Experimental studies on the erodibility and transport behaviour of dreissenid mussel deposits in an annular flume

by

Kelly McLean

A thesis
presented to the University of Waterloo
in fulfilment of the
thesis requirement for the degree of
Master of Science
in
Geography

Waterloo, Ontario, Canada, 2011

AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Dreissenid mussels alter particle transport dynamics in the near shore environment of the Great Lakes by intercepting, retaining and recycling suspended solids that might otherwise be exported to the offshore environment (Hecky et al., 2004). Particulate materials filtered from the water column by dreissenids are subsequently released as either feces or pseudofeces (Walz, 1978). This bio-transformation process alters the nature (grain size distribution, settling velocity and density) and transport properties (critical shear stress for erosion, erosion rates and bed stability) of particulate matter in surficial sediments. While knowledge of the transport characteristics of this material is required to refine particle transport dynamics and energy flow models in the Great Lakes, few studies have been specifically conducted to directly quantify these processes. An annular flume was used to determine the bed stability, rate of erosion and critical shear stress for erosion of dreissenid biodeposits. Materials studied in the flume consisted of 1) a combination of biodeposits and surface sediments collected from dreissenid beds and 2) biodeposits harvested in a weir box with dreissenids. The results show that erosion characteristics and sediment transport properties were strongly influenced by bed age; however particle sizes did not increase in the presence of mussels as originally speculated. Bed stability increased after 7 days, with a τ_{crit} of 0.26 Pa compared to the 2 and 14 day consolidation periods (τ_{crit}= 0.13 and 0.15 Pa respectively). In 2010, following a 2 day consolidation period, pure biodeposits harvested in the weir box had a critical shear stress for erosion of 0.052 Pa. The decrease in bed stability found in biodeposits from 2010 compared to the 2008 biodeposit mixture, may be a result of a more diffuse biofilm developing on the highly organic substrate. The mixture of biodeposits collected in 2008 were a combination organic and inorganic materials which may be creating a nutrient limited environment, where biofilm structure consists of more tightly organized biofilm cells and as a result enhance stability in the bed sediments. The decrease observed after 14 days is likely a result of the microbes depleting their resources and dying off. Due to the added roughness the mussels created in the flume, τ_{crit} could not be measured and critical revolutions per minute (RPM) for erosion are reported for flume runs with mussels. During experiments conducted in 2009 with pure biodeposits and mussels the critical RPM was 5.83 while in 2010 in the presence of mussels a critical RPM was not observed. Settling experiments found biodeposits from both years (2008 and 2010) had decreased settling velocities when compared to different sediment types from lacustrine environments. I speculate that the added enrichment of the surficial sediments by mussel biodeposits is enhancing the process of biostabilization and increasing the bed stability and that the presence mussels themselves may additionally be enhancing bed stability by inhibiting flow from reaching the surface sediments/biodeposits.

Acknowledgments

First and foremost, I would like to offer my sincerest gratitude to my co-supervisor Dr. Mike Stone, for his constant encouragement, immense knowledge and adaptability to "Kelly time". I attribute a great deal of my completion of my M.Sc. to his endless support and efforts. Mike's tireless enthusiasm for his research and students made him a great advisor and mentor. Coming from a background in biology, Mike introduced me to the foreign physical world of sediment transport dynamics. I will never again underestimate the complexity of cohesive sediments and the interdisciplinary nature of their study. I greatly appreciate the opportunity given to me and the experience which has led me to the encounter a group of incredible individuals along the way.

At the beginning of this journey, I was introduced to my co-supervisor Dr. Ralph Smith, who as well I attribute much of the completion of this work to. It is difficult to overstate my gratitude to Ralph for his endless encouragement, sound advice and great ideas. I have never left a conversation with Ralph without having two or three new ideas. His enthusiasm and excitement for all things limnological are constant reminders of why I chose this field. I would also like to add my genuine gratitude to my committee members, Dr. Bommanna Krishnappan, Dr. Houston Saunderson and Dr. Ian Mckenzie for taking the time to provide useful feedback and comments.

I am grateful to Dr. Ian Droppo for his expertise and knowledge on flumes and for use of the annular flume at the Canadian Center for Inland Waters. Brian Trapp and Christine Jaskot have contributed tremendously to my study and without their skills and hard work none of the flume experiments would have been possible or have run so seamlessly.

I must also thank Dr. Sairah Malkin and Dr. Astrid Schwalb for their helpful insights and expertise. Meeting Sairah at the beginning of my time here, she offered great advice and helped me to start thinking about my research questions. I owe much gratitude to Astrid for advising me on water flow from a biologist's perspective. She clarified much of the confusion I had concerning processes surrounding flow.

It is my pleasure to thank my lab mates, Eric Camm, Donnie Allin and Kirstin Damude. Spending long periods of time in the lab were much more bearable having you as my lab mates. I must offer my regards to all of the friends I have made here at UW for all your support in any respect throughout the extent of my project. I would also like to express my gratitude to Niina Luus and Cathy King for their editing skills, without them I would still be trying to edit my first draft.

I would like to thank my parents, Doug and Marbeth McLean, for supporting me and accepting my decisions in life. There will come a day when I won't be a broke student. To my brother Darcy and his wife Meghan, for always supporting me and encouraging me to follow my dreams. Finally I must thank my partner Joshua King. You inspire me to be a better person and challenge me to strive. Your love and support has helped throughout this entire process and I appreciate everything you have done for me.

Dedication

Dad, this has been dedicated to you from the very beginning no matter the circumstances. I am thankful for the time we had and am honoured to have called you my dad. You made me proud every day of my life and I can only hope to have done the same for you. Your selflessness taught me strength, perseverance and commitment I hope you know how amazing you were.

You asked me once "What is the green slime covering the rocks and beds of the rivers I am fishing in?" I didn't know the answer at the time, but I think you knew and were trying to challenge me. I found out what it was. You always found interest in the work I was doing and I am more grateful than you will ever know for that.

You will always be in my heart. Thank you for giving me the courage to keep striving.
I love you.

Table of Contents

List of Figures

List of Tables

List of Symbols and Abbreviations

Symbol/Abbreviation	Definition
τ_o	Bed shear stress
τ_{crit}	Critical bed shear stress for erosion
ρ	Water density (g cm^{-3})
ϵ	Erosion rate
RPM	Revolutions per minute
Pa	Pascals

Chapter 1: Introduction

1.1 Problem Statement

The Laurentian Great Lakes provide a range of resource services and values such as drinking water, waste disposal, industrial water supply, commercial fisheries and recreation (Smith, 2007). Due to an increasing variety of land use pressures, these resources have been negatively impacted resulting in degraded water quality, loss and change of habitat and loss of native biota (Mills et al., 2003). From the 1940s to the 1970s, some of Great Lakes experienced cultural eutrophication resulting in nuisance algal blooms and low dissolved oxygen levels (Snodgrass, 1987; Matisoff and Ciborowski, 2005). International efforts were directed towards identifying the contributing factors and quantifying impacts of eutrophication on the Great Lakes ecosystems (Mills et al., 2003). In 1972, the Great Lakes Water Quality Agreement (GLWQA) set controls on permissible phosphorus loadings from watersheds to each of the Great Lakes (Mills et al., 2003). The desired outcomes were to reduce the incidence and extent of harmful algal blooms, total algal biomass and hypoxic conditions (Conroy and Culver, 2005). Primarily due to efforts that reduced loadings from sewage treatment plants (Smith, 2007), total phosphorus concentrations in Lake Ontario had decreased by 50% from 20-25 $\mu g\ L^{-1}$ in the early 1970s to 9.9 $\mu g\ L^{-1}$ in 1986 (Mills et al., 2003). Makarewicz (1993a) also reported a decrease in total algal biomass and a reduction in the abundance of eutrophic indicator species in Lake Erie by the mid-1980s.

In 1988, the non-indigenous zebra mussel, *Dreissena polymorpha*, was first observed in Lake St. Clair (Hebert et al., 1989). These organisms are believed to have been introduced via ballast water from international shipping vessels (Hebert et al., 1989). By 1990, zebra mussels extended their range throughout Lake Erie and were found in the western basin and southern shorelines of Lake Ontario (Griffiths, 1993). In 1991, the quagga mussel, *Dreissena (rotriformis) bugensis*, was observed in the Erie Canal and Lake Ontario (May and Marsden, 1992). The life cycles of dreissenids include a planktonic larval stage and benthic sedentary juvenile stage followed by an adult stage (Mackie, 1991). Accordingly, the various life stages provide transport opportunities that expand the range of the organism in any stage, either through unintentional transport of veliger larvae in ship ballast water, or mussel attachment to ship hulls, engines or anchors (Brown and Stepien, 2010). Dreissenids have proliferated throughout the Great Lakes and interconnecting waterways, including the Mississippi, Hudson and the St. Lawrence Rivers (Neary and Leach, 1992; Strayer et al., 1999).

In some areas of the Great Lakes such as western Lake Erie, dreissenid densities can exceed 700,000 m^{-2} (Kovalak et al., 1993). MacIsaac et al. (1991) reported slightly lower densities (341,000 m^{-2})

on reefs in Lake Erie, while densities of 4,000 m^{-2} were reported in the Hudson River (Strayer et al., 1996b). Coupled with their high densities, dreissenids can filter large volumes of water in relatively short periods of time. Estimated filtration rates vary amongst studies, between 0.2 to 700 mL hr^{-1} (Kryger and Riisgard, 1988; Noordhuis et al., 1992; Bunt et al., 1992). Collectively, high density dreissenid populations can result in very high total filtration rates (Kryger and Riisgard, 1988; Noordhuis et al., 1992; Bunt et al., 1993; Klerks et al., 1996; Roditi et al., 1996) that remove a wide range of particles from the water column. Typically, dreissenids filter phytoplankton, bacteria, zooplankton, detritus and silt particles varying in size from 1-1200 µm (Horgan and Mills, 1997; Strayer et al., 1999). Dreissenid populations have significant impacts on the physical, chemical and biological processes within benthic habitats due to their large biomass; efficient filtration rates and bio-deposition of suspended particulates and nutrients (Klerks et al., 1996; Hecky et al., 2004).

Dreissenids are non-selective filter feeders and internally sort materials for digestion. Once in the mantle cavity of the mussel, particles are either rejected or selected as food items. Rejected particles are bound together with mucous and released through the inhalant siphon as pseudofeces (Reeders and bij de Vaate, 1992). Particles selected as food items are, ingested and later released as feces (Reeders and bij de Vaate, 1992). Various studies demonstrate that dreissenid induced bio-deposition of suspended materials is significantly greater than natural sedimentation rates (Jaramillo et al., 1992; Klerks et al., 1996; Dobson and Mackie, 1998). In particular, Dobson and Mackie (1998) found that bio-deposition rates were up to eight times higher than natural sedimentation rates. The increased flux of particulate materials to the benthic community enhances species richness and abundance (e.g. amphipods (Bially and MacIsaac, 2000)), while negatively impacting other organisms (e.g. native bivalves (Schloesser et al., 1996)) and leading to losses in natural populations (Riccardi et al., 1996).

In recent years, near shore zones of the Great Lakes have experienced eutrophic conditions associated with excess nutrient loading (Hecky et al., 2004). The benthic filamentous algae, *Cladophora*, an indicator of P enrichment, has proliferated on the shorelines of Lake Erie, while the offshore conditions of P concentrations are consistent with the standards set by the 1972 GLWQA (Hecky et al., 2004). At the outflows of Lake Erie, particulate phosphorus (PP) concentrations are higher than in the 1980s, with no evidence of increased loading from allochthonous sources (Hecky et al., 2004). Accordingly, Hecky et al. (2004) attribute increased P levels to dreissenid activity in the near shore zone. They hypothesize that mussels have reengineered the near shore environment by intercepting, detaining and recycling suspended nutrients that were previously exported to the offshore environment. Hecky et al. (2004) referred to this bio-modification process as the "nearshore shunt". The nearshore shunt hypothesis suggests that prior to dreissenid invasion, the near shore weakly retained PP which was a net source to the offshore pelagic zone (see Figure 1.1a) (Hecky et al., 2004). However, after dreissenid

establishment, it is hypothesized that the near shore benthic zone retains PP from the offshore pelagic because PP is being re-packaged into larger particles and therefore less particulate matter is transferred directly to the offshore pelagic and instead is moving directly to the basin outlet (see Figure 1.1b) (Hecky et al., 2004). This transfer of particulate matter to the benthic environments not only affects the rates and magnitudes of nutrient cycling (Hecky et al., 2004) but also influences contaminant transport (Klerks et al., 1996) and water quality (Dean, 1994).

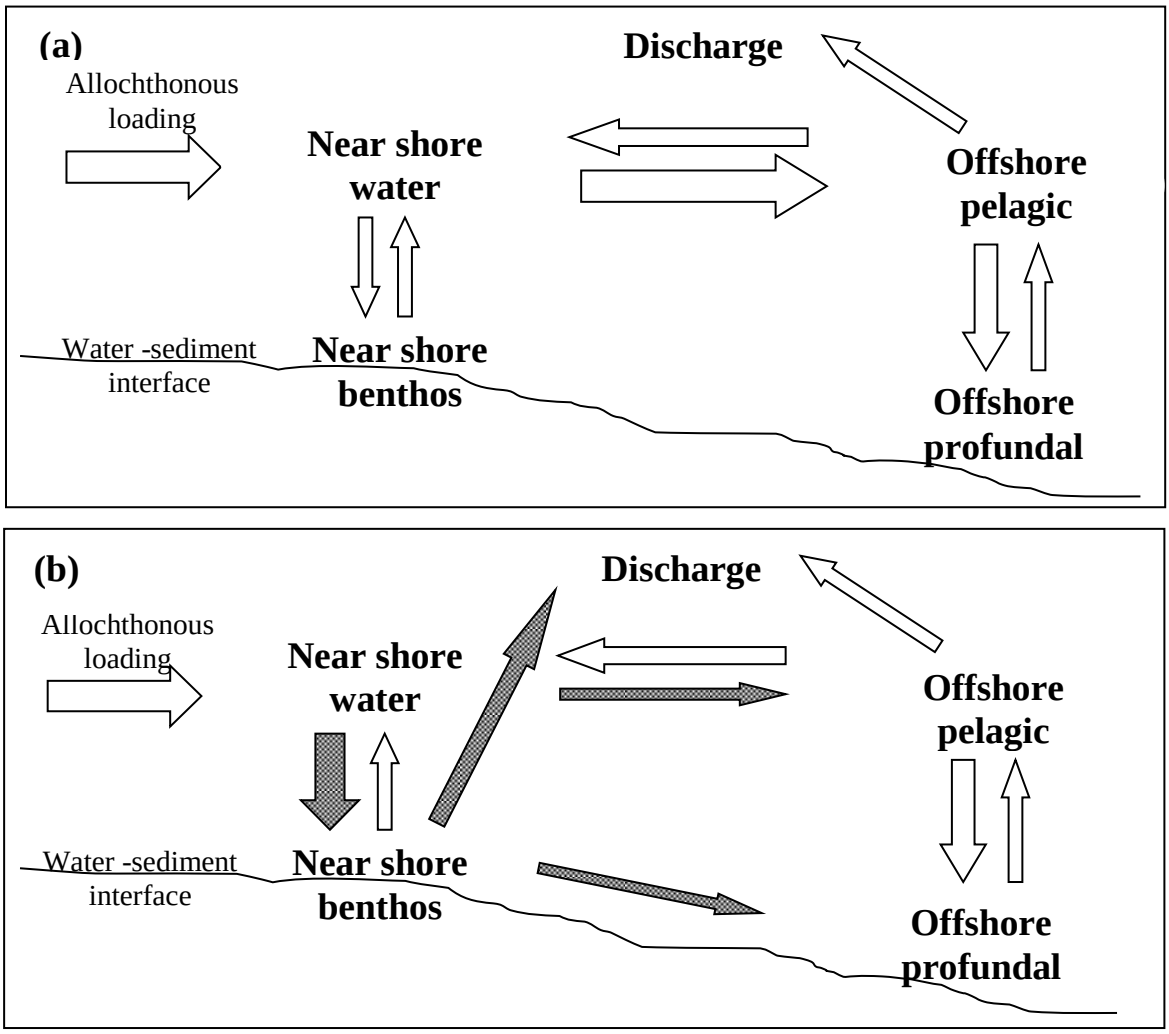


Figure 1.1: Nearshore shunt hypothesis proposed by Hecky et al. 2004, conditions (a) Pre and (b) Post dreissenid establishment. Arrows indicate P pathways in the system, hatched arrows indicate changes following dreissenid establishment. (a) The nearshore weakly retained P and was a net source to the offshore pelagic. (b) The nearshore benthos detaining much of the P from the offshore pelagic.

In aquatic systems, cohesive sediments (<63 μm) are the primary transport vector for phosphorus and many other nutrients and pollutants (Huang et al., 2006; Clifton, 2005). Accordingly, the transport, fate and effect of these sediment associated pollutants are strongly linked to the transport properties of cohesive sediment and the hydrodynamics of the aquatic environment (Clifton, 2005). Despite the

importance of cohesive sediment for P transport dynamics in the near shore zone, little is known about the influence of dreissenids on the re-packaging of suspended solids in the water column into biodeposits and their transport dynamics (see Figure 1.2). The goal of this thesis is to quantify the physical properties (settling velocity, porosity, density, grain size) and transport characteristics (critical shear stress for erosion and erosion rate) of dreissenid biodeposits. The transport characteristics of dreissenid biodeposits are quantified and the effect of biostabilization and mussel bed roughness on the transport of mussel biodeposits is determined using an annular flume. Results of this thesis can be used to elucidate transport parameters required to refine particle transport models in lakes.

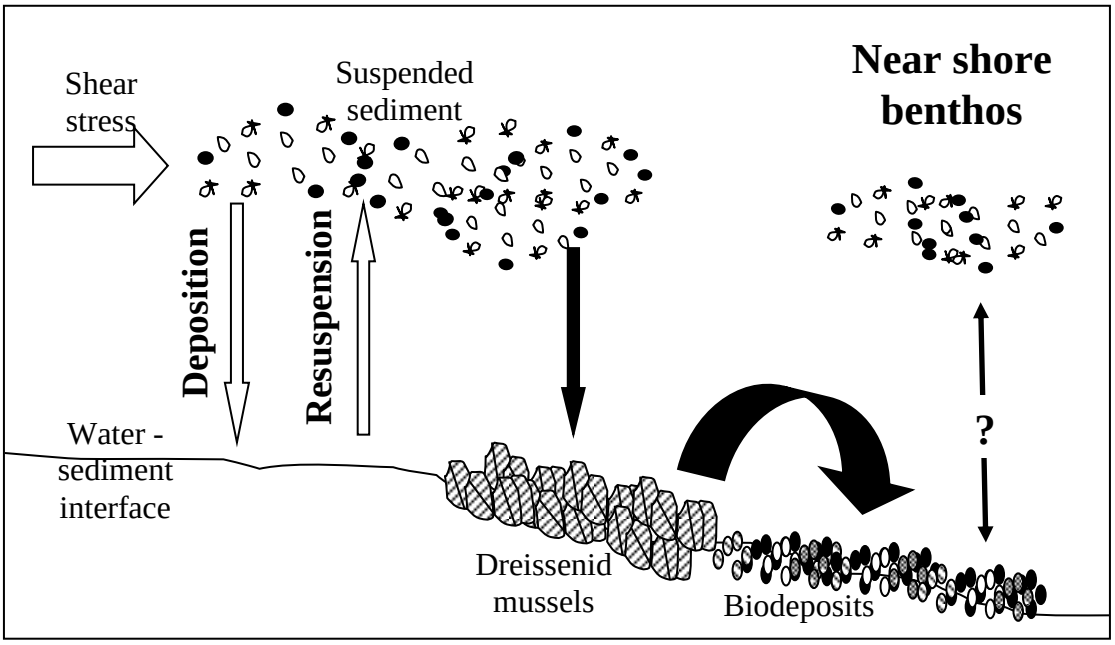


Figure 1.2: Conceptual diagram of dreissenid mussel particle interactions in the near shore zone.

1.2 Research Objectives

The objectives of this study are to:

1. Evaluate the transport properties (critical shear stress for erosion, erosion rate) of dreissenid biodeposits in an annular flume with and without dreissenids.
2. Characterize and quantify the physical properties (porosity, density, settling velocity, and particle size) of dreissenid biodeposits.

1.3 Literature Review

Understanding the transport dynamics of fine grained suspended sediment (<63μm) is important in determining the physical transport characteristics of mussel biodeposits because; 1) mussels preferentially filter cohesive sediments between the size range of 15-40 μm (Ten Winkel and Davids, 1982; Klerks et al., 2001) although they have been observed filtering materials as large as 1500 μm (Sprung and Rose, 1988; Horgan and Mills, 1997); and 2) P and other nutrients and contaminants preferentially bind to sediments <63μm. Accordingly, cohesive sediment transport models incorporating biotic impacts (i.e. mussel feeding and biotransformation of particles) are necessary to predict the distribution of dreissenid biodeposits and associated contaminants. However, many uncertainties surround the hydrodynamic force necessary to erode, suspend and transport cohesive sediments, especially the critical conditions necessary for initiation and subsequent deposition (Milburn and Krishnappan, 2003). The erodibility of cohesive bed sediments is dependent on the balance between erosive forces (i.e. hydrodynamic forces) and resistive forces within the bed sediment (Grabowski et al., 2011). Eroded fine-grained sediments have a cohesive nature and tend to form flocs, which changes the porosity, size and shape of particles (Droppo, 2001). The process of flocculation is dependent on a number of physical, chemical and biological factors within the flow field that constantly change (Droppo, 2001). Accordingly, the transport and settling velocity of a floc changes making it difficult to accurately predict the distribution of particles and associated contaminants within aquatic systems.

All aquatic sediments contain microorganisms (e.g. bacteria, protozoans, fungi and diatoms) and/or macroorganisms (e.g. bivalves, polychaetes, and amphipods) (Grabowski et al., 2011). Collectively, these organisms can have significant impacts on the sediment transport dynamics thus, altering the hydrodynamics (i.e. biogenic structures), sediment stability (i.e. biostabilization) and particle settling velocity (i.e. filtration processes) (Jumars and Nowell, 1984; Karlsson et al., 2003; Widdows et al., 2009; Grabowski et al., 2011). Dreissenids for example, are considered both allogenic and autogenic engineers. Their feeding ecology can change the nature of suspended materials, and their dense mats can alter the morphological and physical properties of areas invaded (Hecky et al., 2004; Coleman and Williams, 2002). Accordingly, the goal of this study is to gain further understanding of how dreissenids influence particle transport dynamics in the near shore environment by examining the effects of dreissenids on sediment transport dynamics (i.e. erodibility of bed sediments) and the physical nature of the biodeposits produced by mussels. This chapter reviews current literature on the nature of cohesive sediments, the main physical and biological processes influencing cohesive sediment transport and the impacts dreissenids have on particle transport dynamics in aquatic systems.

1.3.1 Nature of Cohesive Sediments

Sediments play an integral role in hydrological, geomorphological and ecological processes in aquatic systems (Forstner and Owens, 1997). Produced as a result of weathering rock, transport and biological processes (Grabowski et al., 2011), sediments provide substrates for biota, control water chemistry and clarity and are sites for biogeochemical cycling in rivers and lakes (Forstner and Owens, 1997; Grabowski et al., 2011). Due to anthropogenic emissions (i.e. excessive metals, nutrients, organic pollutants etc.) and hydrogeomorphological modifications (e.g. dams, channels and dredging), the quality and quantity of aquatic sediments has degraded, negatively impacting aquatic ecosystems (Forstner and Owens, 1997; Wood and Armitage, 1997; Huang et al., 2006; Grabowski et al., 2011).

The majority of toxic chemicals including US-EPA priority pollutants are transported primarily by cohesive sediment (<63 μm) and/or biological substrates (Droppo et al., 2001; Chapman, 1982). Contaminants can be delivered directly through point sources and soil erosion or gradually adsorb to fine-grained cohesive sediment within the water column (Droppo et al., 2001). Fine-grained cohesive sediments are clay-sized particles consisting of organic and inorganic materials (Huang et al., 2006). Due to size and surface ionic charges, inter-particle forces dominate the behaviour of cohesive particles over gravitational and drag forces (Huang et al., 2006). As the particle size decreases, the surface area begins to become significant relative to its volume (Gregory, 2005). The large surface area provides more opportunity for contaminant adsorption (Stone and Droppo, 1994). Additionally, compared to larger grain size fractions, cohesive sediments (< 63μm) are more geochemically active (Ongley et al., 1982; Stone and Droppo, 1994; Stone et al., 1995) and remain in suspension for longer periods of time (Lick, 1982). Accordingly, a precise understanding of cohesive sediment transport processes is necessary to model the transport and fate of sediment associated contaminants.

The settling velocity and grain size distribution of particles in suspension, govern the transport of cohesive sediments (Berlamont et al., 1993). These parameters vary in response to the physical, chemical and biological attributes of an individual system and sediment source (de Boer et al., 2005). Various studies have documented the tendency for cohesive suspended sediments to flocculate/aggregate in transport, characterizing their transport in aquatic systems (Droppo et al., 1997, 1998, 2001; Droppo, 2001; de Boer et al., 2005). Aggregation is thought to occur outside of the aquatic system, and aggregates are transported as water stable soil aggregates into a system (Wall et al., 1978; Droppo, 2001). Flocculation refers to the joining of particles in the water column through a series of complicated physical, biological and chemical processes (Droppo, 2001). Both processes refer to the formation of larger particles from smaller particle collisions, and hereafter the terms aggregation/aggregate and flocculation/floc will be used interchangeably (Droppo, 2001).

The hydrodynamic properties of the constituent particles are altered during flocculation which causes changes in particle density, porosity and settling velocity affecting the fate of particle bound contaminants (Droppo et al., 1997; Droppo et al., 2002; de Boer et al., 2005). Droppo (2001) developed a conceptual model to describe floc form and behaviour and illustrate processes affecting floc structure and behaviour. He defined floc behaviour "as any physical, biological, or chemical process that brings about change in the state of the floc or its transport characteristics". His conceptual model describes a floc as having four main components; inorganic, biological, water and pore components (Droppo, 2001). The characteristics and resulting behaviour of each of the aforementioned components are presented, thus illustrating a link between the structural components of a floc and its interrelated behaviour within aquatic environments (Droppo, 2001).

1.3.2 Cohesive Sediment Transport

To effectively assess the impacts of cohesive sediments and associated contaminant transport on aquatic systems, it is necessary to understand and quantify factors that affect the spatial and temporal distribution of particulate matter in the water column and on benthic substrates (Clifton, 2005). The relationship between sediment transport processes and hydrodynamic flow conditions is well understood for cohesive sedimenting materials (Clifton, 2005). Mathematical numerical models are routinely used to examine environmental fluid dynamic problems (i.e. contaminant transport, fate and bioaccumulation) in aquatic systems (Paterson and Black, 1999). Transport processes and parameters important for modelling cohesive sediment transport are illustrated in Figure 1.3.

Key processes influencing the transport and fate of cohesive sediments are flocculation, settling, deposition, consolidation and resuspension (Clifton, 2005). These mechanisms are not solely dependent on the physico-chemical characteristics of the sediment, but also on the particle-particle interactions within the flow field and related biological activity that influence flocculation processes (Paterson and Black, 1999; Clifton, 2005). Accordingly, the interdependence of these processes makes cohesive sediment transport complex to model (Clifton, 2005). The effectiveness in these numerical models relies upon a rigorous understanding of these processes. However, because the interplay between these processes is not fully understood or quantified, models predicting the behaviour of cohesive sediment are not fully developed (Paterson and Black, 1999).

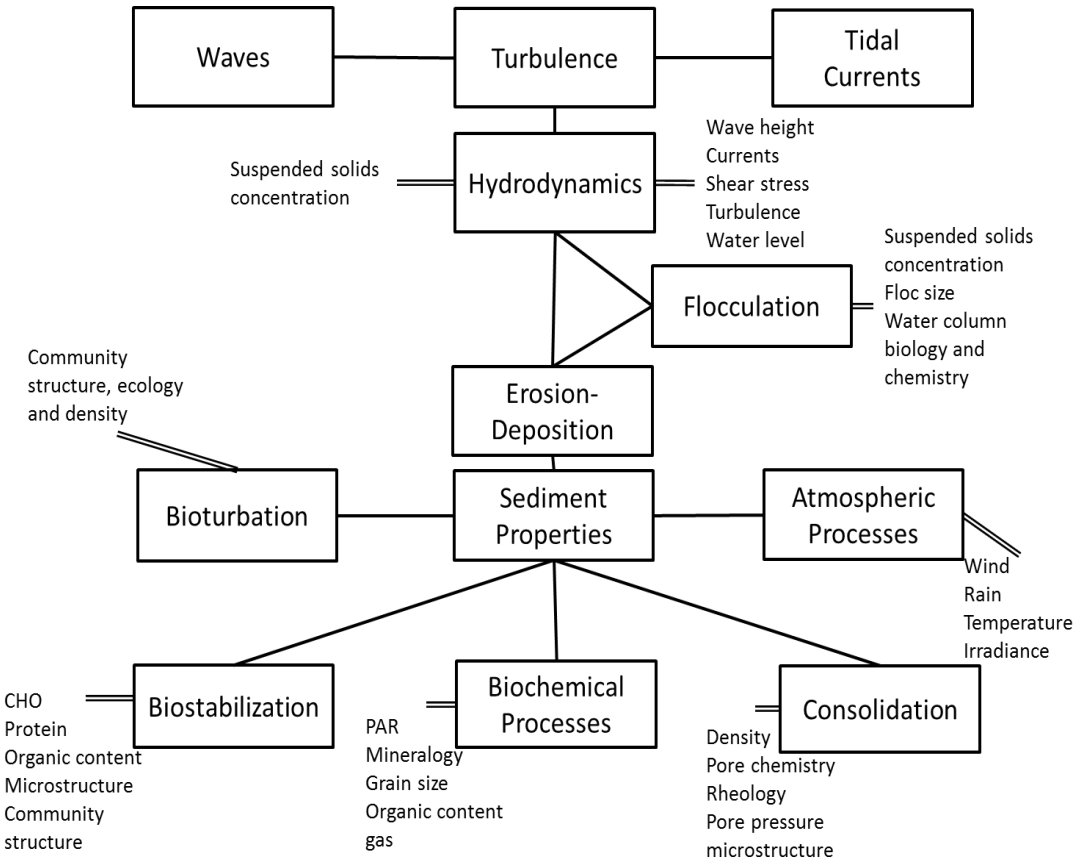


Figure 1.3: Conceptual diagram of interactions between processes and parameters influencing cohesive sediment erosion and deposition on tidal flats (Black and Paterson, 1996).

Parameterization of processes such as erosion and deposition can be determined by direct measurement in laboratory flumes or field (de Boer et al., 2005). The following section reviews cohesive bed sediment stability and the factors influencing erodibility of cohesive bed sediments.

1.3.3 Erodibility of Cohesive Bed Sediments

In aquatic systems, cohesive sediment beds consist of deposited particles and flocs that were previously in suspension (Droppo and Stone, 1994; Stone et al., 2011). Particle settling rates depend on the hydrodynamic and biogeochemical characteristics of the aquatic environment (Lau, 1990; Stone and Droppo, 1994; Droppo et al., 1997; Krishnappan and Marsalek, 2002; Stone et al., 2011). The stability of bed sediment is dependent upon the balance between the hydrodynamic forces causing erosion and the resistive forces within the sediment bed (Grabowski et al., 2011). Boundary layer shear stress and turbulence are some of the erosive forces acting on the bed sediments (Grabowski et al., 2011). These forces are a function of flow characteristics and solid transmitted stresses from saltating materials known

as bedload (Amos et al., 1998; Grabowski et al., 2011). Physical and biological features of the near-bed environment, such as increased surface roughness and filtration by organisms, further influence these forces by altering the flow velocity and transport of particles (Nowell et al., 1981; Eckman and Nowell, 1984; Nowell and Jumars, 1984; Sousa et al., 2009; Grabowski et al., 2011). The influence of biota on flow characteristics and particle transport will be discussed later in this chapter. The force exerted by water flowing over exposed bed sediments is defined as the bed shear stress (τ_o). Bed shear stress increases with channel slope and flow depth according to the Du Boy's equation:

$$\tau_o = \rho g h S \tag{1.1}$$

where τ_o is the spatially averaged bed shear stress (Pa), ρ is water density (kg m^{-3}), g is the acceleration due to gravity, h is the depth of flow (m) and S is the slope. Shear forces acting on cohesive sediment beds is counteracted by the submerged weight of the particle, frictional interlocking of grain sediments and inter-particle cohesive forces (Black et al., 2002). The driving and resisting forces acting on a single particle are illustrated in Figure 1.4. If the hydrodynamic forces exceed the resistive forces, erosion occurs (Black et al., 2002; Grabowski et al., 2011).

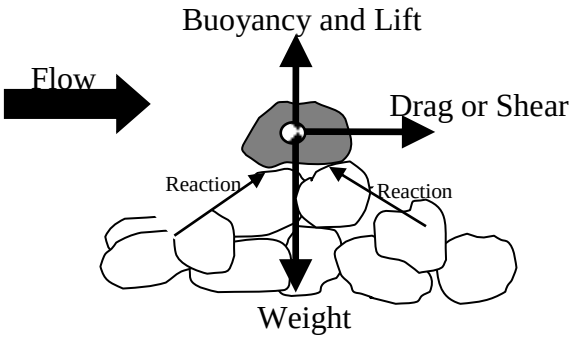


Figure 1.4: Driving and resisting forces acting on a sand grain (Willis and Krishnappan, 2004).

The erodibility of cohesive sediment is often defined based on two principle parameters; critical shear stress (or threshold for erosion) and erosion rate (Amos et al., 1992). The threshold for erosion is the water velocity or the critical bed shear stress (τ_{crit}) that initiates sediment erosion (Grabowski et al., 2011) that changes over time (Amos et al., 1988; Christian, 1990; Paterson et al., 1990; Amos et al., 1992) and depth within the sediment bed (Amos et al., 1992). The erosion rate is defined as the mass of sediment eroded per unit area and time (kg m^{-2} t^{-1}) (Grabowski et al., 2011). Amos et al. (1992) classify erosion with two categories. Type I erosion decreases exponentially with time. In this case, floc erosion decreases with time to a point where the bed shear stress (τ_o) is equal to the bed shear strength (τ_s) (Paterson and Black, 1999). Type II erosion is constant with time and tends to occur when the bed shear

stress greatly exceeds the erosion threshold ($\tau_o >> \tau_{crit}$) (Amos et al., 1992; Paterson and Black, 1999). Experiments performed by Amos et al. (1997) describe a third type of erosion, type Ia, where a surface 'fluff' layer, is eroded under low shear conditions (Amos et al., 1997). This 'fluff' layer has been referred to as the surficial fine-grained laminae (SFGL) which consists of a low density high water content layer (Stone and Droppo, 1994; Stone et al., 2011). Droppo and Stone (1994), reported that this top layer consists of flocculated materials up to 8 mm thick which forms a transient blanket over the existing sediment bed (Droppo and Amos, 2001). Once eroded, this layer can return significant amounts of sediment and associated contaminants back into the water column potentially resulting in further distribution of contaminants (Lambert and Walling, 1988; Stone and Droppo, 1994; Phillips and Walling, 1999; Droppo and Amos, 2001).

McAnnally and Mehta (2001) developed a numerical model to describe the erosion rate of cohesive sediment layers. This model is applied to freshly deposited beds in the process of initial self-weight consolidation by Black et al. (2002):

$$\ln\left(\frac{\epsilon}{\epsilon_f}\right) = \alpha\left[\left(\tau_o - \tau_{o_{crit(z)}}\right)\right]^{\beta} \tag{1.2}$$

where τ_o is the bed shear stress (Pa), $\tau_{o\ crit(z)}$ is the critical erosion stress at depth z (m), ϵ_f is termed the floc erosion rate (kg m^{-2} s^{-1}) and β and α are the exponent and rate coefficient, respectively (Black et al., 2002). Black et al. (2002) describe a second model commonly used for the erosion of consolidated beds or mechanically emplaced beds, where bed properties are uniform within the uppermost centimeters of the bed:

$$\epsilon = \epsilon_M \left(\frac{\tau_o - \tau_{o_{crit}}}{\tau_{o_{crit}}}\right)^{\delta} \tag{1.3}$$

where ϵ_M is the rate coefficient and δ the exponent.

1.3.4 Factors Influencing Bed Sediment Stability

The stability of bed sediments has classically been attributed to factors such as consolidation, dewatering and electrochemical forces resisting erosion (Droppo, 2009). More recently linkages have been recognized between erosion potential and the biological properties of soft sediment bottoms (Black et al., 2002). Although the physical forces necessary to erode sediment beds are typically orders of magnitude greater than biological forces, the biological effects become increasingly important during quiescent (low flow) conditions (Black et al., 2002). One biotic force contributing to bed stability is biostabilization,

which is contrary to destabilization (Black et al., 2002). The following sub sections review literature describing the physical force of consolidation and the biological force of biostabilization that influences the stability of cohesive sediment and methods used to quantify them.

1.3.4.a Physical Processes; Consolidation

The consolidation process is important to consider when modeling the erosion of cohesive sediments (Berlamont et al., 1993; Huang et al., 2006). Consolidated beds become less susceptible to erosion with time due to an increase in bed shear strength and the increase in density changes the mass of sediment eroded per unit bed thickness (Mehta et al., 1989). There are two types of consolidation: primary and secondary (Mehta et al., 1989). Primary consolidation refers to the self-weight of a particle and begins when the self-weight exceeds the seepage force of upward flow of pore water from the underlying sediments (Mehta et al., 1989; Huang et al., 2006). As the self-weight of the particle expels the underlying pore water, the particles move closer together (Huang et al., 2006). The seepage force gradually decreases with time until it is completely dissipated and primary consolidation ends (Huang et al., 2006). Under the constant overburden of stress, plastic deformation of the sediments occurs, which is referred to as secondary consolidation (Mehta et al., 1989; Huang et al., 2006). Secondary consolidation begins during the primary phase and can last for up to months after primary consolidation ends (Huang et al., 2006).

Models incorporating consolidation represent the sediment bed with a number of layers (Huang et al., 2006). Each layer has a specific thickness, consolidation time and critical shear stress (Huang et al., 2006). Nicholson and O'Connor (1986) developed an idealized consolidation model, linking the bulk density (ρ) to the consolidation time (*t*):

$$\rho_b = \begin{cases} \rho_{f,} & t \le t_f \\ \rho_f + (\rho_\infty - \rho_f)\{1.0 - exp[-A_2(t - t_f)]\}^{B,} & t_f < t < t_\infty \\ \rho_{\infty,} & t \ge t_\infty \end{cases} \quad (1.4)$$

where ρ_b is the dry bulk density (kg m^{-3}), *t* represents time (seconds) and A_2 and B_2 are the coefficients that account for the influence of mud type and salinity (Huang et al., 2006). Subscripts f and ∞ represent freshly deposited and fully consolidated states, respectively (Huang et al., 2006).

1.3.4.b Biological Processes; Biostabilization

Increasingly the influence of benthic biota as contributors to bed sediment stability (biostabilization) or factoring against it (biodestabilization) are becoming prevalent in the literature (Black et al., 2002; Le Hir et al., 2007; Stone et al., 2011). Biostabilization influences sediment erodibility and is a process in which sediment associated bacteria, microalgae and macrofauna, produce extracellular polymeric substances (EPS) which coat particles and bridges interstitial gaps to form a cohesive network (Stone et al., 2011; Black et al., 2002; Paterson, 1997). Due to the adhesive nature of the EPS, grains and flocs stick together increasing the bed sediment stability (Stone et al., 2011; Droppo et al., 2001).

The development of biofilms on aquatic sediments can change the physical characteristics (e.g. grain size, structure, morphology, porosity, shape, degree of consolidation) (Stone et al., 2011) and influence its behaviour (i.e. erodibility) (Droppo and Amos, 2001; Droppo et al., 2007; Stone et al., 2011). Numerous studies have demonstrated the increased stability of biostabilized sediments, with the increase in horizontal shear required to initiate erosion (Dade et al., 1996; Amos et al., 2004; Gerbersdorf et al., 2008; Stone et al., 2011). In marine clays for example, Dade et al. (1996) found that the critical shear stress for erosion increased up to 60 % for biostabilized sediments over control sediments. Droppo et al. (2001) observed a 10-fold increase in shear required to induce erosion of biostabilized sediments versus control sediments after 5 day consolidation time. There is abundant literature on the effects of biostabilization as a stabilizing factor, yet few sediment transport models incorporate their effects (Le Hir et al., 2007).

Le Hir et al. (2007) discuss a number of reasons why modeling the effects of biostabilization is difficult. They suggest the type of microbial community inhabiting the benthic sediments and their stabilizing effects vary considerably in aquatic environments depending on the amount of EPS secreted by different types of microbes (Le Hir et al., 2007). Yallop et al. (2000) found that algae secreted more exopolymer than bacteria and as a result increased the stability of the algae dominant sediment. Chlorophyll *a* or colloidal carbohydrates are often used as proxies for an index of EPS. Relationships between these have been made but parameters are environmentally specific (Friend et al., 2003a; Le Hir et al., 2007). Although EPS is the parameter of interest, chl *a* can also provide ecological information and can be measured quickly and efficiently (Murphy et al., 2004; Le Hir et al., 2007). Le Hir et al. (2007) presented data from the literature and illustrated correlations between the critical shear stress for erosion and chl *a* (Figure 1.5). This figure shows the erosion thresholds are much lower for chl *a* concentration below 30 mg m^{-2} (0.1-0.8 Pa) than for concentrations above that limit (0.4-1.7 Pa chl *a*=50 mg m^{-2}) (Le Hir et al., 2007). All plots show similar trends, with stability in chl *a* enriched sediments, increasing up to a factor of 4 (chl a=100 mg m^{-2}) (Defew et al., 2003; Le Hir et al., 2007). The correlation between

critical shear stress for erosion and chl *a* are varied (Le Hir et al., 2007). One common thread amongst them is that with higher chl *a* levels, the shear stress required to erode the bed sediments increases (Le Hir et al., 2007).

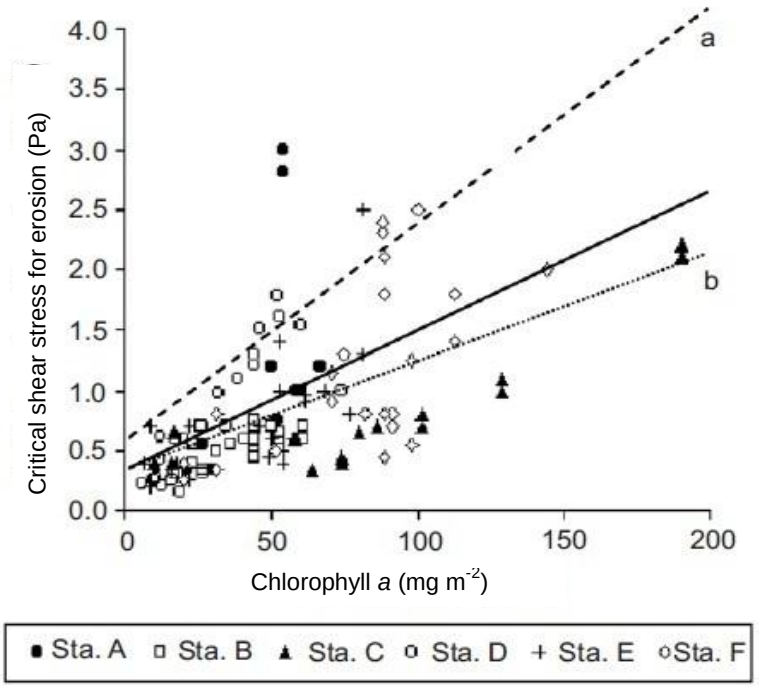


Figure 1.5: Data collected from several studies comparing chlorophyll *a* concentrations and the critical shear stress for erosion (Le Hir et al., 2007).

A second problem regarding modeling the effects of biostabilization is that once the shear stress is sufficient to lift the biofilm from the top layers of the sediment, the underlying sediment may have the same erosive behaviour as bare sediments (Le Hir et al., 2007). The erodibility is then dependent on the physical characteristics of the sediment, which are variable over space and time (Le Hir et al., 2007). The third and final issue Le Hir et al. (2007) suggests, is related to the experimental devices employed and the resulting variability in data. Widdows et al. (2007) conducted erodibility experiments on intertidal sediments, using five devices to measure the critical shear stress for erosion. Even when the methods for calculating type 1a and 1b erosion were defined and standardized, there was little agreement between the results from different devices (Widdows et al., 2007). Similarly, Tolhurst et al. (2000) compared four erosion devices to determine erosion thresholds in the Humber estuary, UK. Their study showed erosion rates varied by orders of magnitude between the different devices (Tolhurst et al., 2000). Using biological and physical parameters, many recent studies have attempted to identify a proxy to characterize sediment behaviour, relating to erosion threshold, but the above mentioned problems make this a difficult

task (Le Hir et al., 2007). The devices for measuring sediment stability will be reviewed in the following section including the use of annular flumes for erosion studies.

1.3.5 Quantifying Cohesive Sediment Transport using Experimental Flumes

To model the transport of cohesive sediments, some quantitative measurements of sediment characteristics and transport parameters are necessary to make inferences on the distribution of sediments within an aquatic system (Willis and Krishnappan, 2004). Since erosion and deposition of cohesive sediment cannot be predicted based on environmental parameters alone, empirical measurements are needed in addition to knowledge of sediment characteristics to validate models. Knowledge of cohesive sediment erodibility and deposition is critical for developing fate and transport models of cohesive sediment and associated contaminants (Ravens, 2007). Many investigators have extensively used laboratory and in-situ flumes, where selected variables can be held constant and flow conditions can be controlled (Schumm et al., 1987). Erosion experiments performed by Kuijper et al. (1989) follow an experimental design presented in Figure 1.6.

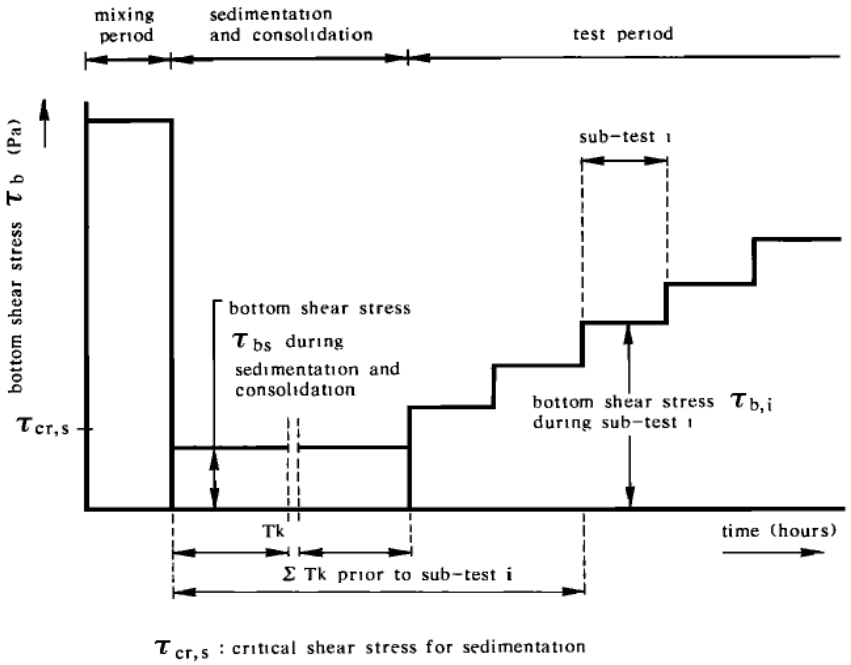


Figure 1.6: Experimental procedure for erosion experiments (Kuijper et al., 1989).

Using an annular flume, Kuijper et al. (1989) conducted erosion experiments on cohesive bed sediment stability as a function of consolidation time. Following a one day mixing period in the flume, sediments settled and went through an 8 day consolidation period (Kuijper et al., 1989). Erosion

experiments consisted of incremental steps in bed shear stress (i.e. increase in rotational speed of flume) to erode the consolidated bed sediments (Figure 1.6) (Kuijper et al., 1989). To measure the depositional rates of cohesive sediments, experiments begin with a mixing period of high turbulence, followed by decrements in shear stress over time (Chan et al., 2006). With each decrease in shear stress, suspended solids concentrations are measured to determine the critical shear stress for deposition (Chan et al., 2006). Chan et al. (2006) performed depositional studies on cohesive sediments from Mai Po, Hong Kong, using an annular flume. The study produced a typical graph displaying decreasing velocity and suspended sediment concentration as a function of time (Chan et al., 2006).

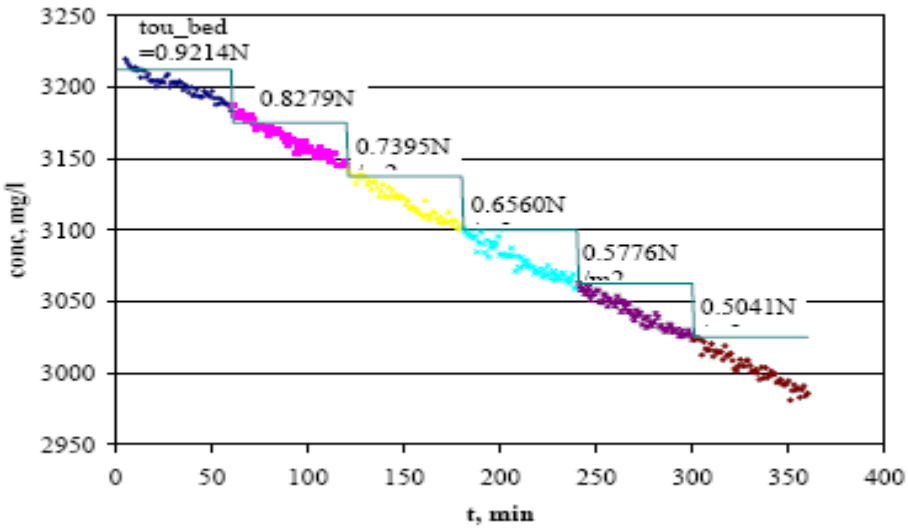


Figure 1.7: Results of settling experiments, solid lines indicate change in velocity (Chan et al., 2006).

Various laboratory flumes have been used in the study of cohesive sediment transport to elucidate the phenomena associated with the transport of cohesive sediments (Lau, 1990; Krishnappan, 1993; Lee and Mehta, 1994). Results from experiments provide relevant input parameters (i.e critical shear stress for erosion and deposition, erosion rate, settling velocity) for mathematical models predicting the transport and fate of sediment associated contaminants (Krishnappan, 1993; Lau et al., 2001). Droppo et al. (2001) described typical flume experiments as characterizing sediment depositional rates by: 1) collecting sediment from site of interest and placing it into a flume; 2) completely mixing the sediment under high shear stress; 3) allowing particles to settle and consolidate; and 4) applying flow of known shear to provide information on sediment resuspension. Experiments for the current study were performed in an annular flume, thus the next section will review studies using laboratory and in-situ annular flumes.

1.3.5. a Experiments using Annular Flumes

Annular flumes have been extensively used to characterize sediment transport characteristics, such as the critical shear stress for erosion (τ_{crit}) and erosion rate (ϵ) of cohesive sediments from marine and aquatic systems (Fukuda and Lick, 1980; Lick, 1982; Parchure and Mehta, 1985; Kuijiper et al., 1989). Traditionally these studies investigated the erodibility of bed sediments deposited under quiescent conditions, consolidated for a period of time and then eroded at incremental bed shear stresses (Droppo and Amos, 2001). While these studies have been useful in quantifying the mechanics of bed failure, they do not provide realistic estimates of bed failure in environments where sediments are deposited under flowing conditions or have experienced a complex stress history (Droppo and Amos, 2001). Lau and Droppo (2000) and Droppo et al. (2001) have shown that depositional history (i.e. shear stress at which bed is deposited); biostabilization and the SFGL floc structure and strength have considerable effects on the stability of bed sediments (Droppo and Amos, 2001). For example, Droppo et al. (2001) used an annular flume with both contaminated sediments and commercial kaolinite clay to determine to what degree depositional history and structure of the sediment influences bed stability. Their results found biostabilization and depositional history had pronounced effects on the stability of contaminated bed sediments (Droppo et al., 2001). Using an annular flume, Droppo (2000) found cohesive sediments deposited under shear stress, had a critical shear stress for erosion value up to eight times larger than quiescently deposited beds. Erosion experiments performed by Lau et al. (2001) found depositional history influenced erosional characteristics of bed sediments. Further they observed the rate of erosion and amount eroded was a function of bed structure and the flocs which created it (Lau et al., 2001).

Experimental annular flumes have been used to develop models that predict the transport and fate of cohesive sediments (de Boer et al., 2005). Milburn and Krishnappan (2003) collected riverine sediments in the spring from Hay River, Northwest Territories, Canada to determine the movement of sediment under ice before breakup. Using an annular flume, they were able to determine the processes influencing erosion and deposition. Erosion experiments provided quantitative information on the critical shear stress for erosion and erosion rates as a function of bed shear stress and age of bed sediments (Milburn and Krishnappan, 2003). With the results of their study, Milburn and Krishnappan (2003) were able to provide a modeling strategy to calculate the under-ice transport of cohesive sediments in Hay River. Similarly, Krishnappan and Marsalek (2002a) performed annular flume experiments to determine the transport characteristics of cohesive sediments from a stormwater management pond. They determined the critical shear stress for deposition and erosion, and were able to develop empirical relationships to estimate the sediment deposition and erosion as a function of bed shear stress. The results

of this study can be used for future modeling of cohesive sediment transport in stormwater management ponds. Table 1 displays various erosion experiments using laboratory and in-situ annular flumes.

Table 1.1: Experimental flumes and investigators researching cohesive sediment transport processes.

Experimental Flume	Investigators	Sediment type(s)	Investigating	Results
Annular flume (two component-rotation of flume lid and base)	Pachure and Mehta, 1985	Cohesive sediments from an estuarine environment	Influence of sediment type, consolidation period and salinity on critical shear stress for erosion	Results found bed shear strength increased with depth, sediment type, consolidation period and salinity
	Kuijper et al., 1989	Cohesive sediments from marine system	Erodibility of sediments in steady flow	Analyzed results using an erosion rate function and found good agreement
	Krishnappan and Marsalek, 2002	Cohesive sediments collected from a storm water management pond	Depositional and erosional characteristics of sediments influenced by consolidation	Critical shear stress for deposition τ_{cd}= 0.05 Pa τ_{crit}= 0.12 Pa
Annular flume (one component-rotation of flume lid only)	Lau and Droppo, 2000	Kaolinite clay and contaminated lacustrine sediments	Stability and transport characteristics of sediments with different depositional histories (i.e. deposited under quiescent conditions or shear)	Found depositional history influenced bed stability. The critical shear stress for beds deposited under shear were 8 times larger than quiescent deposited beds
	Droppo et al., 2001	Kaolinite clay and contaminated lacustrine sediments	Examine the effect depositional history and biostabilization on contaminated bed sediment	Biostabilized beds deposited under shear conditions was more resistant to erosion than non-biostabilized sediments deposited and beds deposited under quiescent conditions
	Chan et al., 2006	Cohesive sediments from a natural reserve	Deposition behavior of cohesive sediment	Suspended sediment concentrations declined slowly and steadily
	Droppo et al., 2007	Waste bed sediment collected beneath a discontinued aquaculture operation	Erosional characteristics of sediments under different flow conditions and consolidation and biostabilization	2 day τ_{crit}= 0.06 Pa 7 day τ_{crit}= 0.06 Pa 14 day τ_{crit}= 0.10 Pa
	Droppo, 2009	Contaminated lacustrine, storm water pond, fluvial, aquaculture waste and kaolin sediment	Comparing 5 different aquatic sediments for erosional strength and varying biofilm development	τ_{crit} after 7 days Contaminated τ_{crit}= 0.22 Pa Storm water τ_{crit}= 0.23 Pa Fluvial τ_{crit}= 0.19 Pa Aquaculture τ_{crit}= 0.06 Pa Kaolin τ_{crit}= 0.10 Pa
	Stone et al., 2011	Cohesive sediments from wildfire-affected stream and undisturbed stream (reference)	Erosion characteristics of cohesive sediments and factors influencing bed stability (i.e. biofilm development and consolidation)	Wildfire-affected sediments 2 day τ_{crit}= 0.12 Pa 7 day τ_{crit}= 0.23 Pa 14 day τ_{crit}= 0.31 Pa

In-situ annular flume	Amos et al., 1992	Cohesive sediments from two regions of the Bay of Fundy. Regions had differing biology, ice loading and wave effects	Erosion characteristics of sediments	Observed three patterns of erosion Type Ia, Ib and II. Type Ib erosion occurred at all sites with critical shear stress values between 1.0 and 4.4 Pa
	Droppo and Amos, 2001	Contaminated lacustrine sediments	Structure, stability and transformation of surface fine-grained laminae	SFGL τ_{crit}= 0.32 Pa, time series of erosion thresholds showed SFGL reconsolidated rapidly and increased in strength
	Amos et al., 2003	Contaminated lacustrine sediments	To evaluate three different methods of deriving erosion; compare physical behavior of sediments with marine counterparts and examine the effects of ploughing and chemical treatment	Mean erosion thresholds for three methods were determined

1.3.6 The Effects of Biota on Sediment Transport Dynamics

Dreissenids increase sedimentation rates significantly over natural rates in the near shore zones of lakes (Klerks et al., 1996; Dobson and Mackie, 1998). Using sedimentation traps, Dobson and Mackie (1998) found biodeposition rates 8 times higher in traps with mussels, than in control traps (no mussels). Similarly in Lake Erie, Klerks et al. (1996) observed biodeposition rates to be 50% higher than natural sedimentation rates. Although the majority of filtered materials are released back into the water column as feces or pseudofeces (undigested materials), the biodeposits tend to more rapidly settle out (Klerks et al., 1996) resulting in reduced seston levels in the water column. Biodeposits produced by filter-feeding bivalves are agglomerated aggregates of particles from the water column and their settling rates can be much greater than their constituent particles (Haven and Morales-Alamo, 1966; McCall, 1979; Giles and Pilditch, 2004). Thus filter-feeding bivalves such as dreissenids increase the flux of materials to the benthos, increasing sedimentation rates, and deposit materials which may otherwise not settle due to their hydrodynamic or chemical characteristics (Taghon et al., 1984). Upon deposition local hydrodynamics can initiate resuspension of biodeposits and bed sediments, thus redistributing the materials (Giles and Pilditch, 2004). Walz (1978) estimate that roughly only 8% of biodeposits produced by dreissenids become resuspended into the water column. Accordingly, this has consequences for the existing sediment bed (chemically and biologically) and the abundance and diversity of benthic biota (Giles and Pilditch, 2004).

Biodeposits are readily available sources of repackaged organic carbon and very abundant in aquatic systems (Wotton and Malmqvist, 2001) yet much less attention has been given to their contribution in the transport of nutrients, pollutants and other particles to benthic environments (Urban et

al., 1993). Several factors contribute to the effects biodeposits have on benthic sediments and biota including; the composition of the benthic community, the hydrodynamics of the local system, the quality and quantity of materials in suspension and the physical and chemical composition of the existing sediment bed (Giles and Pilditch, 2004). Several studies have focused on the enrichment of sediments surrounding bivalve communities due to their biodepositional processes. For example, Norkko et al. (2001), found seafloor sediments in close proximity to a large pinnid bivalve, *Atrina zelandica*, community to be enriched with carbon and nitrogen which supported a more diverse community of macrofaunal assemblages. Studies by Kautsky and Evans (1987) and Stewart and Haynes (1994) report increased abundance in macroinvertebrate communities, associated with dreissenid biodeposits. While these studies focus on enrichment effects that biodeposits have on benthic environments, few studies have quantified, the resuspension and transport characteristics of mussel derived biodeposits (Giles and Pilditch, 2004).

1.3.7 Experimental Studies Quantifying the Physical Characteristics of Biodeposited Materials

Few studies have quantified the physical characteristics of benthic fauna biodeposits. Taghon et al. (1984) measured the density, settling velocity and particle size of biodeposits produced by a deposit feeding tube worm, *Amphicteis scaphobanchiata*. Giles and Pilditch (2004) quantified the settling velocity and erodibility of biodeposits produced by the common mussel, *Perna canaliculus*. Both studies sought to quantify the physical characteristics of biodeposits due to their significant impacts (i.e. flux and redistribution of particles to the benthos which may not otherwise deposit (Giles and Pilditch, 2004)) on the benthic bed. Roditi et al. (1997) performed one of the first comprehensive studies to physically characterize biodeposits produced by zebra mussels. Flow through chambers, were deployed into the Hudson River to collect mussel biodeposits, one with mussels and one without mussels (control). Naturally settling materials and biodeposits settled through rigid mesh and accumulated in a tray below. On average, mussel associated chambers contained 39% more sediment than controls and deposition rates per individual of 2.3 ± 0.4 mg hour^{-1} were estimated (Roditi et al., 1997). The study did not however quantify the physical characteristics (particle size, porosity, density) of the biodeposits produced by the mussels nor determine their transport characteristics (τ_{crit}, erosion rates) within the system.

In a laboratory setting, Roditi et al. (1997) examined the resuspension of biodeposited materials by placing the collected biodeposit mixture (combination of biodeposits and naturally settling particles) and control mixture (naturally settling particles without mussels) into 1-L water filled beakers. Suspended stirring rods were placed into each, rotated and rotational speed was increased at 4 minute

intervals (Roditi et al., 1997). Suspended sediment concentrations were measured using light absorbance. They observed sharp increases in suspended sediment concentrations at lower RPMs for biodeposit mixture versus the control mixture. They reported ranges of critical bed shear stress force of 0.41 Pa and 0.74 Pa for biodeposit mixture and control mixture, respectively (Roditi et al., 1997). These results do not take into account the effects of mussels on reuspension dynamics, thus the following section will review literature regarding the influence of biogenic structures on sediment transport dynamics.

1.3.8 Experimental Studies Determining the Transport Properties of Biodeposited Materials

Various experimental flumes, tunnels and erosion devices have been used to quantify material fluxes and erodibility of sediments associated biological processes in marine benthic habitats (Black and Paterson, 1997). These devices have their strengths and limitations and are constructed to suit the specific needs of each study (Widdows et al., 1998a). At the Plymouth Marine laboratory, UK, an experimental flume was designed and constructed for both in-situ and laboratory measurements of biodeposition rates and erosional potential of estuarine sediments (Widdows et al., 1998a; 1998b). In addition to determining critical shear stress for erosion of sediments and resuspension velocities of biodeposited materials, there are a range of biological and physical processes that can be quantified using the annular flume which are presented in Widdows et al. (1998a). Widdows et al. (1998b) used an experimental benthic annular flume, to investigate the impacts infaunal bivalves, *Macoma balthica*, and epifaunal bivalves, *Mytilus edulis*, had on seston and sediment fluxes of intertidal sediments (Widdows et al., 1998b). They found that the bioturbator *M. balthica* increased resuspension and/or erodibility by 4-fold over the control. Additionally they found a strong correlation between *M. balthica* densities and sediment resuspension, concluding that *M. balthica* enhance bed sediment erodibility (Widdows et al., 1998b). Alternatively, when determining erodibility of sediments associated with epifaunal bivalves *M. edulis*, Widdows and colleagues (1998b) found a 10-fold reduction in sediment eroded in the presence of *M. edulis* at mussel bed densities with 50-100% coverage versus 0% coverage (Figure 1.8).

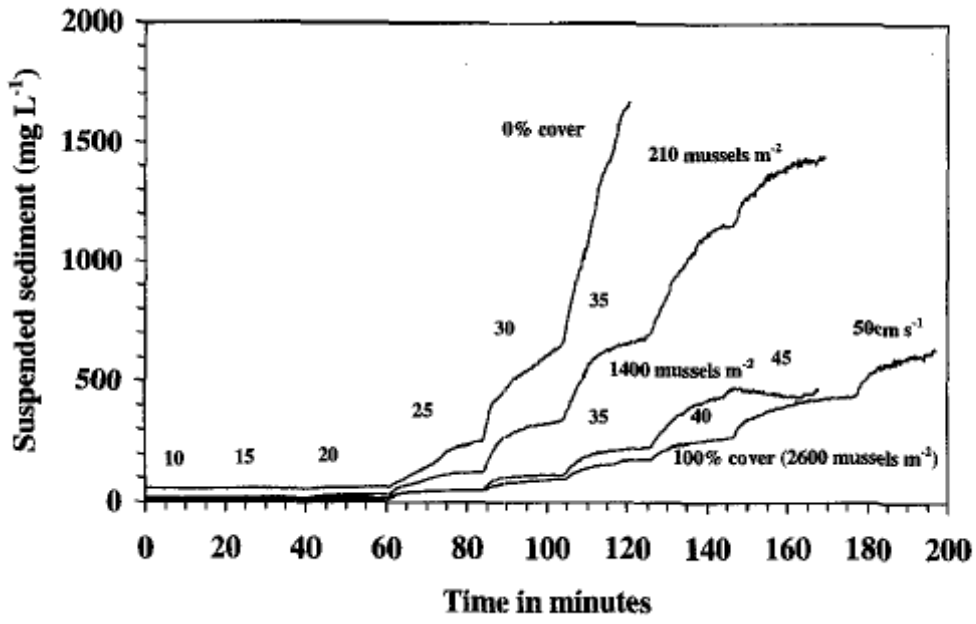


Figure 1.8: The effect of Mytilus edulis density (numbers m^{-2}) on sediment resuspension time-course following a stepwise increase in current velocity from 10 cm s^{-1} to 50 cm s^{-1} in 5 cm s^{-1} increments, each with a duration of 20 minutes (Widdows et al., 1998).

The study also found that mussel beds with <50% coverage had suspended sediment concentrations almost similar to 0% covered beds at velocities of 30-35 cm s^{-1} (Widdows et al., 1998b). They attribute the high suspended sediment concentrations to the water flow scouring around the shells and removing sediments (Widdows et al., 1998b). Vogel (1994) presents three scenarios for flow over surfaces; 1) independent flow; 2) interactive flow and 3) skimming flow. If shells for example are well spaced (i.e. their heights much less than their distances) or isolated, then each shell acts independently, with anterior and posterior eddies forming (Vogel, 1994). The isolated shells are expected to increase shear stress causing erosion in the bed area immediately surrounding them (Eckman and Nowell, 1984). If the shells are more closely spaced, the rear eddy interacts with the anterior eddy of the next, creating a less stable flow pattern (Vogel, 1994). If shell densities are beyond a threshold density and closely packed, water can be expected to flow over rather than through the shells (i.e. skimming flow) with decreasing flow velocities and turbulence levels among them (Nowell and Jumars, 1984).

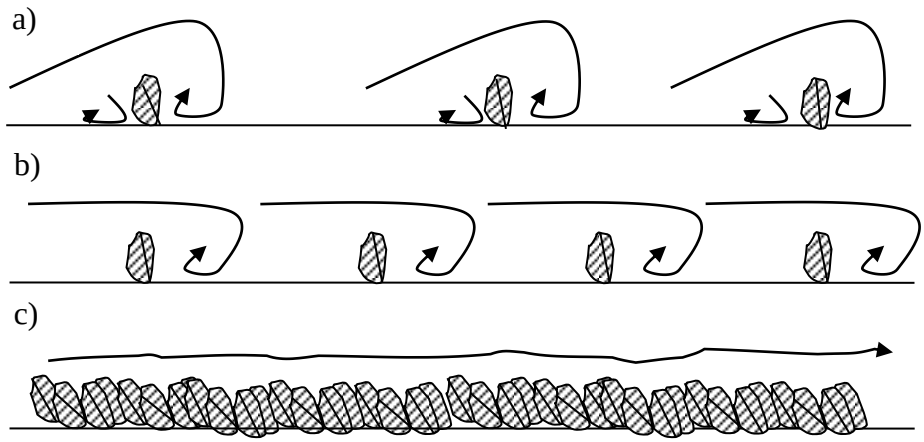


Figure 1.9: Types of flow over 3 separate regimes; a) independent flow; b) interactive flow; c) skimming flow (Vogel, 1994).

Widdows et al. (2009) characterized sediment erosion and deposition associated with mussel beds in the Menai Strait, UK, with high tidal currents. In this study, four cores forming an annulus which fit into the annular flume were used to dig up sediment and attached biota (i.e. mussels) and samples were transported to the laboratory (Widdows et al., 2009). Comparisons were made between bare sediments, sediments with 55% mussel coverage and 95% mussel coverage in a flume (Widdows et al., 2009). Their results indicated that bare sediments are more stable, and the critical shear stress for erosion is significantly higher for bare sediments than mussel associated sediments (Widdows et al., 2009). They attributed the increased erodibility of mussel associated sediments to their self-organised structure and the turbulent kinetic energy (TKE) produced by the mussels (Widdows et al., 2009). Through resuspension mussels enhance their food availability, to offset phytoplankton depletion directly over mussel beds (Widdows et al., 2009). When determining the erodibility and transport of biodeposits, it is important to consider the surface roughness and associated turbulence produced to accurately quantify erosional characteristics of sediments.

Extensive research has been conducted on the impacts of invading dreissenids on the ecology and economics of aquatic systems (Lundyanski et al., 1993). Many studies have investigated their origins, biology, environmental requirements and potential impacts (Hebert et al., 1989; MacIsaac et al., 1991; Lundyanski et al., 1993; Strayer et al., 1999). Dreissenid distribution and spread has been modeled and future recommendations have been established (Fahnential et al., 2010). While some studies have acknowledged dreissenid impacts on suspended sediment concentrations and increased biodeposition rates (Klerks et al., 1996; Dobson and Mackie, 1998), no studies have quantified the physical characteristics and erosional behaviour of biodeposited materials produced by dreissenids.

1.4 Chapter Outline

Chapter 2 describes the experimental design collection methods, flume experiments and methodology as well as data analysis.

Chapter 3 reports the findings of flume experiments on the biodeposit mixture (lake bed sediment and biodeposits), pure biodeposits and runs with and without dreissenids. Data on the influence of consolidation and development of biofilm on the biodeposit mixture and biodeposits only were observed and recorded. The PHOENICS model was applied to describe bed shear stress over smooth flume bed. The model could not however be used for flume beds with added roughness, and therefore results from erosions experiments with mussels are reported in revolutions per minute (RPM). The physical characteristics of the biodeposits are presented, with data collected from the density, porosity and settling velocity. Grain size distributions and images are also illustrated.

The above mentioned results are then discussed in Chapter 4. The trends and relationships of current results to other published studies are discussed and the mechanisms and applications for processes are explained.

Chapter 5 states the conclusions made from the current study. The relevance of the current results and the implications they have on the nearshore shunt hypothesis are discussed. Additional research needs are identified.

Chapter 2: Methods

2.1 Experimental design

The goal of this thesis was to quantify the physical nature and erodibility of dreissenid mussel biodeposits to further examine their effects on nutrient transport dynamics in aquatic systems. To achieve these goals, experiments were conducted in an annular flume to quantify the effects of consolidation time and the physical presence of dressenids on erosional characteristics of mussel related biodeposits. During each flume experiment, the speed at which the flume rotated was controlled and increased in step-wise increments. Samples were collected directly from the flume to determine the suspended sediment concentrations at each incremental step. Critical shear stress for erosion of the bed materials was determined by an increase in measured suspended sediment concentrations and through visual observations. In 2008, flume experiments were conducted with a mixture of lake bottom sediments and biodeposits collected directly from dreissenid beds (biodeposit mixture). Materials were given 2, 7, and 14 day consolidation times to determine bed characteristics and the physical nature of the materials. The factorial design for all years, runs, and parameters measured is presented in Table 2.1.

Various studies have found that bivalve beds physically modify and alter the transport of suspended particles in the near bottom environment (Zaiko et al., 2010; Gutierrez et al., 2003). To examine this parameter, mussels were collected in 2009 and 2010 and placed into a flow through weir box, on flume fitted glass plates, to acclimate and harvest biodeposits. Mussels and biodeposits were transferred on glass plates into the same annular flume as used in the previous year, with filtered lake water. Experiments were conducted in the same manner as the previous year, but without consolidation periods, due to animal care limitation.

Table 2.1: Experimental factorial design of study.

Year	Run #	Treatment (Biodeposits +)	Consolidation period (days)	τ_{crit}	Erosion rate	Settling velocity	Density	Porosity	Grain size
2008	1a	Sediment	2	x	x	-	-	-	x
	1b	(without mussels)	7	x	x	-	-	-	x
	1c		14	x	x	x	x	x	x
2009	2	With mussels	-	x	x	x	x	x	x
2010	3a	With mussels	-	x	x	-	-	-	x
	3b	Without mussels	2	x	x	x	x	x	x

2.2 Weir box description

Mussels collected during 2009 and 2010 were placed into flow-through weir box (Figure 2.1). The weir box consisted of a horizontal fiber glass trough approximately 4.2 m in length, 0.6 m in width and 0.46 m

in height (Figure 2.1). At the upstream end of the trough, a submersible pump provided Hamilton harbour water into the trough. To regulate the flow through the weir box, a v-notch was located on the downstream wall of the trough. From the bottom of the weir to the tip of the notch was 0.18 m.

Figure 2.1: Design and dimensions of the weir box.

The residence time was determined using equation (2.1):

$$RT = \frac{V}{Q} \tag{2.1}$$

where V is the volume (L) and Q is the flow rate (L min^{-1}). The mean residence time was 114.4 minutes. After mussels were placed in the weir box, cut Styrofoam™ was placed on top of the water to avoid predation and reduce sunlight (i.e. cool the water).

2.3 Experimental flume description

A 2 m stainless steel annular flume (Lau, 1995) was used to measure τ_{crit}, erosion rate and resuspension behaviour of mussel biodeposits. The flume consisted of a stainless steel annular trough with an outside diameter of 2 m, width of 0.2 m and wall height of 0.12 m (Droppo et al., 2007). The top cover was attached to a center shaft that could be raised, lowered and rotated by a motor (Lau and Droppo, 2000). When locked into place, the lid sits on top of the water in the trough. The rotation of the top lid exerts shear and generates flow (Droppo et al., 2007).

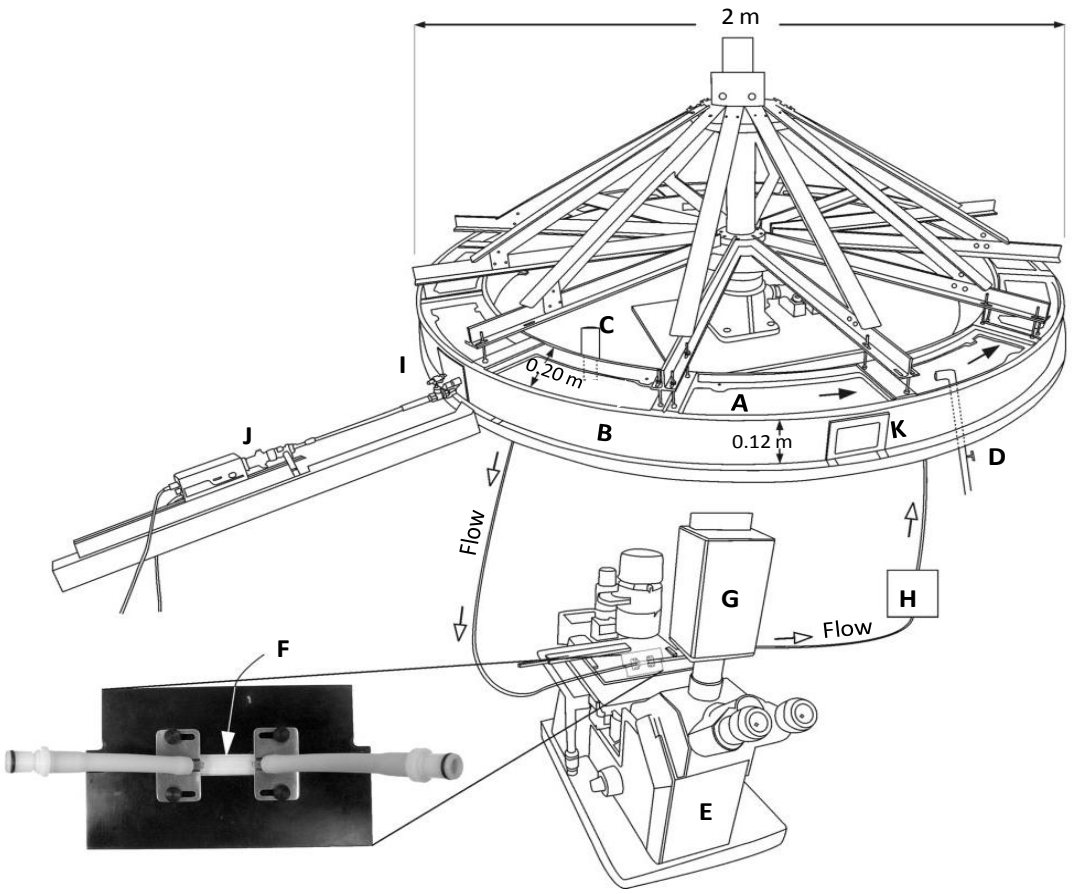


Figure 2.2: Experimental flume used in all three experiments. A) rotating lid, B) flume trough, C) optical backscatter probe, D) Sampling port, E) microscope, F) flow cell, G) digital camera, H) peristaltic pump to pull in eroded materials, I) borescope, J) video camera, K) flume window (Droppo et al., 2009).

Velocity profiles spanning the bottom 10% of the flow depth were used to calculate values of bed shear stress for a smooth bed (Droppo et al., 2007). Lau and Droppo (2000) performed similar annular flume experiments and calculated the average bed shear stress (τ_o) against the cover speed (L_S) and generated equation (2.2):

$$\tau_o = 0.1 \times (0.164 L_s)^2 \qquad (2.2)$$

where τ_o is the bed shear stress in pascals (Pa) and Ls the lid speed in revolutions per minute (RPM). Figure 2.3 elucidates the flows generated by the flume over a smooth bed with nearly evenly distributed bed shear stresses across the width of the trough (Krishnappan and Engel, 2004; Krishnappan, 2007). With the rotational speeds of the flume as a parameter, the bed shear stresses are plotted as a function of transverse distance across the width of the flume (Krishnappan, 2007). In Figure 2.3 the points are measured values and the lines are predictions from a three dimensional (3D) hydrodynamic model called the PHOENICS (Rosten and Spalding, 1984; Krishnappan and Engel, 2004; Krishnappan, 2007). There

is good agreement between the measured and predicted values (Figure 2.3) and the flow is considered two dimensional (Krishnappan and Engel, 2004).

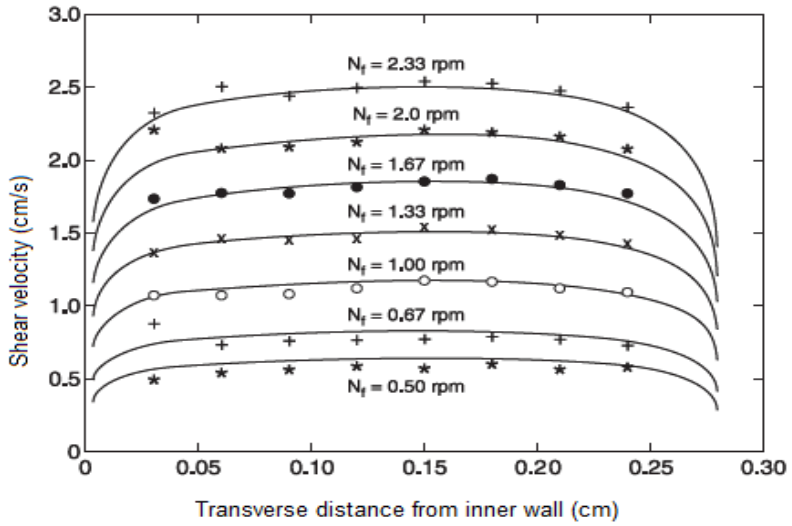


Figure 2.3: Shear velocity distributions as a function of flume speed for the flow depth of 0.12 m in smooth bed flows with 2 component rotational flow (Krishnappan and Engel, 2004).

When the PHOENICS model was applied over a rough bed, the shear stress distributions in this flume were skewed towards the outer wall of the flume (Figure 2.4). Krishnappan and Engel (2004) attribute the skewness to the development of a secondary circulation cell during the rotation of the flume over rough surfaces. To correct for this, they added roughness to the bottom of the rotating ring to compensate the roughness on the bottom of the flume (Krishnappan and Engel, 2004). In the current study, the roughness of the rotating ring was not altered and the distribution of shear stress was not linear. Accordingly, the changes in suspended sediment concentrations under changing rotational speeds will be presented as a function of revolutions per minute (RPM) rather than shear stress.

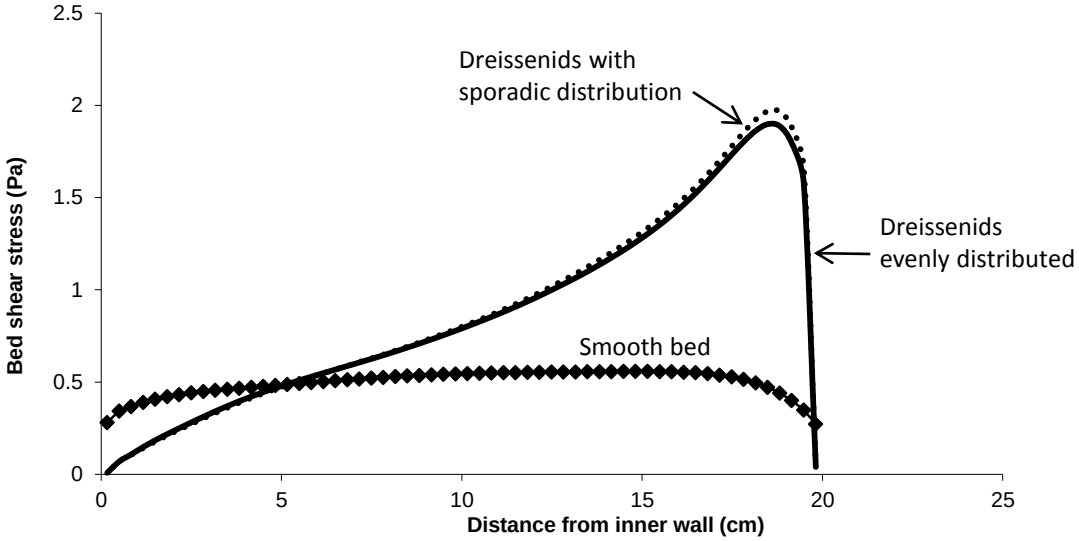


Figure 2.4: Prediction of the effect of roughness on shear stress from mussels using the PHOENICS model for a one component rotational flow in the annular flume (Source Dr. B. G. Krishnappan).

2.3.1 Flume Instrumentation

An optical backscatter (OBS) turbidity meter was used to continuously monitor changes in turbidity with increasing shear stress in the flume experiments (Figure 2.2C) (Droppo et al., 2007). Mounted on the inside wall of the flume facing upstream, the sensor was calibrated against the suspended sediment concentrations collected during experiments (Droppo et al., 2007). Within the flume trough, there are four, 3 mm sampling ports located at approximately mid-channel and mid-depth (Figure 2.2D) (Droppo et al., 2007). Suspended sediment samples were collected three times during each incremental increase, filtered onto pre-weighed 0.45 μm filters and stored for grain size analysis.

A Zeiss Axiovert 100 inverted microscope with a flow cell and digital camera for imaging were connected to the flume via a series of solenoid y-valves and 5 mm internal diameter tubing (Droppo et al., 2007b). Samples were drawn from the flume into the flow cell, allowed to settle and pumped back with clean water when particle imaging and analysis were complete (Figure 2.2E-H) (Droppo et al., 2007).

A borescope was inserted through the outer wall of the flume to directly view sediment deposited on the flume bottom. The borescope was equipped with a color CCD video camera (interfaced with digital recorder) (Figure 2.2I, J) (Droppo, 2009). Four windows were located on the outside walls of the flume, to monitor suspended sediment concentrations and sediment bed structure (Figure 2.2K).

2.4 Sample Collection

2.4.1 Lake Bed Sediment and Biodeposits Collection (2008)

SCUBA divers carefully collected particulate matter from dreissenid mussel beds located on the eastern shores of Lake Ontario, near Pickering, Ontario, Canada (43°48′42″N 79°03′57″W). Samples were transported to University of Waterloo, Waterloo, Ontario, Canada, where particulate matter greater than 100 µm were removed with the use of a mesh sieve. The materials were then transferred into a 2 m stainless steel annular flume located at the Canadian Centre for Inland Waters (CCIW) in Hamilton, Ontario, Canada.

2.4.2 Dreissenid Collection and Biodeposit Harvesting (2009 and 2010)

Mussels were collected with a galvanized steel scraper and a basket from Hamilton Harbour, Ontario, Canada (43°17′21.35″N, 79°50′7.73″W) (Figure 2.5). Shell quality was examined and attached debris was removed. Mussels were then placed into a cooler with lake water and transported directly to CCIW. Approximately 3000 live mussels were collected in 2009 and 1500 (11.61 mm SD± 2.05) in 2010.

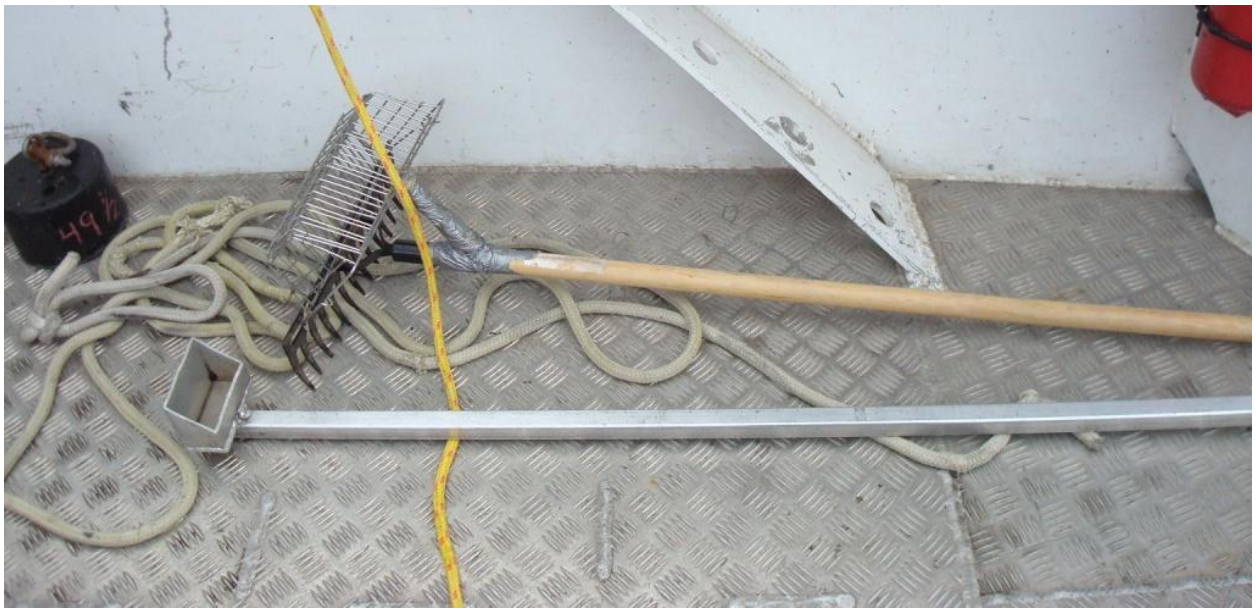


Figure 2.5: Tools used for collecting mussels from rocks in 2009 and 2010.

Mussels were then placed into the weir box (Figure 2.1) and allowed to settle and attach to frosted glass plates, which fit into the annular flume (Figure 2.6). Water quality was monitored daily at the inflow and outflow of the weir (temperature, O_2, conductivity, pH). Mussels were given approximately 2

weeks to establish on frosted glass plates. Mussels observed migrating off plates onto the bottom and walls of the weir box were gently removed and placed back onto the plates.

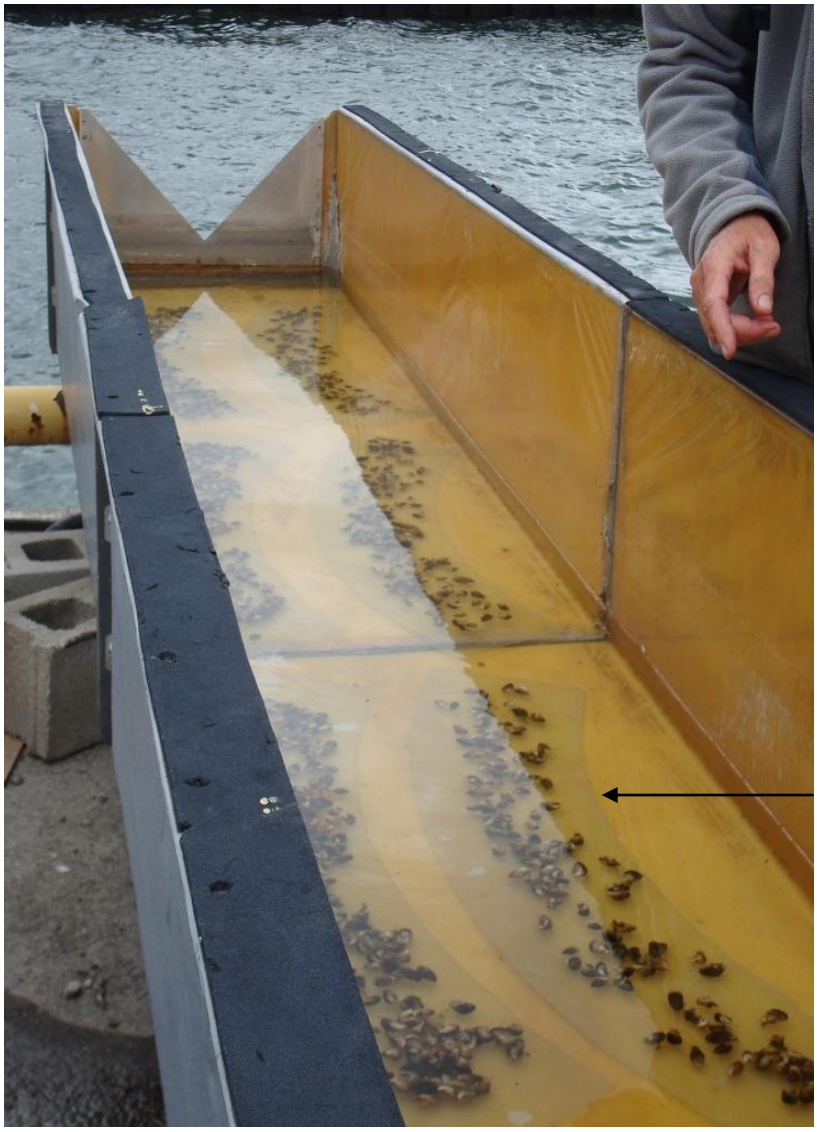


Figure 2.6: Weir box with dreissenid mussels established on glass plates.

2.5 Experimental Procedure

Experiments 1a, 1b, and 1c consisted of placing a mixture of collected lake bottom sediments and biodeposits into the annular flume (Table 2). Prior to the first experimental run, the sediment and biodeposit mixture (which hereafter be referred to as biodeposit mixture), was placed into the flume with filtered lake water. The flume lid was lowered, locked into place and rotated at a high speed to entrain and mix all the biodeposits, gradually the speed was reduced, then stopped to settle the biodeposit mixture

(Droppo et al., 2007; Stone et al., 2011). Consolidation periods of 2, 7 and 14 days were used in three experimental flume runs (see Table 2). In the two subsequent years, experiments 2 (2009) and 3a (2010), dreissenids on frosted glass plates and biodeposits produced (hereafter referred to as pure biodeposits) were transferred from the weir box to the flume, and the flume was filled with filtered lake water. For the final experiment (3b), biodeposits were siphoned out of the flume following experiment 3a. Mussels attached to glass plates were removed, the flume was cleaned with clear tap water, and glass plates were placed back into the flume without mussels. Siphoned biodeposits were placed back into the flume with filtered lake water (see Table 2).

2.5.1 Erosion Experiments

Erosion experiments were conducted by rotating the flume from rest and increasing its rotational speed (and therefore shear stress) in 9 minutes increments until bed sediments were completely eroded (Stone et al., 2011). The critical shear stress for erosion was determined through visual confirmation and the detection of suspended sediment concentrations above ambient levels (Droppo et al., 2007). Suspended sediment concentrations were measured by collecting samples of suspended sediments through sampling port every 8 minutes from the start of a 9 minute interval (Stone et al., 2011). Samples were also collected at 8 minute intervals for grain size analysis and to examine particle morphology.

2.5.2 Settling Velocity Measurements

The settling velocity of biodeposits was determined following the methods of Droppo et al. (1997; 2007). A wide mouth pipette (3.74 mm) was used to collect a drop of sediment which was then placed into an insulated 2.5 L capacity settling column (Droppo et al., 2007). A Nikon™ stereoscopic microscope paired with a digital Hamamatsu™ video camera were used to capture digital images of settling flocs passing through the field of view (Droppo et al., 2007). The settling velocity was then measured digitally by overlaying two digitally captured frames containing an identified floc separated by a known time interval using Open Lab™ (Droppo et al., 2007).

2.6 Sample Analysis

2.6.1 Measured Suspended Solids Concentration

Suspended sediment samples were collected pre-, during and post- flume run for all experiments. Approximately 125 mL samples were collected through sampling port. Water samples were then

transported back to the University of Waterloo and stored at 4° C until analysis. Applying vacuum filtration, water samples were filtered through pre-dried and weighed 0.45 µm filters. Prepared filters were placed into an oven at 100°C for 24 hours. Following the drying period, dried filters were weighed again. The following equation was used to calculate the suspended solids concentration:

$$\text{TSS} = \frac{(\text{A} - \text{B}) \times 1000}{\text{V}} \tag{2.3}$$

where TSS is total suspended solids (mg L^{-1}), A is the post filter weight (mg), B is the pre filter weight (mg), V is the volume of filtered sample (mL). To determine the organic content of the samples, following the second weighing, filters were placed into a muffle furnace at 500° C. The following equation was used to determine the ash free dry weight (AFDW) of the sample:

$$\text{AFDW} = \text{B} - \text{A} \tag{2.4}$$

where B is the pre weight of the filter (mg), and A is the post weight of the filter (mg).

2.6.2 Erosion Rates

Erosion rates were calculated for each experimental run using the following equation:

$$\text{Erosion rate} = \frac{\left(\frac{[SS] \times V}{A}\right)}{T} \tag{2.5}$$

where [SS] is the suspended sediment concentration (mg L^{-1}), V is the volume of water in the flume (L), A is the area of the flume (m^2), and T is the time elapsed throughout experiment (seconds).

2.6.3 Grain Size Analysis

The grain size distributions of suspended materials at each incremental shear stress were measured using image analysis. At each sampling interval (8 minutes), one to two milliliters of suspended sediment was collected from the flume's sampling port, into a 50mL settling column fitted with a 0.45 µm Millipore HA filter, filled with distilled water (Stone et al., 2008). Using low vacuum filtration, to prevent floc breakage, the sample was settled onto the filter, then placed into a petri dish and dried at room temperature for approximately 48 hours (Shantz et al., 2004; Stone et al., 2008).

Similar to the methods of Stone et al. (2008) particles settled onto filters were viewed under a Zeiss Axiovert 100 microscope fitted with a Sony XC75 CCD camera connected to a computer running Northern Eclipse™ image analysis software. To distinguish particles from filter background, filters were rendered (semi) transparent by applying three drops of Stephens Scientific low viscosity immersion oil (Shantz et al., 2004; de Boer and Stone, 1999). Particles were sized to a lower resolution of 2 μm (10x objective), and analyzed until approximately 2500 particles were collected. The digital output converts the area data to Equivalent Spherical Diameter (ESD) which is used to determine particle size and shape and grain size distributions by number and by volume (Droppo and Ongley, 1992).

2.6.3 Porosity and Density Measurements

Measured settling velocity and aggregate size were used to estimate density using Stoke's Law (Droppo et al., 2007). Although Stoke's Law assumes laminar conditions and single spherical particles (Droppo et al., 2007), it has been shown to provide information on how aggregate settling velocity, density and porosity are related to aggregate size (Droppo et al., 1997). Following the methods of Droppo et al. (2007), aggregate porosity was calculated using a mass balance equation assuming typical density of dried silt and clay of 1.65 g cm^{-3}:

$$\varepsilon = (\rho s - \rho f)/(\rho s - \rho w) \qquad (2.6)$$

where ε is the aggregate porosity (%), ρs is the density of the dried solid material (g cm^{-3}), ρf is the wet density of the aggregate (g cm^{-3}) and ρw is the density of the water (g cm^{-3}) (Li and Ganczarczyk 1987).

Chapter 3: Results

Results from the flume experiments are presented in this chapter. The first section presents data on biodeposit erodibility as a function of shear stress and consolidation time. The second section presents data on the effects of surface roughness (i.e. dreissenids present in flume) on biodeposit erodibility. The final section presents porosity, density and settling velocity data.

3.1 Erodibility of Biodeposits with Varying Consolidation Times

For different sediment types (biodeposit mixture and pure biodeposits) and three consolidation periods (2, 7 and 14 days), time series plots were generated for each experiment (Figure 3.1 and Figure 3.2). The critical shear stress (τ_{crit}) for erosion was determined for each using visual observations and plots. Type 1b erosion was used to define the critical shear stress for erosion, a point at which sediments begin to creep, saltate and detach from the bed. In Figure 3.1(b, d, f) and Figure 3.2b, erosion rates are displayed.

The critical shear stress for erosion after a two day consolidation period was 0.13 Pa. Prior to the erosion threshold being reached, minor entrainment of the SFGL was observed for 46 minutes. The peak erosion rate was 0.0867 g m^{-2} s^{-1} after 59 minutes at a bed shear stress of 0.43 Pa. For the 7 day experimental flume run, the initial type 1b erosion was observed for up to 61 minutes until the critical shear stress for erosion increased to 0.26 Pa with peak erosion rate of 0.045 g m^{-2} s^{-1} after 79 minutes at a bed shear of 0.32 Pa. For the 14 day consolidation period experiments, the critical shear stress for erosion was 0.15 Pa after 56 minutes. The highest observed erosion rate for the 14 day sediments was 0.08 g m^{-2} s^{-1} and occurred at a bed shear stress of 0.4 Pa after 99 minutes (see Table 3.1).

For sediments harvested from weir box (pure biodeposits), the critical shear stress for erosion was 0.052 Pa, for a 2 day consolidation period, which occurred at approximately 69 minutes. Peak erosion was observed after 109 minutes at a rate of 0.00498 g m^{-2} s^{-1} with a bed shear stress of 0.12 Pa (see Table 3.1).

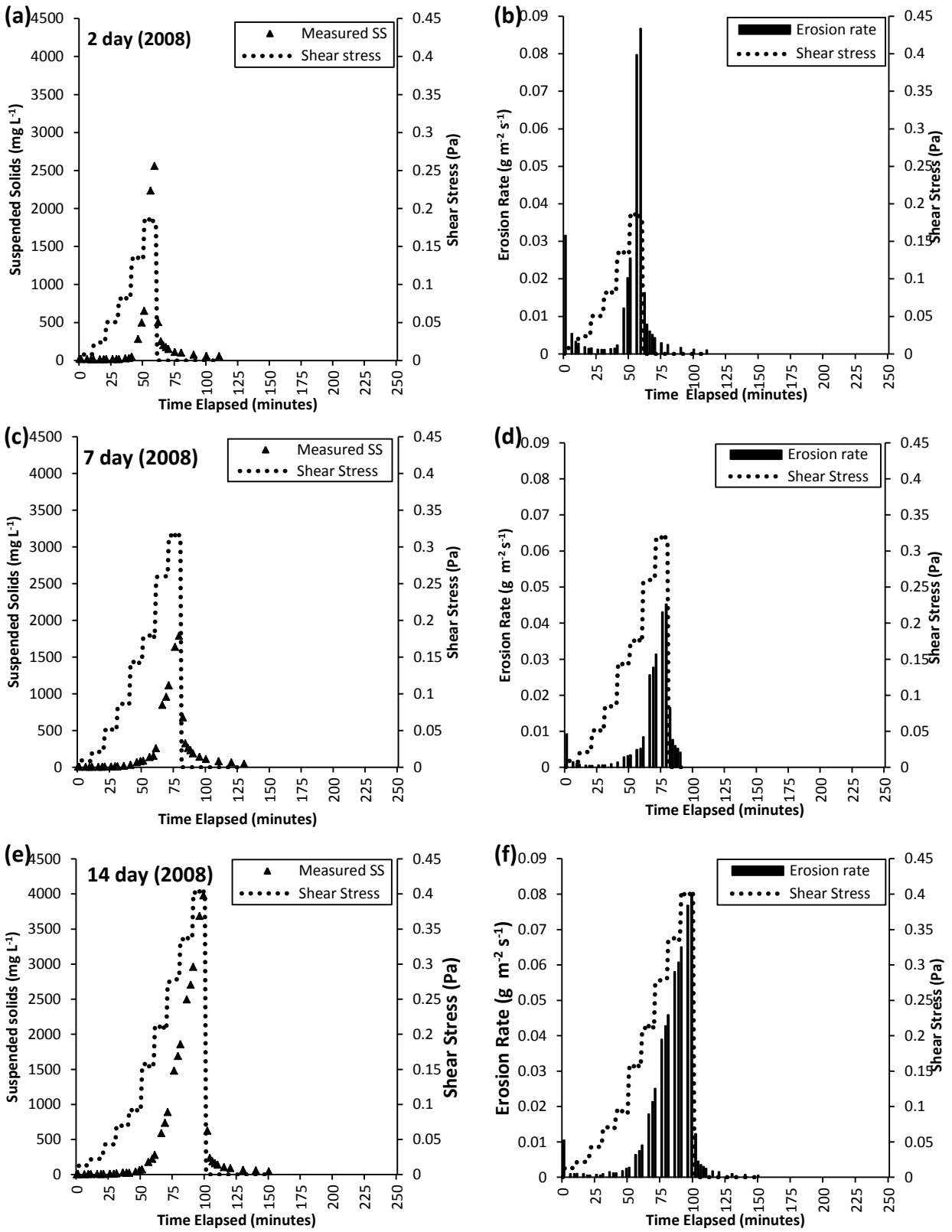


Figure 3.1: Changes in sediment concentration (a, c, e) and erosion rates (b, d, f) as a function of shear stress for 2, 7 and 14 day consolidation periods for the biodeposit mixture collected in 2008.

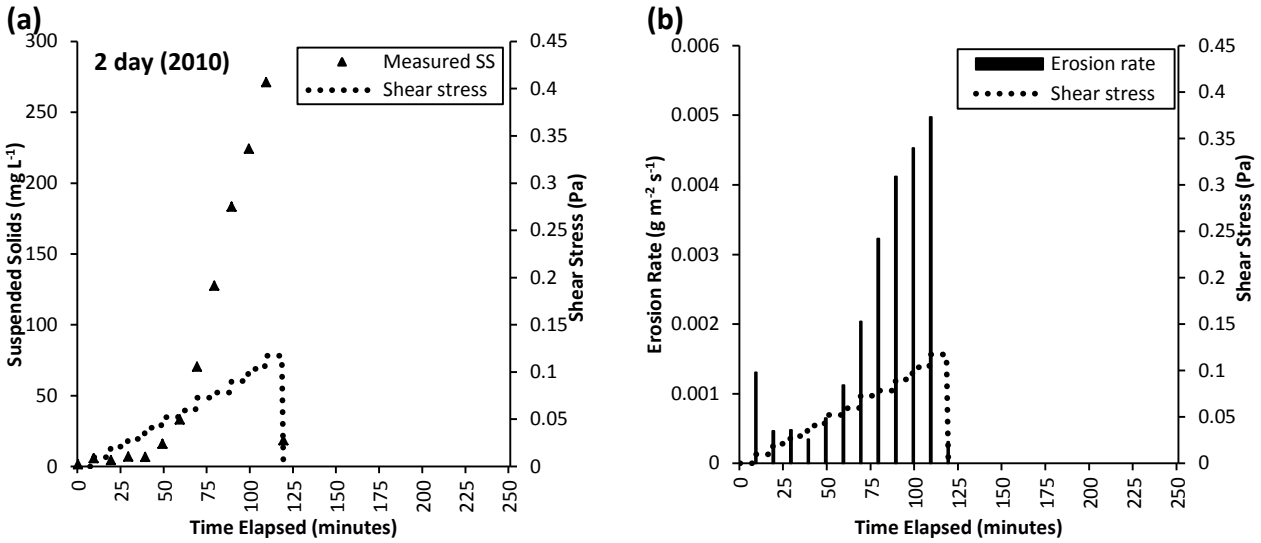


Figure 3.2: Changes in sediment concentration (a) and erosion rate (b) as a function of shear stress for 2 day consolidation period for pure biodeposits harvested in 2010.

3.2 Erodibility of Biodeposited Sediments in the Presence of Dreissenids

In 2009 and 2010 experiments, dreissenids attached to glass plates were placed directly into the flume and offered a 48 hour acclimation period then experiments commenced. Mussels collected in both years were observed distributed in clumps and strings with patches of bare glass in between (Figure 3.3). On the bare surfaces, settled materials and strands of EPS were observed. Revolutions per minute (RPM) are reported for runs with mussels.

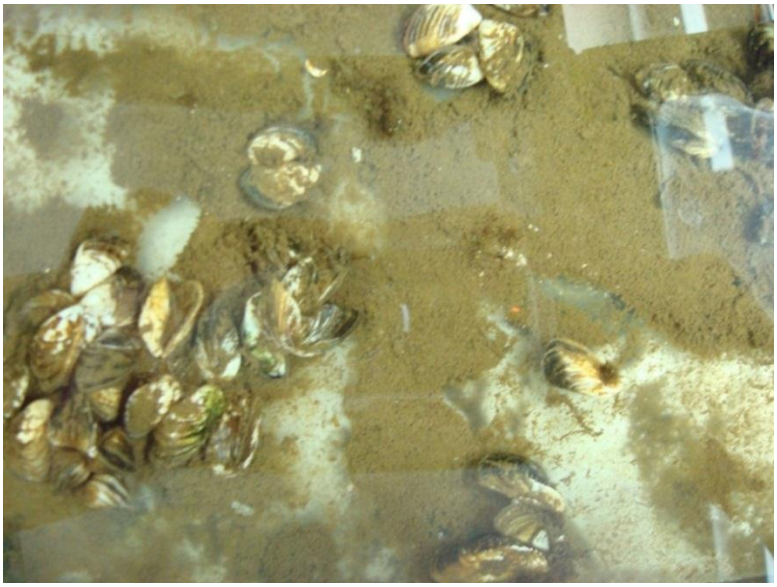


Figure 3.3: Distribution of dreissenids in the flume during 2010 experiments.

In 2009, approximately 3,000 mussels were placed into the flume (Figure 3.4 a, b). The critical RPM for erosion was 5.83 RPM and the peak erosion rate was 0.0023 g m^{-2} s^{-1} after 93 minutes at a revolution of 10.17. During the 2010 experimental flume run (run 3a), approximately 1500 mussels were placed into the flume, and a critical RPM for erosion was never reached, peak erosion rates reached 0.0007 g m^{-2} s^{-1} after 39 minutes at 3.9 RPM. Figure 3.4 a, c, illustrates the change in suspended solids concentrations as a function of time and RPM. Figure 3.4 b, d, shows the calculated erosion rate as a function of time and shear stress (see Table 3.1).

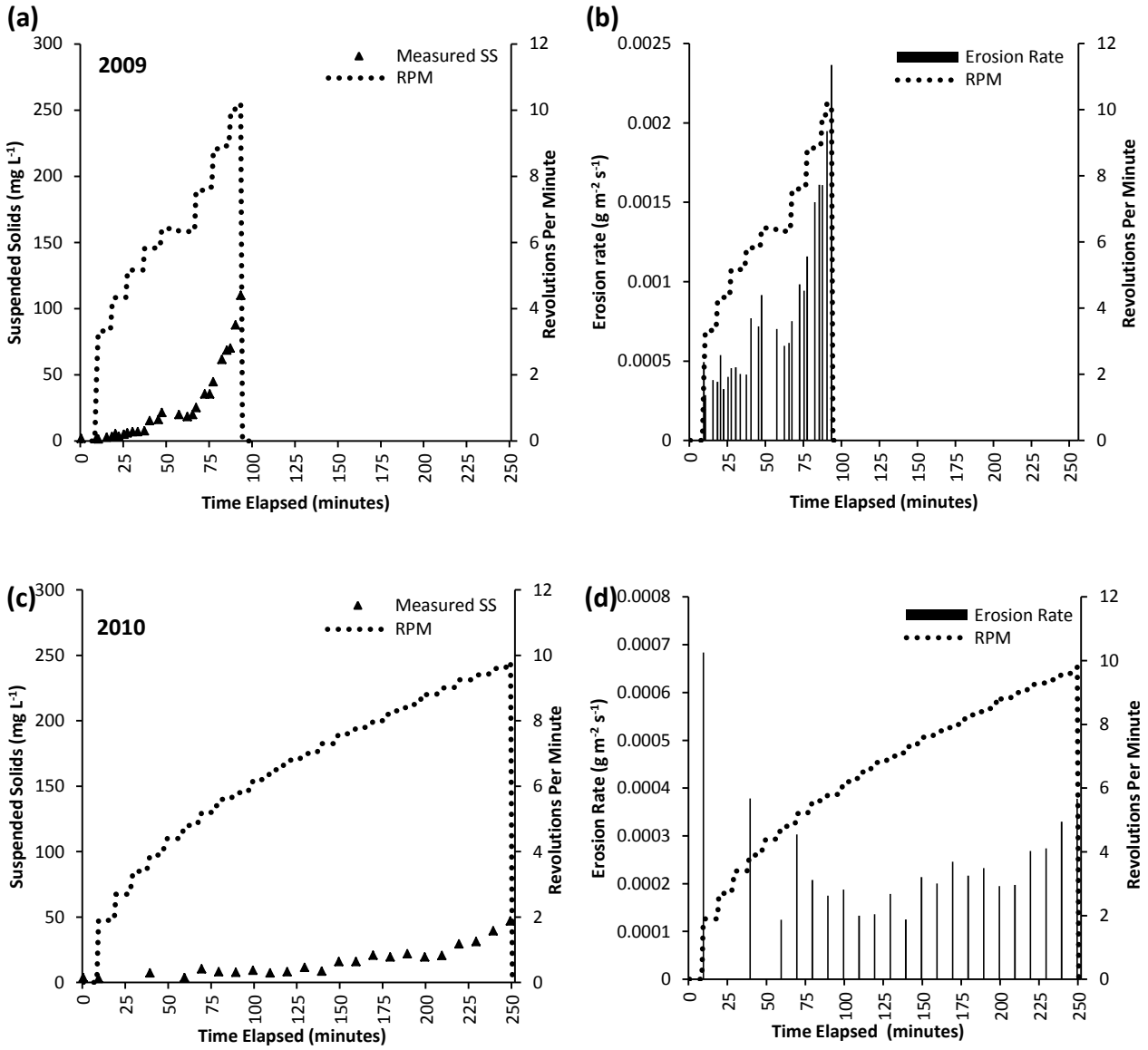


Figure 3.4: Changes in sediment concentration (a, c) and erosion rates (b, d) as a function of time (minutes) and shear stress (Pascals) for flume experiments with dreissenids in 2009 (a, b) and 2010 (c, d).

3.3 Physical Characteristics of Dreissenid Biodeposits

3.3.1 Settling Velocity

The floc settling velocities for the biodeposit mixture and pure biodeposits collected in 2008 and 2010 respectively are illustrated in Figure 3.5. In 2008, the average settling velocity was 2.56 mm s^{-1} (SD±2.30 mm s^{-1}) with a range between 0.54 mm s^{-1} and 10.82 mm s^{-1}. The median floc size (D_{50}) was 232.05 µm (SD±115.43µm) with floc size ranging between 63.1 µm to 602.6 µm. In 2010, the average floc settling velocity was 2.53 mm s^{-1} (SD±3.27 mm s^{-1}) with a range of 0.19 mm s^{-1} to 16.25 mm s^{-1}. The D_{50} was 309.6 µm (SD±229.0 µm) with floc size ranging between 71.3 µm and 1376.6 µm (Table 3.1).

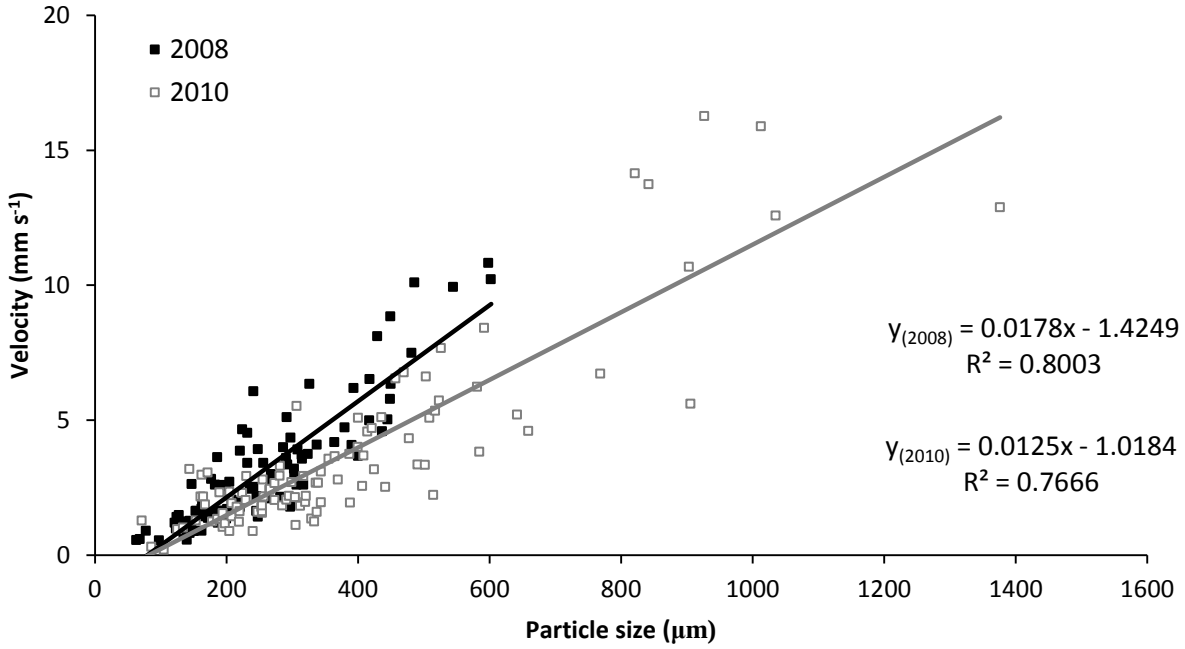


Figure 3.5: Floc settling velocity as a function of particle size from experimental flume runs performed in 2008 and 2010.

3.3.2 Porosity and Density

In 2008, the median density was 1.08 g cm^{-3}(SD±0.05 g cm^{-3}) and ranged from 1.04g cm^{-3} and 1.27 g cm^{-3}. The median porosity was 88% (SD±7.12 %) with a range of 58.7 to 94.3% (Figure 3.6). In 2010, the median excess density was 1.05 g cm^{-3} (SD±0.06 g cm^{-3}) with a range of 1.01g cm^{-3} to 1.46 g cm^{-3}. The porosity ranged from 29% to 98.1% with a median of 92.7% (SD±9.0 %) (Table 3.1).

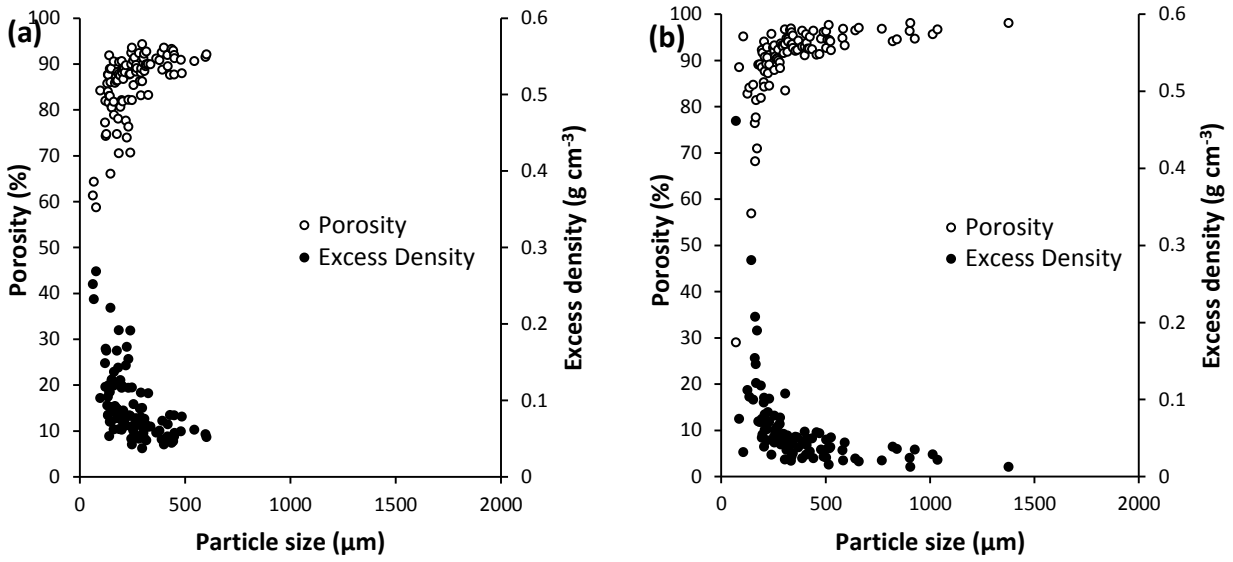


Figure 3.6: Changes in eroded floc density and porosity with size from (a) biodeposit mixture (2008) and pure biodeposits (2010).

Table 3.1: Results from flume experiments conducted on i) biodeposit mixture and ii) pure biodeposits.

Year	Run #	Treatment (Biodeposits +)	Consolidation (days)	τcrit (Pa)	Average erosion rate (g m^{-2} s^{-1})	Median settling velocity (mm s^{-1})	Median excess Density (g cm^{-3})	Median porosity (%)
2008	1a	Sediment (without mussels)	2	0.13	0.003 SD± 0.01191	-	-	-
	1b		7	0.26	0.003 SD± 0.00828	-	-	-
	1c		14	0.15	0.004 SD± 0.01379	2.56 SD± 2.30	1.08 SD±0.05	88.0 SD±7.12
2009	2	With mussels	-	5.83 RPM	0.0002 SD± 0.00046	-	-	-
2010	3a	With mussels	-	x	0.00002 SD± 0.00008	-	-	-
	3b	Without mussels	2	0.052	0.0002 SD± 0.00079	2.53 SD±3.27	1.05 SD±0.06	92.7 SD±9.0

(x indicates no critical RPM was reached)

3.3.3 Grain Size Analysis

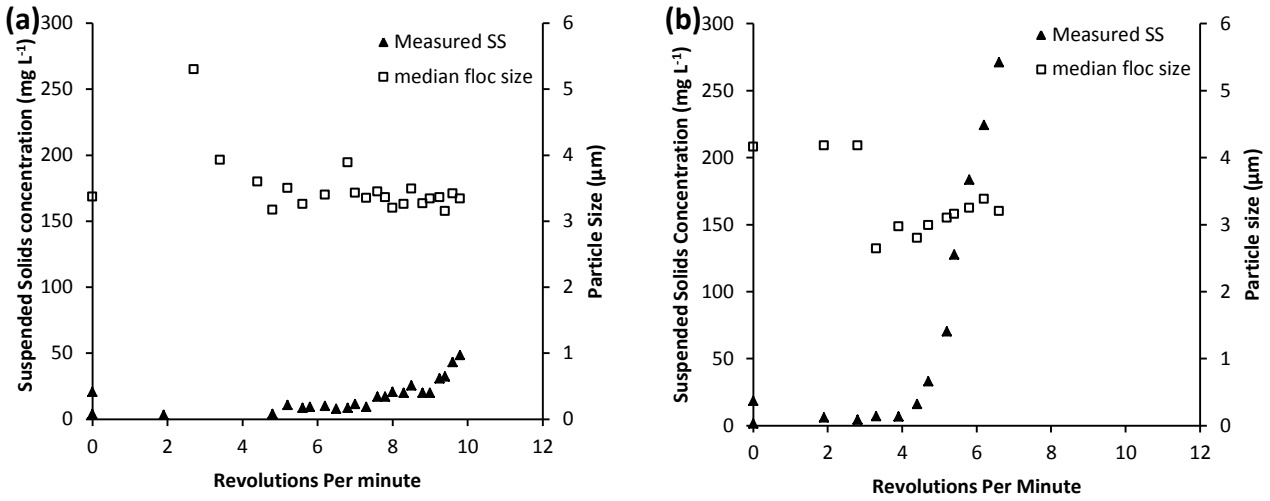
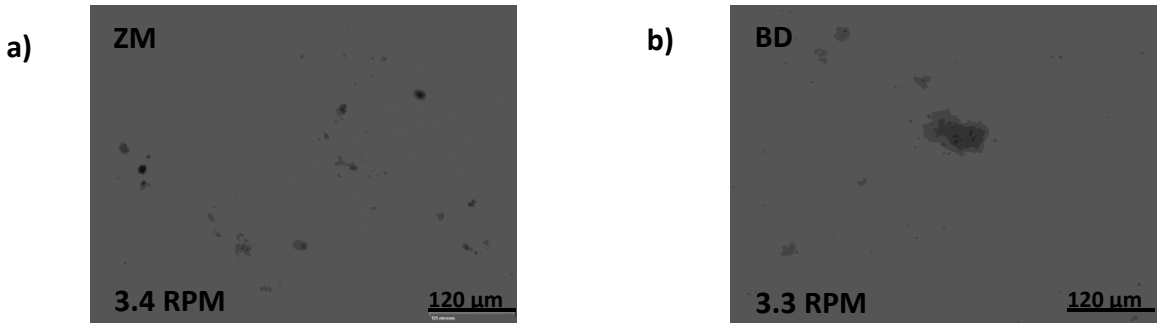


Figure 3.7: Median grain size of samples collected throughout flume experiments on pure biodeposits (a) with mussels and (b) without mussels.

In the flume experiment with dreissenids in 2010, the median grain size decreased with increasing revolutions. However there was little variation in the median grain sizes over the course of the experiment (Figure 3.7a). Similar results were observed for biodeposits without mussels (2010b), where the initial median grain size was larger and with increasing revolutions and time the flocs became smaller. The larger grain sizes initially could be a result of SFGL sloughing off with little force. Images of suspended flocs during each flume run can be seen in Figure 3.8.



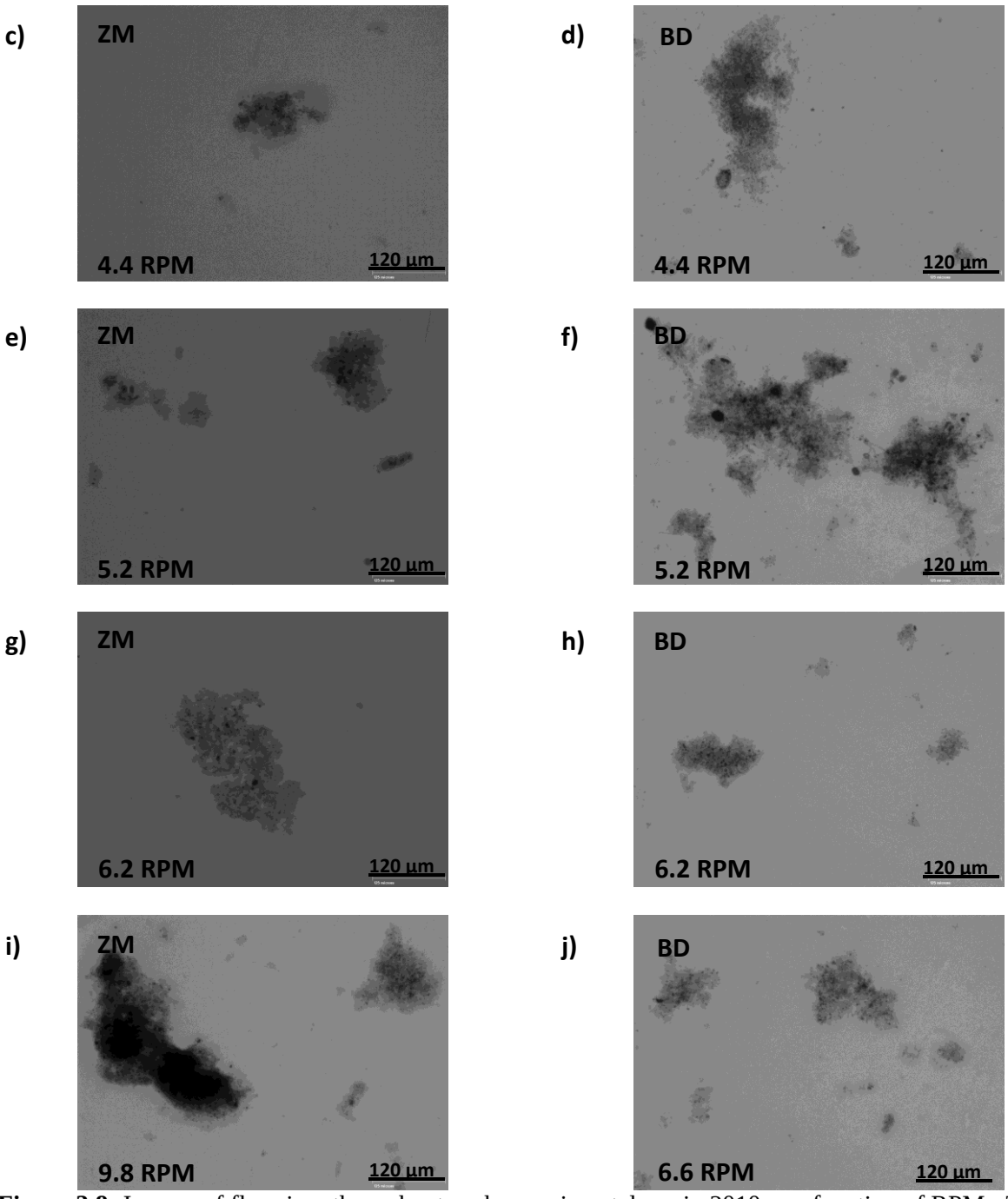


Figure 3.8: Images of floc sizes throughout each experimental run in 2010 as a function of RPM. ZM indicates flume experiment with mussels and BD indicates biodeposits only without mussels.

Petri dishes were placed into the weir box beside plates with attached mussels to collect naturally settling particles and biodeposits produced by dreissenids. Figure 3.9 are images of materials collected in the weir box.

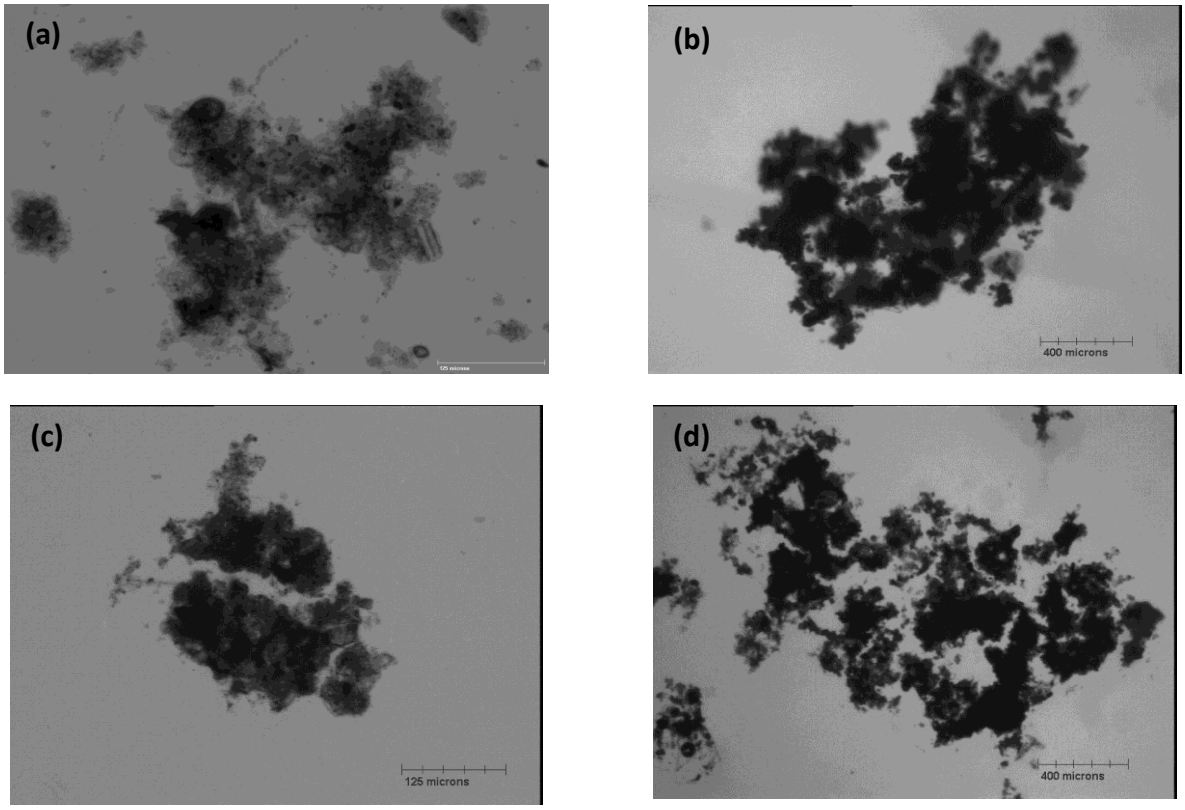


Figure 3.9: Images of materials that settled in the weir box at various locations. (a) and (c) scale is 125 μm while (b) and (d) scale is 400 μm.

Chapter 4: Discussion

Mussels substantially alter the movement of suspended particles in the near shore zone by altering the size distribution of particles in the water column (Nowell et al., 1981). Biodeposit production by mussels alters the mode of transport for previously suspended particles by aggregating particles and transporting them as bedload (Nowell et al., 1981). Accordingly, this biotransformation (bioengineering) of particulate matter has implications for the transport of cohesive sediments that are primary transport vectors for P and many other nutrients and pollutants (Huang et al., 2006; Clifton, 2006). Dreissenids preferentially filter fine-grained suspended sediments (Walz, 1978; Reeders et al., 1989) and excrete them as feces or pseudofeces to the benthic bed sediments at much higher rates than naturally sedimenting particles (Klerks et al., 1996; Roditi et al., 1997; Dobson and Mackie, 1998). While the biological (Roditi et al., 1997) and chemical (Dobson and Mackie, 1998) compositions of the biodeposits have been examined, few studies have quantified the physical transport characteristics of these re-packed materials. Greater knowledge of the physical characteristics and transport properties of these biodeposits are necessary to refine the transport parameters required for modeling the transport of particulate matter and nutrients in lakes.

In the present study an annular flume was used to evaluate the transport properties (critical shear stress for erosion and erosion rate) of dreissenid biodeposits under the influence of varying depositional times (2, 7, 14 days) and added surface roughness (with/out dreissenids). Additionally, settling experiments were conducted to examine the physical properties (density, porosity, settling velocity) of the biodeposits. In the following discussion, the results of from flume and laboratory experiments are presented in the context of the relevant literature to highlight advances in particle transport processes resulting from the present study.

4.1 Effects of Consolidation on the Transport Dreissenid Biodeposits

Previous studies on sediment erosion indicate that τ_{crit} increases as a function of bed age (Pachure and Mehta, 1985; Mehta et al., 1989a). In the current study, erosion experiments performed on the biodeposit mixture were a function of consolidation period (2008). Results indicate that bed sediments had the greatest resistance to erosion over a 7 day consolidation period (τ_{crit}=0.26 Pa) compared to the 2 and 14 day periods (τ_{crit}=0.13 and 0.15 respectively). Consolidation and biostabilization are two possible causes for the increase in bed sediment stability (Stone et al., 2008). However, using an annular flume, Stone et al. (2008) considered the effects of consolidation to contribute minimally to bed stability, due to the depth of the sediment deposit (~1.1 mm). Accordingly, Stone et al. (2008) report that it is unlikely that the

mass of the overlying flocculated materials could produce any significant change in density within the bed sediments. In the present study, the biodeposit mixture was approximately 1.5 mm in depth and it is likely consolidation had little effect on the stability of the bed. Given that mussel deposits are organically enriched (Grenz et al., 1990), it is likely that biofilm development played a role in stabilizing these deposits.

Microbial and/or bacterial populations in sediments excrete extracellular polymeric substances (EPS) composed of DNA, protein, carbohydrates and uronic acids (Bura et al., 1998). EPS often binds particles and fills interstitial gaps due to swelling of hydrated fibrils during exudation, creating a gel-like matrix (Wotton, 2004; Droppo, 2009). The smooth greenish-brown layer that developed on the surface of the 7 day bed sediments detached from the bed by rolling up on itself and sloughing off at a high applied bed shear stress (τ_{crit}=0.26 Pa). The observed development of biofilm, its behaviour and subsequent erosion agrees well with other studies quantifying biogenic mediated bed sediments and the increase in bed stability after 5 to 7 day consolidation periods (Table 4.1) (Droppo, 2009; Stone et al., 2011).

Table 4.1: Critical shear stress for erosion in (Pa) after 2, 7 and 14 day consolidation periods compared with other studies using the same annular flume with one component rotational flow.

Sediment type	**Consolidation period (days)**		
	2	**7**	**14**
Current study (2008; 2010)	0.13; 0.052	0.26	0.15
Aquaculture sediments (Droppo et al., 2007)	0.06	0.06	0.10
Kaolin (Droppo, 2009)	0.04	0.10*	-
Hamilton harbour (Droppo, 2009)	0.06	0.22*	-
South Nation River (Droppo, 2009)	0.14	0.19	0.23
Storm water pond (Droppo, 2009)	0.12	0.23	-
Wildfire-affected sediments (Stone et al., 2011)	0.12	0.23	0.31
Castle river sediments (unburned) (Stone et al., 2011)	0.11	0.14	0.17

(* indicates 5 day consolidation period)

Using an annular flume, Stone et al. (2011) studied the effects that wildfire and bed age have on cohesive sediment erodibility and the effects of biofilms on erosion, transport and deposition of sediments. Cohesive sediments were collected from a wildfire-affected stream and a reference (unburned) stream to quantify their physical and geochemical properties and to characterize the microbial communities inhabiting these two sediment types (Stone et al., 2011). Findings showed that erodibility and sediment properties were strongly influenced by wildfire, consolidation and biostabilization (Stone et al., 2011). In the current study, similar results were observed where the bed stability increased after 7 days. However whereas critical shear stress for erosion is greatest over a 14 day consolidation period in Stone et al. (2011), the current study found that after 14 days the bed sediment stability decreased to a similar value to that of the 2 day experiment (which will be further discussed later in chapter) (Table 4.1). Through visual observations and flume data, Stone et al. (2011) found that the development of biofilm

was greater on the wildfire-affected sediments than on unburned sediments. With the higher τ_{crit} required to erode burned sediments, Stone et al. (2011) suggest the sediment-pore biofilm complex is more integrated and the biological community associated with SFGL and eroded flocs in these deposits is more active than in the unburned sediments. Droppo (2009) performed erosion experiments on five different types of sediment (kaolin, storm water pond, contaminated lacustrine, aquaculture waste and fluvial sediment) with different physical and biological characteristics. All sediments went through consolidation and biofilm growth phases to quantify the relative erosion resistance of different sediment types and to determine to what degree biological activities mediate erosion processes (Droppo, 2009). Similar to the present study, after a 2 day consolidation period, type 1b erosion occurred at low shear stress values and varied between beds (Table 4.1). With increasing time for consolidation, within bed sediment stability increased in all sediment types other than the aquaculture sediments (Table 4.1). Droppo (2009) also attributes the increased stability to the sediment possessing a more sediment-pore integrated biofilm, due to the contribution of depositional flocs having active biological communities. He further adds the structure of the biofilm is likely more tightly organized due to the lower levels of nutrients in these sediment types (Droppo, 2009).

In flume runs with 2 day consolidation periods, τ_{crit} was 1.8 times higher for collected lake bed sediments (biodeposit mixture; 2008) than for pure biodeposits (2010) (2008 τ_{crit}= 0.13 Pa; 2010 τ_{crit}= 0.052 Pa). In a series of erosion experiments, Droppo et al. (2007) found similar results for nutrient enriched aquaculture sediments, with erosion occurring at 0.06 Pa after 2 and 7 day consolidation experiments. They attributed the low critical shear stress to the development of diffuse biofilms on the sediment bed (Droppo et al., 2007). In high organic environments with labile carbon, metabolization of the substrate does not rely upon the tight communal associations of biofilm cells (Karthikeyan et al., 1999; Droppo et al., 2007). It is disadvantageous for biofilm cells to be aligned in close proximity to each other, because cells would have to compete for carbon/nutrient sources (Droppo et al., 2007). Accordingly, biofilms are weaker due to the loose association of bacteria and consequently stability of bed sediment is decreased (Droppo et al., 2007). Alternatively, in environments with nutrient limited resources, bacteria are more tightly organized and develop a more structured biofilm which increases bed stability (Karthikeyan et al., 1999; Droppo et al., 2007). In 2010 experiments biodeposits were harvested from surface waters pumped into the weir box. Slow settling phytoplankton cells found in the surface waters (Burns and Rosa, 1980; Ronzio, 2007) may increase the amount of organic content found in biodeposits compared to biodeposits produced at the benthic surface by dreissenids. In 2008, the materials collected by divers consisted of biodeposits and sediments from dreissenid beds located in the benthos. Settling and biodeposition near the surface of the mussel bed, of organic and inorganic materials may create a nutrient limited environment, increasing the bed stability of the biodeposit mixture. The

large difference in bed stability between the two years could be attributed to the source of particles (e.g. surface versus benthic water) and the increased organic content in the pure biodeposits (2010) produced from the surface waters.

In 2008, after 14 days the bed stability decreased to that of the 2 day consolidation period. For biostabilization to occur, substantial concentrations of organic compounds must be available for the microbial community to inhabit sediments (Schmidt et al., 1985; Stone et al., 2011). Organic compounds drive microbial metabolisms which maintain energy and microbial enzyme induction and produce biofilms (Schmidt et al., 1985; Stone et al., 2011). It is possible that after 14 days, the microbial community inhabiting the biodeposited sediments depleted its source of organic compounds. While these microbial communities consume the organic constituents of the biodeposited materials, P bound to bed sediments can be released back into the water column as orthophosphate (i.e. the form of P that is assimilated by algae, bacteria and plants) (Correll, 1998). In the near shore zone, this biotransformation provides an available source of bioavailable P for benthic algae (Hecky et al., 2004). To improve knowledge regarding their potential redistribution in the near shore zone, the extent to which biodeposits are influenced by the physical roughness produced by mussels is discussed in the next section.

4.2 Bed Stability in the Presence of Dreissenids

Suspended sediment concentrations varied considerably between sediment treatments (i.e. with and without mussels) for biodeposits for all years. In flume experiments with biodeposits without mussels (2010), total suspended sediment concentrations were 2.6 and 1.4 times higher than flume runs with mussels in 2010 and 2009, respectively. These results are comparable to the results of flume experiments performed by Widdows et al. (1998b), where total suspended sediment concentrations were 10-fold higher in runs without mussels compared to runs with 100% mussel bed coverage. Similarly, Widdows et al. (2002) reported suspended sediment concentrations to be 3 times lower over mussel beds with 100% coverage than runs with no mussels. The results from both studies can be attributed to the high mussel bed coverage, creating skimming flow (Widdows et al., 2002), which decreases the potential for sediments to be eroded compared to sediments without mussels.

Bottom morphology influences the velocity gradients close to the bed sediments and levels of turbulence (van Duren et al., 2006). In shallow ecosystems the bottom roughness is determined largely by biogenic structures (Wright et al., 1997). These biogenic structures may be built by the organism, or be the organisms themselves (van Duren et al., 2006). Benthic organisms modify their environment through the production of hard pieces such as shells or tubes (Day et al., 1989), the formation of high density matrices increase the resistance to erosion (Rhoads et al.,1978), and the entrapment of suspended

solids by oyster and mussel reefs that increase deposition rates (Gutierrez and Iribarn, 1999). Further, Meyer et al. (1997) found that artificial beds of oyster shells reduce erosion and vegetation loss in intertidal salt marshes. Crooks (1998) found the presence of the mussel *Musculista senhousia* and the construction of their byssal thread mats on the surface of soft sediments changed the structural complexity of benthic habitats. His study found that mussel mats stabilized sediments on substrates as a result of their physical presence and the binding of sediments and other materials by byssal threads (Crooks, 1998).

Suspension feeders act as both physical obstacles to flow and sources of bed roughness, thereby altering hydrodynamic form and frictional drag (Wright et al., 1997; van Duren et al., 2006; Folkard and Gascoigne, 2009). Roughness elements tend to slow velocities at the bed surface and generate turbulence. Butman et al. (1994) measured vertical velocity profiles over a smooth flume bottom and over a mussel bed within the flume. During slow flow conditions, turbulent stress was three times higher over the mussel bed than over the smooth bed and up to ten times higher during fast flow. Using blue mussels *Mytilus edulis,* van Duren et al. (2006) further compared the turbulent kinetic energy (TKE) over smooth bottoms, inactive mussels (closed shell) and active mussels (filtering). In comparison to the flat bottom, both inactive and active mussels increased turbulence levels, with active mussels significantly increasing TKE over the other two bed types. The presence of the mussels also increased the flux of momentum towards the bed and increased bed shear stress (van Duren et al., 2006). During the 2009 flume experiment with mussels, the critical RPM for type 1b sediment erosion was 5.83 while in 2010 (with mussels) no critical RPM value was obtained as sediment concentrations did not increase with increasing time or revolution. It is possible that the differences reflect the variability in mussel coverage, density, distribution and further the TKE generated by the mussels between the two flume experiments.

During the 2009 flume experiment, the total mass of biodeposits eroded (703.33 mg L^{-1}) was 1.9 times higher than in 2010 (370.55 mg L^{-1}). The observations of Widdows et al. (2009) were similar where the mass of sediment eroded was 2-fold higher for mussel bed coverage of 55% versus 95% coverage; however, these differences were not statistically significant. They found high variability in the replicates for 55% coverage and attribute this to the variability found in the size and shape of the mussel clumps and strings at 55% coverage (Widdows et al., 2009). Although mussel were not analyzed between the two years of the present study, perhaps during the 2010 flume experiments individual mussels covered a larger area, reducing areas of bare sediments which may have resulted in less sediment being eroded. In annular flume experiments performed by Widdows et al. (2002) with blue mussels, *Mytilus edulis,* the density of mussels had an effect on the erodibility of sandy bed sediments. Sediment resuspension was lower in the absence of mussels than with 25% mussel bed coverage (Widdows et al., 2002). Widdows et al. (2002) attribute the lowered resuspension in the absence of mussels to the lower turbulence resulting

from lower bed roughness, the movement of sand primarily as bed load and the lower silt content due to the absence of biodeposits. Low critical velocity (U_{crit}) and high erosion rates were associated with intermediate mussel bed coverage (25% and 50% mussel coverage) as a result of water flow scouring around clumps of mussels (Widdows et al., 2002).

The presence of bivalve shells in benthic environments affects the structure of the habitat by modifying its heterogeneity and complexity, ultimately changing the availability of resources to other organisms (Gutierrez et al., 2003). Studies performed by Botts et al. (1996) and Stewart et al. (1998) quantified the effects of dreissenid induced habitat and increased food supply (i.e. organic matter), as two potential mechanisms for increasing benthic macroinvertebrate biomass. Both studies found the habitat created by dreissenid shells to be the primary cause for increases in total macoinvertebrate abundance and density (Botts et al., 1996; Stewart et al., 1998). As dreissenid shells provide suitable substrates for settlement, grazing and refuge for smaller invertebrates from predators (Stewart et al., 1998), little is known about the influence they have on water flow and more importantly how shells affect the deposition and retention of particles near the bed. Table 4.2 displays the findings of studies conducted on other types of mussels and their effects on sediment transport (Gutierrez et al., 2003). More research on the impacts the added roughness of dreissenids have on the hydrodynamics of a system would help refine hydrodynamic models.

Table 4.2: Summary of study results on the effects mussel beds have on particle transport (modified from Gutierrez et al., 2003).

Shell structure	Effects	Authors
Shells of sea scallop, *Placopecten magellanicus*	Enhanced phytoplankton retention by sediments in areas of increased skim friction	Pilditch et al., (1998)
Mussel mats, *Musculista senhousia*	Enhanced retention of fine-grained sediments and organic particles	Crooks and Khim, (1999)
Beds of Holocene clams, *Tagelus plebeius*	Increase bed sediment stability	Gutierrez and Iribarne, (1999)

4.3 Physical Characteristics of Dreissenid Biodeposits

4.3.1 Settling Velocities

Settling experiments performed on collected biodeposit mixture from 2008 demonstrate a positive relationship of floc size to floc settling velocity, with little variability (r^2=0.80) (see Figure 3.5). When conducting settling experiments on bed sediments from Hamilton Harbour, Lake Ontario, Amos and Droppo (1996) observed a similar positive relationship between floc size and floc settling velocity;

however, there was a greater amount of variability (r^2=0.47). They attribute the high variability to the majority of sediment being in flocculated form (Amos and Droppo, 1996). Factors that can influence the settling rate of a floc and create variability in the results are floc composition (i.e. proportion of organic and inorganic content), shape, porosity and water content (Droppo et al., 1997).

The settling velocity of the biodeposit mixture was similar to that observed by Stone et al. (2011) when examining wildfire-affected sediments after 14 day consolidation. Examining the settling velocities of sediment from a wildfire-affected and unburned reference streams, Stone et al. (2011) found median settling velocities after 14 days to increase in both sediment types. The settling velocity of unburned sediment was significantly higher (3.82 mm s^{-1}) than the wildfire-affect sediment (2.96 mm s^{-1}), which they attribute to the spatial variation and degree of biostabilization between the two sediment types (Stone et al., 2011). In the current study, the settling velocity of the biodeposit mixture was 2.56 mm s^{-1}. Stone et al. (2011) hypothesize that once eroded into the water column, the wildfire-affected flocs with associated biofilms formed organically rich, low porosity flocs, which as a result decreased the settling velocity. When comparing dreissenid biodeposit mixtures to natural sediments in the Hudson River, NY, USA, Roditi et al. (1997) found the organic content of biodeposits to be 22% higher than in natural sediment. Further, biodeposits contained four times more algae by weight than natural sediments (Roditi et al., 1997). Accordingly, in the current study, the biodeposit mixture likely contains more organic material and algae producing low porosity flocs with decreasing settling velocities similar to the findings of Stone et al. (2011).

The median settling velocity for pure biodeposits collected in 2010 was 2.53 mm s^{-1} and there was a positive relation between floc settling velocity and floc size with slightly higher variability (r^2=0.77) compared to the biodeposit mixture. The range in settling velocities for both sediment types fall within the range observed for biodeposits produced by other aquatic invertebrates, of copepods (0.2 mm s^{-1}) to polychaetes (59.4 mm s^{-1}) (Wotton and Malmqvist, 2001). Figure 4.1 illustrates settling velocities of biodeposits produced by other marine and aquatic invertebrates (Wotton and Malmqivst, 2001) the current study included.

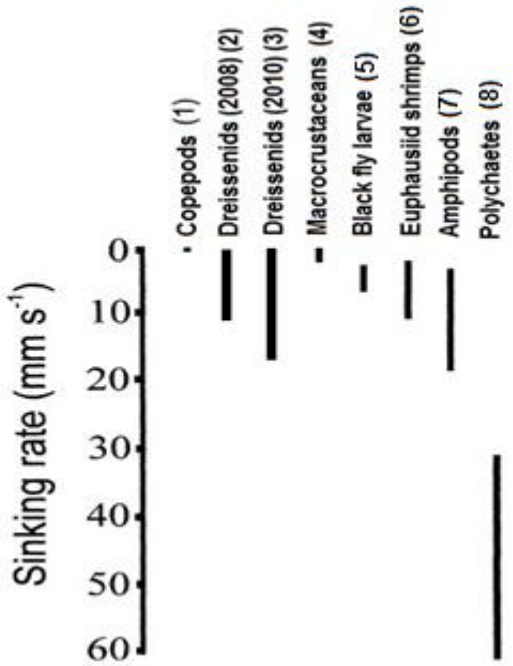


Figure 4.1: Settling rates of fecal pellets (given as ranges) for the named taxa. Fecal pellets are organized according to increasing size from (1) to (8). Data are from (1) Turner 1977, (2) Current study 2008, (3) Current study 2010, (4) Alldredge et al. 1987, (5) Ladle et al. 1987, (6) Fowler and Small 1972, (7) Ladle et al. 1987 and (8) Taghon et al. 1984. Adapted from Wotton and Malmquist (2001).

Giles and Pilditch (2004) observed settling velocities of biodeposits produced by the New Zealand green lipped mussel, *Perna canaliculus*, to range between 1 to 45 mm s^{-1}. Callier et al. (2006) and De Jong (1994) found average settling velocities of biodeposits produced by *P. canaliculus* of 10 ± 1 mm s^{-1} and 12 ± 1 mm s^{-1}, respectively. Variation in settling velocities within species is likely due to the food source, while between species variability can be attributed to differences in organism size, physiology and feeding habits (Wotton and Malmqvist, 2001; Giles and Pilditch, 2004; Callier et al., 2006). For the horse mussel *Atrina zelandica*, Miller et al. (2002) found mussels fed high quality diets with higher concentrations of algae produced biodeposits with lower settling velocities compared to mussels fed an algae and silt diet. Similar experiments on the effects of diet on *P. canaliculus* found mussel diets with a higher silt content produced biodeposits which sank more rapidly (Giles and Pilditch 2004). In the current study, mussel diets were not examined. However, Roditi et al. (1997) did examine dreissenid biodeposits and found them to be enriched with algae and organic materials, which could be the reason for the low settling velocities found in the biodeposit mixture collected in 2008 compared to other studies on biodeposit settling velocities (Wotton and Malmqvist, 2001; Giles and Pilditch, 2004; Callier et al., 2006; Stone et al., 2011). The biodeposits produced in the weir box (2010) were created from surface waters, where slow settling phytoplankton cells are abundant (Burns and Rosa, 1980;

Ronzio, 2007) the resultant lowered settling velocities found in the current study could be due to the organically rich nature of the biodeposits produced. As such, settling velocities of feces and pseudofeces produced by dreissenids could vary spatially (e.g. different food sources) and temporally (e.g. seasonally) and more settling experiments should be conducted to predict the dispersal of biodeposits in benthic environments.

4.3.2 Density and Porosity

In the current study, biodeposit densities were 1.08 g cm^{-3} (SD ±0.05) and 1.05 g cm^{-3} (SD ±0.06) for biodeposit mixture and pure biodeposits respectively. Taghon et al. (1984) measured the densities of biodeposits produced by *Amphicteis scaphobranchiata*, a deposit-feeding polychaete worm. Worms fed a diet of <61μm sediment fractions produced biodeposits with an average density of 1.19 g cm^{-3} while biodeposits produced on sediment fractions between 61-250 μm had an average density of 1.14 g cm^{-3} (Taghon et al., 1984). The authors attribute the denser biodeposits of <61 μm diet to biodeposits containing less fluid-filled porespace (Taghon et al., 1984). The porosity of biodeposits from 2008 and 2010 were high at 88% and 92.7% respectively.

4.3.3. Grain Size Distributions

Grain size analysis was performed on suspended sediment samples collected from the flume at the beginning of each increase in RPM for both runs in 2010 (with and without mussels). Grain size distributions show that 80% of the particles were finer than 10 μm for runs both with and without dreissenids. Particle size distributions did not vary with increasing revolutions. Figure 3.7 illustrates the median grain size for each increment in revolution with time. In the River Meuse, Reeders and Bij de Vaate (1992) observed that over 90% of all biodeposits produced by zebra mussels consisted of pseudofeces. With increasing concentrations of suspended sediments, various studies have observed the increase in pseudofeces production in both laboratory (MacIssac and Rocha, 1995) and field settings (Reeders and Bij de Vaate, 1992; Klerks et al., 1996). Particle size analysis on pseudofeces performed by Reeders and Bij de Vaate (1992) found 82.3% of the total number of pseudofeces collected was less than 10 μm. They attribute the smaller particle sizes to the mussels preferentially filtering smaller particles (Reeders and Bij de Vaate, 1992). Similar findings were observed by Howell et al. (1996) who found grain size distributions progressively decreased following the invasion of dreissenids. Marvin et al. (2000) observed dreissenid colonized sediments to be composed of silt sized fractions (7.5-20μm) while non-colonized sediments consisted of much coarser grains (30-60 μm) in the western basin of Lake Erie.

In the weir box, four petri dishes were placed beside the glass plates to collect passively settling and biodeposited materials from the mussels. Collected materials were analyzed for grain size distributions and imaging. Median grain sizes between petri plates ranged from 2.15 to 15.0 μm. Grain size distributions show that 50% of the particles in all samples were <30 μm. The increase in grain size distribution compared to the other results is likely due to particles settling and not being resuspended and potentially being broken apart. Kautsky and Evans (1987) reported that mussel biodeposits consisted of packages of fine-grained materials which could easily be broken up. Given the degree of stress applied from the flume, biodeposit grain sizes from both experimental runs in 2010 could be a result of biodeposits being repeatedly resuspended and disaggregated.

Size, density and settling velocity are important in determining whether and in what mode a particle will be transported by a given flow (Smith, 1977; Taghon et al., 1984), yet few studies have quantified these parameters for dreissenid biodeposits. By determining these values we can predict the dispersal of biodeposits and hence their flux to the benthic environments. In marine environments it is crucial to determine the influence that biodeposits from shellfish farms have on the surrounding habitat (Giles et al., 2009). Cultured mussels have been observed to modify mass and energy fluxes in coastal ecosystems by linking the upper pelagic waters in which they live, attached to suspended structures and to the benthos (Giles et al., 2009). They feed on suspended phytoplankton and organic particles and egest biodeposits which potentially sink to the seabed, enriching sediments and supply food to the benthic environment. Accordingly, modeling the transport and dispersal of biodeposits is important economically (e.g. number of individuals that can be processed) and ecologically (e.g. energy pathway modified from pelagic to benthic). Giles et al. (2009) determined four factors that control the accumulation of biodeposits in the vicinity of a farm: 1) the rate of biodeposit production; 2) initial dispersal (sinking velocity of biodeposits); 3) redistribution of biodeposits on the sediment surface (i.e. creep, saltation and/or resuspension); and 4) biodeposit decay. In order to successfully model the distribution of mussel biodeposits, Giles et al. (2009) recommend quantifying these four parameters. While the rates of biodeposit production in terms of population density have been measured (Klerks et al., 1996; Dobson and Mackie, 1998), the current study has quantified two of the other factors controlling biodeposit dispersion; initial dispersal and redistribution of biodeposits (i.e. erosion). Since the current study examined benthic dreissenids and not suspended mussel cultures, another factor controlling biodeposit transport is the added roughness dreissenids themselves create which can influence all the other factors. A summary of the study results is illustrated in Figure 4.2.

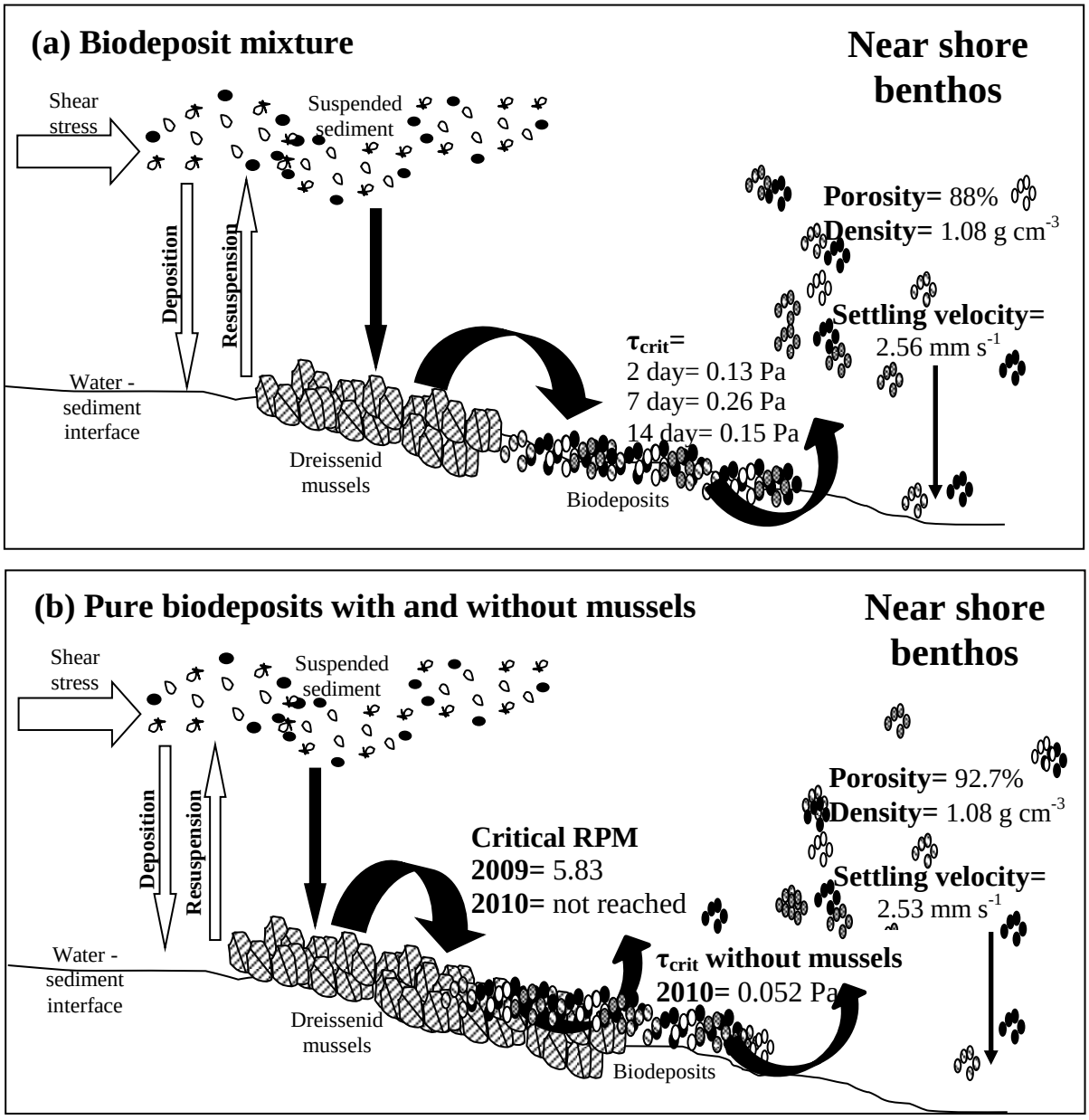


Figure 4.2: Results from flume experiments conducted on (a) biodeposit mixture (2008) and (b) pure biodeposits with and without mussels (2009 and 2010).

Chapter 5: Conclusions

In the current study, an annular flume was used to characterize the physical transport characteristics of dreissenid biodeposits. In the first experiments, a mixture of lake bed sediments and biodeposits were placed into the flume to determine the effects of consolidation and biostabilization on the erodibility of these sediments. In the 2 subsequent years, pure biodeposits and dreissenids were placed directly into the flume to quantify the effects mussel presence had on the erosional characteristics of the biodeposits. Settling velocity, density and porosity were quantified for biodeposit mixture and for pure biodeposits harvested in 2010. This is the first study to quantify the transport characteristics of dreissenid biodeposits using an annular flume. The conclusions of this research are presented below for each of the objectives.

Objective 1: Evaluate the transport properties (critical shear stress for erosion, erosion rate) of biodeposits in an annular flume with and without dreissenids.

With respect to the effects consolidation time has on dreissenid biodeposits, there was an increase in bed stability after a 7 day consolidation period and decrease after 14 days. Accordingly, ageing biodeposits can significantly alter erosion thresholds which can further influence the redistribution of P associated with biodeposits. In their natural environment, high densities of dreissenids increase the flux of particulate matter from the water column to the benthos (Klerks et al., 1996). The constant filtering and biodeposition of suspended materials by mussels would cause layers upon layers of sediment and biodeposits after 14 days. The effects of consolidation and biostabilization would surely be greater than the results found in the current study. In the presence of dreissenids, the amount of resuspended materials was much lower than without mussels. This in part could be due to the amount of materials within the flume, but I suspect the process of biostabilization and the physical presence of mussels (i.e. creating skimming flow) are contributing factors to the enhanced stability of the bed sediments. With dreissenid shells preventing water flow from penetrating the bed sediments and the persistent biodeposition of particulate materials, the process of biostabilization/consolidation could have profound effects on the erodibility of these materials in the near shore zones of lakes.

Objective 2: Characterize and quantify the physical properties (porosity, density, settling velocity, and particle size) of dreissenid biodeposits.

The settling rates of both the biodeposit mixture and pure biodeposits were lower than natural suspended sediment in lacustrine environments. The higher organic content of this material may cause the decreased

settling rates. Biodeposits in marine and aquatic systems provide an important food source for the benthos and can be repackaged by animals, including the individuals who produced them (Giles and Pilditch, 2004). In addition, biodeposits have been observed to mineralize and degrade quickly (Fabiano et al., 1994). Biodeposit modification is dependent on the composition of the benthic environment (Giles and Pilditch, 2004). Processes such as ageing and repackaging alter the biodeposit density, porosity and sinking velocities (Yoon et al., 1996; Roditi et al., 1997; Giles and Pilditch, 2004) and further influences the erosional rates of these materials (Giles and Pilditch, 2004). Therefore, determining the benthic processes modifying biodeposits can help in understanding the redistribution of biodeposits.

5.1 Relevance to the Nearshore Shunt Hypothesis

One assumption of the nearshore shunt hypothesis is that particulate matter produced by dreissenids is often larger and denser. Conversely in the current study these results were not observed. However, this does not mean that the biodeposits are not being retained within the near shore benthos. The high biodeposition rates of dreissenids increase the amount of sediment on the bed surface. These fine-grained materials are enriched with organic matter potentially enhancing biostabilization. Dreissenids typically, are densely aggregated, with this added physical roughness, the flow of water may be skimming the tops of the mussels and not contacting the bed sediments/biodeposits. Furthermore, if biodeposits are undergoing remineralization and developing biofilms while being retained in the near shore zone for long periods of time, large pieces of SFGL may slough off in the event of a large storm or increased wave action. The eroded flocs could potentially travel as saltating flocs over the bed sediments instead of in suspension.

5.2 Future Research Recommendations

Further research is required to determine the effects of dreissenids on sediment transport dynamics and their role in influencing nutrient dynamics within the near shore. The following are suggestions for future research which arise from the current study:

1) To perform similar erosion experiments with mussels where bed shear stress can be quantified to allow for comparisons between mussel free conditions.
2) To distinguish between feces and pseudofeces produced by dreissenids. Studies have found mussel pseudofeces have lower settling velocities than feces (Giles and Pilditch, 2004). Differentiating between the two types of biodeposits would help determine the amount of repackaged pollutants versus organic matter being released to the benthos.

3) To determine the biological and chemical characteristics of pure biodeposits produced by dreissenids. Quantifying these would provide more information on the organic content and nutrients associated with biodeposited materials. Furthermore determining microbial content would help to estimate mineralization rates of ageing biodeposits.
4) To conduct erosion experiments with different mussel densities and percent mussel coverage. Mussels tend to be found in high densities. Determining their spatial coverage could help determine to what degree their presence regulates the erodibility of sediments.
5) To measure the turbulent kinetic energy over dreissenid mussel beds. This would help determine how much mussels and their filtration affect the hydrodynamics and whether they contribute to increasing or decreasing bed shear stress.
6) To perform erosion experiments with an in-situ flume. Results from in-situ experiments would be more realistic of the biological, chemical and physical conditions.

References

Alldredge, A. L., Gotschalk, C. C., & MacIntyre, S. (1987). Evidence of sustained residence of macrocrustacean fecal pellets in surface waters off Southern California. *Deep-Sea Research Part A: Oceanographic Research Papers, 34*, 1641-1652.

Amos, C. L., Bergamasco, A., Umgiesser, G., Cappucci, S., Cloutier, D., DeNat, L., et al. (2004). The stability of tidal flats in Venice Lagoon- the results of in-situ measurements using two benthic, annular flumes. *Journal of Marine Systems, 51*, 211-241.

Amos, C. L., Brylinsky, M., Sutherland, T. F., O'Brien, D., Lee, S., & Cramp, A. (1998). The stability of a mudflat in the Humber estuary, South Yorkshire, UK. In K. S. Black, D. M. Paterson, & A. Cramp (Eds.), *Sedimentary Processes in the Intertidal Zone* (Vol. 139, pp. 25-43). London: Geological Society.

Amos, C. L., Daborn, G. R., Christian, H. A., Atkinson, A., & Robertson, A. (1992). In situ erosion measurements on fine-grained sediments from the Bay of Fundy. *Marine Geology, 108*, 175-196.

Amos, C. L., Feeney, T., Sutherland, T. F., & Luternauer, J. L. (1997). The stability of fine-grained sediments from the Fraser River Delta. *Estuarine, Coastal and Shelf Science, 45*, 507-524.

Andersen, T. J., Lanuru, M., van Bernem, C., Pejrup, M., & Riethmueller, R. (2010). Erodibility of a mixed mudflat dominated by microphytobenthos and Cerastoderma edule, East Frisian Wadden Sea, Germany. *Estuarine, Coastal and Shelf Science, 87*, 197-206.

Berlamont, J., Ockenden, M., Toorman, E., & Winterwerp, J. (1993). The characterisation of cohesive sediment properties. *Coastal Engineering, 21*, 105-128.

Berlamont, J., Ockenden, M., Toorman, E., & Winterwerp, J. (1993). The characterisation of cohesive sediment properties. *Coastal Engineering, 21*, 105-128.

Bially, A., & MacIsaac, H. J. (2000). Fouling mussels (Dreissena spp.) colonize soft sediments in Lake Erie and facilitate benthic invertebrates. *Freshwater Biology, 43*(1), 85-97.

Black, K. S., & Paterson, D. M. (1996). LISP-UK: An holistic approach to the interdisciplinary study of tidal flat sedimentation. *Terra Nova, 8*(4), 304-308.

Black, K. S., Tolhurst, T. J., Paterson, D. M., & Hagerthey, S. E. (2002). Working with natural cohesive sediments. *Journal of Hydraulic Engineering*, 1-8.

Brown, J. E., & Stepien, C. A. (2010). Population genetic history of dreissenid mussel invasions: expansion patterns across North Ameria. *Biological Invasions, 12*, 3687-3710.

Bunt, C. M., MacIsaac, H. J., & Sprules, W. G. (1993). Pumping rates and projected filtering impacts of juvenile zebra mussels (Dreissena polymorpha) in Western Lake Erie. *Canadian Journa of Fisheries and Aquatic Sciences, 50*, 1017-1022.

Bura, R., Cheung, M., Liao, B., Finlayson, J., Lee, B. C., Droppo, I. G., et al. (1998). Composition of extracellular polymeric substances in the activated sludge floc matrix. *Water Science and Technology, 37*(4-5), 325-333.

Burns, N. M., & Rosa, F. (1980). In situ measurement of the settling velocity of organic carbon particles and 10 species of phytoplankton. *Limnology and Oceanography, 25*(5), 855-864.

Callier, M. D., Weise, A. M., McKindsey, C. W., & Desrosiers, G. (2006). Sedimentation rates in a suspended mussel farm (Great-Entry Lagoon, Canada): biodeposit production and dispersion. *Marine Ecology Progress Series, 322*, 129-141.

Chan, W. Y., Wai, O. W., & Li, Y. S. (2006). Critical shear stress for deposition of cohesive sediments in Mai Po. *Conference of Global Chinese Scholars on Hydrodynamics*, (pp. 300-305). Hong Kong.

Christian, H. A., Gillespie, D., & Amos, C. L. (1990). Results of a new device to measure the in-situ shear strength of cohesive sediments, Minas Basin, Bay of Fundy. *13th International Sedimentology Congress*, (p. 91). Nottingham, U.K.

Clifton, H. (2005). Cohesive sediment transport. In M. L. Schwartz (Ed.), *Encyclopedia of coastal Sciences* (pp. 327-330). Dordrecht, The Netherlands.

Cognie, B., & Barille, L. (1999). Does bivalve mucus favour the growth of their main food source, microalgae? *Oceanologica Acta, 22*(4), 441-450.

Coleman, F. C., & Williams, S. L. (2002). Overexploiting marine ecosystem engineers: potential consequences for biodiversity. *Trends in Ecology and Evolution, 17*(1), 40-44.

Conroy, J. D., & Culver, D. A. (2005). Do Dreissenid mussels affect Lake Erie ecosystem stability processes? *Amerian Midland Naturalist, 153*(1), 20-32.

Conroy, J. D., Kane, D. D., Dolan, D. M., Edwards, W. J., Charlton, M. N., & Culver, D. A. (2005). Temporal trends in Lake Erie plankton biomass: Roles of external phosphorus loading and dreissenid mussels. *Journal of Great Lakes Research, 31*(2), 89-110.

Crooks, J. A. (1998). Habitat alteration and community-level effects of an exotic mussel, Musculista senhousia. *Marine Ecology Progress Series, 162*, 137-152.

Dade, W. B., Nowell, A. R., & Jumars, P. A. (1992). Predicting erosion resistance of muds. *Marine Geology, 105*, 285-297.

Dade, W. B., Self, R. L., Pellerin, N. B., Moffet, A., Jumars, P. A., & Nowell, A. R. (1996). The effects of bacteria on the flow behavior of clay-seawater suspensions. *Journal of Sedimentary Research, 105*, 39-42.

Dame, R. F. (1996). *Ecology of marine bivalves. An ecosystem approach.* Boca Raton, Florida: CRC Press.

de Boer, D. H., Hassan, M. A., MacVicar, B., & Stone, M. (2005). Recent (1999-2003) Canadian research on contemporary processes of river erosion and sedimentation, and river mechanics. *Hydrological Processes, 19*, 265-283.

Dean, D. M. (1994). Investigations of biodeposition by dreissena polymorpha. *MSc dissertation*. Guelph, ON: University of Guelph.

Defew, E. C., Tolhurst, T. J., Paterson, D. M., & A, H. (2003). Can the stability of intertidal sediments be predicted from proxy parameters? An in situ investigation. *Estuarine and Coastal Sciences Association, Coastal Zone Topics, 5*, 61-70.

Dobson, E. P., & Mackie, G. L. (1998). Increased deposition of organic matter polychlorinated biphenyls, and cadmium by zebra mussels (Dreissena polymorpha) in western Lake Erie. *Canadian Journal of Fisheries and Aquatic Sciences, 55*, 1131-1139.

Droppo, I. G. (2001). Rethinking what constitues suspended sediment. *Hydrological Processes, 15*, 1551-1564.

Droppo, I. G. (2009). Biofilm structure and bed stability of five contrasting freshwater sediments. *Marine and Freshwater Research, 60*, 690-699.

Droppo, I. G., & Amos, C. L. (2001). Structure, stability, and transformation of contaminated lacustrine surface fine-grained laminae. *Journal of Sedimentary Research, 71*(5), 717-726.

Droppo, I. G., & Ongley, E. D. (1992). The state of suspended sediment in the freshwater fluvial environment: a method of analysis. *Water Resources, 26*, 65-72.

Droppo, I. G., Jaskot, C., Nelson, T., Milne, J., & Charlton, M. (2007b). Aquaculture waste sediment stability: Implications for waste migration. *Water, Air and Soil Pollution, 183*, 59-68.

Droppo, I. G., Lau, Y. L., & Mitchell, C. (2001). The effect of depositional history on contaminated bed sediment stability. *The Science of the Total Environment, 266*, 7-13.

Droppo, I. G., Walling, D. E., & Ongley, E. D. (1998). Suspended sediments structure: implications for sediment and contaminant transport and modelling. *Modelling soil erosion, sediment transport and closely related hydrological processes. 249*, pp. 437-444. Vienna: IAHS.

Eckman, J. E., & Nowell, A. R. (1984). Boundary skin friction and sediment transport about an animal-tube mimic. *Sedimentology, 31*, 851-862.

Fahnenstiel, G., Nalepa, T., Pothoven, S., Carrick, H., & Scavia, D. (2010). Lake Michigan lower food web: Long-term observations and Dreissena impact. *Journal of Great Lakes Research, 36*, 1-4.

Folkard, F. A., & Gascoigne, C. J. (2009). Hydrodynamics of discontiuous mussel beds: Laboratory flume simulations. *Journal of Sea Research*, 250-257.

Forstner, U., & Owens, P. N. (2007). Sediment quantity and quality issues in river basins. In B. Westrich, & U. Forstner, *Sediment dynamics and pollutant mobility in rivers: An interdisciplinary approach* (pp. 1-34). Verlag, Berlin, Heildelberg: Springer.

Fowler, S. W., & Small, L. F. (1972). Sinking rates of euphausiid fecal pellets. *Limnology and Ocreanography, 17*, 293-296.

Friend, P. L., Ciavola, P., Cappucci, S., & Santos, R. (2003). Bio-dependent bed parameters as a proxy tool for sediment stability in mixed habitat intertidal areas. *Continental Shelf Research, 23*, 1899-1917.

Gerbersdorf, S. U., Jancke, T., Westrich, B., & Paterson, D. M. (2008). Microbial stabilization of riverine sediments by extracellular polymeric substances. *Geobiology, 6*, 57-69.

Giles, H., & Pilditch, C. A. (2004). Effects of diet on sinking rates and erosion thresholds of mussel Perna canaliculus biodeposits. *Marine Ecology Progress Series, 282*, 205-219.

Giles, H., Broekhuizen, N., Bryan, K. R., & Pilditch, C. A. (2009). Modelling the dispersal of biodeposits from mussel farms: The importance of simulating biodeposit erosion and decay. *Aquaculture, 291*, 168-178.

Grabowski, R. C., Droppo, I. G., & Wharton, G. (2011). Erodibility of cohesive sediment: The importance of sediment properties. *Earth Science Reviews, 105*, 101-120.

Grenz, C., Hermin, M. N., Baudinet, D., & Daumas, R. (1990). In situ biochemical and bacterial variation of sediments enriched with mussel biodeposits. *Hydrobiologia, 207*, 153-160.

Grenz, C., Hermin, M. N., Baudinet, D., & Daumas, R. (1990). In situ biochemical variation of sediments enriched with mussel biodeposits. *Hydrobiologia, 207*, 153-160.

Griffiths, R. W. (1993). Effects of zebra mussels (Dreissena polymorpha) on the benthic fauna of Lake St Clair. In T. F. Nalepa, & D. W. Schloesser (Eds.), *Zebra mussels. Biology, impacts and control* (pp. 415-437). Bocan Raton, Fla: Lewis Publisers.

Gutierrez, J. L., Jones, C. G., Strayer, D. L., & Iribarne, O. O. (2003). Mollusks as ecosystem engineers: the role of shell production in aquatic habitats. *Oikos, 101*, 79-90.

Gutierrez, J., & Iribarne, O. (1999). Role of Holocene beds of the stout razor clam Tagelus plebeius in structuring present benthic communities. *Marine Ecology Progress Series, 185*, 213-228.

Haven, D. S., & Morales-Alamo, R. (1966). *Aspects of biodeposition by oysters and other invertebrate filter feeders.* Gloucester Point, Virginia: U.S. Atomic Energy Commission.

Hebert, P. D., Muncaster, B. W., & Mackie, G. L. (1989). Ecological and genetic studies on Dreissena polymorpha (Pallas): a new mollusc in the Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences, 46*, 1587-1591.

Hecky, R. E., Smith, R. E., Barton, D. R., Guildford, S. J., Taylor, W. D., Charlton, M. N., et al. (2004). The nearshore phosphorus shunt: a consequence of ecosystem engineering by dreissenids in the Laurentian Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences, 61*, 1285-1293.

Henderson, A. R., & Ross, D. J. (1995). Use of macrobenthic infaunal communities in the monitoring and control of the impact of marine cage fish farming. *Aquaculture Research, 26*, 659-678.

Herman, P. M., Middelburg, J. J., & Heip, C. H. (2001). Benthic community structure and sediment processes on an intertidal flat: results from the ECOFLAT project. *Continental Shelf Research, 21*, 2055-2071.

Horgan, M. J., & Mills, E. L. (1997). Clearance rates and filtering activity of zebra mussel (Dreissena polymorpha): implications for freshwater lakes. *Canadian Journal of Fisheries and Aquatic Sciences, 54*(2), 249-255.

Howell, E. T., Marvin, C. H., Bilyea, R. W., Kauss, P. B., & Somers, K. (1996). Changes in environmental conditions during dreissena colonization of a monitoring station in eastern Lake Erie. *Journal of Great Lakes Research, 22*(3), 744-756.

Huang, J., Hilldale, R. C., & Greimann, B. P. (2006). Cohesive sediment transport. In *Erosion and sedimentation manual* (pp. 4-1-4-46). US Department of the Interior Bureau of Reclamation .

Jaramillo, E., Bertran, C., & Bravo, A. (1992). Mussel biodeposition in an estuary in southern Chile. *Marine Ecology Progress Series, 82*, 85-94.

Jumars, P. A., & Nowell, A. R. (1984). Effects of benthos on sediment transport: difficulties with functional grouping. *Continental Shelf Research, 3*(2), 115-130.

Karlsson, O., Jonsson, P. R., & Larsson, A. I. (2003). Do large seston particles contribute to the diet of the bivalve Cerastoderma edule? *Marine Ecology Progress Series, 261,* 161-173.

Karthikeyan, S., Woolfaardt, G. M., Korber, D. R., & Caldwell, D. E. (1999). Functional and structural responses of a degrative microbial community to substrates with varying degrees of complexity in chemical structure. *Marine Ecology, 38*, 215-224.

Kautsky, N., & Evans, S. E. (1987). Role of biodeposition by Mytilus edulis in the circulation of matter and nutrients in a Baltic coastal ecosystem. *Marine Ecology Progressive Series, 38*, 201-212.

Klerks, P. L., Fraleigh, P. C., & Lawniczak, J. E. (1996). Effects of zebra mussels (Dreissena polymorpha) on seston levels and sediment deposition in western Lake Erie. *Canadian Journal of Fisheries and Aquatic Sciences, 53,* 2284-2291.

Klerks, P. L., Fraleigh, P. C., & Sinsabaugh, R. (2001). *The impact of zebra mussels on the dynamics of heavy metals.* The Ohio State University. Columbus, Ohio: Ohio Sea Grant College Program.

Kovalak, W. P., Longton, G. D., & Smithee, R. D. (1993). Infestation of power plant water systems by the zebra mussel (Dreissena polymorpha Pallas). In T. F. Nalepa, & D. W. Schloesser (Eds.), *Zebra mussels. Biology, impacts and control* (pp. 359-380). Boca Raton, Fla: Lewis Publishers.

Krishnappan, B. G. (2007). Recent advances in basic and applied research in cohesive sediment transport in aquatic systems. *Canadian Journal of Civil Engineering, 34*, 731-743.

Krishnappan, B. G., & Marsalek, J. (2002). Transport characteristics of fine sediments from an on-stream stormwater management pond. *Urban Water, 4*, 3-11.

Krishnappan, B. G., Marsalek, J., Watt, W. E., & Anderson, B. C. (1999). Seasonal size distributions of suspended solids in a stormwater management pond. *Water Science and Technology, 39*(2), 127-134.

Kryger, J., & Riisgard, H. U. (1988). Filtration rate capacities in 6 species of European freshwater bivalves. *Oecologia, 77*(1), 34-38.

Kuijper, C., Cornelisse, J. M., & Winterwerp, J. C. (1989). Research on erosive properties of cohesive sediments. *Journal of Geophysical Research, 94*(10), 14341-14350.

Ladle, M., Welton, J. S., & Bell, M. C. (1987). Sinking rates and physical properties of faecal pellets of freshwater invertebrates of the genera Simulium and Gamarus. *Archiv Fur Hydrobiologie, 108*, 411-424.

Lambert, C. P., & Walling, D. E. (1988). Measurement of channel storage of suspended sediment in a gravel-bed river. *Catena, 15*, 65-80.

Lau, Y. L. (1990). Temperature effect on settling velocity and deposition of cohesive sediments. *Journal of Hydraulic Research, 30*(5), 673-684.

Lau, Y. L. (1995). Relative importance of mean velocity and bed shear on biofilm accumulation in open-channel flows. *Water Science and Technology, 32*(8), 193-198.

Lau, Y. L., & Droppo, I. G. (2000). Influence of antecedent conditions on critical shear stress of bed sediments. *Water Resources, 34*(2), 663-667.

Lau, Y. L., Droppo, I. G., & Krishnappan, B. G. (2001). Sequential erosion/deposition experiments-demonstrating the effects of depositional history on sediment erosion. *Water Research, 35*(11), 2767-2773.

Le Hir, P., Monbet, Y., & Orvain, F. (2007). Sediment erodability in sediment transport modelling: Can we account for biota effects? *Continental Shelf Research, 27*, 1116-1142.

Lee, S. C., & Mehta, A. J. (1994). *Cohesive sediment erosion.* Washington, DC: US Army Corps of Engineers.

Li, D. H., & Ganczarczyk, J. J. (1987). Stroboscopic determination of settling velocity, size and porosity of activated sludge flocs. *Water Resources, 21*(3), 257-262.

MacIsaac, H. J., Sprules, W. G., Johannsson, O. E., & Leach, J. H. (1991). Filtering impacts of larval and sessile zebra mussels (Dreissena polymorpha) in western Lake Erie. *Oecologia, 92*, 30-39.

Mackie, G. L. (1991). Biology of the exotic zebra mussel, Dreissena polymorpha, in relation to native bivalves and its potential impact on Lake St. Clair. *Hydrobiologia, 219*, 251-268.

Makarewicz, J. C. (1993). A lakewide comparison of zooplankton biomass and its species composition in Lake Erie, 1983-87. *Journal of Great Lakes Research, 19*(2), 275-290.

Malmqvist, B., Wotton, R. S., & Zhang, Y. (2001). Suspension feeders transform massive amounts of seston in large northern rivers. *Oikos, 92*(1), 35-43.

Marvin, C. H., Howell, E. T., & Reiner, E. J. (2000). Polychlorinated dioxins and furans in sediments at a site colonized by dreissena in western Lake Ontario, Canada. *Environmental Toxicology and Chemistry, 19*(2), 344-351.

Marvin, C. H., McCarry, B. E., Villella, J., Allan, L. M., & Bryant, D. W. (2000). Chemical and biological profiles of sediments as indicators of sources of genotoxic contamination in Hamilton Harbour. Part I: analysis of polycyclic aromatic hydrocarbons and thia-arene compounds. *Chemosphere, 41*, 979-988.

Matisoff, G., & Ciborowski, J. J. (2005). Lake Erie trophic status collaborative study. *Journal of Great Lakes Research, 31*(2), 1-10.

May, B., & Marsden, J. E. (1992). Genetic identification and implications of another invasive species of dreissenid mussel in the Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences, 49*, 1501-1506.

May, B., & Marsden, J. E. (1992). Genetic identification and implications of another invasive species of dreissenid mussel in the Great Lakes. *Canadian Journal of Fisheries and Aquatic Sciences, 49*(7), 1501-1506.

McAnnally, W. H., & Mehta, A. J. (2002). Significance of aggregation of fine sediment particles in their deposition. *Estuarine, Coastal and Shelf Science, 54*, 643-653.

Mehta, A. J., Hayter, E. J., Parker, W. R., Krone, R. B., & Teeter, A. M. (1989a). Cohesive sediment transport. I: Process description. *Journal of Hydraulic Engineering, 115*(8), 1076-1093.

Mehta, A. J., McAnally, W. H., Hayter, E. J., Teeter, A. M., Schoellhamer, D., Heltzel, S. B., et al. (1989b). Cohesive sediment transport. II: Application. *Journal of Hydraulic Engineering, 115*(8), 1094-1112.

Meyer, D. L., Townsend, E. C., & Thayer, G. W. (1997). Stabilization and erosion control value of oyster cultch for intertidal marsh. *Restoration Ecology, 5*, 93-99.

Milburn, D., & Krishnappan, B. G. (2003). Modelling erosion and deposition of cohesive sediments from Hay River, Northwest Territories, Canada. *Nordic Hydrology* (pp. 125-138). Saariselka, Finland: Northern Research Basins.

Miller, D. C., Norkko, A., & Pilditch, C. A. (2002). Influence of diet on dispersal of horse mussel Atrina zelandica biodeposits. *Marine Ecology Progress Series, 242*, 153-167.

Mills, E. L., Casselman, J. M., Dermott, R., Fitzsimons, J. D., Gal, G., Holeck, K. T., et al. (2003). Lake Ontario: food web dynamics in a changing ecosystem (1970-2000). *Canadian Journal of Fisheries and Aquatic Sciences, 60*, 471-490.

Murphy, R. J., Tolhurst, T. J., Chapman, D. J., & Underwood, A. J. (2004). Estimation of surface chlorophyll a on an exposed mudflat using digital colour infrared (CIR) photography. *Estuarine, Coastal and Shelf Science, 59*, 625-638.

Neary, B. P., & Leach, J. H. (1992). Mapping the potential spread of the zebra mussel (Dreissena polymorpha) in Ontario. *Canadian Journal of Fisheries and Aquatic Sciences, 49*(2), 406-415.

Neumeier, U., Friend, P. L., Gangelhof, U., Lunding, J., Lundkvist, M., Bergamasco, A., et al. (2007). The influence of fish feed pellets on the stability of seabed sediment: A laboratory flume investigation. *Estuarine, Coastal and Shelf Science, 75*, 347-357.

Nicholson, J., & O'Connor, B. A. (1986). Cohesive sediment transport model. *Journal of Hydraulic Engineering, 112*(61), 621-641.

Noordhuis, R., Reeders, H., & Bij de Vaate, A. (1992). Filtration rate and pseudofeces production in zebra mussels and their applicaiton in water quality management. In D. Neumann, & H. A. Jenner (Eds.), *The zebra mussel Dreissena polymorpha* (pp. 101-114). New York: Gustave Fischer Verlag.

Norkko, A., Hewitt, J. E., Thrush, S. F., & Funnell, G. A. (2001). Benthic-pelagic coupling and suspension-feeding bivalves: Linking site-specific sediment flux and biodeposition to benthic community structure. *Limnology and Oceanography, 46*(8), 2067-2072.

Nowell, A. R., & Jumars, P. A. (1984). Flow environments of aquatic benthos. *Annual Review of Ecology andl Systematics, 15*, 303-328.

Nowell, A. R., Jumars, P. A., & Eckman, J. E. (1981). Effects of biological activity on the entrainment of marine sediments. *Marine Geology, 42*, 133-153.

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

Parchure, T. M., & Mehta, A. J. (1985). Erosion of soft cohesive sediment deposits. *Journal of Hydraulic Engineering, 111*(10), 1308-1326.

Paterson, D. M. (1990). The influence of epipelic diatoms on the erodibility of an artificial sediment. *Proceedings of the 10th International Symposium on Living and Fossil Diatoms*, (pp. 345-355). Koenigstein.

Paterson, D. M. (1997). Biological mediation of sediment erodability: ecology and physical dynamics. In N. Burt, R. Parker, & J. Watts, *Cohesive Sediments* (pp. 215-229). Chichester, UK: Wiley and Sons.

Paterson, D. M. (1997). Biological mediation of sediment erodibility, ecology and physical dynamics. In N. Burt, R. Parker, & J. Watts (Eds.), *Cohesive Sediments* (pp. 215-229). New York: Wiley.

Paterson, D. M., & Black, K. S. (1999). Water flow, sediment dynamics and benthic biology. In D. B. Nedwell, D. G. Raffaelli, & A. H. Fitter (Eds.), *Estuaries. Advances in ecological research* (Vol. 29, pp. 155-186). San Diego, Ca: Academic Press.

Phillips, J. M., & Walling, D. E. (1999). The particle size characteristics of fine-grained channel deposits in the Tiver Exe Basin, Devon, UK. *Hydrological Processes, 13*, 1-19.

Pilditch, C. A., Emerson, C. W., & Grant, J. (1997). Effect of scallop shells and sediment grain size on phytoplankton flux to the bed. *Continental Shelf Research, 17*(15), 1869-1885.

Poulton, D. J. (1987). Trace contaminant status of Hamilton Harbour. *Journal of Great Lakes Research, 13*(2), 193-201.

Ragueneau, O., Chauvaud, L., Moriceau, B., Leynaert, A., Thouzeau, G., Donval, A., et al. (2005). Biodeposition by an invasive suspension feeder impacts the biogeochemical cycle of Si in a coastal ecosystem. *Biogeochemistry, 75*, 19-41.

Ravens, T. M. (2007). Comparison of two techniques to measure sediment erodibility in the Fox River, Wisconsin. *Journal of Hydraulic Engineering*, 111-115.

Reeders, H. H., & Bij de Vaate, A. (1992). Bioprocessing of polluted suspended matter from the water column by the zebra mussel (Dreissena polymorpha Pallas). *Hydrobiologia, 239*, 53-63.

Ricciardi, A., Whoriskey, F. G., & Rasmussen, J. B. (1996). Impact of the (Dreissena) invasion on native unionid bivalves in the upper St. Lawrence River. *Canadian Journal of Fisheries and Aquatic Sciences, 53*(6), 1434-1444.

Roditi, H. A., Caraco, N. F., Cole, J. J., & Strayer, D. L. (1996). Filtration of Hudson River water by the zebra mussel (Dreissena polymorpha). *Estuaries, 19*(4), 824-832.

Ronzio, B. S. (2007). Phytoplankton community composition effects on phosphorus sedimentation dynamics in Lake Erie. *MSc dissertation*. Waterloo, ON: University of Waterloo.

Schlichting, H. (1979). *Boundary layer theory* (7th ed.). New York: McGraw-Hill.

Schloesser, D. W., Nalepa, T. F., & Mackie, G. L. (1996). Zebra mussel infestation of unionid bivalves (Unionidae) in North America. *American Zoologist, 36*, 300-310.

Schmidt, S. K., Alexander, M., & Shuler, M. L. (1985). Predicting threshold concentrations of organic substrates for bacterial growth. *Journal of Theoretical Biology, 114*, 1-8.

Schumm, S. S., Mosley, M. P., & Weaver, W. E. (1987). *Experimental fluvial geomorphology.* New York: Wiley.

Shantz, M., Dowsett, E., Canham, E., Tavernier, G., Stone, M., & Price, J. (2004). The effect of drawdown on suspended solids and phosphorus export from Columbia Lake, Waterloo, Ontario. *Hydrological Processes, 18*(5), 865-878.

Smith, R. H. (2007). *Benthic-pelagic coupling and cladophora problems in large lakes.* Waterloo: NSERC application grant.

Snodgrass, W. K. (1987). Analysis of models and measurements for sediment oxygen demand in Lake Erie. *Journal of Great Lakes Research, 13*(4), 738-756.

Sousa, R., Gutierrez, J. L., & Aldridge, D. C. (2009). Non-indigenous invasive bivalves as ecosystem engineers. *Biological Invasions, 11*, 2367-2385.

Sprung, M., & Rose, U. (1988). Influence of food size and food quantity on the feeding of the mussel Dreissena polymorpha. *Oecologia, 77*, 526-532.

Stewart, T. W., & Haynes, J. M. (1994). Benthic macroinvertebrate communities of southwestern Lake Ontario following invasion of Dreissena. *Journal of Great Lakes Research, 20*(2), 479-493.

Stewart, T. W., Miner, J. G., & Lowe, R. L. (1998). Quantifying mechanisms for zebra mussel effects on benthic macroinvertebrates: Organic matter production and shell-generated habitat. *Journal of the North American benthological society, 17*(1), 81-94.

Stone, M., & Droppo, I. G. (1994). Flocculation of fine-grained suspended solids in the river continuum. *Variability in stream erosion and sediment transport. 224*, pp. 479-489. IAHS.

Stone, M., Emelko, M. B., Droppo, I. G., & Silins, U. (2011). Biostabilization and erodibility of cohesive sediment deposits in wildfire-affected streams. *Water Research, 45*, 521-534.

Stone, M., Krishnappan, B. G., & Emelko, M. B. (2008). The effect of bed age and shear stress on the particle morphology of eroded cohesive river sediment in an annular flume. *Water Research, 42*, 4179-4187.

Strayer, D. L., Caraco, N. F., Cole, J. J., Findlay, S., & Pace, M. L. (1999). Transformations of freshwater ecosystems by bivalves. *BioScience, 49*(1), 19-27.

Strayer, D. L., Powell, J., Ambrose, P., Smith, L. C., Pace, M. L., & Fischer, D. T. (1996). Arrival, spread, and early dynamics of a zebra mussel (Dreissena polymorpha) population in the Hudson River estuary. *Canadian Journal of Fisheries and Aquatic Sciences, 53*, 1143-1149.

Taghon, G. L., Nowell, A. R., & Jumars, P. A. (1984). Transport and breakdown of fecal pellets: Biological and sedimentological consequences. *Limnology and Oceanography, 29*(1), 64-72.

Teisson, C., Ockenden, M., Le Hir, P., Kranenburg, C., & Hamm, L. (1993). Cohesive sediment transport processes. *Coastal Engineering, 21*, 129-162.

Ten Winkel, E. H., & Davids, C. (1982). Food selection by Dreissena polymorpha Pallas (Mollusca: Bivalvia). *Freshwater Biology, 12*(6), 553-558.

Tolhurst, T. J., Black, K. S., Paterson, D. M., Mitchener, H. J., Termaat, G. R., & Shayler, S. A. (2000). A comparison and measurement standardisation of four in situ devices for determining the erosion shear stress of intertidal sediments. *Continental Shelf Research, 20*, 1397-1418.

Turner, J. T. (1977). Sinking rates of fecal pellets from the marine copepod Pontella meadii. *Marine Biology, 40*, 249-259.

Urban, J. L., Deibel, D., & Schwinghamer, P. (1993). Seasonal variations in the densities of fecal pellets produced by Oikopleura vanhoeffeni (C. Larvacea) and Calanus finmarchicus (C. Copepoda). *Marine Biology, 117*, 607-613.

van Duren, L. A., Herman, P. M., Sandee, A. J., & Heip, C. H. (2006). Effects of mussel filtering activity on boundary layer structure. *Journal of Sea Research, 55*, 3-14.

Vogel, S. (2004). *Life in moving fluids. The physical biology of flow* (2nd ed.). Princeton, New Jersey: Princeton University Press.

Wall, G. J., Wilding, L. P., & Smeck, N. E. (1978). Physical, chemical and mineralogical properties of fluvial unconsolidated bottom sediments in northwestern Ohio. *Journal of Environmental Quality, 7*, 319-325.

Walz, N. (1978). The energy balance of the freshwater mussel Dreissena polymorpha Pallas in laboratory experiments and in Lake Constance. *Archiv Fur Hydrobiologie, 55*, 83-105.

Widdows, J., Brinsley, M. D., & Salkeld, P. N. (1998b). Use of annular flumes to determine the influence of current velocity and bivalves on material flux at the sediment-water interface. *Estuaries, 21*(4A), 552-559.

Widdows, J., Brinsley, M. D., Bowley, N., & Barrett, C. (1998a). A benthic annular flume for in situ measurement of suspension feeding/biodeposition rates and erosion potential of intertidal cohesive sediments. *Estuarine, Coastal and Shelf Science, 46*, 27-38.

Widdows, J., Brinsley, M., & Elliott, M. (1998). Use of in situ flume to quantify particle flux. In K. S. Black, D. M. Paterson, & A. Cramp, *Sedimentary processes in the intertidal zone* (Vol. 139, pp. 85-97). London: Geological Society.

Widdows, J., Friend, P. L., Bale, A. J., Brinsley, M. D., Pope, N. D., & Thompson, C. E. (2007). Inter-comparison between five devices for determining erodability of intertidal sediments. *Continental Shelf Research, 27*, 1174-1189.

Widdows, J., Lucas, J. S., Brinsley, M. D., Salkeld, P. N., & Staff, F. J. (2002). Investigation of the effects of current velocity on mussel feeding and mussel bed stability using an annular flume. *Helgoland Marine Research*, 3-12.

Widdows, J., Pope, N. D., Brinsley, M. D., Gascoigne, J., & Kaiser, M. J. (2009). Influence of self-organised structures on near-bed hydrodynamics and sediment dynamics within a mussel (Mytilus edulis) bed in the Menai Strait. *Journal of Experimental Marine Biology and Ecology, 379*, 92-100.

Willis, D. H., & Krishnappan, B. G. (2004). Numerical modelling of cohesive sediment transport in rivers. *Canadian Journal of Civil Engineering, 31*, 749-758.

Wotton, R. S. (2004). The ubiquity and many roles of exopolymers (EPS) in aquatic systems. *Scientia Marina, 68*(1), 13-21.

Wotton, R. S. (2004). The ubiquity and many roles of exopolymers in aquatic systems. *Scientia Marina, 68*, 13-21.

Wotton, R. S., & Malmqvist, B. (2001). Feces in aquatic ecosystems. *BioScience, 51*(7), 537-544.

Wright, L. D., Schaffner, L. C., & Maa, J. P. (1997). Biological mediation of bottom boundary layer processes and sediment suspension in the lower Chesapeake Bay. *Marine Geology, 141*, 27-50.

Zaiko, A., Paskauskas, R., & Krevs, A. (2010). Biogeochemical alteration of the benthic enviroment by the zebra mussel Dreissena polymorpha (Pallas). *Oceanologia, 52*(4), 649-667.