Assessing the Sampling Design of the Community Aquatic Monitoring Program (CAMP)

by

Jessica Kidd

A thesis
presented to the University of Waterloo
in fulfillment of the
thesis requirement for the degree of
Master of Science
in
Biology

Waterloo, Ontario, Canada, 2017

AUTHOR'S DECLARATION

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

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

Abstract

The Community Aquatic Monitoring Program (CAMP) is a community based monitoring program that involves local stakeholders to monitor estuaries and bays in the southern Gulf of St. Lawrence (sGSL). Implemented in 2003, CAMP continues to be administered by Fisheries and Oceans Canada (DFO) in collaboration with the Southern Gulf of St. Lawrence Coalition on Sustainability (Coalition-SGSL). Data are collected annually from up to 36 sites, and include counts of nearshore fish, shrimp, and crabs (i.e., nekton) along with measures of aquatic vegetation, water quality and sediment. The CAMP dataset has potential to inform decision-makers on the relationship between the health of an estuary and its nekton assemblage. However, concerns have been raised regarding the CAMP station selection method, as the majority of station locations was selected to provide easy road access for volunteers. Also, a standard number of six stations was established, regardless of estuary size, to allow for community groups to complete each sampling event within one day. The objective of this study was to assess the ability of CAMP to provide a measure of littoral nekton that represents the overall littoral nekton community of the estuary. The adequacy of the CAMP sampling design was tested by comparing it to a sampling program that applied a stratified random design. A subset of ten estuaries that are monitored by CAMP were selected. Twelve stations were sampled within each estuary with six stations located where CAMP samples, and another six stations randomly located and stratified among the upper, middle, and lower estuary. Differences between the nekton community data were assessed using a cluster analysis, non-metric Multidimensional Scaling (nMDS) ordination, permutational MANOVA (PERMANOVA) and a test of homogeneity of dispersions (PERMDISP). The adequacy of six sample stations was tested by comparing the number of species detected by both sampling designs, and then combining the datasets to predict how many stations would be required to detect all species. The combined dataset was analysed using a one-way PERMANOVA to determine if having nekton assemblage data from more stations would alter the conclusions regarding the differences between sites. The potential need to increase the number of stations was determined by assessing the precision of CAMP in estimating the abundance of influential species, as defined by the similarity percentages (SIMPER) routine. In general, significant differences in nekton assemblages were not detected between sampling designs. Six stations are sufficient to detect the moderately and highly abundant nekton species that contribute to the dissimilarities of estuaries. Increasing the number of CAMP stations would not alter the conclusion about the dissimilarity of sites based on the nekton community assemblages, or greatly increase the precision in estimating counts of influential

species. The results indicate the application of CAMP is not limited by station selection bias and would not benefit from increasing the number of stations. Furthermore, programs designed to accommodate volunteers can produce comparable data to scientific studies if designed appropriately. Future analysis of the entire CAMP dataset can be used to determine if there is a relationship between the degree and type of anthropogenic activities influencing an estuary and the littoral nekton assemblages within it.

Acknowledgements

I would like to thank my supervisors Simon Courtenay and Mark Servos for all of their support and guidance over the past two years. I consider myself incredibly fortunate to have had two supervisors who both truly care about the success of their students and make themselves so available. I would also like to thank my committee members Robert Bailey and Heidi Swanson. Robert helped me understand the wonderful world of statistics, and Heidi provided valuable feedback on my study design, and also advice on life decisions and career trajectories.

A massive thank you to those who helped me complete my fieldwork! Michael van den Heuvel's lab group from the University of Prince Edward Island, including Christina Pater, Kyle Knysh, and Carissa Grove, helped me collect all of my data from the Prince Edward Island sites. Monica Boudreau and Michelle Maillet from the Department of Fisheries and Oceans' Gulf Fisheries Centre made sampling in New Brunswick possible. The CAMP coordinators Jolène Mazerolle and Taylor Sheidow were extremely helpful in the field and in providing me with CAMP gear. I also thank Michael van den Heuvel and the Gulf Fisheries Centre for allowing me the use of their boats and field equipment. And of course, thank you to my supervisor Simon Courtenay who joined me in the field for the entire month of July. You hauled those seine nets like a boss Simon. Without all of your help I would not have been able to collect my field data and my thesis would have been quite boring. Also, I would not have learned about Mummichogs and Pipe Fish, which would have been tragic, because they are the coolest.

Leslie Bragg of Mark Servos' lab provided me with lots of help and support over my two years in the Biology department, especially when it came time to applying for an animal care application. I thank Antòin O'Sullivan for taking the time to teach me the ways of ArcGIS. My fellow lab-mate Nicole Stamnes made coming into the office on weekends and working full-days in front of the computer screen so much more enjoyable than if I had been there alone; thanks for listening to my rants Nicole! And finally, I thank my good friend Phil DeWitt who fielded numerous late night phone calls to discuss ideas and questions regarding statistics.

Dedication

I would like to dedicate this thesis to Karen Stroebel, Shona Derlukewich and Kent Kristensen. Karen took a chance on a girl from the suburbs of southern Ontario and hired me for field work in the mountains of Grande Cache, Alberta. Without this challenging and wonderful experience, I may have never realized my love for field work. Karen also inspired me to pursue experience in the field before continuing in academia, which is a decision I strongly attribute my success in graduate school to. Shona is both my fishing-buddy and mentor. Like Karen, Shona took a chance on me, and transferred me to the Aquatics team when I had no formal background or training in fish ecology. Kent is another person I am happy to know as both a friend and mentor. Kent's thorough knowledge of both the science and the policy has always inspired me to continue to learn and do better. Without the training and learning opportunities I received from Shona and Kent I may have never discovered my passion for aquatic sciences. All three of these people instilled in me the importance of collecting and documenting field data accurately. All three also embody what it means to lead by example, earn respect rather than demand it, and are incredibly kind to those who work with them. If I ever find myself in a management position, it is their example I will strive to follow.

Table of Contents

List of Figures

List of Tables

Chapter 1
Introduction

Estuaries are partially enclosed water bodies where fresh water from the land meets and mixes with salty water from the sea. The term estuary encompasses a variety of coastal environments, including fjords, sounds, drowned river valleys, lagoons, coastal inlets, embayments, deltas, and tidal creeks (Thrush et al. 2013; Hallett et al. 2016). The European Union *Water Framework Directive* (European Communities 2000) refers to estuarine environments as "transitional waters", as estuaries are essentially where freshwater environments transition to salty marine environments. Estuaries can be hard to define due to their transitional nature, but many writers define an estuary as being where a salinity gradient becomes apparent (Telesh and Khlebovich 2010; Thrush et al. 2013).

The salinity gradient within an estuary can be highly variable. Salinity is low (as low as 0.5 PPT) in the upper estuary, where fresh water from rivers and surface runoff dilute the salinity of marine water creating brackish water (Butler et al. 1996; Kennish 2002). Salinity concentrations increase (up to 30-40 PPT) towards the mouth of the estuary where tidal influence is the greatest (Butler et al. 1996; Kennish 2002). However, salinity concentrations are continuously changing within an estuary. In northern temperate estuaries, the balance of fresh and salt water is seasonally altered by large inputs of fresh water during the spring when the snow melts and precipitation rates are high, and then a decline of fresh water inputs during the dry summer months and frozen winter (Butler et al. 1996). On a daily and lunar basis, the salinity gradient fluctuates due to the influence of the tides (Butler et al. 1996).

Salinity gradients vary among estuaries as a result of differences in geomorphology, location, and tidal exposure (Thrush et al. 2013). Since fresh water is lighter than salt water, deep estuaries tend to have a salt wedge, where the fresh water from the river floats above the marine water (Butler et al. 1996). Hence, deep estuaries typically have a strong vertical salinity gradient that contributes to high habitat variability (Thrush et al. 2013). Conversely, shallow estuaries are typically well-mixed and have a weak vertical salinity gradient (Thrush et al. 2013; Staehr et al. 2017). In the northern hemisphere, the ocean currents and fresh waters flow in the opposite direction of those in the southern hemisphere due to the Coriolis effect. The clockwise or counter-clockwise flow of the outgoing fresh water and incoming marine water causes one side of the estuary to typically have higher salinity concentrations than the other (Butler et al. 1996). The shape of the estuary mouth and shorelines along with natural and engineered structures influence the speed and direction at which the tides enter the estuary and transport saline waters (Thrush et al. 2013). Tidal influences increase the variability of the salinity gradient among and within estuaries.

Tidal influence on the estuarine environment varies temporally. Since tides are based on the lunar day, which is 24 hours and 50 minutes long, each tidal cycle has a delay in relation to the diel cycle, causing each tidal cycle to be unique and variable amongst the months and years (Krumme 2009). Tidal amplitudes gradually change over the lunar month with a maximum tide height (spring tide) occurring approximately every 15 days when the sun, earth and moon are aligned during new and full moons, and minimum tide height (neap tide) when the moon is at a 90-degree angle to the earth (Wilcockson and Zhang 2008). Some estuaries experience a semi-diurnal tide where high and low tides occur twice daily at intervals of 12.4 hours (Wilcockson and Zhang 2008; Krumme 2009) whereas other estuaries experience diurnal tides with only one daily tidal cycle (Krumme 2009). Hence, like salinity, the variability of tides also varies among estuaries.

The continuing flux in environmental conditions caused by the salinity gradient and tidal cycle amplifies the variability of environmental conditions to which estuarine biota are adapted (Elliott and Quintino 2007; Wilcockson and Zhang 2008; Castellanos-Galindo and Krumme 2015; Porter and Scanes 2015). Due to the high variability of environmental conditions, community composition within an estuary tends to be highly variable as well (Elliott and McLusky 2002). The fish species richness of estuaries is influenced by freshwater species that are able to inhabit estuaries during periods of high freshwater input and marine species that utilize estuarine habitats when salinity increases during dry seasons (Castellanos-Galindo and Krumme 2015). The salinity gradient determines how far up the estuary marine organisms can travel and how far down freshwater aquatic species can go (Butler et al. 1996). The alternating exposure and inundation of intertidal habitats alters salinity, temperature, hydrostatic pressure, turbulence and food availability (Wilcockson and Zhang 2008). As a result, organisms that inhabit estuaries modulate their behaviour to the ebb and flow of the tides (Wilcockson and Zhang 2008). Tidal currents are a mode of transport for organisms that move in synchrony with the tidal cycle (Gibson 2003; Krumme 2009). These movements vary from a few millimeters to kilometers (Gibson 2003). Estuarine fauna also use tidal migrations for feeding, predator avoidance and reproduction (Gibson 2003). Planktonic organisms concentrated by the tides in fronts or eddies create areas within the estuary that are popular feeding habitat for planktivorous fish and their predators, including piscivorous fish, birds, and mammals (Thrush et al. 2013). Thus, while estuaries are a dynamic and complex environment, various flora and fauna species have adapted to thrive in these transition zones (Whitfield 1999; Porter and Scanes 2015).

1.1 Importance of estuaries

Numerous species of birds, mammals and fish depend on estuaries to carry out their life histories (Davidson et al. 1991). While the fluctuation in salinity is a source of stress that certain animals must adapt to (Castellanos-Galindo and Krumme 2015), for others it prepares them for migration from a fresh

to salt water environment, or vice versa (Castellanos-Galindo and Krumme 2015). Diadromous species require both fresh water and marine environments to carry out their life cycle. For example, Atlantic Salmon (*Salmo salar*) travel as juveniles from upstream natal habitats to estuaries where they undergo physiological changes that prepare them for adulthood in the ocean, and then later return to undergo the reverse transformation as they migrate to their riverine spawning habitat (Levings 2016). Other species remain in the estuary to spawn, where abundant food sources and sheltered waters provide optimal spawning and nursery habitat for a variety of fish, crustaceans, and wildlife species (Whitfield 197; USEPA 2012). For this reason, estuaries are aptly named "nurseries of the sea" (Jaureguizar et al. 2004; Hanson 2009; USEPA 2012; Strydom 2015; Costalago et al. 2015). The spawning, nursery, and rearing habitats that facilitate the growth and development of fish species sought after by the lucrative commercial and recreational fisheries contribute to the economic importance of estuaries (Thrush et al. 2013; Goncalves et al. 2015).

Estuaries are economically important as they harbour an array of highly valued flora and fauna. In developing countries, estuarine fisheries often constitute the main source of both food and income for people living along the coast (Blaber et al. 2000). Estuaries also provide ideal conditions for raising shellfish species, which make estuaries a prime area for aquaculture activities (Thrush et al. 2013). In 2010, the total value of Atlantic Canada's aquaculture industry was 486 million dollars (Gardner Pinfold 2013). Estuarine biota are also harvested by the pharmaceutical industry for agar, kelp powder, chitin, fish oil, calcium powder, and mussel extract (Thrush et al. 2013). Estuarine vegetation is harvested for use as fertilizer, fish food, and grazing material for livestock (Thrush et al. 2013). In a review of the global value of ecosystem services, estuaries were found to have the highest value per hectare of any ecosystem (Costanza et al. 1997).

Flora and fauna of estuaries are also highly valued for their as water purifiers as they filter, bind, sequester and bury nutrients, pollutants, and suspended sediments (Barbier et al. 2011; Thrush et al. 2013). Estuarine vegetation, including seagrasses, macroalgae and mangroves, sequester so much carbon that vegetated coastal habitats are believed to sequester up to 50% of the total carbon stored in marine sediments (Duarte et al. 2013). Bacteria within the sediments can detoxify heavy metals, and some shellfish species can sequester heavy metals, which limits toxicity to other organisms (Thrush et al. 2013). Sewage wastes are broken down as food resources through microbial, plant, and animal activities (Thrush et al. 2013). These recycled nutrients fuel primary production (Thrush et al. 2013). Estuaries are believed to be amongst the most productive environments on earth (Tecchio et al. 2015). The high productivity of estuaries makes them attractive to a large number of fish, shore and sea birds, and marine mammals (Thrush et al. 2013).

The myriad ecosystem services provided by estuaries attract human development (Barbier et al. 2011; Dafforn et al. 2012; Sheaves et al. 2012; Temmerman et al. 2013; Goncalves et al. 2015). Throughout human history people have settled near estuaries (Lotze et al. 2006), which is why in many regions estuaries are culturally significant (Butler et al. 1996; Thrush et al. 2013). The establishment of large population centres near estuaries is facilitated by the supply of fresh water for drinking and industrial processes, along with the removal of wastes and access to the sea for transportation and shipping. The economic potential and sheltered access to the ocean continue to attract human development to estuaries. Estuaries protect upland developments from storm and flood damage through the absorption of flood water, and the dissipation of storm surges by estuarine soils and plants (Temmerman et al. 2013). Costanza et al. (2008) estimate coastal wetlands provide $23.2 billion per year in storm protection services. Additionally, estuarine plants prevent erosion and stabilize shorelines (Barbier et al. 2011). Other than just functionality, the aesthetically pleasing landscapes of estuaries have a positive impact on property prices and land value (Thrush et al. 2013). The draw of estuaries for human development is confirmed with twenty-two of the thirty-two largest cities in the world being located on estuaries (NOAA 2015).

1.2 Risks to estuaries

The popularity of estuaries as locations for anthropogenic development and activities has subjected estuarine habitats to high levels of stress (Dafforn et al. 2012; Sheaves et al. 2012). These stressors are increasing in severity due to rapid population growth and development, with approximately 4 billion people currently living within 60 km of the world's coastlines (Kennish 2002). Estuaries are changing in response to these stressors and consequently, the structures of biotic communities within estuaries are threatened (Kennish 2002). Estuaries are currently believed to be one of the most altered and at-risk aquatic environments (Blaber et al. 2000).

Inputs of pollutants are among the most serious stressors currently affecting estuarine environments (Kennish 2002). Industrial and domestic wastewaters release hydrocarbons and heavy metals to the estuarine environment that harm biota and reduce populations of sensitive species (Kennish 2002). Domestic wastewaters release pharmaceuticals that accumulate in estuarine sediment (Liang et al. 2013) and have been found to cause hormone disruption in mollusks and commercially important fish species (Oberdorster and Cheek 2001). Industrial and domestic wastewaters can also cause bacteriological and chemical contamination of organisms that are harvested by aquaculture, which can lead to closures of areas for harvesting due to human health concerns (Butler et al. 1996). Commonly used pesticides transported by agricultural runoff have been observed to cause fish kills and mortality of shellfish (Fulton et al. 1999). Nutrients are transported to estuaries through a variety of vectors, including agricultural

runoff, land clearing activities, the release of human and animal wastes, urban runoff, and atmospheric deposition (Bowen and Valiela 2001; Cloern 2001; Bricker et al. 2008).

Nutrients are naturally occurring and not inherently toxic, but in high concentrations can have significant adverse effects on the estuarine environment (Cloern 2001). Excessive nutrient loading exceeds the natural limiting rate for aquatic plant growth, which stimulates the growth of phytoplankton and macroalgae that causes an imbalance between plant production and consumption (Bowen and Valiela 2001; Cloern 2001; Proffitt 2017). This imbalance creates eutrophic conditions where large amounts of expired plant material accumulate in the benthic habitat as organic matter, which stimulates microbial decomposition and consequently depletes bottom waters of oxygen (Kemp et al. 2005). Eutrophication can lead to severely depleted dissolved oxygen concentrations that threaten the health of fish and other aquatic life (Rothenberger et al. 2014). The increase in biomass of phytoplankton and macrophytes decreases water transparency and limits the light energy reaching benthic habitats (Cloern 2001), which can lead to decreases in the cover of light dependent plant species such as eelgrass (*Zostera marina*) (Short and Burdick 1996; Bowen and Valiela 2001). The decline in eelgrass, which is important for both shell- and fin-fish, alters the rest of the food web (Bowen and Valiela 2001). Depending on the phytoplankton species that become prolific, harmful algal blooms can form that cause toxic conditions for estuarine organisms and humans (Hoagland et al. 2002). Instances of harmful algal blooms have been increasing in recent decades and are predicted to continue to worsen as coastal human populations grow (Bricker et al. 2008). Overall, eutrophic conditions can cause fish kills, warnings against consumption of shellfish, and a reduction in recreation and tourism expenditures (Hoagland et al. 2002; Bricker et al. 2008). The susceptibly to the negative effects of nutrient loading varies among estuaries, as it is controlled by the rate of horizontal transport within the estuary that is regulated by the tides, wind, bathymetry, geography and river flows (Cloern 2001; Bricker et al. 2008).

River flows are naturally variable, but are also heavily manipulated by human activities (Cloern 2001). Rising human populations increase the demand for fresh water, and corresponding water withdrawals from the surrounding watershed may alter estuarine salinity, sediment regimes, and nutrient inputs (Cloern 2001; Kennish 2002; Gorecki and Davis 2013). Changes in salinity during periods of low freshwater inflow can change the abundance of economically important fish species (Tsou and Matheson 2002). The negative effects of manipulated flows are further exacerbated during drought conditions, which are expected to become more frequent in some regions as the climate changes (Kennish 2002).

The projected effects of climate change threaten to intensify the damage to estuaries from anthropogenic influences. Predicted increases in the frequency of drought conditions and water temperatures will likely increase the frequency and intensity of toxic algal blooms (Lehman et al. 2017). Lehman et al. (2017) studied the drought conditions experienced by the San Francisco Estuary in 2014 to

test the potential effects of severe drought on harmful algal blooms. Water temperature was found to be one of the primary factors influencing the severity of the blooms (Lehman et al. 2017). Increases in temperature will also extend the growing season and promote earlier spring algal blooms (Staehr et al. 2017). Forecasted sea level rise from thermal expansion threatens to permanently inundate and remove estuarine habitats. For example, salt marshes in southern New England are vulnerable to sea level rises, as marsh submergence is occurring and vegetation communities are experiencing shifts in species composition due to elevated water levels (Watson et al. 2017). Coastal developments exacerbate the effects of climate change by preventing the landward migration of estuary vegetation (Watson et al. 2017).

Coastal developments alter the estuarine environment and remove important habitats (Duarte 2002). Large constructed embankments are known to change tidal circulation and subsequently, the salinity gradient within the estuary (Xu and You 2017). Estuary habitat has been destroyed by the physical alteration of flood plains and river bank stabilization (Raposa et al. 2003; Harrison and Whitfield 2006a). Estuarine habitat is drained or filled to make way for agricultural, industrial or residential land (Adam 2002; Duarte 2002). Land reclamation, and the construction of ports and marinas involve dredging and landfilling construction activities that physically alter and remove habitat (Adam 2002; Duarte 2002). Approximately one-third of intertidal estuarine habitat in Great Britain has been lost since the Roman occupation (Davidson et al. 1991). The alteration and removal of habitat can lead to reductions in estuarine biodiversity (Kennish 2002).

Estuarine biodiversity is also threatened by intensive fisheries activities (Whitfield 1997; Kennish 2002). Many estuarine fisheries are believed to be over-exploited due to the growth in number of fishers and the development of more efficient gear (Blaber et al. 2000). Overfishing decreases the abundance of sought-after species, which reduces the overall diversity of the fish community and causes trophic shifts (Blaber et al. 2000). Fisheries activities threaten the nursery function of estuaries when juvenile species are captured as by-catch in the pursuit of target organisms (Blaber et al. 2000). While some fisheries deplete the native fish populations, others introduce non-native species that can reduce species diversity, shift trophic organization, and alter habitats (Kennish 2002).

Establishment of non-native species threaten the native estuarine flora and fauna through predation and competition for resources (Williams and Grosholz 2008). Vectors for non-native species invasions include ship transportation, aquaculture activities, and shipment of live seafood or bait (Cohen and Carlton 1998). San Francisco Bay is believed to be one of the most invaded estuaries in the world, with 234 exotic species of plants and animals identified in the estuary (Cohen and Carlton 1998). A review by Williams and Grosholz (2008) found that the costs associated with loss of native species and

structural damage to shipping infrastructure caused by non-native species can reach 250 million dollars US annually.

As human populations expand and further develop along estuaries, the current threats to the estuarine environments are predicted to worsen if continued unabated. Thus, monitoring and assessment of these ecologically and economically important ecosystems are urgent and crucial (Rothenberger et al. 2014; Chariton et al. 2015). Well-designed estuary monitoring programs are used to guide and measure the success of management efforts, and contribute to ongoing adaptive management (OEH 2013; Porter and Scanes 2015).

1.3 Monitoring the health of estuaries

A current focus of estuary monitoring and assessment research is the development of indices of ecosystem health (Ellis et al. 2015). The term health has been used synonymously with status, integrity, and quality (Hallett et al. 2016). Essentially, ecosystem health reflects the degree to which an ecosystem has been altered from its pristine state (Hallett et al. 2016). A review of ecosystem health by Tett et al. (2013) concluded that good ecosystem health can be defined as "the condition of a system that is self-maintaining, vigorous, resilient to externally imposed pressures, and able to sustain services to humans. It contains healthy organisms and populations, and adequate functional diversity and functional response diversity. All expected trophic levels are present and well interconnected, and there is good spatial connectivity amongst subsystems." Although some scientists do not support the use of the term health to define ecosystem condition, it is known to be a useful term in communicating the condition of an ecosystem to the public and decision-makers (Deeley and Paling 1999).

The assessment of estuary health is complicated due to the high spatial and temporal variability of the estuarine environment. Accordingly, the selection of appropriate indicators is critical to the assessment of estuary health (Ysebaert and Herman 2002; Elliott and Quintino 2007; Porter and Scanes 2015). Ideal indicators help discern trends and effects resulting from anthropogenic activities influencing the estuarine ecosystem (Deeley and Paling 1999). A thorough knowledge of the natural state of the selected indicators is required to identify when a change from the natural state has occurred (Deeley and Paling 1999). Hence, long-term monitoring programs are ideal, because they provide critical historical data that can define the typical natural variability of indicators and detect meaningful patterns of change (Deeley and Paling 1999; Tsou and Matheson 2002; Gorecki and Davis 2013).

Traditionally, water quality indicators are monitored to establish the health of an estuary (Oberdorff and Hughes 1992; Scanes et al. 2007). Water temperature and salinity are principal parameters to monitor (Scanes et al. 2007), because they are thought to be the primary factors naturally affecting the distribution and occurrence of estuarine fishes (Harrison and Whitfield 2006b). Nutrient concentrations

are measured to identify stressors present in an estuary, monitor specific discharges, and better understand nutrient dynamics (OEH 2013). Measures of chlorophyll *a* and turbidity provide indications of short-term responses to a range of pressures (Scanes et al. 2007). Yet, water quality parameters alone are ineffective indicators of estuary health (Whitfield and Elliott 2002; Scanes et al. 2007). An essential component of health assessments is measuring biological integrity, which typically focus on the analyses of plankton, benthic invertebrates, macroalgae, and fish (Borja and Dauer 2008).

Currently, the majority of estuary health assessments incorporate biological indicators (Dafforn et al. 2012; Porter and Scanes 2015). Biota are advantageous indicators of estuary health, because they integrate a range of environmental effects, including water quality degradation and habitat loss (Harrison and Whitfield 2006a). However, utilizing biological indicators of ecosystem health can be complicated for estuaries. Organisms that inhabit estuaries have strategies for coping with a variable environment (Elliott and Quintino 2007; Porter and Scanes 2015). The adaptability of estuarine organisms to fluctuating environmental conditions can make it difficult to detect impacts of anthropogenic stress (Elliott and Quintino 2007; Valesini 2017). Ideal biological indicators of estuary health are sensitive to degraded conditions, and provide clear evidence of anthropogenic impacts that are distinct from the influences of natural environmental variability (Ellis and Bell 2013).

Nekton are actively swimming aquatic animals that include fish and crustaceans, which are used as biological indicators for estuary health (Raposa et al. 2003; Staehr et al. 2017). Nekton are attractive indicators, because public familiarity with nekton species simplifies the communication of environmental degradation to the non-scientific community (Whitfield and Elliott 2002; Harrison and Whitfield 2004; Ellis and Bell 2013). The public also tends to value these species more than the less-charismatic species, such as benthic invertebrates or zooplankton that they are less familiar with (Whitfield and Elliott 2002; Harrison and Whitfield 2004; Ellis and Bell 2013). Nekton are also attractive indicators of estuary health, because they are relatively easy to identify, large, taxonomically well-understood, respond to multiple levels of stress (e.g. individual, population, community), and have members in multiple trophic levels (Whitfield and Elliott 2002; Harrison and Whitfield 2004). Nekton communities respond to habitat alterations through changes in competitive interactions (e.g., invasive species expansion, decline in rare species), changes in production (e.g., reduced breeding, abundance, diversity), and changes in predator-prey interactions (e.g., trophic shifts) (Whitfield and Elliott 2002; Raposa et al. 2003). Numerous studies have detected changes in nekton assemblages as a response to anthropogenic influences (Raposa et al. 2003; Aguilar et al. 2004). The multitude of direct and indirect impacts of anthropogenic activities on nekton communities reinforces the selection of nekton as a biological indicator for estuaries (Whitfield and Elliott 2002).

There has been a shift from simple composite measures of communities, such as diversity indices, towards complex multivariate measures (Sheaves and Johnston 2012). Multivariate models developed to focus on community composition incorporate the number and type of taxa that structure the community at the site along with their relative abundance or biomass (Ellis et al. 2015). Assessments at the community level are advantageous, because they assess the response of several species that have a range of sensitivities to human influences, capture changes in species among different estuarine uses and habitats, identify species that are most sensitive to environmental change, and provide a more comprehensive representation than can be achieved by an assessment only focusing on a single species (Attrill and Depledge 1997; Valesini et al. 2017). While absolute values are not sufficient as a measure of health, the change observed in a community over time is an indicator of stress (Deeley and Paling 1999). These multivariate techniques preserve the information on abundance of each species, which provide a more sensitive and ecologically meaningful response to environmental change than traditional univariate diversity indices (Ellis et al. 2015).

It has been recommended that several approaches should be incorporated into the assessment of estuary health, including assessment of fish at the community-level and the response of individual indicator species (Aguilar et al. 2004; Valesini et al. 2017). Relying on just community-level assessment may inhibit the detection of a clear response, due to the high variability of schooling species, influence of non-resident marine and freshwater species, and the highly resilient resident species (Valesini et al. 2017). Some studies have found that the assessment of indicator species can provide a more sensitive signal of environmental degradation than the analysis of the relative abundance of each fish species within the community (Valesini et al. 2017). Indicator species have been recommended to be resident estuarine species that are long-lived, large and abundant and can provide a clear indication of environmental degradation through changes in their growth and body condition (Valesini et al. 2017). However, a study by Finley et al. (2013) found the abundance of Mummichog (*Fundulus heteroclitus)* was more informative of environmental degradation than body condition. The population density of Mummichog combined with lower species richness has been found to be a good indicator of estuary eutrophication (Finley 2008). A study on the effects of Havana Harbour linked changes in fish community composition to the proximity to Havana Harbour, with reduced populations of Bluehead Wrasses (*Thalassoma bifasciatum*) and increases in abundance of Slippery Dick (*Halichoeres bivittatus*) (Aguilar et al. 2004).

Overall, multiple indicators are generally necessary to accurately quantify estuary health (Scanes et al. 2007; Porter and Scanes 2015; Staehr et al. 2017). Due to the number of indicators required to adequately monitor estuaries, the study of estuarine health is known to be difficult and costly (Scanes et al. 2007; Porter and Scanes 2015). Time and cost constraints can lead programs to select only abiotic

indicators (Chariton et al. 2015), and many estuary monitoring programs are at risk of being cancelled due to budget constraints (Mahoney and Bishop 2017).

1.4 Community based monitoring programs

One way managers can both reduce costs associated with monitoring programs and engage local community members is to design a monitoring program that can be executed by local volunteers (Forrester et al. 2015). Community based monitoring programs are a type of citizen science where local community members volunteer their time to assist in the collection of environmental monitoring data (Fernandez-Gimenez et al. 2008). These programs are becoming increasingly popular among government and non-profit agencies internationally (Conrad and Hilchey 2011). A review by Theobald et al. (2015) estimated up to 2.28 million people annually volunteer in 388 community based monitoring programs worldwide.

Not all community based monitoring programs are designed for the same purpose. Some programs are established primarily as stewardship initiatives to engage local communities and promote environmental awareness and learning, where long-term data are not maintained or used for official purposes (Fernandez-Gimenez et al. 2008; Kanu et al. 2016). Alternatively, other programs are meant to better inform decision-makers, are designed to use specific methods, and can involve collaborations with government agencies and academic institutions (Kanu et al. 2016). These community based monitoring programs are increasingly looked to as the solution to the lack of data stemming from reduced government funding for monitoring programs, limited resources available to academia, and a hesitation of knowledge sharing by private industry (Kanu et al. 2016). Although data collected by citizen scientists have a great potential to inform environmental research, these data are not commonly being incorporated into the scientific literature and may be a missed opportunity for science and society (Theobald et al. 2015).

Professional scientists and decision-makers have expressed concerns regarding the quality of data collected by community members (Forrester et al. 2015; Kanu et al. 2016; Savage et al. 2017). Community based monitoring programs are challenged when financial resources are limited, sampling protocols are inadequate or lacking, and access to scientific expertise is limited (Sharpe and Conrad 2006). Volunteers must be provided with valid protocols, appropriate equipment, and adequate training if the program is to collect scientifically-defensible data (Sharpe and Conrad 2006). Indeed, community members are capable of collecting data that are comparable to those collected by professional scientists when they receive adequate training (Fore et al. 2001). Previous studies have assessed the accuracy of data collected by community members through the comparison of data collected by professional

scientists, and found no significant differences (e.g., Fore et al. 2001; Thériault et al. 2008; Danielsen et al. 2014; van der Velde et al. 2017).

Despite quality concerns, community based monitoring programs are gaining international recognition for their potential to fill data gaps, inform decision-makers, and educate communities (Kanu et al. 2016). Globally, organizations exist that were created to develop and support community based monitoring programs, and ensure policy makers support citizen science initiatives, including the European Citizen Science Association and the Citizen Science Network Australia (Kanu et al. 2016). Community based monitoring programs have the capacity to run long-term (>10 years) (Sharpe and Conrad 2006; Ryan et al. 2017), and have been found to run on average seven years longer than the average scientific monitoring program (Theobald et al. 2015). Therefore, community based monitoring programs may be a good option for estuary monitoring, because they have the propensity to run long-term, collect scientifically-defensible data, and are a cost-effective option to collect the various indicators required for health assessments. An example of such an estuary monitoring program is the Community Aquatic Monitoring Program (CAMP).

1.5 Community Aquatic Monitoring Program (CAMP)

In 2003, Fisheries and Oceans Canada (DFO) set out to create a practical estuary monitoring program that would involve local stakeholders in the southern Gulf of St. Lawrence (sGSL) (Weldon et al. 2007). This was in response to the Canada's Oceans Strategy document (DFO 2002), which called upon DFO to collaborate with local stakeholders to create stewardship activities and promote the protection of marine and coastal environments (Weldon et al. 2007; DFO 2011). The result was the creation of CAMP, which was implemented in 2003 and has continued to be administered by Fisheries and Oceans Canada (DFO) in collaboration with the Southern Gulf of St. Lawrence Coalition on Sustainability (Coalition-SGSL). This community based monitoring program evolved from a pilot project with four sites into a long-term monitoring program encompassing 37 estuaries along the sGSL coasts of New Brunswick, Prince Edward Island, and Nova Scotia (DFO 2011). DFO and Coalition-SGSL personnel work alongside volunteers from over twenty-nine watershed groups, three First Nation groups, and maritime universities each year from June to August to collect data (DFO 2011). Data include littoral nekton, aquatic vegetation, water quality and sediment (Thériault and Courtenay 2010). Littoral nekton are collected using a 30 m x 2 m beach seine with a 6 mm mesh size and central bag 2 x 1 m (Weldon et a. 2005). The seine net is deployed once at each station, and six stations are sampled in each estuary. From the shoreline, the net is pulled into the water perpendicular to the shoreline for half the length of the net, then the net is pulled parallel to the shoreline until the entire net is in the water, and then the net is pulled back towards the shore. The seine is then hauled by pulling both ends towards the shore and the contents of the bag are

placed in a live-box with water exchange. The method of seine net deployment samples an area of 225 m^2.

The initial objective of CAMP was to provide an avenue for community outreach and interaction with Environmental Non-Government Organizations (ENGOs) to raise awareness of estuary ecology (DFO 2011). As the program grew and demonstrated potential to collect baseline and long-term monitoring data, it is now looked at as a tool for assessing estuarine health using nekton as an indicator (Thériault and Courtenay 2010; DFO 2011). A goal for the CAMP dataset is to determine if it can be used to assess the relationship between the health of an estuary and the diversity and abundance of nekton within it (Weldon et al. 2007).

In March 2010, DFO biologists and research scientists joined university researchers, Environment Canada biologists, and the executive director of Coalition-SGSL at a science advisory meeting to review CAMP (DFO 2011). A concern was raised about the CAMP sampling design (DFO 2011). For CAMP, a standard number of six stations was established, regardless of estuary size, to allow for community groups to complete the sampling of an estuary within one day (DFO 2011). The main selection criteria for the six stations were: an area that is conducive for seining that has a gradual slope with predominantly mud and sand substrate composition, presence or past occurrence of eelgrass (*Zostera marina L.*) and road access to the shore (DFO 2011). Additional criteria included similar habitat and salinity patterns, and stations within the upper, middle, and lower estuary (DFO 2011). Ultimately, most station locations were selected solely to allow for community groups to easily access them from the road, because most community groups did not have access to a boat (DFO 2011). For several estuaries, up to two of the six stations were established to monitor a potential source of pollution rather than to be representative of the estuary (DFO 2011). Consequently, questions were raised regarding how representative the stations are of the overall condition of the estuary (DFO 2011). Large estuaries may need more stations and smaller ones may be over sampled, or stations may be clumped (DFO 2011). Another suggestion was to add stations to increase the number of "representative' stations to six at each site, and to cover as much of the estuary as possible in order to retain statistical comparability (DFO 2011). The conclusion of the review recommended the station locations and numbers should be assessed to determine if coverage is appropriate for each estuary (DFO 2011).

1.6 Thesis objective

The objective of this research project was to assess the ability of CAMP to provide a measure of littoral nekton that represents the overall littoral nekton community of each estuary. To accomplish this, the sampling design and effort of CAMP was tested by comparing nekton data collected from CAMP stations

to data collected from stations located through a stratified random design (SRD). The effect of increasing the number of stations on the precision of nekton abundance estimates was tested as well.

Hypotheses:

1. Sampling estuaries with a SRD will not produce significantly or substantively different nekton assemblages than those collected from CAMP stations.
2. Sampling a greater number of stations will not significantly or substantively alter the estimate of nekton community composition within each site.

Chapter 2
Assessing the sampling design of the Community Aquatic Monitoring Program (CAMP)

2.1 Introduction

Estuaries are partially enclosed water bodies where fresh water from the land meets the salty water from the sea. These dynamic environments provide numerous ecosystem services that attract human development (Barbier et al. 2011; Dafforn et al. 2012; Sheaves et al. 2012; Temmerman et al. 2013; Goncalves et al. 2015). As human populations expand and further develop along estuaries, assessment and monitoring of these ecologically and economically important ecosystems are urgent and crucial (Rothenberger et al. 2014).

The assessment of estuary health is complicated due to the high spatial and temporal variability of the estuarine environment. Accordingly, the selection of appropriate indicators is critical to the assessment of estuary health (Ysebaert and Herman 2002; Elliott and Quintino 2007; Porter and Scanes 2015). Estuary health signifies the degree to which the ecosystem has been altered from its natural state (Hallett et al. 2016). Comprehensive knowledge of the natural state of the selected indicators is required to identify when a change has occurred due to anthropogenic influences (Deeley and Paling 1999). Long-term monitoring programs have the capacity to detect meaningful patterns of change in these highly variable environments, because they provide critical historical data that can define the typical natural variability of an estuary (Deeley and Paling 1999; Tsou and Matheson 2002; Gorecki and Davis 2013). Both abiotic and biotic indicators are required to adequately monitor estuary health (Scanes et al. 2007). Consequently, the study of estuary health is known to be difficult and costly (Porter and Scanes 2015). Time and cost constraints can lead programs to select only abiotic indicators (Chariton et al. 2015), and many estuary monitoring programs are at risk of being canceled due to budget constraints (Mahoney and Bishop 2017).

One way managers can reduce costs associated with monitoring programs is to design a monitoring program that can be executed by local volunteers (Forrester et al. 2015). These community based monitoring programs are not only cost effective, but also promote public education and engagement (Sharpe and Conrad 2006). Community based monitoring programs also typically run longer than scientific monitoring programs (Theobald et al. 2015). These programs are becoming increasingly popular internationally among government and non-profit agencies (Conrad and Hilchey 2011). However, while funds are spared, concerns are raised regarding the scientific integrity of these programs (Forrester et al. 2015).

Previous studies have assessed the accuracy of community based monitoring programs by comparing data collected by community members with data collected by professionals. Fore et al. (2001) detected no significant difference between benthic macroinvertebrate samples collected by community members and professionals. Danielsen et al. (2014) went further and compared data collected by community members and scientists from resource monitoring programs across four countries and found no significant differences. The data collected by community members through a national program in Australia, which involves 7000 community members to collect marine debris, were assessed and found to be of comparable quality to those collected by researchers (van der Velde et al. 2017). The Community Aquatic Monitoring Program (CAMP), a long-term community based monitoring program that involves local stakeholders to monitor estuaries in the southern Gulf of St. Lawrence (sGSL), was assessed for its accuracy in nekton identification and abundance estimates by comparing data collected by volunteers with data collected by government biologists (Thériault et al. 2008). Taxonomic identifications were generally similar and the differences between the abundance estimates were less than 10% (Thériault et al. 2008). Therefore, community based monitoring programs, such as CAMP, may be ideal to monitor estuaries, because they have the capacity to collect scientifically-defensible data, run long-term, and are a cost-effective option to collect the biotic indicators required for assessments of estuary health.

Implemented in 2003, CAMP continues to be administered by Fisheries and Oceans Canada (DFO) in collaboration with the Southern Gulf of St. Lawrence Coalition on Sustainability (Coalition-SGSL). This community based monitoring program evolved from a pilot project with four sites into a long-term monitoring program encompassing 37 estuaries along the coasts of New Brunswick, Prince Edward Island, and Nova Scotia (DFO 2011). DFO and Coalition-SGSL personnel work alongside volunteers from over 29 watershed groups, three First Nation groups, and maritime universities each year from June to August to collect data (DFO 2011). Data include littoral nekton (i.e., fish, shrimp, and crabs), aquatic vegetation, water quality and sediment characteristics (Thériault and Courtenay 2010). The initial objective of CAMP was to provide an avenue for community outreach and interaction with Environmental Non-Government Organizations (ENGOs), and to raise awareness of estuarine ecology (DFO 2011). As the program grew and demonstrated potential to collect baseline and long-term monitoring data, it is now looked at as a tool for assessing estuarine health using nekton as an indicator (Thériault and Courtenay 2010; DFO 2011).

During a DFO science advisory meeting in March 2010, a concern was raised about the CAMP station selection methods (DFO 2011). Ultimately, the majority of station locations was selected primarily to allow for community groups to easily access them from the road (DFO 2011). For several estuaries, up to two of the six stations were established to monitor a source of direct anthropogenic influence rather than to be representative of the estuary (DFO 2011). In heterogeneous habitats, such as estuaries, a

stratified random sample is recommended where the total area is divided into equal plots and an even number of units is selected randomly from each plot (Dytham 2011). Additional concerns were raised regarding the number of sampling stations. Six stations are sampled in each estuary, regardless of estuary size, because that is the number of stations that volunteers were assumed to be capable of sampling in one day (DFO 2011). However, are six stations adequate to assess the condition of each estuary? Increasing sample size is one way to address nekton variability and increase sampling precision to detect biological differences among sites (Raposa et al. 2003). The conclusion of the review recommended the assessment of station locations to determine if coverage is appropriate for each estuary (DFO 2011).

The objective of this study was to assess the ability of CAMP to provide a measure of littoral nekton that represents the overall littoral nekton community of the estuary. To accomplish this, the sampling design of CAMP was tested by comparing nekton data collected from CAMP stations to data collected from stations located through a stratified random design (SRD). Secondly, the potential need to increase the number of stations was tested by assessing if six stations can detect the typical littoral nekton species within each estuary, and the precision of abundance estimates of each influential species as defined by data analyses.

2.2 Materials and methods

2.2.1 Description of sites

Ten estuaries (sites) were selected within the sGSL (Table 2.1, Figure 2.1). Six sites were located in New Brunswick and four sites were located in Prince Edward Island (PEI). Only sites that were scheduled for the 2016 CAMP program were considered. Sample collection was supported by personnel from DFO's Gulf Fisheries Centre in Moncton, New Brunswick and the University of Prince Edward Island (UPEI) in Charlottetown, PEI. As such, sites were selected where either DFO or UPEI planned to concurrently collect samples. Consequently, estuaries in Nova Scotia and northern New Brunswick were not considered.

Table 2.1 Site information for 2016 sample sites, including estuary area, watershed area, and station placement.

Site	Estuary Area (km^2)[1]	Watershed Area (km^2)	Station Placement
Bouctouche	Large	478.6	Clustered
Brudenell	Medium	260.4	Clustered
Cocagne	Medium	332.6	Clustered
Richibucto	Large	1138.5	Spread out
Scoudouc	Small	158.6	Clustered
Shediac	Small	246.3	Clustered
Souris	Small	53.3	Spread out
St. Louis de Kent	Medium	360.1	Spread out
Summerside	Medium	388.1	Spread out
Trout River	Small	93.1	Spread out

[1]Estuary area categories are based on estimated sizes of all CAMP estuaries. Small (size ≤ 25th percentile), Medium (25th percentile < median <75th percentile), Large (size ≥ 75th percentile).

Figure 2.1 Map of the ten sites sampled in 2016. Image created using ArcGIS (ESRI 2015).

2.2.2 Stratified random sampling design

Twelve stations were designated within each site. Six stations were the established CAMP station locations, and an additional six stations were randomly located and stratified among the upper, middle, and lower estuary. Each estuary was mapped and delineated using ArcGIS. The lower extent of the estuary was marked at the mouth of the estuary or to the lowest CAMP station when sampling extended into the bay. The upper extent of the estuary was marked where (when information was available) the salinity is known to be 10 PPT, or where the estuary narrows to a stream channel. A minimum salinity of 10 PPT was selected as the upper estuary benchmark, as that is the lowest average salinity that CAMP samples. Both shorelines were divided into three equal sections and overlaid by a grid that comprised 50 m^2 grid squares. Numbers were assigned to each grid square (Figure 2.2). One station location was randomly assigned to each section using a random number generator. Once a number was randomly selected, the aerial imagery beneath the corresponding grid square was inspected to ensure there were no obvious impediments to seining (e.g., piers). If an obstruction was clearly present, a new site was assigned using the random number generator.

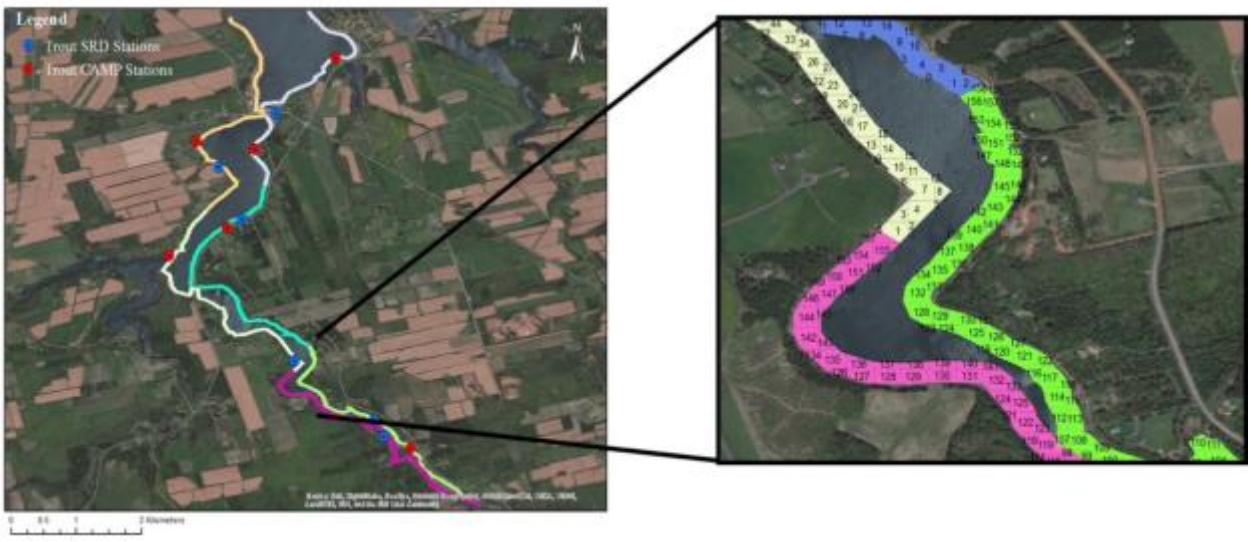

Figure 2.2 Example of estuary delineation and numbered grid employed to randomly select and stratify station locations among the upper, middle, and lower estuary. Image created using ArcGIS (ESRI 2015).

Richibucto is the only exception to the delineation method used, as it is the longest estuary selected as a site. As such, caution was taken to not stratify sampling among the entire 35 km of the Richibucto River estuary, which could result in stations being located over 20 km upstream of the most upstream CAMP station. Such a design would likely detect the effect of varying anthropogenic influences

and salinity rather than differences due to station randomization. Hence, the marine estuary, as defined by Turcotte-Lanteigne and Ferguson (2008), was delineated with the upper reach of the estuary being placed just above the community of Rexton.

2.2.3 Field data collection

Sites were sampled once in either July or August, 2016. Sampling dates were scheduled to avoid conflicts with the 2016 CAMP program. The core sampling crew remained the same throughout the sampling program. Stratified random design (SRD) stations were accessed using a 19-foot Carolina Skiff at New Brunswick sites and a 17-foot Carolina Skiff at PEI sites. If a station location was found to be unsuitable in the field, then the nearest suitable sample location was selected. A total of 15 stations were relocated and the reason for relocation recorded. The greatest number of station relocations occurred in Scoudouc (four stations) due to the large number of shoreline developments and extensive riprap that impeded seining. Other reasons for relocating stations were the presence of aquaculture, insufficient water depths, and extreme density of sea lettuce (*Ulva lactuca*). One CAMP station at Souris was not sampled, because it is located on a public beach and members of the public were swimming during the sampling time. One Summerside SRD station could not be sampled due to unsafe weather conditions. Figures S2.1 to S2.10 display site maps with finalized station locations.

At each station, nekton and water quality parameters were collected, and substrate composition was estimated using CAMP methods, as outlined by Weldon et al. (2005). Nekton species were captured using a 30 m by 2 m beach seine with a mesh size of 6 mm and central bag measuring 2 m by 1 m, which samples a standardized area of 225 m^2 at each station. A seine net with a mesh size of 3 mm was used for sampling Brudenell and Summerside. All captured nekton were placed in a live-box with water exchange, identified, classified as either young-of-the-year (YOY) or adult, enumerated and then released. Species not routinely identified by CAMP were not recorded, including Common Starfish (*Asterias rubens*), Hermit Crabs (*Pagurus* sp.), Mysids (*Mysidopsis* sp.), and Common Periwinkle (*Littorina littorea*). Certain nekton species were preferentially counted and removed from the live-box first, including Atlantic Silversides (*Menidia menidia*) and Green Crabs (*Carcinus maenas*). Atlantic Silversides were removed before other fish, because they succumb more quickly to stress induced by crowding. Green Crabs were removed as soon as possible due to their predatory behaviour. Large catches of Green Crabs could result in large numbers of nekton being physically damaged. As a result, only heads of damaged individuals were counted to reduce the chance of duplicate counts. All fish were handled in accordance with the approved University of Waterloo animal care protocol (AUPP #14-15). All fish collection activities were in compliance with DFO Gulf Region License to Fish for Scientific Purposes, License No. SG-RHQ-16-016C.

Water quality data, including temperature, dissolved oxygen (DO) (mg/L), and salinity (PPT), were collected using a handheld YSI Professional Plus model at New Brunswick sites and a YSI 6600M model at PEI sites. Water quality was measured from the middle of the water column within the seined area. The substrate composition (% cover of sand, gravel, rock, and mud) was estimated visually by walking within the seined area. Tides were visually assessed in the field as either ingoing or outgoing, and at low, mid, or high height. The tide height (m) for each station at the time of sampling was documented by accessing the tide tables available on the DFO website (DFO n.d.).

2.2.4 Data analysis and statistics

The following species were pooled together for data analysis due to difficulty of field identification:

- Alewife (*Alosa pseudoharengus*) YOY and Blueback Herring (*Alosa aestivalis*) YOY counts were pooled as Gaspereau YOY.
- Blackspotted Stickleback (*Gasterosteus wheatlandi*) YOY and Threespine Stickleback (*Gasterosteus aculeatus*) YOY counts were pooled as Gasterosteus YOY.
- Mummichog (*Fundulus heteroclitus*) YOY and Banded Killifish (*Fundulus diaphanous*) YOY counts were pooled as Fundulus YOY.
- Winter Flounder (*Pseudopleuronectes americanus*) YOY and Smooth Flounder (*Pleuronectes putnami*) YOY counts were pooled as Flounder YOY.

PRIMER is a multivariate statistical software package commonly used by researchers assessing aquatic and marine environmental and biological data (Clarke et al. 2014). PRIMER 7 with the PERMANOVA add-on package was used to complete multivariate analyses to test for differences between the CAMP and the SRD data. The stations were treated as replicates within each site. All univariate analyses were completed using RStudio version 0.99.489.

2.2.4.1 Data treatment

A square-root transformation was applied to the data to reduce the dominance of the highly abundant species, such as Mummichog and Sand Shrimp (*Crangon septemspinosa*), and allow other species to also influence the similarity calculation (Figure S2.11). Similarities between pairs of samples were defined with a similarity matrix generated using the Bray-Curtis similarity coefficient. The resulting Bray-Curtis similarity matrix was the basis for all of the multivariate analyses. The Bray-Curtis similarity coefficient was chosen, because the joint absence of a species does not increase the similarity of two samples, and it produces a value of 0 when two samples a no species in common (Clarke et al. 2014).

2.2.4.2 Detecting differences in nekton assemblages between the Community Aquatic Monitoring Program (CAMP) and Stratified Random Design (SRD) sampling designs

The nekton data collected from the SRD and CAMP stations were compared to assess the effect of implementing a stratified random sampling design. A cluster analyses was performed to determine if both sampling designs separate sites into the same groups. The cluster analysis generates a dendrogram that displays the sites in hierarchical groups based on the similarity between each cluster, which is based on the Bray-Curtis dissimilarities (Clarke et al. 2014). A hierarchical cluster analysis using a group average linkage was performed. The success of the cluster analyses was measured using a similarity profile (SIMPROF) test, which assesses if the groups are significantly different. SIMPROF significance level was set at 5% with 9999 permutations.

The differences between the two sampling designs were portrayed using non-metric Multi-Dimensional Scaling (nMDS) ordinations. The nMDS plots visually display the 2-dimensional spatial relationships between the samples based on the ranks created by the Bray-Curtis similarity matrix. An nMDS plot is essentially a map in which the distances between pairs of sites represents the relative dissimilarity of community composition. The accuracy of the nMDS is measured with a stress coefficient. The acceptable level of stress is less than 0.2, which indicates the nMDS is a good to excellent representation with a low risk of misinterpretation (Clarke et al. 2014).

A two-way crossed permutational MANOVA (PERMANOVA) was used to formally test the hypothesis of no difference in nekton community assemblage between the two sampling designs as defined by the Bray-Curtis similarity matrix on square-root transformed data. The two factors of the analysis were sampling design and site. Sampling design was treated as a fixed factor, and site was treated as a random factor since only a subset of all CAMP estuaries was sampled. A Type III sums of squares was used, because it is the most conservative approach to partitioning variability, which is appropriate for unbalanced designs (Anderson et al. 2008). However, since only two observations are missing, it is unlikely the choice of the type of sums of squares had an effect on the overall conclusions (Anderson et al. 2008). P-values were obtained by applying 9999 permutations of residuals under a reduced model, because it yields the best power and most accurate type I error (Anderson et al. 2008). The differences between the nekton assemblages of the two sampling designs were explored within each estuary by applying a pair-wise test among the factor sampling design within the factor site. The differences between the two sampling designs' comparisons of sites were also explored by applying a pair-wise test among the factor site within the factor sampling design.

A test of homogeneity of dispersion (PERMDISP) using the group factor site was used to test if the differences detected between the sites were influenced by differences in the dispersion of the data. Another PERMDISP was run using the group factor sampling design to test if there was a significant

difference in the variability of the replicates between the two sampling designs. The PERMDISP test compares the distances measured from samples to their group centroid (Anderson et al. 2008). P-values were obtained through 9999 permutations of least-square residuals.

2.2.4.3 Assessing the adequacy and precision of six stations

The optimal sub-sampling effort was determined by generating species accumulation plots using PRIMER. Species accumulation plots were created for both sampling designs using data from the CAMP and SRD stations, which were permuted 720 times to determine how many species are typically gained with each additional station. The data from the two sampling designs were then combined to predict the potential for increasing the number of stations to alter conclusions regarding the dissimilarity of sites. A species accumulation plot was generated using the combined dataset to assess at what station number are all species detected. A PERMANOVA and PERMDISP were performed using the combined dataset to assess if an increase in station numbers results in different conclusions regarding the dissimilarity of sites and if the variability within sites is significantly changed.

The similarity percentages routine (SIMPER) in PRIMER was used to assess which species have the greatest influence on the dissimilarities between the estuaries as defined by the CAMP data. The SIMPER analysis measures the contribution of each species to the Bray-Curtis dissimilarity between each pair of samples (Clarke et al. 2014).

Using the CAMP data, for each influential species identified by the SIMPER analysis, one-way, Model II ANOVAs were used to partition the total variance in counts of each species into among and within site components, as introduced by Bailey and Byrnes (1999). The within site mean square (MS_{within}) is an estimate of the variance among stations within a site (s^2_{within}). The among group mean square (MS_{among}) includes both among site and within site variability so among site variance (s^2_{among}) is calculated as follows:

$$s^2_{among} = \frac{MS_{among} - MS_{within}}{6 \text{ samples per estuary}}$$

The variance of the mean (s^2_{mean}) was calculated using the within and among site component of variance, where n is the number of sites sampled and m is the number of stations sampled.

$$s^2_{mean} = \frac{s^2_{within}}{nm} + \frac{s^2_{among}}{n}$$

Values of m were then substituted with values of 7 through 12 to measure if a substantive reduction in the variance of the mean would be obtained by sampling more than 6 stations at each site.

Confidence intervals on the mean of each species count were calculated by taking the square-root of the variance of the mean and multiplying it by its corresponding t-value. The resulting confidence intervals were used to assess the precision gained with increasing station numbers in estimating the abundance of each influential species. The optimal station number was considered to be the one that yielded confidence intervals small enough to detect the average differences in abundances of the influential species among sites.

2.3 Results and discussion

2.3.1 Summary of environmental data

The environmental data collected at each station are summarised in Table 2.2. The greatest difference in salinity between sampling designs was 3.6 PPT (Bouctouche), and the average was 1.9 PPT. The greatest difference in tide height between sampling designs was 0.6 m (Cocagne) and the typical difference was 0.1 m. The greatest difference in water temperature between sampling designs was 2.9°C (Trout River), and the average was 1.2°C. The greatest difference in DO concentrations between sampling designs was 2.0 mg/L (Scoudouc), and the average was 0.7 mg/L.

Table 2.2 Summary of environmental data collected with the Community Aquatic Monitoring Program (CAMP) and Stratified Random Design (SRD) sampling design at each site sampled in 2016. Data are averages of station data collected for each sampling design.

Estuary	Salinity (PPT)[1]		Tide Height[1, 2] (m)		Water Temperature[1] (°C)		DO Concentration[1] (mg/L)		Substrate (%Composition)[1, 3]	
	CAMP	SRD	CAMP	SRD	CAMP	SRD	CAMP	SRD	CAMP	SRD
Cocagne	29.0	26.1	1.0	0.4	20.8	23.3	8.5	8.4	S:81, G:13, R:6, M:0	S:79, G:5, R:1, M:15
St. Louis de Kent	23.8	22.3	0.6	0.7	18.4	19.5	7.6	8.5	S:82, G:7, R:2, M:10	S:68, G:7, R:2, M:24
Trout River	20.4	21.9	0.5	0.6	21.9	19.0	9.6	8.8	S:47, G:3, R:18, M:33	S:55, G:8, R:8, M:28
Souris	25.8	23.4	0.9	0.9	23.3	21.3	8.6	8.9	S:94, G:6, R:0, M:0	S:70, G:9, R:18, M:3
Richibucto	26.7	26.4	0.4	0.3	19.9	20.9	8.6	8.1	S:76, G:8, R:13, M:3	S:68, G:3, R:6, M:23
Bouctouche	24.3	20.7	0.7	0.8	21.1	21.0	8.4	7.9	S:54, G:5, R:8, M:33	S:81, G:7, R:0, M:13
Scoudouc	26.9	27.5	1.2	1.3	22.0	21.1	9.2	7.2	S:83, G:2, R:0, M:15	S:43, G:5, R:7, M:45
Brudenell	29.4	27.3	1.5	1.4	22.3	21.7	9.9	8.8	S:78, G:12, R:11, M:0	S:58, G:5, R:5, M:32
Shediac	27.8	24.6	1.3	1.2	25.1	25.4	7.1	7.5	S:82, G:8, R:4, M:6	S:65, G:1, R:1, M:33
Summerside	25.9	25.0	1.8	1.8	22.4	23.1	8.5	8.6	S:88, G:10, R:2, M:0	S:92, G:2, R:3, M:3

[1] Mean of sampling stations
[2] Tide height information collected from DFO tidal predictions website (DFO n.d.)
[3] S: sand, G: gravel, R: rock, M: mud

2.3.2 Detecting differences in nekton assemblages between the Community Aquatic Monitoring Program (CAMP) and Stratified Random Design (SRD) samples

The cluster analysis grouped the two sampling designs together for each site, except for Cocagne (Figure 2.3). The cluster analysis displays the results of the SIMPROF test by connecting the samples that are not significantly different with dotted red lines. The cluster analysis indicates that the nekton assemblages do not significantly differ among sampling designs, other than for Cocagne. Separate nMDS ordination plots for the data collected from the CAMP stations (Figure 2.4) and SRD stations (Figure 2.5) visually display the differences in the degree of dissimilarity between sites among the sampling designs. The stress for both ordination plots is 0.11 which is a good representation of the distances between sites based on the dissimilarity of their nekton assemblages. The position of Cocagne moves from being closest to Brudenell in the CAMP sampling design to being closest to Richibucto in the SRD sampling design. Scoudouc, Shediac, and Bouctouche are more tightly clustered together in the SRD sampling design. Overall, the general pattern of sites is consistent among the sampling designs. The degree to which these differences would affect the interpretation of the community data, and consequently, management decisions were explored using PERMANOVA.

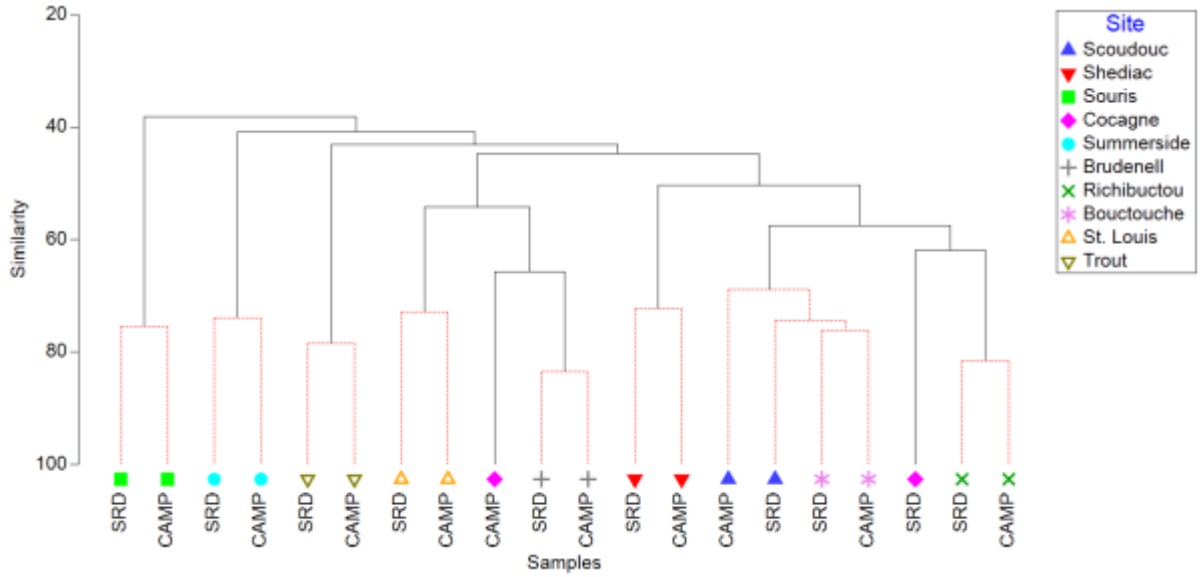

Figure 2.3 Cluster analysis on Bray-Curtis similarities for nekton abundance data (square-root transformed) for each sampling design (Community Aquatic Monitoring Program [CAMP] vs. Stratified Random Design [SRD] - average of 6 stations each) for each site. Dotted red lines represent similarity profile (SIMPROF) results where groups of samples are not significantly different (5% significance).

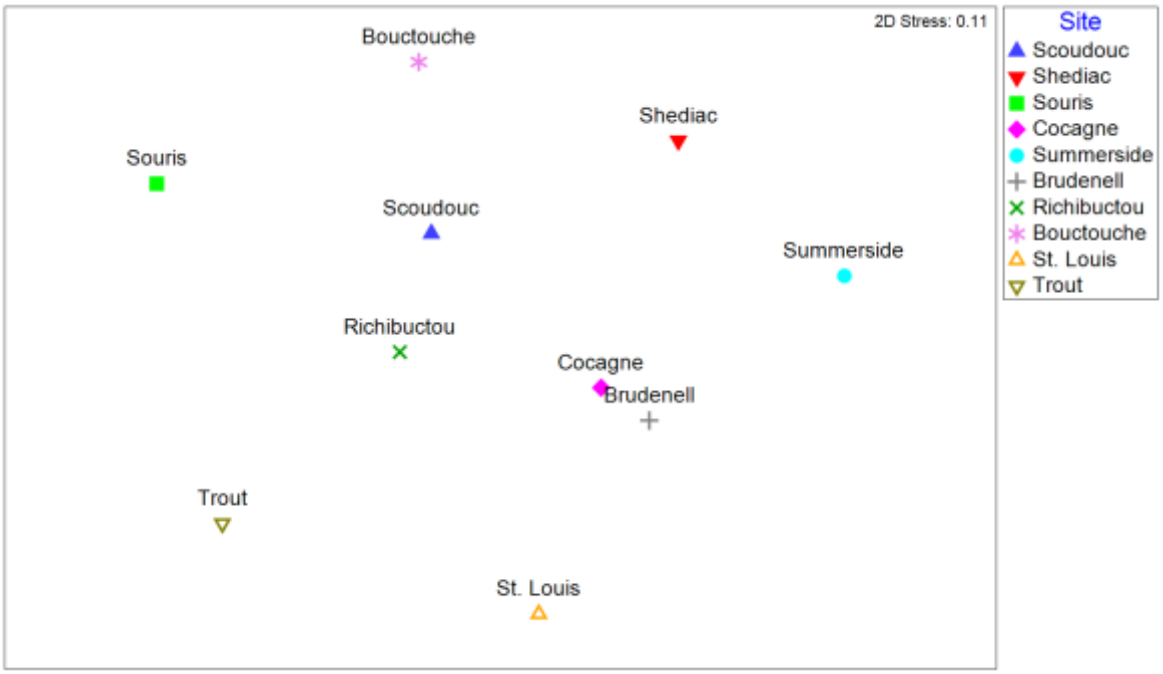

Figure 2.4 non-metric Multidimensional Scaling ordination plot of square-root transformed nekton data collected from Community Aquatic Monitoring Program (CAMP) stations. Nekton data are averages of station data for each site.

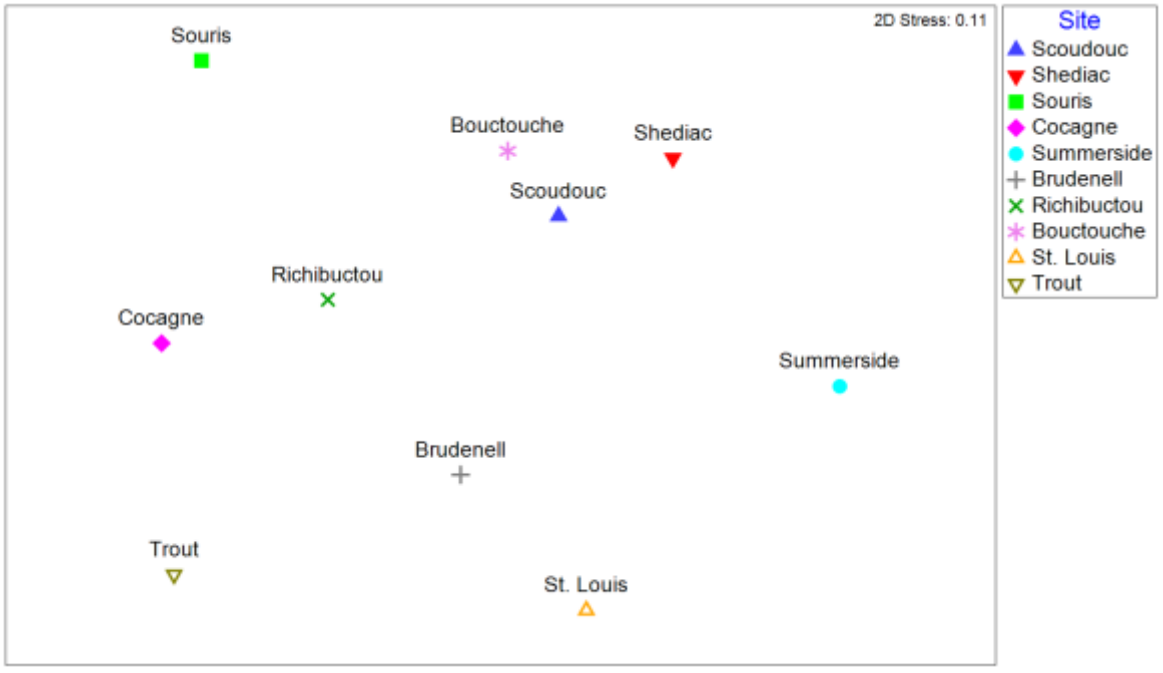

Figure 2.5 non-metric Multidimensional Scaling ordination plot of square-root transformed nekton data collected from Stratified Random Design (SRD) stations. Nekton data are averages of station data for each site.

The results of the PERMANOVA (Table 2.3) show there are significant differences between nekton assemblages among sites (F=12.95, P=0.0001), but there is no significant difference between sampling designs (F=1.44, P=0.2073). However, the degree of differences between the sampling designs was somewhat dependent on the site (marginal interaction between Site x Sampling Design; F=1.28, P=0.0475). Thus, a pair-wise test was performed to look at the specific differences detected between the sampling designs within each site. The results of the pair-wise test (Table 2.4) show there are significant differences detected between the sampling designs within Cocagne and Shediac. The differences detected in Cocagne (t=1.819, P=0.002) support the findings of the cluster analysis and nMDS ordinations. There was a marginal difference detected between the sampling designs in Shediac (t=1.527, P=0.035), which is not convincing as PERMANOVA is known to be an over-powered test.

A pair-wise test was also performed to look at the specific differences detected between sites for each sampling design (Table 2.5). The results of this pair-wise test show there are significant differences detected between all sites for both sampling designs. The one exception is Cocagne and Richibucto, where no significant differences was detected (t=1.27, P=0.173) for the SRD sampling design.

Table 2.3 Two-way crossed permutational-MANOVA (PERMANOVA) results for the analysis of nekton community data with factors Site and Sampling Design.

Factor	d.f.	*MS*	*F*	*P*
Site	9	15443	12.96	0.0001
Sampling Design	1	2197	1.44	0.2073
Site x Sampling Design	9	1528	1.28	0.0475
Residuals	98	1192		
Total	117			

Table 2.4 Permutational-MANOVA (PERMANOVA) pair-wise test results for factor Sampling Design within factor Site.

Site	Sampling Design	*t*	*P*
Cocagne	CAMP, SRD	1.819	0.002
Shediac	CAMP, SRD	1.527	0.035
Bouctouche	CAMP, SRD	1.366	0.127
Souris	CAMP, SRD	1.270	0.168
Trout River	CAMP, SRD	1.165	0.208
Brudenell	CAMP, SRD	1.055	0.345
Scoudouc	CAMP, SRD	1.023	0.401
Richibucto	CAMP, SRD	0.746	0.781
St. Louis de Kent	CAMP, SRD	0.787	0.784
Summerside	CAMP, SRD	0.664	0.940

Table 2.5 Comparison of permutational-MANOVA (PERMANOVA) pair-wise tests among factor Site within factor Sampling Design for nekton data collected from the Community Aquatic Monitoring Program (CAMP) and Stratified Random Design (SRD) stations.

Sites	CAMP		SRD	
	t	***P***	***t***	***P***
SCOU vs SHED	2.830	0.005	1.841	0.021
SCOU vs SOUR	3.22	0.010	2.937	0.007
SCOU vs COCA	2.261	0.003	1.851	0.010
SCOU vs SUMM	2.508	0.003	2.041	0.006
SCOU vs BRUD	2.691	0.004	2.269	0.005
SCOU vs RICH	2.092	0.003	1.943	0.004
SCOU vs BOUC	1.652	0.017	1.906	0.012
SCOU vs STLO	2.849	0.005	2.374	0.001
SCOU vs TROU	2.997	0.005	3.094	0.005
SHED vs SOUR	4.399	0.003	3.916	0.003
SHED vs COCA	2.723	0.005	2.603	0.005
SHED vs SUMM	2.524	0.002	2.642	0.002
SHED vs BRUD	3.001	0.003	2.751	0.002
SHED vs RICH	3.493	0.003	2.592	0.002
SHED vs BOUC	2.664	0.002	1.504	0.035
SHED vs STLO	3.420	0.002	2.698	0.001
SHED vs TROU	4.886	0.003	4.389	0.002
SOUR vs COCA	3.427	0.002	2.602	0.008
SOUR vs SUMM	3.541	0.003	4.053	0.004
SOUR vs BRUD	2.507	0.003	2.227	0.004
SOUR vs RICH	1.961	0.009	2.230	0.003
SOUR vs BOUC	2.679	0.002	2.764	0.006
SOUR vs STLO	3.483	0.001	2.867	0.003
SOUR vs TROU	4.413	0.002	4.132	0.003
COCA vs SUMM	2.372	0.002	2.730	0.002
COCA vs BRUD	1.765	0.021	1.758	0.014
COCA vs RICH	2.140	0.009	1.266	0.173
COCA vs BOUC	2.566	0.007	2.128	0.005
COCA vs STLO	2.039	0.015	2.107	0.002
COCA vs TROU	3.348	0.003	1.852	0.001
SUMM vs BRUD	2.574	0.006	2.499	0.005
SUMM vs RICH	2.884	0.005	2.851	0.002
SUMM vs BOUC	2.404	0.002	2.625	0.004
SUMM vs STLO	2.487	0.002	2.346	0.009
SUMM vs TROU	3.425	0.004	3.449	0.001
BRUD vs RICH	2.102	0.003	1.949	0.009
BRUD vs BOUC	2.568	0.008	2.277	0.002
BRUD vs STLO	2.136	0.008	1.622	0.031
BRUD vs TROU	3.507	0.004	2.663	0.004
RICH vs BOUC	2.009	0.006	1.779	0.009
RICH vs STLO	2.545	0.003	2.273	0.003
RICH vs TROU	2.898	0.003	2.441	0.002
BOUC vs STLO	2.957	0.001	2.607	0.002
BOUC vs TROU	3.419	0.003	3.919	0.005
STLO vs TROU	2.998	0.003	2.333	0.002

While the results indicate the nekton assemblages do not significantly differ among sampling designs in general, it is apparent that the variability within the sites differs among sampling designs (Table 2.6). The greatest differences between sampling designs in within-site similarity were observed in Cocagne (9.4%), Scoudouc (9.1%), St. Louis de Kent (9.1%), and Trout River (4.7%). The reduced similarity observed in the SRD sampling design for Cocagne, Scoudouc, and St. Louis de Kent may be due to a greater distance between stations compared to the CAMP sampling design (i.e., SRD stations were more spread out along the estuary than were CAMP stations). Increased distance between the upper and lower stations for Cocagne, Scoudouc and St. Louis de Kent are 1.8, 0.9, and 3.4 km, respectively. However, for Trout River the distance between stations within the SRD sampling design is approximately 1.1 km less than the CAMP stations. The greatest difference in the spread of stations between sampling designs is in Bouctouche where the maximum distance between SRD sampling stations is 8.1 km greater than CAMP stations. The large difference in stations spread between sampling designs in Bouctouche resulted in a difference of only 3% in within site variability. Hence, the variability of the nekton data within each site may be more controlled by the habitat variability within each estuary rather than the extent of clustering or spread of stations, or the size of the estuary.

Table 2.6 Comparison of the average similarity (%) of nekton assemblage data within sites between the Community Aquatic Monitoring Program (CAMP) and Stratified Random Design (SRD) sampling designs.

Site	Average Similarity (%)	
	CAMP	SRD
Scoudouc	60.9	51.8
Shediac	63.0	59.9
Souris	61.9	61.0
Cocagne	55.6	46.2
Summerside	42.1	46.1
Brudenell	44.7	43.5
Richibucto	52.6	52.5
Bouctouche	50.5	53.5
St. Louis de Kent	51.9	42.8
Trout River	65.9	61.2

The results of the PERMDISP (Table 2.7) indicate that the variability among the sampling designs is not significantly different (F=0.021, P=0.9), and the variability among the sites is not significantly different for the CAMP sampling design (F=2.512, P=0.1) or SRD sampling design (F=1.727, P=0.3). Therefore, the differences detected between the sites are attributed to a difference in location of the data and not the dispersion of the data. Likewise, there is no significant difference between the variability within sites between the sampling designs.

Table 2.7 Test of homogeneity of dispersion **(**PERMDISP) results for the analysis comparing the dispersion of the Community Aquatic Monitoring Program (CAMP) and Stratified Random Design (SRD) sampling designs with group factor Sampling Design. Additionally, results of PERMDISP performed separately on the dispersion of the CAMP and SRD station nekton community data with group factor Site.

Group Factor	d.f.1	d.f.2	*F*	*P*
Sampling Design	1	116	0.021	0.9
Site (CAMP)	9	49	2.512	0.1
Site (SRD)	9	49	1.727	0.3

Overall, I accept the null hypothesis that the two sampling designs do not produce significantly different nekton assemblages. Regardless of sampling design, the findings would indicate all of the sites are different. Consequently, it is unlikely management decisions would change based on which sampling design were used if the differences between sites are assessed based on nekton community composition as recently completed by Reynoldson et al. (2016).

The results are evidence that a lack of station stratification and randomization do not limit the utility of CAMP for decision-makers. Cocagne is the only site where there appear to be significant differences between the nekton assemblages of the two sampling designs. Consequently, variables potentially influencing the differences between the Cocagne sampling designs were explored. Cocagne CAMP stations are clustered in the bay (Figure S2.3), which results in those stations experiencing higher salinity concentrations than the majority of the SRD stations that are spread throughout the estuary. Salinity is a primary factor naturally affecting distribution and occurrence of estuarine fishes (Harrison and Whitfield 2006b) and significant differences in nekton assemblages have been detected between regions that differed based on salinity (Gorecki and Davis 2013). The salinity measured at the CAMP stations was on average 2.9 PPT greater than the salinity measured at the SRD stations (Table 2.2). Yet, a similar difference in salinity was also measured in Bouctouche and Shediac, which had average differences of 3.6 and 3.2 PPT, respectively.

Tides are another environmental variable believed to influence nekton assemblages (Castellanos-Galindo and Krumme 2015). In Cocagne, the average difference in tide height between the sampling designs was 0.6 m. Conversely, the next largest difference in tide height between sampling designs was 0.1 m. The large difference between tide heights in Cocagne resulted from sampling logistics rather than station placement. The sampling designs were sampled only a day apart, but the CAMP sampling began 1.75 hours before the SRD sampling and finished 4.00 hours before the SRD sampling. The difference in time was a consequence of the late start of the SRD sampling, and longer sampling time due to shallow waters preventing boat access to the shoreline. The potential influence of tides on variability of nekton

community assemblages should be explored further to determine if CAMP should start standardizing tide height for each estuary to reduce variability within the dataset. Currently, the CAMP protocol is to sample sites around the same date each year, and to meet volunteers at 8:00 AM regardless of tide height.

Station stratification amongst the upper, middle, and lower estuary was anticipated to influence differences between the sampling designs, especially in estuaries where CAMP stations are clustered (Bouctouche, Brudenell, Cocagne, Scoudouc, and Shediac). While overall significant differences were not detected between the sampling designs, the effect of station stratification was explored to see if nekton communities do differ between the upper and lower estuary. The possible effect of station stratification was explored with another PERMANOVA using only SRD data and defining the location of each station as either upper, middle, or lower estuary. The factors were Site crossed with Location (Table 2.8). The results indicate there are no significant differences between the nekton communities based on location in estuary. The lack of differences between the nekton communities in the upper and lower estuary may be due to the majority of sampled estuaries being shallow and well-mixed, and not characterised by the typical steep horizontal salinity gradient observed in some estuaries. The lack of a steep salinity gradient may be a factor in why station stratification does not appear to be essential when monitoring these estuaries.

Table 2.8 Two-way crossed permutational-MANOVA **(**PERMANOVA) results for the analysis of nekton community data collected from the Stratified Random Design (SRD) stations with factors Site and Location (upper or lower estuary).

Source	d.f.	*MS*	*F*	*P*
Site	9	7967	6.84	0.0001
Location	2	1075	0.76	0.7022
Site x Location	18	1414	1.21	0.0916
Residuals	29	1166		
Total	58			

2.3.3 Assessing station numbers

The results of this study suggest that CAMP does not need to re-locate stations, as the random/stratified site selection does not result in a different assessment from CAMP of which sites are different in most cases. The next question that was addressed was whether stations should be added to CAMP sites? An ideal number of CAMP stations is the minimum number that provides an adequate characterization of the nekton assemblage by detecting the majority of littoral nekton species present in each estuary. A sufficient number of stations would also provide sufficient precision in the estimate of species counts. Precision is gained with the decreasing variability of species counts. One method to reduce variability is

the addition of samples. Hence, the potential benefit of increasing the number of CAMP stations was assessed by determining the ability of the current six CAMP stations to both detect and estimate counts of littoral nekton species.

Species accumulation plots were used to see at which station number the dataset stops accumulating species. Both the species accumulation plots for the CAMP data (Figure 2.6) and SRD data (Figure 2.7) suggest six stations are sufficient, as species are typically not gained after the 5th station. The exceptions are Richibucto, Summerside and Souris. The two sampling designs did not capture all of the same species and neither of the sampling designs detected all possible nekton species present (Table 2.9). The greatest numbers of discrepancies in the species detected between sampling designs are in Shediac, St. Louis de Kent, and Souris. These sites range in size from small to medium. One of the most severe discrepancies was in Trout River where 16 Cunner (*Tautogolabrus adspersus*) were captured over 3 CAMP stations, but none were captured at SRD stations. Trout River is a small estuary and the CAMP stations are spread throughout. Therefore, the potential benefit of increasing the number of stations sampled within an estuary may not be dependent on the size of the estuary.

Table 2.9 Species richness by sampling design, and the species that were captured by one sampling design and not detect by the other.

Estuary	Number of Species Captured			Additional/Different Species	
	CAMP	SRD	Total	CAMP	SRD
Scoudouc	15	14	15	9SS (6 @ 1stn)	
Shediac	13	13	16	BSS (1 @ 1 stn) GASP (7 @ 1 stn) 3SS (3 @ 3 stn)	9SS (1 @ 1 stn) KIL (8 @ 2 stn) FLOU YOY (5 @ 2 stn)
Souris	14	14	16	SFL (4 @ 2 stn) FUND YOY (1 @ 1 stn)	GRUB (1 @ 1 stn) MUM (2 @ 1 stn)
Cocagne	13	12	14	TOM (1 @ 1 stn) SFL (7 @ 1 stn)	SBA (1 @ 1 stn)
Summerside	13	14	14		MCR (16 @ 3 stn)
Brudenell	12	13	13		FUND YOY (7 @ 1 stn)
Richibucto	20	20	21	FUND YOY (1 @ 1 stn)	EEL (1 @ 1 stn)
Bouctouche	17	15	17	SMEL (19 @ 1 stn) 3SS (2 @ 2 stn)	
St. Louis de Kent	15	15	18	GCR (2 @ 1 stn) WFL (2 @ 1 stn) FLOU YOY (1 @ 1 stn)	EEL (1 @ 1 stn) PER (8 @ 4 stn) WNFL (1 @ 1 stn)
Trout River	14	15	16	CUN (17 @ 3 stn)	SFL (13 @ 1 stn) EEL (8 @ 2 stn)

3SS: Threespine Stickelback (*Gasterosteus aculeatus*)
9SS: Ninespine Stickleback (*Pungitius pungitius*)
BSS: Blackspotted Stickleback (*Gasterosteus wheatlandi*)
CUN: Cunner (*Tautogolabrus adspersus*)
EEL: American Eel (*Anguilla rostrata*)
FLOU: Flounder (*Pseudopleuronectes americanus or Pleuronectes putnami*)
FUND: Fundulus (*Fundulus heteroclitus or Fundulus diaphanous*)
GASP: Gaspereau (*Alosa pseudoharengus or Alosa aestivalis*)
GCR: Green Crab (*Carcinus maenas*)
GRUB: Grubby (*Myoxocephalus aeneus*)
KIL: Banded Killifish (*Fundulus diaphanous*)
MCR: Mud Crab (*Neopanope sayi, Rhithropanopeus harrisi)*
MUM: Mummichog (*Fundulus heteroclitus*)
PER: White Perch (*Morone americana*)
SBA: Striped Bass (*Morone saxatilis*)
SFL: Smooth Flounder (*Pleuronectes putnami*)
SMEL: Rainbow Smelt (*Osmerus mordax*)
TOM: Atlantic Tomcod (*Microgadus tomcod*)
WFL: Winter Flounder (*Pseudopleuronectes americanus*)
WNFL: Windowpane Flounder (*Scophthalmus aquosus*)
YOY: Young of Year

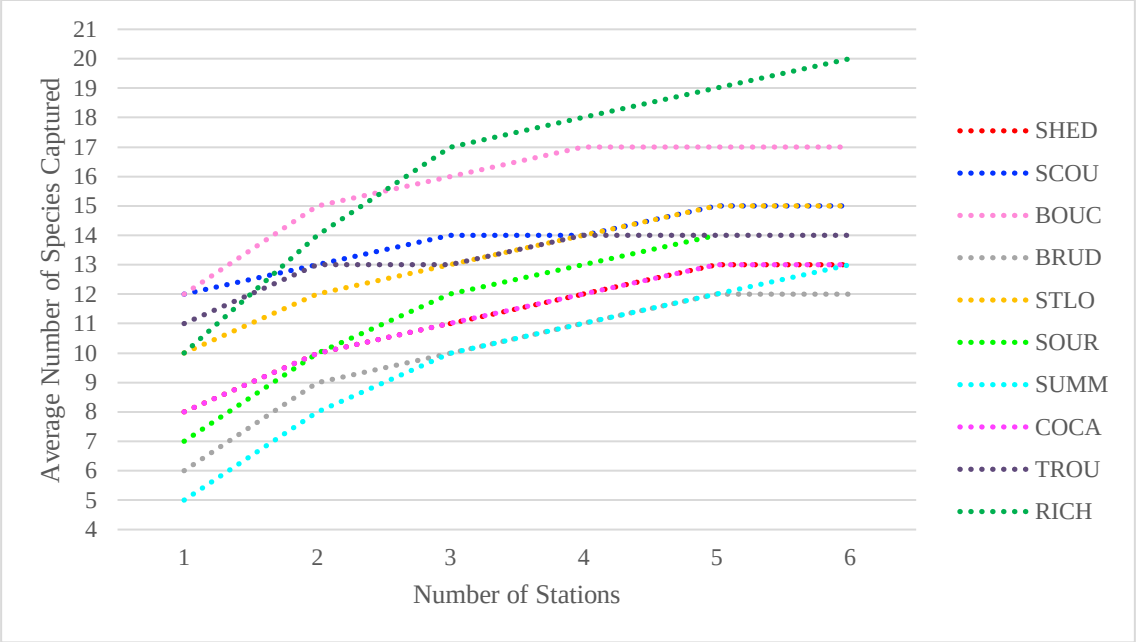

Figure 2.6 Species accumulation plot generated using nekton data collected from the Community Aquatic Monitoring Program (CAMP) stations at each site. The plot displays the average number of new species detected with each increasing station number as defined by 720 permutations of station data.

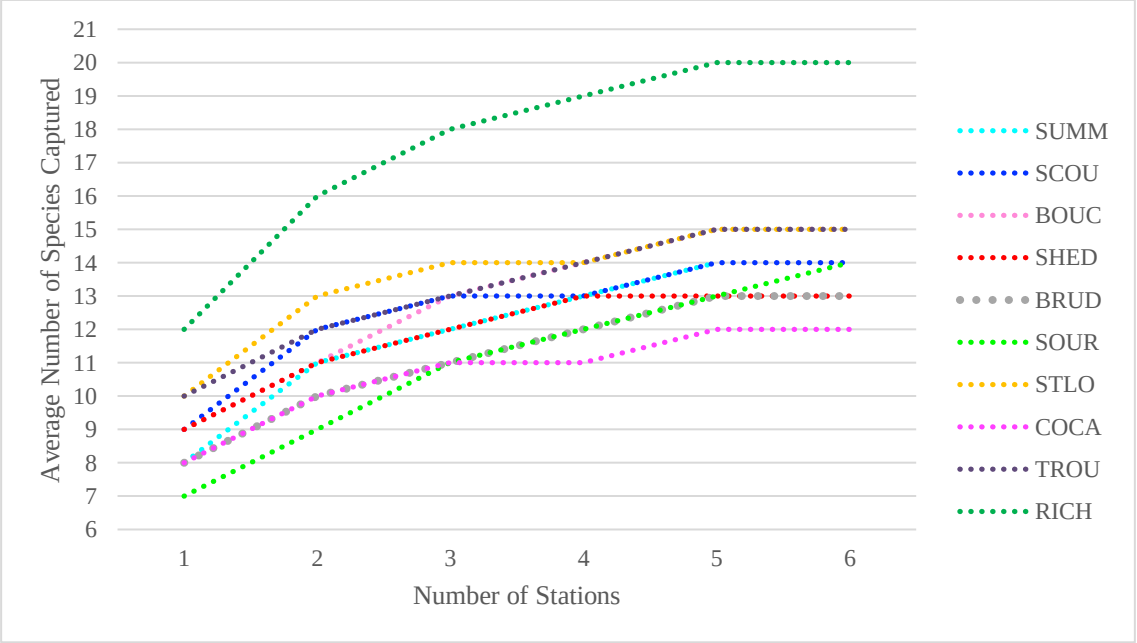

Figure 2.7 Species accumulation plot generated using nekton data collected from the Stratified Random Design (SRD) stations at each site. The plot displays the average number of new species detected with each increasing station number as defined by 720 permutations of station data.

The data from the two sampling designs were combined to generate a species accumulation plot displaying the accumulation of species with twelve stations in each estuary (eleven for Souris and Summerside) (Figure 2.8). The plot suggests ten is a sufficient number of stations, as it is the average number of stations where the maximum number of species is attained. Yet, is the increased effort to sample four additional stations warranted in order to detect the nekton species otherwise missed? The potential influence of additional station data was further evaluated by generating an nMDS ordination plot with nekton community data averaged for all twelve stations sampled at each site (Figure 2.9). Overall, combining the twelve stations does not appear to alter the general position of the sites other than Cocagne and Brudenell. A one-way PERMANOVA using the combined dataset revealed all sites are still significantly different (F=12.6, P=0.0001) (Table 2.10). The standard errors generated by separate PERMDISP tests on the CAMP and combined dataset show that combining the station data generally reduces the variability of the station data within the majority of sites (Table 2.11). A pair-wise PERMDISP was run directly comparing the dispersion of the CAMP data and the combined dataset (Table 2.12). The results of the pair-wise PERMDISP indicate the reduction in dispersion within the combined dataset does not provide a significant reduction in the variability of the site data.

These results suggest that adding stations to CAMP sites will likely increase the number of species detected and reduce variability of the nekton data, but will not alter the conclusion that all sites are different or significantly reduce within site variability. Therefore, rather than assess the adequacy of CAMP based on its ability to detect all species within the estuary, a more pertinent analysis would be to assess the precision of CAMP in estimating the counts of species that do influence the dissimilarities of sites.

A SIMPER analysis was performed to determine which species influence the dissimilarities among the estuaries for the CAMP dataset. The results of the SIMPER analysis (Table 2.13) reveal the dissimilarities between the estuaries are governed by several abundant species. The four species that are most influential in defining the differences between groups of sites are adult Mummichog, Sand Shrimp, and Fourspine Stickleback (*Apeltes quadracus*), and YOY Atlantic Silversides. Figure 2.10 displays the abundance of these species among the sites. One-way ANOVAs were completed for the counts of each of the influential species (Table S2.1). Souris data were excluded from the ANOVAs, because only five CAMP stations were sampled for that site. The information from the one-way ANOVAs was used to calculate how the variance of the mean and confidence intervals shrink with increasing stations numbers (Tables S2.2). The average differences in abundance of influential species that contributed to a minimum of 10% of the dissimilarity between sites (Table S2.3) were used to assess the desired confidence interval. The desired confidence interval was determined to be 50% (+/-) of the average difference in counts between sites, as a confidence interval of that size would prevent overlap of the group means.

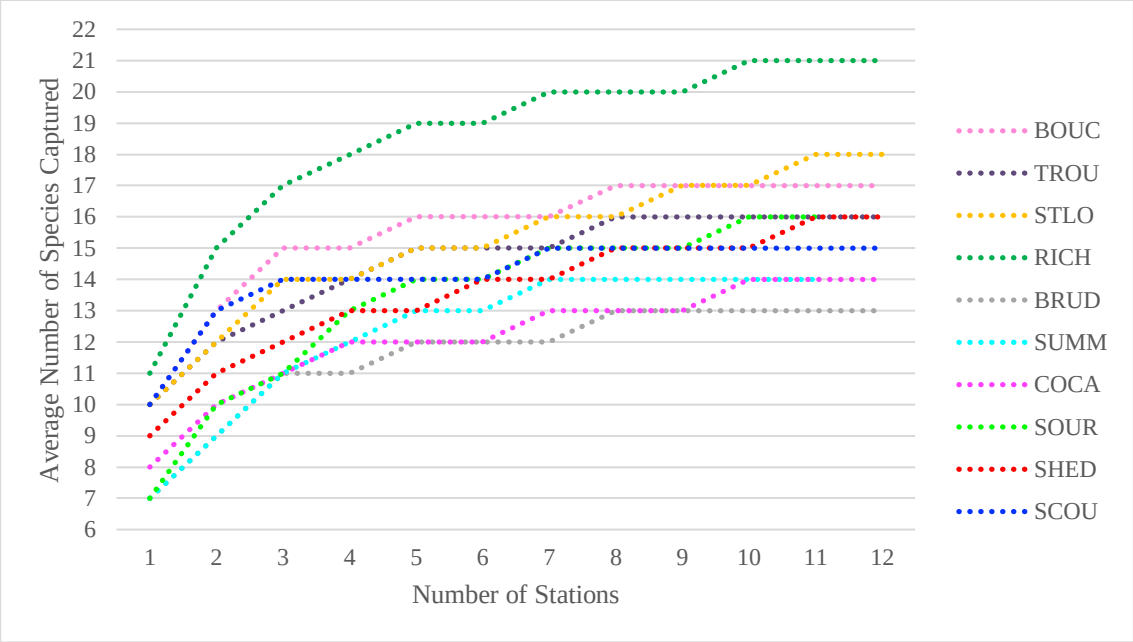

Figure 2.8 Species accumulation plot generated using the combined nekton data collected from the Community Aquatic Monitoring Program (CAMP) and Stratified Random Design (SRD) stations at each site. The plot displays the average number of new species detected with each increasing station number as defined by 9999 permutations of station data.

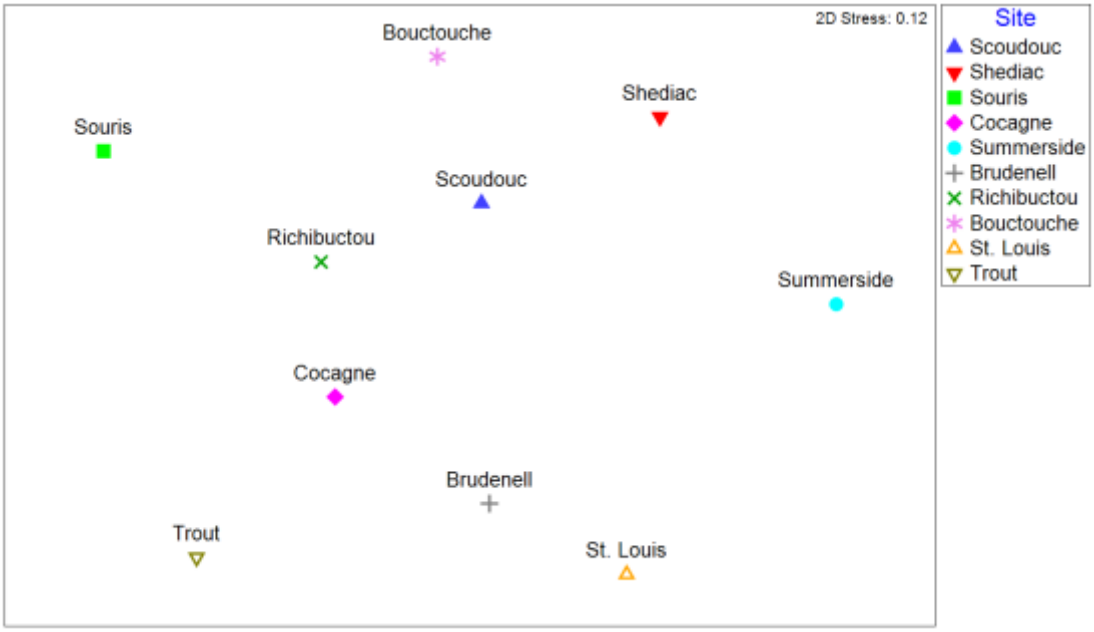

Figure 2.9 non-metric Multidimensional Scaling ordination plot of square-root transformed nekton data collected from Community Aquatic Monitoring Program (CAMP) and Stratified Random Design (SRD) stations. Nekton data are averages of combined station data for each site.

Table 2.10 One-way permutational-MANOVA (PERMANOVA) results for the analysis of the combined data set of data collected from the Community Aquatic Monitoring Program (CAMP) and Stratified Random Design (SRD) stations nekton community data using factor Site.

Source	d.f.	*MS*	*F*	*P*
Site	9	15448	12.57	0.0001
Residuals	108	1229		
Total	117			

Table 2.11 Standard errors generated by separate PERMDISP tests with group factor Site on the Community Aquatic Monitoring Program (CAMP) and combined dataset (data collected from the CAMP and Stratified Random Design [SRD] stations).

Site	Standard Error (SE)	
	CAMP	CAMP + SRD
Scoudouc	2.32	2.80
Shediac	2.38	2.00
Souris	0.97	1.21
Cocagne	3.22	2.62
Summerside	6.1	4.0
Brudenell	2.39	2.02
Richibucto	2.77	1.78
Bouctouche	3.55	1.33
St. Louis de Kent	3.77	3.13
Trout River	2.73	1.77

Table 2.12 Pair-wise test of homogeneity of dispersion (PERMDISP) among factor Sampling Design within factor Site for the data collected from the Community Aquatic Monitoring Program (CAMP) stations and the combined dataset (data collected from CAMP and the Stratified Random Design [SRD] stations).

Site	Sampling Design	*t*	*P*
Scoudouc	CAMP, Combined	0.988	0.4057
Shediac	CAMP, Combined	1.158	0.3278
Souris	CAMP, Combined	1.318	0.3023
Cocagne	CAMP, Combined	1.949	0.1370
Summerside	CAMP, Combined	0.132	0.9233
Brudenell	CAMP, Combined	0.6074	0.6181
Richibucto	CAMP, Combined	0.279	0.8183
Bouctouche	CAMP, Combined	0.786	0.5086
St. Louis de Kent	CAMP, Combined	0.755	0.5486
Trout River	CAMP, Combined	1.047	0.3700

Table 2.13 Results of the similarity percentages routine (SIMPER) measuring the contribution of each species to the dissimilarities between sites as defined by the Bray-Curtis dissimilarity between each pair of sites based on nekton data collected from the Community Aquatic Monitoring Program (CAMP) stations.

Species	Average % Contribution to Dissimilarity
Sand Shrimp (Adult)	18
Atlantic Silverside (YOY)	13
Mummichog (Adult)	12
Fourspine Stickleback (Adult)	11
Sand Shrimp (YOY)	8
Grass Shrimp (Adult)	6
Black Spotted Stickleback (Adult)	4
Three Spine Stickleback (Adult)	4
Killifish (Adult)	3
Green Crab (Adult)	3
Fundulus (YOY)	3
Atlantic Silverside (Adult)	3

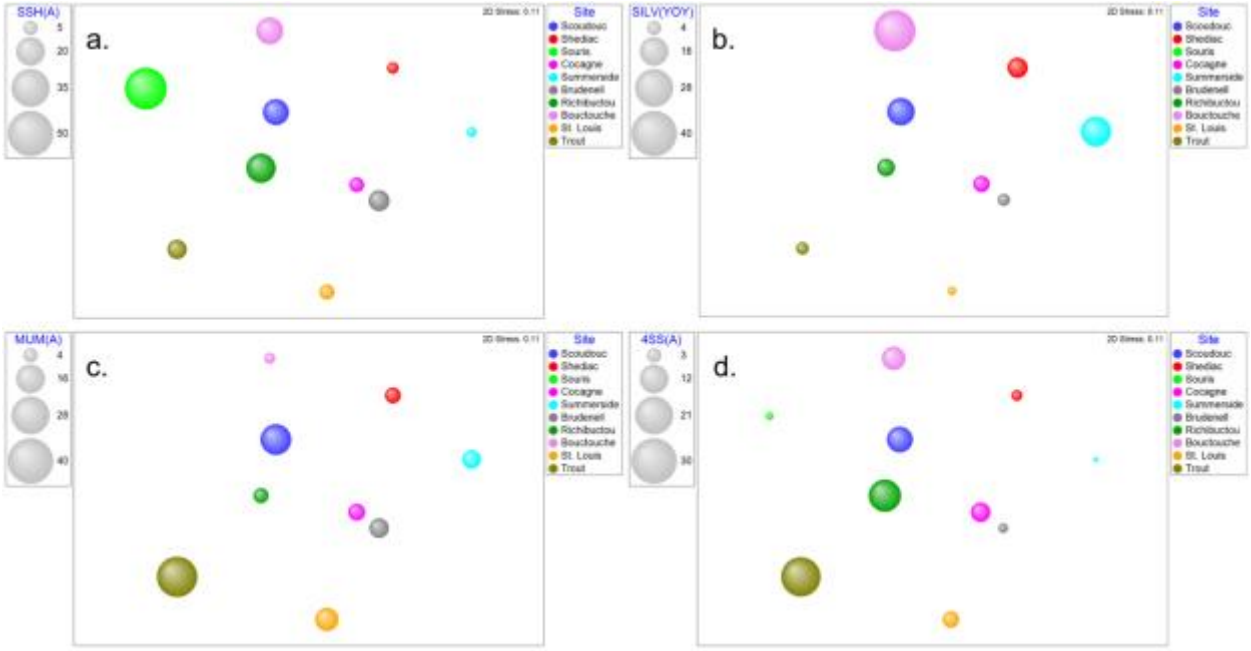

Figure 2.10 non-metric Multidimensional Scaling ordination plots of square-root transformed Community Aquatic Monitoring Program (CAMP) nekton data averaged over sample design within each site, overlaid with bubble plots depicting species abundances. a: abundance of Sand Shrimp adults, b: abundance of Atlantic Silverside YOYs, c: abundance of Mummichog adults, d: abundance of Fourspine Stickleback adults.

The average difference in adult Mummichog counts between groups ranged from 11 to 994. The current six CAMP stations have the precision to detect a difference of +/- 276 adult Mummichogs, which is adequate to detect the larger differences of ≥661. However, even twelve stations would not be sufficient to detect the smaller differences of ≤333 (Figure 2.11).

The average difference in adult Sand Shrimp counts between groups ranged from 21 to 1785. The current six CAMP stations have the precision to detect a difference of +/- 133 adult Sand Shrimp, which is adequate to detect the larger differences of ≥283. However, seven stations would be required to detect the difference of 260, seven stations for the differences of 256, eight stations for 239, nine for 230, and even twelve stations would be insufficient to detect the smaller differences of ≤163 (Figure 2.12).

The average difference in YOY Atlantic Silverside counts between groups ranged from 18 to 1096. The current six CAMP stations have the precision to detect a difference of +/- 303 YOY Atlantic Silversides, which is adequate to detect the larger differences of ≥787. However, even twelve stations would not be sufficient to detect the smaller differences of ≤309 (Figure 2.13).

The average difference in adult Fourspine Stickleback counts between groups ranged from 14 to 505. The current six CAMP stations have the precision to detect a difference of +/- 143, which would be sufficient to detect the differences ≥411. However, even twelve would not be sufficient to detect the smaller differences of ≤228 (Figure 2.14).

Overall, the results suggest adding up to six stations to CAMP would not greatly increase the precision of count estimates for any of the influential species. The difference between the confidence interval of six stations versus the confidence interval of twelve stations (72 adult Mummichog, 23 adult Fourspine Stickleback, 26 adult Sand Shrimp, and 57 YOY Atlantic Silversides) is small when considering the typical variability in counts of these species within sites. For example, adult Mummichog counts collected in Scoudouc ranged from 8 to 1200. The variability in catch size of these species is in part due to their schooling behaviour. Thus, a difference in 72 Mummichogs likely does not signify a change in environmental conditions. Much larger differences in counts of these species would be considered biologically significant. Therefore, the results do not provide compelling evidence to suggest more stations should be added to CAMP sites.

The methods employed in this study to address questions regarding the adequacy of station numbers are not ideal. The SRD stations are not truly random, since they are stratified. A study with randomly assigned stations throughout the estuaries would provide a stronger assessment of the potential effects of adding stations to CAMP. Ultimately, studies should oversample using the same methodology to determine at which sample number the variability within the data is reduced to a desirable level.

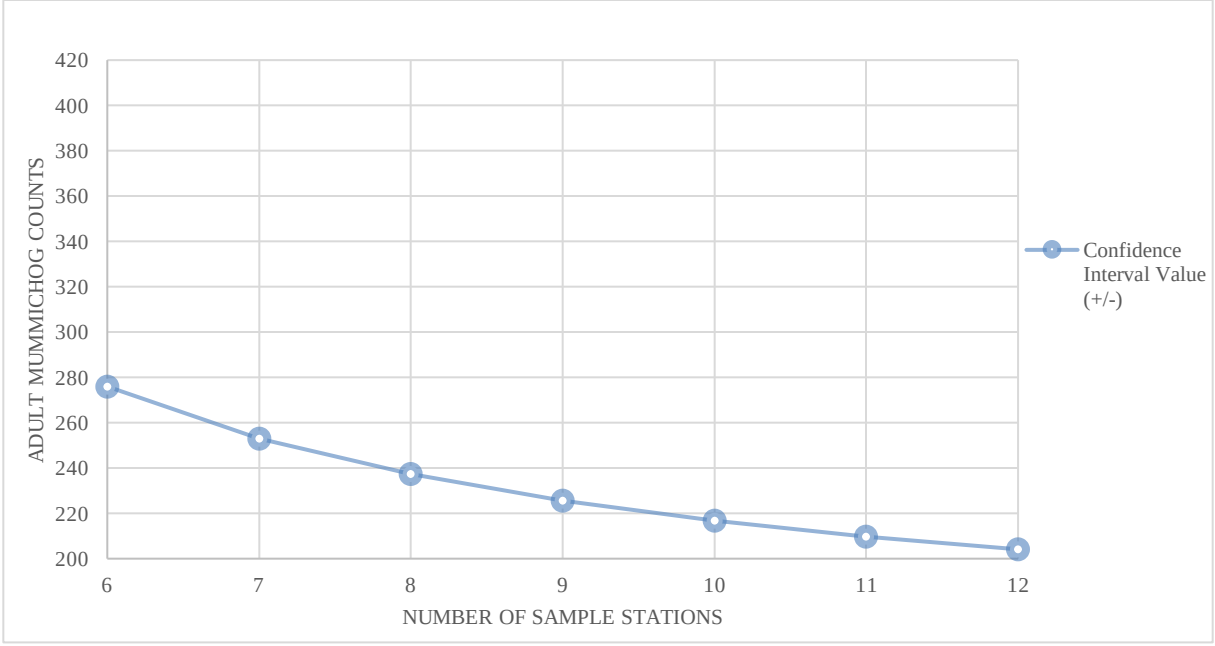

Figure 2.11 Confidence interval values (+/-) of estimates of adult Mummichog abundances calculated using Mummichog data collected from the Community Aquatic Monitoring Program (CAMP) stations.

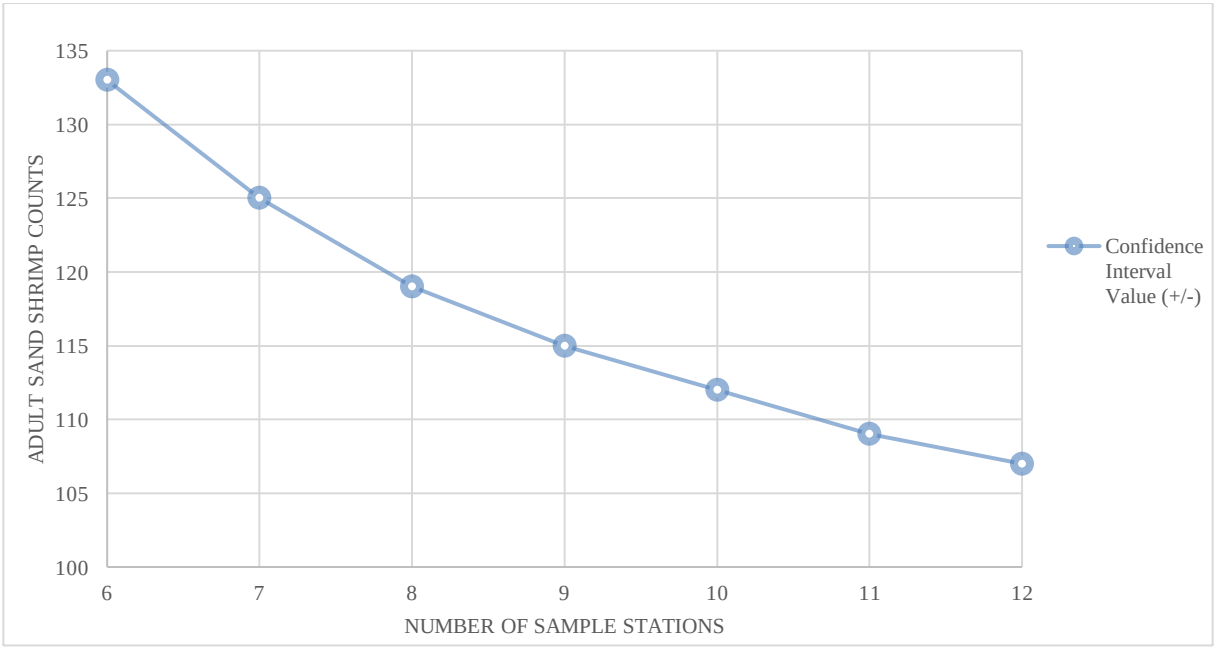

Figure 2.12 Confidence interval values (+/-) of estimates of adult Sand Shrimp abundances calculated using Sand Shrimp data collected from the Community Aquatic Monitoring Program (CAMP) stations.

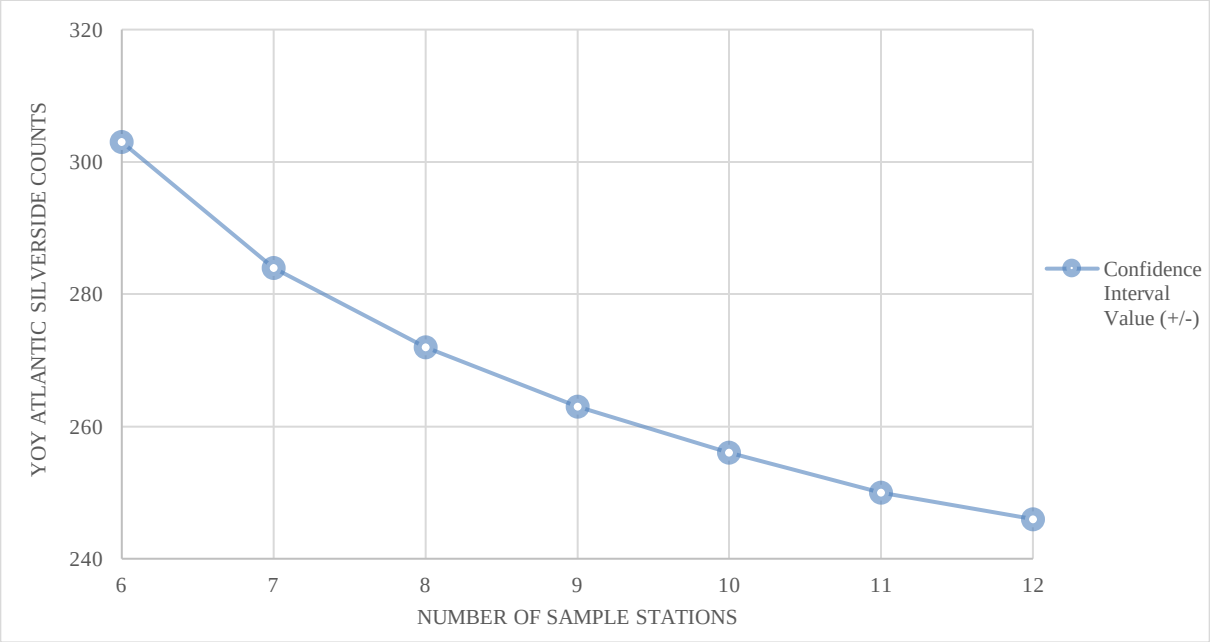

Figure 2.13 Confidence interval values (+/-) of estimates of YOY Atlantic Silverside abundances calculated using YOY Atlantic Silverside data collected from the Community Aquatic Monitoring Program (CAMP) stations.

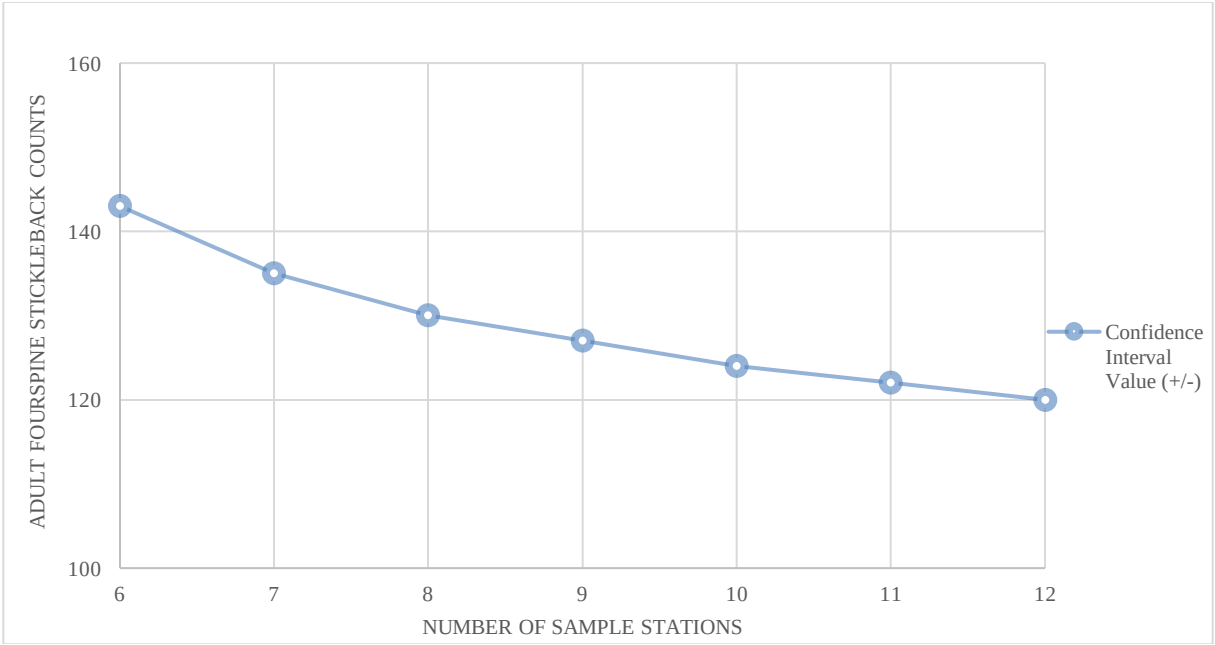

Figure 2.14 Confidence interval values (+/-) of estimates of adult Fourspine Stickleback abundances calculated using adult Fourspine Stickleback data collected from the Community Aquatic Monitoring Program (CAMP) stations.

2.4 Conclusions and recommendations

The results of this study indicate that if estuary health is to be assessed based on nekton community composition then the application of CAMP is not limited by station selection bias. Furthermore, this study demonstrates monitoring programs designed to accommodate volunteers can produce data comparable to scientific studies. There is no evidence that increasing the number of CAMP stations would significantly increase precision of the description of nekton community or influential species counts. It is clear that the nekton communities are highly variable within sites, but this variability will not be significantly reduced by relocating or adding stations. Also, the variability of the nekton community within estuaries does not appear to be dependent on estuary size. The variability within an estuary may be more dependent on the relative degree of the patchiness of the habitat within each site. Sampling within a smaller standardized range of salinity or within a defined tidal range may help reduce variability within nekton data. More research is needed to determine how best to reduce the variability of estimates of nekton abundances within estuaries in the sGSL.

Subsequently, the CAMP dataset needs to be explored to determine if littoral nekton assemblages are reflective of the health of an estuary and how they are influenced by environmental stressors. The results would assist CAMP managers in determining the utility of the program and its future objectives. A critical component will be collecting updated land use data for each watershed. The land use data can be analysed along with the nekton community data to determine if there are correlations between the clustering of sites based on nekton assemblages, and the type and degree of surrounding anthropogenic influences. The land use component will clarify which of the influential species correlate with heavily influenced conditions. A suite of indicator species whose relative densities signal either healthy or degraded conditions for estuaries in the sGSL can then be developed. If large differences between the abundance of indicator species distinguish degraded sites, then CAMP is an appropriate tool for managers to assess estuarine health and can provide useful information to decision-makers as to which estuaries should be the focus of management and restoration initiatives. However, if managers of CAMP decide to assess estuary health by only using a suite of indicator species rather than community composition, then the adequacy of station locations to detect those indicator species should be reassessed.

Chapter 3
Conclusions and recommendations

This thesis is comprised of research that assessed the sampling design of the Community Aquatic Monitoring Program (CAMP). Initially designed as a stewardship initiative, CAMP has developed into a long-term monitoring program that collects annual biological and environmental data from up to 36 estuaries in the southern Gulf of St. Lawrence (sGSL). The principal research objective was to assess whether CAMP is limited in its scientific application due to a sampling design that facilitates community member involvement. The two aspects of the CAMP sampling design that were tested were the placement of stations within a site, and the number of stations sampled.

The nekton assemblage data collected from CAMP stations were compared to data collected from stations located using a stratified random design (SRD). The purpose was to test the effect of CAMP station locations being selected based on easy road access for volunteers. In general, the nekton assemblages for each estuary did not differ significantly between the two sampling designs. The variability of the nekton data was also not significantly different among sampling designs. Both sampling designs yielded results that suggest all sites contain significantly different littoral nekton assemblages. Therefore, the conclusion that each estuary is dissimilar based on littoral nekton assemblages would remain the same regardless of the sampling design employed. These results suggest CAMP is not limited in its scientific application due to a bias in the location of stations, and there is no evidence to suggest existing stations should be relocated.

Several approaches were taken to examine the question of whether six stations provide an adequate estimate of nekton community composition and relative abundances of species within each estuary, to permit discrimination of estuaries of different environmental quality. Species accumulation plots were generated, using the data from the six CAMP and SRD stations separately, to determine if the six stations sufficiently capture the characteristic nekton species of each estuary. Secondly, the magnitude of the differences in nekton assemblages between estuaries that are desirable to detect in order to distinguish estuaries, and how many sampling stations are required to detect these differences were considered. This was accomplished by examining mean differences in abundance of those species that contributed most to differences between sites. The optimal number of stations is the minimum that provides sufficient precision to discriminate among nekton assemblages characterizing estuaries of suspected different environmental quality. Finally, the data collected from the CAMP and SRD stations were combined to consider the potential influence of increasing the number of stations on the differences

among sites. Overall, the results of these analyses indicate the current six CAMP stations are sufficient to detect the species that are characteristic of each estuary, and increasing the number of stations will not increase the precision in a biologically meaningful way. Combining the data collected from the two sampling designs did not alter the perception that all estuaries are different. Therefore, the results indicate the precision gained by increasing the number of stations would likely not alter management decisions that are based on differences in littoral nekton assemblages.

These research findings support the validity of the CAMP sampling design, which will give credence to any past, present and future studies completed using the CAMP dataset to assess littoral nekton assemblages. This study has implications for the future of CAMP, as the conclusions of this research will help managers of CAMP decide if the program is to continue in its current form and if it will be implemented as a tool to assess estuarine health and inform management decisions. Accordingly, these findings may have great implications for the numerous community members who annually participate in CAMP.

In broader terms, this research will help fill a current knowledge gap in the study of community based monitoring programs by demonstrating the utility of these programs. As such, this work may have implications for future monitoring programs that wish to involve community members in data collection. This research also contributes to the overall study of fish-based estuarine indices as it advances the understanding of the effect of sampling design and effort on estimates of nekton abundances and community composition. Specifically, these findings are applicable to furthering research in assessing the condition of shallow, well-mixed, temperate estuaries. Recommended factors to consider in the design and execution of estuary monitoring programs and future considerations for CAMP are presented below.

3.1 Recommendations for the design of estuary monitoring programs

3.1.1 Consideration of salinity in sampling designs

Salinity is known to be one of the most influential contributing factors to the variability of nekton communities within estuaries (Marshall and Elliott 1998; Jaureguizar et al. 2004; Araujo 2017). As such, monitoring programs that employ nekton as indicators should consider salinity in their design. The objective of station stratification in this study was to capture the range of salinity between 10 and 30 PPT. It is important to sample within a standardized range of salinity in order to compare nekton communities among estuaries and distinguish the influence of an anthropogenic effect from natural variability (e.g., Araujo et al. 2017).

The results of this study indicate that the samples collected from the upper estuary do not differ significantly from those collected in the lower estuary, which would suggest station stratification is unnecessary in these estuaries. The estuaries sampled are generally shallow and well-mixed; hence, these estuaries lack the steep salinity gradient characteristic of deeper estuaries with a higher degree of vertical stratification of the water column. These well-mixed estuaries are characterised by a gradual change in salinity throughout the lower and middle reaches and a sudden drop in the upper reach where the range of 0-15 PPT encompasses only a small area (Coffin et al. 2017). Only one of the SRD stations in the present study had a salinity value less than 15 PPT. Future research assessing the influence of the salinity gradient in these well-mixed estuaries should delineate the estuaries with greater precision by deploying salinity probes prior to sampling to better define where the salinity gradient drops to 10 PPT. As well, data loggers moored at each station would be helpful to describe the variance in salinity (and potentially other water characteristics such as temperature) associated with the diurnal and lunar tidal cycles and seasonal variance in river freshwater discharge.

3.1.2 Station stratification

The results of this research indicate there is no need for station stratification in the estuaries sampled, including instances where CAMP stations are clustered in the lower estuary. Nonetheless, if CAMP is to be used as a tool to assess the health of estuaries, it is logical to sample stations in the upper estuary to provide an early warning sign of contamination. Previous studies have demonstrated the effects of degradation in estuaries are most severe and/or linger in the upper estuary. The upper estuary tends to have an increase in pollutant concentrations due to the low dilution of fresh water by marine water (Araujo 2017). Liang et al. (2013) found concentrations of antibiotics in estuarine substrate to be the greatest in the upper estuary. Dense mats of macrophytes are typically found in the upper estuary at sites exposed to high nutrient inputs (Coffin et al. 2017). Staehr et al. (2017) found nutrient concentrations remained above threshold levels in the upper estuary several years after implementation of management initiatives to greatly reduce nutrient inputs. During the 2016 sampling program, eutrophic conditions, in the form of dense mats of macroalgae, were noted to be most severe at the upper estuary stations. Therefore, managers of CAMP should consider adding new stations, or relocating an existing station, to the upper estuary (10-15 PPT) at sites where stations are clustered in the lower reaches to provide an early indication of deleterious effects from anthropogenic influences upriver.

3.1.3 Consideration of tides in the execution of estuary monitoring programs

Like salinity, tides are known to influence the distribution of nekton communities within an estuary (Gibson 2003; Wilcockson and Zhang 2008; Krumme 2009). However, the variability introduced by the tides fluctuates at a finer temporal scale than salinity. Hence, while salinity needs to be considered in the design of a monitoring program, the tidal influence must be accounted for in its execution. So, sampling programs that incorporate nekton as indicators should coordinate daily sampling schedules with consideration of a standardized or minimum tide height.

The field crew for this study met at 8:00 AM every day. However, the first day of sampling was delayed and resulted in an average difference in tide height of 0.6 m between the Cocagne sampling designs. Although CAMP maintains consistent sampling start times and attempts to sample each site on the same day of the month each year, it does not guarantee consistent tide heights for each sampling event. The tidal range fluctuates daily, monthly, and annually. For example, if Summerside were to be sampled from 8:00 AM to 12:00 PM on July 1, 2016, the tidal range would be 1.2 to 2.0 m; conversely, if Summerside were to be sampled at the same time and date in 2017 the projected tide height is 0.8 to 1.6 (DFO n.d.). Consequently, there would be an average difference of 0.4 m in tide height between the annual sampling events. Substantive differences in tidal range among sampling events could increase the natural variability influencing the data that could result in false conclusions of altered nekton assemblages. Standardizing the tide height may help control for some of the natural variability. Hence, efforts should also be made to sample within a consistent tidal range.

The difference in tide height that appears to have generated a difference in nekton assemblages in sampling of Cocagne estuary in the present study is 0.6 m, and it appears from sampling of other sites that an average difference of 0.1 m is acceptable. Further research is required to test for the maximum difference in tide height that will not result in significant differences between nekton assemblages. Such research could be accomplished by repeatedly sampling one station throughout a day to assess how the captured nekton assemblages change as the tides rise and fall. The results could be used to define a maximum range in tide height that does not result in significantly different nekton samples. Then each estuary can be evaluated based on its typical tidal range to determine how much time samplers have to collect data. Once an appropriate sampling timeframe is established then the number of stations a monitoring program should sample can be addressed.

3.1.4 The sufficient number of sampling stations for monitoring estuaries

Six stations were originally proposed for each CAMP site, because that was the number of stations that volunteers were predicted to be able to sample within one day (DFO 2011). CAMP sampling is typically completed in approximately four hours, so there are more hours in the day to sample additional stations if deemed necessary. However, while sampling more stations might increase the precision of nekton estimates it would also involve greater effort and more time, which could introduce more variance associated with tide heights. The consideration of tide height limits the time available for sampling each day and consequently will limit the number of stations that can be sampled.

Regardless of tidal range consideration, the results of this study indicate increasing the number of stations will not substantively increase the precision of CAMP. These results were obtained through statistical techniques. A more effective method would have been to over-sample each site using the same sampling design to sample all stations. If twelve SRD stations had been sampled, then the six SRD stations and the full SRD dataset could have been compared to conclude if within-site variability is significantly reduced with increased station numbers. Future monitoring programs should over-sample within a specified tidal range to define the optimal number of stations for each site.

3.1.5 Reducing variability in data collected for estuary monitoring

The high variability of nekton assemblages has been reported in previous studies as a difficulty in using nekton as an indicator (Ellis and Bell 2013). Based on the results of this study, altering the spread of locations throughout the estuary or increasing the number of stations does not necessarily lead to either an increase or decrease in within site variability. Since station number and placement do not appear to reduce variability, other factors that can be controlled must be considered; perhaps sampling within a smaller range of salinity (e.g., 15-25 PPT) within each site, sampling at a consistent tide height, or within a specific habitat, such as eelgrass beds. These methods all require thorough knowledge of the habitat of each estuary and proper planning, preparation and execution of sampling events. A good method to employ before designing an estuary monitoring program may be to develop a comprehensive habitat map of each estuary to be monitored.

3.2 Recommendations for Future CAMP research

The initial objective of CAMP was to provide an avenue for DFO community outreach and interaction with Environmental Non-Government Organizations. The involvement of over 29 watershed groups throughout the past 14 years clearly demonstrates the program has succeeded in its initial objective. Since

this thesis has tested the concerns regarding the CAMP sampling design, questions about the utility of CAMP as a scientific decision making tool can now be addressed. A goal for CAMP is to use it to assess estuarine health using nekton assemblages as an indicator (Thériault and Courtenay 2010; DFO 2011). Accordingly, the question to ask of the CAMP dataset is: is there a relationship between the degree and type of anthropogenic activities influencing an estuary and its nekton assemblages?

A study by Ellis and Bell (2013) using nekton assemblages as indicators of mangrove removal concluded that nekton assemblages may not be a sensitive indicator of environmental degradation and cited several other studies with similar conclusions. However, these studies were relatively short-term, and other studies have highlighted the benefits of nekton as indicators (Whitfield and Elliott 2002; Harrison and Whitfield 2004). The long-term dataset of CAMP presents a unique opportunity to assess the natural variability of these nekton communities in order to discern changes triggered by anthropogenic influences. The next step in defining the utility of CAMP is the compilation of updated land use data that can be analysed to determine if sites can be differentiated based on their nekton assemblages, and these differences related to a gradient of anthropogenic influences. If so, then there will be validation of the use of littoral nekton assemblages as an indicator of estuarine health and the use of CAMP as an adequate vehicle for the assessment of estuarine health.

If the assessment of the entire CAMP dataset finds littoral nekton assemblages are not a sensitive indicator of anthropogenic influences, then alternative indicators should be explored. In this study, Mummichog were highlighted as one of the most influential species that define the dissimilarities between sites. Finley et al. (2013) concluded that Mummichog abundances have the potential to be a good indicator for estuaries in the sGSL. Mummichog have also been used as an indicator species to assess the influence of pollutants from pulp and paper mills. However, these studies measured reproductive parameters, including gonad size and hormones, rather than estimates of abundance (Leblanc et al. 1997; Dubé and MacLatchy 2001). Other studies have found growth and body condition of indicator species are sensitive indicators of anthropogenic influences (Valesini et al. 2017). The weight of individual fish was not measured in this study, but visual assessments of fish condition did suggest that sites experiencing eutrophic conditions (e.g., Trout River) contained larger Mummichogs on average than those collected elsewhere. So, if Mummichogs are considered as an indicator species in the sGSL, then further research is required to assess which parameters (e.g., abundance or body condition) would be most appropriate to measure.

Ultimately, more research is required to determine which species would be the most appropriate indicators for assessing the health of estuaries in the sGSL. Once candidate indicator species are

established, the dataset of this study can be reassessed for the suitability of CAMP stations in sampling those particular species. If results indicate CAMP stations are inadequate for use in monitoring indicator species, then it can either be re-designed or continue to collect community data with the objective of assessing changes within rather than among estuaries. Nekton assemblages have been suggested as better indicators of estuarine health within an estuary rather than among estuaries (Sheaves and Johnston 2012).

In conclusion, the research completed for this thesis contributes to the assessment of the CAMP sampling design. The overall conclusion of this research is that CAMP is not limited in its scientific application due to station placement or the number of stations sampled. This study demonstrates the difficulty in estuary monitoring resulting from the inherent variability of biotic communities within estuaries. Variability within estuary monitoring data may be reduced by sampling within a standardized range of salinity, a standardized range in tide height, or specific habitat type. While results suggest there is no need for CAMP to add or relocate stations, it would be logical to either add or relocate a station to the upper estuary in sites where stations are clustered in the lower estuary. Stations located in the upper estuary may provide an early warning sign of watershed impact on the estuary. Finally, this study demonstrates monitoring programs designed to accommodate volunteers can collect data that can be contributed to scientific studies. The conclusions of this research can be used to motivate other government agencies to implement similar monitoring programs that both engage the local community and produce data to inform management decisions.

Bibliography

Adam, P. 2002. Saltmarshes in a time of change. *Environmental Conservation* 29 (1) 39-61.

Aguilar, C., G. Gonzalez-Sanson, K. R. Munkittrick, and D. L. MacLatchy. 2004. Fish assemblages on fringe coral reefs of the northern coast of Cuba near Havana Harbor. *Ecotoxicology and Environmental Safety* 58: 126-138.

Anderson, M. J., R. N. Gorley, and K. R. Clarke. 2008. *PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods*. PRIMER-E: Plymouth, UK.

Araujo, A. V., C. O. Dias, and S. L. C. Bonecker. 2017. Differences in the structure of copepod assemblages in four tropical estuaries: Importance of pollution and the estuary hydrodynamics. *Marine Pollution Bulletin* 115: 412-420.

Attrill, M. J., and M. H. Depledge. 1997. Community and population indicators of ecosystem health: targeting links between levels of biological organisation. *Aquatic Toxicology* 38: 183-197.

Bailey C., and J. Byrnes. 1990. A new, old method for assessing measurement error. *Systematic Zoology* 39 (2): 124-130.

Barbier, E. B., S. D. Hacker, C. Kennedy, E. W. Koch, A. C. Stier, and B. R. Silliman. 2011. The value of estuarine and coastal ecosystem services. *Ecological Monographs* 81 (2): 169-193.

Blaber, S. J. M., D. P. Cyrus, J.-J. Albaret, C. V. Ching, J. W. Day, M. Elliott, M. S. Fonesca, D. E. Hoss, J. Orensanz, I. C. Potter, and W. Silvert. 2000. Effects of fishing on the structure and functioning of estuarine and nearshore ecosystems. *ICES Journal of Marine Sciences* 57: 590-602.

Borja, A., and D. M. Dauer. 2008. Assessing the environmental quality status in estuarine and coastal systems: Comparing methodologies and indices. *Ecological Indicators* 8: 331-337.

Bowen, J. L., and I. Valiela. 2001. The ecological effects of urbanization of coastal watersheds: Historical increases in nitrogen loads and eutrophication of Waquoit Bay estuaries. *Canadian Journal of Fisheries and Aquatic Sciences* 58 (8) 1489-1500.

Bricker, S. B., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner. 2008. Effects of nutrient enrichment in the nation's estuaries: A decade of change. *Harmful Algae* 8: 21-32.

Butler, M., R. D. Chiasson, R. W. Daury, S. Dean, S. B. Dietz, N. MacKinnon, and J. Steel. 1996. Module 3: Estuaries. In *By the Sea: A Guide to the Coastal Zone of Atlantic Canada*. Tabusintac: Corvus Consultants Inc.

Castellanos-Galindo, G. A., and U. Krumme. 2015. Tides, salinity, and biogeography affect fish assemblage structure and function in macrotidal mangroves of the neotropics. *Ecosystems* 18: 1165-1178.

Chariton, A. A., S. Stephenson, M. J. Morgan, A. D. L. Steven, M. J. Colloff, L. N. Court, and C. M. Hardy. 2015. Metabarcoding of benthic eukaryote communities predicts the ecological condition of estuaries. *Environmental Pollution* 203: 165-174.

Clarke, K. R., R. N. Gorley, P. J. Somerfield, and R. M. Warwick. 2014. *Change in marine communities: an approach to statistical analysis and interpretation.* 3rd ed. PRIMER-E; Plymouth, UK.

Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223-253.

Coffin, M. R. S., K. M. Knysh, E. F. Theriault, C. C. Pater, S. C. Courtenay, and M. R. van den Heuvel. 2017. Are floating algal mats a refuge form hypoxia for estuarine invertebrates? *Peer J* DOI 10.7717/peerj.3080.

Cohen, A. N., and J. T. Carlton. 1998. Accelerating invasion rate in a highly invaded estuary. *Science* 279 (5350): 555-558.

Conrad, C., and K. Hilchey K. 2011. A review of citizen science and community-based environmental monitoring: issues and opportunities. *Environmental Monitoring and Assessment* 176: 273-291.

Costalago, D., N. Strydom, C. Frost, and C. Clemmesen. 2015. Preliminary insight into the relationship between environmental factors and the nutritional condition and growth of *Gilchristella aestuaria* larvae in the upper reaches of South African estuaries. *Environmental Biology of Fishes* 98: 2367-2378.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253-260.

Costanza, R., O. Perez-Maqueo, M. L. Martinez, P. Sutton, S. J. Anderson, and K. Mulder. 2008. The value of coastal wetlands for hurricane protection. *AMBIO: A Journal of the Human Environment* 37 (4): 241-248.

Dafforn, K. A., S. L. Simpson, B. P. Kelaher, G. F. Clark, V. Komyakova, C. K. C. Wong, and E. L. Johnston. 2012. The challenge of choosing environmental indicators of anthropogenic impacts in estuaries. *Environmental Pollution* 163: 207-217.

Danielsen, F., P. M. Jensen, N. D. Burgess, R. Altamirano, P. A. Alviola, H. Andrianndrasana, J. S. Brashares, A. C. Burton, I. Coronado, N. Corpuz, M. Enghoff, J. Fjeldsa, M. Funder, S. Holt, H. Hubertz, A. E. Jensen, R. Lewis, J. Massao, M. M. Mendoza, Y. Ngaga, C.B. Pipper, M. K. Poulsen, R. M. Rueda, M. K. Sam, T Skielboe, M. Sorensen, and R. Young. 2014. A multicountry assessment of tropical resource monitoring. *BioScience* 64: 236 251.

Davidson, N. C., D. d'A Laffoley, J. P. Doody, L. S. Way, J. Gordon, R. Key, M. W. Pienkowski, R. Mitchell, and K. L. Duff. 1991. Introduction. Chap. 1 in *Nature Conservation and Estuaries in Great Britain*. Peterborough: Nature Conservancy Council.

Deeley, D. M., and E. I. Paling. 1999. *Assessing the ecological health of estuaries in Australia*. Canberra: Land and Water Resources Research and Development Corporation.

Department of Fisheries and Oceans Canada (DFO). n.d. 7 day tidal predictions *Fisheries and Oceans Canada*.
http://www.waterlevels.gc.ca/eng/find/zone/4 (accessed January 20, 2017)

Department of Fisheries and Oceans Canada (DFO). 2002. Canada's Oceans Strategy: Policy and Operational Framework for Integrated Management of Estuarine, Coastal and Marine Environments in Canada. *DFO Oceans Directorate*. Cat. No. Fs77-2/2002E.

Department of Fisheries and Oceans Canada (DFO). 2011. Proceedings of a Regional Advisory Process to Review the Community Aquatic Monitoring Program (CAMP) and its Use to Infer the Ecological Health of Bays and Estuaries in the Southern Gulf of St. Lawrence, March17-18, 2010. *DFO Canadian Scientific Advisory Secretariat Proceed. Ser.* 2011/029.

Duarte, C. M. 2002. The future of seagrass meadows. *Environmental Conservation* 29 (2): 192-206.

Duarte, C. M. I. J. Losada, I. E. Hendriks, I. Mazzarrasa, and N. Marba. 2013. The role of coastal plant communities for climate change mitigation and adaptation. *Nature Climate Change* 3: 961-967.

Dubé, M., and D. L. MacLatchy. 2001. Identification and treatment of a waste stream at a bleached-kraft pulp mill that depresses a sex steroid in the mummichog (*Fundulus heteroclitus*). *Environmental Toxicology* 20 (5): 985-995.

Dytham, C. 2011. *Choosing and using statistics: A Biologist's guide*. 3rd ed., 26-27. West Sussex, UK: John Wiley & Sons Ltd.

Elliott, M. and D. S. McLusky. 2002. The Need for Definitions in Understanding Estuaries. *Estuarine, Coastal and Shelf Science* 55: 815-827.

Elliott, M. and V. Quintino. 2007. The Estuarine Quality Paradox, Environmental Homeostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. *Marine Pollution Bulletin* 54: 640-645.

Ellis, W. L., and S. S. Bell. 2013. Intertidal fish communities may make poor indicators of environmental quality: Lessons from a study of mangrove mangrove habitat modification. *Ecological Indicators* 24: 421-430.

Ellis, J. I., J. E. Hewitt, D. Clark, C. Taiapa, M. Patterson, J. Sinner, D. Hardy, and S. F. Thrush. 2015. Assessing ecological community health in coastal estuarine systems impacted by multiple stressors. *Journal of Experimental Marine Biology and Ecology* 473: 176 187.

Environmental Systems Research Institute (ESRI) (2015). ArcGIS 10.4.1

European Communities. 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. *Official Journal of the European Communities* L327: 1-73.

Fernandez-Gimenez, M. E., H. Ballard, and V. E. Sturtevant. 2008. Adaptive management and social learning in collaborative and community-based Monitoring: A study of give community-based forestry organizations in western USA. *Ecology and Society* 13 (2): 4.

Finley, M. A. 2008. Investigating the suitability of Northern Mummichog (*Fundulus heteroclitus Macrolepidotus*) for the assessment of cumulative and non-point source pollution in Prince Edward Island Estuaries. Master of Science thesis, University of Prince Edward Island.

Finley, M. A., S. C. Courtenay, K. L. Teather, L. M. Hewitt, D. A. Holdway, N. S. Hogan, and M. R. van den Heuvel. 2013. Evaluating cumulative effects of anthropogenic inputs in Prince Edward Island estuaries using the Mummichog (*Fundulus Heteroclitus*). 2013. *Integrated Environmental Assessment and Management* 9 (3): 496-507.

Fore, L. S., K. Paulsen and K. O' Laughlin. 2001. Assessing the performance of volunteers in monitoring streams. *Freshwater Biology* 46: 109-123.

Forrester, G., P. Bailey, D. Conetta, L. Forrester, E. Kintzing, and L. Jarecki. 2015. Comparing monitoring data collected by volunteers and professionals shows that citizen scientists can detect long-term change on coral reefs. *Journal for Nature Conservation* 24: 1-9.

Fulton, M. H., D. W. Moore, E. F. Wirth, G. T. Chandler, P. B. Key, J. W. Daugomah, E. D. Strozier, J. Devane, J. R. Clark, M. A. Lewis, D. B. Finley, W. Ellenberg, K. J. Karnacky Jr., and G. I. Scott. 1999. Assessment of risk reduction strategies for the management of agricultural nonpoint source pesticide runoff in estuarine ecosystems. *Toxicology and Industrial Health* 15: 201-214.

Gardner Pinfold. 2013. *Socio-Economic Impact of Aquaculture in Canada: 2013 Edition*. Report prepared for Fisheries and Oceans Canada, Aquaculture Management Directorate. Halifax: Gardner Pinfold Consultants Inc.

Gibson, R. N. 2003. Go with the flow: tidal migration in marine animals. *Hydrobiologia* 503: 153-161.

Goncalves, R., A. D. Correia, N. Atanasova, M. A. Teodosio, R. Ben-Hamadou, and L. Chicharo. 2015. Environmental factors affecting larval fish community in the salt marsh area of Guadiana Estuary (Algarve, Portugal). *Scientia Marina* 79 (1): 25-34.

Gorecki, R., and M. B. Davis. 2013. Seasonality and spatial variation in nekton assemblages of the Lower Apalachicola River. *Southeastern Naturalist* 12 (1): 171-196.

Hallett, C. S., F. Valesini, and M. Elliott. 2016. A review of Australian approaches for monitoring, assessing and reporting estuarine condition: I. International context and evaluation criteria. *Environmental Science and Policy* 66: 260-269.

Hanson, J.M., 2009. Predator-prey interactions of American lobster (*Homarus americanus*) in the southern Gulf of St. Lawrence, Canada. *New Zealand Journal of Marine and Freshwater Research* 43 (1): 69-88.

Harrison, T. D., and A. K. Whitfield. 2004. A multi-metric fish index to assess the environmental condition of estuaries. *Journal of Fish Biology* 65: 683-710.

Harrison, T. D., and A. K. Whitfield. 2006a. Application of a multimetric fish index to assess the environmental condition of South African estuaries. *Estuaries and Coasts* 29: 1108 1120.

Harrison, T. D., and A. K. Whitfield. 2006b. Temperature and salinity as primary determinants influencing the biogeography of fishes in South African estuaries. *Estuarine, Coastal and Shelf Sciences* 66: 335-345.

Hoagland, P., D. M. Anderson, Y. Kaoru, and A. W. White. 2002. The economic effects of harmful algal blooms in the United States: Estimates, assessment issues, and information needs. *Estuaries* 25 (4): 819-837.

Jaureguizar, A. J., R. Menni, R. Guerrero, and C. Lasta. 2004. Environmental factors structuring fish communities of the Rio de la Plata estuary. *Fisheries Research* 66: 195-211.

Kanu, A., C. DuBois, E. Hendriks, K. Cave, K. Hartwig, J. Fresque-Baxter, K. Trembath, and E. Kelly. 2016. Realizing the potential of community based monitoring in assessing the health of our waters. *Our Living Waters*.
http://www.ourlivingwaters.ca/cbmreport_sep2016 (accessed March 1, 2017).

Kemp, W. M., W. R. Boynton, J. E. Adolf, D. F. Boesch, W. C. Boicourt, G. Brush, J. C. Cornwell, T. R. Fisher, P. M. Gilbert, J. D. Hagy, L. W. Harding, E. D. Houde, D. G. Kimmel, W. D. Miller, R. I. E. Newell, M. R. Roman, E. M. Smith, and J. C. Stevenson. 2005. Eutrophication of Chesapeake Bay: Historical trends and ecological interactions. *Marine Ecology Progress Series 303*: 1- 29.

Kennish, M. J. 2002. Environmental threats and environmental future of estuaries. *Environmental Conservation* 29 (1): 78-107.

Krumme, U. 2009. Diel and tidal movements by fish and decapods linking tropical coastal ecosystems. In *Ecological Connectivity Among Tropical Coastal Ecosy*stems, ed. I. Nagelkerken, 271 - 324, New York: Springer Science.

Leblanc, J., C. M. Couillard, and J.-C. F. Brethes. 1997. Modifications of the reproductive period in mummichog (*Fundulus heteroclitus*) living downstream from a bleached kraft pulp mill in the Miramichi Estuary, New Brunswick, Canada. *Canadian Journal of Fisheries and Aquatic Sciences* 54 (11): 2564-2573.

Lehman, P. W., T. Kurobe, S. Lesmeister, D. Baxa, A. Tung, and S. J. Teh. 2017. Impacts of the 2014 severe drought on the Microcystis bloom in San Francisco Estuary. *Harmful Algae* 63: 94-108.

Levings, C. D. 2016. *Ecology of Salmonids in Estuaries Around the World: Adaptations, Habitats, and Conservation*. Vancouver: UBC Press.

Liang, X., B. Chen, X. Nie, Z. Shi, X. Huang, and X. Li. 2013. The distribution and partitioning of common antibiotics in water and sediment of the Pearl River Estuary, South China. *Chemosphere* 92: 1410-1416.

Lotze, H. K., H. S. Lenihan, B. J. Bourque, R. H. Bradbury, R. G. Cooke, M. C. Kay, S. M. Kidwell, M. X. Kirby, C. H. Peterson, and J. B. Jackson. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 312 (5781): 1806-1809.

Mahoney, P. C., and M. J. Bishop. 2017. Assessing risk of estuarine ecosystem collapse. *Ocean & Coastal Management* 140: 46-58.

Marshall, S., and M. Elliott. 1998. Environmental influences on the fish assemblage of the Humber Estuary, U.K. *Estuarine, Coastal and Shelf Science* 46: 175-184.

National Oceanic and Atmospheric Administration (NOAA). 2015. *What is an estuary?* http://oceanservice.noaa.gov/facts/estuary.html. (accessed January 15, 2015).

Oberdorff, T. and R. M. Hughes. 1992. Modification of an index of biotic integrity based on fish assemblages to characterize rivers of the Seine Basin, France. *Hydrobiologia* 228: 117-130.

Oberdorster, E., and A. O. Cheek. 2001. Gender benders at the beach: endocrine disruption in marine and estuarine organisms. *Environmental Toxicology and Chemistry* 20 (1): 23-36.

Office of Environment and Heritage (OEH). 2013. Assessing estuary ecosystem health: sampling, data analysis and reporting protocols. *NSW Natural Resources Monitoring, Evaluation and Reporting Program* 2013/0125.

Porter, A., and P. Scanes. 2015. Scavenging rate ecoassay: a potential indicator of estuary condition. *PLOS ONE* 10 (5): e0127046.

Proffitt, C. E. 2017. Phytoplankton biomass in a subtropical estuary: Drivers, blooms, and ecological functions assessed over space and time using structural equation modeling. *Marine Ecology Progress Series* 569: 55-75.

Raposa, K. B., C. T. Roman, and J. F. Heltshe. 2003. Monitoring nekton as a bioindicator in shallow estuarine habitats. *Environmental Monitoring and Assessment* 81: 239-255.

Reynoldson, T. B., J. L. Bailey, and R. C. Bailey. 2016. *Preliminary review of data collected by the CAMP nearshore monitoring programme 2004-2015*. Report prepared for the Department of Fisheries and Oceans Canada. GHOST Environmental Consulting.

Rothenberger, M. B., T. Swaffield, A. J. Calomeni, and C. D. Cabrey. 2014. Multivariate Analysis of Water Quality and Plankton Assemblages in an Urban Estuary. *Estuaries and Coasts* 37: 695-711.

Ryan, W. H., E. S. Gornish, L. Christenson, S. Halpern, S. Henderson, G. Lebuhn, and T. E. Miller. 2017. Initiating and managing long-term data with amateur scientists. *The American Biology Teacher* 79 (1): 28-34.

Savage, J. M., P. E. Osborne, and M. D. Hudson. 2017. Effectiveness of community and volunteer based coral reef monitoring in Cambodia. *Aquatic Conservation: Marine and Freshwater Ecosystems* 27: 340-352.

Scanes, P., G. Coade, M. Doherty, and R. Hill. 2007. Evaluation of the utility of water quality based indicators of estuarine lagoon condition in NSW, Australia. *Estuarine, Coastal and Shelf Science* 74: 306-319.

Sharpe, A., and C. Conrad. 2006. Community based ecological monitoring in Nova Scotia: Challenges and opportunities. *Environmental Monitoring and Assessment* 113: 395-409.

Sheaves, M., R. Johnston, and R. M. Connolly. 2012. Fish assemblages as indicators of estuary ecosystem health. *Wetland Ecology Management* 20: 477-490.

Short, F. T., and D. M. Burdick. 1996. Quantifying eelgrass habitat loss in relation to housing development and nitrogen loading in Waquoit Bay, Massachusetts. *Estuaries* 19 (3): 730-739.

Staehr, P. A., J. Testa, and J. Carstensen. 2017. Decadal changes in water quality and net productivity of a shallow Danish estuary following significant nutrient reductions. *Estuaries and Coasts* 40: 63-79.

Strydom, N. A. 2015. Patterns in larval fish diversity, abundance, and distribution in temperate South African estuaries. *Estuaries and Coasts* 38: 268-284.

Tecchio, S., A. T. Rius, J.-C. Dauvin, J. Lobry, G. Lassalle, J. Morin, N. Bacq, M. Cachera, A. Chaalali, M.C. Villanueva, and N. Niquil. 2015. The mosaic of habitats of the Seine estuary: Insights from food-web modelling and network analysis. *Ecological Modelling* 312: 91-101.

Telesh, I. V., and V. V. Khlebovich. 2010. Principal processes within the estuarine salinity gradient: A review. *Marine Pollution Bulletin* 61: 149-155.

Temmerman, S., P. Meire, T. J. Bouma, P. M. Herman, T. Ysebaert, and H. J. De Vriend. 2013. Ecosystem-based coastal defence. *Nature* 504: 79-83.

Tett, P., R. J. Gowen, S. J. Painting, M. Elliott, R. Forster, D. K. Mills, E. Bresnan, E. Capuzzo, T. F. Fernandes, J. Foden, R. J. Geider, L. C. Gilpin, M. Huxham, A. L. McQuatters-Gollop, S. J. Malcom, S. Saux-Picart, T. Platt, M.-F. Racault, S. Sathyendranath, J. van der Molen, and M. Wilkinson. 2013. Framework for understanding marine ecosystem health. *Marine Ecology Progress Series* 494: 1-27.

Theobald, E. J., A. K. Ettinger, H. K. Burgess, L. B. DeBey, N. R. Schmidt, H. E. Froehlich, C. Wagner, J. HilleRisLambers, J. Tewksbury, M. A. Harsch, and J. K. Parrish. 2015. Global change and local solutions: Tapping the unrealized potential of citizen science for biodiversity research. *Biological Conservation* 181: 236-244.

Thériault, M.-H., S. Courtenay and J. Weldon. 2008. Quality Assurance / Quality Control (QA/QC) program for the Community Aquatic Monitoring Program (CAMP). *Canadian Technical Report or Fisheries and Aquatic Science* 2823: v + 29 p.

Thériault, M. H., and S. Courtenay. 2010. Overview analysis of the Community Aquatic Monitoring Program (CAMP) in the Basin Head Lagoon from 2002 to 2008. *Canadian Science Advisory Secretariat Research Document* 2010/001: iv + 34 p.

Thériault, M.-H. and S.C. Courtenay. 2012. Calibration exercise for the Community Aquatic Monitoring Program (CAMP) nutrient analyses: establishing variability between filtered and unfiltered water samples and two analytical laboratories. *Canadian Technical Report of Fisheries and Aquatic Science*s 2980: ix + 29 p.

Thrush, S. F., M. Townsend, J. E. Hewitt, K. Davies, A. M. Lohrer, C. Lundquist, and K. Cartner. 2013. The many uses and values of estuarine ecosystems. In *Ecosystem Services in New Zealand*, ed. J. R. Dymond, 226-237. Lincoln: Manaaki Whenua Press.

Tsou, T-S., and R. E. Matheson. 2002. Seasonal changes in the nekton community of the Suwannee River Estuary and the potential impacts of freshwater withdrawal. *Estuaries* 25: 1372-1381.

Turcotte-Lanteigne, A., and E. Ferguson. 2008. Ecosystem overview of the Richibucto Watershed in New Brunswick. *Canadian Manuscript Report on Fisheries and Aquatic Sciences* 2847: xiv + 167 p.

United States Environmental Protection Agency (USEPA). 2012. Basic Information about Estuaries. *Estuaries*.
http://water.epa.gov/type/oceb/nep/about.cfm. (accessed January 10, 2016).

Valesini, F. J., A. Cottingham, C. S. Hallett, and K. R. Clarke. 2017. Interdecadal changes in the community, population and individual levels of the fish fauna of an extensively modified estuary. *Journal of Fish Biology* 90: 1734-1767.

van der Velde, T., D. A. Milton, T. J. Lawson, C. Wilcox, M. Lansdell, G. Davis, G. Perkins, and B. D. Hardesty. 2017. Comparison of marine debris data collected by researchers and citizen scientists: Is citizen science data worth the effort? *Biological Conservation* 208: 127-138

Watson, E. B., K. B. Raposa, J. C. Carey, C. Wigand, and R. S. Warren. 2017. Anthropocene survival of southern New England's salt marshes. *Estuaries and Coasts* 40: 617-625.

Weldon, J., D. Garbary, S. Courtenay, W. Ritchie, C. Godin, M-H. Thériault, M. Boudreau, and A. Lapenna. 2005. The Community Aquatic Monitoring Project (CAMP) for measuring marine environmental health in coastal waters of the southern Gulf of St. Lawrence: 2004 Overview. *Canadian Technical Report of Fisheries and Aquatic Sciences* 2624: vii + 53 p.

Weldon, J., S. Courtenay, and D. Garbary. 2007. The Community Aquatic Monitoring Program (CAMP) for measuring marine environmental health in coastal waters of the southern Gulf of St. Lawrence: 2005 Overview. *Canadian Technical Report of Fisheries and Aquatic Sciences* 2708: viii + 47 p.

Whitfield, A. K. 1997. Fish conservation in South African estuaries. *Aquatic Conservation: Marine and Freshwater Ecosystems* 7: 1-11.

Whitfield, A. K. 1999. Icthyofaunal assemblages in estuaries: A South African case study. *Reviews in Fish Biology and Fisheries* 9: 151-186.

Whitfield, A. K., and M. Elliott. 2002. Fishes as indicators of environmental and ecological changes within estuaries: a review of progress and some suggestions for the future. *Journal of Fish Biology* 61: 229-250.

Wilcockson, D., and L. Zhang. 2008. Circatidal clocks. *Current Biology* 18 (17): 753-755.

Williams, S. L., and E. D. Grosholz. 2008. The invasive species challenge in estuarine and coastal environments: Marrying management and science. *Estuaries and Coasts* 31 (1): 3-20.

Xu, T., and X. You. 2017. Effects of large-scale embankments on the hydrodynamics and salinity in the Oujiang River Estuary, China. *Journal of Marine Science and Technology* 22: 71-84.

Ysebaert, T., and P. M. Herman. 2002. Spatial and temporal variation in benthic macrofauna and relationships with environmental variables in an estuarine, intertidal, soft-sediment environment. *Marine Ecology Progress Series* 244: 105-124.

Appendix A

Supplementary data for Chapter 2

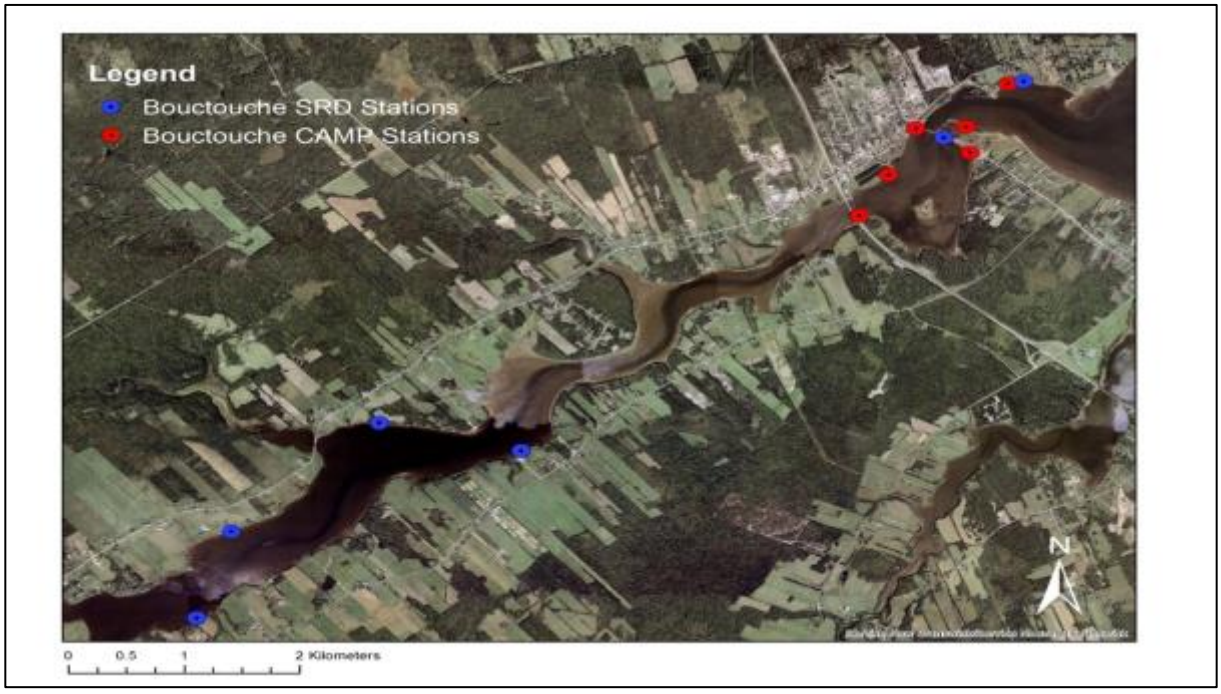

Figure S2.1 Map of Bouctouche station locations. Image created using ArcGIS (ESRI 2015).

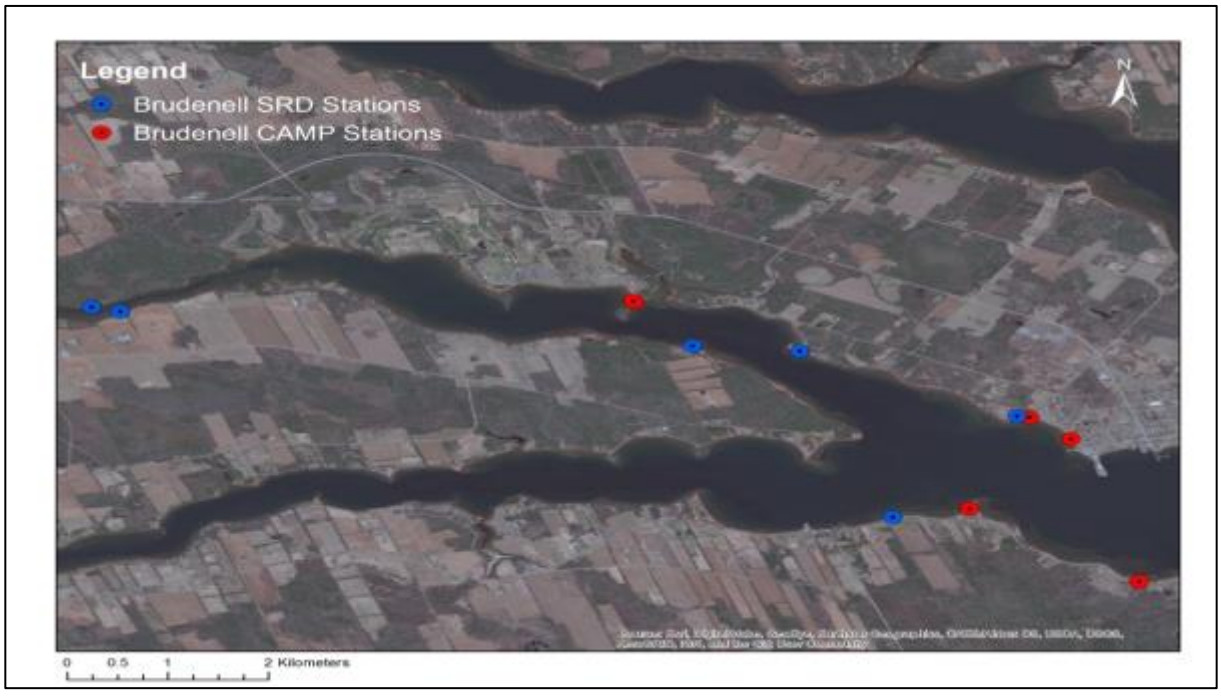

Figure S2.2 Map of Brudenell station locations. Image created using ArcGIS (ESRI 2015).

Figure S2.3 Map of Cocagne station locations. Image created using ArcGIS (ESRI 2015).

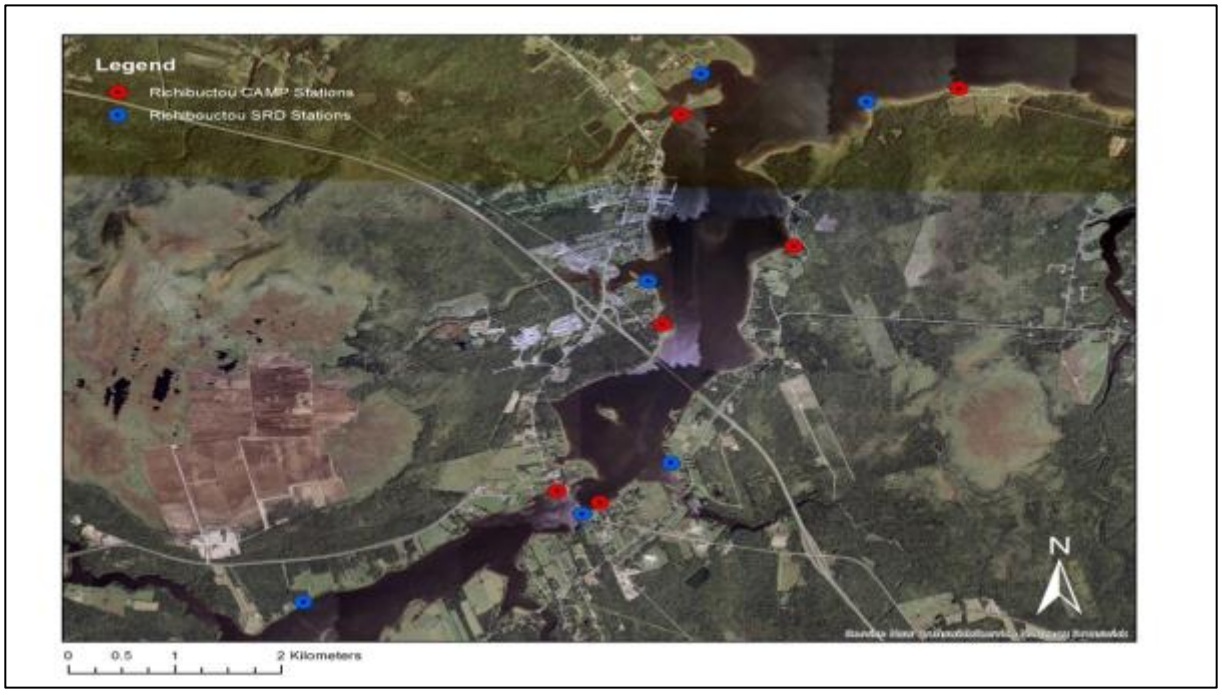

Figure S2.4 Map of Richibucto station locations. Image created using ArcGIS (ESRI 2015).

Figure S2.5 Map of Scoudouc station locations. Image created using ArcGIS (ESRI 2015).

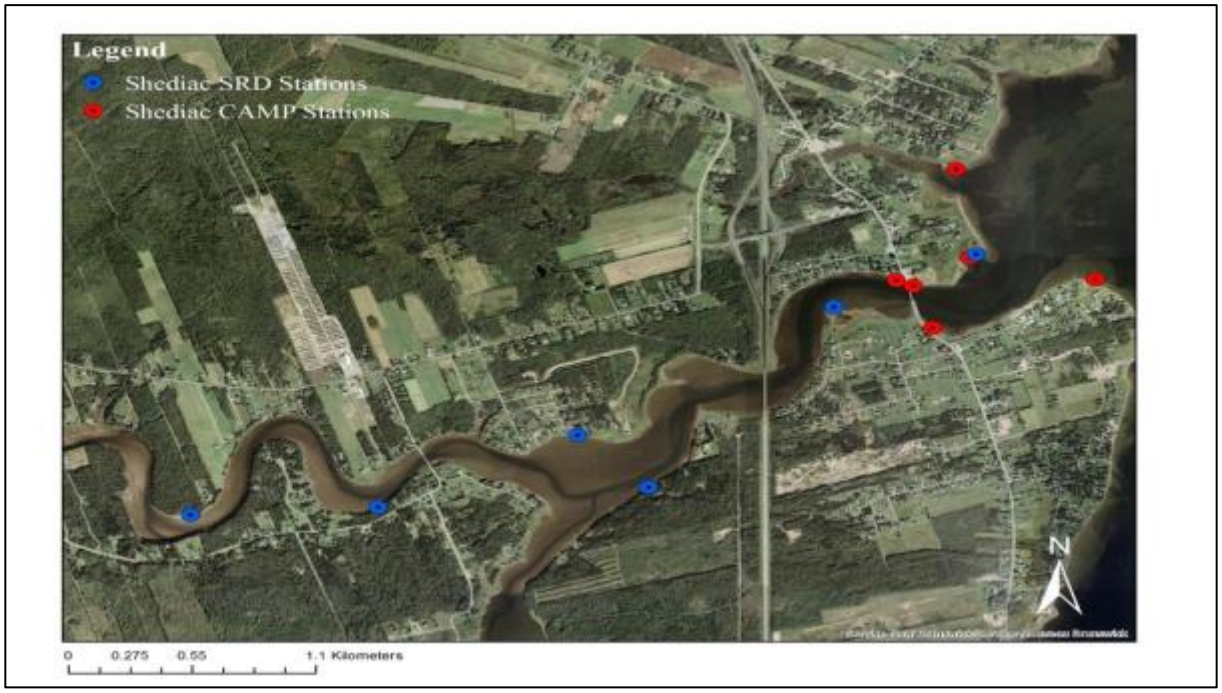

Figure S2.6 Map of Shediac station locations. Image created using ArcGIS (ESRI 2015).

Figure S2.7 Map of Souris station locations. Image created using ArcGIS (ESRI 2015).

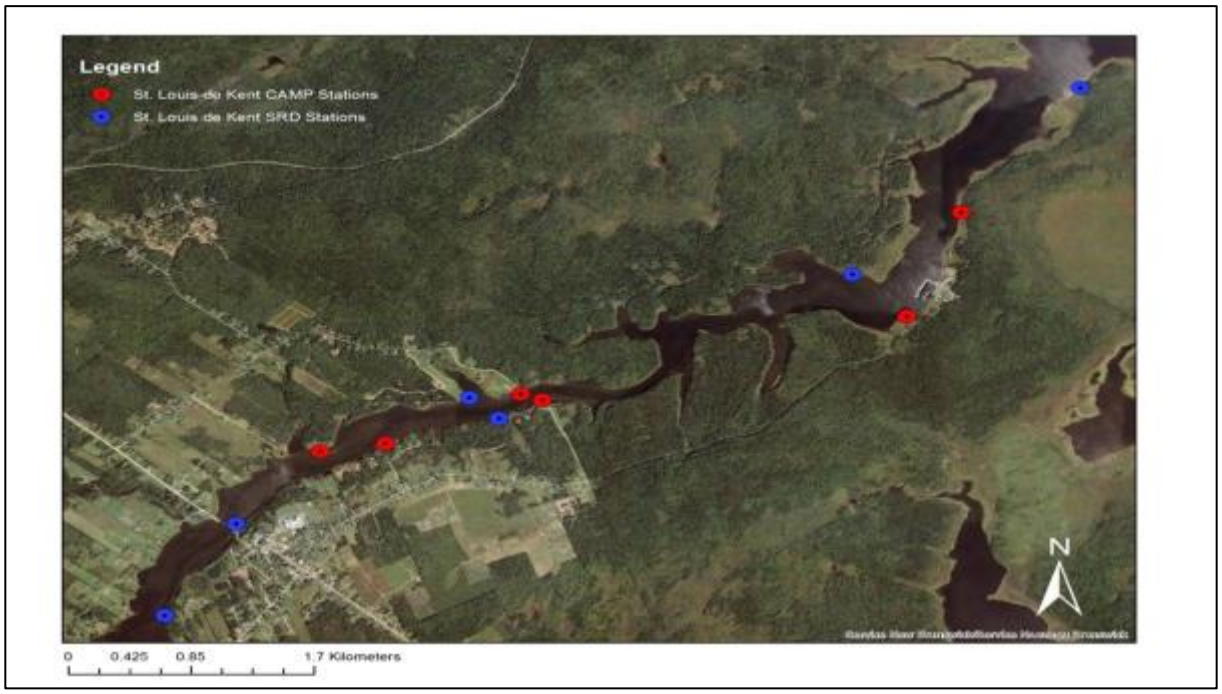

Figure S2.8 Map of St. Louis de Kent station locations. Image created using ArcGIS (ESRI 2015).

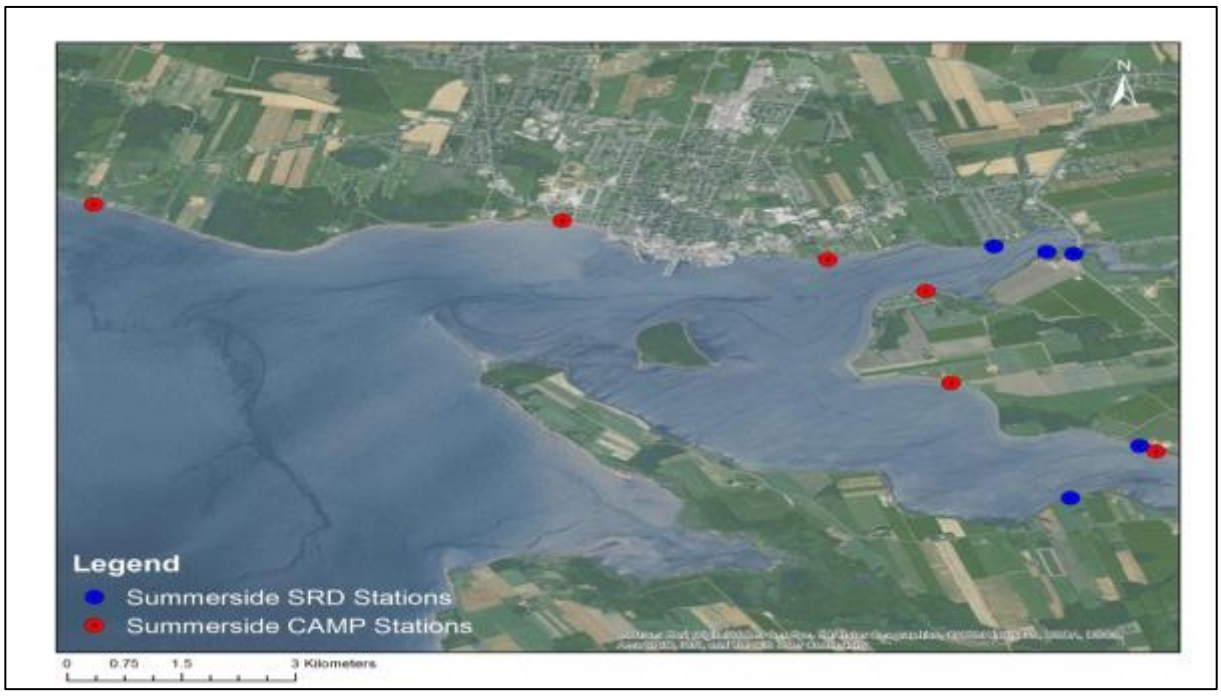

Figure S2.9 Map of Summerside station locations. Image created using ArcGIS (ESRI 2015).

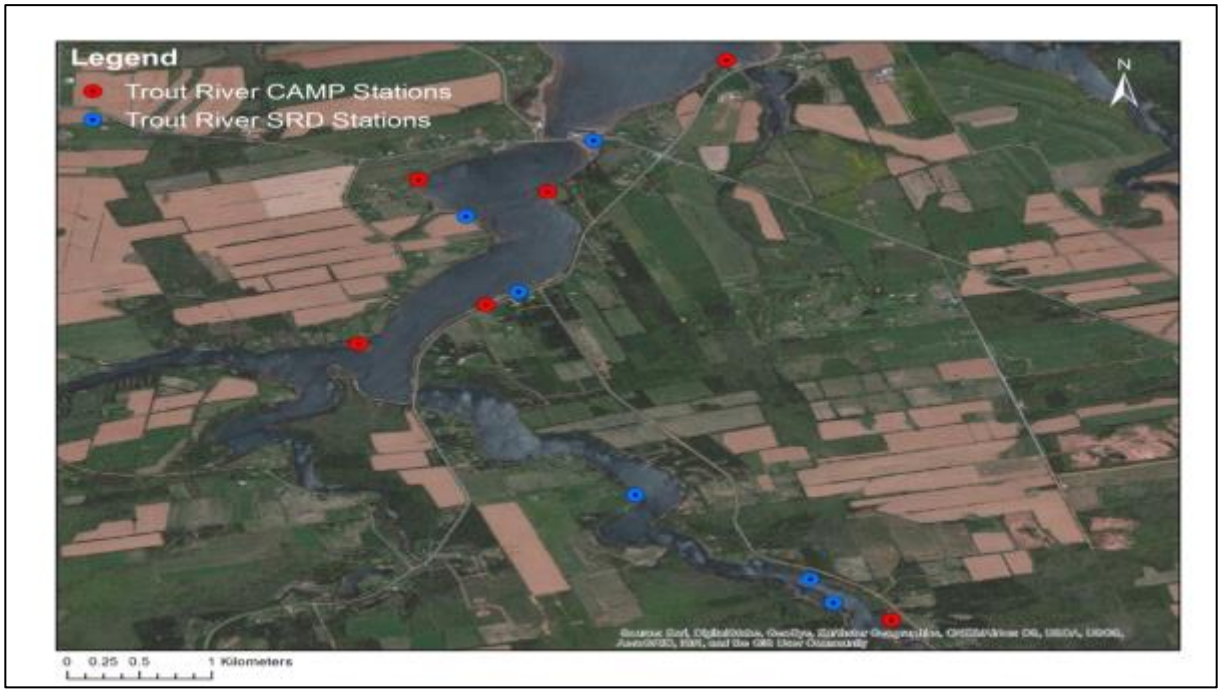

Figure S2.10 Map of Trout River station locations. Image created using ArcGIS (ESRI 2015).

Figure S2.11 Shade plots displaying the non-transformed abundance of each nekton species (y-axis) per sampling design (Community Aquatic Monitoring Program [CAMP] and Stratified Random Design [SRD]) for the 10 sites (x-axis). The intensity of the shade signifies the relative abundance of each species and the contribution of each species to the similarity calculation.

Figure S2.12 Shade plot displaying the square-root transformed abundances of each nekton species (y-axis) per sampling design (Community Aquatic Monitoring Program [CAMP] and Stratified Random Design [SRD]) for the 10 sites (x-axis). The intensity of the shade signifies the relative abundance of each species and the contribution of each species to the similarity calculation

Table S2.1 One-way analysis of variance of data collected from the Community Aquatic Monitoring Program (CAMP) stations using factor Site for the number of each species captured.

Source	d.f.	*Sum Sq*	*Mean Sq*
Mummichog			
Site	8	4980725	622591
Residuals	45	14237469	316388
Adult Fourspine Stickleback			
Site	8	1334058	166757
Residuals	45	514836	11441
Adult Sand Shrimp			
Site	8	1158866	144858
Residuals	45	1484940	32999
YOY Atlantic Silversides			
Site	8	6016357	752045
Residuals	45	7053897	156753

Table S2.2 Variance of the mean and confidence interval calculations for each influential species.

Number of Stations	Variance of the mean	T-value	Confidence Interval (+/-)
Adult Mummichog			
6	11529.5	2.57	276
7	10692.5	2.45	253
8	10064.7	2.37	237
9	9576.5	2.31	226
10	9185.9	2.26	217
11	8866.3	2.23	210
12	8599.9	2.20	204
Adult Fourspine Stickleback			
6	3088.1	2.57	143
7	3057.8	2.45	135
8	3035.1	2.37	130
9	3017.5	2.31	127
10	3003.3	2.26	124
11	2991.8	2.23	122
12	2982.2	2.20	120
Adult Sand Shrimp			
6	2682.6	2.57	133
7	2595.3	2.45	125
8	2529.8	2.37	119
9	2478.9	2.31	115
10	2438.1	2.26	112
11	2404.8	2.23	109
12	2377.0	2.20	107
YOY Atlantic Silverside			
6	13926.8	2.57	303
7	13512.1	2.45	284
8	13201.1	2.37	272
9	12959.2	2.31	263
10	12765.6	2.26	256
11	12607.3	2.23	250
12	12475.3	2.20	246

Table S2.3 The average differences in influential species counts between sites calculated using nekton data collected from Community Aquatic Monitoring Program (CAMP) stations.

	SCOU	***SHED***	***SOUR***	***COCA***	***SUMM***	***BRUD***	***RICH***	***BOUC***	***STLO***
Mummichog									
SHED	310*								
SOUR	333*	23							
COCA	303*	7	30						
SUMM	290*	20	43	13*					
BRUD	279*	31*	54	24*	11*				
RICH	314*	4	19	11	24	35			
BOUC	327*	17	6	24	37	48	13		
STLO	223*	87*	110*	80*	67*	56*	91*	104	
TROU	661*	971*	994*	964*	951*	940*	975*	988*	884*
Adult Fourspine Stickleback									
SHED	91								
SOUR	93	2							
COCA	64	27	29						
SUMM	94*	3	1	30*					
BRUD	92*	1	1	28*	2				
RICH	134*	225*	227*	198*	228*	226*			
BOUC	29	62	64	35	65	63	163*		
STLO	78	13	15	14*	16	14	212*	49	
TROU	411*	502*	504*	475*	505*	503*	277*	440*	489*
Adult Sand Shrimp									
SHED	256*								
SOUR	1525*	1781*							
COCA	239*	17	1764*						
SUMM	260*	4	1785*	21*					
BRUD	155*	101*	1680*	84*	105*				
RICH	163*	419*	1362*	402*	423*	318*			
BOUC	48*	304*	1477*	287*	308*	203*	115*		
STLO	235*	21	1760*	4	25	80*	398*	283*	
TROU	185	71	1710*	54	75	30	348*	233	50
YOY Atlantic Silversides									
SHED	150*								
SOUR	212*	62							
COCA	186*	36*	26						
SUMM	97*	247*	309*	283*					
BRUD	204*	54*	8	18*	301*				
RICH	176*	26	36	10	273*	28			
BOUC	884*	1034*	1096*	1070*	787*	1088*	1060*		
STLO	210*	60*	2	24	307*	6	34	1094*	
TROU	203*	53	9	17	300*	1	27	1087*	7

*≥10% influence on dissimilarity of Group