Development of Resuspension Technique for On-site Phosphorus Remediation of Eutrophic Lakes

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Abstract

Development of Resuspension Technique for On-site Phosphorus Remediation of Eutrophic Lakes

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Eutrophication is the enrichment of nutrients, especially phosphorus, in aquatic systems, with harmful effects on surface water bodies such as lakes. Usually, 80-90% of the phosphorus is incorporated into lake bottom sediments, which could be released to the water body through different mechanisms.

Resuspension technique has been introduced as a new approach to solving the eutrophication issue by treating the bottom sediments. The basic idea of resuspension method is that finer sediment particles tend to adsorb more contamination due to their higher effective surface area and ionic attraction. In resuspension systems, finer particles are targeted for removal from the aquatic environment. In this experimental study, the resuspension mechanism was simulated in a confined water column. This study aims to identify the most effective particle sizes and the optimum condition to capture the sediment particles, and to investigate the feasibility and efficiency of the resuspension method.

The results show that sediment particles of around 16 µm to 30 µm in size carried higher concentrations of phosphorus than sediment particles of other sizes. Resuspension technique could successfully reduce the total concentration of phosphorus contaminations. The optimum settling time to decrease the phosphorus level by 15% was calculated as 14 minutes, considering 30% slurry removal. The phosphorus concentration decreased by 15% by removing only 7% of the fine particles. Phosphorus contamination carried by the suspended sediment particles was about 2.4 times higher than the bulk sediments. The experimental results confirmed the accuracy of theoretical calculations. The suspended particle matter (SPM) removed through the resuspension system was passed through one layer of different filters. Filters with a pore size of 13.3 μm decreased the phosphorus level of SPM from 6.57 mg/L to 0.027 mg/L. Generally, filters with smaller pore sizes were more effective for decreasing the phosphorus contamination.

The resuspension method showed desirable results for the removal of phosphorus from bottom sediments. A small amount of sediment removed from the system, and no chemical substances were employed. Consequently, the resuspension method causes less destruction in the aquatic ecosystem.

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List of Abbreviations

Dissolved Oxygen (DO)

Dry Weight (DW)

Effective Microorganism (EM)

Environmental Protection Agency (EPA)

Heavy Metals (HMs)

Inductively Coupled Plasma Mass Spectrometry (ICP-MS)

Lightweight Aggregate (LWA)

Loss on Ignition (LOI)

Polynuclear Aromatic Hydrocarbons (PAHs)

Soluble Reactive Phosphate (SRP)

Total Nitrogen (TN)

Total Phosphorus (TP)

Total Suspended Solids (TSS)

1 Introduction

1.1 Background

Sediments are a crucial element in the aquatic ecosystem, and enormous biodiversity exists in the sediments (Bartram & Ballance, 1996). Settlement and accumulation of suspended solids over time form most sediments (Kim et al., 2007). Sediment quality impacts aquatic life, the food chain and human health since it is a source of nutrients for different kinds of species and contains various habitats.

Sediments interact with the overlying water and play a crucial role in lake water quality. The binding of many contaminants to the sediment particles is not permanent, and the contaminants can be released to the overlying water through different release mechanisms. Water quality assessments are incomplete without considering the sediment quality (Bartram & Ballance, 1996). The sources of toxic pollutants to lakes could be natural or due to human activities. Examples of natural contamination sources are volcanoes, earthquakes and forest fires. Moreover, agricultural, urban and industrial activities, spill and energy production are considered anthropogenic-related contamination sources (Mulligan et al., 2009).

Sediment contaminants are classified into eight categories by the U.S. EPA as polynuclear aromatic hydrocarbons (PAHs), pesticides (such as DDT), chlorinated hydrocarbons (such as PCBs), mononuclear aromatic hydrocarbons (benzene and its derivatives), phthalate esters and metals (such as mercury and lead), nutrients and other contaminants, such as cyanides and organo-metals (Renholds, 1998).

Among all types of contaminations, phosphorus has received special attention due to its impact on surface water quality (Kim et al., 2007). Phosphorus availability is an essential factor in determining lakes' water quality (Søndergaard et al., 2003). The enrichment of nutrients, especially phosphorus, in the aquatic systems, can result in eutrophication. It has been recognized that phosphorus is the controlling nutrient for planktonic and attached algae and macrophytes in almost all lakes (Lee et al., 1978).

The trophic state management of shallow lakes is more complicated compared to deep lakes. In the shallow lakes, phosphorus recycled from sediments is more available to the photic zone (Welch & Cooke, 2005).

The natural attenuation is not a valid option to address the lake eutrophication problem if the deposit of low phosphorus content sediment is not sufficient. Various studies investigated the rate of internal phosphorus loading decrease after the external load reduction. No considerable improvement was observed in the lake water quality after cutting or reducing phosphorus's external loading for many years. High internal phosphorus release from sediments could be the explanation (Søndergaard et al., 2003).

Sediment treatment methods are essential to tackle the lake sediment phosphorus enrichment issue, and they are categorized into in-situ and ex-situ techniques. The most common in-situ methods are sediment capping, sediment bioremediation, aeration, solidification and flushing.

The common in-situ sediment treatment techniques aim to reduce the mobilization, resuspension and availability of phosphorus contamination that have accumulate in sediments. They are temporary methods and need long term monitoring and maintenance. The treatment of sediment phosphorus store requires long-term management of shallow lakes (Welch & Cooke, 2005).

Dredging is the basis of all ex-situ sediment treatment methods. The ex-situ sediment treatment approach removes the contaminated sediments directly to reduce the internal load. Dredging is a rough method, with many disadvantages. The major problem of dredging operations is a large amount of dredged materials, which requires identifying proper storage sites and disposal methods (Yoobanpot et al., 2020). Possible environmental impacts of dredging are the potential for a significant disturbance and release of contaminants into the water body during the operations, an increased oxygen demand, a decrease of light penetration and losses of natural habitat. Besides, dredging operations are costly and cause managerial, economic and social problems (Hickey & Gibbs, 2009; Manap & Voulvoulis, 2015; Qin et al., 2006).

The resuspension technique was introduced to address the phosphorus enrichment issue of lake sediments. The resuspension method is based on the concept that finer sediments have a greater tendency to adsorb the contamination due to the high specific surface area and ionic attraction. The resuspension method aims to remove the finer sediments instead of dredging the whole contaminated area.

The resuspension method works in multiple stages: First, air jets create a strong turbulence in the confined water column and mix the sediments and water. Next, aeration is stopped, and sediment settling is allowed for a period of time. During this period, the coarser sediments settle, and the finer sediments stay suspended in the water column. Lastly, a portion of sediment-water slurry is removed from the water column by a vacuum pump and is directed to a filtration system. The result is that a part of the fine sediment particles, which is believed to have a high concentration of contaminants, is removed from the lake in question.

Unlike other in-situ techniques, the resuspension method provides a permanent solution to lake contamination. One advantage of the method is that it removes contamination with a minimum disturbance to the aquatic life and do not add any chemical reagents. Another advantage is that aeration operations create an aerobic condition in the lower sediment layers. Resuspension technique can also be applied for the remediation of inorganic contaminants as they have low density and fine texture (Mulligan et al., 2009; Fukue et al., 2012)

1.2 Motivation

The province of Quebec has more than a half million lakes, representing a significant portion of freshwater resources. Therefore, their protection is very important. Lake Caron is a shallow eutrophic lake, located in the Sainte-Anne-des-Lacs municipality in Quebec. The lake has been categorized as hypereutrophic since 2008 (Veetil et al., 2019b).

Lake Caron is an artificial lake. Similar to many other shallow lakes, this lake has experienced maximum summer TP concentrations. Lake Caron has no recent history of high external phosphorus loading. The resuspension technique could be a viable solution to address Lake Caron’s eutrophication issue.

1.3 Scope and application

The main focus of this study was on improving the resuspension technique for reducing the phosphorus contamination of lake sediment samples. The result of this study can be employed to remediate contaminated sediments in contaminated shallow water bodies such as lakes and harbours.

1.4 Objective

The specific objectives of this research thesis are to:

- evaluate the sediment behaviours and characterize water-sediment slurry during the settlement process to determine the optimum settlement time for pumping out the water-sediment slurry,
- evaluate the feasibility and efficiency of the resuspension technique by comparing the concentrations of phosphorus before and after resuspension treatment for the contaminated sediments,
- investigate the performance of filters of different pore sizes for improving the water-sediment slurry removed from the resuspension water column.

1.5 Thesis Organization

The contents of this thesis are organized into five chapters. Chapter one provides a description of the research problem and objectives.

Chapter two is a review of the pertinent literature. The review covers lake classification, lake water and sediment quality, sediment contamination, phosphorus release mechanisms, eutrophication, and sediment treatment methods.

Chapter three describes the materials and methods used in this thesis. This chapter includes detailed explanations of the experimental setup and methods, a description of the site of study, and a description of the key instruments.

Chapter four presents the results and discussion. The results were from a series of experiments conducted in this study and analyses of the experimental data. The discussion contains interpretations of the results.

Chapter five draws conclusions from the analyses of the experimental results and provides recommendations and suggestions for further work.

2 Literature Review

2.1 Classification of lakes

Thomas et al. (1996) defined lakes as "an enclosed body of water (usually freshwater) totally surrounded by land and with no direct access to the sea. A lake may also be isolated, with no observable direct water input and, on occasions, no direct output." The first level of classification of lakes which is defined by their origin are glacial lakes, tectonic lakes, fluvial lakes, shoreline lakes, dammed lakes, volcanic lakes, and solution lakes. Many lakes have been constructed artificially by humans, to fulfill various demands.

Lakes are prime regions for human settlement and habitation. They have numerous uses such as drinking and municipal water supply, industrial and cooling water supply, power generation, navigation, commercial and recreational fisheries, body contact recreation, boating, agricultural irrigation, waste disposal and other aesthetic recreational uses (Thomas et al., 1996; Yong et al., 2014).

2.2 Lake water and sediment quality

Lake water quality has significant impacts on public health and biodiversity, mainly where the lake water serves for drinking and recreation purposes (Veetil et al., 2019b). The surface water comprises all water that is naturally exposed to the atmosphere (Yong et al., 2014). Common problems facing the lake water quality are eutrophication, health-related issues and organic wastes, contaminants, lake acidification and salinization (Thomas, et al., 1996). The physical, chemical, and biological characteristics of lake water is an asset in the assessment of water quality (Bharti et al., 2011).

Settlement and accumulation of suspended solids create most of the sediments over the years (Kim et al., 2007). Sediments are a combination of sediment particles (solid), pore water (liquid), and gas (Mulligan et al., 2009). Sediments are very important in various aspects. They are a key element in the aquatic ecosystem, and large biodiversity exists in the sediments. Sediments provide the nutrients for different kinds of species, and the quality of sediments has an impact on aquatic species, the food chain and eventually on human health (Bartram & Balance, 1996). Sediments have a wide range of applications in different fields. It has been used in construction areas and as the filling material in hydraulic structures or land reclamations, sand for beaches, and farmland and wetland nutrients (Yong et al., 2014).

It is crucial to include sediment measurements and sampling into lake monitoring and assessment programs. The United States Environmental Protection Agency (EPA) has listed 128 different pollutants as the priority pollutants. Approximately 65% of these pollutants were found mainly, or exclusively, in association with sediment and biota. Limiting the contamination assessment controls to water would result in missing most of the more toxic contaminants which are in the sediments (Bartram & Ballance, 1996; Thomas et al., 1996).

Sediments interact with the overlying water. Binding of many contaminants to the sediment particles is not permanent and the contaminants can be released to the overlying water through different release mechanisms. The physicochemical properties of both pollutants and sediments are the most important factors to the abiotic reactions and transformations. To handle such complicated issues and provide information to predict system behaviour, analytical computer modelling techniques are beneficial (Mulligan et al., 2009).

2.3 Sediment Contamination

The sources of toxic pollutants to lakes could be natural or by human activities. The sources can be a “point source” or a “non-point source”. Natural rock substrates may release high levels of toxic metals. Given that natural cycles of weathering, transport and deposition are slow processes, human activities are mostly responsible for lake pollution. Municipal and industrial effluent discharges due to human activities are direct point sources of contamination. Contaminated wash-off and soil erosion from agricultural lands carrying materials applied during agricultural land use, mainly herbicides and pesticides might diffuse to the lake water body. Wash-off from city streets, horticultural and gardening activities, and from industrial sites and storage areas, could also end up in lakes. Other water bodies (e.g. riverine sources, groundwater sources) are also possible to act as contamination sources and transfer the contamination to the lakes (Thomas et al., 1996).

The U.S. EPA identified and classified the sediment contaminants into eight categories: polynuclear aromatic hydrocarbons (PAHs); pesticides (such as DDT); chlorinated hydrocarbons (such as PCBs); mononuclear aromatic hydrocarbons (benzene and its derivatives); phthalate esters and metals (such as mercury and lead); nutrients; and other contaminants, such as cyanides and organo-metals (Renholds, 1998). More details follow regarding sediment contaminants, which are more related to the scope of this study.

2.3.1 Heavy metals

Heavy metals (HMs) are inorganic geoenvironmental pollutants. The more notable HMs are lead (Pb), cadmium (Cd), copper (Cu), chromium, (Cr), nickel (Ni), iron (Fe), mercury (Hg), and zinc (Zn) (Mulligan et al., 2001; Yong et al., 2014). The sources of pollution by heavy metals could be from mining activities, industrial and domestic effluents, or from the combustion of fossil fuels (Renberg, 1986).

The heavy metal distribution in soils and sediments can indicate the potential harm to the environment through the chemical associations. The degree of mobility is an important factor in evaluating the risks associated with heavy metals. Generally, those with high mobility are believed to cause more risks for the environment.

Remediation methods to remove heavy metals are mostly common with soil treatment techniques. Pretreatment, physical separation, thermal processes, biological decontamination, stabilization/solidification and washing are examples of possible sediment treatment methods effective for heavy metal removal (Mulligan et al., 2001).

2.3.2 Nitrogen

Many forms of nitrogen exist in the sediments and water of lakes such as inorganic, organic, dissolved, and particulate. Summation of all forms of nitrogen present in sediment is indicated as total nitrogen (TN). It is challenging to manage and control nitrogen because of the existence of a great amount of N in nature and the fact that it is not easily adsorbed on materials. Releasing nitrogen from sediment into the water is possible by denitrification (Mulligan et al., 2009).

Denitrification is the conversion of NH_4^+ to NO_3^-. Excess NO_3^- is an aquatic pollutant and a potential health risk.

N is one of the main nutrients for algae growth and an essential component of proteins, enzymes, and other biologically important compounds. According to the Strauss et al. (2002), due to an excessive amount of available nitrogen compared to the quantity of nitrogen needed by the aquatic plants for optimum growth nitrogen is usually not a controlling nutrient in lakes.

2.3.3 Phosphorus

“Phosphorus (P) is a highly reactive, multivalent, non-metal of the nitrogen group that is never found free in nature” according to the Karim et al. (2012). It is a very dynamic, biologically active element (Almroth et al., 2009). Aquatic systems have faced phosphorus delivery as a mixture of dissolved and particulate forms. The sum of these two forms of phosphorus represents the total phosphorus (TP) concentration (Correll, 1999).

Nearly all total dissolved phosphorus is bioavailable and can be utilized by plants and bacteria. The bioavailability of particulate P is variable and might be relatively high if P is bound to clays, easily degradable organic matter or if it is only weakly sorbed to particles (Lizhi Wang et al., 2018). The particulates might be deposited in the bottom sediments, and with the presence of microbial communities and various P compounds, might chemically or enzymatically be hydrolyzed to orthophosphate and released back to the water column (Correll, 1999). Orthophosphate (PO_4^{3-}) is a fraction of total phosphate, which is directly available for uptake by cyanobacteria, plants and algae. Orthophosphate is required for all known forms of life. It plays a significant role in biological molecules (Chorus et al., 1999; Almroth et al., 2009).

It has been recognized that the amounts of the biologically available forms of phosphorus, relative to other required nutrients in the surface waters, are small compared to the quantity of phosphate needed by the aquatic plants for optimum growth. Therefore, phosphate is the controlling nutrient for planktonic and attached algae and macrophytes in almost all lakes (Lee et al., 1978).

Phosphorus can enter the water bodies through point-sources or non-point sources of contamination. Examples of phosphorus point-source contaminations are effluents or wastewater from sewage treatment plants, industrial effluents, agricultural run-off, run-off from lawn mowing and grass cutting. On the other hand, non-point sources do not enter into the freshwater from any definite point. Stormwater and runoff due to agricultural and land cleaning activities and other diffuse sources are common examples. Other groups of non-point source phosphorus contaminations diffuse to the water bodies from within the water system due to many incidents such as washout from riverbanks and resuspension from sediments as internal loading (Karim et al., 2012). Søndergaard et al. (2003) defined the phosphorus availability as the most critical factor for determining the water quality of lakes.

In the aquatic environment, there are numerous reactions and several transport processes that naturally tend to make phosphorus unavailable for aquatic plant growth. The transport of phosphorus to the sediment by chemical and biological activities is an example of these processes. In this case, sediment acts as a sink for phosphorus. There is some recycling of phosphorus from sediments to the overlying waters and sediment acts as an internal source of phosphorus. Usually 80-90% of the phosphorus is incorporated into the sediments in water bodies with the residence time of more than a few months (Lee et al., 1978; Correll, 1998).

The settling of phosphorus compounds into the sediment occurs by various mechanisms. Examples are settling of insoluble (particulate) inorganic and organic P, uptake of soluble reactive P (SRP) by primary producers such as algae and its subsequent settling into the sediment, sorption of soluble inorganic or organic P onto particles that settle onto the sediments (especially the fine sediments) (Belmont et al., 2009).

Various studies reported no considerable improvement in the lake water quality after cutting or reducing the external loading of phosphorus for many years. High internal phosphorus release from sediment could be the explanation. The duration and importance of internal loading relate to the flushing rate, loading history and chemical characteristics of the sediment (Søndergaard et al., 2003). Phosphorus could transport from sediment to the overlying water through different mechanisms, and several variables affect the release rate of phosphorus from sediment (Blond et al., 2015).

2.4 Phosphorus Release mechanisms

Phosphorus release is greatly controlled by different in-situ physical, chemical and microbiological conditions (Veetil et al., 2019).

2.4.1 Resuspension

Resuspension could lead to a phosphorus release directly or indirectly into the water body for most of the cases. Resuspension increases the contact and interactions between sediment and overlying water (Selig, 2003). P release from sediments depends on many factors such as the actual equilibrium conditions between sediment and water, and the capability of phytoplankton to take up phosphorus (Søndergaard et al., 1992). In general, resuspension increases the turbidity, but does not necessarily lead to an increased release of phosphorus (Søndergaard et al., 2003). Søndergaard et al. (1992) conducted laboratory experiments to investigate the impact of resuspension on internal phosphorus loading in the shallow and eutrophic Lake Arreso, Denmark. The results showed that the P release caused by sediment resuspension was 20-30 times greater, compared to the release from undisturbed sediment cores. Laboratory experiments were conducted on the bottom sediments of the Gulf of Finland (GoF), Baltic Sea, to investigate the effect of the sediment resuspension. They reported no P release into the water body as a direct result of the sediment resuspension (Almroth et al., 2009). The impact of the sediment resuspension on the phosphorus release might vary for different lakes, and an accurate judgement requires further information about the lake characteristics. Bioturbation is the disturbance of sediment by benthic organisms, which might release the contamination into the water column by the same processes as in resuspension (Chapman, 1996).

2.4.2 Temperature

Temperature could also be an effective factor in phosphorus release. A temperature increase could accelerate the mineralization of organic matter, the release of inorganic phosphate, and an increased sedimentation of organic material related to the seasonal variation in phytoplankton productivity (Søndergaard et al., 2003; Jiang et al., 2008).

2.4.3 Redox

Phosphorus release happens in both oxic and anoxic conditions due to various in-situ physical, chemical and microbiological conditions. The initial rate of phosphorus release is much more rapid under anoxic conditions, compared to the oxic conditions; however, in the long term, oxic release is approximately equal to the P release under anoxic conditions (Lee et al., 1976; Kim et al., 2007; Rahman & Al Bakri, 2018; Veetil et al., 2019).

Redox-sensitive iron dynamics can explain phosphorus release into the surface water under anoxic conditions. Phosphorus sorbs to iron (III) (ferric iron) compound under oxidized conditions and settles to the sediments. In the anoxic condition, iron (III) is reduced to iron (II) (ferrous) and subsequently, both iron and sorbed phosphate liberate into the solution (Lee et al., 1976; Correll, 1998; Søndergaard et al., 2003). The results of a study on phosphate adsorption properties of surface sediments from 12 lakes revealed that "the capability of aerobic sediments to buffer phosphate concentration correlated with the Fe:P ratio while the maximum adsorption capacity correlated with total iron." (Jensen et al. 1992) supported the idea of Fe:P ratio importance in the control of the phosphorus release. According to the results collected from 15 Danish lakes, they claimed that the lake sediments with Fe:P ratio (by weight) above 15 successfully controlled the phosphorus release under the oxic conditions, although lakes with Fe:P ratio less than 10 seemed unable to retain the phosphorus.

2.4.4 pH

pH is important in P release especially when the capacity to retain phosphorus depends on iron. In the high pH, hydroxyl ions compete with phosphorus ions and phosphorus binding capacity of the sediment layer in decreases in the oxic conditions (Søndergaard et al., 2003). Lake pH is dependent on many factors. pH value is relatively high in summer, while the general condition (e.g. light, high temperature, high sedimentation rate of organic matter) is in favour of photosynthesis. The consumption of CO_2 within photosynthesis processes increases the pH (Morgan, 1985).

2.4.5 Mineralization and microbial processes

Microbial activities affect the phosphorus cycling in lake sediments directly and indirectly. Sediment bacteria may play a significant role in the uptake, storage and release of phosphorus (Søndergaard et al., 2003). A considerable proportion of TP in soils and sediments is present in organic forms, which have to be mineralized. Microorganisms can mineralize organic P inside and outside their cells. Sometimes, the microbial C demand would result in the consumption of organic phosphorus compounds as a C source. Hence, the microbial microorganisms mineralize P without incorporating it (Spohn & Kuzyakov, 2013).

On the other hand, a high organic input leads to the potential for a high mineralization rate while the supply of oxidizers such as oxygen or nitrate is sufficient. The consumption of oxygen and nitrate would create an anoxic condition and result in phosphorus release due to redox reactions. If nitrate concentrations are low and sulphate levels are high, desulphurization happens in the presence of a sufficient supply of biodegradable organic matter. In the above-mentioned process, sulphate reduction leads to the formation of iron sulphide and decreases the potential of phosphorus sorption. It follows the potential phosphorus release from the sediment (Søndergaard et al., 2003). The sources of sulphide may differ with the sediment site (Jones et al., 1982; Parnell et al., 2018).

2.4.6 Submerged macrophytes

“Macrophytes are aquatic plants that generally inhabit lentic ecosystems and play an essential role in natural processes, such as nutrient cycle and metabolism” (Moreira et al., 2018). They could have negative or positive influence on the phosphorus release. They are capable of enhancing the oxic conditions by releasing oxygen from their roots. Consequently, the redox-sensitive phosphorus sorption to iron increases. P release could also lessen in the case of high abundance of macrophytes by reducing the rate of resuspension of bottom sediment (Søndergaard et al., 2003).

Wang et al. (2018) studied decomposition rates and P release from the submerged macrophyte. They observed an increase in the N and P contents and a decrease in the C content, by the end of the decomposition phase. They believed that an efficient management strategy could include the harvesting of this submerged macrophyte before its decomposition in order to control eutrophication in lakes.

2.5 Eutrophication

2.5.1 Causes of eutrophication

Eutrophication is the enrichment of nutrients, especially phosphorus and nitrogen, in the aquatic systems, followed by biological responses. It has been recognized that phosphorus is the controlling nutrient for planktonic and attached algae and macrophytes in almost all lakes. However, the fact that algae typically need 106 carbon atoms and 16 nitrogen atoms for each phosphorus atom for growth and reproduction should not mislead us. Although there is a great algal demand for carbon, compared to the phosphorus, carbon rarely controls the total algal mass. Phosphorus is the limiting factor because the amounts of the biologically available forms of phosphorus compared to the quantity of P needed by the aquatic plants for optimum growth is small (Lee et al., 1978). Phosphorus is a non-toxic element to people or animals, and it does not have any direct adverse effects. However, the excessive availability of phosphorus will lead to serious indirect effects and concerns such as eutrophication (Karim, 2010).

Excessive available phosphorus results in high primary production. “Primary production is a complex set of processes in which chemical or solar energy is converted to produce biomass (Sinclair, 2002).” Accelerated production of phytoplankton algae will lead to the algae bloom. High primary production is followed by death, settling, decay and deposition of the phytoplankton to the lake sediment. The decomposition of these dead residues increases the oxygen consumption in the water body and decreases dissolved oxygen concentration to the extent that oxygen is completely used up and the hypolimnetic waters become anoxic in many lakes.

It has been calculated that in the decay process, 1.5 to 1.8 g of oxygen is used per gram dry weight of mineralized organic matter. Anoxic conditions might even cause more phosphorus release into the water body. Intense or complete oxygen discharge of hypolimnetic has severe impacts on the lake system. Possible adverse effects are the release of Fe, Mn, NH_4 and metals in the hypolimnion and large-scale threat to benthic organisms such as fish and loss of biotic diversity at all trophic levels (Lee et al., 1978; Chapman, 1996).

2.5.2 Blue-Green Algae

Phytoplankton has many different structures, but all of them are unicellular algae except for cyanobacteria. Cyanobacteria are known as blue-green algae. They are photosynthetic bacteria that have some common properties with algae and exist naturally in lakes, streams, ponds, and other surface waters. Cyanobacterial blooms can be harmful to the environment, animals, and human health. The reason is that some species of cyanobacteria produce toxic secondary metabolites, called cyanotoxins. Many pieces of evidence are available from all over the world, reporting the death of dogs, birds and livestock as the result of consumption of surface water with cyanobacterial blooms (Environmental Protection Agency, 2014).

There are many types of exposure to cyanobacteria and their toxins (cyanotoxins) such as the ingestion of contaminated drinking water, direct contact, inhalation or ingestion during recreational activities. This exposure can result in variable symptoms in humans, such as fever, headaches, muscle and joint pain, blisters, stomach cramps, diarrhea, vomiting, mouth ulcers, and allergic reactions. In severe cases, seizures, liver failure, respiratory arrest, and death may occur. The cyanotoxins include neurotoxins (that affect the nervous system), hepatotoxins (that affect the liver), and dermatoxins (that affect the skin) (Environmental Protection Agency, 2014).

2.5.3 Trophic state

In addition to the P concentration of the lake, the chlorophyll α concentration (an indicator of algal abundance) and transparency of water (measured by Secchi disk) are necessary for the assessment of the trophic status (Sarma et al., 2016). Eutrophic lakes range from oligotrophic to hypereutrophic (ranging from very low to very high productivity):

- *Oligotrophic lakes:* The overlying water of these lakes is saturated with O_2. They have low biomass associated with low concentrations of nutrients (N and P) and low primary productivity.
- *Mesotrophic lakes:* These lakes are in transition between oligotrophic and eutrophic state. A decrease in O_2 concentrations may occur in the hypolimnion during summer stratification.
- *Eutrophic lakes:* These lakes contain high concentrations of nutrients followed by high biomass production and low transparency. As a result, the oxygen concentrations could get as low as 1 mg/l or less in the hypolimnion during summer stratification.
- *Hypereutrophic lakes:* These lakes contain extremely high nutrient concentrations and high biomass production. They are at the extreme end of the eutrophic range. The use of the water is severely impaired, and anoxia or complete loss of oxygen often occurs in the hypolimnion during summer stratification (Chapman, 1996).

2.6 Sediment treatment methods

Eutrophication is not a phenomenon due to water conditions only. Recent studies proved that the quality of sediments also influences the eutrophication of surface water because sediments contain nutrients that can release into the overlying water (Mulligan et al., 2009). Eutrophication management in lakes generally focuses on controlling phosphorus (P) inputs as the key element to surface waters (Ellison & Brett, 2006). The choice of phosphorus remediation approaches is influenced by the suitability of the remediation method and the nature of the lake environment in order to reach the management objectives for a specific lake. Key lake factors include the depth, morphometry, hydraulic flushing, stratification depth and period, wind fetch, water quality (e.g., alkalinity, pH, clarity), sediment characteristics, and aquatic organisms. The treatment characteristics such as thickness, area treated, application techniques, the formulation should also be taken into consideration. Considering the combination of these lake factors and the treatment characteristics would influence the efficacy and outcome of the remediation (Hickey & Gibbs, 2009).

The two main categories of the sediment treatment methods are in-situ and ex-situ methods. Treatment methods could also be classified as physical, chemical and biological. The in-situ remediation techniques can eliminate the eutrophication by having one or a combination of different approaches such as immobilization of contaminants or contaminated sediments, reducing or ceasing the release of contaminants from sediments, and contaminant extraction from the sediments (Hickey & Gibbs, 2009; Mulligan et al., 2009).

2.6.1 Ex-situ methods for sediment treatment

Dredging is the basis of all ex-situ sediment treatment methods. It removes the contaminated sediment directly to reduce the internal load. The main stages of dredging activities are excavation, transport, dewatering and disposal of sediments. Dredging could be applied to fulfill other purposes such as navigation, mining, construction and reclamation (Childs et al., 2005; Mulligan et al., 2009; Manap & Voulvoulis, 2015). Although dredging has been effective in many cases, it is a rough method and has many disadvantages, especially when it comes to dealing with the contaminated dredged materials. Possible environmental impacts of dredging are the potential for significant disturbance and release of contaminants into the water body during the operation, the increase of oxygen demand and decrease of light penetration. Another issue could be the loss of natural habitat since excavation changes the sediment type and forces polychaeta species to change their habitat. In addition, the dredging operation is costly and causes managerial, economic and social problems (Hickey & Gibbs, 2009; Manap & Voulvoulis, 2015; Qin et al., 2006). Therefore, some of the contaminated sediments might not be possible to be removed by dredging due to financial and physical limitations (Murphy et al., 1999). The environmental impacts and disturbance of dredging can be decreased by the proper design of the dredging project (Mulligan et al., 2009).

The major problem of the dredging operation is a large amount of dredged material, which requires identifying the proper storage sites and disposal methods (Yoobanpot et al., 2020). Different methodologies are available to deal with the dredged material. Disposal in a secure landfill, treatment of the contaminated sediments and reuse of the treated sediment are examples. Landfills have been used widely for disposal of dredged materials, especially when the sediment is highly contaminated, and remediation is not feasible. Sediment dewatering is important to reduce the volume of dredged sediments as the capacity of landfills is limited. On the other hand, landfill facilities cannot handle slurries (Mulligan et al., 2009; Yong et al., 2014). Most of the time, dewatering the dredged material undergoes a physical transformation from "sediment" to a "soil" after dewatering is completed (Childs et al., 2005). In the case of open disposal, it is generally not permitted for highly contaminated sediments (Manap & Voulvoulis, 2015). Open water disposal may result in a short-term risk of contaminated sediment suspension. The desorption or dissolution of contaminants also threaten the water body during the disposal process. Additionally, the potential continuous long-term release of pollutants is a serious issue with open water disposal (Jacobs & Förstner, 2001).

Different techniques are available to treat the dredged contaminated sediment based on different approaches such as physical, chemical, thermal and biological. Some of these techniques are:

- *Physical approach:* Different methods have been invented based on different physical characteristics of sediment grains and their capability of adsorbing the contamination. Physical separation processes are good examples. These methods are based on the size and density of sediment particles, and they aim to remove fine particles that are believed to be more contaminated. These methods include centrifugation, flocculation, hydrocyclones, screening and sedimentation. Sediment washing is another physical method which generally uses hot water or the mixture of water and surfactants to separate contaminants from sediments. Flotation and ultrasonic cleaning are other examples of physical remediation technologies (Mulligan et al., 2009).

- *Chemical/thermal remediation approach:* Chemical remediation includes oxidation, which uses oxidants to destroy different kinds of wastes and ideally generates no harmful byproducts. Solidification, stabilization and vitrification techniques are also beneficial to reduce the mobility of the contaminants. Electrokinetic remediation and thermal extraction are other examples of chemical remediation techniques (Mulligan et al., 2009).

- *Biological approach:* Ex-situ biotreatment systems include the use of slurry bioreactors, biopiles, landfarming, and composting, bioleaching, bioconversion and phytoremediation (Mulligan et al., 2009).

The dredged sediments are usually considered as waste materials. However, this material could be valuable and reused in many industries for economic reasons and long-term sustainability of maintenance practices. Generally, dredged material has beneficial usage in upland, wetland, and aquatic environments (Childs et al., 2005). The use of dredged sediments in the construction industry is common due to the physical and chemical characteristics of the sediments and their perpetual availability. Dredged sediments can be used as raw material to manufacture lightweight aggregate (LWA). The sediments contain a high portion of clay particles and meet the low bulk density requirements for LWA. Sediment is also suitable for structural concrete to the extent of full replacement of raw materials with sediments. Due to the high proportion of clay particles, sediments can also be used as a clay substitute for brick production and ceramics with the condition of considering economic, technical and social limitations. Another application of sediment is as a binder in concrete. This technology was developed by Gas Technology Institute and Unitel Technologies, USA, in 2008 (Junakova & Junak, 2019).

Reuse of dredged sediment in road and pavement construction may be considered with appropriate treatment and conditioning. Dredged sediment cannot be reused directly as pavement materials, and they need to undergo stabilization processes. The reason is that most of them are classified as fine-grained materials such as silts and clays, with high void ratios and compressibilities, weak strengths, and low bearing capacities (Yoobanpot et al., 2020). Agricultural use and soil applications are one of the best alternatives. It could make good use of the sediment contents of clay and silt, nutrients and organic matter and somehow bring the material back to its original place (Canet et al., 2003).

2.6.2 In-situ methods for sediment treatment

2.6.2.1 Sediment bioremediation

Sediment bioremediation is a biological approach that benefits from effective microorganism (EM) to remove total nitrogen (TN) and total phosphorus (TP) (Mohamad et al., 2017). A field-scale test was carried out by (Liangkai et al., 2019) to verify the feasibility of the sediment bioremediation technology. The initial concentrations of TP were 14.6 mg/L in the lake sediment. It decreased to 6.2 mg/L, based on the approach of sediment aeration combined with immobilized EM, after a seven month experiment.

2.6.2.2 Sediment Capping

Sediment capping is an in-situ method. There are physical and chemical approaches to sediment capping due to the different capping materials. The capping materials may be either a passive physical barrier such as sand, gravel, clay or an active barrier. The active materials are capable of binding contaminants by adsorption or precipitation processes. Physical barriers are mostly used to control the mobilization of contaminants or contaminated sediments (Hickey & Gibbs, 2009; Mulligan et al., 2009). Physical capping generally refers to placing a layer of clean materials on the sediment surface. It isolates the contaminated sediment to minimize resuspension and transport of sediment particles, stabilize sediments, and reduce the diffusion of dissolved contaminants into the overlying waters (Gu et al., 2017). Traditionally, clean sand and gravel have been adopted for in situ capping. The isolating material must be placed on the lake sediment in a uniform and accurate manner. The nutrient release from the sediment may be eliminated only for a short time, because of the time lag for advection and diffusion of nutrients from the sediment. On the other hand, new contaminated sediments might settle on the applied cap and release the contamination into the overlying water. The cap also can be gradually eroded by water currents and waves. Therefore, a few years after applying the method, the effects might not be significant (Mulligan et al., 2009).

Active capping is another in-situ method. One of the objections to the traditional sand capping is that the thick and heavy caps compact the soil matrix and causes pore water advection from contaminated sediments. To improve the effectiveness of capping for specific contaminants, Jacobs & Förstner (1999) suggested capping contaminated sediments with an active barrier system using natural zeolites. They aimed to provide a low-cost, effective alternative to the traditional capping material. Due to the reactive nature of these materials, it is possible to apply the relatively thin layers (2–5 mm) compared with physical capping (>0.1 m) (Hickey & Gibbs, 2009). Other examples of active capping agents are alum salt, iron salt, allophane, and Phoslock (Lin et al., 2017). This method aims to replace the reversible P-binding of Fe and Mn with an irreversible P-binding to the capping agent (Hickey & Gibbs, 2009).

Gu et al. (2017) conducted an experiment on the contaminated sediments of a lake located in Anseong City, Korea. They evaluate the feasibility of using activated carbon (AC) and nonwoven fabric mats for thin-layer capping to interrupt nutrient release from lake sediments. They claimed that the method was effective for interrupting the release of T-N, PO_4-P, and T-P and can be used for remediation of lake sediments with high amounts of nitrogen and phosphorus.

2.6.2.3 Aeration

As discussed in the phosphorus release mechanism, the level of dissolved oxygen is a key regulating factor in most lakes. Aeration is a short-term management strategy, and it is capable of regulating the phosphorus in the water column. The two types of aeration are full lake mixing and hypolimnetic aeration without mixing (Mulligan et al., 2009).

2.6.2.4 In-situ solidification/stabilization

Solidification reduces the resuspension of sediment and decreases the contact area of sediment particles with overlying water. This results in a reduction to the potential for contamination release. Cement and lime are generally used as solidifiers. The long-term performance of this method might be negatively affected by erosion and diffusion (Renholds, 1998).

2.6.2.5 Flushing

The flushing method aims to minimize and dilute the nutrients in lakes with physicomechanical approaches. In this method, fresh water is introduced to the contaminated lake, from a water source which is not currently entering the lake. The freshwater could be from a river in an adjacent catchment diverted or piped water from a river into the lake. Injecting the additional water to a lake increases flushing and reduces the natural retention time of lake water. In some cases, the retention time controls in-lake phosphorus concentrations and, consequently, affects the amount of phytoplankton (Hickey & Gibbs, 2009; Qin et al., 2006).

The flushing method is more effective for small and shallow lakes. Using this method for large lakes might be uneconomic. This method cannot control the eutrophication essentially and is applicable in the emergent situation as a temporary countermeasure. Another disadvantage of the flushing method is discharging nutrient-enriched water from the eutrophic lake downstream. Therefore, it is vital to consider the impacts on the downstream environment.

2.6.2.6 Resuspension Method

The resuspension method is an in situ physical separation technique. In general, physical separation processes are technically simple methods for separating dredged solids based on size and density (Mulligan et al., 2009). This technique targets and removes the most effective particles which carry a higher concentration of contaminants. Therefore, removing a small percentage of the sediment will remove significant amounts of contamination from the polluted area. The idea behind this technique is that the finer sediment particles have more tendency to adsorb the contamination due to the high specific surface area and ionic attraction (Pourabadehei & Mulligan 2016b). This method is also effective to remove organic compounds from sediment, since these materials have low density.

Regarding the effect of surface area on contamination absorption, the specific surface area is the total surface area of all the solids or particles per unit volume or unit weight. It usually is evaluated by the amount of gas or liquid (adsorbate) that forms a monolayer coating on the surface of the particles. The grain size of sediment particles is one of the most important physical properties, and it can impact the specific surface area. The specific surface area increases with a decrease in grain size. A higher surface area would adsorb more contamination (Mulligan et al., 2009; Yong et al., 2014).

The resuspension method aims to remove the finer sediments instead of dredging the whole contaminated area (Pourabadehei & Mulligan 2016a). The first stage is to create a strong turbulent flow to suspend the sediments. This can be achieved mechanically using a stir, water jet, or other techniques. The water jet could be more desirable if the sediment is under the anaerobic condition, as it is effective in changing the sediment status to aerobic. After a period of time, the coarser sediments are allowed to settle, whereas the finer sediments remain suspended. The reason is that the smaller-grain-size particles have lower specific gravities and settling velocities, and thus they remain as suspended solids longer.

In the second stage, the fine suspended sediments are pumped out. The pumped suspended sediments include a large amount of water. Therefore, dewatering is needed by filtration or sedimentation. Sedimentation could take longer for smaller or lighter particles. In the end, the contaminated solid should be properly disposed (Mulligan et al., 2009; Fukue et al., 2012; Yong et al., 2014). This technique is capable of reducing the total concentration of sediment contaminants, unlike the common in situ sediment remediation techniques (Pourabadehei & Mulligan, 2016b).

The resuspension technique was initially created and employed to address the sediment contamination issue of a channel in Fukuyama City, Japan. The main objective of the study was to remove the organic pollutants from the surface sediments and to create an environment where no hydrogen sulfide production occurs. Due to the environmental features at the time, a few techniques, such as aeration, removal of organic matter and nutrients and sediment improvement, were combined and the resuspension technique was developed. They applied the technique on the area of 3000 m^2 with a high organic content (30% loss on ignition). By removing only 3% of the re-suspended contaminated sediments, they successfully reduced COD by 27.6%, T-P by 14%, T-N by around 31% and sulphide by around 21% for re-deposited sediments in comparison to the original sediments.

Pourabadehei and Mulligan (2016) identified the resuspension technique as a desirable treatment option for treating the contaminated sediments in a harbour in Quebec, Canada. They put this method into practice and evaluated the feasibility of the resuspension technique to improve the quality of sediments and reducing the concentration of heavy metals. Severe contamination and shallowness were the main issues of this study area. The harbour was a semi-motionless area and a perfect destination for over-loading the suspended sediments, which made this harbour shallower over the years. Residues of the boat painting washout from the maintenance area were believed to be the primary source of the contamination.

Sand capping was not applicable to address the contamination issue of the harbour, because shallowness was the main issue of this harbour and capping would further decrease the depth. In addition, due to the fine texture of sediment in the harbour, capping is not effective since the sand layer can be compromised, and the contamination can leach. If one considers ex-situ treatment methods, dredging the contaminated sediment is required. Dredging the contaminated sediment can increase the risk of mobility and availability of heavy metals in the harbour and impacts on the disposal sites that receive the dredged sediment. Additionally, the disposal of contaminated dredged material can be very costly and not every landfill has space.

There was an urgent need to remove the sediments from stations around the dock and passageways to facilitate the passage of the larger yachts. In that case, potential dredging could be performed safely and more cost-effectively after applying the resuspension technique. Fukue et al. (2012) reported the average removal efficiency of more than 21% for all samples with a maximum of 25.9% for Zn and a minimum of 17.6% for Co, by removing only 4% of the contaminated sediments.

In the resuspension method, the main focus is on treating the surface sediments. The reason is that the surface sediment is in direct contact with the overlying water. The surface sediment layer's top portion is the pollution source, and any turbulence or remobilization in the system could result in the release of the contamination into the surface water. On the other hand, surface sediment is generally more contaminated compared to core sediment samples. In shallow lakes, phosphorus is mainly released from the upper 2 to 6 cm layers (Selig, 2003).

Pourabadehei and Mulligan (2016c) compared the core sediment samples and the surface sediment samples of a harbour in Quebec, Canada. On the basis of particle size analysis and images of SEM, they claimed that the surface sediments contained a finer texture. Toxicity test results of selected heavy metals for all stations showed that the surface sediment was about 10% more contaminated than the core sediments. Another study on contaminated sediment of Lake Arreso, Denmark conducted by Søndergaard et al. (1992). In this study, total P decreased from 2.7 mg P in 1 g dry weight (DW) sediment in the upper 2 cm to 1.4 mg P g DW^{-1} at a 20-30 cm depth.

As for advantages and disadvantages, the resuspension method is an in-situ method, and unlike other common in-situ methods, it decreases the total contamination of the bottom sediments and is a permanent solution to the sediment contamination. The resuspension method aims to target the most contaminated sediment and remove the small percentage of the bottom sediment instead of draining the whole area. As a result, costs for dewatering, transportation and tipping fees for handling the contaminated disposal materials in landfill are reduced. Since no chemical substances needed, less destruction to the aquatic ecosystem would occur (Pourabadehei & Mulligan, 2016c).

Using air jets to resuspend the sediments could increase the DO in the system. The DO plays an essential role in the determination of water and sediment quality. With the sufficient DO value, the organic matter can decompose under aerobic conditions (Fukue et al., 2012). DO is vital for aquatic habitats and can decrease the phosphorus release into the surface water. Under oxic conditions, hydrogen sulfide production is also eliminated.

This technique is designed for shallow water bodies such as lakes and harbours. It would be difficult to apply if the water depth was too high. Although the resuspension technique generally causes less disturbance to aquatic life compared to sand capping and dredging, further studies are still needed on the potential adverse effects on sensitive species (Pourabadehei & Mulligan, 2016c). Besides analyzing the toxicological responses of organisms to resuspended contaminated sediments, it is important to also consider the frequency, magnitude and duration of sediment disturbance events (Roberts, 2012).

The resuspension method is a new technique and thus needs further study. This work aims to develop the resuspension technique. Designing an efficient resuspension system to decrease the sediment contamination to a desirable level is favorable. For this purpose, having a good understanding of the various effective factors such as sediment behaviour of the sediment particles is necessary. Other effective parameters could be initial sediment contamination, sediment size range and size distribution, desirable phosphorus level after the treatment and the maximum capacity of the filtration system. The resuspension method's main objective is to target the effective particles and decrease the contamination level by removing the minimum amount of sediment. Therefore, the most effective grains and the best strategy to capture them should be identified.

3 Material and Methods

3.1 Introduction

This chapter provides a description of the methodologies and experimental procedures of this study. The description includes details of the site of study, field condition, physical parameters of Lake Caron, and sample collections. The procedures cover water parameter measurements and equipment used.

Figure 3.1 Photo of Lake Caron and the excessive algae growth in the surface water of the lake during the summer 2018

3.2 Site of Study

Lake Caron is an artificial and closed water body, located at Ste. Anne-des-Lacs, Quebec. The approximate surface area and water volume of the lake are 35,300 m^2 and 46,400 m^3, respectively. Lake Caron is a shallow lake, and the maximum and average depths are 2.6 and 1.4 m, respectively. The primary source of water of the Lake Caron is rainwater, snowmelt and surface runoff.

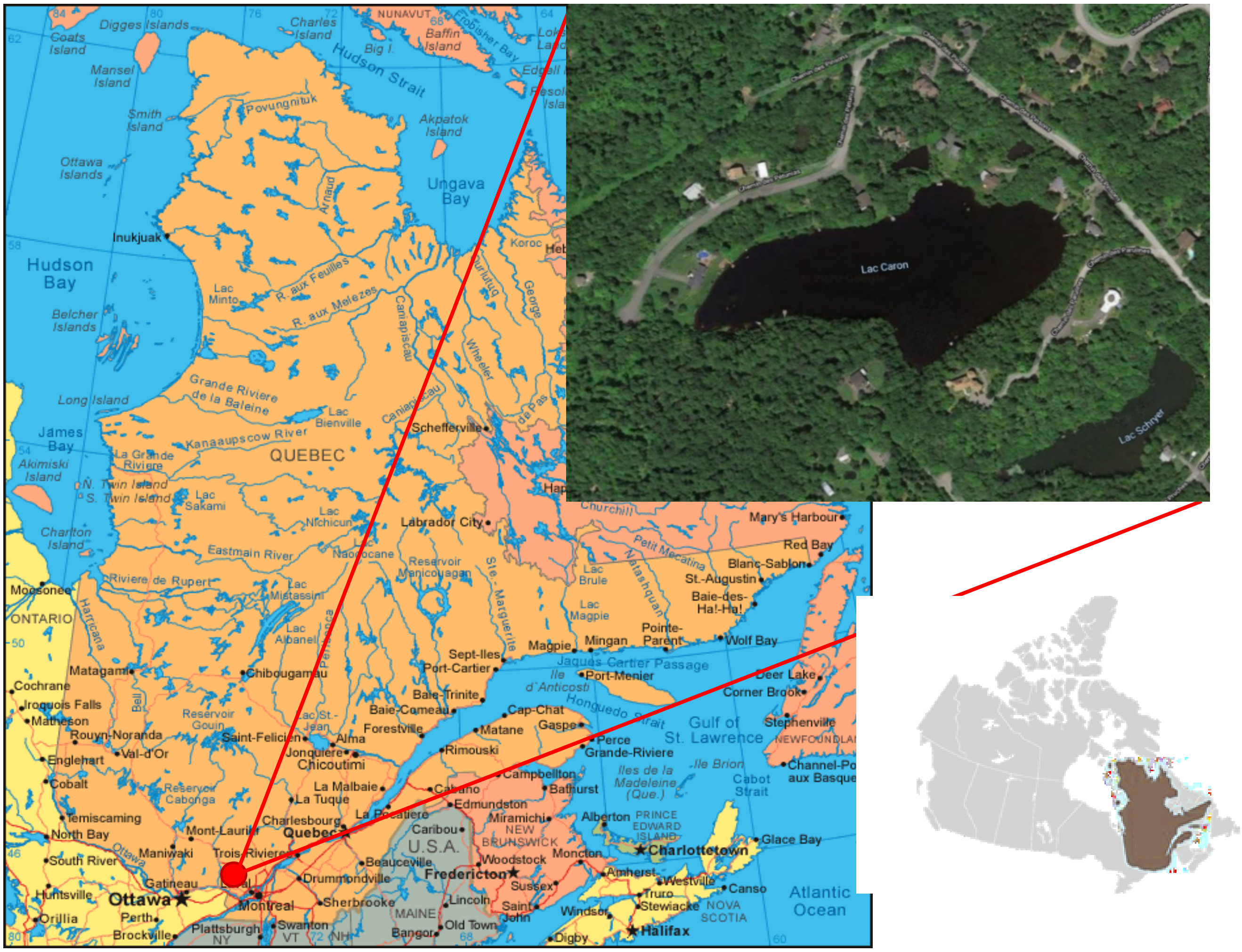

Figure 3.2 Location of lake Caron in The Quebec province, Canada (geology.com; Google Maps, 2020; globalimmigrantivest.com)

Wild trees dominate the watershed, and there are some privately owned houses surrounding the lake. Lake Caron is a eutrophic lake duo to human activities like the addition of fertilizer, phosphate detergents and septic tank discharges from the nearby neighborhood. The runoff from the forested area and plant growth, death and decomposition in the water are another source of internal phosphorus loading. Due to the high nutrient concentrations in the water, Lake Caron led to eutrophic conditions, and it has suffered from the excessive growth of algal bloom since 2008. Consequently, the lake is restricted to only recreational purposes like boating since then (Veetil et al., 2016).

3.3 Sample collection and storage

Sediment samples were collected during the summer and fall of 2018 (from June 2018 to September 2018) when the lake water quality showed the worst conditions with the highest amounts of blue-green algae blooms.

Sediment samples were obtained using a Birge-Ekman grab sediment surface sampler. The sampler had two thin jaws, hinges, overlapping on the top, which opens during descent to let water pass through, and they close at the retrieval by water pressure to minimize the washout. There are two other hinged jaws on the side, set apart forcefully and set on the springs as shown in Figure 3.3. They were capable of grabbing the sediments from the bottom surface when they close. The sampler was lower inside the water in the accurate sampling location, until it reached to the sediments.

Figure 3.3 Birge-Ekman grab sediment surface sampler

A heavy object that slides along the string attached to the sampler, which worked as the messenger, was released, releasing the spring so that the hinged side can close and grab sediment samples. The sediment samples after removal from the sampler were kept in an iced-cool box and transported to the Environmental Engineering laboratory at Concordia University, where they were stored at 4°C before further analysis.

3.4 Size analysis of suspended sediments during the settlement procedure

This experiment aims to provide a better understanding of the lake sediment behavior, and characteristics at different times and depth during the settlement process. It is designed to evaluate the effect of the size distribution of sediment particles on the phosphorus concentration of lake sediments. To verify the results, all following experiment steps are implemented in two runs.

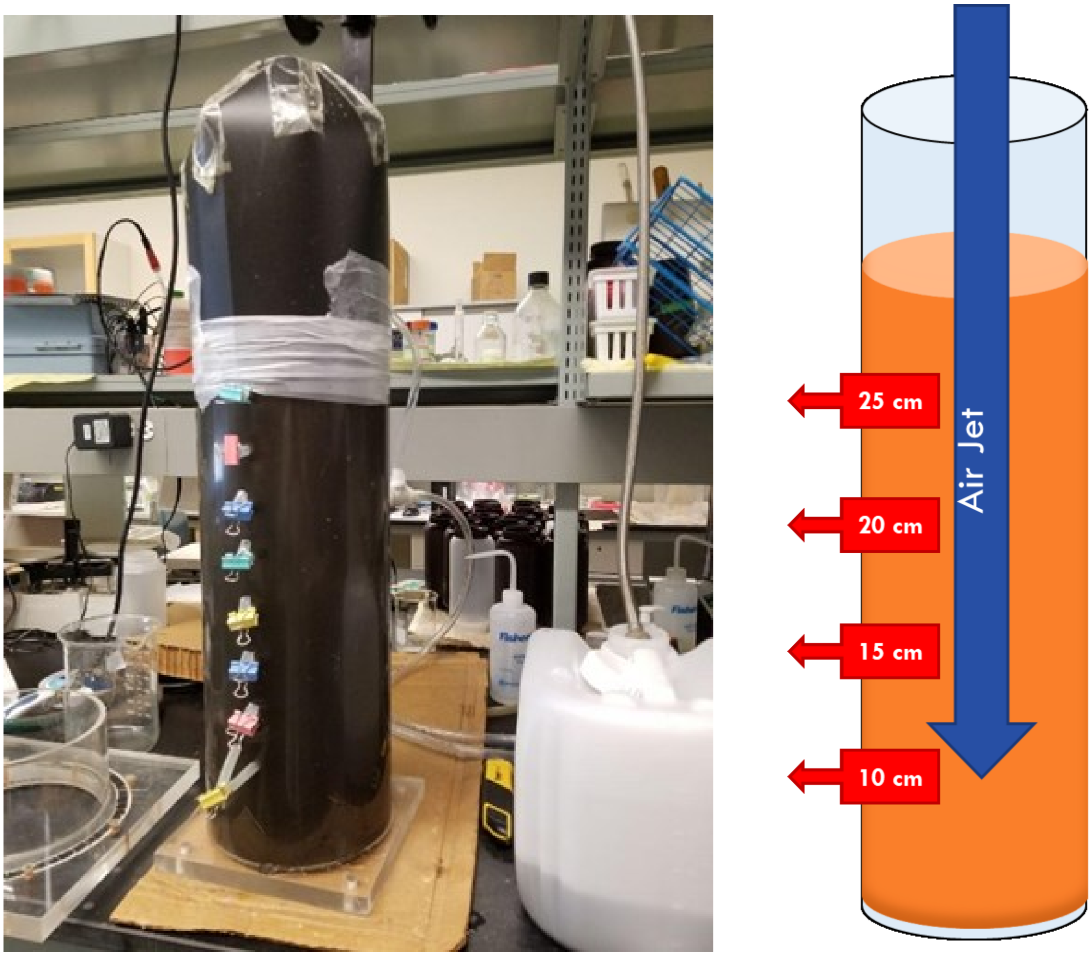

Figure 3.4 Resuspension reactor with outlets at different heights

3.4.1 Experimental setup

The resuspension setup is a vertical cylinder with the diameters of 20 cm and 50 cm in terms of internal diameter and height, respectively. There are several outlet holes in the cylinder sidewall located at different heights of 10, 15, 20 and 25 cm (Figure 3.4). During the experiments, sediment samples from the lake were homogenized and then mixed with tap water. The ratio of the sediment sample to tap water was 1:10 (vol./vol.) in the reactor. Tap water was used in all experiments due to its similar characteristics to the river freshwater samples.

In the first step, resuspension/aeration was performed by an air jet connected to the central compressing air system in the laboratory. The air jet was injected at the bottom in to provide the uniform sediment distribution in the water column. The injected air contributed to an increase in the water/sediment volume by approximately 30% (Figure 3.5).

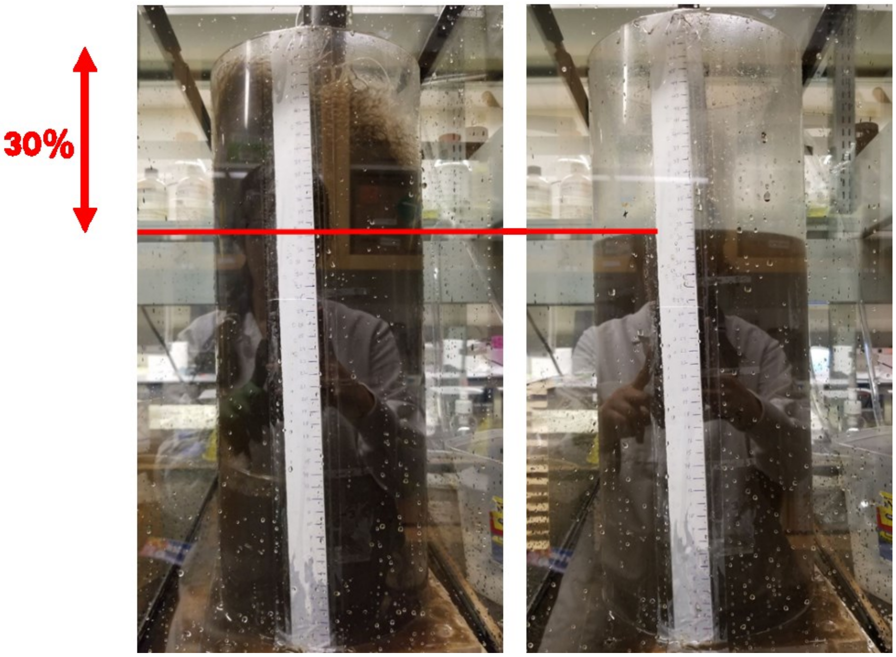

Figure 3.5 The volume increase caused by aeration process in the resuspension technique

The plastic lid on top of the reactor prevented water from spitting outside in the aeration step since the water jet created high turbulence. The aeration stopped after two hours, and immediately after this action, initial samples of well-mixed sediments and water were taken through the outlet tube at the height of 10 cm. The outlet tube was installed in the outlet hole.

The tube was long enough so that the end of the tube reached the cylinder's central line to extract the samples from the middle of the cylinder and minimize the effect of the cylinder walls on extracted samples during the settling process.

Subsequent samples were obtained every 5 minutes for 30 minutes in the 50 ml centrifuge tubes during the sediment settling process. The volume of extracted samples was relatively low comparing to the total volume of the resuspension system. Seven samples were extracted from a height of 10 cm at different times during the settling process. The experiments were repeated for the other outlet holes at the heights of 15, 20 and 25 cm, respectively, from the bottom. Therefore, 28 different samples were extracted from the setup.

3.5 Evaluating the feasibility and efficiency of the resuspension technique to treat the phosphorus contamination of lake sediments

Different laboratory-scale experiments were performed to analyze the resuspension technique's feasibility and efficiency on the lake sediment treatment. This study's main focus was on removing the phosphorus contamination by targeting and removing the most effective sediment particles. Different filters were used in this experiment to evaluate the effect of filters' pore size on improving the quality of discharged slurry from the resuspension system. To verify the results, all following experiment steps are implemented in two runs and the average values calculated with the percentage error of ±7%.

3.5.1 Experimental setup

Sediment samples were first homogenized and deposited into the reactor, followed by adding the tap water. The ratio of sediment sample to tap water was 1:10 (vol./vol.) in the reactor. The total height of the sediment and water mixture in the reactor was 40 cm. The experimental procedures for sediment resuspension consisted of three steps. The first step was resuspension/aeration, which was the same as the first experiment (Experiment 1). The air jet created strong turbulence to suspend sediment particles. After two hours of aeration, the sediment-water suspension was homogenous.

The second step was that aeration stopped after two hours, and coarser sediments started to settle. In the first experiment, the optimum settling time is calculated as 20 minutes and the volume of removed slurry is considered as 30% of the entire column. Similarly, after 20 minutes of settlement, 30% of the slurry was removed from the middle of the reactor (20 cm depth), using a pump. It took less than one minute for the suction pipe to remove 30% of the slurry from the reactor. The slurry contained water and suspended particulate matter (SPM) and some insoluble organic matter.

The slurry was directed to a filtration setup (Figure 3.6a). Geotextile filters from Texel Company were used in this experiment. Texel Matériaux Techniques, Inc. is recognized as one of the largest manufacturers of technical nonwoven materials in North America. Texel's products are marketed internationally through six market segments: geosynthetics, filtration, transport, industrial, medical-wipes, CIPP (texel.ca/fr/a-propos-de-texel, 2020).

Six different geotextiles of different pore sizes were used in this step. **Error! Reference source not found.** presents details of the filters, including pore sizes according to the manufacturer. The pore sizes ranged from 13.33 µm to around 70 µm. The filtration setup contained one layer of geotextile for the filtration purpose and a net to keep the filter in position. Water-sediment samples were collected before and after passing through filters for further analysis.

Table 3.1 Filters of Texel geotextiles used in this study

Filter number	**Filter name**	**Geotextile Number/ Media**	**Pore Size (µm)**
1	FIN01933	PEAFE001	13.33
2	FIN02144	PEAXLFE001	16.44
3	FIN02152	POAXL001	18.07
4	FIN02870	POAF001	19
5	FIN01808	F-300	60
6	FIN01584	F-200	70

In each of the filtration experiments, 100 ml of the slurry were passed through one layer of the filter. The dry filters were weighted before the experiment. After passing the entire slurry through the filter, the filter was placed in the oven and dried for 16 hours at 105 °C (ASTM D 2974-00, 2011). The weight of the contaminated filters was measured after drying. The slurry was also collected after passing the filters for further analysis. It is essential to analyze the water quality of the slurry before discharging it to the environment. The discharge should meet the water quality criteria, which shows the exact levels of specific chemicals allowable in a water body.

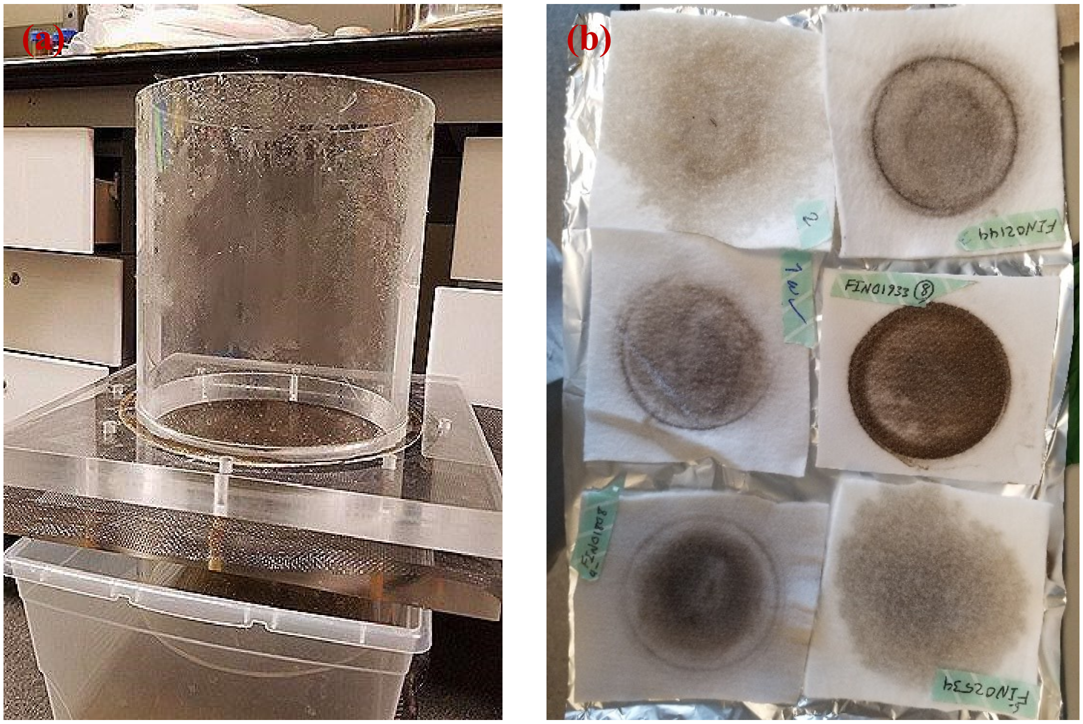

Figure 3.6 Photos of (a) filtration setup, (b) contaminated filters after the filtration process

3.6 Analysis methods

3.6.1 Turbidity

Turbidity was a parameter that was chosen for monitoring process. Turbidity is the optical property of a sample that causes light rays to be scattered and absorbed (ASTM, 2002). Each water sample was diluted ten times to meet the range of the turbidity meter (Oakton turbidity meter, T-100). The turbidity of each sample was measured three times, and the average value of readings was reported as the turbidity of the diluted sample. Afterwards, each value was multiplied by ten to calculate the actual turbidity of each sample before dilution.

3.6.2 Total suspended solids (TSS)

The amount of total suspended solids of samples is an essential parameter. Although turbidity gives us an understanding of the amount of suspended solid in samples, it is not an accurate measure of sediment mass or concentration (ASTM, 2002).

The standard method ASTM 2540-D was used to measure the total suspended particles. A known volume of each water-sediment sample was passed through the pre-weighed glass microfiber filters using the vacuum filtration method (Figure 3.7). It was then placed in aluminum dishes and dried in the oven at 105°C for 16 hours. Then the oven dried samples covered tightly, cooled in a desiccator, and the mass was recorded afterwards. The total suspended solid (TSS) was calculated by using the initial volume of samples and the weight of dry filters and aluminum dishes.

$$\text{TSS (mg/L)} = (m_f - m_i)/V \times 1000 \qquad \text{Equation 3-1}$$

where m_i = mass of the filter paper before filtration (mg); m_f = mass of the filter paper after filtration and drying in the oven (mg); V= total volume of water that was passed through the filter (ml).

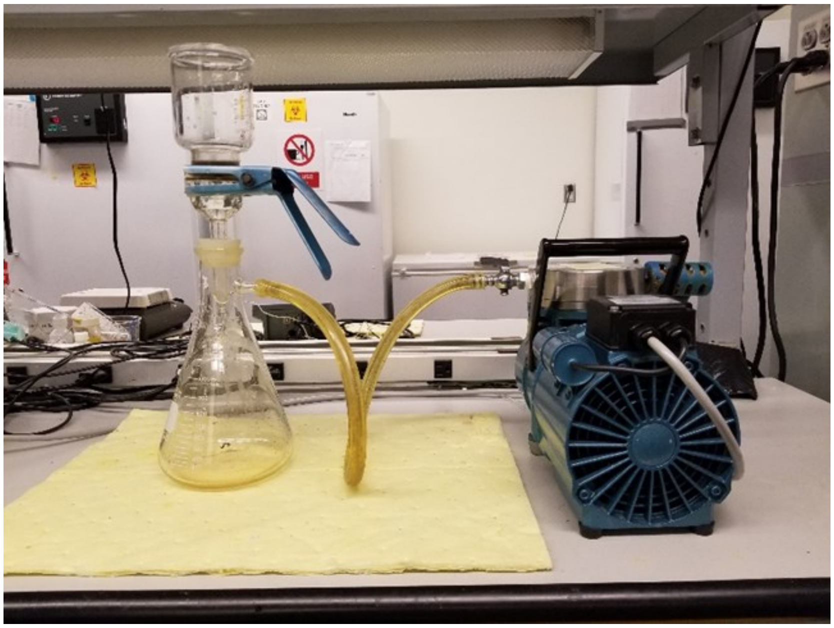

Figure 3.7 Picture of the vacuum filtration setup

3.6.3 Loss on ignition (LOI)

The organic and carbonate contents of the sediment were measured, according to ASTM D2974-00 (American Society for Testing and Material, 2000). Oven-dried sediment samples at 105 ℃, were placed in a furnace at 550 ℃ for 4 hours. After the sediments were cooled down in a desiccator, and their weights (W) were measured, loss on ignition (%) in each sample was calculated based on the following equation:

$$\mathrm{LOI}\,\% = ((W_{105^{\circ}\mathrm{C}} - W_{550^{\circ}\mathrm{C}}) \,/\, W_{105^{\circ}\mathrm{C}}) \times 100\% \qquad \text{Equation 3-2}$$

where $W_{105^{\circ}\mathrm{C}}$ is the weight of the oven dried sediment samples, and $W_{105^{\circ}\mathrm{C}}$ is the weight of cooled ash.

3.6.4 pH

The pH is one of the common parameters of water quality. The pH was measured using a Fisher Scientific pH meter model AR25, dual channel pH/Ion meter. Water sediment samples from experiments 1 and 2 were well mixed by a shaker for one hour. Samples were allowed to settle for another hour and afterwards the pH was measured using the pH meter (AR25).

3.6.5 Particle Size Distributions

The mean particle size distributions of sediment samples were analyzed using a laser scattering analyzer (HORIBA, LA-950V2) (Figure 3.8).

Figure 3.8 Laser Diffraction Particle Size Analyzer (HORIBA LA 950-V2)

This instrument measures the particle size of wet or dry samples within a range of 0.01 - 3000 µm (HORIBA, 2019). A proper amount of well mixed water-sediment samples was injected into the analyzer and the mean particle size was measured for all of samples from the experiments.

3.6.6 Total Phosphorus Concentration

Phosphorus can be found in many forms, and not all of them are directly available for aquatic creatures. As discussed in the previous chapters, orthophosphate or soluble reactive phosphate (SRP) is the only fraction of total phosphorus directly available for uptake by cyanobacteria, plants and algae. Therefore, orthophosphate was measured and addressed in most studies historically (Correll, 1998; Chorus et al., 1999).

Measurements of only SRP form would result in underestimating the total quantity of phosphorus that is likely to influence the water body (Lee et al., 1978). One should not assume that other fractions of total P such as particulate P inert in the aquatic systems since they might convert to dissolved orthophosphate under appropriate conditions. (Almroth et al., 2009)
Some scientists believe that available phosphorus load quantity lies between the soluble orthophosphate load and the total phosphate load. They provided the biologically available phosphorus calculation, which is approximately equal to the soluble orthophosphate plus 0.2 times the difference between the total phosphorus and soluble orthophosphate (Lee et al., 1978). For the sake of simplification, total phosphorus (TP) concentration has been widely used to represent total phosphate phosphorus in studies of the effects of phosphorus in the eutrophication process (Lee et al., 1978; Chorus et al., 1999)
The total phosphorus concentration was measured by Inductively Coupled Plasma Mass Spectrometry (ICP-MS, Agilent 7700x). In order to use the ICP MS for solid samples such as sediments, acid digestion is required.

In order to prepare the samples for injecting in ICP-MS and measure the total phosphorus concentration, acid digestion was necessary. A SMT protocol developed by the Standard Measurements and Testing Program of the European Commission was applied in this study (Ruban et al., 2001). Two hundred milligrams of oven-dried sediment were weighed in a porcelain crucible. Samples were calcinated at 450º C for 3 hours in the furnace. Afterwards, the samples were directed to the desiccator to cool down. Cold ash was poured into a centrifuge tube, and 20 mL of 3.5 M HCl were added to each tube with a pipette. All tubes were covered and stirred for 16 h. Finally, all samples were centrifuged at 2000g for 15 min, and the extract was collected in a centrifuge for total P analysis. After the acid digestion, to prepare the ICP-MS feed, samples were diluted with the deionized water with a specific dilution factor before injecting it to the ICP-MS. The raw results from the ICP-MS were in µg/L, and it was related to the amount of phosphorus available in the liquid feed. The following calculations were used to convert the raw data from ICP-MS into the phosphorus concentration of sediments in mg/g.

$C_p = C_f \times D_{digestion}$ Equation 3-3

$C_f = C_{ICP\text{-}MS} \times DF$ Equation 3-4

$D_{digestion} = V_t/m_{sediment}$ Equation 3-5

where:

C_p (mg/g) = Phosphorus concentration of sediment sample

C_f (µg/L) = Phosphorus concentration of the ICP-MS feed before dilution with water

$D_{digestion}$ (ml/g) = Dilution factor for the acid digestion

$C_{ICP\text{-}MS}$ (µg/L) = Phosphorus concentration of the ICP-MS feed

DF = Dilution factor used to dilute the digested liquid samples with water before injection to the ICP-MS

V_t (mL)= total volume of sample after the digestion

$m_{sediment}$ (g)= mass of the dry sediment samples used to prepare the ICP-MS feed

4 Results and Discussions

4.1 Introduction

This chapter discusses the results from a series of laboratory-scale tests of sediment samples from Lake Caron. The experiments were designed and conducted to address the lake sediment contamination issue. This chapter also discusses sediment behaviour during the settlement process and contamination distributions of particles of different grain sizes. The characteristics of 28 samples from the resuspension system were reported and discussed. During the settling process, these samples were extracted in time intervals of 5 minutes from four different heights for 30 minutes.

This chapter also investigated the feasibility and efficiency of the resuspension method for treating phosphorus contamination from the lake sediments through experimental tests and theoretical calculations. Finally, the filtration results demonstrate the effectiveness of different filters in improving the slurry quality from the resuspension system.

To verify the results, all the mentioned experiment steps are implemented in two runs and the average data is reported.

4.2 Size analysis of suspended sediments during the settlement procedure

Results from this experiment aim to develop the design strategy of the resuspension method and increase its efficiency. Removing maximum phosphorus contamination by capturing the minimum amount of sediment is desirable. For this purpose, a good understanding of sediment behaviour and sediment characteristics during the settlement process is needed. These findings are necessary to estimate the optimum time and depth to capture the most contaminated grains.

4.2.1 Loss on ignition (LOI)

The LOI was measured for the homogenized surface sediment samples of Lake Caron. The average LOI was 31.7%, which represents a high amount of organic and carbonate content. The excessive amount of dead plants and algae, which settle down regularly to the lake bottom, could explain the high organic content of Lake Caron surface sediment. Organic matter has a very fine texture and could act as strong adsorbents for contaminations such as phosphorus. The resuspension technique can remove the organic matter since they stay suspended in the water column for longer time than coarser sediment.

4.2.2 pH

The sediment samples of Lake Caron were slightly acidic, and the average pH of the sediment samples was 5.40. A parallel study on Lake Caron water quality by Veetil et al. (2019b) reported the average lake pH around 6.7.

The pH value affects the phosphate sorption rate by the sediment, and sediments in different trophic states tend to have similar trends of change. (Wang et al., 2005). According to Wu et al. (2014), P release is higher under alkaline conditions than acidic conditions from sediment to eutrophic lakes' water body. They reported the least amount of P release was under neutral pH.

Naturally, as pH increases from acidic conditions to around 7, the phosphate sorption rate increases sharply at first. By increasing the pH to 9, the phosphate sorption rate tends to decrease suddenly (Wang et al., 2005).

Accordingly, through alkalinity, the sediment of Lake Caron tends to adsorb phosphorus contamination rather than release it to the surface water.

4.2.3 Total suspended solids (TSS)

As explained in the previous chapter, sediment samples were mixed with water in the resuspension setup and 28 samples were extracted from the system from four different depths in seven different times. The TSS value is necessary for calculating the quantity of removed SPM from the resuspension setup. TSS measurements were performed on all extracted water sediment samples. The average TSS value was 7.90 g/L for initial samples extracted right after aeration from the resuspension system. After 5 minutes of settlement, a significant decrease of the TSS value with a rate of 17.17 mg/L.s occurred. The average amount of TSS after 5 minutes of settlement was 2.75 g/L. The TSS value decreased at a lower rate of 1.03 mg/L.s in the last 25 minutes of settlement. The average amount of TSS after 30 minutes of settling reached to around 1.20 g/L (Figure 4.2). It shows that the large sediment grains settle down in the first minutes and fine particles stay suspended in the system for a longer time.

The results also indicate that the TSS value decreases from the lower depths to the higher depths with an insignificant rate at the specific settling time.

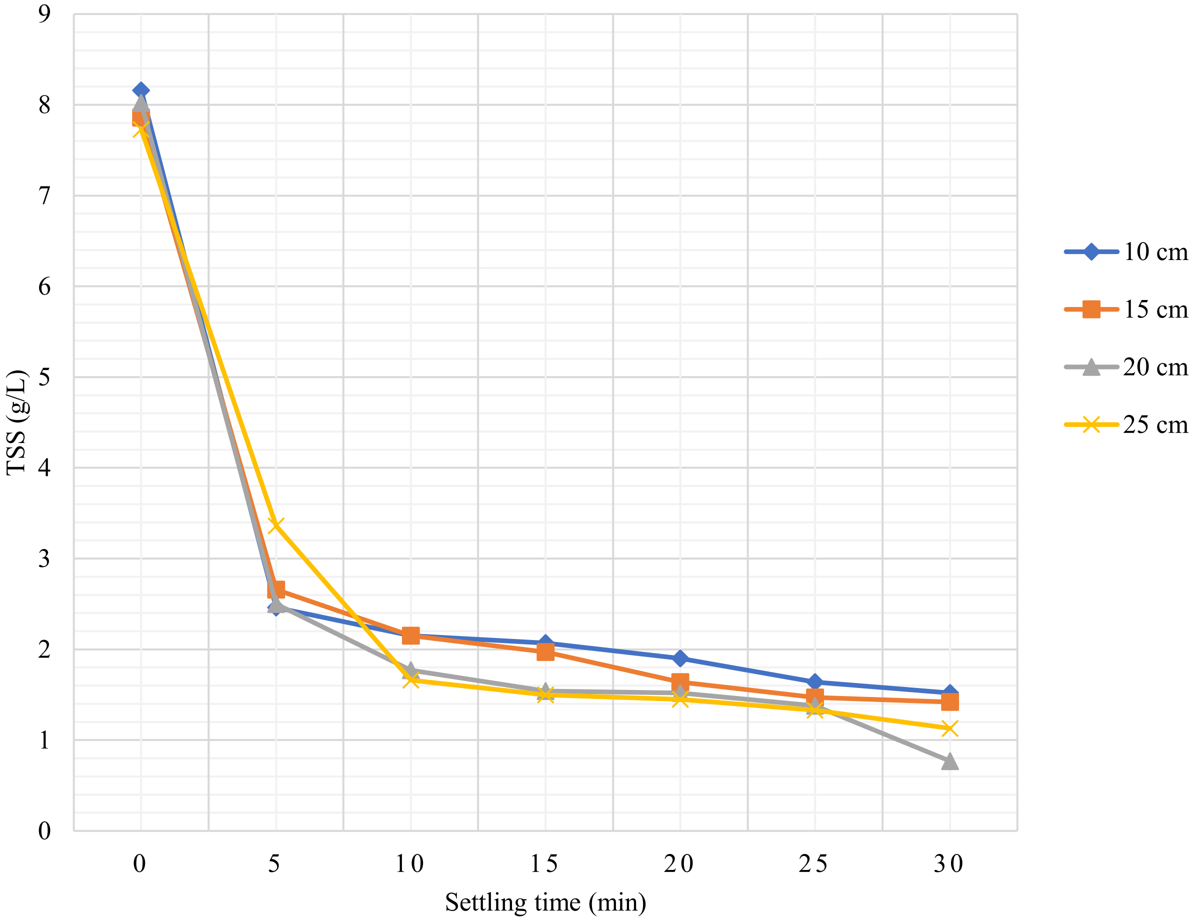

Figure 4.1 TSS values at different settling times and depths

In this experiment, the volume of removed slurry from different depths for the sampling purpose is very small compared to the slurry's total volume. The sample removal does not make any significant change in the entire water column's height. Therefore, it is logical to associate the amount of TSS value to a specific height.

However, in the second step of the resuspension method, around 30% of the slurry is removed from the system. The height of the water column would decrease by 30% after this process, respectively. Therefore, due to the high volume of the removed slurry, the average TSS of removed SPM does not necessarily equal the TSS value at the specific height where the vacuum pump is located. On the other hand, comparing the TSS value at different heights indicates that the TSS difference at different depths in the fixed settling time is negligible.

Consequently, the TSS value is more dependent on the settling time rather than the settling height. The average TSS values of the entire column at different heights were calculated and presented in Figure 4.2.

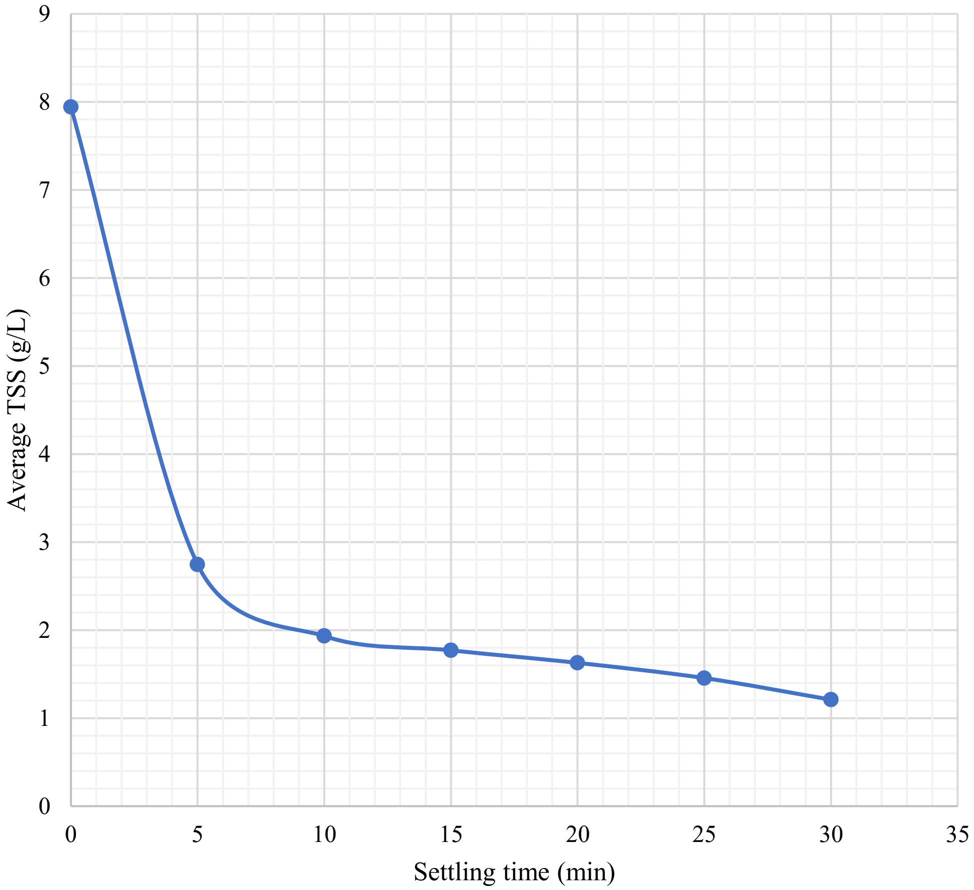

Figure 4.2 Average TSS values of the water-sediment suspension of the entire system at different settling times

4.2.4 Turbidity

The average turbidity of sediment-water samples had the initial value of around 3903 NTU. The turbidity decreased to around 1370 NTU after 30 minutes of settlement (Figure 4.3).

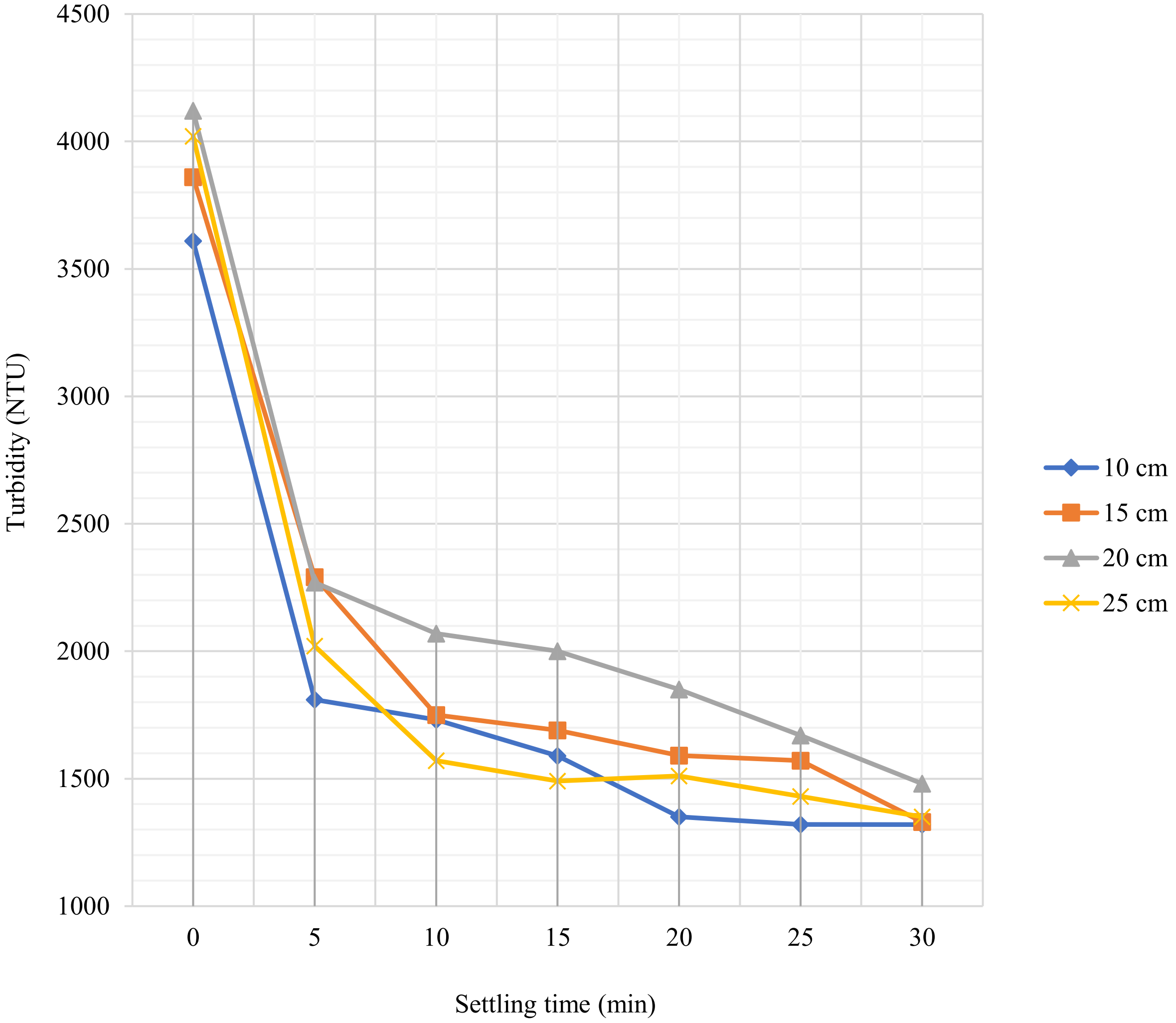

Figure 4.3 Turbidity of the water sediment suspension extracted at different times and different depths

In the first 5 minutes of the settling, a significant decrease was observed in the turbidity value since coarser sediments settled down rapidly. The average turbidity decrease of coarser sediments in the first five minutes was around 45% of the initial turbidity. The turbidity decreased with a lower rate in the last 25 minutes of the settlement procedure. The settling velocity of fine sediment is not high, and the sediment tends to stay suspended in the water column.

4.2.5 Particle Size analysisff

The mean particle size of the initial water-sediment samples extracted immediately after aeration was around 106 µm (Figure 4.4). The particle size distribution of the initial sample measured as 34.1% clay, 58.6% silt and 7.3% sand, which was determined as poorly graded. The mean particle size value decreased over time, at a specific height during the settling process. The mean particle size of sediments decreased to around 46.3 µm after 5 minutes of settlement, which illustrates the fast reduction of particle size in the first few minutes of settlement. After 30 minutes, the mean particle size reached to around 14 µm.

Pursuing the results at the settling time of 20 min to 30 min illustrates that the mean particle size of sediment grains indicates a sort of equilibrium when it reaches the size of around 14 µm. Therefore, the particles with the mean size of around 14 µm remain longer in the mixture. These particles approximately were found at the height of 25 cm after 20 to 30 minutes of settlement and at the height of 20 cm after 25 to 30 minutes of settlement.

Sediment particle size is an essential parameter to analyze before considering the resuspension technique as a solution for the sediment treatment for any lake. The resuspension method is based on removing the fine effective sediment grains. Therefore, it is important that the predestinated lake's surface sediment naturally has a fine texture and contains a high percentage of fine particles.

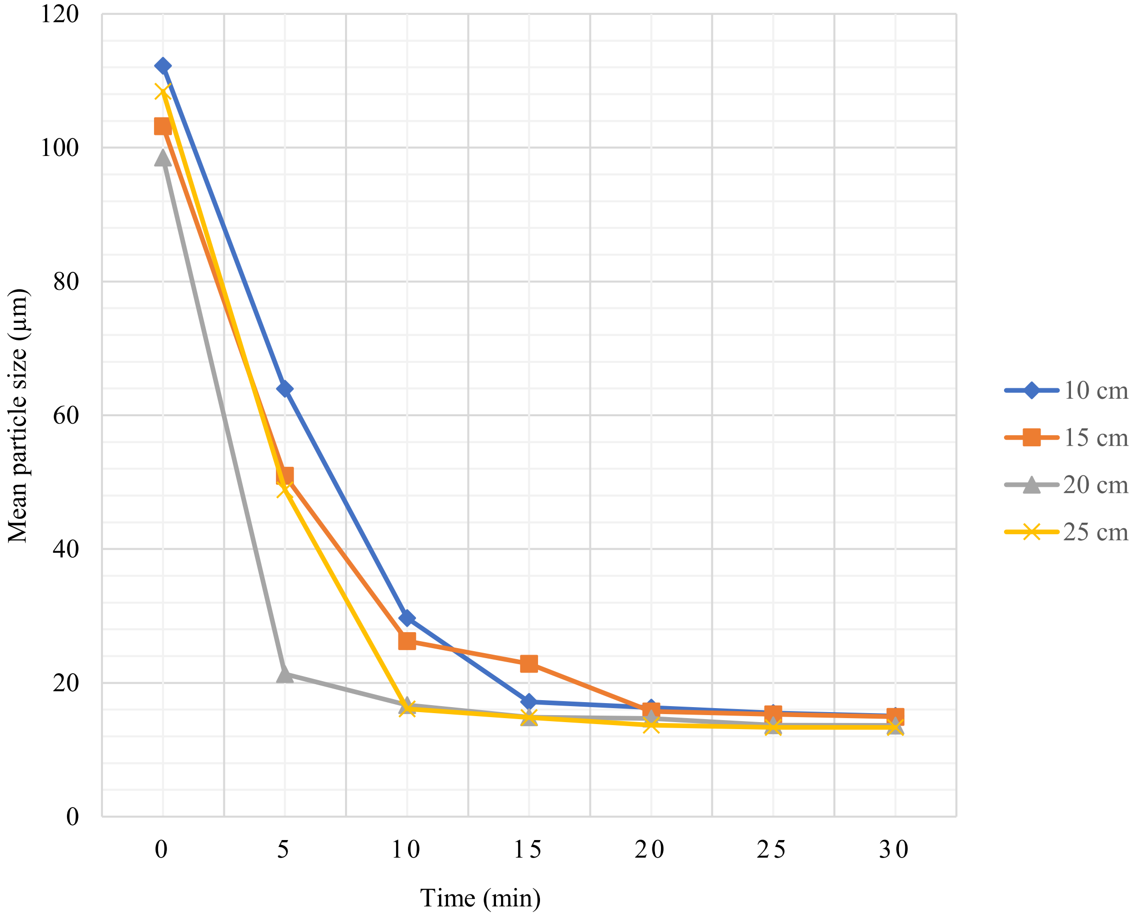

Figure 4.4 Mean particle size of water-sediment samples at different heights and different times

The sediment particle size range also plays an essential role in the resuspension technique's efficiency and feasibility. Lake sediments with a wide range of particle sizes are desirable for this technique. If the sediment particles have a narrow size range, the contamination distribution is expected to be uniform on the sediment particles, and the resuspension technique is not recommended. In this case, all sediment particles are equally contaminated approximately, and the approach of removing the most contaminated grains is not practical.

Therefore, the sediment particle size analysis is a critical factor in assessing the contamination sites and selecting a suitable management strategy for remediation.

4.2.6 Total Phosphorus Concentration

The total phosphorus concentration of all 28 extracted samples was measured using ICP-MS. The values of the total phosphorus concentration presented in Figure 4.5. The results of this study are reported as total phosphorus, and it should not be confused with the term of total phosphate. In order to use the ICP-MS for solid sediment samples, acid digestion was required. The acid dilution factor applied in this experiment was 100 ml/g. After the acid digestion, to prepare the ICP-MS feed, samples were diluted with the deionized water with a specific dilution factor. The raw results from the ICP-MS were in µg/L, and it was related to the amount of phosphorus available in the liquid feed. The phosphorus concentration of sediments was then calculated and is reported in Figure 4.5.

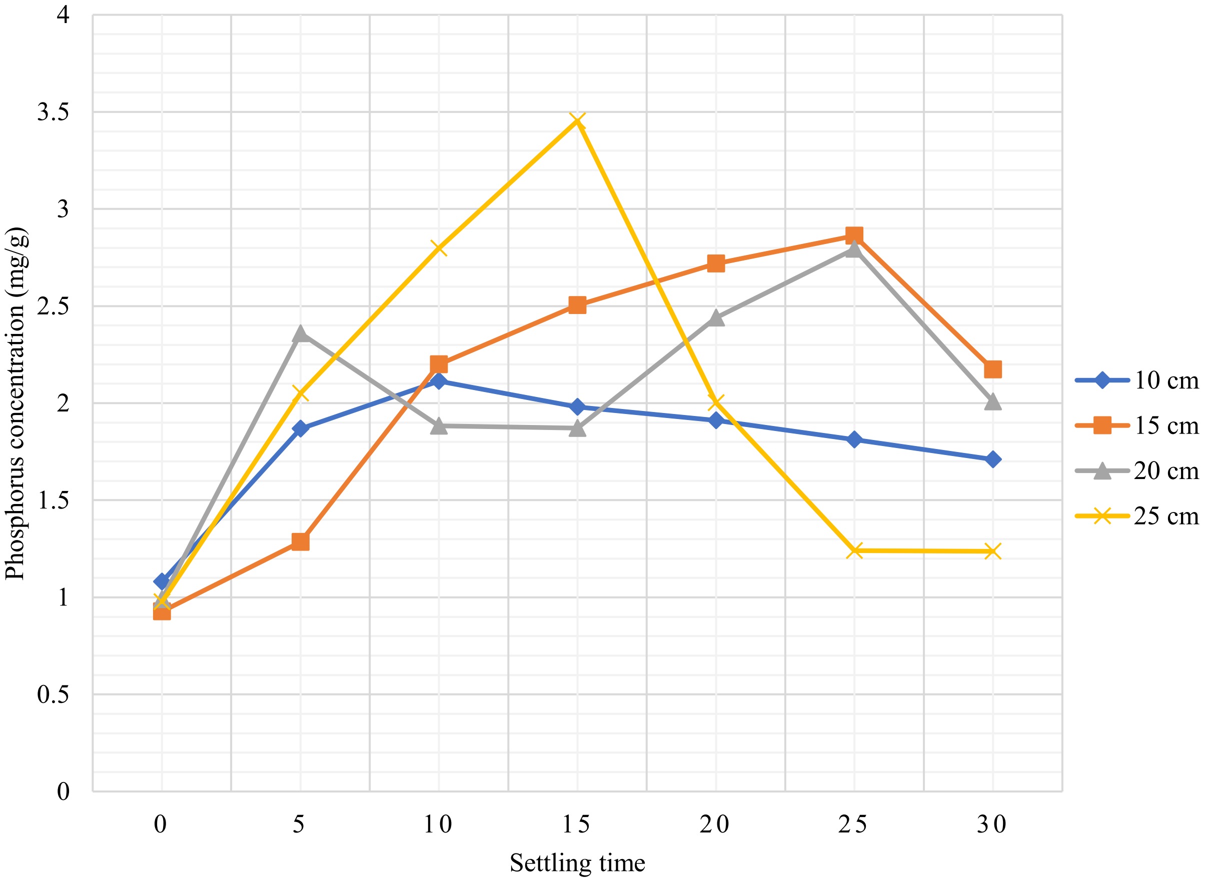

Figure 4.5 Phosphorus concentration on sediment (mg/g) in different times and different settling depths

The phosphorus concentration was measured for the initial samples extracted from the system immediately after aeration. The average amount of phosphorus concentration of the initial samples was around 1 mg/g. After 5 minutes of settlement, phosphorus concentration on suspended sediment particles increased to 1.868, 1.284, 2.359 and 2.051 at depths of 10, 15, 20 and 25 cm, respectively. These increases continued for 10 to 25 minutes of settlement. Phosphorus concentration then started to decrease as the settling process continued. After 30 minutes of settlement, the phosphorus concentration on suspended sediment particles decreased to 1.710, 2.173, 2.008 and 1.238 mg/g at the depths of 10. 15, 20 and 25 cm, respectively.

In summary, the amount of phosphorus concentration of sediment particles initially increased during the settlement, and then decreased as the settling process continued. An explanation for the initial increase in the phosphorus concentration could be coarser sediment particles' settlement, which increases the refined effective grains' percentage in the total suspended solids. At some point, by settling more sediment grains, the phosphorus concentration starts to decrease. It declares that although fine particles absorb more contamination due to their high surface area in general, their capability for carrying contamination could moderately decrease for very fine grains. Therefore, the fine effective particles are mostly suspended in the water-sediment slurry for around 15 to 25 minutes. The phosphorus concentration on the total suspended sediment mass starts to decrease afterwards as the effective particles settle down.

At this point, we only detected the time and height in which the most effective particles could be found in the water column and during the settlement process. These particles could be mostly found at a depth of 25 cm after around 10 to 20 minutes of settlement, and at a depth of 15 cm after 20 to 25 minutes of settlement.

The size of the effective particles and their concentration in the slurry is also essential.
The phosphorus concentration data was combined with the particle size distribution data to realize what particle sizes are associated with the most phosphorus concentration. The association of P absorption with different particle sizes is essential and has been investigated in many studies (Selig, 2003). A combination of the data from the phosphorus concentration and

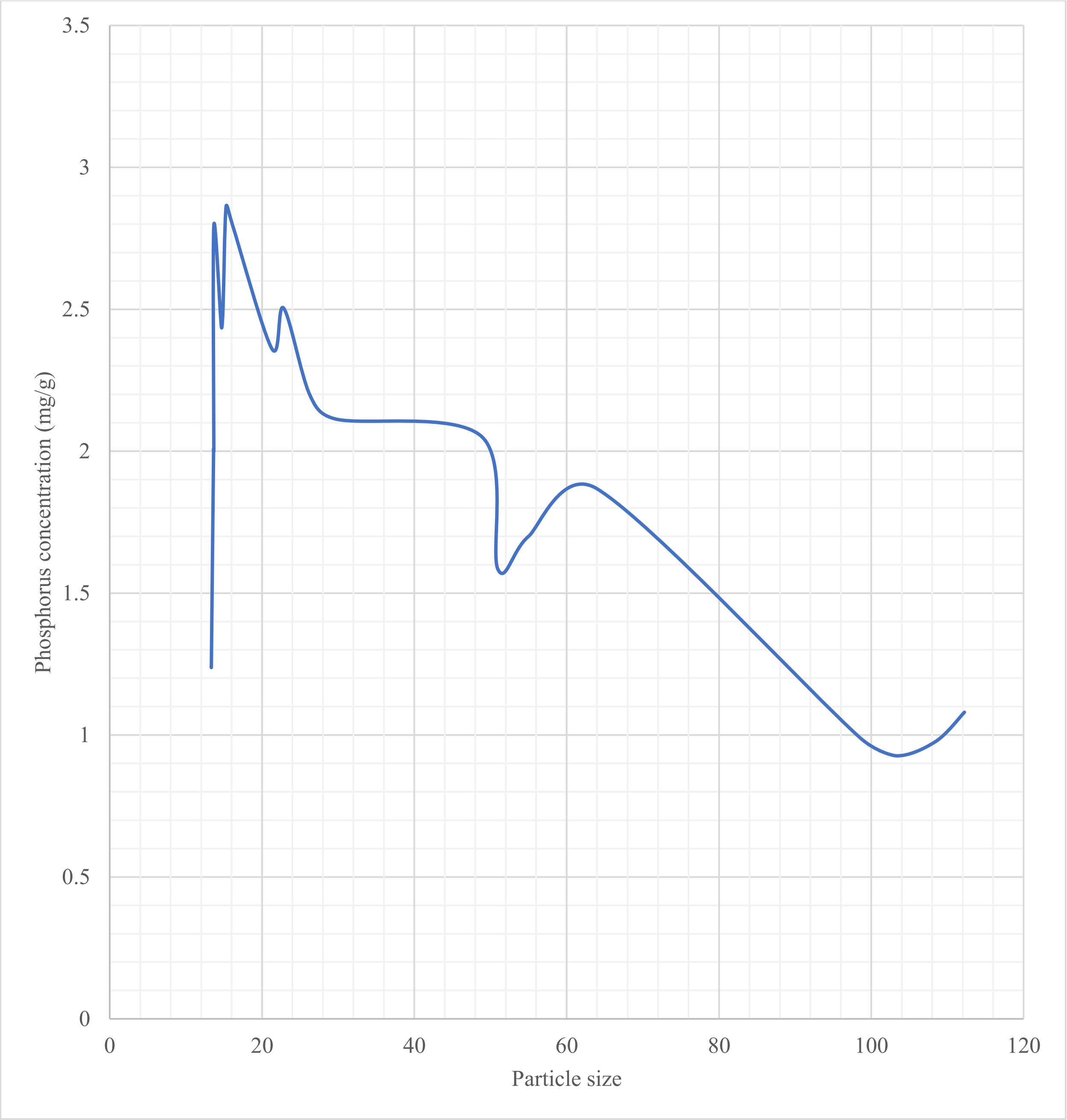

Figure 4.6 Phosphorus concentration (mg/g) on different particle sizes (µm)

mean particle size analysis was used to detect the most effective sediment grains Figure 4.6. The results show that the particles with the size of around 16 µm to 30 µm carry the highest concentration of phosphorus on them. The phosphorus concentration started to decrease for the sediment particles of 14 µm and smaller compared to those particles in the range of 16 µm to 30 µm. Additionally, the concentration of phosphorus is lower for the particles larger than 50 µm. The phosphorus concentration on different particle sizes depends on the particle size range, particle size distribution of the sediment samples and the total level of phosphorus contamination. These amounts could be different for the sediment samples with different characteristics and/or different pollution levels from Lake Caron. The phosphorus distribution results in different sediment particle sizes are applicable to other lakes' sediments with the characteristics similar to the Lake Caron.

So far, the most contaminated grain sizes and the best time and height to capture them were determined. However, this information is not sufficient for designing an optimal resuspension strategy. Removing the sufficient amount of contaminated sediment grains from the system is essential for decreasing the sediment's total contamination level.

The settling time and depth in which the phosphorus concentration on sediment particles is the highest might not be the optimal condition for pumping out the slurry in the second step of the resuspension technique. The amount of the effective particle should also be sufficient to decrease the system's total contamination to a desirable level. The TSS value plays an important role in calculating the total mass of phosphorus contamination removal from the system, and it is continuously decreasing during the settling process. The phosphorus concentration in the slurry was calculated as follows:

$$C_p\text{ (water column)} = \text{TSS} \times C_p\text{ (Sediment particles)} \qquad \text{Equation 4-1}$$

where:

C_p (water column) (mg/L): the phosphorus concentration in the slurry

C_p (Sediment particles) (mg/g): the phosphorus concentration on sediments

TSS (g/L): total suspended solids of the slurry

To calculate the total amount of phosphorus removal, the average amount of phosphorus contamination in the entire water column is needed. For that purpose, the phosphorus concentration at different heights calculated using Equation 4-1 for four different heights, and then the average of four values was considered as the phosphorus concentration of the entire column.

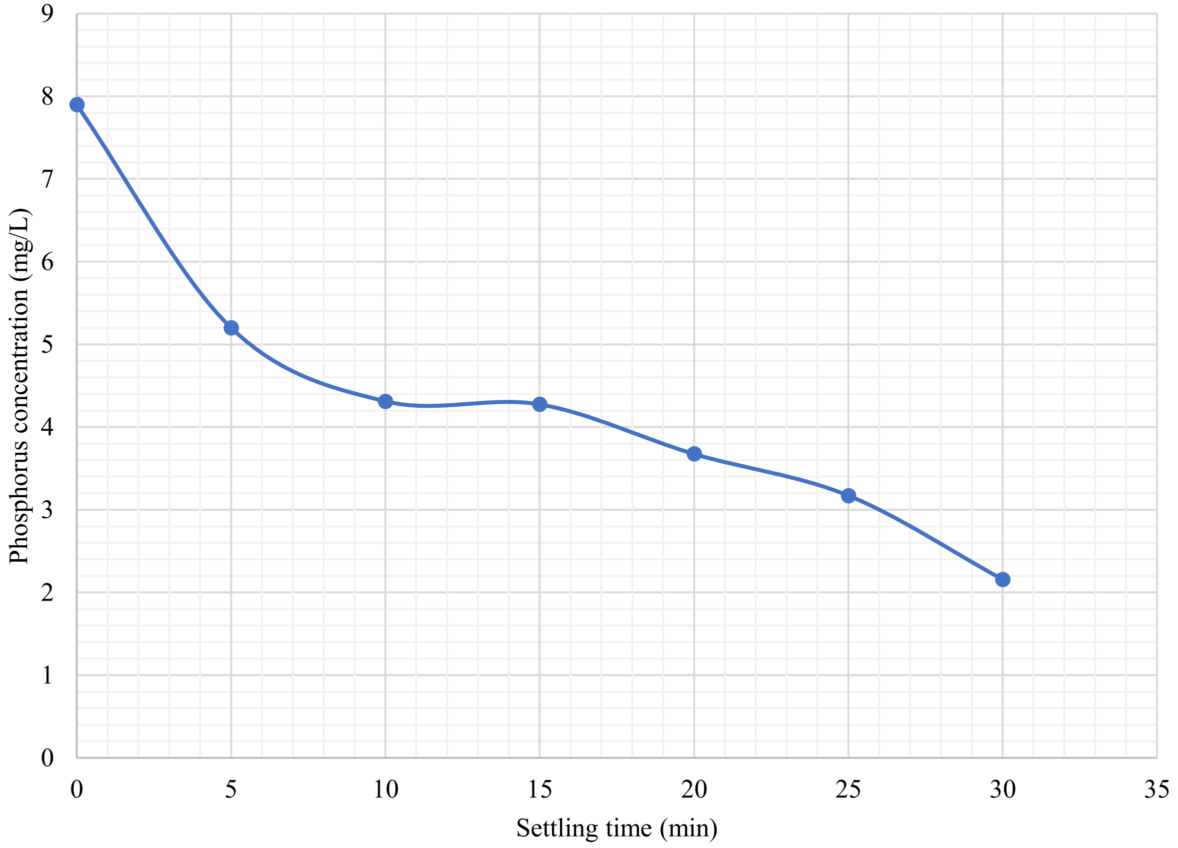

Figure 4.7 Phosphorus concentration of entire column in different settling times

The average concentration of phosphorus in the well-mixed water-sediment suspension was around 7.90 mg/L, and it continuously decreased to around 2.17 mg/L. The phosphorus concentration decreased with a higher rate in the first 5 minutes of settlement. A high mass of sediment is settling down and removing from the suspension in the first minutes of settlement. This rate is 48% lower than the total sediment settlement rate since the removed particles are less contaminated due to their larger particle size. After 20 minutes of settlement, the phosphorus concentration decreased to 3.67 mg/L, which means 46% of the total phosphorus contamination stayed suspended in the water column after 20 minutes of settlement. The amount of suspended sediments after 20 minutes is around 21% of the total sediment's mass from the TSS results. Accordingly, 46% percent of the total sediment contamination is accumulated on only 21% of the fine effective particles. By targeting and removing these suspended effective particles, we would be able to eliminate a higher amount of contamination, which is the objective of the resuspension technique.

4.2.7 Analytical calculations

In order to decrease the phosphorus concentration to a desirable level, an appropriate amount of phosphorus should be removed from the system. Several parameters affect the phosphorus removal level. Designing the optimal remediation strategy requires a good understanding of all effective factors.

The volume of removed slurry has a direct effect on the phosphorus removal amount. Pumping out the more volume of contaminated slurry would result in a higher contamination removal. This slurry will be directed to the filtration system afterwards. The volume of removed slurry is usually predetermined as the filtration system has a limited capacity. The maximum capacity of the filtration system can be used in the design calculations.

The difference between the initial and desirable amount of sediment contamination is another key factor in the strategic calculations. These two parameters and, consequently, their differences are also predetermined in most of the cases. The initial level of contamination is relevant to the nature of the lake sediments. The final desirable phosphorus level is also specified by considering many factors, such as the lake's application and its aquatic life demands.

The phosphorus concentration in the slurry is another effective factor. As discussed earlier, this value is decreasing during the settling process. The reason is that the contaminated suspended sediment particles are settling down continuously, but decreasing by time.

The third-degree polynomial equation fitted to the experimental data of phosphorus concentration of the slurry at different settling times.

$$C_p\,(\text{water column}) = p_1 \times t^3 + p_2 \times t^2 + p_3 \times t + p_4 \qquad \text{Equation 4-2}$$

$p_1 = -0.0006829$

$p_2 = 0.03512$

$p_3 = -0.6298$

$p_4 = 7.794$

Where C_p (water column) is the concentration of phosphorus in the slurry in mg/L, and t is the settling time in minutes.

While the volume percentage of removed slurry and phosphorus removal percentage are defined, the minimum amount of C_p (water column) could be calculated through the following equations.

$$\mathrm{C_p\ (water\ column)} = \frac{\mathrm{C_p\ (sediment)} \times \mathrm{m_{total}} \times \mathrm{Contamination\ decrease\%}}{\mathrm{V_{total}} \times \mathrm{Slurry\ removal\%}} \qquad \text{Equation 4-3}$$

$$\mathrm{Contamination\ decrease\%} = \frac{\mathrm{C_p\ (sediment)} - \mathrm{C_p\ desirable\ (sediment)}}{\mathrm{C_p\ (sediment)}} \qquad \text{Equation 4-4}$$

where:

C_p (water column) (mg/L): minimum phosphorus concentration in order to decrease the phosphorus level to the specific desirable amount by removing a defined amount of slurry.

C_p (sediment) (mg/g): the initial phosphorus concentration in the sediment before treatment

Slurry removal%: the percentage of slurry pumping out over the resuspension test

C_p desirable (sediment) (mg/g): the desirable sediment phosphorus concentration after the treatment.

m_{total} (g): the total mass of sediment particles in the system

V_{total} (L): the total volume of slurry in the resuspension column

Our specific system's optimal settling time could be calculated using the minimum phosphorus concentration value and Equation 4-5. Pumping out the slurry in the settling times less than the optimal settling time would remove more contamination from the entire system, while all other parameters stay constant. Despite this, we have to note that TSS value increases and more sediment mass would be removed from the system for the same slurry volume by decreasing the settling time. However, this is not favourable since the resuspension technique approach decreases contamination by minimum sediment removal. Pumping out sediment after the optimal time is not also favourable as the removed SPM might not be sufficient to decrease the total contamination to the desirable level. Considering 15% desirable phosphorus concentration decrease in sediments by 30% slurry removal, the optimum settling time and amount of removed sediments were calculated for this experiment's sediment samples.

Using Equation 4-3, the optimum phosphorus concentration was calculated as 3.97 mg/L. Solving Equation 4-2 for the settling time determines the optimum settling time, while the optimal concentration of phosphorus in the slurry is known. The optimal settling time was calculated as 14 minutes. TSS value after 14 minutes of settlement is around 1.77 g/L, and the initial TSS is 7.94 g/L; therefore, pumping out 30% of the slurry after 14 minutes of the settlement will remove about 7% of the total suspended sediments. In conclusion, the optimal settling time to decrease the phosphorus concentration for 15% is calculated as 14 minutes. The theoretical calculations indicated that by removing just 7% of the SPM from the reactor, phosphorus concentration decreases by 15%.

4.3 Evaluating the feasibility and efficiency of the resuspension technique to treat the phosphorus contamination of lake sediments

To validate the accuracy of experiment one calculations and evaluate the feasibility of the resuspension technique, a set of lab-scale experiments was performed on the sediments of Lake Caron and the entire process of the resuspension technique was simulated in the laboratory.

Lake's sediment samples were mixed with the tap water in the resuspension system with a ratio of 1:10 (vol./vol.). The total height of sediment and water mixture in the system was 40 cm. The total volume was 11.4 L. Aeration occurred for 2 hours to homogenize the water-sediment mixture. A 50 ml sample of well-mixed water and sediment suspension was collected from the system right after the aeration process. After 15 minutes of settlement, 30% of the slurry was collected from the resuspension system from the height of 20 cm. Several analytical experiments were performed on the removed slurry. The quality and quantity of removed SPM can have a significant effect on resuspension efficiency. **Error! Reference source not found.** illustrates the characteristics of the water-sediment samples before and after the settlement process.

The TSS of water-sediment suspension was 8.38 g/L immediately after the aeration process. It decreased to 2.10 g/L after the 15 minutes settlement of coarser sediments. Around 25% of the sediment particles stayed suspended in the water column. It is essential to estimate the amount of sediment that was removed from the resuspension system. By removing 30% of the slurry after the settlement procedure, approximately 7.5% of the total sediment particles' mass was removed from the system.

The phosphorus concentration in the water column before the settlement of coarser sediments was 10.99 mg/L, and it decreased to 6.57 mg/L after the settling process. These results indicate that although about 75% of the sediment settled down in the first 15 minutes, around 60% of the phosphorus contamination stayed suspended in the water column.

In other words, 60% percent of the total phosphorus contamination is absorbed by only 25% of the fine sediment particles. This shows the importance of particle size analysis as it leads to the proper management strategy for remediation of contaminated sediments.

Table 4.1 Characteristics of the water-sediment samples before and after the settlement process

	TSS (g/L)	**Mean Particle Size (µm)**	**Turbidity (NTU)**	**TP on sediments (mg/g)**	**TP in water (mg/L)**
Initial sediment-water suspension	8.36	98	3810	1.31	10.99
Sediment-water suspension after settlement process (SPM)	2.10	30	1420	3.13	6.57

The suspended sediment particles' total phosphorus concentration also increased from 1.31 mg/g to 3.13 mg/g after the settlement. It emphasizes that fine particles carries more contamination compared to the coarser sediment particles. The phosphorus removal value was calculated using the following equation.

$$\text{Removal\_Percentage} = \text{Cp (SPM)} \times \text{SPM removed (\%)} / \text{Total Cp} \qquad \text{Equation 4-5}$$

Where C_p (SPM) is the total phosphorus concentration of the SPM in the water column in mg/L, SPM removed (%) is the volumetric percentage for the SPM removal, which is 30% in this experiment, and total Cp is the phosphorus concentration of initial water-sediment suspension after aeration in the water column in mg/L.

Removing 7.5% of the fine contaminated sediments by weight decreased the phosphorus concentration by around 18%. These results are relatively similar to the theoretical results from experiment one and indicate the accuracy of the theoretical calculations.

The removed slurry was then directed to the filtration system (figure 4.8). One layer of the filter was used in the filtration setup. This experiment was repeated six times for six different geotextiles with different pore sizes. The geotextile pore sizes varied from 13.33 µm to around 70 µm. The filters with smaller pore sizes were more effective for decreasing the turbidity and the phosphorus concentration in the suspension. Table 4.2 illustrates the characteristics of the slurry after filtration.

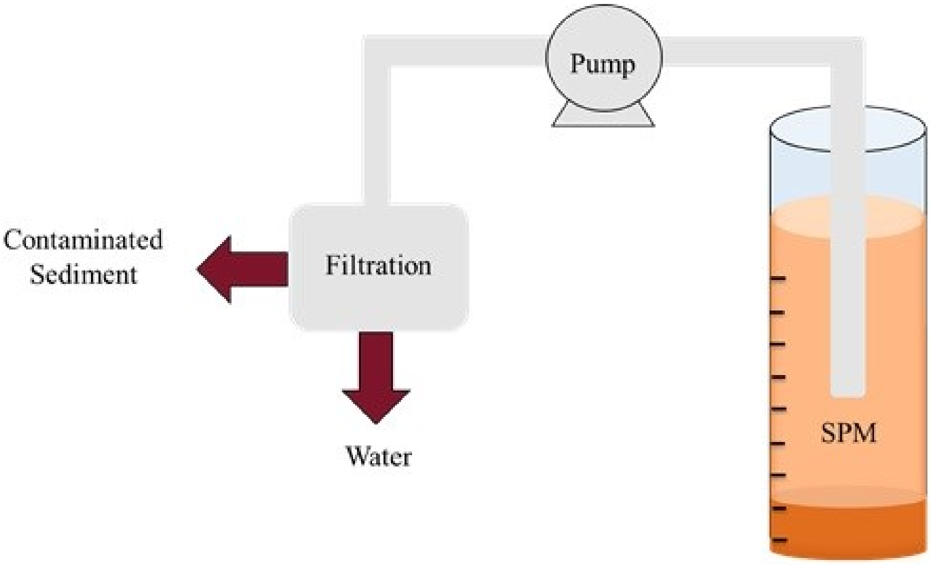

Figure 4.8 Resuspension column along with filtration setup

Table 4.2 Characteristics of the slurry after filtration

filter number	Filter name	Pore size (µm)	Turbidity (NTU)	Phosphorus concentration (mg/L)	TSS (g/L)
1	FIN01933	13.33	286	0.027	0.310
2	FIN02144	16.44	519	0.280	0.420
3	FIN02152	18.07	630	0.260	0.510
4	FIN02870	19	717	0.342	0.500
5	FIN01808	60	945	0.760	0.720
6	FIN01584	70	1130	0.905	0.730

The turbidity decreased from 1420 NTU to 286 NTU, 519 NTU and 630 NTU for the filters with the pore sizes of 13.33 µm, 16.44 µm and 18.07 µm, respectively. For filter #6, the turbidity had the minor decrease to 1130 NTU, since the mean particle size of the sediment grains in the slurry was 30 µm, and most of them were able to pass the filter with a pore size of 70 µm.

Total phosphorus concentration decreased in the slurry discharged from the filtration setup. The phosphorus concentration decreased from 6.57 mg/L to 0.027 mg/L after passing through a filter layer. According to the Environment and Climate Change Canada (2019), the water body's allowable phosphorus level is 0.03 mg/L. With a phosphorus concentration of 0.027 mg/L, the slurry is allowed to be discharged back into the lake in terms of phosphorus concentration. No further phosphorus treatment procedures are needed in this case. However, the slurry's other characteristics should be checked before discharging it back to the lake's water body. The phosphorus concentration was 0.760, and 0.905 mg/L for the slurry passed through filters #5 and #6, respectively. The mentioned filters were not sufficient for filtration purposes considering the fine nature of lake sediments. They might be beneficial to use as an upper layer of multi-layer filtration systems, but they cannot reduce the phosphorus concentration to the acceptable level alone.

The total mass of sediment captured by each filter calculated as $m_S = m_B - m_A$, while m_S is the total mass of sediment, and m_A and m_B are the mass of dry filter and the mass of polluted filter after drying, respectively.

The total suspended solid concentration of the slurry after passing the filters calculated by having m_S and through the following equation:

$$TSS_2 = (TSS_1 \times V_t - m_S)/V_t \qquad \text{Equation 4-6}$$

Where TSS_1 and TSS_2 are the total suspended solid concentration before and after passing through the filter, and V_t is the total volume of the sample. TSS values decreased from 2.10 g/L to 0.31, 0.42, 0.510, 0.50, 0.72 and 0.73 g/L for the filters #1 to #6 respectively. Canadian Council of Ministers of the Environment (CCME, 2002) declares the allowable amount of TSS as 0.025 g/L for the surface water bodies. The slurry passed through the filtration setup used in this experiment could not meet these criteria. More layers of filters and multiple filtration operations are needed to fulfill the acceptable TSS level.

The initial turbidity after filtration was 1420 NTU and it decreased to 286, 519, 630, 717, 945 and 1130 NTU for the filters #1 to #6, respectively. It is crucial to choose the appropriate filters in this step to capture fine particles as they carry a higher amount of contamination.

Environment and Climate Change Canada (2019) indicates that the allowable amount of turbidity as 10 NTU for the surface water. The turbidity of the slurry after filtration does not meet these criteria.

5 Conclusions

This study's main focus was developing the resuspension technique to address the sediment contamination of shallow water bodies (particularly shallow lakes). The resuspension technique's main objective is removing a small percentage of sediment (e.g. 7% in this study) containing the largest concentration of contaminants.

As the first step to fulfill this purpose, influential parameters were identified by assessing several characteristics of surface sediments of Lake Caron. The average pH of the lake sediment samples was acidic and around 5.40. The amount of LOI was about 31.7%. It represents the high amount of organic and carbonate contents in the sediment. Organic compounds could act as a strong adsorbent for phosphorus. The resuspension technique can remove the organic compounds as they have a fine texture and they can stay suspended in the system after the aeration. The amount of total phosphorus concentration of sediment was 1.31 mg/g.

Sediment behaviour during the settling process following resuspension was investigated in this study. For this purpose, several samples were extracted during the settling process in time intervals of 5 minutes for 30 minutes from four different heights. The TSS amount decreased with a high rate of mg/L.s during the first minutes of settlement. The initial TSS value of well-mixed sediment and water slurry was 7.9 g/L. This amount contains both coarse and fine sediment particles. TSS decreased to 2.75 g/L after 5 minutes of settlement. The amount of TSS in the last 15 minutes of the settlement was relatively uniform and was about 1.43 g/L.

The turbidity of water-sediment slurry during the settling process shows a similar trend as the TSS. The average turbidity decreased from 3903 NTU to 2108 NTU after the first 5 minutes of settlement. The decrease continued to around 1370 NTU after 30 minutes of settlement. The decreasing rate was about 29 times higher in the first 5 minutes comparing to the last 25 minutes of settlement.

The size analysis of suspended sediments during the settlement procedure indicated that the initial mean particle size was around 106 µm. Coarser sediment grains settled down with a high settling velocity. Therefore, the mean particle size of sediment gains decreased to 46.3 µm after 5 minutes of settlement. This reduction continued with a lower rate to around 14 µm after 30 minutes. The particles with a size of about 14 µm stayed suspended in the water column for a more extended period.

Phosphorus concentration on sediment was measured for all samples. The values were combined with particle size analysis to obtain the phosphorus distribution on different particle sizes.

The results show that the particles with the size of around 16 µm to 30 µm carry the higher concentration of phosphorus on them. Sediment particle with the particle size between the mentioned range was mostly found at a depth of 25 cm after around 10 to 20 minutes of settlement, and at a depth of 20 cm after 15 to 25 minutes of settlement in the water column.

This shows the importance of particle size analysis and its crucial role in assessing the contamination sites and then choosing the proper management strategy for remediation. These results can be used for designing other lakes' treatment strategies with similar sediment characteristics. The tests should be repeated if the size range, the size distribution or the contamination level of the specified lake sediments were very different from the sediment samples used in this study.

It is essential to take the TSS amount into account while designing the best strategy to target and pump out the effective sediment grains to achieve the desirable phosphorus level after the treatment. TSS value should be considered in the calculations to estimate the total phosphorus removal.

Considering that the TSS amount continuously decreases by the time during the settling process, the optimal settling time for pumping out the SPM from the resuspension system is estimated by applying this theory on Lake Caron's sediment characteristics.

The desirable phosphorus removal from the sediment was considered as 15% of the initial level, and the slurry removal was considered as 30% of the total volume of the slurry. The optimum settling time was calculated as 14 minutes. Based on the theoretical calculations, sediment removal for decreasing the total contamination of sediment for 15% estimated to be around 7% of the total sediments.

In the second phase of this study, the theoretical calculations' accuracy was validated. These experiments were beneficial for evaluating the feasibility and general performance of the resuspension technique.

For this purpose, a series of lab-scale experiments of the resuspension technique's entire steps were simulated and performed. The influential factors such as sediment type, water-sediment ratio (or the initial TSS) and the removed volume of SPM were similar to the first experiment. Similar to the optimal settling time calculation in experiment one, the settling time of 15 minutes was considered in the experiment.

The quality and characteristics of the sediment were measured before and after the remediation.

The resuspension technique successfully reduced the total concentration of phosphorus contaminants in the lake sediments. The 18% contamination decreased observed after removing only 7.5% of the suspended sediments. Fine sediment particles were suspended and removed over the resuspension test. Although only 25% of the sediment particles remained suspended, they carried about 60% of the total contamination. The amount of phosphorus contamination carried by the suspended sediment particles was about 2.4 times higher than the bulk sediment.

The removed slurry from the resuspension technique contains the SPM with high phosphorus contamination. The amount removed SPM is relative to the filtration system capacity. Due to the high contamination level of the removed slurry, appropriate filtration is necessary.

In the last step of the resuspension technique, the removed slurry from step 2 was directed to the filtration system. The effectiveness of different filters with a size range of 13.3 to 70 µm was investigated. Among different filters used in this step, filter #1 with the pore size of 13.33 µm showed the best results. Various characteristics of the slurry were measured before and after filtration. Using one layer of filter #1 the turbidity decreased from 1420 NTU to 286 NTU, the phosphorus concentration decreased from 6.57 mg/L to 0.027 mg/L, and the TSS values decreased from 2.10 gr/L to 0.31 g/L. The slurry had a minor improvement after passing the last filter with the pore size of 70 µm, as this filter was not capable of capturing the SPM. In general, filters with smaller pore sizes were more effective for increasing the removed slurry quality.

In conclusion, the resuspension technique proved to be effective in treating the lake sediments. Unlike other in-situ methods, the resuspension technique decreases the total contamination level of sediments. Consequently, in the case of potential sediment disturbances by natural events or human activities, less contamination would be released into the water body.

A positive side effect of the resuspension technique is the DO increase in the system by using air jets to resuspend the sediments in the first step. The DO plays an essential role in the determination of water and sediment quality.

The resuspension technique could act as a good alternative for the dredging. In the case of the unavoidable dredging, the resuspension technique could be beneficial for the sediment pre-treatment. The dredging could be performed safer since the risk of mobility and availability of contaminants would be decreased. Besides, the dredging operation would be cost-effective because the costs of dewatering, transportation and disposal of contaminated sediments in landfills would be minimized.

5.1 Recommendations for future work

The resuspension technique is a new method and has a lot of potential for the further studies. The following suggestions could be beneficial to expand this study.

- Performing resuspension technique on different types of sediments with different size distributions and various contamination levels is recommended. The resuspension technique is expected to offer stronger performance on more contaminated sediments. On the other hand, for sediments with a wide size distribution a more significant distribution of contaminants is expected between SPM and bulk sediment. These considerations are essential when it comes to the strategic lake management and choosing the best treatment option. Considering the specific characteristics of the intended lake is necessary for the decision makers.

- The results of this work are based on laboratory-scale experiments. Scaling up computations by using the laboratory data presented in this study and other similar studies, is recommended. Pilot tests are essential as the next step to evaluate and develop the resuspension technique since laboratory-scale experiments cannot reflect all complications and challenges of the method.

- To work towards commercializing this technology, more effort is needed toward designing the large scale system including operation procedures and the necessary equipment. In addition to science and technology aspects, the feasibility of the resuspension method should be evaluated in the economic aspect.

- Despite the fact that the resuspension technique generally causes less damage to aquatic life compared to other treatment methods such as sand capping and dredging, further studies still are needed on the potential adverse effects on sensitive species. The toxicological responses of organisms to resuspended contaminated sediments should be considered while designing the magnitude and duration of the aeration step of the resuspension process.

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