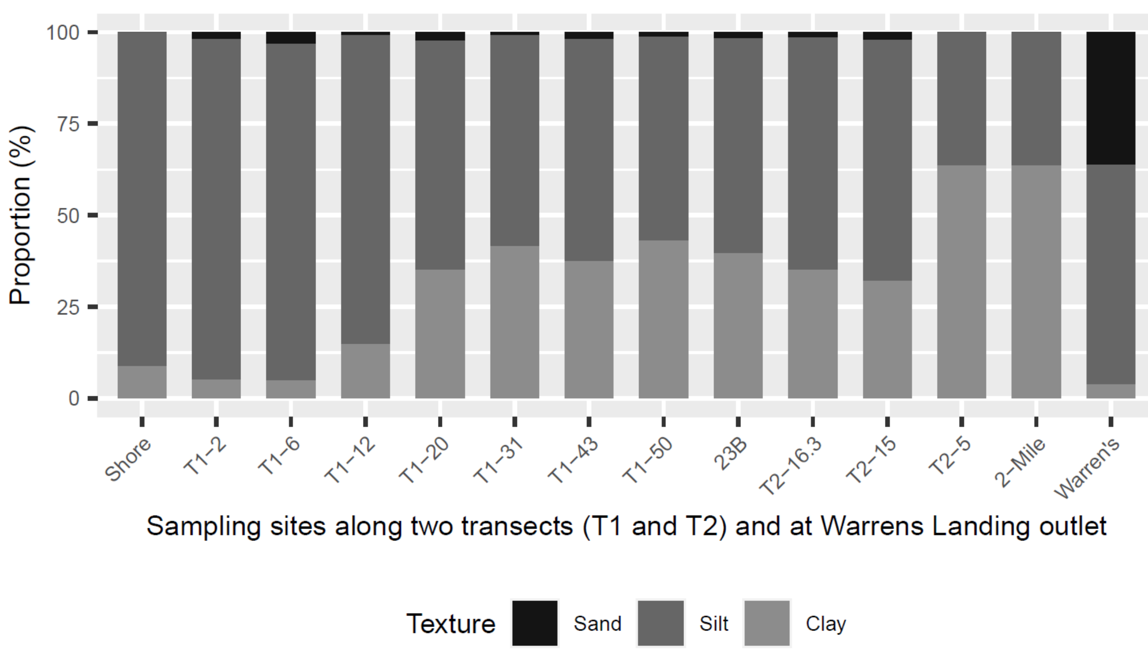

Figure 4-7 Spatial comparison of different primary particle size classes of top 5 cm sediment cores along transects T1 and T2 and at Warren's Landing outlet. Distances from the shore increase along T1 until T1-50 (50 km from shore; W1) then decrease from this point along T2 to the shore at the location of 2-Mile Channel.

In contrast to sediments along transect T1, no evidence of the north shore materials can be observed along transect T2 and the bottom sediments at the entrance to the 2-Mile Channel (e.g., T2-5) are predominantly clay-sized and may be glacial clay materials rather than contemporary mobile sediment (~60% < 2 µm; Figure 4-7). Kimiaghalam and Clark (2017) and Manitoba Hydro (2014b) found that there was considerable erosion of the lake shoreline around the 2-Mile Channel. This, and the absence of coarser particles of the north shore materials in sediment cores along transect T2, suggests that a considerable portion of the north shore eroded materials in that area are likely exported through the 2-Mile Channel (i.e., due to high flow velocity) to the downstream channel system without substantial interaction with the lake bottom (i.e., minimal net deposition).

Additional insight into the sedimentation dynamics in the regions of Lake Winnipeg can be obtained using the Pejrup diagram (Pejrup, 1988). This diagram classifies the hydrodynamic conditions of water bodies as low, moderate, high, and very high using the particle size composition of the bottom sediment (Gadkar et al., 2019). Considering the percentage of particles less than 4 µm in the sediment cores, according to the Pejrub diagram (Appendix B-B10), the South Basin, Narrows, NB offshore, entrance to the 2-Mile Channel, and NB nearshore (including Warren's Landing outlet; Figure 4-7) are classified as moderate, high, moderate, low, and very high hydrodynamic condition, respectively. The high hydrodynamic conditions in the Narrows is in agreement with the low DMAR in this region and it may, therefore, add further strength to the conclusion that riverine suspended sediments tend to stay in suspension under the high-flow conditions through the Narrows.

Considering the reported high kinetic energy of flow and wind–induced bi–directional water exchanges at the 2-Mile Channel (Bijeljanin, 2013; Kimiaghalam and Clark, 2017), a high percentage of particles smaller than 4 µm in the sediment cores at the entrance to the 2-Mile Channel (e.g., T2-5; Figure 4-7) is not consistent with low (i.e., calm) hydrodynamic conditions during sediment accumulation (Appendix B-B10). This reveals that the area uplake of the 2-Mile Channel: a) is a scour zone; b) may not reflect the contemporary deposition processes; and c) requires further research.

4.5.4 Lake Winnipeg total annual dry mass accumulation and sediment budget

Brunskill and Graham (1979) identified major sedimentary basins in Lake Winnipeg based on: a) bathymetry; b) particle size distribution in the surface sediments; c) qualitative interpretation of sonar penetration into sediments; and d) textural stratigraphy of selected cores (n = 14). With this information, in this study it was possible to estimate the total annual dry mass accumulation (Tg/yr) for each sedimentary basin by (area-weighted) extrapolating DMARs from individual sites within the Thiessen polygons (Figure 4-8). These estimates of regional sediment deposition along with estimates of fluvial sediment loading and sediment export were combined to develop a total sediment budget (i.e., organic and inorganic) for the lake (Figure 4-8).

No estimate has been considered for loading by internal biomass productivity or the deposition of atmospheric dust onto the lake surface. Lake Erie, with similar surface area, average depth, and external sediment loading in its west and central basins (Anderson et al., 2017; Rea et al., 1981), is also eutrophic, but there, most autochthonous biomass is apparently decomposed before deep burial (Kemp, 1971). This may also be the case in Lake Winnipeg, but because neither primary production nor organic matter degradation in the sediments are well quantified, there was no attempt to include autochthonous material as a source. Also, in Lake Erie atmospheric loading is estimated to contribute 0.5 Tg/yr of fine sediment (i.e., dust) (Rea et al., 1981). However, most of Lake Winnipeg is more remote than Lake Erie from such sources of airborne dust as intensive agriculture and industrial pollution (Anderson et al., 2017), so that atmospheric loading is likely to contribute less to the sediment budget. Nonetheless, although investigation of either sources is outside of the scope of this study, it is apparent from the Lake Erie example that they warrant further research.

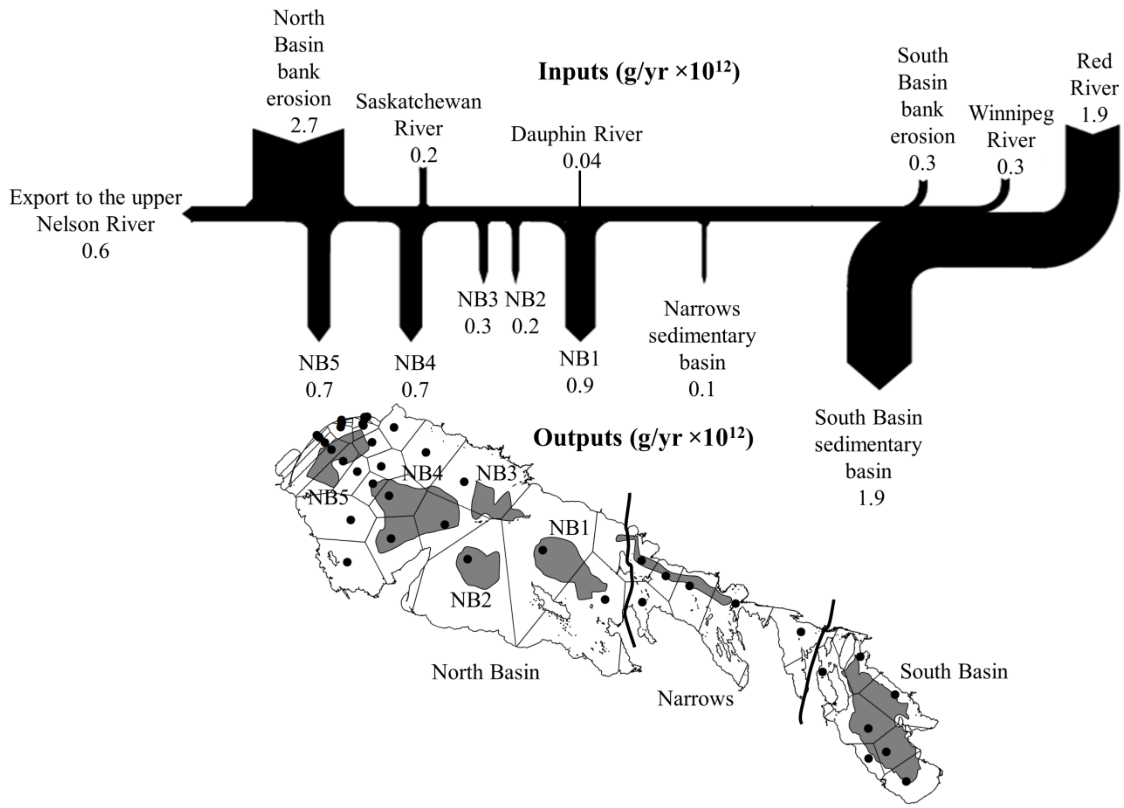

Figure 4-8 Top: sediment budget for Lake Winnipeg; bottom: Lake Winnipeg, sampling sites, three main areas of the lake, seven major sedimentary basins (gray shaded areas), and Thiessen polygons associated with sampling sites.

Sediment loading by bank erosion in the South Basin exacerbated by gradually increasing water levels associated with differential isostatic rebound (20 cm/century; Nielsen, 1998; see also Appendix B-B11) were also included in the sediment budget. The contribution of the bank erosion in the Narrows region was considered to be relatively small, both because the shores are mostly bedrock-controlled, and because wave energy is limited by the relatively short fetches in this region. The east shore of the lake (except the southeast shore of the South Basin) is almost entirely bedrock-controlled (Precambrian granitic assemblages) with broad, stable water-washed zones protecting shallow overburden (Brunskill and Graham, 1979). Some of the west side and island shores in the North Basin, including all of Long Point (a glacial end moraine), are heavily armored with cobbles and boulders winnowed from glacial till (Brunskill and Graham, 1979). Even were this not the case, North Basin shores are not subject to water level rise due to differential isostatic rebound to the degree that South Basin shores are (Nielsen 1998). For all of these reasons, we have assumed here that the west shore and islands of the North Basin contribute little additional loading to the sediment budget. However, the north shore is distinguished by 8–12 m high, actively eroding banks of glacio-lacustrine sediments overlain by peat which do contribute a very significant sediment load to the lake (see Figure B7 in Appendix B-B11).

4.5.5 Sedimentation

Deposition in sedimentary basins delineated by Brunskill and Graham (1979) was estimated from sedimentation rates in cores, collected for this study, by the Thiessen polygon spatial interpolation method (Figure 4-8). The point data within the polygons were aggregated to the closet sedimentary basin. It was assumed that sediment deposited outside these basins is subject to resuspension and focussing (i.e., there is negligible net deposition outside these regions). In consequence, total and regional sedimentation values are sensitive to the assumptions that Brunskill and Graham (1979) used to define regions of net sediment accumulation.

4.5.6 Sediment loading and export to the upper Nelson River

The fluvial sediment loading during the 2004 to 2017 period was estimated using daily discharge (downloaded from the WSC's online data portal) multiplied by TSS values (interpolated from monthly observations supplied on request by the MB DARD) (see Appendix B-B11 for more details). The fluvial loading of the Red, Winnipeg, Dauphin, and Saskatchewan Rivers was 2.8, 0.3, 0.04, and 0.2 Tg/yr, respectively. Brunskill et al. (1980) sediment loading estimation for the period 1969 to 1974 from the three largest tributaries (Red, Winnipeg, Saskatchewan Rivers) was ~2.9 Tg/yr which is broadly similar to the loading estimated for the recent period.

The annual sediment load reported for the Red River by Brunskill et al. (1980) and re-calculated for this study was determined from observations at Selkirk, 30 km upstream of the lake, that is, upstream of the large Netley–Libau marsh complex. TSS measured near the river mouth, downstream of the marshes, is on average 32% lower than at Selkirk (flow-weighted average difference in 27 pairs of samples collected no more than 9 days apart, from 2006 to 2016; see Appendix B-B11). Therefore, it was considered that 0.9 Tg/yr of the sediment load in the lower Red River is diverted into the marshes and only 1.9 Tg/yr is delivered directly into Lake Winnipeg.

To the best of our knowledge, no previous studies have postulated such a large sediment load to these marshes. This raises the question of whether they can sequester such a large load without obvious diminution. Over the last century, differential isostatic rebound and increasing runoff in the watershed have together increased the water level of the South Basin of Lake Winnipeg by an average rate of 0.005 m/yr (Water Survey of Canada records; see Appendix B-B11). The total area of the marshes is about 260 km^2 of which about 190 km^2 is open water or emergent cattails (Grosshans et al., 2004). It would require that sediment deposition be spread over less than 140 km^2 of this open-water, wetland complex to absorb the full 0.9 Tg of sediment each year with no net change in either inundated area or open water volume over the last century (Appendix B-B11). It is beyond the scope of this study to refine the estimation of sedimentation rates in the Netley–Libau marsh complex, or of water level rise. However, it is recommended that the sediment budget of the Netley-Libau marsh complex itself be the subject of further study.

Penner and Swedlo (1974) estimated that shore erosion in the South Basin contributes an additional 0.28 of silt and clay, and 0.23 Tg/yr of sand and gravel to Lake Winnipeg (Nielsen and Conley, 1994). It was assumed that the coarse material is confined largely to the nearshore zone and, therefore, it is not considered further in the fine-grained sediment budget. By examination of Landsat TM satellite data from 1984 to 2006 (i.e., in the post-regulation period) we estimated that 1.1–3.2 Tg/yr is supplied from actively eroding banks along the north shore to Lake Winnipeg (Appendix B-B11). By comparison with results from shore erosion studies at South Indian Lake (Newbury et al., 1978), Brunskill and Graham (1979) estimated a similar 1.5–3 Tg/yr erosional input from the north shore. Considering that the fallout radionuclide dating methods to represent ~60–100 years of sedimentation, these fluvial and erosional sediment input data (i.e., 1969–1974 and 2004–2017) are broadly representative of the accumulation rate periods in the core sections analyzed for this study.

To provide an estimate independent of the input-minus-output budget, inter–basin sediment fluxes were estimated from historical tributary discharge and TSS (WSC and MB DARD data) for the period 2004–2016. Average open water (under-ice) discharge was 1,717 (1,407) and 2,026 (1,513) m^3/s from the South Basin to the Narrows, and from the Narrows into the North Basin, respectively. Corresponding median TSS concentrations, at stations W9 (South Basin to the Narrows) and W7 (Narrows into the North Basin) (Figure 4-2) were 8.0 (2.5) and 6.5 (3.0) g/m^3 [n = 37 (13) and 38 (10)], respectively (open water and under-ice values, with the latter in parentheses; Appendix B-B11). Annual sediment fluxes by this calculation (i.e., $Q \times TSS$) were 0.3 Tg both into and out of the Narrows.

Sediment exported from the lake was calculated in a similar manner. In samples collected during the open water season from 2013 to 2019, median TSS in the 2-Mile Channel and Warren’s Landing were 13.6 (n = 37) and 8 g/m^3 (n = 26), respectively. Neither channel was sampled in winter. Given that neither bank nor bottom sediments are exposed to wind/wave energy under ice, it was assumed that in winter, TSS in the outlets was the same as the median 2.5 g/m^3 (n = 21) recorded under-ice at stations W1 and W3 (central and east side in the NB offshore, Figure 4-2; see Appendix B-B11 for map). Over the same 2013–2019 period, average discharges were 2,830 and 2,530 m^3/s in open water and under-ice, respectively (WSC records at downstream stations). One third was assumed to flow through the 2-Mile Channel, and the rest through the Warren’s Landing outlet (Kimiaghalam and Clark, 2017). Thus, total annual export into the Nelson River was estimated to be 0.6 Tg/yr.

4.5.7 Sediment budget

Overall, fluvial loading and bank erosion supply 2.5 Tg of sediment to the South Basin annually. It was estimated that 1.9 Tg/yr, or 76% of the total loading to the South Basin, is retained in bottom sediments. By subtraction, 0.6 Tg/yr is exported into the Narrows.

The sediment exported through the Narrows, calculated independently as the product of net water discharge and median total suspended solids ($Q \times TSS$) is 0.3 Tg/yr. This value functions as a rough check on the South Basin sediment budget. The difference between the two estimates of export corresponds to 12% of the total loading, or 16% of the estimated deposition. The error may equally well reside in the alternate flux calculation, or in some combination of the three budget terms. However, TSS is typically higher at within-Narrows stations than at the two between-basin stations, so that median TSS at the outlet of the South Basin may be an underestimate (Appendix B-B11). Furthermore, the gross inter-basin discharge greatly exceeds the net flow (due to flow generated during wind-generated setup events, when TSS is likely to exceed the median). Therefore, the higher sediment flux estimate derived from the budget may be the more reasonable value.

Based on the sediment budget, only 0.1 Tg/yr, or 17% of the sediment transported into the Narrows is deposited there. Based on the alternate influx and outflux calculations ($Q \times TSS$), however, there is no measurable net sedimentation. The difference between two results is within the precision of the budget term (0.1 Tg/yr) and, in either case, support the argument that the Narrows sedimentary environment is dominated by no net deposition.

Only 0.3 Tg/y (by $Q \times TSS$) to 0.5 Tg/yr (by the sediment budget) of the sediment load is carried through to the North Basin. The Saskatchewan River contributes another 0.2 Tg/yr, and the Dauphin River a near-negligible 0.04 Tg/yr. It is estimated that 2.8 Tg/yr, is deposited in five sedimentary basins. It is worth noting that using the same spatial interpolation method (i.e., Thiessen polygon) and data from Matisoff et al. (2017), the average annual sediment accumulation is estimated to be ~4.5 Tg/yr, throughout the whole lake, which is in good agreement with the total sediment accumulation of 4.8 Tg/yr based on data presented in this contribution. Finally, it is estimated (i.e., by $Q \times TSS$) that 0.6 Tg/yr is exported into the Nelson River.

It requires an additional load of 2.7–2.9 Tg/yr to balance the budget. Although some of this may derive from erosion of till along western or island shores or the load of unaccounted small tributary rivers, most must be from the actively eroding glacio-lacustrine banks along the north shore. The 2.7–2.9 Tg/yr is within, although near the high end of the range estimated by inspection of satellite imagery, i.e., 1.1–3.2 Tg/yr. Unfortunately, the range is too large to serve as a useful check on other terms of the sediment budget. It is beyond the scope of this study to improve on this estimate. It is, however, recommended that a more precise determination of bank erosion be undertaken as a further check on the estimate of sediment deposition in the NB offshore.

Most of the sediment load carried through the Narrows is probably deposited in the southern sedimentary basins (NB1 to NB3, Figure 4-8) with little reaching the northern sedimentary basins or the outlet. The inter-annual variability of the lake water circulation due to episodic strong wave action during high winds may also play a considerable role in the movement of north shore eroded sediment, which ultimately transport these sediments through the outflows or settle in the deeper sedimentary basins. The sediment deposited near the north shore (NB5 in Figure 4-8; deposition = 0.7 Tg/yr), therefore, must derive almost exclusively from the adjacent eroding banks, in which case this gently sloping region captures 24–26% of the coarser north shore sediments. Up to 0.6 Tg/yr, or 22% of the eroded bank sediments may be exported directly (or indirectly by resuspension and transport of ephemeral nearshore deposits). Zhao et al. (2012) (their

Figure 4-8) demonstrate that the average circulation is to the east along the north shoreline, i.e. preferentially transporting eroded bank materials towards the 2-Mile Channel. Their result is supported by the wind record at Norway House, just north of the lake, where prevailing winds are from the southwest in most summer months (Environment Canada; https://climate.weather.gc.ca/climate_normals/index_e.html). By this accounting, about half (>1.4 Tg/yr) of the sediments eroded from the north shore are transported either directly or gradually by focussing processes to the offshore sedimentary basins. Further investigation of the spatial distributions and fate of sediment from the different sources (South Basin, Saskatchewan River and bank erosion) – possibly using a tracer or fingerprinting approach in order to compare physical and/or geochemical properties of bottom and exported sediments with fluvial and north shore sources – is recommended.

It can be concluded that the load of sediment exported from Lake Winnipeg into the upper Nelson River is mainly derived from erosion of banks very near the outlet channels. Almost all the sediment load derived from the lake's contributing watershed (i.e., 953,250 km^2; Figure 4-1) is, therefore, sequestered in Lake Winnipeg, along with nutrients and contaminants bound to them. Sediments, nutrients, and contaminants will, for the most part, be buried and stored in the long term, but can also be released and/or resuspended cyclically due to internal lake processes, thereby contributing to the functioning of the lake ecosystem for a number of years (Matisoff et al., 2017; Nürnberg and LaZerte, 2016). Consequently, only a small amount of the sediment load from the Lake Winnipeg watershed is transferred downstream to the Nelson River (i.e., there is minimal sediment connectivity). Sequestration of upstream sediments in the lake, and resupply of the Nelson River sediment load by internal processes (bank erosion), together have important implications not only for Lake Winnipeg itself, but also for the Nelson River down to its estuary in Hudson Bay.

4.5.8 Comparison between Lake Winnipeg and Laurentian Great Lakes sediment budgets

A number of published studies report DMAR and sediment budgets for the various Laurentian Great Lakes (Colman and Foster, 1994; Corcoran et al., 2018; Eadie and Robbins, 2005; Joshi et al., 1992; Kemp and Harper, 1977; Kemp et al., 1976; Kemp et al., 1977; Kemp et al., 1978; Klump et al., 2006; Rea et al., 1981; Robbins, 1980). The DMARs for the Laurentian Great Lakes were derived using different methods (including fallout radionuclides). The major morphometric and hydrological characteristics of these lakes are summarized in Appendix B-B12.

The dominant source of the sediment loading to all of these large lakes, including Lake Winnipeg, is shoreline erosion and the sedimentation rate is related to sediment load and lake surface area (Figure 4-9). However, for Lake Winnipeg, this picture is altered when the two major Basins are considered separately. Shoreline erosion contributes 3–4 times more sediment than rivers to the North Basin; however, this source only contributes 20% of the total sediment loading in the South Basin. This is similar to Lake Erie (with an overall shoreline erosion over river supply ratio of 1.9; Rea et al., 1981) where almost all of the sediment from shoreline erosion is supplied to, and deposited in, the central and eastern basins, and almost all of the fluvial sediment is supplied to the western basin (Kemp et al., 1977).

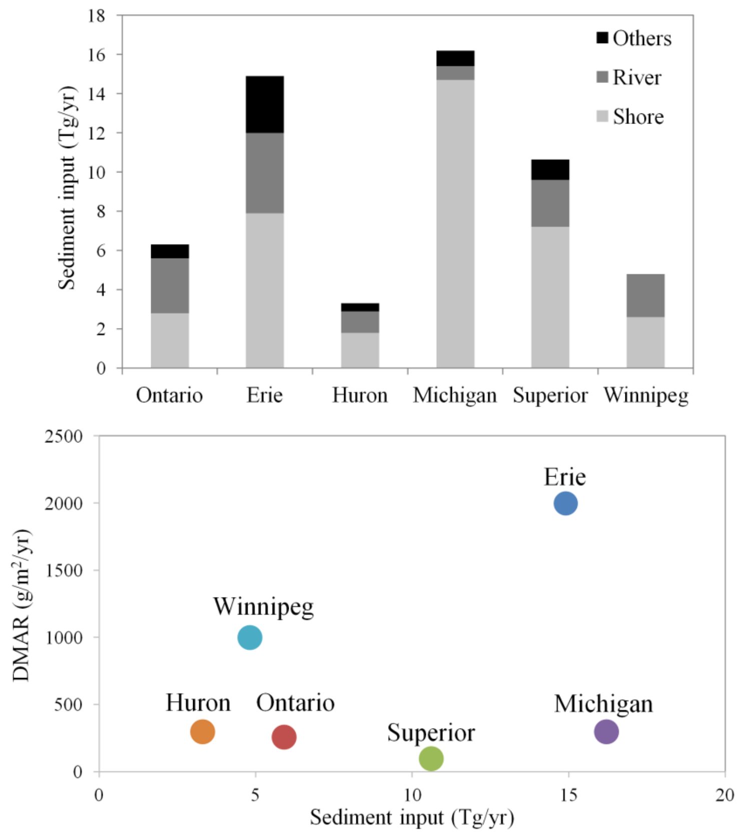

Figure 4-9 Top: contribution of different sediment sources to the total annual sediment input (Tg) for Lake Winnipeg and the Great Lakes; bottom: relation between the DMAR and total sediment input (Tg/yr) for Lake Winnipeg and the Great Lakes. The DMAR data for Lake Winnipeg are based on this study; Bunting et al., 2016; Lockhart et al., 1998, 2000; Matisoff et al., 2017; and Wilkinson and Simpson, 2003. Great Lakes data (i.e., DMAR and/or total sediment input) are based on Colman and Foster, 1994; Corcoran et al., 2018; Eadie and Robbins, 2005; Joshi et al., 1992; Kemp and Harper, 1976; ; Kemp et al., 1977, 1978; Rea et al., 1981; and Robbins, 1980.

As described above, the two lakes are analogous in other ways. They are similar in surface area, average depth, residence times and land use in their watershed (i.e., a large amount of agricultural activity) (Appendix B-B12; Scavia et al., 2019). Not surprisingly, given that the sediment loading to Lake Erie is about three times that to Lake Winnipeg (14.9 and 5.4 Tg/yr, respectively), then the average DMAR measured in Lake Erie is also approximately three times the average DMAR in Lake Winnipeg (2,227 and 863 g/m^2/yr, respectively). The DMARs are higher in the western basin of Lake Erie, near the mouths of the largest riverine sources, than in the central basin (Figure 3 in Kemp et al., 1977); similarly, DMARs tend to be higher in the South Basin of Lake Winnipeg – which, like the western basin in Lake Erie, is fed by the largest riverine source; the Red River –than in the North Basin. The major difference is that Lake Erie has a third large, eastern basin, where DMARs are as high or higher than in the western basin, in spite of there being no really important local source. Kemp et al. (1977) attribute this to circulation carrying sediments from the eroding bluffs along the north shore of the central basin into the deeper water in the eastern basin. This may be possible because the bluffs are formed in tills and glacio-lacustrine sediments that are on average 50% and 31% clay, respectively. (Rukavina and Zeman, 1987). In Lake Winnipeg, the analogous destination would be the upper Nelson River downstream of the lake. However, most of the north shore materials are transported lakeward into sedimentary basins in the NB offshore. These materials are predominantly silt-sized (~90%; Figure 4-7), hence, likely to be transported gradually downslope along the bottom.

4.6 Conclusions

Analysis of the spatial variability of bottom sediment properties and dry mass accumulation rate (DMAR; determined using $^{210}Pb_{xs}$ and ^{137}Cs radionuclides) within Lake Winnipeg were used to investigate the sedimentation dynamics in this large lake. In addition, the role of sedimentation within Lake Winnipeg within the broader sediment transport dynamics in the Nelson River continental-scale watershed was assessed by establishing a total (i.e., organic and inorganic) sediment budget for the lake. Transportation of sediment dominates in the Narrows, whereas sedimentation is dominant in the South Basin and the NB offshore regions. This reflects the role of local morphology (i.e., the relatively small lake width and river-like path with numerous islands) and hydrodynamics on sedimentation in the Narrows. In addition, the considerable contribution of the eroded materials from the north shore in the lake is documented by: a) the high accumulation rate with low inventories of fallout radionuclides in bottom sediment along the slope between the 12 and 16 m isobath in the northernmost part of the North Basin; and b) the NB nearshore-NB offshore gradient in the bottom sediment properties. These differences indicated that the signature of the north shore materials can reach at least ~20 km southward into the lake.

Considering the Lake Winnipeg sediment budget and sedimentation dynamics, it is also concluded that the most of the riverine sediment inputs are being stored in Lake Winnipeg. Given the lines of evidence provided in this study, it is inferred that: a) Lake Winnipeg causes a decoupling of riverine sediment sources between the upstream contributing watershed and the downstream channel system; and b) a considerable amount of the materials eroded from the north shore are exported from the lake without interaction with the lake bottom and this local source is the dominant source of sediment for the downstream system. Given the association between mineralogy and surface area of fine-grained sediment and nutrients (e.g., phosphorus), carbon and contaminants (e.g., metal(loid)s like mercury) then the decoupling (also termed disconnectivity; Fryirs, 2013) of riverine sediment sources also has implications for the transport of these materials.

This paper also identifies areas of further research to provide an improved understanding of sedimentation dynamics within Lake Winnipeg and sediment transport in the downstream river system (i.e., the upper Nelson River system) including: a) constructing an independent sediment budget for the Netley-Libau marsh complex as a key feature in the Lake Winnipeg watershed; b) studying the influence of the effective particle size distribution (i.e., composite particles) of suspended sediment on the sedimentation processes in Lake Winnipeg; c) reviewing the boundaries of major sedimentary basins as well as zones of transport and erosion within Lake Winnipeg, and collecting more sediment cores in each major sedimentary basin to calculate total annual dry mass accumulation with greater precision; d) assessing the effects of 2-Mile Channel on the contemporary deposition processes in the area immediately upstream of the channel; e) investigating the relative contribution of the riverine sediment and the north shore eroded materials to the NB offshore bottom sediment as well as sediments exported to the Nelson River system using diagnostic physical, biological, and geochemical properties of sediment and source materials and the source fingerprinting approach; f) obtaining a more precise estimate of the erosion of the north shoreline bank in order to check the estimate of sediment deposition in Lake Winnipeg; and g) incorporating additional sources of sediment not addressed in this study to the sediment budget, including atmospheric deposition and internal biomass productivity.

4.7 References

Anderson, P.J., Warrack, S., Langen, V., Challis, J.K., Hanson, M.L. and Rennie, M.D., 2017. Microplastic contamination in lake Winnipeg, Canada. Environmental Pollution. 225, 223-231.

Baskaran, M., Miller, C.J., Kumar, A., Andersen, E., Hui, J., Selegean, J.P., Creech, C.T., Barkach, J., 2015. Sediment accumulation rates and sediment dynamics using five different methods in a well-constrained impoundment: Case study from Union Lake, Michigan. Journal of Great Lakes Research. 41(2), 607-617.

Bijeljanin, M., 2013. Application of CRISSP-2D finite element modelling in predicting ice formation processes upstream of the Jenpeg Generating Station. Unpublished MSC Thesis, University of Manitoba, Winnipeg, Canada.

Binding, C.E., Greenberg, T.A., McCullough, G.K., Watson, S.B., Page, E., 2018. An analysis of satellite-derived chlorophyll and algal bloom indices on Lake Winnipeg. Journal of Great Lakes Research. 44(3), 436-446.

Blais, J.M., Kalff, J., 1995. The influence of lake morphometry on sediment focusing. Limnology and Oceanography. 40(3), 582-588.

Brunskill, G.J., B.W. Graham., 1979. The offshore sediments of Lake Winnipeg. Canadian Fisheries and Marine Science Manuscript Report No. 1540. Department of Fisheries and Oceans, Ottawa. 82 pp.

Brunskill, G.J., Elliott, S.E.M., Campbell, P., 1980. Morphometry, hydrology, and watershed data pertinent to the limnology of Lake Winnipeg. Canadian Fisheries and Marine Science Manuscript Report No. 1556. Department of Fisheries and Oceans, Ottawa. 39 pp.

Bunting, L., Leavitt, P.R., Simpson, G.L., Wissel, B., Laird, K.R., Cumming, B.F., St. Amand, A., Engstrom, D.R., 2016. Increased variability and sudden ecosystem state change in Lake Winnipeg, Canada, caused by 20th Century agriculture. Limnology and Oceanography. 61, 2090–2107.

Capelle, D.W., Kuzyk, Z.Z.A., Papakyriakou, T., Guéguen, C., Miller, L.A., Macdonald, R.W., 2020. Effect of terrestrial organic matter on ocean acidification and CO2 flux in an Arctic shelf sea. Progress in Oceanography. 185, 102319.

Colman, S.M. and Foster, D.S., 1994. A sediment budget for southern Lake Michigan: source and sink models for different time intervals. Journal of Great Lakes Research. 20(1), 215-228.

Corcoran, M., Sherif, M.I., Smalley, C., Li, A., Rockne, K.J., Giesy, J.P. and Sturchio, N.C., 2018. Accumulation rates, focusing factors, and chronologies from depth profiles of 210Pb and 137Cs in sediments of the Laurentian Great Lakes. Journal of Great Lakes Research. 44(4), 693-704.

Déry, S.J., Stadnyk, T.A., MacDonald, M.K., Koenig, K.A., Guay, C., 2018. Flow alteration impacts on Hudson Bay river discharge. Hydrological Processes. 32(24), 3576-3587.

Duboc, Q., St-Onge, G., Lajeunesse, P., 2017. Sediment records of the influence of river damming on the dynamics of the Nelson and Churchill Rivers, western Hudson Bay, Canada, during the last centuries. The Holocene. 27(5), 712-725.

Eadie, B.J. and Robbins, J.A., 2005. Composition and accumulation of recent sediments in Lake Michigan. State of Lake Michigan: Ecology, Health and Management, Ecovision World Monograph Series. 89-111.

Environment Canada, Manitoba Water Stewardship., 2011. State of Lake Winnipeg: 1999 to 2007. Manitoba Stewardship Board & Environment Canada, Winnipeg, Manitoba. 209 pp.

Environment and Climate Change Canada, Manitoba Agriculture and Resource Development., 2020. State of Lake Winnipeg-2nd Edition-Highlights. Manitoba Agriculture and Resource Development & Environment and Climate Change Canada, Winnipeg. 175 pp.

Fryirs, K., 2013. (Dis)Connectivity in catchment sediment cascades: a fresh look at the sediment delivery problem. Earth Surface Processes and Landforms. 38, 30–46.

Gadkar, N.S., Nayak, G.N., Nasnodkar, M.R., 2019. Assessment of metal enrichment and bioavailability in mangrove and mudflat sediments of the tropical (Zuari) estuary, west coast of India. Environmental Science and Pollution Research. 26(24), 24998-25011.

Goharrokhi, M., 2015. Effect of hydraulic shear stress on the banks of the Red River. Unpublished MSC Thesis, University of Manitoba, Winnipeg, Canada.

Goharrokhi, M., Pahlavan, H., Lobb, D.A., Owens, P.N., Clark, S.P., 2019. Assessing issues associated with a time-integrated fluvial fine sediment sampler. Hydrological Processes. 33(15), 2048-2056.

Goharrokhi, M., Lobb, D.A., Owens, P.N., 2020. Evaluation of high-flow rate continuous-flow centrifugation and filtration devices for sampling and concentrating fine-grained suspended sediment. Hydrological Processes. 34(19), 3882-3893.

Grosshans, R. E., Wrubleski, D. A., Goldsborough, L. G., 2004. Changes in the emergent plant community of Netley-Libau Marsh between 1979 and 2001. Delta Marsh Field Station (University of Manitoba) Occasional Publication No. 4, Winnipeg, Canada. 52 pp.

Guéguen, C., Mokhtar, M., Perroud, A., McCullough, G., Papakyriakou, T., 2016. Mixing and photoreactivity of dissolved organic matter in the Nelson/Hayes estuarine system (Hudson Bay, Canada). Journal of Marine Systems. 161, 42-48.

Håkanson, L., 1977. The influence of wind, fetch, and water depth on the distribution of sediments in Lake Vänern, Sweden. Canadian Journal of Earth Sciences. 14(3), 397-412.

He, Q., Walling, D.E., 1996. Interpreting particle size effects in the adsorption of 137Cs and unsupported 210Pb by mineral soils and sediments. Journal of Environmental Radioactivity. 30(2), 117-137.

Hjulström, F., 1935. Studies of the morphological activity of rivers as illustrated by the river Fyris. Bull. Geol. Inst., Uppsala. 25, 221-527.

Jweda, J., Baskaran, M., 2011. Interconnected riverine–lacustrine systems as sedimentary repositories: Case study in southeast Michigan using ^{210}Pb and ^{137}Cs-based sediment accumulation and mixing models. Journal of Great Lakes Research. 37(3), 432-446.

Joshi, S.R., Shukla, B.S., Bobba, A.G., 1992. Lead-210 sedimentation in lake Ontario. Environmental Geology and Water Sciences. 19(2), 121-126.

Kemp, A.L.W., 1971. Organic carbon and nitrogen in the surface sediments of Lakes Ontario, Erie and Huron. Journal of Sedimentary Research. 41(2), 537-548.

Kemp, A. L. W. and N. S. Harper. 1976. Sedimentation rates and a sediment budget for Lake Ontario. Journal of Great Lakes Research. 4(2), 324–339.

Kemp, A.L.W., MacInnis, G.A., Harper, N.S., 1977. Sedimentation rates and a revised sediment budget for Lake Erie. Journal of Great Lakes Research. 3(3-4), 221-233.

Kemp, A.L.W., Dell, C.I., Harper, N.S., 1978. Sedimentation rates and sediment budget for Lake Superior. Journal of Great Lakes Research. 4, 276–287.

Kenney, B.C., 1979. Lake surface fluctuations and the mass flow through the Narrows of Lake Winnipeg. Journal of Geophysical Research. 84 (B3), 1225–1235.

KGS Group., 2019. Netley-Libau Marsh restoration pilot project. Manitoba Environment Act Proposal. v + 59 p. + appendices. https://gov.mb.ca/sd/eal/registries/6004netleylibau/eap.pdf.

Kimiaghalam, N., Clark, S.P., Ahmari, H., Hunt, J., 2015. Wave-current induced erosion of cohesive riverbanks in northern Manitoba, Canada. Proceedings of the International Association of Hydrological Sciences. 367, 134-140.

Kimiaghalam, N., Clark, S.P., 2017. Morphodynamics of diversion channels in Northern Manitoba, Canada. Canadian Water Resources Journal / Revue canadienne des ressources hydriques. 42:2, 149-162, DOI: 10.1080/07011784.2016.1249961.

Kling, H.J., Watson, S.B., McCullough, G.K., Stainton, M.P., 2011. Bloom development and phytoplankton succession in Lake Winnipeg: a comparison of historical records with recent data. Aquatic Ecosystem Health and Management. 14(2), 219-224.

Klump, J.V., Weckerly, K., Edgington, D., Anderson, P., Szmania, D., Waples, J. and Eadie, J.D., 2006, February. Historical sedimentation rate determinations in lake Erie. In Lake Erie Millennium Network Conference. Windsor, ON (Vol. 28).

Lévesque, L., Page, E. (Eds.), 2011. State of Lake Winnipeg: 1999–2007, Environment Canada and Manitoba Water Stewardship Report (Available at: http://gov.mb.ca/waterstewardship/water_quality/state_lk_winnipeg_report/index.html).

Liu, J., Hirose, T., Kapfer, M., Bennett, J, McCullough, G.K., Hocheim, K., Stainton, M., 2007. Operational water quality monitoring over Lake Winnipeg using satellite remote sensing data. Our Common Borders–Safety, Security, and the Environment Through Remote Sensing October 28–November 1, 2007, Ottawa, Ontario, Canada.

Lockhart, W.L., Wilkinson, P., Billeck, B.N., Stern, G.A., Danell, A.A., DeLaronde, J., Muir, D.C.G., 2000. Studies of dated sediment cores from Lake Winnipeg, 1994. In: B.J. Todd, C.F.M. Lewis, D.L. Forbes, L.H. Thorleifson, E. Nielson (Eds.), 1996 Lake Winnipeg Project: Cruise Report and Scientific Results, pp. 257–267. Geological Survey of Canada, Open File Report 3470.

Lockhart, W.L., Wilkinson, P., Billeck, B.N., Danell, R.A., Hunt, R.V., Brunskill, G.J., Delaronde, J., Louis, V.S., 1998. Fluxes of mercury to lake sediments in central and northern Canada inferred from dated sediment cores. Biogeochemistry. 40(2-3), 163-173.

Lou, J., Schwab, D.J., Beletsky, D., Hawley, N., 2000. A model of sediment resuspension and transport dynamics in southern Lake Michigan. Journal of Geophysical Research: Oceans. 105(C3), 6591-6610.

Mabit, L., Benmansour, M., Abril, J.M., Walling, D.E., Meusburger, K., Iurian, A.R., Bernard, C., Tarján, S., Owens, P.N., Blake, W.H., Alewell, C., 2014. Fallout ^{210}Pb as a soil and sediment tracer in catchment sediment budget investigations: a review. Earth-Science Reviews. 138, 335-351.

Manitoba Department of Mines and Natural Resources., 1970. Interim license for the regulation of water levels for water power purposes: Lake Winnipeg, Playgreen and Kiskitogisu. https://www.gov.mb.ca/sd/waterstewardship/licensing/pdf/lake_wpg_reg/lwrint.pdf.

Manitoba Hydro., 2014a. Lake Winnipeg regulation; A document in support of Manitoba Hydro's Request for a Final Licence under the Manitoba Water Power Act. Winnipeg, MB: Manitoba Hydro.

Manitoba Hydro., 2014b. Lake Winnipeg regulation project; assessment of 2-Mile & 8-Mile channel surveys. Winnipeg, MB: Manitoba Hydro.

Manitoba Watershed Stewardship. 2006. Literature review related to setting nutrient objectives for Lake Winnipeg. Winnipeg, Manitoba, Canada. 186 pp.

Matisoff, G., Watson, S.B., Guo, J., Duewiger, A., Steely, R., 2017. Sediment and nutrient distribution and resuspension in Lake Winnipeg. Science of the Total Environment. 575, 173-186.

Mayer, T., Simpson, S.L., Thorleifson, L.H., Lockhart, W.L., Wilkinson, P., 2006. Phosphorus geochemistry of recent sediments in the South Basin of Lake Winnipeg. Aquatic Ecosystem Health and Management. 9(3), 307-318.

McCullough, G.K., Hochheim, K., Cooley, P., 2001. Retrospective study of suspended sediment patterns on Lake Winnipeg using NOAA AVHRR satellite imagery. Department of Fisheries and Oceans, Winnipeg. 96 pp.

McCullough, G.K., Page, S., Hesslein, R.H., Stainton, M.P., Kling, H.J., Salki, A., Barber, D.G., 2012. Hydrological forcing of a recent trophic surge in Lake Winnipeg. Journal of Great Lakes Research. 38, 95–105.

McKindles, K.M., Zimba, P.V., Chiu, A.S., Watson, S.B., Gutierrez, D.B., Westrick, J., Kling, H., Davis, T.W., 2019. A multiplex analysis of potentially toxic cyanobacteria in Lake Winnipeg during the 2013 Bloom Season. Toxins. 11(10), p. 587.

Newbury, R.W., Beaty. K.G., and McCullough, G.K., 1978. Initial shoreline erosion in a permafrost affected reservoir, Southern Indian Lake, Canada, in: (Eds.), Third International Conference on Permafrost, Edmonton, Alberta, Canada, 833-839.

Nielsen, E., 1998. Lake Winnipeg coastal submergence over the last three centuries. Journal of Paleolimnology 19: 335–342.

Nielsen, E. and Conley, G., 1994. Sedimentology and geomorphic evolution of the south shore of Lake Winnipeg (Vol. 94, No. 1). Manitoba Energy and Mines. Geological Services.

Nürnberg, G.K., LaZerte, B.D., 2016. More than 20 years of estimated internal phosphorus loading in polymictic, eutrophic Lake Winnipeg, Manitoba. Journal of Great Lakes Research. 42(1), 18-27.

Owens, P.N., Walling, D.E., Leeks, G.J., 1999. Use of floodplain sediment cores to investigate recent historical changes in overbank sedimentation rates and sediment sources in the catchment of the River Ouse, Yorkshire, UK. Catena. 36(1-2), 21-47.

Owens, P.N., Walling, D.E., 2002. The phosphorus content of fluvial sediment in rural and industrialized river basins. Water Research. 36, 685-701.

Phillips, J.M., Russell, M.A. and Walling, D.E., 2000. Time-integrated sampling of fluvial suspended sediment: a simple methodology for small catchments. Hydrological Processes. 14(14), 2589-2602.

Pejrup, M., 1988. The triangular diagram used for classification of estuarine sediments: a new approach. Tide-influenced sedimentary environments and facies. Reidel, Dordrecht. 289-300.

Penner, F., Swedlo, A., 1974. Lake Winnipeg shoreline erosion, sand movement, and ice effects study; The hydrologic, hydraulic and geomorphic studies (Vol. 1). technical report appendix 2.

Rea, D.K., Owen, R.M. and Meyers, P.A., 1981. Sedimentary processes in the Great Lakes. Reviews of Geophysics and Space Physics. 19(4), 635-648.

Robbins, J.A., 1980. Sediments of southern Lake Huron: Elemental composition and accumulation rates. United States Environmental Protection Agency. Environmental Research Laboratory, Duluth. 333 pp.

Rukavina, N.A. and Zeman, A.J., 1987. Erosion and sedimentation along a cohesive shoreline–the 4entral shore of Lake Erie. Journal of Great Lakes Research. 13(2), 202-217.

Scavia, D., Bocaniov, S.A., Dagnew, A., Hu, Y., Kerkez, B., Long, C.M., Muenich, R.L., Read, J., Vaccaro, L. and Wang, Y.C., 2019. Detroit River phosphorus loads: Anatomy of a binational watershed. Journal of Great Lakes Research. 45(6), 1150-1161.

Scott, K.J., McCullough, G.K., Stainton, M., Ayles, B., Hann, B., 2011. Lake Winnipeg- State of the Science, What is the Scientific Basic for Understanding and

Protecting Lake Winnipeg. Lake Winnipeg Research Consortium, Winnipeg. 188pp.

Siev, S., Yang, H., Sok, T., Uk, S., Song, L., Kodikara, D., Oeurng, C., Hul, S., Yoshimura, C., 2018. Sediment dynamics in a large shallow lake characterized by seasonal flood pulse in Southeast Asia. Science of the Total Environment. 631, 597-607.

Valipour, R., Boegman, L., Bouffard, D. and Rao, Y.R., 2017. Sediment resuspension mechanisms and their contributions to high-turbidity events in a large lake. Limnology and Oceanography. 62(3), 1045-1065.

Wilkinson, P., Simpson, S.L., 2003. Radiochemical analysis of Lake Winnipeg 99-900 cores 4 and 6, p. 91–119. In S.L. Simpson, L.H. Thorleifson, C.F. Michael Lewis, J.W. King [Eds.], 1999 Lake Winnipeg project: Cruise Report and Scientific Results. Geological Survey of Canada Open File 4196.

Yamada, I., 2016. Thiessen polygons. International Encyclopedia of Geography: People, the Earth, Environment and Technology: People, the Earth, Environment and Technology. 1-6.

Zhao, J., Rao, Y.R., Wassenaar, L.I., 2012. Numerical modeling of hydrodynamics and tracer dispersion during ice-free period in Lake Winnipeg. Journal of Great Lakes Research. 38, 147-157

CHAPTER 5: Sediment sources and transport dynamics in large, regulated river systems with multiple lakes and reservoirs in the subarctic region of Canada

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5.1 Abstract

The Burntwood River (BR) and Upper Nelson River (UNR) are regulated rivers in the subarctic region of Canada. They merge at Split Lake and then discharge into Hudson Bay via the Lower Nelson River (LNR). The BR water discharge was increased eight-fold by a cross-watershed diversion in 1976. The UNR drains the 11th largest lake in the world, Lake Winnipeg, which itself receives discharge from a large North American Interior Plains watershed. Sediment loads and the source fingerprinting approach in these rivers were used to: a) identify the sediment sources; b) examine the impact of climate and flow regulation on the BR and UNR sediment loads; and c) assess the influence of Split Lake on downstream delivery of sediment into the LNR. Lake Winnipeg effectively decouples the UNR from the sediment sources in its prairie watershed. Fluvial riverbank

and reservoir shoreline erosion in the UNR increased in the late 1990s, in response to a multi-decadal increase in discharge forced by climate change in the Lake Winnipeg watershed. The BR sediment load was increased seven-fold by diversion. Since diversion, flow regulation near the licenced limit has muted the response to variability in local precipitation and runoff; however, erosion processes independent from discharge (bank failures and subaerial processes) add variability in the sediment load. Based on sediment budgeting, Split Lake conveys almost 80% of the BR and UNR sediment load into the LNR. The greater sediment load in the UNR (~1100 Gg/yr, compared to ~530 Gg/yr from the BR) reveals that the UNR is the primary sediment source into the LNR, so that downstream sediment transport dynamics are more sensitive to the environmental changes in the UNR than to disturbances in the BR. Whether this may change in the future depends on engineering responses to increasing demand for hydroelectric power.

5.2 Introduction

It is well documented that fine-grained suspended sediment (i.e., <63 µm) has a high capacity for binding and transporting natural and anthropogenically derived constituents due to its large specific surface area and high chemical reactivity (He and Walling, 1996; Owens and Walling, 2002). Spatial and temporal variability of sources and fluxes of fine-grained suspended sediment due to natural or human-induced controls – including climate forcing of hydrology and construction of dams and diversions with associated water level and discharge regulation – affect: a) the land–ocean flux of contaminants (e.g., metals and persistent organic pollutants) and nutrients (e.g., carbon and phosphorus); b) the fate of these substances; c) the water quality and ecological status of aquatic systems; and d) the geomorphology of fluvial systems.

The Nelson River is an important freshwater–marine corridor from the Interior Plains of North America to the Arctic Ocean via Hudson Bay. Its watershed is the largest by area (i.e., 1,125,520 km^2) and discharge (i.e., average annual discharge (Q_{ave}) at ~170 km upstream of Hudson Bay = 3,540 m^3/s) in the Hudson Bay drainage basin (Déry et al., 2018; Figure 5-1). The climate of a considerable portion of this large watershed (i.e., from Lake Winnipeg to Hudson Bay; discussed in Section 5.3) is subarctic and the major rivers in this region are characterized by numerous riverine lakes which are: a) typically ice-covered for seven months of the year (Bodaly et al., 2007); and b) generally shallow with short water residence times (Bodaly et al., 1984). Natural riverbanks and the shorelines of lakes in this region are largely dominated by wave-washed bedrock overlain with clayey tills or glacio-lacustrine fine sediments deposited by Lake Agassiz.

Over the last half century, the region has experienced climate change in the form of increasing summer temperatures, longer ice-free periods and increasing river discharges (Environment and Climate Change Canada and Manitoba Agriculture and Resource Development, 2020; McCullough, 2015; McCullough et al., 2012; Stadnyk and Déry, 2021). In addition, most reaches in major rivers are heavily regulated by a combination of a cross-watershed water diversion system, hydroelectric generating stations, control structures, and constructed channels. Increasing water discharge due to climate change and/or impoundments supporting hydroelectric generation stations typically raised the water–land interface above wave-washed bedrock into fine sediments, making the fine glacio-lacustrine sediments highly erodible. It is, therefore, fundamentally important to develop an improved understanding of the impacts of these natural and human-induced controls on the sources and fluxes of suspended sediment transported to Hudson Bay, and ultimately to the Arctic Ocean.

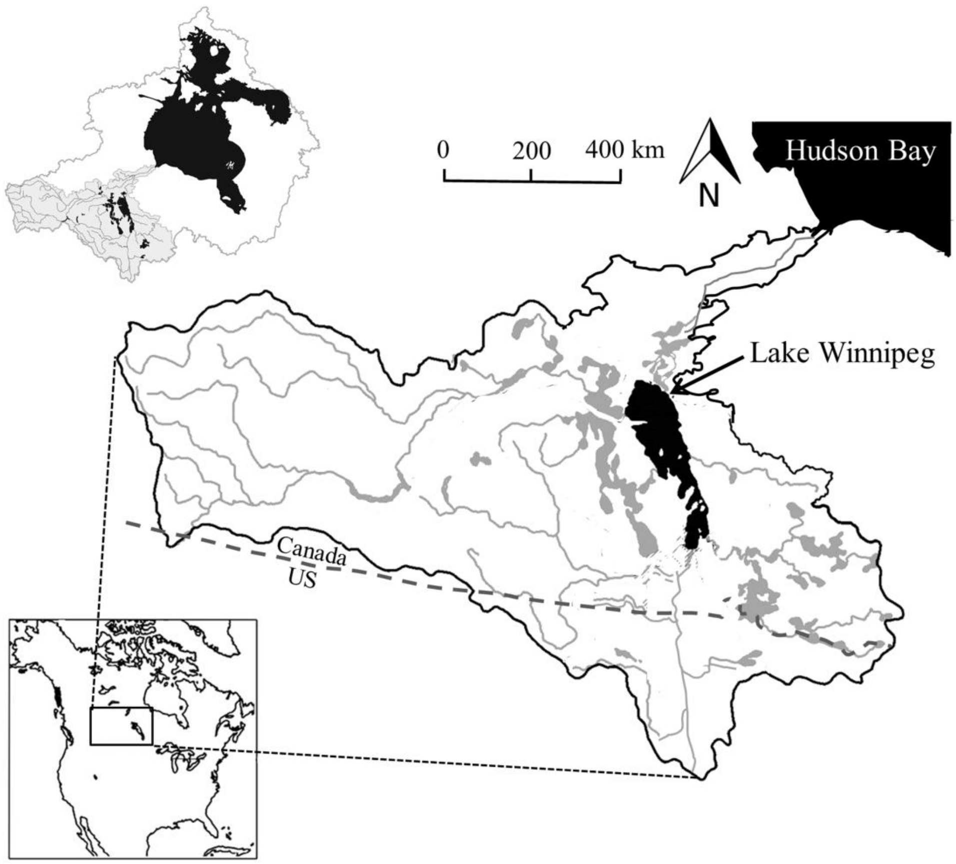


Figure 5-1 Hudson Bay drainage basin and the Nelson River watersheds. The scale bar refers to the Nelson River watershed.

This research forms part of the BaySys project, which focuses on the effects of environmental change, including hydroelectric regulation, on freshwater–marine coupling in the Hudson Bay system (Capelle et al., 2020; Déry et al., 2018; Goharrokhi et al., 2021; Guéguen et al., 2016). Colour- and geochemical-based fingerprints were used to provide information on the sources of the sediment at several key locations along the two major regulated rivers of the subarctic section of the Nelson River watershed, namely, the Burntwood River (BR) and the Upper Nelson River (UNR). The sediment source fingerprinting results and information on recent historical changes in the suspended sediment loads in these two large rivers were used to: a) develop a total (organic and inorganic) suspended sediment budget for Split Lake (which represents the outlet of the BR and UNR watersheds) to assess the influence of this lake on the downstream delivery of sediment to Hudson Bay; and b) assess the sensitivity of the BR and UNR to recent natural and human-induced environmental changes. This study also provides a basis for predicting the impact of future environmental changes on the fluxes of sediment and associated elements in the main river systems contributing to Hudson Bay. This study uses the concept of sediment delivery ratios (e.g., Walling, 1983), sediment source fingerprinting techniques (e.g., Walling, 2013) and sediment budget approaches (e.g., Walling et al., 2001, 2003) to investigate factors influencing land–ocean sediment transport dynamics (e.g., Walling and Fang, 2003; Walling, 2006) in this remote region of the world.

5.3 Study area

The study area includes Lake Winnipeg (watershed area = 953,250 km^2), the BR (watershed area = 25,500 km^2; Q_{ave} = 917 m^3/s), the Grass River (watershed area = 15,400 km^2; Q_{ave} = 66 m^3/s), and the UNR (watershed area = 30,800 km^2 excluding the Grass River watershed (discussed below); Q_{ave} = 2,390 m^3/s) (Figure 5.2). With a surface area of 23,750 km^2 Lake Winnipeg is the 11th largest freshwater lake in the world (Environment and Climate Change Canada and Manitoba Agriculture and Resource Development, 2020) and the third-largest hydroelectric reservoir in the world (Environment Canada and Manitoba Water Stewardship, 2011). It comprises three distinct regions, the South Basin, Narrows, and North Basin with average depths of only 10, 7, and 13 m, respectively (Goharrokhi et al., 2021). It has been regulated to better match seasonal discharge to hydroelectric demand, within historical water levels, since 1976 mainly by constructing: a) 2-Mile and 8-Mile Channels at the outlet and ~60 km downstream from the lake outlets, respectively; b) the Jenpeg hydroelectric generating station (hereafter referred to as Jenpeg) ~100 km downstream of Lake Winnipeg; and c) a control structure at Cross Lake (average depth = 4 m; surface area = 177 km^2; volume = 0.7 km^3; Coordinated Aquatic Monitoring Program (CAMP), 2008a) ~130 km downstream of Lake Winnipeg.

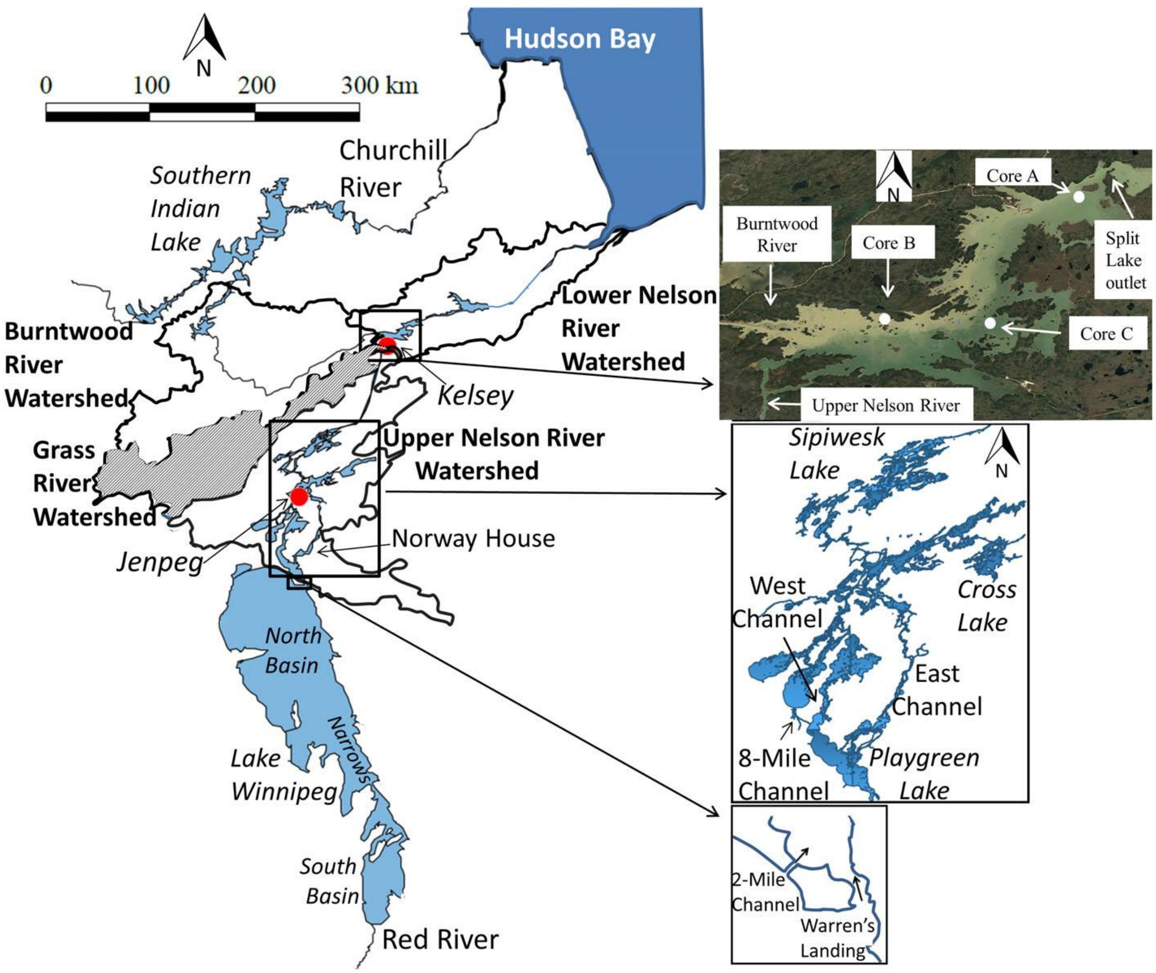


Figure 5-2 Left: study area including Lake Winnipeg and its three distinct regions (South Basin, Narrows, and North Basin), Upper Nelson River (UNR), Grass River, Burntwood River (BR), and Lower Nelson River (LNR) watersheds (red dots represent the Jenpeg and Kelsey generating stations); right (up): Google Earth map: Split Lake and locations of the sediment cores collected by Manitoba Hydro in 1997 and 1998 (Core A, B, and C) and the visible turbid plume extended

from the BR along the northern margin of the lake; right (middle): the UNR between Lake Winnipeg and Sipiwesk Lake; right (below): Lake Winnipeg outlets (2-Mile Channel and Warren's Landing). The scale bar refers to the left side of the figure.

The UNR originates from Lake Winnipeg and, therefore, it receives a considerable amount of water from the Lake Winnipeg watershed. The outflow from the lake into the UNR passes through two outlets: a) the relatively shallow natural outlet at Warren's Landing; and b) the deeper 2-Mile Channel which was constructed to improve hydraulic connectivity between the lake and the control structure at Jenpeg (Goharrokhi et al., 2021). Downstream of these outlets, the UNR splits into two channels which rejoin at Cross Lake. Approximately 85% of the flow passes through the West Channel and the Jenpeg generating station, and the rest flows through the uncontrolled East Channel (Bijeljanin, 2013). Downstream of Cross Lake, the UNR flows through Sipiwesk Lake (surface area = 496 km^2; EC/DFO, 1992) and from then via a narrow, straight channel to Split Lake (average depth = 3.9 m; surface area = 274 km^2; volume = 1.5 km^3; CAMP, 2008b; Lawrence et al., 1999).

Just upstream of Split Lake, the UNR is impacted by the Kelsey generating station located ~7 km upstream of Split Lake. It was constructed in 1961 and created a 166 km^2 reservoir (volume = 130,408 m^3; EC/DFO, 1992). The Grass River joins the UNR between the Kelsey generating station and Split Lake. The Grass River is unregulated and does not receive any water external from its watershed. While the Grass River watershed increases the total UNR watershed area to 46,200 km^2 (i.e., 50% increase), the UNR discharge at the Split Lake inlet increases by less than 5%. Hence, the discharge in the UNR is mainly controlled by the Lake Winnipeg contributing watershed and the local sub-watersheds within the UNR watershed do not contribute greatly to the UNR discharge.

The BR flows into Split Lake and it also receives water external from its watershed to augment flows for hydroelectric generating stations constructed between Split Lake and Hudson Bay (discussed below). Excess water was diverted from the Churchill River watershed (i.e., area = 242,000 km^2) at Southern Indian Lake in the BR in 1976 (Figure 5-2; Manitoba Hydro, 2015; Newbury et al., 1984). Southern Indian Lake was impounded and its water level was raised by ~2 m to divert about three-quarters of the annual average water discharge of the Churchill River to the BR (Manitoba Hydro, 2015). The upper BR (i.e., >112 km upstream of Split Lake at Thompson; watershed area = 18,500 km^2 excluding the Churchill River watershed; Q_{ave} = 883 m^3/s) is in essence a series of lakes joined by short riverine reaches and only ~36% of the distance is riverine. Conversely, about 86% of the lower BR (between Thompson and Split Lake) may be considered riverine stretches (Manitoba Hydro, 2015; Figure 5-2). After merging at Split Lake, both the BR and UNR drain into the Lower Nelson River (LNR). The LNR flows into southwestern Hudson Bay, 300 km downstream of Split Lake. The LNR is also regulated for hydroelectric power generation, and water passes through four more dams before discharging to Hudson Bay.

5.4 Methods

Sediment source fingerprinting approaches were used to determine sources of fine-grained sediment at the Lake Winnipeg outlets and in the BR. Potential sediment sources and suspended sediment samples were collected from several locations between May 2016 and October 2017. Suspended sediment from the Red River as the main riverine input to Lake Winnipeg and sediment eroded from the north shore of the lake (i.e., the ~8–12 m of glacio-lacustrine sediments) were identified as the two main sediment sources at the Lake Winnipeg outlets (Goharrokhi et al., 2021). The Red River suspended sediment samples were collected using a time-integrated suspended sediment sampler between May and November 2016. The sampler was designed by Phillips et al. (2000) and evaluated for this watershed by Goharrokhi et al. (2019). The *MV Namao* research vessel was used to access and collect the north shore materials from the middle height of the shore line using a small, stainless steel trowel.

Discrete or point-in-time suspended sediment samples at the Lake Winnipeg outlets were collected from the *Namao* using high-flow rate continuous-flow centrifugation and filtration devices (for details on these samplers see Goharrokhi et al. (2020)). Suspended sediment samples were collected during whole-lake spring, summer, and fall cruises of the *Namao* and, thus, are representative of any seasonal variability in the contribution of sediment sources.

In addition, the colour properties of north shore materials and suspended sediment samples at the Lake Winnipeg outlets collected by and described in Theroux (2017) were used. These samples were collected in 2014 and are included in the current study to: a) increase the sample number of north shore materials; and b) allow for examination of temporal variability of sediment source fingerprinting results. It is worth noting that in Theroux (2017), the time-integrated sampler was used to collect suspended sediment samples at the Lake Winnipeg outlets from June to October 2014.

To determine the sources of sediment in the BR, suspended sediment and potential source type (topsoil and riverbank materials) samples were collected at five sites in the BR in late July and early August of 2016 and 2017. The sites were located 2, 24, 112, 172, and 259 km upstream of Split Lake. Suspended sediment site coordinates are reported in Appendix C-C1. Suspended sediment samples were collected using the same high-flow-rate samplers described above. Close to each site, a small stainless-steel trowel was used to collect riverbank samples from the middle height of the bank. At the riverbank profile sample site 259 km upstream of Split Lake, sub-samples were collected at 10-cm increments. In the upland area close to each of the riverbank locations, topsoil samples (i.e., surface mineral layer; soil A horizon) were collected from pits dug using a stainless-steel shovel and a soil probe to a depth of 5-30 cm based on the horizon depth.

In addition, information on the sediment accumulation rate in Split Lake was determined using three cores collected by Manitoba Hydro in 1997 and 1998 (Figure 5-2; Core A, B, and C) so as to provide a better understanding of sedimentation dynamics in this lake (Bezte and Lawrence, 1999; Manitoba Hydro, 2015). Core A is about 2 km upstream of the outlet, Core B is in the visibly turbid plume of the BR, about 12 km downstream of the inlet (and 30 km upstream of the outlet by the most direct path), and Core C is centrally located. Bezte and Lawrence (1999) used the unsupported lead-210 ($^{210}Pb_{xs}$) linear fit and constant flux models to estimate the average sediment accumulation rates. Also, the average annual sediment accumulation rates for the cores were estimated using the depth-profiles of caesium-137 (^{137}Cs) concentrations. As these sediment cores were collected in 1997 and 1998, the chronology method based on the location of the ^{137}Cs fallout peak in the cores represents the average sediment accumulation rate for the period of 1963-1997/98 (for more details on the sediment dating methods using ^{137}Cs, see Goharrokhi et al., 2021).

All source and suspended sediment samples were placed in prelabeled plastic bags and stored in coolers (i.e., the Red River, the BR, and Split Lake samples) or in a refrigerator (at 4^{o}C) on board the *Namao* (i.e., Lake Winnipeg samples) for further analyses. In the laboratory, suspended sediments were recovered from the filters as described by Goharrokhi et al. (2020) and the approach described by Perks et al. (2014) was used to obtain samples from the time-integrated sediment sampler. All the samples were dried at 40^{o}C and manually disaggregated with a mortar and pestle. To provide a more direct comparison between suspended sediment and source type samples, all the latter samples were passed through a 63 μm sieve (Laceby et al., 2017).

The volumetric primary particle size distribution of the suspended sediment samples was determined in triplicate using a Malvern Mastersizer 2000 (Malvern, UK) laser diffraction particle size analyser at the University of Manitoba, Canada. Prior to measurement, the organic fraction of ~1 g subsamples was removed using 35% hydrogen peroxide and further dispersed with sodium hexametaphosphate following the procedures outlined in Kroetsch and Cang (2007). Clay, silt, and sand-sized particles in this paper indicates <2 μm, 2-63 μm, and 63-2,000 μm, respectively.

The organic matter content of the suspended sediment samples was determined using the approach described by Siev et al. (2018) by drying ~3 g subsamples at 105^{o}C for 24 hr followed by combustion of the subsample at 550^{o}C for 16 hr. Subsamples of the 1-L water samples collected in the BR were filtered using Whatman GF/F pre-weight and pre-ashed glass fiber filters according to the ASTM standard method (D3977-97; ASTM, 2013) to determine total suspended solids.

The visible and near-infrared (Vis-NIR) reflectance of the samples were measured using a portable spectroradiometer (ASD FieldSpec Pro, Analytical Spectral Device Inc., Boulder, CO, USA) at the University of Manitoba. Samples were placed into Petri dishes and smoothed before conducting the colour test. A white panel (Spectralon diffuse reflectance standard) was used to determine downwelling spectra before performing each test, after which 10 spectra were recorded for each sample. For each sample, the average percent reflectance spectrum (radiance from sample divided by radiance from the white panel) was then determined and used to calculate 15 colour coefficients over the visible wavelength range (i.e., 360-830 nm) (for analytical and calculation details, see Barthod et al., 2015).

The concentrations of 46 geochemical elements of the sediment source and suspended sediment samples were measured using ICP-MS following a microwave-assisted digestion with nitric acid at the Northern Analytical Laboratory Services, University of Northern British Columbia, Canada (for analytical details, see Owens et al., 2019).

Colour- and geochemical-based sediment source fingerprinting approaches were both employed to identify the sources of sediment. This allows colour-based results to be compared with the more conventional geochemical-based approach (Martinez-Carreras et al., 2010). The standard MixSIAR model framework, as a flexible Bayesian model and an open-source R package, was used to estimate the relative contribution of each potential source to the suspended sediment samples (for details, see Blake et al., 2018; Stock et al., 2018; and Stock and Semmens, 2016). One of the main advantages of Bayesian modelling is that the covariance structure of MixSIAR addresses redundancy (i.e., fingerprint selection by discriminant function analysis is not required; Blake et al., 2018).

However, prior to modelling, non-conservative fingerprints were identified and removed using the range test (Collins et al., 1997).

For suspended sediment samples collected in 2014, only a residual error term was included in the mixing model as the time-integrated suspended sediment sampler was continuously capturing sediment over time (Stock and Semmens, 2016). However, for the discrete suspended sediment samples collected by high-flow rate samplers in 2016 and 2017, a process error term was also included. An uninformative prior was specified and the Markov Chain Monte Carlo parameters were set as: chain length = 100,000, burn = 50,000, thin = 50, chains = 3. Model convergence was assessed using the Gelman-Rubin diagnostic (< 1.05).

Suspended sediment fluxes at selected sites in the BR and UNR were calculated using fluvial discharges reported by Environment and Climate Change Canada hydrometric records (https://wateroffice.ec.gc.ca/) and long-term records of total suspended solids data supplied on request by Manitoba Agriculture and Resources Development. Environment and Climate Change Canada hydrometric stations and Manitoba Agriculture and Resources Development suspended sediment sampling site coordinates are reported in Appendix C-C2. At hydrometric stations in the study area, daily and monthly discharges were obtained using hourly water discharge data. Suspended sediment sampling intervals varied between sampling stations and years from bi-weekly to five-six samples per year. Quantification of precision in total suspended solid measurements in the BR and the UNR using point sampling method were carried out by Bezte and Lawrence (1999) and Stainton (2019), respectively. Both studies showed little variation in total suspended sediment concentration due to vertical and horizontal velocity gradients in a cross-section and a precision error of ±1 g/m was estimated for both river systems. Considering the fine particle size composition of suspended sediment in the BR and the UNR (discussed below) these findings are consistent with Horowitz et al. (1990) who indicated that large variation in total suspended solid concentration in vertical and horizontal directions in a river cross-section is mostly due to coarse-grained particles (>63 µm).

The following interpolation-based sediment load estimation algorithm (i.e., method 17 of Phillips et al. (1999)) was used to estimate the annual sediment loads.

$$\text{Total load} = k \left(\sum_{i=1}^{n} \frac{C_i}{n}\right) \overline{Q_r}$$

where K is a conversion factor, C_i is the instantaneous total suspended sediment concentration, n is the number of samples, and Q_r is the average discharge for the period of record.

While the infrequent suspended sediment sampling is concerning with respect to sediment load estimation over longer time-scales, this is the best available data set for the BR and the UNR. The remoteness and logistical difficulties of the subarctic region severely limits data collection. It is important to note that extrapolation-based methods (i.e., rating curves) were not employed as these algorithms may not be able to reflect non-stationarity in the annual sediment load time series in response to a number of factors including environmental changes (Walling, 2006).

Sediment loads at Norway House on the East Channel of the Nelson River were estimated using hydrometric records at Sea River Falls (~30 km downstream of Norway House) and suspended sediment records at Norway House. Discharges for the Sipiwesk Lake outlet were estimated from the hydrometric stations at Jenpeg, Sea River Falls, and the Kelsey generating station. Moreover, Water discharge of the BR at the Split Lake inlet was estimated as the sum of discharge at Thompson and plus the discharge of the Odie River (watershed area = 6,110 km^2; Q_{ave} = 34 m^3/s) which joins the BR between Thompson and Split Lake. In addition, sediment input from the UNR to Split Lake was estimated by using a linear regression method between sediment loads calculated at the Sipiwesk Lake outlet and at the Split Lake inlet below the Kelsey generating station (discussed below).

The relation between annual river discharge and sediment load due to natural and/or human-induced environmental changes was investigated by using a double cumulative mass plot. This plot is a tool used for providing additional insights into the sediment transport dynamics in a river system by identifying whether there are changes in the relation between these two variables (Walling, 2006, 2012; Walling and Fang, 2003).

5.5 Results

5.5.1 Sediment source contributions and properties

The colour- and geochemical-based source contribution results from MixSIAR for the suspended sediment collected at the Lake Winnipeg outlets (i.e., 2-Mile Channel and Warren's Landing) are shown in Figure 5-3 (top panel). Using the median value of the posterior distribution, sediments derived from erosion of glacio-lacustrine fine sediments along the north shore of the lake contributed >85% of the suspended sediment load; sediments derived from the Red River, comprised <15% of the load in the outlets. The entire posterior distributions are shown in Figure 5-3 and highlight the uncertainty surrounding the median values of each sediment sources. The seasonal relative contributions of these two sediment sources at the outlets are presented in Appendix C-C3.

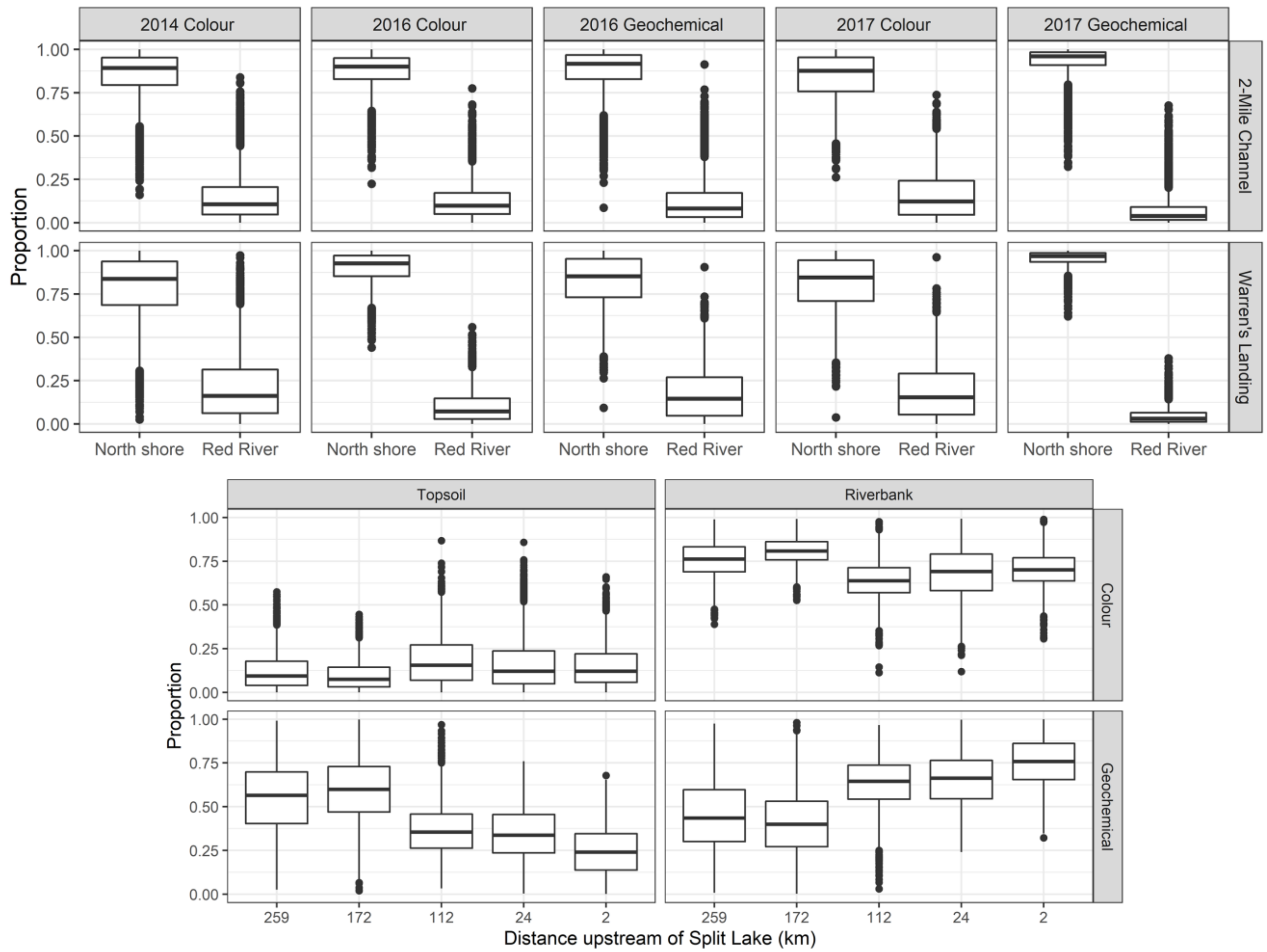


Figure 5-3 Box and whisker plot of the relative contribution from (top panel) the Red River suspended sediment and Lake Winnipeg north shore materials to the suspended sediment at the outlets of the lake; and (bottom panel) from individual source types along the Burntwood River (BR) to the suspended sediment in the BR; using colour and geochemical based fingerprints and MixSIAR.

Both colour and geochemical fingerprinting results showed that riverbank material (as opposed to topsoil) is the source of over two-thirds of suspended sediment in the lower BR ($\leq$ 112 km upstream of Split Lake) and the relative contribution of this source gradually increases downstream (Figure 5-3; bottom panel). In the upper BR and using the geochemical approach, topsoil and riverbank material are nearly equally represented as a source of sediment; however, by the colour approach riverbank material comprises over three quarters of the suspended load. The dissimilar results may possibly be explained by geochemical differences in the source materials that are not indicated by colour fingerprinting. This is not unreasonable, given underlying geology differences between the upper BR, where banks are predominantly granitic with shallow, mainly clayey till overburden, and lower BR, where banks are predominantly metagreywacke overlain by deeper glacio-lacustrine sediments.

Total suspended solids concentration in the BR increase in the downstream direction, from 8–18 g/m^3 at stations 259 and 172 km upstream of Split Lake, to 27 g/m^3, at stations 24 and 2 km above Split Lake. As could be expected given the longitudinal increase in the share of riverbank material in the suspended sediment load, the organic matter content of suspended sediment samples decreases from 4.5% at the upstream stations to 2.5% at stations near Split Lake (Appendix C-C4). The volumetric primary particle size distribution of the inorganic fraction of suspended sediment collected at the Lake Winnipeg outlets and five sites in the BR, samples is dominated by fine-grained materials; the percent silt (2-63 µm) ranges from 70 to 91% (data not shown).

5.5.2 Temporal change in water discharge and suspended sediment load

Time series of annual river discharge and sediment load, limited to the periods of available total suspended solids records, in the UNR at Norway House, Jenpeg, and Sipiwesk Lake outlet, as well as Split Lake outlet at the head of the LNR are shown in Figure 5-4. With the exception of Jenpeg, both time series of discharge and sediment loads show evidence of a statistically significant upward trend over the period of record at the 95% level.

The significant increases in the estimated annual sediment loads are further confirmed by the cumulative double mass plots in the UNR at these sites (Figure 5-4a-d). Slope breaks in the plots indicate that in the UNR the increase in sediment loads relative to water discharges began in the late 1990s (i.e., about 1998). Because data were recorded at Jenpeg only since 2001, it is not possible to identification of a similar late 1990s divergence there. However, the available data do indicate a later divergence at this station, beginning in 2010.

The effect of diverting Churchill River flow in the BR (opened in 1976) had a great impact on erosion and sediment transport. The BR annual river discharge and sediment load at Thompson from 1958 to 2019 are shown in Figure 5-4e. The average annual sediment load since 1977 is more than seven-fold of that in the period 1958 to 1972, primarily as a result of the construction of the Churchill River Diversion between 1974 and 1976. This cross-watershed water diversion in the BR increased the average annual water discharge to about eight-fold of its former value.

In the BR, the annual discharges show a statistically significant ($p < 0.1$) increase over the period 1979 to 2019. However, the trend line fitted to the sediment loads over the same period, suggests that sediment loads have declined. Although the annual sediment loads showed no statistically significant trend, the double mass plot also suggests that the sediment load has decreased over the past ~15 years. This departure from the initial trend in the double mass plot can be seen as a gradual shift rather than the sharp break in slope that is evident in the UNR plots.

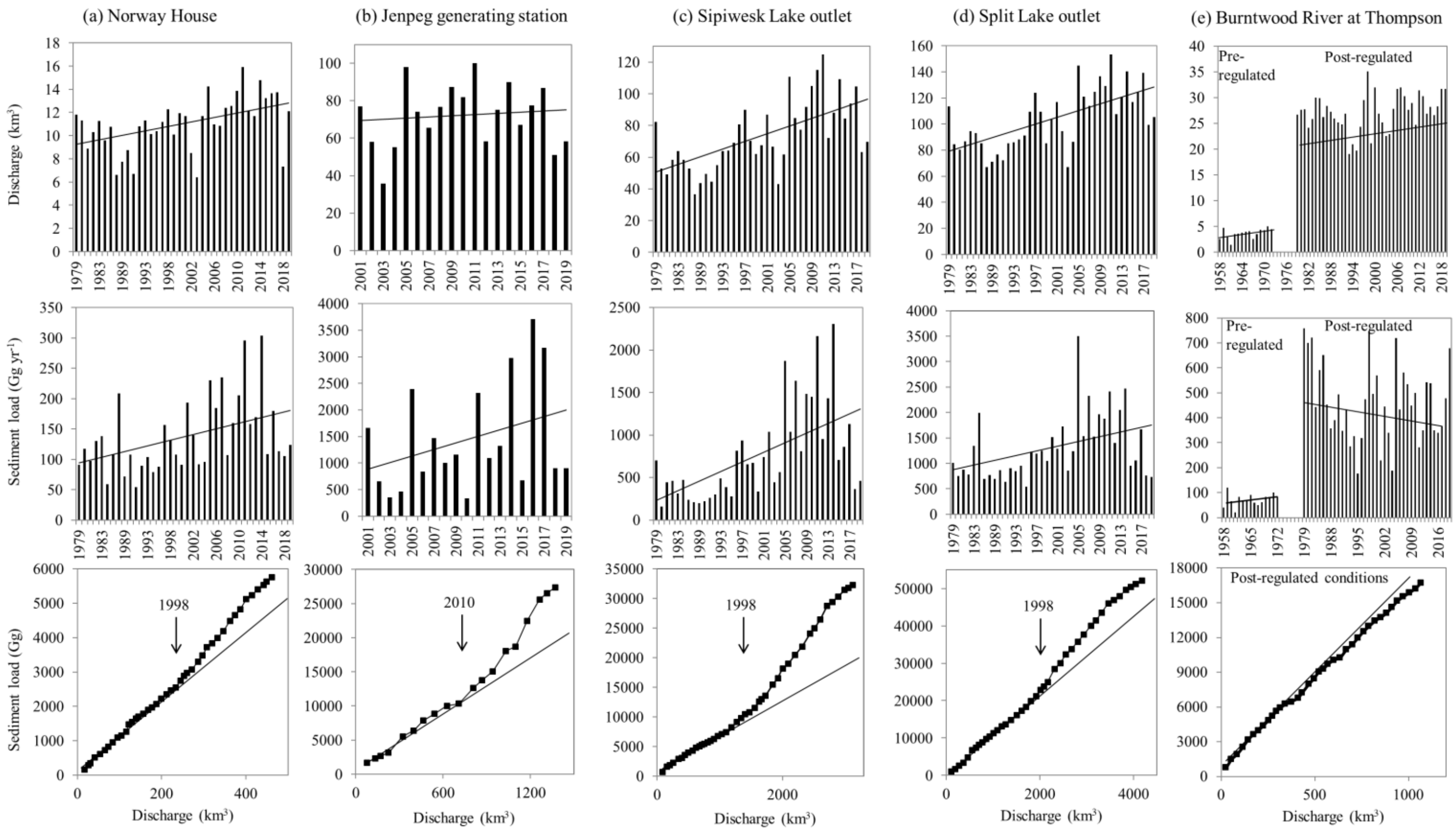


Figure 5-4 Recent trends in the annual water discharge and suspended sediment loads of: (a) the East Channel of the Upper Nelson River (UNR) at Norway House; (b) the West Channel of the UNR at Jenpeg generation station; (c) the UNR at the Sipiwesk Lake outlet; (d) at the Split Lake outlet; and (e) at Thompson in the Burntwood River (BR) and their cumulative double mass plots of annual sediment load vs annual discharge.

5.5.3 Split Lake sediment budget

Water discharge and sediment loads delivered by local tributaries to Split Lake are negligible compared to the BR and the UNR (Manitoba Hydro, 2015). Moreover, considering that most of the shoreline of Split Lake is bedrock-controlled and overlaid by a protective vegetation cover, the shoreline is resistant to erosion during low to normal water levels and minor erosion may occur locally at high water levels (Manitoba Hydro, 2015). Thus, the shoreline materials are unlikely to be a significant source of additional sediment.

There are uncertainties regarding the proportion of sediment loads measured at Thompson that will reach Split Lake, as an unknown part of the load is deposited in shoals at the mouth of the BR in Split Lake (Manitoba Hydro, 2015). Moreover, recurring mass wasting events have been documented at several local sites in the reach of the BR between Thompson and Split Lake (Kellerhals Engineering, 1988). However, precise determination of in-stream losses by sedimentation at the inlet to Split Lake, and the magnitude of local riverbank mass wasting between Thompson and Split Lake are beyond the scope of this study, although they do warrant further research. Nevertheless, a comparison between available total suspended solids data between Thompson and the Split Lake inlet demonstrates that, on average, the total suspended solids at the latter point is greater than that at the former by about 15%. Increasing contributions of riverbank material to the sediment load in a downstream direction is supported by the sediment source fingerprinting results (Figure 5-3), as well as the downstream values of total suspended solids and the organic matter content of suspended sediment (Appendix C-C4; Section 5.4.1).

The average sediment load delivered into Split Lake from 1979–2019 is 530 and 1,100 Gg/yr from the BR and UNR, respectively, which gives a total of 1,630 Gg/yr. The average sediment load exported from the lake is 1,300 Gg/yr. The total amount of sediment sequestered by Split Lake is, therefore, about 330 Gg/yr (i.e., the difference). This is equivalent to 20% of the total annual sediment load delivered to the lake from the BR and UNR (Appendix C-C5).

5.6 Discussion

5.6.1 Decoupling of the North American interior plains watershed at Lake Winnipeg

Sediment source fingerprinting results at the Lake Winnipeg outlets indicate that <15% of the suspended sediment load in the outlets is from the Red River with the remainder being primarily derived from the erosion of the north shore of the lake. Approximately 600 Gg of sediment is exported from Lake Winnipeg into the UNR each year (Goharrokhi et al., 2021). Therefore, using colour and geochemical fingerprinting, ~90 Gg/yr (i.e., 15% of 600 Gg/yr) of the exported sediment is derived from the Red River sediment load. The total incoming sediment load from the Red River is about 2,800 Gg/yr (Goharrokhi et al., 2021). If it is assumed that sediment samples collected from the Red River are representative of the incoming sediment load from the entire Lake Winnipeg watershed – as the other major tributaries supply only 16% of total watershed sediment load – it can be concluded that only about 3% of the Red River sediment load is transferred to the UNR. Even by considering the total incoming sediment load from all major tributaries (~3,340 Gg/yr; including the Red River), the fraction of the watershed sediment load transferred to the UNR is the same, 3%. These results demonstrate that this large lake essentially decouples the UNR from the suspended sediment load derived from its upper watershed. Thus, linking the sediment source fingerprinting and sediment budget approaches identifies Lake Winnipeg as playing a major role in modifying sediment sources and transport in this continental-scale watershed.

5.6.2 Sediment sources and transport dynamics in the Burntwood River (BR)

As mentioned above, the most upstream sites in the BR (i.e., sites at 259 and 172 km above Split Lake) are reasonably representative of the predominantly fluvial-lacustrine regime (64% of mainstem length) and, conversely, sites at 2, 24, and 112 km above the lake are in riverine stretches in the lower BR (86% of mainstem length) (Manitoba Hydro, 2015). The lower total suspended solids at stations 259 and 172 km upstream of Split Lake (Appendix C-C4) can partly be explained by the lower overall sediment transport capacity of the upper BR. This is likely due to the system being a series of linked lacustrine environments rather than a continuous riverine reach. However, the higher total suspended solid concentration in the lower BR is supported by the higher sediment transport capacity of the faster flowing, predominantly riverine regime. The lower BR is also known for several large rotational bank failures of ice-rich clay banks, activated as part of riverbank re-adjustment to accommodate the eight-fold increase in discharge. That is, the reach includes intermittent (spatially and temporally) sources of fine sediments to the BR (Kellerhals Engineering, 1998). In another early study (i.e., Northwest Hydraulic Consultants Ltd., 1987) the authors noted that the primary sediment source in the lower BR was riverbank materials, in contrast to the upper BR where erosion of reservoir banks was the primary source.

The higher organic matter content of the suspended load at sites in the upper BR (Appendix C-C4) may derive from the degradation of the extensive wetlands and forests inundated during the construction of these reservoirs (i.e., Notigi and Wuskwatim). Moreover, the low-relief glacio-lacustrine clays and silts that border the reservoirs are overlain by deep peat. Therefore, both inundated wetlands and forests, and eroding peat bordering reservoirs in the upper BR may explain the higher organic matter content of the suspended load at sites in the upper BR.

Overall, the low organic matter content of the lower BR suspended load is consistent with the low organic matter content of the suspended sediment collected by sediment traps at 20 sites in Split Lake during the open-water season of 2009 and 2012 conducted by Manitoba Hydro, i.e., on average ~<3% of the total sediment mass (data not shown) (Manitoba Hydro, 2015). This supports the sediment source fingerprinting results which indicate that the materials with high organic matter content (eroded topsoil) are unlikely to be a primary source of sediment in the lower BR and Split Lake.

5.6.3 Sensitivity of the Burntwood River (BR) and Upper Nelson River (UNR) to recent natural and human-induced environmental changes

5.6.3.1 Importance of environmental changes in the Upper Nelson River (UNR)

Several studies have documented increasing water discharge since the 1990s in regulated and unregulated rivers that flow into Lake Winnipeg, and have attributed this to increasing precipitation in major contributing watersheds (e.g., Environment and Climate Change Canada and Manitoba Agriculture and Resource Development, 2020; McCullough, 2015; McCullough et al., 2012). Therefore, a change in climatic conditions is the main cause for the increasing discharge in the UNR over ~1999-2019 (Figure 5-4).

Given that sediment transport in the UNR is largely decoupled from its upper watershed due to sequestration of riverine sediment in Lake Winnipeg, as indicated by both the Lake Winnipeg sediment budget and fingerprinting results (Section 5.5.1), the increased sediment load in the UNR, unlike the source of increase in the discharge, must be derived from local sources. This strongly suggests that land and water management practices impacting erosion of fields and riverbanks upstream of Lake Winnipeg has little impact on sediment loading in the UNR.

The spatial records of annual discharges and sediment loads can also be used to assess the existence of inter-regional variability in response to environmental change. For example, at all four sites in the UNR, the average annual discharge during 2010-2019 increased between 7-16% compared to the period 2001-2009. However, the average annual sediment loads during the former period at Jenpeg, Norway House, and Sipiwesk Lake outlet increased by about 57%, 10%, and 10% relative to those in the period 2001-2009, respectively. The equivalent value at the Split Lake outlet was a 13% reduction (Table 5-1). These sediment loads results demonstrate that the West Channel of the UNR is more responsive to water level fluctuation and increasing discharge than both the East Channel and the lower reach of the UNR. The West Channel is impounded above Jenpeg generating station; its shores have had only decades to stabilize to new (and variable) water levels. The lower responsiveness of sediment load in the East Channel may be because its shores have been washed clean in response to fluctuating water levels over centuries, if not millennia. It is not clear why the UNR at the outlet of Sipiwesk Lake was less responsive to higher discharge. Kelsey generating station impounded the Nelson River, destabilizing shorelines as far upstream as Sipiwesk Lake, where some banks are still observed to be actively eroding. However, the impoundment is 15 years older than the Jenpeg forebay, and it may be that fewer unstable banks remain within reach of waves, even at higher water levels (Manitoba Hydro, 2015).

5.6.3.2 Importance of Environmental Changes in the Burntwood River (BR)

The low inter-annual discharge variability of the BR after 1979 (top panel in Figure 5-4e) is explained by regulation of flows diverted from the Churchill River. Diversion discharge has been fairly constant since an augmented flow regime was instituted under annually renewed Provincial operating licences (Manitoba Hydro, 2010). Since the diversion contributes most of the flow in the BR and the river is generally operated near the licenced limit, it has partially masked potential climate-forced increase in local runoff. This can be seen in Table 5-1 as the average annual water discharge and sediment load at Thompson for the recent period (1999–2019) changed from those for the period prior to 1999, by 7% and -7%, respectively

Other published studies provide further evidence that the sediment load has decreased despite the increased discharge in the BR under post-diversion conditions (CAMP, 2018; Stange, 1990; Vitkin and Penner, 1979). Data for these studies were collected independently of the long-term Provincial water quality data that were used to create Figure 5-4. Using these studies, the BR average annual sediment load at the Split Lake inlet for 1977-1979, 1987-1989, and 2008 is estimated as ~830, 640, and 510 Gg/yr, respectively. Thus, while annual average discharge was 26% and 11% higher in 2008 than in 1977-1979 and 1987-1989 respectively, the annual average sediment load decreased by about 40% and 20%, respectively.

The higher sediment loads in the first years after the diversion (i.e., ~700–750 Gg/yr at Thompson and ~830 Gg/yr at the Split Lake inlet; Figure 5-4) were rare later in the time series. While the average sediment load at Thompson from 1982–2019 was less than two-thirds that reported for 1979-1981, it has been highly variable, and in some years has approached or matched the magnitude reported for the early post-diversion years. This indicates that intermittent erosion processes, related to, but quasi-independent from water discharge, such as subaerial processes and bank failure, are significant sources of sediment to the system. In localized areas in the BR, previous studies have linked specific instances of large sediment loads to episodic events, including sudden bank or bluff slumping, and wave action associated with high wind events (Northwest Hydraulic Consultants Ltd. 1987, 1988).

Table 5-1 Average water discharges and suspended sediment loads, percent changes of discharges and sediment loads, and sediment load to discharge multiplier for different periods (i.e., 1979-1998, 1999-2019, 2001-2009, and 2010-2019; column 2) in the Burntwood River (BR) at Thompson (column 3); in the Upper Nelson River (UNR) at Norway House (East Channel; column 4), at Jenpeg generating station (West Channel; column 5), summation of the East and West Channels (column 6), at Sipiwesk Lake outlet (column 7), and at Split Lake outlet (column 8).

1	2	3	4	5	6	7	8
	Period	Thompson	Norway House	Jenpeg	Column 4 + 5 (West + East Channels)	Sipiwesk Lake outlet	Split Lake outlet
Discharge (m^3/s)	1979-2019	883	350	1,949	2,299	2,332	3,316
	1979-98	853	317	1,637	1,954	1,912	2,849
	1999-2019	911 (7%)†	377 (19%)	2,246 (37%)	2,623 (34%)	2,692 (41%)	3,671 (29%)
	2001-09	891	350	2,211	2,561	2,563	3,544
	2010-19	938 (5%)	407 (16%)	2,366 (7%)	2,773 (8%)	2,935 (14%)	3,952 (11%)
Sediment load (Gg/yr)	1979-2019	464	138	NA‡	>138	769	1,304
	1979-98	482	111	NA	>111	419	960
	1999-2019	448 (-7%)	162 (46%)	NA (NA)	>162 (NA)	1,070 (155%)	1,600 (68%)

	2001-09	435	160	1,111	1,271	1,070	1,772
	2010-19	442 (2%)	176 (10%)	1,740 (57%)	1,916 (51%)	1,182 (10%)	1,536 (-13%)
Multiplier	Between 1979-98 and 1999-2019	-1.04	2.43	NA	NA	3.81	2.35
	Between 2001-09 and 2010-19	0.3	0.61	8.1	6.13	0.7	-1.16

†: Percent changes of discharges and sediment loads given in parentheses.

‡: Not Available (owing to the lack of data).

5.6.4 Influence of Split Lake on downstream delivery of sediment

Considering Section 5.5.3, it is possible to establish a sediment budget for Split Lake for 1979-1998 and 1999-2019. This demonstrates the stability of the lake sediment budget in spite of environmental change. From 1979-1998, the BR and UNR contributed similar sediment loads (i.e., 550 and 620 Gg/yr, respectively). However, during the subsequent period a 40% increase in the average annual discharge in the UNR was observed, mainly due to climate change (increased runoff due to changes in precipitation) in the Lake Winnipeg watershed. This increase in discharge resulted in a 155% increase in the average annual sediment load in the UNR (Figure 5-5). By comparison, between-period differences in water discharge and sediment load of the BR were minor. Therefore, in the second period, the UNR represented the primary sediment source for Split Lake with an average contribution of 1,500 Gg/yr. Nevertheless, in spite of this large increase in sediment supply from the UNR, the sediment trapping efficiency for Split Lake (i.e., ~20%) remained constant. This may relate to the short water residence time in the lake (discussed below). Given the magnitude of sediment loads of the BR and UNR in the second period and the low sediment trapping efficiency in Split Lake, the UNR is the primary source of sediment (~73%) supplied to the LNR in this period.

The results of continuous monitoring of sediment flux using turbidity sensors at the BR and UNR mouths and the Split Lake outlet from July through September 2008 (CAMP, 2018) were used to validate the estimation of sediment trapping efficiency in Split Lake. While the average daily sediment load at the outlet of the lake was 5.3 Gg, the equivalent values for the BR and UNR inlets were 1.4 Gg and 5.1 Gg, respectively, over that study period (CAMP, 2018). Assuming negligible sediment input from other rivers and the

Split Lake shoreline, about 20% of the sediment load from the BR and UNR was sequestered in the lake, i.e., the same as reported herein for the period 1999-2019. It can also be estimated from the CAMP (2018) data that during the open-water season of 2008, the UNR supplied about 80% of the sediment that was delivered to the LNR, only slightly more than the 73% estimated herein for the period 1999–2019.

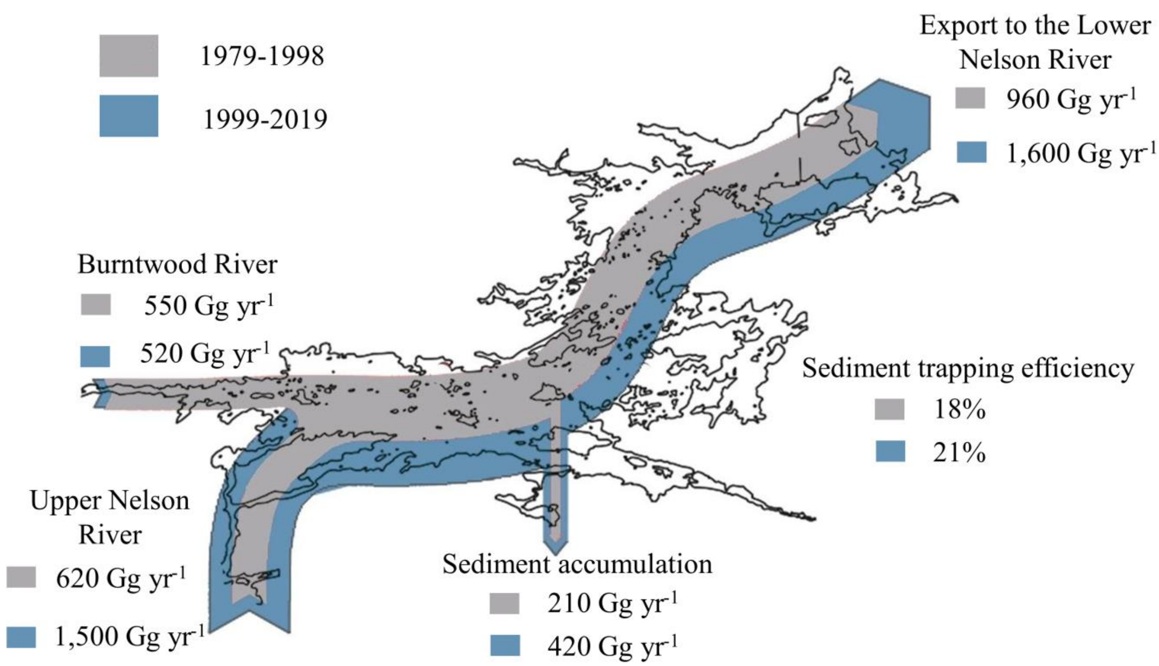


Figure 5-5 The sediment budget for Split Lake under post-regulated conditions for the periods 1979-1998 and 1999-2019.

The use of theoretical methods for estimating sediment trapping efficiencies in lakes and reservoirs has been well documented (e.g., Foster and Walling, 1994; Walling et al., 2003). An assessment of the sediment trapping efficiency in Split Lake under post-regulated conditions was, therefore, performed by: a) calculating the ratio of the lake volume (1.5 km^3) over the average annual water discharge at the lake outlet (104 km^3); b) considering the particle size distribution of the BR and UNR suspended sediment (Section 5.5.1); and c) using the fine-grained sediment curve developed by Brune (1953). This provides an estimate of the sediment trapping efficiency of Split Lake of <40%, suggesting that the sediment trapping efficiency estimated by the sediment mass balance and the theoretical method are in only rough agreement. While the value using the latter method is greater than the former, albeit still low (i.e., <40%), several studies have shown that the Brune theoretical method considerably overestimates sediment trapping efficiencies for lakes and reservoirs (Bashar et al., 2010; Lewis et al., 2013; Revel et al., 2015; Ward, 1980).

Sedimentation rates determined by Bezte and Lawrence (1999) and reported by Manitoba Hydro (2015) from Split Lake cores generally supports the Split Lake sediment budget results. Using concentrations of $^{210}Pb_{xs}$, Bezte and Lawrence (1999) estimated the average 100-year sediment accumulation rate range as 788 to 1,070 g/m^2/yr. Using the ^{137}Cs chronology model, sedimentation rates varied from 1,117 to 1,269 g/m^2/yr (Table 5-2). The ^{137}Cs rates are the average sediment accumulation from about 1963 to 1997 or 1998, the years the cores were collected. The rate of sediment sequestration in Split Lake was 216 Gg/yr over ~100 years (using $^{210}Pb_{xs}$) and 348 Gg/yr over 34–35 years (based on the ^{137}Cs peak). These rates are the product of lake surface area and the minimum and maximum sediment accumulation rates in cores A to C, which are widely spaced through the lake. Moreover, the two most centrally located cores (Cores A and C) fall inside and outside the visible turbid plume of the BR (Appendix C-C6). Even so, in such a large, complex lake, the derived average lake-wide sediment accumulation rates must be interpreted with caution.

Table 5-2 Average annual sediment accumulation rates using the $^{210}Pb_{xs}$-linear fit and $^{210}Pb_{xs}$-constant flux models (adapted from Bezte and Lawrence (1999)) and the peak ^{137}Cs values for the cores collected in Split Lake in 1997 and 1998.

Sample	Sediment accumulation rate (g/m^2/yr)		
	$^{210}Pb_{xs}$ (linear fit)	$^{210}Pb_{xs}$ (constant flux)	^{137}Cs
Core A	1,070	ND†	1,117
Core B	994	904	1,269
Core C	788	834	ND

†: ND = Not datable.

It is worth noting that Bezte and Lawrence (1999) measured the current velocity along the main stem of the lake (i.e., locations of Cores A and B; Figure 5-2) as 10 cm/s at the time of sampling in 1997 and 1998. This current velocity may exceed the threshold value for fine-grained materials (<63 μm) to stay in suspension in the water column (Goharrokhi, 2015; Hjulström, 1935). The particle size distribution of the sediment samples collected in 2016 and 2017 were dominated by fine-grained materials which is in agreement with previous studies reporting the particle size distribution of the sediment collected by sediment traps at 20 sites within Split Lake in 1997-1998, 2008-2010, and 2012 (Manitoba Hydro, 2015). This high current velocity provides supporting evidence for the short water residence time in the lake and the low and relatively constant sediment trapping efficiency in Split Lake.

From the above, it is suggested that: a) a large quantity of the seven-fold increase in the BR suspended sediment load is intercepted by Split Lake; b) riverbank and lake shore erosion associated with diversion of the Churchill River does not measurably add to the sediment load transported from the BR into the LNR; c) the UNR is currently the primary sediment source for both Split Lake and the LNR; and d) the LNR sediment transport dynamics is more sensitive to the variability of sources and fluxes of sediment in the UNR due to environmental changes.

5.6.5 Comparison between the sediment loads in the Burntwood River (BR) and Upper Nelson River (UNR) and selected large rivers

Numerous studies have considered the influence of dams on the sediment loads in the BR and the UNR with other large rivers in the world. For example, Syvitski et al. (2022), estimated that, on average, large dams and their associated reservoirs on 34 major rivers are responsible for a reduction of ~74% in the sediment loads in those rivers. Dethier et al. (2022) determined that sediment loads for major rivers in the global North (i.e., above 20 ^{o}N) had declined by an average of 49% compared to pre-dam conditions. Lake Winnipeg, the third largest hydroelectric reservoir in the world, is also among the most effective reservoirs in retaining sediments; in that it sequesters ~97% of upstream riverine sediment loads. However, this is not, at least not entirely, due to regulation. As a reservoir, the water level of Lake Winnipeg is managed within its historical level range, so that the annual residence time is not altered. Although there is insufficient pre-regulation sediment data to demonstrate this, it is unlikely that the sediment trapping efficiency of the lake has been measurably increased. It differs from most large reservoirs of the world, which more typically involve converting a low residence time riverine reach into a higher residence time lake (e.g., dams on the Missouri River (Walling 2002) and on major Chinese rivers (Syvitski et al., 2022)).

The BR and the UNR are also somewhat atypical in that they are very dilute systems. In flow-weighted suspended sediment concentrations, they rank 7th and 8th most dilute among 34 large rivers in a compilation by Best (2019) (Figure 5-6). The UNR is most similar in suspended sediment concentration to the St. Laurence River. They have similar watershed areas (i.e., both ~1,100,000 km^2), both including a large part in glaciated terrain with shallow overburden (limited source materials). Perhaps most significantly, both flow through large lakes in their middle or lower reaches which interrupt the transport of sediment loads developed from headwater sources.

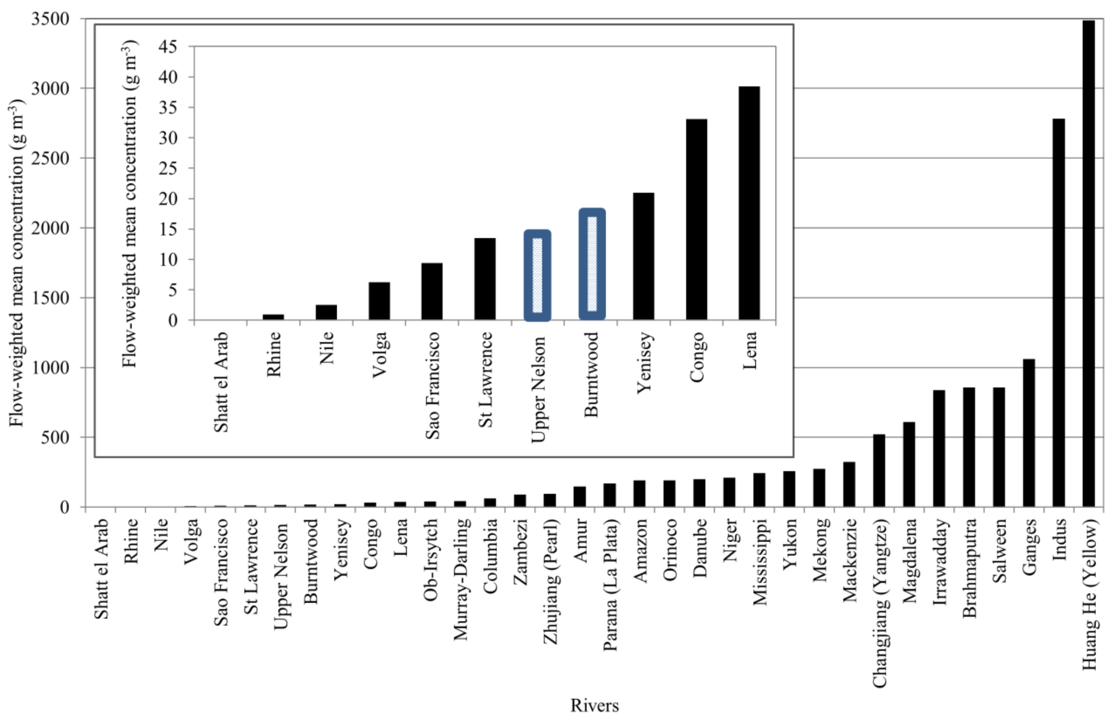


Figure 5-6 Flow-weighted means concentration of 34 large rivers including the Burntwood River (BR) and the Upper Nelson River (UNR).

The initial response of the UNR to initial regulation, at least in Sipiwesk Lake upstream of Kelsey generating station, and in the local forebay above Jenpeg generating station, may have been like at Southern Indian Lake, which was impounded in 1976, raising it by ~2 m to facilitate the Churchill River diversion. In the early years after impoundment, sediment loading by shore erosion exceeded tributary loading by more than an order of magnitude, with the result that sediment export increased three–five fold (Hecky and McCullough 1984) in spite of an increase in sediment trapping efficiency from ~50% to ~87%. This is likely not an uncommon impact of reservoir creation in dilute river systems. It is the case in the BR, where shoreline erosion and channel rebuilding associated with an eight-fold increase in water discharge resulted in a seven-fold increase in sediment export from the system. Effects of dams in such dilute systems appear not to be well-represented in global studies of the impacts of river regulation.

The effect of climate change on changing water discharge and sediment load is a global concern. Using data from eight large rivers in China, Lu et al. (2013) found that – on average – every 1% change in water discharge due to climate change from 1991-2007 resulted in a 1.6% change in sediment loads. The equivalent value for the UNR at the Norway House and Sipiwesk Lake outlet was ~1-2% higher. Under post-regulation conditions, a 1% change in water discharge resulted in a 2.4% and 3.8% increase in sediment loads at the Norway House and Sipiwesk Lake outlets, respectively (Table 1). This suggests that the subarctic region of Canada (including the UNR) is more sensitive to climate change (Stadnyk and Déry, 2021).

5.7 Conclusions

Analysis of the spatiotemporal variability of sediment loads over >40 years and sediment source fingerprinting results in two large, regulated rivers (the Burntwood River (BR) and the Upper Nelson River (UNR)) in the subarctic region of Canada were used to identify the dominant sediment sources and their significance in sediment transport dynamics. In addition, they were used to investigate the importance of natural and human-induced environmental changes in altering sediment loads and budgets. Moreover, the role of Split Lake (which is the outlet of both the BR and the UNR watersheds) on downstream delivery of sediment was assessed by constructing a total (i.e., organic and inorganic) suspended sediment budget for the lake.

Colour- and geochemical-based fingerprints in combination with the sediment budget for Lake Winnipeg indicated that only 3% of the predominant riverine sediment source, the Red River, reaches the Lake Winnipeg outlets and, thus, that Lake Winnipeg causes a significant decoupling in sediment transport through the Nelson River watershed. Also, the sediment properties, source fingerprinting, and sediment load results showed that in the BR, riverbank material is the primary sediment source and its contribution progressively increases downstream.

In the BR and the UNR, the average annual water discharge for the recent period (1999-2019) was generally found to exceed that for the period 1979-1998. For the UNR, the average annual discharge and sediment load for the former period (1999-2019) was generally found to exceed that for the latter period, by up to 40% and 155%, respectively. The temporal increases in the UNR water discharges appeared to mainly reflect climate forcing of hydrological changes in the watershed contributing to Lake Winnipeg. Unlike the source of the increase in water discharge, the source of increased sediment load in the UNR appeared to be local as Lake Winnipeg effectively causes a dis-connectivity in sediment transport from its contributing watershed.

In contrast, the cross-watershed water diversion in the BR caused a seven-fold increase in the sediment load. Operation near the licenced limit caused a low inter-annual variability in water discharge in the BR and has muted the response to variability in local precipitation and runoff. Nevertheless, other erosion processes that are independent from water discharge (bank failures and subaerial processes) may exert a large control on inter-annual variability in the BR sediment load despite the low inter-annual variability in water discharge. However, the trend line fitted to the sediment loads over the post-regulated conditions, suggests that sediment loads have declined.

Given the Split Lake sediment budget, it is suggested that: a) this lake sequesters only ~20% of the total annual sediment loads entering from the BR and UNR; b) the UNR sediment is the primary source of sediment contributing to Split Lake and the downstream system; and c) the increasing sediment flux within the UNR watershed in response to human-induced and natural environmental changes can be transferred to the LNR.

While the methods used in this study suggest that the climate forcing of hydrological changes is a key control on the stability of the UNR discharge and sediment load, precise quantitative assessments of the human-induced controls on changes to the sources and fluxes of suspended sediment and the possible consequences of these changes on aquatic system and land–ocean fluxes of contaminants and nutrients warrants further rigorous research. The large size, remoteness, complexity, and irregular monitoring of these watersheds present significant research challenges. However, given the rapidly changing climate in the subarctic regions and increasing demand for hydroelectric power generation, this should be a research priority.

5.8 References

ASTM standard D3977-97, 2013. Standard Test Methods for Determining Sediment Concentration in Water Samples. ASTM International, West Conshohocken, PA, USA.

Barthod, L. R., Liu, K., Lobb, D. A., Owens, P. N., Martínez-Carreras, N., Koiter, A. J., ... Gaspar, L. (2015). Selecting color-based tracers and classifying sediment sources in the assessment of sediment dynamics using sediment source fingerprinting. Journal of Environmental Quality. 44(5), 1605-1616.

Bashar, K. E., Eltayeb, A., 2010. Sediment Accumulation in Roseires Reservoir. Nile Basin Water Science & Engineering Journal. 3(3), 46-55.

Best, J., 2019. Anthropogenic stresses on the world's big rivers. Nature Geoscience. 12(1), 7-21.

Bezte, C.L., Lawrence, M.J. 1999. Sediment trap, sediment core, and total suspended solids data from Split Lake, Manitoba, 1998. TEMA Data Report 99-02. A report prepared for Tataskweyak Environmental Monitoring Agency by North/South Consultants Inc., Winnipeg, Manitoba. 52 pp.

Bijeljanin, M., 2013. Application of CRISSP-2D Finite Element Modelling in Predicting Ice Formation Processes Upstream of the Jenpeg Generating Station. Unpublished MSc Thesis University of Manitoba, Winnipeg, Canada.

Blake, W.H., Boeckx, P., Stock, B.C., Smith, H.G., Bodé, S., Upadhayay, H.R., Gaspar, L., Goddard, R., Lennard, A.T., Lizaga, I., Lobb, D.A., Owens, P.N., Petticrew, E.L., Kuzyk, Z.Z., Gari, B.D., Munishi, L., Mtei, K., Nebiyu, A., Mabit, L., Navas, A., Semmens, B.X., 2018. A deconvolutional Bayesian mixing model approach for river basin sediment source apportionment. Scientific Reports. 8(1), 1-12.

Bodaly R.A., Rosenberg D.M., Gaboury M.N., Hecky R.E., Newbury R.W., Patalas K., 1984. Ecologic effects of hydroelectric development in northern Manitoba, Canada: The Churchill-Nelson River diversion. In: Sheehan P.J., Miller D.R., Butler G.C., Bourdeau P., (eds) Effects of pollutants at the ecosystem level. SCOPE 22. Wiley, New York, NY, 273–310.

Bodaly, R.A., Jansen, W.A., Majewski, A.R., Fudge, R.J.P., Strange, N.E., Derksen, A.J., Green, D.J., 2007. Post-impoundment time course of increased mercury concentrations in fish in hydroelectric reservoirs of northern Manitoba, Canada. Archives of Environmental Contamination and Toxicology. 53(3), 379-389.

Brune, G.M., 1953. Trap efficiency of reservoirs. Eos, Transactions American Geophysical Union, 34(3), 407-418.

Collins, A.L., Walling, D.E., Leeks, G.J.L., 1997. Source type ascription for fluvial suspended sediment based on a quantitative composite fingerprinting technique. Catena. 29(1), 1-27. https://doi.org/10.1016/S0341-8162(96)00064-1.

Coordinated Aquatic Monitoring Program (CAMP). 2008a. Habitat Map Poster (Cross Lake). Report prepared for Manitoba/Manitoba Hydro MOU Working Group by North/South Consultants Inc., Winnipeg, Manitoba. 1 pp.

Coordinated Aquatic Monitoring Program (CAMP). 2008b. Habitat Map Poster (Split Lake). Report prepared for Manitoba/Manitoba Hydro MOU Working Group by North/South Consultants Inc., Winnipeg, Manitoba. 1 pp.

Coordinated Aquatic Monitoring Program (CAMP). 2018. Sediment Transport Using Continuous Turbidity on the Burntwood and Nelson River System, 2016. Winnipeg, Manitoba. 60 pp.

Capelle, D.W., Kuzyk, Z.Z.A., Papakyriakou, T., Guéguen, C., Miller, L.A., Macdonald, R.W., 2020. Effect of terrestrial organic matter on ocean acidification and CO_2 flux in an Arctic shelf sea. Progress in Oceanography. 185, 102319. 102319. https://doi.org/10.1016/j.pocean.2020.102319.

Déry, S.J., Stadnyk, T.A., MacDonald, M.K., Koenig, K.A., Guay, C., 2018. Flow alteration impacts on Hudson Bay river discharge. Hydrological Processes. 32(24), 3576-3587.

Dethier, E.N., Renshaw, C.E., Magilligan, F.J., 2022. Rapid changes to global river suspended sediment flux by humans. Science. 376(6600), 1447-1452.

EC/DFO (Environment Canada and Department of Fisheries and Oceans Canada). 1992. Federal Ecological Monitoring Program Final Report. Volume 1, Environment Canada, Conservation and Protection Western and Northern Region, and Department of Fisheries and Oceans, Central and Arctic Region, 1-58. https://waves-vagues.dfo-mpo.gc.ca/Library/167974_v1.pdf

Environment Canada, Manitoba Water Stewardship. 2011. State of Lake Winnipeg: 1999 to 2007. Manitoba Stewardship Board & Environment Canada, 2011. Winnipeg, Manitoba 209 pp.

Environment and Climate Change Canada, Manitoba Agriculture and Resource Development, 2020. State of Lake Winnipeg-2nd Edition-Highlights. Manitoba Agriculture and Resource Development & Environment and Climate Change Canada, Winnipeg, Manitoba. 175 pp.

Foster, I.D.L., Walling, D.E., 1994. Using reservoir deposits to reconstruct changing sediment yields and sources in the catchment of the Old Mill Reservoir, South Devon, UK, over the past 50 years. Hydrological Sciences Journal. 39(4), 347-368.

Goharrokhi, M., 2015. Effect of Hydraulic Shear Stress on the Banks of the Red River MSc Thesis. University of Manitoba, Winnipeg, Canada.

Goharrokhi, M., Pahlavan, H., Lobb, D.A., Owens, P.N., Clark, S.P., 2019. Assessing issues associated with a time-integrated fluvial fine sediment sampler. Hydrological Processes. 33(15), 2048-2056.

Goharrokhi, M., Lobb, D.A., Owens, P.N., 2020. Evaluation of high-flow rate continuous-flow centrifugation and filtration devices for sampling and concentrating fine-grained suspended sediment. Hydrological Processes. 34(19), 3882-3893.

Goharrokhi, M., McCullough, G.K., Owens, P.N., Lobb, D.A., 2021. Sedimentation dynamics within a large shallow lake and its role in sediment transport in a continental-scale watershed. Journal of Great Lakes Research. 47(3), 725-740.

Guéguen, C., Mokhtar, M., Perroud, A., McCullough, G., Papakyriakou, T., 2016. Mixing and photoreactivity of dissolved organic matter in the Nelson/Hayes estuarine system (Hudson Bay, Canada). Journal of Marine Systems. 161, 42-48.

He, Q., Walling, D.E., 1996. Interpreting particle size effects in the adsorption of ^{137}Cs and unsupported ^{210}Pb by mineral soils and sediments. Journal of Environmental Radioactivity. 30(2), 117-137.

Hecky, R.E., McCullough, G.K., 1984. Effect of impoundment and diversion on the sediment budget and nearshore sedimentation of Southern Indian Lake. Canadian Journal of Fisheries and Aquatic Sciences. 41(4), 567-578.

Hjulström, F., 1935. Studies of the morphological activity of rivers as illustrated by the river Fyris. Bull. Geol. Inst., Uppsala. 25, 221-527.

Horowitz, A. J., Rinella, A. R., Lamothe, P., Miller, T. L., Edwards, T. K., Roche, R. L., and Rickert, D. A. 1990. Variations in suspended sediment and associated trace element concentrations in selected riverine cross sections, Environmental Science and Technology. 24, 1313-1320.

Kellerhals Engineering Services Ltd. 1988. Morphological effects of the Churchill River Diversion, Volume 2. Ecological Report Series No. 88-4. Prepared for Environment Canada, Inland Waters Directorate. 136 pp.

Kroetsch, D., Cang, C., 2007. Particle size distribution. In: Carter, M.R., Gregorich, E.G. (Eds.), Soil Sampling and Methods of Analysis. CRC Press, Boca Raton, FL, USA, pp. 713–727.

Laceby, J.P., Evrard, O., Smith, H.G., Blake, W.H., Olley, J.M., Minella, J.P. and Owens, P.N., 2017. The challenges and opportunities of addressing particle size effects in sediment source fingerprinting: a review. Earth-Science Reviews. 169, 85-103.

Lawrence, M.J., Fazakas, C.R., Zrum, L., Bezte, C.L., Bernhardt, W.J., 1999. The Split Lake Aquatic Ecosystem: A synthesis of Split Lake biological and environmental data, January 1997 - October 1998. A report prepared for the Taskaskweyak Environmental Monitoring Agency by North/South Consultants Inc: xii + 87 pp.

Lewis, S.E., Bainbridge, Z.T., Kuhnert, P.M., Sherman, B.S., Henderson, B., Dougall, C., Cooper, M., Brodie, J.E., 2013. Calculating sediment trapping efficiencies for reservoirs in tropical settings: a case study from the Burdekin Falls Dam, NE Australia. Water Resources Research. 49(2), 1017-1029.

Lu, X.X., Ran, L.S., Liu, S., Jiang, T., Zhang, S.R., Wang, J.J., 2013. Sediment loads response to climate change: A preliminary study of eight large Chinese rivers. International Journal of Sediment Research. 28(1), 1-14.

Manitoba Hydro, 2010. Churchill River Diversion Report in Support of a Request for a Final Licence 183 under the Water Power Act and Regulations. 53 p. + appendices. Winnipeg, Manitoba.

Manitoba Hydro, 2015. Regional Cumulative Effects Assessment for Hydroelectric Developments on the Churchill, Burntwood and Nelson River systems: Phase II Report. Winnipeg, Manitoba.

Martínez-Carreras, N., Udelhoven, T., Krein, A., Gallart, F., Iffly, J.F., Ziebel, J., Hoffmann, L., Pfister, L., Walling, D.E., 2010. The use of sediment colour measured by diffuse reflectance spectrometry to determine sediment sources:

application to the Attert River catchment (Luxembourg). Journal of Hydrology. 382, 49-63.

McCullough, G.K., 2015. Climate in the Lake Winnipeg watershed and the level of Lake Winnipeg. Manitoba Clean Environment Commission, Winnipeg, Manitoba. 44 pp.

McCullough, G.K., Page, S., Hesslein, R.H., Stainton, M.P., Kling, H.J., Salki, A., Barber, D.G., 2012. Hydrological forcing of a recent trophic surge in Lake Winnipeg. Journal of Great Lakes Research. 38, 95–105.

Newbury, R. W., McCullough, G. K., Hecky, R. E., 1984. The Southern Indian Lake impoundment and Churchill River diversion. Canadian Journal of Fisheries and Aquatic Sciences, 41, 548-557.

Owens, P.N., Walling, D.E., 2002. The phosphorus content of fluvial sediment in rural and industrialized river basins. Water Research. 36 (3), 685–701.

Owens, P.N., Gateuille, D.J., Petticrew, E.L., Booth, B.P., French, T.D., 2019. Sediment-associated organopollutants, metals and nutrients in the Nechako River, British Columbia: a current study with a synthesis of historical data. Canadian Water Resources Journal. 44(1), 42-64.

Perks, M. T., Warburton, J., Bracken, L., 2014. Critical assessment and validation of a time-integrating fluvial suspended sediment sampler. Hydrological Processes. 28, 4795–4807.

Phillips, J.M., Webb, B.W., Walling, D.E., Leeks, G.J.L., 1999. Estimating the suspended sediment loads of rivers in the LOIS study area using infrequent samples. Hydrological Processes. 13(7), 1035-1050.

Phillips, J.M., Russell, M.A., Walling, D.E., 2000. Time-integrated sampling of fluvial suspended sediment: a simple methodology for small catchments. Hydrological Processes. 14(14), 2589-2602.

Revel, N.M.T.K., Ranasiri, L.P.G.R., Rathnayake, R.M.C.R.K., Pathirana, K.P.P., 2015. Estimation of Sediment Trap Efficiency in Reservoirs-An Experimental Study. Engineer: Journal of the Institution of Engineers, Sri Lanka. 48(2), 43-49.

Siev, S., Yang, H., Sok, T., Uk, S., Song, L., Kodikara, D., Oeurng, C., Hul, S., Yoshimura, C., 2018. Sediment dynamics in a large shallow lake characterized by seasonal flood pulse in Southeast Asia. Science of the Total Environment. 631, 597-607.

Stadnyk, T.A., Déry, S.J., 2021. Canadian continental-scale hydrology under a changing climate: A review. Water, 13(7), 906.

Stainton, T.M., 2019. An initial investigation into the sources and transport of particulate organic matter in the Nelson River system, Manitoba. MSc Thesis. University of Manitoba, Winnipeg, Canada.

Stock, B.C., Semmens, B.X., 2016. MixSIAR GUI User Manual. Version 3.1. https://github.com/brianstock/MixSIAR. doi:10.5281/zenodo.1209993.

Stock, B.C., Jackson, A.L., Ward, E.J., Parnell, A.C., Phillips, D.L., Semmens, B.X.. 2018. Analyzing mixing systems using a new generation of Bayesian tracer mixing models. PeerJ 6:e5096 https://doi.org/10.7717/peerj.5096

Strange, N., 1990. Water quality data for sites affected by Churchill River Diversion and Lake Winnipeg regulation, northern Manitoba, 1986-89. North/South Consultants Inc., Winnipeg, Manitoba.

Syvitski, J., Ángel, J.R., Saito, Y., Overeem, I., Vörösmarty, C.J., Wang, H., Olago, D., 2022. Earth's sediment cycle during the Anthropocene. Nature Reviews Earth and Environment. 3(3), 179-196.

Theroux, J., 2017. Characterizing turbidity and identifying sediment sources in Norway House Cree Nation drinking water using sediment fingerprinting. Unpublished MSc Thesis, University of Manitoba, Winnipeg, Canada.

Vitkin, N., Penner, F. 1979. 1979 review of suspended sediment sampling in Manitoba. Water Resources Branch Report. Winnipeg, Manitoba. 79-22. 21 pp.

Walling, D.E., 1983. The sediment delivery problem. Journal of Hydrology. 65(1-3), 209-237.

Walling, D.E., 2006. Human impact on land–ocean sediment transfer by the world's rivers. Geomorphology. 79(3-4), 192-216.

Walling, D.E., 2012. The role of dams in the global sediment budget. IAHS-AISH Publication. no.356., IAHS Press, Wallingford, pp. 3-11.

Walling, D.E., 2013. The evolution of sediment source fingerprinting investigations in fluvial systems. Journal of Soils and Sediments. 13, 1658–1675. https://doi.org/10.1007/s11368-013-0767-2.

Walling, D.E., Collins, A.L., Sichingabula, H.M., Leeks, G.J.L., 2001. Integrated assessment of catchment suspended sediment budgets: a Zambian example. Land Degradation and Development. 12(5), 387-415.

Walling, D.E., Fang, D., 2003. Recent trends in the suspended sediment loads of the world's rivers. Global and Planetary Change. 39(1-2), 111-126.

Walling, D.E., Owens, P.N., Foster, I.D., Lees, J.A., 2003. Changes in the fine sediment dynamics of the Ouse and Tweed basins in the UK over the last 100–150 years. Hydrological Processes. 17(16), 3245-3269.

Ward, P.R., 1980. Sediment transport and a reservoir siltation formula for Zimbabwe-Rhodesia. Civil Engineer in South Africa. 22(1), 9-15.

CHAPTER 6: Conclusions and recommendations

6.1 Summary of conclusions

In this thesis, sediment sources and transport dynamics in Lake Winnipeg – located in north-central Manitoba, Canada – and the Nelson River – the largest river (by watershed area and freshwater discharge) contributing to Hudson Bay – were investigated using the sediment source fingerprinting and sediment budget techniques. Moreover, the collection of a representative sample of ambient suspended sediment using a well-established time-integrated sampler and two discrete samplers were investigated. These samplers were examined in terms of total collected mass, particle size composition, and geochemical and colour properties of sediment. The major contributions of this research are presented below, followed by recommendations.

6.1.1 Sedimentation dynamics within Lake Winnipeg and its role in sediment transport in the downstream river system

This was the first comprehensive study to construct a total organic and inorganic sediment budget for Lake Winnipeg (area: 23,750 km^2). This study also explored the sources of sediment, sediment transport and connectivity between the two main basins of the lake, and the pattern of sediment deposition within the lake. This was based on a large number of dated sediment cores collected in three unique regions of the lake. One of the important findings was that almost all the sediment load derived from the prairies area is sequestered in Lake Winnipeg, along with nutrients and contaminants bound to them. Another key finding was that sediment derived from bluff erosion on the northern shore of the lake is the major source for sediment being exported in the lake's outflow. Sequestration of upstream sediments in the lake, and resupply of the Nelson River sediment load by bluff erosion, together has important implications not only for Lake Winnipeg itself, but also for the Nelson River down to its estuary in Hudson Bay. An additional major contribution was found by comparing Lake Winnipeg and Laurentian Great Lakes (i.e., Lakes Erie, Huron, Michigan, Ontario, and Superior) sediment budgets as it highlighted that the dominant source of the sediment loading to all of these large lakes is shoreline erosion.

6.1.2 Sediment sources and transport in the Burntwood River and the Upper Nelson River

This work focused on the Nelson River (between Lake Winnipeg and Split Lake) and the Burntwood River (between Southern Indian Lake and Split Lake; Figure 1-1). As part of the BaySys research project, the major contributions of this study were to a) assess the sediment sources and sinks in these systems; b) examine the influence of Split Lake on downstream delivery of sediment to the Lower Nelson River; and c) distinguish the effects of climate change from changes in flow regimes in both river systems due to regulation on sediment transport. Colour- and geochemical-based fingerprints indicated that: a) riverbank material is the primary sediment source in the BR; and b) Lake

Winnipeg effectively decouples the Upper Nelson River sediment transport processes from its upstream watershed.

Temporal analysis of sediment loads suggested that the Upper Nelson River is characterized by increases in sediment loading as a result of hydrological changes, forced by climate. In the Burntwood River, cross-watershed water diversion caused a seven-fold increase in sediment discharge, and that variability in sediment discharge has been mostly driven by episodic erosion events (bank collapse, wind-event-driven reservoir bank erosion) rather than flow-induced erosion processes. Since diversion, flow regulation near the licenced limit has muted the response to variability in local precipitation and runoff.

The Split Lake sediment budget suggested that the lake has a limited impact on reducing downstream sediment transport as it intercepts only ~20% of the total incoming sediment load. This low estimated sediment trapping efficiency and the magnitude of the Burntwood River and Upper Nelson River sediment loads highlighted that the latter river is the primary sediment source contributing to the downstream Lower Nelson River system. This work demonstrated that the sediment transport dynamics in the Lower Nelson River is more sensitive to environmental changes in the Upper Nelson River, rather than changes occurring in the Burntwood River.

6.1.3 Assessing a time-integrated fluvial suspended sediment sampler

This study assessed some hydrodynamic issues about the validity of the widely used time-integrated fluvial fine sediment sampler (i.e., Phillips et al., (2000)). To fully understand the performance of the sampler, two complementary studies were conducted: a) flume experiments using an acoustic Doppler velocimeter in controlled laboratory conditions to identify the relationship between the ambient and inlet velocity, and thus to establish whether it is likely to sample isokinetically; and b) deployment of the sampler in the Red River, Manitoba, for a three-day period to assess its mass collection efficiency. This study was the first study to measure the mass collection efficiency of the sampler under field conditions by an acoustic Doppler current profiler. It was found that under the original development of the sampler (i.e., Phillips et al, 2000), the mass collection efficiencies of the sampler were overestimated and, therefore, the mass of sediment collected by sampler cannot be used as an indicator for estimating the absolute time-integrated mass flux of sediment during the period of field deployment. The acoustic Doppler velocimeter measurements illustrated that the sampler inflow efficiency was 87% and, therefore, the sampler is isokinetic. In other words, a representative sample of the fine particles enters into the sampler.

6.1.4 Evaluation of continuous-flow centrifuge and continuous-flow filtration system techniques for sampling suspended sediment

This study provided the first comparison of the performance of two high-flow rate active samplers (i.e., continuous-flow centrifugation and continuous-flow filtration systems) in terms of mass collection efficiency and particle size distribution. The ability of these devices to collect a representative sample from freshwater systems was also examined by comparing the colour and geochemical properties of the sediment collected by the samplers with those of the ambient suspended sediment. These two samplers were compared under a range of field conditions in Manitoba (i.e., Red River, Lake Winnipeg, and Nelson River) as an alternative approach to the more common low-flow rate continuous-flow centrifugation samplers. The development and rigours testing of this equipment will help advance the understanding of surface hydrology and landscape erosion processes, particularly by presenting new approaches for collecting bulk samples of suspended sediment. The results of this study confirmed that the filtration system collected a representative sample of ambient suspended sediment, in terms of particle size composition and other biogeochemical properties (e.g., geochemistry, spectral reflectance). It was also found that the continuous-flow centrifugation preferentially collected particles of a certain size range (>1 µm) and accordingly this may affect the collection of a representative sample in waters containing a high proportion of finer particles with low suspended sediment concentrations. The main outcome of this study was to provide guidance on the use of equipment to collected suspended sediment samples and highlight the advantages that filtration systems have over centrifugation as the filtration system is more portable, cost-effective, has a lower power demand, and collects a more representative sample.

6.2 Projected changes in water discharge of the Burntwood River (BR) and Upper Nelson River (UNR) and their effects on sediment transport dynamics

Looking towards the future, the impacts of climate change on the BR and UNR water discharge are expected to have the potential to alter sediment fluxes and sources. Stadnyk et al. (2021) used an ensemble of GCM-RCP model simulations to estimate that by 2070 the whole Nelson River watershed (i.e., 1,111,890 km^2) will show small statistically significant increasing trends of water discharge (0.21 km^3/yr; 6.7 m^3/s). This increase in water discharge for the Nelson River is anticipated primarily due to more increases in precipitation in the Nelson River watershed (Stadnyk and Déry, 2021).

The future trend of water discharge in the Burntwood River (i.e., Churchill River above Leaf Rapids) and the UNR have been studied by Manitoba Hydro for the 2050s relative to 1981-2010 using GCM simulations (147 simulations from 18 GCMs available at the time) (Manitoba Hydro, 2021). The simulations indicate that median annual water discharge in the BR and UNR will increase by 4% and 4.6% in 2040-2069 relative to 1981-2010, respectively. Manitoba Hydro (2021) also studied future seasonal water discharge changes and reported that spring water discharge will decrease whereas winter water discharge will increase in both the BR and the UNR. They attributed increasing discharge in winter to: a) increasing temperature and snowmelt in winter; b) reducing the duration for snow fall as a result of warmer temperatures in winter; and c) the increase rainfall events.

Considering the projection of the water discharge in the BR and UNR, the potential impacts of climate change on the sedimentary processes should be considered in light of past and current sediment transport dynamics in these regulated rivers. In the BR, as the flow diversion from the Churchill River contributes most of the flow in the BR and the river is generally operated near the licenced limit, the inter-annual variability after 1979 has been low (see top panel in Figure 5-4e). Given the licenced limit, it can be speculated that diversion discharge will not change and not be beyond Provincial operating licences. Therefore, similar to the current situation and considering the percentage of water discharge changes in the future, the sediment flux trend and source of eroded materials will not be altered considerably. In addition, the still augmented flow regime will partially mask any potential climate-forced increase in local runoff.

However, it can be speculated that prediction of a 4% increase in the average annual discharge in the UNR, mainly due to climate change in the Lake Winnipeg watershed, will result in roughly 10% increases in the average annual sediment load (Figure 5-5; Table 5-1). Since: a) the increase in water discharge in the sub-watersheds within the UNR is not considerable and its magnitude will not cause significant increases in fluvial erosion (see Chapter 5); and b) these sub-watersheds within the UNR are well-covered with vegetation and likely resistant to water erosion, then sub-watersheds in the UNR watershed will have a limited impact on changing sediment load in future and the source of "increased sediment load" in the UNR will be the main stem of the UNR. It can also be speculated that, similar to the period 1998-2019, the UNR will represent the primary sediment source for Split Lake in the future.

6.3 Impact of the study

The effects of key drivers (e.g., climate change, human activities, and natural lakes and reservoirs at different scales) on sediment fluxes and sources in large watersheds in the Canadian subarctic region are poorly documented. Environmental assessment of the Nelson River system and understanding how this river system responds to environmental changes are particularly: a) important as the Nelson River is a unique water resources for First Nation communities; and b) complicated due to the large spatial scales involved. Considering the long history of anthropogenic activities in the Nelson River system, the combined effects of such activities and climate change on the quality of water resources in the region highlight a strong research need to advance the knowledge of sediment sources and transport processes which is required for many government agencies to implement integrated watershed-scale planning and management strategies. The thesis provides new knowledge of the dominant sediment sources and their significance in sediment transport processes in the BR and UNR as well as sediment dynamics in Lake Winnipeg.

Altering the quality and quantity of sediment due to environmental changes (e.g., increased turbidity, increased pollution, and degradation of aquatic habitats) may have important effects on First Nations communities' water resources, which in turn will have social and economic implications (Arsenault et al., 2018; see Chapter 1). The thesis introduced an inexpensive sediment sampler (i.e., high-flow rate continuous-flow filtration system) which can easily be used by First Nations communities to collect representative samples of suspended particulate matter. Robust sediment sampling protocols and instrumentation along with simple methodologies (e.g., colour properties of the collected samples) can be used to quantitatively assess the physical, chemical, and biological properties of these materials and how they change in response to natural and/or anthropogenic environmental changes.

This thesis also provides further insights on the transfer of sediment from a large watershed to the Arctic Ocean. This study successfully documented the non-stationary nature of sediment flux just 300 km upstream of Hudson Bay (i.e., Split Lake outlet). This thesis can be used to provide a basis for predicting the impact of future environmental changes on the fluxes of sediment and associated elements in the last segment of the Nelson River before entering to Hudson Bay.

6.4 Recommendation and future work

In Section 4.5 a list of recommended research was presented for Lake Winnipeg. This suggested research includes: a) constructing an independent sediment budget for the Netley-Libau marsh complex as a key feature in the Lake Winnipeg watershed; b) reviewing the boundaries of major sedimentary basins as well as zones of transport and erosion within Lake Winnipeg, and collecting more sediment cores in each major sedimentary basin to calculate total annual dry mass accumulation with greater precision; c) investigating the relative contribution of the riverine sediment and the north shore eroded materials to the North Basin offshore bottom sediment using diagnostic physical, biological, and geochemical properties of sediment and source materials and the source fingerprinting approach; d) obtaining a more precise estimate of the erosion of the north shoreline bank in order to check the estimate of sediment deposition in Lake Winnipeg; and e) incorporating additional sources of sediment not addressed in this study to the sediment budget, including atmospheric deposition and internal biomass productivity.

As mentioned above, this thesis mainly focused on sediment sources and transport in Lake Winnipeg, the Upper Nelson River, and the Burntwood River. While this research provided information on the Split Lake sediment budget, areas of further research were also identified to provide an improved understanding of sedimentation dynamics within Split Lake and sediment transport in the downstream river system (i.e., the Lower Nelson River system and its estuary).

With methods and objectives similar to this study, the impacts of different scales of natural lakes and hydroelectric reservoirs on sediment sources and transport in the Lower Nelson River (including downstream of Split Lake) can be examined. The suggested methods for the Lower Nelson River study include accessing long-term suspended sediment data and using sediment source fingerprinting technique. In addition, the sediment source fingerprinting approach can be used to determine the relative contributions of the fluvial load and erosion of tidal mudflats to the sediment plume in the Nelson River estuary.

As Split Lake is a key feature in the Nelson River system, further research can also be focused on providing an improved understanding of sedimentation and sediment transport dynamics in this lake. The sediment source fingerprinting technique and the Split Lake sediment budget can be used to investigate the relative contribution of the main sediment sources to surficial bottom sediments in Split Lake and to sediments exported into the Lower Nelson River. This objective can be achieved by collecting samples of: a) contemporary suspended sediment at the Upper Nelson River and the Burntwood River inlets and the Split Lake outlet; b) the Split Lake surficial bottom sediments; and c) materials from shorelines with evidence of active erosion.

In the context of sediment transport dynamics in Split Lake, sediment resuspension of the lake – as an important internal process – can also be examined. This can be performed by using available reports and data on ~20 sediment traps deployed in the lake in six years by Manitoba Hydro (i.e., 1997, 1998, 2008, 2009, 2010, and 2012). Sediment mass balance (Sections 5.4.3 and 5.5.4), net sediment accumulation rates estimation using sediment traps deployed by Manitoba Hydro (discussed in Section 5.5.4), and gross sedimentation (i.e., the trap settling flux) can be used to quantify spatiotemporal variability in the resuspension of Split Lake bottom sediment. The main factors (e.g., incoming sediment load and wind) which may exert a large control on this process should also be investigated.

The influence of human-induced environmental change in the Burntwood River (since 1976) and climate change in the Upper Nelson River (since 1998) on Split Lake sedimentation dynamics can be investigated by: a) assessment of the lake surficial sediment properties (i.e., particle size distribution, organic matter/carbon content, and colour); and b) sediment accumulation rates in key locations. One of the key locations should be at the west end of the lake near the mouth of the Burntwood River as this location was chosen for conducting a 20 by 20 m sand-seeding plot in 1997 by Manitoba Hydro. Collecting a sediment core from this reference site will help to: a) improve the Split Lake sediment budget by quantifying the BR mouth sedimentation (this was also suggested as future research in Section 5.4.3); b) determine a time series of sediment sources and loads from the Burntwood River system; and c) examine the effects of cross-watershed water diversion and post-diversion flow variability on the Split Lake sedimentation processes.

Sediment source fingerprinting by different tracers (e.g., colour and geochemical fingerprints) can be applied at the Nelson River estuary to establish the relative importance of the upstream fluvial load and local mudflat sediments in the Nelson River estuary sediment plume zone. Sediment fingerprinting techniques can be applied to samples of: a) suspended sediment delivered from the Nelson River into Hudson Bay; b) sediment from tide flats in the Nelson estuary; and c) suspended sediment from the estuary to determine the relative contribution of each source to the suspended sediment load in the estuary. Coupling of the sediment source fingerprinting results with the available sediment flux data at the Nelson River mouth and its estuary will afford a means of estimating the mass of sediment derived from the riverine and mudflat sediments in the sediment plume zone.

Regarding time-integrated fluvial suspended sediment sampler assessment, future work should examine the effect of higher flow velocities (>60 cm/s) on the performance of this sampler. It is well documented that a large proportion of the suspended sediment flux occurs during high run-off events and flood flows. This will, therefore, be a valuable study as the magnitude of water discharge in natural rivers under flood conditions may be greater than the range of flume flow velocities that were conducted in the laboratory in the paper of Phillips et al. (2000) (i.e., 15.4 to 60 cm/s).

Both continuous-flow centrifugation and filtration devices were tested under limited inflow rates. The filtration device, for example, was evaluated under 26.5 and 53 L/min. However, the inflow rate can reach up to ~150 L/min, with the manufacturer's recommended inflow rate of ~130 L/min. Performance of these devices under different inflow rates and assessment of the optimum inflow rate may, therefore, be useful areas of further research. In addition, the ability of these devices to provide a truly representative sample of suspended sediment was evaluated by comparing three geochemical properties and three colour coefficients of sediment collected by these systems with those of ambient suspended sediments. Expanding the assessment of the suite of diagnostic properties for sediment collected by both devices is also suggested as future work.

It is highly recommended that for any future work, researchers will inform First Nations communities on the possible work on their land prior to conducting any study and will take benefit of their valuable knowledge. This could be achieved by using different methods including sending emails and scheduling information sessions. Consistent with the recommendations and suggestions provided by First Nations, the research should be conducted and if it is possible, researchers should develop an engagement process to provide the First Nations with meaningful opportunities to participate in the study. The impacts of the research findings on First Nations communities should also be communicated to these Nations by the end of project using a number of meetings. The findings of this thesis were shared with one of the members of the Councillor of the Tataskweyak Cree Nation, Split Lake, MB using phone calls, emails, and a Zoom meeting.

Overall, this thesis demonstrated that Lake Winnipeg essentially decouples the Upper Nelson River from its upstream watersheds. It is also determined that climate change is the dominant controlling factor in changing sediment loads in the Upper Nelson River, whereas cross-watershed water diversion is the governing factor causing changes in the sediment transport dynamics in the Burntwood River. This thesis also provides several recommendations for further research related to each chapter.

6.5 References

Arsenault, R., Diver, S., McGregor, D., Witham, A. and Bourassa, C., 2018. Shifting the framework of Canadian water governance through Indigenous research methods: Acknowledging the past with an eye on the future. Water, 10,49, w10010049.

Climate Change Report: Insight into the Strategies Making Manitoba Hydro an Industry Leader in Responding to Climate Change; Winnipeg, MB, Canada., 2021, https://www.hydro.mb.ca/environment/pdf/climate_change_report_2020.pdf.

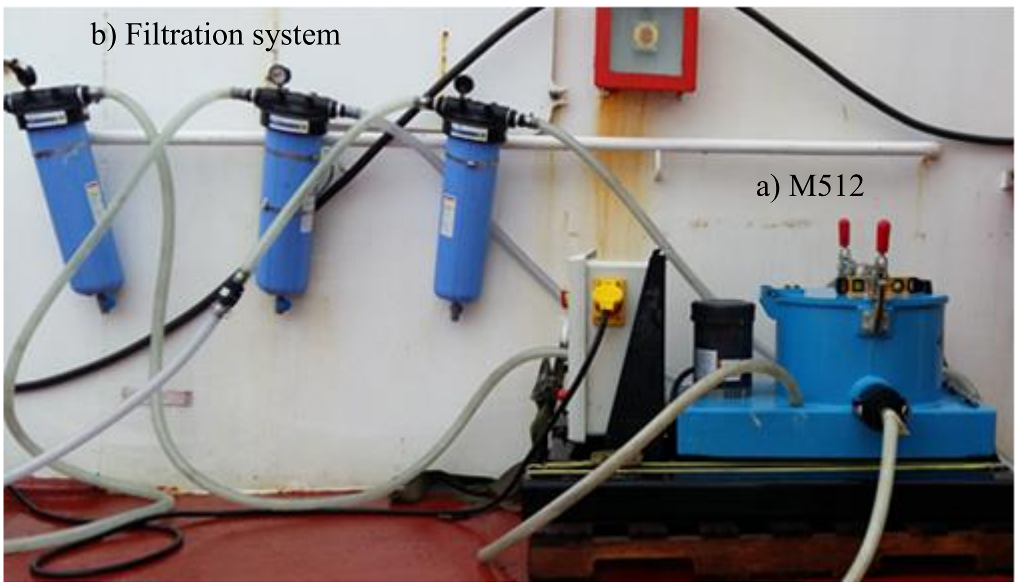
Phillips, J.M., Russell, M.A., Walling, D.E., 2000. Time-integrated sampling of fluvial suspended sediment: a simple methodology for small catchments. Hydrological Processes, 14, 2589-2602.

Stadnyk, T.A., Tefs, A., Broesky, M., Déry, S.J., Myers, P.G., Ridenour, N.A., Koenig, K., Vonderbank, L. and Gustafsson, D., 2021. Changing freshwater contributions to the Arctic: A 90-year trend analysis (1981–2070). Elementa Science of the Anthropocene, 9(1), DOI: https://doi.org/10.1525/elementa.2020.00098.

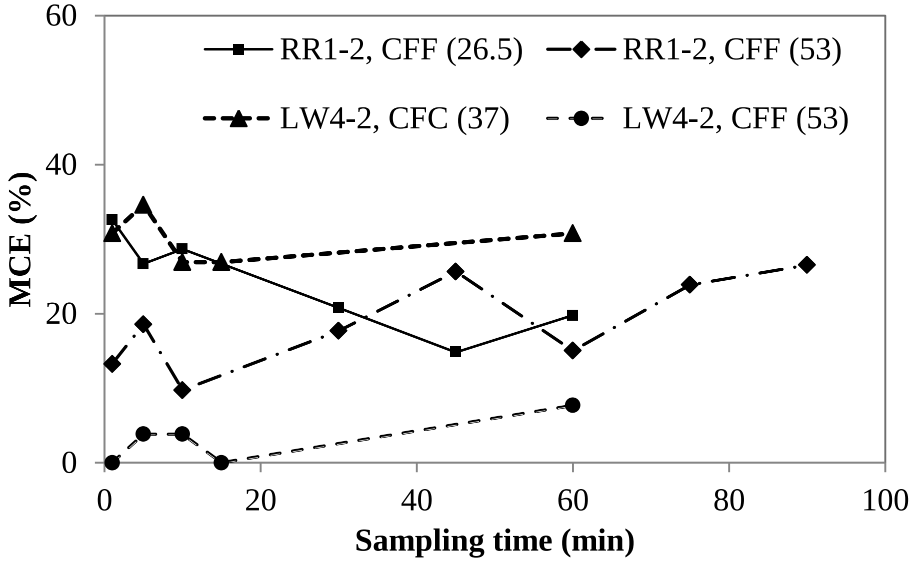
Stadnyk, T.A. and Déry, S.J., 2021. Canadian continental-scale hydrology under a changing climate: A review. Water, 13, p.906.

Appendix A: Supplementary materials for Chapter 3

A1: Photograph of the PENTEK filtration (left) M-512 C centrifuge (right) samplers. In this situation, each device is simultaneously sampling ambient water–sediment mixture from a ship (Hudson Bay, Canada, from the Canadian Coast Guard Ship *(CCGS) Des Groseilliers*)



A2: Instantaneous mass collection efficiency (MCE) of methods 2 and 3 at RR1-2 (on July 24th, 2017) and of methods 1 and 3 at LW4-2 (August 7th, 2017) sites



A3: Ambient TSS concentration, processed volume of ambient water–sediment mixture by the M512 and filtration system at different sites, MCE (%), and actual air-dried mass of collected suspended sediment

Site ID	Ambient TSS (g/m^3)	Method employed†	MCE (%)	Processed volume (l)	Air-dried mass (g) ††
RR1-1	250	CFC (37)	78	2,000	380
RR1-2	85	CFF (26.5)	24	2,385	50
		CFF (53)	20	4,770	55
LW4-1	35	CFC (37)	20	6,000	40
		CFF (26.5)	13	4,700	20
		CFF (53)	5	4,200	6
LW4-2	25	CFC (37)	30	2,220	18
		CFF (53)	5	3,180	4
		CFF 4*(53)	11	3,180	11

†: See Table 1 for description of the sampling methods

††: Air-dried mass may be biased high owing to the presence of unevaporated water within the collected sediment

A4: Comparison of the filtration system's mass collection efficiency (MCE) between different methods

Site	Method employed (Table 1)	Ambient TSS (g/m^3)	MCE (%)					Average MCE (%)	Air-dried# mass (g)
			t = 1 min	t = 5 min	t = 10 min	t = 15 min	t = 60 min		
LW4-2	CFF (53)	25	0	6	7	0	11	5	4
	CFF*4 (53)		10	14	5	18	10	11	11
LW5	CFF (53)	18	17	11	39	22	N.D.*	22	3.2
	CFF*2 (53)		22	39	39	44	N.D.	36	5.2
LW6	CFF (53)		11	11	0	22	N.D.	11	1.7
	CFF*2 (53)		6	28	28	6	N.D.	17	2.5
NR1	CFF (26.5)	22	0	22	33	N.D.	27	21	N.R.**
	CFF*2 (53)		17	22	17	N.D.	22	20	N.R.

*: N.D. = not determined

**:N.R. = Not reliable: at this site the submersible pump fell to the bottom of the river and the collected mass may not be accurate.

#: Air-dried mass may be biased high considering the presence of unevaporated water in the processed samples.

A5: Comparison of the required times to collect a 10 g sample of suspended sediment under different ambient TSS concentrations between a hypothetical low-flow CFC (6 L/min), M512 (37 L/min), and filtration system with two filters in series (53 L/min) with realistic MCEs (see text for details)

Ambient TSS (g/m^3)	Required times (in min) to collect a 10 g of ambient suspended sediment sample			
	CFC (6) MCE = 65%	CFC (6) MCE = 90%	CFC (37) MCE = 15%	CFF*2 (53) MCE = 10%
20	128	93	90	96
35	73	53	51	55
60	43	31	30	32
100	26	19	18	19
150	17	12	12	13
300	9	6	6	6

A6: The Kolmogorov-Smirnov statistical test (K-S test) results for the particle size distribution (PSD) for 4 sediment samples collected with filtration systems in series at LW4-2 (on August 7th, 2017).

Case	K-S test value*
Filter bag 1 vs. filter bag 2	0.052
Filter bag 1 vs. filter bag 3	**0.206**
Filter bag 1 vs. filter bag 4	**0.273**
Filter bag 2 vs. filter bag 3	0.170
Filter bag 2 vs. filter bag 4	**0.231**
Filter bag 3 vs. filter bag 4	0.082

*: The PSD difference between two distributions is significant at 95 % ($\alpha = 0.5$) if the test value is less than the critical value of 0.17; bold values indicates significant difference.

Appendix B: Supplementary materials for Chapter 4

B1: Core length, lake water depth, section interval, and UTM coordinate for the sites in the South Basin, Narrows, and North Basin

Region	Site #	Sampling site ID	Core length (cm)	Lake water depth (m)	Section interval (cm)	Site UTM coordinate (14 U)	
						Easting (m)	Northing (m)
South Basin	1	2	5	5.8	5	653809	5589083
	2	3B	5	5.8	5	664056	5592129
	3	3C	5	5.5	5	668846	5592737
	4	W12	30	8.0	1	653519	5598300
	5	60C	5	8.5	5	665252	5603352
	6	60B	5	8.5	5	644928	5611096
	7	59	25	8.8	1	656092	5618350
	8	36S	23	9.4	1	646132	5622212
	9	W10	24	10.7	1	657050	5635093
	10	W11	5	8.2	5	679541	5626349
	11	9	15	9.1	1	681148	5639584
	12	W9	25	11.6	5	669467	5655050
	13	12B	10	10.7	5	666418	5667294
	14	46S	19	8.8	1	680375	5668722
	15	44S	24	6.4	1	658909	5676394
Narrows	16	13B	10	15.0	5	668639	5690185
	17	W13	19	9.1	1	664472	5701557
	18	54	21	12.2	1	634436	5762926
	19	53	16	11.0	1	628113	5776147
	20	W14	17	10.7	1	623491	5791744
	21	68	15	12.19	1	608432	5773787
North Basin	22	64	27	15.5	1	593624	5788740
	23	W7	10	15.2	5	613014	5810566
	24	W5	5	14	5	596887	5850590
	25	W6	28	16.8	1	585694	5833211
	26	W4	32	16.8	1	550475	5857583
	27	19	10	17.1	5	591292	5895402
	28	W3	32	15.2	1	577725	5891924
	29	39	36	17.1	1	553628	5880924
	30	23S	38	16.5	1	525648	5895010
	31	W2	30	14.6	1	498372	5901494
	32	31	5	11.3	5	504950	5938999
	33	28	8.5	8.8	5	484433	5895957
	34	W1 (T1-50)	45	17.1	1	540821	5914151

	35	21	23	16.5	1	572561	5918781
	36	26S	38	15.8	1	515513	5918162
	37	T1-43	30	16.8	1	538475	5925280
	38	T1-31	30	16.8	1	536226	5936410
	39	34S (T1-20)	25	15.2	1	534169	5946107
	40	T1-12	20	17.4	1	533640	5955305
	41	T1-6	16	12.2	1	533231	5960910
	42	23B (T2-30)	31	14.9	1	548341	5929565
	43	T2-16.3	20	16.1	1	552541	5941097
	44	T2-15	23	16.1	1	553385	5943305
	45	T2-5	5	11	5	556077	5953717
	46	33 (T1-2)	20	9.4	1	532905	5965112
	47	T1-0.3	5	3.8	5	559641	5956766
	48	T1-0	5	-	5	559823	5957002
	49	T2-0.3	8	4.6	5	532793	5966926
	50	T2-0	5	-	5	532791	5967215

B2: Porosity and Organic matter (OM) content determination

- **Porosity (φ_j; non-dimensional) calculation**

Considering the water content for each layer, porosity (φ_j; non-dimensional) was calculated using the following equation:

$$\varphi_j = \frac{W_{wj} \times \rho_s}{W_{wj} \times \rho_s + \left(1 - W_{wj}\right)\rho_w} \qquad (1)$$

where W_{wj} is the water content (i.e., non-dimensional; $\frac{\text{weight of water}}{\text{total weight}}$) for each sediment layer (j) , ρ_s is the density of dry sediment (i.e., assumed to be 2.5 g/cm^3), and ρ_w is the density of the pore water (i.e., 1 g/cm^3) (Baskaran et al., 2015).

- **Organic matter (OM) content (non-dimensional) determination**

The OMC was determined by combustion of ~ 3 g subsample at 550°C for 16 hr after drying the subsample at 105°C for 24 hr using the following equation:

$$\mathrm{OMC} = \frac{W_1 - W_2}{W_1} \qquad (2)$$

where W_1 and W_2 are the weight of subsample (g) after drying at 105°C and after combustion at 550°C, respectively (Siev et al., 2018).

- **Cumulative mass depth (g/cm^2) calculation for each sediment core**

The cumulative mass depth for each core sample was calculated using:

$$M = \sum_{j=1}^{n}\left(\left(1 - \varphi_j\right) \times \delta_j \times \rho_s\right) \qquad (3)$$

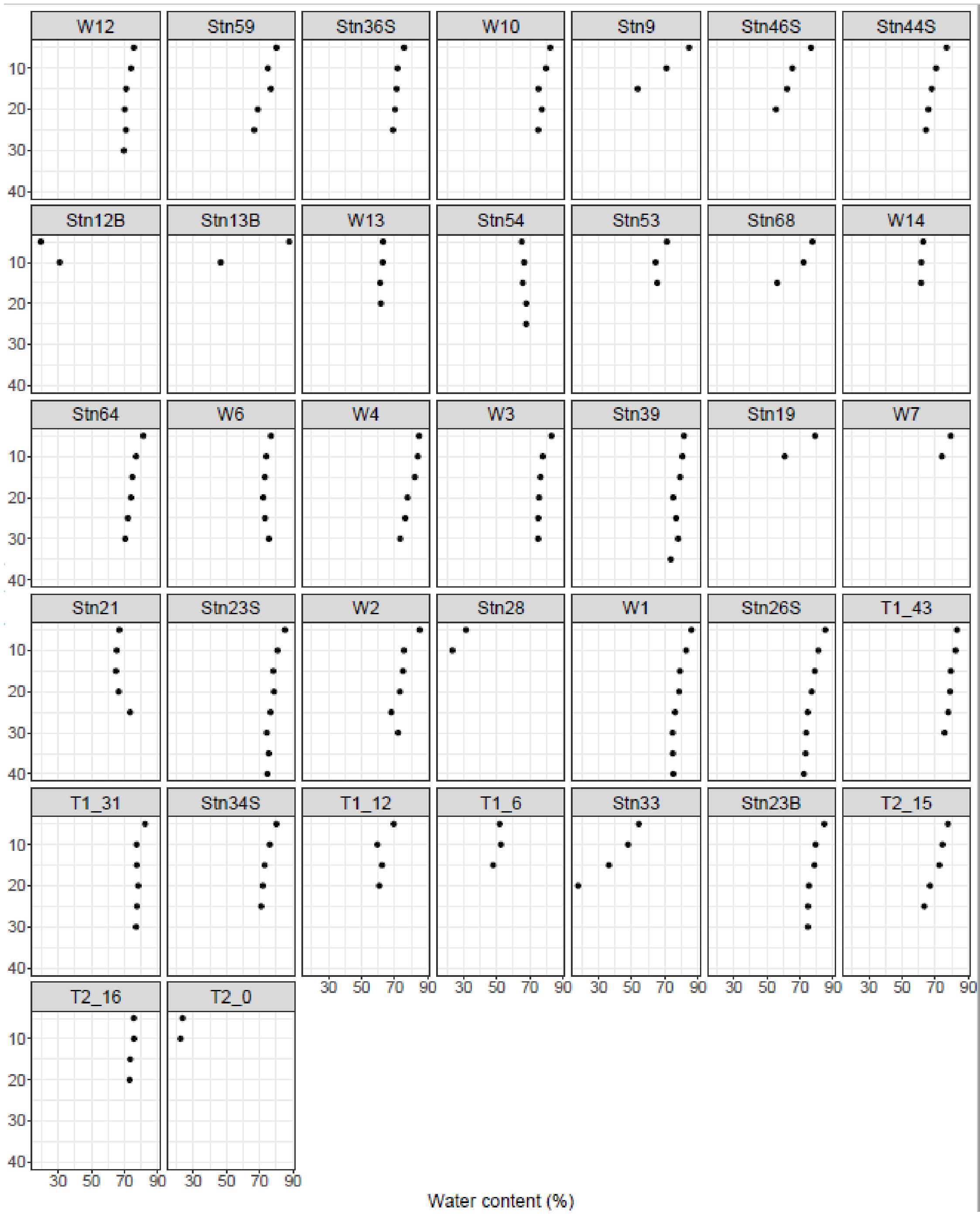
where M is the cumulative mass depth (g/cm^2), φ_j and δ_j are the porosity and the thickness (cm) of each sediment section, respectively (Baskaran et al., 2015).

References

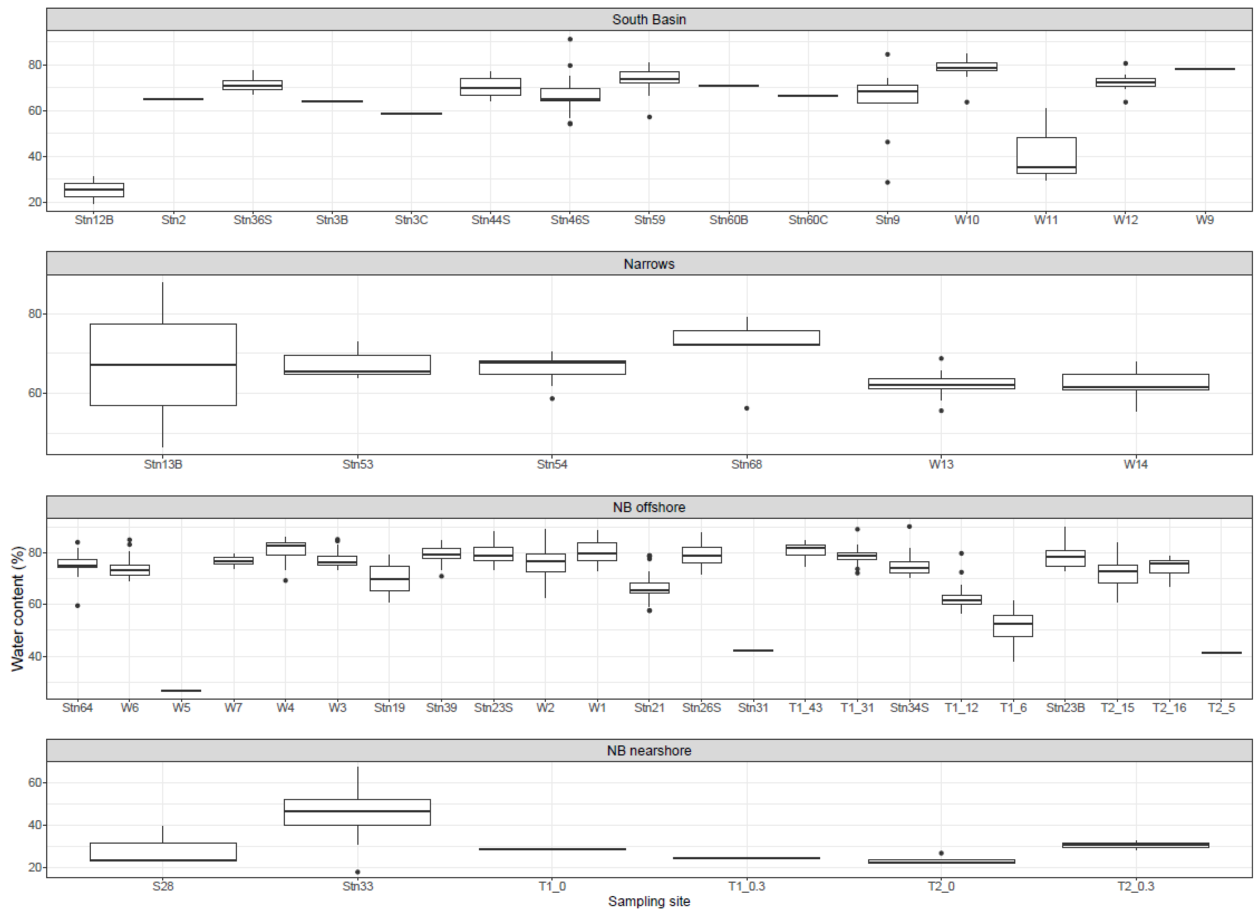
Baskaran, M., Miller, C.J., Kumar, A., Andersen, E., Hui, J., Selegean, J.P., Creech, C.T., Barkach, J., 2015. Sediment accumulation rates and sediment dynamics using five different methods in a well-constrained impoundment: Case study from Union Lake, Michigan. Journal of Great Lakes Research. 41(2), 607-617.

Siev, S., Yang, H., Sok, T., Uk, S., Song, L., Kodikara, D., Oeurng, C., Hul, S., Yoshimura, C., 2018. Sediment dynamics in a large shallow lake characterized by seasonal flood pulse in Southeast Asia. Science of the Total Environment. 631, 597-607.

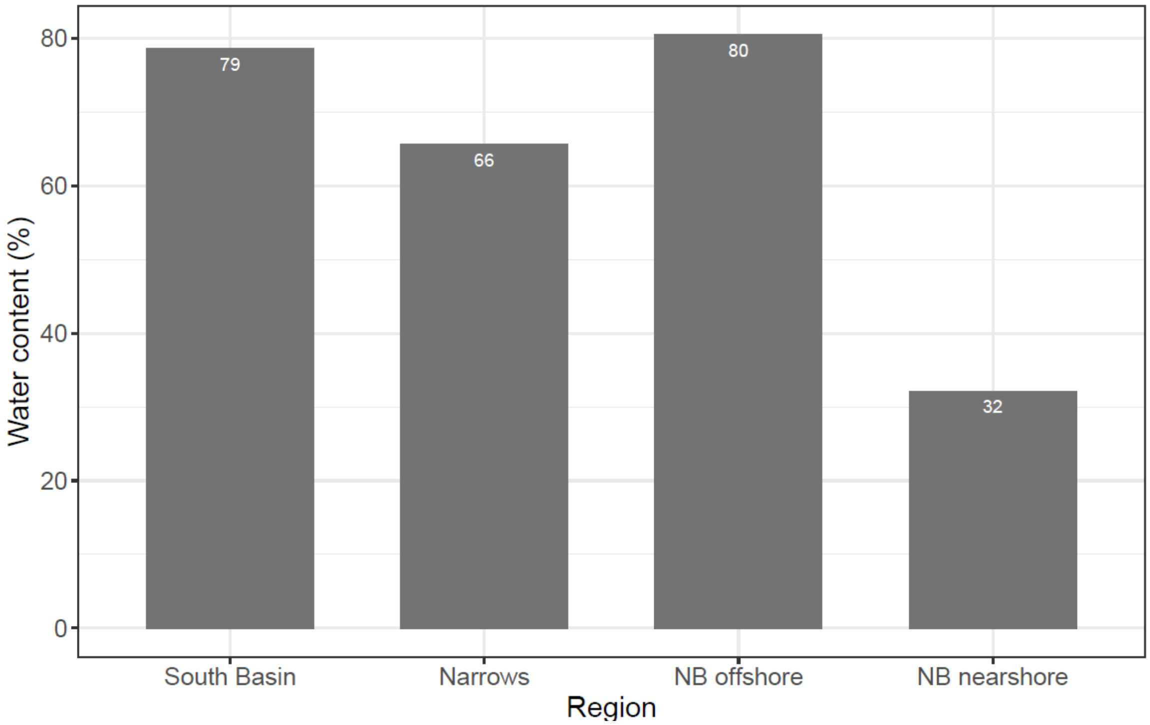
B3: Profiles of water content versus depth for select sediment cores (longer than 5 cm) in Lake Winnipeg (in some cores, water content in 1 cm increments were averaged to present 5 cm water content values)



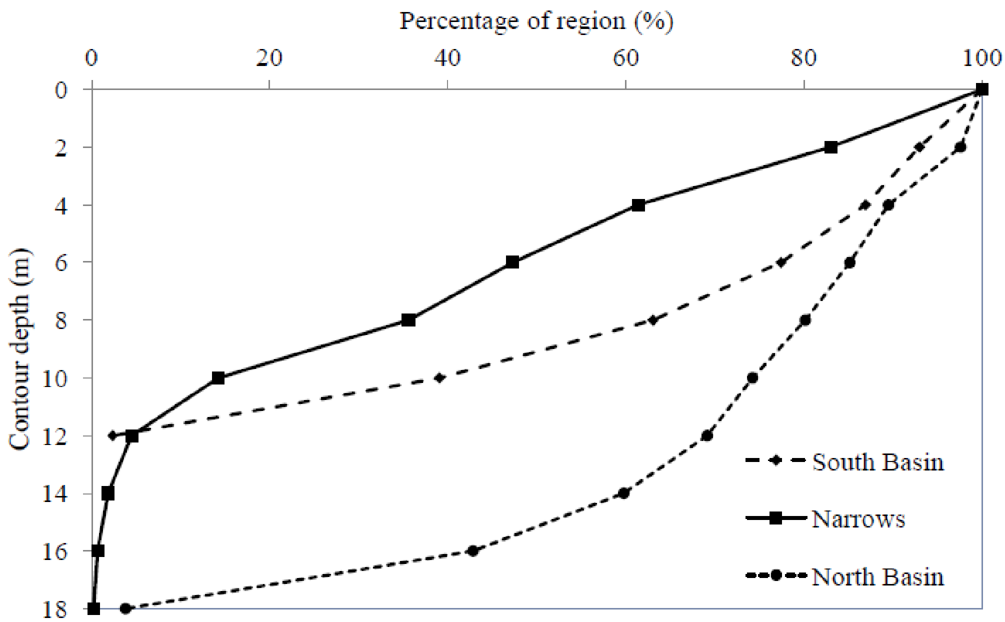
B4: Box and Whisker plot showing the spatial (vertical and horizontal) variability of the average values of water content of sediment cores for individual sampling sites within the four regions of Lake Winnipeg



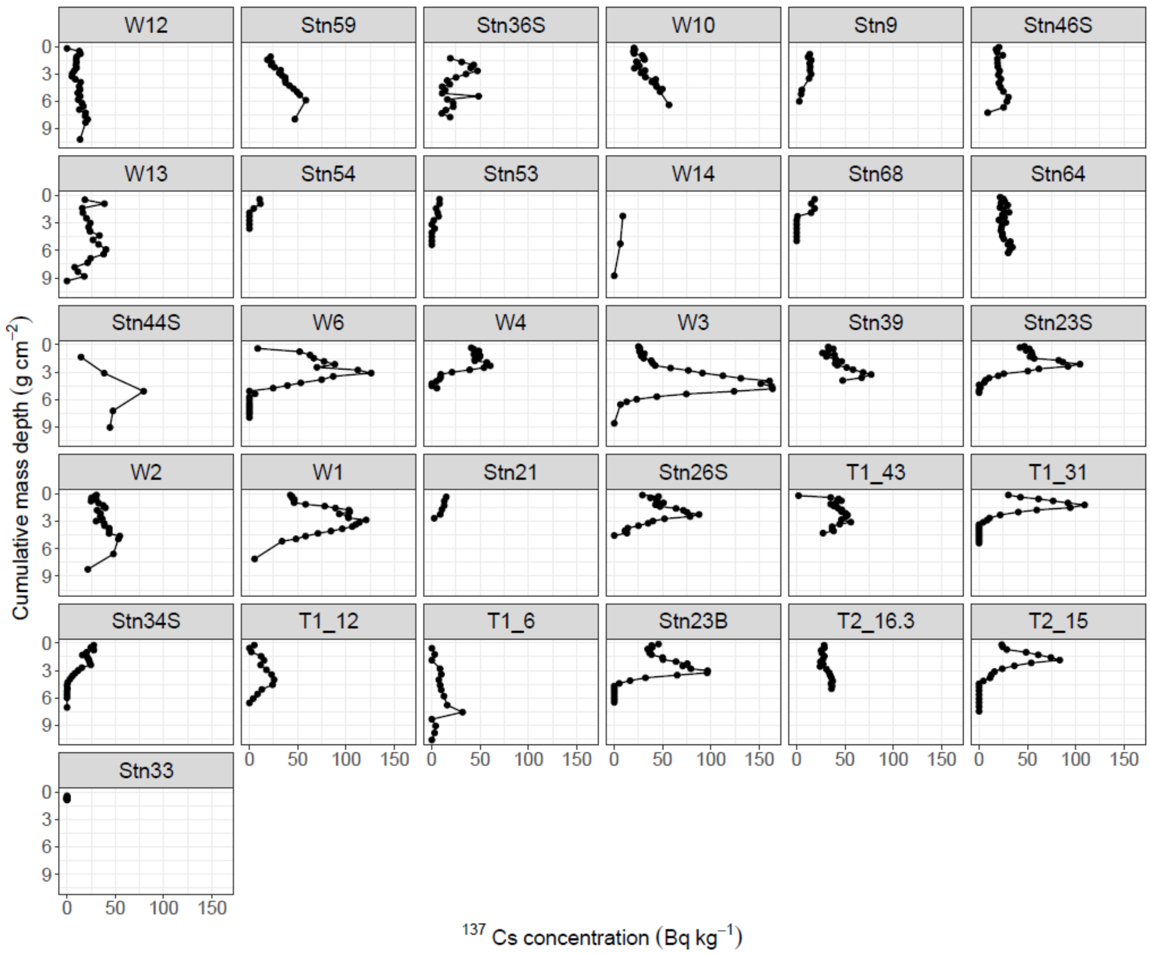
B5: Longitudinal variation in average water content for the top 5 cm of sediment cores collected from four regions in Lake Winnipeg

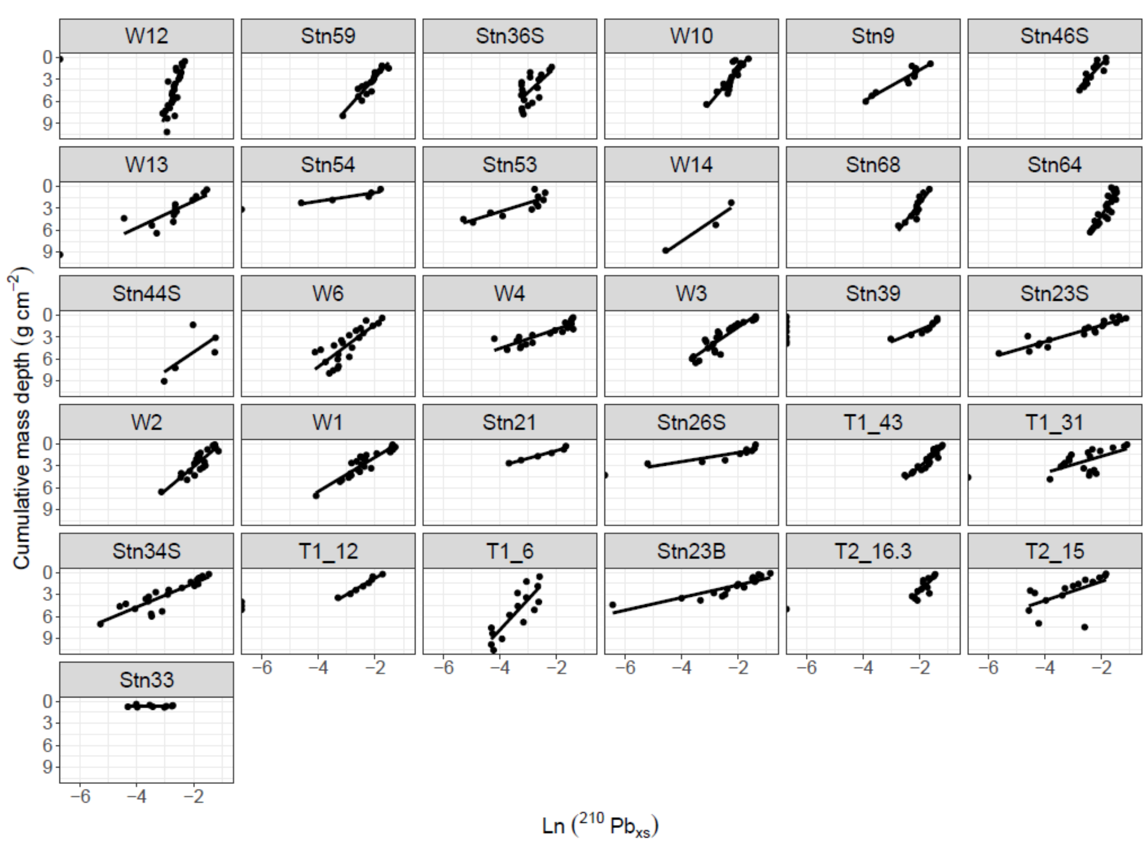


B6: Contours of Lake Winnipeg at 2 meter depth intervals versus percentage surface area of the main areas of the lake (i.e., South Basin, Narrows, and North Basin; adapted from Brunskill et al. (1980))

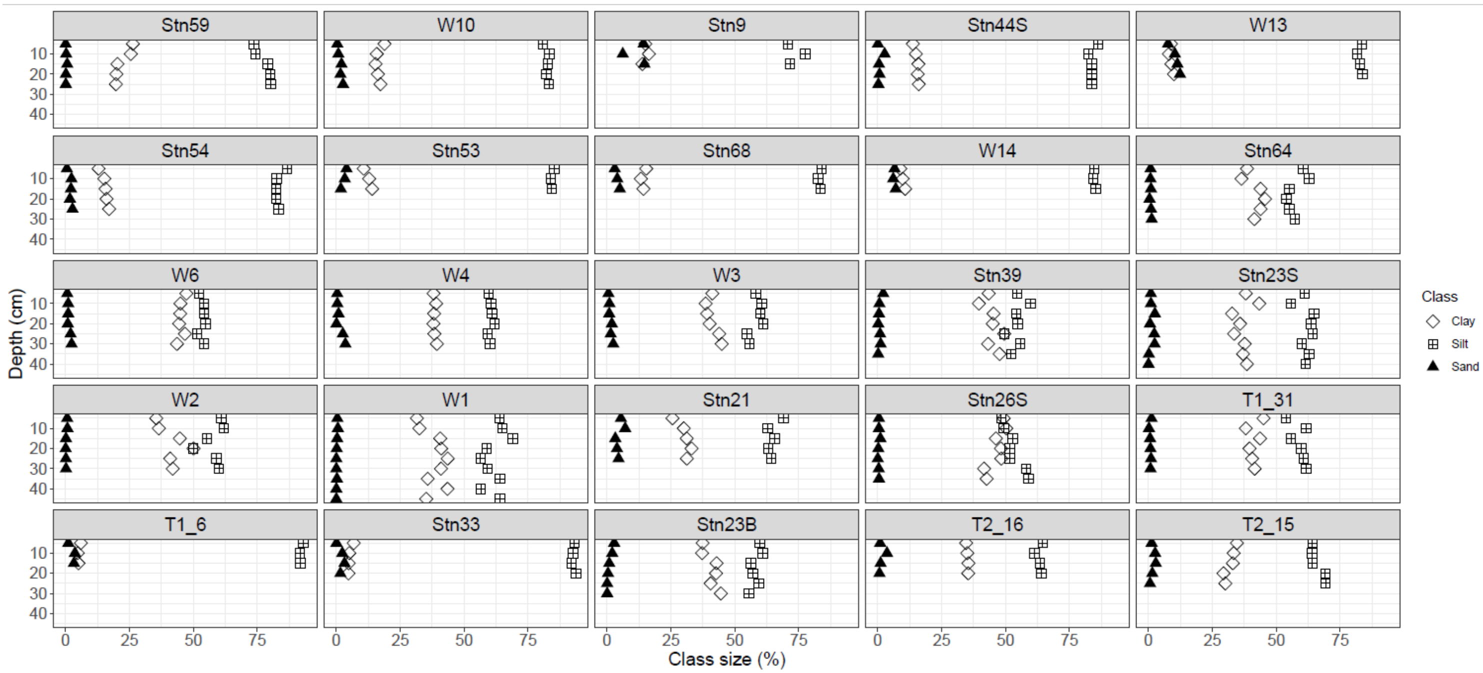


B7: Profiles of ^{137}Cs and $^{210}Pb_{xs}$ activity concentrations versus cumulative mass depth for select sediment cores in Lake Winnipeg (all the slices of sediment cores at Stn 44S and W14 were not analysed)

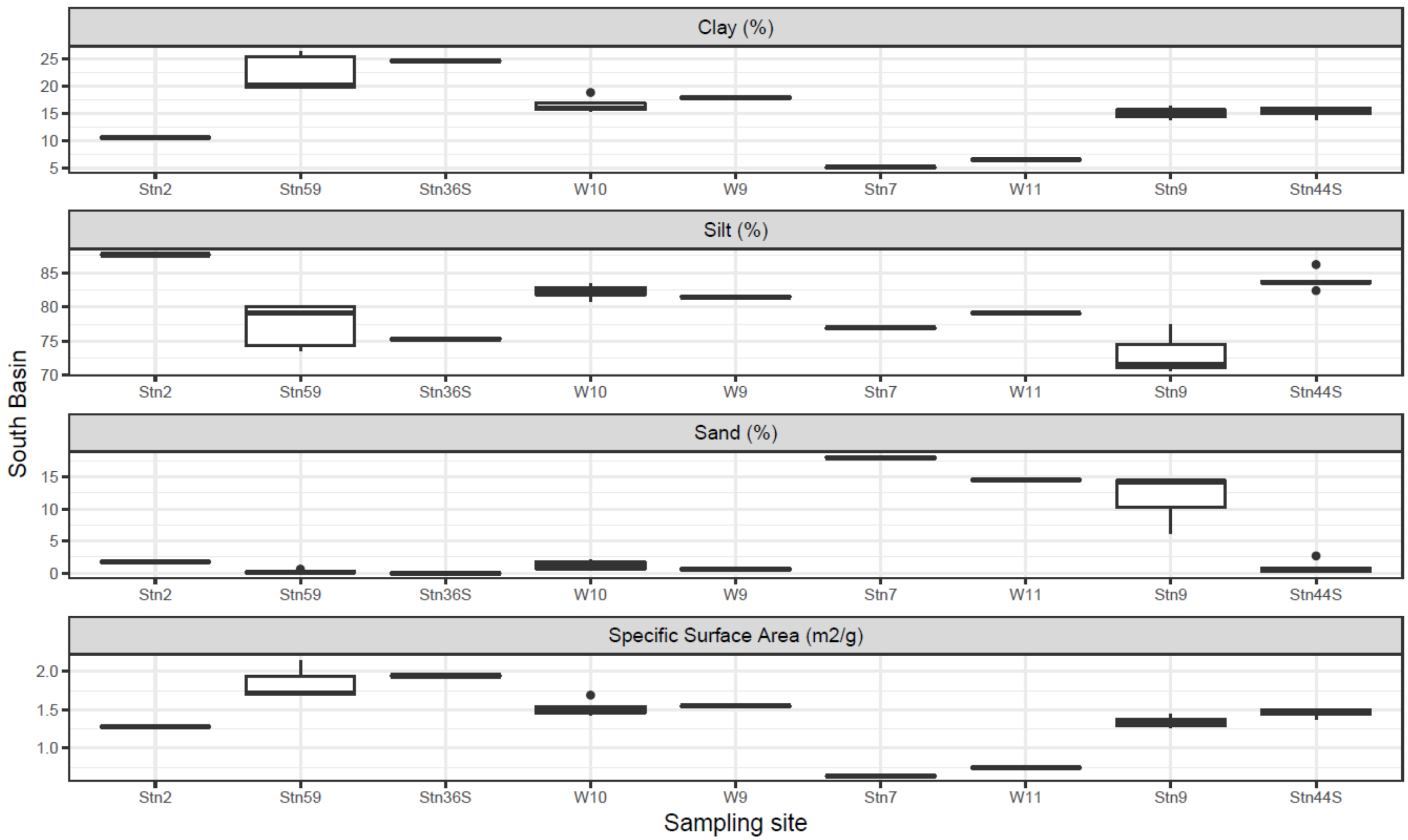


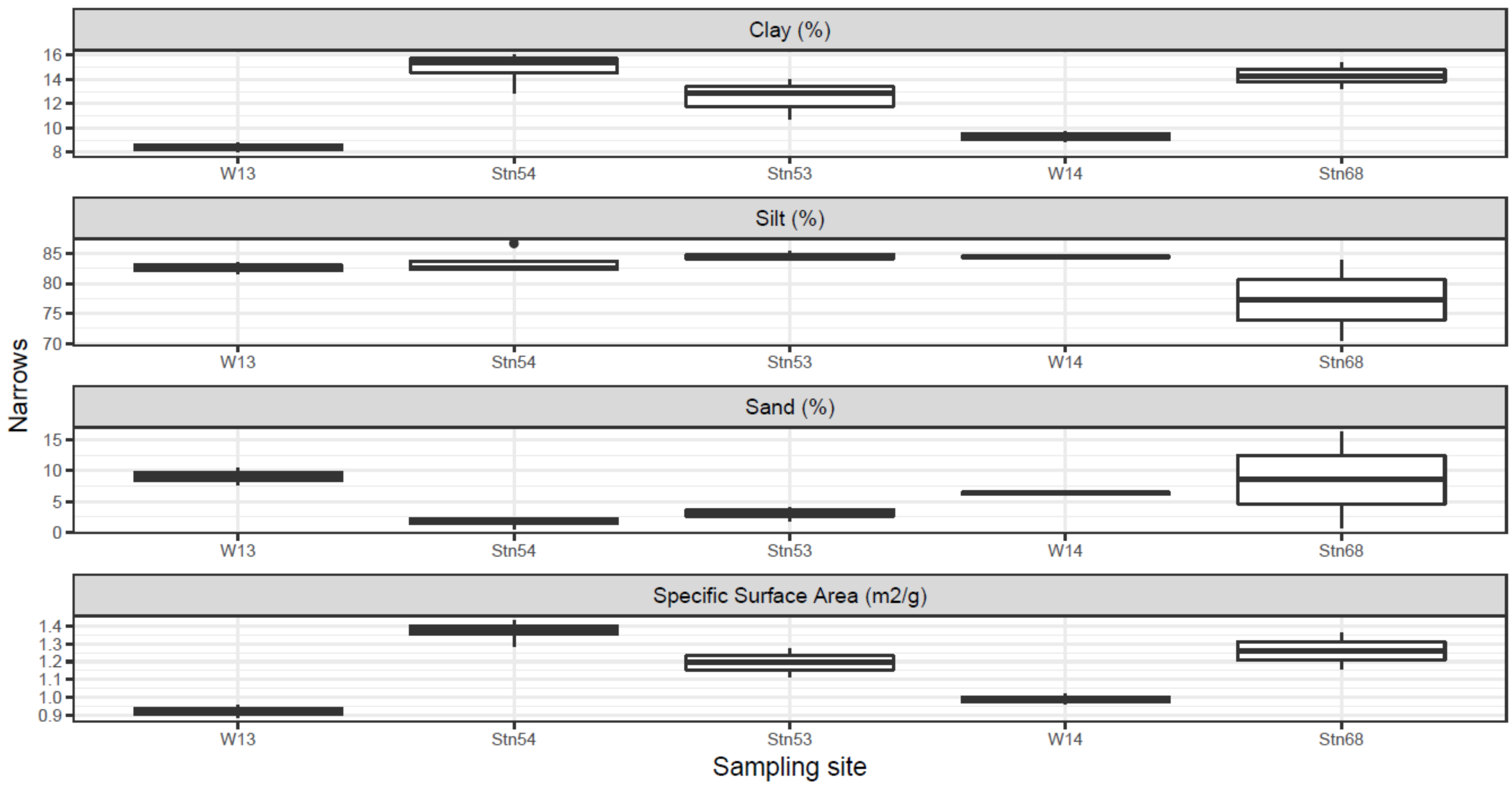

W12
Stn59
Stn36S
W10
Stn9
Stn46S
W13
Stn54
Stn53
W14
Stn68
Stn64
Stn44S
W6
W4
W3
Stn39
Stn23S
W2
W1
Stn21
Stn26S
T1_43
T1_31
Stn34S
T1_12
T1_6
Stn23B
T2_16.3
T2_15
Stn33
Cumulative mass depth (g cm−2)
Ln (210 Pbxs)

B8: Vertical variation of the primary particle size composition of select sediment cores (i.e., longer than 10 cm) in Lake Winnipeg. Clay is < 2 μm, silt is 2-63 μm, and sand is 63-2000 μm (in some cores, primary particle size composition in 1 cm increments were averaged to present 5 cm particle size composition values)

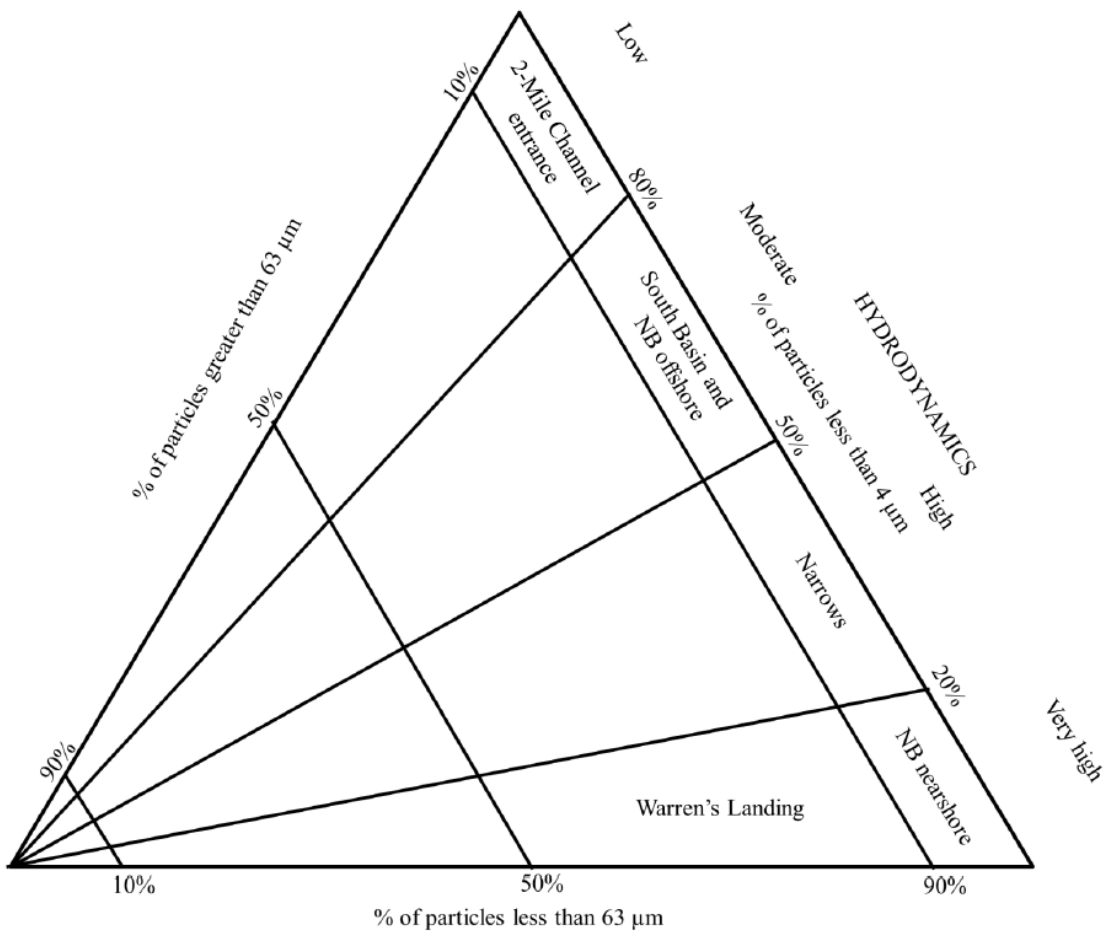


B9: Box and Whisker plot showing the variability of the average values of select measures of particle size composition of representative sediment cores from the South Basin and Narrows.




Clay (%)
Silt (%)
Sand (%)
Specific Surface Area (m2/g)
Narrows
W13
Stn54
Stn53
W14
Stn68
Sampling site

B10: Hydrodynamic condition of each region during sedimentation using particle size composition of the sediment cores and the Pejrup diagram.



B11: Supplementary Material related to the Lake Winnipeg sediment budget

Fluvial sediment loads

Locations (Figure B11-1)

- Saskatchewan at Grand Rapids
- Red River at Selkirk
- Winnipeg River at Powerview Dam.

Data sources

- Discharge: Water Survey of Canada (https://www.canada.ca/en/environment-climate-change/services/water-overview/quantity/monitoring/survey.html)
- Total suspended solids concentration (TSS, determined by filtration using GF/C 1.2 μm filters) supplied on request by the Manitoba Department of Agriculture and Resource Development.

Methods

- Daily TSS were estimated by linear interpolation between observations at approx. monthly intervals, and typically more frequently during flood periods. (On average, n = 15, 21 and 17 observations per year from 2004–2017 for the Saskatchewan, Red and Winnipeg Rivers, respectively.)
- Estimated daily TSS were multiplied by daily average discharge and the product (load) reported in terragrams per year.

Results

- Annual average discharge and total sediment loads are shown for each tributary in Figure B11-2.
- Average annual loads for the three tributaries, from 2004–2017, were 0.2, 2.8 and 0.3 Tg/yr for the Saskatchewan, Red and Winnipeg Rivers, respectively.
- Significant losses occur in the Netley-Libau marsh complex at the mouth of the Red River. These are discussed below and in the text of this paper.

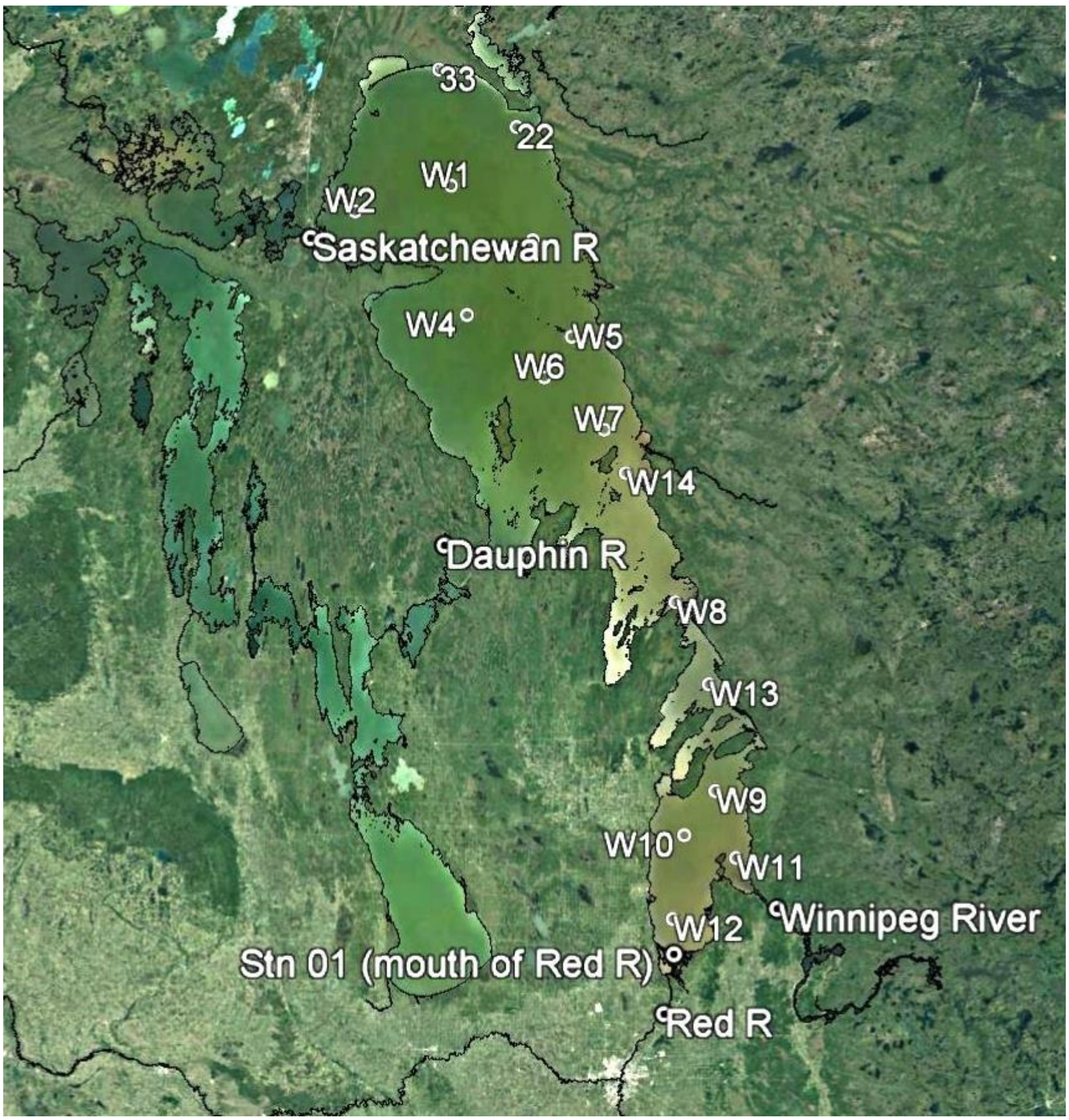


Figure B11-1. Locations of discharge and water quality stations near the mouths of major tributaries, and selected water quality stations in Lake Winnipeg. Station i.d.s are those assigned by the Manitoba Department of Agriculture and Resource Development.

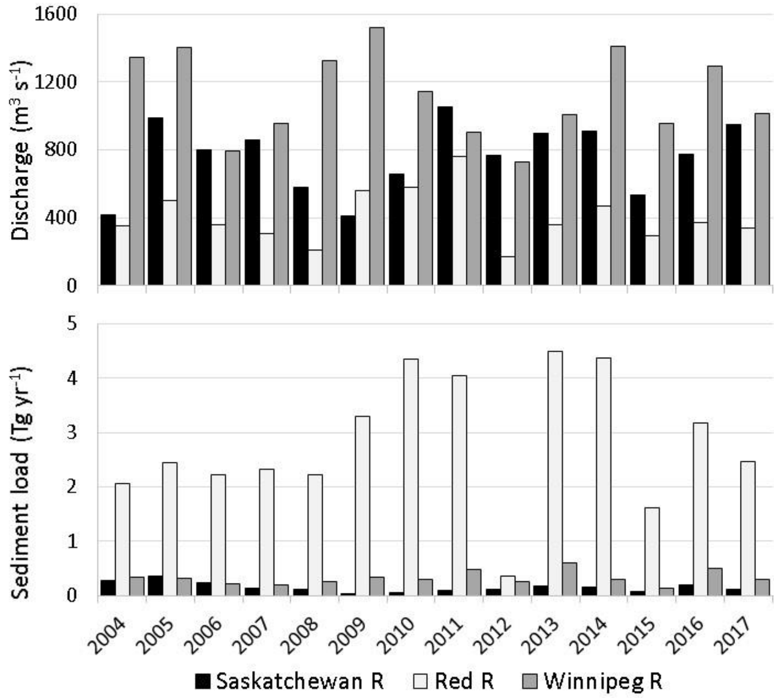


Figure B11-2. Annual average discharge (above) and total sediment loads (below) in the Saskatchewan River at Grand Rapids (4 km upstream of Lake Winnipeg), the Red River at Selkirk (30 km upstream) and the Winnipeg River at Powerview Dam (12 km upstream).

Sediment losses in the Netley-Libau marsh complex (NLM)

Locations

- TSS measured in the Red River at Selkirk and at the mouth (Figure B11-3).

Data sources

- TSS was supplied on request by the Manitoba Department of Agriculture and Resource Development.

Methods

- TSS was sampled at Selkirk at ≥ monthly frequency, but only seasonally through the open water season at the river mouth (in the course of spring, summer and fall whole lake surveys).

- Of 27 paired observations where the two stations were sampled ≤ 9 days apart, the ratio between TSS at the river mouth and at Selkirk ranges from 0.08 to 5.1 (Table B11-1). The ratios tend to be lower at higher than median discharge (Figure B11-3) indicating that a greater proportion of the load is sequestered in the NLM during high discharge events than during periods of low flow. At less than median discharge, the ratios are centred at unity (mode in the range 0.8 to 1.2) indicating that only a small proportion of the sediment load is lost in the marshes during periods of low flow. The few high positive ratios (> 2) may be due to turbid lake water flowing back up into the river during or soon after wind-driven setup of the South Basin water level.

Results

- TSS at Selkirk is variable and skewed towards high values (averages greatly exceed medians, Table 1) and is correlated with discharge (Figure B11-4). It is typically lower at the river mouth, and more weakly correlated with discharge. The ratio between the two tends to be lower at higher discharges (Figure B11-4).
- Given the discharge effects, we consider the flow-weighted average ratio to be a reasonable estimator of the proportion of TSS at Selkirk transported directly into Lake Winnipeg. This flow-weighted average ratio is 0.68 (n = 27).
- Through the period 2004–2017, the average sediment load in the Red River at Selkirk was 2.8 Tg/yr. Assuming that the losses by sequestration in the reach between Selkirk and Lake Winnipeg (mainly in the NLM) are proportional to the decrease in TSS, then the annual average sediment load from the Red River transported directly into Lake Winnipeg is 1.9 Tg.

Table B11-1 Descriptive statistics for TSS measured in the Red River at Selkirk and at Station 01 near the mouth, and ratios between paired observations (including only observations separated by <= 9 days). Data are restricted to the period for which both were routinely monitored, i.e. open-water season, 2006–2016.

	Average	Median	Min.	Max.	n
Selkirk	162	113	12	2250	195
Station 01	90	51	15	405	30

Figure B11-3. Upper images: the lower Red River, including the NLM. Note subsidiary deltas indicating deposition at mouths of distributary channels (lower right) and in Lake Winnipeg at the mouths of the west and main channel (upper right). Source: Google Earth; image recorded 29 June 2019. Lower picture: shoals of newly deposited sediment among cattails, apparent during a low water period following inundation during the spring flood of 2015.

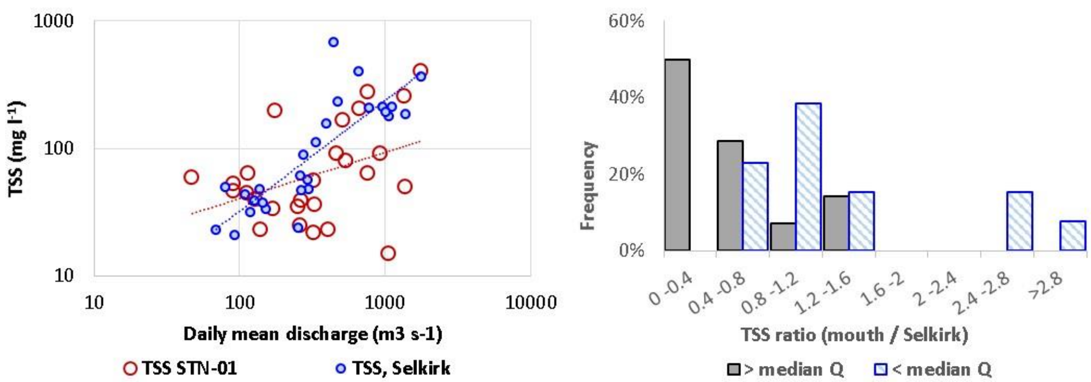


Figure B11-4. Left: TSS at Station 01 (at the river mouth) and at Selkirk as functions of daily average discharge at Selkirk. Right: distribution of ratios of TSS at Station 1 to TSS at Selkirk, with data grouped according to whether discharge (Q) was less (n = 14) or greater than (n = 13) the median for the sample. In both charts, the data set was limited to observations <=9 days apart, all during the open water period, recorded over the period 2006–2016.

Comment on the sediment capacity of the NLM

- We have inferred from the decrease in TSS concentrations in the Red River between Selkirk and the river mouth that 0.9 Tg/yr is diverted into the NLM. If distributed over the entire wet area of the marshes (area of open water plus emergent macrophytes; Grosshans et al., 2004), this load would support a sedimentation rate of 0.002–0.004 cm/yr (Table B11-2). Since it is unlikely that the load is distributed throughout the whole NLM, this represents the minimum likely range of sedimentation rates.

- Due to differential isostatic rebound and increasing runoff from the watershed, the water level in the NLM has been estimated to have risen by 0.002 m/yr over the last 300 years (Nielson 1998). By the hydrometric record in the South Basin of Lake Winnipeg, it has risen by ~ 0.005 m/yr over the last century (Figure B11-5) due partly to large increases in discharge from southern tributaries (McCullough et al. 2012). At the value estimated for sedimentation due to 0.9 Tg/yr loading, 0.002 m/yr (Table B11-2) is in equilibrium with Nielsen's estimate of the average rate of water level rise over the last 300 y. At the upper end of the range, 0.004 m/yr sedimentation is less than the average 0.005 m/yr rise recorded at hydrometric stations over the last century, and would require only about $^{4}/_{5}$ ths of the total wetted area of the marshes to absorb the load without diminishing the area and volume of marshland.
- It is beyond the scope of this study to refine our estimates of sedimentation rates in the NLM, or of water level rise. For the purposes of this paper, we can conclude that the NLM has had the capacity to absorb the load that we estimate to be diverted from the Red River over at least the last century, and probably over a much longer period of time.

Table B11-2. Average sedimentation rate in the NLM due to the load diverted from the Red River, a settled in situ density of 1.3–2.3 g/cm^3 (i.e. the range observed in our Lake Winnipeg cores). The wet area includes the areas of open water and emergent macrophytes, measured in 2001 (Table 2 in Grosshans et al., 2004).

Wet area (km^2)	Diverted sediment mass (Tg/yr)	Settled, *in situ* density (g/cm^3)	Diverted sediment volume (km^3/yr)	Sedimentation rate (m/yr)
190	0.9	1.3	0.00069	0.0036
190	0.9	2.3	0.00039	0.0021

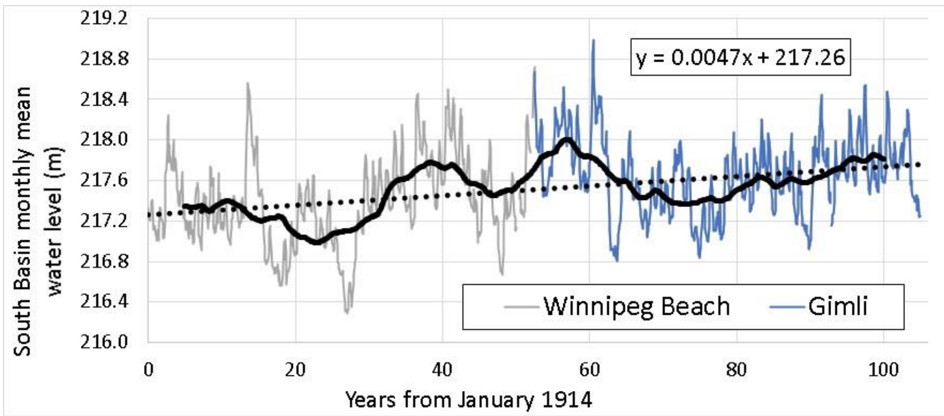


Figure B11-5. Water level in the South Basin. Source: Water Survey of Canada hydrometric records for stations at Winnipeg Beach and Gimli.

Inter-basin sediment fluxes and sediment export

Locations

- Interbasin fluxes: TSS measured at stations W9 (for estimating load from the South Basin to the Narrows) and W7 (from the Narrows to the North Basin);
- Export: TSS measured in the outlet channels and at W1 and W3 in the northern North Basin (Figure B11-1).

Data sources

- Discharge: Water Survey of Canada (https://www.canada.ca/en/environment-climate-change/services/water-overview/quantity/monitoring/survey.html)

- TSS was supplied on request by the Manitoba Department of Agriculture and Resource Development.

Methods

- *Interbasin sediment fluxes:* Sediment fluxes were determined as the product of average net discharge, and median TSS for the open water season and under-ice seasons (May–November and December–April, respectively) from 2004–2017 (Table B11-4). Seasonal median TSS was determined from observations at the nearest long term monitoring stations in the lake, where typically one observation was recorded each winter, and three during each open water season. TSS from surface and bottom samples was averaged prior determining medians (and observations were excluded if both surface and bottom were not sampled).
  - *South Basin to Narrows:* Net northward discharge was estimated as the sum of measured discharges of the Red River at Selkirk and the Winnipeg River at Powerview (2004–2017) plus discharge from ungauged drainage near the mouth of each estimated by assuming the same unit area discharge as the nearest gauged river (Table B11-3). TSS in the discharge was determined from records at Station W9.
  - *Narrows to North Basin:* Net northward discharge was estimated as the sum of discharge from the South Basin, plus discharge of the Bloodvein River at Bloodvein Bay from 2004–2017, plus discharge from ungauged drainage estimated by assuming the same unit area discharge as the Berens River. TSS in the discharge was determined from records at Station W7.

- *Sediment export from the lake:* The nearest hydrometric stations downstream of Lake Winnipeg are at Jenpeg and Sea River Falls on the west and east channels of the Nelson, 100 and 70 km downstream of the Two-Mile Channel, respectively. There is also a hydrometric record on the Gunisao River at Jam Rapids, which joins the Nelson River between the outlets of the lake and the downstream stations. Outflow from the lake was calculated as the sum of the discharge of the Nelson River at the two Nelson River stations, minus 2.13 times the discharge of the Gunisao River (where 2.13 is the ratio of the total drainage area between the outlet of the lake and the two stations, to the drainage area of the Gunisao River at the hydrometric station). The average seasonal discharges were calculated for the period 2013–2019, this being the entire period for which we were able to obtain TSS data. One third of the discharge were assumed to flow through the Two-Mile Channel (Kimiaghalam and Clark, 2017) and the remainder through the natural outlet at Warren's Landing. Median TSS was determined separately for each outlet. TSS in the under-ice discharge was assumed to equal the median in surface and bottom samples at stations W1 and W3 in the northern North Basin (Figure B11-1).

Results

- Results of intermediate computations are reported in Table B11-4. The sediment fluxes out from the South Basin to the Narrows, and from the Narrows to the North Basin were in each case 0.3 Tg/yr. Export into the upper Nelson River was 0.6 Tg/y.

Table B11-3. Drainage areas of gauged rivers tributary to the South Basin and Narrows Region of Lake Winnipeg.

Basin	Tributary River	Gauged d.a. (km^2)	Associated ungauged d.a. (km^2)
South Basin	Red River	287,000	9,699
	Winnipeg River	136,000	5,730
Narrows	Bloodvein River	9,090	18,560*

* Not including the Berens River watershed, which flows into Lake Winnipeg at the northern boundary of the Narrows.

Table B11-4. Seasonal average discharge, TSS and sediment flux from the South Basin to the Narrows, from the Narrows to the North Basin, and through the outlet channels from Lake Winnipeg into the upper Nelson River. Open-water = May–November; under-ice = December–April.

	South Basin to Narrows	Narrows to North Basin	Lake Winnipeg outlet, 2-Mile Channel	Lake Winnipeg outlet, Warren's Landing	Units
Open water					
Discharge	31.6	37.3	16.7	33.4	km^3
TSS	8.0	6.5	13.6	8.0	g/m^3
Flux	0.25	0.24	0.23	0.27	Tg/yr
Under-ice					
Discharge	18.5	19.9	10.2	20.4	km^3
TSS	2.5	3.0	2.5	2.5	g/m^3
Flux	0.05	0.06	0.03	0.05	Tg/yr
Annual flux	0.3	0.3	0.6		Tg/yr

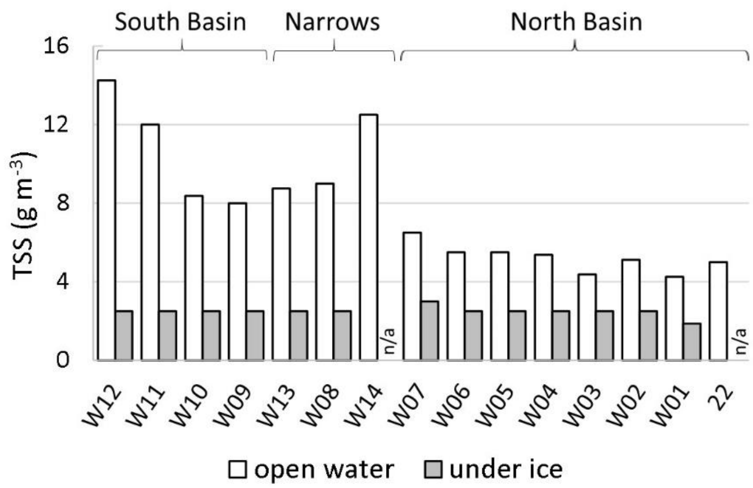


Figure B11-6. Median TSS recorded from 2004–2016 at selected monitoring stations on Lake Winnipeg. Sample size ranges from 14–39 in open water; from 6–9 under-ice. Data source: Manitoba Department of Agriculture and Resource Development, supplied on request.

Bank erosion along the north shore of Lake Winnipeg

The north shore is formed by erosion of high banks of glacio-lacustrine sediments, predominantly silt, overlaid by about 1 m of sphagnum peat. High, actively eroding banks occur over a 50 km reach, stretching westward from near Warren's Landing (Figure B11-7). To check the rate of bank erosion estimated by Brunskill and Graham (1979) we compared bank locations in Landsat TM images retrieved in 1984 and 2011 (Figure B11-8). Over a 50 km reach with visibly active bank erosion (observed from the vessel Namao) we estimate that the average recession rate was between 1.3–2.4 m/yr (and up to 2X as much near the Two-Mile Channel). Over the same reach, the bank height (excluding peat) ranged from 8–10 m, so that between 10 and 24 m^3 of sediments were eroded per meter/m of shoreline per year. Our rough estimate is similar to the earlier estimate of 10–20 m^3 estimated by Brunskill and Graham (1979). In situ density in several samples ranged from 2140–2670 kg/m^3. We conclude that from 1.1 to 3.1 Tg of sediment is supplied to Lake Winnipeg (i.e., annually) by erosion of banks along the north shore.

Figure B11-7. Actively eroding glacio-lacustrine sediments along the north shore of Lake Winnipeg.

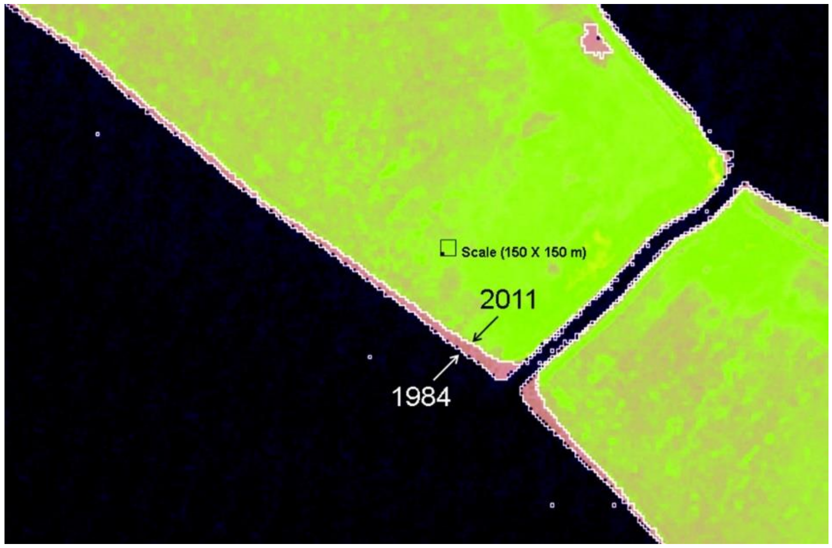


Figure B11-8. Bank retreat based on overlaid Landsat TM images retrieved in 1984 and 2011. Scale: the outer, open square indicates 150 X 150 m on the ground; the inner black square is 30 m X 30 m.

References

Brunskill, G.J. and B.W. Graham. 1979. The offshore sediments of Lake Winnipeg. Canadian Fisheries and Marine Service Manuscript Report 1540. vi + 75 p.

Grosshans, R.E., D.A. Wrubleski and L. G. Goldsborough. 2004. Changes in the Emergent Plant Community of Netley-Libau Marsh Between 1979 and 2001. Delta Marsh Field Station (University of Manitoba) Occasional Publication No. 4. 40 p. + appendices.

KGS Group. 2019. Netley-Libau Marsh Restoration Pilot Project Manitoba Environment Act Proposal. Final Report. KGS 18-3471-001. V + 59 p. + appendices. https://www.gov.mb.ca/sd/eal/registries/6004netleylibau/eap.pdf (accessed 8 Nov. 2020)

Kimiaghalam, N., Clark, S.P., 2017. Morphodynamics of diversion channels in Northern Manitoba, Canada. Canadian Water Resources Journal / Revue canadienne des ressources hydriques. 42:2, 149-162, DOI: 10.1080/07011784.2016.1249961.

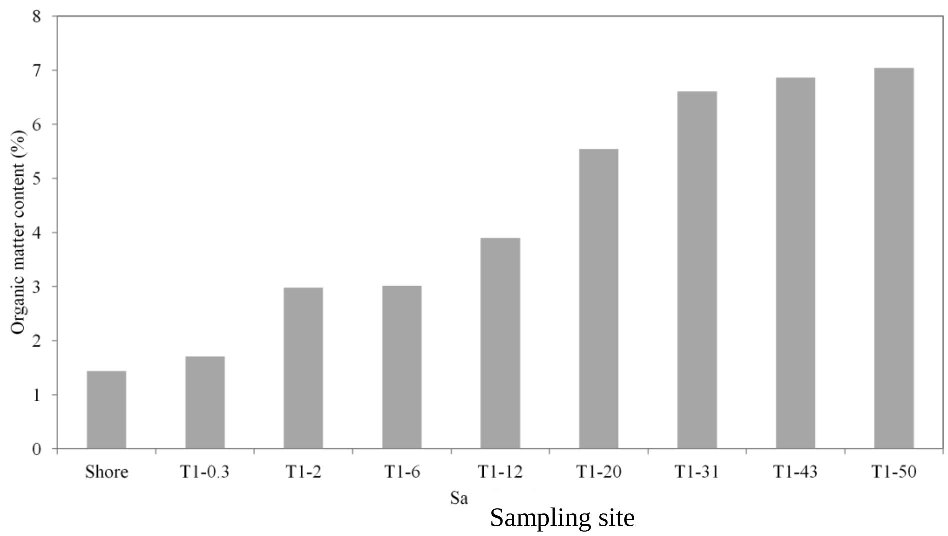
McCullough, G.K., S.J. Page, R.H. Hesslein, M.P. Stainton, H.J. Kling, A. Salki, D.G. Barber. 2012. Hydrological forcing of a recent trophic surge in Lake Winnipeg. J. of Great Lakes Research. 38:95-105.

Nielsen, E. 1998. Lake Winnipeg coastal submergence over the last three centuries. Journal of Paleolimnology 19: 335–342.

B12: A comparison of different physical characteristics of Lake Winnipeg and the Great Lakes (y axes are the ratio of properties for each lake with Lake Winnipeg).

B13: Spatial analysis of organic matter (OM) content of bottom sediment

Similar to the vertical variation in water content, the expected trend of decreasing OM with increasing sediment depth was observed in nearly all sediment cores (data not shown). Despite the large area of the lake, the OM content of the sediment cores show relatively limited spatial variations. The average OM contents in the South Basin, Narrows, and NB offshore is 5.2% ± 1.1% (n = 15), 6.3% ± 0.8% (n = 6), and 6.8% ± 0.3% (n = 15), respectively. However, the OM content for the top 5 cm of sediment cores along transect T1 (see below) generally increase with an increase in distance from the north shore from 1.7% to 7%. The low level of OM in the north shore materials (i.e., 1.4%) and the increase from NB nearshore to NB offshore in a relatively short distance (~ 50 km) provide further evidence of the contribution of the north shore materials (as an active sediment source) to the North Basin bottom sediments.



Spatial comparison of the organic matter (OM) content of the top 5 cm of sediment cores collected along transect T1.

Appendix C: Supplementary materials for Chapter 5

C1: UTM coordinates of the suspended sediment sites in the Burntwood River (BR).

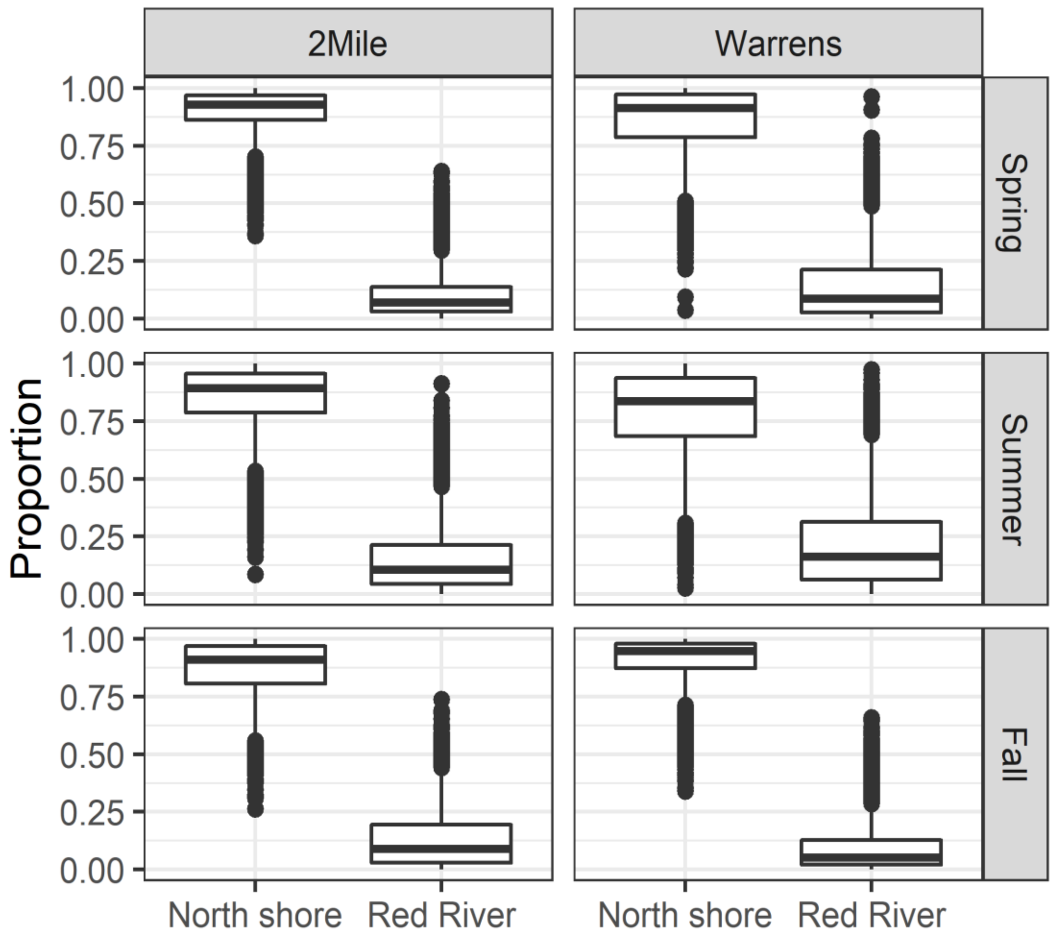
Distance from the Split Lake inlet (km)	UTM Zone	Easting	Northing
259	14 U	478,960	6,191,495
172	14 U	532,599	6,154,843
112	14 U	572,723	6,179,289
24	14 V	630,848	6,219,519
2	14 V	648,649	6,224,492

C2: UTM coordinates of Environment and Climate Change Canada (ECCC) hydrometric stations and Manitoba Agriculture and Resources Development (MB ARD) water sampling sites in the Burntwood River (BR) and Upper Nelson River (UNR).

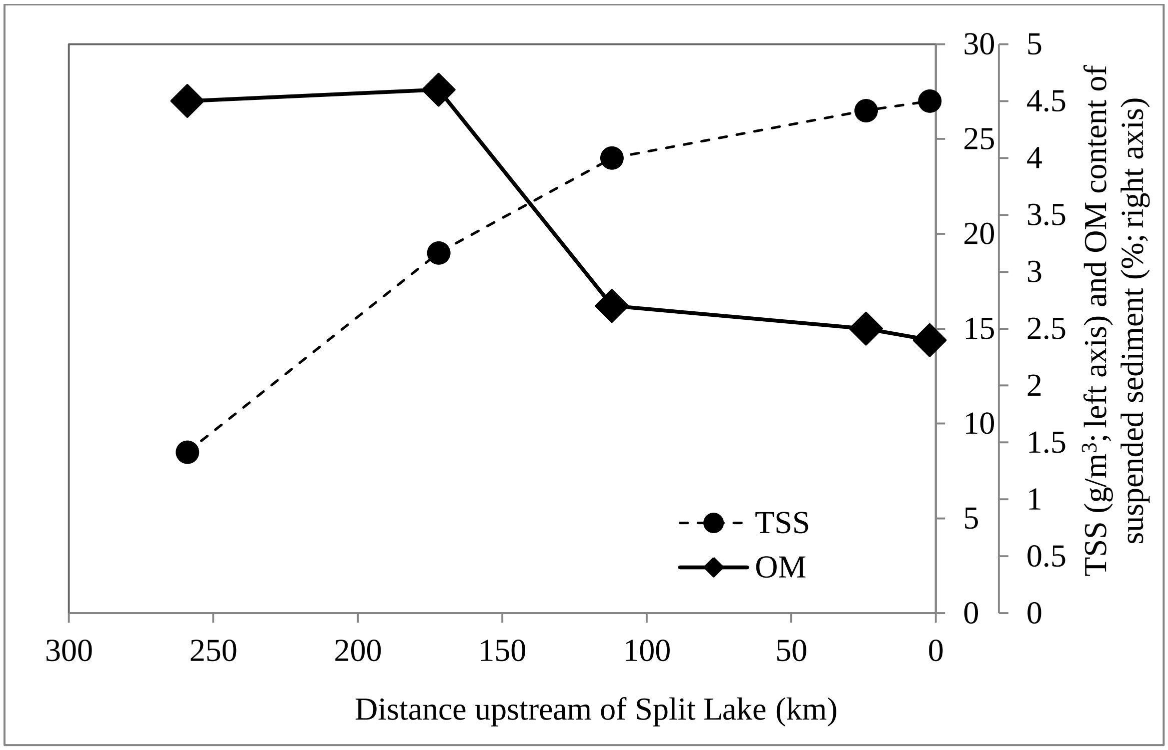
	Hydrometric stations		
	UTM Zone	Easting	Northing
Thompson (Burntwood River)	14 U	569,062	6,177,690
Sea River Falls (Upper Nelson River, East Channel)	14 U	591,889	6,011,142
Jenpeg Generating Station (Upper Nelson River, West Channel)	14 U	561,522	6,039,569
Kelsey Generating Station (Upper Nelson River)	14 U	650,522	6,202,025

Site	Water sampling stations		
	UTM Zone	Easting	Northing
Thompson (Burntwood River)	14 U	572,682	6,179,263
Split Lake inlet (Burntwood River)	14 V	650,585	6,224,394
Playgreen Lake (Upper Nelson River, West Channel)	14 U	561,679	5,970,773
Norway House (Upper Nelson River, East Channel)	14 U	576,424	5,982,692
Jenpeg Generating Station (Upper Nelson River, West Channel)	14 U	562,895	6,043,049
Sipiwesk Lake (Upper Nelson River)	14 U	595,899	6,107,800
Sipiwesk Lake outlet (Upper Nelson River)	14 U	607,458	6,119,739
Split Lake inlet (Upper Nelson River)	14 V	654,222	6,212,270
Split Lake outlet	14 V	680,875	6,236,433

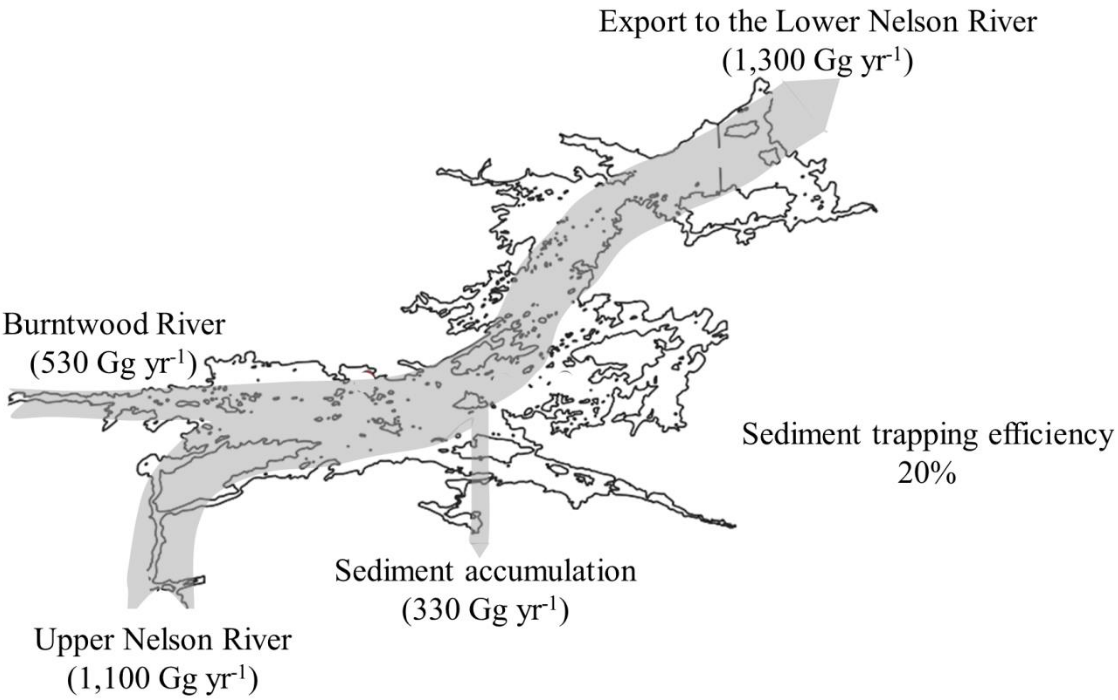
C3: Box and whisker plot of the seasonal relative contribution from the Red River and Lake Winnipeg north shoreline in suspended sediment at the outlets of the lake using colour- and geochemical fingerprints and MixSIAR



C4: Spatial distribution of total suspended solid (TSS) and organic matter (OM) content of suspended sediment collected in the Burntwood River (BR).



C5: Sediment budget for Split Lake under post-regulated conditions



C6: Top (table): estimation of sediment accumulation rate for cores collected by Manitoba Hydro in 1997 (Core A) and 1998 (Cores B and C) using the ^{137}Cs and excess ^{210}Pb chronology models; Google Earth map: Locations of the Split Lake sediment cores collected in 1997 and 1998, and the plume extended from the Burntwood River (BR) along the northern margin of the lake

Sample	Sedimentation rate ($g\ m^{-2}\ yr^{-1}$)		
	$^{210}Pb_{xs}$ (linear fit)	$^{210}Pb_{xs}$ (constant flux)	^{137}Cs
Core A	1,070	ND#	1,117
Core B	994	904	1,269
Core C	788	834	ND

#: ND = Not datable.