Master's thesis

Absence of recovery in a degraded eelgrass (*Zostera marina* L.) bed in Nova Scotia, Canada

Results from a transplant study

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Declaration

I hereby confirm that I am the sole author of this thesis and it is a product of my own academic research.

Erin Wilson

Abstract

By the early 2000s, the invasion of the European green crab (*Carcinus maenas*) had contributed to severe decline of eelgrass (*Zostera marina*) beds in eastern Canada. Some eelgrass beds have recovered, but Benoit Cove, Nova Scotia has failed to rebound. The primary objectives of this study were to determine whether the cove has reached a state at which it can no longer return to a healthy eelgrass bed, to evaluate the economic importance of restoring eelgrass in Atlantic Canada, and to explore Canadian policies and management practices for seagrass protection. From 3 July – 29 August 2018, I conducted an eelgrass transplant experiment in Benoit Cove and the donor site, Tracadie Harbour, using a modification of the TERFS method. Transplant survival, rhizome growth, and above-ground growth (canopy height, blade length, number of blades per shoot, and blade width) were measured. Above-ground growth declined in both sites, but all blade-size variables were significantly smaller in Benoit Cove. Negative transplant growth trends may have been tied to higher water temperature and epiphytic algal cover. Tracadie Harbour had a final transplant survival rate of 91.6% which was significantly greater than Benoit Cove (15.4%). Sediment composition, organic matter, and associated biota were also evaluated. The sediment from both sites was composed mainly of silt (> 28%). It is probable that the absence of eelgrass in Benoit Cove induced sediment resuspension and turbidity. The combination of turbidity and estimated drift algal cover (50–75%) may have increased light attenuation to a point at which the transplants could no longer tolerate their surroundings. The biota survey confirmed that Benoit Cove no longer has the faunal community of a typical eelgrass system. Tracadie Harbour was more species-rich and had larger populations of fish and meiofauna, which is expected in seagrass habitats. These species were essentially absent in Benoit Cove; instead, species of molluscs and annelids dominated. The number of green crabs counted in Benoit Cove was roughly 0.01 crabs m^{-2}, which was lower than the recorded number in 2013 (0.03 crabs m^{-2}). Tracadie Harbour had a significantly higher green crab count (0.075 crabs m^{-2}), suggesting that green crab populations are influenced by the presence of dense eelgrass beds.

Atlantic Canada has an economic dependence on commercial fisheries and many of these species depend on the support of *Z. marina* beds at some point in their life. Therefore, the benefits of monitoring and investing in eelgrass restoration can certainly outweigh the costs. For several decades, eelgrass populations have suffered from lax policies and management practices in Eastern Canada. Considering the increasing anthropogenic activities that are negatively affecting coastal habitats, I hereby recommend that Canada establishes strict policies aimed directly at protecting valuable eelgrass habitat, improves data collection and monitoring of *Z. marina*, and increases social awareness and community involvement.

Keywords: Atlantic Canada, eelgrass bed, epiphytic algae, European green crab, seagrass management, transplants, *Zostera marina*

I dedicate this thesis to my late brother, Cameron, who gave me the strength to pursue this master's degree and always taught me to smile even through the toughest of times.

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Acronyms

ANOVA: Analysis of variance

CAMP: Community Aquatic Monitoring Program

CSR: Carbohydrate Storage Reserves

CRF: Coastal Restoration Fund

DFO: Department of Fisheries and Oceans Canada

EBSA: Ecologically and Biologically Significant Areas

ESS: Ecologically Significant Species

EFH: Essential Fish Habitat

HRM: Horizontal Rhizome Method

LOI: Loss on Ignition

NB: New Brunswick

NL: Newfoundland and Labrador

NOAA: National Oceanic and Atmospheric Administration

NS: Nova Scotia

OM: Organic Matter

PEI: Prince Edward Island

PLL: Percent Light at the Leaf

PLW: Percent Light through Water

s: Standard Deviation

TERFS: Transplanting Eelgrass Remotely with Frames

TNV: Total Net Value

USA: United States

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1 Introduction

1.1 Eelgrass (*Zostera marina* L.)

Zostera marina Linnaeus (Fig. 1), commonly known as eelgrass, is a vascular marine macrophyte that forms extensive perennial beds in subtidal and intertidal habitats (Vandermeulen et al. 2012). Eelgrass is distributed along the coasts and in estuaries of North America, Europe, and eastern Asia (Burkholder and Doheny 1968). It is absent in the Antarctic because of extensive ice cover, but numerous observations of *Zostera marina* have been recorded in Arctic regions including Alaska, Iceland and southwest Greenland (Holmer 2009). Eelgrass habitats occur along all three Canadian coastlines but are most abundant in Atlantic Canada (Vandermeulen et al. 2012). Atlantic Canada consists of the easternmost province, Newfoundland and Labrador, and the three Maritime provinces: New Brunswick (NB), Nova Scotia (NS) and Prince Edward Island (PEI). Eelgrass beds are also common in the St. Lawrence estuary and the Gulf of St. Lawrence on coastlines of both Québec and New Brunswick (DFO 2009). In Nova Scotia, *Z. marina* occurs around most of the province but is limiting along the outer portion of the Bay of Fundy (Short and Short 2003). Eelgrass habitat also surrounds PEI and Newfoundland.

Although it is not a true “grass”, *Z. marina* has similar features such as long, slender leaves, roots, stems, flowers and seeds (Burkolder and Doheny 1968). In terms of reproduction, eelgrass can reproduce both vegetatively through clonal growth, and sexually through production of flowers and seeds (Cabaço and Santos 2010). Vegetative propagation involves belowground rhizomes and nutrient-absorbing roots and is the most common reproductive strategy in mature eelgrass beds (Bertness 2007). Sexual reproduction is carried out to colonize new areas, and is essential in maintaining genetic diversity within seagrass meadows (Cabaço and Santos 2010).

Seagrasses, including eelgrass, play a major role in maintaining the biological, physical, and chemical structure of coastal and estuarine habitats. Eelgrass beds are considered nursing grounds because they are highly productive and provide shelter for an abundance of floral and faunal communities (Burkholder and Doheny 1968). Waterfowl species feed directly on the leaves and rhizomes of *Z. marina* (Seymour et al. 2002), whereas most invertebrates prefer to feed on bacteria, epiphytes and smaller organisms that are found on eelgrass leaves (Burkolder and Doheny 1968). These complex habitats implement essential resources that are extremely important for a variety of aquatic organisms including commercially valuable fish and shellfish species. Along the eastern coast of North America, eelgrass beds support important commercial species such as soft-shell clams and juvenile Atlantic cod (Neckles 2015; McCain et al. 2016).

Eelgrass differs from other marine plants in that it is anchored into the substrate with horizontal, branching rhizomes (Malyshev and Quijón 2011). Its unique rooting system provides sediment stabilization and its leaves baffle wave energy, all of which helps prevent coastal erosion (Davis and Short 1997). The stabilization of sediment, especially very fine particles, assists in maintaining water clarity which reduces stress on the system. Eelgrass habitats help maintain water quality and clarity by cycling nutrients and trapping suspended particulates from the water column (Orth and McGlathery 2012). Seagrass meadows are responsible for approximately one-fifth of total oceanic carbon burial (Röhr et al. 2016). Since *Z. marina* is a highly abundant seagrass species on a global scale, it plays a major role with the uptake of anthropogenic carbon dioxide emissions and reducing impacts of ocean acidification (Neckles 2015).

In 2009, the Department of Fisheries and Oceans Canada (DFO) concluded that *Zostera marina* possesses the necessary characteristics required to be classified as an “Ecologically Significant Species” (ESS). High density eelgrass beds in the Atlantic Coastal region of Nova Scotia are also considered as Ecologically and Biologically Significant Areas (EBSA) (Fig. 2; Hastings et al. 2014). In addition to being ecologically significant, eelgrass systems are also economically important. The economical importance of eelgrass in North America began in the late 1800s. Eelgrass blades that were washed up and dried out formed extensive wrack along the shores. This wrack was used as feed and bedding for domestic livestock, and for insulating and sound-proofing houses (Cottam and Munro 1954). From 1907 to 1960, two American companies began making seagrass “quilts” for infrastructure insulation and sound-proofing (Wyllie-Echeverria and Cox 1999). This commercial harvesting of *Z. marina* leaf litter occurred in the small coastal community of Yarmouth, Nova Scotia, Canada.

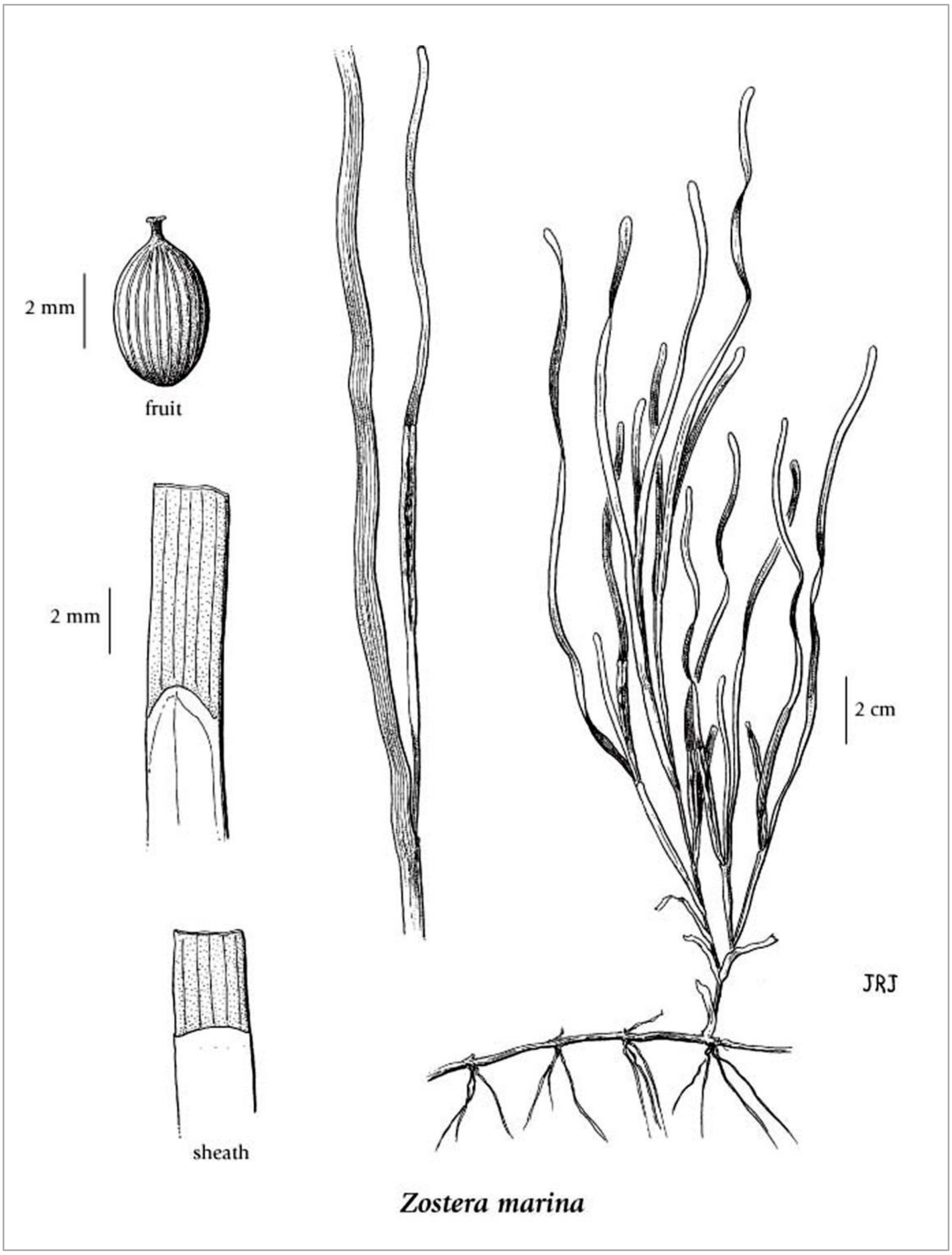


Fig. 1. Illustration of *Zostera marina* showing morphological features. Mature eelgrass blades can reach up to 80 cm long and 3–12 mm wide (figure from Douglas et al. 2001).

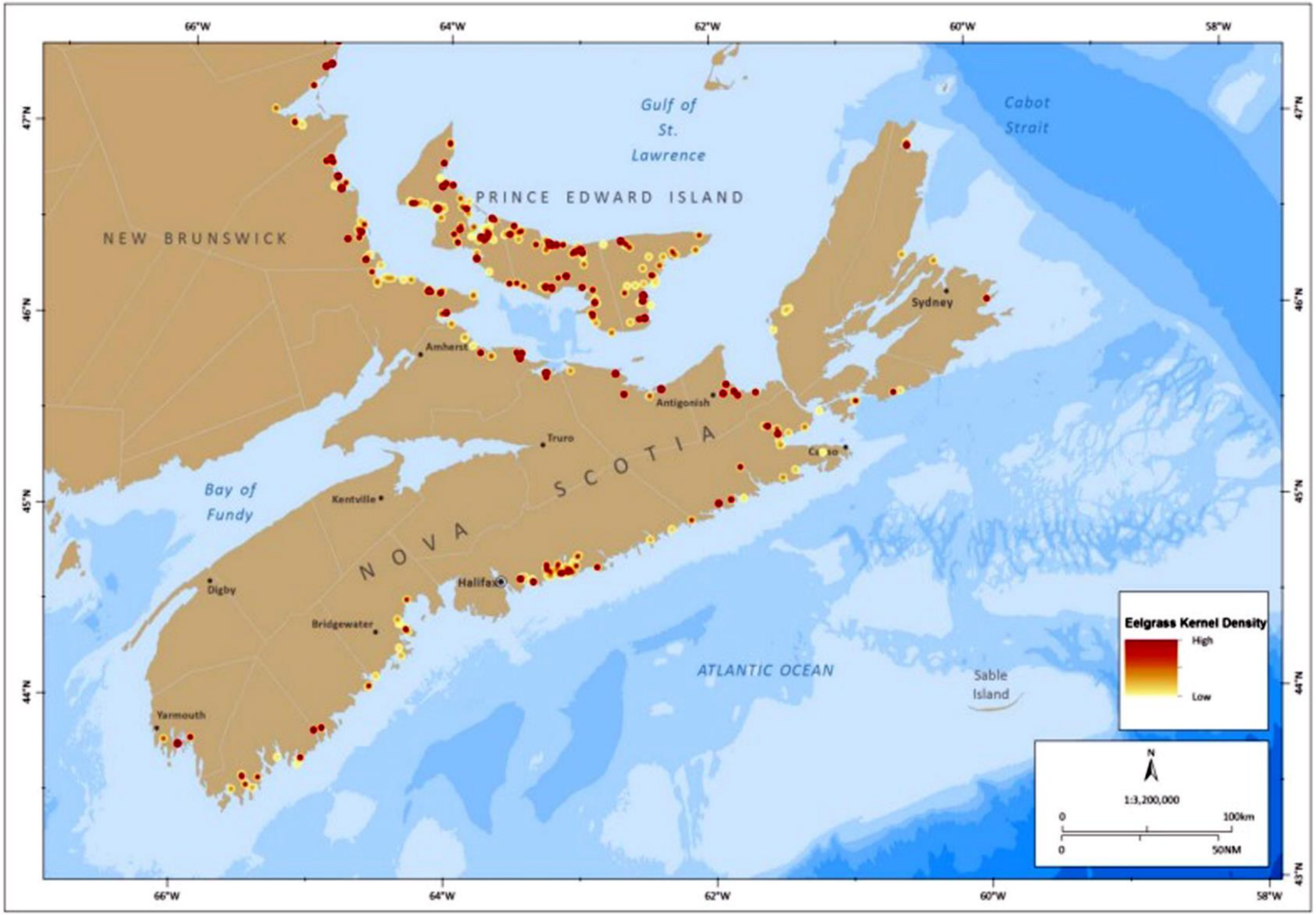


Fig. 2. Extent of eelgrass density within the Maritime region (figure from Allard et al. 2014).

1.2 Decline in Eelgrass Habitats

1.2.1 The Beginning of the Decline: Wasting Disease

The importance of eelgrass was not widely understood until there were reports of significant declines in the distribution and abundance of *Zostera marina*. In the early 1930s, eelgrass beds in North America experienced a decline of over 90% which then brought attention to its contribution to marine food webs (Bertness 2007). The extreme die-off of eelgrass was caused by "wasting disease", an infection of a slime mold pathogen called *Labyrinthula macrocystis* (Short and Short 2003). Wasting disease worsened during years with warmer water temperatures in the summer and winter (Bertness 2007). According to Short and Short (2003), it took roughly 30–40 years for eelgrass habitats to fully recover from this event and some habitats have not yet recovered to their original extent. Wasting disease is always present in low amounts, but reports of episodic wasting disease still occur. Currently, there are several other causes of decline in eelgrass habitat.

1.2.2 Anthropogenic Impacts

On a global scale, seagrass cover has declined by 7% each year since the 1990s (Waycott et al. 2009). Over the last two decades, direct and indirect anthropogenic impacts have been responsible for nearly 18% of the globally reported seagrass declines (Duarte et al. 2004). Within the last century, many reductions in eelgrass habitat have been recorded across all of the Maritime provinces (Valdemarsen et al. 2010). Declines in eelgrass beds have been tied to direct impacts such as fishing and aquaculture, boating activities, marina development and coastal construction (Short and Short 2003). The growth of human activity along the coastline is causing significant alterations to sediment stability and composition, which can disturb water quality and clarity. With the increase in human population along coastlines, there is an increase in development of infrastructure as well. These activities have become major contributors to nutrient loading, and the subsequent eutrophication has a huge impact on the health of eelgrass beds (Short and Burdick 1996). Nutrient loading decreases eelgrass populations because they become smothered by algal species, e.g. *Ulva lactuca*, that thrive in environments with excess nutrients (Short and Burdick 1996).

Eelgrass bed declines have also occurred from indirect anthropogenic threats such as climate change. Without immediate action and management of human population, economic motives, and standard of living, greenhouse gas emissions will continue to increase (WMO 2018). On a global scale, climate change impacts are already noticeable based on increased air and water temperatures, sea level rise, CO_2 levels and the frequency and intensity of storms (Duarte et al. 2004). An increase in water temperature can have a major impact on eelgrass growth, nutrient uptake, reproduction success, and enhance the severity of wasting disease (Kaldy 2014).

1.2.3 Invasive Species

Another indirect anthropogenic impact known to influence the distribution and health of eelgrass meadows is the introduction of invasive species. Increased populations of the European green crab, *Carcinus maenas*, have had significant impacts on the biodiversity of coastal habitats and the density of eelgrass beds (Malyshev and Quijón 2011). The European green crab, is a small crab that originates from the coastal waters of Europe and North Africa (Fig. 3; DFO 2018*a*). In 1817, the green crab was first reported in North America (Say 1817). The green crab reached the western Atlantic through the ballast waters of ships (Locke et al. 2003). In the early 1980s, these crabs had become abundant in Nova Scotia and by 2007, were present along harbours and estuaries in Newfoundland and Labrador (DFO 2011).

Green crabs are extremely aggressive territorial creatures that feed on a variety of shellfish, crustacean and finfish species (DFO 2018*a*). The removal of eelgrass shoots is caused by the green crabs' natural behaviour. In soft-sediment habitats, adult and juvenile green crabs dig pits into the substrate for food and shelter which leads to the uprooting of eelgrass shoots (e.g. Malyshev and Quijón 2011; Garbary et al. 2014). From 2001 to 2002, 13 estuaries along the southern Gulf of St. Lawrence had a mean above-ground biomass decline of nearly 40% (Hanson 2004). A Before-After-Control-Impact (BACI) study in Newfoundland showed that eelgrass habitats with green crabs experienced a 50% decline in biomass since 1998, and up to a 100% loss for sites that had large populations of green crabs for an extended period (Matheson et al. 2016). In Antigonish Harbour, NS, there are a number of eelgrass beds that serve as important foraging sites for migrating birds such as Canadian geese (*Branta canadensis*) and the common goldeneye (*Bucephala clangula*; Seymour et al. 2002). In one year, the eelgrass beds in Antigonish Harbour underwent a 95% loss of biomass which caused a 50% decline in the number of migrating geese and duck species that depended on those beds (Seymour et al. 2002). Since the ecological and socioeconomical threats are now well understood, *C. maenas* was added to the list of top 10 most unwanted species in the world (DFO 2011).

Invasive tunicates (Ascidiacea) have also become a problem in coastal ecosystems because of their bio-fouling abilities which negatively impact native species and their surrounding environment (Sephton et al. 2011). In Atlantic Canada, there are four non-native tunicate species that have become well-established (e.g. *Botrylloides violaceus* and *Botryllus schlosseri*), and several others are still being detected (Sephton et al. 2017). Recent studies have shown that tunicates can attach to eelgrass blades which weakens the shoots and could limit the distribution and density of eelgrass beds (Carman et al. 2016).

There is also increased concern for the introduction of invasive seaweeds and their impacts on coastal communities, including seagrass beds (Williams 2007). A Japanese green alga, *Codium fragile*, also known as the "oyster thief", has become a nuisance plant for oyster beds and eelgrass habitats in Canada (Matheson et al. 2014). In the 1950s, *Codium fragile* was introduced to the east coast of North America through Long Island Sound, New York, and continued to spread throughout New England (Mathieson et al. 2003; Garbary et al. 2004). In 1989, *C. fragile* was present in southern Nova Scotia and reached the Gulf of St. Lawrence by 1996 (DFO 2011). The abundance of *C. fragile* is positively correlated with increasing populations of *Z. marina* (Drouin et al. 2016). A study on four sites in Caribou Harbour, NS, revealed that 57–100% of *C. fragile* plants were attached to the shoots or rhizomes of eelgrass (Garbary et al. 2004). Due to the rapid growth and photosynthetic processes by this invasive seaweed, *C. fragile* becomes increasingly buoyant which causes it to uproot eelgrass and carry oysters away (Loosanoff 1975; Garbary et al. 2004).

Fig. 3. Two adult European green crab (*Carcinus maenas*) individuals competing along the shoreline in Tracadie Harbour, Nova Scotia.

1.3 Research Purpose

Since the early 2000s, the invasion of the European green crab has contributed to severe declines in eelgrass beds among estuarine habitats in Nova Scotia, Canada. While nearby habitats in Antigonish Harbour and Pomquet Harbour have largely recovered over the years, the small inlet of Benoit Cove appears to have entered a new stable state in which it is unable to recover what was once a healthy eelgrass bed. In Benoit Cove, roughly 50 000 m^2 of the total area (68 400 m^2) was comprised of a dense bed of *Z. marina* (Garbary et al. 2014). According to Garbary et al. (2014), on 24 July 2002, the cove's eelgrass density was 175 shoots m^{-2} which declined to less than 50 shoots m^{-2} by 14 September 2002. It was determined that the European green crab caused this rapid decline of the eelgrass bed in Benoit Cove (Garbary et al. 2014). Local scientists re-visited Benoit Cove over the next 11 years and found an increase of approximately 86 shoots per m^2 in 2005 (Garbary et al. 2014). However, in the fall of 2009, eelgrass density had declined to roughly one-third of that observed in 2005, and by June 2013 a complete eelgrass die-off was observed (Garbary et al. 2014). The benthos of the cove in 2018 was comprised of unvegetated (i.e. no macrophytes) sediment with a microalgal and bacterial biofilm, with snails (periwinkles and mud snails), and occasional crabs being the conspicuous fauna.

1.4 Research Aims and Objectives

The aim of this study was to discover whether Benoit Cove has transitioned from a healthy eelgrass habitat to one in which some species can no longer grow. Thus, has the cove entered a new stable state in which it can no longer recolonize? This will be determined by transplanting eelgrass from a nearby habitat into Benoit Cove. Evaluations of environmental conditions will also be carried out in order to determine what is preventing Benoit Cove from returning to its original state. The abiotic and biotic conditions in Benoit Cove will be compared with a nearby eelgrass bed. Findings from this study could help close the research gaps on eelgrass habitat recovery in Eastern Canada. This is important because eelgrass is an Ecologically Significant Species (DFO 2009). The outcome of this research could also help identify whether anthropogenic activities have contributed to the prevention of eelgrass recovery, or if natural causes are playing a significant role. It is important to recognize that if conditions improve in an area, eelgrass can return over time (Leschen et al. 2010). Therefore, this study could help identify whether it is worth taking the next steps towards restoration of the site from environmental and economic perspectives. The hypothesis for this study is that Benoit Cove has reached a state in which the habitat cannot support the return of eelgrass via transplantation of shoot and rhizome fragments.

Research Questions:

I. Has Benoit Cove reached a new stable state at which eelgrass restoration is not a viable option?

II. Are there differences in the ecological and physical environment between Benoit Cove and Tracadie Harbour that might explain the non-recovery in Benoit Cove subsequent to its extirpation between 2005 and 2013?

III. Is it economically beneficial for Atlantic Canada to invest in eelgrass restoration efforts?

IV. Are there any flaws in Canadian policies or management approaches that could negatively affect eelgrass habitats in Eastern Canada?

Research Objectives:

I. To evaluate the growth and survival success of eelgrass in Benoit Cove and Tracadie Harbour through an eelgrass transplant experiment.

II. To identify floral and faunal diversity and abundance in both study sites.

III. To assess the physical sediment conditions in Benoit Cove for comparison with other local eelgrass sites.

IV. To analyze the economic value of eelgrass transplantation based on a literature review.

V. To evaluate Canadian policies for seagrass management, and develop recommendations for managing and restoring eelgrass habitat in eastern Canada.

With regards to coastal management, there needs to be an evaluation on Canadian policies for seagrass management. In Canada, efforts have been made to mitigate the invasion of the European green crab, which is helpful for the overall health of eelgrass beds. However, direct restoration and conservation initiatives for eelgrass habitats in Canada are extremely limited. Recently (2017), DFO released a restoration program for eelgrass mapping and recovery in

Placentia Bay, Newfoundland (NL). In May 2017, DFO also launched a $75 million Coastal Restoration Fund (CRF) under the Ocean Protection Plan to support projects for the next five years (DFO 2019). Rehabilitation of eelgrass beds on the east and west coasts of Canada are listed as priority areas for the CRF projects (DFO 2019). Aside from these relatively new projects, Canada has failed to produce restoration and conservation programs for eelgrass systems in the Maritimes. On a global scale, transplanting seagrasses has become a well-known management tool for restoring disturbed habitats. Nevertheless, Canada has paid little attention to eelgrass transplantation, and therefore lacks significant experience. Suggested management approaches for Canada will be further discussed in Chapter 5.

1.5 Thesis Structure

This thesis is divided into six chapters. Chapter 1 introduces *Zostera marina* and its declining trends on the east coast of North America. Chapter 2 contains a literature review on some of the major physical, chemical, and biological interactions within seagrass habitats which contribute to increasing eelgrass survival and production. Further, a theoretical overview on existing eelgrass transplant methodologies and the techniques chosen for this research experiment are included. In Chapter 3, field, laboratory, and statistical research methods are explained in detail. Chapter 4 describes results from all of the experiments that were carried out from the end of June 2018 until the end of August 2018. The future of eelgrass in Benoit Cove is assessed in Chapter 5 by evaluating its habitat conditions and comparing the results with the donor site in Tracadie Harbour West Arm. In Chapter 6, there is an overview of existing Canadian policies for seagrass protection and what needs to be changed. After the policies, there is a brief evaluation of the economic benefits and values of transplanting and restoring eelgrass beds in Atlantic Canada. Lastly, there is a series of recommendations for improving eelgrass management in Canada.

2 Background

2.1 Site-Selection

When planning seagrass restoration projects, site selection is one of the key components involved in successful transplanting (Fonseca 1998). In the past, unsuccessful transplant projects for *Zostera marina* resulted because sites with limited light availability or poor water quality were selected (Dennison et al. 1993; Fonseca 1998; Short et al. 2002a). Restoring eelgrass on a large-scale is often expensive, which is why there needs to be high likelihood that the sites will respond following the transplant process. Conservation and restoration of eelgrass habitat is much more successful when physiological thresholds are fully understood (Ewers 2013). Knowing the biotic and abiotic thresholds will help identify sites that are suitable for supporting eelgrass systems (Short et al. 2002a). There are several different environmental factors that can impact eelgrass survival and growth (Fig. 4). These factors may vary depending on the region, which is why it is important to evaluate them before transplanting. The most frequently studied variables include, but are not limited to: light attenuation, sediment conditions, water temperature and salinity, and nutrient levels.

2.2 Light Attenuation

Light is one of the most important factors for seagrass distribution because it is required for photosynthesis. In marine environments, light attenuates exponentially with increasing depth (Greve and Binzer 2004). This explains why there is a maximum depth at which growth of *Z. marina* becomes compromised. Typically, eelgrass will not grow beyond 3 to 4 m (Burkolder and Doheny 1968). Furthermore, the ideal depth for eelgrass growth can vary depending on region.

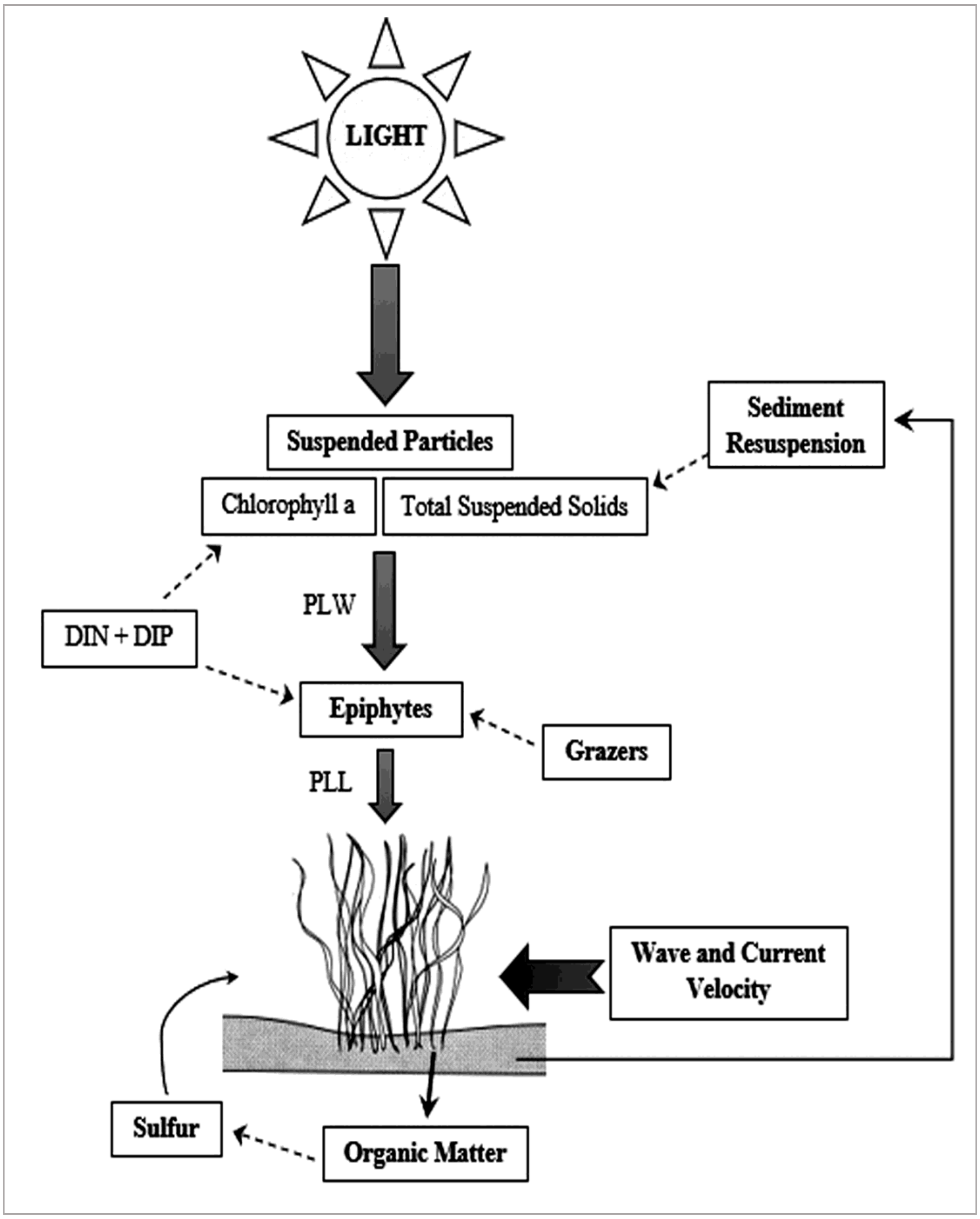


Fig. 4. A conceptual model showing some of the major interactions that can affect the health of eelgrass beds if they are not balanced. PLW = Percent light through water, PLL = Percent light at the leaf, DIN = Dissolved inorganic nitrogen, and DIP = Dissolved inorganic phosphorus (adapted from Batiuk 2000; Simpson and Dahl, 2017).

For example, on the east coast of North America, eelgrass beds grow mostly ≤ 3 m, whereas on the Pacific coast of North America, they are found at depths > 5 m (Murphy et al. 2012). However, with increasing sea level rise, it is expected that the depth limit for plant growth will change over time and beds of *Z. marina* will be found closer to shorelines (Boyer and Wyllie-Echeverria 2010).

Eelgrass is an ideal indicator species for water quality and clarity because eelgrass has a much higher light requirement than other marine plants (Dennison et al. 1993). According to Moore and Short (2006), the minimum light requirement for eelgrass survival in Chesapeake Bay, USA is approximately 15% of surface light at leaves (PLL). This value is equal to 22% of surface irradiance at the maximum depth of eelgrass growth along the east coast of North America (Kemp et al. 2004). There are several different ways in which water column light levels are reduced. Some of the most common factors include sediment turbidity, epiphytic growth, and drift macroalgal cover (e.g. Griffin 1997; Brush and Nixon 2002; Vandermeulen et al. 2012).

Current strength and wave action in coastal regions can prevent eelgrass beds from thriving due to resuspension of sediment particles (Greve and Binzer 2004). Sediment resuspension increases the amount of total suspended solids (TSS) in the water column, thus altering water quality and light availability. This concept is known as turbidity and is one of the main contributors to eelgrass mortality (Vandermeulen et al. 2012). This is why the stabilizing effect of *Z. marina* on sediments is crucial for plant survival and productivity. In Chesapeake Bay, suggested TSS levels for eelgrass is recommended to be < 15 mg l^{-1} (Moore and Short 2006).

In estuaries with *Z. marina,* it is not uncommon to find epiphytic algae attached to eelgrass leaves or as drifting forms of macroalgae (Irlandi et al. 2004). The most common eelgrass epiphytes are green algae, red algae and clumps of cyanobacteria (Burkholder and Doheny 1968). Epiphytic algae can be present as diatoms, crusts of calcified or non-calcified red algae, or filamentous red, green and brown macroalgae (Irlandi et al. 2004; Garbary et al. 2019). A large portion of primary production within seagrass food webs comes from epiphytic algae which support many invertebrate communities. In fact, most grazers prefer epiphytic algae over eelgrass (Cook et al. 2011). In a healthy eelgrass meadow, epiphytic algal loading is often controlled by the abundant herbivores which feed on the plant (Hily et al. 2004). High epiphytic loading and drift algal cover can add significant stress to *Z. marina* beds by increasing shading and decreasing ability to photosynthesize (Miller-Myers and Virnstein 2000). The interaction between light and algae has proven to be one of the most important concepts for understanding eelgrass health. From 1991 to 2001, approximately 62% of the published literature on seagrasses concentrated on light and epiphytes (Koch 2001). Research by Hauxwell et al. (2001) found that surface light was reduced by 63% when eelgrass leaves carried approximately 8 mg cm^{-2} of epiphytic matter. In addition, Höffle et al. (2011) found that epiphytic and drift algae provide additional organic material to the sediment with consequent lowering of oxygen concentrations in the water column. As a result, the lack of sufficient oxygen can create an anoxic environment (Höffle et al. 2011). Although the rooting system of *Z. marina* is quite tolerant to anoxic sediments, these plants can still suffer if a site develops extreme anoxic conditions (Vandermeulen 2005). In addition, the organisms that not only depend on eelgrass beds, but contribute to those systems, will die under such conditions and the ecosystem structure will be impacted.

2.3 Sediment Conditions

Eelgrass can grow in various marine sediments from fine silt to coarse sands, and their morphological structure varies with sediment type. Habitats with fine sediment tend to produce eelgrass with long luxurient leaves and extensive root systems; whereas, sandy sediments tend to produce eelgrass with shorter leaves and elongated rhizomes (Burkolder and Doheny 1968). In coarser substrates, eelgrass beds are better anchored into the substrate which makes them less likely to become uprooted. Some studies have found that because of this, eelgrass beds grow mostly in mixed sediment types or coarse substrata, such as sand (e.g. Burkolder and Doheny 1968; Kemp et al. 2004). According to Kemp et al. (2004), seagrass beds are generally denser in habitats where silt and clay represent less than 20–30% of sediment composition. However, Koch (2001) found healthy eelgrass beds in areas with 2.3–56.3% silt. It is apparent that poorly anchored eelgrass beds can thrive in soft sediments, but that they are highly dependent on wave and current velocities at site level (Wicks et al. 2009).

Sediment types also differ in terms of organic content which can influence eelgrass distribution. Coarse sediments are typically low in organic matter and fine sediments such as silts are rich in organic matter (Burkolder and Doheny 1968). Kemp et al. (2004) determined that seagrasses are typically absent in sites with sediment organic content > 5%, concluding that these plants thrive in coarser substrata. Some studies have found healthy eelgrass beds in higher concentrations of organic matter. Krause-Jensen et al. (2011) determined an organic matter threshold level of 13%, above which eelgrass growth may be limited or compromised. Healthy eelgrass beds have also been observed in sediment organic matter of 16.4% (Short et al. 1993). However, there appear to be more eelgrass beds at sites with organic matter < 5%, and the reason is not well understood (Koch 2001).

Eelgrass systems are intolerant of anoxic and eutrophic environments which include high levels of sediment sulfide (Vandermeulen et al. 2012). Holmer and Nielson (1997) found a direct correlation between sulfur cycling and the density of *Z. marina*. With efficient photosynthesis, eelgrass can oxidize the sediment through its roots, which limits sulfide accumulation (Holmer et al. 2005). In a habitat with lower productivity and available oxygen, sulfur can exceed an acceptable limit and result in plant mortality (Korhonen et al. 2012). Dooley et al. (2012) found that survival of seeds and productivity of adult eelgrass plants are reduced tremendously in habitats with sulfide concentrations greater than 680 μM (micromolar). Conversely, earlier studies found toxic effects at sulfide levels of 400 μM and greater (Goodman et al. 1995).

2.4 Nutrient Levels

Eelgrass uses ammonium, nitrate and phosphate as essential resources for nitrogen and phosphorus which are found in the water column and sediment porewater (Hemminga and Duarte, 2000). In the water column of coastal habitats, inorganic nitrogen is typically a limiting resource; however, in excess it can cause significant damage (Simpson and Dahl 2017). Currently, one of the leading contributors to seagrass declines is increased nutrient input. Over the last 25 years, the decline in eelgrass biomass, canopy height, shoot density, and leaf area in the Great Bay Estuary of New Hampshire and Maine, USA, had been tied to a 59% increase in dissolved inorganic nitrogen (Beem and Short 2009). In Waquoit Bay, Massachusetts, USA, poor housing development and nitrogen loading caused severe losses of surrounding *Z. marina* beds (Short and Burdick 1996). Environmental conditions, including nitrogen levels, can vary from site to site. In Chesapeake Bay, the recommended dissolved inorganic nitrogen level for habitats of *Z. marina* is < 0.15 mg l^{-1} (Moore and Short 2006). Following a large four-season study at 70 sites in southeastern Massachusetts, USA, the sites with over 75% transplant survival success shared a mean total nitrogen level of 0.39 mg l^{-1} (Benson et al. 2013). Because these nitrogen levels are vastly different, it's important to develop thresholds at which species can tolerate their surroundings.

Studies have shown that with an increase in nitrogen levels, there is an increase in macroalgal cover and epiphytic algae which both indirectly and directly destroys eelgrass beds (e.g. Orth and Moore 1983; Short et al. 1993; Short and Burdick 1996; Beem and Short 2009; Kaldy 2014). A nearby example of this occurred in Waquoit Bay, Massachusetts, USA, where anthropogenic nitrogen and increased macroalgal canopies contributed to the decline of several eelgrass beds (Fig. 5; Hauxwell et al. 2001). Hauxwell et al. (2001) hypothesized that increased cover of macroalgal canopies creates unfavorable biogeochemical conditions and reduces light levels which negatively impact eelgrass beds.

The occurrence of increased algal growth in response to increased nutrient input is known as eutrophication (Simpson and Dahl 2017). In an eelgrass bed, eutrophication not only reduces light levels, but also increases carbon loading in sediment, decreases dissolved oxygen levels, and increases hydrogen sulfide concentrations (Vandermeulen 2005). A study on the effects of eutrophication by Goodman et al. (1995) found that the photosynthetic rate of *Z. marina* was reduced to roughly one tenth of its potential in an area with high hydrogen sulfide concentration (800–1000 μM) and poor light availability (15% of solar irradiance).

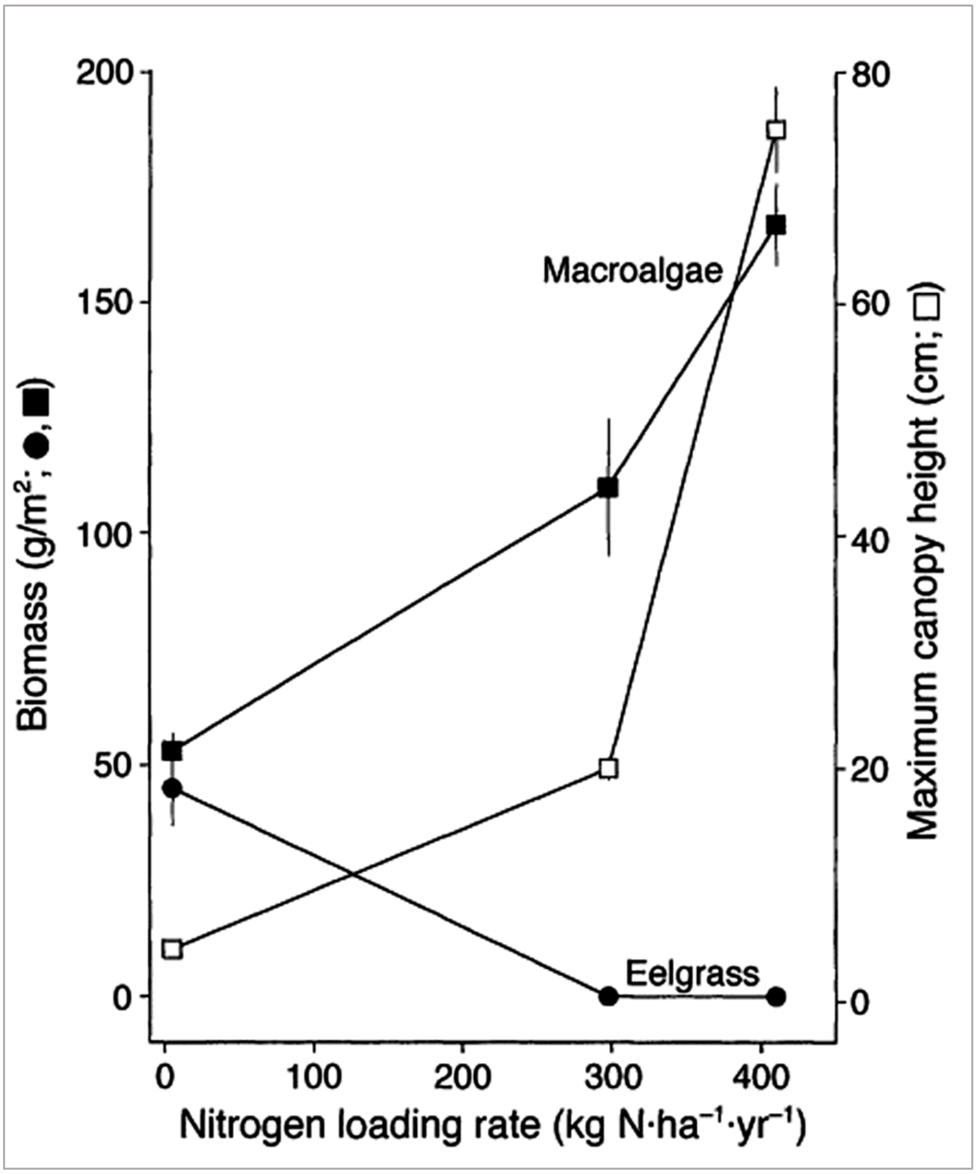


Fig. 5. Relationship between the biomass of eelgrass and the biomass and canopy height of macroalgae with increasing nitrogen loading in the Waquoit Bay, Massachusetts, region (figure from Hauxwell et al. 2001).

2.5 Water Temperature and Salinity

According to DFO (2009), eelgrass can tolerate temperatures from 0°C to 35°C. The necessary water temperature range associated with successful eelgrass reproduction is between 10°C and 20°C (Ewers 2013). Water temperatures over 30°C are considered dangerous and can adversely affect eelgrass beds. In areas where water temperature reaches the upper limit of thermal tolerance, there is a decrease in productivity and distribution of *Z. marina* (Moore and Short 2006). Despite tolerance to high water temperature, eelgrass is more susceptible to wasting disease starting at temperatures from 18 to 25°C (Kaldy 2014).

Eelgrass is typically well adapted to sudden and extreme changes in salinity (Greve and Binzer 2004). Eelgrass beds have been observed in salinities varying from hyposaline to oceanic water (Salo et al. 2014). According to DFO (2009), eelgrass can tolerate low salinities (~5‰), high salinities (~35‰) and even freshwater for short periods but the optimal range is 20–26‰. Recommended salinity range can vary depending on the region. In Chesapeake Bay, USA, the suggested salinity regime for eelgrass transplant survival is approximately 18‰ (Kemp et al. 2004). According to Vandermeulen (2005), the salinity threshold at which *Z. marina* habitats could be damaged is 26 – 30‰ and salinities > 30‰ could result in mortality.

2.6 Eelgrass Transplant Methodologies

The general understanding and application of techniques involved with seagrass restoration has been developed over the last 30 years. The United States, Europe, Australia, and South-East Asia dominate this field and have been highly successful with seagrass transplantation (Paling et al. 2007). Most eelgrass transplant studies take between 1 and 5 years before measuring success because plant growth depends on seasonal variability (e.g. Bach 1993; Orth et al. 2006; Kwak and Huh 2009).

There are several protocols for transplanting eelgrass, but each comes with advantages and disadvantages (Table 1). These techniques vary depending on site conditions, time, labour and funding. In North America, there are three well-known transplanting styles used in eelgrass restoration. Davis and Short (1997) identified these as: transplanting eelgrass shoots with original sediment still attached (sods and cores), transplanting adult shoots with bare roots, and the dispersal of mature seeds (Fig. 6).

The advantage to using sods or cores is that it creates a natural anchoring system which keeps the eelgrass shoots stabilized (Orth et al. 2006). Sod or core units are collected mechanically or by using a hand coring device (Davis and Short 1997). Orth et al. (2006) found that within the first 1–2 months of transplanting cores in Chesapeake Bay, eelgrass survival ranged between 94 and 100%. The downside of this method is that it can destroy the donor bed (Davis and Short 1997).

The bare-root technique can be performed with or without anchors. Both techniques can be successful depending on habitat conditions (Phillips 1990). Planting shoots without anchors can be done by carefully pushing the rhizomes into the first 2–5 cm of the sediment on a slight angle which mimics the natural anchoring system (Davis and Short 1997; Eriander et al. 2016). In Chesapeake Bay, single shoots were transplanted without anchors and had a 73% survival rate after the first month (Orth et al. 1999). According to Eriander et al. (2016), using that type of method is highly recommended for shallower habitats.

Davis and Short (1997) created a variation of the bare-root method called the Horizontal Rhizome Method (HRM), which involves overlapping two eelgrass shoots around a bamboo staple with their rhizomes growing in opposite directions. HRM requires fewer transplants and reduces damage to the donor site, but the technique may not be best suited to sites with high current and wave exposure (Davis and Short 1997). In eight New Hampshire sites, transplant survival after the first year ranged between 75% and 99% (Davis and Short 1997). In Korea, oyster shells were also used as anchors for transplanting shoots of *Z. marina* (Lee and Park 2008). Lee and Park (2008) revealed that after the first 2–3 months, over 78% of transplants survived, whereas 71–72.7% survived when using two older techniques.

Table 1. Existing eelgrass transplant methodologies with advantages and disadvantages for each technique.

Technique	Advantages	Disadvantages	Supporting Studies
Cores and Sods	• Preserves original rooting system • High survival rate (first 2 months) • Unaffected by seasonal variation	• Damages donor site • Labour intensive • Costly	Phillips (1990) Davis and Short (1997) Orth et al. (2006)
Bare-Root Method	• Less damaging to donor site • High overall growth and survival • Similar to natural anchoring system	• Time and labour (SCUBA) intensive • Intense environmental conditions can uproot the unanchored transplants	Davis and Short (1997) Orth et al. (1999)
Horizontal Rhizome Method (HRM)	• Lower costs and disturbance on donor site than BRM • High overall success rate • Prevents uprooting of shoots	• Labour intensive and time consuming • 2-month survival is lower than other methods	Davis and Short (1997) Orth et al. (1999) Orth et al. (2006) Eriander et al. (2016)
Seeding	• Feasible • Labour and time efficient • Increases genetic biodiversity • Little impact on donor site	• Vulnerable to intense habitat and weather conditions – laterally transported or buried • Low survival and germination success	Phillips (1990) Williams & Orth (1998) Davis and Short (1997) Valdemarson et al. (2010)
Frames (TERFS)	• Protects from bioturbators • Lower costs and less intensive	• Extremely site-specific • Cannot be used in soft-sediment habitats	Short et al. (2002*b*) Leschen et al. (2010)

Transplanting seagrasses using seeds is beneficial because it maintains strong genetic diversity within an eelgrass bed (Fonseca et al. 1998). Conserving genetic diversity in a population can increase resilience to disease, disturbance, and climate change (Williams and Orth 1998; Ehlers et al. 2008). Reproductive shoots can be placed in mesh bags or the seeds can be dispersed by hand. According to Tanner and Parham (2010), both techniques are typically less costly than planting adult shoots. However, direct seeding often leads to limited success. In North America, up to 5% of seedings of *Z. marina* survived a large-scale transplant experiment (Marion and Orth 2010). The reduced survivorship of eelgrass seeds has been tied to various factors including predation (Fishman and Orth 1996; Infantes et al. 2016), unsuccessful germination (Moore et al. 1993), failed seed transportation due to unfavourable hydrodynamic conditions, intense burial through sediment resuspension and cover by intense macroalgal drift (Orth et al. 1994; Valdemarsen et al. 2010).

2.6.1 TERFS™ Method

The TERFS™ (Transplanting Eelgrass Remotely with Frames System) method, was established in the late 1990s by Short et al. (2002*b*) and is a registered trademark of the University of New Hampshire (Fig. 6). This method was created as a low-cost technique that is performed in shallow waters where scuba divers are not required (Short et al. 2002*b*). It is also beneficial for transplanting at deeper depths or polluted sites because of the original TERFS design. These frames have weights on either side which allow them to be dropped off from a boat rather than placed by a scuba diver. The initial method attaches 50 eelgrass shoots (200 shoots per m^2) to a 60 cm x 60 cm x 15 cm wired frame (Evans and Leschen 2010). Eelgrass shoots can be attached to the frames with biodegradable thread so that they are held down until the plant roots into the sediment. The frames can then be removed, leaving the eelgrass transplants in the sediment. These frames are also designed to protect transplants from bioturbating organisms (Short et al. 2002*b*). The first TERFS experiment revealed a one-month transplant survival between 53% and 86% (Short et al. 2002*b*). A TERFS study in Korea had a survival rate between 58.7 and 69.0% after almost one year (Park and Lee 2007).

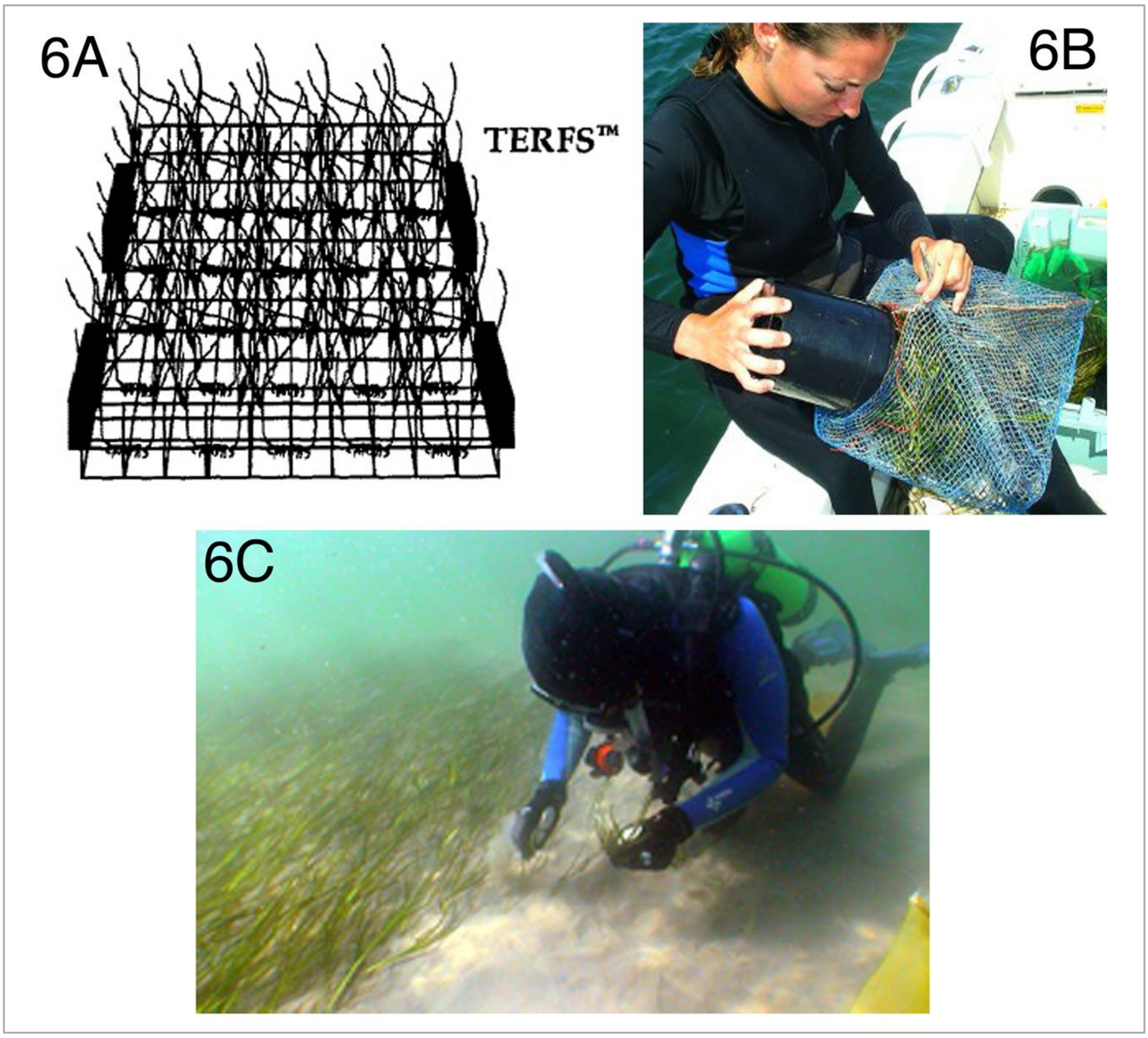


Fig. 6. Eelgrass transplant methods. (A) Free-planting eelgrass shoots by hand (figure from CCE 2010), (B) Reproductive shoots are placed into a mesh bag for buoy-deployed seeding (figure from Pickerell et al. 2006), and (C) TERFS design with eelgrass shoots attached (figure from Short et al. 2002*b*).

3 Methodology

3.1 Study Sites

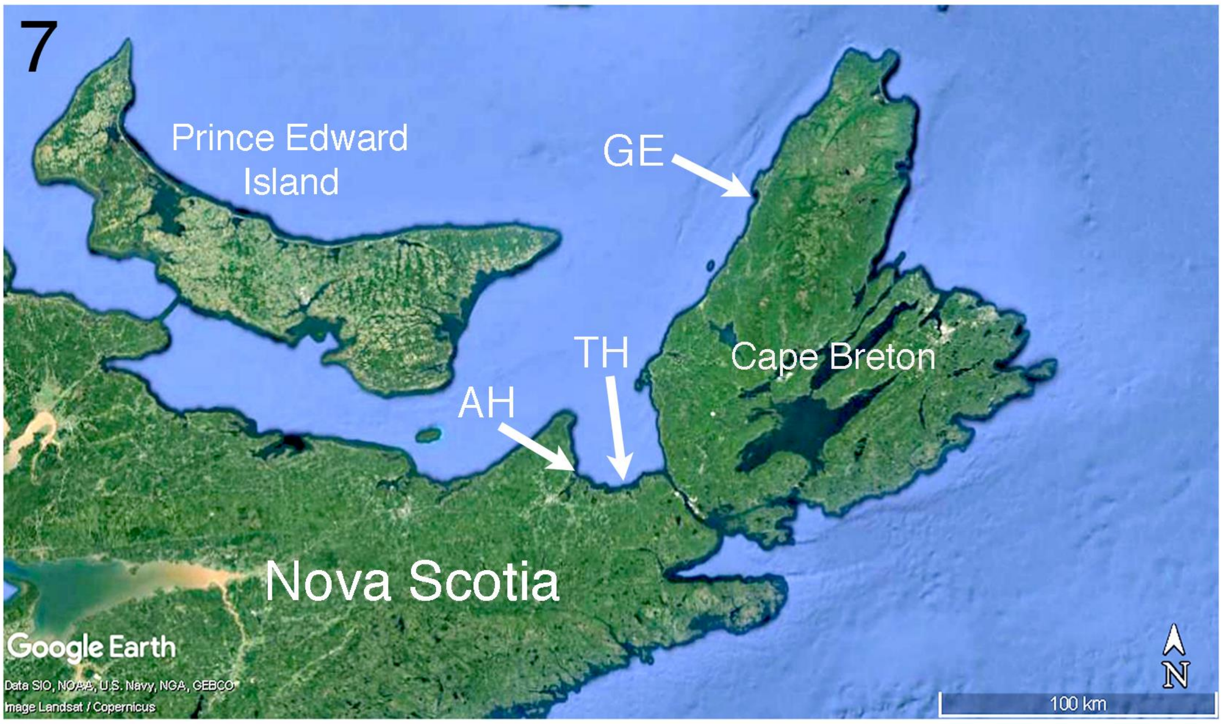
The study included four sites in Nova Scotia (NS), Canada. Three of the sites are in Antigonish County and one is on Cape Breton Island (Figs 7–8). Two of the three sites in Antigonish County are in Tracadie Harbour and the other is in Antigonish Harbour. Both harbours exit to St. Georges Bay, a body of water 25 km wide that separates the mainland Nova Scotia peninsula from Cape Breton Island. St. Georges Bay leads into the Northumberland Strait, which is in the southern part of the Gulf of St. Lawrence. The eelgrass transplant experiment occurred at two sites that were both associated with Tracadie Harbour (Fig. 9).

Tracadie Harbour (6.65 km^2) has a westerly extension called the Tracadie West Arm, which is divided from the remaining harbour by a gravel road. Limited water exchange occurs between the two areas via a small culvert under the road which connects the Tracadie West Arm to the larger basin (see Fig. 9). The donor bed was located within the comparison site (hereafter, Tracadie Harbour). Tracadie Harbour is a small sheltered area located in the north-eastern part of Tracadie West Arm (45°63.82'N; 61°66.05'W). The land surrounding the study site (area of ~25 000 m^2) is a salt marsh and largely covered at high tide. There is one gravel road which passes the site and connects all of the islands in the West Arm. There are a few houses and cottages scattered along the shore. An eelgrass community follows the harbour shoreline and is continuous roughly 10–15 m from the salt marsh border. We selected Tracadie Harbour as the donor and comparison site for the experiment because it supports a mature eelgrass bed and has been resilient following almost two decades of disturbances by the green crab. We replicated the eelgrass transplant study in three different subsites called subsites A, B and C (Fig. 10).

The test site, Benoit Cove (45°37.92′N; 61°37.67′W), is a small, sheltered cove, which feeds into East Tracadie Harbour through a narrow passage (Figs. 8 and 10). Most of the passage is occupied by a sand bar that is exposed at low tide. A small freshwater stream runs into the head of the cove. Presently, Benoit Cove no longer supports eelgrass and is depauperate in macrophyte species throughout the entire cove (Garbary et al. 2014). Biofilm and diatom species form a layer across the sediment surface. Distributed along the shoreline intertidal are developing oyster beds (*Crassostrea virginica*) and rocks with bladder wrack (*Fucus vesiculosus*). An abundance of trees and shrubs shade a large portion of the cove on the north side and are less than 5 m from the water. The cove on the south side is bordered by at least 5 m of grass and shrub vegetation before the start of a large hayfield, but there are no intense farming or roads near the study site. In Benoit Cove, we also replicated the transplant study in three different subsites (Fig. 10).

We compared various features in Tracadie Harbour and Benoit Cove with an eelgrass bed in Antigonish Harbour and an eelgrass bed in Grand Étang Estuary (Table 2). Antigonish Harbour is a large estuary located 20 km north-east of the town of Antigonish and 20 km west of Tracadie Harbour. The comparison site in Antigonish Harbour (45°68.14′N, 61°90.93′W) was previously named Green Crab Cove by local scientists because it had also experienced a severe decline in eelgrass due to a green crab invasion in the early 2000s. According to Garbary et al. (2014), eelgrass meadows in Antigonish Harbour had recovered to approximately 60% of their original extent before the green crab invasion.

The second comparison site, Grand Étang (46°53.28′N, 61°03.89′W), is a small estuary located in Inverness County on Cape Breton Island. This estuary is approximately 2.20 km long, has a maximum width of 0.36 km, and opens directly into the Gulf of St. Lawrence via a rock-walled channel. The eelgrass meadow of interest is located at the top of the estuary near a small stream. In this estuary, eelgrass blades can be up to 1 m long. Over the last decade, there have been reports and counts of the European green crab in the Grand Étang and near Chéticamp Harbour, but there are no records or current evidence of significant impacts on the eelgrass bed.



Figs 7–8. Map of north-eastern Nova Scotia. **Fig. 7.** Locations of Antigonish Harbour (AH), Tracadie Harbour (TH) and Grand Étang Estuary (GE). **Fig. 8.** Map of Antigonish County showing locations of Green Crab Cove (GCC) in Antigonish Harbour (AH) and Tracadie Harbour (TH). (Maps from Google Earth Pro).

Figs 9–11. Maps indicating locations of the two sites involved in the transplant. **Fig. 9.** Map of Tracadie Harbour showing test site (BC) and comparison site (TH). **Fig. 10.** Enlarged map of Tracadie Harbour showing three transplant subsites and donor bed (D). **Fig. 11.** Map of Benoit Cove showing subsites A, B and C. (Maps from Google Earth Pro).

Table 2. *Zostera marina* study sites with coordinates, dates visited and parameters measured.

Site	**Coordinates**	**Date(s) Visited**	**Parameters**
Tracadie Harbour	45°63.82′N; 61°66.05′W	28/06/18 to 4/9/18	• Eelgrass transplant success • Sediment composition • Sediment organic matter • Biota communities • Green crab abundance
Benoit Cove	45°37.92′N; 61°37.67′W	28/06/18 to 4/9/18	• Eelgrass transplant success • Sediment composition • Sediment organic matter • Associated biota • Green crab abundance
Antigonish Harbour (Green Crab Cove)	45°68.14′N, 61°90.93′W	23/10/18 to 27/10/18	• Sediment composition • Sediment organic matter • Green crab presence
Grand Étang Estuary	46°53.28′N, 61°03.89′W	8/08/18 to 9/08/18	• Sediment composition • Green crab presence

3.2 Eelgrass Transplanting

3.2.1 Frame Design and Eelgrass Collection

The procedure for transplanting eelgrass was similar to the TERFS method (Short et al. 2002*b*) but we used a modified frames technique described by Leschen et al. (2010). Each frame measured 0.25 m x 0.25 m, and was constructed of PVC pipe (1.5 cm internal diameter) and stiff plastic webbing (mesh size 3 cm × 3 cm). Two holes were drilled into each arm of the frame to ensure that it would sink. Every second column of the webbing was cut off to facilitate the growth of the plants. We used an assortment of coloured cable ties to create unique colour patterns for easy frame identification. Both sites had 12 frames.

Due to the amount of effort and time required to complete the transplant procedure, collection and transplanting were carried out on different days. We conducted the transplant experiment for the comparison site, Tracadie Harbour, on 3 July 2018. The experiment for the test site, Benoit Cove, began on 6 July 2018. Both sites had a total of 108 adult eelgrass shoots collected from the donor bed at low tide (Fig. 10). The specific area in Tracadie Harbour was selected for plant removal because it was easily accessible and located far enough away that harvesting would not interfere with the subsites. Only non-reproductive shoots were collected for attachment to the frames. As per Zhou et al. (2014), eelgrass shoots had a rhizome length of 2–3 cm with roots. Collected shoots were stored in a cooler with cold seawater to avoid desiccation and exposure. The cooler with the Benoit Cove transplants was placed on top of ice packs and immediately transported to the site. The eelgrass shoots were transplanted within a maximum of 5–6 hr after harvesting.

3.2.2 Transplanting Procedure

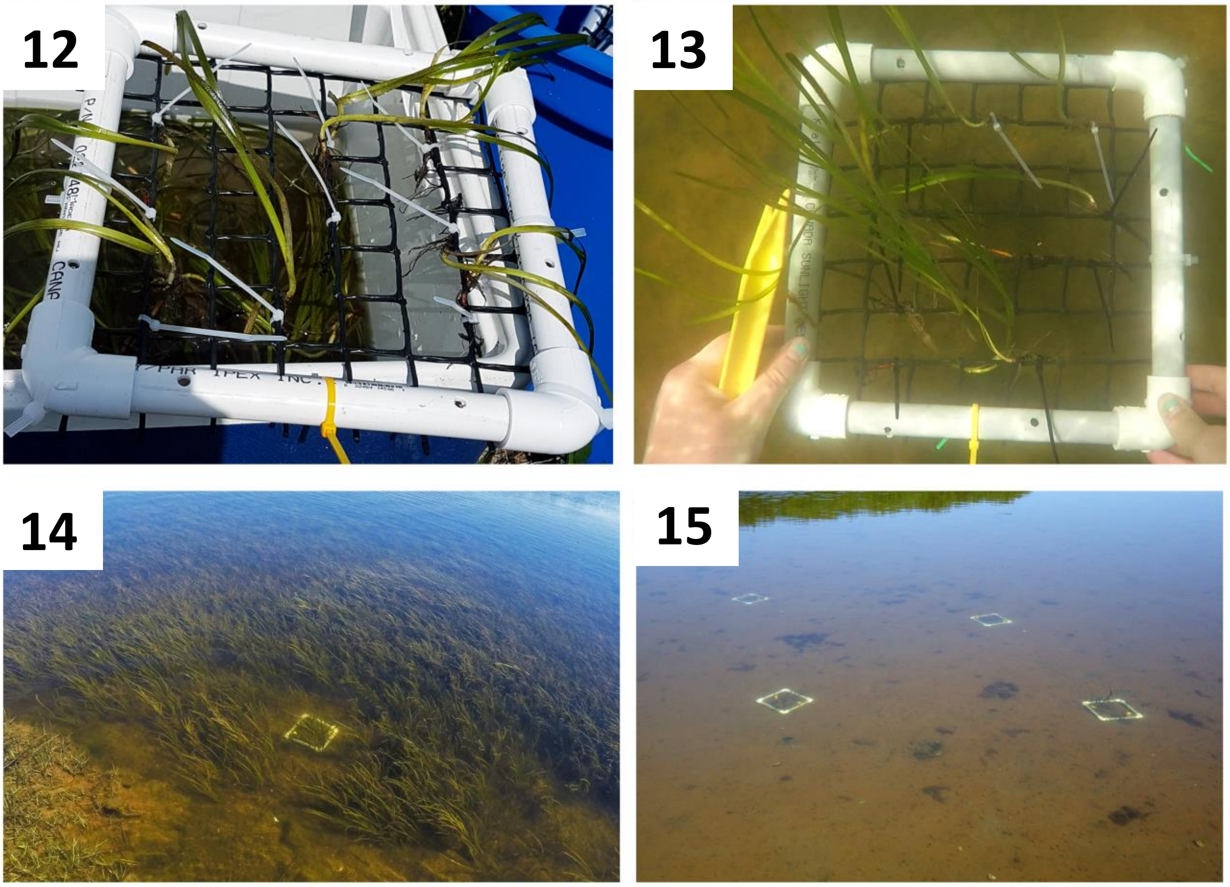
Nine adult eelgrass shoots were attached to each frame. Before attaching each individual plant to a frame, rhizome length, blade length, and the number of blades per shoot were recorded. Blade length was measured from the base of the sheath to the tip of the longest blade. The rhizome was carefully fastened to the bottom of the wired mesh using a cable tie. This allowed us to plant the rhizome horizontally into the sediment at a depth of 1–2 cm, mimicking the natural position of eelgrass rhizomes (similar to Davis and Short 1997). In each row, three plants were tied along the column, 5 cm apart. During this process, we placed the frame into the cooler in seawater so that the eelgrass blades were submerged. Once all eelgrass transplants were attached, the frames were secured to the substratum using 30-cm tent pegs.

In Tracadie Harbour and Benoit Cove, we transplanted four frames per subsite (~8 m apart). Frame placement in Tracadie Harbour was altered because of the dense eelgrass bed. There was not enough open area to accommodate a systematic placement of the frames in groups of four. Therefore, each frame was placed in any available bare patch (without eelgrass) within the subsite. These frames were placed in a linear formation which was separated by 1–5 m. In Benoit Cove, we systematically placed the frames in a square, spaced by 1 m. The eelgrass transplanting procedure is illustrated in Figs 12–15.

3.2.3 Measurements and Observations

The eelgrass transplants were visited once every 1 to 2 weeks from 3 July to 29 August for observation. For exact visiting dates, see Table 3 in Section 4.1. To avoid disturbing the sediment, causing stress to plants, and impacting underwater vision, I used a mask and snorkel and an inflated air mattress for sampling and measurement. While floating over each frame, the length of the longest blade per transplant was measured and recorded. During each visit, we also recorded the number of surviving and missing shoots per frame. Transplants with only the rhizome still attached to the frame were recorded as dead. All eelgrass transplants that slipped out of the cable ties were recorded as missing and were omitted from the initial population size.

During each visit, we collected nine water samples within the area of the three subsites. The custom-designed water sampling device allowed us to collect samples near the surface, at half depth and full depth, with minimal disturbance to the surrounding environment. Water temperature was measured using a glass alcohol thermometer to the nearest 0.5 °C. Salinity was measured to nearest 0.5‰ using a Portable Refractometer (Aqueous Lab, Las Vegas, Nevada, USA). Water temperature and salinity data are included in Appendix 11.



Figs 12–15. Transplanting adult eelgrass shoots using PVC frames (0.25 m²). **Fig. 12.** Fully constructed frame with nine eelgrass shoots attached and ready for transplantation. **Fig. 13.** Securing the frame with tent pegs in Benoit Cove. **Fig. 14.** Transplant frame placed in a bare patch within a mature eelgrass bed in Tracadie Harbour. **Fig. 15.** Subsite B transplants from Benoit Cove in a square grid formation.

After seven weeks, we carefully removed the frames with the remaining eelgrass transplants attached and transported them back to the Jack McLachlan Laboratory of Aquatic Plant Resources. Transplant survival and mortality were calculated per frame for each observation date. Trends in transplant growth throughout the study were assessed using multiple parameters:

- Eelgrass blade length (cm): for the oldest (longest) blade per transplant.
- Blade abundance: initial and final number of leaf blades per transplant.
- Rhizome length (cm): for each transplant. Initial and final rhizome lengths represented the combined length of the horizontal and vertical rhizome structure.
- Canopy height (cm): for each frame. We calculated the mean of the longest two-thirds of the plants in the frame (similar to Hansen and Reidenbach 2013).
- Blade width (mm): for the longest blade of each transplant that survived the seven-week experiment.

3.3 Sediment Analysis

3.3.1 Sediment Composition

Sediment samples were collected using a 10-cm-diameter custom-designed coring device. We collected sediment samples at a depth of 10–15 cm. Samples closer to shore were collected by wading, and samples further out were collected by snorkeling. To decrease stress on the eelgrass transplants from sediment resuspension, we sampled roughly 5 m away (downstream if there were currents) from the frames. Sediment composition was estimated using 1-l Imhoff Settling Cones (Wheaton 06340-02, Vernon Hills, Illinois, USA). We collected 12 sediment cores from both transplant sites and six cores from Antigonish Harbour and Grand Étang Estuary. The sediment solution (900 ml) which used 300 ml of the core sediment and 600 ml of tap water, was mixed to form a slurry. The solution was then poured into one Imhoff cone. The amount of sediment that settled out after 1 min (sand), 1 h (silt), and 1 d (clay) was recorded. This procedure was repeated for all four sites.

3.3.2 Organic Matter Content

Organic matter content was estimated using the loss on ignition (LOI) method (as per SFU Soils Lab 2011). We collected 12 cores from both transplant sites and six cores from Antigonish Harbour. We measured approximately 50 cm^3 of the sediment from the original core and used a scale to record the wet weight. We put the cores into pre-weighed porcelain crucibles (CoorsTek 60110, Vernon Hills, Illinois, USA) and placed them in a drying oven (Heratherm-Thermo Scientific, Waltham, Massachusetts, USA) at 70 °C for 72 h. The dry weight of the cores was then recorded. We determined total OM content by ashing the sediment core in a muffle furnace (Barnstead Thermolyne 6000, Waltham, Massachusetts, USA) at 550 °C for 4.5 hours. The mass weight of the remaining material was subtracted from the initial dry mass to determine the total OM content. We repeated this procedure for each site. Original sediment organic matter data and analysis can be found in Appendices 16–17.

3.4 Species Richness

Macrofaunal and macroflora were identified and noted during each visit at both sites. Estimated abundance for each species was assigned based on its frequency throughout the experiment compared to other species. We used a classification of abundance for each animal and algal species: absent, low, low-moderate, moderate, moderate-high, or high. Classification was determined based on the frequency that species were observed and whether they were found in large masses or not. Using a stereoscope (Nikon SMZ-1, Tokyo, Japan), we observed smaller invertebrates on sample eelgrass shoots, on algae, and burrowed into epiphytes. Using the identification key, *A Practical Guide to the Marine Animals of Northeastern North America* (Pollock 1998), we identified the various invertebrate species. We also recorded European green crab abundance at subsites A–C for both study sites on each visit. We were careful to avoid counting the same green crab multiple times, by completing one swim through the three transplant sites (8 m^2 each) and counting every green crab individual observed at that time.

In Tracadie Harbour, we collected samples of wild eelgrass with algal epiphytes and brought them back to the lab for taxonomic identification. Algae attached to the Benoit Cove transplants and frames were identified after the experiment. At both sites, we collected drift macroalgae from around the eelgrass transplant frames for subsequent observation. We stained fragments of algal species with iodine and used with a microscope (Olympus BH-2 Series, Tokyo, Japan) to determine characteristics such as cell size, shape, and number of chloroplasts and pyrenoids for the identification using the *Illustrated Key to the Seaweeds of New England* (Villalard-Bohnsack 2003).

Changes in epiphyte cover were observed and noted throughout the study. We took a visual estimate of epiphytic and drift algal percent cover on the eelgrass blades and on the frames *in situ*. I also reviewed underwater photos of the frames on each observation date. Algal cover was considered in five categories: Absent (0%), Low (1–25%), Moderate (25–50%), High (50–75%), and Intense (75–100%). By estimating percent cover for each frame per visit, we derived cover values for the three subsites in Tracadie Harbour and Benoit Cove.

3.5 Data Analysis

All data was compiled and analyzed using the Data Analysis Tool Package in Microsoft Excel, SPSS Software 25, and Minitab 18. When comparing the sample means between Tracadie Harbour and Benoit Cove, a two-sample Student's *t*-test was performed using Microsoft Excel. Before the *t*-test, an F-test was carried out to determine whether equal variance occurred between the two samples (Zar 2010).

Minitab 18, was used to carry out a Nested Analysis of Variance (ANOVA), also known as a Hierarchical ANOVA. A nested ANOVA is carried out when there is one factor among multiple levels or subgroups (Zar 2010). This statistical method was carried out for mean transplant survival and canopy height recorded after seven weeks. Results from the nested ANOVA would reveal whether the factor was statistically different between the study sites and subsites. If the test detected at least one significant difference between levels, a multiple comparisons test for means was used. The widely accepted Tukey's Honest Significant Differences test (hereafter Tukey HSD test) was chosen to determine which site or subsites were significantly different (Zar 2010). Data tables and analyses for transplant survival can be found in Appendices 1–2 and in Appendices 9–10 for canopy height.

Sediment data was analysed using several tests from the software SPSS 25. A one-way ANOVA was carried out for sediment composition and organic matter content because both variables involved more than two sites (Zar 2010). Before conducting the ANOVA, I used the general tests on SPSS 25 to ensure that the assumptions of normal distribution and equal variance were met. A Tukey HSD test was used for analyzing which sites were significantly different. If the data had unequal variances but normal distribution, I used a Welch's ANOVA (Zar 2010). Since the Tukey test is less resistant to heterogeneous variances, the Games-Howell non-parametric post-hoc test was used instead to determine which sites were significantly different from each other (Zar 2010).

4 Results

4.1 Eelgrass Transplant Study

4.1.1 Observation Dates and Missing Plants

The transplant experiment lasted 7 weeks. The two study sites were observed on different days because of the time required at each site (Table 3). The initial goal was to visit the sites bi-weekly, but this was not always possible because of weather and site conditions.

Both sites started with 108 adult eelgrass shoots. In Tracadie Harbour, two frames from subsite A had four reproductive shoots inadvertently included instead of vegetative ones. These four shoots were not included in the initial population. Transplants that went missing over duration of the experiment were omitted from the initial population size. After seven weeks, five transplants had slipped out of the Tracadie Harbour frames and 17 went missing in Benoit Cove. In Tracadie Harbour, two frames from subsite A and one from subsite B could not be found due to intense cover by macroalgae. The transplants from these missing frames were also ommited from the initial population size in Tracadie Harbour and will be searched for during the spring of 2019. The initial eelgrass transplant population for Tracadie Harbour was reduced to 72 transplants and Benoit Cove was reduced to 91 transplants.

The missing transplants were not included for most of the analyses, however, there were exceptions. Blade length of transplants before going missing were used when comparing growth trends with the plants that survived or eventually died off. Since the nested ANOVA typically requires even sampling sizes for all groups and subgroups, the final mean transplant survival and canopy height results for the missing frames were derived using the mean imputation method. From this technique, the survival and canopy height means from the frames in the same subsite as the missing frame were used to calculate means which replaced the absent data.

Table 3. Dates for eelgrass transplant measurements in Tracadie Harbour and Benoit Cove.

Visit (#)	Tracadie Harbour	Benoit Cove
1	3 July, 2018	6 July, 2018
2	19 July, 2018	17 July, 2018
3	27 July, 2018	24 July, 2018
4	14 August, 2018	16 August, 2018
5	28 August, 2018	29 August, 2018

Visit 1 = Transplanting date, Visit 5 = Removal of frames

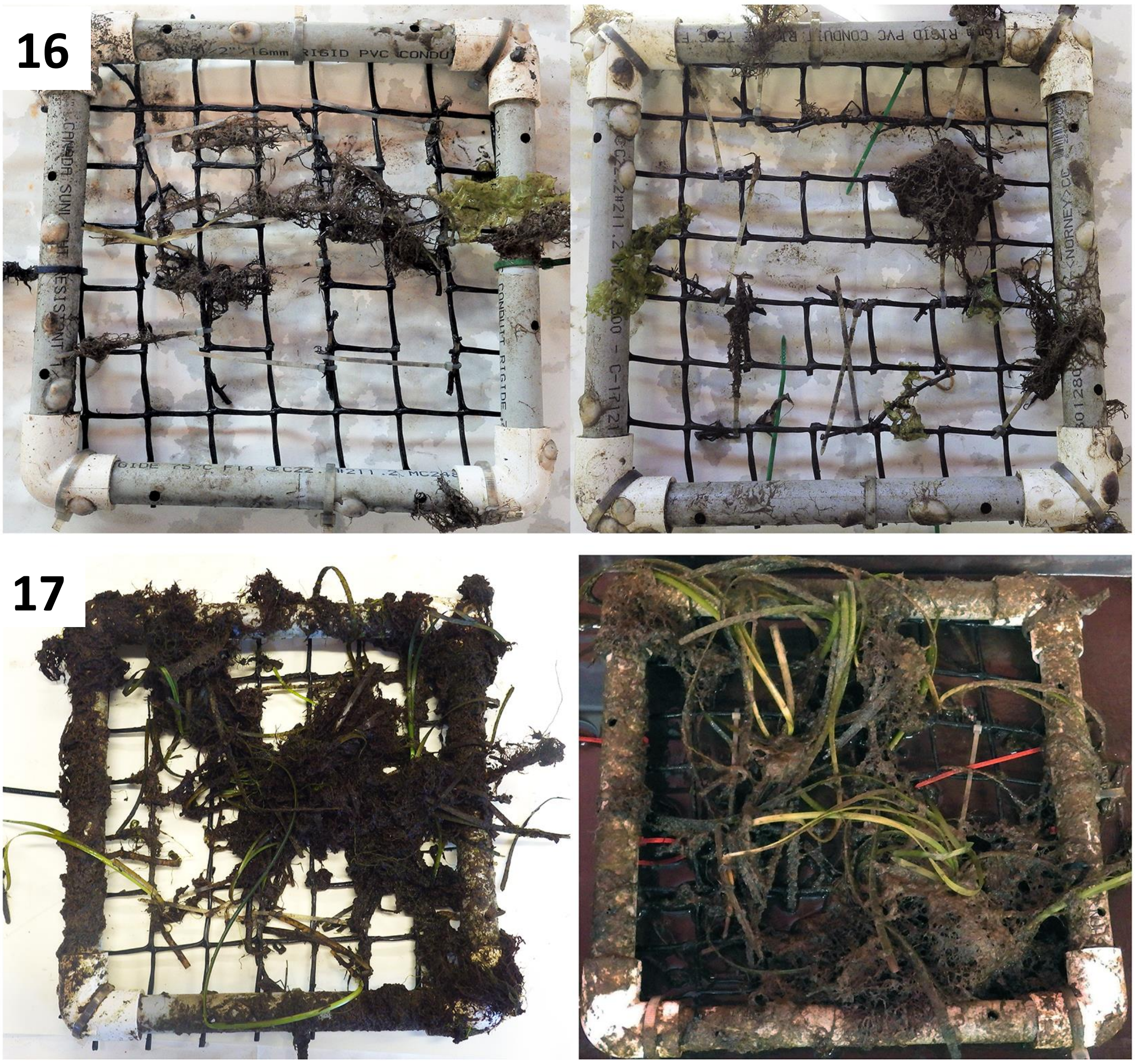
4.1.2 Transplant Survival

After the first two weeks of the experiment, eelgrass transplant survival rate differed between Tracadie Harbour and Benoit Cove (Figs 16–17). On 24 July, Benoit Cove had a transplant mortality of 55.0% which decreased to 38.5% by 16 August. Transplant survival in Benoit Cove after 7 weeks was 15.4%. The subsites' mean survival ranged between 4.17 ± 8.3% and 24.8 ± 8.9% after 7 weeks (Fig. 18). Tracadie Harbour had a final plant survivorship of 91.6% and 6 shoot mortalities. The three subsites ranged between 83.0 ± 0.0% (subsite A) and 96.3 ± 6.4% (subsite B); subsite C had a mean survival of 89.0 ± 15.5%, indicating some variation between the frames. One of the subsite C frames had a survival of 66.5% and the rest were over 83.0%. A nested ANOVA revealed that transplant survival after 7 weeks was significantly different between Tracadie Harbour and Benoit Cove (F = 122.025, $P < 0.001$). There was also a statistical difference in survival between subsites (F = 3.659, $P = 0.024$). The Tukey HSD test revealed that there was no significant difference between the subsites from Tracadie Harbour, but subsite A and subsite B in Benoit Cove were statistically different from one another.

4.1.3 Transplant Growth

The raw data and analyses for above-ground and below-ground growth variables can be found in Appendices 3–10. Below-ground growth was evaluated from the initial and final rhizome lengths of the transplants that survived. After 7 weeks, there were no noticeable changes in rhizome length. The initial mean rhizome length for both sites was 7.1 cm. Tracadie Harbour had a final mean length of 7.2 ± 4.4 cm and Benoit Cove had a final length of 7.0 ± 3.8 cm.

For both study sites, a negative trend occurred in all above-ground growth variables (Table 4). The total number of eelgrass blades and the number of blades per shoot transplant declined at both study sites. Each shoot transplant in Benoit Cove initially had approximately 5.5 ± 1.8 blades and Tracadie Harbour had 5.0 ± 1.2 blades per shoot. The two sites' initial blade-to-shoot ratios were not significantly different ($P = 0.12$). After 7 weeks, both sites had a ratio of roughly 2.6 blades per shoot which was significantly lower than the initial ratio ($P < 0.001$). There was no significant difference in the final blade to shoot ratio between the two sites ($P = 0.78$).



Figs 16–17. Eelgrass transplant survival in Tracadie Harbour and Benoit Cove after 7 weeks. **Fig. 16.** Two transplant frames from Benoit Cove each with one transplant survival. **Fig. 17.** Tracadie Harbour frames both with 100% transplant survival.

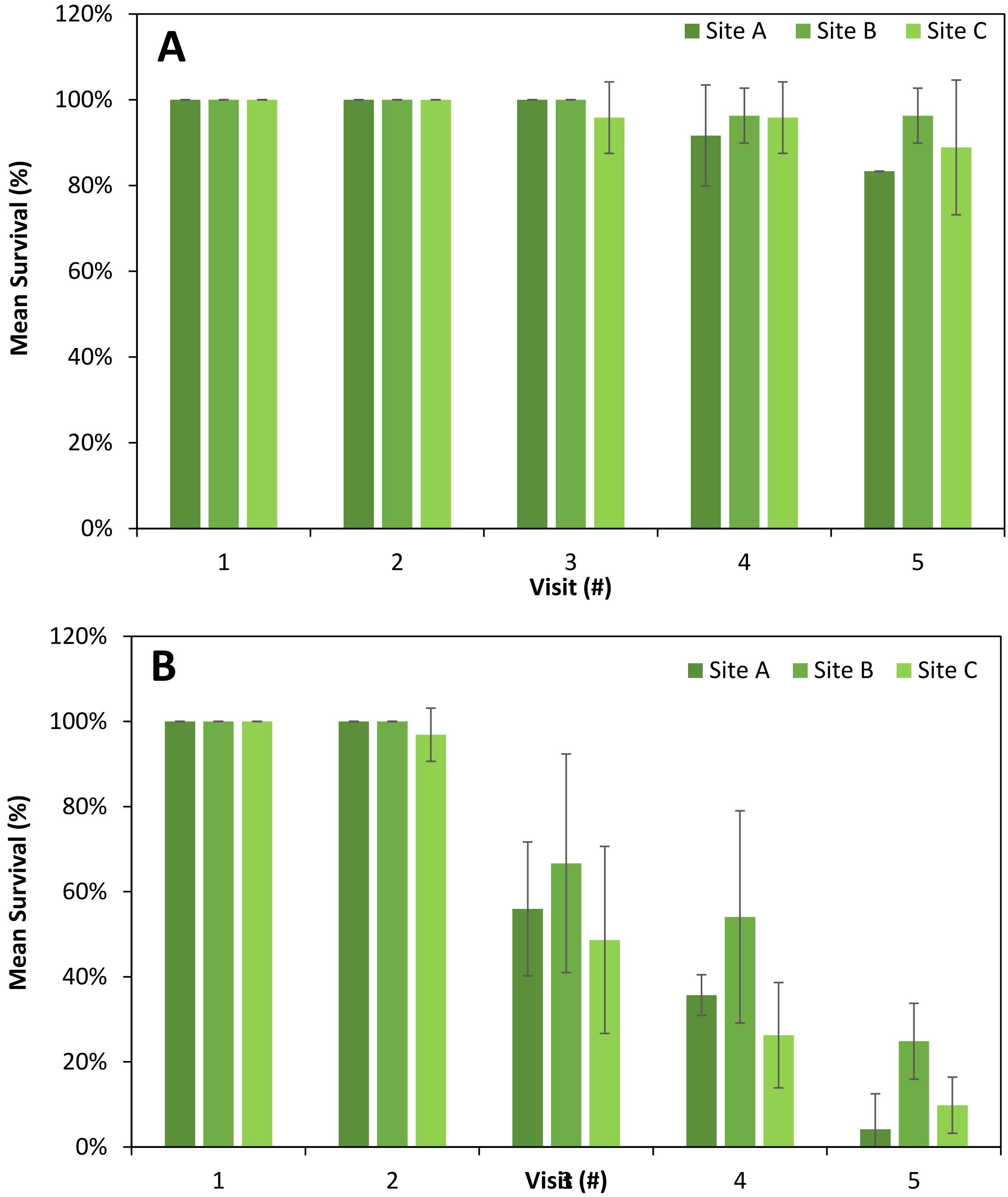


Fig. 18. Transplant survival (mean ± *s*) trends recorded per visit for subsites in (A) Tracadie Harbour and (B) Benoit Cove. Visit 1 represents the dates 3–6 July; Visit 2 = 17–19 July, Visit 3 = 24–27 July, Visit 4 = 14–16 August, and Visit 5 = 28–29 August.

Table 4. Summary of eelgrass transplant above-ground growth characteristics from Tracadie Harbour and Benoit Cove.

	Tracadie Harbour		Benoit Cove	
Plant Characteristics	Initial	Final	Initial	Final
Number of shoots	72	66	91	14
Total number of blades	388	173	579	38
Blades per shoot	5.0 ± 1.2	2.6 ± 1.2	5.4 ± 1.6	2.6 ± 0.8
Blade length (cm)	27.5 ± 8.1	20.9 ± 8.5	29.8 ± 8.3	13.2 ± 6.1

After 7 weeks, eelgrass transplants in Tracadie Harbour were longer than the transplants in Benoit Cove. An example of the differences in *Z. marina* length between the frames from both study sites can be found in Fig. 19. Benoit Cove started with a mean canopy height of 33.8 ± 3.9 cm and had a final height of 3.8 ± 5.4 cm. The initial mean canopy height in Tracadie Harbour (32.0 ± 5.0 cm) declined by roughly 7.4 cm after 7 weeks (Fig. 20). All but one frame in Tracadie Harbour had a final canopy height that ranged between 19.6 cm and 31.4 cm. In Benoit Cove, 11 of the 12 frames had a mean canopy height < 8.0 cm and four of them were recorded as 0.0 cm. The nested ANOVA for canopy height had a $P < 0.001$, which indicates that the final mean canopy height in Tracadie Harbour was significantly greater than in Benoit Cove. A Tukey HSD test determined that subsite A and B in Benoit Cove had significantly different mean canopy heights. The canopy height means of all three Tracadie Harbour subsites were not significantly different from one another.

The rate of blade length loss between the transplants that survived and before transplants went missing or died was observed (Figs 21–22). This pattern was observed for both study sites. In Benoit Cove, the transplants that died or went missing during the experiment had a similar rate of decline as the surviving transplants. On 16 August, blade length of survived transplants had declined by 13.3 ± 7.6 cm, the dead transplants decreased by 13.6 ± 10.7 cm, and the missing transplants declined by 18.4 ± 16.3 cm. In mid-August, blade length loss of the transplants that died in Tracadie Harbour followed a slower trend than the other transplants (0.35 ± 0.2 cm). The successful eelgrass shoots in Tracadie Harbour had a blade length loss of 4.0 ± 3.3 cm and the missing transplants had a similar decline of 3.4 ± 3.0 cm.

Blade width was measured for the longest leaf of each surviving transplant. The initial widths prior to transplanting were not collected. Based on the *Z. marina* samples collected from the comparison site, initial widths of the longer eelgrass blades would have ranged between 3–6 mm. There was a noticeable change in blade width size after 7 weeks, especially in Benoit Cove. The final mean blade width in Tracadie Harbour was 3.2 ± 0.7 mm and in Benoit Cove it was 2.6 ± 0.5 mm. A two-sample *t*-test revealed that blade width after 7 weeks was significantly larger in the transplants from Tracadie Harbour than Benoit Cove ($P = 0.009$).

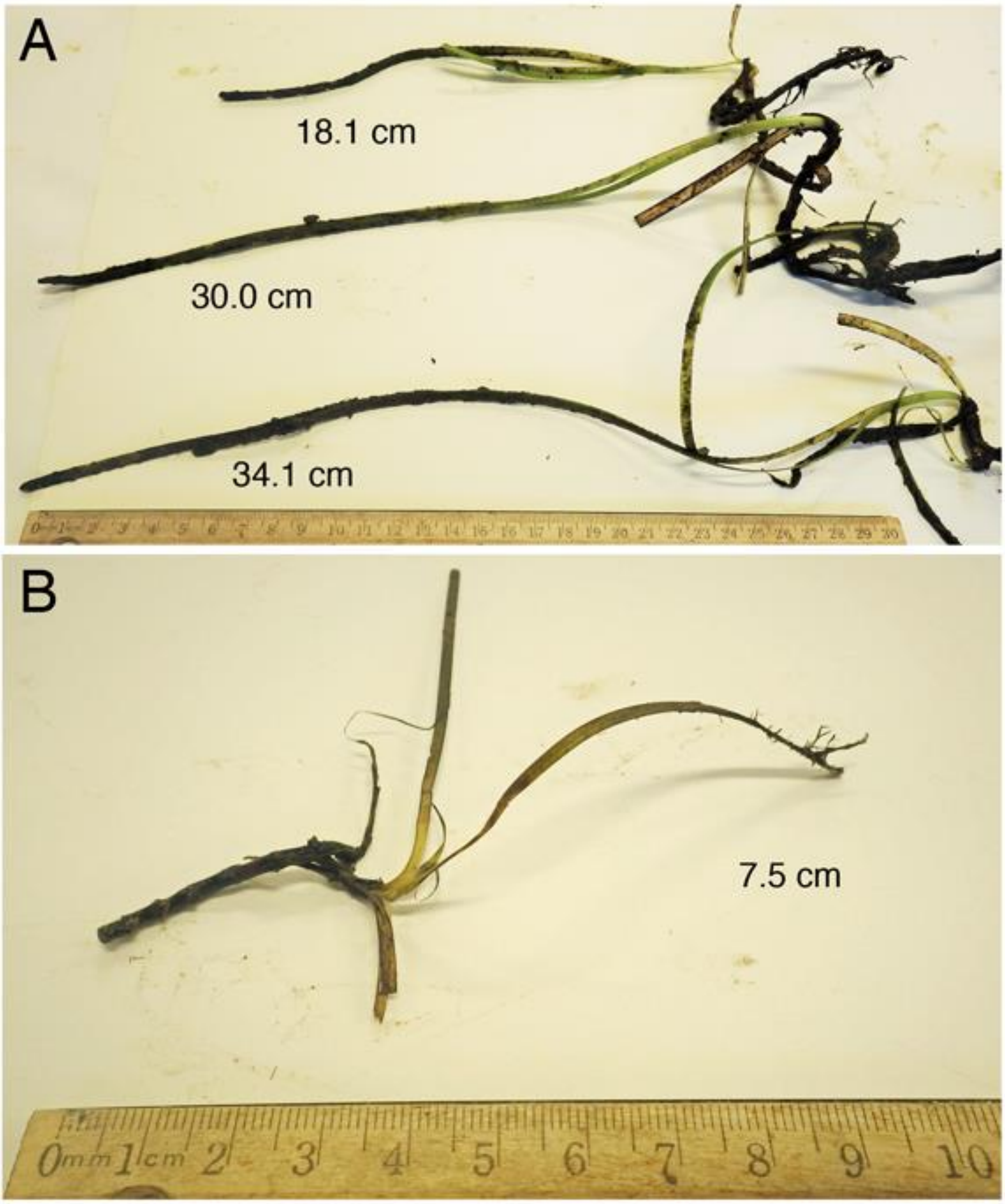


Fig. 19. Example of differences in transplant blade length after seven weeks between (A) Tracadie Harbour and (B) Benoit Cove.

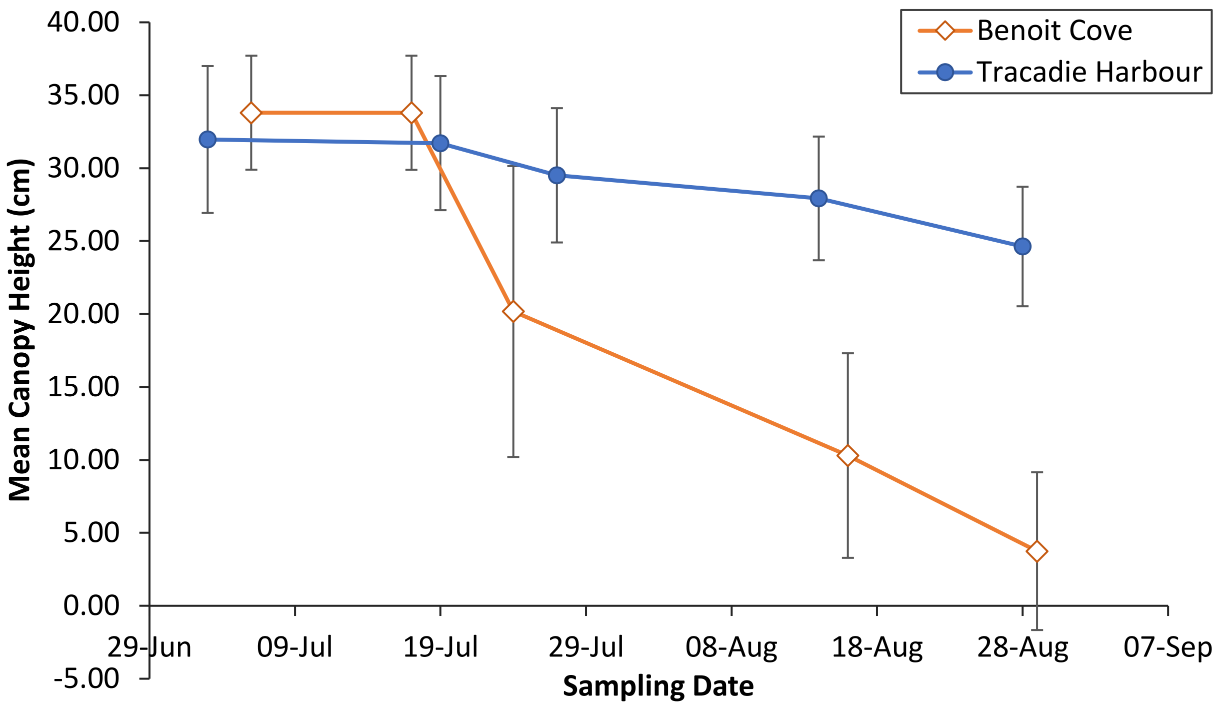
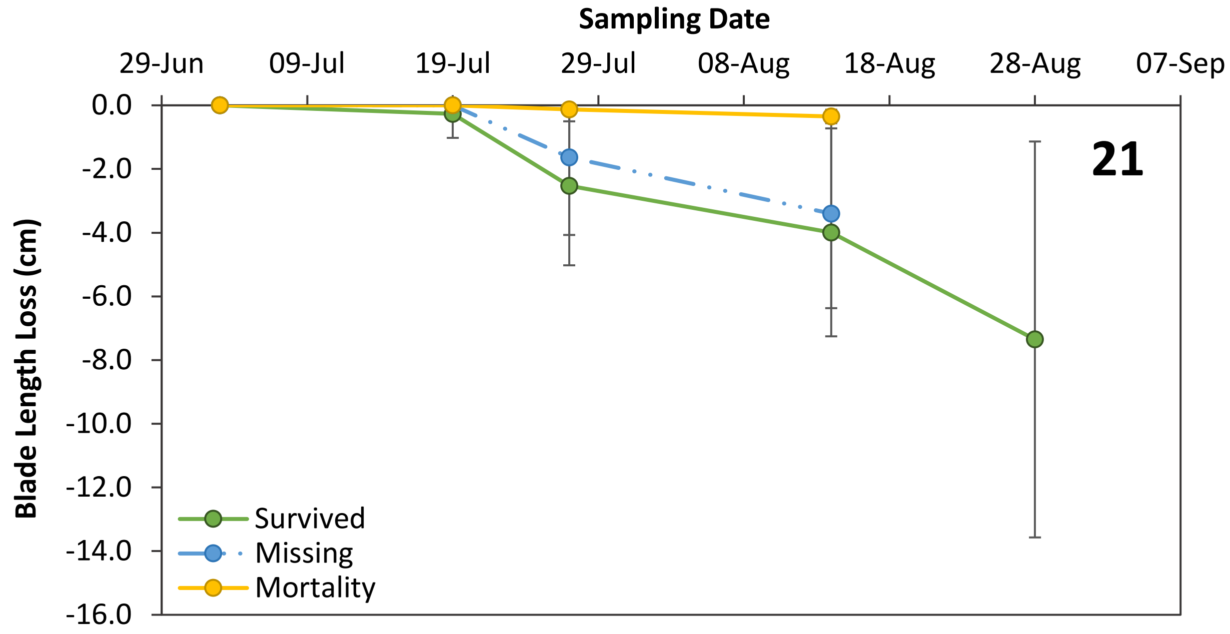
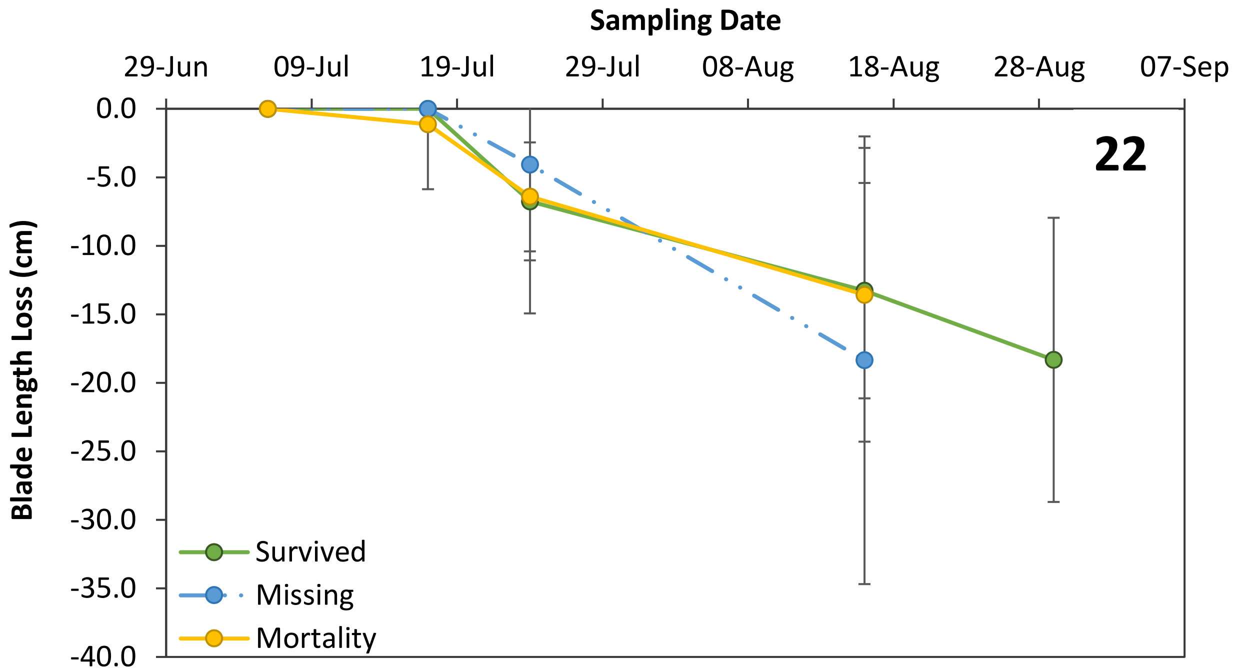


Fig. 20. Mean canopy height (± *s*) calculated on each sampling date in Tracadie Harbour and Benoit Cove.



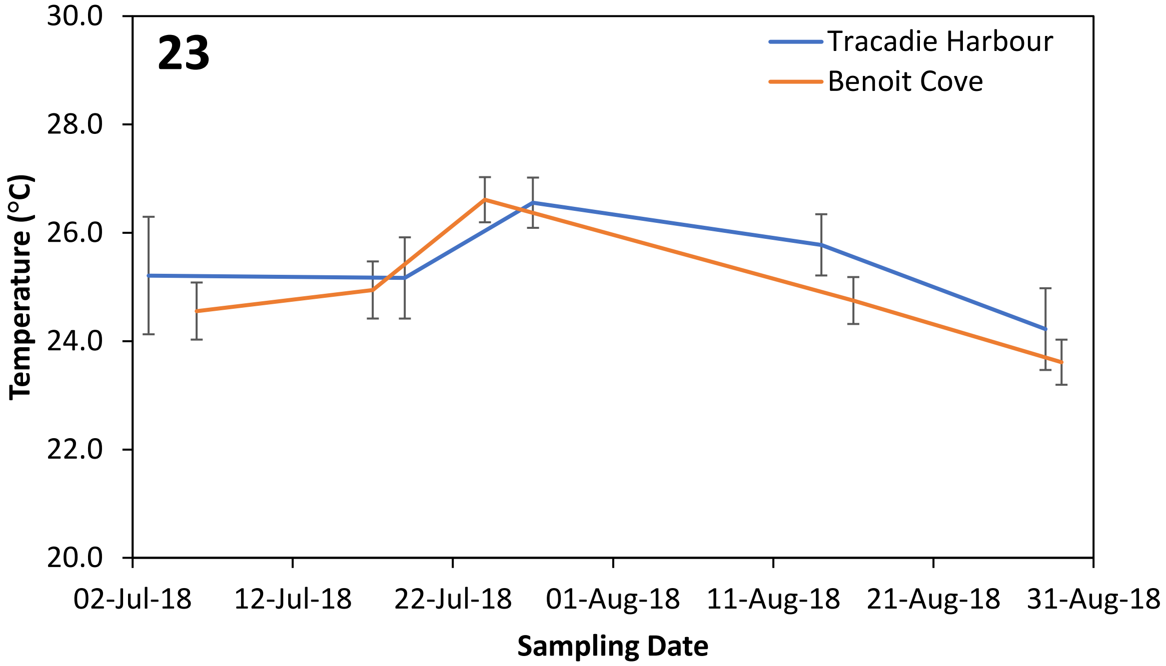


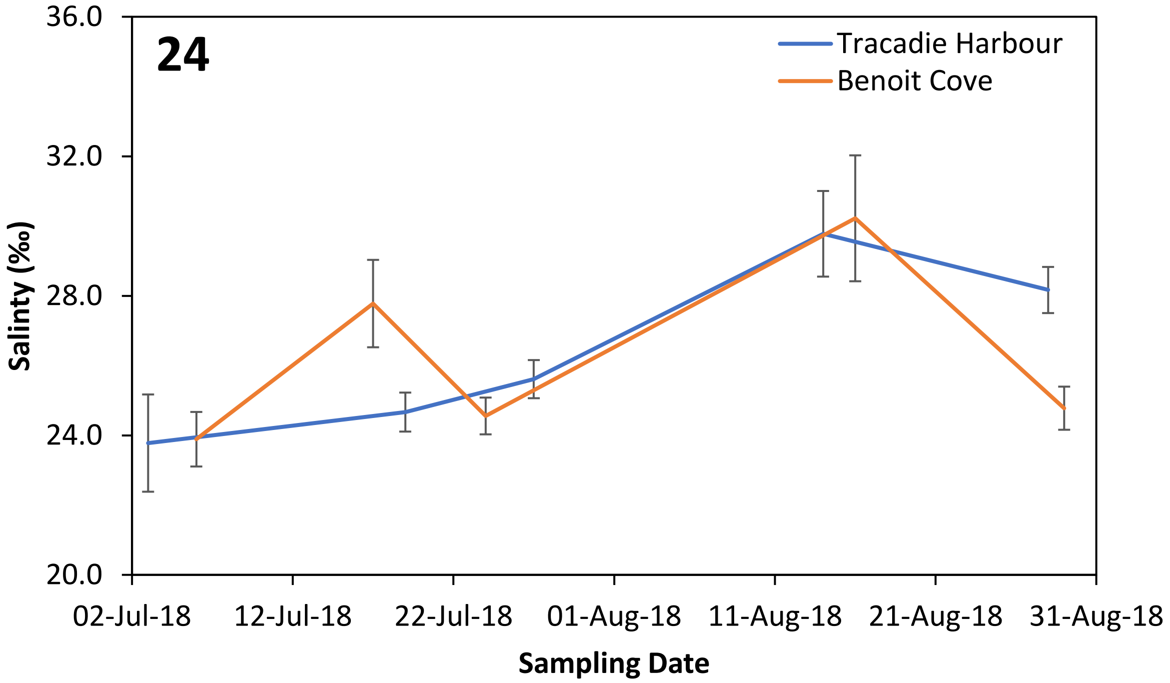
Figs 21–22. Decline in mean transplant blade length (± *s*) observed on each sampling date for 7 weeks. **Fig. 21.** Blade length trends recorded for survived (n = 66), missing (n = 5) and dead (n = 6) eelgrass transplants from Tracadie Harbour. **Fig. 22.** Blade length trends observed for survived (n = 14), missing (n = 17) and dead (n = 77) transplants from Benoit Cove.

4.2 Water Temperature and Salinity

In both study sites, water temperature and salinity were recorded nine times per visit. Tracadie Harbour and Benoit Cove had similar water temperature trends over the summer (Fig. 23). Mean water temperature in Tracadie Harbour was 25.4 ± 1.1°C and in Benoit Cove it was 24.9 ± 1.1°C. The difference in water temperature per visit between Tracadie Harbour and Benoit Cove was not statistically tested because the sites were visited on different days and measurements would not account for the daily and hourly temperature variations. Over the 7-week study, both sites had a maximum recorded water temperature of 27°C during the third week of July and a minimum of 23°C at the end of August.

Salinity was also similar between Benoit Cove and Tracadie Harbour (Fig. 24). Benoit Cove had a mean salinity of 26.2 ± 2.6‰ and Tracadie Harbour had a recorded mean of 26.4 ± 2.4‰. Both sites had a maximum salinity of 33‰ and a minimum of 22‰.





Figs 23–24. Trends of water quality for both study sites from 03 July–29 August. **Fig. 23.** Mean (± *s*) water temperature in Tracadie Harbour and Benoit Cove. **Fig. 24.** Mean (± *s*) salinity in Tracadie Harbour and Benoit Cove.

4.3 Sediment Analysis

4.3.1 Sediment Composition

The amount of sediment that had settled after one minute, one hour and 24 hours indicated the relative composition of sand, silt and clay (Fig. 25). The results showed that some of the sites differ in sediment composition. The amount of clay in all four sites was low, ranging from 0.3 ± 0.2% to 0.6 ± 0.3%. A single-factor ANOVA revealed that there was no significant difference in percent clay composition of all four sites ($F = 2.89$, $P = 0.05$). The sediment in Tracadie Harbour and Benoit Cove was composed mainly of silt. Tracadie Harbour had a mean silt composition of 34.1 ± 12.8% which was slightly higher than Benoit Cove (28.3 ± 4.4%). The mean silt percentage in Grand Étang Estuary was 14.0 ± 2.6% which was less than half the amount found in both Tracadie Harbour and Benoit Cove. Antigonish Harbour had the lowest silt composition of the four sites with a mean of 10.6 ± 1.6%. Raw data and analysis for sediment composition can be found in Appendices 12–15.

Among the four eelgrass sites, a Welch's ANOVA revealed that there was a significant difference between the mean silt compositions ($F = 56.29$, $P < 0.001$). The Games-Howell Method revealed that there was no significant difference in silt composition between Tracadie Harbour and Benoit Cove, but their means were significantly different from Antigonish Harbour and Grand Étang Estuary. The post-hoc test revealed that there was no significant difference in silt percent composition in Antigonish Harbour and Grand Étang.

The sediment in Antigonish Harbour and Grand Étang Estuary had higher concentrations of sand than Tracadie Harbour (0.8 ± 0.4%) and Benoit Cove (2.1 ± 1.1%). Antigonish Harbour had a mean sand composition of 14.9 ± 1.2% which was the highest among the four sites. The average amount of sand that settled from the Grand Étang Estuary sediment samples was 8.67 ± 0.61%. A Welch's ANOVA revealed that there was a significant difference between the sites' mean sand compositions ($F = 245.45$, $P < 0.001$). According to a Games-Howell Multiple Comparisons test, percent sand composition for all four sites was significantly different.

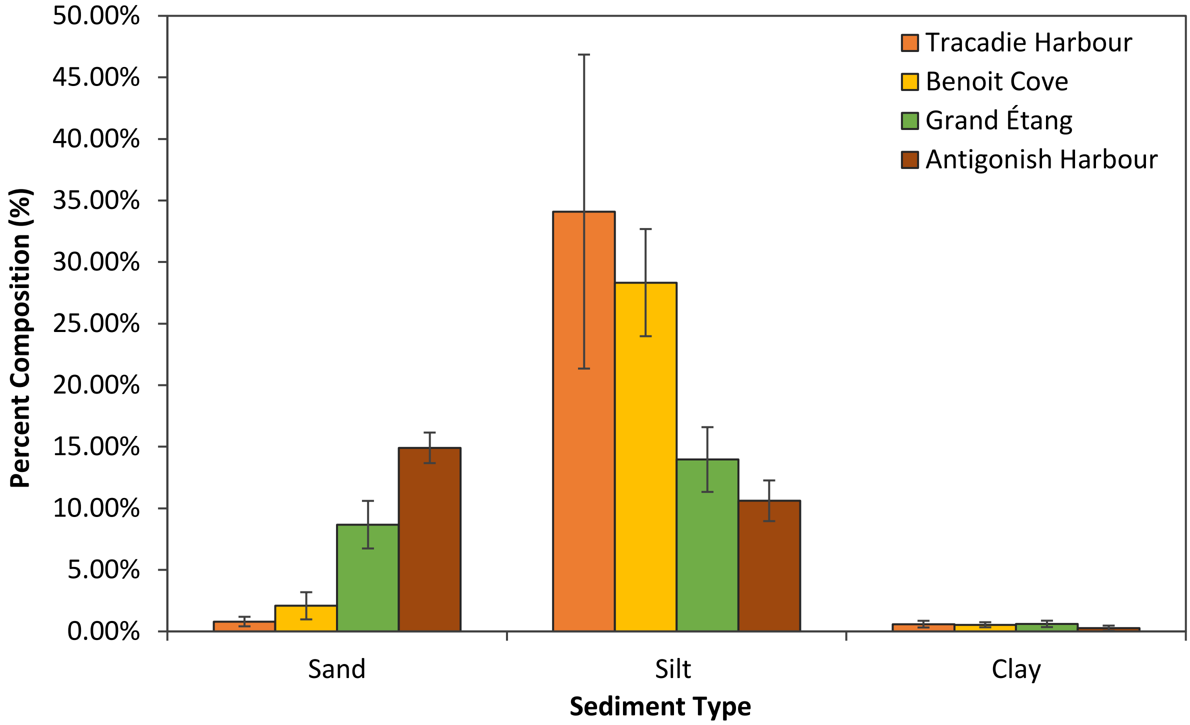


Fig. 25. Percent composition of sand, silt and clay of sediment samples from Tracadie Harbour (n = 12), Benoit Cove (n = 12), Grand Étang Estuary (n = 6), and Antigonish Harbour (n = 6). Values are represented as means ± standard deviation.

4.3.2 Organic Matter Content

Organic matter (OM) content was measured for Tracadie Harbour, Benoit Cove and Antigonish Harbour (Table 5). Organic matter for Tracadie Harbour was 33.0 ± 10.1% which was 21.1% higher than the mean in Benoit Cove (11.9 ± 5.3%). Organic content from Tracadie Harbour's original sediment sample ranged from 2.53 g to 7.58 g, and in Benoit Cove, it was between 1.92 g and 4.63 g. Antigonish Harbour had the lowest organic matter content with 1.99 ± 0.83%. A Welch's ANOVA which assumed unequal variances, showed that the mean percent organic matter of the three sites were significantly different ($F = 53.59$, $P < 0.001$). OM from the three sites were significantly different based on Games-Howell Comparison test.

Table 5. Mean dry weight, ash weight, organic matter weight and calculated mean organic matter content of 50-g sediment samples from Tracadie Harbour (n = 9), Benoit Cove (n = 9) and Antigonish Harbour (n = 9). Values are means with standard deviations.

Site	**Dry Weight (g)**	**Ash Weight (g)**	**Organic Matter (g)**	**Organic Matter (%)**
Tracadie Harbour	12.39 ± 1.65	8.17 ± 0.67	4.21 ± 1.19	32.95 ± 10.08%
Benoit Cove	24.86 ± 6.34	22.09 ± 6.57	2.77 ± 1.10	11.90 ± 5.31%
Antigonish Harbour	57.18 ± 2.98	56.053.05	1.13 ± 0.46	1.99 ± 0.83%

*Ash Weight (g) = mineral content

4.4 Species Richness

At the two sites, in addition to eelgrass, 40 different species were observed (Tables 6 and 7). These organisms consisted of algae, invertebrates and fish species. Tracadie Harbour had 92.5% of the species found which was more than Benoit Cove (52.5%). Arthropoda represented 62% of the invertebrate species observed and approximately one-third of the species were smaller crustaceans, classified as Amphipoda or Tanaidacea. Tracadie Harbour had all seven of the small crustacean species identified, but the most abundant were *Corophium volutator, Corophium* sp. and *Gammarus* sp. (Table 8). In Benoit Cove, *Corophium* sp. and *Gammarus* sp. were the only species observed and they were present in low abundance.

4.4.1 Marine Algae

The majority of identified algal species were green algae (n = 7). Fewer species of red algae (n = 4) and brown algae (n = 3) were present. For both sites, the red alga *Polysiphonia subtilissima* was the most dominant seaweed. In Benoit Cove, *P. subtilissima* was found along the first 5 m of shoreline and on top of the eelgrass transplant frames. Some of the *P. subtilissima* mats included tangled fragments of the green algae *Ulva intestinalis* and *Ulva prolifera*. The frames were also accompanied by unanchored *Ulva lactuca*. In low abundance, small filaments of the red alga *Stylonema alsidii* were found tangled in the algal mats.

In Tracadie Harbour, large floating algal mats were often observed on the water surface and throughout the eelgrass bed. These mats were primarily composed of *P. subtilissima* and large fragments of other species including *U. prolifera*, *Cladophora* sp., *Chaetomorpha picquotiana* and *Ulothrix flacca*. In late August, there were a couple of branched fragments of red alga *Gracilaria* sp. sitting on top of the frames. In moderately-high abundance, a red and white striped alga called *Ceramium diaphanum* was found growing on the eelgrass blades and entangled with *P. subtilissima*.

Table 6. Relative abundance of the algal, invertebrate and fish species found in Tracadie Harbour from 3 July to 28 August.

Taxon	Abundance Level
Marine Algae	
Ulva lactuca (Sea lettuce)	Low
Ulva intestinalis (Gutweed)	Moderate
Ulva prolifera	Moderate-high
Cladophora sp.	High
Chaetomorpha picquotiana	High
Ulothrix flacca	Moderate
Gomontia polyrhiza	Low-moderate
Gracilaria sp.	Low-moderate
Polysiphonia subtilissima	High
Ceramium diaphanum	Moderate-high
Pylaiella littoralis (Angel hair)	Low
Fucus vesiculosus (Bladder wrack)	Low
Invertebrates	
Carcinus maenas (European green crab)	High
Rhithropanopeus harrisii (Harris mud crab)	Low-moderate
Pagurus sp. (Hermit crab)	Low
Crangon septemspinosa (Sand shrimp)	Moderate
Palaemonetes vulgaris (Grass shrimp)	Low-moderate
Idotea balthica (Baltic isopod)	Moderate-high
Littorina littorea (Periwinkle snail)	Low
Tritia obsoleta (Eastern mud snail)	Low
Crassostrea virginica (Eastern oyster)	Moderate
Mercenaria mercenaria (Northern quahog)	Low-moderate
Mytilus edulis (Blue mussel)	Low
Hediste diversicolor (Ragworm)	Low-moderate
Capitella sp. (Polychaete)	Moderate
Chordates	
Anguilla rostrata (American eel)	Low
Gasterosteus aculeatus (Three-spined stickleback)	High
Fundulus heteroclitus (Mummichog)	High
Fundulus diaphanous (Killifish)	High
Menidia menidia (Atlantic silverside)	High

Table 7. Relative abundance of algae, invertebrate and fish species found in Benoit Cove from 6 July to 29 August.

Taxon	Abundance Level
Marine Algae	
Ulva lactuca (Sea lettuce)	Moderate-high
Ulva intestinalis (Gutweed)	Low-moderate
Ulva prolifera	Moderate
Cladophora sp.	Low-moderate
Polysiphonia subtilissima	High
Stylonema alsidii	Low
Fucus vesiculosus (Bladder wrack)	Moderate
Chorda filum (Sea lace)	Low
Invertebrates	
Carcinus maenas (European green crab)	Moderate
Rhithropanopeus harrisii (Harris mud crab)	Moderate
Crangon septemspinosa (Sand shrimp)	Low
Littorina littorea (Periwinkle snail)	Low-moderate
Tritia obsoleta (Eastern mud snail)	High
Crassostrea virginica (Eastern oyster)	High
Crepidula fornicata (Common slipper limpet)	High
Mercenaria mercenaria (Northern quahog)	Low
Hediste diversicolor (Ragworm)	Moderate-high
Capitella sp. (Polychaete)	Moderate-high
Chordates	
Gasterosteus aculeatus (Three-spined stickleback)	Low

Table 8. Estimated abundance of identified species from the orders Amphipoda and Tanaidacea in Benoit Cove and Tracadie Harbour from 17 July to 29 August.

Species	Tracadie Harbour	Benoit Cove
Amphipoda		
Corophium volutator	High	N.P.
Corophium sp.	High	Low
Gammarus mucronatus	Moderate	N.P.
Gammarus finmarchicus	Low	N.P.
Gammarus sp.	High	Low
Tanaidacea		
Chondrochelia sp.	Moderate	N.P.
Leptochelia sp.	Low-moderate	N.P.

N.P. indicates that the species were not present

Both sites had two algal species that were not associated with the eelgrass bed or transplants. A green alga, *Gomontia polyrhiza*, grew in fragments in shells of the Eastern oyster (*Crassostrea virginica*) from Tracadie Harbour. Along the shoreline in Benoit Cove, there were a few lengthy strands of *Chorda filum* (Sea lace). The brown alga *Fucus vesiculosus* (Bladderwrack) was rare in Tracadie Harbour. In Benoit Cove, the thalli of *F. vesiculosus* were small but moderately abundant attached to rocks along the shoreline in the mid- to upper intertidal zone.

4.4.2 Epiphytic and Drift Algal Cover

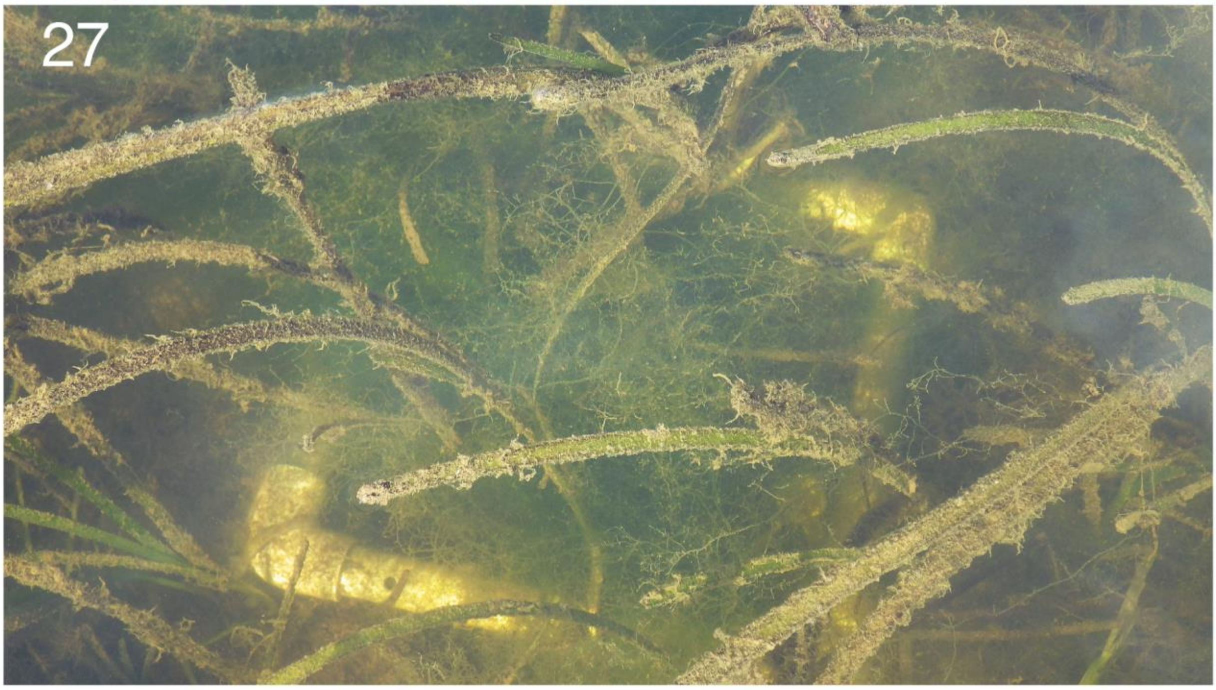
During the transplanting process, eelgrass blades were a healthy green colour with virtually no epiphytic algae. In both sites, algal loading increased during the experiment, but cover was higher in the comparison site (Table 9). In Tracadie Harbour, *P. subtilissima* covered the eelgrass as a drift alga and epiphytic alga, whereas cover in Benoit Cove was strictly drift alga. On 19 July, epiphyte cover in Tracadie Harbour was low (< 25%). One week later, subsites B and C had moderate epiphyte cover (25–50%), and 50–75% of transplant frames and blades from subsite A were covered. Floating mats of green algae, composed of *U. prolifera* and *Cladophora* sp. were also distributed across the eelgrass bed in larger blooms than before (Figs 26–27). On 14 August, there were estimates of 75–100% cover. The thick epiphyte layers of *P. subtilissima* supported large blooms of blue-green cyanobacteria (*Calothrix* sp.), green tuffs of *Cladophora* sp. and high concentrations of sediment particles. Due to the intense progression of epiphytic algae in Tracadie Harbour (Fig. 28), frames were removed on 29 August. Epiphyte cover throughout the eelgrass bed decreased by 4 September and large floating mats consisting of dead eelgrass blades, *P. subtilissima*, *U. prolifera*, *U. intestinalis* and *Cladophora* sp. appeared at the surface (personal observation).

In Benoit Cove, percent cover remained absent or low for the first 3 weeks. From 16 to 28 August, subsite C maintained a percent drift algal cover of 25–50%. After seven weeks, percent algal cover was < 75%. There was one frame with 75–100% cover (Fig. 29), but each subsite estimate was calculated based on the mean percent cover in all four frames. The second most abundant drift alga was *U. lactuca* (Fig. 30). Although the distribution of drift algae was high around frames and the first 3 m of shoreline, algae were limited throughout the rest of the site.

Table 9. Estimated percent cover of epiphytic and drift algae on eelgrass transplants and within frames for each subsite in Tracadie Harbour and Benoit Cove.

	Tracadie Harbour			**Benoit Cove**		
Visit Dates	**Subsite A**	**Subsite B**	**Subsite C**	**Subsite A**	**Subsite B**	**Subsite C**
Jul 3 – 6	Absent	Absent	Absent	Absent	Absent	Absent
Jul 17 – 19	Low	Low	Low	Absent	Low	Absent
Jul 24 – 27	High	Moderate	Moderate	Low	Low	Low
Aug 14 – 16	Intense	Intense	Moderate	Moderate	High	Moderate
Aug 24 – 25	Intense	Intense	Intense	High	High	Moderate
Aug 28 – 29	High	High	High	High	High	Moderate

Absent = 0%, Low = 1–25%, Moderate = 25–50%, High = 50–75%, Intense = 75–100%



Figs 26–27. Increase in epiphytic and drift algal species in subsite B of Tracadie Harbour on 27 July 2018. **Fig. 26.** Epiphyte cover of *Polysiphonia subtilissima* and green floating mats composed of *Ulva prolifera* and *Cladophora* sp. **Fig. 27.** Change in colour of eelgrass leaf blades with increased epiphyte loading.

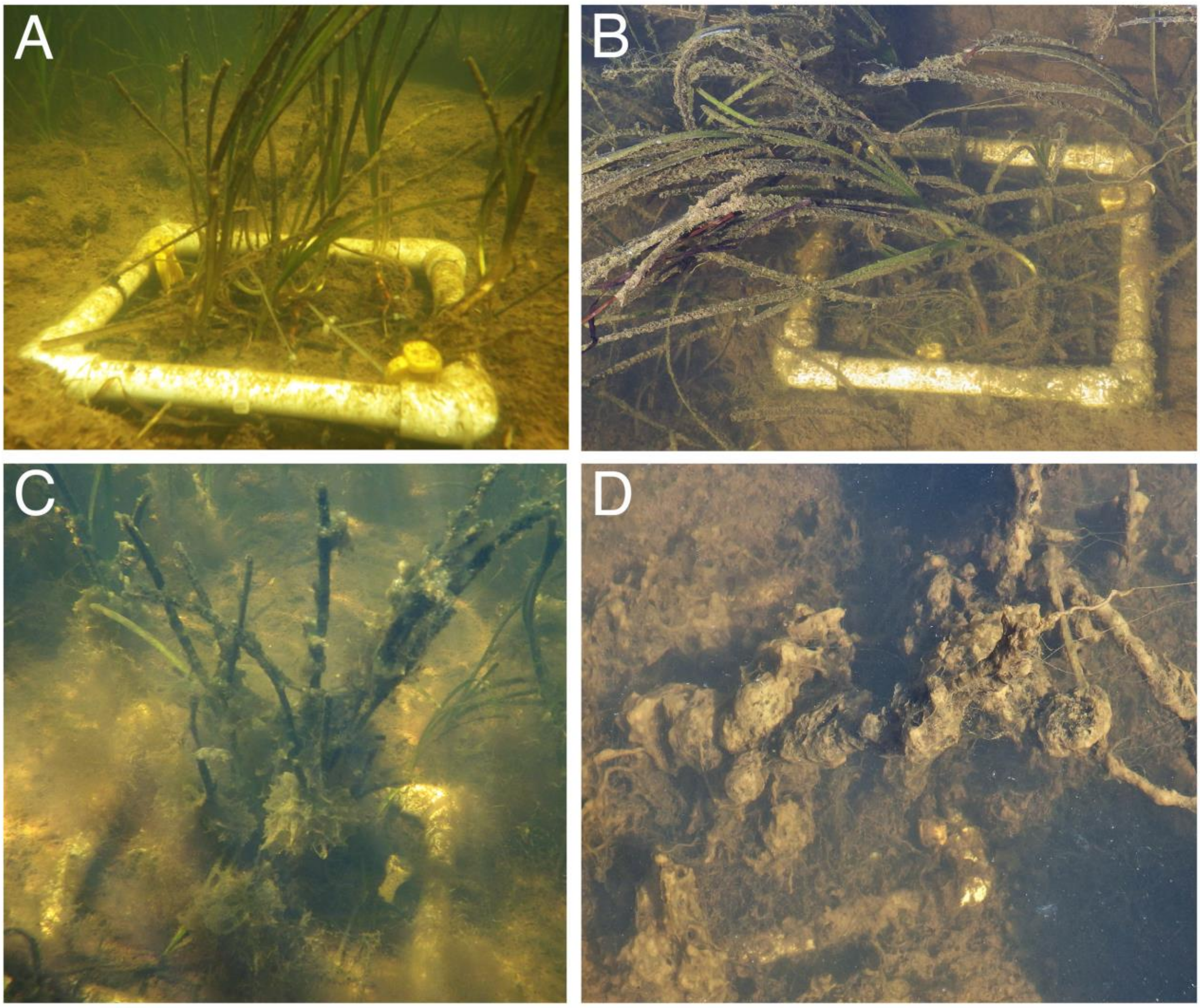


Fig. 28. Changes in algal epiphyte loading on eelgrass transplants in Tracadie Harbour on (A) 6 July (Absent), (B) 19 July (Low), (C) 27 July (Moderate – High), and (D) 24 August (Intense).

Figs 29–30. Drift macroalgal species that contributed to increased epiphytic loading on the transplant frames in Benoit Cove. **Fig. 29.** Frame "blue" dominated by drift of the red alga *Polysipnonia subtilissima*. **Fig. 30.** The second most abundant drift algal species, *Ulva lactuca*.

4.4.3 European Green Crab

The invasive European green crab (*Carcinus maenas*) was present in both sites but at different frequencies (Table 10). A two-sample *t*-test revealed that the mean count of green crab individuals per visit was significantly greater in Tracadie Harbour than in Benoit Cove ($P = 0.0086$). In Tracadie Harbour, between 24 and 62 green crabs were observed per visit. Subsite C had the highest green crab count with a mean of 19.4 ± 12.8 crabs observed which was more than double the mean counts of the other subsites. In Tracadie Harbour, green crabs were also found near the transplant frames but did not appear to hide or use the frames as often as the green crabs in Benoit Cove.

In Benoit Cove, between four and eight green crabs were counted per visit. Subsite B had a mean count of 2.8 ± 1.1 green crabs per visit, which was the highest of the three subsites. Most of the green crabs were found beside or in the transplant frames along with several crab carapaces, legs, and empty shells. When macroalgal loading was high, the green crabs were often found hiding under the algal mats that were covering the frames.

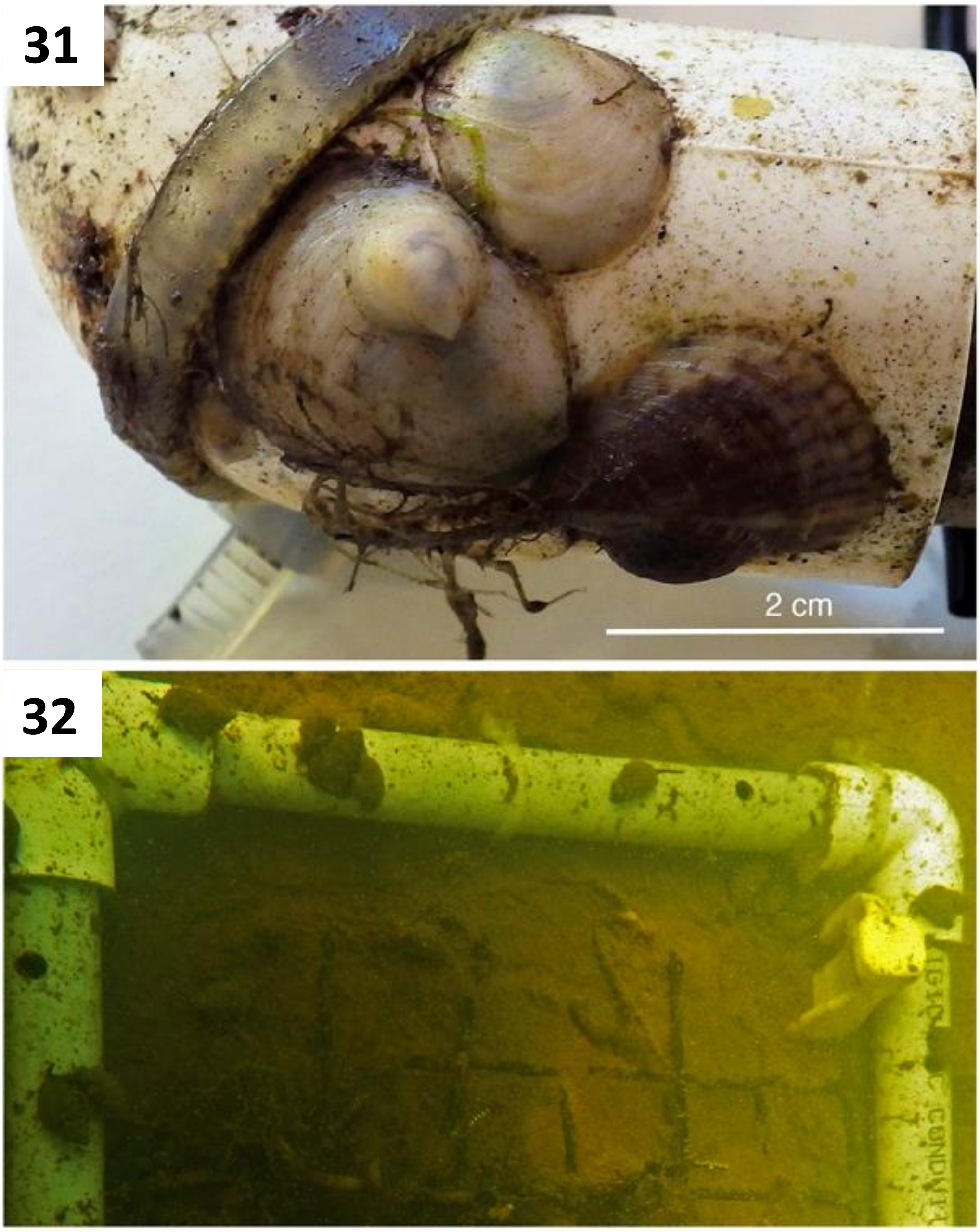
Table 10. European green crab count from each observation date per subsite in Tracadie Harbour and Benoit Cove.

	Tracadie Harbour				**Benoit Cove**			
Date	**Subsite A**	**Subsite B**	**Subsite C**	**Total**	**Subsite A**	**Subsite B**	**Subsite C**	**Total**
3–6 Jul	10	7	7	24	1	2	1	4
17–19 Jul	7	4	10	21	1	2	2	5
24–27 Jul	6	9	14	29	1	4	3	8
14–16 Aug	5	7	30	42	2	4	1	7
28–29 Aug	8	18	36	62	2	2	2	6
Mean ± *s*	7.2 ± 1.9	9.0 ± 5.3	19.4 ± 12.8	35.6 ± 16.8	1.4 ± 0.6	2.8 ± 1.1	1.8 ± 0.8	6 ± 1.6

4.4.4 Molluscs

In Benoit Cove, 38.5% of the animal species found were molluscs and they were all highly abundant. In order of most to least abundant, the species were: Eastern mud snail (*Tritia obsoleta*), Common slipper limpet (*Crepidula fornicata*), Eastern oyster (*Crassostrea virginica*), Periwinkle snail (*Littorina littorea*) and Northern quahog (*Mercenaria mercenaria*). The blue mussel (*Mytilus edulis*) was the only absent mollusc in Benoit Cove. Tracadie Harbour also had five species but they were all in lower abundances than in Benoit Cove.

In Benoit Cove, slipper limpets and mud snails were attached to the PVC frames in high abundance (Figs 31–32). Neither species was attached to the Tracadie Harbour frames. The frames were dominated by *T. obsoleta* for the first month before decreasing in frequency. Common slipper limpets were identified in August and after three weeks, the PVC pipe had become covered. A total of 213 slipper limpets were found attached to the eelgrass frames and 48.8% of them came from subsite C. The limpets ranged from roughly 4–25 mm long, but most were 20 mm. *C. fornicata* was the only animal from Benoit Cove that was not found in Tracadie Harbour.



Figs 31–32. Transplant frames in Benoit Cove serving as a substratum for several species. **Fig. 31.** Common slipper limpets (*Crepidula fornicata*) attached to frames on 29 August in high abundance. **Fig. 32.** Coverage of Eastern mud snails (*Tritia obsoleta*) on the frames.

4.4.5 Annelids and Fish

Similar to the amphipod identification, the number of annelid and fish species may represent a small fraction of the individuals actually present. In both sites, two bright red annelid species were observed on the transplant frames and algal mats. These species were identified as the polychaete (*Capitella* sp.) and the ragworm (*Hediste diversicolor*). Ragworms, a highly mobile annelid, were most commonly found on the arms of the frames during the transplant removal process. Polychaetes were wrapped around the eelgrass blades and macroalgae. In Benoit Cove, *H. diversicolor* and *Capitella* sp. had a moderately high abundance whereas in Tracadie Harbour, the species were less frequent.

Fish species included the Three-spined stickleback (*Gasterosteus aculeatus*), Mummichog (*Fundulus heteroclitus*), Killifish (*Fundulus diaphanous*), Atlantic silverside (*Menidia menidia*) and the American eel (*Anguilla rostrata*). All five species were found in Tracadie Harbour and only one was observed in Benoit Cove (*G. aculeatus*). In Benoit Cove, there were a few three-spined stickleback individuals observed, but not in large schools. Estimated population size for each fish species was not determined, but the frequency of large schools indicated that the chordates were one of most abundant macro-animal groups in Tracadie Harbour. The abundance of fish species appeared to have increased throughout the summer experiment. Throughout the seven-week experiment, only two individuals of *Anguilla rostrate* were observed in the entire estuary. The rest of the species were found in moderately high to high abundance in Tracadie Harbour.

5 Discussion

5.1 Eelgrass Transplant

5.1.1 Transplant Survival in Tracadie Harbour

To my knowledge, this study is the first eelgrass transplant experiment in the southern Gulf of St. Lawrence of Canada. The transplant results from this study are consistent with my initial hypothesis. As expected, the comparison site out-performed the test site in terms of eelgrass survival. Tracadie Harbour had transplant survival of approximately 91.6% whereas only 15.4% of transplants in Benoit Cove survived the full 7 weeks. Within the first month, survival success in Tracadie Harbour followed a similar trend to other transplant experiments that used frames (Short et al. 2002*b*). The initial TERFS experiment by Short et al. (2002*b*) used a donor bed that was located within the comparison site and the one-month survival was highly successful. Mean survival results for the subsites in Tracadie Harbour ranged from 84.7 ± 2.8% to 95.7 ± 5.4% which was slightly higher than the experiment by Short et al. (2002*b*). There was no significant difference in transplant survival between the three subsites, which suggests that the site conditions were suitable for transplants.

5.1.2 Regime Shift in Benoit Cove

Results from the eelgrass transplant experiment and sediment composition assessment suggest that the decrease in survival and growth in Benoit Cove was caused by increased resuspension of fine sediment and resulting turbidity due to lack of vegetation. It can be difficult to grow eelgrass transplants in areas with fine-grained sediment and limited vegetation because sediment turbidity disrupts water clarity which reduces light transmission (Leschen et al. 2010). At both study sites, fine silts formed >28% of sediment composition. Areas that have high silt composition will attenuate more light during sediment resuspension than sand-dominated sites because sand settles faster than silt (Herrera-Silveira et al. 2000). Since silt is lighter than sand, it is more susceptible to being resuspended into the water column, disrupting water quality and clarity. There were no quantitative measurements for water clarity but on numerous occasions, I could not collect data in Benoit Cove because of poor visibility.

Density of seagrass meadows can substantially influence the severity of sediment resuspension (Hansen and Reidenbach 2013). In dense *Z. marina* habitats, most of the current is deflected over or around the eelgrass. This prevents resuspension of sediment particles, protecting the shoots within that patch (Lawson et al. 2012). The Tracadie Harbour frames were located in bare patches surrounded by the extensive rooting system of the wild eelgrass populations which stabilizes the soft, fine-grained sediment and prevents resuspension. In areas with high flow and low-density of *Z. marina*, the flow moves predominantly through the bed, causing sediment resuspension and turbidity (Lawson et al. 2012). The eelgrass transplants were not well protected in Benoit Cove because they were the only stable source of vegetation. In sparse eelgrass habitats, stress abrasion measured from sediment particle suspension is almost equivalent to that of unvegetated regions (Luhar et al. 2008). Increased sediment turbidity also causes eelgrass to reduce its carbon reserves and begin losing biomass, growth, shoot density, and eventually to die-off (Vandermeulen et al. 2012).

In Benoit Cove, transplant survival was significantly different between subsite A (4.2 ± 8.3%) and B (24.9 ± 8.9%). The results from subsite A were likely caused by the same factors mentioned above, but the intensity of these factors was amplified because of differences in depth. The frames in subsite A were transplanted closer to shore, at a shallower depth. At low tide, the frames in subsite A were at ≤ 0.5 m, whereas the frames in subsites B and C were located at ≥ 1 m. In shallow coastal habitats, hydrodynamic exposure (i.e. to wave and current velocity) increases the frequency of sediment resuspension and turbidity which contributes significant stress to eelgrass shoots (Carr et al. 2010; Vandermeulen et al. 2012). In addition, the estimated percent algal cover in subsites A and B was 50–75% towards the end of August. As previously mentioned, eelgrass mortality and reduced growth are common responses to increased loading of algae because they increase light attenuation which impacts the photosynthetic process of *Z. marina* (Vandermeulen 2005). Although both subsites had a high estimated algal cover, it appears that subsite A was under increased stress because of the intensity of sediment turbidity at a shallower depth. When there is an increase in the depth limits for eelgrass, turbidity and algal accumulations decrease (Moore and Short 2006). If the frames in subsite A had been placed at a slightly greater depth, the transplants may have had less exposure to turbidity and shading by drift algae, resulting in increased survival.

It is possible that increased sediment resuspension and turbidity in Benoit Cove impeded transplant survival by reducing light availability (indirect) and covering or burying the eelgrass blades (direct). This trend would also suggest that Benoit Cove has entered a new stable state at which it is unlikely to return to its previous condition. Sediment resuspension and hydrodynamic conditions have caused local regime shifts in habitats that once supported a dense eelgrass bed, and complicate eelgrass recovery both naturally and through restoration efforts (Moksnes et al. 2018). If this hypothesis is true, this could also explain Benoit Cove's rapid loss of eelgrass from 2002 – 2013 following the disturbance of a dense European green crab population. Garbary et al. (2014) showed that the green crabs initiated the decline of eelgrass in 2002, but for the next 11 years there were significant declines in green crab density. In Benoit Cove, shoot density may have fallen below a critical threshold, where sediment resuspension caused a rapid decrease in eelgrass density that prevented recovery (Fig. 33; Moksnes et al. 2018).

5.1.3 Transplanting Season

A limitation of my study was the season when eelgrass shoots were transplanted. Transplanting eelgrass during the summer is not recommended because there is a higher chance of large-scale mortality (CCCCEP 2011). Additional physiological stress is put on plants during the summer because of increasing water temperature and decreasing light levels caused by epiphytic and drift algae (Park and Lee 2007; CCCCEP 2011). Park and Lee (2007) used three transplant techniques (including TERFS) and found that regardless of method, all eelgrass shoots planted during summer had died-off due to higher water temperatures. It is unlikely that water temperature was responsible for the die-off of transplants in Benoit Cove because Tracadie Harbour was highly successful and both sites had similar temperatures throughout the summer. This result also suggests that the eelgrass bed in Tracadie Harbour had a relatively high resistance to temperature stress which is important in the face of climate change (Ehlers et al. 2008).

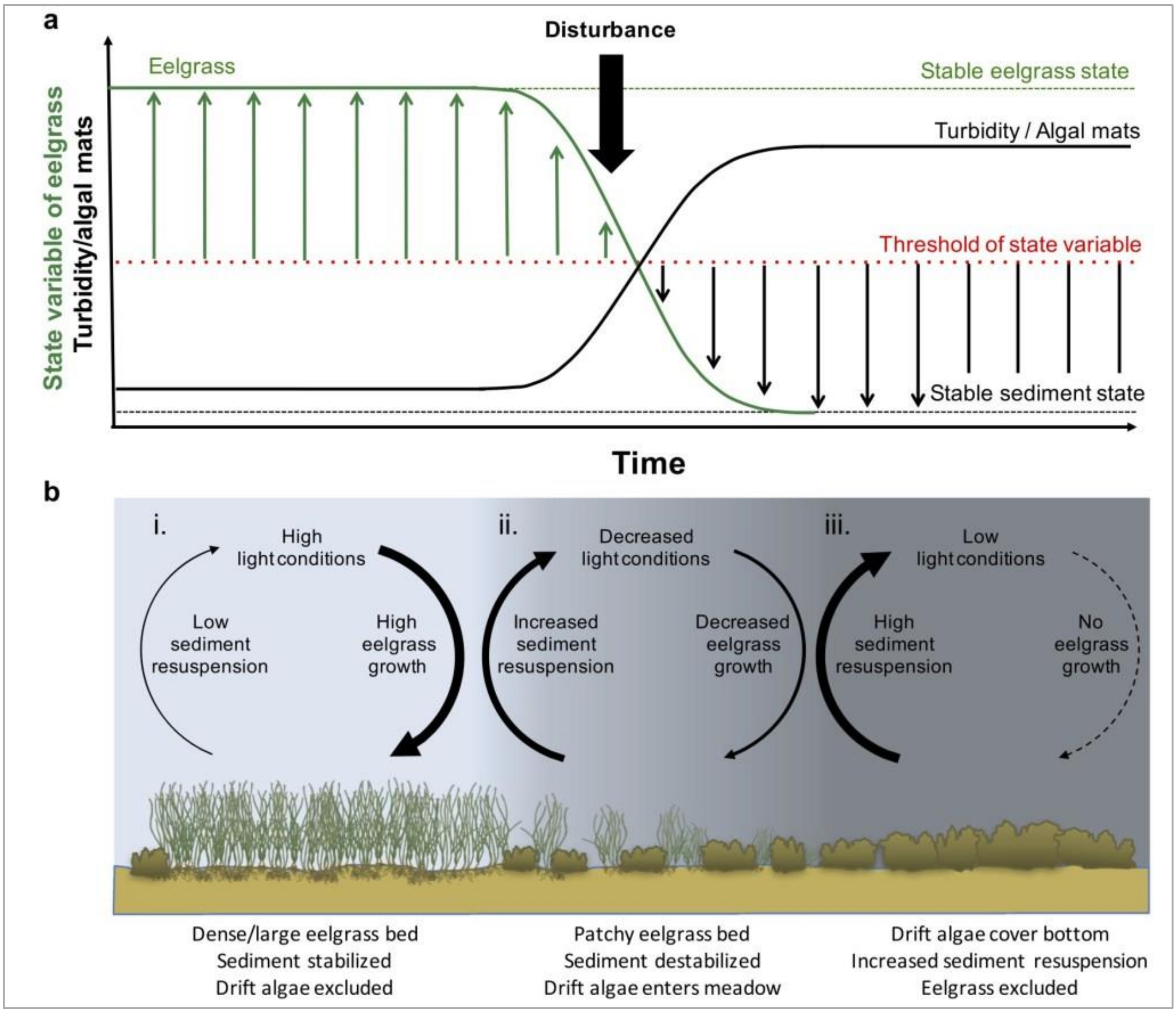


Fig. 33. Illustration and conceptual model showing how sediment resuspension and turbidity, and algal mats can cause a regime shift once an eelgrass bed decreases below its critical threshold density over time (figure from Moksnes et al. 2018).

Genetic variation does exist among seagrass meadows, indicating that eelgrass beds could be less resilient to environmental conditions, such as increasing temperatures, than nearby eelgrass beds (Ehlers et al. 2008; Unsworth et al. 2015). The eelgrass transplants used in both sites came from the same donor bed which was also located in the comparison site. Results suggest that transplants in Benoit Cove would have been more tolerant to warmer water temperatures, and that the rapid mortality was caused by environmental pressures other than temperature.

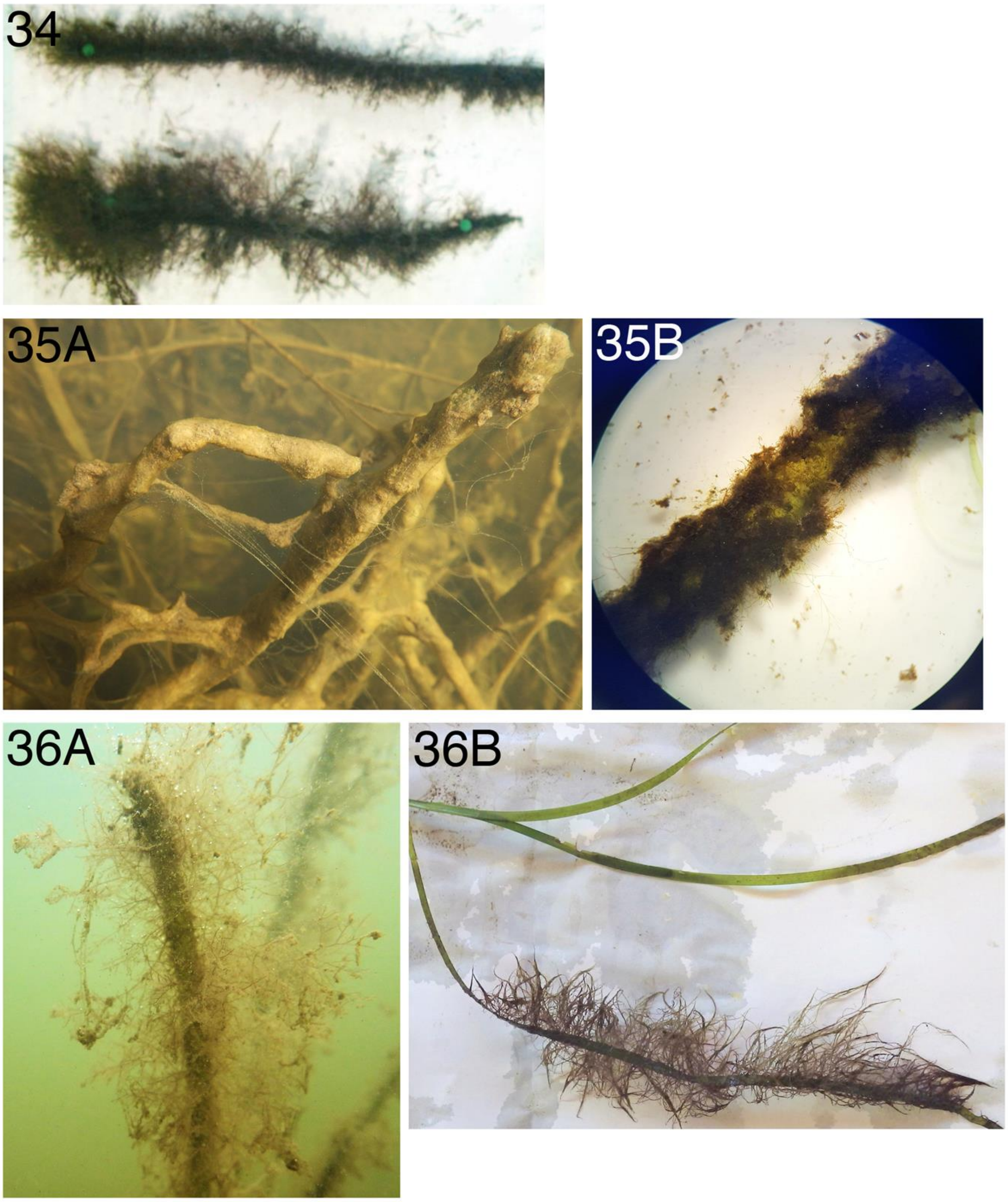
Since production of *Z. marina* is seasonally dependent, this could explain why there was no positive growth at either study site. Evaluating transplant growth during summer is not advised because eelgrass undergoes heat rigor above 20°C and therefore stops growing (Burkholder and Doheny 1968). In Nova Scotia, coastal water temperatures during summer can reach well over 20°C; therefore, the absence of eelgrass growth was anticipated. In Tracadie Harbour, the mean water temperature was 25.4 ± 1.1°C and in Benoit Cove it was 24.9 ± 1.1°C. Although eelgrass production is compromised after 20°C, plants can still tolerate temperatures between 20°C and 30°C. It was expected that transplant growth would remain relatively consistent in both sites, but this was only partially true for Tracadie Harbour. Tracadie Harbour had a final mean canopy height of 24.6 ± 4.1 cm which was approximately 20.9 cm taller than Benoit Cove (3.8 ± 5.4 cm). It is important to recognize that canopy height results from Benoit Cove were also influenced by the massive die-off of transplants. Between 25°C and 30°C, eelgrass individuals may lose shoot weight as a response to stress (Touchette et al. 2003; Moreno-Marín et al. 2018). After this process, eelgrass individuals will have shorter and fewer leaves. This trend occurred in both study sites.

Both sites also had maximum water temperatures around 26°C. At 25°C and above, eelgrass blades are more susceptible to intense wasting disease (Kaldy 2014). It did not appear that wasting disease played a significant role in survival and growth of eelgrass shoots because the eelgrass leaves in Benoit Cove were not dark green or black in colour. In Tracadie Harbour, some of the longer leaves began turning black or dark green, but it was often difficult to tell whether that was a result from extensive shading by algal cover or Labyrinthula. The rhizomes, sheath and base of the blades were still healthy, suggesting that it was the epiphytic algae causing the change in colour and shedding of blades.

5.1.4 Influence of Epiphytic Algae and Drift Algae

At both sites, marine algae represented over 30% of identified species. Benoit Cove was occupied predominantly by free-floating algae, whereas eelgrass from the comparison site was covered in epiphytic algae and was surrounded by abundant drift algae. At both sites, the dominant alga was *Polysiphonia subtilissima*, which is not uncommon in eelgrass habitats along the Atlantic coast of North America (Figs 34–36). In eastern New Brunswick, *P. subtilissima* and blue-green algae were frequently attached to eelgrass blades (McCullough et al. 2005). A report by the Cape Cod Cooperative Extension Marine Program (2011) also found extensive cover of *Polysiphonia* sp. (approximately 40%) in an eelgrass bed during late August 2009. Roughly one year later, the longer leaves that were covered by *Polysiphonia sp.* had been shed, resulting in a decrease in canopy height of approximately one-third to one-half (Cape Cod Cooperative Extension Marine Program 2011). Similarly, the trend of decreasing canopy height with increased loading of *Polysiphonia* occurred at both study sites.

As previously mentioned, extensive shading by *P. subtilissima* and other algae reduces photosynthetic production in eelgrass which can lead to mortality (Höffle et al. 2011). Brush and Nixon (2002) found an exponential relationship between light transmission and epiphyte biomass in both *Polysiphonia* sp. and *Cladophora* sp. Light level measurements were not collected for this study; hence, the influence of epiphytic and drift algae on the transplants is unclear. Regardless, it is highly probable that the algal epiphytes and drift algae impacted the transplants at both sites. The eelgrass from Tracadie Harbour also had a dense accumulation of sediment particles tangled with the epiphytic algae. Eelgrass mortality can occur when the leaves are covered by excessive sediment (Vandermeulen 2005). The donor site had a transplant mortality of six shoots, likely caused by stress from increased sediment particles and intense cover of epiphytic algae.



Figs 34–36. Interactions between the epiphytic red alga *Polysiphoni*a with *Zostera marina*. **Fig. 34.** The impact of *Polysiphonia* sp. cover and light attenuation on eelgrass health (figure from Brush and Nixon 2002). **Fig. 35.** Intense epiphyte cover by *P. subtilissima* in Tracadie Harbour, (A) *in situ* and (B) using a stereoscope. **Fig. 36.** Benoit Cove transplants with a high drift algal cover of *P. subtilissima*, (A) *in situ* and (B) at the end of the 7-week experiment.

By the end of August, Tracadie Harbour had between 75% and 100% epiphytic algal cover and Benoit Cove had an estimated 25% to 75%. There are a few possible explanations for how Tracadie Harbour had only 6 shoot mortalities and a high canopy height despite an intense epiphytic algal cover. With increasing epiphyte density, light penetration can eventually level off because the algae will start to grow over the edges of the *Z. marina* blades (Brush and Nixon 2002). Also, with increased epiphyte loading, eelgrass becomes more dependent on mesograzers (e.g. amphipods, copepods, and isopods) to control epiphytic algal growth (Unsworth et al. 2015). In Tracadie Harbour, seven species of amphipods and one isopod species (Baltic isopod) were abundant. This community, which feeds on algae, may have prevented the epiphytes from suffocating the eelgrass or further reducing light availability (Hily et al. 2004; Unsworth et al. 2015).

The experimental results from Tracadie Harbour suggest that the eelgrass bed is highly resilient because transplant survival success occurred over the summer where environmental stressors (i.e. water temperature, light attenuation, and epiphytic algal cover) were high (Unsworth et al. 2015). Seagrasses depend on light during spring to preserve carbohydrate storage reserves (CSR) which help the plants tolerate periodic occurrence of negative carbon balance (Simpson and Dahl 2017). This process is extremely important during summer when high light levels are often compromised by shading of epiphytic algae. Tracadie Harbour had a transplant survival of 91.6% and an estimated epiphytic algal cover between 75% and 100%. This suggests that the eelgrass CSR resisted the negative carbon balance from intense algal shading.

Nonetheless, the plants' effectiveness at tolerating such conditions can be reduced when additional stressors are introduced. These stressors include turbidity, increased water temperature and high sediment sulfide (Vandermeulen 2005; Unsworth et al. 2015; Simpson and Dahl 2017). Although algal cover in Benoit Cove was less than in Tracadie Harbour, frequent turbidity and a potential increase in sediment sulfide levels may have taken the transplants to a point at which they could no longer tolerate the cove's environment.

5.1.5 Transplant Methodology

It is unlikely that the transplant frames had a significant impact on survival results in Benoit Cove because the same methodology was used in Tracadie Harbour and was highly successful. The TERFS method was designed to help protect eelgrass transplants from various bioturbators (Short et al. 2002*b*). The frames in this design were made from PVC pipe, so the sides were not as tall as the original design, and organisms were still able to get in. The two bioturbators, Harris mud crab (*R. harrisii*) and European green crab (*C. maenas*), were both found in the Tracadie Harbour and Benoit Cove frames. The frames provided some protection for the transplants because the bioturbators were unable to dig through the mesh to make pits.

It is possible that the eelgrass transplants that died were unhealthy or at a disadvantage prior to selection for the transplant experiment. Based on blade length measurements, there was a similar trend in negative growth of blade length between survived transplants and transplants that eventually died in both study sites. This trend supports the hypothesis that transplant mortality occurred because of abiotic or biotic factors and not because they were already unhealthy.

In Benoit Cove, 17 shoots went missing over the course of the experiment. The eelgrass transplants were securely fastened to the frames using cable ties. Tightening them any further would have caused the rhizomes to break. If a different method had been used, more eelgrass shoots would probably have gone missing in Benoit Cove. The sediment was very fine and there was no vegetation to deflect the water current. In Tracadie Harbour, the eelgrass bed protected the transplants from physical exposures which explains why only seven shoots went missing.

5.2 Sediment Organic Matter Content

Organic matter (OM) content of sediment was significantly different between the two sites. Tracadie Harbour had a mean organic matter content of 32.95 ± 10.08% which was roughly 21% higher than Benoit Cove. McGlathery et al. (2012) found that nine-year-old eelgrass meadows that were restored had double the organic matter content than bare, unvegetated habitats nearby. Despite the statistically greater mean from Tracadie Harbour, OM content from both study sites was considered to be high. The accumulation and preservation of organic matter is often higher in finer grained sediment (Keil et al. 1994). This explains why OM content was significantly richer in both transplant sites than in Antigonish Harbour. Tracadie Harbour and Benoit Cove had high concentrations of fine silt whereas sandier Antigonish Harbour had OM of 1.99 ± 0.83%.

Although sediment OM in Benoit Cove (11.90 ± 5.31%) fell within the upper end of the ideal threshold, organic matter may have been too high for the transplants when exposed to other intense abiotic conditions. If light levels and photosynthetic performance are strong, eelgrass can release oxygen to its roots which maintains sediment chemistry and prevents increases in organic content (Koch 2001; Leschen et al. 2010). Since light availability was poor in Benoit Cove, it is possible that the eelgrass transplants were unable to maintain neutral sediment conditions. Organic enrichment can also change the chemical composition of the surrounding environment by increasing sediment sulfide and ammonium concentrations, and lowering oxygen levels which impacts the growth and survival of eelgrass (Krause-Jensen et al. 2011; Eriander et al. 2016). These chemical cycles are often regulated by dense eelgrass beds, but this did not occur in Benoit Cove because vegetation was essentially absent. In Tracadie Harbour, organic matter content was higher than in Benoit Cove, but the density of eelgrass was also significantly greater which may have allowed the site to complete these chemical cycles more efficiently.

In previous studies, seagrass habitats with high silt composition had lower OM content than my results. The variability in OM content between this experiment and other studies can be explained by the differences in depth at which the sediment cores were sampled. In this study, the sediment samples were collected at between 10 cm and 15 cm whereas in other studies sediment core samples were collected at a depth of 30 cm (e.g. Herrera-Silveira et al.

2000; Green et al. 2018). With increasing depth, sediment OM decreases; this may explain the lower OM content in similar sediment types. Sediment cores were collected at this depth because I initially thought that since the eelgrass rhizomes only grow within the first 2-5 cm of the sediment, there was no need to collect at a greater depth. Also, if the organic matter content was poor at a shallower depth, then it would be clear that something was wrong.

There are a few explanations for why OM was so high. There may have been increased terrestrial input entering both study sites. The eelgrass bed in Tracadie Harbour is located beside a large salt marsh that fills with seawater during high tide and filters back into the harbour during low tide. In Benoit Cove, a small salt marsh and freshwater stream are at the head of the cove and there are several trees and shrubs closer to the shore. This could have provided additional terrestrial input. In addition, a lack of flushing may have contributed to the increase of OM content in both sites.

The increased epiphytic and drift algal cover may have also had an impact. Drift algae can contribute organic material to the sediment (Höffle et al. 2011). Towards the end of the experiment, the estimated epiphytic and drift algal cover was intense in Tracadie Harbour and high in Benoit Cove. Sediment cores used for determining the OM content were sampled after the eelgrass transplant experiment. At this time, there could have been a build-up of organic matter on the sediment surface because of the steady increase in algal loading over the 7 weeks.

5.3 Species Richness

5.3.1 Tracadie Harbour

Results from the biota survey revealed that the study sites had different floral and faunal species. Tracadie Harbour supported an assortment of species with varying abundances. Eelgrass beds are typically species-rich, with abundant annelid, arthropod, mollusc and fish species (Burkholder and Doheny 1968). Tracadie Harbour had invertebrate species from all groups, showing that the site had the typical species composition of an eelgrass system (Fig. 37). Fifteen species were abundant that were either absent or rare in Benoit Cove. Of this group, nearly 50% were small crustaceans (amphipoda and tanaidacea) and 36% were fish. These small crustaceans were most often found grazing in the algal epiphyte layers attached to the eelgrass blades. During the study, the population of amphipods, tanaidaceans, fish, Baltic isopods (*I. balthica*) and sand shrimps (*C. septemspinosa*) had increased (personal observation). An increase in species-richness and diversity from late spring to late summer is often observed in seagrass habitats with additional nutrients (Prado et al. 2008). Previous studies found a significant positive relationship between amphipod abundance and density of algal epiphytes (Saunders et al. 2003). This makes sense because in Tracadie Harbour, there was a large increase in epiphytic algae and drift algal cover across the entire habitat.

According to Joseph et al. (2013), higher fish densities are more often observed in eelgrass meadows than above bare substrata. This trend occurred between the two study sites. Tracadie Harbour, an eelgrass habitat, had five fish species, four of which were highly abundant. The fish feed on small crustaceans such as amphipods, copepods, and shrimps (Hemminga and Duarte 2000). The frequency of amphipods, isopods, and shrimps in Tracadie Harbour could indicate that these small crustaceans had high nutritional value and were preferred by fish in this habitat. In contrast, Benoit Cove had very few amphipods and fish. The three-spined stickleback (*G. aculeatus*) was the only fish species identified in Benoit Cove and it had a low abundance. Benoit Cove also had a low density of sand shrimp and amphipods, which may have been a contributing factor to the low fish diversity.

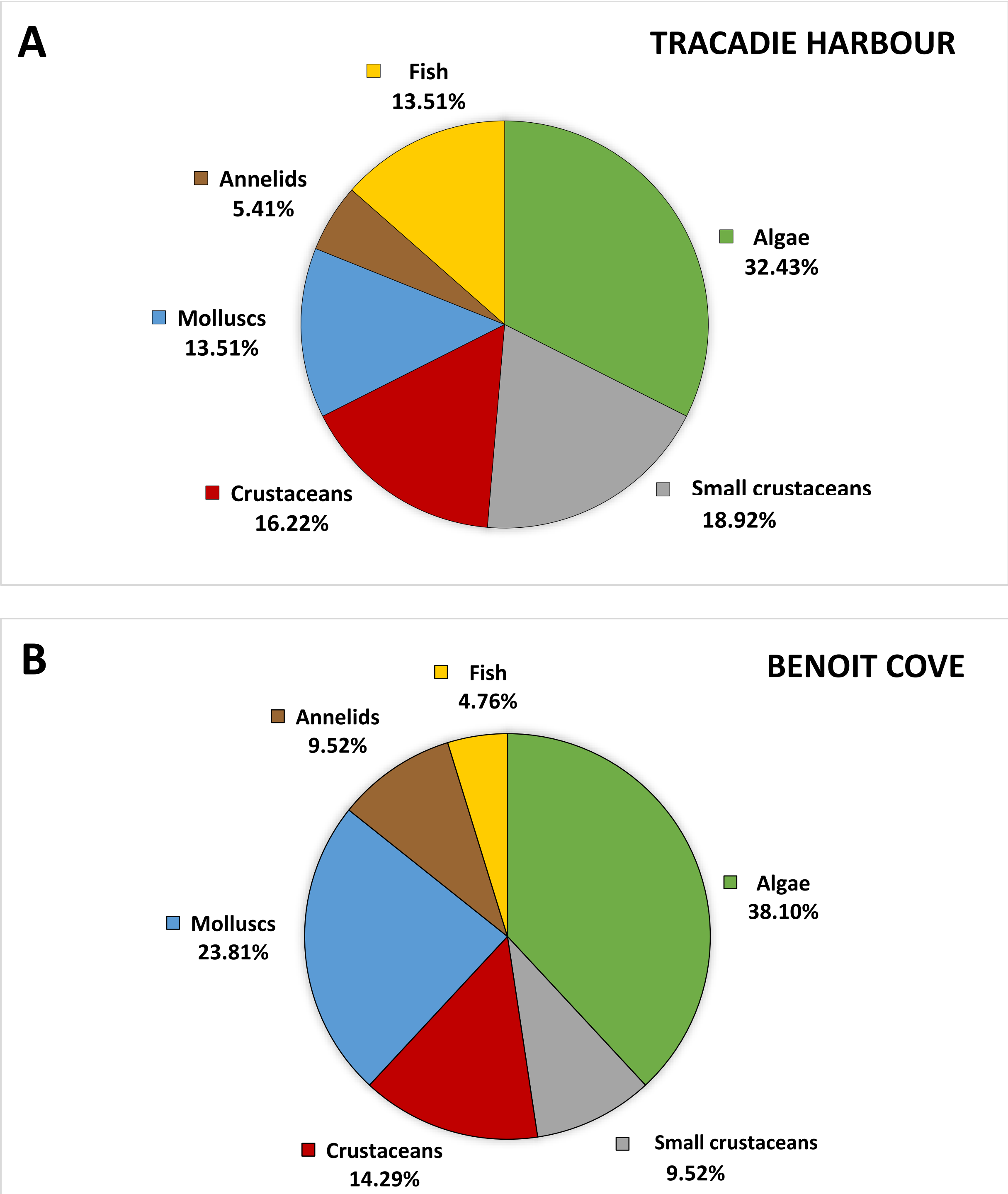


Fig. 37. Percentage of taxonomic groups that represent the number of species found in (A) Tracadie Harbour and (B) Benoit Cove.

The clear differences in species composition can be attributed to the denser eelgrass bed in Tracadie Harbour. In addition to being highly productive, eelgrass is a structure-providing species that adds habitat complexity and protects animals from predation (Hastings et al. 2014). These characteristics attract a variety of invertebrate species. Subsequent studies have concluded that predation foraging success often declines with high seagrass shoot density (Hemminga and Duarte 2000). This finding suggests that the populations of species in Tracadie Harbour were balanced because predation pressures were lower. The observed species, especially juvenile fish and amphipods, used the leaf canopy to hide from predators. Seagrass canopy is positively correlated with the abundance of both epifaunal and infaunal species (Orth et al. 1984). In Benoit Cove, the low abundance and diversity of species may have been associated with predation pressure in a highly exposed habitat.

Despite the high transplant survival success in Tracadie Harbour, the high macroalgal cover raises concern for the future health of the eelgrass bed. The increase in epiphytic algae most often occurs in polluted waters during late summer (Burkholder and Doheny 1968). Polluted waters can indicate that there are additional nutrients being released into the area which leads to eutrophication. When a site becomes increasingly eutrophic, there is often a shift in the macroalgal species composition. With increased nutrient levels, fast-growing green algae, including *Ulva* spp., tend to dominate (Vandermeulen et al. 2012). In Tracadie Harbour, there were six different green macroalgal species, three of which were *Ulva*. Increased cover of these species can also change the biogeochemical dynamics within a site, which is problematic for eelgrass. As epiphytic and drift algae decompose in Tracadie Harbour, the process will reduce the amount of oxygen available, creating dead zones which can contribute to fish kills (Yarrington et al. 2013). Nutrient input, especially nitrogen, has become a major global concern for coastal ecosystems. A recent study by Murphy et al. (2018) found that 64% of their study sites in Nova Scotia are at risk for eelgrass declines because of nitrogen loading.

5.3.2 Benoit Cove

Following the assessment of biota in Benoit Cove, it was obvious that the biota of Benoit Cove no longer matched the expectations of a typical seagrass habitat. Only a few invertebrate species dominated the site, and these were not the typical dominant organisms in eelgrass beds. This was expected because eelgrass is important in maintaining a diverse faunal assemblage. Following the disappearance of eelgrass in the 1930s, the Woods Hole area lost approximately one third of its species (Burkholder and Doheny 1968). The same type of observation for Benoit Cove was made in 2013 by Garbary et al. (2014), who reported that the cove was depauperate in benthic invertebrates. The most abundant animal species were the Eastern mud snail (*Tritia obsoleta*), Common slipper limpet (*Crepidula fornicata*), Eastern oyster (*Crassostrea virginica*), European green crab, Harris mud crab, and two annelid species. In Woods Hole, Burkholder and Doheny (1968) also found that mud snails and species of crustaceans adapted to the loss of eelgrass by moving to the sediment surface, whereas several other organisms completely disappeared. Benoit Cove showed the same trend.

After the eelgrass shoots were planted in Benoit Cove, nearly all the animals identified in the cove were found attached to, within or near the transplant frames. Although the abundance of organisms was still low, increased animal diversity occurred even with a limited amount of eelgrass (Fonseca et al. 1990). Orth (1977) observed that objects placed into the sediment supported several epifaunal communities and attracted a variety of infaunal species to these areas. This result confirms previous observations of a strong positive correlation between macroinvertebrates and eelgrass transplant units (e.g. Smith et al. 1989; Fonseca et al. 1990). There are several possible explanations, but it appears that the frames provided a more complex ecosystem with increased food resources, shelter and sediment stability.

The animal assemblage associated with the frames in Benoit Cove may indicate that the organisms migrated to the transplant frames from other areas where resources were more limited. Just 1 week after the eelgrass transplanting, mud snails were abundant on the frames. *T. obsoleta* has well-developed sensory systems that allows it to quickly detect food resources such as high levels of detritus in which they form dense aggregations (Watson et al. 2018). In and around the frames, increased detrital matter may have resulted from the accumulation of diatoms, drift macroalgae, disintegrating eelgrass transplants, and carrion, all of which attract mud snails and other animals. At all three subsites, there were numerous mucus trails created by the mud snails (personal observation). Trail-following is a known behaviour that helps the snails find areas with sufficient food resources (Trott and Dimock 1978), which explains why there was a rapid increase of mud snails in the frames.

It appears that the test site has entered a new state, one that resembles a macroalgal bed or mudflat. *P. subtilissima* and *Ulva lactuca* did not form extensive macroalgal beds in the cove, but dominated the transplant frames and patches along the shoreline. In the United States, several sites that initially supported dense eelgrass meadows failed to recover and over time became dominated by macroalgal beds of *Ulva* spp. (Cottam and Munro 1954). It is possible that Benoit Cove will develop into a macroalgal bed. Coastal habitats that are occupied by *U. lactuca* have been shown to be important for faunal communities in areas that lack *Z. marina* (Sogard and Able 1991). In habitats with limited fixed nitrogen, *U. lactuca* serves as a food source for marine invertebrates. The downfall with increased development of macroalgal mats of *U. lactuca* is their impact on fish and crab larval species because they lower dissolved oxygen levels (Sogard and Able 1991).

Adult and juvenile mud snails were abundant throughout the test site. Mud snails are common in the sheltered, soft-sediment habitats of North America. When abundant, they can serve as indicator species for water quality and sediment conditions (Kelaher et al. 2003; Watson et al. 2018). Several juvenile mud snails were found attached to drifting *U. lactuca*. This observation can be simply explained by the fact that mud snails are a dominant species and often coexist with *U. lactuca* (Giannotti and McGlathery 2001) which was also common to abundant. The increased cover of *U. lactuca* in Benoit Cove occurred after the discovery of mud snails in high abundance along the frames. Studies have suggested that mud snails can influence the growth and distribution of macroalgae through epiphyte removal or by providing additional nutrients (e.g. Connor et al. 1982; Yarrington et al. 2013). From mid-August onwards, there were fewer mud snails on the frames, and the number of *Capitella* sp. had increased. When there is an increase in *Ulva* detritus and a low abundance of *T. obsoleta*, there is often a large increase in deposit-feeding polychaetes such as *Capitella* sp. (Kelaher et al. 2003).

Benoit Cove also had a moderately high number of ragworms (*Hediste diversicolor*). A previous transplant study of *Zostera noltii* revealed that transplant success was best in habitats that excluded ragworms (Hughes et al. 2000). According to Hughes et al. (2000), herbivory and sediment displacement by ragworms influenced transplant success of *Zostera* species. A similar observation occurred with studies of other saltmarsh plants such as *Spartina angelica* (Emmerson 2000). The high abundance of *H. diversicolor* and other species may have contributed to the poor transplant survival of the *Z. marina* transplants in Benoit Cove. Another abundant species that can impact estuarine habitats is the common slipper limpet (*Crepidula fornicate*; Blanchard 2009). Towards the end of the experiment, slipper limpets were found all over the transplant frames. Past studies expressed concern with increasing populations of *C.*

fornicata because they form cohesive anoxic mud which can negatively affect other organisms (Blanchard 2009).

5.3.3 European green crab

At each visit, 4–8 crabs were counted in the test site. The green crab count in Benoit Cove went from 0.6 crabs m^{-2} in 2002 to 0.03 crabs m^{-2} in 2013 (Garbary et al. 2014). These observations were determined using a 400 m^2 surveying area. In my study, the snorkelling survey covered approximately 600 m^2 which included all three subsites. Approximately 0.01 crabs m^{-2} were counted which indicates that the green crab population has continued to decline in Benoit Cove. Throughout the last 17 years, the abundance of green crabs in Benoit Cove appears to have steadily declined with the decrease in eelgrass density. With an eelgrass density of essentially zero, it appears that most of the green crabs either died or moved to a new habitat with more substantial food resources and better security.

Tracadie Harbour had a significantly greater population of green crabs with roughly 21–62 green crabs counted per visit. Subsite C was the most abundant (area of 200 m^2) with 0.1 crabs m^{-2} and the least abundant was subsite B with 0.045 crabs m^{-2} (200 m^2). In terms of green crab density, there is no existing baseline data for the comparison site which makes it difficult to determine whether the eelgrass bed is entering a more vulnerable state. Based on the observations for just a small portion of the harbour, it appears that the green crabs are highly abundant in Tracadie Harbour and may pose a significant threat to eelgrass habitat.

5.4 Future Research and Limitations

Transplanting eelgrass during summer is not advised. Planting a few weeks sooner would have helped, but this was not possible. The most suitable time to transplant eelgrass is during winter or spring; however, this can be difficult in eastern Canada due to ice and freezing temperatures. In the eastern United States, scientists claim that transplanting eelgrass in the fall is preferable (Orth et al. 1999). If these shoots were planted during the fall or late spring, I believe the survival and growth results at both sites would have been more telling.

To strengthen my conclusions, additional baseline data would confirm whether Benoit Cove is a suitable site for eelgrass restoration. Thorough evaluations of light levels, wave and current velocity, and sediment chemistry are necessary. Suggested variables to measure these factors include surface light irradiance, biomass of epiphytic and drift algae, nutrient levels, sediment sulfur concentration, and frequency of sediment resuspension. This study had aimed to evaluate the dissolved oxygen and sediment sulfide levels in both sites using core samplers, and measure the thickness of the oxic and anoxic layers. Sampling at both transplant sites was delayed due to harsh weather conditions, scheduling conflicts with transportation, and lack of congruence between actual water levels and predicted tide levels. When these constraints were overcome, the evaluation was not completed because the two sites had frozen over for the winter.

It is plausible that the initial shoot density of 9 shoots per 0.25 m^2 was not enough to counteract the physical conditions in Benoit Cove. Since transplant survival increases when the planting unit size is increased (Sheridan et al. 1998; Kim and Park 2007), having designed a larger frame with more shoots may have helped with sediment stabilization leading to improved plant survival. Therefore, increasing the number and size of frames should improve transplant survival, especially in sites like Benoit Cove.

For Tracadie Harbour, further monitoring and research on nitrogen levels and green crab populations is needed. Eutrophication may be a serious issue and having quantitative data on nitrogen levels could help determine whether there is a problem and its cause.

5.5 Economic Analysis of Eelgrass Restoration in Canada

5.5.1 Economic Value of Transplanting Eelgrass

During this research, it became increasingly obvious that nearly all existing data and knowledge on eelgrass bed declines and transplanting efforts in eastern North America have come from the United States. Recently, Atlantic Canada has started to evaluate coastal habitat conditions that support *Z. marina* beds, but they lag behind the US. Without doubt, the US is the leader in seagrass management and restoration using various transplanting techniques (Hemminga and Duarte 2000). Canada has the longest coastline in the world, yet eelgrass transplant efforts are few and far between. The only accessible example from eastern Canada is from a local environmental group located in Souris, PEI, who tried transplanting eelgrass with oyster shells as anchors (Novaczek 2016). This project occurred in 2015 and there are still no results or updates available. The neglect and lack of transparency raises concern because eelgrass habitat provides important ecosystem services that are ecologically and economically important to the Maritime provinces.

Most commercial species depend on seagrass beds during at least one stage of their life cycle (Hughes et al. 2009). Thus, the decline of these habitats has serious repercussions on commercial species. However, there is a lack of Canadian literature on the economic importance of eelgrass meadows on commercially valuable fish species, but sufficient evidence has been found elsewhere. In the Mediterranean, seagrass beds have an estimated economic value at US$93.5 million yr^{-1} for commercial fishing because 30–40% of the commercial landings are represented by seagrass-related species (Jackson et al. 2015). In Sweden, seagrass currently has an estimated value of US$5,300 ha^{-1} for enhancing the production of five commercial fish species and 97% of that value is represented by Atlantic cod (*Gadus morhua*), Goldskinny wrasse, and Corkwing wrasse (Cole and Moksnes 2016). In North America, there is also a strong correlation between the extent of seagrass cover and abundance of commercial species (Plummer et al. 2013). In Chesapeake Bay, the Virginia hard-shelled blue crab (*Callinectes sapidus*) is a commercial species that relies on eelgrass habitat (Anderson 1989). According to Anderson, if the eelgrass habitat were returned to its original state from 1960, the net economic benefit to the blue crab fishermen would be approximately US$1.8 million per year and an additional $2.4 million to US crab consumers. This is roughly US $4.3 million yr^{-1} for just one commercial species (Anderson 1989).

In the western US, Plummer et al. (2013) found that the commercial net value for hatchery salmonids, mature crabs and geoduck species is influenced by changes in eelgrass cover (Table 11). Salmonids that are harvested both commercially and recreationally (hereafter hatchery salmonids) have a total commercial harvest net value of US$3,054 km^2 (Plummer et al. 2013). Plummer et al. (2013) predicted that if eelgrass cover were increased by 100%, the total commercial net value of hatchery salmonids would increase by US$1,747 km^2. Salmonids, Atlantic cod, and crustaceans depend on eelgrass habitats and these species are all important to Nova Scotia fisheries. A case study in the Maritimes concluded that eelgrass meadows should be considered as Essential Fish Habitat (EFH), especially for Atlantic cod (McCain et al. 2016). McCain et al. (2016) stressed that large declines in the abundance of juvenile cod coincides with a decrease in adult stocks. If eelgrass habitats are properly preserved, juvenile fish that are found in these systems could contribute to recovering the adult stocks.

Additionally, large populations of adolescent American lobsters (*Homarus americanus*) are found in eelgrass meadows along the east coast (White et al. 2012). Commercial landing for lobster on the Canadian Atlantic coast in 2016 was 90,624 metric tonnes, with the province of Nova Scotia providing more than half this amount on its own (DFO 2018*c*). Nova Scotia provides CDN$729,982 of the total ($1,296,336) commercial lobster landings (DFO 2018*c*). In order to maintain a strong economic status of the American lobster fishery in the Maritimes, eelgrass restoration and management in Nova Scotia is vital.

Fish production is one of the most valued ecosystem services, but it does not represent the highest economic value (Table 12). Cole and Moksnes (2016) found that fish production represented 25% of the total value, whereas nitrogen cycling had 46% and carbon sequestration had 30%. The majority of valuation studies have been conducted on the value of seagrass habitats for enhancing commercial fishing. This leaves out other important ecosystem services and suggests that most seagrass beds are undervalued (Cole and Moksnes 2016). Most valuations for seagrasses are missing the non-consumptive value of carbon sequestration because the significance of seagrasses as a major carbon sink was only recently discovered (Dewsbury et al. 2016).

Table 11. Total and incremental values for commercial harvest groups depending on eelgrass biomass (data retrieved from Plummer et al. 2013).

Commercial harvest groups	**Commercial harvest TNV ($/km²) (at eelgrass baseline)**	**Incremental commercial net value ($/km²)**		
		Eelgrass (50% decline)	Eelgrass (20% increase)	Eelgrass (100% increase)
Subadult hatchery, pink and wild salmon	$3,045	-$746	$316	$1,747
Geoducks	$2,840	-$18	$4	$12
Crab (> age 1)	$317	-$36	$15	$68

Table 12. The economic value of seagrass habitats from studies on commercial fishing, restoration, nutrient cycling, and carbon sequestration (data retrieved from Unsworth and Cullen-Unsworth 2010).

Service	**Location**	**Value ($US)**
Fisheries Exploitation	Queensland, Australia	\$3,500 ha^{-1} yr^{-1}
Fisheries Production	South Australia	\$1,436 ha^{-1} yr^{-1}
Fisheries Production	Indian River Lagoon, USA	\$1,862 ha^{-1} yr^{-1}
Restoration	United States	\$1,236 ha^{-1}
Restoration	Florida, USA	\$140,752 ha^{-1}
Nutrient Cycling	Globally	\$19,004 ha^{-1} yr^{-1}
Carbon Sequestration	Globally	\$28 ha^{-1} (avg)

5.5.2 Costs and Effectiveness of Transplant Techniques

Seagrass habitats are by far one of the most expensive ecosystems to restore (Bayraktarov et al. 2015). The United States has invested in some of the most expensive seagrass restoration programs ranging from US \$1.9 to \$3.39 million ha^{-1} (Paling et al. 2009). There is an extensive price range in seagrass restoration because of the differences in seagrasses transplant methodologies (Blandon and zu Ermgassen 2014). In Atlantic Canada, eelgrass transplanting efforts are near absent and it is unclear as to why this is the case. Perhaps there is an assumption that the efficiency of eelgrass restoration can be achieved by only the costlier techniques, and therefore Canada will not invest. This is not entirely true because several studies have achieved high transplant survival from low restoration costs. For example, the estimated cost for the Horizontal Rhizome Method (HRM) is US \$85,470 and the TERFS method is \$42,735 per acre (Short et al. 2002*b*). These two techniques were used at the same sites and TERFS had a much higher eelgrass transplant survival rate (Short et al. 2002*b*).

A study on coastal restoration by Bayraktarov et al. (2015) found that 22 sites located in developed countries had an average total restoration cost of US\$699,525 ha^{-1} over one year. Survival rate was approximate 38%, which was the lowest success rate of all coastal ecosystems involved in the study. Once again, the more expensive technique was the least successful. The most cost-effective method from this study was the use of sediment cores and plugs which cost US\$29,749 ha^{-1} and the least cost-effective method was the use of mechanical transplantation which costed US\$1,196,896 ha^{-1} (Bayraktarov et al. 2015). Therefore, restoration costs are not necessarily correlated with the survival of seagrass transplants, and survival is highly dependent on site-selection and the fit of the restoration technique for each site (e.g. Short et al. 2002*a*; Bayraktarov et al. 2015).

5.6 Canadian Seagrass Management Policies

5.6.1 Eelgrass as a Significant Species and Habitat

From an environmental perspective, the importance of seagrass habitats and the significance of restoration are becoming understood globally. Over the last several decades, involvement in eelgrass management and protection has been limited in Canada. This is apparent as there have been no policies developed aimed directly at monitoring and restoring beds of *Zostera marina* in eastern Canada (Short and Short 2003). As previously mentioned, eelgrass systems are now considered an Ecologically Significant Species (ESS) because they meet necessary criteria (DFO 2009). This shows that Canada has started to address the importance of seagrass habitats.

While this is a step forward, there are some flaws with this classification. In 1996, the United States' National Oceanic and Atmospheric Administration (NOAA) listed eelgrass as an Essential Fish Habitat (EFH) under the *Magnuson-Stevens Fishery Conservation and Management Act* (NOAA 2014*a*). The difference between the two listings is that the EFH in the United States is legally protected, whereas Canada's EBSA offers no legal protection (McCain et al. 2016). Ecologically and Biologically Significant Areas need to acquire legal protection so that species under this listing are immediately protected and further restoration or mitigation acts can be enforced.

5.6.2 Protection of Eelgrass Habitat

In 2012, the *Fisheries Act* changed its legislation so that the protection and mitigation of marine habitats would be recognized only if commercial fishing acts were involved. The main issue is that with declining seagrass cover, fish abundance also decreases, which forces commercial fisheries to move elsewhere (McCain et al. 2016). As a result, the degraded and vulnerable habitats that were once fished, are now less likely to receive the protection and restoration needed. Recently, *Fisheries Act* section 35 was revised such that all fish and fish habitats will be protected (DFO 2018*b*). This crucial change may help bring more attention towards vulnerable fish habitats, including eelgrass beds. However, modifications are necessary so that eelgrass systems become a main priority rather than an afterthought. Changes to existing policies within the *Fisheries Act* and *Oceans Act* could promote and force the protection of eelgrass meadows. Vulnerable or degraded seagrass sites that are valuable to commercial fisheries and other services should be listed as Marine Protected Areas in the *Oceans Act*. This would help prevent anthropogenic disturbances and allow scientists to monitor habitats on a regular basis.

5.6.3 "No Net Loss" Policy

NOAA (2014*a*) recently developed a new guideline known as the *California Eelgrass Mitigation Policy* which ensures a "no net loss" of eelgrass habitat function caused by federal agencies. This policy also uses compensation mitigation in which restoration acts will create around 20% more eelgrass habitat than what was originally destroyed (NOAA 2014*b*). According to NOAA (2014*b*), this will compensate for the loss of beneficial eelgrass services since it will take many years for the habitat to reach full maturity. DFO also has a "no net loss" policy for marine habitats that support significant commercial fish species, but there are no policies focused on direct protection of eelgrass beds (McCain et al. 2016). It would be beneficial for DFO to follow the policy of NOAA and establish new policy guidelines for mitigation compensation for eelgrass beds. Such a policy could lead to development of restoration projects that compensate for the loss of cover by increasing eelgrass habitat to more than what was originally present. This type of policy would also create funding for restoration projects.

5.7 The Future of Eelgrass Management in Canada

5.7.1 Transplant Implementation and Improved Data Collection

With trends of declining eelgrass in the Maritimes, we need to adapt in order to save this valuable ecosystem. It is time for Canada to start investing and implementing eelgrass transplant projects along the Atlantic coast before things deteriorate further. There have been numerous eelgrass transplant studies just south of the border that provide insight into successful restoration efforts.

Before transplanting eelgrass, site conditions must be fully understood in order to determine which sites are most promising and which transplant methodologies are best suited. My study is a suitable example for explaining why site selection is important for transplanting *Z. marina*. Eelgrass was transplanted into Benoit Cove before having all of the necessary data and most of the transplants died because of abiotic conditions. In Canada, it is difficult to determine whether recovery in a declining eelgrass bed is possible because of a lack of baseline data (Hanson 2004; Namba et al. 2018).

According to Short and Short (2003), Canada established a map of eelgrass habitats in the 1980s but it was never digitized. They also believe that Atlantic Canada has more eelgrass cover than the United States, but there is no quantitative data to support this (Short and Short 2003). Eelgrass mapping has become more widespread on the west coast of Canada but it is limited in Atlantic Canada (Rao et al. 2014). In order to accurately monitor the status of eelgrass beds in terms of density, distribution and health, Canada needs to improve its mapping procedure. This is especially important for the Atlantic region where the only recent eelgrass mapping project is Placentia Bay, NL (Rao et al. 2014; DFO 2017).

5.7.2 Nutrient Loading in Atlantic Provinces

The concerns of increased nutrient input and eutrophication have already been discussed. In Atlantic Canada; several studies found increased nitrogen loading in eelgrass estuaries which poses a major threat to these habitats. In Nova Scotia, nitrogen loading has become a major risk factor for eelgrass declines (Fig. 38). According to recent studies from Atlantic Canada, the largest contributor of increased nitrogen in eelgrass estuaries is from direct and indirect atmospheric deposition (Cullain et al. 2018; Murphy et al. 2018; McIver et al. 2019). Atmospheric deposition represented 73–100% of the total nutrient loading in 21 bays from Nova Scotia that support significant eelgrass beds (Nagel et al. 2018).

Another major concern with nutrient enrichment and its impact on eelgrass habitats is the presence of nearby aquaculture farms. According to the Department of Fisheries and Aquaculture (2018), the aquaculture industry reached $100 million in sales which was a new record for Nova Scotia. With the success of the aquaculture industry, companies are now looking to increase profits by expanding across the province. Currently, there is a large knowledge gap on the interactions between aquaculture leases and nearby eelgrass beds in Atlantic Canada. Eelgrass habitats located in proximity to finfish aquaculture farms have shown increased nutrient loading and epiphytic algal cover, and reduced biomass and cover compared to eelgrass beds that are further away (Cullain et al. 2018). Eelgrass beds that are located under or near suspended bivalves have also been subjected to increased nutrient levels on the sediment surface (McIver et al. 2019). Therefore, continuous monitoring of eelgrass areas with respect to increased nitrogen levels is important. In addition, protection of eelgrass beds near proposed aquaculture farms needs to be mandated.

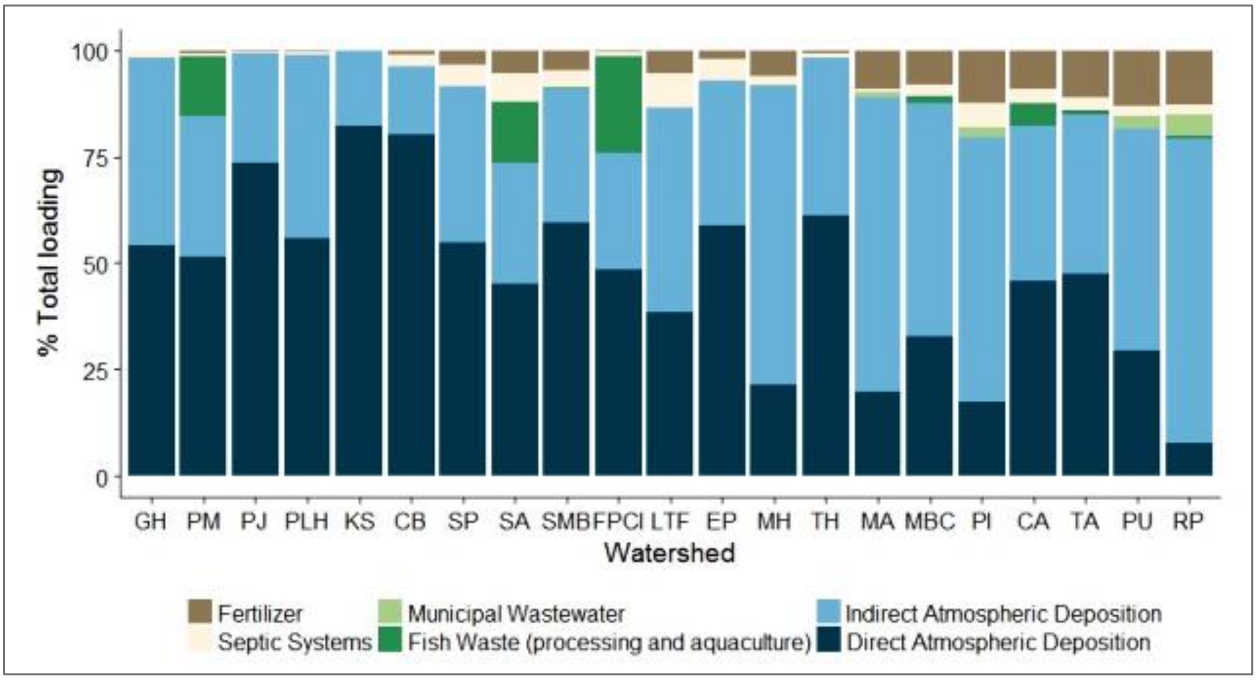


Fig. 38. Total nutrient loading (%) for 21 sites in Nova Scotia showing percent derived from fertilizer, septic systems, municipal wasterwater, fish waste, and indirect and direct atmospheric deposition (Nagel et al. 2018). Watersheds include Green Harbour (GH), Port Mouton (PM), Port Joli (PJ), Port L'Hebert (PLH), Kejimkujik Seaside (KS), Crescent Beach (CB), Second Peninsula (SP), Sambro (SA), St. Margaret's Bay (SMB), False Passage–Cable Island (FPCI), Lower Three Fathom (LTF), East Petpeswick (EP), Musquodoboit Harbour (MH), Taylor's Head (TH), Mabou (MA), Merigomish Big Cove (MBC), Pictou (PI), Caribou (CA), Tatamagouche (TA), Pugwash (PU), and River Philip (RP).

5.7.3 Increasing Community Involvement

The maintenance and regulation of eelgrass is required for continued coastal marine health and economic development, yet there is currently a lack of public awareness and appropriate management action to support the protection of seagrass systems (Cullen-Unsworth and Unsworth 2013). In Atlantic Canada, many locals have come in contact with eelgrass at some point in their lives but do not know what it is or the significance of this species. Most people confuse seagrasses with seaweed, which are completely different (Unsworth et al. 2018). Educating the community about the importance of eelgrass beds is a primary step towards the recovery of seagrass habitats. It can help people understand what anthropogenic actions are negatively impacting these eelgrass systems. This increases public awareness on how their lifestyles are contributing to the loss of valuable coastal habitats. For example, in areas that are experiencing higher nutrient input levels and epiphytic algal loading, locals who understand the impacts of housing development on nutrient input are more likely to repair or replace old sewage systems (LISS 2004).

Community involvement can also provide a “hands on” learning experience while improving the health of eelgrass beds. As previously mentioned, eelgrass transplanting can be time-consuming and labour intensive. These disadvantages can be improved by increasing the involvement of local members in eelgrass transplant projects and management. It would also provide an increased sense of importance and worth while educating others. For example, in Frenchman Bay, Maine, USA, Disney and Kidder (2010) conducted an eelgrass transplant project which relied on public outreach and education programs. In 2007, the site had an eelgrass cover of less than 1% and after using an adaptation of the TERFS method, eelgrass cover increased by 8.6% (Disney and Kidder 2010).

The Community Aquatic Monitoring Program (CAMP) is a program carried out along several estuaries in the Southern Gulf of St. Lawrence (SGSL 2009). The purpose of CAMP is to encourage local communities to participate in scientific assessment of the biota using a beach seine, monitor water quality, vegetation, nutrient levels and turbidity (SGSL 2009). These types of programs are extremely useful for increasing community involvement, but they need to be better advertised and promoted.

6 Conclusions

Findings from this study have highlighted the challenges of transplanting *Z. marina* in habitats with soft-sediment. Both study sites had a high percentage of silt (> 28%), but Tracadie Harbour had a transplant survival of 91.6% whereas Benoit Cove had a survival of 15.4%. It appears that the primary contributing factor was the difference in vegetation between the two sites. The transplants in Tracadie Harbour were surrounded by a dense eelgrass bed which protected them from physical exposure. Conversely, Benoit Cove had limited vegetation to stabilize the sediment, thus increasing sediment resuspension and turbidity. Additionally, the transplants in Benoit Cove may have become increasingly stressed leading to mortality due to reduced light levels from turbidity. For both sites, there was an increase in epiphytic and drift algal cover which would have also reduced light levels. Tracadie Harbour had a higher accumulation of epiphytic algae than Benoit Cove and still maintained a significantly greater transplant survival. It can be hypothesized that the extreme eelgrass mortality in Benoit Cove occurred because the site had too many environmental stressors which prevented the transplants from thriving. Since eelgrass stops growing at water temperatures above 20°C and can lose shoot weight at 25–30°C, plant growth was not an accurate measurement of transplant success in this experiment. Conducting this study during the fall or spring instead of in summer would have given a more accurate representation of eelgrass transplant growth.

Benoit Cove appears to have entered a new stable state at which it is unlikely to recolonize naturally because of the recurring suspension of fine sediments and turbidity. While these conditions also make eelgrass restoration incredibly difficult (Unsworth et al. 2015), the odds of achieving a successful transplant in Benoit Cove remain unclear and further research is necessary.

The study sites may have had similar habitat characteristics prior to the green crab disturbance, but this is no longer the case. Eelgrass habitats, including Tracadie Harbour, have a higher abundance of species and are typically more diverse than unvegetated habitats like Benoit Cove (McCullough et al. 2005).

There is a strong correlation between the extent of eelgrass cover and the abundance of commercial species (Plummer et al. 2013). Eelgrass restoration can be costly, but it would be worth the investment for the Maritime provinces that financially depend on commercial fisheries. Over the last several decades, eelgrass policies and management in Canada have been lacking. It is strongly recommended that Canada establish stronger policies and management practices aimed directly at sustaining eelgrass habitats. Improving data collection is essential for monitoring plant health so that the early warnings of eelgrass decline are noticeable and appropriate measures can occur to reverse the effects (Unsworth et al. 2018). In Atlantic Canada, augmenting social awareness on the ecological and economical importance of eelgrass habitat is incredibly important. Having public support can also influence government and organizations' decisions on eelgrass management and restoration practices (Unsworth et al. 2018). Restoring and conserving valuable eelgrass habitat are also extremely important on a global scale where there have been increases in nitrogen loading, invasive species, fisheries and aquaculture demands, CO_2emissions, sea level, and water temperature.

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Háskólasetur
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Appendix A

Research ethics training and clearance

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This letter certifies that **Erin Kathleen Wilson** has completed the following modules of:

(x) Basic ethics in research
(x) Human subjects research
(x) Animal subjects research

Furthermore, the Masters Study Committee has determined that the proposed masters research entitled **Absence of recovery in a degraded eelgrass (Zostera marina L.) bed in Nova Scotia, Canada. Results from a transplant study** meets the ethics and research integrity standards of the University Centre of the Westfjords. Throughout the course of his or her research, the student has the continued responsibility to adhere to basic ethical principles for the responsible conduct of research and discipline specific professional standards.

University Centre of the Westfjords ethics training certification and research ethics clearance is valid for one year past the date of issue.

Effective Date: 15 June 2018
Expiration Date: 15 June 2019

Prior to making substantive changes to the scope of research, research tools, or methods, the student is required to contact the Masters Study Committee to determine whether or not additional review is required.

Appendix B

Appendix 1: Eelgrass survival data from the transplant sites.

Tracadie Harbour – 3 July 2018 – 28 August 2018

Time (Days)	Population	Mortality (#)	Mortality (%)	Survival (%)
0	72	0	0.00%	100.00%
16	72	0	0.00%	100.00%
24	71	1	1.39%	98.61%
42	68	4	5.56%	94.44%
56	66	6	8.45%	91.55%

Eelgrass Transplant Survival (%)

Subsite	Frame	03-Jul-18	19-Jul-18	27-Jul-18	14-Aug-18	28-Aug-18
A	1	100.00%	100.00%	100.00%	83.33%	83.33%
A	2	100.00%	100.00%	100.00%	100.00%	83.33%
A	3	100.00%	100.00%	*	*	*
A	4	100.00%	100.00%	*	*	*
B	5	100.00%	100.00%	100.00%	88.89%	88.89%
B	6	100.00%	100.00%	100.00%	100.00%	100.00%
B	7	100.00%	100.00%	100.00%	100.00%	100.00%
B	8	100.00%	100.00%	*	*	*
C	9	100.00%	100.00%	100.00%	100.00%	100.00%
C	10	100.00%	100.00%	83.33%	83.33%	66.67%
C	11	100.00%	100.00%	100.00%	100.00%	100.00%
C	12	100.00%	100.00%	100.00%	100.00%	88.89%
	MEAN	**100.00%**	**100.00%**	**98.15%**	**95.06%**	**90.12%**
	s	**0.00**	**0.00**	**0.06**	**0.08**	**0.11**

*Data not available

Benoit Cove – 6 July 2018 – 29 August 2018				
Time (Days)	**Population**	**Mortality (#)**	**Mortality (%)**	**Survival (%)**
0	91	0	0.00%	100.00%
11	90	1	1.10%	98.90%
18	50	41	45.05%	54.95%
41	35	56	61.54%	38.46%
54	14	77	84.62%	15.38%

Eelgrass Transplant Survival (%)

Subsite	Frame	06-Jul-18	17-Jul-18	24-Jul-18	16-Aug-18	29-Aug-18
A	1	100.00%	100.00%	66.67%	33.33%	16.67%
A	2	100.00%	100.00%	66.67%	33.33%	0.00%
A	3	100.00%	100.00%	33.33%	33.33%	0.00%
A	4	100.00%	100.00%	57.14%	42.86%	0.00%
B	5	100.00%	100.00%	66.67%	55.56%	33.33%
B	6	100.00%	100.00%	62.50%	50.00%	25.00%
B	7	100.00%	100.00%	100.00%	85.72%	28.58%
B	8	100.00%	100.00%	37.50%	25.00%	12.50%
C	9	100.00%	100.00%	75.00%	25.00%	12.50%
C	10	100.00%	100.00%	37.50%	25.00%	12.50%
C	11	100.00%	87.50%	25.00%	12.50%	0.00%
C	12	100.00%	100.00%	57.15%	42.59%	14.29%
	MEAN	**100.00%**	**98.96%**	**57.09%**	**38.69%**	**12.95%**
	s	**0.00**	**0.04**	**0.21**	**0.19**	**0.12**

Appendix 2: Nested ANOVA and Tukey Pairwise comparisons test results for transplant survival between Tracadie Harbour and Benoit Cove and their subsites.

Nested ANOVA: Survival versus Site, Subsite, Frame

Analysis of Variance for Survival

Source	DF	SS	MS	F	P
Site	1	35415.5568	35415.5568	122.025	0.000
Subsite	4	1160.9304	290.2326	3.659	0.024
Frame	18	1427.6541	79.3141		
Total	23	38004.1413			

Variance Components

Source	Var Comp.	% of Total	StDev
Site	2927.110	95.68	54.103
Subsite	52.730	1.72	7.262
Frame	79.314	2.59	8.906
Total	3059.154		55.310

Expected Mean Squares

1	Site	1.00(3) + 4.00(2) + 12.00(1)
2	Subsite	1.00(3) + 4.00(2)
3	Frame	1.00(3)

Tukey Pairwise Comparisons: Subsite(Site)

Grouping Information Using the Tukey Method and 95% Confidence

Subsite(Site)	N	Mean	Grouping		
2(0)	4	95.7175	A		
3(0)	4	88.8900	A		
1(0)	4	84.7200	A		
2(1)	4	24.8525		B	
3(1)	4	9.8225		B	C
1(1)	4	4.1675			C

Means that do not share a letter are significantly different.

Appendix 3. Eelgrass transplant blade and rhizome length data from Tracadie Harbour.

Site A								
Frame ID	Eelgrass (#)	Sampling Dates					Rhizome Length (cm)	
		July 3	July 19	27 July	14 Aug	28 Aug	Initial	Final
Black	1	26.40	26.40	25.00	15.30	14.20	7.60	6.40
	2	17.40	17.40	17.80	0	0	4.60	0
	3	24.20	R	R	R	R	5.50	R
	4	28.70	28.70	28.90	29.00	20.20	8.00	8.00
	5	20.80	20.80	20.20	21.00	17.00	4.00	4.70
	6	21.60	R	R	R	R	3.30	R
	7	27.20	27.20	22.30	22.30	20.60	8.20	8.00
	8	32.10	R	R	R	R	4.70	R
	9	14.30	14.30	11.40	11.20	10.60	2.60	3.40
Blue	10	15.20	15.20	15.30	15.30	6.50	9.20	9.20
	11	18.20	18.20	18.00	18.00	0	4.60	0
	12	24.30	24.30	21.80	21.70	20.90	10.00	10.50
	13	30.10	M	M	M	M	3.60	M
	14	28.60	28.60	28.60	22.00	21.6	4.10	4.60
	15	34.70	34.70	30.20	26.50	25.1	7.00	6.60
	16	32.30	R	R	R	R	4.00	R
	17	34.30	34.30	33.70	33.00	32.1	3.70	3.80
	18	20.60	20.50	20.00	19.40	18.7	3.10	3.50
Green	19	47.20	47.20	M	M	M	10.60	M
	20	54.20	54.20	M	M	M	10.00	M
	21	32.60	32.60	M	M	M	7.40	M
	22	37.50	37.50	M	M	M	8.00	M
	23	30.40	30.40	M	M	M	4.70	M
	24	23.30	23.30	M	M	M	4.20	M
	25	33.20	33.20	M	M	M	2.60	M
	26	29.40	29.40	M	M	M	4.70	M
	27	14.30	14.30	M	M	M	3.90	M
Yellow	28	27.40	27.40	M	M	M	4.30	M
	29	34.70	34.70	M	M	M	5.70	M
	30	32.20	32.20	M	M	M	6.30	M
	31	37.50	37.50	M	M	M	5.20	M
	32	42.30	42.30	M	M	M	4.20	M
	33	36.10	36.10	M	M	M	2.50	M
	34	29.50	29.50	M	M	M	3.20	M
	35	26.50	26.50	M	M	M	2.00	M
	36	16.80	16.80	M	M	M	1.30	M
	MEAN	**28.78**	**28.89**	**22.55**	**21.23**	**18.86**	**5.24**	**6.25**
	s	**8.95**	**9.35**	**6.45**	**6.13**	**6.91**	**2.42**	**2.45**

Site B								
Frame ID	Eelgrass (#)	Sampling Dates					Rhizome Length (cm)	
		July 3	July 19	27 July	14 Aug	28 Aug	Initial	Final
Black and Green	1	24.80	24.50	22.10	20.80	17.40	3.70	3.90
	2	16.30	16.30	15.80	0	0	5.10	0
	3	26.70	26.70	24.10	22.70	21.20	6.10	6.30
	4	22.10	22.10	20.80	20.00	19.30	3.60	4.00
	5	24.80	24.80	23.70	22.10	21.60	6.80	6.90
	6	26.70	26.70	24.00	23.10	22.50	2.70	2.70
	7	28.20	28.20	23.10	17.80	5.10	6.90	7.00
	8	18.30	18.30	16.40	15.90	15.70	4.60	4.70
	9	20.40	20.40	18.40	16.90	15.50	2.80	3.00
Black and Yellow	10	22.80	22.50	21.30	20.60	18.20	3.00	3.00
	11	22.80	22.80	22.20	21.70	21.20	3.10	3.20
	12	30.10	28.10	25.50	20.70	15.60	8.80	8.90
	13	23.20	23.00	21.70	20.10	18.40	2.50	2.70
	14	29.70	29.20	25.10	24.90	23.50	5.30	5.60
	15	36.40	36.40	36.70	36.70	9.10	8.70	8.80
	16	35.80	35.20	33.50	31.90	30.20	2.90	3.00
	17	38.20	38.10	37.30	37.00	36.70	2.90	3.10
	18	27.20	22.70	17.40	16.80	5.20	5.60	5.50
Black with Red	19	22.10	22.10	20.90	18.80	17.20	11.20	10.90
	20	17.50	17.50	17.50	17.50	17.70	4.30	5.50
	21	36.20	36.10	31.20	29.50	27.40	9.00	9.80
	22	33.50	33.50	33.50	33.50	27.10	12.50	13.20
	23	31.20	31.20	27.10	23.80	16.00	4.00	4.80
	24	23.70	23.70	19.20	16.80	13.60	2.90	3.50
	25	20.90	20.90	16.00	15.40	13.80	4.50	5.20
	26	24.00	24.00	23.70	23.70	23.00	8.90	8.50
	27	28.70	28.20	27.80	26.00	25.00	7.20	6.70
Black and Blue	28	8.20	8.20	M	M	M	1.20	M
	29	27.80	27.80	M	M	M	2.00	M
	30	23.90	23.90	M	M	M	3.20	M
	31	25.10	25.10	M	M	M	4.40	M
	32	30.20	30.20	M	M	M	2.80	M
	33	18.40	18.40	M	M	M	7.40	M
	34	29.30	29.30	M	M	M	4.80	M
	35	12.50	12.50	M	M	M	2.80	M
	36	24.60	24.60	M	M	M	2.80	M
	MEAN	**25.34**	**25.09**	**23.93**	**22.87**	**19.12**	**5.03**	**5.78**
	s	**6.59**	**6.51**	**6.10**	**6.20**	**7.12**	**2.71**	**2.81**

Site C								
Frame ID	Eelgrass (#)	Sampling Dates					Rhizome Length (cm)	
		July 3	July 19	27 July	14 Aug	28 Aug	Initial	Final
Red (X2)	1	11.20	11.20	8.60	8.00	7.30	2.00	2.40
	2	37.10	37.10	32.80	28.40	24.20	17.50	17.20
	3	26.40	26.40	22.80	18.40	13.30	3.00	2.40
	4	36.50	34.20	30.10	30.10	29.50	9.00	8.90
	5	46.80	46.80	45.10	44.40	43.80	5.60	5.50
	6	40.00	37.20	32.50	32.50	32.00	22.10	22.50
	7	30.20	30.20	28.70	28.00	27.70	9.60	10.50
	8	31.50	31.50	28.80	25.10	23.00	6.90	5.00
	9	20.30	20.30	17.40	16.90	15.80	2.10	3.40
Yellow (X2)	10	32.40	32.40	25.80	24.00	23.90	9.50	10.00
	11	26.80	26.80	M	M	M	3.20	M
	12	37.20	37.20	35.80	35.00	34.80	8.20	10.40
	13	16.50	16.50	16.30	16.00	0	6.50	0
	14	37.30	37.30	32.90	31.80	M	12.80	M
	15	41.20	41.20	35.20	30.40	22.60	14.80	16.50
	16	17.30	17.30	0	0	0	7.80	0
	17	33.20	33.20	32.80	26.90	21.40	13.60	11.50
	18	35.80	35.80	36.00	M	M	3.40	M
White and Grey	19	20.30	20.30	19.60	19.00	M	2.70	M
	20	34.80	34.80	30.20	28.40	23.20	12.50	14.00
	21	36.50	36.40	35.10	34.70	34.10	3.90	8.50
	22	13.50	12.80	12.20	11.90	11.70	2.10	2.20
	23	24.90	24.90	24.20	23.70	22.20	3.60	4.00
	24	25.80	25.80	25.00	25.00	9.00	3.40	2.50
	25	16.20	16.00	15.40	14.20	13.50	6.00	5.50
	26	23.80	22.20	21.50	20.10	18.10	3.70	4.90
	27	33.40	33.10	32.40	31.80	30.20	19.70	20.50
Black (X2) White Frame	28	35.00	35.00	35.00	32.00	30.50	12.50	12.20
	29	26.00	26.00	12.70	12.40	7.20	8.20	8.20
	30	21.40	21.40	20.50	0	0	5.60	0
	31	37.20	37.30	34.10	34.0	31.90	16.40	13.50
	32	20.60	20.60	18.40	18.2	16.50	5.30	3.50
	33	42.70	42.70	40.20	40.0	39.50	10.60	9.50
	34	43.70	43.70	43.30	43.6	17.20	4.80	6.60
	35	37.30	36.90	36.30	35.2	34.90	8.60	6.10
	36	16.30	16.30	15.90	15.2	14.10	8.00	7.50
	MEAN	**29.64**	**29.41**	**28.03**	**26.10**	**23.21**	**8.20**	**8.81**
	s	**9.45**	**9.40**	**8.90**	**9.35**	**9.79**	**5.25**	**5.48**

Appendix 4: Eelgrass transplant blade and rhizome length data from Benoit Cove.

Site A

Frame ID	Eelgrass (#)	Sampling Dates					Rhizome Length (cm)	
		July 6	July 17	24 July	16 Aug	29 Aug	Initial	Final
White Black	1	44.00	44.00	30.50	M	M	4.20	M
	2	29.10	29.10	0	0	0	2.90	0
	3	40.00	40.00	M	M	M	5.20	M
	4	19.80	19.80	0	0	0	2.60	0
	5	26.00	26.00	27.00	0	0	1.60	0
	6	26.90	26.90	26.00	16.20	1.70	4.70	5.00
	7	29.30	29.30	34.10	0	0	3.30	0
	8	29.80	29.80	17.40	8.20	0	3.20	0
	9	36.00	36.00	M	M	M	2.20	M
White Blue	10	40.00	40.00	0	0	0	6.70	0
	11	50.00	50.00	49.70	0	0	8.80	0
	12	15.20	15.20	4.70	4.30	0	3.20	0
	13	16.20	M	M	M	M	1.90	M
	14	27.80	27.80	14.20	0	0	5.80	0
	15	41.00	41.00	39.30	11.10	M	3.10	M
	16	37.00	37.00	35.80	30.20	M	3.90	M
	17	44.00	44.00	8.40	8.20	0	4.00	0
	18	14.40	14.40	0	0	0	1.30	0
White Yellow	19	24.00	24.00	0	0	0	4.10	0
	20	26.30	26.30	0	0	0	2.20	0
	21	25.00	25.00	20.00	18.70	0	3.50	0
	22	35.00	35.00	0	0	0	2.70	0
	23	21.60	21.60	25.00	24.30	0	8.70	0
	24	27.30	27.30	0	0	0	3.00	0
	25	38.50	38.50	10.00	10.30	0	5.20	0
	26	33.00	33.00	0	0	0	6.00	0
	27	39.00	39.00	0	0	0	5.20	0
White Green	28	34.00	M	M	M	M	6.10	M
	29	38.70	38.70	0	0	0	5.40	0
	30	29.00	29.00	22.00	12.20	0	4.90	0
	31	31.70	31.70	0	0	0	5.00	0
	32	29.80	29.80	0	0	0	5.10	0
	33	34.30	34.30	M	M	M	4.60	M
	34	23.60	23.60	0	0	0	2.00	0
	35	27.30	27.30	9.20	9.00	0	13.30	0
	36	21.10	21.10	4.50	4.70	0	1.70	0
	MEAN	**30.71**	**31.04**	**22.22**	**13.12**	**1.70**	**4.37**	**5.00**
	s	**8.54**	**8.39**	**13.01**	**7.89**	**0.00**	**2.38**	**0.00**

Site B								
Frame ID	Eelgrass (#)	Sampling Dates					Rhizome Length (cm)	
		July 6	July 17	24 July	16 Aug	29 Aug	Initial	Final
Blue with Red	1	23.00	23.00	18.50	17.40	15.60	3.00	3.00
	2	40.50	40.50	0	0	0	4.00	0
	3	42.00	42.00	34.90	25.10	17.10	8.10	8.90
	4	35.00	35.00	28.50	0	0	6.20	0
	5	29.90	29.90	17.70	12.70	9.70	8.20	8.00
	6	26.00	26.00	21.30	18.00	17.60	10.70	10.90
	7	45.40	45.40	0	0	0	2.10	0
	8	35.60	35.60	0	0	0	2.30	0
	9	35.10	35.10	31.00	22.80	20.00	8.00	8.00
Blue and Green	10	28.30	28.30	30.00	14.60	0	9.10	0
	11	34.80	34.80	0	0	0	4.00	0
	12	34.50	34.50	23.70	16.90	5.00	7.10	7.00
	13	37.00	37.00	0	0	0	3.60	0
	14	31.30	31.10	20.60	14.60	11.10	2.50	3.30
	15	22.00	22.00	0	0	0	4.00	0
	16	38.30	38.30	36.20	30.10	0	3.20	0
	17	33.50	33.50	28.20	0	0	4.00	0
	18	18.00	18.00	M	M	M	5.20	M
Blue and Yellow	19	34.10	34.10	27.50	22.30	17.40	14.20	15.30
	20	37.40	37.40	36.80	0	0	5.00	0
	21	29.60	29.60	29.30	3.40	0	4.10	0
	22	37.90	M	M	M	M	5.20	M
	23	19.80	19.80	10.30	3.80	0	1.60	0
	24	17.00	17.00	M	M	M	6.30	M
	25	43.00	43.00	34.20	14.90	9.60	4.20	4.00
	26	36.30	36.30	34.10	33.90	0	2.00	0
	27	39.20	39.20	39.00	7.80	0	4.00	0
Blue (X2)	28	15.60	15.60	0	0	0	2.50	0
	29	34.30	34.30	30.10	0	0	6.20	0
	30	22.00	22.00	8.30	8.00	0	3.70	0
	31	34.20	34.20	0	0	0	7.20	0
	32	26.30	26.30	26.40	M	M	5.00	M
	33	26.40	26.40	13.20	13.00	12.10	5.20	6.20
	34	31.00	31.00	0	0	0	6.20	0
	35	27.80	27.80	0	0	0	4.70	0
	36	40.60	40.60	0	0	0	5.20	0
	MEAN	**31.74**	**31.56**	**26.35**	**16.43**	**13.52**	**5.22**	**7.46**
	s	**7.69**	**7.73**	**8.69**	**8.50**	**4.73**	**2.63**	**3.76**

Site C								
Frame ID	Eelgrass (#)	Sampling Dates					Rhizome Length (cm)	
		July 6	July 17	24 July	16 Aug	29 Aug	Initial	Final
Green (X2)	1	19.10	19.10	17.40	0	0	5.60	0
	2	34.00	34.00	0	0	0	4.20	0
	3	24.70	24.70	22.00	0	0	2.50	0
	4	46.60	46.60	36.80	21.70	13.00	6.80	3.00
	5	31.00	31.00	28.10	0	0	4.00	0
	6	13.60	M	M	M	M	2.80	M
	7	29.20	29.20	0	0	0	4.00	0
	8	24.50	24.50	16.40	0	0	7.70	0
	9	28.90	28.90	24.10	24.00	0	2.50	0
Yellow with Red	10	22.60	22.60	0	0	0	2.20	0
	11	25.70	25.70	0	0	0	3.10	0
	12	41.10	41.10	36.10	0	0	6.40	0
	13	25.50	25.50	0	0	0	6.30	0
	14	25.00	M	M	M	M	12.20	M
	15	38.30	38.30	0	0	0	5.90	0
	16	24.00	24.00	23.80	21.10	0	2.00	0
	17	31.30	31.30	0	0	0	8.20	0
	18	16.50	16.50	14.90	12.70	7.50	4.80	3.40
Yellow with Green	19	35.90	35.90	0	0	0	6.20	0
	20	23.20	23.20	21.70	0	0	5.10	0
	21	28.50	28.50	0	0	0	4.20	0
	22	21.00	21.00	0	0	0	5.20	0
	23	34.20	34.20	32.90	30.10	0	13.00	0
	24	25.50	25.50	0	0	0	6.00	0
	25	31.00	31.00	0	0	0	3.10	0
	26	16.60	16.60	0	0	0	1.50	0
	27	23.50	0	0	0	0	5.30	0
Clear	28	33.00	33.00	M	M	M	6.30	M
	29	23.00	23.00	23.20	24.00	24.20	11.50	12.10
	30	31.20	31.20	30.00	0	0	6.20	0
	31	4.60	4.60	0	0	0	3.50	0
	32	32.30	M	M	M	M	5.10	M
	33	28.00	28.00	24.70	23.80	0	8.20	0
	34	26.10	26.10	0	0	0	7.60	0
	35	36.60	36.60	0	0	0	4.20	0
	36	19.50	19.50	14.30	12.20	0	2.80	0
	MEAN	**27.09**	**27.53**	**24.43**	**21.20**	**14.90**	**5.45**	**6.17**
	s	**8.06**	**8.12**	**7.21**	**6.04**	**8.51**	**2.75**	**5.14**

Appendix 5: Blade abundance and width data for each individual eelgrass transplant.

Tracadie Harbour- 3 July, 2018 – 1 September, 2018

Eelgrass #	Initial (# / shoot)	Final (# / shoot)	Blade width (mm)
1	4	3	3.70
2	8	0	0
3	R	R	R
4	6	4	4.10
5	5	3	3.30
6	R	R	R
7	7	4	4.00
8	R	R	R
9	6	3	3.00
10	4	1	2.50
11	5	0	0
12	6	3	2.90
13	7	M	M
14	6	3	3.30
15	6	6	2.90
16	R	R	R
17	5	2	3.70
18	5	1	3.10
19	4	2	3.80
20	5	2	3.00
21	5	4	3.40
22	4	3	2.90
23	4	2	3.20
24	6	3	3.00
25	5	3	3.50
26	7	3	3.40
27	5	2	2.60
28	3	1	2.80
29	3	2	2.90
30	4	2	2.50
31	4	3	2.20
32	3	1	2.40
33	4	3	2.30
34	5	2	2.80
35	5	2	3.10
36	4	2	3.00
37	4	2	4.30
38	5	0	0
39	5	3	2.40
40	3	1	2.30
41	4	2	3.00

42	5	2	2.90
43	4	2	1.90
44	4	1	2.10
45	5	2	3.10
46	8	4	3.90
47	7	4	2.20
48	6	0	0
49	5	2	4.40
50	5	3	3.10
51	4	2	4.00
52	6	2	3.40
53	4	3	4.20
54	4	2	2.40
55	5	2	3.80
56	4	M	M
57	5	5	4.50
58	6	0	0
59	6	M	M
60	7	2	3.70
61	5	0	0
62	5	2	2.90
63	5	M	M
64	6	M	M
65	5	2	4.10
66	6	4	5.00
67	3	2	2.90
68	4	2	2.40
69	4	2	2.50
70	7	5	3.50
71	5	2	2.60
72	6	4	4.90
73	3	2	2.20
74	8	6	3.90
75	4	2	2.30
76	5	2	3.60
77	6	6	3.30
78	7	4	4.60
79	4	1	3.40
80	4	2	2.80
81	5	2	2.70
TOTAL	**388**	**173**	
MEAN	**5.04**	**2.64**	**3.19**
s	**1.23**	**1.18**	**0.72**

Benoit Cove- 6 July, 2018 – 1 September, 2018

Eelgrass #	Initial (# / shoot)	Final (# / shoot)	Blade Width (mm)
1	5	0	0
2	4	0	0
3	7	0	0
4	4	0	0
5	5	0	0
6	6	0	0
7	5	0	0
8	8	0	0
9	4	0	0
10	8	M	M
11	7	0	0
12	7	0	0
13	7	0	0
14	8	0	0
15	6	M	M
16	6	0	0
17	9	0	0
18	5	0	0
19	8	M	M
20	7	0	0
21	6	M	M
22	5	0	0
23	8	0	0
24	7	4	2.20
25	7	0	0
26	5	0	0
27	9	M	M
28	6	0	0
29	4	0	0
30	4	0	0
31	5	M	M
32	6	0	0
33	6	M	M
34	5	M	M
35	7	0	0
36	3	0	0
37	4	2	2.00
38	7	0	0
39	5	3	3.00
40	8	0	0
41	5	2	2.90
42	5	3	2.70

43	8	0	0
44	6	0	0
45	5	3	3.00
46	4	0	0
47	5	0	0
48	6	2	3.60
49	8	0	0
50	5	3	2.90
51	3	0	0
52	7	0	0
53	5	0	0
54	3	M	M
55	6	3	3.10
56	7	0	0
57	5	0	0
58	6	M	M
59	3	0	0
60	3	M	M
61	5	1	2.80
62	5	0	0
63	7	0	0
64	3	0	0
65	5	0	0
66	4	0	0
67	6	0	0
68	4	M	M
69	5	3	2.20
70	7	0	0
71	4	0	0
72	6	0	0
73	3	0	0
74	6	0	0
75	4	0	0
76	7	3	1.90
77	7	0	0
78	2	M	M
79	4	0	0
80	3	0	0
81	6	0	0
82	5	M	M
83	6	4	2.90
84	6	0	0
85	8	0	0
86	5	M	M
87	4	0	0
88	4	0	0

89	7	0	0
90	3	0	0
91	6	0	0
92	3	0	0
93	4	0	0
94	3	0	0
95	5	0	0
96	5	0	0
97	7	0	0
98	3	0	0
99	4	0	0
100	3	0	0
101	3	0	0
102	6	0	0
103	5	0	0
104	4	M	M
105	7	0	0
106	3	0	0
107	5	0	0
108	4	2	1.90
TOTAL	**579**	**38**	
MEAN	**5.36**	**2.62**	**2.65**
s	**1.61**	**0.77**	**0.52**

Appendix 6: F-test and two-sample *t*-test for the number of blades per shoot transplant between Tracadie Harbour and Benoit Cove.
*No significant difference at the 0.05 level (two-tailed).

F-Test Two-Sample for Variances

Initial	*Benoit Cove*	*Tracadie Harbour*
Mean	5.361111111	5.038961039
Variance	2.588006231	1.511619959
Observations	108	77
df	107	76
F	1.712074662	
P(F<=f) one-tail	0.006868081	
F Critical one-tail	1.428904564	

t-Test: Two-Sample Assuming Unequal Variances

Initial	*Benoit Cove*	*Tracadie Harbour*
Mean	5.361111111	5.038961039
Variance	2.588006231	1.511619959
Observations	108	77
Hypothesized Mean Difference	0	
df	182	
t Stat	1.542917262	
P(T<=t) one-tail	0.0622939	
t Critical one-tail	1.653269024	
P(T<=t) two-tail	0.124587801	
t Critical two-tail	1.973084077	

F-Test Two-Sample for Variances

Final	*Tracadie Harbour*	*Benoit Cove*
Mean	2.621212121	2.714285714
Variance	1.438927739	0.681318681
Observations	66	14
df	65	13
F	2.111974585	
P(F<=f) one-tail	0.0681938	
F Critical one-tail	2.289907042	

t-Test: Two-Sample Assuming Equal Variances

Final	*Tracadie Harbour*	*Benoit Cove*
Mean	2.621212121	2.714285714
Variance	1.438927739	0.681318681
Observations	66	14
Pooled Variance	1.312659563	
Hypothesized Mean Difference	0	
df	78	
t Stat	-0.276084115	
P(T<=t) one-tail	0.39160682	
t Critical one-tail	1.664624645	
P(T<=t) two-tail	0.78321364	
t Critical two-tail	1.990847069	

Appendix 7: F-test and two-sample *t*-test for the initial and final number of blades per shoot transplant.

*Significant difference at the 0.05 level (two-tailed).

F-Test Two-Sample for Variances

Tracadie Harbour	*Initial*	*Final*
Mean	5.038961039	2.621212121
Variance	1.511619959	1.438927739
Observations	77	66
df	76	65
F	1.050518326	
P(F<=f) one-tail	0.421063774	
F Critical one-tail	1.490666632	

t-Test: Two-Sample Assuming Equal Variances

Tracadie Harbour	*Initial*	*Final*
Mean	5.038961039	2.621212121
Variance	1.511619959	1.438927739
Observations	77	66
Pooled Variance	1.478109361	
Hypothesized Mean Difference	0	
df	141	
t Stat	11.8551547	
P(T<=t) one-tail	3.16418E-23	
t Critical one-tail	1.655732287	
P(T<=t) two-tail	6.32835E-23	
t Critical two-tail	1.976931489	

F-Test Two-Sample for Variances

Benoit Cove	*Initial*	*Final*
Mean	5.361111	2.714286
Variance	2.588006	0.681319
Observations	108	14
df	107	13
F	3.798525	
P(F<=f) one-tail	0.004824	
F Critical one-tail	2.257874	

t-Test: Two-Sample Assuming Unequal Variances

Benoit Cove	*Initial*	*Final*
Mean	5.361111	2.714286
Variance	2.588006	0.681319
Observations	108	14
Hypothesized Mean Difference	0	
df	28	
t Stat	9.821353	
P(T<=t) one-tail	7.15E-11	
t Critical one-tail	1.701131	
P(T<=t) two-tail	1.43E-10	
t Critical two-tail	2.048407	

Appendix 8: F-test and two-sample t-test for the final blade widths between both study sites.

*Significant difference at the 0.05 level (two-tailed).

F-Test Two-Sample for Variances

	Tracadie Harbour	*Benoit Cove*
Mean	3.189393939	2.65
Variance	0.522808858	0.270384615
Observations	66	14
df	65	13
F	1.933574723	
P(F<=f) one-tail	0.094032188	
F Critical one-tail	2.289907042	

t-Test: Two-Sample Assuming Equal Variances

	Tracadie Harbour	*Benoit Cove*
Mean	3.189393939	2.65
Variance	0.522808858	0.270384615
Observations	66	14
Pooled Variance	0.480738151	
Hypothesized Mean Difference	0	
df	78	
t Stat	2.643886214	
P(T<=t) one-tail	0.004952263	
t Critical one-tail	1.664624645	
P(T<=t) two-tail	0.009904526	
t Critical two-tail	1.990847069	

Appendix 9: Canopy height (cm) data from each transplant frame in the study sites. *Derived values from mean imputation for 28 August, 2018 (not included in mean).

Tracadie Harbour

Frame	03-Jul	19-Jul	27-Jul	14-Aug	28-Aug
1	25.78	25.78	24.10	21.90	18.00
2	28.50	28.48	26.86	25.80	24.93
3					*23.50
4					*23.13
5	25.55	25.50	22.57	21.08	19.62
6	34.04	33.40	31.62	30.44	26.00
7	30.72	30.60	28.66	27.30	24.04
8					*24.26
9	38.38	37.36	33.86	32.68	31.44
10	37.88	37.88	34.98	31.03	25.68
11	29.87	29.53	28.07	27.28	23.55
12	36.98	36.93	34.90	33.83	28.42
MEAN	31.97	31.72	29.51	27.93	24.63
s	5.04	4.60	4.60	4.24	4.10

Benoit Cove

Frame	06-Jul	17-Jul	24-Jul	16-Aug	29-Aug
1	28.78	28.78	26.13	6.10	0.43
2	40.45	40.45	24.10	3.13	0.00
3	34.56	34.56	9.17	8.88	0.00
4	32.30	32.30	8.93	6.48	0.00
5	38.93	38.93	28.93	21.97	18.23
6	35.62	35.62	31.47	20.53	8.05
7	38.98	38.98	36.03	19.73	6.75
8	32.38	32.38	8.60	3.50	2.02
9	32.40	32.40	24.13	7.62	2.17
10	30.98	30.98	12.47	5.63	1.25
11	29.77	29.72	9.10	5.02	0.00
12	30.48	30.48	23.05	15.00	6.05
MEAN	33.80	33.80	20.17	10.30	3.75
s	3.91	3.91	9.97	7.01	5.41

Appendix 10: Nested ANOVA (General Linear Model) and Tukey Comparisons test for mean canopy height between the transplant sites.

General Linear Model: Canopy Height versus Site, Subsite

Method

Factor coding	(-1, 0, +1)

Factor Information

Factor	Type	Levels	Values
Site	Fixed	2	0, 1
Subsite(Site)	Fixed	6	1(0), 2(0), 3(0), 1(1), 2(1), 3(1)

Analysis of Variance

Source	DF	Adj SS	Adj MS	F-Value	P-Value
Site	1	2554.4	2554.44	187.31	0.000
Subsite(Site)	4	212.8	53.20	3.90	0.019
Error	18	245.5	13.64		
Total	23	3012.7			

Model Summary

S	R-sq	R-sq(adj)	R-sq(pred)
3.69286	91.85%	89.59%	85.52%

Coefficients

Term	Coef	SE Coef	T-Value	P-Value	VIF
Constant	14.062	0.754	18.65	0.000	
Site					
0	10.317	0.754	13.69	0.000	1.00
Subsite(Site)					
1(0)	-1.86	1.51	-1.23	0.233	1.33
2(0)	-1.03	1.51	-0.68	0.502	1.33
1(1)	-3.64	1.51	-2.41	0.027	1.33
2(1)	5.02	1.51	3.33	0.004	1.33

Regression Equation

Canopy Height = 14.062 + 10.317 Site_0 - 10.317 Site_1 - 1.86 Subsite(Site)_1(0)
- 1.03 Subsite(Site)_2(0) + 2.89 Subsite(Site)_3(0)
- 3.64 Subsite(Site)_1(1)
+ 5.02 Subsite(Site)_2(1) - 1.38 Subsite(Site)_3(1)

Fits and Diagnostics for Unusual Observations

Obs	Canopy Height	Fit	Resid	Std Resid	
17	18.23	8.76	9.47	2.96	R
20	2.02	8.76	-6.75	-2.11	R

R Large residual

<u>Tukey Pairwise Comparisons: Subsite(Site)</u>

Grouping Information Using the Tukey Method and 95% Confidence

Subsite(Site)	N	Mean	Grouping		
3(0)	4	27.2704	A		
2(0)	4	23.3467	A		
1(0)	4	22.5188	A		
2(1)	4	8.7625		B	
3(1)	4	2.3667		B	C
1(1)	4	0.1062			C

Means that do not share a letter are significantly different.

Appendix 11: Water temperature and salinity data from the transplant sites.

Tracadie Harbour – Temperature (°C)

	03-Jul-18	19-Jul-18	27-Jul-18	14-Aug-18	28-Aug-18
	23.00	24.00	27.00	26.00	25.00
	26.00	26.50	26.50	25.50	24.50
	25.90	25.00	26.00	26.50	23.00
	26.00	25.00	27.00	26.00	23.50
	24.00	24.50	26.50	25.00	25.00
	25.00	26.00	26.00	25.50	24.00
	26.00	25.00	27.00	26.00	24.50
	25.00	25.50	27.00	26.50	23.50
	26.00	25.00	26.00	25.00	25.00
MEAN	**25.21**	**25.17**	**26.56**	**25.78**	**24.22**
s	**1.08**	**0.75**	**0.46**	**0.57**	**0.75**

Tracadie Harbour – Salinity (ppt)

	03-Jul-18	19-Jul-18	27-Jul-18	14-Aug-18	28-Aug-18
	25.00	24.00	26.00	28.00	28.50
	22.00	25.00	25.00	32.00	29.00
	22.00	25.00	25.50	29.50	28.00
	24.00	24.50	25.50	30.00	28.00
	26.00	24.00	26.00	30.00	29.00
	25.00	25.50	25.00	31.00	27.00
	23.00	24.00	26.50	29.00	27.50
	24.00	25.00	25.00	30.00	28.00
	23.00	25.00	26.00	28.50	28.50
MEAN	**23.78**	**24.67**	**25.61**	**29.78**	**28.17**
s	**1.39**	**0.56**	**0.55**	**1.23**	**0.66**

Benoit Cove – Temperature (°C)					
	06-Jul-18	**17-Jul-18**	**24-Jul-18**	**16-Aug-18**	**29-Aug-18**
	25.00	25.50	27.00	24.50	23.50
	24.00	25.00	26.50	25.50	24.00
	24.00	25.00	26.00	24.75	23.00
	25.00	25.50	27.00	25.00	24.00
	25.00	24.50	26.00	25.00	23.50
	25.00	24.50	27.00	24.50	23.00
	25.00	25.00	26.50	25.00	23.50
	24.00	25.50	26.50	24.00	24.00
	24.00	24.00	27.00	24.50	24.00
MEAN	**24.56**	**24.94**	**26.61**	**24.75**	**23.61**
s	**0.53**	**0.53**	**0.42**	**0.43**	**0.42**

Benoit Cove – Salinity (ppt)					
	06-Jul-18	**17-Jul-18**	**24-Jul-18**	**16-Aug-18**	**29-Aug-18**
	24.00	26.00	24.00	28.00	25.00
	24.00	30.00	25.00	29.00	24.00
	25.00	27.00	24.00	33.00	24.50
	22.00	26.50	24.50	31.00	24.00
	24.00	27.50	25.50	30.00	25.00
	24.00	28.50	25.00	33.00	25.00
	24.00	28.00	24.50	29.50	24.50
	24.00	29.00	24.00	28.50	26.00
	24.00	27.50	24.50	30.00	25.00
MEAN	**23.89**	**27.78**	**24.56**	**30.22**	**24.78**
s	**0.78**	**1.25**	**0.53**	**1.80**	**0.62**

Appendix 12: Sediment composition data from each site in Nova Scotia.

Tracadie Harbour- 20 July, 2018 – 7 August, 2018				
Trial #	**Time (min)**	**Sediment Type**	**Amount Settled (mL)**	**Composition (%)**
1	1	Sand	5	0.56%
	60	Silt	205	22.77%
	1440	Clay	5	0.56%
2	1	Sand	10	1.11%
	60	Silt	240	26.60%
	1440	Clay	10	1.11%
3	1	Sand	6	0.67%
	60	Silt	269	29.88%
	1440	Clay	5	0.56%
4	1	Sand	2.5	0.27%
	60	Silt	117.5	13.05%
	1440	Clay	5	0.56%
5	1	Sand	7	0.78%
	60	Silt	293	32.56%
	1440	Clay	10	1.11%
6	1	Sand	6	0.67%
	60	Silt	484	53.78%
	1440	Clay	5	0.56%
7	1	Sand	5.5	0.61%
	60	Silt	369.5	41.05%
	1440	Clay	5	0.56%
8	1	Sand	4	0.44%
	60	Silt	446	49.55%
	1440	Clay	5	0.56%
9	1	Sand	10	1.11%
	60	Silt	480	53.30%
	1440	Clay	5	0.56%
10	1	Sand	10	1.11%
	60	Silt	300	32.22%
	1440	Clay	3	0.33%
11	1	Sand	15	1.67%
	60	Silt	260	27.22%
	1440	Clay	3	0.33%
12	1	Sand	5	0.56%
	60	Silt	250	27.22%
	1440	Clay	2	0.22%

Benoit Cove- 23 July, 2018 – 16 August, 2018				
Trial #	**Time (min)**	**Sediment Type**	**Amount Settled (mL)**	**Composition (%)**
1	1	Sand	16	1.78%
	60	Silt	294	32.67%
	1440	Clay	5	0.56%
2	1	Sand	10	1.11%
	60	Silt	240	26.60%
	1440	Clay	5	0.56%
3	1	Sand	40	4.44%
	60	Silt	200	22.22%
	1440	Clay	5	0.56%
4	1	Sand	30	3.33%
	60	Silt	250	38.88%
	1440	Clay	3	0.33%
5	1	Sand	20	2.22%
	60	Silt	240	26.67%
	1440	Clay	5	0.56%
6	1	Sand	25	2.78%
	60	Silt	275	30.56%
	1440	Clay	10	1.11%
7	1	Sand	24	2.67%
	60	Silt	250	25.11%
	1440	Clay	5	0.56%
8	1	Sand	15	1.67%
	60	Silt	260	27.22%
	1440	Clay	5	0.56%
9	1	Sand	10	1.11%
	60	Silt	260	27.78%
	1440	Clay	3	0.33%
10	1	Sand	5	0.56%
	60	Silt	275	30.00%
	1440	Clay	4	0.44%
11	1	Sand	10	1.11%
	60	Silt	260	27.78%
	1440	Clay	5	0.56%
12	1	Sand	20	2.22%
	60	Silt	240	24.44%
	1440	Clay	3	0.33%

Grand Étang Estuary- 8 August, 2018				
Trial #	**Time (min)**	**Sediment Type**	**Amount Settled (mL)**	**Composition (%)**
1	1	Sand	60	6.67%
	60	Silt	150	16.67%
	1440	Clay	10	1.11%
2	1	Sand	75	8.33%
	60	Silt	190	12.78%
	1440	Clay	5	0.56%
3	1	Sand	55	6.11%
	60	Silt	145	10.56%
	1440	Clay	5	0.56%
4	1	Sand	92	10.22%
	60	Silt	130	14.44%
	1440	Clay	5	0.56%
5	1	Sand	95	10.55%
	60	Silt	155	17.22%
	1440	Clay	5	0.56%
6	1	Sand	91	10.11%
	60	Silt	109	12.11%
	1440	Clay	3	0.33%

Antigonish Harbour- 23 October, 2018				
Trial #	**Time (min)**	**Sediment Type**	**Amount Settled (mL)**	**Composition (%)**
1	1	Sand	120	13.33%
	60	Silt	120	13.33%
	1440	Clay	1	0.11%
2	1	Sand	140	15.56%
	60	Silt	98	10.89%
	1440	Clay	0.75	0.08%
3	1	Sand	150	16.67%
	60	Silt	95	10.56%
	1440	Clay	0.5	0.06%
4	1	Sand	125	13.89%
	60	Silt	80	8.89%
	1440	Clay	3	0.33%
5	1	Sand	130	14.44%
	60	Silt	100	11.11%
	1440	Clay	5	0.56%
6	1	Sand	140	15.56%
	60	Silt	80	8.89%
	1440	Clay	4	0.44%

Appendix 13: Levene's test and One-way ANOVA analysis to assess clay compositions between Tracadie Harbour, Benoit Cove, Grand Étang Estuary and Antigonish Harbour.

Test for Equal Variances: Clay versus Site

Method

Null hypothesis	All variances are equal
Alternative hypothesis	At least one variance is different
Significance level	$\alpha = 0.05$

95% Bonferroni Confidence Intervals for Standard Deviations

Site	N	StDev	CI
0	12	0.0027285	(0.0012715, 0.0073941)
1	12	0.0020670	(0.0007170, 0.0075250)
2	6	0.0026013	(0.0004531, 0.0255845)
3	6	0.0021062	(0.0008326, 0.0091276)

Individual confidence level = 98.75%

Test

Method	Test Statistic	P-Value
Multiple comparisons	—	0.936
Levene	0.21	0.888

One-way ANOVA: Clay versus Site

Method

Null hypothesis	All means are equal
Alternative hypothesis	Not all means are equal
Significance level	$\alpha = 0.05$

Equal variances were assumed for the analysis.

Factor Information

Factor	Levels	Values
Site	4	0, 1, 2, 3

Analysis of Variance

Source	DF	Adj SS	Adj MS	F-Value	P-Value
Site	3	0.000050	0.000017	2.89	0.050
Error	32	0.000185	0.000006		
Total	35	0.000235			

Model Summary

S	R-sq	R-sq(adj)	R-sq(pred)
0.0024038	21.34%	13.97%	0.44%

Means

Site	N	Mean	StDev	95% CI
0	12	0.005850	0.002728	(0.004437, 0.007263)
1	12	0.005383	0.002067	(0.003970, 0.006797)
2	6	0.00613	0.00260	(0.00413, 0.00813)
3	6	0.002635	0.002106	(0.000636, 0.004634)

Pooled StDev = 0.00240377

Appendix 14: Levene's test, Welch's ANOVA and Games-Howell post-hoc test analysis to assess the silt compositions between Tracadie Harbour, Benoit Cove, Grand Étang Estuary and Antigonish Harbour.

*Tracadie Harbour (0), Benoit Cove (1), Grand Étang (2), Antigonish Harbour (3)

Test for Equal Variances: Silt versus Site

Method

Null hypothesis	All variances are equal
Alternative hypothesis	At least one variance is different
Significance level	α = 0.05

95% Bonferroni Confidence Intervals for Standard Deviations

Site	N	StDev	CI
0	12	0.127531	(0.0799378, 0.256939)
1	12	0.043518	(0.0187991, 0.127218)
2	6	0.026297	(0.0117750, 0.100613)
3	6	0.016531	(0.0061763, 0.075803)

Individual confidence level = 98.75%

Tests

Method	Test Statistic	P-Value
Multiple comparisons	—	0.000
Levene	5.20	0.005

One-way ANOVA: Silt versus Site

Method

Null hypothesis	All means are equal
Alternative hypothesis	Not all means are equal
Significance level	α = 0.05

Equal variances were not assumed for the analysis.

Factor Information

Factor	Levels	Values
Site	4	0, 1, 2, 3

Welch's Test

Source	DF Num	DF Den	F-Value	P-Value

Site	3	16.4607	56.29	0.000

Model Summary

R-sq	R-sq(adj)	R-sq(pred)
60.24%	56.51%	52.45%

Means

Site	N	Mean	StDev	95% CI
0	12	0.3410	0.1275	(0.2600, 0.4220)
1	12	0.2833	0.0435	(0.2556, 0.3109)
2	6	0.1396	0.0263	(0.1120, 0.1672)
3	6	0.10612	0.01653	(0.08877, 0.12347)

Games-Howell Pairwise Comparisons

Grouping Information Using the Games-Howell Method and 95% Confidence

Site	N	Mean	Grouping	
0	12	0.3410	A	
1	12	0.2833	A	
2	6	0.1396		B
3	6	0.10612		B

Means that do not share a letter are significantly different.

Appendix 15: Levene's test, Welch's ANOVA and Games-Howell post-hoc test analysis to assess the sand compositions between Tracadie Harbour, Benoit Cove, Grand Étang Estuary and Antigonish Harbour.

Test for Equal Variances: Sand versus Site

Method

Null hypothesis	All variances are equal
Alternative hypothesis	At least one variance is different
Significance level	$\alpha = 0.05$

95% Bonferroni Confidence Intervals for Standard Deviations

Site	N	StDev	CI
0	12	0.0038626	(0.0019377, 0.0097232)
1	12	0.0110373	(0.0058649, 0.0262314)
2	6	0.0193285	(0.0066999, 0.0955267)
3	6	0.0124014	(0.0055546, 0.0474337)

Individual confidence level = 98.75%

Tests

Method	Test Statistic	P-Value
Multiple comparisons	—	0.014
Levene	7.08	0.001

One-way ANOVA: Sand versus Site

Method

Null hypothesis	All means are equal
Alternative hypothesis	Not all means are equal
Significance level	$\alpha = 0.05$

Equal variances were not assumed for the analysis.

Factor Information

Factor	Levels	Values
Site	4	0, 1, 2, 3

Welch's Test

Source	DF Num	DF Den	F-Value	P-Value

Site	3	11.0399	245.45	0.000

Model Summary

R-sq	R-sq(adj)	R-sq(pred)
95.96%	95.58%	94.55%

Means

Site	N	Mean	StDev	95% CI
0	12	0.00796	0.00386	(0.00551, 0.01042)
1	12	0.02083	0.01104	(0.01382, 0.02785)
2	6	0.08665	0.01933	(0.06637, 0.10693)
3	6	0.14909	0.01240	(0.13607, 0.16210)

Games-Howell Pairwise Comparisons

Grouping Information Using the Games-Howell Method and 95% Confidence

Site	N	Mean	Grouping			
3	6	0.14909	A			
2	6	0.08665		B		
1	12	0.02083			C	
0	12	0.00796				D

Means that do not share a letter are significantly different.

Appendix 16: Sediment organic matter content data collected from the sites in Nova Scotia.

Tracadie Harbour- 15 August, 2018 – 28 August, 2018							
Trial #	Cup Weight (g)	Total Wet Weight (g)	Wet Weight (g)	Dry Weight (g)	Final Weight (g)	Organic Matter (g)	Percent OM (%)
1	67.168	119.081	51.913	15.698	8.122	7.576	48.26%
2	67.323	116.522	49.199	14.210	6.990	7.220	50.81%
3	67.217	127.740	60.523	10.469	7.941	2.528	24.15%
4	67.348	123.220	55.872	11.287	7.846	3.441	30.49%
5	64.635	123.573	58.938	12.652	9.559	3.093	24.45%
6	70.655	126.120	55.465	11.082	8.355	2.727	24.61%
7	68.018	126.202	58.184	11.537	8.389	3.148	27.29%
8	67.481	123.208	55.728	12.419	8.172	4.248	34.20%
9	67.209	124.399	57.190	12.127	8.212	3.915	32.28%
Mean	**67.450**	**123.341**	**55.890**	**12.387**	**8.176**	**4.211**	**32.95%**
s	**1.530**	**3.555**	**3.509**	**1.648**	**0.668**	**1.887**	**10.08%**

Benoit Cove- 15 August, 2018 – 28 August, 2018							
Trial #	Cup Weight (g)	Total Wet Weight (g)	Wet Weight (g)	Dry Weight (g)	Final Weight (g)	Organic Matter (g)	Percent OM (%)
1	68.922	128.917	59.995	26.079	21.565	4.514	17.31%
2	67.180	122.545	55.365	22.426	17.800	4.626	20.63%
3	67.500	103.832	36.332	14.120	11.710	2.410	17.07%
4	68.291	121.192	52.901	21.296	18.527	2.769	13.00%
5	66.912	130.309	63.397	34.288	31.814	2.474	7.22%
6	66.639	129.128	62.489	34.497	32.573	1.924	5.58%
7	67.270	129.420	62.150	23.431	21.452	1.979	8.45%
8	69.122	128.216	59.094	23.408	21.167	2.241	9.57%
9	67.730	124.195	56.465	24.201	22.189	2.012	8.31%
Mean	**67.730**	**124.195**	**56.465**	**24.861**	**22.089**	**2.772**	**11.90%**

s	**0.873**	**8.321**	**8.331**	**6.342**	**6.565**	**1.055**	**5.31%**

Antigonish Harbour- 23 October, 2018 – 27 October, 2018							
Trial #	Cup Weight (g)	Total Wet Weight (g)	Wet Weight (g)	Dry Weight (g)	Final Weight (g)	Organic Matter (g)	Percent OM (%)
1	66.718	148.574	81.86	55.66	54.96	0.70	1.26%
2	67.715	157.848	90.13	63.23	62.04	1.18	1.87%
3	68.838	155.467	86.63	61.35	60.24	1.11	1.81%
4	66.975	146.945	79.97	55.28	54.07	1.21	2.19%
5	66.017	147.710	81.69	55.73	53.99	1.74	3.12%
6	66.594	149.363	82.77	56.62	56.24	0.38	0.67%
7	67.313	148.967	81.65	56.20	55.46	0.74	1.32%
8	66.056	147.433	81.38	55.88	54.31	1.57	2.81%
9	67.798	146.122	78.32	54.72	53.16	1.56	2.85%
Mean	**67.11**	**149.83**	**82.71**	**57.18**	**56.05**	**1.13**	**1.99%**
s	**0.91**	**4.05**	**3.57**	**2.98**	**3.05**	**0.46**	**0.83%**

Appendix 17: Levene's test, Welch's ANOVA and Games-Howell post-hoc test analysis to assess the organic matter content in sediment samples from Tracadie Harbour, Benoit Cove and Antigonish Harbour.

Test for Equal Variances: Organic Matter versus Sites

Method

Null hypothesis	All variances are equal
Alternative hypothesis	At least one variance is different
Significance level	α = 0.05

95% Bonferroni Confidence Intervals for Standard Deviations

Sites	N	StDev	CI
0	9	0.100805	(0.0392187, 0.352998)
1	9	0.053090	(0.0314364, 0.122149)
2	9	0.008291	(0.0051881, 0.018053)

Individual confidence level = 98.3333%

Tests

Method	Test Statistic	P-Value
Multiple comparisons	—	0.000
Levene	4.60	0.020

One-way ANOVA: Organic Matter versus Sites

Method

Null hypothesis	All means are equal
Alternative hypothesis	Not all means are equal
Significance level	α = 0.05

Equal variances were not assumed for the analysis.

Factor Information

Factor	Levels	Values
Sites	3	0, 1, 2

Welch's Test

Source	DF Num	DF Den	F-Value	P-Value
Sites	2	10.9904	53.59	0.000

Model Summary

R-sq	R-sq(adj)	R-sq(pred)
81.17%	79.60%	76.16%

Means

Sites	N	Mean	StDev	95% CI
0	9	0.3295	0.1008	(0.2520, 0.4070)
1	9	0.1190	0.0531	(0.0782, 0.1598)
2	9	0.01989	0.00829	(0.01352, 0.02626)

Games-Howell Pairwise Comparisons

Grouping Information Using the Games-Howell Method and 95% Confidence

Sites	N	Mean	Grouping		
0	9	0.3295	A		
1	9	0.1190		B	
2	9	0.01989			C

Means that do not share a letter are significantly different.