# EVALUATING HABITAT AND HABITAT USE BY AMERICAN EEL (*ANGUILLA ROSTATA*)/KATEW IN THE BRAS D'OR LAKE /PITU'PAQ ESTUARY USING BENTHOSCAPE MAPS AND ACOUSTIC TELEMETRY

by

Caelin Murray

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Dalhousie is located in Mi'kma'ki,
the ancestral and unceded territory of the Mi'kmaq.
We are all Treaty people.

*To my friends and family who supported me during this thesis.*

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# ABSTRACT

The American eel/Katew is a culturally significant and endangered species that has faced population declines on a global scale. The Bras d'Or Lake (BdOL)/Pitu'paq, Cape Breton, Nova Scotia, offers a unique environment for eel, yet habitat information for this species in the BdOL is limited. American eels are primarily a benthic species and habitat information is required to identify risks to the population. Using a Two-Eyed Seeing/Etuaptmumk approach, this study developed a benthic habitat map of the BdOL using multibeam echosounder survey bathymetry and backscatter data, relying on both existing data and through collection of new data. Acoustic telemetry was paired with local and Mi'kmaw knowledge to overlay eel presence and habitat across seasons. Eels used vegetated habitats in summer and overwintered on *Shallow Silt/Mud* habitat *(≤ 50 m)*. Using results from this study, co-management recommendations can be developed to provide stewardship of eel and eel habitat in this region.

# LIST OF ABBREVIATIONS USED

| | |
|---|---|
| BoF | Bay of Fundy |
| BPI | Benthic Positioning Index |
| BTM | Benthic Terrain Modeler |
| BdOL | Bras d'Or Lake |
| CHS | Canadian Hydrographic Service |
| DFO | Fisheries and Oceans Canada |
| FSC | Food, social, and ceremonial |
| GEE | Google Earth Engine |
| GSC | Geological Survey of Canada |
| HSI | Habitat Suitability Index |
| IKS | Indigenous Knowledge System |
| $Km^2$ | Square kilometers |
| m | Meters |
| MBES | Multibeam echosounder data |
| OBIA | Object Based Image Analysis |
| OTN | Ocean Tracking Network |
| ROV | Remotely Operated Vehicle |
| TASSE | Terrain Attribute Selection for Spatial Ecology |
| UINR | Unamak'i Institute of Natural Resources |
| WKS | Western Science Knowledge System |

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# CHAPTER 1: INTRODUCTION

## 1.1 STUDY SITE: BRAS D'OR LAKE/ PITU'PAQ

The Bras d'Or Lake (BdOL) located in the center of Cape Breton Island/ Unamak'i, Nova Scotia/Mi'kma'ki is a large (1,099 km $^2$) estuary with only three outlets to the sea and salinity ranging from 20-26 ppt (Fig. 1.1; Lambert, 2002). By comparison, the nearby Sydney Bight, which is more fully open to the North Atlantic Ocean, ranges in salinity from 28-32 ppt (Denny et al., 2013; Lambert, 2002). Many of the fish, invertebrate, and vegetative species found within the BdOL are representative of species found to occur along the Atlantic coast of Nova Scotia. However, the BdOL consists of many warm and shallow bays (< 30 meters) and deep (> 280 m) and cold pockets of water, which provide home to both arctic and sub-tropical species that arrived during historic events and still arrive (and thrive) today by the Labrador Current and the Gulf Stream (Hatcher, 2018; Lambert, 2002). Given its unique structure, the BdOL is home to a diverse range of species deemed rare outside this estuary (Hatcher, 2018; Lambert, 2002).

Due to its confined watershed geography, the BdOL experiences relatively low fetch and offers a semi-exposed to sheltered environment (Petrie & Bugden, 2002). Shorelines of the BdOL are necklaced with an abundance (> 400) of coastal lagoons (Fig 1.1; Ross, 2018; Taylor & Shaw, 2002). Coastal lagoons are nearshore water bodies separated from the main water body by a sediment barrier beach (Taylor & Shaw, 2002). These habitats consist of a mix of fresh to brackish water and are extremely vulnerable to anthropogenic impacts (Lambert, 2002). Throughout the BdOL, these habitats vary in their connection to the estuary, being partially to fully enclosed and, in some cases, serve as buffer zones, collecting sediment and pollutant run-off from near-by roads and preventing materials from entering the BdOL (Peterson et al., 1985; Ross, 2018). Coastal lagoons provide important habitat to several fish species throughout their life history as they offer natural protection from wind and waves, refuge from pelagic predators, foraging and overwintering grounds, and spawning and nursery habitats (Franco et al., 2006; Kjerfve, 1994; Ross, 2018).

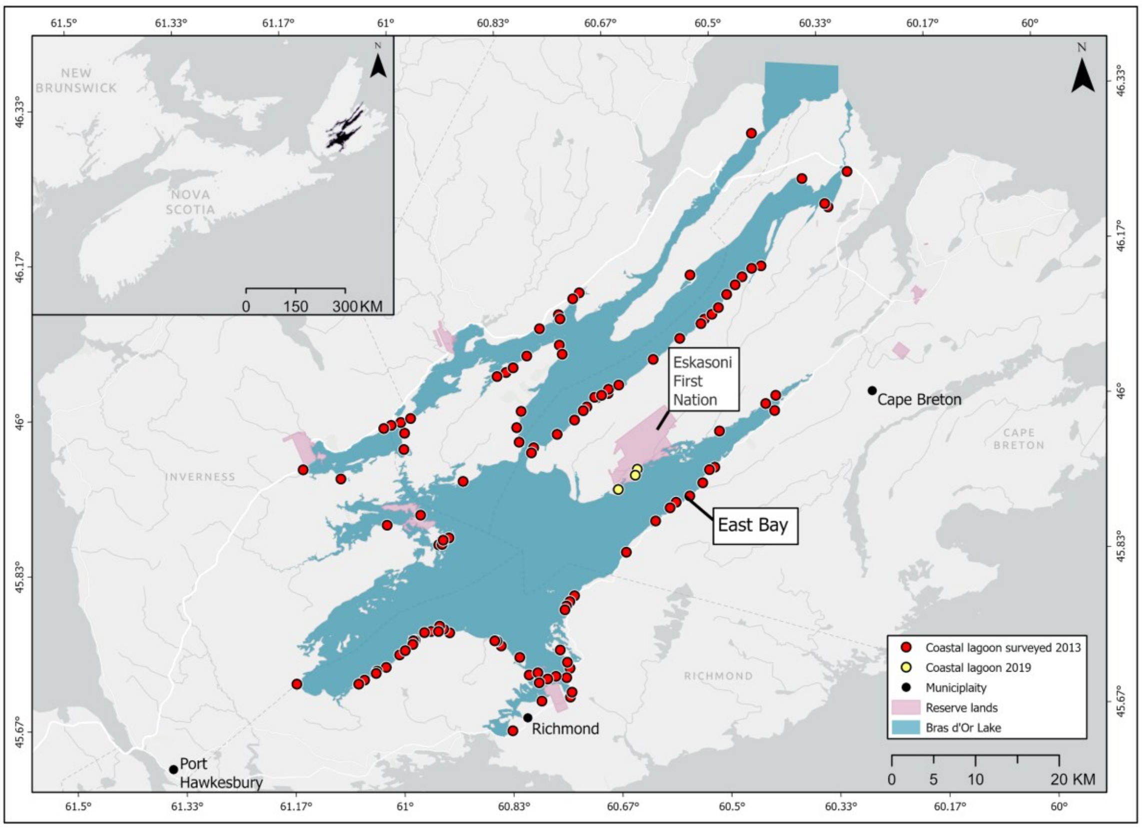



**Figure 1.1.** Map of the Bras d'Or Lake (BdOL) and a subset of coastal lagoons Cape Breton, Nova Scotia. While more than 400 coastal lagoons are documented to occur along the BdOL shoreline, this map represents only those surveyed in 2013 (red) as well as three used in this study (yellow) as this was the only file available to plot coastal lagoon locations in the BdOL. More sites can be found in Ross et al., 2018.

In the shallow sublittoral regions, there is a mix of soft sandy to fine grain sediments, with eelgrass (*Zostera marina*) to small cobble are present along sheltered shores and with large cobble to boulders and seaweeds present along more exposed areas and deeper regions (Lambert, 2002; Parker et al., 2007; Tremblay, 2002; Tremblay et al., 2005). Most of the seabed found in sublittoral regions of the BdOL is mud, however a mixture of soft and hard bottom substrata occurs as well (Shaw et al., 2006a). Information regarding habitats throughout the BdOL is rich but scattered. Previous work has collected information for nearly the entirety of the BdOL, however there has been no comprehensive map of habitats and existing data are not readily available to researchers (Shaw et al., 2006b; Vandermeulen et al., 2016).

The BdOL and its watershed were designated as a United Nations Educational, Scientific and Cultural Organization (UNESCO) Biosphere Reserve in 2011 (Hatcher, 2018). This was in part due to the work and advocacy of Elder Albert Marshall, who serves as the environmental spokesperson for all five of the Mi'kmaw communities in Unamak'i. These communities are highly dependent on the BdOL and coastal lagoons for commercial and subsistence fisheries, as well as for the transmission, practice, and adaptation of Mi'kmaq fishing knowledge (Giles et al., 2016; Hatcher, 2018). Several species, including American eel (*Anguilla rostrata*) and American lobster (*Homarus americanus),* are fished for food by Mi'kmaq in the BdOL (Denny et al., 2020).

## 1.2 STUDY SPECIES: AMERICAN EEL/KATEW

The American eel is a long-lived and single breeding species (Pratt et al., 2014). American eels have a wide geographic distribution in the Western Atlantic region, ranging from northern South America to Iceland (COSEWIC, 2012; Engler-Palma et al., 2013). In Canada, the species inhabits terrestrial watersheds, estuaries, and coastal marine waters connected to the Atlantic Ocean up to the mid-Labrador coast (COSEWIC, 2012). Anguillid eels are understood to be a facultatively catadromous species meaning that catadromy, the movement from sea to freshwater, is not required for eels to complete their life cycle (Daverat et al., 2006). American eels, therefore, demonstrate a variety of movement behaviors and some may be residents of freshwater or saline environments while others shift between the two until they mature and migrate as a silver eel (Daverat et al., 2006; Jessop & Iizuka, 2002; Thibault et al., 2007; Tsukamoto & Arai, 2001). It is hypothesized that anguillid eels at high latitudes may rely more heavily on estuaries for food, because of the comparatively high productivity of these areas (Beck et al., 2001; Tsukamoto & Arai, 2001). American eels also demonstrate greater growth rates and mature at younger ages in estuarian habitats compared to those in freshwater habitats (Cairns et al., 2009; Jessop et al., 2008; Morrison & Secor, 2003; Oliveira, 1999).

The American eel has five major life stages: leptocephali (larvae), glass eel, elver, yellow eel, and silver eel (Appendix A; COSEWIC, 2012). As leptocephali, eels drift by ocean currents throughout the western Atlantic, Gulf of Mexico, and Caribbean Sea to the continental shelf where they metamorphose into transparent glass eels (Atlantic States Marine Fisheries Commission, 2000; Walker et al., 2019). Glass eels migrate inshore toward freshwater rivers or estuaries to feed and become progressively more pigmented (brown in color) as they mature into elvers. After a few months, elvers develop into yellow eels and obtain a dark back and a yellow to white underside (Atlantic States Marine Fisheries Commission, 2000; COSEWIC, 2012; Walker et al., 2019). This stage is the longest phase in the eel's life cycle (5 to 40+ years) and is understood as the growth stage where sexual determination occurs (Atlantic States Marine Fisheries Commission, 2000; COSEWIC, 2012; Jessop, 1987; Walker et al., 2019). Yellow eels then develop into silver phase eels with a clean white underside and silvering sides. The silver phase is the final metamorphosis event when eels become sexually mature. During the silver phase, their eyes become larger and the stomach degenerates to prepare for the long ocean migration towards the Sargasso Sea to spawn (COSEWIC, 2012).

American eels are opportunistic omnivores and stomach content analysis has revealed that they are top predators in saltmarsh food webs (Eberhardt et al., 2015). In the BdOL, yellow eels may feed on detritus, small fish such as Atlantic silversides (*Menidia menidia*), cunner (*Tautogolabrus adspersus*), Atlantic salmon (*Salmo salar*), sticklebacks (*Gasterosteus aculeatus* ), alewife (*Alosa pseudoharengus*), and other eels, as well as invertebrates such as shrimps (*Crangon septemspinosa*) and green crabs (*Carcinus maenas*) (Denny et al., 2012; Lambert, 2002; Parker et al., 2007). American eels also serve as prey for other fish species such as striped bass (*Morone saxatilis*), salmon, and trout (*Salvelinus fontinalis , Oncorhynchus mykiss)*, mammals including phocid seals, mink (*Neovison vison*) and American martens (*Martes americana*), and fish-eating birds such as eagles (*Haliaeetus leucocephalus*), osprey (*Pandion haliaetus*), mergansers (*Mergus serrator*), and double crested cormorants (*Phalacrocorax auritus*) (Atlantic States Marine Fisheries Commission, 2000; Tomie, 2011; Weiler, 2011).

The American eel was listed as 'Special Concern' by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 2006 and was re-assessed and re-listed in 2012 as 'Threatened' due to population declines and the existing anthropogenic challenges from fishing and development that may continue to restrict the recovery of this species (COSEWIC, 2006, 2012; Pratt et al., 2014). For example, between 1996 to 2010 American eels in Lake Ontario and the St. Lawernce River, Canada, exhibited dramatic population declines, with a reported 65% decrease in the number of eels near maturity and in some parts of Ontario, greater than 90% decline in two generations (COSEWIC, 2006). Declines of American eel have also occurred in parts of the Maritimes, including the BdOL in 2008 and 2009 (Denny et al., 2013). Noticeable declines of eels in the BdOL were documented again in 2012 with Mi'kmaw fishers stating an increase in fishing effort with a decrease in catch (Denny et al., 2012). Anthropogenic effects, particularly dams that may block access to habitat or passages used for seaward migration, contaminants, and overfishing of juvenile life stages are listed as some contributors to eel decline (Castonguay et al., 1994; COSEWIC, 2006).

## 1.3 SIGNIFICANCE OF EELS TO INDIGENOUS PEOPLES

The American eel is a culturally significant species to Mi'kmaw for food and sustenance as well as cultural and ceremonial practices (Giles et al., 2016). Socially, eels are documented to bring community members together through fishing and feasting activities that strengthen community bonds and allow for the adaptation and transfer of Mi'kmaw fishing knowledge and language (Giles et al., 2016). These gatherings are important for sharing the catch with extended family, Elders, and other community members who cannot fish for themselves (SRSF, 2002). Guided by *Msit No'kmaq*, which translates to "all my relations", Mi'kmaq understand that in return for being nourished by these beings, humans should treat these spirits with respect and gratitude through their own behavior and through the offering of ceremonial gifts (SRSF, 2002; Weiler, 1990).

While the cultural significance of eels lies in Mi'kmaq relationships with the environment, the Mi'kmaq also have an Aboriginal right to fish for food, social, and ceremonial (FSC) purposes due to prior use and historical occupation of the lands and waters of Canada and a Treaty right to earn a moderate livelihood (Denny & Fanning, 2016; Unamak'i Institute of Natural Resources, 2007b). Treaty rights are negotiated rights that arose from formal agreements between the British Crown and Aboriginal leaders and are supported under the Peace and Friendship Treaties signed in the 1700's (Government of Canada, 2013a, 2013b). Two major court decisions, the 1990 Sparrow Decision and the 1999 Marshall Decision, have played a significant role in the recognition of Indigenous inherent and Treaty rights by the Canadian government, and these rights are recognized and affirmed under Section 35(1) of the Constitution Act (Denny & Fanning, 2016) (Giles et al., 2016; Supreme Court of Canada, 1990, 1999). The Supreme Court Decision in R.v. Sparrow was the first to apply Section 35 of the Canadian Constitution Act, 1982, which acknowledged and reaffirmed inherent rights to harvest resources for FSC purposes, limited only by conservation concerns (Giles et al., 2016). This decision established an obligation for Aboriginal people's to be consulted by the Crown when there is a possibility of infringement of Aboriginal rights. This decision for obligation to consult stemmed from previously recognized Aboriginal rights but is not a right itself.

Following this, the Mi'kmaq held renewed interest in commercial fisheries and established Mi'kmaq jurisdiction over their fishing activities (Milley & Charles, 2001). In response, the Government of Canada introduced the Aboriginal Fishing Strategy (AFS) agreements which provide access for FSC, yet restricts the number of fish that can be caught, fishing methods, and seasons (Unamak'i Instituite of Natural Resources, 2007b). Following the arrest of Donald Marshall Junior in 1993 for fishing and selling eels without a government-issued license, the Supreme Court of Canada, in R.v. Marshall (1999), acknowledged the Treaty right to hunt and fish for a moderate livelihood, a decision referred to as Marshall I. Only two months later, a second decision, referred to as Marshall II, was reached stating that conservation and other compelling and substantial public objectives would allow regulation over Treaty rights (Government of Canada, 2021). Through recognition of Aboriginal and Treaty rights, Indigenous Peoples across Canada have experienced a “different legal relationship with fisheries than non-Aboriginal Canadians" (Denny & Fanning, 2016; Harris & Millerd, 2010).

## 1.4 MANAGEMENT OF AMERICAN EEL FISHERIES

### 1.4.1 Federal management

The federal American eel fishery is managed by Fisheries and Oceans Canada (DFO) and is divided into two separate fisheries by life stage, one for elvers (elver fishery), and one for yellow and silver eels (adult fishery) (Table 1.1; Chaput et al., 2014). Elvers are heavily fished commercially and for commercial communal purposes, while adults are fished for FSC, as well as commercial, communal commercial, and recreational purposes (Fisheries and Oceans Canada, 2019a). Both the commercial and recreational elver and adult eel fisheries are regulated by DFO through a licensing policy with gear restrictions, size retention limitations, and catch retention limits (Fisheries and Oceans Canada, 2020a).

**Table 1.1**. Management of American eel fisheries in Canada.

| Fishery type | Elver fishery (Glass eels & Elvers) | Adult eel fishery (Yellow/Silver) | Managed by | Management measures |
|---|---|---|---|---|
| **Commercial** | Commercial<br>Commercial communal | Commercial<br>Commercial communal | DFO | Quotas, limited licensing, effort controls, mandatory catch reporting |
| **Recreational** | Individual | Individual<br>Unlicensed | DFO | Licensing (pots & traps),<br>Unlicensed (spearing & angling)<br>Effort controls, mandatory catch reporting |
| **Aboriginal** | Moderate livelihood (2021 Acadia & Bear River First Nation) | Food, Social, Ceremonial<br>Moderate Livelihood | DFO<br>Aboriginal Fishing Strategy | *Netukulimk:* taking only what is needed<br>*Msit no'kmaq:* all my relations<br>Ensuring enough for the next seven generations |
| **Catch value** | $5,200 per kg<br>Normally ~$3,000 per kg | ~$188 per kg | | |

Within the American eel fishery, there is no recreational fishery for elvers. However, the adult eel recreational fishery is further divided by gear type, with unlicensed spearing and angling and licensed eel pots and traps (Table 1.1). Recreational fishing via angling or spearing for adult eel in tidal waters is open year-round, though there is a 2-day closure time to allow for any changes (such as the opening or closing) to the current fishing season, and these fishers are not required to report their catch (Fisheries and Oceans Canada, 2020a; Bradford, 2013). Therefore, it is unclear the extent to which eel are caught via angling and spearing for food or bait, yet it is understood that many recreational fishers do not directly target eel (Atlantic States Marine Fisheries Commission, 2000; Bradford, 2013). Recreational fisheries that use eel pots and eel traps are regulated under licensing policy by DFO. There are currently 92 recreational licenses for adult eel in the Maritime Region of which 67 have been relinquished in exchange for a green crab license (R. Curwin, personal communication, 2020). Fisheries using eel pots have a one-day closure time to allow for any changes (such as the opening or closing) to the current fishing season (Fisheries and Oceans Canada, 2020a). Fisheries using eel traps occur between Aug 15- Oct 31 to coincide with the departure of silver eels migrating to the Sargasso Sea. In addition, it is likely that some economic incentive was also at stake in this decision as larger eels command a higher price (R. Curwin, personal communication, 2020).

Adult eels are significantly less profitable, valued at ~ \$188/kg compared to elvers valued at >\$5000/kg (Withers, 2021). The Maritimes region holds the only active commercial elver fishery in Canada with eight commercial elver fishery licenses (one occurs in the BdOL) and one commercial communal license (Fisheries and Oceans Canada, 2019a). These nine commercial license holders share a total allowable catch of just less than 1 tonne, or 9,960 kilograms (Withers, 2021). In the Maritimes region (Nova Scotia, New Brunswick, and P.E.I.), 173 metric tons of adult eel were caught in 2018 and valued at \$1,092,000 with Nova Scotia representing only about 10% of that catch (Fisheries and Oceans Canada, 2020b). Within Nova Scotia there are approximately 100 commercial eel licenses and 10 communal commercial licenses although it is unclear which of those are currently active (R. Curwin, personal communication, 2020; Denny et al., 2012; Giles et al., 2016).

Commercial data available for adult yellow eels in the BdOL indicated that the highest reported landings occurred in district 6 of East Bay (Appendix B) with 4.09 mt from 2000-2007 (Bradford, 2013). In the 2013 status of American eel report, data after 2007 in the BdOL were unavailable as there were too few active licenses (<5), and landings data were not yet available. Additionally, in 2008, no logbooks were distributed to commercial eel license holders in the Maritimes nor were records in 2009 available at the time information on the status of American eel in the Maritimes region was published (Bradford, 2013). Therefore, information on the extent of commercial fishing in the Maritimes and specifically the BdOL is not publicly available beyond 2007.

While commercial fisheries are regulated by licensing policy, gear restrictions, and catch limits by DFO, commercial communal licenses and the FSC fishery are managed under a separate section within DFO using AFS agreements (Table 1.1; Unamak'i Instituite of Natural Resources, 2007b). FSC licenses are provided for adult eels, but none are provided for elvers (Table 1.1; Fisheries and Oceans Canada, 2019a).

In Canada, Mi'kmaq have a Treaty right to fish in pursuit of a moderate livelihood, yet the quota of a moderate livelihood has yet to be defined. On November 18, 2021, the Government of Canada reaffirmed their commitment to advancing reconciliation and stated they are actively working with First Nations across the Maritimes and Gaspé region of Quebec to further support and implement the Treaty right while maintaining a healthy fishery for all harvesters for future generations (Government of Canada, 2021a). On that same day, the Government of Canada joined Acadia First Nation of Nova Scotia in announcing that members will be fishing in pursuit of a moderate livelihood during the 2021-2022 commercial seasons for lobster in LFA 33,34 and 35. Then, on June 25, 2021, Acadia First Nation and Bear River First Nation, both located in Nova Scotia, jointly presented the country's first-ever exploratory interim plan for a moderate livelihood fishery for elvers (Withers, 2021).

The elvers plan, conducted outside of federal management, is being discussed among Mi'kmaw community members and harvesters stating the purpose would be to "observe and document the upstream migration of the elvers on selected rivers in southwest Nova Scotia to identify potential locations and appropriate gear types for a future elver fishery on those rivers." (Withers, 2021). The plan would allow district harvesters to retain a small amount of those elvers for the livelihood fishery while also establishing an index for rivers that do not have an established fishery (Withers, 2021). DFO has not commented on the proposal, but did state that they are working with bands on their fishing plans and that moderate livelihood plans may include elvers, however the discussion is still ongoing (Withers, 2021). These efforts brought forward by Acadia and Bear River First Nations are initial key steps in helping to develop a co-management livelihood plan for elvers.

### 1.4.2 Mi'kmaw management

Traditionally, Mi'kmaq manage eels through *Netukulimk*, which translates to taking only what is needed and ensuring there is enough for future generations (Denny & Fanning, 2016). Guided by *Netukulimk*, Mi'kmaq implement several fishing strategies to manage the local eel population, including a seasonal rotation of fishing sites, fishing with traditional gear (spears, pots, traps), and adjusting fishing strategies annually based on the need and availability of eels (Denny et al., 2012). Spearing is the most common method for eeling in the BdOL, though eel pots and traps are also used (Denny et al., 2012). Spearing targets large eels that are 45 cm or greater; small eels are not targeted but if caught, they are considered a gift and communal sharing takes place after the catch (Denny et al., 2012).

Spearing for eels occurs between April and October and then begins again once there is ice cover (Denny et al., 2012). In the summer, Mi'kmaq have suggested that the first thunderstorm of the year indicates it is safe to begin eeling (S. Denny, personal communication, 2021) and they do not fish eels during blueberry season (August to early September) out of respect that eels may be migrating to spawn (Denny et al., 2012; A. Sylliboy, personal communication, 2021). In early to mid-fall, Mi'kmaq adjust their fishing strategies to capture eels moving into the lagoons to overwinter or those migrating out of the BdOL (Denny et al., 2012). Spearing occurs at night or during the early morning in coastal lagoons and along beaches where eelgrass is present (Denny et al., 2012).

## 1.5 IDENTIFYING & FILLING SHARED KNOWLEDGE GAPS

Canada has a legal obligation to consult with Indigenous peoples on matters that may impact Aboriginal and Treaty rights (Supreme Court of Canada, 2004a, 2004b, 2005). Indigenous peoples hold valuable knowledge and within the setting of true co-governance can improve decision-making for more equitable sharing of our marine and aquatic resources (Iverson, 2019; Supreme Court of Canada, 2004a, 2004b, 2005). Recently proposed amendments to the *Fisheries Act* have stated that traditional knowledge must inform fish habitat protection decisions and that these decisions must consider the adverse effects on the rights of Indigenous peoples (Government of Canada, 2019, 2021b). However, federal management, which is driven by western science, has not been effective in communicating with Indigenous communities nor making space for knowledge shared by these communities in government assessments and management plans (Giles et al., 2016; Iverson, 2019). This lack of connection and communication between government and Indigenous communities has prompted researchers to begin to acknowledge issues within the consultation process and to develop better relationships with Indigenous people.

Partnerships have been formed in academia by collectively gathering members from government, stakeholders, academics, and First Nations representatives together to share knowledge and gain a stronger collaborative understanding of the movements and distributions of culturally, ecologically, and economically significant species (Iverson et al., 2019; Nguyen et al., 2019; N. Young et al., 2016). These partnerships have been used to address socioeconomic and resource management issues in Canada for Greenland halibut (*Reinhardtius hippoglossoides*), Pacific salmon (*Oncorhynchus spp*., several species), and Atlantic salmon smolt (Béguer-Pon et al., 2017; Béguer-Pon, et al., 2015; Crossin et al., 2016; Iverson et al., 2019; Strople et al., 2018).

One such partnership is Apoqnamtulti'k (Mi'kmaw: "we help each other"), a three-year collaborative study with the goal of engaging diverse knowledge holders and embracing multiple ways of knowing to support the establishment of a fisheries co-management framework. The aim of Apoqnamtulti'k has been to pair Mi'kmaw and local knowledge with western science methods to learn more about the movements and habitat use of three commercially, ecologically, and culturally important species: American eel/katew, American lobster/jakej, and Atlantic tomcod/punamu (*Microgadus tomcod*), in the BdOL and the Bay of Fundy/Pekwitapa'qek. Apoqnmatulti'k places a key emphasis on co-learning and the exchange of knowledge which is critical for building trust and developing a cross-cultural understanding among members as each individual works within their own knowledge system, experiences, values, and biases. For example, while

I have learned to appreciate and value Indigenous Knowledge Systems (IKS), I myself am not Mi'kmaq and my own knowledge and background stem from a western science knowledge system (WKS).

WKS is based upon the scientific method which uses testable hypotheses and empirical data and has been elevated to the primary way of knowing with regards to the natural environment and resource management (Giles, 2014; Hassan & Hanapi, 2013; Weiss et al., 2013). This type of knowledge is derived from scholarly articles or books and is often shared in the form of written results, such as a report or a peer reviewed journal article, or an oral presentation (Giles, 2014). On the other hand, IKS weave spirituality, culture, beliefs, and environmental knowledge into daily life and practices (Carm, 2014; Giles, 2014). IKS are shared between individuals through stories or cultural and ceremonial practices where knowledge is transferred from one generation to another. Local knowledge stems from place-based knowledge and individual experiences and can be held by those within a western or Indigenous worldview (Berkes, 2003).

One approach to address integrating knowledge systems is through Two-Eyed Seeing/Etuaptmumk. The concept of Two-Eyed Seeing was developed by Mi'kmaw Elder Albert Marshall and Elder Dr. Murdena Marshall and is described as "learning to see the strengths of Indigenous knowledge from one eye, and with the strengths of western knowledge from the other eye and using both eyes together to benefit all" (Bartlett et al., 2012). Several other frameworks similar to Two-Eyed Seeing have been developed to integrate knowledge systems, yet, Two-Eyed Seeing is unique in its "notion that knowledge transforms the holder, and that the holder bears a responsibility to act on that knowledge", encouraging those learning to see through this lens and to take action

(Hatcher et al., 2009; Reid et al., 2020). Mi'kmaw values, which are guided by *Netukulimk* and *Msit No'kmaq*, acknowledge the interconnectedness of systems and the responsibility that is bestowed on us to care for those with whom we share territory (Giles et al., 2016).

Apoqnmatulti'k has incorporated a Two-Eyed Seeing approach to guide the use of western and Mi'kmaw knowledge systems in a way that supports collaboration and enhances the sustainability of shared resources. Using this framework, shared knowledge has been embedded in Apoqnmatulti'k from inception by using a community-based study design, as well as continued communication between all partners on decisions regarding study species, site selection, the placement of acoustic receivers, and methods for animal capture and tagging. Research questions and objectives for this thesis were continuously re-evaluated and driven by questions of this community to ensure research goals were aligned.

## 1.6 RATIONALE

American eels demonstrate a wide geographic range and a diverse use of habitat which has been suggested as a critical life history trait for the resilience and survival of this species (Daverat et al., 2006; MacGregor et al., 2008, 2009). Currently, there is a lack of knowledge available on habitat used by eels in estuaries. The importance of filling this knowledge gap has been identified by both western and Mi'kmaw knowledge holders, with the goal of preserving eels and eel habitat in culturally significant and historical fishing areas (Denny et al., 2013; Giles et al., 2016). Further in-field investigations are needed to provide specific information on habitat use at a finer scale in order to assess habitat used by eels and identify any potential risks to eels or eel habitat (Béguer-Pon et al., 2018; Denny et al., 2013; Giles et al., 2016).

Mi'kmaw knowledge and acoustic telemetry both demonstrate the presence of eels in coastal lagoons and nearshore habitats, with Mi'kmaw knowledge of eel movements being gathered across thousands of years and telemetry data representing only recent studies over the past decade. Previous work regarding habitat and newly collected ground-truthing video footage emphasized the range of suitable habitat available to eels in the BdOL (Nixon, 2015; Shaw, Taylor, et al., 2006; Taylor & Shaw, 2002; Tremblay, 2002; Tremblay et al., 2005; Vandermeulen et al., 2016). This thesis seeks to develop a deeper understanding of American eel movements and habitat use in the East Bay of the BdOL and specifically, within culturally significant nearshore habitats where relatively little knowledge is available on the spatial and temporal distribution of eels. Through this research, we seek to gain knowledge that will contribute to enhanced stewardship of eels, assess risks to this species and its habitat, and contribute to the equitable management of this declining species that acknowledges the Mi'kmaq relationship with eels and respecting their knowledge and values.

## 1.7 THESIS OVERVIEW

In Chapter 2, I aimed to determine whether benthic habitat types found in the BdOL could be classified using previously collected multibeam backscatter data along with previously and newly collected seafloor imagery datasets. Second, whether satellite imagery could be used to fill nearshore data gaps within East Bay to monitor eel habitat use in nearshore regions (performed in Chapter 3). Chapter 2 focuses on habitat classification and quantification through the use of the generated benthoscape map. The benthoscape map was then used to better understand eel habitat relationships in Chapter 3.

Eel movements and habitat associations were examined using acoustic telemetry data paired with knowledge shared by Mi'kmaq and local knowledge holders in Chapter 3. In Chapter 3, I aimed to determine whether eels captured and released in the coastal lagoon remained in that location year-round which can be useful to identify eel habitat use and threats to eel habitat. Chapter 3 focused on determining the spatial and seasonal movement of American eels in the BdOL, where eels are considered an estuarine resident species, to determine whether eels associated with certain habitats are linked to changes in season. I also aimed to identify which areas of the Lake and East Bay are used by eels and whether this could provide insights into habitat use during their yellow stage.

Understanding the movement and habitat use of eels through telemetry, mapping, and knowledge sharing is essential for developing co-management recommendations that guide stewardship of eels in a way that values and respects diverse values and ways of knowing. Through Apoqnmatulti'k, research questions addressed in this thesis were guided by knowledge shared with Mi'kmaq project partners and are driven by questions of the local community to encourage co-learning of the spatial distribution of eel and their associated habitats. This information will help fill shared knowledge gaps regarding habitat to contribute to identification of any potential risks to eels and eel habitat in this region. In Chapter 4, I discuss conclusions of the study, limitations, and directions of future research.

## 1.8 RESEARCHER POSITIONALITY

I am a non-Mi'kmaq female researcher from Unamak'i/Cape Breton, Mi'kma'ki/ Nova Scotia which is located on the ancestral and unceded territory of the Mi'kmaq People. I was born and raised in Unamak'i. My home is within driving distance to six of the thirteen Mi'kmaw communities in Nova Scotia. However, my educational background is derived from western science-based knowledge. My interest in science, mapping, the BdOL, and surrounding communities has led to my participation in this master's project.

During this project with local Mi'kmaw communities, I strived to actively understand and recognize my own knowledge system and bias while also learning to see using Indigenous ways of knowing. I believe that coastal communities, those that are most impacted by, and share an intimate relationship with place and species who inhabit it, should be incorporated into consultation and research projects as they are the ones who are most affected by these decisions and hold valuable knowledge of the area.

It was important to me to learn from and listen to the concerns and questions of the community during this project and when given a window to meet during the COVID-19 pandemic, an opportunity to hear stories shared by our community liaison as well as learn the Mi'kmaw names and meanings for species found in the BdOL. I am grateful for the experiences and relationships that have been built throughout this project as they have embraced the cultural differences and grit that comes with growing and learning from diverse cultures together all while reassessing the way we are taught to conduct research and define how species and ecosystems are managed. I am continuously impressed by Mi'kmaq and non-Mi'kmaw partners' willingness to share knowledge and readily ask questions. The willingness of partners to learn how to incorporate Mi'kmaw values into science-based research has been the most impactful to me.

# CHAPTER 2: HABITAT CHARACTERIZATION OF THE BRAS D'OR LAKE/PITU 'PAQ USING ACOUSTIC SONAR DATA AND SATELLIE IMAGERY

## 2.1 INTRODUCTION

The associations of marine animals and their habitats are often used by management to assess the quality and quantity of the habitat available to local species and to identify any potential risks to species and their habitat (Rudolfsen et al., 2021). Although observing and measuring the availability of habitat is important to understanding species-environment relationships, doing so in marine environments is difficult as it requires baseline knowledge of the ecosystem characteristics that are present in the region (Proudfoot et al., 2020).

Over the past 20 years, improvements in remote sensing technologies have enabled marine scientists to map seafloor environments through the adoption of a landscape scale approach, now commonly referred to as seascape ecology to generate benthoscape maps (Brown et al., 2011, 2012; Pittman et al., 2021). Benthoscape maps are fine scale biophysical maps of the seafloor which integrate both physical and biological elements that can be distinguished and delineated using remote sensing methods (Brown et al., 2012;  Pittman et al., 2021; Wilson et al., 2021; Zajac, 2008).

In deeper waters (approximately ≥ 30 m), beyond the reach of optical satellite remote sensing methods, multibeam echosounders (MBES) have become the survey tool of choice in mapping continuous baseline information of the seafloor (Brown et al., 2011, Brown et al., 2012; Harris and Baker, 2011; Misiuk et al., 2021). Bathymetry, collected by MBES data, and bathymetry derivatives, such as seafloor slope, curvature, and rugosity, can be used to understand the geomorphology of the seafloor (Brown et al., 2011; Lecours et al., 2017). MBES backscatter, a measure of the acoustic signal strength that is returned from the seabed, can be used to distinguish between substrate composition such as hard or soft bottoms and in some cases, biogenic components of the seafloor (e.g., biogenic reefs, dense algal beds, dense bivalve beds, corals) (Brown et al., 2011; Lurton et al., 2015; Wilson et al., 2021). The use of bathymetry, bathymetric derivatives, and backscatter data together is valuable for understanding benthic habitat and predicting species distribution within a region (Becker et al., 2020; Brown et al., 2011, 2012; Lecours et al., 2017; Monk et al., 2010; Rudolfsen et al., 2021; Wilson et al., 2021; Proudfoot, 2020).

Benthic habitat is described as an area of seafloor that is defined by specific abiotic characteristics such as substrate type or oceanographic conditions where bottom dwelling organisms live (Brown et al., 2011; Wilson, 2020). It is described by a species' realized niche with a set of conditions that are used by the organism or a place where an organism lives after interacting with other species (Hutchinson, 1957; Molles & Cahill, 2011). It is scale dependent, meaning that the use of benthic habitat may vary with life stage and size of the organism (Dutil et al., 1988; Morrison et al., 2003; Oliveira, 1999; Pratt et al., 2014). A study by Brown et al. (2011) emphasized the complexity and confusion associated with the term benthic habitat: “The term benthic habitat has no fixed definition and therefore can be confusing as it can be used to describe a range of attributes from the same geographical space and at different spatial and temporal scales such as a boulder providing habitat for a barnacle and the sand beneath the boulder providing habitat for polychaetae worms while a region of habitat may provide feeding habitat for demersal fish species.” (Brown et al., 2011). The terms 'benthic habitat mapping' and 'benthoscape mapping' are often used interchangeably in published literature to describe these types of mapping products, yet benthic habitat mapping is different than benthoscape mapping.

Marine benthic habitat mapping has been defined as "plotting the distribution and extent of habitats to create a map with complete coverage of the seabed showing distinct boundaries separating adjacent habitats" (Brown et al., 2011; MESH, 2008). Furthermore habitat is defined as "...both the physical and environmental conditions that support a particular biological community together with community itself…" (Brown et al., 2011; MESH, 2008). According to Brown et al. (2011) this definition of habitat suggests that to map biological patterns spatially, technologists need to impose distinct boundaries between adjacent and discrete habitat types. These boundaries can be mapped using remote sensing methods (acoustic or satellite) and these areas can be divided into spatial units with distinct boundaries representing discrete sediment or bedform characteristics (Brown et al., 2011). Benthoscapes are one such product generated from mapping the seafloor.

The term benthoscape is used to describe the geomorphology and biophysical features of the seabed such as coral reefs or mussel beds (Brown et al., 2011). Benthoscape maps represent the minimum mapping unit that can be spatially characterized and derived from remote sensing methods and represent discrete boundaries of habitat types. Differentiation between benthic habitat and benthoscape mapping can be as simple as the incorporation of seagrasses. For example, satellite imagery can detect the presence of vegetation but cannot see the substrate beneath this vegetation making it more than simply a benthic substrate map. Moreover, benthoscapes focus on the geomorphology and physical components of the seabed and organisms may use many benthoscape classes (mud and seagrass), or they may select a benthoscape class based on a combination of complex variables (e.g., temperature, salinity, oxygen, life stage, size) associated with habitat that may not be resolved using remote sensing methods (Brown et al., 2011).

Benthoscape classes represent distinct patches of habitats with clear boundaries distinguishing one habitat patch from another which may not be reflected in nature as they represent the minimal mapping unit of what can be derived from using this technology and spatial information. As a result they may not be truly reflective of how an organism might interact with or be present on a given benthoscape class as organisms may select habitat based on many complex variables (Brown et al., 2011; Strong et al., 2019; Wilson, 2020). Despite this, benthoscape mapping using remote sensing technologies has provided a valuable method to generate baseline data to characterize benthic habitats and to measure changes to these environments (Brown et al., 2011; Lecours et al., 2015; Proudfoot, et al, 2020a; Proudfoot et al., 2020b; Wilson et al., 2021).

Understanding how characteristics of, and changes in, benthic habitats affect distribution, abundance and life histories of fish and other benthic dwelling species is required to make informed decisions surrounding fisheries management and to enable better stewardship of benthic habitats. The ability to identify the range and composition of seabed characteristics and associated biodiversity in the form of benthoscape maps provides important baseline information that can be used to measure anthropogenic stressors such as increased sedimentation and harmful run-off of nutrients from surrounding landscapes. Resulting changes in benthic habitat can lead to changes in the ecology of species and community assemblages. As species die or migrate away from an area due to unsuitable benthic habitat conditions, species who are better suited to the new bottom conditions will replenish the area (Brown et al., 2012; Unamak'i Institiute of Natural Resources, 2007a).

Recent examples of benthoscape mapping approaches have demonstrated the benefits that these forms of spatial information can offer, including fisheries management applications, marine conservation, and the planning of Marine Protected Areas (Brown et al., 2012; Buhl-Mortensen et al., 2015; Caldwell, 2012; Copeland et al., 2013; Kostylev et al., 2001; Lacharité et al., 2018; Novaczek et al., 2017; Proudfoot et al., 2020; Smith et al., 2017; Walton et al., 2017; Wilson et al., 2021; Young & Carr, 2015). Specifically, benthoscapes can be used to help identify vulnerable or threatened habitats for species that are listed as special concern and can be used to guide monitoring and other restoration activities (Novaczek et al., 2017; Proudfoot et al., 2020; Rengstorf et al., 2013).

Similarly, advancements in the accessibility and affordability of satellite imagery have increased researchers' ability to remotely map and increase our understanding of benthoscapes in shallow (<30 m) coastal waters (Brown et al., 2011; Forsey et al., 2020; Traganos et al., 2018; Wilson, 2020). Several studies have demonstrated the use of satellite imagery to observe the distribution and predict the biomass of seagrasses in coastal waters (Forsey et al., 2020; Traganos et al., 2018; Webster et al., 2020; Wilson et al., 2020). Nearshore coastal areas are globally important habitats as they provide protection from predators, along with resting, foraging, and nursery grounds for juvenile fish and invertebrate species (Joseph et al., 2006; Lambert, 2002; Olson et al., 2019). Despite the importance of nearshore habitats and advancements in technology available for benthoscape mapping, nearshore areas generally remain poorly mapped (Forfinski-Sarkozi & Parrish, 2019; Leon et al., 2013). There is a need to begin combining methodologies to generate seamless benthoscape maps that cover the diversity of habitats from very shallow coastal waters to deep water which may be used by mobile species whose range varies both spatially and temporally throughout their life stages (Becker et al., 2020). Increased understanding of species habitat use can contribute knowledge needed towards developing recovery strategies for threatened species such as American eel *(Anguilla rostrata*) in the Bras d'Or Lake (BdOL), Cape Breton, Nova Scota (Becker et al., 2020).

Due to its diversity of habitats including the abundance of coastal lagoons, several channels and straits, many deep (> 250 m ) and shallow bays, and limited connection to the ocean the BdOL offers a unique ecological area and home for a variety of artic and sub-artic species not found along the Atlantic coast (Lambert, 2002; Parker et al., 2007; Petrie & Bugden, 2002). Several migratory pelagic and resident fish species occur throughout the BdOL, yet many of the resident species are demersal or bottom dwelling. Although appropriate habitat conditions may exist outside the BdOL for resident species, many of them do not appear to leave the system and instead complete their life cycles, or a portion of their life cycles, within the BdOL (Parker et al., 2007).

Of the 46 known fish species found to occur in the BdOL, 15% have been designated as endangered or of special concern by the Committee of Endangered Wildlife in Canada (COSEWIC) while many other species have also declined (see Appendix C). COSEWIC is comprised of an independent advisory panel which reports to the Minster of Environment and Climate Change Canada (COSEWIC, 2022a; 2022b). The advisory panel, consisting of academia, government, non-governmental organizations and members from the private sector meet twice a year and are responsible for compiling and analyzing the best available information about a species status in Canada and to provide this information to the federal government of Canada which will make the final decision of whether or not to assign a designation to a given species. Species must be listed by COSEWIC in order to be considered under the Species at Risk Act (SARA, COSEWIC, 2022a; 2022b).

Currently, the identification of habitats deemed essential or vital to population recovery (e.g., nursery, spawning, feeding, and wintering grounds) remains a key information gap for many commercial, recreational, and culturally significant marine species (Novaczek et al., 2017). Multiple sources of geospatial data have been collected on the geology and marine habitats occurring throughout the BdOL, yet a comprehensive habitat map of the BdOL does not currently exist (Nixon, 2016; Shaw et al., 2002, 2006; Taylor & Shaw, 2002; Tremblay, 2002; Tremblay et al., 2005; Vandermeulen et al., 2016; Vilks, 1967). It is crucial to understand and monitor the variety of habitats found within the BdOL, as this estuary plays a key role in supporting and maintaining a variety of marine life (Parker et al., 2007).

This study characterizes the benthic habitat found in the BdOL through the creation of a benthoscape map, to enable our understanding of species-habitat relationships and species distributions within this unique and valued region. Remote sensing methods such as those used in this study to generate the present benthoscape map are valued by Mi’kmaw as these methods provide non-destructive means to monitor, observe and measure habitat required to understand species-environmental relationships that honors the concept of *Netukulimk* (taking only what is needed and ensuring there is enough for future generations) and which is central to the Mi’kmaw worldview (Unamak’i Institiute of Natural Resources, 2020). Managers can also use this map to enhance stewardship and recovery strategies for other species, especially those that are threatened or at-risk in this region. The specific objectives of this study were to: 1) identify the benthic habitat types found in the BdOL from seafloor imagery datasets, 2) map shallow water (≤ 3 m) benthoscapes from satellite remote sensing data, 3) map deeper regions of the BdOL (> 3 m) using MBES datasets, and 4) combine shallow and deep water benthoscape maps to generate a seamless benthoscape map of the BdOL estuary.

## 2.2 METHODS

### 2.2.1 Apoqnmatulti’k

This study was part of a 3-year collaborative project ("Apoqnmatulti’k”, Mi’kmaw for “we help each other”) that is built on both Two-Eyed Seeing/Etuaptmumk, which combines the strengths of Indigenous knowledge with those of western knowledge, and a community-based study design in order to understand the movements and habitat use of ecologically, commercially, and culturally significant species in relation to their ecosystem characteristics. In the context of Apoqnmatulti’k, the generation of a continuous benthoscape map will allow Mi’kmaw decision makers, along with non-Mi’kmaw representatives, to observe changes to benthic habitats outside places of observation and in deeper waters while providing baseline knowledge needed to measure the rates of these changes especially in coastal lagoons and along the shoreline.

Additionally, a more comprehensive map of the seabed may aid in decisions regarding choice of fishing gear for key species. For instance, eel spears are designed for different habitats such as *Nikoql* where one type is used for hard bottom and another for soft, as well as *Netawemkewe'l* which is used for mud bottoms (Denny et al., 2012). However, fishing gears may also be adjusted based on season, with the hard bottom spear used in winter to penetrate deeper into the bottom to capture eels that bury deeper in winter (Denny et al., 2012).

### 2.2.2 Study site: The Bras d'Or Lake ecosystem

The BdOL ecosystem is a large (1,099 km²) and unique estuary located in the center of Cape Breton Island, Nova Scotia, Canada (Fig. 2.1). The unique physical structure of the BdOL offers very shallow to very deep (>250 meters) pockets of water, limited connection to the nearby Atlantic Ocean, and miniscule tidal impact with low flushing rates (Yang et al., 2007). The BdOL ranges in salinity from 20-26 ppt, with more enclosed inlets near 18ppt and areas more fully connected to the North Atlantic Ocean ranging from 28-32 ppt (Lambert, 2002; Strain & Yeats, 2002; Yang et al., 2007). In 2011, the BdOL was designated a United Nations Educational, Scientific and Cultural Organization (UNESCO) Biosphere Reserve and is currently under consideration by Canada for designation as some form of Marine Protected Area (Environment and Climate Change Canada, 2018; Hatcher, 2018). In this study, an emphasis was placed on the East Bay region of the BdOL as this area was outlined by Giles et al. (2016) and surrounding Mi'kmaq communities as important areas to preserve for American eel/Katew, an ecologically, commercially, and culturally significant species.

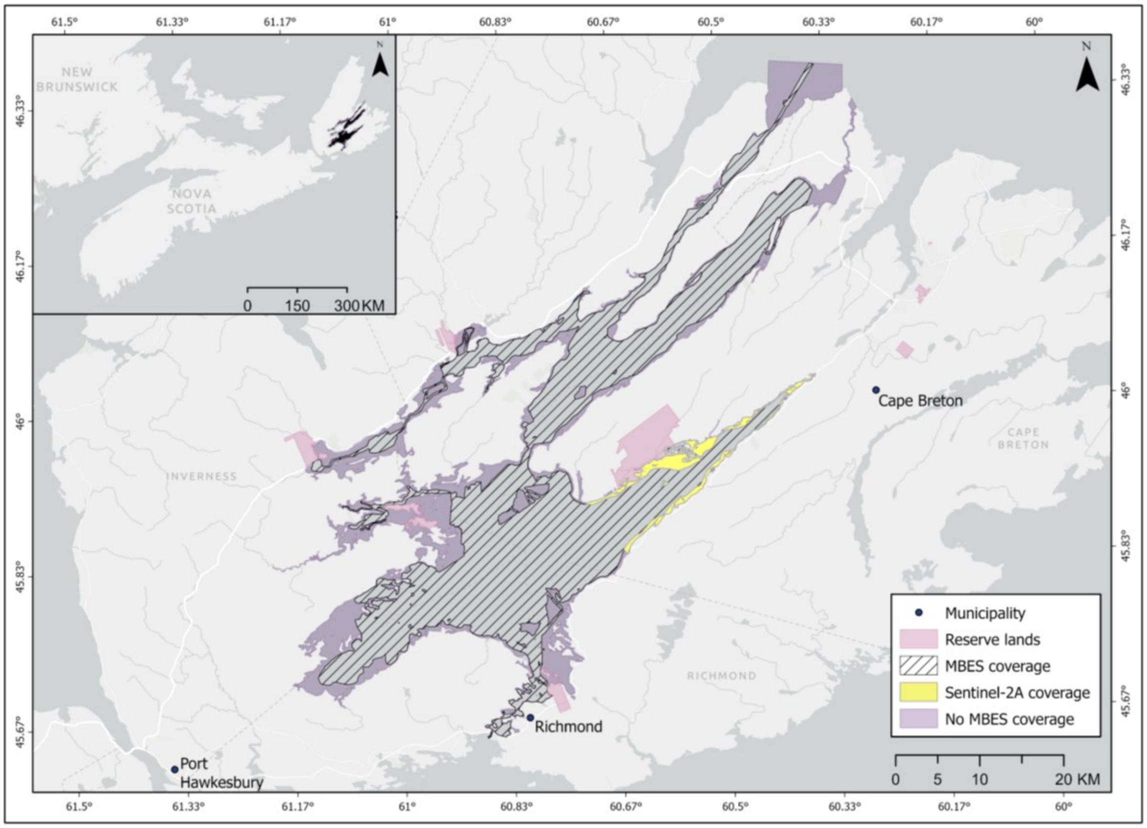



**Figure 2.1.** Map of the Bras d'Or Lake (BdOL), Cape Breton, Nova Scotia. The stripped black lines symbolize the Multibeam Echosounder Data (MBES) coverage collected by the Canadian Hydrographic Service (CHS) and the Geological Survey of Canada (GSC) from 1999-2003. The pink areas represent Reserve lands. The yellow area represents the Sentinel-2 level 2A satellite imagery coverage used to fill nearshore habitat gaps in this study, while purple areas represent no MBES data coverage.

### 2.2.3 Acquisition and preprocessing of remotely sensed data

Benthic habitats found in the Eastern region of the BdOL were characterized using a combination of previously and newly collected data. These data were used to guide an unsupervised classification, resulting in a continuous classified benthoscape map. The workflow of these methods can be found in Figure 2.2 and is further explained below. Benthoscapes within the BdOL were classified for the entire area that had MBES coverage including all three entrances to the nearby ocean.

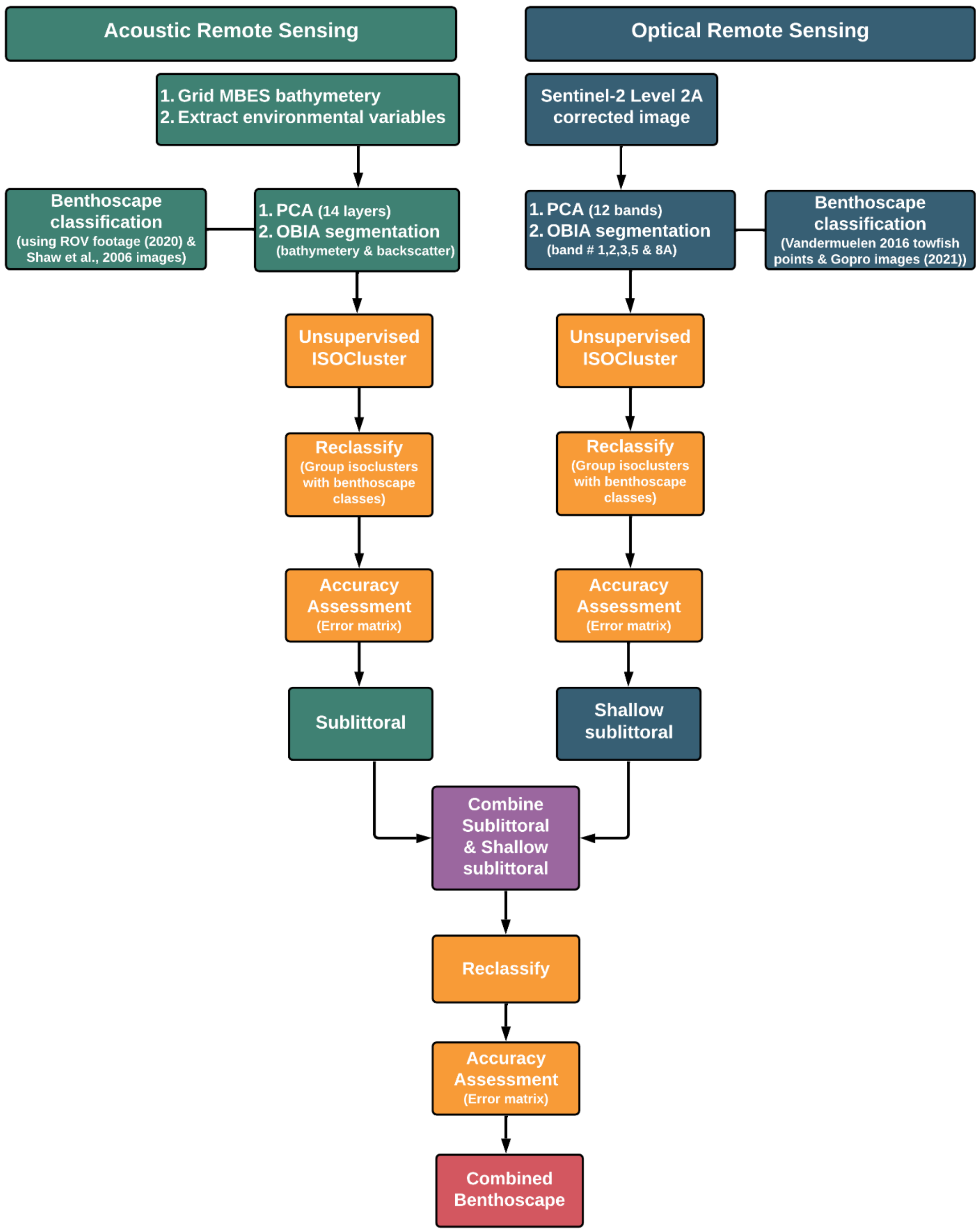


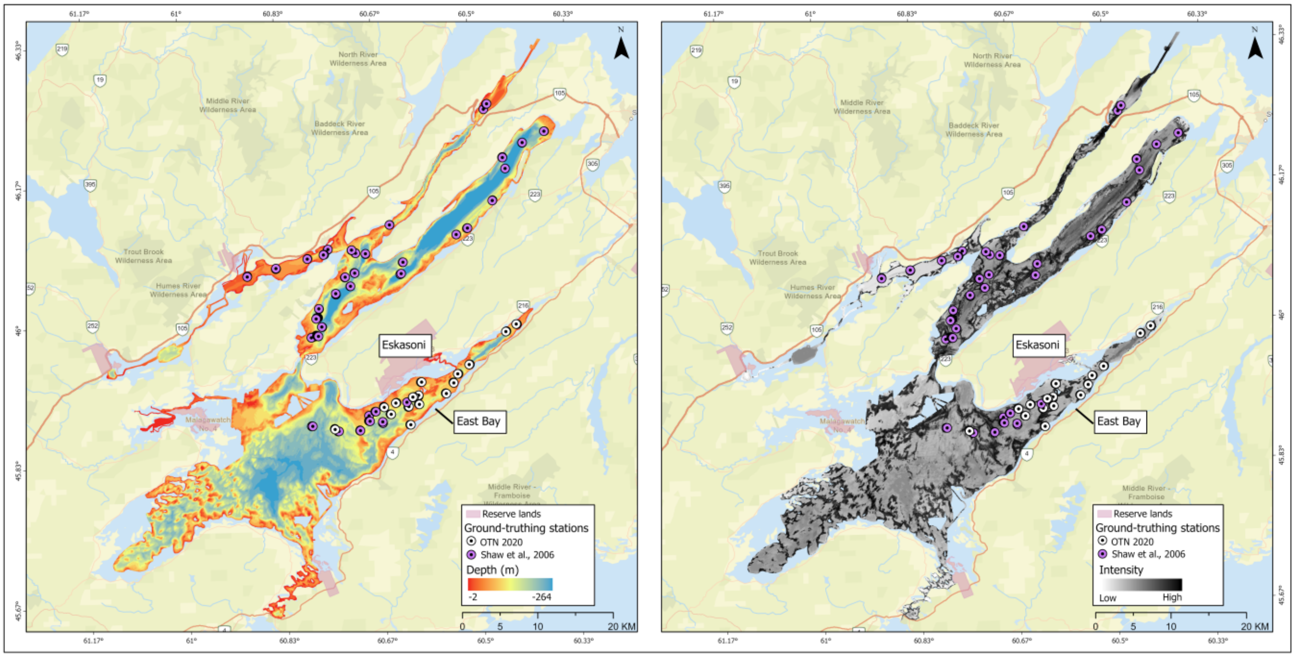

**Figure 2.2.** Methodological workflow showing both the acoustic remotely sensed data and satellite remotely sensed data using Object Based Image Analysis segmentation and benthoscape classification process.

### 2.2.4 Acoustic remotely sensed data

This study drew heavily on the data generated from previous multibeam echosounder (MBES) surveys conducted within the BdOL over four years:1999, 2000, 2002, and 2003, in depths ranging from >2 to 264 meters and covering approximately 777.6 $km^2$ (Shaw *et al*., 2005). Surveys were conducted by the Canadian Hydrographic Service (CHS) and the Geological Survey of Canada (GSC). Bathymetry and backscatter data were collected using two Kongsberg MBES systems: EM1000 (95kHz), and EM3000 (300kHz) (Griffin, 2003; Shaw et al., 2005). The EM3000 data were collected from the CCGS Matthew which was a base for the hydrographic survey launch CSL *Plover*, deployed in 1999. The EM1000 data were collected from the CCGS Fredrick G. Creed in 2000, 2002, and 2003 (Griffin, 2003; Shaw et al., 2005). Positional accuracy for all surveys was between 2 to 10 m horizontal accuracy and a vertical accuracy of 1 cm (Shaw et al., 2005). Survey lines were conducted at various spacing throughout the BdOL to obtain a 200% coverage of the seafloor in depths greater than 20 m.

MBES data used in this study were collected by CHS and processed by GSC. Bathymetric data were processed by CHS using Caris HIPS v.5.0 to apply sound velocity and tidal corrections using a tidal station in Baddeck provided by the CHS (Shaw *et al*., 2005). Geometric and radiometric corrections were applied to the MBES backscatter data by the GSC using inhouse tools. Backscatter data were gridded by GSC at 10 m resolution and a processed .asc file was provided to CHS from GSC (Shaw et al., 2005). Bathymetry data used in this study were provided by CHS in ASCII xyz format at 2 m resolution. The 2 m bathymetric data were gridded and resampled to 10 m resolution and clipped to the extent of the backscatter data using Global Mapper v.22.1. A resolution of

10 m was chosen as this was the highest resolution that could be achieved given the processed backscatter data that were provided by CHS (Fig. 2.3).




a) b)

**Figure 2.3.** Multibeam Echosounder Data (MBES) from 1999-2003 collected by Canadian Hydrographic Service (CHS) and Geological Survey of Canada (GSC). a) Multibeam bathymetry and b) Multibeam backscatter. The white circles represent newly collected ground-truthing sites by OTN in 2020 and purple circles represent ground-truthing sites used for habitat classification collected by Shaw et al (2006).

The environmental layers (Table 2.1) local mean, standard deviation of bathymetry, easterness, northerness, slope, and relative deviation from the mean value were generated from the bathymetry using the Terrain Attribute Selection for Spatial Ecology (TASSE) toolbox using a neighborhood window of 3 (default settings) and saved as geographical un-projected .asc files (Lecours et al., 2016). Lecours (2017) stated that these six variables, when used together, can describe most of the variation in terrain properties and local topographic features. Fine scale Benthic Position Index (BPI) (Table 2.1) was derived using an inner radius of 5 and an external radius of 10 and a scale factor of 100. Broad Scale BPI (Table 2.1) was derived using an inner radius of 10 and an outer radius of 50 and a scale factor of 500. These variables have been successful in other benthic habitat mapping studies to generate benthoscape maps (Lacharité et al., 2018; Wilson et al., 2021), and were therefore also incorporated into the data analyses.

### 2.2.5 Satellite remotely sensed data

Sentinel-2 level 2A satellite, launched 23 June 2015 (European Space Agency, 2021a), was chosen for this study as this satellite offered open-source imagery with a high (5 day) temporal and spatial (10-60 m) resolution that was at a comparable resolution to the previously collected MBES data. Copernicus Sentinel-2 Level 2A top of atmosphere and bottom of atmosphere reflectance corrected satellite data were downloaded and clipped to the East Bay region of the BdOL using Google Earth Engine (GEE) and used to supplement areas where there was no MBES coverage (Fig. 2.1, Fig. 2.2). All 12 bands of Sentinel-2 satellite imagery were downloaded and resampled to 10 m upon export from GEE. Upon searching for images within the Sentinel 2A library, the search was filtered to only look for and select images with less than 5% cloud cover in GEE. Bands 1, 2, 3, 4, 5, and 8 were brought into SNAP, an open-source desktop version software for ESA Toolboxes, to make use of and explore Earth Observation data (European Space Agency, 2021b). The Sen2Coral toolbox in SNAP was used to remove glint for bands 1, 2, 3, 4, 5, and 8 as a reference band. The glint corrected bands (1, 2, 3, 4, 5, 8) and remaining bands (6, 7, 8A, 9, 10, 11) were then brought into ArcGIS Pro for subsequent analyses (section 2.4 below).

**Table 2.1.** Environmental variables used in the Principal Component Analysis for classification. Bathymetry and backscatter data were derived from MBES data while layers 2-14 were derived from bathymetry.

| Layer # | Component | Description | Units | Toolbox |
|---|---|---|---|---|
| 1 | **Bathymetry** | Water depth. | meters | - |
| 2 | **Backscatter** | A measure of the intensity of the acoustic signal returned from MBES. Provides information on bottom characteristics such as softness or hardness. | dB | - |
| 3 | **Relative deviation from mean value (bathymetry)** | A measure of relative position that identifies peaks (positive high values) and pits (negative low values) (Lecours et al., 2017). | meters | TASSE |
| 4 | **Easterness** | A component of aspect that informs on the orientation of the slope, i.e., its deviation from east. It ranges between -1 (fully West) and 1 (fully East) (Lecours et al., 2017). | - | TASSE |
| 5 | **Northerness** | This is the second component of aspect that informs on the orientation of the slope, i.e., its deviation from north. It ranges between -1 or fully South and 1 or fully North (Lecours et al., 2017). | - | TASSE |
| 6 | **Slope** | Identifies steepness or gradient (ESRI, 2016c). | degrees | TASSE |
| 7 | **Standard deviation** | A measure of roughness (Lecours et al., 2017). | meters | TASSE |
| 8 | **Local mean** | Mean water depth, useful if the original bathymetry layer is noisy (Lecours et al., 2017). | meters | TASSE |
| 9 | **Aspect** | Identifies the slope direction/maximum rate of change of the downslope direction from one cell to its neighbors (ESRI, 2021). | degrees | TASSE |
| 10 | **Curvature** | Direction of maximum slope (ESRI, 2016a). | (1/100) of a z unit (meters) | Curvature Tool |
| 11 | **Planar Curvature** | Curvature of surface in the direction of the slope (ESRI, 2016a). | (1/100) of a z unit (meters) | Curvature Tool |
| 12 | **Profile Curvature** | Curvature of surface perpendicular to the slope direction (ESRI, 2016a). | 1/100 of a z unit (meters) | Curvature Tool |
| 13 | **Fine BPI** | A measure of where a referenced location is relative to the locations surrounding it. Fine-scale BPI identifies smaller features within the benthic landscape such as narrow crests or lateral mid-slope depressions (Goes et al., 2019; Weiss, 2000). | meters | BTM |
| 14 | **Broadscale BPI** | A measure of where a referenced location is relative to the locations surrounding it. A broad-scale BPI identifies larger features within the benthic landscape such as large depressions or significant changes in slope or elevation (Goes et al., 2019; Weiss, 2000). | meters | BTM |

### 2.2.6 Ground-truthing data

Ground-truthing points (n = 721) were compiled from 174 stations using previously collected ground-truthing photos by Shaw et al (2006) (n=72 stations) and Vandermuelen et al (2007; 2016) (n=77 stations), and newly collected ground-truthing photos by the Ocean Tracking Network (OTN, 2020) (n= 17 stations) and using the GoPro (n= 8 stations) to develop a benthoscape schema to classify benthic habitat in the BdOL (Table 2.2; Fig. 2.2). Ground-truthing of MBES data made use of previously collected ground-truthing images (n=262 images) by Shaw et al. (2006) (see Shaw et al., 2006 for survey methodology). Ground-truthing for Sentinel 2A satellite imagery was comprised of previously collected and classified points by Vandermuelen (2007; 2016, see Appendix D) obtained from underwater video clips (see Vandermeulen et al., 2007, 2016 for methodology). In 2020, several sites were selected to collect new ground-truthing data in deeper regions. These sites were chosen based on the MBES backscatter data that represented several types of hard and soft substrates in the Eastern region of the BdOL.

New ground-truthing in these deeper regions was collected by OTN's field team in 2020 using two separate remotely operated vehicles (ROV) each fitted with a forward-facing camera (Table 2.2; Fig. 2.2). Sixteen transects, 100 m in length, were collected between October 20-22, 2020, using a Blue ROV2. Duration of each transect was approximately 10-15 minutes with 6 minutes of total bottom time. An additional transect, 69 m in length, was collected on June 24 and 27, 2020 using a Falcon Saab Seaeye ROV. This transect was approximately 30 minutes in duration, with 11.5 minutes of total bottom time. In nearshore areas, a series of 8 locations was chosen based on accessibility from shore to collect new ground-truthing photos in depths ≤ 3 m (Table 2.2; Fig. 2.2). New ground-truthing data on shallow nearshore areas were collected using a GoPro Hero 7 Silver camera and a 73x73 cm quadrat for a reference frame (n=57 photos) between July 13 and August 02, 2021. Quadrant drops were collected randomly from the waterline with a minimum of three quadrant drops per station.

**Table 2.2.** Seafloor images assembled from previously collected and newly collected ground-truthing datasets used to guide classification for the benthoscape map. The first column shows the ground-truthing data source, followed by the number of images classified in each benthoscape class, the total number of stations, and the device which captured the image used for ground-truthing.

| | Benthoscape class | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | **Coarse Sed.** | **Silt/Mud with < 50 % Gravel** | **Mixed Sed.** | **Shallow Silt/Mud (≤ 50 m)** | **Deep Silt/Mud (≤ 50 m)** | **Cont. Veg.** | **Patchy Veg.** | **Total # of images** | **Total # stations** | **Source of image** |
| Shaw et al., 2006 | 2 | 8 | 45 | 169 | 38 | 0 | 0 | 262 | 72 | • Photos of sediment samples on deck collected using a 0.1 cubic mere van Veen grab sampler (n=36 images).<br>• Scorpio underwater camera and an "ice-hole" camera developed by GSCA (n=125).<br>• Video stills (n=101 images). |
| Ocean Tracking Network, 2020 | 9 | 15 | 12 | 145 | 0 | 0 | 0 | 167 | 17 | • BLUE ROV2 video still (n=160 images)<br>• Saab Seaeye Falcon ROV video still (7 images) |
| Vandermeulen 2007;2016 | 63 | 0 | 0 | 44 | 0 | 48 | 80 | 235 | 77 | • Classified point by towfish with Shark Marine underwater video SV-16 camera. |
| GoPRO, 2021 | 4 | 0 | 0 | 15 | 0 | 26 | 12 | 57 | 8 | • GoPRO quadrant drops (0.75 x 0.75 quadrant) |
| **Total** | **78** | **23** | **57** | **373** | **38** | **74** | **92** | **721** | **174** | |

Positioning for ROV transects was achieved using known coordinates at the beginning of each transect and followed a known heading and bearing for the length of the transect. Transect waypoints were plotted in Google Earth and coordinates were measured and extracted at distance intervals of 10 m and used to geo-reference images. To calculate timestamps at 10 m intervals, speed was calculated as total time in seconds divided by distance covered and was considered constant. A still image was extracted at each 10 m interval time stamp and georeferenced. In nearshore areas, coordinates were collected by positioning a handheld Garmin GPS over the center of the quadrat. Both the coordinates and time were recorded in a field notebook for each image. Extracted and geo-referenced seafloor images were then classified based on their biophysical characteristics such as the dominant substrate type according to the Wentworth scale and Folk method (Folk, 1954; Wentworth, 1922). Once data were extracted and initially analyzed, the final dataset was compiled into a 'master file' and validated or reclassified if needed against the extracted image. Once ground-truthing data were complete, the mode or most frequent benthoscape class was extracted from each station or transect in R so that only a single image would fall within each segmented object. These aggregated images were then used to validate the unsupervised assigned class against ground-truthing images to obtain the overall accuracy of the classification performed.

### 2.2.7 Image classification

Once all ground-truthing images that would be used to guide classification in the deeper regions of the BdOL were compiled into a master file, images were re-grouped into four general benthoscape classes based on what was observed to be consistent and distinguishable among both the ROV images and the ground-truthed images collected by Shaw et al. (2006) (Fig. 2.4). Class 1: *Coarse Sediments* was comprised of mostly hard bottom (cobble, pebble gravel, occasional boulder) with coarser fine sediments present and were not covered with a veneer of silt/mud (Fig. 2.4). Class 2) *Mixed Sediments* were comprised of a mix of soft and hard bottoms (cobble, pebble gravel, occasional boulder, often covered with a veneer of mud) (Fig. 2.4). This class was similar to Class 1 but contained a fine sediment component (Fig. 2.4). Class 3) *Silt/Mud with < 50% Gravel* consisted of classes that were comprised predominately of silt and mud with a small proportion of coarse-grained substrate (**c**obble, pebble gravel, occasional boulder) (Fig. 2.4). Finally, Class 4): *Shallow Silt/Mud (≤ 50 m)* was the most abundant ground-truthing class collected. The *Shallow Silt/Mud (≤ 50 m)* class consisted of soft substrata such as fine silt, mud and in some cases likely clay bottom with no evidence of hard sediments (Fig. 2.4). Notes on attached vegetation were made for all photos in the deeper region which consisted of 0 - < 2% appearing as either single strands of dead eelgrass (*Zostera marina*) or red algae's in nearly all photos while no kelp was observed in any of the deeper regions sampled.

The approach of compiling all ground-truthing images into a master file and re-grouping into distinct classes was also performed for the shallow region in East Bay. The shallow region made use of previously collected and classified points by Vandermuelen (2007; 2016) and newly collected ground-truthing by Murray (2021). Using the ground-truthing points collected and classified by Vandermuelen et al. (2007; 2016; n = 613 points) 235 points were randomly selected to combine with the 2021 nearshore ground-truthing data (n=57 photos) to guide the shallow water benthoscape classification (Table 2.1; Fig.2.2). In the shallow regions, the density of vegetation was described as patchy (sparse, 0 - 25% coverage) or continuous (dense 75-100% coverage). Initially, vegetation from the Murray (2021) dataset was classified as sparse (0-25% coverage), medium (25-75% coverage), and dense (75-100% coverage). However, there were not enough photos classified as medium vegetation to guide classification and these classes were re-assigned to match those observed from the Vandermuelen et al., (2016) dataset.

Overall, Vandermuelen et al. (2007; 2016) identified four main nearshore classes: Continuous eelgrass, Patchy eelgrass, Rocky eelgrass, and Sandy Mud bottom. These classes, including the non-vegetated benthoscapes, outlined by Vandermuelen (2007; 2016) served as a baseline for naming shallow water classes in this study. As a result, the compiled data sets for the shallow region grouped images into four main classes: *Continuous Vegetation*, *Patchy Vegetation*, *Coarse Sediments*, and *Shallow Silt/Mud (≤ 50 m)* (Fig. 2.4). Using these classes, the non-vegetated classes in the shallow region overlapped with the classes observed in datasets of the deeper region of the BdOL and therefore would allow for a seamless benthoscape map once shallow and sublittoral regions were combined.

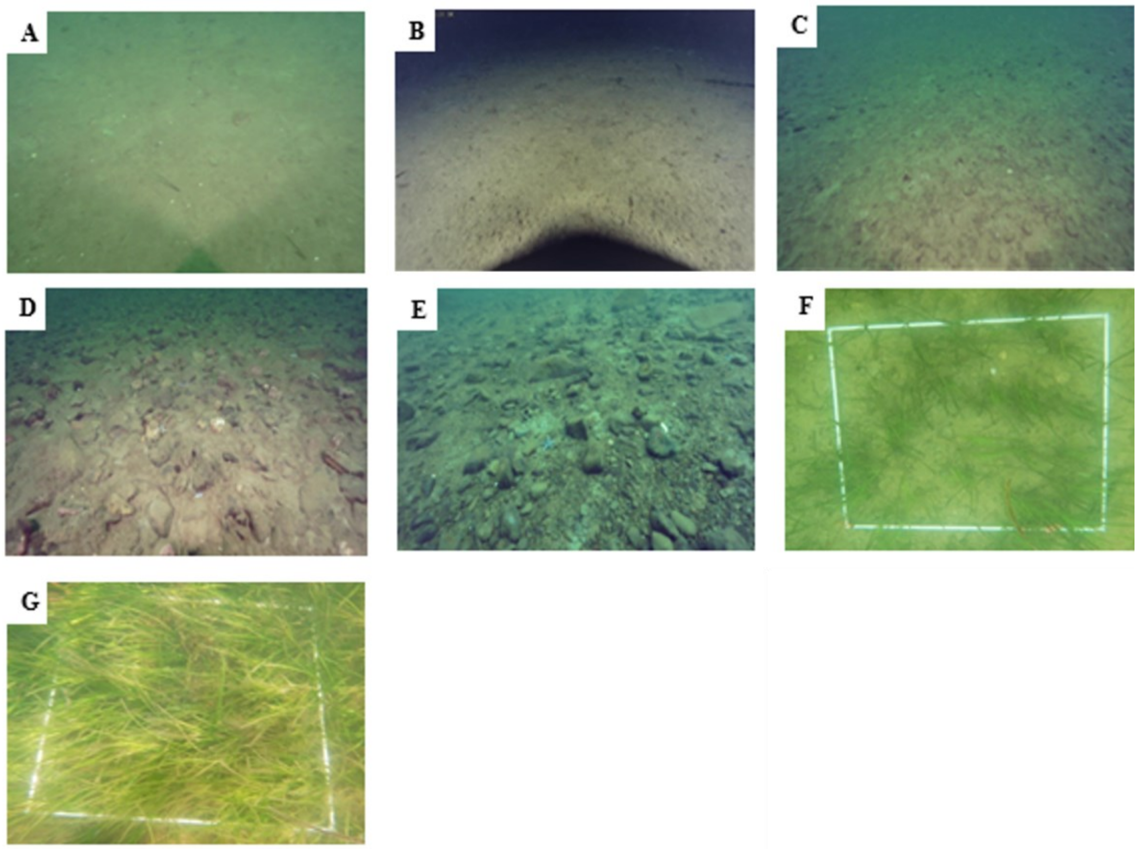



**Figure 2.4.** Images representing benthoscape classes in the Bras d'Or Lake: (A) *Shallow Silt/Mud (≤ 50 m)*; (B) Deep Silt/Mud (≥ 50 m); (C) *Silt/Mud with < 50% Gravel*; (D) *Mixed Sediments* (Gravel and cobble with a veneer of mud and without visible presence of fine sediments); (E) *Coarse Sediments* (Gravel and cobble without visible presence of fine sediments); (F) *Patchy Vegetation* (Sparse: 0-25% cover); and (G) *Continuous Vegetation* (Dense: 75-100% cover). Field of view (FOV) in images A-E is approximately 10° horizontally with a tilt range of ± 90°. Quadrant dimensions in the nearshore habitat photographs (F and G) are approximately 73x73 cm.

### 2.2.8 Unsupervised classification

### Unsupervised classification and segmentation of MBES

All layers derived from bathymetry and backscatter data (Table 2.1) were rescaled from 0 to 1 using the Rescale tool in ArcGIS Pro applying a linear method. Using the rescaled data as input layers, a principal component analysis (PCA) was run on all 14 MBES data layers and used to generate a raster based on the first three principal components that accounted for at least 95% of the variance (Fig. 2.2). The three principal components were used to generate a RGB color composite raster to be used as an input raster for the Iso Cluster analysis. A PCA was run to reduce correlation between variables used for classification (Jollife & Cadima, 2016; Shlens, 2014).

The Iso Cluster method performs clustering of the multiband raster (PCA) and the output results in a signature file that can be used as the input for the classification tool that guides and generates the an unsupervised classification raster (ESRI, 2016b, 2022a). The Iso data clustering algorithm determines the characteristics of the natural groupings of cell attributes in space and stores the results in an output ascii signature file (ESRI, 2022a). The Iso Cluster analysis is similar to K-means clustering.

The Iso Cluster algorithm is an iterative process that computes the minimum Euclidean distance when assigning each cell to a cluster (ESRI, 2022a).The process begins with arbitrary means being assigned by the software with one for each cluster and every cell is assigned to the closest of these means in multidimensional attribute space (ESRI, 2022a). Using this algorithm, new means are re-calculated for each cluster based on the attribute distances of the cells that belong to the cluster after the first iteration (ESRI, 2022a).This process is then repeated with each cell being assigned to the closest mean in multidimensional attribute space and new means are calculated for each cluster based on the membership of cells from the iteration (ESRI, 2022a). Using the Iso Cluster analysis, a minimum of 2 clusters is required and the default setting is set at 20 clusters. There is no maximum number of clusters. Finally, the specified number of classes value is the maximum number of clusters that can be generated from the clustering process.

The number of clusters output in the signature file described earlier may not be the same as the number specified for the number of desired classes for several reasons. First, the values of the input multiband raster data and the initial cluster means may not be evenly distributed. For example, in certain ranges of cell values, the frequency of occurrences for these clusters may be next to none (ESRI, 2022a). Consequently, some of the originally predefined cluster means may not have a chance to absorb enough cell members (ESRI, 2022a). Secondly, clusters consisting of fewer cells than the specified minimum class size value (2 clusters) will be eliminated at the end of the iterations (ESRI, 2022a). Third, clusters merge with neighboring clusters when the statistical values are similar after the clusters become stable (ESRI, 2022a). Some clusters may be so close to each other and have such similar statistics that keeping them apart would be an unnecessary division of the data (ESRI, 2022a). While Iso Cluster analysis was used, this clustering and classification was also driven by segmentation derived from the bathymetry and backscatter data in efforts to capture heterogeneity between classes in multidimensional space. Segmentation of the MBES dataset was completed using the bathymetry and backscatter data with a spatial detail of 20, a spectral detail of 10, and a pixel size of 10 m using ArcGIS Pro (Fig. 2.2; Fig. 2.5).

An object-based image analysis (OBIA) and unsupervised Iso Cluster classification was applied to the PCA and segmented raster of the MBES dataset in ArcGIS Pro (Fig. 2.2). Using the Iso Cluster classification, the default number of clusters is 20 and the minimum number of clusters is 2 yet the maximum number of clusters remains undefined. Therefore, this study used the multivariate clustering tool to determine the optimal number of clusters to set for the Is Cluster algorithm. To determine the optimal number of clusters, the Multivariate clustering tool within ArcGIS Pro was run on the PCA RGB input raster. The Multivariate clustering tool evaluates the optimal number of clusters given the input data by computing a pseudo-F-statistic for clustering solutions between 2 and 30 clusters and a default of 20 clusters (ESRI, 2022b). In the resulting chart computed by the Multivariate statistics tool, the largest pseudo-F-statistic values indicate solutions that perform best at maximizing both within and between cluster similarities (ESRI, 2022b).

Since the Iso Cluster procedure does not specify a guide to determining the optimal number of clusters, the number associated with the largest pseudo-F statistic values can be used (ESRI, 2022b). The largest pseudo-F-statistics in the MBES were greatest between 20 and 26 clusters with no significant decreases in the pseudo-F-statistic values beyond 26 clusters. Iso Cluster classification was then run on the PCA RGB multiband raster and segmented bathymetry and backscatter raster for several iterations using an optimal number of clusters between 20 and 26 clusters. The final iteration settings used a maximum of 22 classes, a maximum of 20 iterations, a maximum number of cluster merges per iteration set to 10 cluster merges, a maximum merge distance of 0.2, a minimum samples per cluster set to 2, and a skip factor of 10 was used with segment attributes checked for active chromaticity color, standard deviation, and compactness. It is suggested that when increasing the number of clusters, the number of iterations should also increase as the value should be large enough to ensure the migration of cells from one cluster to another is minimized and allow clusters to become stable (ESRI, 2020).

The mode or most frequent benthoscape class from each station was used to facilitate merging and reduction of the 22 unsupervised Iso Clusters into four benthoscape classes and produce a new classified map. The purpose of using the mode of each ground-truthing station was used to ensure only a single image was assigned to a given object. An error matrix was generated in ArcGIS Pro upon each iteration of reducing Iso Cluster classes to ensure the highest overall accuracy could be achieved and to determine Cohen's Kappa Statistic (Fig. 2.2).

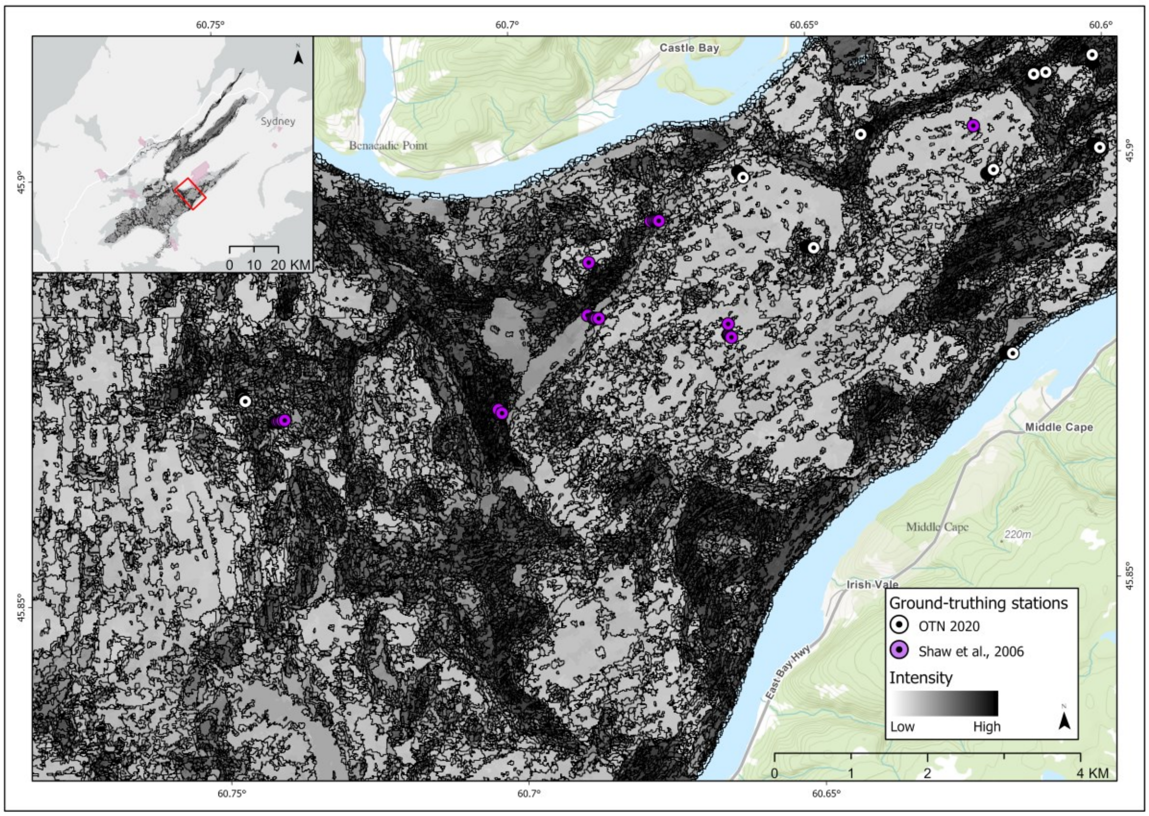
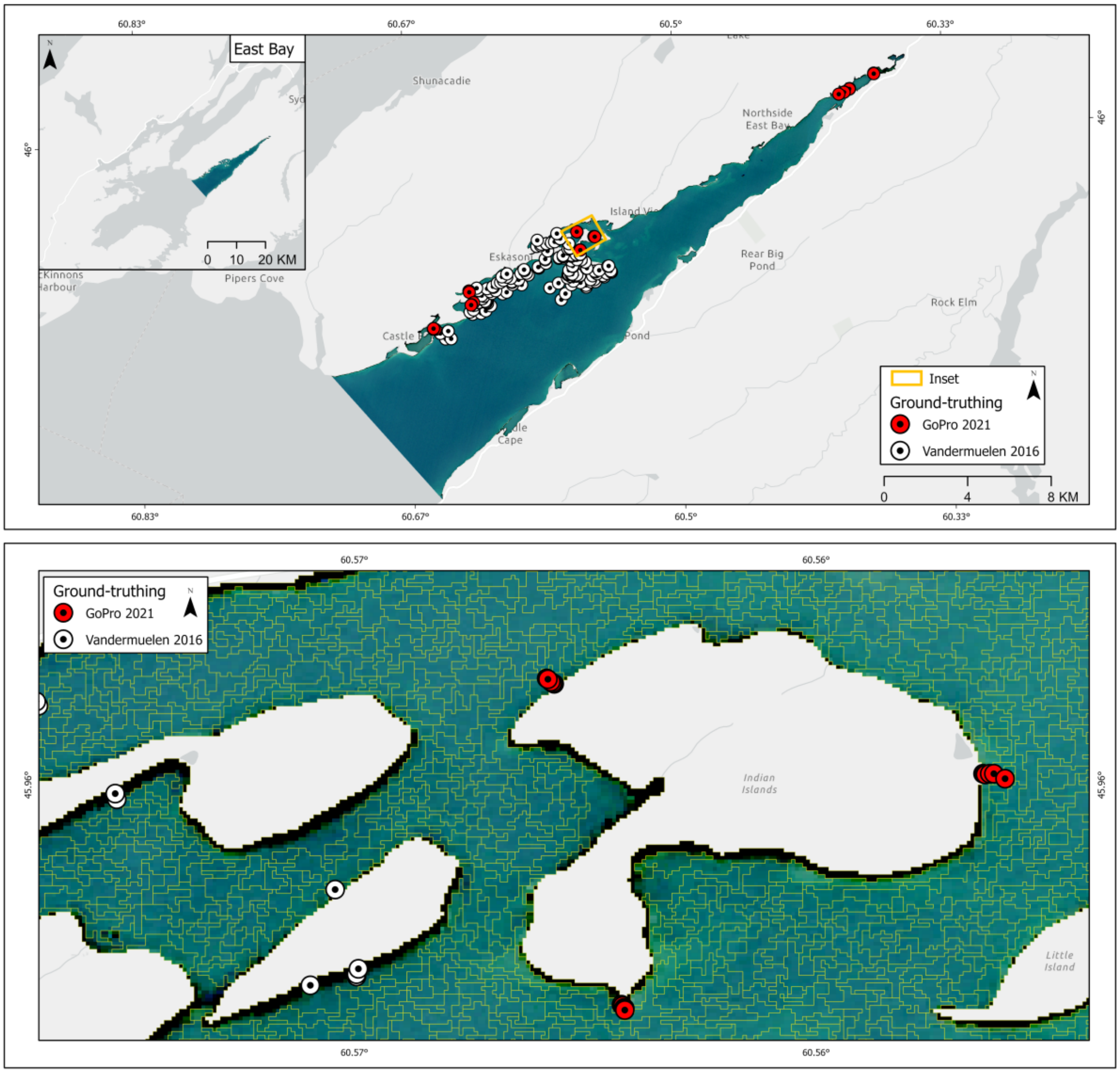



**Figure 2.5.** MBES bathymetry and backscatter combined raster used for segmentation with the segmentation file overlaid. Ground-truthing stations from Shaw et al. (2006) are in purple circles and ground-truthing stations from OTN, 2020 represented by white circles. The red outline inset map represents the area of the main map.

## Unsupervised classification and segmentation of satellite imagery

For satellite imagery, the Jeffries-matusia distance method was used in GEE to examine the 12 original bands and a suite of band indices similar to Forsey et al. (2020) to identify the best band selection for classification of my data in addition to other studies. The purpose of the Jeffries-matusia distance method is to assess whether desired classes are not only spectrally different but are spectrally and significantly different from one another (Richards, & Jia, 2006). Following the Jeffries-matusia distances, the final decision was to make use of all 12 original bands in the Sentinel-2 satellite imagery as no spectral difference between classes or band indices were found. A PCA was run on all 12 bands, which were rescaled from 0 to 1 using the *Rescale* tool and used as input bands to generate a raster based on the first three principal components that accounted for at least 95% of the variance. The three principal components were used to generate a RGB color composite for the Iso Cluster analysis. Segmentation of the satellite imagery was generated using the glint corrected bands (1,2,3,4,5 and 8A) and used a spectral detail of 20, a spatial detail of 15, and pixel size of 10 m (Fig. 2.2; Fig. 2.6).

An OBIA and unsupervised Iso Cluster classification was applied to the PCA output of the 12-band satellite imagery in ArcGIS Pro (Fig. 2.2). To determine the optimal number of clusters, the Multivariate clustering tool within ArcGIS Pro was run on the segmented raster dataset. The largest pseudo-F-statistics in the satellite datasets were greatest between 20 and 26 clusters with no significant decreases in the pseudo-F-statistic values beyond 26 clusters. Iso Cluster classifications were then run on the satellite imagery input data (PCA and segmented raster's) for several iterations using an optimal number of clusters between 20 and 26 clusters. Once an optimal number of clusters was found, assessment of the best matched classes between each Iso Cluster and ground-truthed images was used to facilitate merging and reduction of Iso Clusters into benthoscape classes and produce a new classified map. An error matrix was generated upon each iteration of reducing Iso Cluster classes to ensure the highest overall accuracy could be achieved and to determine Cohen's Kappa Statistic (Table 2.3).




a)

b)

**Figure 2.6.** (a) Corrected Sentinel-2A satellite image with ground-truthing stations from Vandermuelen et al. (2016) (white circles) and newly collected ground-truthing in 2021 (red circles); b) Segmentation overlaid on corrected satellite image. The orange outline in figure (a) represents the inset displayed in figure (b).

### 2.2.9 Integration of maps

Following classification of the MBES and satellite imagery separately, the two classified benthoscape maps were combined into a single classified benthoscape map (Fig. 2.9) in ArcGIS Pro using *Mosaic to new raster.* A ≥ 6 m depth contour generated from the multibeam bathymetry was applied to the nearshore classification, and the *reclassify* tool was used to reclassify vegetated pixels (*Patchy* or *Continuous*) ≥ 6 m as *Shallow Silt/Mud (≤ 50 m)* and recorded the area ($km^2$) and count of pixels reclassified (Fig. 2.9). In the BdOL, it is documented that eelgrass is not common in depths ≥ 6 m (Tremblay et al., 2005; Vandermeulen et al., 2016). The same method above was also used for reclassifying *Shallow Silt/Mud (≤ 50 m)* class into *Shallow Silt/Mud (≤ 50 m)* and *Deep Silt/Mud (≥ 50 m)* and the area ($km^2$) and count of pixels reclassified were recorded (see Appendix E). A final accuracy assessment (error matrix) was conducted using the merged raster and compiling all ground-truthing validation points and training areas (Table 2.5).

## 2.3 RESULTS

### 2.3.1 Multibeam echosounder data classification

Unsupervised classification of the 14 data layers derived from the MBES data (Table 2.1), using the object-based image analysis approach for segmentation and the Iso Cluster procedure, resulted in a classified image with an optimum of 22 Iso Cluster classes (Fig. 2.7). These 22 Iso Cluster classes were reduced into four classes to match the four benthoscape classes determined by our ground-truthing datasets (n=161 objects) (Table 2.2; Table 2.3). *Coarse Sediments* (16 objects, Table 2.2) corresponded with IsoCluster classes 0,1,4,9,11 and 20 (14.3%) (Table 2.3; Fig. 2.7), and were located on areas with the strongest backscatter returns. This class was often confused with *Shallow*

*Silt/Mud (≤ 50 m)* (Table 2.3). Class 2: *Mixed Sediments* (Gravel and cobble with a veneer of mud) (2 objects; Table 2.2), occurred on areas with high to medium backscatter intensities and was equally confused with *Coarse Sediments*. *Mixed Sediments* and corresponded with IsoCluster classes 5 and 12 (4.8%) (Table 2.3; Fig. 2.7). Class 3: *Silt/Mud with < 50% Gravel* (18 objects; Table 2.3) corresponded with IsoCluster classes 6,13,15 and 21 (33.3%) (Table 2.3; Fig. 2.7) and were associated with lower MBES backscatter returns. *Silt/Mud with < 50% Gravel* was equally confused with *Mixed sediments* and often confused with *Coarse Sediments* and less confused for *Shallow Silt/Mud (≤ 50 m)* (Table 2.3). Class 4: *Shallow Silt/Mud (≤ 50 m)* (124 objects; Table 2.3) corresponded with IsoCluster classes 2,3,7,8,10,14,16,17,18, and 19 (85.0%), in areas associated with very low MBES backscatter. This class had the most objects identified corerctly yet some confusion occurred among all three classes: *Mixed sediments*, *Silt/Mud with < 50% Gravel*, and occasionally *Coarse Sediments* (Table 2.3; Fig. 2.7). Overall accuracy for the MBES sublittoral benthoscape was determined to be 62.7% with a kappa statistic of 0.57% (Table 2.3). A kappa value of 0.40-0.80 indicates moderate agreement while a kappa < 0.4 represents poor agreement and >0.80 is considered strong agreement (Landis & Koch, 1977). *Shallow Silt/Mud (≤ 50 m)*depth were reclassified as *Deep Silt/Mud (≥ 50 m)* (1,391,225 pixels, area= 139.13 km$^2$, see Appendix E).

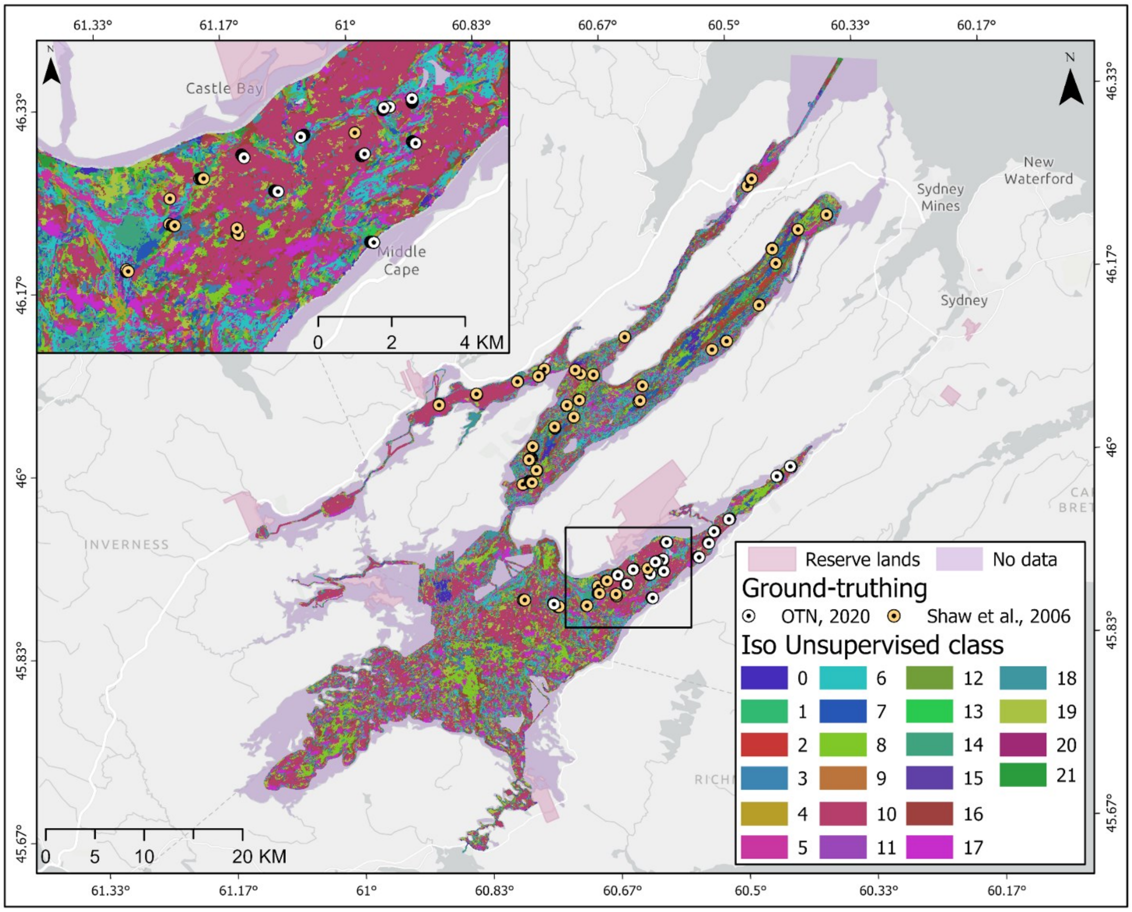



**Figure 2.7.** Results of the Iso Cluster unsupervised classification of the PCA raster derived from the MBES input layers (Table 2.1). Image objects derived from application of an object-based image analysis segmentation of the bathymetry and backscatter layers were classified into 22 classes from the PCA raster. The white circles represent ground-truthing sites collected in 2020 by OTN and the orange circles indicate ground-truthing sites derived from Shaw et al., 2006

**Table 2.3.** Error matrix for the multibeam sublittoral benthoscape classification.

| | Map (Iso Cluster) Classes MBES data | | | | | | |
|---|---|---|---|---|---|---|---|
| | **Coarse Sediments** | **Mixed Sediments** | **Silt/Mud with ≤ 0% Gravel** | **Shallow Silt/Mud (≤ 50 m)** | | | |
| **Ground-truth (Benthoscape) class** | **0+1+4+9+11+20** | **5+12** | **6+13+15+21** | **2+3+7+8+10+14+16+17+18+19** | **Total (no. of objects)** | **User's Accuracy (%)** | **Omission Error (%)** |
| Coarse Sediments | **2** | 0 | 0 | 14 | 16 | 12.5 | 87.5 |
| Mixed Sediments | 1 | **1** | 0 | 0 | 2 | 50.0 | 50.0 |
| Silt/Mud with ≤ 50% Gravel | 4 | 6 | **6** | 2 | 18 | 33.3 | 66.7 |
| Shallow Silt/Mud (≤ 50 m) | 7 | 14 | 12 | **91** | 124 | 73.4 | 26.6 |
| **Total objects** | 0 | 0 | 0 | 0 | **161** | **Overall Accuracy: 62.7%** | |
| **Producer's Accuracy (%)** | 14.3 | 4.8 | 33.3 | 85.0 | | **Kappa Statistic: 0. 57** | |

## 2.3.2 Sentinel 2A satellite classification

Unsupervised classification of the 12 original bands derived from Sentinel 2A satellite imagery (PCA output) using an object-based image analysis approach on the segmented raster and the Iso Cluster procedure resulted in a classified image with an optimum of 22 Iso Clusters (Fig. 2.8). Similar to the MBES data, the most frequent benthoscape class from the georeferenced satellite imagery was matched against the unsupervised Iso Cluster raster and the 22 Iso Cluster classes were reduced and grouped into four benthoscape classes as determined by our ground-truthing datasets (n= 292 objects; Table 2.2) (Fig. 2.8; Table 2.4). Class 1: *Continuous Vegetation* were located along areas closest to shore and along the perimeters of islands (Fig. 2.8). *Continuous Vegetation* (76 objects; Table 2.2) corresponded with Iso Cluster classes 1,2,5, and 14 (57.9%) (Table 2.4. Fig. 2.8). This class was confused with both *Patchy Vegetation* and in some cases bare *Shallow Silt/Mud (≤ 50 m)* bottom (Table 2.4). Class 2: *Patchy Vegetation* (119 objects; Table 2.2), were associated with Iso Clusters classes

0,4,10,12,15,16,18 (63.8%) (Table 2.4; Fig. 2.8) and was occasionally confused with *Continuous Vegetation* and *Shallow Silt/Mud (≤ 50 m)*. Class 3: *Shallow Silt/Mud (≤ 50 m)* (68 objects; Table 2.2) corresponded with most Iso Cluster classes (3,6,8,9,13,17,19,20) (57.6%) and was nearly equally confused with *Patchy Vegetation* and less confused with *Continuous Vegetation* (Table 2.4; Fig. 2.7). Class 4: *Coarse Sediments* (29 objects; Table 2.2) was associated with Iso Cluster classes 7,11, and 21 (100%), was equally confused with *Continuous Vegetation* and *Shallow Silt/Mud (≤ 50 m)* and was most confused with *Patchy Vegetation* (Table 2.4; Fig. 2.7). The overall accuracy for the shallow sublittoral benthoscape was determined to be 61.3% with a kappa statistic of 0.55% (Table 2.4) indicating moderate agreement.

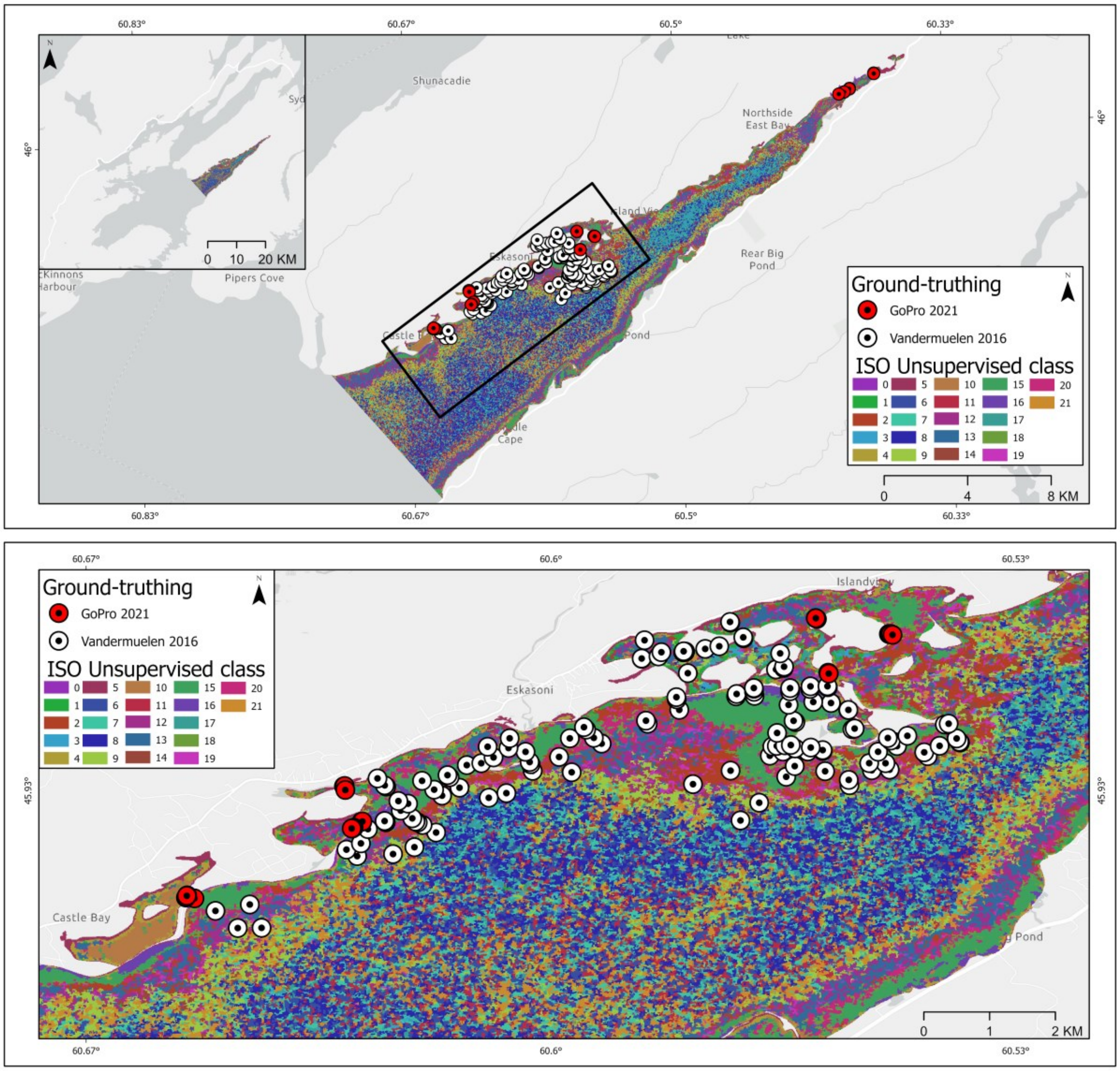



**Figure 2.8.** Iso Cluster unsupervised classification of the PCA raster derived from the satellite imagery input layers (12 bands). Image objects derived from application of an object-based image analysis segmentation of bands 1,2,3,5 and 8 were classified into 22 classes from the PCA raster. The white circles represent ground-truthing sites Vandermuelen et al. (2007; 2016) and the red circles represent ground-truthing stations collected by Murray (2021) via GoPro and quadrat drops. (a) Represents the Iso Cluster for all East Bay and (b) demonstrates the inset outlined by the black rectangle of Eskasoni First Nation located within East Bay.

**Table 2.4.** Error matrix for the satellite shallow sublittoral benthoscape classification used to compliment MBES sublittoral benthoscape.

| | Map (Iso Cluster) Classes Satellite Imagery | | | | | | |
|---|---|---|---|---|---|---|---|
| | Conti Veg. (≤ 6 m) | Patchy Veg. (≤ 6 m) | Shallow Silt/Mud (≤ 50 m) | Coarse Sediments | | | |
| **Ground-truth (Benthoscape) class** | **1+2+5+14** | **0+4+10+12+15+16+18** | **3+6+8+9+13+17+19+20** | **7+11+21** | **Total (no.of objects)** | **User's Accuracy (%)** | **Omission Error (%)** |
| Continuous Vegetation (≤ 6 m) | **44** | 21 | 11 | 0 | 76 | 57.9 | 42.7 |
| Patchy Vegetation (≤ 6 m) | 15 | **97** | 7 | 0 | 119 | 81.5 | 18.5 |
| Shallow Silt/Mud (≤ 50 m) | 10 | 24 | **34** | 0 | 68 | 50.0 | 50.0 |
| Coarse Sediments | 7 | 10 | 7 | **4** | 29 | 13.8 | 86.2 |
| **Total objects** | 76 | 152 | 59 | 4 | **292** | **Overall Accuracy: 61.3%** | |
| **Producer's Accuracy (%)** | 57.9 | 63.8 | 57.6 | 100.0 | **Kappa Statistic: 0.55** | | |

## 2.3.3 Map integration

The final combined benthoscape map (MBES and satellite data) achieved an overall accuracy of 59.3% and a kappa statistic of 0.45 and classified 806.6 km² of habitat in the BdOL and 125.8 km² in East Bay (Fig. 2.9; Table 2.5; Table 2.6). The combined map resulted in seven benthoscape classes (Fig. 2.9). *Shallow Silt/Mud (≤ 50 m)* was the dominant substrate throughout the BdOL, followed by *Deep Silt/Mud (≤ 50 m)*, *Silt/Mud with ≤50% Gravel*, *Coarse Sediments*, *Mixed Sediments*, *Continuous Vegetation*, and *Patchy Vegetation* (Table 2.6). Slight variation in habitat was found in East Bay; East Bay was dominated by *Shallow Silt/Mud (≤ 50 m)*, followed by *Continuous Vegetation*, *Silt/Mud with ≤ 50% Gravel*, *Coarse Sediments*, *Mixed Sediments*, *Patchy Vegetation*, and *Deep Silt/Mud (≥ 50 m)* (Table 2.5). In the combined benthoscape approximately 139.1 km $^2$ of the Silt/Mud class was reclassified as *Deep Silt/Mud (≥ 50 m)* (see methods section 2.5; Appendix E, Table E.1.). Furthermore, 11.61

km $^2$ of the *Continuous* (4.7 km$^2$) and *Patchy Vegetation* (6.7 km $^2$) was reclassified as *Shallow Silt/Mud (≤ 50 m)* (see methods section 2.5; Appendix E, Table E.1.).

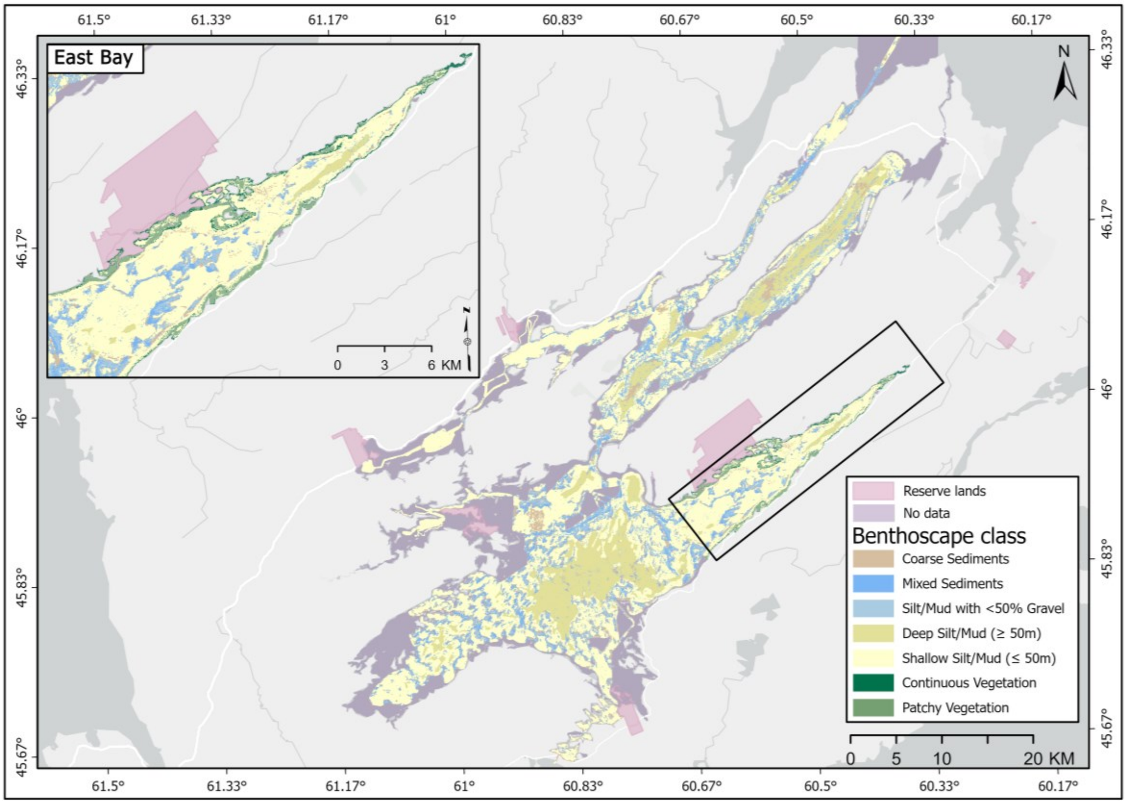



**Figure 2.9.** The classified and combined benthoscape map comprised of the shallow sublittoral region (satellite imagery) and the sublittoral region (MBES data) of the Bras d'Or Lake, Nova Scotia. The benthoscape was produced using an object-based image analysis segmentation and an unsupervised Iso Cluster analysis.

**Table 2.5.** Error matrix for the final combined benthoscape map of the Bras d'Or Lake where the classified shallow sublittoral (satellite imagery) region and the classified sublittoral (MBES data) regions were mosaicked into a single combined benthoscape. These results reflect changes to each benthsocape including a 50 m depth gradient in the sublittoral map between *Shallow* and *Deep Silt/Mud (≥ 50 m)* and a 6 m depth gradient for vegetated areas in the shallow sublittoral region.

| **Benthoscape class** | **Coarse sed.** | **Mixed sed.** | **Silt/Mud <50% gravel** | **Shallow Silt/Mud (≤ 50m)** | **Deep Silt/ Mud** | **Cont veg** | **Patch. veg** | **Total (no. of objects)** | **Users' Accuracy (%)** | **Errors of omission (%)** |
|---|---|---|---|---|---|---|---|---|---|---|
| Coarse sediments | **4** | 2 | 4 | 9 | 14 | 5 | 6 | 44 | 9.1 | 90.9 |
| Mixed sediments | 1 | **1** | 0 | 0 | 0 | 0 | 0 | 2 | 50.0 | 50.0 |
| Silt/Mud <50% gravel | 4 | 6 | **6** | 2 | 0 | 0 | 0 | 18 | 33.3 | 66.7 |
| Shallow Silt/Mud (≤ 50 m) | 6 | 6 | 13 | **130** | 0 | 8 | 19 | 182 | 71.4 | 28.6 |
| Deep Silt/Mud (≥ 50 m) | 0 | 10 | 0 | 0 | **0** | 0 | 0 | 10 | 0.0 | 100.0 |
| Cont. veg | 0 | 0 | 0 | 13 | 0 | **41** | 22 | 76 | 53.9 | 46.1 |
| Patchy veg | 0 | 0 | 0 | 21 | | 11 | **83** | 115 | 72.2 | 27.8 |
| **Total objects** | 15 | 25 | 23 | 175 | 14 | 65 | 130 | **448** | **Overall Accuracy: 59.3%** | |
| **Producer's Accuracy (%)** | 26.7 | 4.0 | 26.1 | 74.3 | 0.0 | 63.1 | 63.8 | | | **Kappa Statistic: 0.51** |

**Table 2.6.** Area of classified habitat derived from the combined benthoscape for the Bras d'Or Lake and East Bay.

| | **Bras d'Or Lake** | | **East Bay** | |
|---|---|---|---|---|
| **Habitat type** | **Area (km$^2$)** | **Area (%)** | **Area (km$^2$)** | **Area (%)** |
| Coarse Sediments | 76.4 | 9.5 | 5.9 | 4.7 |
| Silt/Mud with < 50% Gravel | 96.1 | 11.9 | 8.6 | 6.8 |
| Mixed Sediments | 64.7 | 8.0 | 5.2 | 4.1 |
| Shallow Silt/Mud (≤ 50 m) | 415.8 | 51.5 | 88.4 | 70.3 |
| Deep Silt/Mud (≤ 50 m) | 139.1 | 17.2 | 3.2 | 2.5 |
| Continuous Vegetation (≤ 6 m) | 9.9 | 1.2 | 9.9 | 7.9 |
| Patchy Vegetation (≤ 6 m) | 4.6 | 0.6 | 4.6 | 3.7 |
| **Total** | **806.6** | **100** | **125.7** | **100** |

## 2.4 DISCUSSION

### 2.4.1 Combined benthoscape of the Bras d'Or Lake

The developed benthoscape map of the BdOL, combining MBES and satellite data, represents one of the few integrated mapping datasets in this region. This benthoscape map serves as continuous classified baseline data that can be used to better understand seafloor characteristics and species-environmental relationships in the BdOL and can be used to inform management and marine spatial planning activities.

Benthoscape mapping via remote sensing methods is an example of non-destructive methods that can be used by researchers to guide collaborative decision making without harming the environment, which is of critical value for projects incorporating Indigenous knowledge and respecting Indigenous values. Indigenous values such as *M'sit No'kmaq*, which translates to "all my relations", acknowledges that Mi'kmaq people are related to those they share territory with and acknowledges the spiritual and cultural ties to the land as well as reciprocal responsibility to care for it (Denny & Fanning, 2016; Giles, 2014). Studies on marine conservation have demonstrated how combining Indigenous ecological knowledge, via participatory mapping, with remote sensing technologies, such as satellite and aerial imagery with geographic information systems, has helped bridge the gap between scientists and local people. This has encouraged conversation and acceptance of conservation projects while fostering collaborative management decisions of coastal ecosystems (Aswani & Lauer, 2006a, 2006b; Lauer & Aswani, 2008).

This study identified seven benthoscape classes that are consistent with previously published research in this region and the types of habitats found to exist within the BdOL (Shaw et al., 2006; Tremblay et al., 2005). Also consistent was the finding that the East Bay region of the BdOL was dominated by soft sediments (mud) with some patches of cobbles and isolated boulders (Fig. 2.9). However, results from this study did not find higher structure seafloor types (extensive boulder with macrophyte coverage) as described previously (Tremblay, 2002; Tremblay et al., 2005). The lack of higher structure seabed in this study is likely due to the limited amount of ground-truthing coverage which focused predominately in the northern and eastern regions of the BdOL compared to previous studies where ground-truthing occurred in the western and southern regions of the BdOL where more cobble-boulder habitats were observed (Tremblay et al., 2005). Additional ground-truthing extending to the southern and western regions of the BdOL and the collection of higher image quality would be valuable to capture the diversity of benthic habitats and communities that exist throughout the BdOL.

At least two benthoscape classes in this study were nearly identical to classes interpreted by Shaw et al., 2006. For example, my *Shallow Silt/Mud (≤ 50 m)* class was most similar to Shaw's et al. (2006) class 3: Mud: clayey silt to sandy silt clay. Similarly, my *Mixed Sediments* class was most similar to Shaw et al. (2006) class 4: Undifferentiated gravel, respectively (Table 2.3; Fig. 2.4). My *Shallow Silt/Mud (≤ 50 m)* class was distinguished as a featureless soft, muddy, and silt bottom and was easily distinguished in instances when the ROV video would occasionally hover towards the bottom and stir the sediments. Disruption of sediments created a plume of poor visibility, suggesting lighter and softer sediments, rather than settling quickly and clearly as seen with coarser fine sediments such as sand.

In this study, the *Mixed Sediments* in newly collected ROV footage consisted of a densely packed mixture of large and small subrounded and subangular pebbles and gravel and was most similar to the Glacial diamict (till) class described by Shaw et al. (2006). In some cases, *Mixed Sediments* were associated with encrusting red algae (*Lithothamnion* sp.), small green sea urchins (*Stronglyocentrotus droebachiensis*), and sea stars (*Asterias* sp.). The *Silt/Mud with < 50% Gravel* class identified in this study is likely similar to class 2: Littoral gravel by Shaw et al. (2006), as this study did not include the collection of in-situ grab or core samples and the ROV was not equipped with measuring lasers to classify grain size, so it was not possible to accurately differentiate between fine sediments such as sands, silts, and mud. Bedrock was not observed in any still images despite being identified by Shaw et al. (2006) in at least one area of the BdOL.

While this study obtained similar class names compared to Shaw et al. (2006), that study was a geological classification and classes were examined using more thorough techniques such as sediment grab samples which can delineate between glacial diamict, ice-contact sediments, gas charged sediments, fluvial deposits, lacustrine and marine mud, littoral deposits, etc., whereas this study used a more general approach of classification of images of substrates by using the Wentworth (1922) and Folk (1954) methods.

Currently there is no one standard method for classifying marine habitat based on ground-truthed data. Instead, several versions of schemas, seascape classifications, or benthoscapes are used to define habitat from images and sediment samples (Strong et al., 2019). For example, European Nature Information System (EUNIS), Coastal and Marine Ecological Classification Standard (CMECS), HELCOM Underwater Biotope and Habitat classification system, Hierarchical Framework of Marine Habitat Classification for Ecosystem-Based Management (HFMHC), and National Intertidal Subtidal Benthic Habitat classification (NISB) are some schemas that exist and aim to provide a framework for habitat classification (Strong et al., 2019). Moreover, the Potential Habitat Classification Schemes is designed to address the delineation of fisheries habitats, while others may specifically include fish habitats of conservation importance (Greene et al., 2005; Greene et al., 1999, 2007; Strong et al., 2019). While hierarchical schemas provide a framework for habitats to be grouped into coarser levels, comparisons between different studies using the same scheme (if one is used) can be difficult. Furthermore, comparisons between studies are only possible if the habitat schema classification is interpreted consistently and rests upon a thorough understanding of the schema used as well as knowledge of how to best classify information using that schema (Strong et al., 2019).

The benthoscape classification applied a 50 m depth threshold to the *Shallow Silt/Mud (≤ 50 m)* class to separate it into two classes: *Shallow Silt/Mud (≤ 50 m)* and *Deep Silt/Mud (≥ 50 m)*. This 50 m division was based on knowledge observed during in-field surveys in deeper regions of the BdOL, which suggested that soft sediment habitats in the BdOL are likely different at depth. However, differences in benthoscape classes at the 50 m contour could not be distinguished acoustically alone. For example, differences in benthoscape classes at the 50 m contour include variation in species composition inhabiting these areas which may be a result of depth alone or a combination of depth and associated environmental characteristics such as temperature, and oxygen. However more sampling information is required to better assess these depth differences in benthoscape classes. For example, the vertical profile of the water column in the northern region of the BdOL is relatively warm (~6–8°C) and fresher (~20–21 psu) in the first 5 m depth while demonstrating colder (~0.5°C) and more brackish (~ 25-26 psu) water below 30 m, with a strong thermocline occurring from ~5 to 30 m (Yang et al., 2007). Differentiating *Silt/Mud* classes at this contour could serve as the basis for the addition of a new category to the benthoscapes identified here, if assessing the benthoscapes of deep bottom dwelling demersal fishes. However, in this study the 50 m contour was chosen due to a combination of some sampling in the northern region as well as stations which were deployed as deep as 38.5 m. A 10 m buffer was allotted for any change in receiver placement after the first roll over of detections. The combined benthoscape map also applied a ≤ 6 m depth threshold to the *Continuous* and *Patchy Vegetation* classes in order to reclassify pixels ≥ 6 m from these classes as *Shallow Silt/Mud (≤ 50 m)*. This decision was based on information from Tremblay (2005) which stated that eelgrass did not extend beyond 6 m depth in the BdOL. Furthermore, in-field sonar sampling around the perimeter of nearshore islands in the BdOL have suggested a 5 to 6 m depth contour is indeed the depth at which one can expect the presence of eelgrass, with no eelgrass present deeper than 6 m (B. Hatcher, personal communication, 2021). Therefore, the 6 m depth contour was used as a boundary of which to observe attached vegetation, such as eelgrass, in the final benthoscape map. Overall, the combined benthoscape map that was developed represented seven distinct classes with reasonable accuracy and corresponded with previous datasets of habitats found in the BdOL (Table 2.5; Table 2.2).

### 2.4.2 Multibeam echosounder data

There are challenges to using multisource MBES datasets collected at different operating frequencies due to the lack of calibration of the backscatter data and system-specific settings (Brown et al., 2019; Lacharité et al., 2018). The data provided in this study were too old to re-process to examine whether discrepancies of the backscatter data collected between MBES systems occurred. However, the data used for classification that were processed by GSC and provided by CHS appeared to be high quality. However, misclassifications when using MBES benthoscape can occur (Lacharité et al., 2018). Sediment stratigraphy, particularly in areas where hard substrate exists beneath a thin, soft substrate layer, can further complicate the ability to interpret backscatter intensity and generate benthoscape maps from MBES datasets when mapped with lower frequency

MBES systems (Brown et al., 2019; Hillman et al., 2018; Lacharité et al., 2018; Lurton et al., 2015; Misiuk et al., 2021).

Within this study, several ground-truthing locations were misclassified as "soft bottoms" as they corresponded with areas of high MBES backscatter indicative of hard substrata (*Coarse sediments* or *Silt/Mud with ≤ 50% gravel*). Similarly, several areas identified as "hard bottom" in the ground-truthing were observed as "soft bottom" in the MBES backscatter. Multibeam backscatter data used in this study were collected using two different MBES Kongsberg systems at two different operating frequencies: EM1000 (95kHz) and EM3000 (300kHz) (Griffin, 2003; Shaw et al., 2005). Multibeam frequency and its ability to distinguish between different seafloor characteristics has been extensively studied in recent years, and requires careful consideration when using MBES backscatter for benthic habitat characterization (Lacharité et al., 2018; Lurton et al., 2015).

In fine-grained substrates (e.g., silts and muds) signal penetration of lower frequency systems (e.g., EM1000, EM1002) will penetrate deeper into the sediment than for high frequency systems (e.g., EM3000). This can result in misclassification of seafloor substrates, particularly where there are sediment stratifications with fine sediments overlayed on coarser substrates. As a result, areas mapped using multiple sources of MBES data (e.g. different surveys and/or operating frequencies) can result in lower classification accuracies depending on the local seafloor characteristics (Lacharité et al., 2018).

Greater penetration using lower frequency systems may therefore enable characterization of sub surface sediments (via volume scattering, variation in grain size, sediment strata, or presence of infauna and bioturbation), while higher frequency systems, that do not penetrate as deep into the seafloor, will enable better characterization of seafloor surface features and rugosity (Lacharité et al., 2018). Despite the challenges of using multiple frequency MBES datasets, the benefits of using MBES data as a tool to map benthic habitats has exceeded the challenges of its use (Brown et al., 2011, 2012; Misiuk et al., 2018, 2019, 2020, 2021)(Brown et al., 2011, 2012). These challenges simply illuminate the importance of ground-truthing as part of the validation methods for benthoscape maps.

### 2.4.3 Sentinel 2A satellite imagery

Nearshore areas in this study were classified using the best Sentinel-2A image acquired between August 2019 and June 2021 that were deemed low-cloud for the study area. This study focused only on identifying areas with attached vegetation in the East Bay region of the BdOL which is comprised of soft-bottom substrates and a less exposed shoreline (Fig. 2.9). Distinguishing between eelgrass and seaweeds, as well as macroalgae and other brown algae's, remains a challenge in classifying nearshore areas using satellite imagery (Immordino et al., 2019; Poursanidis et al., 2019; Traganos et al., 2018; Wilson et al., 2020). However, eelgrass is the only species of seagrass found to occur in the BdOL and is the most common marine plant in this system, making confusion between seagrasses unlikely (Lambert, 2002; McLachlan & Edelstein, 1971).

Nearshore areas classified in the shallow sublittoral benthoscape map were consistent with previous research by Tremblay et al. (2005) and were predominately comprised of eelgrass, macroalgae, and some brown algae. More exposed and rocky shores of the BdOL like those found in the northern and western regions, consisted of a mix of seaweeds but were located outside the East Bay extent. Areas classified as *Patchy* and *Continuous Vegetation* in this study were in line with those classified by Vandermuelen et al. (2016) (Fig. 2.9; Appendix E). In both this study and Vandermuelen (2016) areas closer to the shoreline and in shallower depths consisted of *Continuous Vegetation* becoming *Patchy* farther from shore and in deeper depths.

Nearshore areas mapped in this study were also consistent with the results of Wilson et al. (2020) in the Eastern Shore Islands in Nova Scotia, which revealed challenges in the ability of Sentinel-2A imagery to distinguish between dark bare muddy substrates and vegetated habitats. Using Sentinel-2A imagery, a variety of band indices that are known to aid in reflectance of vegetation were tested, however these indices did not improve separation of soft sediments such as sand, mud, and silt bottoms from densely vegetated areas. Furthermore, this study also demonstrated no spectral differences between *Shallow Silt/Mud (≤ 50 m)* and densely vegetated areas when using original and glint corrected bands. The results of this study agree with results by Traganos et al. (2018) in clear tropical waters where they used a supervised pixel-based classification approach to identify several types of seagrasses in the Aegean and Ionian seas. Traganos et al. (2018) found no spectral separability between non-vegetated and vegetated areas. Based on the results of this study as well as those of Wilson et al. (2020) and Traganos (2018), Sentinel-2A satellite imagery appears to inconsistently distinguish between densely vegetated areas and bright soft unvegetated sediments and is likely due to the coarse resolution (10 m) remotely sensed data. Despite this, the use of Sentinel-2A imagery in this study has provided useful results regarding the presence of vegetated areas that are consistent with previously published datasets such as Vandermulelen et al. (2007; 2016) and Tremblay et al. (2005).

Work by Vandermuelen et al. (2016) is restricted to five main areas, one of which overlapped with our study area in East Bay, yet they did not provide coverage for all of East Bay. Therefore, this study could not measure changes in vegetated areas for the entirety of East Bay, yet these changes could be measured and monitored given the area classified in both Vandermuelen et al. (2016) and this study. Furthermore, it is important to note that this classification was driven and validated using a significant amount of older ground-truthing data and newly collected satellite imagery (2019). Due to the inability to access many of the ground-truthing stations that had been classified by Vandermulen (2007; 2016), these sites could not be re-surveyed to monitor changes in density or loss of eelgrass. However, the classified dataset by Vandermulen (2007; 2016) significantly contributed to the quantity of ground-truthing sites required for the nearshore classification (Table 2.2).

The classified nearshore benthoscape map generated in this study may best be used as baseline information outlining areas with attached vegetation in the region. Future studies should consider expanding nearshore habitat classification outside of East Bay and be mindful of changes in aquatic vegetative species composition and assemblages that may occur along more exposed shorelines of the BdOL. This study generally achieved good accuracy of classified nearshore habitats (Table 2.5, Table 2.6). However, a combination of additional ground-truthing sites and higher resolution imagery, such as Worldview 2 or Worldview 3, would result in greater overall accuracy of these habitats. Higher resolution imagery would be required to distinguish between eelgrass and seaweed in the BdOL as Sentinel 2 satellite imagery does not have the spectral resolution to distinguish between these plants in this area (Wilson et al., 2020). Higher resolution satellite imagery would also be beneficial to build confidence in the distinctions between densely vegetated areas and bright and soft bare sediment, such as *Shallow Silt/Mud (≤ 50 m)*. Alternatively, researchers may consider the use of bathymetric LiDAR to distinguish between bottom types and species of submerged aquatic vegetation. Using bathymetric LiDAR would also allow for the inclusion of bathymetric information for the nearshore areas as satellite derived bathymetry is not open source and available tools require extensive ground-truthing.

### 2.4.4 Unsupervised classification

In this study the results of the MBES unsupervised classification resulted in both moderate overall accuracy (62.7%) and a moderate Kappa statistic (0.57). Similarly, in the satellite imagery the nearshore classification achieved an overall accuracy of (61.3%) and a moderate Kappa statistic of (0.55%). In supervised methods which require training data, labelled observations used to train the algorithm how to identify classes often require extensive human interaction. Meanwhile unsupervised classification methods, as used in this study, use only testing data, which are observations that are used to evaluate the performance of the classification (in this case ground-truth images) but are not used to train the algorithm used for classification itself. As a result, unsupervised classification methods which do not use training data and do not require guidance from human interaction can be less accurate than supervised methods, yet unsupervised classification methods are beneficial as they are reproducible and repeatable by any user using the same input information. Since these two-classification methods differ, the overall accuracy of a supervised classification and an unsupervised classification method is not equally comparable and therefore the kappa statistic is an overall better measure of accuracy.

Kappa statistics are an index of agreement that demonstrate how well the classification performed based on what would be expected if it was conducted by random chance (McHugh, 2012; Pykes, 2020). For example, if Iso Clusters were randomly assigned to benthoscape classes one might get some correct by chance. In this study, once the Iso Cluster algorithm helped to find the natural grouping of clusters, ground-truthing images (testing data) was used to help reduce the 22 Iso Cluster classes into four benthoscape classes allow the quantification and averaging of the exact number of correctly identified clusters. The kappa statistic is therefore lower than the overall accuracy and is a more conservative measure. As a result, it is possible that one might have a high accuracy but a low to moderate kappa or in this case a moderate accuracy and moderate kappa. In these results, the kappa statistic of the nearshore suggests results are 45% better than a random assignment of Iso Cluster clusters to assigned benthoscape classes (McHugh, 2012; Pykes, 2020).

### 2.4.5 Benthoscape mapping in the Bras d'Or Lake to support future research

Previous studies have demonstrated the value of incorporating habitat characteristics for understanding species' spatial and temporal distribution patterns (e.g., Becker et al., 2020; Finn et al., 2014; Rudolfsen et al., 2021). Several studies in the BdOL have examined or described species-habitat associations (Lambert, 2002; Parker et al., 2007; Petrie & Bugden, 2002; Tremblay, 2002; Tremblay et al., 2005). Most of this work has focused on invertebrates, such as crustaceans, echinoderms and other mollusks

(Breen & Metaxas, 2009; Tremblay, 2002; Tremblay et al., 2005), though, several finfish and invertebrate species distributions have been documented according to depth (Breen & Metaxas, 2009; Lambert, 2002), migration patterns (Crossin et al., 2016), geographic location (Giles et al., 2016), and movement selection patterns (Landovskis, 2021). However, to date, a comprehensive habitat map of the BdOL has not previously existed and therefore was not available for researchers or managers to conduct studies regarding movement and habitat use of a wide variety of species.

With the production of this comprehensive benthoscape map, it has recently allowed the examination of habitat associations and distributions of American lobster in the BdOL. This study, using acoustic telemetry in the BdOL, demonstrated that adult lobsters show little preference to substrate class and, contrary to predictions, used all available habitats patterns (Landovskis, 2021). Currently, there is a lack of ideal habitat in the BdOL for American lobster, a culturally, ecologically, and commercially important species to the area and its communities. The BdOL consists predominately of mud and has experienced significant (60%) declines of hard bottom habitats with increased rates of sedimentation (Shaw et al., 2006; Unamak'i Institute of Natural Resources, 2007a). Given the low salinity and minuscule tidal influence in the BdOL, a loss of hard bottom habitat may further contribute to the low productivity of lobster and threaten the recruitment of other invertebrate species that surrounding communities rely on for food and sustenance (Parker et al., 2007; Shaw et al., 2002). Longer-term study, combining tracking with benthoscape mapping, should allow an assessment of such expectations.

Benthoscape maps may also offer more realistic understanding of species-habitat associations when paired with oceanographic parameters. For instance, studies on

American eel and Atlantic cod (*Gadus morhua*) outside the BdOL have demonstrated the influence temperature can have on habitat selection (Freitas et al., 2016; Tomie et al., 2017). Atlantic cod were found to select vegetated (eelgrass and macroalgae) nearshore habitats during favorable temperatures, whereas under increased sea surface temperatures cod selected non-vegetated rocky bottoms and sand habitats available in deeper and colder areas (Freitas et al., 2016). It has also been suggested that American eels overwintering in estuaries may prefer mud substrates with freshwater upwelling, as mud acts as a thermal layer increasing winter survival under freezing water temperatures (Tesch, 2003; Tomie et al., 2017).

The final combined benthoscape map (MBES and satellite imagery data together), can be used to support future studies that examine species-habitat associations or temporal changes in species distribution in relation to substrate. With baseline knowledge of habitat, researchers can begin to match habitat selections to associated seasons, temperatures, or movement patterns. Furthermore, outputs generated in this chapter can be used to determine the amount of available and suitable habitat in the BdOL (Table 2.6) for species of interest. The benthoscape maps from this study can contribute valuable information for fisheries management and monitoring of culturally, commercially, and ecologically important species in the BdOL.

Species may use many benthoscape classes based on a combination of complex variables (temperature, salinity, oxygen, life stage, size) associated with habitat that may not be resolved using remote sensing methods alone. Future work should consider the inclusion of oceanographic environmental variables such as temperature, salinity, and oxygen to develop a more realistic understanding of species-habitat relationships in addition to the identification of benthoscapes provided (Freitas et al., 2016; Rudolfsen et al., 2021). Furthermore, benthoscape classes themselves represent distinct patches of habitats with clear boundaries distinguishing one habitat patch from another and therefore these boundaries between habitat types may not be truly reflective in nature (Brown et al., 2011; Strong et al., 2019; Wilson, 2020). Understanding classification methods of how benthoscape maps and classes are created can aid in understanding the limitations of products that are generated and in interpreting species habitat relationships.

# CHAPTER 3: HABITAT USE BY AMERICAN EEL/KATEW IN THE BRAS D'OR LAKE/ PITU'PAQ

## 3.1 INTRODUCTION

American eels (*Anguilla rostrata*) in the Bras d'Or Lake (BdOL) estuary, Cape Breton/ Unamak'i, Nova Scotia/ Mi'kmak'i, exhibit different life histories compared to those found in other Nova Scotia rivers (R. Bradford, personal communication, 2022; Jessop, 1987; Medcof, 1966; Smith & Saunders, 1955). For example, in the BdOL, there appears to be no extensive use of freshwater habitat by yellow stage American eels. Additionally, the BdOL appears to represent an optimal habitat for American eel as habitat characteristics found there are similar to those reported in other studies such as in a sheltered to semi-exposed shoreline, mud substratum, and at depths ≤ 30 m (R. Bradford, personal communication, 2022; Cairns et al., 2012; Parker et al., 2007). However, little knowledge has been documented on the movement of eels or their habitat associations in the BdOL as information on habitat has not been readily available.

The shorelines of the BdOL are necklaced with coastal lagoons (> 500) which are extremely vulnerable to anthropogenic impacts as they serve as a buffer zone trapping sediment and anthropogenic run-off from near-by roads, keeping materials from entering the main body of the BdOL (Peterson et al., 1985; Ross, 2018). These habitats, which offer refuge for a variety of species including eel, are also susceptible to the buildup of excessive, rich nutrients (eutrophication) that can lead to harmful algal blooms which can cause low oxygen areas and may be toxic to species that feed on aquatic vegetation (Kennish, 2002; Peterson et al., 1985; Ross, 2018; Skei, 2000).

Such anthropogenic impacts may lead to changes in water quality which can subsequently impact sediments where bottom dwelling species live or burrow and could result in a loss of, or alteration to, suitable habitat (Kennish, 2002; Peterson et al., 1985; Ross, 2018; Skei, 2000). Furthermore, American eels in the BdOL have been infested with an invasive nematode swim bladder parasite *Anguillicola carassus* (Denny et al., 2013). This parasite has been documented to reduce eels' resilience in low oxygenated areas, which may pose increased risks for eels burrowed in areas with oxygen limitations (Denny et al., 2013; Gollock et al., 2005; Lefebvre et al., 2007; Tomie, 2011). In the BdOL, localized build ups of both natural and anthropogenic nutrients have caused eutrophication in at least one bay as well as some coastal lagoons (Strain & Yeats, 2002). While the overall water quality of the BdOL is generally good, maintaining good environmental water quality within the BdOL is important to preserve habitats and reduce environmental stressors to species inhabiting them, especially in areas where *A. carassus* is present (Strain & Yeats, 2002).

American eels are often captured in salt marshes and coastal lagoons, and in some cases may comprise the main fish biomass in these habitats (Dionne et al., 1999; Ford & Mercer, 1986). Eels have also been documented to depend heavily on salt marsh secondary production as an energetic resource over time, and thus can be considered salt marsh residents (Eberhardt et al., 2015). Coastal lagoons and nearshore habitats, which are often vegetated, are important to eels throughout their life history as they provide refuge from predators in shallow depths (<6 m) especially during daylight hours (COSEWIC, 2012; Murphy et al., 2021).

The BdOL coastal lagoons, referred to locally as barachois ponds, serve as primary areas for several Mi’kmaq communities to conduct their food, social, and commercial (FSC) and winter eel fisheries (Denny et al., 2012). These nearshore habitats have been documented as important areas to preserve for the retention of language and transmission of Mi’kmaq fishing knowledge and were identified by Mi’kmaq as places young eelers learn to eel or go eeling for the first time (Giles, 2014). The East Bay, a long narrow arm that extends in the east of the main body of the BdOL, represents 125 $km^2$ of the 1,099 $km^2$ lake (see 2.3.3. Chapter 2). This region is culturally significant, as it is comprised of several primary eel fishing areas used by Mi’kmaq and is adjacent to Eskasoni First Nation, the largest Mi’kmaw community in Cape Breton and the largest Mi’kmaq speaking community in the world (Fig 3.1;  Denny et al., 2012; Fish-WIKS, 2022; Giles et al., 2016). Declines of the American eel have been suggested in the BdOL and the Mi’kmaq have reported requiring a higher fishing effort for fewer eels, as well as a notable decline in eel abundance (Denny et al., 2013; Denny et al., 2012).

Mi’kmaw knowledge holders in communities surrounding the BdOL have understood eel movements and habitat use for thousands of years and adapt their fishing strategies and gear according to the seasonal movements of eels (Denny et al., 2012). For example, Mi’kmaq fishers move from one fishing site to another to avoid repeatedly fishing the same area and in doing so aim to reduce depleting eel at a given location (Denny et al., 2013). Mi’kmaq eelers have reported waiting 5 to 7 days before returning to the same location to fish, while others have reported rotating fishing sites annually, or if abundance was low, not fishing an area for many years or cycles (seasons), or not fishing at all out of respect for the eel (Giles et al., 2016; Denny et al., 2013). During the spring, summer and fall, spearing occurs at night or during the early morning in coastal ponds where eelgrass is present until the growth of eelgrass makes it too difficult to see the eels (Denny et al., 2013). In early to mid-fall, Mi'kmaq adjust their fishing strategies to capture eels moving into the ponds to overwinter or those migrating out (Denny et al., 2013).

Mi'kmaw fishers also undergo a period of observation out of respect for the eel, to learn patience as well as proper eeling techniques and respect for place (Giles et al., 2014). For example, proper summer eeling technique is described as having eelers target the tail to help ensure eels punctured by the spear may survive if they escape and to ensure the head, body, and organs including the swim bladder are not punctured (Giles et al., 2014). Additionally, this period of observation described by Giles (2014) demonstrates how eelers learn to value the transmission of knowledge obtained through oral traditions such as through stories and experiential learning and illustrates how values and beliefs are transmitted and adapted over time and integrated into practices (Giles et al., 2014). However this period of observation and gaining experience in eeling using traditional gear is limited to shallow ( $\leq$ 4 m) waters (Denny et al., 2012).

Relatively little knowledge is available on the spatial and temporal distribution of eels and their habitat use in deeper waters (> 4 m) of the BdOL, as well as the amount of time eels spend in or moving between the coastal lagoons and East Bay. Previous studies in the St. Jean River watershed in eastern Canada, which conducted otolith analysis on yellow American eels, have suggested that eels considered to be brackish water residents may actually demonstrate seasonal migration patterns occupying brackish water in summer to feed and then migrate to freshwater rivers to overwinter as water temperatures begin to cool (Thibault et al., 2007). However, it is relatively unknown whether eels in estuarine environments, and specifically in the BdOL estuary, are indeed residents or demonstrate this seasonal migration pattern. Furthermore, the ability to understand this pattern in such a highly diverse species is likely restricted to the environment being studied and whether seasonal movement is accessible, available, or even required by eels. For example, American eels in the Great Lakes, one of the world's largest freshwater ecosystems, cannot exhibit this pattern as the region is comprised of a series of connected lakes linked to the Atlantic Ocean by the St. Lawrence River, with no other access to brackish waters or ponds (F. Whoriskey, personal communication, 2022).

The Mi'kmaq of Eskasoni seek to learn more about how eels are using the coastal lagoons as they adapt their strategic rotation of eel fishing sites based on the observance of eels and to guide their practices using the Mi'kmaq values *Msit No'kmaq* "all my relations" and *Netuklimk* "taking only what is needed" to ensure eels will be present for the next seven generations (Denny et al., 2012; Giles et al., 2016). These methods align with Western Knowledge Systems (WKS), knowledge that is driven by the scientific method, also undergo a period of observation to guide questions and learning yet using telemetry to track eel movements we can extend this period of observation throughout the seasons and in depths greater than 4 meters. The primary objective of this study is to examine how American eels use the East Bay region of the BdOL, and specifically coastal lagoons, to observe what habitat eels may be associated with and whether changes in movement patterns during certain seasons occur. This chapter addresses these objectives through six questions: 1) Do eels captured and released in the coastal lagoons remain in that location year-round? 2) Do eels that leave the Inner Pond then leave East

Bay? 3) Do eels spend more time in coastal lagoons compared to East Bay? 4) Do eels use all habitat available to them? 5) Are eels associated with certain habitats linked to changes in season? and 6) Do eels use certain habitat habitats more in the day compared to the night?

## 3.2 METHODS

### 3.2.1 Apoqnmatulti'k

This study was part of a 3-year collaborative project ("Apoqnmatulti'k", Mi'kmaw for "we help each other") that is fostered on a combination of Two-Eyed Seeing/Etupaptmumk (combining the strengths of Indigenous knowledge with those of western knowledge) and a community-based study design to understand the movements and habitat use of the American eel, an ecologically, commercially, and culturally significant species. Apoqnmatulti'k's Mi'kmaw partners of Eskasoni First Nation, East Bay, BdOL, seek knowledge on how eels use the estuary and specifically coastal lagoons as these communities rely on the estuary for food and sustenance as well as for the adaptation and transfer of Mi'kmaq fishing knowledge (Giles et al., 2016). Mi'kmaw partners in Apoqnmatulti'k wish to identify eel habitat associations in several historical fishing areas that can help further identify any potential anthropogenic risks to eels and eel habitat in this region. This study aims to help contribute a collective understanding of American eel habitat use in the BdOL.

### 3.2.2 Study site and design

This study was conducted within the Eastern region (East Bay, 45.911902° N, -60.602913° W) of the BdOL (45.848131° N, -60.818953° W; Fig. 3.1). Placement of stationary acoustic receivers was chosen based on local and Mi'kmaw knowledge shared by project partners at the Unamak'i Institute of Natural Resources (UINR) and knowledge of acoustic telemetry by members of the Ocean Tacking Network (OTN). A previous study by Giles et al (2014) contributed to site selection for receivers placed in nearshore areas, which included John Paul's Lane Pond, East Bay Sandbar and beaches, and Goat Island and surrounding islands, each deemed culturally significant and important to Mi'kmaq.

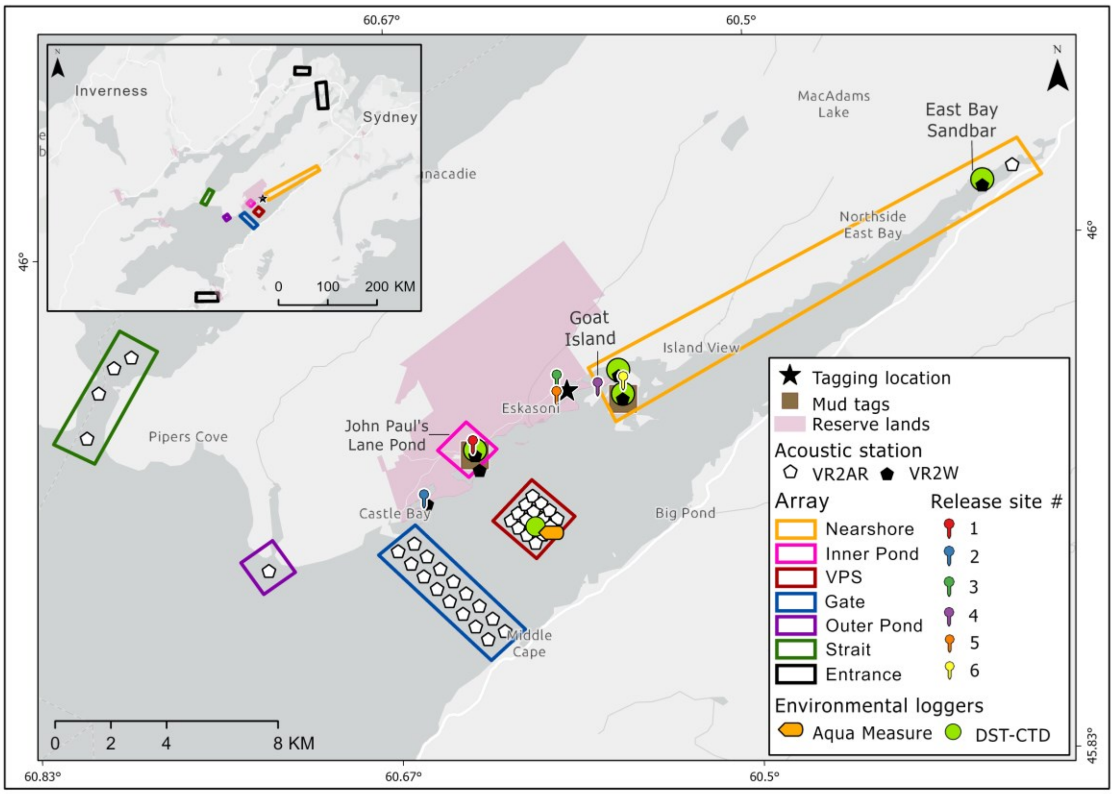



**Figure 3.1.** Acoustic receiver locations (n=47) within the Bras d'Or Lake. White pentagons represent Innovasea VR2ARs while black pentagons represent Innovasea VR2Ws. Environmental data loggers (n=6) consist of DST-CTD loggers (n=5) in green circles and the aquaMeasure (n=1) dissolved oxygen logger with the orange pin. Buried mud tags (n=6) are represented by brown squares. Release locations of tagged eels are represented by pins. Release sites (RS) are differentiated by color: RS #1 (red), RS #2 (blue), RS #3 (green), RS #4 (purple) RS #5 (orange) and RS #6 (yellow). The tagging location (black star) represents Eskasoni Fish and Wildlife Center where all eels were tagged and held in holding tanks until release back into the BdOL. The inset map shows the placement of acoustic stations at the entrances to the Bras d'Or Lake.

This study deployed a total of 47 acoustic receivers (Vemco/Innovasea VR2ARs and VR2Ws) with the help of the OTN field team and members of the UINR between June 2019 and October 2020 (Fig. 3.1). VR2Ws were placed in nearshore areas as these receivers did not have an acoustic release to bring the station to surface and instead are deployed and retrieved manually, except for one which was placed in the head of East Bay (1.2 m depth) and was not equipped with a release mechanism (Fig. 3.1).

Sixteen receivers (VR2ARs) were arranged into a VEMCO Positioning System (VPS) array situated in a gridded pattern in East Bay to collect fine-scale movement data of eels and lobsters (Landovskis, 2021). Another sixteen receivers (VR2ARs) were arranged into two lines across the mouth of East Bay forming a double gate and were used to observe species migrating in and out of East Bay and the main body of the BdOL (Fig. 3.1; (Brownscombe et al., 2019). Stations placed in the VPS and the Gate were placed 500 m apart. Remaining receivers were placed in the Barra Strait (VR2AR's, n=4), at each of the exits of the BdOL to the Ocean (n=3), and at several nearshore locations including beaches, headwaters, coves (VR2AR, n=1 and VR2W's, n=5) and at entrances to coastal lagoons referred here as the Inner and Outer Ponds (VR2AR, n=1 and VR2W, n=1) (Fig. 3.1).

Several environmental loggers were also deployed. In 2019, three Star-Oddi DST-CTDs were deployed at a single coordinate located in the center of the East Bay VPS array at 6.4 m, 14.4 m, and 20.4 m depths to measure salinity, temperature, and conductivity (Starr-Oddi, 2017). One aquaMeasure data logger was deployed at the same location at 20.9 m depth to record dissolved oxygen. In July 2020, additional Star-Oddi DST-CTD tags (n=4) were deployed and attached to stationary receivers to collect oceanographic data in coastal lagoons and nearshore areas. All environmental data loggers were set to record at 10-minute intervals. To convert percent oxygen concentrations into mg/L an online converter was used (Loglio Systems, 2022). V9 acoustic transmitters (V9P-2x-069k-1, n=6, not equipped with a pressure sensor to record depth) were buried 30-45 cm beneath the substrate ("mud tags", Fig. 3.1) on December 17 and 20, 2019, at two locations. Each individual tag was placed in a Ziploc baggie and then placed inside another Ziploc baggie filled with water and sealed with duct tape. These were meant to resemble a porous sac such as a ventral cavity of a fish to assess if tagged eels could be detected if they remained beneath the substrate during periods of winter dormancy.

### 3.2.3 Range testing

Range testing of receivers in the BdOL was conducted by Crossin et al (2016) using V13 and V16 tags which occurred at the same frequency as those used in this study (V9 at 69kHz) as well as range testing of sentinel tags by receivers placed in the same or similar locations as those used in this study. Sentinel tags maintained good detection efficiencies despite different environmental conditions, such as surface wind, bottom tide, and a stratified water column. Detection efficiency ranged from 92.8% in a channel to 98.6% in a small bay, however no estimated detection range was provided, beyond what was stated as the estimated range in the receiver manual (Crossin et al., 2016). Stations placed close to nearshore areas may reduce the detection range from 500 m to as little as 50 m as they are confounded by land masses and in some cases occupied with dense aquatic vegetation (Bangley et al., 2018).

### 3.2.4 Animal Capture and tagging

Forty tags were allocated to the project over a 3-year period, though actual numbers of eels tagged varied by year due to difficulties in capture success. In total, 33 eels were tagged over three summers: 18 between August 1 and 28 2019, 7 between August 14 and September 17, 2020, and 8 between June 4 and September 22, 2021. All eels were captured near Eskasoni Reserve Lands and nearshore areas within East Bay of the BdOL (Fig. 3.1) by the Apoqnmatulti'ks Community Liaison and a local Mi'kmaw harvester. In 2019, eels were captured using un-baited fyke nets, while in 2020 and 2021, eels were captured using baited pop-up cylinder nets. Pop-up nets were baited with crushed blue mussels and green crab in 2020 and with a single alewife in 2021. This fishing gear and bait was selected according to local and Mi'kmaw knowledge and modified due to the low numbers of eels captured in 2020 and 2021.

Immature yellow staged eels ≥ 50 mm in total length and ≥ 140 g in weight (for tags to be ≤ 2% of body weight) that were in good health were retained for tagging; smaller eels were returned to the BdOL (see Appendix A.1&A2). After capture, eels were brought to the Eskasoni Fish and Wildlife Center for tagging (Fig. 3.1) and held for no more than three days in holding tanks equipped with a covering to prevent animals from jumping out and to minimalize any visual stress. Eels were tagged externally with a T-bar Floy tag labelled with an ID number "BLE 0000#" and UINR's phone number and tagged internally with a unique acoustic transmitter (model V9P-2x-069k-1: 9mm diameter, 26.5mm length, 2.8g weight, a range of 34 meters and estimated tag life 335 days; Innovasea, 2020). All tags were equipped to record pressure (depth) and were set to transmit a uniquely codded signal every 45 to 75 seconds. Because eels are important to the local FSC fishery external Floy tags were critical to indicate that the individual would be unsafe for human consumption given the anesthetic used for internal tagging. Information on the total length, weight, body condition, capture location, date and time, tag ID and Floy tag ID were recorded for all eels.

For tag implantation, eels were anesthetized using 2-phenoxy-ethanol (0.5 mg/L) and a 20 mm incision was made using a small scalpel blade (size 10) 50 mm anterior to the anus and offset from the mid-ventral line. Tags were inserted into the visceral cavity of the abdomen and a 3-0 nylon-monofilament suture was used to close the incision with 2-3 sutures. The abdomen was disinfected with Betadine before and after tag implantation. Tagging and capture procedures were carried out in accordance with the approved animal use protocol from Dalhousie University's Animal Care Committee (protocol I19-17). After tagging, eels were placed into a recovery tank and monitored for 6 to 24 hours after surgery to ensure they remained dorsal side up and then returned to their capture location for release (Fig. 3.1).

### 3.2.5 Determining life stage for tagging

Morphometrics used to ensure only yellow eels were tagged were not performed on the first 18 eel tagged. In 2020, morphometrics such as body color (having a yellow to white ventral surface and a dark green dorsal), with non-melanized pectoral fins and no presence of a visible lateral line, was used to aid in determining yellow eels used for tagging from silvering eels in the BdOL (Acou et al., 2005; Okamura et al., 2007). Furthermore, in 2020, photos were gathered for the remaining 15 tagged eels and were used to collect measurements for the left eye diameter (Dv: vertical and Dh: horizontal) to calculate an ocular index (OI = (($\pi$ /TL) × ((Dv + Dh) / 4)) x 100), as well as measurements of the left pectoral fin to calculate the pectoral fin index (PFI = (PF/ TL) x 100) was conducted in ImageJ to help measure the degree of silvering (Béguer-Pon, Castonguay, Benchetrit, et al., 2015; Pankhurst, 1982). The Pankhurst ocular index is particularly useful for identifying the level of maturity in eels using morphometric data in addition to body color, with an ocular index of ≤ 6.5 mm indicating immature or maturing adult silver eels and an index of > 6.5 indicating mature eels (Pankhurst, 1982). Other measurements such as head length (measured from the tip of the lower jaw to the lower point of the gill openings or tip of snout to back edge of left gill operculum), head width (distance between the outside of the jaw hinges to the nearest 0.1mm (Barry et al., 2016) and Girth width (tip of the lower jaw to the farthest inside corner of the mouth) were measured. Though not categorized, head width has been assigned in previous studies as broad at >0.33mm and as narrow at <0.33mm (Barry et al., 2016; Ide et al., 2011; Lammens & Visser, 1989; Proman & Reynolds, 2000).

### 3.2.5 Detection data: Filtering and analysis

Acoustic detection data were imported into R and the GLATOS package was used to filter any potential false detections from the dataset (Holbrook, et al., 2019). False-positive detections or “false-detections” occur when a signal from two or more transmitters collide and result in a different tag ID code by the receiver station (Simpfendorfer et al., 2015). False detections were removed using the “min_lag” function within the GLATOS package which calculates the minimum time interval (seconds) to the next closest detection (either previous or subsequent) of the same transmitter on the same receiver (Holbrook, et al., 2019). The “false_detections” function within the

GLATOS package was then used with a time frame of 1,350 seconds which is 30 times the minimal step delay (45 sec) of the tags used. The “false_detections” filter also categorizes each detection assigning a “passed_filter” of 0 or 1 indicating 1 as a true detection and 0 for receivers with only a single detection. The data were then filtered to remove detections with “passed_filter == 0”. Therefore, detections that have a passed filter of 1 demonstrate the eel was detected at least twice at two different timestamps by the same receiver or was detected by more than one receiver within the study array.

Once filtered, detection data were analyzed using the Refining Shortest Paths (RSP) package to generate eel tracks (Niella et al., 2020; R Core Team, 2021). RSP uses a least-cost path analysis to calculate the shortest path between locations where animals were detected and is constrained by landmasses to allow for longer, but more realistic distances travelled by tagged animals (Niella et al., 2020). This study applied a 500 m buffer to the land shapefile as several receivers were close to shore and this was the smallest buffer accepted by the package given our study design. RSP then applies a dynamic Brownian Bridge Movement Model to calculate utilization distribution (UD) areas with 50% representing the core use area and 95% representing the home range (Niella et al., 2020). This accounts for the time differences between sampled positions and their respective accuracy while using conditional random walks to detect behavioral changes along trajectories used in estimating utilization distributions (Kranstauber et al., 2012; Niella et al., 2020).

### 3.2.6 Mortalities

At least two detected eels were assumed to have died and thus were removed from the analysis. One determination of mortality was if elapsed time between the first detection timestamp and the last detection timestamp was greater than 90 days during the spring (Mar-May), summer (Jun-Aug), or fall (Sept-Nov) months, or was greater than 182 days during the winter (Dec-Feb) dormancy period (Denny et al., 2012). Mortality was also assumed if an individual demonstrated a high number of detections (≥ min step rate of tag at 45 seconds) at a single location for the given detection period, no horizontal movement to or between adjacent arrays, little to no changes in pressure (depth) sensor, or if a mortality report was made by local and Mi'kmaw fishers (Klinard & Matley, 2020). Detection data for the two individuals removed from analysis are found in Appendix F.

### 3.2.7 Categorization of movements

For nearshore and largescale movement analysis eels were grouped as either sedentary or vagrant, similar to Beguer Pon et al., (2015). Sedentary eels were categorized as those that demonstrated no movements between arrays yet were repeatedly detected at a single location (receiver closest to their release site). Vagrant eels were categorized as eels that were detected by at least two arrays.

To determine whether eels tagged in the Inner Pond remained in that area, data were filtered to include only eels captured/released in the Inner Pond. A chi square goodness of fit test was used to determine whether there was a significant difference in the number of eels that left the pond compared to those that stayed. For expected values, a 3:1 ratio for the number of individuals that would leave the Inner Pond (75%) compared to those that may stay (25%) was used based upon the prediction that eels in the Inner Pond are not always there but rather use these areas for foraging or winter refuge (Denny et al., 2012). A two-sample t-test was then used to determine whether the total length and weights of individuals that stayed were significantly different from those that left the Inner Pond.

To determine whether vagrant eels remained in East Bay the data were filtered to include only eels that left the Inner Pond. A chi square goodness of fit test was then used to determine whether there was a significant difference in the number of eels that left East Bay and were detected in the Strait compared to those that remained in East Bay. For expected values, a 3:1 ratio for the number of individuals that would leave the East Bay (75%) compared to those that may stay (25%) in East Bay was used based upon the prediction that the BdOL offers suitable habitat for wintering (ex. Coastal lagoons with freshwater upwelling and predominately mud substrate (Ross, 2018; Taylor & Shaw, 2002). A two-sample t-test was then used to determine whether the total length and weights of individuals that remained in East Bay were significantly different from those that left East Bay.

### 3.2.8 Residency

To determine whether eels spend more time in coastal lagoons compared to East Bay and the amount of time eels spent in each array, all receivers that detected eels were grouped into seven arrays: 1) Nearshore, 2) Inner Pond), 3) VPS, 4) Gate, 5) Outer Pond, 6) Strait, 7) Ocean (Fig 3.1; Wickham et al, 2022). Data were summarized by eel ID, the array, and the month collected and arranged by the timestamp of the detection (Wickham et al., 2022). Detections were filtered separately for time of first detection and time of last detection for each individual within each array. The filtered data consisting of array, eel ID, residency time and time of first detection timestamp were merged with that of last detection timestamp and a new column was made for residency time (seconds), which was determined by subtracting the last detection timestamp from the first detection timestamp. Residency time was summarized for all eels in each array and expressed as a proportion to determine the amount of time eels spent in each array. No eels were detected at the entrances/exits to the BdOL and therefore this array was not included in analyses (Fig. 3.1).

To determine whether eels spent more time in one array over others in the BdOL, a non-parametric Kruskal-Wallis test was used to assess whether there was a significant difference in residency time in each of the six arrays (Fig 3.1). When significant differences in residency between arrays were found, Dunn's (1964) test for multiple comparisons was used to determine where significant differences occurred and in which arrays. Another two-sample t-test was used to determine whether there was a significant difference in the total length and weight of individuals that left the BdOL (last detected in the Strait) compared to those that remained in the East Bay.

### 3.2.9 Habitat Use

Habitat availability and suitability were calculated in this study following methods used in Rudolfsen et al (2021). Briefly, a habitat suitability index (HSI) is a value ranging from 0 to 1 and is used to estimate the suitability of different habitat types for a population based on the species observed presence or absences within a dataset. An HSI was calculated based on the number of detections that occurred on a given benthoscape for each individual eel in each season. Individual HSI was determined according to Rudolfsen et al (2021) by determining the index of preference where the number of detections within a benthoscape class was divided by the benthoscape class with the maximum number of detections and then was averaged to scale the HSI between

0 and 1. These individual HSIs were then compiled for each season and an average HSI was determined for each benthoscape class (see Chapter 2 sections 2.1 and 2.2.5; Rudolfsen et al., 2021). For each receiver, a benthoscape class was extracted at the receiver's coordinates using the extract function from the *raster* package (Hijmans, 2019). Habitat availability was determined using a 500 meter (m) detectable radius based on the receiver's detection range and the extracted benthoscape class. Then, 95% confidence intervals were calculated using the standard error of the individual HSI values grouped by benthoscape class. Eel detections for all years (2019, 2020, and 2021) were used and seasons were categorized in this study as: winter (Dec. 01-Feb. 28/29), spring (Mar. 01-May 31), summer (Jun. 01-Aug. 31), and fall (Sept. 01-Nov. 31).

Areas (km$^2$) for each classified benthoscape class (H) were determined in Arc GIS Pro v.2.8 and were used to determine the probability that eels would be detected on a given benthoscape, calculated as:

$$P(Strata_i)=\frac{Da}{H}$$

Where the detectable area (Da) for each benthoscape class was:

$$Da = \frac{(\pi * 500m^2)* \text{ \# hydrophones}}{100}$$

The conditional detectability in Strata$_i$ within a given benthoscape was calculated as P(Strata$_i$) within a given benthoscape multiplied by the area (%) for a given benthoscape. Following the conditional detectability in P(Strata$_i$), the expected distribution of detections was calculated as conditional detectability in Strata$_i$ within a given benthoscape divided by the total conditional detectability in Strata$_i$ for all benthoscape classes. A chi-squared goodness-of-fit-test was used to test whether there was a significant difference between the habitat used (observed) and the habitat available (expected). Another, chi-squared test was used to compare habitat use against available habitat for each season. Benthoscape classes were then further categorized as “low use habitat” or “high use habitat” for each season and a chi-squared test was used to compare habitat use against available habitat for each season. Low use habitat areas were categorized as those that were rarely or never used while high use habitats were categorized as habitats that were used more frequently.

To determine whether eels used certain habitats more in the day compared to the night, detection data were converted from UTC to ADT using the *lubridate* package in R (Spinu, 2021). Sunrise and sunset times were obtained using the *suncalc* package and merged with detection data (Thieurmel & Elmarhraoui, 2019). Diel period was classified as “day” if the time of detection in ADT was after sunrise and before sunset or ‘night’ if the time of detection was after sunset and before sunrise. The number of detections for each diel period were summed for each individual, and the proportion of detections for day and night was determined.

After the diel period was determined, the benthoscape raster generated in Chapter 2 was used to extract benthoscape classes to determine the habitat type for each detection and for individual using the *raster* package in R (Hijmans, 2019). The number of detections per individual for each habitat in each diel period was then summarized, and a proportion was determined as above to determine which diel period eels may be using certain habitats.

## 3.3 RESULTS
### 3.3.1 Detections of tagged eels

Detections from 19 of 33 tagged eels were recorded from Aug 02, 2019 to July 11, 2021 (Tables 3.1, 3.2). A single eel was tagged after the data roll over period and therefore no detections were recorded for this individual. Seven of the 18 eels tagged in 2019 were detected in 2020: one was detected in both 2019 and 2020, while the other four were detected in 2020. At least four eels that were tagged in 2019 were not detected until the next calendar year in spring and summer of 2020 (Fig 3.2). No eels that were tagged in 2020 were detected in 2021. Individual data for the 19 eels (mean ± S.D.: total length = 659 ± 78.8 mm, body weight = 474 ± 198.8 g) that were tagged and subsequently detected are presented in Table 3.2. Furthermore, two individuals were removed from final analysis due to assumed tagging induced mortality (see section 3.2.3) and twelve were never detected.

**Table 3.1.** Tagging effort and subsequent detection of all acoustically tagged American eels (n = 33) in the Bras d'Or Lake estuary. Capture/Release locations are designated by release site number (RS #) and N represents number of eels tagged.

| Year | Tagging Period | Capture/ Release Location | *N* | Number of eels detected eel | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | 2019 | 2020 | 2021 | Detected in 2019 & 2020 | Detected in 2020 & 2021 | Total # of individuals detected |
| 2019 | 02 Aug to 26 Oct | RS #1 | 15 | 5 | 4 | - | 1 | - | 9 |
| 2019 | 02 Aug to 26 Oct | RS #2 | 1 | 0 | 0 | - | 0 | - | 0 |
| 2019 | 02 Aug to 26 Oct | RS #3 | 2 | 2 | 0 | - | 0 | - | 2 |
| **Total** | | | **18** | **7** | **4** | **0** | **1** | **0** | **11** |
| 2020 | 14 Aug to 14 Sep | RS #4 | 3 | - | 1 | 0 | - | 0 | 1 |
| 2020 | 14 Aug to 14 Sep | RS #5 | 2 | - | 1 | 0 | - | 0 | 1 |
| 2020 | 14 Aug to 14 Sep | RS #6 | 2 | - | 0 | 0 | - | 0 | 0 |
| **Total** | | | **7** | **0** | **2** | **0** | **0** | **0** | **2** |
| 2021 | 04 Jun to 21 Sep | RS #1 | 7 | - | 0 | 6 | - | - | 6 |
| 2021 | 04 Jun to 21 Sep | RS #4 | 1 | - | 0 | 0 | - | - | 0 |
| **Total** | | | **8** | **0** | **0** | **6** | **0** | **0** | **6** |
| **Total all years** | | | **33** | **7** | **6** | **6** | **1** | **0** | **19** |

**Table 3.2.** Individual American eels tagged and subsequently detected (n=21 of 33 tagged) in the Bras d'Or Lake estuary. Individuals highlighted in grey were assumed to have died. #Uniq. receiver represents the number of unique receivers on which each individual was detected.

| Floy tag ID | Tag ID | Body mass (g) | Total length (mm) | Release date & time (UTC) | Date & time (UTC) of first detection | Date & time (UTC) of last detection | # Days Detect. | # Days since release | # Uniq. receiver |
|---|---|---|---|---|---|---|---|---|---|
| BLE 00001 | 7778 | 528 | 690 | 2019-08-02 13:30 | 2020-05-27 1:02 | 2020-07-06 7:43 | 41 | 339 | 1 |
| BLE 00003 | 7771 | 207 | 531 | 2019-08-02 13:30 | 2020-07-04 6:34 | 2020-07-05 1:57 | 2 | 338 | 1 |
| BLE 00004 | 7766 | 366 | 620 | 2019-08-02 13:30 | 2020-07-03 6:01 | 2020-07-03 6:02 | 1 | 336 | 1 |
| BLE 00008 | 7773 | 389 | 616 | 2019-08-02 13:30 | 2019-10-21 17:13 | 2020-07-04 16:54 | 259 | 337 | 1 |
| BLE 00009 | 7772 | 980 | 795 | 2019-08-02 13:30 | 2019-10-17 22:21 | 2019-11-04 12:06 | 8 | 94 | 24 |
| BLE 00010 | 7777 | 688 | 700 | 2019-08-02 13:30 | 2019-09-12 4:18 | 2019-10-01 22:19 | 12 | 60 | 29 |
| BLE 00015 | 7774 | 715 | 750 | 2019-08-29 14:34 | 2019-09-09 6:04 | 2019-09-09 6:17 | 1 | 11 | 1 |
| BLE 00017 | 7779 | 501 | 670 | 2019-08-29 14:34 | 2019-10-21 18:04 | 2020-10-02 14:33 | 340 | 400 | 8 |
| BLE 00019 | 7776 | 565 | 645 | 2019-08-29 14:34 | 2019-08-31 2:16 | 2019-09-01 5:17 | 2 | 3 | 5 |
| BLE 00021 | 7767 | 204 | 525 | 2019-08-29 14:34 | 2019-10-21 17:18 | 2020-06-16 11:10 | 240 | 292 | 1 |
| BLE 00023 | 8584 | 1115 | 826 | 2019-10-18 18:02 | 2020-07-16 3:56 | 2020-07-21 1:17 | 6 | 276 | 5 |
| BLE 00024 | 8595 | 528 | 700 | 2019-10-26 16:10 | 2019-10-26 20:53 | 2019-11-20 4:56 | 20 | 25 | 15 |
| BLE 00025 | 8583 | 412 | 630 | 2019-10-26 16:10 | 2019-10-26 21:45 | 2019-11-26 2:50 | 3 | 30 | 9 |
| BLE 00027 | 10627 | 767 | 710 | 2020-08-15 15:00 | 2020-09-17 7:44 | 2020-09-24 16:18 | 4 | 40 | 3 |
| BLE 00029 | 10628 | 544 | 665 | 2020-09-01 15:29 | 2020-10-18 23:17 | 2020-10-21 2:06 | 2 | 49 | 5 |
| BLE 00035 | 11691 | 489 | 677 | 2021-06-05 16:00 | 2021-06-05 20:46 | 2021-07-31 6:47 | 57 | 56 | 1 |
| BLE 00036 | 11687 | 716 | 740 | 2021-06-05 16:00 | 2021-06-05 16:00 | 2021-06-11 4:44 | 7 | 6 | 2 |
| BLE 00039 | 10632 | 420 | 610 | 2021-06-05 16:00 | 2021-06-05 16:00 | 2021-06-05 19:59 | 1 | 0 | 1 |
| BLE 00040 | 11690 | 564 | 740 | 2021-06-05 16:00 | 2021-06-05 16:31 | 2021-06-16 16:11 | 13 | 11 | 1 |
| BLE 00042 | 10626 | 568 | 730 | 2021-06-08 14:00 | 2021-06-08 21:02 | 2021-07-02 2:43 | 20 | 24 | 2 |
| BLE 00044 | 10633 | 354 | 610 | 2021-06-08 14:00 | 2021-06-08 20:58 | 2021-07-01 10:50 | 24 | 23 | 1 |

### 3.3.2 Mud tags

Acoustic transmitters buried 30-45 cm beneath the substrate were detected consistently throughout the winter and continued to transmit for the remainder of each tag's battery life (see Appendix G). Mud tags demonstrated good detection efficiency with approximately 85.5 – 87.9% of detections recorded (Table 3.3). These findings confirm that signals would be detected by burrowed eels that are within 500 m detection range of a receiver (Table 3.3).

**Table 3.3.** Actual and expected number of detections obtained from mud tags buried 30-45 cm beneath the substrate at two locations in East Bay, Bras d'Or Lake estuary.

| Location | # of Tags | # of Actual detections | # of Expected detections | % of Actual detections |
|---|---|---|---|---|
| Inner Pond | 3 | 402,786 | 457,920 | 87.9 |
| Nearshore | 3 | 470,880 | 550,508 | 85.5 |

### 3.3.3 Coastal movements

More than half of the tagged eels were captured and released in the Inner Pond (Fig. 3.2, 14 of the 19 eels subsequently detected). It was predicted that that more eels (75%) would leave the Inner Pond compared to those that would remain (25%). Contrary to predictions, a significant difference in the number of individuals that remained in the Inner Pond compared to those that left was found ($\chi 2$= 33.3, df= 13, $p \leq 0.005$). Approximately half of the eels released in the Inner Pond left. Sedentary eels were detected at the Inner Pond between May 27 and July 31 in 2020 (Fig. 3.2). At least two sedentary eels (BLE 00003 and BLE 0004) were undetected after tagging in summer of 2019 until the next calendar year in summer (3.2) and were only detected for up to 2 days of the entire detection period (Table 3.1).

Of the seven vagrant eels that left the Inner Pond, six demonstrated uni-directional (linear) movements and were last detected in the Gate or the Strait while one revealed multi-directional movements travelling from the Inner Pond to the Nearshore array where it remained for 18 days before returning to the Inner Pond. This eel was last detected in the Inner Pond on December 12, 2019 and was detected on December 16, 2019 in the Gate (Fig. 3.2). Additionally, one vagrant eel tagged in 2019 (BLE 0001) and released in the Inner Pond was not detected until the next calendar year in summer (Fig. 3.2).

Eels were highly variable in the amount of time they spent in the Inner Pond. Among individuals, 42.9% were detected to be within range of the array for less than 1 day while 57.1% were detected to spend 11 to 57 days in the pond (Table 3.1). No eels remained in the Inner Pond beyond 57 days, suggesting eels do not use the Inner Pond for extended periods of time such as winter dormancy. Significant differences in total length were found between eels that left the Inner Pond compared to those that stayed, with smaller eels tending to remain in the pond (676 ± 158.3 mm and 719 ± 50.9 mm; respectively, $t= -2.45$, $df=12$, $p \leq 0.05$). Significant differences in total weight were also found between eels that left the Inner Pond compared to those that stayed with heavier eels tending to leave the pond (418 ± 122.3 g and 640 ± 68.2 g, respectively: $t= -3.41$, $df=12$, $p \leq 0.005$).

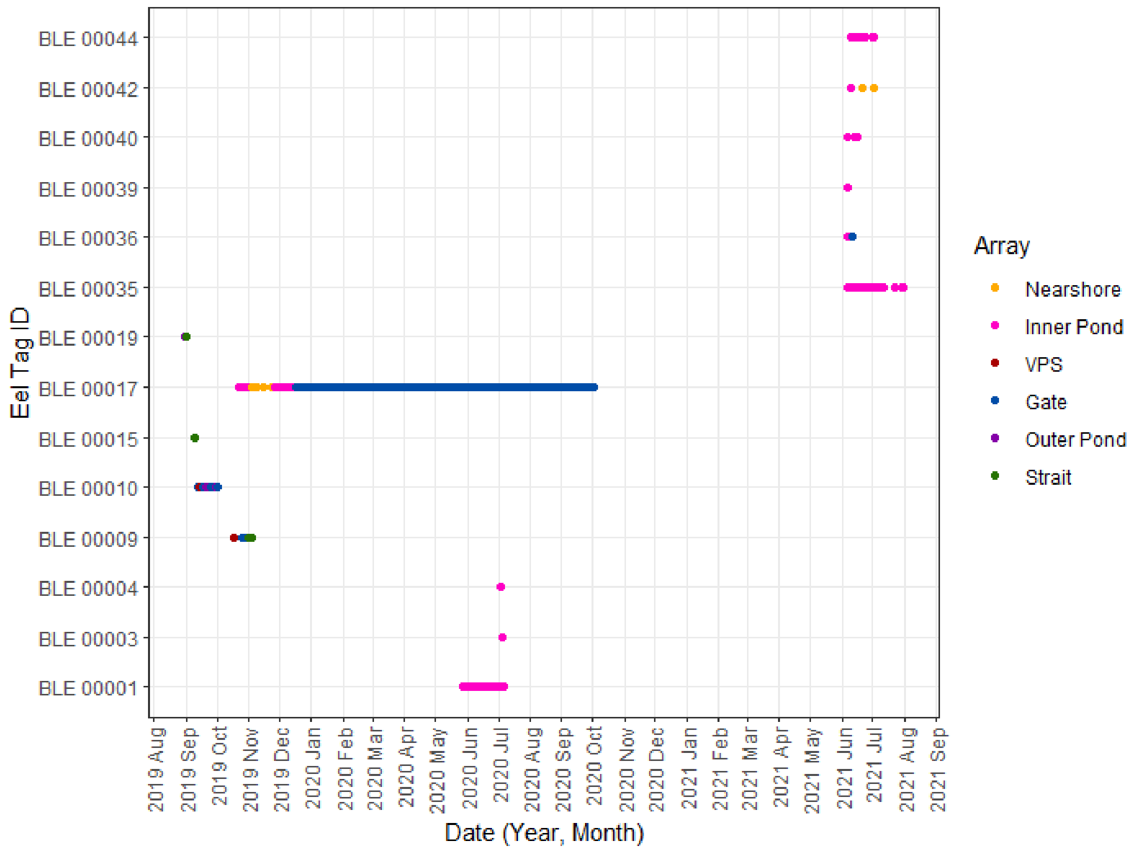



**Figure 3.2** Abacus plot of each individual eel tag ID by year and month coloured by array. This figure is based on a subset of eels that were captured and released in the Inner Pond (n=14 of 19 eels detected after release).

### 3.3.4 Large scale movements

It was predicted that more eels would remain in the BdOL compared to those that left. Contrary to these predictions, a significant difference was found in the number of eels that left East Bay (n=9) and possibly the BdOL (last detected in the Strait), compared to those that remained in East Bay (n=3; $\chi^2$=108, df= 11, $p \leq 0.005$). No significant difference in total length was found between eels that left the BdOL (last detected in the Strait, 581 ± 226 mm) compared to those that remained in East Bay or were last detected at the Gate (662 ± 64 mm; t= 1.13, df=17, $p > 0.05$). However, eels that left the BdOL

(733.8 ± 127.0 g) were significantly heavier than those that remained in East Bay (478 ± 134.8 g; $t= -4.16$, $df=17$, $p \leq 0.005$) indicating eels that weighed more left the BdOL.

Large scale analysis further examined the movements of vagrant eels (n=12) by grouping them as either uni-directional (n=9) or multi-directional (n=3). Vagrant eels that exhibited uni-directional movements and left East Bay (n=9) did so via the west side of the Gate, hugging the shore to the Outer Pond, and then travelling through the Strait (Fig. 3.3). Seven of the vagrant uni-directional eels that passed the Strait left in Fall between September 01 and November 26, and one (BLE 00017) was detected to leave as early as July 21. One uni-directional vagrant eel was detected in the nearshore array six days after it was tagged and was not detected anywhere else.

Overall eels that displayed uni-directional movements from release site to the Strait travelled at least 23.0 km and spent an average of 67 days in East Bay and the main BdOL before their final detection in the Strait. Most of the vagrant uni-directional eels passed the Strait in less than 11 days (n= 6), while two took between one and three months, and one eel (BLE 00023) went undetected after tagging until the next calendar year and was only detected for 6 days of the 276-day detection period (Table 3.1). One vagrant uni-directional eel tagged in 2019, was detected outside the BdOL in the Laurentian Channel, 211 km from its release site. This eel (BLE 00025) took approximately 29 days to travel from RS 4 to the Laurentian Channel (Fig 3.1). No eels that exhibited uni-directional movements and were last detected at the Strait returned. Furthermore, no eels that were detected at the entrances to the BdOL, likely due to late deployment of receivers at these locations (October 2020) and the small numbers of eels tagged in 2020 and 2021.

Vagrant eels that demonstrated multi-directional movements (n= 3) revealed some individual variability. One of these vagrant eels (BLE 00010) travelled back and forth between the Gate and VPS array while the other made several attempts to leave through the Strait yet returned to the Gate at least twice. BLE 00010 was last detected at the Gate suggesting it went somewhere else in the BdOL, and one eel travelled between the Inner Pond and the Nearshore array and was last detected in the Nearshore array (Fig. 3.3). The eel (BLE 00017) that travelled back and forth between the Inner Pond to the Nearshore array settled in the Gate on December 16 where it wintered and emerged in the summer on June 14th when it travelled to the VPS array and back to the Gate before the tag's battery died (Fig. 3.3).

Utilization distributions of vagrant eels derived from RSP analysis revealed variation in home ranges. The 50% UD core range of tagged eels varied from 0.3 to 29 $km^2$ (mean ± SD = 7.1 ± 8.6 $km^2$), while the 95% UD home range varied from 1.6 to 109.1 $km^2$ (mean ± SD = 29.2 ± 34.2 $km^2$). Utilization distributions were largest for eels that demonstrated multi-directional movements and for those that were released in the Nearshore area.

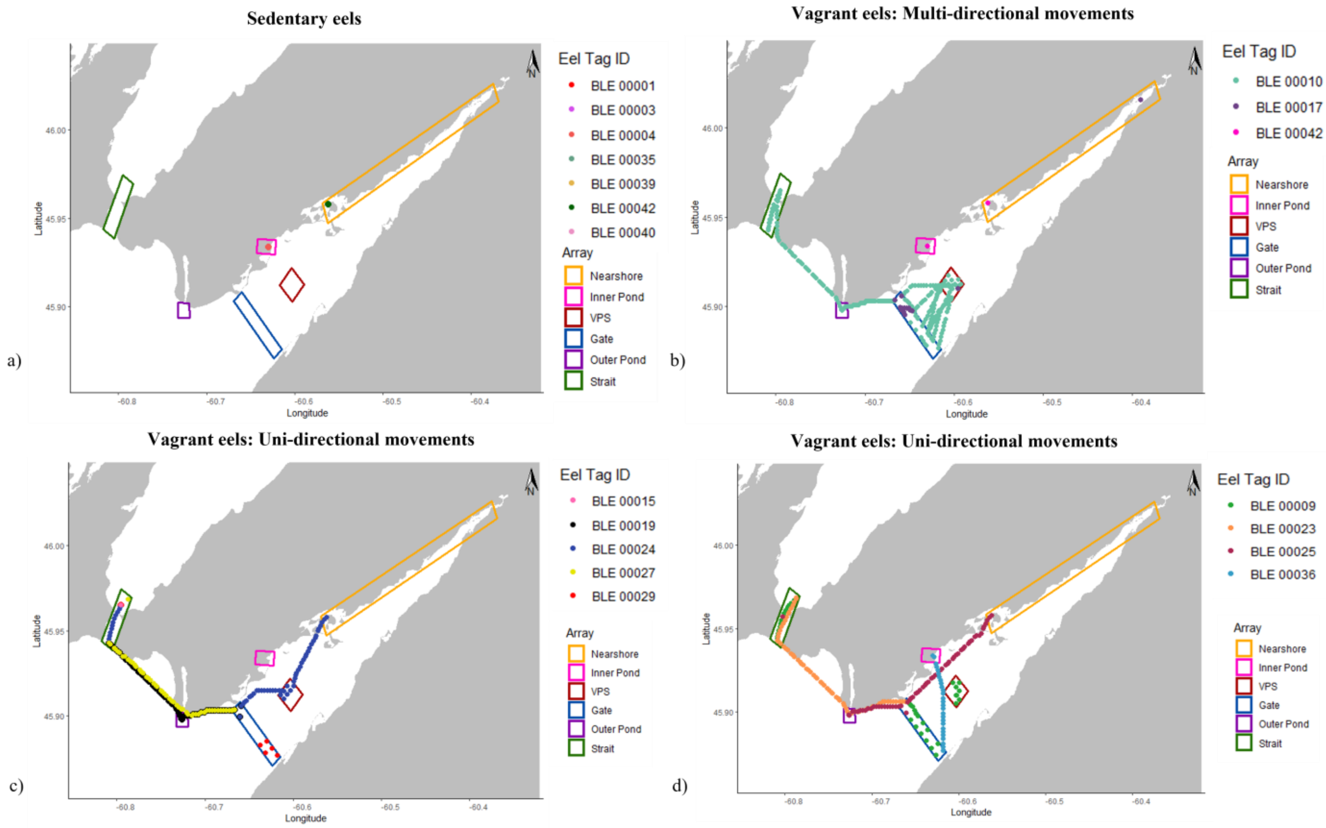



**Figure 3.3.** Movement patterns of detected American eels in the Bras d'Or Lake (n=19 of 33 tagged). Three patterns are demonstrated a) sedentary eels (n=7, e.g., all eels detected at the Inner Pond were only detected at this array and thus overlap), b) multi-directional vagrant eels (n=3), and uni-directional vagrant eels (n= 9) (c, and d). Uni-directional movements were split into two figures to avoid overlapping paths.

### 3.3.5 Eel and environmental data

American eels have a wide tolerance for temperature (0-31 °C) and a favorable temperature in fresh waters between 17-20°C (Fisheries and Oceans Canada, 2013). In this study, environmental data derived from DST-CTD data loggers were matched to eel detections. Eels were detected at average monthly temperatures ranging from 10 – 11 °C between July 23 to Sept 07 in 2019 and ranging from 14-18 °C between July 8, 2020, and July 22, 2021 (see Appendix J.1). Eel detections matched with environmental data in the coastal lagoon and nearshore arrays and occurred during an average monthly temperature ranging from 15.8 -20.7 °C between October 10, 2019 and July 31, 2021 with a mean salinity ranging from 7.1 -10.7 psu (mean=10.5°C psu). In the deeper regions of East Bay (> 2 m), average monthly temperatures ranged from 6.3 to 9.4°C (mean= 9.0 °C) while the average monthly salinity ranged from approximately 20.5 to 20.6 psu (mean= 20.6 psu) between July 8, 2020, and July 22, 2021 (see Appendix J.2). Furthermore, oxygen data were only collected at a single depth and at one location for all years. In 2019, oxygen data collected from July 23 to September 07 demonstrated that the average monthly dissolved oxygen ranged from 8.4 to 10.8 mg/L.

The preferred oxygen tolerance of eels in freshwater is reported to be > 4 mg/L and has not been reported for estuaries (Fisheries and Oceans Canada, 2013). While the effect of low oxygen is species specific, most fish species become stressed when concentration falls below 4 mg/L (Francis-floyd, 2003; Stevens, 2015). Furthermore, catches of American eels in North Carolina and Okland Lake Nova Scotia have reported greater catches in waters with dissolved oxygen levels above 4 mg/L (Stevens, 2015). No temperature data were available to convert percent oxygen concentrations to mg/L from October 2019 until the next DST-CTD data logger deployment. Meanwhile the average monthly dissolved oxygen data from July 15 to July 22, 2021, ranged from 9.0 to 12.75 mg/L. In winter of 2020-2021 the average dissolved oxygen was 12.3 mg/L.

### 3.3.6 Residency by array

Results of this study found eels spent more time in East Bay (in the Gate 54.5%) than the Inner Pond (31.9%). Furthermore, eels spent more time in the Gate and the Inner Pond compared to the other arrays (Nearshore = 31.7%, VPS = 2.9%, Outer Pond = 2.2, Strait = 5.0%, and the Ocean = 0.1%). Significant differences in residency time were found for eels that were detected in the Pond, the Gate, and the Strait ($p < 0.005$, df= 2,

$\chi^2$=11.757). Multiple comparisons between groups demonstrated no significant differences in residency time for eels that remained in East Bay compared to eels that left the BdOL (Strait) ($p > 0.05$) or eels that remained in East Bay and in the Inner Pond ($p > 0.05$). However significant differences were found in residency times between those detected in the Inner Pond and those that left the BdOL ($p < 0.05$) as well as the Inner Pond compared to the Ocean ($p < 0.05$).

### 3.3.7 Habitat associations

Habitat associations were assessed for eels within East Bay (18 of the 19 detected eels). Eels were found on all benthoscape classes determined in Chapter 2 except for *Deep Silt/Mud (≥ 50 m)*. Individual variability among eels was found within the classified habitats (Table 3.4). Nearly half (7 of 18 detected eels) of the eels were found to occur only on *Patchy Vegetation* (Table 3.4). Sedentary eels that were detected in the Inner Pond or the Nearshore array occurred only on *Patchy* or *Continuous Vegetation* benthoscape classes (Table 3.4). Meanwhile, vagrant eels, especially those that travelled back and forth between the VPS and the Gate, were found to occur predominately on hard bottom benthoscape classes (Table 3.4; BLE 0009, BLE 00010). Overall, the proportions of detections were greatest on *Shallow Silt/Mud (≤ 50 m)* for five eels (Table 3.4; BLE 0009, BLE 00010, BLE 00017, BLE 00023, BLE 00024) while two eels had nearly equal detections on *Shallow Silt/Mud (≤ 50 m)* and *Mixed Sediments* (Table 3.4; BLE 00019, BLE 00025). Additionally, the single eel that was found to overwinter in the gate (BLE 00017, Figs. 3.2; 3.4) occurred predominately on *Shallow Silt/Mud (≤ 50 m)* habitat.

**Table 3.4.** Proportions (%) of detections that occur per habitat type for each American eel detected. No data represent areas where habitat information was unavailable. All receivers within the Bras d'Or Lake were used.

| **Eel ID** | **Benthoscape class** | | | | | |
|---|---|---|---|---|---|---|
| | **Patchy Veg.** | **Coarse Sed.** | **Mixed Sed.** | **Silt Mud ≤ 50% Gravel** | **Shallow Silt Mud (≤ 50 m)** | **Cont. Veg.** |
| BLE 00001 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| BLE 00003 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| BLE 00004 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| BLE 00009 | 0.0 | 12.6 | 4.8 | 15.2 | 67.4 | 0.0 |
| BLE 00010 | 0.0 | 13.7 | 28.0 | 11.9 | 46.4 | 0.0 |
| BLE 00017 | 2.4 | 0.0 | 0.0 | 0.0 | 97.6 | 0.1 |
| BLE 00019 | 0.0 | 0.0 | 52.0 | 0.0 | 48.0 | 0.0 |
| BLE 00023 | 0.0 | 0.0 | 0.0 | 35.0 | 65.0 | 0.0 |
| BLE 00024 | 0.0 | 0.0 | 13.1 | 7.3 | 55.5 | 24.1 |
| BLE 00025 | 0.0 | 0.0 | 25.3 | 0.0 | 39.6 | 35.2 |
| BLE 00027 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 |
| BLE 00029 | 0.0 | 44.9 | 0.0 | 20.5 | 34.6 | 0.0 |
| BLE 00035 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| BLE 00036 | 99.9 | 0.1 | 0.0 | 0.0 | 0.0 | 0.0 |
| BLE 00039 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| BLE 00040 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| BLE 00042 | 91.0 | 0.0 | 0.0 | 0.0 | 0.0 | 9.0 |
| BLE 00044 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |

### 3.3.8 Habitat availability and habitat use

Eels were detected on six of the seven benthoscape classes, however, no receivers were placed in *Deep Silt/Mud (≥ 50 m)*. Eels used *Shallow Silt/Mud (≤ 50 m)* and *Patchy Vegetation* more than any other benthoscape class in East Bay while eels demonstrated little use of *Coarse Sediments*, *Silt/Mud with < 50% Gravel*, or *Mixed Sediments* benthoscape classes (Fig. 3.4). No significant difference was found in the average habitat used (observed) compared to the habitat available (expected) ($p = \geq 0.05$, df=5, $\chi^2$=0.72). However, it is unknown whether eels use all the habitat available to them as they will only be detected at known receiver locations and within the detectable ratios of the deployed stationary receivers (500 m radius). Only the benthoscape classes within that detectable area can be determined. Therefore, eels may be using more habitat than was able to be identified in this study as it does not account for individuals that could not be recorded outside of these known station's detectable radius.

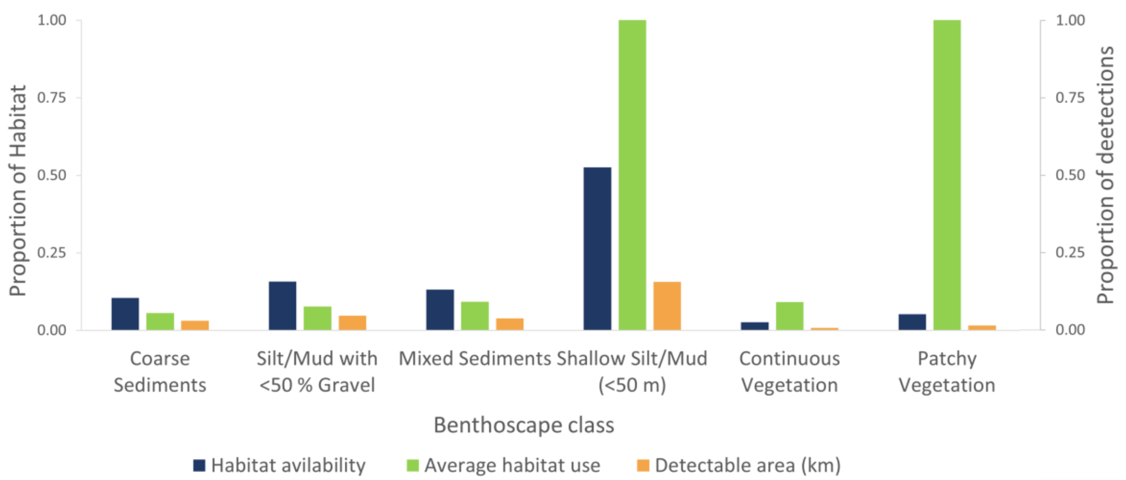



**Figure 3.4.** Habitat availability and the proportion of detectable habitat available to American eel in the Bras d'Or Lake and the average habitat used by tagged eels for each benthoscape class based on telemetry detections from August 02, 2019, to July 11, 2021.

Detections were recorded for one eel during the winter of 2019-2020, for two eels in spring 2020, for 12 eels in summer 2019 to 2021, and for seven eels in fall. No eels were detected to overwinter in 2020-2021 or were detected in spring 2021. Two eels (of the 18 used for analysis) were detected in more than one season: one eel (BLE 00017) tagged in summer of 2019 was detected in all four seasons from 2019 to 2021 and at several arrays, while the other (BLE 0001) was tagged in summer 2019 and detected in both the spring and summer of 2020. The remaining eels (n=16) were detected in only a single season that corresponded to their time of tagging. However, two of these 16 eels

(BLE 00003, BLE 00004) were tagged in summer of 2019 and not detected until summer 2020 (Fig. 3.2).

A significant difference between the expected and observed distribution of detections was found for all eels for each season: winter ($p= \leq 0.05$, df=5, $\chi$=Inf), spring ($p = \leq 0.05$, df=5, $\chi^2$=Inf), summer ($p = \leq 0.05$, df=5, $\chi^2$=Inf), and fall ($p = \leq 0.05$, df=5, $\chi^2$=12.09) (Fig. 3.7). Eels exhibited a strong affiliation with *Shallow Silt/Mud (≤ 50 m)* in all four seasons (estimated average habitat use) followed by *Continuous Vegetation* in the winter and fall and *Patchy Vegetation* in the spring and summer (Fig. 3.5). In summer, eels used *Shallow Silt/Mud (≤ 50 m)* followed by *Patchy Vegetation* (Fig. 3.5). HSIs (0.0-1.0) in this study echoed patterns observed in the average habitat use with greater HSI values in habitats with *Shallow Silt/Mud (≤ 50 m)* in all four seasons followed by *Continuous Vegetation* in winter and fall and *Patchy Vegetation* in spring and summer.

There was little or no habitat use by eels for all four seasons for three of the six benthoscape classes: *Coarse Sediments*, *Silt/Mud with ≤ 50% Gravel*, and *Mixed Sediments*. For low habitat use areas a significant difference in the observed habitat used compared to the habitat available (expected) was found in summer ($p = \leq 0.05$, df=2, $\chi^2$=Inf) and winter ($p = \leq 0.05$, df=2, $\chi^2$=Inf), while no significant difference was found in the observed habitat used compared to the habitat available (expected) in fall ($p = > 0.05$, df=2, $\chi^2$=0.16). In spring eels were only detected in high use habitats. Areas estimated to be high habitat use by eels were *Shallow Silt/Mud (≤ 50 m)*, *Patchy Vegetation* and *Continuous Vegetation*. Significant differences were found between the available habitat and high use habitat areas for eels in winter ($p = > 0.05$, df=2, $\chi^2$= Inf), summer ($p = > 0.05$, df=2, $\chi^2$= 23.82), and spring ($p = > 0.05$, df=2, $\chi^2$= Inf), whereas areas of high use habitat were not significantly different from that available in fall ($p = > 0.05$, df=2, $\chi^2 = 1.39$).

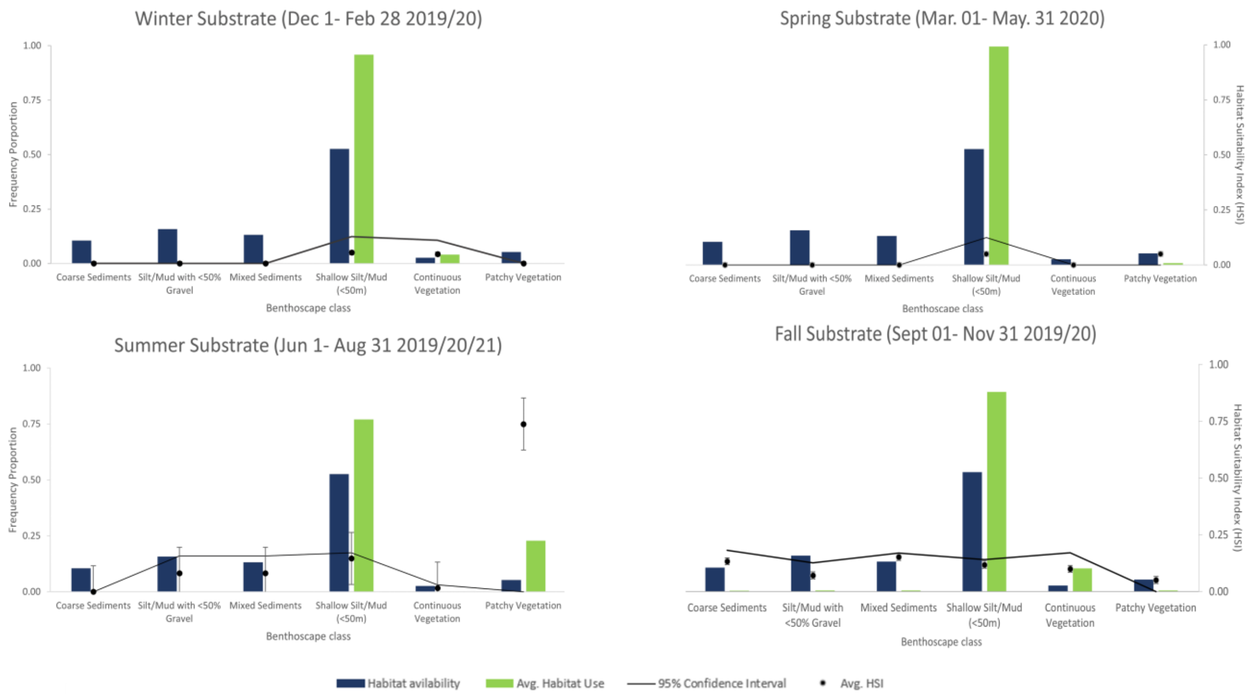



**Figure 3.5.** Habitat suitability indices (HSI) for American eels (n=18) for a given season a) winter, b) spring, c) summer, d) fall by benthoscape class based on telemetry detections from August 02, 2019, to July 11, 2021. Points indicate the average HSI value for each substrate for all eels. The 95% confidence intervals were determined from the standard error of the HSI values. Blue bars represent the proportion of each habitat available in East Bay and were calculated from using hydroacoustic receiver data and a radius of 500 m for each receiver. Green bars represent the average habitat use proportions based on hydroacoustic data within a 500 m radius of where individuals were detected.

### 3.3.9 Diel period and habitat associations

Eels were detected in both the day (46.3%) and the night (53.7%) (Fig. 3.6). However, individual variability was found in the diel period in which eels were most active (Fig. 3.6). Furthermore, eels demonstrated individual variability in diel period and habitat type (Table 3.5). Among sedentary eels, which occurred on *Patchy Vegetation*, approximately half of the individuals moved during the day (42.8%) and half moved at night (42.8%) while some moved in both the day and the night (14.3%). Vagrant uni-directional eels were detected mainly at night and were detected on the greatest range of hard bottom habitats (Table 3.6). Of the vagrant multi-directional eels (n= 2), one eel (BLE 00017) moved less in the day (48.7%) than the night (51.7%), while the other (BLE 00010) moved more during the day (54.5%) than night (45.5%). BLE 00017 occurred only on *Shallow Silt/Mud (≤ 50 m)* and *Patchy Vegetation* during the day and the night with the dominant substrate also being *Shallow Silt/Mud (≤ 50 m)*, whereas BLE 00010 was found on four habitat types: *Shallow Silt/Mud (≤ 50 m)*, *Silt/Mud with < 50% Gravel*, and *Coarse Sediments* in both the day and the night yet predominately occurred on *Shallow Silt/Mud (≤ 50 m)* in both diel periods respectively (Table 3.5).

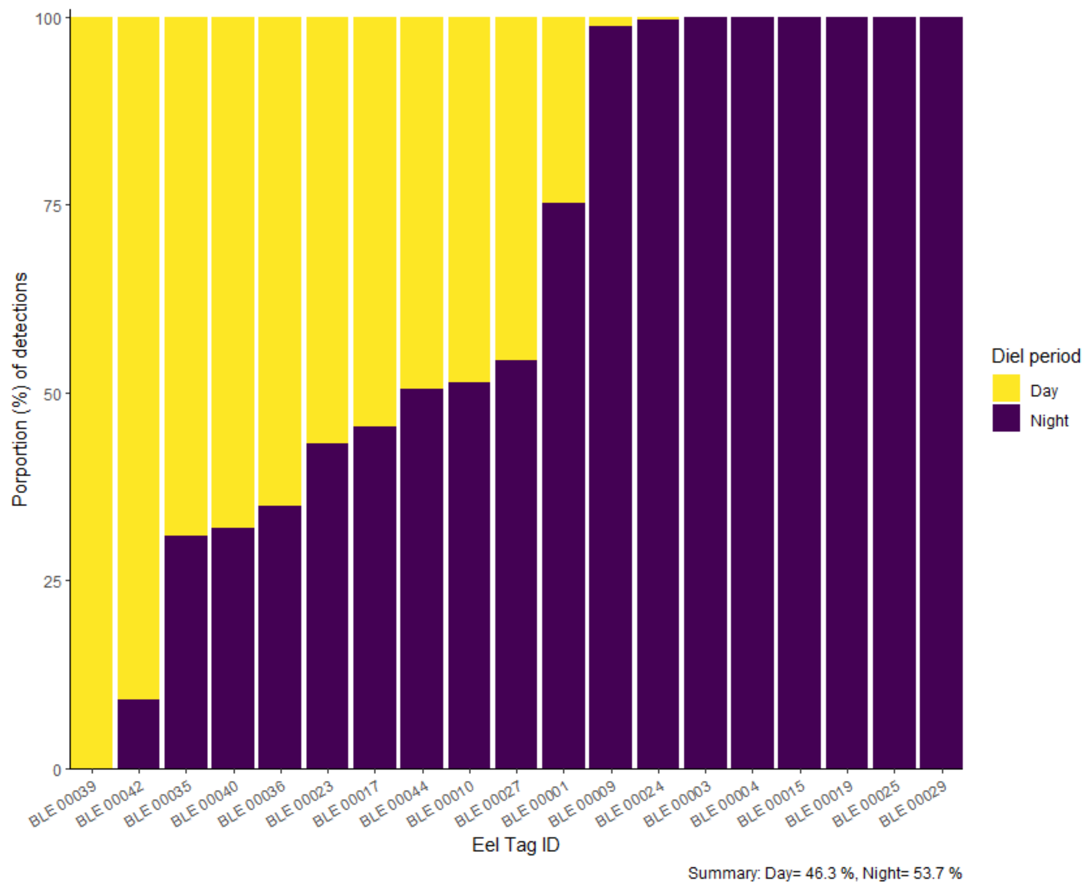



**Figure 3.6.** Proportion of detections for individual American eels (n=19) according to diel period.

**Table 3.5.** Proportion of time individual eels spend on each habitat type in the day and the night. Behaviors of individuals were grouped as either Sedentary (S) or Vagrant (V) demonstrating either unidirectional (U) or multi-directional movements. (M).

| | Day | | | | | | Night | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Eel ID** | **Shallow Silt Mud (≤ 50 m)** | **Silt/ Mud < 50% Gravel** | **Mixed sed.** | **Coar. Sed.** | **Patch Veg.** | **Cont. Veg.** | **Shallow Silt Mud (≤ 50 m)** | **Silt/ Mud < 50% Gravel** | **Mixed Sed.** | **Coarse Sed.** | **Patchy Veg.** | **Cont. Veg.** | **Group** |
| **BLE001** | 0.0 | 0.0 | 0.0 | 0.0 | 24.8 | 0.0 | 0.0 | 75.2 | 0.0 | 0.0 | 0.0 | 0.0 | S |
| **BLE 03** | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 | S |
| **BLE 04** | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 | S |
| **BLE 09** | 0.0 | 0.0 | 1.3 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 9.1 | 3.5 | 11.0 | 65.5 | V, U |
| **BLE 10** | 0.0 | 0.0 | 9.2 | 8.6 | 0.0 | 0.0 | 0.0 | 0.0 | 3.7 | 16.4 | 7.4 | 22.4 | V, M |
| **BLE 17** | 27.5 | 3.4 | 0.0 | 0.0 | 0.9 | 0.0 | 0.1 | 1.5 | 0.0 | 0.0 | 0.0 | 43.9 | V, U |
| **BLE 19** | 53.6 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 29.9 | 0.0 | 70.1 | V, M |
| **BLE 23** | 0.0 | 0.0 | 0.0 | 56.7 | 0.0 | 0.0 | 0.0 | 0.0 | 27.6 | 0.0 | 6.5 | 9.2 | V, U |
| **BLE 24** | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.4 | 11.1 | 0.0 | 0.0 | 21.5 | 3.6 | 58.4 | V, U |
| **BLE 05** | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 18.6 | 0.0 | 0.0 | 13.8 | 0.0 | 29.9 | V, U |
| **BLE 27** | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 29.2 | 25.0 | V, U |
| **BLE 29** | 0.0 | 45.8 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 44.2 | 0.0 | 20.8 | 35.1 | V, U |
| **BLE 35** | 0.0 | 0.0 | 0.0 | 0.0 | 69.0 | 0.0 | 0.0 | 31.0 | 0.0 | 0.0 | 0.0 | 0.0 | V, U |
| **BLE 36** | 0.0 | 0.0 | 0.0 | 0.0 | 65.1 | 0.0 | 0.0 | 34.8 | 0.1 | 0.0 | 0.0 | 0.0 | S |
| **BLE 39** | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | V, U |
| **BLE 40** | 0.0 | 0.0 | 0.0 | 0.0 | 68.1 | 0.0 | 0.0 | 31.9 | 0.0 | 0.0 | 0.0 | 0.0 | S |
| **BLE 42** | 0.0 | 0.0 | 0.0 | 0.0 | 90.9 | 0.0 | 9.1 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | S |
| **BLE 44** | 0.0 | 0.0 | 0.0 | 0.0 | 49.6 | 0.0 | 0.0 | 50.4 | 0.0 | 0.0 | 0.0 | 0.0 | S |

## 3.4 DISCUSSION

### 3.4.1 Coastal movements

Work by Ford and Mercer (1986) found larger eels (>360 mm) occupied wider marsh creeks and restricted small eels (<160 mm) to the narrower creeks in the Great Sippewisset Marsh, Massachusetts and suggested eels demonstrate territorial behavior as a mechanism for maintaining differences in size class distributions and a limited home range. In this study, American eels captured and released in a culturally significant coastal lagoon showed that smaller sedentary eels (639.7 mm, 418.3 g) remained in the pond in summer longer (up to 57 days) while larger (717.8 mm, 539 g) vagrant eels left within the same day as being tagged. These results are consistent with the results of Ford and Mercer (1986) where small and large eels may occupy different areas of the same waterbody. These results are also supported by and overlap with MEK fishing locations in summer in the BdOL and by MEK which suggests larger eels may occupy deeper waters (> 4 m) (see Appendix I; Denny et al., 2012; Giles, 2014). In this study, the Inner Pond is a shallow coastal lagoon (1.2 m) comprised of *Continuous Vegetation*. Vegetated habitats, either *Patchy* or *Continuous* have been linked to increased food availability compared to adjacent bare sediments and to provide adequate protection from predators during daylight hours (COSEWIC, 2012; Murphy et al., 2021; Olson et al., 2019). Therefore, these habitats may provide suitable protection and foraging for smaller eels during their growth phase. Telemetry results, however, were not supported by MEK which suggested eel may use coastal lagoons for winter dormancy as the overwintering eel was found at deeper depths (25 m). Using telemetry data alone, it is unclear whether vegetation or predator-prey dynamics may be driving the difference in size class distribution of eels to the nearshore array and Inner Ponds compared to the other arrays that were situated in the middle and mouth of East Bay.

MEK shared by project partners have also discussed the increase in striped bass in the BdOL, a known predator to American eel. It is unclear whether the behavior of smaller eels occupying coastal ponds reflects the increase of predators such as striped bass as well as other larger eels. However, given information available in this study using the benthoscape map and telemetry data, it is likely that the protection provided by eelgrass beds in nearshore habitats and the Inner Pond provide adequate protection and access to more suitable food resources for smaller eels. Consequently, this study was limited to known receiver locations and therefore eels could only be detected within the 500 m range of a known receiver location (Denny et al., 2012; Giles, 2014). Additionally, at least two receivers placed in nearby coastal lagoons were lost in 2019 and were not replaced, limiting results to a single coastal lagoon. These limitations, as well as the abundance of coastal lagoons found in the BdOL (> 500), demonstrate how knowledge sharing can help deepen our understanding of eel movements and habitat use, by making space for different types of knowledge necessary to balance each other's limitations.

### 3.4.2 Largescale movements

### 3.4.3 Migration

Once mature, silver stage American eels are understood to travel from continental waters to the Sargasso sea to spawn (Béguer-Pon et al., 2015). Although this study aimed to tag only yellow stage American eels, at least one of the six uni-directional vagrant eels, was detected on a neighbouring array in the Laurentian Channel 211 km from its release site, suggesting a mature silver eel was tagged. This eel (BLE 00025) travelled from Eskasoni, East Bay to the Laurentian Channel over approximately 29 days.

Previous studies have calculated river and estuarine migration speeds of silver eels to range between 0.05 and 190 km a day with the average maximal value below 75 km day and a median speed of ≤ 17 km day (Béguer-Pon et al., 2018). Migration speeds for American eels are thought to be higher (between 10 and 52 km/day) compared to European eels (between 2 and 51 km/day) (Béguer-Pon et al., 2018). Studies have suggested that higher migration speeds of 10-52 km/day would be sufficient for American eels to reach the Sargasso Sea at the beginning of the breeding season following the initial outward migration from continental waters (Béguer-Pon et al., 2015; 2018). For other eel species, migration speeds have been documented to range from 15.1 to 31.3 km/day in the New Zealand longfin eel (*A. dieffenbachia*) and to average 10 km/day for the Giant mottled eel (*A. marmaorata*) and the Polynesian long finned eel (*A. megastoma*) (Béguer-Pon et al., 2018; Jellyman & Tsukamoto, 2002; Schabetsberger et al., 2013, 2015).

Previous studies documenting the swimming capacity of eels have reported an optimal speed of 0.81 body lengths per second (Lennox et al., 2018; Palstra & van den Thillart, 2010). In contrast, data from this study suggested eels moved between 0.001 and 0.13 body lengths per second based on the total distance covered and time of first and last detection. These results suggest that uni-directional eels and the outward migrating eel moved slowly compared to their optimal swimming capacity. Thus, estimated speeds of tagged eels in this study suggest these speeds would be too low to migrate to the desired spawning area, yet these slow rates may be due to heightened awareness and risk associated with navigating out of continental waters and that eels studied were not ready to undertake migration, as expected given that immature yellow eels were targeted for tagging.

During migration eels must move through multiple habitats and therefore are exposed to additional risks such as predation and energetic exhaustion to reach the spawning location in a timely manner (Lennox et al., 2018). Lennox (2018) suggested that European eels may comprimise speed for safety in the early marine spawning migration and avoiding migrating during the day to avoid predation. In this study at least five of the 8 uni-directional eels and the outward migrating eel BLE 00025 moved only at night (Fig. 3.6). BLE 00025 travelled from the Strait to the Laurentian channel 2.1 km/day and from the release site in East Bay to the Laurentian channel at 7.2 km/day. For the remaining 8 uni-directional eels, rates of movements ranged only from 0.1-1.8 km/day (mean=0.82 km/day). Given findings of Béguer-Pon et al., 2015 and Béguer-Pon et al., 2018, the swimming rate of BLE 00025 appears to be below what is needed to make the long ocean migraton to the spawning area in the Saragasso Sea. The other 8 uni-directional eels moved even slower and they were not detected to return through the Strait or East Bay. These results could suggest these eels were simply foraging in the BdOL during these night-time linear movement patterns which may conserve energy or avoid detering prey they are seeking while foraging (F. Whorsikey, personal communication, 2022). American eels in the BdOL have several predators including seals, other eels, striped bass, mergansers, eagles, and herons. Consequently, movement at night may suggest eels are more successful at foraging by feeding at night as they may feed on prey items that are also nocturnal foragers avoidng daytime predators themselves

(F. Whorsikey, personal, communication, 2022). It is also possible that the movement pattern displayed by BLE 00025 could indicate leaving the BdOL towards spawning grounds, but not yet having reached its maximum migration speed.

American eels at northern latitudes are understood to be predominately females (Jessop, 2010; 2008). Yellow eels, however, are sexually indetermined and therefore, sex was not collected in this study as this requires dissection or histological criteria of matured eel which is not always reliable (Atlantic States Marine Fisheries Commission, 2000; Jessop, 1987; Peterson et al., 1985; Sinha & Jones, 1966). Therefore, it is unclear whether the outward migrating eel was indeed a female leaving to spawn. In freshwater rivers of Nova Scotia maturing female silver eels have been documented to range between 394-945 mm and weigh 107-1,641 g in the Medway River to 435-939 mm and 163-1,793 g in the Lahave River (Jessop, 1987). Meanwhile, female yellow eels from Eel Brook, another freshwater river in Nova Scotia, ranged in total length of 385-702 mm and weighed 98-693 g (Jessop, 1987). However, these lengths and weights may not be comparable to estuaries as eels may mature at younger ages and experience faster growth rates in brackish waters compared to freshwater environments (Cairns et al., 2009; Jessop et al., 2008; Morrison & Secor, 2003). However, eels tagged in this study appear to be within the range of yellow and silver eel sizes found in other Nova Scotia studies as well as those by Denny et al (2013) in the BdOL, which had a total length of 165-928 mm and a total weight of 107 to 1,503 g. However, while the length (630 mm) of the outward migrating eel in this study correspond to those of silver eels, this eel was lighter (412 g) than other silver migrating eels documented in previous studies (Jessop, 1987).

The remaining five eels that demonstrated uni-directional movements and were not detected to return to the Strait or the East Bay ranged in total length from 645 – 795 mm and left as early as July with most leaving between September and November. The outward migrating eel (BLE 00025) was first detected on October 26, 2019 and was detected to leave the Strait of the BdOL on November 2 and was detected in the Laurentian channel November 26, 2019. The timing of presumed migration of this eel corresponds to timing of eels migrating from other Nova Scotia Rivers (COSEWIC, 2012; Jessop, 1987; Thibault et al., 2007). For example, eels migrating from the Upper St. Lawrence River have left as early as June and as late as November in the Sud-Ouest River (South shore of St. Lawrence) and East River Chester, Nova Scotia (COSEWIC, 2012; Thibault et al., 2007). In Cape Breton, eels have been documented to leave the Margaree Lake, southwest of the BdOL, in September (COSEWIC, 2012; Thibault et al., 2007). No eels that were last detected in the Strait returned to the BdOL, suggesting these eels left to spawn, left to seek freshwater to overwinter, or simply went somewhere else in the BdOL.

### 3.4.4 Unique behaviors

During the yellow phase, eels are generally sedentary (Béguer-Pon et al., 2018; Hedger et al., 2010). However, during certain times of the year, yellow eels may display some restlessness or movement and perform "dry runs" over a period of several migration seasons as they mature into silver eels before migrating to spawn in the Sargasso Sea (Hain, 1975). After each 'dry run', their migratory characteristics may diminish either entirely or to a large degree until the next migratory season, meaning that eels may revert to the yellow stage and continue to feed (Hain, 1975). At least one eel, BLE 00010, demonstrated unique movement patterns as it travelled from the Gate to the Strait, then from the Gate to the Outer Pond and back to the Gate where it was last detected (Appendix H). While I cannot say for certain this eel was performing a dry run as it was detected in only one season, this behaviour is consistent with a dry run. Furthermore, it is unclear whether this eel remained in the BdOL as detections ceased. Future studies should examine eel movements over multiple years using a longer-term study to better understand movement patterns of eels and to gain a deeper understanding of their behaviour in the BdOL ecosystem. Weatherhead (1986) suggested that one risk of short-term studies may be that individuals may demonstrate too many unusual events (those that may occur by chance) and as a result we may overestimate the importance of some of these events when we lack the perspective provided by a longer study.

### 3.4.5 Home range

Home range is defined as the movement an animal travels during its regular activity (Worton, 1989). Estimates of the home ranges of American eels are extremely variable depending on the shape and size of the water body (fresh or estuarine) and the size of the eel, increasing with total body length (Thibault et al., 2007). In this study, sedentary eels were smaller (640 mm, 418 g) and remained in the pond longer while larger (718 mm, 539 g) eels used East Bay and the BdOL demonstrating a greater distribution. However, it is difficult to state whether the home ranges observed in the BdOL are unique compared to other estuaries as this information is limited.

Anguillid eels have demonstrated a variable home range in tidal creeks, estuaries and saltmarshes ranging from 0.01 $km^2$ to 3.25 $km^2$ (Parker, 1995; Bozeman et al., 1985; Helfman et al., 1983; Walker et al., 2014). In freshwater lakes, the home range of short finned eels (*A. australis*) was roughly 1 km$^2$ while in large rivers American eels reached home ranges of up to 1.3 km$^2$. Variation in home ranges are a result of individual varibility in anguillid movements, the methods used to examine locations (passive vs. active tracking), the spatiotemporal coverage of the area studied, as well as the size of the water body being considered (Begeuer Pond et al., 2018; Hedger et al., 2010). Home ranges reported by Parker (1995) of 3.25 km$^2$ covered up to 17 km$^2$ of the 25 km$^2$ narrow estuary while those documented in tidal creeks where the spatial coverage was limited to 1 km$^2$ suggesting eel used all of the habitat in this small area.

Telemetry results derived from RSP analysis in this study demonstrated the 50% utilization distribution (UD), or the core range of tagged eels ranged from 0.3 to 29.0 km$^2$ (mean ± SD = 7.1 ± 8.6 km$^2$), while the 95% UD home range, ranged from 1.6 to 109.1 km$^2$ (mean ± SD = 29.2 ± 34.2 km$^2$). These home ranges were largest for eels that demonstrated multi-directional movements as well as those that were released in the nearshore area and are larger than those documented in previous literature in estuaries. These results may therefore be a result of the RSP analysis as core areas for eels exhibiting multiple movement patterns may exhibit a wider home range or total area covered based on the tracks available at the individual level. The East Bay covers approximately 125 km$^2$ with the widest part of the Bay at the mouth 8 km$^2$. These results suggest eels cover a significant portion of the habitat available in East Bay which may be a result of the diversity of habitats available. More importantly, home ranges may be unique in the BdOL given its geophysical structure.

The BdOL is considered a large estuary which is described as a semi-enclosed body of water with a free connection to the open ocean and where seawater is measurably diluted by inflowing freshwater from the land (Pritchard, 1967; Kennish, 2002; Webb, 2017). Estuaries are therefore partially enclosed coastal water bodies with one or more rivers or streams flowing into them, or, where a freshwater river or stream meets the ocean. The BdOL is unique with several rivers and streams (≥ 17) with six of them being major contributors of freshwater that feed the estuary (Parker et al., 2007; Yang et al., 2007). Given the size of this estuary (1,100 $km^2$) and in this study the size of East Bay (125 $km^2$), the home ranges of eels do not appear to be large given the habitat available in comparison to those documented in other studies where the shapes of estuaries were narrow and long, held a smaller spatial coverage, and did not hold the same diversity of habitats (coastal lagoons, access to freshwater in addition to the nearby connecting ocean as in the BdOL). This study may therefore further highlight key messages outlined by Beguer-Pon et al (2018) emphasizing the need to consider temporal movement patterns and determine home range estimates more carefully and consistently among studies for any meaningful management efforts directed at conserving this species to be determined and especially regarding fish passage. Furthermore, Beguer-Pon et al (2018) state that given the variability in tools and statistical packages used to determine home ranges as well as methods used for tracking species (passive vs active), making the comparison between home ranges of anguillid species, waterbodies, and regions have been difficult.

In this study larger home ranges were found in fall while smaller home ranges were observed in summer and likely reflect the degree of movement patterns observed by eels tagged. In this study eels in fall were either moving/foraging slowly displaying unidirectional movements or multidirectional movements and covering a larger area than sedentary eels remaining in the ponds. Future studies should consider determining seasonal home ranges of eels to identify the spatial and temporal limits of their regional distributions as eels that demonstrated larger home ranges in this study were also eels that presumably left the Lake and left East Bay. The home range of the American eel is extremely variable and given their ability to perform inter-habitat shifting, understanding seasonal home ranges may be valuable to our ability to monitor this species and further examine the possibility of hidden seasonal movements of these brackish residents. This information may help to further examine fishing areas in non tidal waters, alter fishing practices and regulations, and assess any threats to eel habitat to contribute better stewardship of this species.

### 3.4.6 Wintering

American eels have been shown to exhibit seasonal movement patterns where they may feed in brackish water in summer and then migrate to freshwater rivers to overwinter as water temperatures begin to cool (Thibault et al., 2007). However, it has been suggested eels may remain in estuaries in areas with freshwater upwelling and soft, muddy bottoms as mud provides a thermal layer required for winter survival (Tesch, 2003; Tomie et al., 2013, 2017). In estuaries, American eels begin their winter dormancy when water temperature drops below 10 °C and stop feeding and reduce their oxygen consumption at temperatures < 5 °C (Thibault et al., 2007; Walsh et al, 1983). American eels do not possess antifreeze proteins that would allow them to withstand cool water temperatures (Tesch, 2003; Tomie, 2011). Therefore, overwintering habitat may be limited or influenced by the eel's lack of antifreeze capabilities (Tomie, 2011).

In this study, at least one eel (BLE 00017) demonstrated winter dormancy in the Gate from December 16, 2019, to June 14, 2020, at 25 m depth on *Shallow Silt/Mud (≤*

*50 m)* (Table 3.4; see Appendix I). This is the first evidence of an eel being detected throughout winter dormancy in situ but, unfortunately, no temperature data were collected in 2019 due to technical issues with the data loggers deployed. Despite the lack of temperature data in the winter of 2019, results of one eel overwintering in deep water may suggest that eels may not require shallow habitats with freshwater upwelling nor movement to freshwater rivers to survive in winter. The ability to overwinter at 25 m may be supported by the unique characteristics of the BdOL system given that the estuary is comprised predominately of *Shallow Silt/Mud (≤ 50 m)* and the average depth is 30 m. This display of dormancy on *Shallow Silt/Mud (≤ 50 m)* does align with past studies in which substrate choice of American eel was observed in a lab under three different temperature regimes (Tomie et al., 2017). Mud and cobble substrates were equally preferred during wintering periods (1.5 to 0.6 °C), but during cooling (10.3 to 1.5 °C) and warming periods (1.2 to 9.0 °C), eels demonstrated a preference for mud (Tomie et al., 2017).

Another study on eels in small tributaries, such as the Bonnechere River, in the Saint Lawrence River, Ontario, Canada, have observed eels moving downstream in the fall from hard clay bottoms to areas in the lower reaches with mud or silt bottoms (MacGregor et al., 2008, 2009). Furthermore, several telemetry studies have concluded that the lack of eel detections in winter is linked to eels being burrowed beneath the substrate. Buried mud tags and detections of BLE 00017 in this study confirmed that eels can be detected even when burrowed 35 to 45 cm beneath the substrate (see Appendix, G), as long as they are in range of a receiver (up to 500 m radius). Traditional, and local knowledge have shared similar observations of eels in winter.

MEK has reported that larger eels may burrow deeper in winter and Mi'kmaq fishers have described adjusting their thrust when spearing in winter to catch larger "good sized" eels (Denny et al., 2012). Furthermore, evidence of winter dormancy has also been documented by traditional spear fishing through ice in Antigonish Harbour, Nova Scotia, as well as from local commercial eel fisheries (Stevens, 2015). Previous studies on short-finned eels (*Anguilla australis*) in other ecosystems have suggested that eels may congregate at high densities in areas with suitable refuge or vegetative cover due to contact with other eels as several eels have been observed to use the same burrowing tube even when several other tubes were available simultaneously (Aoyama et al., 2002; Tesch, 1977, 2003). However, results in this study found only a single eel to overwinter and that winter dormancy did not occur in primary fishing areas used by Mi'kmaw reducing eel susceptibility to overfishing as tagged eels did not appear to congregate together in these areas yet there may have been untagged eels congregating there.

Project partners shared MEK, and local knowledge shared by project partners have suggested that the sound of spring peepers, presence of dandelions, the budding of pussywillows, and presence of lightning bugs provide environmental cues that signal eels are beginning to emerge from winter dormancy, usually in late April to late May (Denny et al., 2012; S. Jeddore and A. Sylliboy, personal communication, 2020). Scientific advisory reports have also documented a similar time of year for when eels emerge from the mud (COSEWIC, 2012). BLE 00017 became active in the Gate as early as February 17 yet remained in the Gate until after winter when it moved from the Gate to the VPS array on June 14 2020 at 06:18:30 UTC (see Appendix I).

Future studies should consider using telemetry to determine when eel emerge from the mud to better understand the correlation between substrate choice and temperature in estuaries and to understand the role temperature, depth and substrate may play in eel habitat choice. Future work could also explore substrate choice in relation to body size as studies have found habitat preference of eels may vary based on body size, changing with increasing body length and life stage (Chaput et al., 2014; Lloyst et al., 2015; Machut et al., 2007). For example, smaller eels (<150-250 mm) have been associated with small cobble and gravel substrates while larger eels (351-450 mm) have been associated with larger cobble, boulders, and sand substrates (Lloyst et al., 2015; Machut et al., 2007; Stevens, 2015). Understanding the roles that temperature, depth, body size and substrate play in determining where an eel inhabits can help us better identify important habitats to eels and assess any threats either due to anthropogenic impacts or climate change to these habitats such as in nearshore coastal lagoons compared to deeper and more open habitats.

### 3.4.7 Habitat associations

The benthoscape map paired with acoustic telemetry data helped demonstrate which benthoscape classes American eels in East Bay, BdOL are associated with in different seasons. However, to understand habitat associations of eels, detections of tagged eels at known stationary acoustic receivers were used. Therefore, this study was limited as tagged eels may only be detected at known receiver locations and within the detectable radius of these stationary acoustic receivers (500 m radius). Therefore, it is unknown the degree to which tagged eels are present outside of these stationary acoustic receivers and may use the available habitats identified by the continuous benthoscape map. However, examining habitat associations in cases where eels were present within the known acoustic receiver locations can offer insights required to observe eel that extend beyond those available to MEK partners such as in deeper depths greater than 4 m and more closely over four seasons.

The results of this study found that the average habitat use (see 3.2.6 Methods) by eels showed a strong affiliation with *Shallow Silt/Mud (≤ 50 m)* in all four seasons (Fig. 3.7). Results in this thesis are supported by previous literature which has stated that American eels prefer shallow (< 30 m), sheltered embayment's and semi-exposed areas composed of soft sediments (Cairns et al., 2017). The BdOL, with a mean depth of 30 m, a diversity of habitats including an assortment of sheltered embayments and semi-exposed shorelines and consisting predominantly of soft bottom substrates may therefore offer ideal habitat for eel (Cairns et al., 2017; COSEWIC, 2012; Lambert, 2002; Parker et al., 2007). Moreover, given the BdOL is mainly *Silt/Mud* and the acoustic study array was deployed without prior knowledge of benthoscape classes in East Bay, these results are influenced by sampling bias as acoustic receivers may only detect eel at known receiver locations and the distribution of receivers among benthoscape classes could not be controlled.

This study demonstrated *Shallow Silt/Mud (≤ 50 m)* as the dominant habitat type which likely arose because receivers were not evenly distributed across benthoscape classes prior to the generation of the benthoscape map, as this knowledge was not available at the time of deployment. Therefore, more receivers occurred on *Shallow Silt/Mud (≤ 50 m)* (n= 20 receivers) compared to other benthoscape classes: *Silt/Mud < 50% Gravel* = 6 receivers*, Coarse Sediments* =3 receivers*, Mixed sediments= 5* receivers*,* *Deep Silt/Mud (≥ 50 m)* =0 receivers*, Patchy Vegetation* =4 receivers*,* and *Continuous Vegetation* =1 receivers. Furthermore, more eels were detected outside the Inner Pond and Nearshore array (where *Patchy and Continuous Vegetation* are found) for longer periods of time. For example, eels were not found to use the Inner Pond beyond 57 days and spent approximately 31.9% of their time in this area followed by the Nearshore array (31.7%) and most of their time in the Gate (57%). Moreover, several eels tagged in 2019 were detected to leave in fall while two travelled between the VPS and the Gate many times. Therefore, the results of *Shallow Silt/Mud (≤ 50 m)* as the dominant habitat used by eels are as expected given that the BdOL is comprised predominately of mud and the receivers were not evenly distributed across benthoscape classes. Additionally, eels demonstrated use of deeper depths where no vegetation occurred. However, vegetated areas may play a vital role in habitat used by eels as increased food resources such as small fishes reside in areas with greater vegetation (Cottreau, 2013; Murphy et al., 2021; Olson et al., 2019).

Previous work conducted in spring and summer in a small Nova Scotia lake found that American eels preferred areas with a greater proportion of vegetation cover (Cottreau, 2013; Stevens, 2015). These findings by Stevens (2015) and Cottreau (2013) are supported in my results as eels demonstrated a use of *Patchy vegetated* areas in spring and summer and *Continuous Vegetation* in fall and winter. However, in winter 2019-2020, only a single eel (BLE 00017) was detected and detections of this individual on *Continuous Vegetation* are likely because they left the Inner Pond in early winter (December 12) before moving to the deeper waters in the Gate (December 16) where it remained on *Shallow Silt/Mud (≤ 50 m)* with no attached vegetation.

Findings of eels associated with vegetated habitats in this study are not supported by Hallett (2013) which found no significant relationship between eel density and the percent of vegetative cover nor were eels observed in areas with more than 60% vegetative cover in 27 brackish and saltwater estuaries in the Gulf of St. Lawrence yet the study was limited to sheltered shorelines in depths $\leq$ 3 m. It is possible that the results of Hallett (2013) were not significantly correlated with vegetation as the methods used to observe eels were conducted at night using a glass bottom boat which required the ability to see and count eels and would become increasingly difficult with increased vegetation. Furthermore, substrate type was not reported by Hallett (2013), yet some suggestion was made for soft bottoms as it was documented that eels occasionally created a cloud of sediment, making it difficult to count and distinguish eels from other fish during surveys. Moreover, Hallet (2013) did not document the type of vegetative cover (eelgrass, kelp, other aquatic vegetation) and therefore it is difficult to understand whether these areas offer any insights to distinguishable food availability in relation to other behaviours such as hiding or predation.

Stevens (2015) suggested that vegetation may be an important factor to eels residing in areas where substrate type is not beneficial to eels such as hiding or protection. For example, in Oakland Lake, vegetation is absent in early spring to early summer until water temperatures warm and therefore eels may seek shelter based on increased structure for protection when vegetation is absent(Stevens, 2015). Consequently, sites with attached vegetation may allow eels to successfully hunt and ambush prey. Cottreau (2013) found higher abundance of eels in areas with moderate percentage of boulders and sand with attached vegetation (lily pads) and suggested that the combination of protection and potentially increase of available food resources may create optimal habitat for eels. The BdOL being predominately mud with eelgrass growing in soft bottom substrates, a mean depth of 30 m and a semi-exposed shoreline, may offer both suitable habitat and vegetative protection to eels.

The benthoscape map generated in this study which identified areas with attached vegetation paired with telemetry results of eel presence on *Patchy* and *Continuous Vegetation* are supported by eel distributions documented with MEK knowledge (Giles et al., 2016). Results in this study along with those of Giles et al., (2016) together support the hypothesis that eels do use vegetated habitats in the BdOL. These findings demonstrate how knowledge sharing can help strengthen our understanding of eel habitat use in the BdOL and the value that making space for different types of knowledge can have if used to help fill knowledge gaps.  Future studies could consider species distribution modelling using other sampling techniques such as satellite telemetry data, observations and MEK of known eel presence to gain a stronger collective understanding of eel distribution such as in Skroblin et al., (2020). Furthermore, MEK workshops will be valuable to help identify and fill knowledge gaps for eels at regional scales. Understanding eel movements and habitat use is critical to preserve and maintain historical eeling areas as well as for the adaptation and transfer of Mi’kmaq fishing knowledge (Giles et al., 2016). Finally, American eels are habitat generalists (Chaput et al., 2014). Future studies should consider habitat use under different behaviours such as feeding, refuge, or winter dormancy to better understand how eels use truly use these habitats and throughout different life stages to contribute better stewardship of eel and eel habitat.

### 3.4.8 Habitat suitability indices

In some cases, the estimated habitat used may have been greater than the habitat available such as on *Shallow Silt/Mud (≤50 m)* and *Patchy* or *Continuous Vegetation* in all four seasons (Fig. 3.7). This is explained by the methodology used to determine habitat available, which uses the average the number of hydrophones per benthoscape class compared to average habitat use determined by the proportion of detections within the detectable area (500 m) collected by hydrophones on a given benthoscape class. In this study placements of acoustic receivers were not distributed equally among benthoscape classes as this information was not previously classified and therefore not available prior to deployment. As a result, placement of acoustic receivers deployed in East Bay could not be controlled at the time of deployment and therefore occurred more on *Shallow Silt/Mud (<50 m)* than any other benthsocape class: *Shallow Silt/Mud (<50 m)* = 20 receivers, *Coarse Sediments* = 4 receivers, *Silt/Mud with <50% Gravel* =6 receivers, *Mixed Sediments* = 4 receivers, *Continuous Vegetation* =1 receivers, *Patchy Vegetation* =2 receivers). Furthermore, average habitat use was determined by detections of individual eels on a given benthoscape class. While the habitat use was averaged for each individual, these values were used to generate the population level habitat use (Rudolfsen et al., 2021). Therefore, discrepancies in a greater habitat use compared to habitat available may be explained by the detections of one individual on a given benthoscape class, while many individuals may have been detected across a range of benthoscape classes (Rudolfsen et al., 2021). Furthermore, habitat available is limited to receivers placed upon a given benthoscape, individuals may be detected on one-to-many receivers contributing to this discrepancy and at least two eels were detected in more than one season. These discrepancies were also reflected in the HSI values across seasons.

A benthoscape class might have a had a higher average proportion of habitat use relative to habitat available yet still have a low HSI value as observed across all four seasons (Fig 3.7; Rudolfsen et al., 2021). Alternatively, a benthoscape class could have exhibited a lower average proportion of habitat use, but a higher HSI value such as in summer for *Patchy Vegetation* (Fig 3.7; Rudolfsen et al., 2021). These discrepancies can be explained by the averaging of individual HSI to produce a population-level HSI. Although the habitat use to habitat available ratio summarizes the data used to develop HSI values for each individual, it may not reflect the population's averaged HSI score (Rudolfsen et al., 2021). For example, an individual eel may have had on average a greater proportion of detections (habitat use) on a given benthoscape class compared to the proportion of habitat available by other acoustic receivers. This would generate a higher HSI value elsewhere and a lower HSI value for that benthoscape class on average. However, while habitat associations and HSI values may provide valuable information for monitoring and management purposes, results of this study may not provide a true representation of how eels might interact with or be present on a given benthoscape class. For example, organisms may use many benthoscape classes, or they may select a benthoscape class based on a combination of complex variables (depth, temperature, salinity, and oxygen) associated with habitat that were not incorporated in this study. Future studies should consider incorporating these environmental variables into their habitat suitability modelling approach (Freitas et al., 2016; Rudolfsen et al., 2021).

### 3.4.9 Feeding and habitat

Eels, being opportunistic omnivores, demonstrate an enormous range in diet given their adaptability in nearly all water bodies (Tesch, 2003). Previous studies have described variations in movement associated with individual behavior as potentially correlated with morphological characteristics, such as broad or narrow head width, or foraging mechanisms, such as broad or localized movements where larger eels feeding on faster moving prey may travel farther (Barry et al., 2016; Geffroy et al., 2015). Broad-headed individuals are suggested to be nocturnal and to be more piscivorous compared to narrow-headed individuals which are suggested to be crepuscular and to feed more on benthic organisms (Barry et al., 2016;Kloppmann et al., 2003). However, no analysis could be completed on morphological characteristics in this study as these characteristics were not gathered for the first 18 eels tagged, as tagging began prior to my involvement.

All tagged eels that were categorized as sedentary were captured in the Inner Pond and the eels that remained in the Inner Pond for a longer period were smaller (639.7 mm, 418.3 g) than those that left sooner (717.8 mm, 539 g). Based on the results of this study, sedentary eels may demonstrate localized foraging behavior in both the day and the night; however, this knowledge is constrained by receiver placement and presence only detections. Moreover, eels categorized as sedentary were those that were detected at one receiver and therefore it is unknown where these eels went when detections ceased after 57 days.

Sedentary eels, in the Inner Pond, were associated with vegetated habitats which have been linked to increased food availability as these habitats are comprised of many gastropods, crustaceans, and various fish species (Murphy et al., 2021; Olson et al., 2019). Nearshore vegetated habitats, such as *Patchy* or *Continuous Vegetation* found in this study, may provide shelter for eels, especially during daylight hours (COSEWIC, 2012). Therefore, movements of tagged eels that occurred across all diel periods may be due to the increased protection from predators provided by *Patchy* or *Continuous Vegetation* (COSEWIC, 2012). Meanwhile vagrant eels were found on a diverse range of habitats found at deeper depths (2 to 25 m).

Vagrant eels, particularly those that exhibited multi-directional movement patterns and were detected equally in the day and the night, may therefore demonstrate broadscale foraging patterns. These eels moved between the VPS and the Gate and in one case the VPS, the Gate, and the Strait, and therefore occupied deeper depths and were found on harder substrates. Vagrant, uni-directional eels showed linear movement patterns from release to the Strait and therefore it is unclear the degree to which these eels used their associated habitats.

Future studies on the relationships between morphological characteristics and habitat use of eels in the BdOL may contribute to our understanding of individual variability in movement patterns of eels. Understanding the foraging mechanisms of eels in relation to prey items may contribute to our understanding of eel movements. For example, eels that are more localized may feed on worms by sucking while larger eels may consume larger prey by crushing, tearing, and spinning, yet spinning has an associated cost of being highly visible to predators and is energetically costly except for those that may have large enough jaws to crush prey (Helfman, 1995). Future work may therefore consider observing the relationship between habitat, prey, and foraging behavior in relation to nearshore and large-scale movements as done in this study.

Conducting beach seines or having eel stomachs, heads, and or gonads donated either through FSC or other fishing means are one way in which this information regarding prey available or prey consumed by eel may be gathered. However, consultation with Mi'kmaw is essential in discarding remaining contents of eels in an appropriate manner that aligns with Mi'kmaw values so that no part of the eel is wasted for science.

### 3.4.10 Management implications

The management of the American eel is complex given that the fishery is divided by life stage (elvers and adult eel). Furthermore, eels are a long-lived, benthic dwelling and endangered species that it is culturally significant to Indigenous people (COSEWIC, 2012). Management decisions regarding fish and fish habitat decisions require consultation with Indigenous peoples and are aimed at building a nation-to-nation relationship between Indigenous Peoples and the government of Canada (Government of Canada, 2021b). However, federal governing bodies must make space for consultations with Indigenous peoples to identify concerns and knowledge gaps at regional scales. Indigenous people hold valuable knowledge such as the general status (increase or decrease) of local eel abundances, natural threats to eels, or anthropogenic developments derived from personal observation and relationships with eels (R. Bradford, personal communication, 2022).

Management of American eels can be improved by addressing knowledge gaps for this species, such as which habitat types are used by eels and in what seasons. Understanding habitat used by eels compared to the habitat available can allow decision makers to identify important rearing habitat for eels required to improve protections of eel habitat, identify areas that may be under development or may pose risk to eels, or identify areas where risk and fishing areas overlap (R. Bradford, personal communication, 2022). Furthermore, results in this study may contribute to further assessment for eels in tidal waters, which currently hold a 2-day closure time to allow for any changes (such as the opening or closing) to the current fishing season and no licensing requirements for non-Indigenous fishers targeting eel (Fisheries and Oceans Canada, 2022).

Given the complexities of the American eel life cycle, its cultural significance to Mi'kmaw as well as its importance for food and sustenance, it is crucial that management decisions are made based on information gathered on a more local scale. Furthermore, substrate occupancy is a major component of the eel life cycle, therefore conservation measures aimed at assessing risk to eel and eel habitat should aim to identify threats to benthic habitats in these environments.

# CHAPTER 4: CONCLUSION

Obtaining baseline information of habitat is critical to understanding local yellow eel rearing habitat, and in the case of unique rearing habitats like the BdOL to contribute knowledge to gain a collective understanding of the flexibility of this species. However, this information can be difficult to collect in deeper regions (> 30 m), is time consuming, and can be expensive. In Chapter 2, I identified the benthic habitats found to occur in the BdOL from several previously collected and newly collected seafloor imagery data sets. Using these seafloor images and previously collected MBES backscatter and bathymetry data, I was able to classify deeper regions (> 30 m) of the BdOL. Then, using Sentinel-2A satellite imagery I was able to classify shallow water habitats (< 3m) of East Bay to fill data gaps where MBES data did not extend. Through combining both MBES and satellite imagery together, I generated a seamless benthoscape map of the BdOL estuary, which to date did not previously exist for the BdOL.

The final benthoscape map demonstrated that the BdOL is predominately composed of *Shallow Silt/Mud (≤ 50 m)* and within East Bay, the sheltered shore consists of *Patchy* and *Continuous Vegetation*. The benthoscape map produced in this study may now serve as baseline information for other researchers in the BdOL to examine species-habitat relationships such as those conducted by Landovskis (2021) and Kromberg (2022). Additionally, the benthoscape map can be used to monitor changes and identify potential risks to habitat such as in coastal lagoons, or in deeper regions, and may be used to better understand efforts made to enhance habitat for a given species such as lobster (Reynolds, 2021).

In Chapter 3, I used the benthoscape map generated in Chapter 2 and overlaid detections of acoustically tagged American eels to better understand the movements and habitat use of eels in the BdOL. The BdOL, given its unique physical characteristics, diversity of habitats, and cultural significance to surrounding Mi'kmaq communities, provided an ideal study site as previous literature by Giles et al (2016) identified knowledge gaps important to local Mi'kmaq communities to preserve eels and eel habitat in historical and culturally significant fishing areas.

Telemetry data in this study revealed that eels did not use primary fishing areas for winter dormancy, yet they did use these areas for summer foraging. Results using telemetry in this study overlapped with and were supported by Mi'kmaq fishing areas outlined by Giles et al., (2016) (Appendix K). Furthermore, eels spent most of their time in East Bay followed by the coastal lagoon and moved primarily at night, yet individual variability was found in the time of day when eels were active. Eels were not found to use all habitat available to them or demonstrate seasonal preferences for a given benthoscape type, however a seasonal shift from *Patchy* to *Continuous Vegetation* was observed in fall. Meanwhile one eel was found to remain in East Bay in winter at 25 m deep on *Shallow Silt/Mud (≤ 50 m)*. Combining mapping, shared knowledge, and acoustic telemetry together has provided an opportunity to strengthen our collective understanding of eels in this region and identify and fill shared knowledge gaps important to local communities.

Indicators on the status of American eel in Canada have lacked long-term data necessary to assess the status of this species at regional and local scales. While indices of abundance in the St. Lawrence River and Lake Ontario have declined more than 90%

since the 1970’s, long-term data on species abundance is not available for other Maritime regions, including Scotia/Fundy, Newfoundland and Labrador, and the BdOL (COSEWIC, 2006). Yet, because this species is panmictic, meaning that all spawners form a single breeding unit, it is suggested that the recruitment of eels to Canadian waters would be affected by the status of the species in other parts of Canada and the United States (COSEWIC, 2006). However, reports of elver abundance, (also not long term), do not show evidence of decline of eel.

Abundances of eels suggest declines may have ceased in some areas, however, the abundance of eels remains drastically lower than former levels and positive trends are too short-term to provide strong evidence that the abundance of eels is increasing in Ontario (COSEWIC, 2006). These data emphasize the need for more longer-time series data on eels at other regional and local scales. Declines to eel abundance have reported habitat alteration, dams, fishery harvest at multiple life stages, high economic value of elvers, oscillations in ocean conditions, and contaminants, as contributors to decline of eels and which may impede the recovery of the species (COSEWIC, 2006). Moreover, despite being named a freshwater eel, this species is found to use all waterbodies accessible to them.

Within the federal management framework for the American eel, stock assessments and habitat risk assessments have been the guiding principle in advice for exploited species, yet these assessments are focused on and often derived from freshwater systems, as these areas are considered to have the greatest potential harm from human activities (R. Bradford, personal communication, 2022). Therefore this species may receive protection against harmful alteration or destruction of habitat through the

Canadian *Fisheries Ac*t, Canadian *Environmental Protection Act*, or provincial acts such as Ontario *Water Resources Act*, Quebec *Environmental Water Quality Act*, and the New Brunswick *Clean Water Act* (COSEWIC, 2012). Furthermore, habitat that lies within National Parks, Provincial parks, or wildlife and marine protected areas may also be subject to additional protection through the Canada *National Parks Act*, the Loi sur les Parcs in Québec, the *Ontario Provincial Parks and Conservation Reserves Act*, 2006 and the Canada *Wildlife Act* to name a few (COSEWIC, 2012).

More information is needed to understand eel distribution and habitat use in estuaries especially during their growth stage, as eels may mature faster and at much younger ages in estuaries compared to freshwater environments (Cairns et al., 2009; Jessop et al., 2008; Morrison & Secor, 2003; Oliveira, 1999). Secondly, eels may be estuarine or even salt marsh resident species and do not require movement to freshwater environments to complete their life cycle (Daverat et al., 2006; Eberhardt et al., 2015). Therefore, knowledge of eel distribution and habitat use paired with local population assessments is needed in estuaries to contribute to stewardship of eels in these environments and at more local and regional scales. Third, there is a need to better integrate Indigenous knowledge into the assessment and monitoring process as well as the need to consult with Indigenous Peoples in order to develop co-management recommendations for the recovery of species given these processes may infringe on Indigenous peoples aboriginal and or Treaty rights.

Canada has a legal obligation to consult with Indigenous Peoples and recently proposed amendments to the Fisheries act state that traditional knowledge must inform fish habitat decisions and that these decisions must consider the adverse effects on the rights of Indigenous Peoples (Fisheries and Oceans Canada, 2019b; Supreme Court of Canada, 2004a, 2004b, 2005). Future research may consider using MEK workshops to record information in a way that expands our understanding of this data poor species, such as in Eckert et al (2017). Eckert et al (2017) gathered information from government as well as knowledge from MEK and TEK holders via interviews to observe changes to the body sizes (total lengths) and abundance of rockfish on the Central Coast of BC, Canada over the last 60 years, and examine the factors that may be driving these changes. The results of Eckert et al (2017) demonstrated a repeatable method for both using and incorporating traditional and local knowledge to build baseline information for data-poor species and ultimately the value of incorporating Indigenous knowledge for the purpose of fisheries research and management.

Through Aponmatulti'k, and this thesis, I demonstrate the importance of estuarine habitat to yellow stage American eels by using shared knowledge to explore the regional home range, seasonal habitat use and associations with specific habitat types in the BdOL. Knowledge shared by project partners, makes space for diverse knowledge systems and values and can contribute to extending the assessment process for eels in tidal waters and can contribute to help incorporate regional diversity into management and monitoring plans.

This thesis aimed to gain knowledge on the distributions and habitat use of American eels, a data-poor species, in the BdOL through a Two-Eyed Seeing framework. As a result of challenges introduced by the COVID-19 pandemic, this thesis was not able to implement a Two-Eyed Seeing approach to its full potential.  For example, MEK and community input that such an approach was hoped to achieve, were largely lacking, and analyses were largely conducted using western academic methods (e.g., habitat mapping and final telemetry analysis). Due to the pandemic, MEK workshops or interviews with Mi'kmaw elders were not able to occur, which were essential to gather knowledge of local and traditional fishers to identify knowledge gaps that were of most interest to the community and to ensure research goals aligned with the needs of that community. Furthermore, not being able to have frequent face-to face meetings was challenging to build relationships and to gather knowledge, as knowledge is shared through stories, land-based learning, or through cultural or ceremonial practices.

Despite challenges introduced by the pandemic, the Aponmatulti'k study did find meaningful ways to cope with challenges of the pandemic by conducting online monthly meetings to offer a space for students to share their work and discuss with Mi'kmaw partners. This created a safe space for individuals to share their knowledge, ask questions, and ensure research interests were aligned. Ultimately these meetings were very effective and successful due to relationships and trust that had been built by project partners.

Apoqnmatulti'k was built on the concept of Two-Eyed Seeing to encourage the exchange of knowledge sharing and develop a cross-cultural understanding among individuals in order to develop a deeper understanding of our ecologically, commercially, and culturally significant marine resources and to support the establishment of a fisheries co-management framework. Apoqnmatulti'k was comprised of western academics, federal government bodies, local knowledge holders, and M'ikmaq knowledge holders, yet the project was heavily dominated by non-Mi'kmaq members and completed within a western academic setting. Furthermore, the form in which this thesis is shared is written according to western science standards and therefore may not necessarily be easily shared with the local community. However, information gained through this thesis might benefit from the creation of an ArcGIS story map which combines stories, text, interactive maps, and other multimedia content to users in an open-source, friendly, and accessible way, which could be easily shared with Mi'kmaw project partners (ESRI, 2022c).

Although this thesis was not able to fully incorporate a Two-Eyed Seeing approach, Apoqnmatulti'k did demonstrate how this framework may be implemented by other researchers to identify knowledge gaps faced by local communities. For example, Apoqnmatulti'k used a community-based study design that involved communication and consultation with Mik'maw partners on decisions regarding site selection, the placement of acoustic receivers, hiring Mi'kmaw community liaisons, and methods for animal capture and tagging. Throughout this project, research questions and objectives of the research were continuously re-evaluated and driven by questions initially addressed by the community to ensure research goals were aligned. Through Apoqnmatulti'k, several lessons have been learned that I believe are key components to those hoping to incorporate a Two-Eyed Seeing approach.

Apoqnmatulti'k challenged the means in which individuals from western science knowledge systems are taught to conduct research and emphasized the importance of building meaningful and trusting relationships, reconciliation, developing cross-cultural understanding, and respect for diverse knowledges and values, which are critical steps towards co-governance and co-management of our marine resources. Several excellent lessons learned throughout Apoqnmatulti'k have been outlined already by Landovskis (2021). Briefly, Landovskis (2021) stated lessons such as: 1) individuals must be willing to genuinely listen to, respect, and value one another, 2) time must be set aside to build and nurture relationships so that individuals feel valued and safe enough to share their perspectives, and 3) individuals must be willing to work through discomfort that comes from reducing and acknowledging their own knowledge system and biases. Through these lessons learned, a desire for reconciliation, and a commitment to overcoming challenges, I believe others hoping to incorporate a Two-Eyed Seeing framework may find their own versions of success.

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## APPENDIX A: LIFE STAGES OF AMERICAN EEL

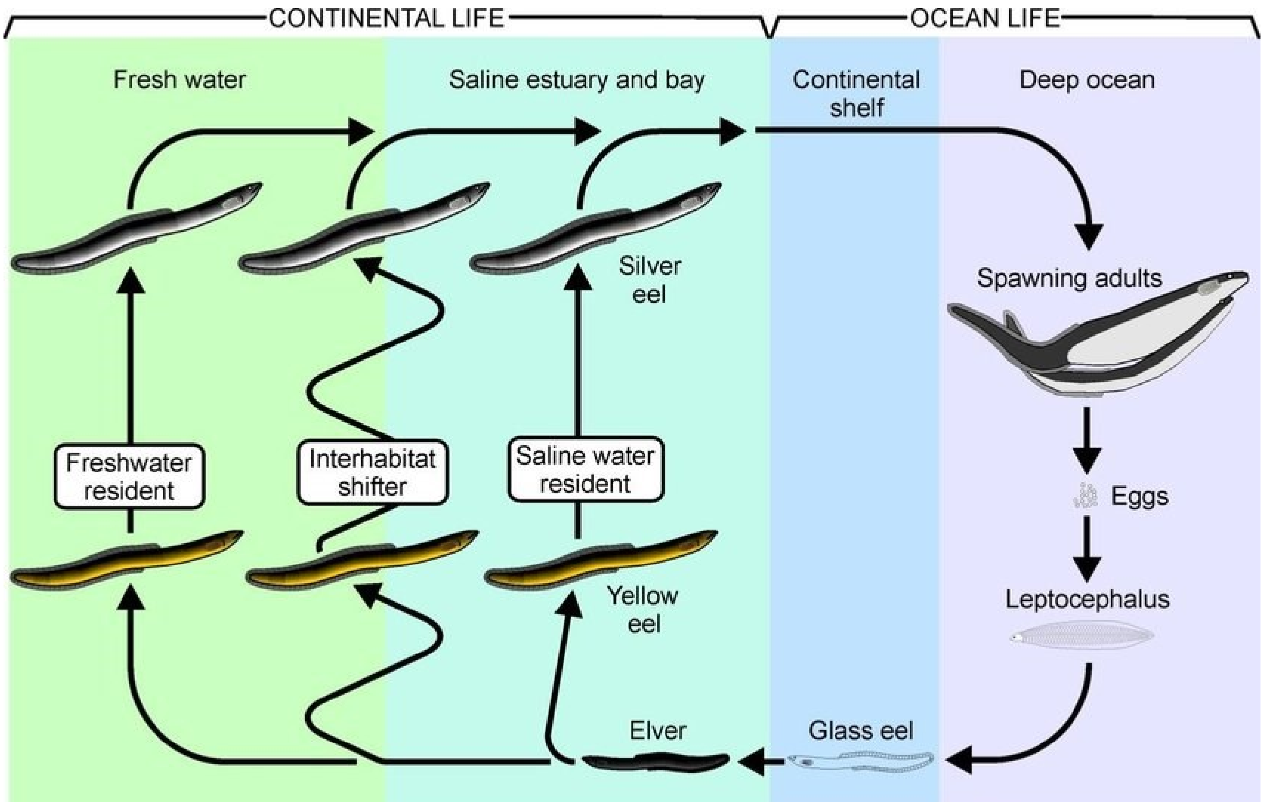
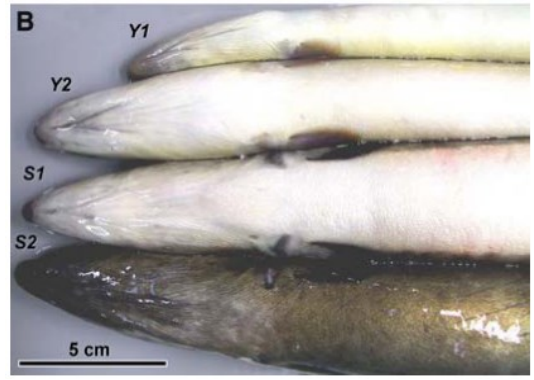
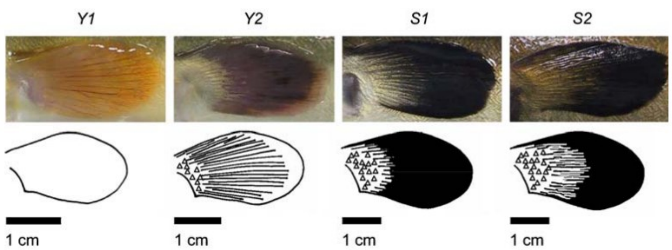



**Figure A.1.** American eel life cycle (CSAS, 2013). American eels begin their life cycle as eggs and hatch into the larval leptocephali. As Leptocephali they drift by ocean currents throughout the western Atlantic, Gulf of Mexico, and Caribbean Sea to the continental shelf where they metamorphose into transparent glass eels (Atlantic States Marine Fisheries Commission, 2000; Pratt et al., 2014). Glass eels migrate inshore toward freshwater rivers or estuaries to feed and become progressively more pigmented (brown in color) as they mature into elvers (6.5-10 cm in length). After a few months elvers develop into yellow eels and obtain a dark back and a yellow to white underside (ASMF, 2000; Chaput et al., 2014). This stage is the longest phase in the eel's life cycle (4- 25 years) and is understood as the growth stage where sexual determination occurs (ASMF, 2000; Chaput et al., 2014) The silver phase is the final metamorphosis event when eels become sexually mature. During the silver phase their eyes become larger and their stomach degenerates to prepare for the long ocean migration towards the Sargasso Sea to spawn (Chaput et al., 2014).

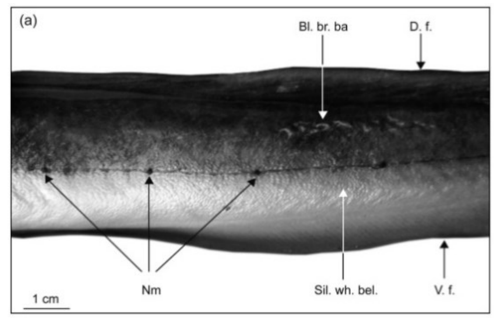

A.2a.




A.2b.




A.2c.




A.2d.

**Figure B.2:** Morphometrics to assess maturity (life stage/silvering process) of eel: a) demonstration of body color and enlargement of eyes from yellow to silver stages (Miller & Casselman, 2020); b) a dark dorsal coloration and white ventral coloration (Okamura et al., 2007), c) degree of melanized pectoral fins (Okamura et al., 2007), and d) the presence of a visible lateral line (Acou et al., 2005), will be used to aid in determining silvering criteria (Durif, et al., 2005).

# APPENDIX B: FISHERY DISTRICTS

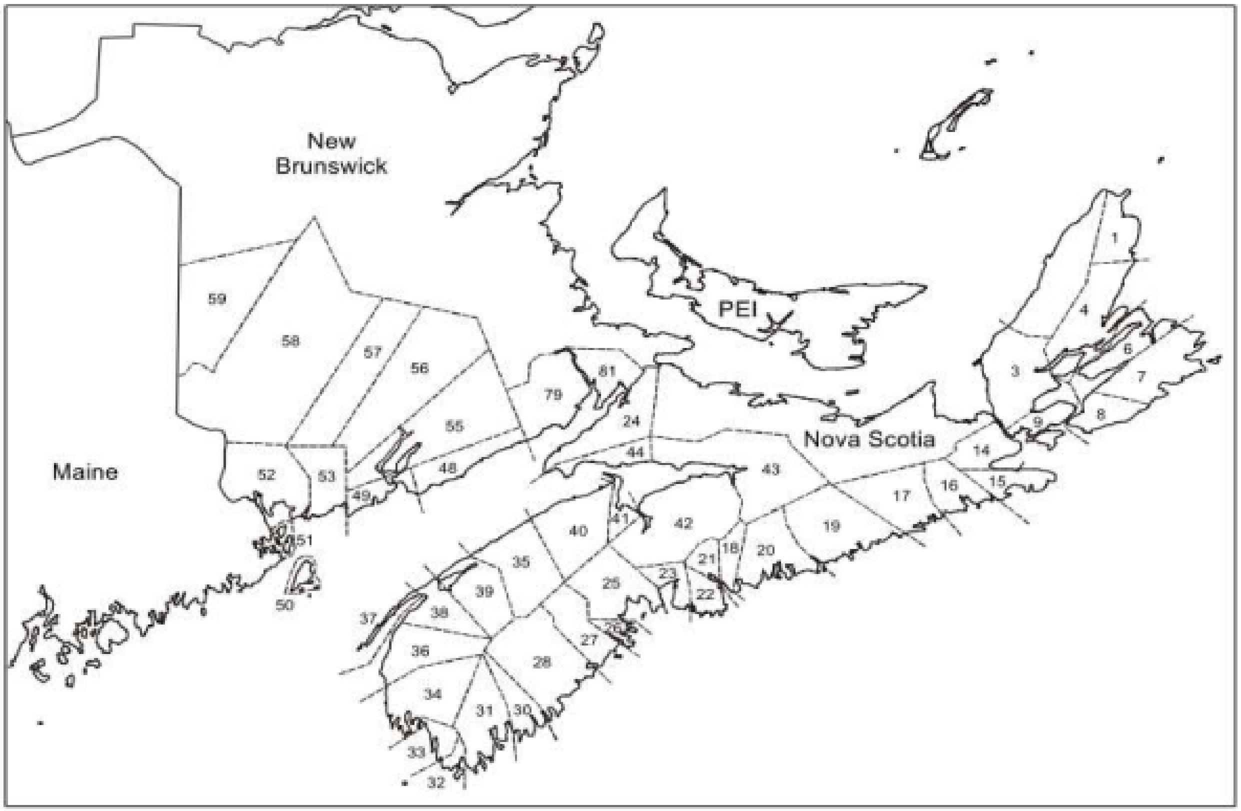



**Figure B.** Map of the Maritime Provinces showing the distribution of Fishery Districts (map proived by Bradford et al., 2013).

# APPENDIX C: STATUS OF FISH SPECIES IN THE BRAS D'OR LAKE

**Table C.** Fish species documented to occur in the Bras d'Or Lake that have declined, increased, or have received a COSEWIC listing.

| Fishery Type | Group | Common Name | Scientific name | COSEWIC listing | Year | Vagrant Migratory Resident | Status (2002) | Status (2007) | Status (TEK) | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| Commercial | Ground fish | Atlantic cod | *Gadus morhua* | Special concern | 2003 | R, M | Common | Declined | Declined | Black, 1976, MacDonald, 1967, Lambert, 2002, Parker, et al., 2007 |
| Commercial | Pelagic | Spiny dogfish | *Squalus acanthius* | Special concern | 2010 | M | Rare | Not found | Declined | Black, 1976, Lambert, 2002, Parker, et al., 2007 |
| Commercial | bottom dwelling | Thorny skate | *Raja radiata* | Special concern | 2012 | - | Low | | | Lambert (2002) |
| Recreational, FSC, moderate livelihood | Anadromous/ pelagic | Atlantic salmon | *Salamo salar* | Endangered | 2015 | M | Rare | Declined | Declined | Black, 1976, Lambert, 2002, Parker, et al., 2007 |
| Commercial | Ground fish | White hake | *Urophycis tenuis* | Endangered | 2013 | - | Medium | | | Black, 1976, MacDonald, 1967, Lambert, 2002, Parker, et al., 2007 |
| Commercial | Ground fish | Winter skate | *Raja ocellata* | Endangered | 2015 | - | Medium | | | Lambert (2002) |
| Commercial, recreational, FSC, moderate livelihood | Catadromous/pelagic/bottom dwelling | American eel | *Anguilla rostrata* | Threatened | 2012 | M | Low | Not found | Declined | Black, 1976, MacDonald, 1967, Lambert, 2002, Parker, et al., 2007 |
| Recreational | Anadromous | Brook trout | *Salvelinus fontinalis* | NA | NA | M | Rare | Declined | Declined | Parker, et al., 2007; Black (1976) |
| Commercial | Anadromous | Gasperau/ Alewife | *Alosa pseudoharengus* | NA | NA | M | Common | Declined | | Parker, et al., 2007; Black (1976); Lambert, 2002 |
| Recreational | Pelagic | Blueback herring | *Alosa aestivalies* | NA | NA | M | Low | Declined | | Parker, et al., 2007; Denny (2001); |
| Commercial Aquaculture, Recreational | Pelagic | Rainbow trout | *Salmo gairdneri* | NA | NA | | Low | Declined | Declined | Parker, et al., 2007; Black (1976); Lambert, 2002 |
| Commercial | Ground fish | haddock | *Melanogrammus aegelinus* | NA | NA | | Rare | Not found | Declined | Parker, et al., 2007; Black (1976); |
| No fishery in Canadian waters | Pelagic | Ocean pout | *Macrozoarces americanus* | NA | NA | | Rare | Not found | Declined | Black (1976); Lambert, 2002 |
| Commercial, recreational, | Pelagic | Pollock | *Pollachius virens* | NA | NA | | Rare | Not found | Declined | Parker, et al., 2007; Black (1976); Lambert, 2002 |

## APPENDIX D: PREVIOUSLY COLLECTED ACOUSTIC SONAR DATA OF NEARSHORE HABITATS

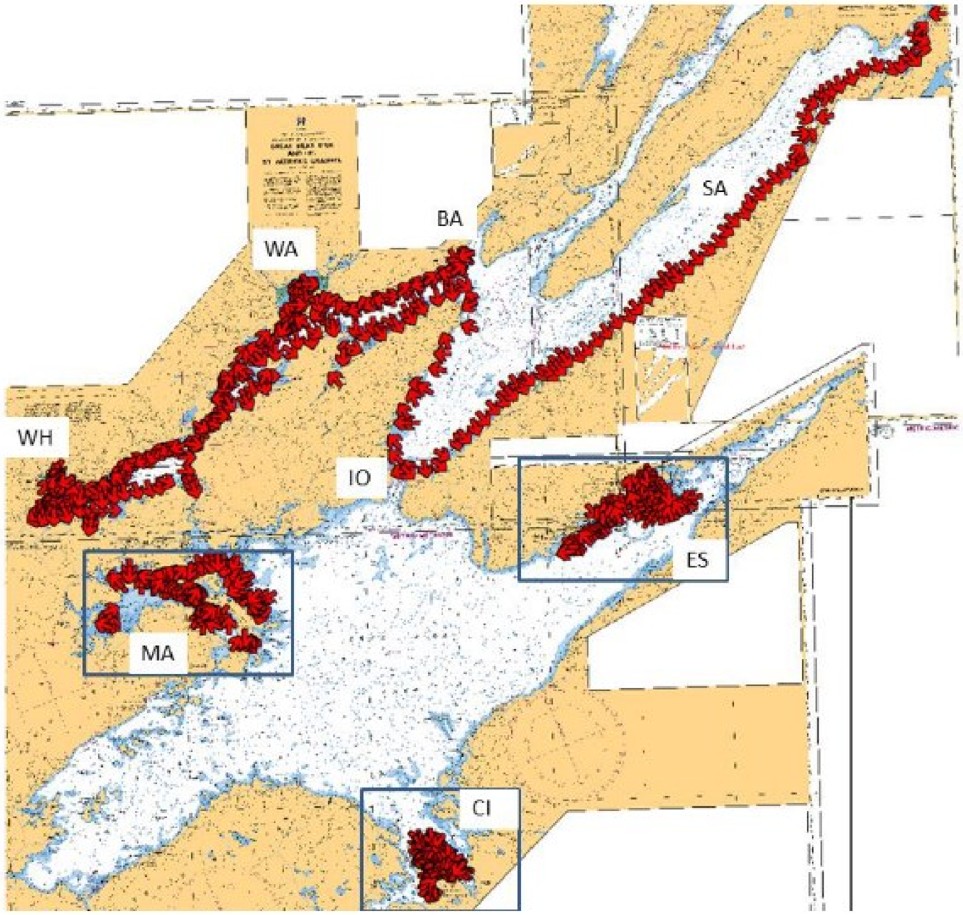



Figure 1: Bras d'Or Lake survey areas. **BA** – Baddeck, **CI** – Chapel Island, **ES** – Eskasoni, **IO** – Iona, **MA** – Malagawatch (including River Denys Basin), **WA** – Wagmatcook, **WH** – Whycocomagh, **SA** – St. Andrew's Channel. The red arrows indicate video clips from the towfish transects.

**Figure D.1.** This figure represents five sites surveyed by Vandermuelen et al (2016) for the purpose of classifying nearshore habitats in the Bras d'Or Lake. Each site was surveyed with a towfish equipped with an underwater video camera and transects running perpendicular to shore.

# APPENDIX E: RECLASSIFIED PIXELS

**Table E.1.** Summary of the number of reclassified pixels and the associated area in km$^2$ in the final benthoscape map. The *Shallow Silt/Mud (≤ 50 m)* class in the MBES dataset was split into two classes at a 50 m depth gradient while areas ≥ 6 m and ≤ 50 m of both *Continuous* and *Patchy Vegetation* were reclassified as *Shallow Silt/Mud (≤ 50 m)* in East Bay.

| | **Reclassified # pixels** | **Reclassified area (km$^2$)** |
|---|---|---|
| Shallow Silt/Mud (≤ 50 m) | 0 | 0.00 |
| Deep Silt/Mud (≥ 50 m) | 1,391,225 | 139.1 |
| Continuous Vegetation | 68,577 | 4.77 |
| Patchy Vegetation | 98,713 | 6.84 |
| **Total** | **1,558,515** | **150.71** |

# APPENDIX F: MORTALITIES

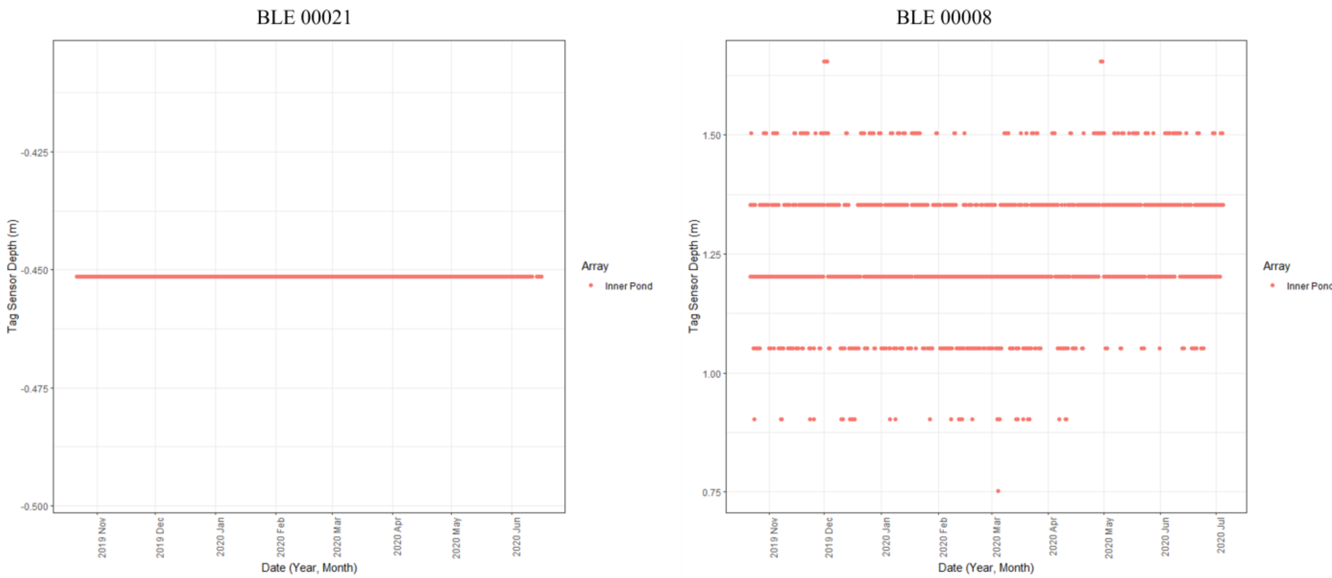



**Figure G.** Detection data for two individual eels that were considered to have died after tag implantation in the Bras d'Or Lake. This plot shows the tag snesor depth (in meters) and the date by year and month colored by array. Both indiviudals BLE 00008 and BLE were captured and detected in the Inner Pond (1.2 m depth). Negative depths (BLE 00021) indicate sensor error as opposed to an eel being burrowed.

# APPENDIX G: MUD TAGS

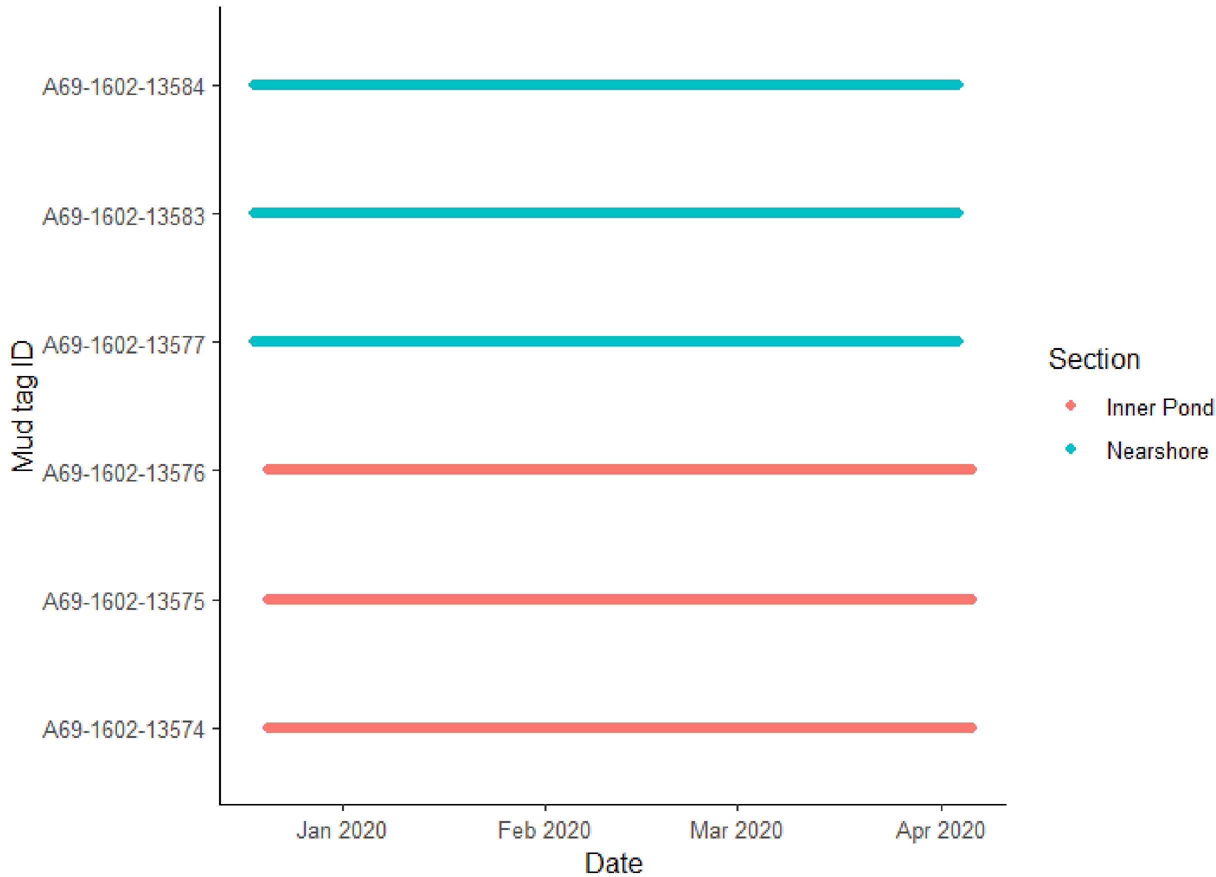



**Figure F.** Mud tags (V9 acoustic transmitters (V9P-2x-069k-1, n = 6) that were buried 30-45 cm beneath the substrate in December 2019 at two locations: nearshore (n=3) section in red and Inner Pond (n=3) in blue were detected throughout the entire deployment period. Deployed mud tags emitted a uniquely codded signal at nearshore until April 03, 2020, and until April 05, 2020, at the Inner Pond.

# APPENDIX H: UNIQUE BEHAVIOUR OF MULTIDRIECTIONAL EEL

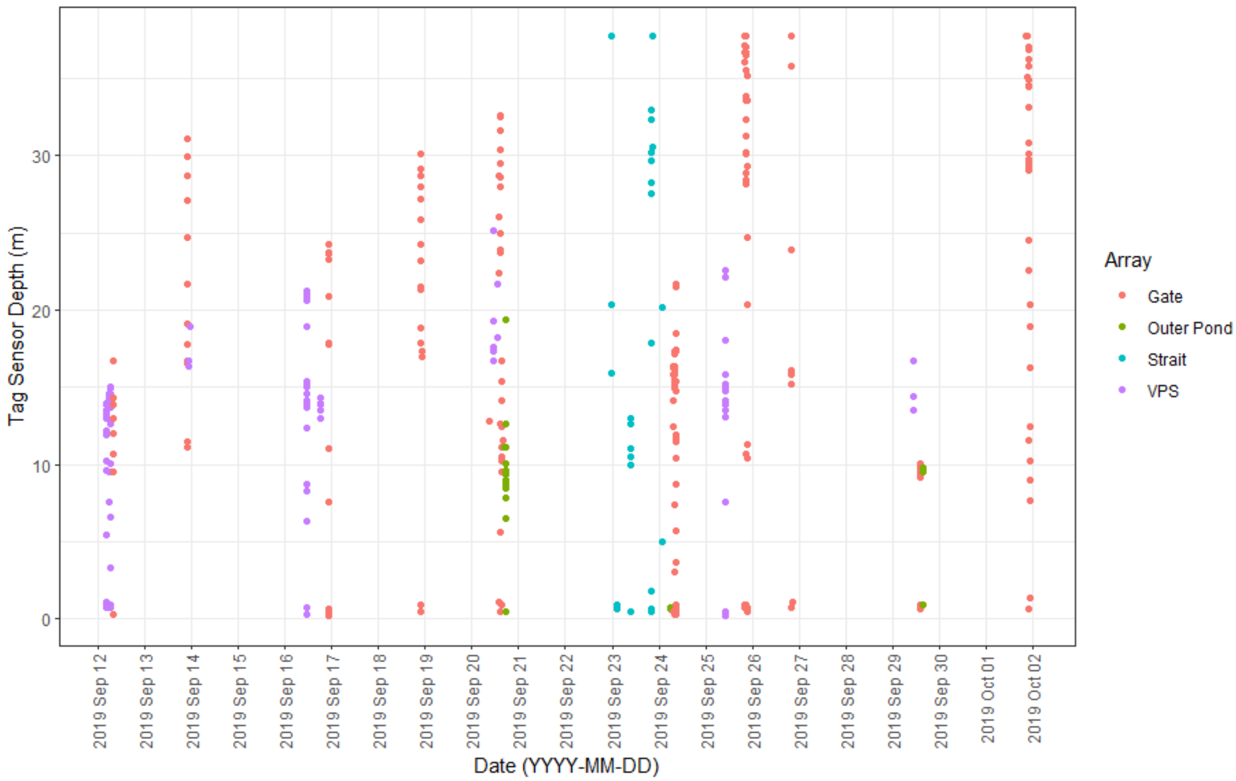



**Figure I.1.** Detection data for BLE 00010 in the Bras d'Or Lake. This plot shows the tag sensor depth (in meters) and the date colored by array.

## APPENDIX I: WINTERING EEL

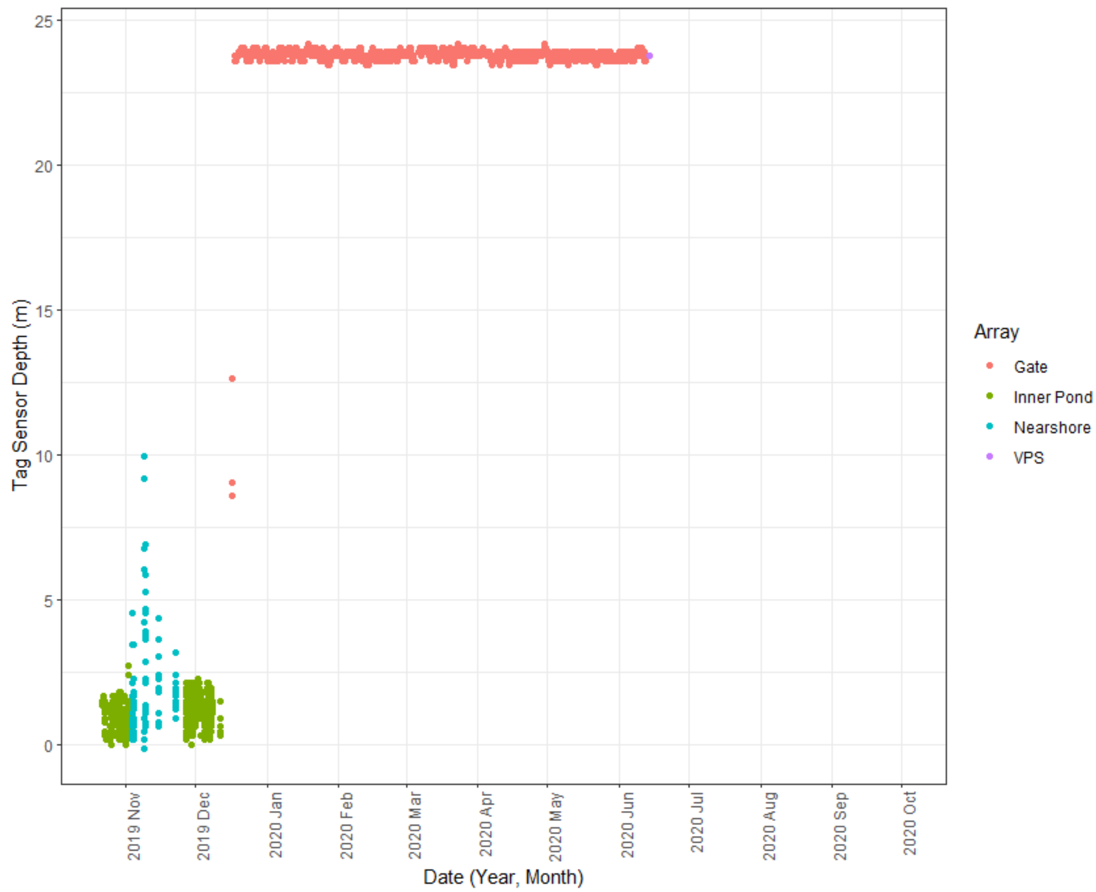



**Figure I.1.** Detection data for the over wintering eel BLE 00017 in the Bras d'Or Lake. This plot shows the tag snesor depth (in meters) and the date by year and month colored by array.

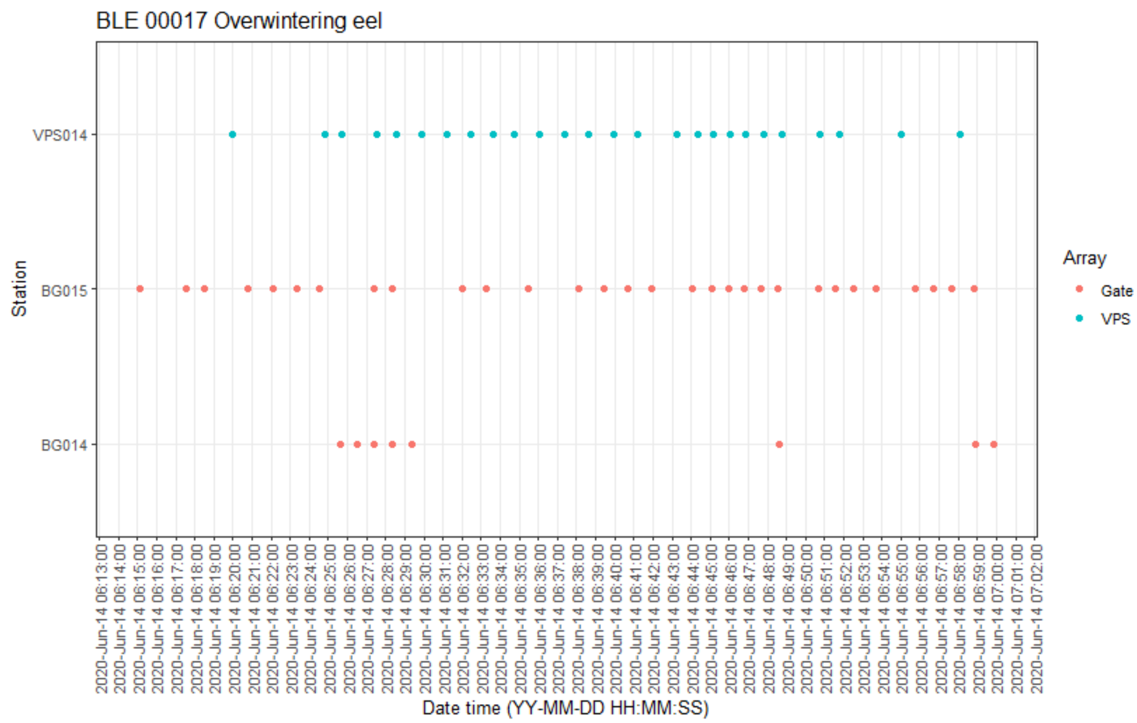



**Figure I.2.** Detection data for BLE 00017 in the Bras d'Or Lake presumed to have overwintered. This plot shows the tag sensor depth (in meters) for June 14, 2020 set at 1 hour intervals and colored by array indicates time of emergence from winter dormancy in the Gate on June 14 at 06:18:30 UTC and detected in the VPS at 06:20:00 UTC.

# APPENDIX J: EEL MOVEMENT, TEMPERATURE, & SALINITY

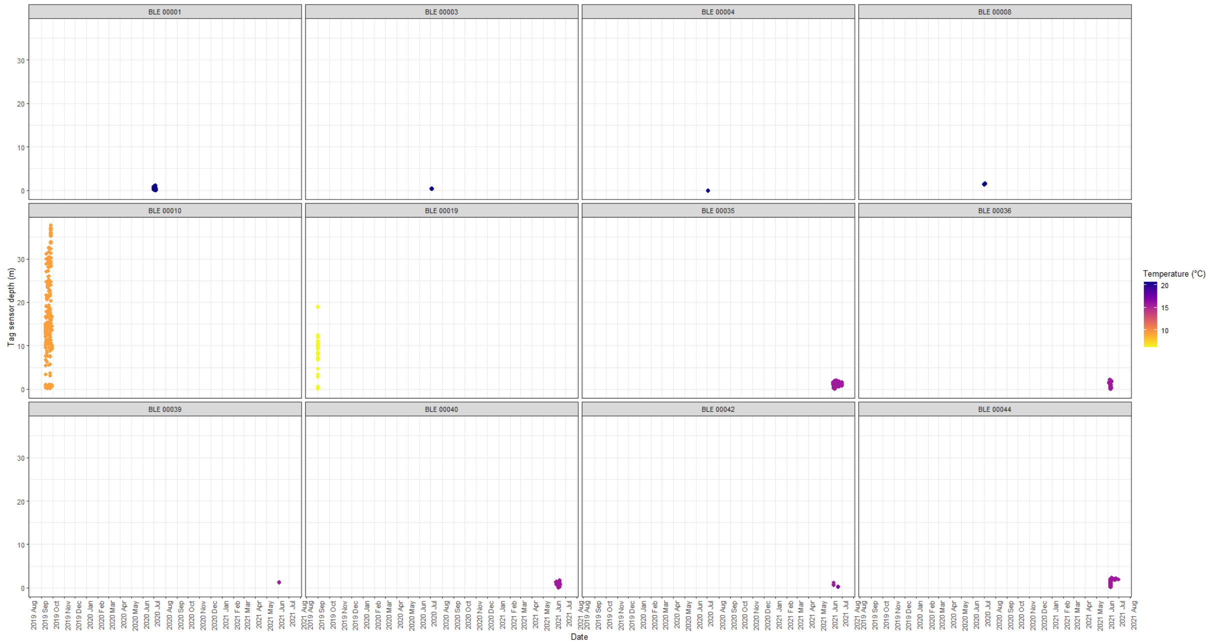



**Figure J.1.** Detections of individual American eels that corresponded with the deployed DST-CTD data loggers deployed from July 23, 2019 to September 07 2019 and July 15 2020 to July 07 2022. This plot demonstrates the average monthly bottom temperature and eels tag sensor depth in meters.

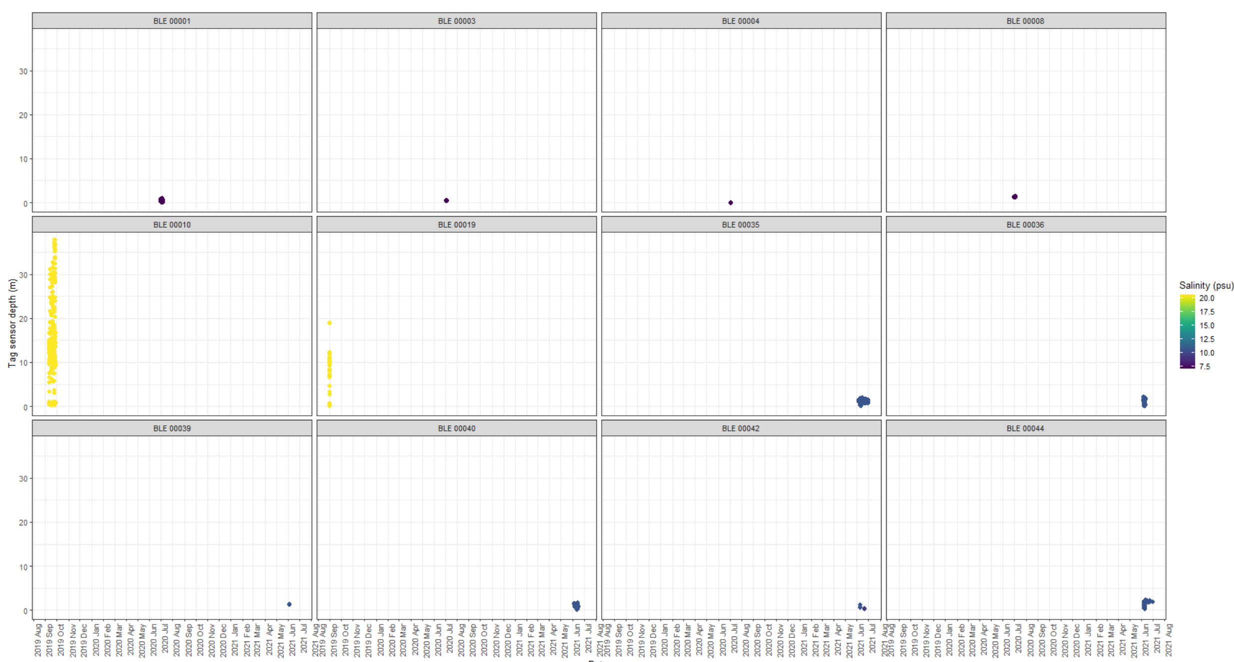



**Figure J.2.** Detections of individual American eels that corresponded with the deployed DST-CTD data loggers deployed from July 23, 2019 to September 07 2019 and July 15 2020 to July 07 2022. This plot demonstrates the average monthly bottom salinity and eels tag sensor depth in meters.

# APPENDIX K: EEL DISTRIBUTION IN THE BRAS D'OR LAKE USING MIKMAW ECOLOGICAL KNOWLEDGE

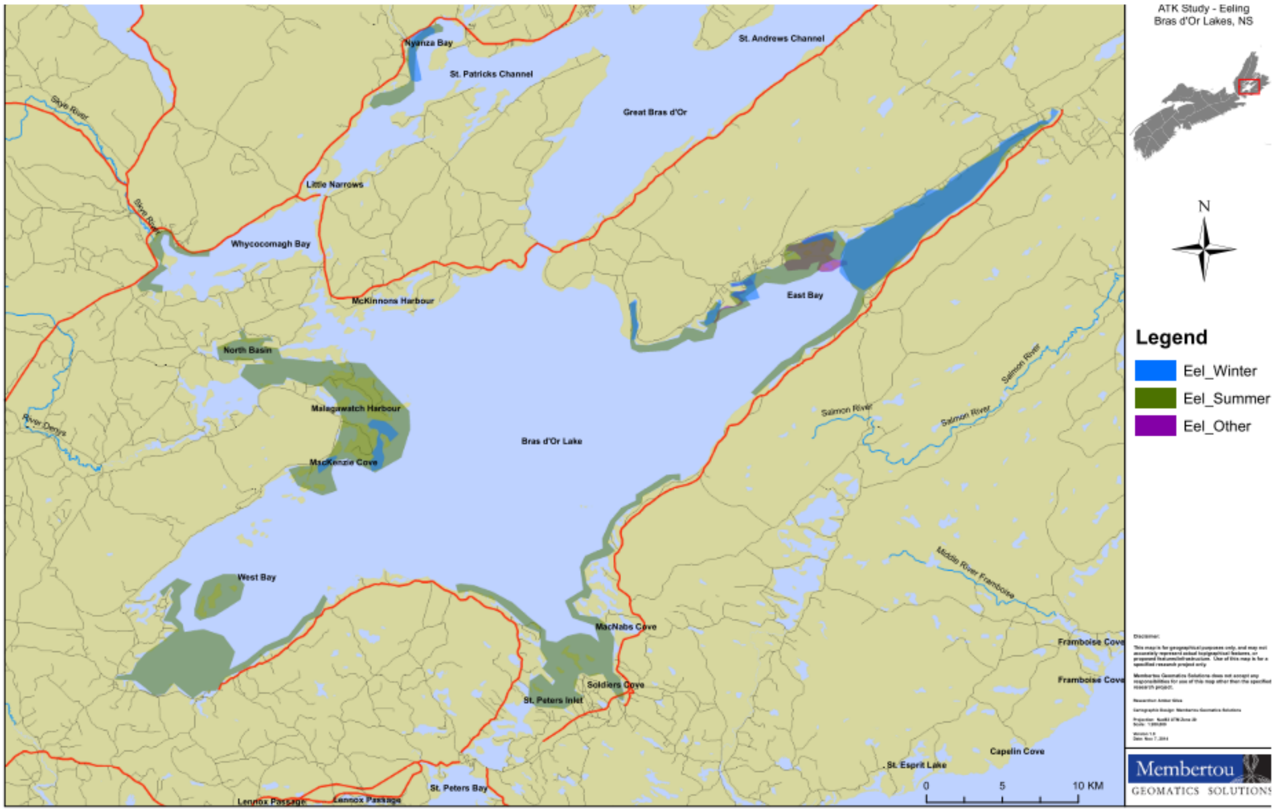



**Figure K.** Map of the Bras d'Or Lake with areas identified as summer, winter, and first-time eeling locations "Eel_Other" by Eskasoni First Nation community members provided by Giles et al., 2014.