MAPPING EELGRASS AND ASSESSING ANTHROPOGENIC STRESSORS IN PLACENTIA BAY AND TRINITY BAY, NEWFOUNDLAND AND LABRADOR, CANADA

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Abstract

Seagrasses are declining globally at an accelerating rate. A lack of information on seagrass condition and spatial extent is hindering seagrass monitoring, conservation, and management. This study provides a baseline of eelgrass' (*Zostera marina*) extent at several sites in Placentia Bay and Trinity Bay, Newfoundland and Labrador, Canada, providing a reference point for future eelgrass monitoring. Eelgrass was mapped using high-resolution UAV imagery and underwater video. The presence of physical disturbance and eutrophication was assessed using visual analysis of both imagery sources. Additionally, changes in eelgrass landscape structure were assessed in areas with and without the invasive European green crab (*Carcinus maenas*). The total area of eelgrass mapped was approximately 1km^2 across seven sites in Placentia Bay and three sites in Trinity Bay. There were few indications of physical disturbance and eutrophication across the study sites. The site with the largest change in eelgrass (a loss of 98.9% of eelgrass area between 2014 and 2020) in Placentia Bay also had the greatest abundance of beach-washed European green crab carapaces. Observations agree with previous research that has shown that eelgrass in Placentia Bay is predominantly threatened by the invasive European green crab.

General Summary

Seagrasses are plants that form large underwater meadows in the oceans. Many animals rely on seagrasses to live. Unfortunately, seagrasses are declining globally from human activities. Many countries, including Canada, lack detailed maps that would help conserve seagrasses. This research seeks to provide information for seagrass conservation in Placentia Bay, an area where seagrass is declining, and Trinity Bay, Newfoundland and Labrador, Canada. Drone images and underwater videos were used to map eelgrass meadows and assess signs of nutrient pollution and human disturbance. Both can be harmful to eelgrass. Additionally, the changes to eelgrass meadows due to the European green crab, an invasive crab that can harm eelgrass, was studied to better understand impacts to eelgrass fragmentation and area loss. There were few signs of human impacts on eelgrass, but the results suggest green crabs removed large amounts of eelgrass at one study site. The results of this research can be used to inform seagrass conservation in Placentia Bay and Trinity Bay.

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List of Abbreviations

5-FCV	Five Fold Cross Validation
ATV	All-Terrain Vehicle
AWMPAR	Area Weighted Mean Perimeter Area Ratio
CBC	Come By Chance
DFA	Department of Fisheries and Aquaculture
DFO	Department of Fisheries and Oceans Canada
DJI	Da-Jiang Innovations
GBP	Great Barasway Pond
GCP	Glennons Cove Pond
GPS	Global Positioning System
LD	Landscape Division
OA	Overall Accuracy
OBIA	Object-Based Image Analysis
OSP	Old Shop Pond
PLS	Placentia Swans
SEP	Spread Eagle Pond
SHB	Ship Harbour
SOH	Southern Harbour
SUN	Sunnyside
UAV	Unoccupied Aerial Vehicle
WPS	Western Placentia Southeast Arm

Co-Authorship Statement

The work presented in this thesis was conducted by Aaron Sneep with guidance from Rodolphe Devillers, Evan Edinger, Katleen Robert, and Arnault Le Bris. Fieldwork was organized by Aaron Sneep with guidance from Katleen Robert, Evan Edinger, and Rodolphe Devillers. Aaron Sneep, with aid from field assistants, collected UAV imagery and conducted transect surveys of beach-washed European green crab carapaces. Aaron Sneep and Evan Edinger collected underwater video for use in groundtruthing and to assess the semi-quantitative abundance of epiphytes. Aaron Sneep was responsible for the design and analysis in Chapter 2 and Chapter 3, with guidance from Rodolphe Devillers and Evan Edinger. The chapters were written by Aaron Sneep, with editorial and intellectual input from Rodolphe Devillers, Evan Edinger, Katleen Robert, and Arnault Le Bris. Chapter 2 has been prepared for submission to Estuarine and Coastal Shelf Science (ECSS) under the co-authorship of Aaron Sneep, Rodolphe Devillers, Katleen Robert, Arnault Le Bris and Evan Edinger. Chapter 3 has been prepared for submission to Continental Shelf Research under the co-authorship of Aaron Sneep, Rodolphe Devillers, Katleen Robert, Arnault Le Bris, and Evan Edinger.

Chapter 1 - Introduction and Overview

1.1 Introduction

Seagrasses, a group of 72 marine angiosperms, form complex habitats along the coasts of all continents, except Antarctica (Green and Short 2004). Seagrasses are ecosystem engineers (Gutiérrez et al. 2011). They create a positive feedback by reducing hydrodynamic energy, which increases sediment deposition and increases water clarity, resulting in conditions that favour seagrass growth (Bos et al. 2007, van der Heide et al. 2011). Seagrass beds may range from extensive, continuous beds, to a mosaic of fragmented patches (Robbins and Bell 1994).

There is greater species diversity in seagrass beds than unvegetated habitats (Edgar et al. 1994, Jenkins et al. 1997). The three-dimensional structure of seagrass meadows provides protection from predators (Canion and Heck 2009). Due to the increased protection from predators and availability of food resources, juveniles from a variety of culturally and economically important fish species use seagrasses as nursery habitat, such as juvenile cod (*Gadus spp.*) and Pacific Salmon (*Oncorhynchus spp.*) (Heck Jr. et al. 2003, Lilley and Unsworth 2014, Kennedy et al. 2018). A variety of herbivorous waterfowl, urchins, gastropods, crustaceans, fishes, reptiles, and mammals use seagrasses as a food source (Valentine and Heck 1999). Many threatened species use seagrass habitat at some stage of their life cycle, such as green sea turtle (*Chelonia mydas*), dugong (*Dugong dugon*), and dwarf seahorse (*Hippocampus zosterae*) (Orth et al. 2006, Hughes et al. 2009).

Seagrass ecosystems also benefit humans by providing a variety of ecosystem services, and as such are considered one of the World's most valuable ecosystems (Costanza and D'Arge 1997). It is estimated that 20% of the World's 25 largest fisheries are supported by nursery habitat provided by Seagrasses (Unsworth et al. 2019b). Sediment deposition is greater within seagrass meadows resulting in improved water quality (van der Heide et al. 2011). Shorelines are stabilized through the reduction of wave energy and retention of sediments within seagrass meadows, which reduce coastal erosion (Ondiviela et al. 2014). Seagrass ecosystems have high rates of primary production (Duarte and Chiscano 1999) and sequester carbon at a rate over 30 times faster than tropical rainforests (McLeod et al. 2011). As such, these ecosystems have been suggested as major carbon sinks with implications for addressing climate change (Fourqurean et al. 2012, Duarte et al. 2013).

Despite their importance, there is a lack of recognition for seagrass habitats and their ecosystem services in many parts of the world (Cullen-Unsworth et al. 2014). This lack of recognition is one of the major challenges facing seagrass ecosystems (Unsworth et al. 2019a). There is less media attention directed towards seagrasses compared to more charismatic ecosystems, such as mangroves, despite comparable amounts of research attention given to each of these ecosystems, and research indicating that seagrasses are rapidly declining globally (Duarte et al. 2008). Seagrass ecosystems declined at a rate of 1.5%yr^{-1}, between 1879 and 2006, a rate accelerating in recent decades to 7%yr^{-1} since 1990 (Waycott et al. 2009). This decline has been attributed to a variety of anthropogenic activities (Orth et al. 2006).

Globally, seagrasses are mostly threatened by changes in water quality that reduce light availability (Grech et al. 2012). Eutrophication leads to algal blooms and epiphyte over-growth which have direct impacts on light availability (Burkholder et al. 2007). Eutrophication in seagrass ecosystems is commonly linked to nutrient pollution from terrestrial sources, such as urban and agricultural runoff (Grech et al. 2012). Similarly, anthropogenic activities that increase water turbidity, such as dredging, negatively impact seagrass by limiting light availability (Erftemeijer and Robin Lewis 2006).

The physical disturbance of many anthropogenic activities is responsible for losses of seagrasses. Direct physical disturbance or removal of seagrasses occurs at broad and fine scales. Coastal development, the construction of coastal infrastructure, can remove seagrass over large areas (e.g., Bull et al. 2010, Unsworth et al. 2018) and has been ranked as the most prominent threat to seagrass in the temperate North Atlantic bioregion (Grech et al. 2012). At a fine scale, a variety of anthropogenic activities can result in physical disturbance, including but not limited to: mooring scars (e.g., Unsworth et al. 2017), propeller scars (e.g., Orth et al. 2017), off road vehicle tracks (e.g., Martin et al. 2008), and trawling (e.g., Kiparissis et al. 2011). Each individual disturbance typically influences a small area, but the same process may occur over a large area with high frequency, creating a substantial cumulative impact (e.g., Eriander et al. 2017, Unsworth et al. 2017, Glasby and West, 2018).

Non-native species may also threaten seagrass ecosystems (Orth et al. 2006). Non-native algae, invertebrates, seagrasses, birds, and fish have been reported in seagrass ecosystems, with invertebrates and algae being the most common invaders (Williams 2007). The impacts of non-native species on seagrass ecosystems are mainly negative and may act through a variety of mechanisms, some of which include: direct competition, preemptive competition, fouling, herbivory, and bioturbation (Williams 2007).

Climate change, encompassing warming and ocean acidification, along with relative sea-level rise will have a variety of impacts on seagrass populations (Short et al. 2016). Rising water temperatures result in reduced photosynthesis and increased respiration as temperatures exceed thermal tolerances of seagrass species (Collier et al. 2011). Species distributions are projected to expand poleward and contract from the tropics in response to warming (Hyndes et al. 2016, Wilson and Lotze 2019). High summer water temperatures in Chesapeake Bay, USA, have already resulted in losses of temperate seagrass species (Moore et al. 2012). Sea-level rise can result in shoreward migration of seagrass meadows, with limited light availability at depth combined with expansion as coastal areas become submerged (Short and Neckles 1999). In the Mediterranean, contraction of the deep edge of *Posidonia oceanica* meadows has been attributed to sea-level rise (Pergent et al. 2015). Lastly, ocean acidification may have positive effects on seagrasses due to an increased availability of dissolved $C0_2$ resulting in increases to above and below ground biomass (Garrard and Beaumont 2014).

Improvements to water quality has proven to be effective in promoting recovery of degraded seagrass ecosystems (Lefcheck et al. 2018, de los Santos et al. 2019). For instance, Lefcheck et al. (2018) found submerged aquatic vegetation in Chesapeake Bay, United States increased 316% since 1984 following a 23% reduction in nitrogen concentrations. However, multiple stressors may impact seagrass communities simultaneously, requiring more holistic approaches to management efforts (Orth et al. 2006, Krause-Jensen et al. 2020).

There is a lack of information regarding the status and condition of many seagrass ecosystems, which was identified as a major challenge for seagrass conservation (Unsworth et al. 2019a). Many regions of the globe lack spatial data delineating the distribution of seagrass (McKenzie et al. 2020), and indicators of seagrass ecosystem health are spatially and temporally limited (Unsworth et al. 2019a). The major threats to seagrasses are well understood with only slight differences across seagrass bioregions (Grech et al. 2012). However, depending on local anthropogenic activities and environmental settings, threats at a local scale may differ from common stressors known to affect seagrasses over broad geographic scales. Management efforts would benefit from understanding threats to seagrass ecosystems at the local scale to better target management efforts (Unsworth et al. 2019a).

In Canada, there is a deficit of information regarding seagrass condition and spatial extent (McKenzie et al. 2020, Murphy et al. 2021). Canada is in the temperate North Atlantic seagrass bioregion, where common threats to seagrass include agricultural runoff, urban runoff, dredging, and coastal development (Grech et al. 2012, Murphy et al. 2021). Within Canada, anthropogenic threats to seagrass vary regionally and locally (Murphy et al. 2021). For instance, forestry activities on the Pacific coast have a greater impact on seagrass, whereas in Atlantic Canada seagrass is primarily threatened by nutrient loading (Murphy et al. 2021). At sub-region or local scales, threats may also vary based on the source of the stressor (e.g., nutrient pollution from aquaculture, agricultural runoff, urban runoff) or the impact of the stressor (Murphy et al. 2021). For instance, the island of Newfoundland is in Atlantic Canada, but Newfoundland eelgrass meadows receive lower anthropogenic nutrient inputs (Murphy et al. 2021). Although threats to seagrass in Newfoundland may differ from other regions of Canada, such data is limited on the island of Newfoundland.

Eelgrass (*Zostera marina*) is considered an ecologically significant species in Atlantic Canada due to its role as an ecosystem engineer (DFO 2009). Current information on eelgrass is limited on the island of Newfoundland to specific locations where studies of eelgrass or associated fauna have occurred and point observations indicating eelgrass presence or likely presence (Rao et al. 2014). Research involving eelgrass in Newfoundland has predominantly focused on the relationship between Juvenile Cod (*Gadus* sp.), a culturally and economically important fish species in Newfoundland, and the role of eelgrass role as nursery habitat (e.g., Gotceitas et al. 1997, Linehan et al. 2001, Laurel et al. 2003, Robichaud and Rose 2006, Schneider et al. 2008, Gorman et al. 2009, Thistle et al. 2010, Warren et al. 2010). This research has provided some information on eelgrass extent and dynamics. For instance, eelgrass cover in Newman Sound, a fjord on the east coast of the island, expanded between 1998 and 2006 (Warren et al. 2010). These studies and other eelgrass data sources have been incorporated into a comprehensive map indicating the presence of eelgrass or the likely presence of eelgrass along the coast of the island of Newfoundland (Rao et al. 2014). However, these point observations do not provide an indication of eelgrass condition or spatial extent, essential information for eelgrass conservation. In some areas of Newfoundland, the condition of eelgrass has been directly assessed, such as Placentia Bay, but without spatial data delineating eelgrass meadow extent (e.g., Matheson et al. 2016).

Eelgrass in Placentia Bay is declining, which has been attributed to the invasive European green crab (*Carcinus maenas*) (Matheson et al. 2016). Bioturbation caused by foraging and burrowing activities of the invasive European green crab decreases eelgrass biomass (Davis et al. 1998, Malyshev and Quijón 2011, Garbary et al. 2014, Neckles 2015). These observations have been linked to eelgrass loss over broad spatial scales (Garbary et al. 2014, Matheson and Mckenzie 2014, Neckles, 2015). The first observation of the European green crab in Newfoundland was in North Harbour, Placentia Bay, in 2007, with initial arrival estimated to be several years earlier, in approximately 2002 (Blakeslee et al. 2010). Subsequently, the European green crab has spread predominantly along the south and west coast of Newfoundland, while other regions are yet to be colonized (DFO 2011).

Despite impacts documented in other parts of the world (Burkholder et al. 2007, Orth et al. 2017, Kelly et al. 2019), there has been no assessment of other anthropogenic stressors, such as eutrophication and physical disturbance that may negatively impact eelgrass in Placentia Bay.

These stressors may vary regionally and locally. For instance, untreated sewage is discharged from coastal communities in Placentia Bay and may accumulate in sheltered embayments (DFA 2007). Additionally, mooring areas have been constructed in areas where eelgrass occurs or is likely to occur. A variety of boating-related activities may result in disturbance to eelgrass, which may be substantial over large areas. The impacts of mooring areas on aquatic vegetation are generally negative, but the response is highly variable, with an increased abundance in some mooring areas (Sagerman et al. 2020). Additionally, anthropogenic activity in Placentia Bay is expected to increase with the development of aquaculture and increasing vessel traffic (DFA 2007).

There is a need to quantify a baseline extent of eelgrass in Placentia Bay and to assess the presence and impacts of anthropogenic stressors. Management efforts and monitoring of future environmental change in Placentia Bay may benefit from assessing the presence of eutrophication and physical disturbance at local scales. Additionally, there is a lack of spatial data providing a baseline extent of eelgrass beds for monitoring efforts to assess temporal and spatial dynamics of eelgrass beds. Despite the declining status of eelgrass due to the European green crab, there has been no spatial assessment of eelgrass extent and how green crab disturbance affects eelgrass landscape structure. Understanding how green crabs influence the spatial pattern of eelgrass landscapes may help management efforts identify priority areas for eelgrass habitat protection and green crab removal efforts.

1.2 Research questions

To address the research gaps described above, I will answer the following questions in my thesis:

1. what is the spatial extent of eelgrass beds in Placentia Bay and Trinity Bay?

2. what is the presence of common anthropogenic stressors known to negatively impact eelgrass, specifically physical disturbance and eutrophication, in eelgrass beds of Placentia Bay and Trinity Bay?

3. how has eelgrass landscape structure changed over recent years in areas with and without the European green crab?

1.3 Research goal and objectives

The goal of this research is to provide a baseline distribution of eelgrass in Placentia Bay and Trinity Bay, NL, as well as local context to anthropogenic stressors that may be impacting eelgrass in this region. The specific objectives of this research are to:

1. delineate the current extent of eelgrass in several embayments of Placentia Bay and Trinity Bay;

2. provide local context to the presence of physical disturbance and eutrophication;

3. assess changes in eelgrass landscape structure in areas with and without the invasive European green crab.

1.4 Method summary

1.4.1 Eelgrass mapping

Unmanned Aerial Vehicle (UAV) imagery and archival aerial images were used to map eelgrass beds in Placentia Bay and Trinity Bay, with groundtruthing conducted using underwater video. An Object Based Image Analysis (OBIA) approach was applied to both image sets. Two different classification approaches were applied to the image segments. In Chapter 2, the random forest algorithm was applied to the UAV imagery alone, and in Chapter 3 a manual classification approach was applied to both the UAV imagery and archival aerial photos.

1.4.2 Anthropogenic stressors

The presence of physical disturbance and proliferation of epiphytes, an indication of nutrient enrichment, was visually assessed using the UAV aerial imagery and underwater groundtruth videos.

1.4.3 Landscape structure

Landscape ecology concepts for measuring fragmentation with landscape pattern metrics were used to assess changes in eelgrass landscape structure. Additionally, the abundance of beach-washed European green crab carapaces was used to estimate the relative abundance of green crabs among sites where changes in landscape structure were assessed.

1.5 Thesis outline

Chapter 2 examines the spatial extent of eelgrass in several embayments of Placentia Bay and Trinity Bay and provides local context into the impact of common anthropogenic stressors affecting eelgrass in Placentia Bay. Chapter 3 examines the impact of the invasive European green crab on eelgrass landscape structure by comparing changes in eelgrass area and landscape fragmentation metrics in areas with green crab (i.e., Placentia Bay) and areas without (i.e., Trinity Bay). Finally, Chapter 4 concludes the thesis, summarizing the main findings, highlighting how the findings can inform eelgrass management in Placentia Bay, and proposing future research directions.

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Chapter 2 - Mapping and characterizing eelgrass beds using UAV imagery in Placentia Bay and Trinity Bay, Newfoundland and Labrador, Canada

Abstract

Detailed information on the fine scale distribution of seagrass habitats and their stressors is required to inform management efforts but is lacking in many regions of the globe. Eelgrass (*Zostera marina*), a species of seagrass found throughout the Northern Hemisphere, has been declining in Placentia Bay, an ecologically significant area of Canada's East coast with an increasing human impact. This research provides baseline information, acting as a reference point for eelgrass monitoring, on the distribution of eelgrass beds and their anthropogenic stressors at seven sites of Placentia Bay and three sites of the adjacent Trinity Bay. Unmanned Aerial Vehicle (UAV) imagery, analysed using an Object-Based Image Analysis (OBIA) approach, was used to create high-spatial-resolution eelgrass distribution maps. Visual analyses of the imagery and underwater videos were conducted to characterize sites based on the presence of physical disturbances and semi-quantitative abundance of epiphytes, an indication of nutrient enrichment. A total eelgrass area of ~1 km^2 was mapped across the 10 sites. User's accuracy (error of commission) and producer's accuracy (error of omission) values for the eelgrass class ranged from 48.8% to 95.1% and 23.3% to 99.2%, respectively. Results indicate rare signs of physical disturbance and eutrophication affecting eelgrass in the region, likely due to the small population size of the communities near the eelgrass beds. These baseline data will inform eelgrass conservation efforts and enable future monitoring of temporal trends.

2.1 Introduction

Seagrasses are marine plants that grow along the shorelines of all continents, except Antarctica (Green and Short 2004). Seagrasses form highly productive ecosystems supporting a wide diversity of marine organisms, including: invertebrates, fishes, reptiles, mammals, and waterfowl (Green and Short 2004). Seagrasses are considered among the most valuable ecosystems globally (Costanza and D'Arge 1997). They provide key ecosystem services in coastal regions, supporting fisheries (Nordlund et al. 2018, Unsworth et al. 2019b), protecting from coastal erosion (Paul 2018), and acting as carbon sinks (Duarte et al. 2013, Fourqurean et al. 2012). Seagrasses are considered biological sentinels (Nordlund et al. 2016) and have been adopted as bioindicators for numerous ecosystem health monitoring programs in a variety of locations, including in Europe, the Mediterranean Sea, and the Caribbean Sea to name a few (e.g., Borum et al. 2004, Güreşen et al. 2020, Kerninon et al. 2021, Martínez-Crego et al. 2008).

Seagrasses are, however, threatened by a variety of anthropogenic stressors, including eutrophication, coastal development, sea level rise, physical disturbances (e.g., propeller scarring, trawling, anchor damage), and increased water turbidity (Orth et al. 2006). As a result, they were found to decline globally between 1879 and 2006 at ~1.5%yr^{-1} (Waycott et al. 2009). While seagrass has been shown to recover following management efforts, such as improving water quality (de los Santos et al. 2019, Lefcheck et al. 2018), seagrass ecosystems are often impacted by multiple stressors (Orth et al. 2006). For instance, Krause-Jensen et al. (2020) assessed that seagrass of the Western Baltic Sea, following mitigation of eutrophication, did not return to its historic levels due to the additional impact of bottom trawling and increased water temperature. This highlights the need for a holistic approach, targeting multiple stressors acting simultaneously rather than individual stressors, for seagrass management.

Information on seagrass distribution would help monitor and manage those ecosystems (McKenzie et al. 2020, Unsworth et al. 2019a). In addition, threats to seagrass ecosystems should be better understood at a local scale (i.e., an eelgrass meadow) to target management efforts effectively (Murphy et al. 2019, Unsworth et al. 2019a). An understanding of threats at local scales can also be used to inform management efforts over a broader area (e.g., Placentia Bay).

Canada, in particular, has been identified as containing extensive seagrass areas (estimated at ~24,170 km^2), but with a lack of spatial information quantifying the distribution (McKenzie et al. 2020, Murphy et al. 2021). Eelgrass has been shown to be declining on both the east and west coasts of Canada (DFO 2009, Murphy et al. 2021, Nahirnick et al. 2020). On Canada's east coast, eelgrass of Placentia Bay in the Province of Newfoundland and Labrador is in decline (Matheson et al. 2016). Across 17 sites in Placentia Bay, average eelgrass percent cover along transects in 2012 was half that observed in 1998 (Matheson et al. 2016). These declines have been largely attributed to the European green crab (*Carcinus maenas*) an invasive species that was first observed in this region in 2007, which reduces eelgrass biomass while burrowing or foraging in soft sediments (Blakeslee et al. 2010, Davis et al. 1998, Matheson et al. 2016). In contrast, eelgrass in other areas along the east coast of the Island of Newfoundland not colonized by green crabs has been reported to be expanding (Warren et al. 2010). Previous eelgrass mapping efforts on the Island of Newfoundland consist mostly of point observations indicating where eelgrass is present or is likely to occur, providing a valuable first estimate in documenting eelgrass locations (Rao et al. 2014). However, this does not quantify or document the spatial extent of eelgrass, which would be valuable in terms of baseline data for monitoring temporal changes in eelgrass extent.

Eelgrass in Placentia Bay could also be impacted by other anthropogenic stressors. Many of the coastal communities along the shores of Placentia Bay do not have modern forms of wastewater secondary treatment, wastewater being typically discharged in the ocean with little or no treatment (DFA 2007). Sewage has been reported to accumulate in some sheltered areas of Placentia Bay (Catto et al. 1999, DFA 2007). Aquaculture and wastewater have been shown to negatively impact seagrasses through eutrophication (Cullain et al. 2018, Jones et al. 2018). In addition, throughout Placentia Bay and the Island of Newfoundland, there are mooring areas with varying degrees of boating intensity, including single docks used for recreational boating, harbours for small fishing vessels, and a shipyard in Marystown. Boating-related activities have been shown to have a negative impact on seagrasses, with damages caused by propeller scars (Hallac et al. 2012, Orth et al. 2017), mooring scours (Glasby and West 2018, Unsworth et al. 2017), dock shading (Eriander et al. 2017, Gladstone and Courtenay 2014), and anchoring (Kelly et al. 2019, La Manna et al. 2015). While the individual disturbances caused by boating-related activities are typically small, an accumulation of these disturbances over a larger region may be substantial (e.g., Unsworth et al. 2017).

This study provides baseline data on the distribution of eelgrass in a region where eelgrass is known to be declining but there has been no quantification of eelgrass extent or prevalence of common anthropogenic stressors. The objectives of this study were 1) to delineate the spatial extent of seven eelgrass beds in Placentia Bay and three eelgrass beds in Trinity Bay, and 2) to provide local context to the occurrence and impact of eutrophication and physical disturbance to eelgrass beds at these sites.

2.2 Methods

2.2.1 Study sites

Study sites were selected based on previous knowledge of eelgrass presence as observed by Matheson et al. (2016) and Rao et al. (2014). The tidal conditions throughout the study region are microtidal, expect for the heads of some shallow embayments such as Come By Chance (Catto et al. 2003). The tidal conditions from available stations on the eastern coast of Placentia Bay and Trinity Bay are presented in Table 2.1. To help understand the prevalence of anthropogenic stressors, five sites were selected in areas having limited human development in the surrounding area, while five other sites were selected in areas experiencing greater levels of human activities where anthropogenic disturbances (e.g., anchor damage, propeller scaring, dock shading) or nutrient pollution could be expected. Sites with higher human presence were located along the shorelines of communities where outfall pipes or docking infrastructure for motorboats were present. Sites with lower human presence had minimal presence of infrastructure except for roads allowing vehicle access. Seven of those study sites were selected in Placentia Bay, and three sites were selected in Trinity Bay. Sites were selected in Trinity Bay to provide baseline data on eelgrass extent outside the current invasion range of the European green crab, to enable future monitoring efforts to assess differences in trends between these two bays. The distribution of field sites is presented in Figure 2.1.

2.2.2 Aerial image collection and processing

Aerial imagery was collected using a Da-Jiang Innovations (DJI) Mavic 2 Professional unmanned aerial vehicle (UAV) from August to September 2020. The UAV image collection method was informed by Nahirnick et al. (2019b) and was conducted when the Sun’s angle was less than 40° and, when possible, cloud cover was <10% or >90%, specifically targeting overcast or clear sky days. DJI's Ground Station Pro app v.2.0.12 was used to plan and conduct the UAV surveys. The survey was designed to obtain images with a forward and lateral overlap of 80% obtained at an altitude of 120 m, resulting in images at about 2.8 cm/pixel resolution. Images were captured at a flight speed of 5 m/s using the hover and capture flight mode. For georeferencing the orthomosaics during image processing, the positions of at least seven Ground Control Points (GCPs), distributed as evenly as possible throughout each field site, were collected with a Garmin eTrex 20x (~3 m accuracy) global positioning system (GPS) using the waypoint averaging function until a sample confidence of 100% was achieved. Two GPS readings were collected for each GCP at least 90 minutes apart to increase positional accuracy.

Table 2.1. Maximum tidal range predictions for 2022 at five tidal stations on the eastern coast of Placentia Bay and one tidal station in Trinity Bay.

Station	High high water	Low low water	Tidal range	Date (yyyy-mm-dd)	Bay name
Hearts Content	1.4 m	0.1 m	1.4 m	2022-06-16	Trinity Bay
Come by Chance	2.6 m	0.2 m	2.4 m	2022-08-13	Placentia Bay
Arnolds Cove	2.7 m	0.4 m	2.3 m	2022-05-17	Placentia Bay
Long Harbour	2.7 m	0.4 m	2.3 m	2022-08-13	Placentia Bay
Argentia	2.5 m	0.3 m	2.2 m	2022-08-13	Placentia Bay
St. Brides	2.5 m	0.3 m	2.2 m	2022-08-13	Placentia Bay

Note: Data available from https://tides.gc.ca/en/stations (accessed March 14, 2022).

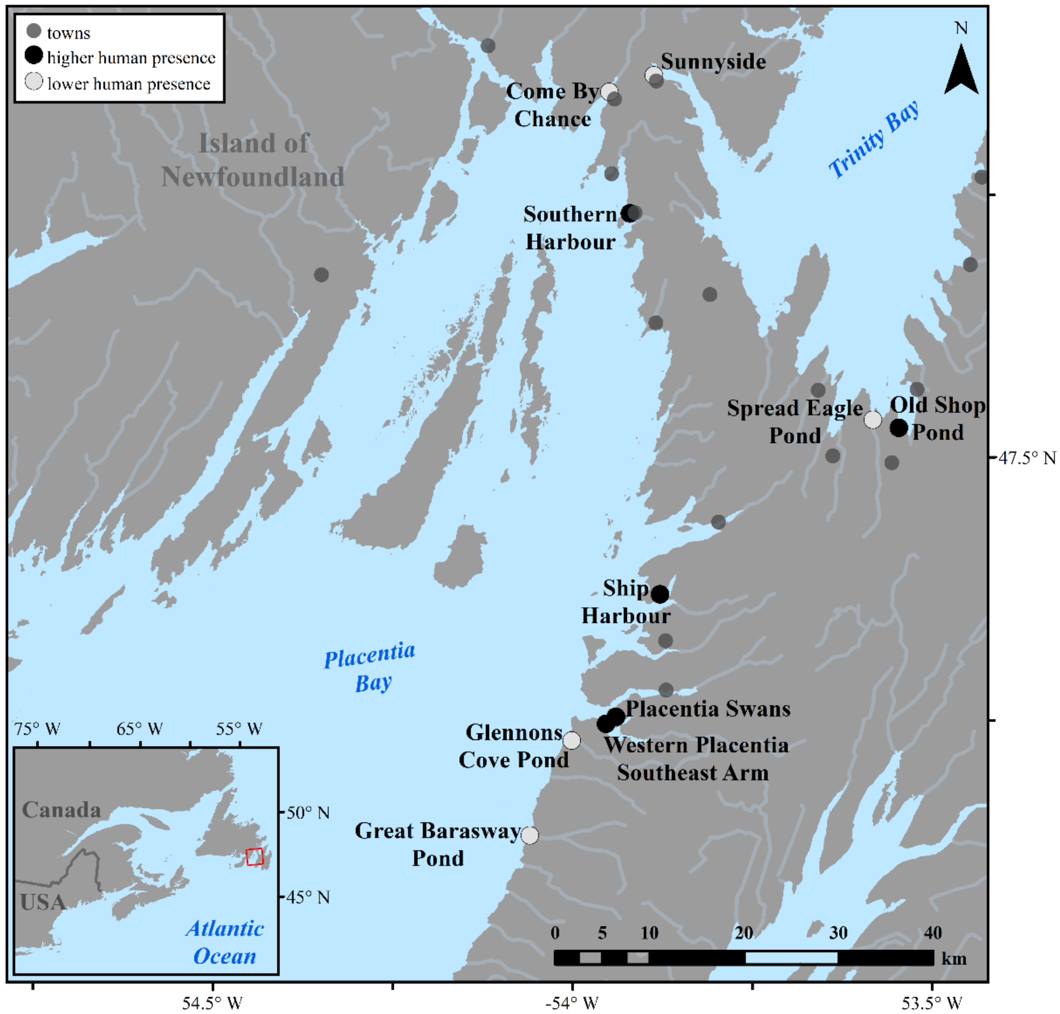

Figure 2.1. Study sites for eelgrass mapping in Placentia Bay and Trinity Bay, Newfoundland and Labrador, Canada.

Agisoft Metashape Professional v.1.6.1 (Agisoft LLC 2019) was used to create orthomosaics of the UAV imagery. All land and anthropogenic features (docks, boats, etc.) were manually masked from each site in ArcGIS 10.7 (ESRI 2019) and the orthomosaics were resampled to a spatial resolution of 25 cm to improve the processing time of image classification.

2.2.3 Groundtruth data

Based on visual inspection of the orthomosaics of each study site, underwater video collection was planned such that reference data was distributed throughout all areas of the field sites to collect reference data of as many cover types as possible. Underwater videos were collected using a Sony FDR-X3000 ActionCam. The camera was operated in an underwater housing from a slow-moving 2-person kayak, while the Garmin eTrex 20X GPS recorded the position of the camera operator. The time stamps of the GPS coordinates were matched in post-processing with the video data by recording a few seconds of a digital clock synchronized with the GPS time at the start or end of each video. Improving on the limitations identified by Nahirnick *et al*. (2019a) for collecting groundtruth data with a camera attached to the bottom of a kayak, the underwater camera was affixed to an Unger 2.5-5 m aluminum telescopic pole. The pole height was adjusted by the camera operator to position the camera near the target features directly under the kayak.

The underwater videos were used in conjunction with photo interpretation to generate a series of training data points for each field site (Table 2.1). Transects were buffered at 3m in ArcGIS, corresponding to the horizontal accuracy of the GPS, and observations were made at least every 5 seconds along the underwater video transects by creating point features within the buffer. Point features were created when a cover type could be identified in both the underwater video and the orthomosaic. Care was taken to position the point features within a uniform cover type and away from edges or transitions.

2.2.4 OBIA and classification

Object-Based Image Analysis (OBIA) is commonly employed to improve the classification accuracy of high-resolution imagery through the process of image segmentation (Blaschke, 2010). The R statistics software package SegOptim was used for image segmentation and classification (Gonçalves et al. 2019). SegOptim uses a genetic algorithm, a machine learning algorithm that emulates the process of natural selection, to determine the optimal parameter settings for the segmentation algorithm and can interface with six third-party software to conduct image segmentation. SegOptim's segmentation_ArcGIS_Mshift function was used to conduct image segmentation due to the comparable performance of the algorithm to others available in SegOptim (Gonçalves et al. 2019) and the wide use of ArcGIS as a GIS software. The mean shift segmentation method used by SegOptim's segmentation_ArcGIS_Mshift function (Comaniciu and Meer 2002) uses three parameters: spatial detail, spectral detail, and minimum segment size. To improve the efficiency of the genetic algorithm, it is important to constrain the parameter space to avoid poor solutions and to improve computation time (Gonçalves et al. 2019). Details of delineating the parameter space are provided in Appendix A.

Classification of the image segments was conducted using SegOptim random forest classifier with the default parameters (ntree = 250, mtry = 2). The random forest classifier was selected for its performance compared to other machine learning classifiers (Belgiu and Drăgu 2016) and was frequently the best preforming classification algorithm for use in SegOptim (Gonçalves et al. 2019).

Training and validation for the random forest classifier were conducted using five-fold cross validation. Millard and Richardson (2015) identified that training data for use in random forest classification should be as large as possible, have a random distribution or class proportions that reflect the actual proportions of the classes on the landscape, and minimal spatial autocorrelation. A dataset with these attributes improves classification results and will reducing model overfitting (Millard and Richardson 2015). To produce a dataset with these attributes, a dataset of 300 observations for each site was deemed the largest possible due to project constraints. To avoid model overfitting, the 300 data points consisted of a combination of randomly sampled transect observations and randomly distributed photo interpretation points. Such combination of ground truth data and photo interpretation points has been used previously in UAV mapping research (e.g., Ellis et al. 2020, Papakonstantinou et al. 2020). These high-resolution images were shown to be interpreted accurately by trained photo interpreters, removing the reliance on field observations (Chabot et al. 2018). A random sample of 100 transect observations, separated by at least 5 m, was taken from the transect data points in R (R Core Team 2020). In ArcGIS, 200 points, separated by at least 5 m, were randomly generated within the study boundary of each site and outside of the 3 m video transect buffers. These two datasets were merged to create a file of observation presences. A file of observed absences was generated from cells in a grid, with a 5m cell size, that did not contain one of the 300 observations. The presence and absence datasets were merged and spatial autocorrelation between observation presence and absence was assessed throughout the study site using a series of 10 global Moran's I tests (Moran 1950), with incrementally increasing threshold distances. Ten threshold distances were used to assess spatial autocorrelation across a range of scales. Threshold distances for the Moran's I tests started at the minimum distance so that each observation had at least one neighbor, followed by 10 m and increasing increments of 5 m up to 50 m. If the distribution of the observations exhibited significant clustering or dispersion (p-value <0.05) at any of these distance thresholds, a new dataset was generated, and the process was iterated. If subsequent resamples exhibited significant clustering or dispersion, the proportion of transect data in the data set was reduced by increments of 25 (i.e., 75 randomly sampled transect observations and 225 randomly distributed photo interpretation points) and the process was iterated until a dataset was produced that had a random distribution at all threshold distances.

Image classification was conducted using the mean and standard deviation of the red, green, and blue values for the image segments (e.g., Chabot et al. 2018, Csillik 2017, Ellis et al.

2020, Husson et al. 2016, Oldeland et al. 2021, Wahidin et al. 2015). However, for some orthomosaics the segmentation parameters were manually specified due to the genetic algorithm being unable to reach an optimal segmentation that did not exhibit under segmentation. Under-segmentation is an issue in OBIA that occurs when image objects encompass multiple target features (e.g., a patch of eelgrass and a patch of algae are in one image object when they should be two separate objects). The maximum amount of detail (i.e., maximum spatial and spectral detail, with a minimum segment size of one pixel) was required to produce a segmentation that was not under-segmenting. In these instances, the classification was conducted using the mean red, green, and blue values for the image segments alone, as a standard deviation cannot be calculated for image segments with one pixel.

The classification scheme for each site varied depending on the cover types present at each site. Each classification scheme started with three classes: bare sediment, eelgrass, and algae. Additional classes (i.e., shadows, optically deep water, green algae, spume, detritus, turbid water, shells) were added depending on their presence in the randomly sampled transect observations or the randomly distributed photo interpretation points. Photo interpretation points were assigned a class corresponding to the 25 cm resolution orthomosaics. The 2.8 cm resolution images were referenced during this process to aid with photo interpretation.

2.2.5 Classification accuracy

Classification accuracy was assessed using SegOptim by calculating Cohen's Kappa coefficient, overall accuracy, eelgrass class producer's accuracy, and eelgrass class user's accuracy (hereafter called accuracy metrics). Overall accuracy is the percent of correctly classified image objects out of the total sample. Eelgrass class producer's accuracy is the commission error, or the accuracy of how often the eelgrass observations are correctly classified on the map. Eelgrass class user's accuracy is the omission error, or the accuracy of how often eelgrass areas classified in the map will be present on the ground. Kappa values were interpreted following the agreement categories outlined by Sim and Wright (2005) (≤0 = poor, 0.1-0.20 = slight, 0.21-0.40 = fair, 0.41-0.60 = moderate, 0.61-0.80 = substantial, and 0.81-1.0 = almost perfect).

The accuracy of the maps was assessed using 5-fold cross validation (5-FCV). To increase the likelihood of generating a permutation with an observation in each fold, additional observations were added via photo interpretation to any class that had less than 10 observations until 10 observations were achieved. Accuracy metrics were calculated for each fold when it was acting as the validation sample. The final accuracy metrics for each site were obtained from the confusion matrices for each fold by calculating the mean and standard deviation of the accuracy metrics.

2.2.6 Presence of anthropogenic stressors

Sites were characterized through the semi-quantitative abundance of epiphytes, an indication of nutrient enrichment (Nelson 2017), and the occurrence and nature of physical disturbance. At sites with a proliferation of epiphytes, the semi-quantitative abundance of epiphytes was visually assessed using the underwater videos. The percent of eelgrass blade surface in the field of view covered by epiphytes was recorded using a 6-point scale (0%, 1-20%, 21-40%, 41-60%, 61-80%, 81-100%). The duration of video time for each cover category was recorded and proportion of video length for each cover category was calculated.

The nature of physical disturbances was determined using photo interpretation from the UAV imagery by looking for disturbance patterns indicative of common sources of physical disturbance in seagrass meadows (i.e., propellor scars are long narrow disturbances in shallow water in areas of boat traffic, anchor scars are typically circular disturbances, and mooring buoys typically cause a roughly circular disturbance around the mooring anchor). During field visits, when time allowed, underwater video was collected to groundtruth potential anthropogenic disturbances identified in the UAV imagery. The area of disturbance was estimated using ArcGIS by manually delineating disturbances observed in the orthomosaics.

2.3 Results

2.3.1 Eelgrass distribution

Maps delineating the distribution of eelgrass at each of the 10 study sites were generated (Figure 2.2, Appendix B). The areal extent of individual study sites ranged from 0.1747 km^2 to 0.3631 km^2. The total areal extent of eelgrass at those 10 sites was estimated to be just over 1 km^2. The mean and median area of eelgrass across the 10 sites was 0.1073 km^2 and 0.0801 km^2, respectively. The largest eelgrass meadow was observed in western Placentia Southeast Arm (Figure 2.2.e), with 0.3331 km^2 of eelgrass. The smallest eelgrass meadow was observed in Glennons Cove Pond (Figure 2.2.f), with 0.0013 km^2 of eelgrass.

2.3.2 Map accuracy assessment

The mean and standard deviation of the classification accuracy assessments metrics from 5-fold cross validation are presented in Table 2.2. Kappa values ranged from 0.22 to 0.81. Western Placentia Southeast Arm and Placentia Swans had “slight” and “fair” agreement levels respectively. Placentia Swans (Figure 2.2.d) was the worst preforming classification, ranking poorly across all accuracy assessment metrics. Western Placentia Southeast arm had the lowest average Kappa value, the overall accuracy, eelgrass class producer’s accuracy, and eelgrass class user’s accuracy had high average values of 86.3%, 99.2%, and 87.5% respectively. The other eight sites showed a “substantial” or better level of agreement, with Great Barasway Pond

(Figure 2.2.c) producing the most accurate classification. Average overall accuracy values ranged from 69.7% to 89.3%. Average eelgrass class producer's accuracy ranged from 23.3% to 99.2% and average eelgrass class user's accuracy ranged from 48.8% to 95.1%. Sites with a lower area of eelgrass generally had the lowest eelgrass class accuracy metrics; the Placentia Swans site was an exception to this, with a moderate amount of eelgrass cover and a poor classification accuracy (28.8%).

Table 2.2. Accuracy assessments of eelgrass distribution maps and the area of eelgrass per site.

Study site	Mean Kappa	Overall Accuracy (%)	Eelgrass Producer's Accuracy (%)	Eelgrass User's Accuracy (%)	Eelgrass area (km^2)
Come By Chance Gut	0.71 ± 0.08	82.2 ± 4.9	89.9 ± 9.0	87.9 ± 7.3	0.1310
Glennons Cove Pond	0.67 ± 0.08	82.2 ± 3.5	60.0 ± 43.5	73.3 ± 43.5	0.0013
Great Barasway Pond	0.81 ± 0.04	89.3 ± 2.8	98.4 ± 2.3	95.1 ± 0.4	0.0597
Old Shop Pond	0.73 ± 0.10	82.0 ± 6.9	88.5 ± 6.5	89.2 ± 9.7	0.0687
Placentia Swans	0.46 ± 0.07	69.7 ± 4.1	56.0 ± 5.2	62.1 ± 8.3	0.0916
Ship Harbour	0.72 ± 0.08	82.2 ± 5.1	91.7 ± 7.3	77.2 ± 10.9	0.0587
Southern Harbour	0.61 ± 0.15	75.8 ± 9.9	23.3 ± 14.9	48.8 ± 36.6	0.0054
Spread Eagle Pond	0.65 ± 0.14	84.7 ± 6.3	95.4 ± 3.8	90.0 ± 3.7	0.1967
Sunnyside	0.69 ± 0.10	80.3 ± 6.3	89.8 ± 5.6	81.0 ± 5.2	0.1265
Western Placentia Southeast Arm	0.22 ± 0.23	86.3 ± 3.1	99.2 ± 1.1	87.5 ± 3.0	0.3331

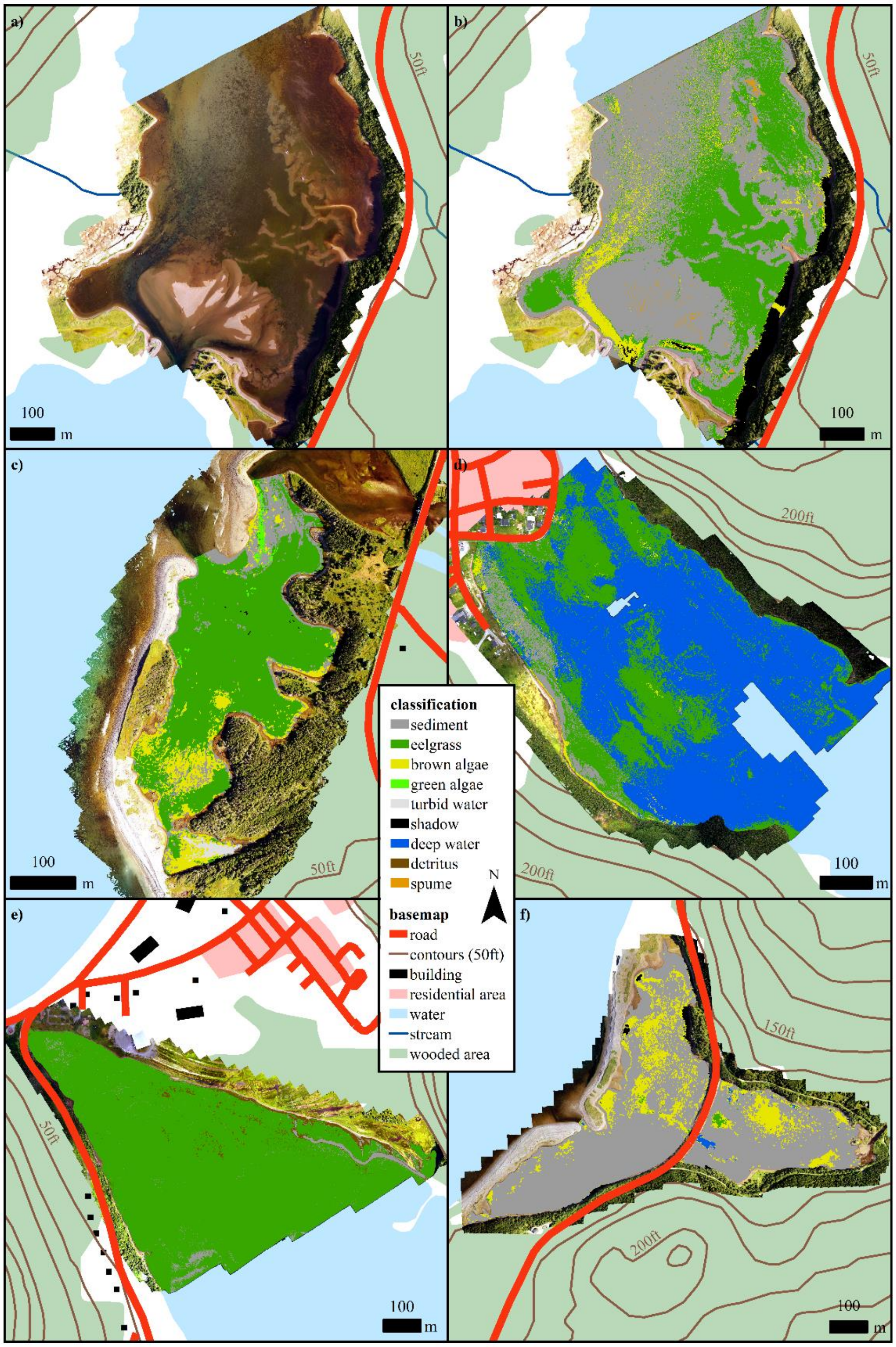

Figure 2.2. Orthomosaic aerial image (**a**) and classified map (**b**) of Come By Chance Gut. Maps of Great Barasway Pond (**c**), Placentia Swans (**d**), Western Placentia Southeast Arm (**e**), and Glennons Cove Pond (**f**).

2.3.3 Anthropogenic disturbances

Five occurrences of anthropogenic physical disturbances were identified from the aerial imagery, caused by three activities: all-terrain vehicles (ATVs) driving in eelgrass areas, buoys, and boat anchoring (Figure 2.3). Physical disturbances affected an approximate area of 132.4 m^2. Relative to the total area of eelgrass mapped, the observed total area of disturbance is inconsequential (0.013%). The area of eelgrass disturbance by activity and relative to the total area of eelgrass is presented in Figure 2.4.

Indications of ATV use were observed at all sites. Relative to the prevalence of ATV use at the sites, disturbances rarely occurred in areas of eelgrass. ATV tracks were common throughout the sites, particularly concentrated on beaches and deltas, away from eelgrass. ATV tracks caused disturbance to eelgrass at Come by Chance Gut and Old Shop Pond. ATV tracks were observed in approximately 87.6 m^2 of eelgrass, corresponding to a disturbance area of 0.008% of total eelgrass area.

Twelve buoys were observed in eelgrass areas, five of them marking fishing gear while seven others were associated with docking infrastructure. Two buoys associated with docking infrastructure at Old Shop Pond and Ship Harbour created disturbances due to ropes dragging in the sediment with a combined area of ~21 m^2, corresponding to a disturbance area of 0.002% of total eelgrass area. The other buoys did not create visible disturbances.

Anchoring within the eelgrass meadow was observed in the field at Ship Harbour. Eelgrass in the surrounding area varied in density with some barren patches. The anchoring disturbance did not however create a disturbance characteristic of anchoring damage. The area disturbance associated with anchoring was estimated to be 23.8 m^2, corresponding to a disturbance area of 0.002% of the total eelgrass area.

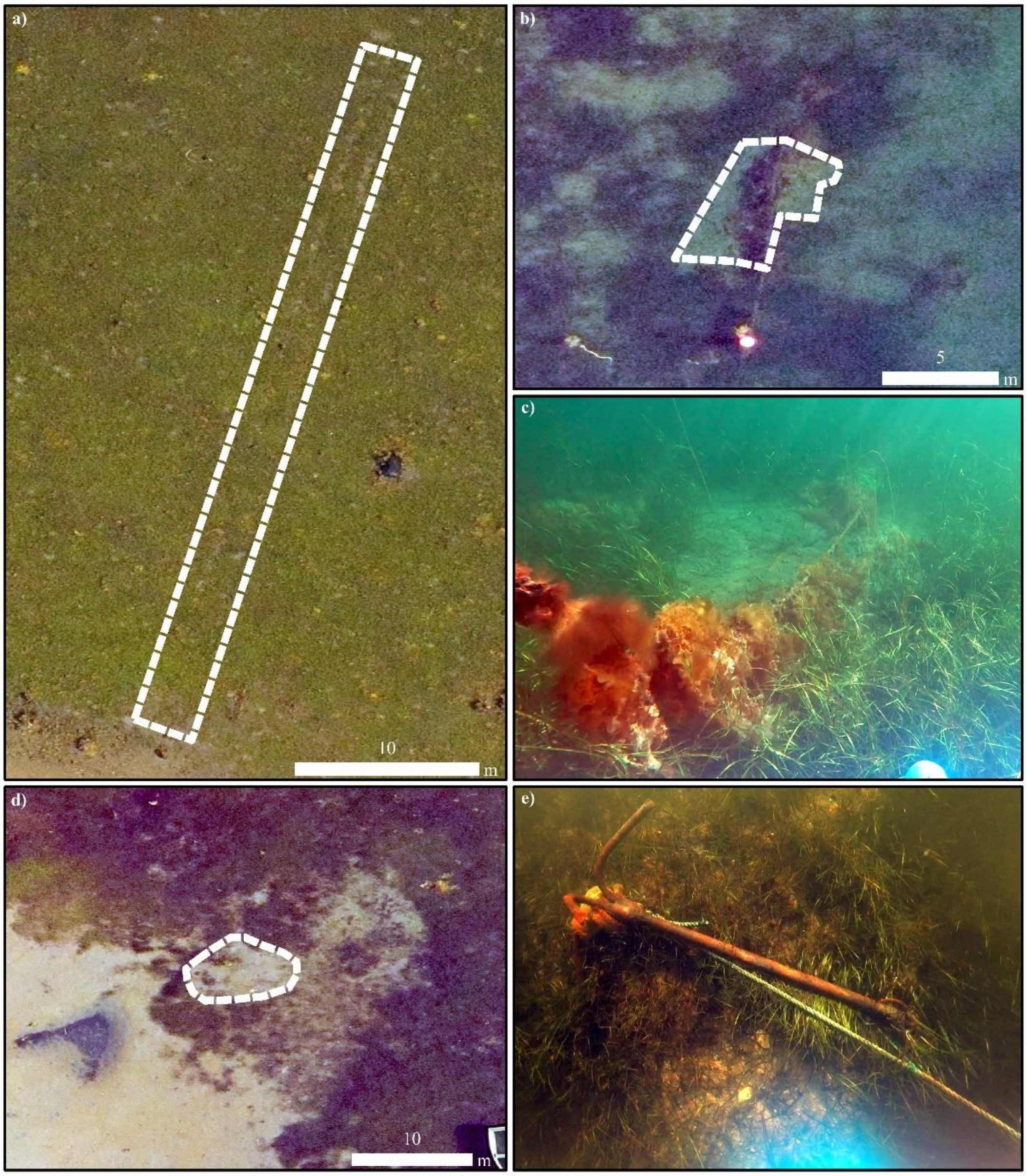

Figure 2.3. **a)** All-terrain vehicle (ATV) disturbance area at Come By Chance, **b)** buoy disturbance area at Ship Harbour, **c)** underwater image of buoy rope disturbance at Old Shop Pond, **d)** anchor disturbance area at Ship Harbour, **e)** underwater image of anchor in eelgrass bed at Ship Harbour.

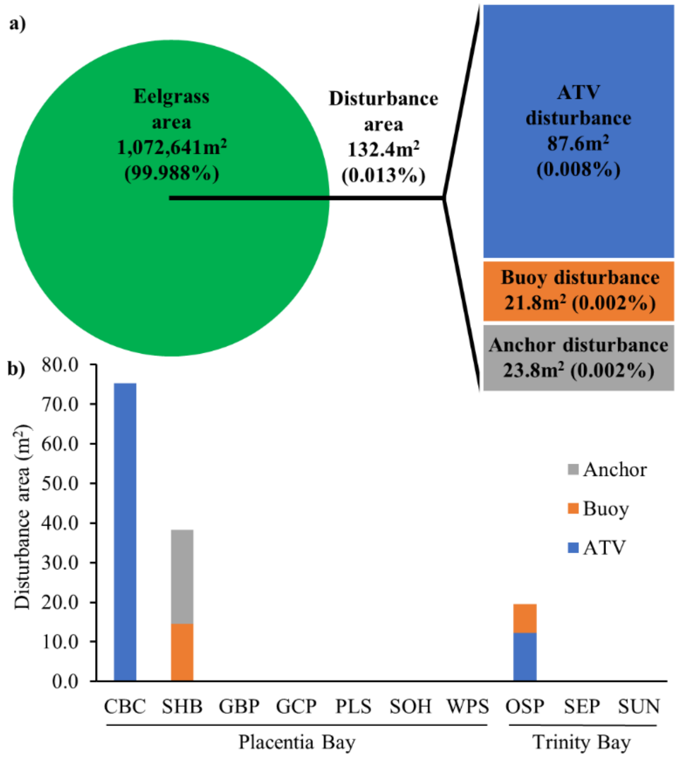

Figure 2.4. **a)** Extent of disturbance area relative to the total area of eelgrass mapped. **b)** Disturbance area at each site caused by anthropogenic activities. No disturbances were observed at GBP, GCP, PLS, SEP, SOH, SUN, and WPS. CBC: Come By Chance Gut, GBP: Great Barasway Pond, GCP: Glennons Cove Pond, PLS: Placentia Swans, OSP: Old Shop Pond, SEP: Spread Eagle Pond, SHB: Ship Harbour, SOH: Southern Harbour, SUN: Sunnyside, WPS: western Placentia Southeast Arm.

2.3.4 Semi-quantitative epiphyte abundance

Placentia Swans was the only site to have a large presence of epiphytes. When eelgrass was present in the video, approximately 27% of the video duration had no epiphyte cover, 49% had between 1% - 20% epiphyte cover, and 24% had over 20% epiphyte cover (half of it with 60% - 80% epiphyte cover). The presence of epiphyte cover categories when eelgrass was present in the groundtruth video for Placentia Swans is presented in Figure 2.5. At all other sites, the presence of epiphytes was minimal. However, at western Placentia Southeast Arm, tannin-rich freshwater inputs from heavy rainfall in the days preceding video collection reduced video quality, and potentially reduced epiphyte detectability. At western Placentia Southeast Arm algal mats were observed during field visits (Figure 2.5.c). Algal mats are an indication of nutrient enrichment and are harmful to eelgrass when persistent (Gustafsson and Boström 2014).

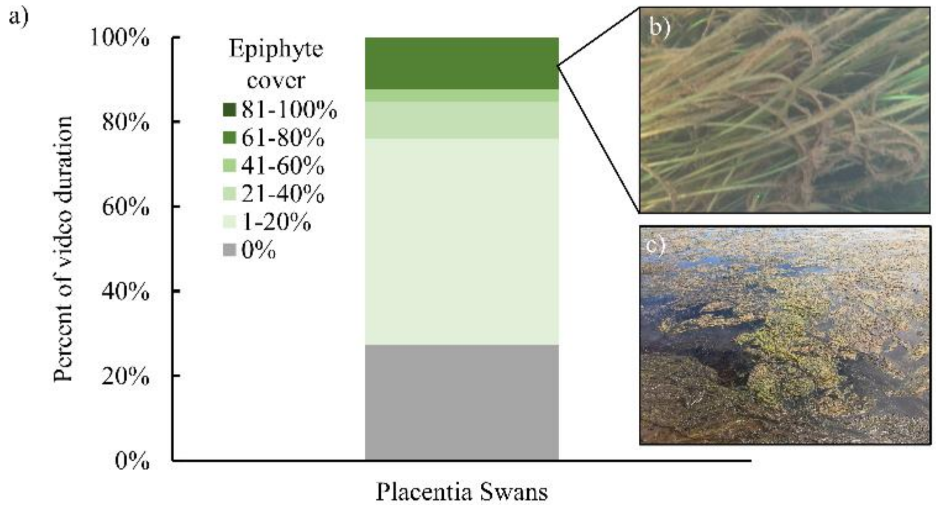

Figure 2.5. a) Percent video duration of epiphyte cover categories, measured as the amount of eelgrass in the video frame covered by epiphytes, when eelgrass was present in the groundtruth video at Placentia Swans. b) a screenshot of the groundtruth video corresponding to the 61-80% epiphyte cover category. c) example of floating algal mats observed during field visits to western Placentia Southeast Arm.

2.4 Discussion

The study allowed mapping the distribution of eelgrass in seven sites of Placentia Bay and three sites of Trinity Bay, offering geographic baseline data for the spatial extent of eelgrass in the Island of Newfoundland. There were very few physical disturbances detected across the sites, and only one site had a proliferation of epiphytes, an indication of nutrient enrichment (Table 2.3). These results suggest that there is minimal impact of physical disturbance and eutrophication affecting eelgrass in Placentia Bay and Trinity Bay.

2.4.1 Classification Accuracy

Eight of the ten classification maps produced substantial or better levels of agreement. The two lowest performing classifications likely resulted from variable cloud cover during image collection causing variable illumination between images, resulting in “streaky” orthomosaics (Figure B.5 and B.10). Variable illumination within the same cover class resulted in mis-classifications with other classes. For instance, variable radiometry of the eelgrass class at Placentia Swans resulted in misclassification between eelgrass and optically deep water (Table B.5.).

To a smaller degree, thematic map accuracy may also have been affected by the results of image segmentation. Lourenço et al. (2021) found the accuracy of thematic vegetation maps produced using ArcGIS’ mean shift segmentation were worse than that of eCognition, a common proprietary software used for OBIA, and Orfeo Toolbox/Monteverdi (OTB), an open source OBIA software. For instance, the difference between overall accuracy (OA) was small between ArcGIS and OTB with values of 84.3% and 87.0%, respectively, with a larger difference when compared to eCognition (OA = 95.7%) (Lourenço et al. 2021). Similarly, for multiclass thematic vegetation maps of UAV imagery, Gonçalves et al. (2019) found that ArcGIS produced a kappa of 0.78 compared to 0.85 for RGISLib’s Shepherd segmentation algorithm and a kappa of 0.96, the highest kappa value, for single-class thematic vegetation maps. Perhaps image segmentation conducted with an alternative segmentation software would result in slight improvements to thematic map accuracy assessments. Such improvements would likely not result in increases to the agreement categories outlined by Sim and Wright (2005). Greater priority should be given to collecting images under favourable weather conditions for aquatic vegetation mapping (i.e., consistent cloud cover, low glare, limited surface effects).

Table 2.3. Summary of the presence of physical disturbance and signs of eutrophication found across the ten study sites in Placentia Bay and Trinity Bay.

Site	Bay	Eelgrass area (km^2)	Disturbance area (m^2)	Source of disturbance	Signs of eutrophication
Come By Chance	Placentia Bay	0.1310	75.4	ATV	/
Glennons Cove Pond	Placentia Bay	0.0013	/	/	/
Great Barasway Pond	Placentia Bay	0.0597	/	/	/
Old Shop Pond	Trinity Bay	0.0687	19.6	ATV, buoy	/
Placentia Swans	Placentia Bay	0.0916	/	/	proliferation of epiphytes
Ship Harbour	Placentia Bay	0.0587	38.3	buoy, anchor	/
Southern Harbour	Placentia Bay	0.0054	/	/	/
Spread Eagle Pond	Trinity Bay	0.1967	/	/	/
Sunnyside	Trinity Bay	0.1265	/	/	/
Western Placentia Southeast Arm	Placentia Bay	0.3331	/	/	floating algal mats

2.4.2 Presence of anthropogenic stressors

Recreational vehicle disturbance has been shown to cause a decrease in saltmarsh vegetation cover (Kelleway 2006). The impact is most evident in areas with high track density, but reductions to vegetation cover can also occur in areas with a single track (Kelleway 2006). Come By Chance and Old Shop Pond were the only sites to have ATV disturbance. There were indications of ATV use at all sites but mostly on land or the intertidal zones, and rarely in areas occupied by eelgrass. Tides throughout Placentia Bay are considered microtidal (less than 2m), expect for Come By Chance which has low-mesotidal conditions (2 – 4 m) (Catto et al. 2003).

The water depth in areas of eelgrass may deter ATV users and may explain why there are few instances of eelgrass disturbance caused by ATVs despite indications of ATV use in other parts of the study sites. During field visits on September 2nd and October 13th, 2020, the Come By Chance had low water levels of 0.56 m and 0.76 m, respectively. During these tidal conditions the eastern half of the eelgrass bed at Come By Chance was submerged by only a few centimeters of water. The shallow nature of the eelgrass bed at Come By Chance during low tide may make the depth of water over the eelgrass area traversable by ATV. Similarly, at Old Shop Pond, ATV disturbance was observed in small patches of shallow eelgrass along the edge of a river delta.

The impact of mooring buoys on seagrass has been well documented (e.g., Walker et al. 1989, Glasby and West 2018, Unsworth et al. 2017, Evans et al. 2018). The disturbance area caused by buoys in this study is minuscule relative to the area of eelgrass mapped and to other estimates of mooring buoy disturbances in the literature. The mooring areas in this study receive less use than other examples from the literature. In contrast to the 7 buoys associated with docking infrastructure in this study creating a disturbance area of $21m^2$, Unsworth et al. (2017) identified 366 scars caused by moorings across 8 sites creating an estimated total disturbance area of 3.71 hectares, and Glasby and West (2018) estimated that leased moorings (1914 moorings) across New South Wales, Australia, caused ~9.4 hectares of disturbance to seagrasses.

Only one instance of anchoring within an eelgrass bed was observed. The disturbance associated with anchoring at Ship Harbour was not a clear disturbance pattern. The presence of a lower density of eelgrass and barren patches suggest anchoring may occur in the same area periodically but could also be natural variation in eelgrass density. This highlights a limitation in identifying physical disturbances using aerial imagery and underwater video without consistent temporal monitoring or groundtruth data collection. If anchoring was not observed in this area during field visits, we may not have been able to identify this barren patch as a potential disturbance. If disturbances are created without a clear pattern, typical of common sources of disturbance (e.g., propeller scarring, mooring chains), they will likely not be detected when the source is no longer present for identification. Therefore, the number of physical disturbances identified may be underestimated due to the limited ability to determine if barren patches are of anthropogenic origin or naturally occurring. For instance, in Ship Harbour two crescent shaped disturbances with diameter of ~6 metres and a disturbance width of ~1 metre were observed in both the ortho imagery and with underwater video (Appendix C). The regular shape of these disturbances suggests an anthropogenic source, but we were unable to identify the source of this disturbance.

Epiphytes, an indication of persistent nutrient enrichment, were mostly observed at one study site: Placentia Swans. This site is adjacent to the town of Placentia, the largest town along the eastern shore of Placentia Bay, with a population of ~3500 (Statistics Canada 2017). Previous reports have indicated that untreated sewage may accumulate in some of the sheltered embayments of Placentia Bay (DFA 2007), which may be the case here as there are outfalls from the town of Placentia that empty directly into this site. The western part of Placentia Southeast Arm, also adjacent to the town of Placentia, had minimal observations of epiphytes in the underwater video. Poor video quality, due to heavy rainfall and a large tannin- rich freshwater input in the days preceding underwater video collection, may have reduced epiphyte detectability at western Placentia Southeast Arm. However, algal mats, an indication of nutrient enrichment, were observed at the site during UAV image collection. Although there were indications of nutrient enrichment at the two sites adjacent to the town of Placentia, both sites have dense and healthy eelgrass beds, with Western Placentia Southeast Arm having the greatest amount of eelgrass across the ten sites. This suggests that current level of eutrophication may not be affecting eelgrass meadows in the area, yet.

This study observed eelgrass beds growing adjacent to anthropogenic development, but there appears to be little impact of anthropogenic disturbance. The communities in the surrounding areas of sites in this study have small human populations, resulting in less disturbance area and nutrient pollution relative to regions where these stressors are more prevalent. Physical disturbances and proliferations of epiphytes appear to occur infrequently in eelgrass beds of Placentia Bay and Trinity Bay.

2.5 Conclusions

This study looked at the spatial distribution of eelgrass in Placentia Bay, an area of Canada’s East coast where eelgrass is declining due to the invasive European green crab (Matheson et al. 2016), and Trinity Bay, an adjacent area where this species is currently absent. Findings indicate variable eelgrass extent in embayments of Placentia Bay and Trinity Bay. The study offers a baseline for monitoring future distributional changes in eelgrass. This baseline data contributes to assessing the extent of eelgrass in Canada. The results also suggest that currently there is little anthropogenic impact from physical disturbance and eutrophication on eelgrass in Placentia Bay and Trinity Bay. The limited number of disturbances and generally low epiphyte load observed is likely due to the low populations of the human communities in proximity to the eelgrass beds. The results of this study can be used in monitoring programs to inform eelgrass conservation in Placentia Bay and Trinity Bay.

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Chapter 3 - Landscape-level assessment of the impacts of the invasive European green crab on eelgrass

Abstract

Seagrasses, including eelgrass, are important habitat provisioning species that have been declining globally at an accelerating rate in recent decades due to anthropogenic stressors. Losses in eelgrass habitat on North America's east coast have been linked to disturbances from the invasive European green crab (*Carcinus maenas*). Those disturbances have been well documented at the plant and patch scales, but less so at the level of eelgrass landscape. Here, we quantified changes to the distribution of eelgrass at six coastal sites of the Island of Newfoundland, Canada. Eelgrass was mapped using aerial imagery and the relative abundance of European green crabs between sites was estimated based on the abundance of beach-washed green crab carapaces. Almost complete loss of eelgrass area (-98.9%) was observed at the site with the highest abundance of beach-washed European green crab carapaces. Two other sites, with varying abundances of beach-washed green crab carapaces, experienced landscape level change comparable to sites with green crab absence. Results corroborate previous research suggesting a link between high green crab abundance and eelgrass loss.

3.1 Introduction

Seagrasses, including eelgrass (*Zostera marina*), is a group of marine flowering plants that form highly productive marine ecosystems, with primary productivity estimates placing them as one of the most productive ecosystems globally (Duarte and Chiscano 1999). Seagrasses provide habitat for a wide diversity of marine organisms, including species of conservation concern such as green sea turtle (*Chelonia mydas*), dugong (*Dugong dugon*), and dwarf seahorse (*Hippocampus zosterae*) (Green and Short 2004, Hughes et al. 2009). Seagrasses function as nursery habitat for many culturally and economically important fish species, such as juvenile cod (*Gadus spp.*) and Pacific Salmon (*Oncorhynchus spp.*) (Heck Jr. et al. 2003, Lilley and Unsworth 2014, Kennedy et al. 2018) and have been estimated to provide nursery habitat for about 20% of the world's 25 largest fisheries (Unsworth et al. 2019). Additionally, the complex three-dimensional structure of seagrass meadows attenuates wave energy, stabilize sediments, and protects from coastal erosion (Paul 2018). The importance of seagrass ecosystems and the ecosystem services they provide makes them one of the most valuable ecosystems globally (Costanza and D'Arge 1997). However, despite their ecological importance, seagrasses are often overlooked in international conservation agendas (Brown et al. 2021).

Invasive populations of green crabs have also been described as allogenic ecosystem engineers due to the alterations they cause to eelgrass (*Zostera marina*), an important coastal habitat (Klassen and Locke 2007, Matheson et al. 2016, Howard et al. 2019). While foraging for prey or burrowing in soft sediments, green crabs cause physical damage to eelgrass rhizomes and uproot eelgrass shoots (Davis et al. 1998, Malyshev and Quijón 2011, Howard et al. 2019), which can result in wide scale habitat destruction (Garbary et al. 2014, Neckles 2015, Matheson et al. 2016). The European green crab (*Carcinus maenas*), originating from coastal European and

North African waters, has been introduced in other regions of the world through shipping and is now found in North America, South America, South Africa and Australia (Young and Elliott 2020). Green crabs are known to negatively impact many ecosystems, outcompeting native species for food resources (MacDonald et al. 2007, Matheson and Gagnon 2012, Griffen and Riley 2015) and introducing new predation pressures on native prey species (Gregory and Quijón 2011, Matheson and Mckenzie 2014). Due to their widespread distribution and their impacts on local ecosystems, green crabs are ranked among the worst global invasive species (Lowe et al. 2004).

The impacts of North American green crab populations on eelgrass have been well described in the literature. Laboratory and field studies have quantified eelgrass loss at the plant and patch scale using green crab enclosures and exclosures (Davis et al. 1998, Malyshev and Quijón 2011, Garbary et al. 2014, Neckles 2015, Howard et al. 2019). Juvenile green crabs have been observed *in-situ* to graze on eelgrass (Malyshev and Quijón 2011). Bioturbation by green crab foraging and burrowing uproots eelgrass shoots (Davis et al. 1998, Malyshev and Quijón 2011) and increases sediment re-suspension, which may increase light attenuation and has been suggested as another mechanism of green crab disturbance (Garbary et al. 2014, Neckles 2015, Matheson et al. 2016). Additionally, green crabs consume eelgrass seeds, reducing seed abundance and seedling establishment (Infantes et al. 2016). Seed predation by green crabs may contribute to a feedback system reducing eelgrass recovery and may lead to a regime shift to an algae dominated state (Infantes et al. 2016).

A few field studies have linked eelgrass loss over large spatial scales to green crab disturbance. Neckles (2015) studied the impacts of European green crab on eelgrass using green crab exclosures and related their results to loss of eelgrass at the landscape level in Maquoit Bay, Maine. Garbary et al. (2014) observed losses of eelgrass shoot density in Tracadie Harbour, Nova Scotia, and noted the formation of barren patches with thinning of the eelgrass bed adjacent to these barren patches. Matheson et al. (2016) used fortuitous eelgrass habitat surveys, before and after the introduction of the European green crab in Newfoundland, to assess changes in eelgrass percent cover. Transects of eelgrass percent cover showed declines of 27% from 1998 to 2012, but declines varied between sites, with declines over 80% occurring at four sites with a high abundance of green crabs (Matheson et al. 2016). While field-based assessments have linked green crab disturbance at the plant and patch scale to declines of eelgrass at the landscape level, structural change of eelgrass landscapes have not been quantified.

Spatial pattern metrics (e.g., number of patches, perimeter to area ratio, etc.), commonly applied in landscape ecology, can help quantify the structural pattern of a landscape. Seagrass landscapes are one of the most studied marine habitats using a landscape ecology approach (Wedding et al. 2011), also called seascape ecology in the marine environment. Assessing changes in eelgrass landscape structure may provide greater insight into the impacts of green crab disturbance than changes to eelgrass area alone. For instance, an eelgrass meadow may transition from a landscape of few large patches to many small patches without much change in area. Measuring area alone would show little change but quantifying changes in landscape structure would capture such changes. The spatial pattern of an eelgrass meadow can influence meadow resilience and feedback processes (Unsworth et al. 2015, Gurbisz et al. 2016), with smaller patches having a greater risk of patch mortality (Olesen and Sand-Jensen 1994, Stipek et al. 2020). Therefore, understanding how green crab disturbance affects eelgrass landscape structure may provide further insight into the impacts of green crab disturbance on eelgrass landscape structure. The goals of this study were to quantify changes in eelgrass landscape structure in areas of coastal Newfoundland characterized by the presence or absence of green crabs as well as assessing the relative abundance of green crab at the different sites.

3.2 Methods

3.2.1 Study area and site selection

Placentia Bay, an ecologically and biologically significant area (DFO 2016, Templeman 2007), is a large bay (~6000 km^2) located in the southeast of the Island of Newfoundland on Canada's East coast (Figure 3.1). The European green crab was first reported in North Harbour, Placentia Bay, in 2007 (Blakeslee et al. 2010). It has since spread throughout the bay in a heterogeneous manner, resulting in areas that vary in terms of green crab abundance and duration of exposure (Matheson et al. 2016). The invasion of the European green crab in Placentia Bay has been linked to declines of eelgrass and subsequent changes in the fish community composition (Matheson et al. 2016). While green crabs have colonized the south and west coasts of the Island of Newfoundland, it has not been reported on the northeast coast including Trinity Bay.

Three study sites were selected in Placentia Bay, where green crabs are present, and three others were selected in Trinity Bay, where green crabs have not yet been reported (Figure 3.1). All six sites selected have eelgrass (Rao et al. 2014) and are distant from common sources of other anthropogenic stressors, such extensive population centers, industrial activities, and agricultural activities. This was done to limit the influence of anthropogenic stressors on eelgrass landscape structure and try to isolate the effects of green crab disturbance. Sites had limited human presence, including road access, off-road vehicle trails, sparse housing, and recreational boat docks. As the hydrodynamic setting (e.g. wave exposure, depth, current speed, tidal range) of an eelgrass meadow influences eelgrass landscape structure (Fonseca and Bell 1998), sites with a similar hydrodynamic environment and geomorphology were selected when possible, in that five of the six sites are coastal lagoons.

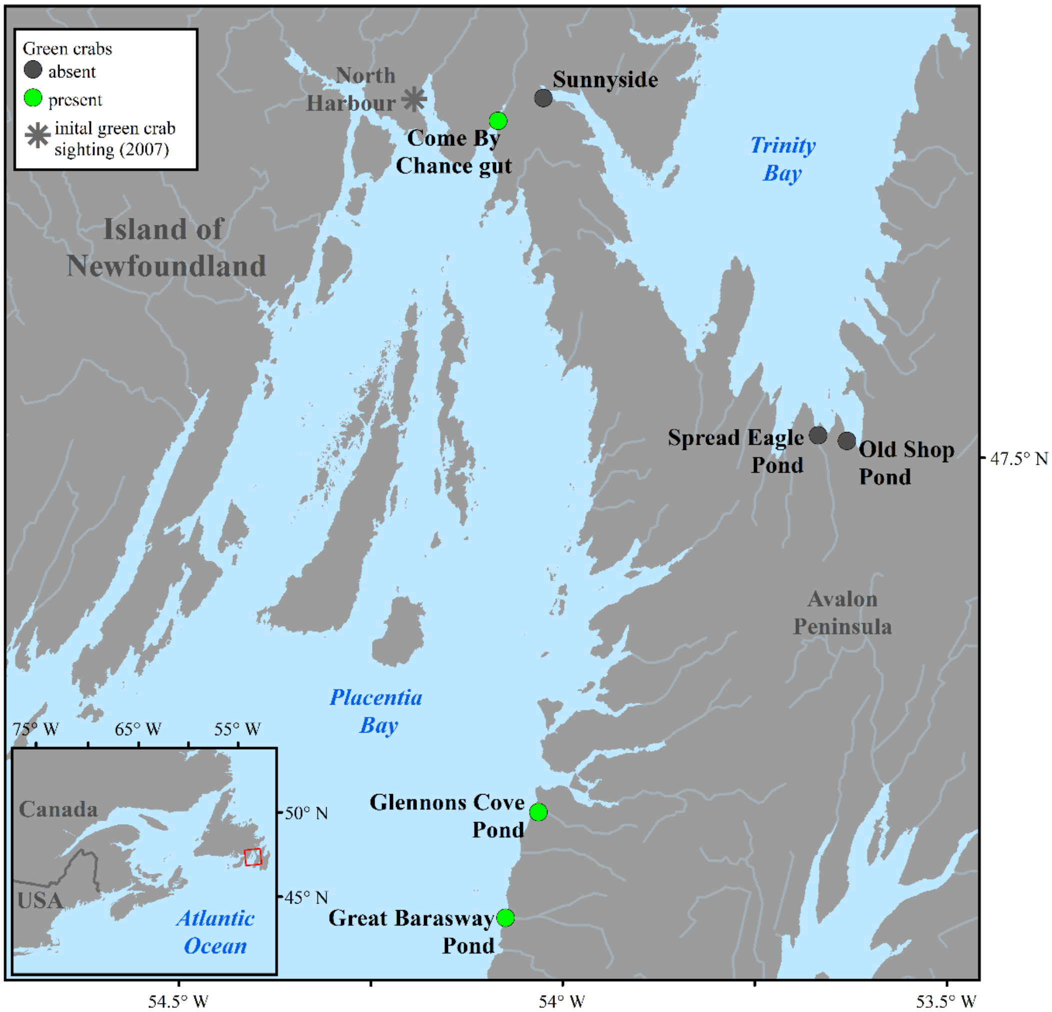

Figure 3.1. Distribution of the six study sites in Placentia Bay and Trinity Bay.

3.2.2 Aerial images and processing

Aerial images used to map eelgrass originated from two data sources: historical georeferenced aerial images and newly acquired unoccupied aerial vehicle (UAV) images. Aerial images from 2009 onward were obtained from the Newfoundland and Labrador Department of Fisheries, Forestry, and Agriculture (FFA). Images collected before 2009 were black and white, and did not allow for an accurate eelgrass detection. While metadata for these images were unavailable, images were collected during the summer months (July or August) (FFA personal communication).

UAV images were collected from August to September of 2020 using a Da-Jiang Innovations (DJI) Mavic 2 Professional UAV. To ensure high quality UAV images, image collection was conducted in the morning, when the Sun's angle was low, and when the sky was clear or overcast (Nahirnick et al. 2019b). UAV images were collected at an altitude of 120 m with a forward and lateral overlap of 80% to enable mosaicking. The position of 7 to 13 ground control points (GCPs) was collected using the waypoint averaging function of a Garmin eTrex 20x (~3 m accuracy) and were used in image processing to georeference the image orthomosaics.

Orthomosaics were created using the Agisoft Metashape version 1.6.1 software (Agisoft 2019). All images (archive images and UAV orthomosaics) were resampled to a cell size of 0.6m, the coarsest spatial resolution of the archive imagery. All images for a site were masked to the same site boundary. If surface effects (e.g., glare) were present in one of the images for a site, which hindered eelgrass detectability, the area of surface effects was masked from all other images for this site to reduce biases caused by weather effects.

3.2.3 Groundtruth data

Groundtruth data were collected using underwater video in September and October of 2020. The underwater camera was operated from a two-person kayak while the position of the camera operator was recorded using the Garmin eTrex 20x. Underwater video collection was guided by visual inspection of the UAV orthomosaics to collect groundtruth data throughout all areas of the field sites and to target as many cover types as possible. A Sony FDR-X3000 ActionCam in a waterproof housing was used to collect groundtruth data. The camera was affixed to an Unger 2.5 – 5 m aluminum telescopic pole and positioned directly under the camera operator, while the pole height was adjusted to maintain the camera's position near the target features. Attaching the camera to a telescopic pole reduced the occurrence of unusable data due to water depth, identified as a limitation for using an underwater camera affixed to the bottom of a kayak (Nahirnick et al. 2019a). Video time stamps were paired to the GPS position in post-processing by recording a few seconds of a digital clock synchronized with the GPS clock.

3.2.4 OBIA and classification

All images were segmented into image objects using the R package SegOptim (Gonçalves et al. 2019). Using the segmentation_ArcGIS_Mshift function, spatial detail, spectral detail, and minimum segment size values of 20, 20, 1, respectively, were specified for all image segmentations. Spatial detail and spectral detail values can range from 1 to 20, with 20 being the maximum amount of detail. Minimum segment size defines the smallest allowable size of an image segment in the units of pixels. Image objects created from image segmentation were manually classified based on photo interpretation of the aerial images in ArcGIS 10.7 (ESRI 2019). The unclassified images for the study sites are presented in Appendix D. The groundtruth data were used to generate point observations that helped guide photo interpretation of the 2020 imagery (Table 3.1). Point observations based on the groundtruth video were created within a 3 m buffer of the GPS transect tracks, corresponding to the accuracy of the GPS. Along the transects, point observations were created at least every 5 seconds, identifying cover types that could be observed in both the video and UAV imagery. No ground truth data were available for the archive imagery. Like similar studies (Evans et al. 2018, Nahirnick et al. 2020), all groundtruth data were used for the classification of the map, preventing from an independent assessment of classification accuracy. Such an approach is justified by the lower spatial accuracy of ground-truth data compared to the image resolution and to the fact that using data from the same set would not allow an independent accuracy assessment.

Table 3.1. Number of groundtruth data points collected per site.

Site	Bay	Number of points	Latitude	Longitude
Come By Chance	Placentia Bay	857	47.834° N	53.996° W
Glennons Cove Pond	Placentia Bay	834	47.217° N	54.015° W
Great Barasway Pond	Placentia Bay	781	47.125° N	54.067° W
Old Shop Pond	Trinity Bay	1417	47.533° N	53.609° W
Spread Eagle Pond	Trinity Bay	1423	47.524° N	53.573° W
Sunnyside	Trinity Bay	1710	47.851° N	53.935° W

3.2.5 Landscape fragmentation metrics

Eelgrass landscape fragmentation metrics were calculated in FragStats version 4.2.1 using binary rasters (eelgrass presence and absence) of the manually classified eelgrass maps (McGarigal et al. 2012). FragStats allows for quantifying landscape metrics at the patch, class, and landscape levels, where class refers to one habitat category and landscape referring to all the habitat categories in a landscape. Many of the landscape metrics at the class and landscape level are calculated in the same manner, while class level considers one habitat category and landscape considers all the habitat categories. Since changes to eelgrass alone were of interest, all landscape metrics were calculated at the class level, as opposed to the landscape level, to quantify changes in the eelgrass class and not other patch types present in the landscape. In other words, class level metrics quantify the spatial pattern of eelgrass and produces one value quantifying the spatial pattern of the eelgrass landscape. The landscape level quantifies the spatial pattern of eelgrass and non-eelgrass areas together and produces one value which quantifies the spatial pattern of this combined landscape. This combined landscape is not relevant to the research questions addressed here as the spatial pattern of eelgrass could remain the same but changes to non-eelgrass areas would result in different values between years.

Landscape Division (LD) and Area-Weighted Mean Perimeter Area Ratio (AWMPAR), recommended by Sleeman et al. (2005), were used for distinguishing fragmentation patterns across a spectrum of fragmented to continuous seagrass landscapes. These metrics have been used previously to quantify fragmentation in seagrass landscapes (Thistle et al. 2010, Santos et al. 2011, 2015, 2020, Abadie et al. 2015, Kaufman and Bell 2020). Landscape division is interpreted as the probability that two randomly chosen pixels in a landscape are not within the same patch (McGarigal et al. 2012). Values of LD range from 0 to 1, with high values indicating a more fragmented landscape. AWMPAR is the sum of the perimeter to area ratio values (patch perimeter divided by patch area) multiplied by a weight based on patch area for all patches in a landscape (McGarigal and Marks 1995). Higher values of AWMPAR indicate more complex patches. Increases to both LD and AWMPAR over time may indicate that a landscape has become more fragmented. Changes to values of eelgrass area and landscape metrics were calculated in R statistical software (R Core Team 2020).

3.2.6 Green crab carapace abundance

The abundance of beach-washed green crab carapaces (moulted or deceased) was used as a proxy for the relative size of green crab populations at each site. Exuviae, the remnants of an exoskeleton following moulting, have been shown to be reliable estimates of aquatic larval insect population density (Foster and Soluk 2004, Heinold et al. 2020). For instance, Heinold et al. (2020) sampled exuviae of larval salmonflies (*Pteronarcys california*) along stream banks and found a strong correlation (R^2=0.88) with total larval density (Heinold et al. 2020). Additionally, beach-cast biological matter has been used to assess spatial and temporal variation in species composition of elasmobranchs (Smith and Griffiths 1997, Schmöle et al. 2020), and marine macrovegetation (Suursaar et al. 2014).

A similar approach to Heinold et al. (2020) was used to sample beach-washed green crab carapaces. At each field site, five 25 m long transects were used to count the number of green crab carapaces along the shoreline. We decided to count carapaces because the number of spines on the carapace can be used to distinguish green crab carapaces from the native Atlantic rock crab (*Cancer irroratus*). Transects were established parallel to the shoreline and distributed as evenly as possible throughout the field sites. If the site was a coastal lagoon, the transects were distributed on landward beaches within the coastal lagoon and not on seaward beaches. Before searching for carapaces, the search area on each side of the transect line was defined by the waterline on the lower side and by the presence of beach grass or shrubs on the upper side. The distance between the upper side and lower side of the transect was measured at the start and at the end of each transect. These measures were used to calculate the area of each transect by taking the mean of the start and end widths and multiplying it by the length of the transect. After establishing a transect, the area was searched and crab carapaces were collected along the transect line and photographed, using the transect line to provide a scale. Since the total area searched at each site differed depending on the width of the transects, the total number of green crab carapaces divided by the total area of transects at each site was used to calculate green crab carapace density.

3.3 Results

3.3.1 Eelgrass landscape change

Four of the six sites studied experienced a loss of eelgrass. Both Placentia Bay and Trinity Bay experienced a net loss of eelgrass area, with cumulative losses of 18.2 ha (-44.5%) and 1.0 ha (-2.5%), respectively. The loss to eelgrass area in Placentia Bay largely occurred at Glennons Cove Pond, with a loss of 19.7 ha (-98.9%) from 2014 to 2020. Both AWMPAR and LD increased for Glennons Cove Pond (Figure 3.2a,c,e), suggesting the landscape became more fragmented. Changes in eelgrass area, AWMPAR, and LD for Placentia Bay sites are presented in Figure 3.2a, c, and e, respectively. From 2014, eelgrass filled Glennons Cove Pond declined to a small, aggregated area of eelgrass patches by 2020 (Figure 3.3c). Come By Chance Gut experienced the largest increase in eelgrass area, with an increase of 2.0 ha (13.4%), a subsequent decrease in LD, and slight increase to AWMPAR. Eelgrass expansion in some areas of Come By Chance Gut connected previously disjointed patches forming larger and more contiguous patches (Figure 3.3a). The third site in Placentia Bay, Great Barasway Pond (Figure 3.3b), lost 0.4 ha (-6.1%) of its eelgrass, with a slight increase in both AWMPAR and LD.

In Trinity Bay, Old Shop Pond had the largest change in eelgrass area with a loss of 1.5 ha (-17.0%) from 2014 to 2020 (Figure 3.2b). Old Shop Pond was the only site with suitable imagery before 2014. From 2009 to 2020, eelgrass at Old Shop Pond lost 2.5 ha (-25.7%) (figure 3.3f). Old Shop Pond had the largest increase to AWMPAR and a minor increase in LD (Figure 3.2d, f). Losses in eelgrass area at Old Shop Pond resulted in more disjointed patches of eelgrass (Figure 3.3d,e,f) . Spread Eagle Pond lost 0.27 ha (-1.2%) of eelgrass and Sunnyside gained 0.73 ha (7.5%) (Figure 3.2b; Figure 3.3g,h). Both sites experienced minimal changes to AWMPAR and LD (Figure 3.2d, f).

3.3.2 Green crab carapace counts

Green crab carapaces were present at all sites in Placentia Bay and absent from all sites in Trinity Bay (Table 3.2). Glennons Cove Pond had the greatest number of green crab carapaces with 140 carapaces observed and the highest density amongst all sites, with 0.11 carapaces/m^2. Come by Chance Gut and Great Barasway Pond had 43 and 17 green crab carapaces respectively, corresponding to densities of 0.06 and 0.01 carapaces/m^2, respectively. The distribution of the green crab carapace transects for the three Placentia Bay sites and the number of green crab carapaces observed in each transect are provided in Figure 3.4 – 3.6.

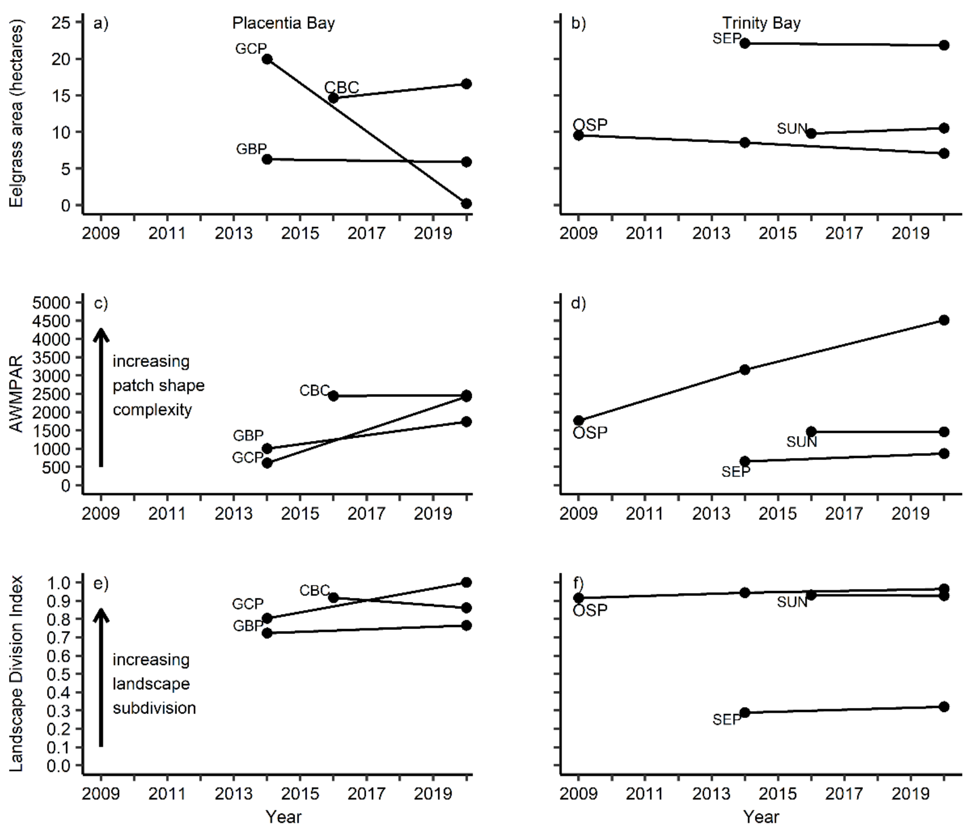

Figure 3.2. Changes in eelgrass landscape metrics for the Placentia Bay and Trinity Bay study sites. CBC: Come By Chance Gut, GBP: Great Barasway Pond, GCP: Glennons Cove Pond, OSP: Old Shop Pond, SEP: Spread Eagle Pond, SUN: Sunnyside.

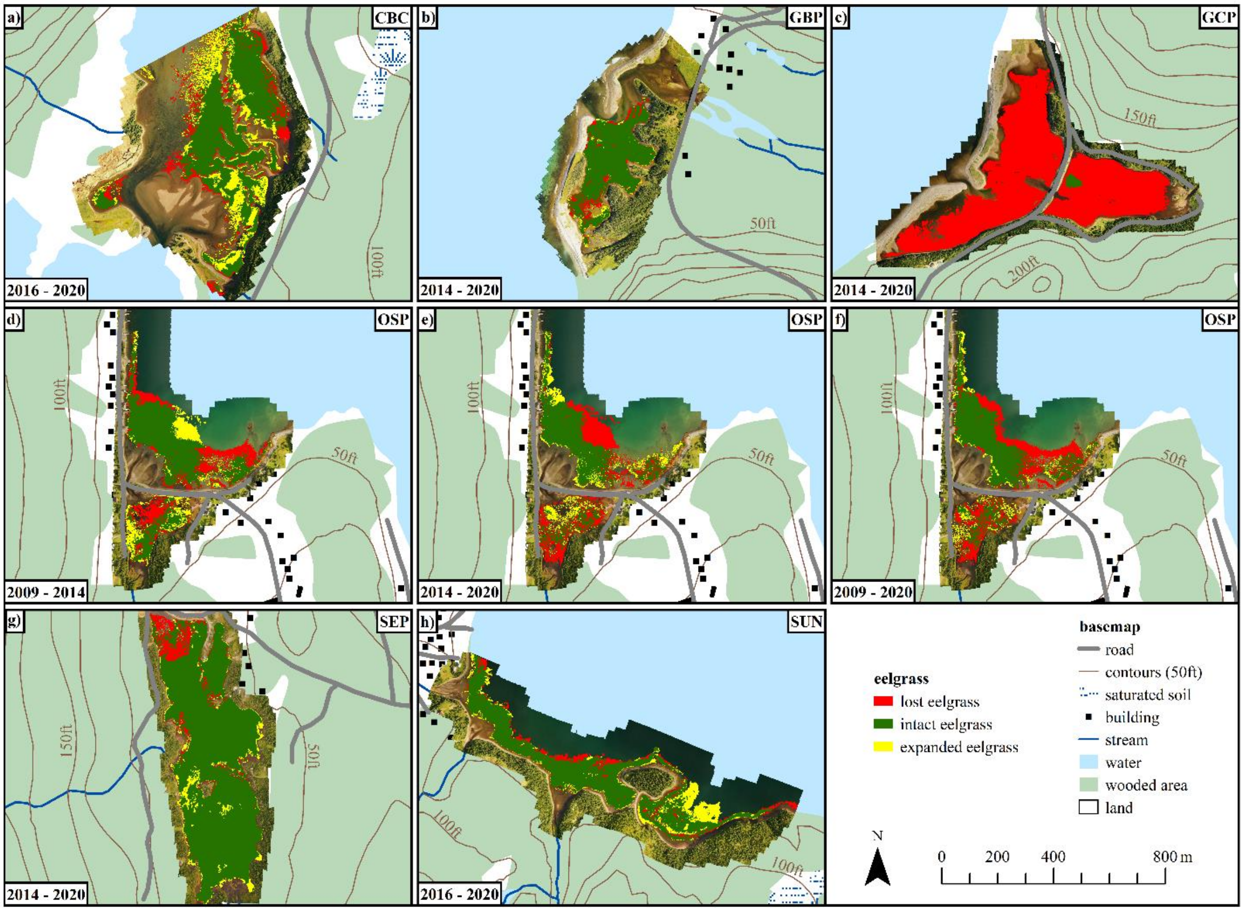

Figure 3.3. Changes to eelgrass landscapes for sites in Placentia Bay (a, b, c) and Trinity Bay (d, e, f, g, h). CBC: Come By Chance Gut, GBP: Great Barasway Pond, GCP: Glennons Cove Pond, OSP: Old Shop Pond, SEP: Spread Eagle Pond, SUN: Sunnyside.

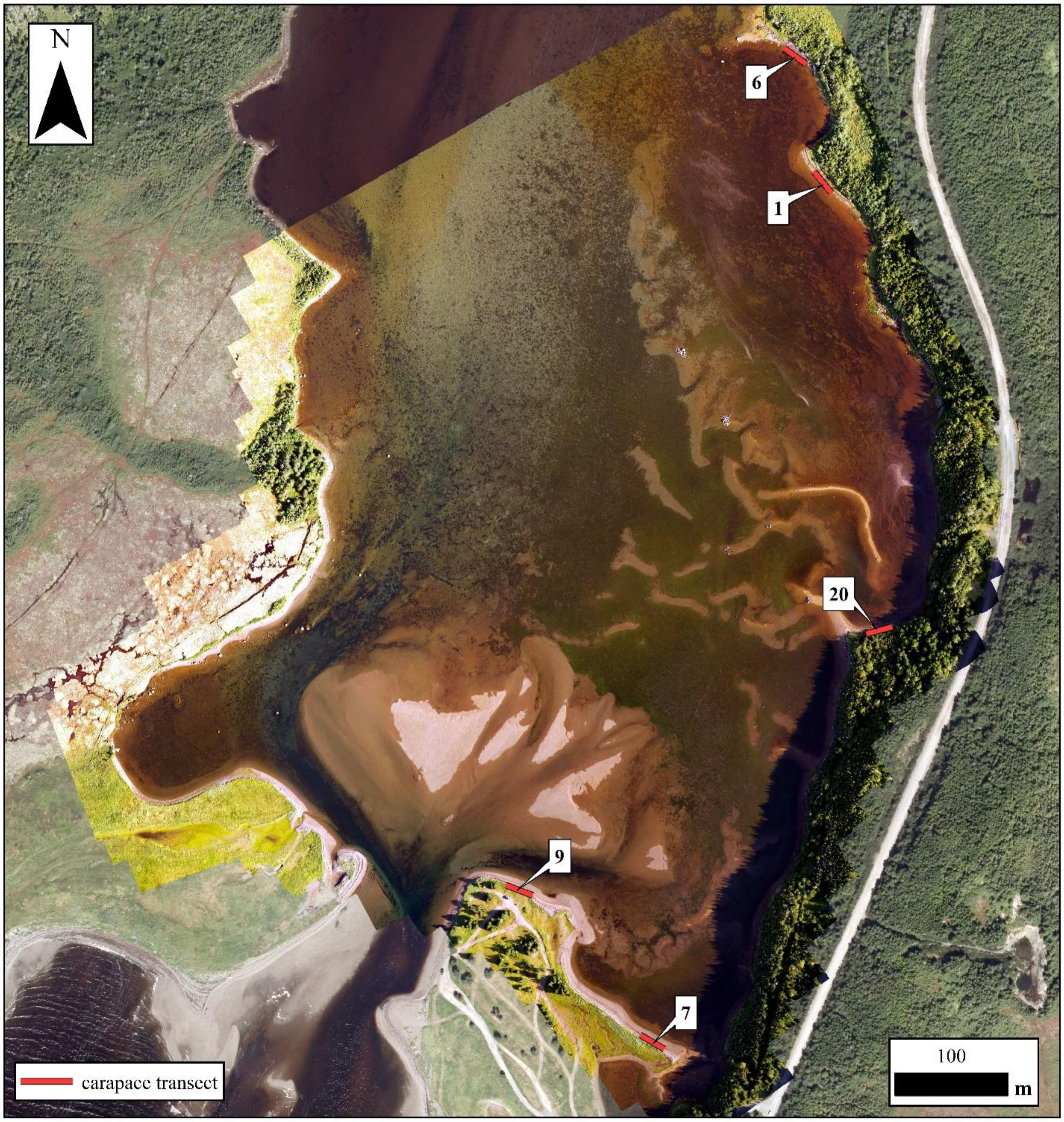

Figure 3.4. The distribution of beach-washed green crab carapace transects at Come By Chance Gut. The values indicate the number of beach-washed green crab carapaces observed within each transect.

Figure 3.5. The distribution of beach-washed green crab carapace transects at Great Barasway Pond. The values indicate the number of beach-washed green crab carapaces observed within each transect.

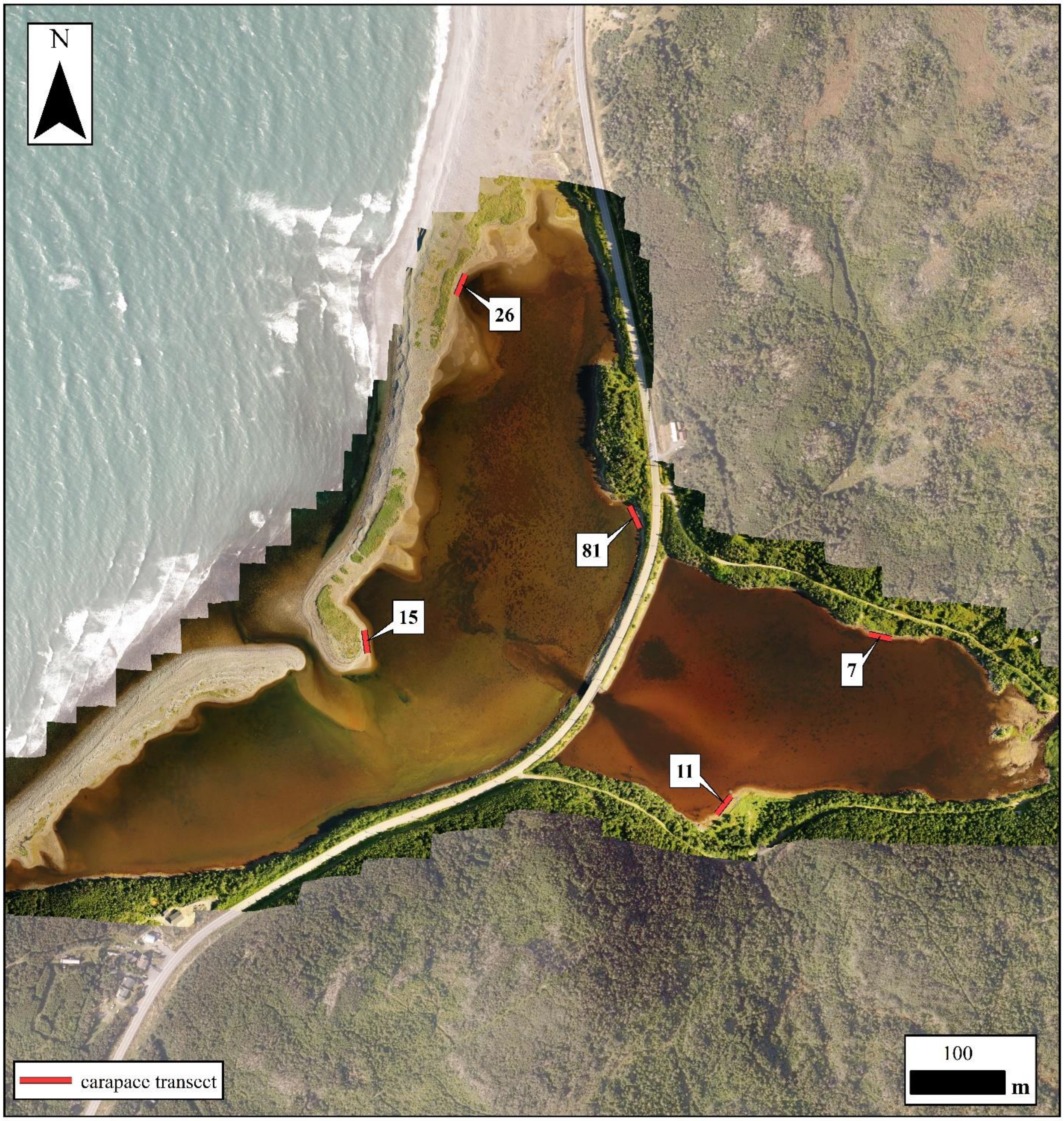

Figure 3.6. The distribution of beach-washed green crab carapace transects at Glennons Cove Pond. The values indicate the number of beach-washed green crab carapaces observed within each transect.

Table 3.2. Results of transect surveys for beach-washed European green crab carapaces.

Site name	Total transect area (m^2)	Total green crab carapaces	Total density (carapaces/m^2)
Come By Chance Gut	758 ($\bar{x}$ = 152 ± 47)	43 ($\bar{x}$ = 8.6 ± 7.0)	0.06 ($\bar{x}$ = 0.075 ± 0.097)
Great Barasway Pond	1098 ($\bar{x}$ = 220 ± 36)	17 ($\bar{x}$ = 3.4 ± 6.0)	0.01 ($\bar{x}$ = 0.016 ± 0.029)
Glennons Cove Pond	1248 ($\bar{x}$ = 250 ± 41)	140 ($\bar{x}$ = 28 ± 30.5)	0.11 ($\bar{x}$ = 0.111 ± 0.123)
Old Shop Pond	964 ($\bar{x}$ = 193 ± 78)	0 ($\bar{x}$ = 0 ± 0)	0 ($\bar{x}$ = 0 ± 0)
Spread Eagle Pond	321 ($\bar{x}$ = 64 ± 18)	0 ($\bar{x}$ = 0 ± 0)	0 ($\bar{x}$ = 0 ± 0)
Sunnyside	646 ($\bar{x}$ = 130 ± 34)	0 ($\bar{x}$ = 0 ± 0)	0 ($\bar{x}$ = 0 ± 0)

Note: enclosed in brackets is the sample mean ($\bar{x}$) and standard deviation.

3.4 Discussion

This study provides an initial assessment of changes to eelgrass landscape structure in sites of Placentia Bay and Trinity Bay, on the Island of Newfoundland, where European green crabs are present and absent. Total eelgrass area was found to decline in Placentia Bay and to a lesser measure in Trinity Bay. The decline in eelgrass area in Placentia Bay was largely dominated by eelgrass loss at one site, Glennons Cove Pond. All the other sites in Placentia Bay and Trinity Bay had relatively small changes in eelgrass coverage.

The high number of beach-washed green crab carapaces at Glennons Cove Pond, along with little human presence and no indications of anthropogenic disturbance, suggests invasive European green crab disturbance resulted in changes to the eelgrass landscape at Glennons Cove Pond. The drastic decline in eelgrass area at Glennons Cove Pond in conjunction with the greatest abundance of beach-washed green crab carapaces is consistent with previous findings in Placentia Bay, where the greatest losses of eelgrass occurred in areas with the highest abundances of trapped green crabs (Matheson et al. 2016). Conversely, changes to the eelgrass landscapes at Come By Chance gut and Great Barasway Pond were not appreciably different from non-green crab sites in Trinity Bay. Differences in the abundance of green crab between sites in Placentia Bay, measured using beach-washed carapace surveys, may explain the differences in landscape level change. For instance, Davis et al. (1998) and Howard et al. (2019) manipulated green crab density in experimental eelgrass enclosures and found that eelgrass loss caused by low densities of green crabs (1 crabs/m^2, 1.4 crabs/m^2, respectively) did not differ from control sites, while densities of 4 crabs/m^2 and 5.6 crabs/m^2, respectively, resulted in significant reductions to eelgrass.

While our study provides a valuable baseline on eelgrass coverage and change in the study region, studying more sites characterized by different densities of green crab would help understand the impact of green crab disturbance at the landscape level. The results of the carapace transects should be interpreted with caution. Beach-washed green crab carapaces were used as a proxy for relative green crab abundance between sites, but beach-washed carapace abundance could be affected by environmental or site-specific factors, such as site-specific mortality rates, coastal morphology, currents, and carapace retention. The carapace surveys used in this study included carapaces of deceased individuals and moulted carapaces. Crab carapaces have been used to assess avian predation on intertidal and subtidal crabs (Ellis et al. 2005). However, avian predation on green crabs may be low, with gull predation on green crabs estimated between 0 and 0.8% in the low intertidal (Ellis et al. 2005). Therefore, the population of green crabs, particularly the number of individuals that moulted, may have a greater effect on beach washed carapace counts than site specific predation rates. Additionally, the distribution of beach-washed debris can be affected by circulation and wind direction (Suursaar et al. 2014, Brennan et al. 2018). Along the eastern shore of Placentia Bay, marine debris is transported by southerly winds from the south of the bay, accumulating in the north (Pink 2004). We did not observe this regional trend with regard to beach-washed green crab carapaces. All of the beaches sampled for beach-washed green crab carapaces in Placentia Bay were within the coastal lagoons, with limited exposure to wave energy. Seaward beaches with higher wave energy in Placentia Bay have self-cleaning properties and transport debris away from the beach, whereas beaches with lower wave energy have higher amounts of debris (Pink 2004). Greater counts of green crab carapaces were found along sections of beach with higher general amounts of debris (e.g., garbage, drift wood, macrovegetation). For instance, one of the transects at Glennons Cove Pond had 81 carapaces where a collection of debris had accumulated as well. High debris areas may indicate high retention rather than source abundance (Brennan et al. 2018). However, the carapace transects were distributed along beaches within the lagoon and spread throughout the site as evenly as possible. Given the sheltered nature of the beaches within the coastal lagoons (i.e., protected by the seaward beach), the transport and deposition of seaward marine debris is likely limited compared seaward beaches. For instance, McNeil (2009) observed minimal marine debris accumulation at Goose Cove beach in Placentia Bay, and attributed this to the sheltered nature of the beach. Additionally, the sheltered nature of these beaches would limit departures of carapaces from a section of beach compared to an exposed section of beach, where carapaces would more likely be transported away from the site. It is assumed that given the sheltered nature of all the beaches within the coastal lagoons and protection by the seaward beach from marine debris transport that the carapaces are of local origin and would be retained similarly across the sites.

Glennons Cove Pond also experienced increases in both LD and AWMPAR, suggesting a transition to a more fragmented landscape. However, the landscape metrics for Glennons Cove

Pond do not reflect the changes in landscape pattern observed. The eelgrass area decreased to a small area of 12 aggregated patches. Class area, the area of a single habitat type (e.g., eelgrass area), affects both AWMPAR and LD with small class areas producing higher values for AWMPAR and LD (Neel et al. 2004, Wang et al. 2014), which would explain the increase in both AWMPAR and LD despite fewer and more aggregated patches at Glennons Cove Pond. In this instance, the landscape metrics appear misleading by suggesting fragmentation occurred when there was only habitat loss without fragmentation. LD and AWMPAR have previously highlighted the importance of assessing changes in both area and landscape fragmentation. For instance, Santos et al. (2015) observed slight losses of submerged aquatic vegetation area (-3%) in Biscayne Bay, Florida, but increases to landscape fragmentation, quantified using LD, AWMPAR, patch density and mean radius of gyration, between 1938 and 2009. In our study, almost complete loss of eelgrass area at Glennons Cove Pond occurred. Matheson et al. (2016) documented complete loss of eelgrass due to green crab disturbance in Placentia Bay at 4 sites and 90% loss at a fifth site, out of 17 total sites. Given that large losses of eelgrass area may occur from green crab disturbance and that LD and AWMPAR are affected by habitat abundance (Neel et al. 2004, Wang et al. 2014), there may be limitations in using these landscape metrics for quantifying changes in eelgrass landscape structure due to green crab disturbance. Metrics that are weakly correlated with habitat abundance and strongly correlated with spatial aggregation (e.g., perimeter area fractal dimension, aggregation index, clumpiness index, Coefficient of Variation of Proximity Index, etc; see Wang et al. 2014) may be more useful in quantifying changes in eelgrass landscape structure.

The habitat change presented in the map figures may be overestimated due to minor misalignments of aerial images (Figure 3.3 – 3.6), resulting from limitations in the spatial accuracy of the GPS (~3 m) used for georeferencing the UAV orthomosaics. However, differences in the spatial accuracy used for georeferencing would not affect the calculations of our results for change in area and landscape metrics over time because these values are calculated for individual landscapes. Additionally, there is a lack of metadata for the archival imagery. Similar limitations of missing metadata have arisen in previous eelgrass studies using aerial imagery not collected for the purposes of habitat mapping (Nahirnick et al. 2020). The archival images were collected during July or August (FFA personal communication) with the UAV imagery being collected in August and September, and therefore, it was assumed that seasonal dynamics did not affect changes in eelgrass area or landscape metrics. Lastly, the classification accuracy of the eelgrass distribution maps could not be assessed due to the lack of groundtruth data in the archive images and the lack of independence between the groundtruth observations and the manual classification of the 2020 UAV orthomosaics. However, all of the aerial images have properties of high image reliability according to the image reliability factors outlined by Nahirnick et al. (2020) and Pasqualini et al. (1997) (e.g., tidal height, eelgrass visualization, surface effects). Four for the six sites were shallow coastal lagoons with tidal height having little impact on eelgrass delineation. In the two other sites, Old Shop Pond, also containing a coastal lagoon, and Sunnyside, the deep edge of the eelgrass is visible in all years of the imagery and throughout the sites with some minor variations in detectability due to water depth along the northeast side of Old Shop Pond and Sunnyside. At both sites there are minor changes in eelgrass area along the northeast side of both sites (Figure 3d, e, f, h) and any classification errors as a result of poorer eelgrass detectability due to water depth are likely minor on the overall change in eelgrass area. There was high eelgrass visualization in all images with some minor variations throughout the images (Appendix D). Areas where surface effects inhibited confident eelgrass delineation were masked from all years and not assessed, increasing reliability of the images. Despite the lack of classification accuracy assessment, the high image reliability properties suggest the suitability of the images for accurate manual classification via photo interpretation.

Differences in resilience, the capacity to resist and recover from disturbance, may also affect the response of eelgrass landscapes to green crab disturbance. The resilience and productivity of eelgrass beds were shown to be reduced by high water temperature, high temperature variability, high light attenuation, shallow water depth, and low water movement (Krumhansl et al. 2021). For instance, Wong et al. (2013) associated a 90% loss in eelgrass area at coastal lagoon in Atlantic Canada, where green crabs are also present, to water flow constriction by a bridge and causeway, resulting in low water movement and high water temperatures. The reduced water flow and high water temperature resulted in reduced eelgrass shoot density (Wong et al. 2013), an indication of long-term stress (McMahon et al. 2013, Roca et al. 2016). Howard et al. (2019) speculated that differences in resilience between Atlantic and Pacific eelgrass beds may explain the lack of reports for eelgrass decline on the Pacific Coast caused by green crab disturbance, despite high densities of green crab being established for more than two decades. However, Howard et al. (2019) also recommend caution by noting that the highest density green crab populations are located in remote areas of the Pacific coast and that challenges in detecting changes to eelgrass beds may explain the lack of reports.

Disturbance to eelgrass meadows may also increase resilience. Disturbance to seagrass meadows from anthropogenic and natural sources have been shown to increase reproductive output and genetic diversity (Cabaço and Santos 2012). For instance, sea otters disturb eelgrass beds while foraging in the sediment for infaunal prey, a similar process to green crab disturbance, which has been shown to increase eelgrass genetic diversity by promoting conditions that favour eelgrass sexual reproduction (Foster et al. 2021). Higher genetic diversity in eelgrass has been shown to increase resilience to temperature warming (Ehlers et al. 2008). Therefore, small amounts of green crab disturbance may promote eelgrass reproductive output, genetic diversity, and resilience.

Site specific interactions, such as the depth of an eelgrass meadow, may play a role in shaping green crab distribution at the fine scale (Cosham et al. 2016). For example, avian predators can reduce crab densities in intertidal and shallow subtidal areas (Ellis et al. 2005, Good 1992). However, green crabs were shown to undergo less predation by gulls compared to other crab species (Dumas and Witman 1993, Ellis et al. 2005). Compared to *Cancer irroratus* and *Cancer borealis*, the survivorship of green crabs increases as depth decreases, suggesting a greater mortality caused by aquatic predators than avian predators (Donahue et al. 2009). The survival of green crabs from predation is also significantly greater when a vegetated refuge is available (Dumas and Witman 1993), such as that provided by eelgrass. The shallow nature of the eelgrass beds at all three sites in Placentia Bay (~1-2 m) would appear to be ideal habitat for green crabs in that they receive little predation by avian predators (Ellis et al. 2005), the eelgrass provides a vegetative refuge (Dumas and Witman 1993), and the shallow depth may limit predation by aquatic predators (Donahue et al. 2009).

Environmental variables and food resources may also affect green crab disturbance to eelgrass landscapes. Matheson et al. (2016) theorized that differences in sediment compactness and infaunal communities may contribute to differences in green crab disturbance to eelgrass landscapes by reducing green crabs' ability to forage or the targeted locations of foraging. Additionally, green crab size and sex have been shown to affect the response of green crabs to environmental variables, especially depth (Cosham et al. 2016). There is some evidence to suggest that green crab disturbance may be affected by green crab demographics, such as age and proportion of males (Malyshev and Quijón 2011, Matheson et al. 2016). Future research could monitor fine-scale changes to eelgrass landscapes in concert with variation in fine-scale site specific factors, resilience, and European green crab densities and demographics to assess how site-specific factors may affect the distribution of green crabs and disturbance to eelgrass landscapes.

Rapid rates of eelgrass loss due to green crab disturbance were described by Neckles (2015) who reported the loss of 474 ha of eelgrass (-83.16% change) in Macquoit Bay, USA, from 2001 to 2013. Little change occurred from 2001 to 2009 based on comparison of aerial photos and the most substantial loss occurred between 2012 and 2013, based on local observations (Neckles 2015). Similarly, 98.9% of eelgrass at Glennons Cove Pond was lost over 6 years. Given the potentially high rate of eelgrass loss observed in this study and in previous studies (e.g., Neckles 2015), the use of UAV for collecting imagery may be beneficial to cost effectively produce a dataset with a high temporal and spatial resolution to capture the process of green crab disturbance on eelgrass landscapes. Capturing the progression of eelgrass loss and potential habitat fragmentation at a high temporal resolution may provide insights into the spatial and temporal dynamics of green crab disturbance, which may be useful in early detection of green crab disturbance and identifying priority areas for eelgrass protection or green crab removal.

3.5 Conclusions

This study explored changes in eelgrass coverage in six sites of Canada's Atlantic coast characterized by the presence or absence of the invasive European green crab. In addition to providing baseline data on eelgrass that will help monitor future changes, this study identified very different dynamics among study sites. Similar changes in eelgrass area were observed amongst five of the six sites. One site in Placentia Bay, Glennons Cove Pond, experienced a loss in eelgrass area of 98.9% between 2014 and 2020. Findings corroborate studies done in the region and others suggesting a potential link between eelgrass loss and invasive green crab disturbance. While the site showing the largest eelgrass loss also shows the largest number of beach-washed green crab carapaces, a larger number of study sites and in-situ estimations of crabs would be required to confirm the causal relationship between the two. In areas where green crab disturbance to eelgrass is a prominent threat, high temporal resolution monitoring is recommended to inform eelgrass habitat protection, green crab removal efforts, and to fully capture the progression of habitat disturbance.

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Chapter 4 - Summary and Conclusions

4.1 General summary

Seagrass are recognized as one of the most valuable marine ecosystems globally (Costanza and D'Arge 1997), but are declining at an accelerating rate (Waycott et al. 2009). There is a need to quantify the spatial distribution of seagrasses and identify threats at local scales to inform effective seagrass management efforts (Unsworth et al. 2019). Eelgrass, a species of seagrass, in Placentia Bay, Newfoundland and Labrador, Canada, is declining (Matheson et al. 2016). The decline of eelgrass in Placentia Bay has been attributed to the invasive European green crab (Matheson et al. 2016), but other stressors that commonly impact seagrass ecosystems have not been assessed, nor has there been an assessment of green crab disturbance on eelgrass landscape structure.

This research provides a baseline of the spatial extent of eelgrass in study sites of Placentia Bay and Trinity Bay, Newfoundland and Labrador, that can be used as a base to monitor future changes in the spatial extent of eelgrass over time, and assesses anthropogenic stressors affecting eelgrass. The specific research questions were (1) what is the spatial extent of eelgrass beds in Placentia Bay and Trinity Bay?, (2) what is the presence of common anthropogenic stressors known to negatively impact eelgrass (i.e., physical disturbance and eutrophication) in eelgrass beds of Placentia Bay and Trinity Bay?, and (3) how has eelgrass landscape structure changed over recent years in areas with and without the European green crab?

Questions one and two were addressed in Chapter 2. Question one was addressed by producing baseline distribution maps of eelgrass extent from UAV imagery at seven sites in

Placentia Bay and three sites in Trinity Bay. Question two was addressed using the UAV imagery and underwater video to quantify the presence of physical disturbances and the semi-quantitative abundance of epiphytes, respectively. My results indicate that physical disturbance to eelgrass rarely occurred at the study sites in Placentia Bay and Trinity Bay, and that there were few signs of eutrophication, with some indications of nutrient enrichment only being present at two sites directly adjacent to the Town of Placentia. The limited populations of the communities surrounding the study sites likely explains the limited presence of these anthropogenic stressors. This study provides an initial assessment of the presence of physical disturbance and eutrophication in eelgrass beds of Placentia Bay and Trinity Bay, Newfoundland and Labrador, Canada.

Question 3 was addressed in Chapter 3, where landscape pattern metrics, in conjunction with transect surveys of beach-washed green crab carapace abundance, were used to quantify changes in eelgrass landscape structure. Glennons Cove Pond had the highest abundance of beach-washed green crab carapaces and experienced the greatest loss in eelgrass area (98.9%). Changes in eelgrass landscape structure at the other two sites where green crabs are present were not appreciably different from sites with no green crabs. The results suggest that disturbance to the eelgrass landscape at Glennons Cove Pond was caused by green crabs. The results also suggest that green crab disturbance to eelgrass meadows can result in a dramatic loss of eelgrass at a rapid rate. The present study provides an initial analysis of how green crab disturbance affects eelgrass landscape structure.

4.2 Limitations and future directions

The present study provides insight for eelgrass monitoring and conservation in Placentia Bay and Trinity Bay. Continued monitoring of eelgrass in Placentia Bay is important given the declining status due to green crab disturbance, and while, currently, physical disturbance and eutrophication appear to occur infrequently in Placentia Bay and Trinity Bay, there are indications of nutrient enrichment adjacent to the Town of Placentia. Further monitoring is needed to assess the degree of impact nutrient pollution is having on eelgrass adjacent to the Town of Placentia.

The assessment of eutrophication in this study provides an initial assessment of the presence of this stressor in Placentia Bay and Trinity Bay, but lacks the depth that other indicators and measures offer. The assessment of eutrophication, using a semi-quantitative abundance of epiphytes as a proxy for nutrient enrichment, suggests that nutrient enrichment is occurring to eelgrass beds adjacent to the town of Placentia, but further research is required to understand the degree of nutrient pollution and the impact it is having on eelgrass in the study region. A variety of metrics exist for assessing eutrophication, such as nitrogen concentrations (Robertson and Savage 2020) and epiphyte loads (Nelson 2017), as well as the degree of impact on eelgrass (e.g., leaf growth, shoot density, ratio of above ground biomass to belowground biomass, etc.; see McMahon et al. 2013). Logistical and financial constraints hindered our ability to assess eutrophication using more quantitative methods, such as collecting water chemistry samples or taking biological measures and samples of eelgrass or epiphytes. Using a more standardized and quantitative method for assessing eutrophication would allow future monitoring efforts to assess more than just the likely presence of nutrient enrichment and would allow for a greater assessment of the impact on eelgrass between sites and may enable comparisons to other studies.

An additional limitation of this research is that the distribution of field sites was restricted to the eastern side of Placentia Bay and southern Trinity Bay due to logistical constraints. The strength of anthropogenic stressors can vary locally (Murphy et al. 2019) and the impacts of some stressors, such as mooring areas, may be variable (Sagerman et al. 2020). Future research and monitoring efforts may benefit from including additional sites in other areas of Placentia Bay and throughout Newfoundland to provide a greater ability to assess trends locally and regionally. Many assessments of eelgrass in Canada are conducted in isolation and this presents challenges for assessing trends between sites regionally and nationally (Murphy et al. 2021). There is increasing acknowledgement of a need to coordinate seagrass monitoring efforts both nationally, in Canada (Murphy et al. 2021), as well as internationally (Duffy et al. 2019, McKenzie et al. 2020).

In concert, chapter 2 and chapter 3 suggest that the invasive European green crab may be the most prominent threat to eelgrass in Placentia Bay, which corroborates the findings of previous research (Matheson et al. 2016). This research also suggests that green crab disturbance may rapidly and extensively alter an eelgrass landscape. Similar observations of a high rate of eelgrass area loss linked to European green crab disturbance is reported in the literature (Neckles 2015).

Future research and eelgrass monitoring in areas where European green crab disturbance is a prominent threat may benefit from high temporal resolution monitoring to capture the progression of changes to eelgrass landscape structure. The ability afforded by UAV to collect high resolution imagery at user specified time intervals and during weather conditions favourable for aerial imagery collection in aquatic environments may be highly beneficial for monitoring green crab disturbance. High temporal resolution monitoring may allow management efforts to detect eelgrass loss before a drastic decline occurs and effectively target green crab removal efforts to these areas.

In chapter 3, the greatest loss of eelgrass area occurred at the site that had the greatest abundance of beach-washed green crab carapaces, whereas the two other sites, with lower abundances of beach-washed green crab carapaces, had minimal change in eelgrass area. Site specific characteristics (e.g., temperature, depth, food resources, sediment hardness, etc.) that affect eelgrass meadow resilience and green crab abundance, demographics, and behaviour, may make some eelgrass beds more susceptible to green crab disturbance. Future studies should focus on assessing threshold abundance levels of green crab and factors that affect eelgrass bed vulnerability to green crab disturbance.

This research contributes to a data gap in the extent and condition of eelgrass in Canada. Future monitoring efforts can build upon the initial baseline of eelgrass extent provided in this research. The baseline data provides opportunities for future monitoring of eelgrass in Placentia Bay and Trinity Bay to assess how eelgrass may respond to current threats and potential changes in environmental conditions.

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Appendix A

SegOptim Parameter Space Delineation

In ArcMap 10.7, spectral detail values can range from 1 – 20, spatial details can range from 1- 20, and minimum segment size specifies the minimum size of a segment, in pixel count. The parameter space was determined through experimentation with a subsample of the orthomosaics by setting spatial and spectral detail to the maximum value of 20 and minimum segment size to 20. Experimentation involved decreasing the value of either spectral detail or spatial detail by increments of 1, while holding the two other parameters constant. When the output segmentation raster exhibited under segmentation (i.e., multiple image features contained in one image segment) the previous value was set as the lower limit of the parameter space. These values were 18 and 17 for spectral detail and spatial detail, respectively. The parameter space for minimum segment size was set with a lower limit of 1 and an upper limit of 100. The lower limit of 1 is the most detailed setting and the upper limit of 100 was determined through preliminary tests of the genetic algorithm. The preliminary tests produced optimal minimum segment sizes less than 100, and therefore it was determined that minimum segment sizes from 1-100 was a sufficiently large parameter space to contain an optimum solution without being too large as to hinder computation time.

Appendix B

Orthomosaic images, classified maps, and confusion matrices for sites in Placentia Bay and Trinity Bay

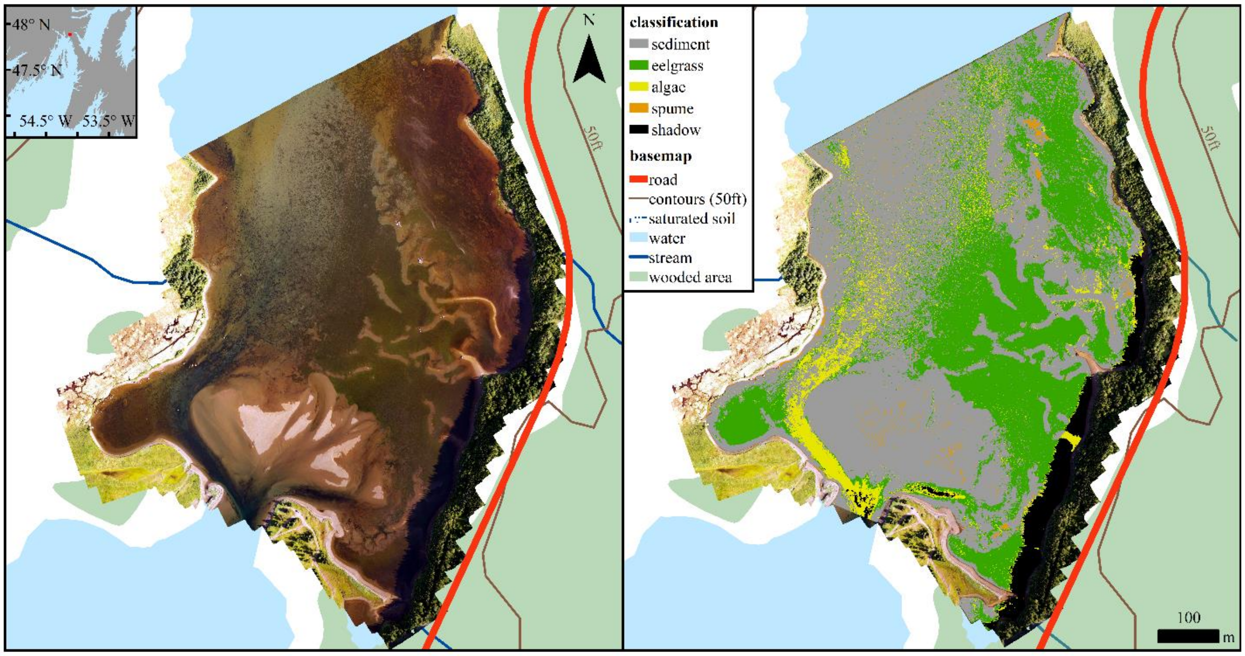

Figure B.1. Orthomosaic (left) and thematic map (right) of Come By Chance Gut, NL, CA.

Table B.1. Full confusion matrix for Come By Chance Gut classification.

		observed				
		sediment	eelgrass	algae	spume	shadow
predicted	sediment	94	11	9	6	0
	eelgrass	11	104	7	0	0
	algae	1	2	16	0	2
	spume	2	0	0	4	0
	shadow	0	0	1	0	5

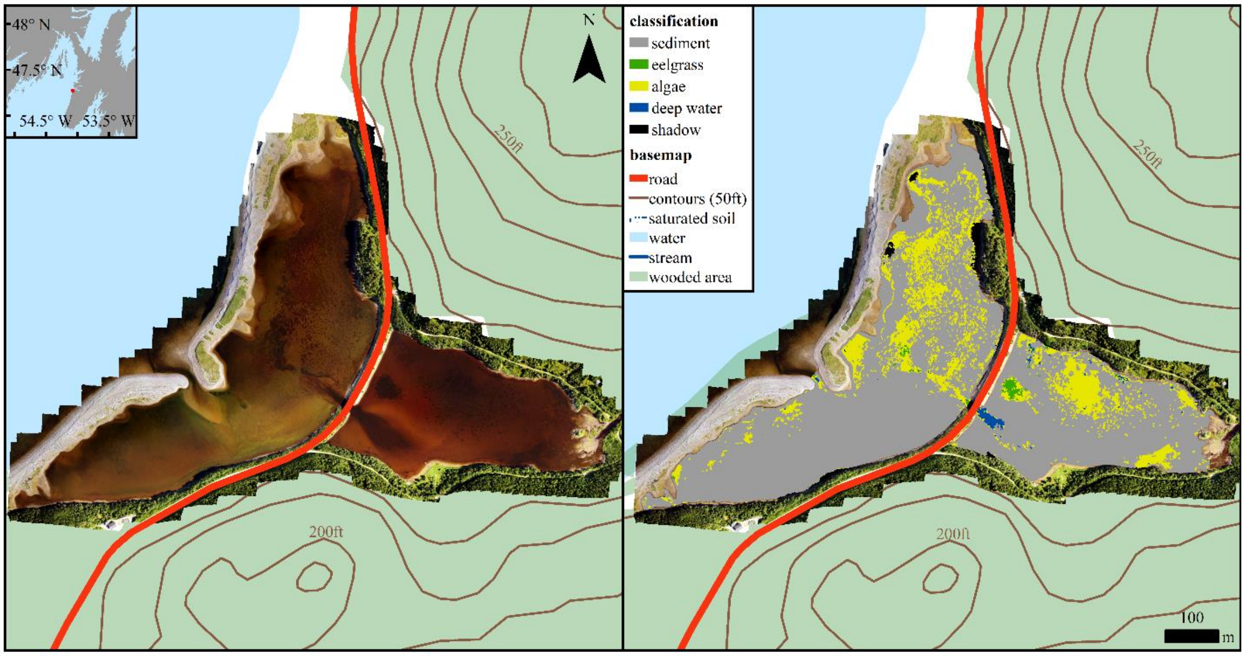

Figure B.2. Orthomosaic (top) and thematic map (bottom) of Glennons Cove Pond, NL, CA.

Table B.2. Full confusion matrix for Glennons Cove Pond classification

		observed				
		sediment	eelgrass	algae	deep water	shadow
predicted	sediment	123	0	18	0	1
	eelgrass	0	5	1	0	0
	algae	10	5	41	3	0
	deep water	0	0	3	6	0
	shadow	0	0	0	0	9

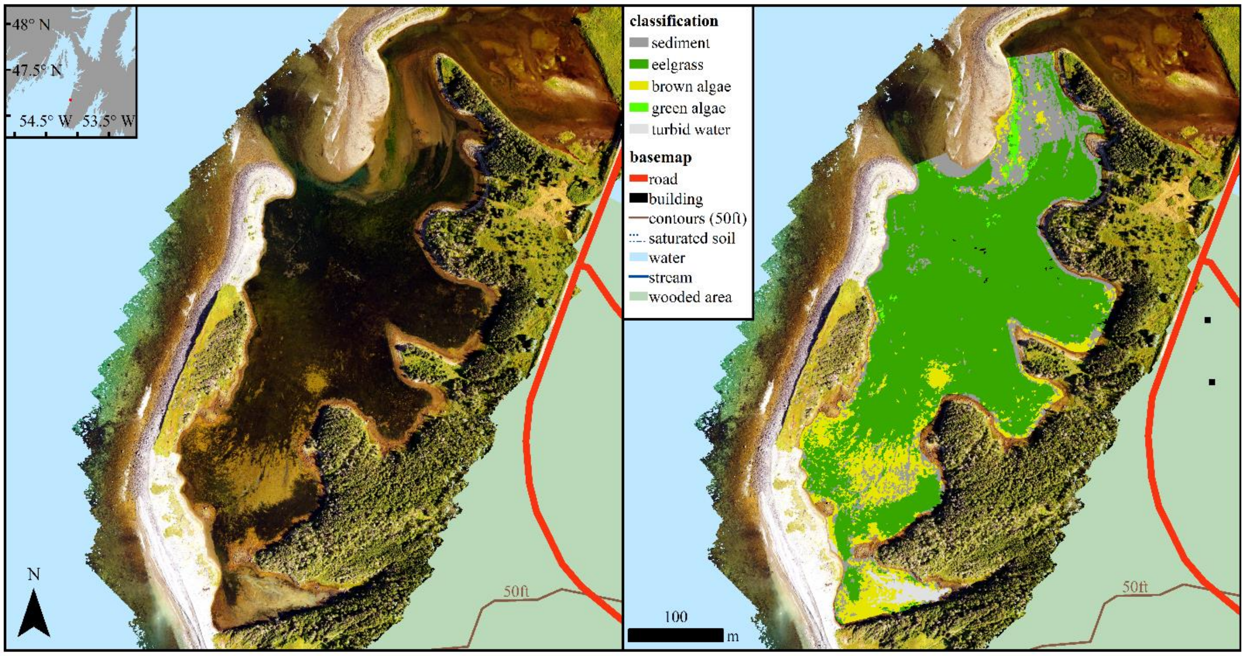

Figure B.3. Orthomosaic (left) and thematic map (right) of Great Barasway Pond, NL, CA.

Table B.3. Full confusion matrix for Great Barasway Pond classification.

		observed					
		sediment	eelgrass	brown algae	green algae	turbid water	shadow
predicted	sediment	30	1	8	1	1	0
	eelgrass	3	196	2	1	0	6
	brown algae	9	1	35	2	0	0
	green algae	0	0	0	6	0	0
	turbid water	1	0	0	0	9	0
	shadow	0	1	0	0	0	4

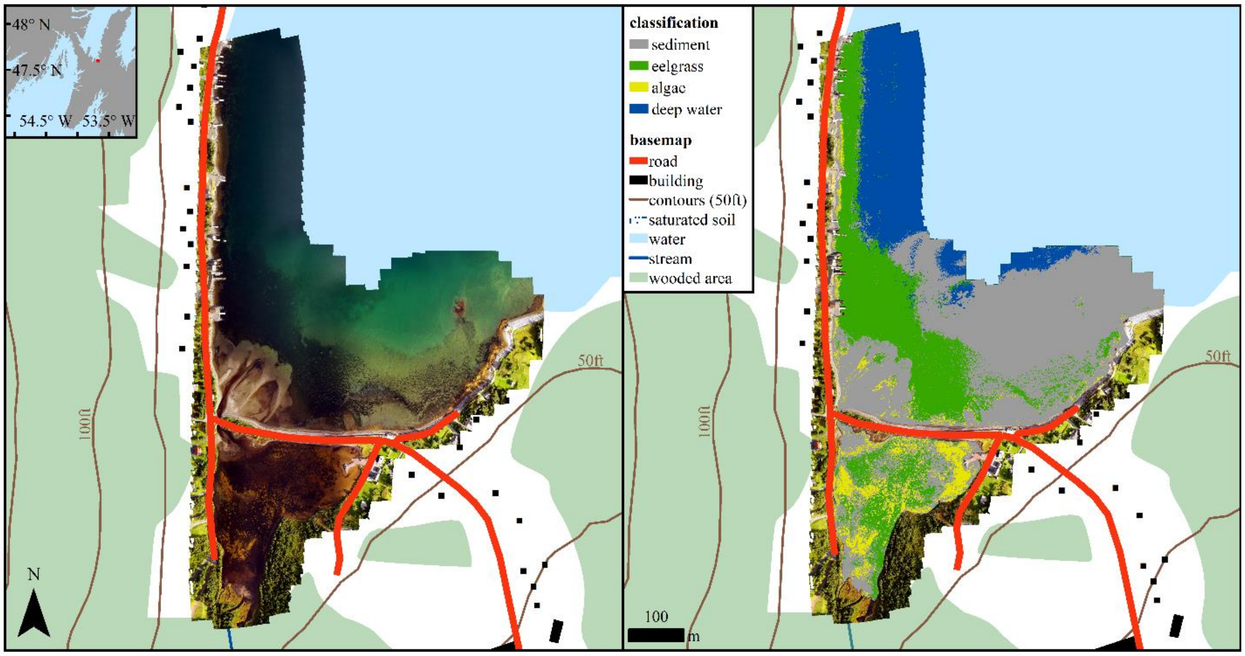

Figure B.4. Orthomosaic (left) and thematic map (right) of Old Shop Pond, NL, CA.

Table B.4. Full confusion matrix for Old Shop Pond classification.

		observed			
		sediment	eelgrass	algae	deep water
predicted	sediment	107	8	14	5
	eelgrass	10	83	1	2
	algae	12	2	17	0
	deep water	4	2	0	33

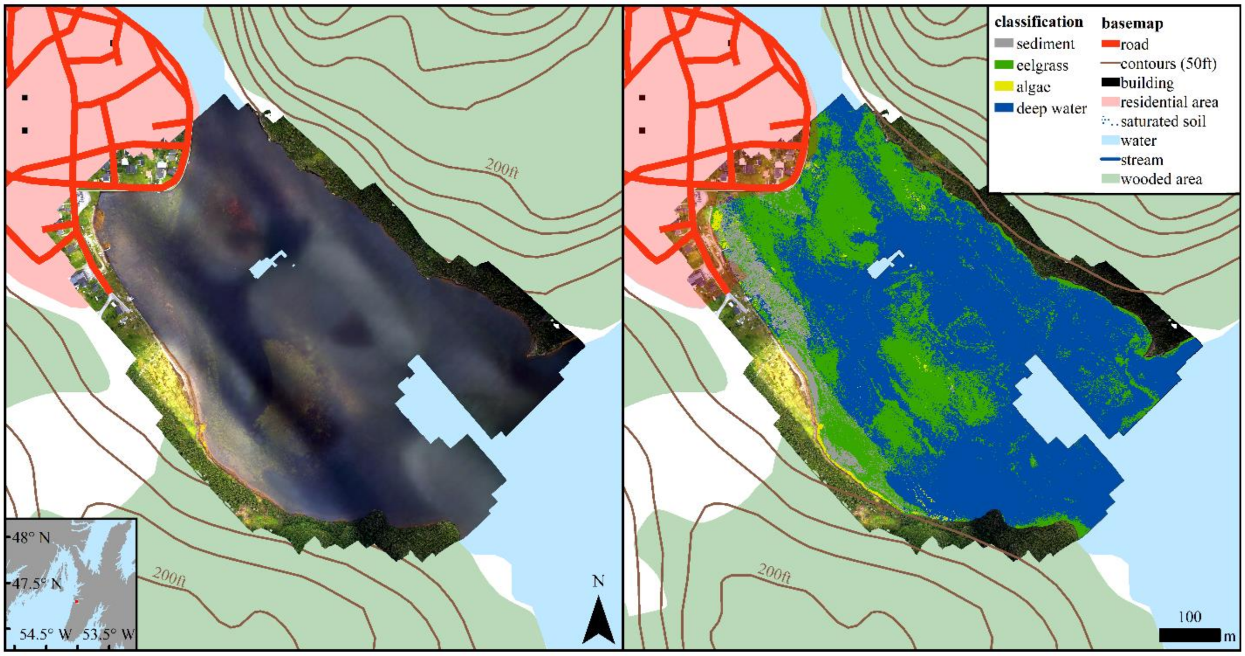

Figure B.5. Orthomosaic (left) and thematic map (right) of Placentia Swans, NL, CA.

Table B.5. Full confusion matrix for Placentia Swans classification.

		observed			
		sediment	eelgrass	algae	deep water
predicted	sediment	8	6	3	1
	eelgrass	11	60	4	19
	algae	0	1	5	0
	deep water	2	37	2	141

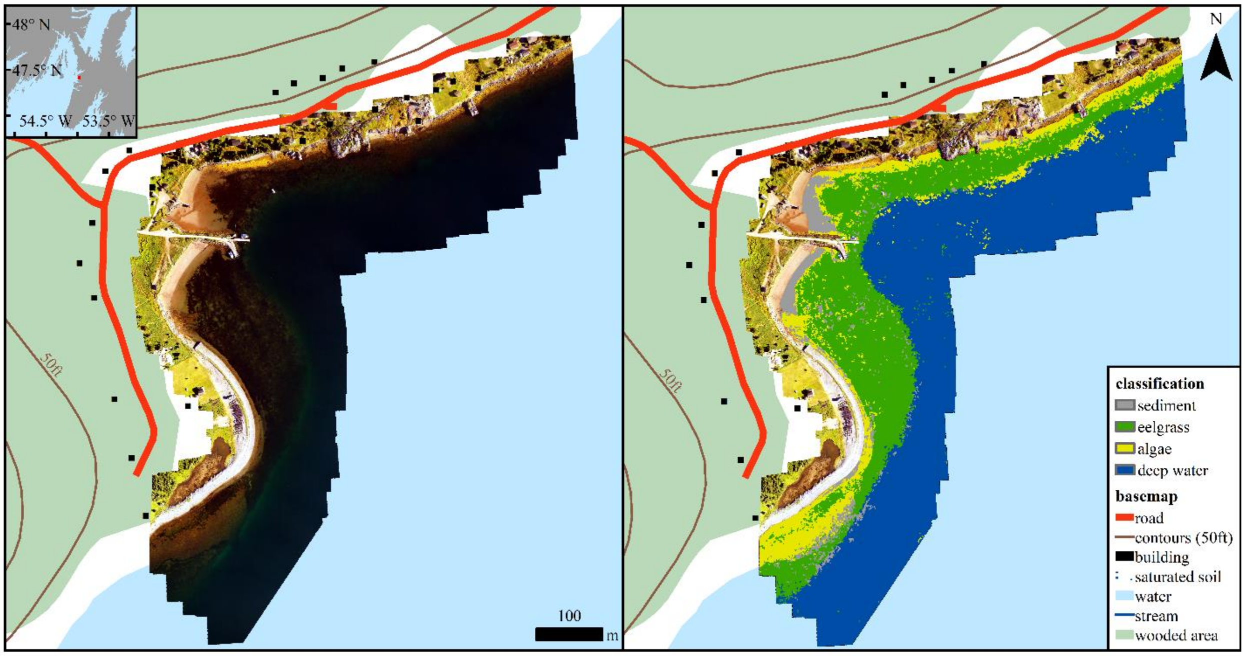

Figure B.6. Orthomosaic (left) and thematic map (right) of Ship Harbour, NL, CA.

Table B.6. Full confusion matrix for Ship Harbour classification.

		observed			
		sediment	eelgrass	algae	deep water
predicted	sediment	10	0	3	0
	eelgrass	4	81	20	1
	algae	4	6	22	1
	deep water	5	3	3	134

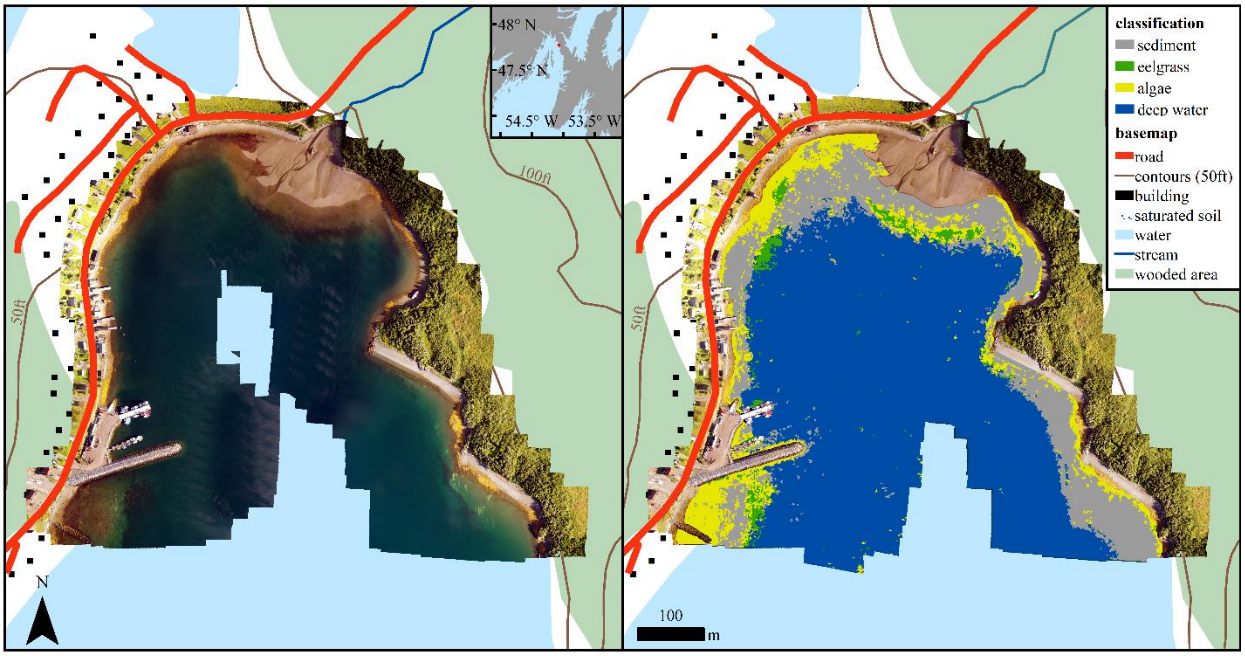

Figure B.7. Orthomosaic (right) and thematic map (left) of Southern Harbour, NL, CA.

Table .7. Full confusion matrix for Southern Harbour classification.

		observed			
		sediment	eelgrass	algae	deep water
predicted	sediment	49	4	15	0
	eelgrass	1	4	6	1
	algae	14	6	24	1
	deep water	10	4	5	153

Figure B.8. Orthomosaic (right) and thematic map (left) of Spread Eagle Pond, NL, CA.

Table B.8. Full confusion matrix for Spread Eagle Pond classification.

		observed			
		sediment	eelgrass	algae	shells
predicted	sediment	5	2	3	0
	eelgrass	8	206	15	0
	algae	8	7	44	0
	shells	1	0	0	10

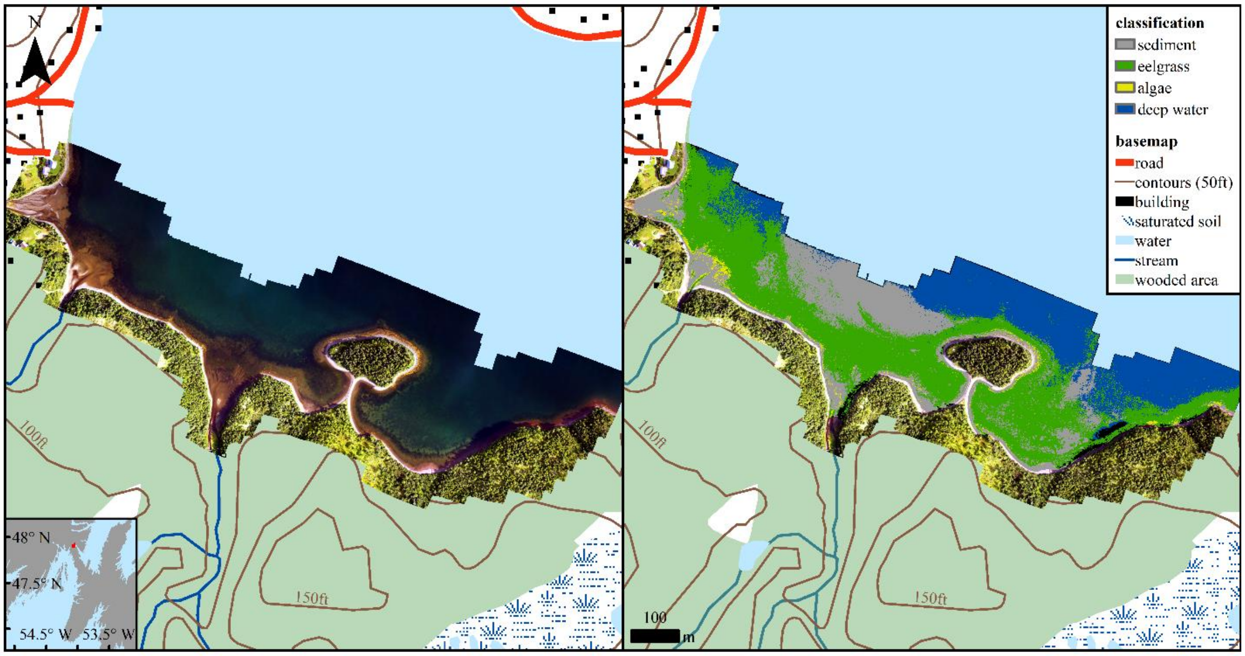

Figure B.9. Orthomosaic (right) and thematic map (left) of Frenchmans Island, Sunnyside, NL, CA.

Table B.9. Full confusion matrix for Frenchmans Island, Sunnyside classification.

		observed				
		sediment	eelgrass	algae	deep water	shadow
predicted	sediment	40	9	6	2	0
	eelgrass	19	134	6	3	4
	algae	3	3	1	0	0
	deep water	4	2	0	62	2
	shadow	0	0	0	0	4

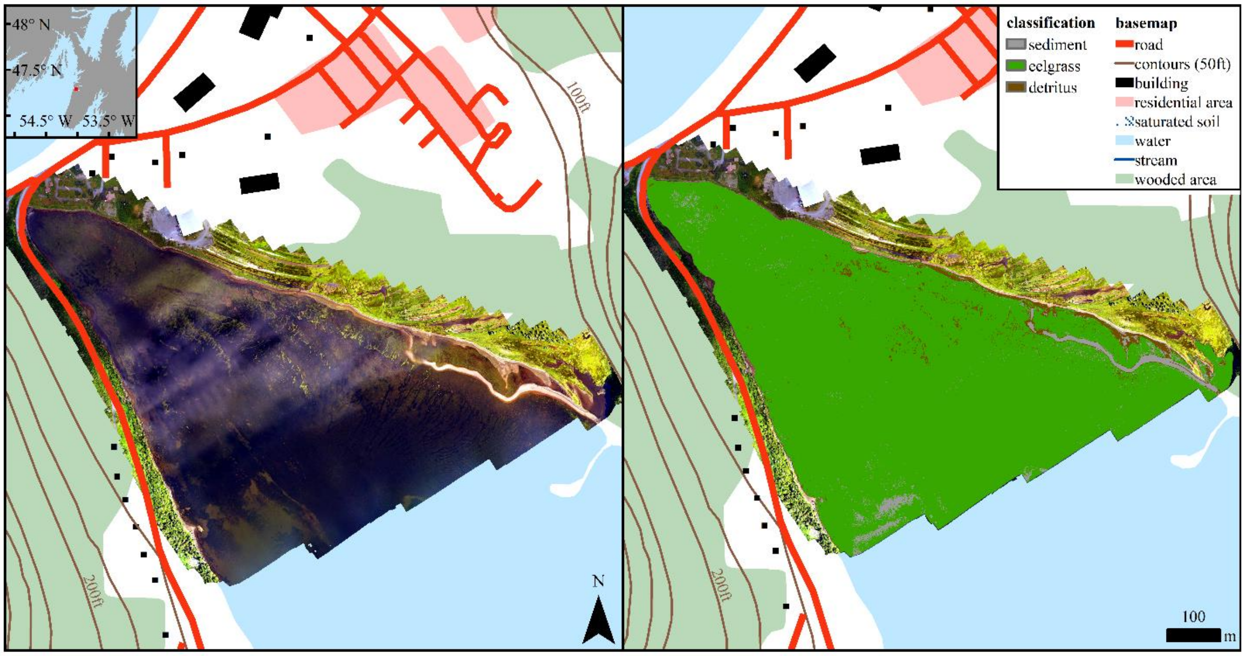

Figure B.10. Orthomosaic (eft) and thematic map (right) of Western Placentia Southeast Arm, NL, CA.

Figure B.10. Full confusion matrix for western Placentia Southeast Arm classification.

		observed		
		sediment	eelgrass	detritus
predicted	sediment	8	2	1
	eelgrass	18	250	12
	detritus	3	2	4

Appendix C

Image of unidentified disturbance at Ship Harbour, NL.

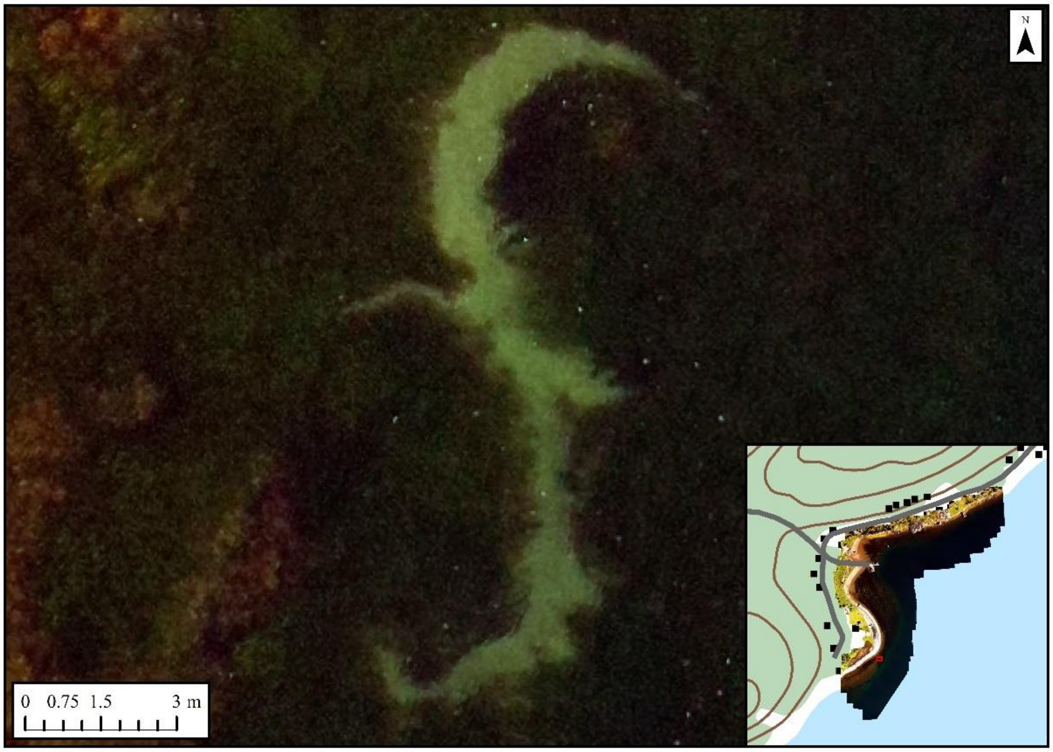

Figure C.1. Two semi-circular disturbances of an unknown source observed at Ship Harbour. The width of the disturbance is approximately 1 metre. The diameter of the semicircular pattern is approximately 6 metres.

Appendix D

Unclassified aerial images used to produce eelgrass distribution maps to assess changes in eelgrass landscape structure

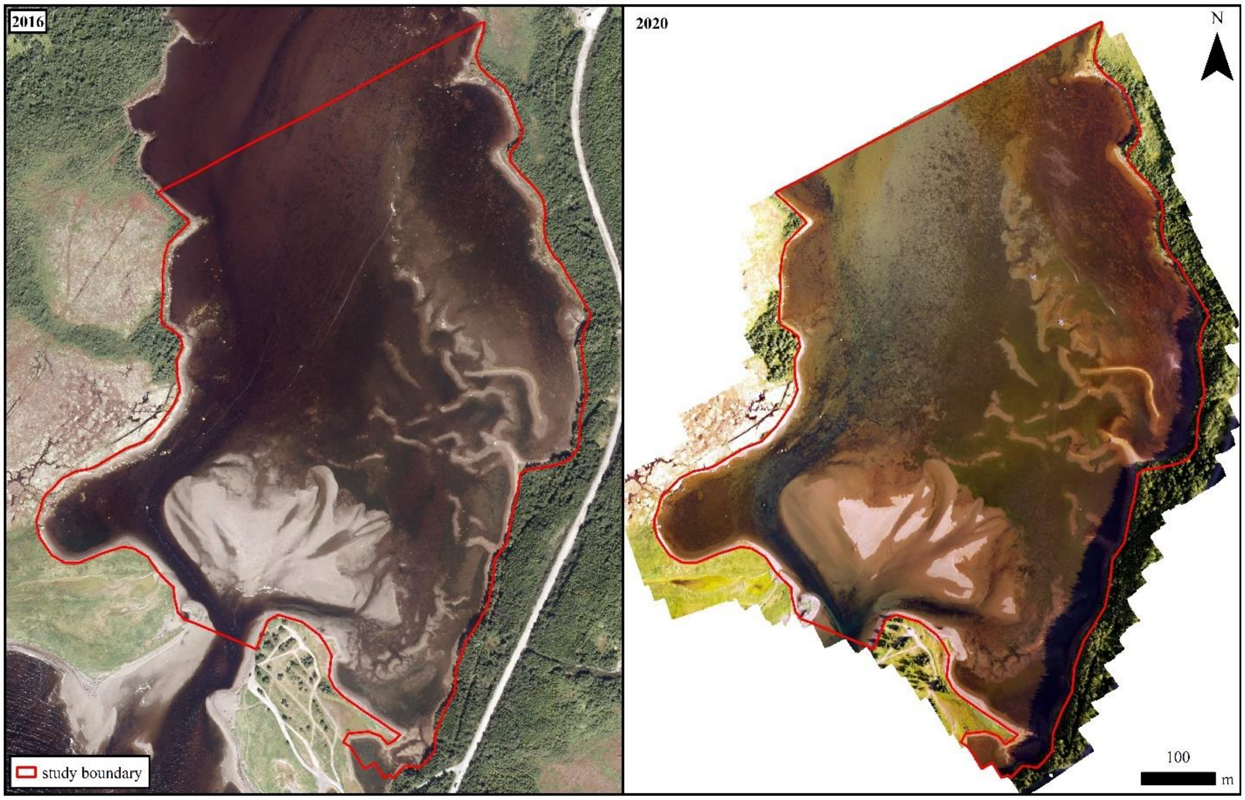

Figure D.1. Aerial imagery for Come By Chance in 2016 and 2020.

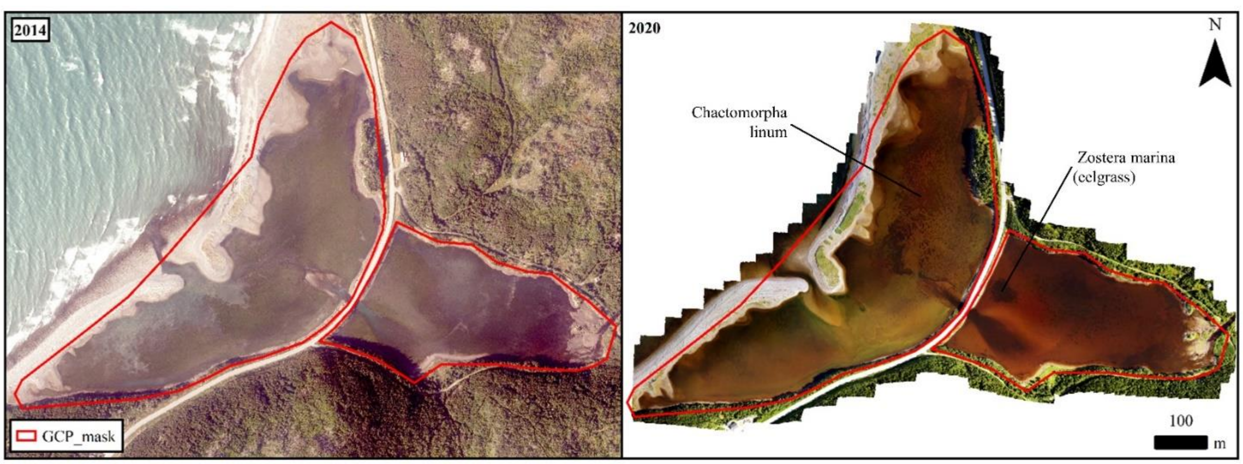

Figure D.2. Aerial imagery for Glennons Cove Pond in 2014 and 2020.

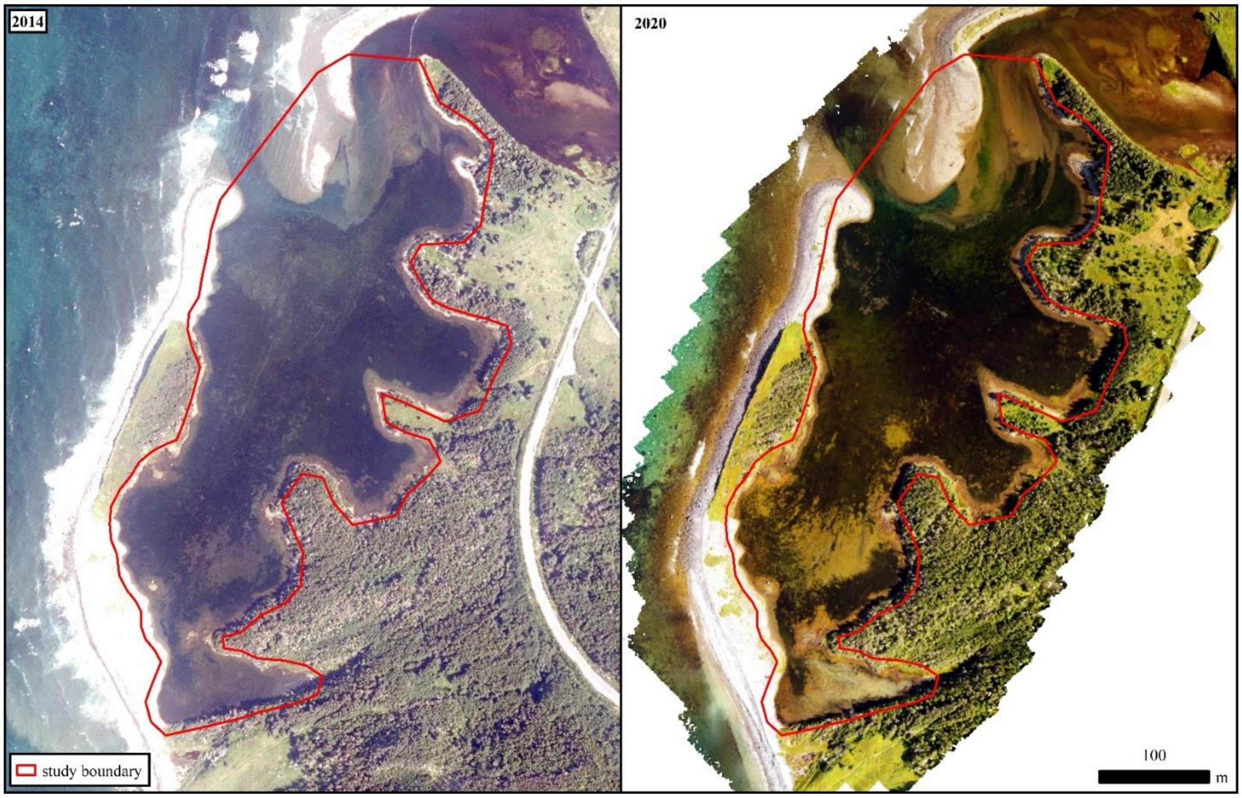

Figure D.3. Aerial imagery for Great Barasway Pond in 2014 and 2020.

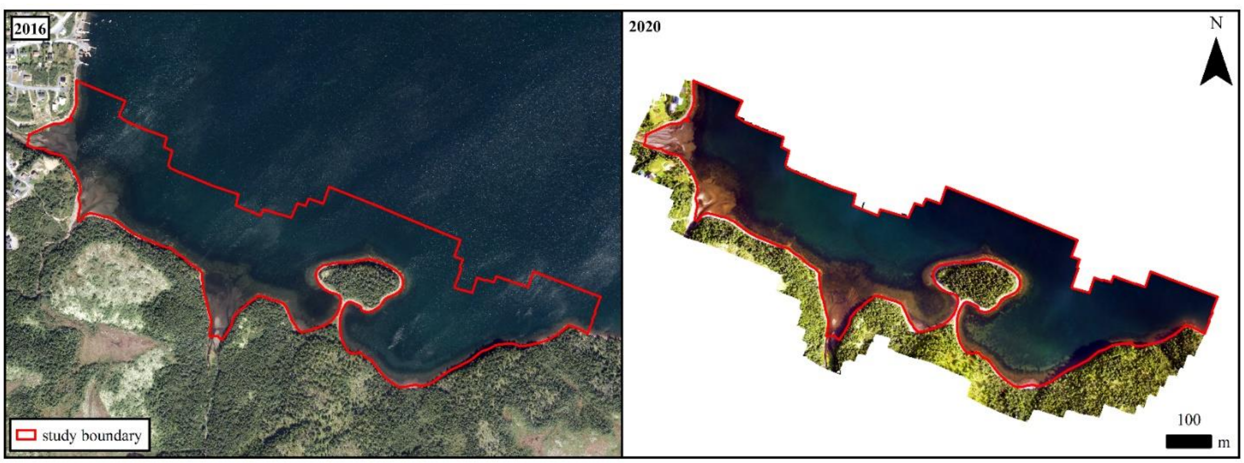

Figure D.4. Aerial imagery for Sunnyside in 2016 and 2020.

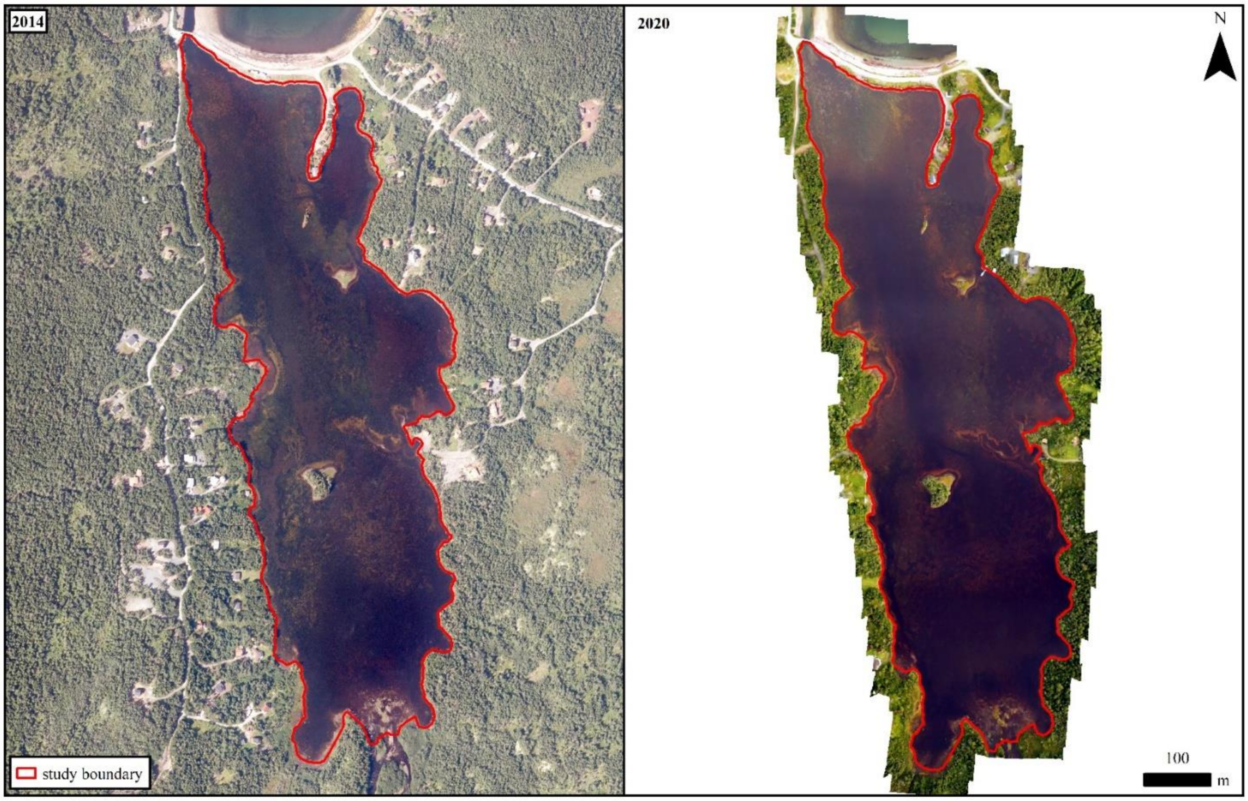

Figure D.5. Aerial imagery for Spread Eagle Pond in 2014 and 2020.

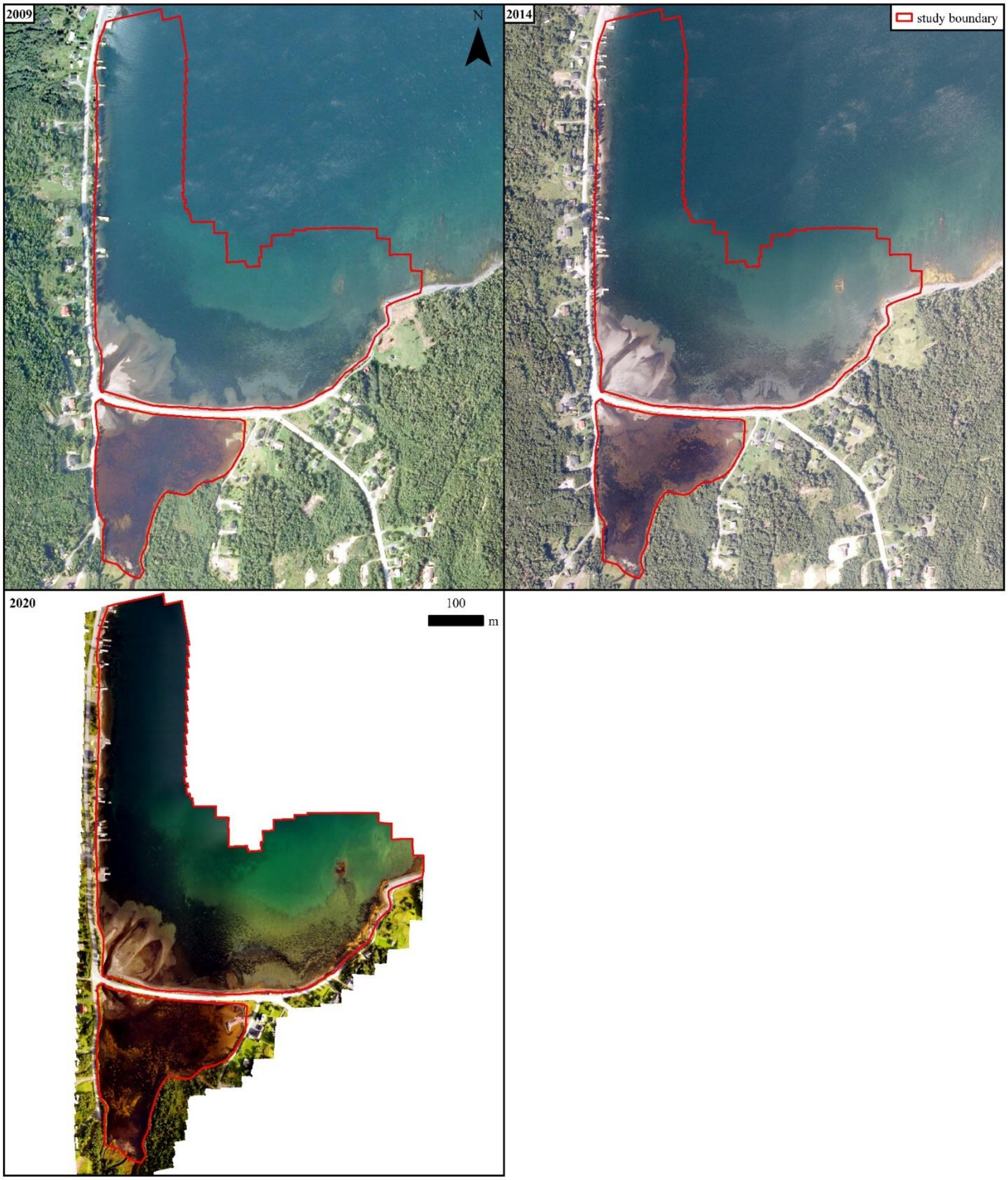

Figure D.6. Aerial imagery for Old Shop Pond in 2009, 2014, and 2020.