

**Effects of Canada Goose (*Branta canadensis*) and
Snow Goose (*Chen caerulescens*) Herbivory on Tidal
Marsh Recession at the Westham Island Marsh**

**by
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Declaration of Committee

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Abstract

In the Fraser River Estuary of British Columbia, tidal marshes have been receding and converting into unvegetated mudflats since the 1980s. While there are many hypotheses for this recession, the effect of avian herbivory is poorly understood. This study assessed how Canada Goose (*Branta canadensis*) and Snow Goose (*Chen caerulescens*) herbivory affected cover of tidal marsh vegetation that was comprised mainly of three-square bulrush (*Schoenoplectus pungens*) in the Westham Island tidal marsh. I conducted two field-based enclosure experiments, marsh edge and mudflat, that used enclosure plots to reduce specific goose herbivory in a randomized block design. Each experiment consisted of four blocks each of which was comprised of four treatments: open to goose herbivory, excluded all goose herbivory, primarily excluded Canada Goose herbivory, or primarily excluded Snow Goose herbivory. The marsh edge experiment used enclosures centered on the vegetated edge of the marsh, while the mudflat experiment was conducted in the unvegetated mudflat and were transplanted with *S. pungens*. Based on results from July to October of 2020, percent cover of tidal marsh vegetation was about 20% lower in plots open to Canada Goose herbivory versus those that excluded geese. Snow Goose herbivory could not be accurately assessed as they arrived when *S. pungens* were dormant. Thus, deterring goose herbivory may be an important consideration for land managers in restoring tidal marshes. Additionally, I compared percent cover from drone-derived remote sensing to traditional ground-based visual estimates of percent cover of *S. pungens* in the tidal marsh. One per month, from July to October of 2020, I used a drone to take photos of the enclosures from the previous experiments, and used pixel counts to calculate the percent cover of *S. pungens*. I then used a *t*-test to compare the drone-derived percent cover to the ground-based estimates and found no significant difference ($t = 0.58$, $p = 0.56$). I then plotted a linear regression model and found a strong correspondence between both methods ($R^2 = 0.99$, $p = 1.3e-139$). So, remote sensing using drones appears to be an effective alternative to visual estimates of percent cover of tidal-marsh vegetation in the Westham island tidal marsh.

Keywords: Tidal marsh recession; Goose herbivory; Canada Goose; Snow Goose; *Schoenoplectus pungens*; Drones

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Chapter 1. Exclosure Experiments

1.0 Introduction

Tidal marshes are among the world's most ecologically productive and economically valuable ecosystems (Costanza *et al.* 1997). These marshes provide food and refuge for fish and wildlife, store appreciable amounts of carbon, filter contaminants, supply organic matter to estuarine and marine environments, and protect shorelines (Boesch and Turner 1984; Bakker *et al.* 2002; Valiela *et al.* 2004; Loomis and Craft 2010). However, impacts associated with urbanization, agriculture, and industrial and residential development have converted many tidal marshes into mudflats (Thorne *et al.* 2015). As we continue to build and develop into the estuary, natural deltaic processes are disrupted which may contribute to tidal marsh recession (Caldicot 2020).

The Fraser River Estuary (FRE) in British Columbia (BC) is designated as a Wetland of International Importance (a Ramsar Site) but has experienced appreciable marsh recession (Balke 2017; Bode 2019). Many fish, birds, and wildlife species within the FRE depend on tidal marshes for one or more stages of their life cycles (Vermeer and Davies 1978; Levings *et al.* 1991). On Sturgeon Bank, in the northern portion of the FRE, at least 160 ha (30%) of marsh vegetation died from 1989 to 2011 (Balke 2017; S. Boyd, Environment and Climate Change Canada, Pacific Wildlife Research Centre, unpubl. data). Similarly, there is a large portion of the Westham Island (WI) tidal marsh that has also become unvegetated (Balke 2017). Thus, it is critical to identify the cause of this recession to prevent further loss and determine if it is feasible to restore the marsh ecosystem.

There are contrasting results as to whether the leading edge of the FRE tidal marsh is expanding or receding. Some results indicated that the leading edge is generally quite stable since the rapid growth of common three-square bulrush (*Schoenoplectus pungens*) is quickly able to revegetate mudflats (Medley and Luternauer 1976; Moody 1978; Hales 2000). Other results indicated that marsh vegetation has been able to grow and stabilise further seaward (Kirwan and Murray 2008). However, the consensus based on recent findings is that the marsh is receding (Boyd *et al.* 2012; Balke 2017; Marijnissen and Aarninkhof 2017; Bode 2019; Caldicot 2020).

Several causes have been proposed (hypothesized) for the marsh recession at Sturgeon Bank, including sea-level rise, changes in sedimentation, changes in salinity, and goose herbivory (Balke 2017; Caldicot 2020). Roman *et al.* (1984), suggested that marsh vegetation may be intolerant of sea-level rise and shifts in salinity, leading to marsh recession. Herbivory by Snow Geese (SNGO; *Chen caerulescens*) is suspected to contribute to the recession of bulrush in the FRE (Giroux and Bédard 1987; Boyd 1995). By grazing on aboveground vegetation and grubbing roots and rhizomes, SNGO can eat all parts of a plant (Giroux and Bédard 1987). This intensive grazing by SNGO, coupled with their tendency to flock in large numbers, may greatly reduce marsh vegetation (Flemming *et al.* 2019).

Resident CAGO on Vancouver Island and in the FRE have greatly increased since the 1970's and are considered by many to be overabundant (Dawe *et al.* 2011). Although CAGO in the FRE were historically migratory, breeding programs created many non-migratory CAGO (Dawe and Stewart 2010). Studies have indicated that CAGO herbivory has had deleterious effects that have led to a reduction in vegetation and increased erosion rates throughout the Little Qualicum River Estuary on Vancouver Island (reviewed by Dawe 2015). However, the role of Canada Goose (CAGO; *Branta canadensis*) herbivory in tidal marsh recession in the FRE is still poorly understood (Kirwan *et al.* 2008).

Distinguishing the relative effects of herbivory by SNGO and CAGO on marsh recession is necessary to identifying appropriate management. Most SNGO have temporary and predictable periods of herbivory, and thus suitable management techniques would focus on short time periods. In contrast, most CAGO in the FRE are year-round residents, and thus anti-herbivory methods must be continuous. Thus, techniques designed to reduce seasonal herbivory by SNOGO may not effectively reduce chronic damage by CAGO.

My Applied Research Project (ARP) examined the role of herbivory in tidal-marsh recession, with a particular focus on contrasting the effects CAGO and SNGO in the FRE near WI, and proposes management strategies to prevent goose herbivory on an operational scale.

1.1 Key Species

1.1.1 Canada Goose (*Branta canadensis*)

CAGO were historically migratory in BC. However, there are now established resident breeding populations of CAGO throughout much of the Lower Mainland and Vancouver Island

(Dawe and Stewart 2010). There are ten recognized subspecies of CAGO, but I mainly observed *B. c. fulva* in the WI tidal marsh. (Lebeda and Ratti 1983).

Starting around the 1970s organized efforts to introduce CAGO to BC became more common (Dawe and Stewart 2010). These introduced geese likely came from subspecies that did not include the native *B. c. fulva* (Dawe and Stewart 2010). This introduction of varied stock likely led to the hybridization of multiple subspecies. While the exact number of CAGO in the FRE is unknown, on nearby Vancouver Island CAGO numbers increased from 532 birds in 1989 to 2,061 in 2006 in the Nanoose, Parksville, and Qualicum Beach areas (Dawe and Stewart 2010).

Most CAGO in the Lower Mainland tend to be non-migratory hybrids of multiple subspecies. Often seen as a nuisance species, CAGO are aggressive, territorial, and show minimal fear towards humans and other potential predators; these traits enable them to rapidly establish populations (Conover and Chasko 1985; Whitford 2003; Dawe and Stewart 2010). The predominant impacts of CAGO include airplane strikes/ crashes, damages to infrastructure, and high fecal bacterial loads in urban and watershed environments, including around homes, parks and drinking water (Woodruff *et al.* 2004; Council 2011; DeSisto 2014).

While many different goose-management strategies have been employed, current strategies are often costly and not well received by the public, due to their invasive nature (Dawe and Stewart 2010). Direct methods such as egg addling, falconry, and physical removal may be the most effective in reducing CAGO population, but these methods are often labour intensive and expensive (Woodruff *et al.* 2004; Council 2011; DeSisto 2014). Dawe (2011) harvested CAGO in cooperation with the Little Qualicum First Nations, and this effort was effective in reducing CAGO numbers while providing food to First Nations (Cameron and Jones 1983; Hunt 2007; Dawe *et al.* 2011). However, active harvest of CAGO is highly controversial among the public (Gale 2012). Key goals for CAGO management in the FRE include enhancing public awareness of impacts associated with overabundant geese and developing economically feasible management strategies.

1.1.2 Snow Goose (*Chen caerulescens*)

In recent years, the dramatic increase in SNGO numbers appears to have caused a loss in tidal-marsh vegetation in parts of western Europe (Fox and Madsen 2017). The Fraser-

Skagit population was estimated to be at 220,000 for the winter of 2020 birds, a nearly 15 times increase from roughly 15,000 birds in the the mid-1970s (S. Boyd, pers. comm.). SNGO are herbivorous and can eat much of the rhizomatous biomass in a tidal marsh (Boyd 1995; Demarchi 2006). They spend most of their waking hours eating vegetation in tidal marshes, agricultural fields, and residential areas (Burton and Hudson 1978; Ankney 1980, 1982; Boyd 1995; Demarchi 2006). With SNGO flocks becoming larger, there is speculation that they may also contribute to the loss of tidal marsh vegetation in the Fraser and Skagit regions (Demarchi 2006; Balke 2017). At their current rate of population growth, SNGO will soon likely exceed the carrying capacity of the FRE (Fox and Madsen 2017; Flemming *et al.* 2019).

SNGO arrive in the FRE in early October, then migrate to the Skagit River Estuary where they stay from December to February; they then return to the FRE before departing for breeding areas in April (Boyd 1995). However, factors such as climate change and direct anthropogenic involvement, including light pollution and supplementary cover-crop feeding, may be changing their behaviours (S. Boyd, pers. comm.). I detected signs of SNGO on the tidal marsh as early as 20 September 2020, with multiple flocks of around 50 geese arriving on 27 September. SNGO are migratory and thus their herbivory entails a short-term impact. This distinction between the chronic herbivory of CAGO and the temporary herbivory by SNGO is an important focus of my ARP. Similarly, being able to increase our understanding of this distinction will allow managers to implement strategies that are appropriate to the species causing the damage and their ecology.

1.1.3 Common three-square bulrush (*Schoenoplectus pungens*)

Previously named *Scirpus americanus*, *S. pungens* often occurs in marshes as large monotypic stands but can also appear in small, clustered forms in suboptimal environments (Bode 2019). This species occupies high-energy flooded and coastal environments and is tolerant of a range of salinities (0-10 ppt; Albert *et al.* 2013). *S. pungens* typically occupies the leading edge of the WI tidal marsh. It spreads via rhizome and can be associated with Lyngby sedge (*Carex lyngbyei*), seacoast bulrush (*Bolboschoenus maritimus*), Baltic rush (*Juncus balticus*), and soft-stem bulrush (*S. tabernaemontani*; Bode 2019). Because of its rapid growth rate, *S. pungens* is often seen as a colonizing species and is therefore important in the establishment of tidal marshes (Bell-Dereske 2008; Albert *et al.* 2013). *S. pungens* provides wave attenuation, erosion control, sediment and nutrient accumulation, and habitat function that supports native fish, bird, and wildlife species (Albert *et al.* 2013).

S. pungens has a yearly cycle of senescence and is eaten by most herbivorous waterfowl in the FRE. From September to mid July, *S. pungens* is typically dormant. During the remainder of summer, it develops stems and fruiting bodies. By about October most of the aboveground components of *S. pungens* have entered senescence and the belowground rhizomes remain dormant until the following spring (Boyd 1995; Balke 2017). Both CAGO and SNGO commonly consume *S. pungens* in the FRE (Balke 2017). Grazing (eating the aboveground biomass) and grubbing (digging to eat the rhizomes) behaviour have been observed in both goose species. This makes *S. pungens* a good indicator to study herbivory effects by the two goose species.

1.2 Focus

The main question my study addressed was:

What are the relative effects of CAGO versus SNGO herbivory on abundance of *S. pungens* on the WI tidal marsh?

To address this question, the overarching goal of my ARP was to test the following hypothesis:

The abundance of *S. pungens* is more strongly regulated by CAGO herbivory than SNGO herbivory.

My ARP investigated the effects of CAGO and SNGO herbivory on the recolonization of *S. pungens* by conducting two field-based enclosure experiments. These were the marsh edge experiment, which tested how *S. pungens* reacts when herbivory is removed, and the mudflat experiment, which tests if the mudflats could be revegetated when herbivory is removed. Each experiment consisted of four blocks, each of which was comprised of four treatments which either excluded one, both, or neither of the two goose species. I selected *S. pungens* as an indicator species due to its high rate of horizontal growth that enables it to colonize new areas within a few months (Bell-Dereske 2008). I also investigated CAGO and SNGO abundance using visual observations and camera-trap monitoring to assess how each species used the study area.

2.0 Methods

2.1 Study Site

My field research began in May and extended into November of 2020. The study was located on WI, within the municipality of Delta, BC (Fig. 1). WI lies in a productive agricultural region with a humid maritime climate, and a mean annual temperature of 9.6 °C and mean annual precipitation of 1,008 mm (Environment Canada 2017). Bordered by the FRE and the Strait of Georgia, the 17-km² island includes the George C. Reifel Bird Sanctuary and the Alaksen National Wildlife Area. Both areas are key locations in the Pacific Flyway for resident and migratory waterbirds (Zhang *et al.* 2017).

This study was located on the western side of the WI foreshore, partially within the Riefel Bird Sanctuary and Alaksen National Wildlife Area (Fig. 1). The 40-ha study site has exhibited an appreciable amount of tidal-marsh recession since the 1990s (E. Balke, South Coast Conservation Land Management Program, pers. comm.). The study site includes a large area of marsh vegetation composed of nearly monotypic *S. pungens*, which serves as the vegetated edge of the marsh in my study.

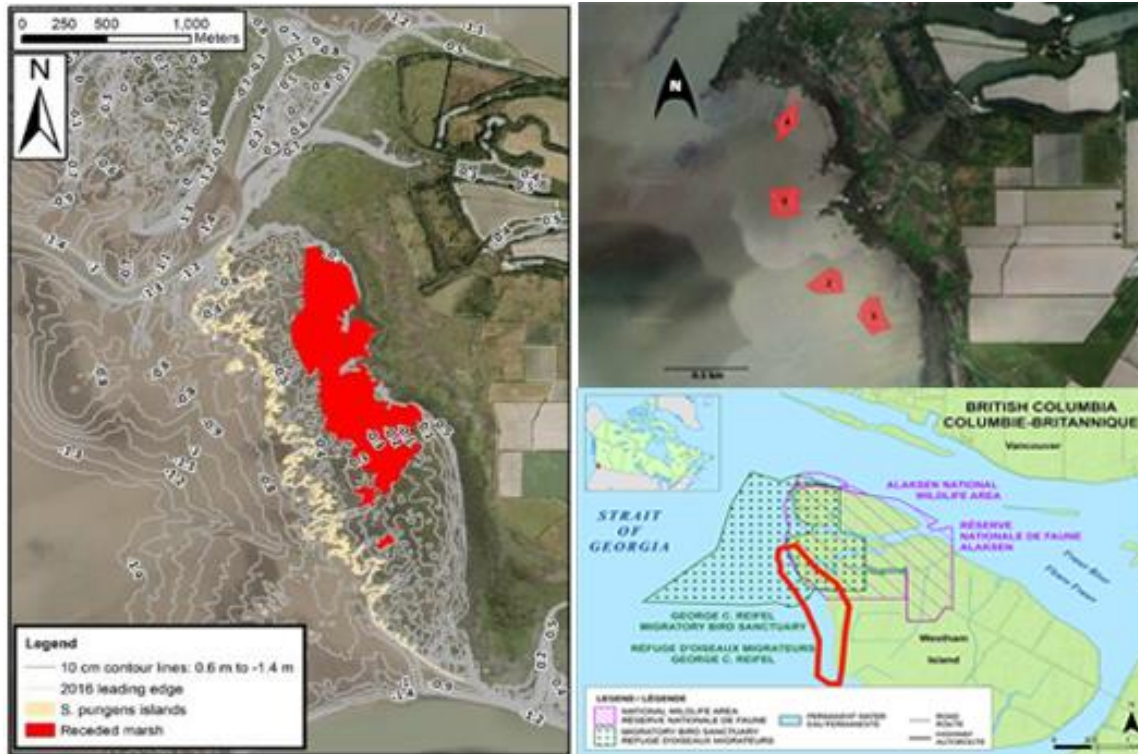


Figure 1. Maps of the Tidal marsh at the Westham Island foreshore in the Fraser River Estuary (Lower Mainland of BC), with red polygons denoting areas of receded marsh (figure from Balke 2017). Left panel: map with 10 cm-interval contour lines from 0.6 m to -1.4 m overlaid on 2013 air photos (CGVD2013 geoid) collected with LIDAR in 2013 (VFPA, 2013a; VFPA, 2013b). Beige lines indicate the 2016 marsh leading edge and *S. pungens* islands (Mason 2016, unpubl. data). Right top panel; map of the brackish marsh with red polygons delineating locations for study-blocks (Google imagery, 2020). Right bottom panel; Location illustrates the map of the study sites at Westham Island (red outline) used to examine Canada Goose and Snow Goose herbivory on bulrush in the estuary from May to November 2020 (map source: ECCC).

I divided the study site into four equal-sized quadrants, within which a single block of experimental treatments was randomly located with the stipulation that blocks be ≥ 200 m apart (Fig. 2). I used exclosures (5 x 10 m) that were a modified version of Balke’s (2017) pilot design. Each exclosure was comprised of six PVC poles dug 1 m into the ground and connected by 1.6-mm diameter stainless-steel aircraft cable. The cable was affixed between poles at 0.3 m and 0.6 m heights (Appendix 1). Holes were dug with a soil auger and pipes were pounded into the substrate using a hammer and wooden block. All exclosures were completed by 16 June 2020. These exclosures were used in two separate experimental designs: a marsh edge exclosure study (section 3.1) and a mudflat exclosure study (section 3.2).

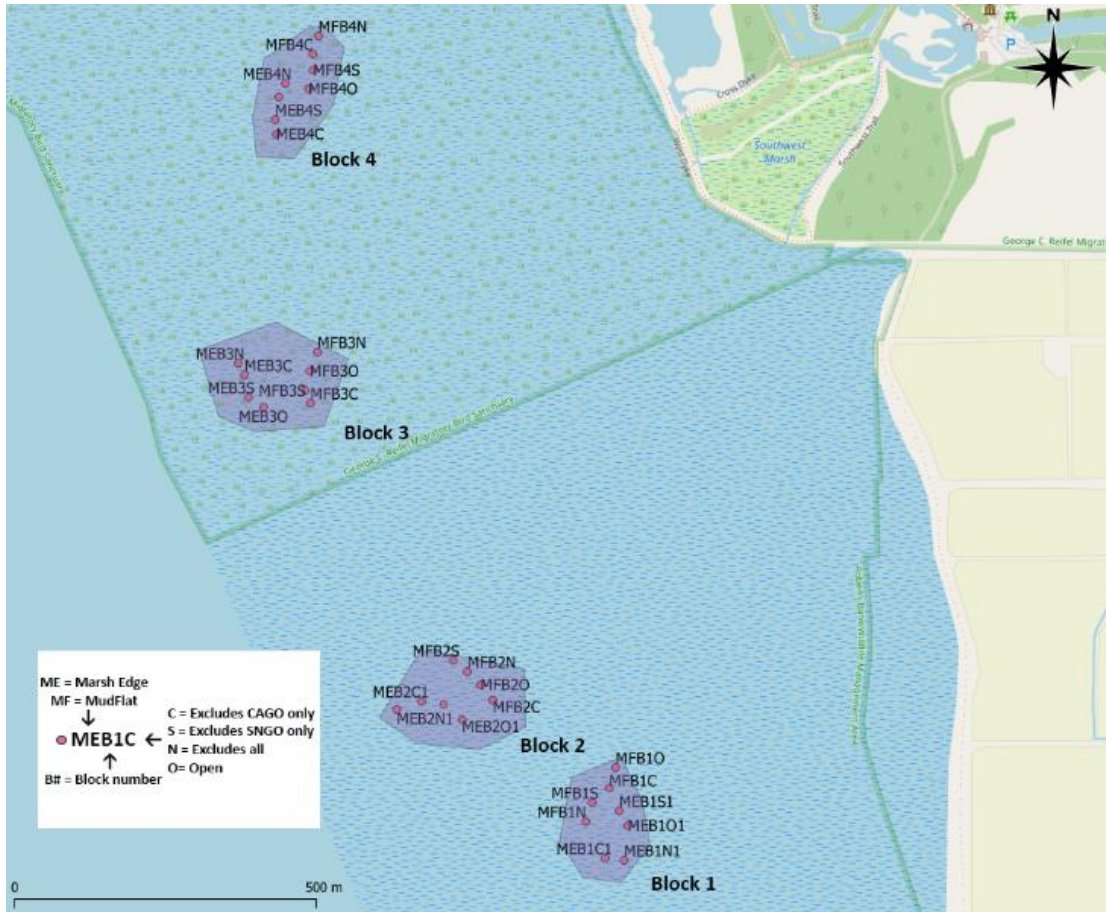


Figure 2. Locations of study blocks and treatments for the two experimental studies (marsh edge and mudflat) at the Westham Island study site in the Fraser River Estuary during May to November 2020 (QGIS, 2020).

2.2 Marsh Edge Exclosure Experiment

For this study, treatments straddled the vegetated edge of the tidal marsh. To assess whether herbivory is preventing the recolonization of *S. pungens*, each block was comprised of four treatments: a plot that was permanently accessible to all geese (control); a permanent exclosure preventing all goose herbivory; a temporary exclosure erected between October to April to restrict SNGO access during their presence in the FRE; and a temporary exclosure from May to September to exclude CAGO during the bulrush growing season (Fig. 3). Treatment locations within each block were assigned at random. To track how the marsh edge changes over time, I marked the existing marsh edge using three wooden stakes inside and four wooden stakes outside the exclosures. Control plots entailed PVC posts but no cables.

I measured percent cover of *S. pungens* in each treatment monthly from June 2020 until late September 2020 when vegetation had begun to senesce. I visually estimated percent cover of tidal marsh vegetation to the nearest 1%, excluding fully aquatic vegetation such as filamentous green algae, widgeon grass (*Ruppia maritima*), or Japanese eel grass (*Zostera japonica*). Additionally, I used a drone to measure the same data using pixel counts (see detailed methods in Chapter 2).

I took photos on the date plots were established (17 June 2020), and on 25 September 2020 just as dormancy had started (Appendix 2). These photos were taken to track how the plots would change over time and can be used as a baseline by future studies. To ensure that photos were taken from the same location, one wooden stake was placed 5 m away from the edge of the plot at each of the four cardinal directions. All photos were taken from the same height and angle using the same camera gear.

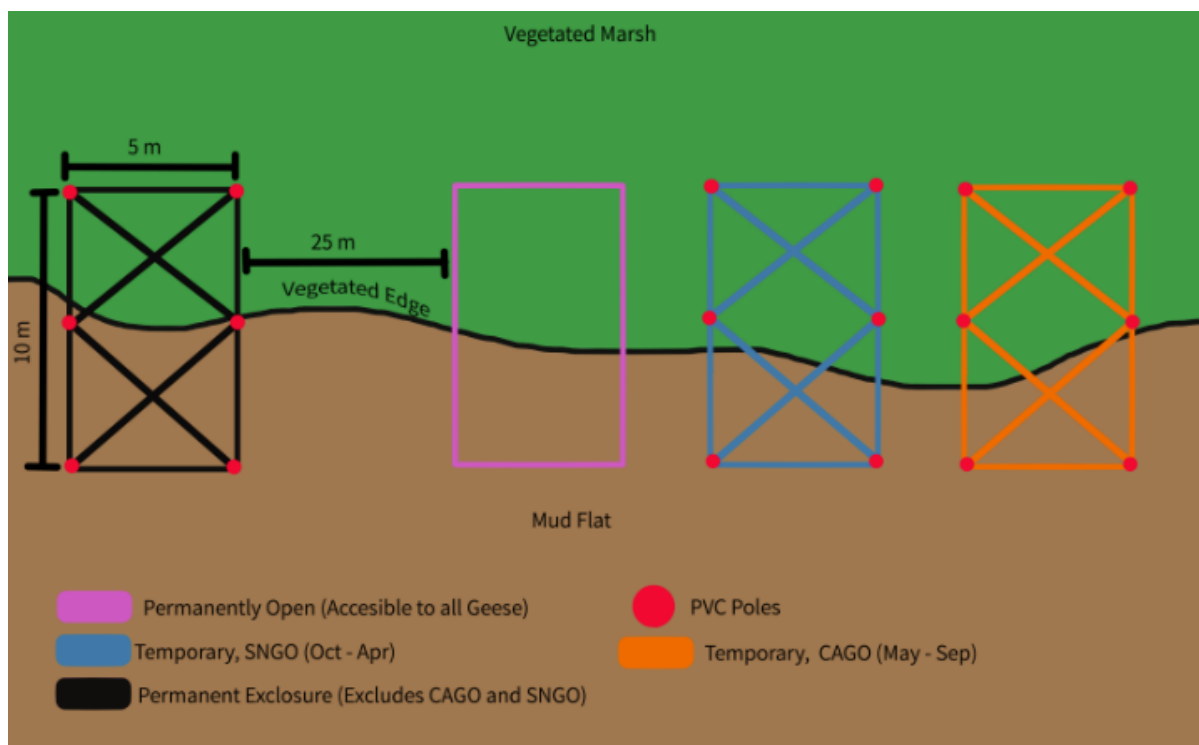


Figure 3. Experimental design of one study block of the enclosure experiment used to assess effects of Canada Geese and Snow Geese herbivory on tidal marsh recession at Westham Island study site in the Fraser River Estuary (Lower Mainland of BC) during the summer and fall of 2020. Plots are centered on the leading edge of marsh vegetation and locations of four treatment types (open, permanent exclusion, seasonal exclusion designed to exclude mainly Snow Goose, and seasonal exclusion designed to exclude mainly Canada Goose) were randomly assigned to each plot.

2.3 Mudflat Exclosure Experiment

This experiment assessed the potential for marsh regeneration on unvegetated mudflats under different grazing regimes. I used the same experimental and exclosure design as described above for the marsh edge experiment. I transplanted eight *S. pungens* cores from the existing tidal marsh into each treatment using a soil corer. Each core consisted of five to eight stems of *S. pungens* and was 20 cm in depth. Cores were taken from the vegetated section of the marsh that was closest to the respective study block and at least 50 m away from any study plot. I planted each core by hand. Only half of each exclosure was planted to see if *S. pungens* can passively colonize unvegetated mudflats in the WI tidal marsh (Fig 4).

Each month from June to September 2020, I counted each visible stem within each treatment. I also conducted photo monitoring using the same method as the marsh edge experiment. Plugs were planted on 11-12 June 2020 right after construction of exclosure posts but before cabling. I also assessed percent cover of *S. pungens* using the methods described for the marsh edge experiment (i.e., using visual estimates to the nearest 1% and aerial photos taken using a drone).

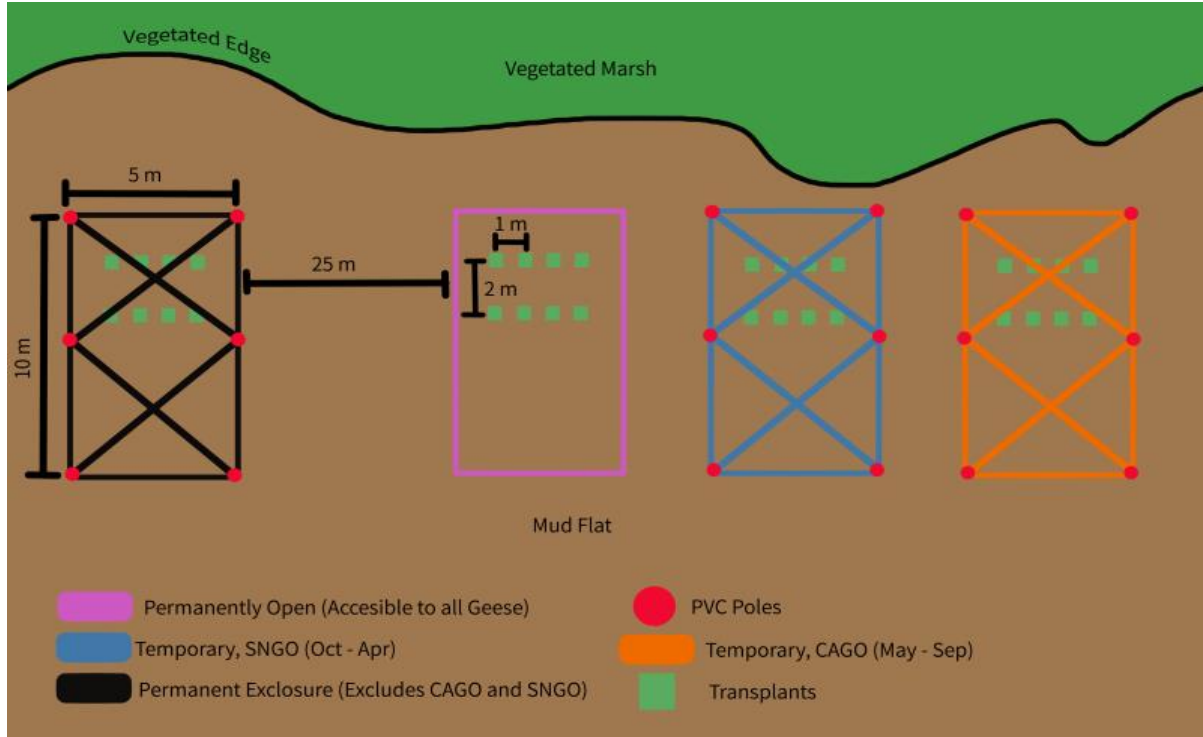


Figure 4. Experimental design of one study block of the exclosure experiment used to assess effects of Canada Geese and Snow Geese herbivory on tidal marsh recession at Westham

Island study site in the Fraser River Estuary (Lower Mainland of BC) during the summer and fall of 2020. Plots are on the unvegetated mudflats and locations of four treatment types (open, permanent enclosure, seasonal enclosure designed to exclude mainly Snow Goose, and seasonal enclosure designed to exclude mainly Canada Goose) were randomly assigned to each plot.

2.4 Statistical Analysis

To examine the effects of the enclosures on both the visual estimates and drone derived percent cover of *S. pungens* in both the mudflat and marsh edge experiments, and the stem counts in the mudflat experiment, I used a randomized complete block design, analyzed using a two-factor analysis of variance (ANOVA). The location of the four replicates (random factor) were blocks, and fixed effects were comprised of the four treatments (i.e., control (open), SNGO and CAGO enclosure (closed), SNGO enclosure, CAGO enclosure). If CAGO are a substantial contributor to marsh recession, one would expect to see lower percent cover and stem counts in the open and SNGO enclosure treatments and low herbivory in the closed and CAGO enclosure treatments. Conversely, if SNGO are a greater contributor to marsh recession, I would expect to see greater herbivory effects in the open and CAGO enclosure treatments compared to the closed and SNGO enclosures. I did not detect any appreciable number of CAGO using the WI marsh until the 27 July 2020. So, the ANOVA analysis was done only for August and September when geese were present on the study site. I then used a Tukey's Honestly Significant Difference (HSD) test to assess pairwise differences between treatments (Appendix 3). I tested for differences in treatments and for the interaction of blocks and treatments using an α level of 0.05 for three datasets: drone images, visual estimates of percent cover for both experiments, and stem counts in the mudflat experiments. Visual estimates and drone images of percent cover of marsh vegetation were highly correlated (Chapter 2), and thus I only conducted statistical analyses for visual estimates. All statistics were done in RStudio (ver. 1.2.5033).

3.0 Results

3.1 Marsh Edge Enclosure Experiment

Initial analysis of the marsh edge data indicated that there was a significant block-by-treatment interaction when all replicates were considered. However, after analysing each block individually, I found that when block 2, one of the replicates for the marsh edge experiment, was

removed from the analysis, the interaction disappears. So, I excluded block 2 and continued the marsh edge analysis with only three blocks. While this reduces the number of replicates, this allows us to analyze the three remaining replicates together using ANOVA.

After removing block 2 from analyses, I found that vegetation cover differed significantly among treatments ($F_{3,2} = 6.78$; $p = 0.005$). The permanent enclosure and the CAGO enclosure had greater cover of *S. pungens* when compared to the SNGO enclosure, after the arrival of CAGO to the study site (Tukey HSD adjusted $p = 0.042$; $p = 0.21$). However, percent cover of *S. pungens* in the open treatment, did not significantly differ from any of the other treatments. Percent cover of *S. pungens* in both open and SNGO enclosure treatments declined earlier in the summer and by greater amounts than the percent cover in treatments that excluded CAGO (Fig. 5).

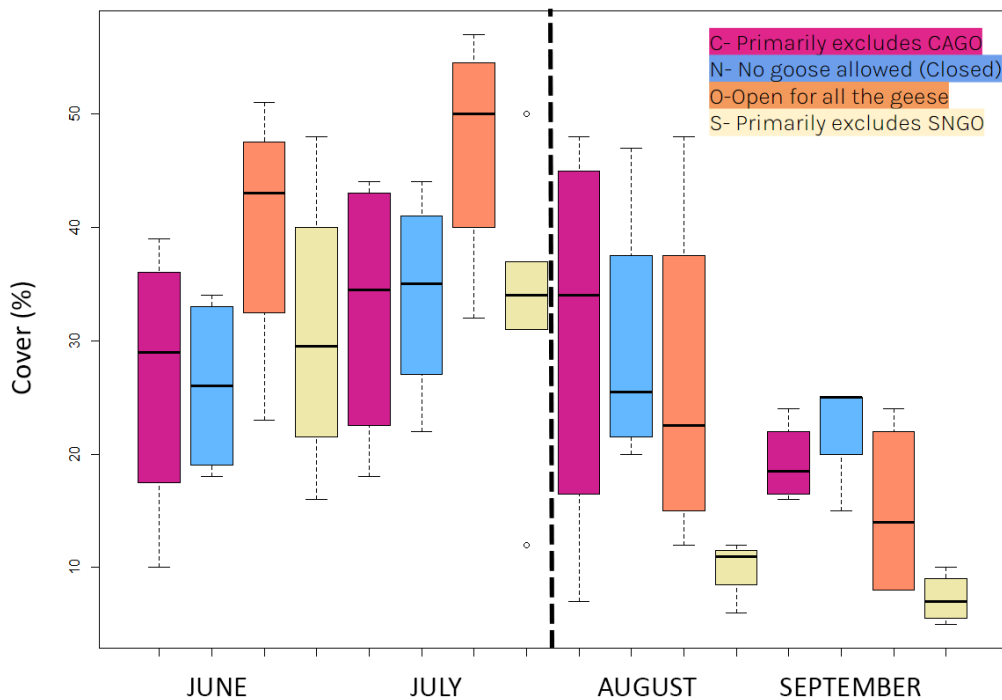


Figure 5. Change in visually estimated percent cover of *S. pungens* over time for the four treatments aimed to assess Canada and snow goose herbivory along the marsh edge at the Westham Island study site in the Fraser River Estuary (Lower Mainland of BC) from May to November 2020. The dotted line represents arrival of large numbers of Canada Geese in the study site.

3.2 Mudflat Exclosure Experiment

In the mudflat experiment, there was a significant difference in visually estimated percent cover of *S. pungens* among treatments ($F_{3,2} = 67.28$; $p = 6.14e-11$) and no block-by-treatment interaction. I found that *S. pungens* cover was similar between the open and SNGO exclosure treatments (Tukey HSD adjusted $p = 0.92$), and between the closed and CAGO exclosure treatments (Tukey HSD adjusted $p = 0.99$). Additionally, percent cover of *S. pungens* was greater in the closed and CAGO exclosure treatments as compared to both the open and SNGO exclosure treatments. Statistical analyses of stem counts shared the same pattern as the visual assessments of percent cover, and thus I have omitted those results. Percent cover of *S. pungens* declined earlier in the summer and by greater amounts in the open and SNGO exclosure treatments, as compared to the closed and CAGO exclosure treatments (Fig 6).

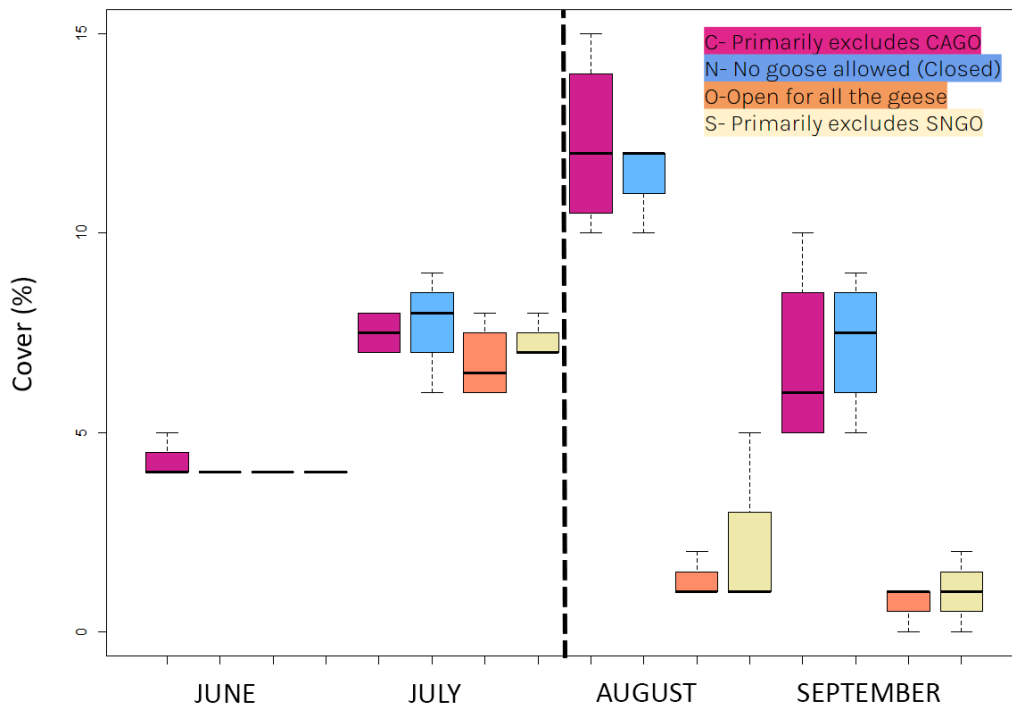


Figure 6. Change in visually estimated percent cover of *S. pungens* over time for the four treatments aimed to assess Canada and snow goose herbivory along the mudflats at the Westham Island study site in the Fraser River Estuary (Lower Mainland of BC) from May to November 2020. The dotted line represents arrival of large numbers of Canada Geese in the study site.

4.0 Discussion

While this study requires replication in space and time, the results from this first year suggest that CAGO herbivory may contribute to loss of tidal-marsh vegetation. The analysis of both experiments also indicated that the open and the SNGO exclosure treatments were statistically similar, while the closed and CAGO exclosure treatments were statistically similar. This was to be expected as senescence occurs at a similar time to the arrival of SNGO, and percent cover cannot be assessed when *S. pungens* is dormant. This makes it difficult to study the effect of SNGO herbivory on the percent cover of *S. pungens* in the same year. However, future analyses based on multiple years of field data may be able to show the effects of the long term exclosures on *S. pungens*. Thus, a comparison across multiple years using this year's data as a baseline are needed to assess the impact of SNGO herbivory. Similarly, further analysis is needed to assess how CAGO herbivory affects the abundance of *S. pungens* in the long term (i.e., the role of CAGO herbivory in marsh recession).

Both experiments showed a pattern where CAGO exclosures and permanent exclosure had greater *S. pungens* cover (as well as stem density in the mudflat experiment), but this pattern was more evident in the mudflat experiment. This observed pattern was likely due to several factors that were common among both experiments. Each exclosure of the marsh edge experiment started at different baseline percent covers due to the nature of the marsh. The mudflat experiment on the other hand, started at similar baseline percent covers between treatments as they consisted of transplants artificially placed in the study area. Moreover, *S. pungens* that has established in the marsh edge may be better at replacing above ground vegetation lost to herbivory as compared to transplants that had just started to establish.

When analyzed individually, blocks 2 and 4 in both experiments displayed the greatest difference percent cover of vegetation among treatments. This makes sense as based on results of the wildlife camera traps, CAGO and to some extent SNGO tended to spend more time in these blocks (Appendix 10).

Although CAGO are year-round residents on WI, my research indicated that their herbivory on marsh vegetation may be greatest from late July to early October. Earlier in the summer most CAGO on WI fed on surplus potatoes in agricultural fields, but then moved to the marsh in late July when farms began using propane cannons as an auditory hazing method. Observations of goose fecal matter in and around the study site confirmed this, as there more

feces in the area after auditory hazing, and a change in fecal colouration, suggesting a shift in diet (Appendix 8). Similarly, observations of goose body condition also changed after farms used propane cannons (Appendix 7). These observation metrics may allow future researchers to get a better understanding of goose diets and bioenergetics in the WI area. However, this pattern may be an effect of the COVID-19 pandemic. In most years, CAGO likely move to the marsh earlier in the summer because farms around the area tend to use auditory hazing earlier in the growing season and there is often less of a crop surplus.

Anthropogenic development and agricultural use of the FRE may be a further threat to tidal-marsh vegetation. Many areas in the FRE are being developed for human settlement. To protect human interest, many of these developments are protected by physical barriers (e.g., dykes) which can prevent the landward spread of tidal-marsh vegetation (Caldicott 2020). With physical barriers in place, tidal-marsh vegetation may be unable to escape the effects of sea-level rise which leads to a loss of tidal marshes, this is known as coastal squeeze (Caldicott 2020). Additionally, the Delta Farm and Wildlife Trust has, since the 1990s, bought and planted parcels of land on WI with lure crops for the consumption of geese (Merkens 2005; Bradbeer 2007). Lure crops steer geese away from areas of human interest (e.g., farms, airports) and bring them into Delta Farm and Wildlife Trust managed fields, thus, reducing the negative economic impacts of geese (Merkens 2005). However, this surplus of food may be a key factor in the exponential growth of the SNGO population and may have contributed to CAGO population growth in the FRE (Gauthier *et al.* 2005; Demarchi 2006; S. Boyd, pers. comm.). More geese means more herbivory and a potentially greater impact on tidal-marsh vegetation. I suggest that future studies investigate the economic and ecological impacts of the dyke enhancement and lure crop programs, and how these programs affect tidal marsh recession.

Management techniques to control the impact of resident CAGO on marsh vegetation that I observed may need to include year-round deterrents. Having year-round anti-herbivory methods may be expensive, so I piloted multiple different deterrent designs that may be more cost effective and require less maintenance (Appendix 9). While this will still need further study, initial observation seems to show that there are some cheap and effective designs.

5.0 Conclusion

Both enclosure experiments indicated that CAGO herbivory has a detrimental impact on the percent cover of *S. pungens* in the WI tidal marsh, though this impact was more apparent in the mudflat experiment. However, because the senescence of marsh vegetation that occurred before SNGO arrival limits our ability to contrast effects of SNGO versus CAGO herbivory. To fully investigate my hypothesis that the percent cover of *S. pungens* is more greatly regulated by CAGO than SNGO, multiple years of continued study will be necessary. By comparing data from the next few years to this year's baseline data, we may be able to see how the percent cover of *S. pungens* is affected by goose herbivory.

Although the ultimate cause(s) of marsh recession remain unknown, results of my study indicate that when CAGO herbivory is reduced percent cover of *S. pungens* increases. As *S. pungens* is an essential part of tidal-marsh vegetation on WI, it may be important consider the use of goose herbivory deterrents in tidal marsh restoration and management. So, for the success of tidal-marsh vegetation in the WI tidal marsh, I would suggest the use of passive deterrent structures to reduce all goose herbivory. Additionally, I suggest that managers work to promote public education about the many impacts associated with overabundant CAGO and SNGO, and to encourage the active harvest of both species in the tidal marsh.

Chapter 2. Drone Analysis

1.0 Introduction

Conducting field work in tidal marshes is logistically difficult and physically demanding. Walking on the soft terrain can entail a safety risk, and the tidal nature of these areas makes them difficult to access by foot when the tide is high. Although boats are an option, the timing windows of tides are often a challenge to work with especially in fall and winter. Scientific research and monitoring have increasingly used drone technology to conduct observations of sites that are challenging to access on the ground (Mury *et al.* 2020).

Drones have been used for vegetation and topographical mapping studies in tidal marsh environments (Mury *et al.* 2020). While successful, most studies used large commercial-grade drones that currently require additional licencing (Giones and Brem 2017). To avoid the need for licensing, I used the DJI Mavic mini. This lightweight drone is also less likely to affect bird behaviour (Brisson-Curadeau *et al.* 2017).

The goal of this portion of my ARP was to see if drone remote sensing is a viable alternative to ground-based assessments of *S. pungens* cover in the enclosure experiments in the WI tidal marsh. I compared field-based visual estimates (Chapter 1) with drone photography of the same enclosures and at the same time periods. I hypothesized that remote sensing via drones can be an effective alternative to visual estimates of percent cover of *S. pungens* in the WI tidal marsh. By using pixel count techniques on drone photography this chapter will show how remote sensing can be a powerful tool in marsh-vegetation monitoring and can increase the efficiency of data collection in the field.

2.0 Methods

I conducted ground-based visual assessments of *S. pungens* percent cover according to methods outlined in Chapter 1. I contrasted these ground-based estimates with cover estimates based on pixel counts of photos that I obtained using a drone equipped with a 12-megapixel sensor and a 3-axis motorized gimbal. Once per month from June to September, I took one photo per plot of the study blocks from 14 m above each treatment. Drone-based photos were taken on the same day as ground-based visual estimates, or on the following day if it was

raining or winds were too strong. I used the following equations to determine that 14 m would achieve an appropriate ground sampling distance (GSD) of 0.05 cm/pixel (Bridgelall *et al.* 2016):

$$\text{GSD} = \frac{\text{Flight Height (m)} \times \text{Sensor Height (m)}}{\text{Focal Length (m)} \times \text{Image Height (m)}}$$

Drone photos were processed in Adobe Photoshop using colour correction and contrast increasing techniques to enhance the difference in colours among the ground, *S. pungens*, and other aquatic plant species (Iqbal *et al.* 2010) (Appendix F). I then conducted a pixel count of *S. pungens* using the selection by colour tool and divided that number by the total pixel count in the plot to get percent cover of *S. pungens* (Luscier 2006).

I decided to analyse percent cover to the nearest 1% for visual estimates, and to the nearest 0.1% for drone analysis. An expert suggested the use of the Braun-Blanquet (BB) scale method to reduce bias and increase efficiency of ground-based estimates (S. Boyd pers. comm., Wikum and Shanholtzer 1978). However, I opted to use continuous measurements because it would be easier to directly compare with drone analysis.

Using RStudio (ver. 1.2.5033) I conducted a paired *t*-test at $\alpha = 0.05$ to determine if estimates of *S. pungens* cover differed significantly between ground-based visual observations and pixel counts from drone-based photos. I conducted a linear regression of ground-based estimates of percent cover versus drone-based pixel counts of percent cover, including block and date as factors. Block and date were not significant (adj. $p = 0.74$), and thus were removed from the regression. This allowed me to pool all the samples for an n of 128.

3.0 Results

There was no significant difference in percent cover of *S. pungens* derived from ground- or drone-based assessments ($t = 0.58$, $p = 0.56$). The linear regression model showed a strong correspondence between results using the two different methods ($p = 1.3\text{e-}139$). However, differences between the methods deviated slightly at higher levels of cover, at which cover estimates for the ground-based estimates were slightly greater than the drone-based estimates (Fig. 7).

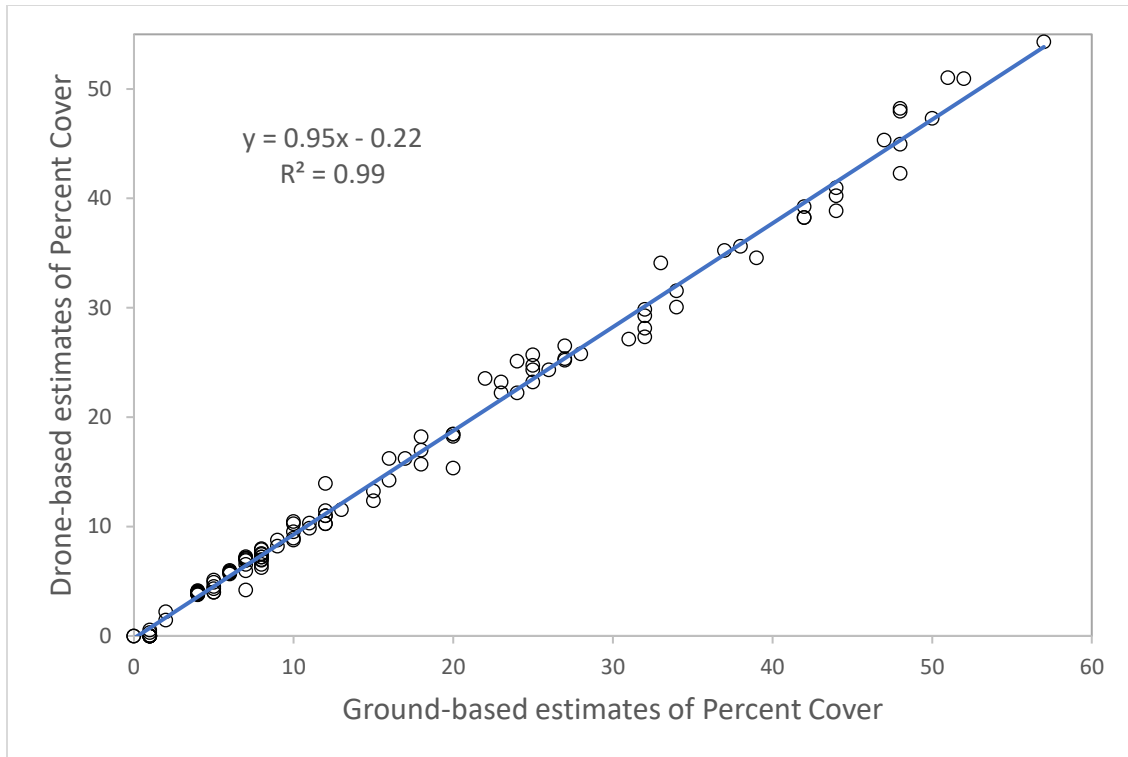


Figure 7. Ground-based estimates of percent cover by drone- based estimates of percent cover with linear regression model attached for the Westham Island study site in the Fraser River Estuary (Lower Mainland of BC) June to November 2020.

4.0 Discussion

The regression analysis indicated that percent cover of *S. pungens* derived from drone photos seem to be a viable alternative to the more traditional visual estimates. However, the drone assessment method seems to slightly underestimate *S. pungens* cover compared to visual assessments. In our testing, none of the values differed by more than 10% from one method to the other at the high end, while at the low percent cover, estimates were within 1%. So, depending on the accuracy required, drone remote sensing may provide quick accurate results.

Although ground-based visual estimates were similar to drone-derived estimates, ground-based estimates might be biased, while drone images can be relied on as having high accuracy. Drone-derived percent cover is based on an exact measure of the number of pixels in the photo. While it may be difficult to completely capture all the intricacies of the 3D world in a 2D image, the high resolution (2.7K and 1080p for Mavic mini) that is achieved by new drone

technologies will capture a single blade of *S. pungens* as a single pixel. Visual estimates require more field-based training and experience, while lab-based assessments of drone photos yield consistent results with little training. For example, in my study three different field observers gave different visual estimates for the same enclosure on the same day yet achieved similar results when they processed drone images. While it can be logistically difficult to work in the marsh environment, the drone method does not necessitate personnel to physically be out in the marsh, given that the site is within drone range. This not only increases efficiency but also safety.

While drone remote sensing has many benefits, it is not without drawbacks. Many drones with similar capabilities have been used in terrestrial vegetation studies (Tang and Shao 2015; Cunliffe *et al.* 2016). However, using a small drone in the WI tidal marsh, led to some problems. First, the drone was extremely susceptible to winds due to its light mass and could not be operated when it rained. Second, predatory birds occasionally attacked the drone necessitating the use of a spotter and leading to expensive repairs. Lastly, I experienced software issues that led to the drone crashing in brackish water (DJI technician, pers. comm).

Consumer drone technology has advanced greatly in recent years. Current models can provide professional grade photography, and aid in long-term monitoring. However, if this study were to be continued, I would suggest the use of drones that can operate at longer ranges and more extreme conditions. This would of course require more advanced drone licences and greater expense. The use of drones may one day become an important tool for the management of the tidal marshes, but multiple years of continued study and development will still be needed to test their effectiveness.

5.0 Conclusion

By comparing drone-derived percent cover to ground-based assessments, we not only ground truthed results from the drone photography, but also showed that drone remote sensing can be an effective alternative to visual estimates of *S. pungens* percent cover in the WI tidal marsh. In this experiment, the drone-derived percent cover was highly similar to results of ground-based visual estimates ($R^2 = 0.99$, $p = 1.3e-139$). However, drone data required a fraction of the time to collect. While I did run into some problems with the use of a drone, the benefits of increased efficiency and safety, in my opinion, are worth the expense. And as drones

become more capable in the future, we may be able to develop even better strategies to monitor marsh vegetation, and the overall state of the marsh. As we continue to develop drone technology and techniques, drones may become an invaluable tool in tidal-marsh monitoring, and ecological restoration.

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Appendix 1.

Exclosure Materials

Ten-foot pieces of 3-inch perforated PVC pipe and 4-foot pieces of 1 x 2 wooden stakes were cut on a 45° angle to permit easy pushing in the muddy substrate. The PVC pipe had two sets of parallel holes drilled at 1.3 m and 1.6 m from the sharpened base of the pipes, then cleaned before installation. Cable was attached using a series of knots to secure each line. Four lines of cable were tied to each corner to provide stability and easy repair.

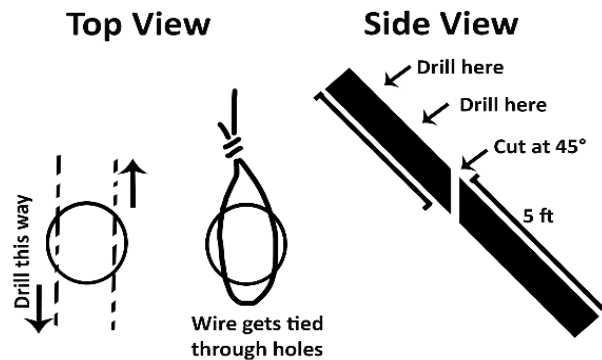


Figure A1- 1. Diagram of where and how holes were drilled and cut in PVC pipes for the exclosures on the Westham Island project site for June to November 2020.

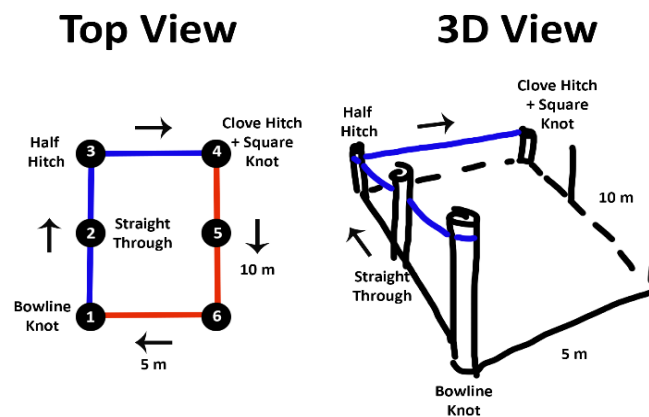


Figure A1- 2. Wiring schematic for CAGO and SNGO exclosures at Westham Island study site in the Fraser River Estuary (Lower Mainland of BC) June to November 2020.

Appendix 2.

Photo Monitoring Examples



Figure A2- 1. Photos showing change in Marsh Edge Block 1 CAGO exclosures at Westham Island study site June to November 2020.



Figure A2- 2. Photos showing change in Marsh Edge Block 1 open exclosures at Westham Island study site June to November 2020.

Appendix 3.

Full R Script for Statistical Analyses

```
> #use this one
> summary(aov(VISUAL...COVER ~ TREATMENT*DATE+as.factor(BLOCKS),ME))
      Df Sum Sq Mean Sq F value    Pr(>F)
TREATMENT      3      907      302.3      3.110  0.03559 *
DATE            3     3898     1299.5     13.366 2.27e-06 ***
as.factor(BLOCKS) 3     1614      538.2      5.535  0.00254 **
TREATMENT:DATE   9     1527      169.7      1.746  0.10634
Residuals      45     4375       97.2
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> summary(aov(DRONE...COVER ~ TREATMENT*DATE+as.factor(BLOCKS),ME))
      Df Sum Sq Mean Sq F value    Pr(>F)
TREATMENT      3      920      306.8      3.598  0.02053 *
DATE            3     3395     1131.7     13.273 2.44e-06 ***
as.factor(BLOCKS) 3     1562      520.8      6.108  0.00141 **
TREATMENT:DATE   9     1473      163.7      1.920  0.07326 .
Residuals      45     3837       85.3
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> T.HSD = TukeyHSD(aov(VISUAL...COVER ~ TREATMENT*period+as.factor(BLOCKS),ME), o
rder=TRUE)
Warning messages:
1: In replications(paste("~", xx), data = mf) :
  non-factors ignored: period
2: In replications(paste("~", xx), data = mf) :
  non-factors ignored: TREATMENT, period
3: In TukeyHSD.aov(aov(VISUAL...COVER ~ TREATMENT * period + as.factor(BLOCKS),
:
  'which' specified some non-factors which will be dropped
> T.HSD
Tukey multiple comparisons of means
 95% family-wise confidence level
factor levels have been ordered

Fit: aov(formula = VISUAL...COVER ~ TREATMENT * period + as.factor(BLOCKS), data
= ME)

$TREATMENT
      diff      lwr      upr      p adj
C-S    5.8125 -4.2017368 15.82674 0.4216508
N-S    6.4375 -3.5767368 16.45174 0.3312714
O-S   10.5625  0.5482632 20.57674 0.0350006
N-C    0.6250 -9.3892368 10.63924 0.9983655
O-C    4.7500 -5.2642368 14.76424 0.5932302
O-N    4.1250 -5.8892368 14.13924 0.6955672

$`as.factor(BLOCKS)`
      diff      lwr      upr      p adj
1-2    7.204457 -2.8097802 17.21869 0.2369724
3-2    9.375000 -0.6392368 19.38924 0.0742573
4-2   13.875000  3.8607632 23.88924 0.0030306
3-1    2.170543 -7.8436934 12.18478 0.9391516
4-1    6.670543 -3.3436934 16.68478 0.3005458
4-3    4.500000 -5.5142368 14.51424 0.6345938
```

```

> par(mfrow=c(1,1))
> plot(T.HSD)
> #BLOCK 1
> anova(lm(VISUAL...COVER ~ TREATMENT,ME1))
Analysis of Variance Table

Response: VISUAL...COVER
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3  682.25  227.42  1.0621 0.4014
Residuals 12 2569.50  214.12
> #BLOCK 2
> anova(lm(VISUAL...COVER ~ TREATMENT,ME2))
Analysis of Variance Table

Response: VISUAL...COVER
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3  571.69 190.562  2.1773 0.1437
Residuals 12 1050.25  87.521
> #BLOCK 3
> anova(lm(VISUAL...COVER ~ TREATMENT,ME3))
Analysis of Variance Table

Response: VISUAL...COVER
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3  273.19  91.063  0.5064 0.6851
Residuals 12 2157.75 179.812
> #BLOCK 4
> anova(lm(VISUAL...COVER ~ TREATMENT,ME4))
Analysis of Variance Table

Response: VISUAL...COVER
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3 1604.2  534.73  3.5457 0.04799 *
Residuals 12 1809.8  150.81
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> boxplot(VISUAL...COVER ~ TREATMENT*period, data=ME, xlab = "Date", ylab = "Visual % Cover", col = c("violetred", "steelblue1", "salmon1", "palegoldenrod"))
> #use this one
> summary(aov(VISUAL...COVER ~ TREATMENT*DATE+as.factor(BLOCKS),MF))
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3  294.42  98.14  68.255 < 2e-16 ***
DATE       3  149.67  49.89  34.698 9.04e-12 ***
as.factor(BLOCKS) 3  4.05  1.35  0.938 0.43
TREATMENT:DATE  9  281.14  31.24  21.725 1.35e-13 ***
Residuals  45  64.70  1.44
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> summary(aov(DRONE...COVER~ TREATMENT*DATE+as.factor(BLOCKS),MF))
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3  269.02  89.67  65.261 < 2e-16 ***
DATE       3  149.23  49.74  36.201 4.65e-12 ***
as.factor(BLOCKS) 3  8.71  2.90  2.113 0.112
TREATMENT:DATE  9  280.60  31.18  22.689 6.27e-14 ***
Residuals  45  61.83  1.37
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> summary(aov(STEM.COUNT~ TREATMENT*DATE+as.factor(BLOCKS),MF))
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3  69484  23161  37.396 2.78e-12 ***
DATE       3  98719  32906  53.130 7.87e-15 ***
as.factor(BLOCKS) 3  3025  1008  1.628 0.196
TREATMENT:DATE  9  88537  9837  15.884 2.80e-11 ***
Residuals  45  27871  619

```



```

---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> T.HSD = TukeyHSD(aov(VISUAL...COVER ~ TREATMENT*period+as.factor(BLOCKS),MF), o
rder=TRUE)
Warning messages:
1: In replications(paste("~", xx), data = mf) :
  non-factors ignored: period
2: In replications(paste("~", xx), data = mf) :
  non-factors ignored: TREATMENT, period
3: In TukeyHSD.aov(aov(VISUAL...COVER ~ TREATMENT * period + as.factor(BLOCKS),
:
  'which' specified some non-factors which will be dropped
> T.HSD
  Tukey multiple comparisons of means
    95% family-wise confidence level
    factor levels have been ordered

Fit: aov(formula = VISUAL...COVER ~ TREATMENT * period + as.factor(BLOCKS), data
= MF)

$TREATMENT
      diff      lwr      upr      p adj
S-O  8.0000 -42.931305  58.93131 0.9755294
C-O 63.4375  12.506195 114.36881 0.0089410
N-O 74.8750  23.943695 125.80631 0.0014804
C-S 55.4375   4.506195 106.36881 0.0278159
N-S 66.8750  15.943695 117.80631 0.0053124
N-C 11.4375 -39.493805  62.36881 0.9332891

> par(mfrow=c(1,1))
> plot(T.HSD)
> #BLOCK 1
> anova(lm(VISUAL...COVER ~ TREATMENT,MF1))
Analysis of Variance Table

Response: VISUAL...COVER
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3  43.687  14.5625  1.9915 0.1691
Residuals 12  87.750   7.3125
> #BLOCK 2
> anova(lm(VISUAL...COVER ~ TREATMENT,MF2))
Analysis of Variance Table

Response: VISUAL...COVER
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3    64  21.3333  3.0843 0.06811 .
Residuals 12    83   6.9167
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> #BLOCK 3
> anova(lm(VISUAL...COVER ~ TREATMENT,MF3))
Analysis of Variance Table

Response: VISUAL...COVER
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3   90.25  30.083  2.3215 0.1269
Residuals 12 155.50  12.958
> #BLOCK 4
> anova(lm(VISUAL...COVER ~ TREATMENT,MF4))
Analysis of Variance Table

Response: VISUAL...COVER
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT  3 110.25  36.750  2.836 0.0829 .

```

```

Residuals 12 155.50 12.958
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> # VD T test
> t.test(VD$VISUAL...COVER,VD$DRONE...COVER)

      Welch Two Sample t-test

data:  VD$VISUAL...COVER and VD$DRONE...COVER
t = 0.58349, df = 253.38, p-value = 0.5601
alternative hypothesis: true difference in means is not equal to 0
95 percent confidence interval:
 -2.522071  4.645798
sample estimates:
mean of x mean of y
 16.39062  15.32876

> #new stats for blocks
> summary(aov(VISUAL...COVER ~ TREATMENT*DATE+as.factor(BLOCKS),ME2m))
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT    3 1447.8   482.6   5.477 0.00651 **
DATE          1  525.1   525.1   5.959 0.02407 *
as.factor(BLOCKS) 3  776.3   258.8   2.937 0.05825 .
TREATMENT:DATE  3  111.6    37.2   0.422 0.73915
Residuals    20 1762.3    88.1
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> T.HSD1 = TukeyHSD((aov(VISUAL...COVER ~ TREATMENT*period,ME2m)))
Warning messages:
1: In replications(paste("~", xx), data = mf) :
  non-factors ignored: period
2: In replications(paste("~", xx), data = mf) :
  non-factors ignored: TREATMENT, period
3: In TukeyHSD.aov(aov(VISUAL...COVER ~ TREATMENT * period, ME2m), :
  'which' specified some non-factors which will be dropped

> T.HSD1
      Tukey multiple comparisons of means
      95% family-wise confidence level

Fit: aov(formula = VISUAL...COVER ~ TREATMENT * period, data = ME2m)

$TREATMENT
      diff          lwr          upr          p adj
N-C    1.00000 -13.83837 15.838374 0.9977176
O-C   -4.37500 -19.21337 10.463374 0.8506336
S-C  -16.71429 -32.07346 -1.355109 0.0291035
O-N   -5.37500 -20.21337  9.463374 0.7555696
S-N  -17.71429 -33.07346 -2.355109 0.0192129
S-O  -12.33929 -27.69846  3.019891 0.1492572

> summary(aov(VISUAL...COVER ~ TREATMENT*DATE+as.factor(BLOCKS),ME2m2))
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT    3 1506.6   502.2   6.780 0.00542 **
DATE          1  797.0   797.0  10.759 0.00597 **
as.factor(BLOCKS) 2  308.7   154.3   2.083 0.16411
TREATMENT:DATE  3  243.2    81.1   1.095 0.38628
Residuals    13  962.9    74.1
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
> T.HSD2 = TukeyHSD(aov(VISUAL...COVER ~ TREATMENT*period,ME2m2), order=TRUE)
Warning messages:
1: In replications(paste("~", xx), data = mf) :

```

```

non-factors ignored: period
2: In replications(paste("~", xx), data = mf) :
non-factors ignored: TREATMENT, period
3: In TukeyHSD.aov(aov(VISUAL...COVER ~ TREATMENT * period, MF2m), :
'which' specified some non-factors which will be dropped

> T.HSD2
Tukey multiple comparisons of means
95% family-wise confidence level
factor levels have been ordered

Fit: aov(formula = VISUAL...COVER ~ TREATMENT * period, data = ME2m2)

$TREATMENT
      diff      lwr      upr      p adj
O-S 16.366667 -2.4146303 35.14796 0.1011262
N-S 19.366667  0.5853697 38.14796 0.0418510
C-S 21.533333  2.7520364 40.31463 0.0212641
N-O  3.000000 -14.9072640 20.90726 0.9645251
C-O  5.166667 -12.7405973 23.07393 0.8483613
C-N  2.166667 -15.7405973 20.07393 0.9860208

> summary(aov(VISUAL...COVER ~ TREATMENT*DATE+as.factor(BLOCKS), MF2m))
      Df Sum Sq Mean Sq F value Pr(>F)
TREATMENT    3  537.3   179.11   67.281 6.14e-11 ***
DATE          1   63.3    63.28   23.770 8.05e-05 ***
as.factor(BLOCKS) 3    2.8     0.95    0.356  0.7853
TREATMENT:DATE  3   35.8    11.95    4.488  0.0139 *
Residuals    21   55.9     2.66
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

> T.HSD3 = TukeyHSD(aov(VISUAL...COVER ~ TREATMENT*period, MF2m), order=TRUE)
Warning messages:
1: In replications(paste("~", xx), data = mf) :
non-factors ignored: period
2: In replications(paste("~", xx), data = mf) :
non-factors ignored: TREATMENT, period
3: In TukeyHSD.aov(aov(VISUAL...COVER ~ TREATMENT * period, MF2m), :
'which' specified some non-factors which will be dropped

> T.HSD3
Tukey multiple comparisons of means
95% family-wise confidence level
factor levels have been ordered

Fit: aov(formula = VISUAL...COVER ~ TREATMENT * period, data = MF2m)

$TREATMENT
      diff      lwr      upr      p adj
S-O 0.500 -1.658035  2.658035 0.9182674
N-O 8.375  6.216965 10.533035 0.0000000
C-O 8.500  6.341965 10.658035 0.0000000
N-S 7.875  5.716965 10.033035 0.0000000
C-S 8.000  5.841965 10.158035 0.0000000
C-N 0.125 -2.033035  2.283035 0.9985056

>

```

Appendix 4.

Growing *S. pungens* With Different Stressors

I grew five small 10 cm plugs of *S. pungens* in a controlled environment and performed different treatments on each plug. The treatments performed ranged from control, “rhizome cutting” using scissors, “grazing” by pulling above ground vegetation with fingers, “rhizome cutting and grazing”, and covering with chicken wire mesh. The plugs were placed in nursery pots on a tray above a 25 g brackish aquarium with a salinity ranging from 12 to 14 ppt. the tray had a hole cut out with a floating standpipe inserted keeping the water level at either a 1 cm or 6 cm wetted depth, a return pump was used to draw water from the aquarium into the tray. The aquarium was stocked with Amano shrimp (*Caridina multidentate*), mollies (*Poecilia sphenops*), and various brackish water plants. The light source came from an Aqua Design Amano grand solar plant grow light, which also kept water temperature at around 22 °C. I started growing the plugs on 30 June 2020. Scaling up this project may be of interest to the marsh recession project in the future.

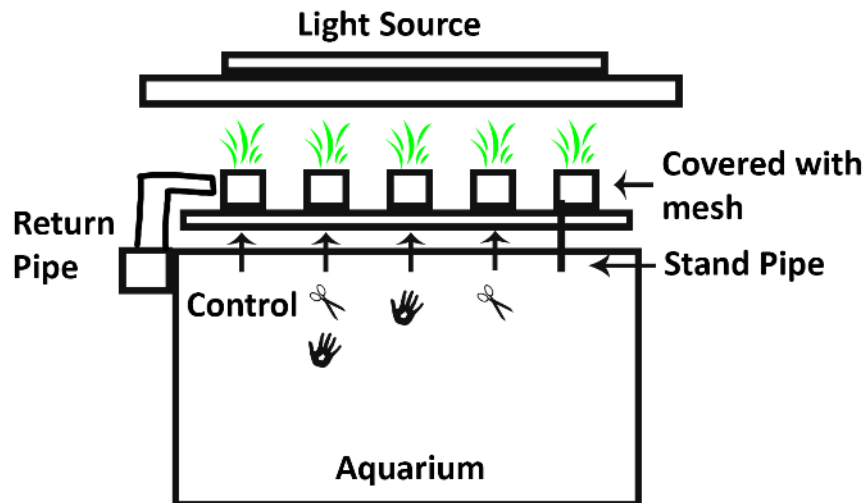


Figure A4- 1. Set up of pilot *S. pungens* growing experiment with the scissor icon representing rhizome cutting, and the hand icon representing grazing via pulling with fingers. The treatments were randomly assigned via a random number generator for the project at the Westham Island study site in the Fraser River Estuary (Lower Mainland of BC) June to November 2020.

Appendix 5.

Wildlife Camera Traps

I installed six 18 MP Bushnell Strike Force Apex trap cameras, mounted on 3-m PVC poles that were buried about 1-m in the mud and secured with stainless steel guy wires, four associated with each of the four study blocks, and two associated with goose deterrents and a deterrent control in the WI study site. The cameras were set to fire in a burst of four shots if both motion and IR sensors were triggered. The traps were set up from June to November, and memory cards were collected monthly. With the help of technicians, each photo was reviewed, and I recorded the location, date, time and number and kind of species in each photo. The data was then summarized by comparing the percent number of geese per location.

Although both species of geese use the site, their distribution was not even. Of the blocks from the enclosure studies, blocks 2 and 4 appeared to be where geese spent most of their time. While in the deterrent test, more geese occurred in the area with no installed deterrents than that with deterrents installed (Appendix 10.)

Although there have been studies using wildlife cameras in mangrove estuaries (Yaney-Keller 2018); most use seems to be in terrestrial ecosystems (Silver *et al.* 2004; Olson 2012; McCleary *et al.* 2014). However, the extreme conditions in the WI tidal marsh caused all the cameras to fail, and for one to be destroyed by an unknown cause. The lack of tidal-marsh specific camera equipment led to many misfires mostly of the tides going up and down, suggesting that the camera is likely capturing the movement and heat differences in the waves. Additionally, from June to October, I noticed a marked decrease in camera performance, seen by misfiring and blurry images. If this study were to continue, I would suggest a more robust camera system to mitigate the extreme conditions in the WI tidal marsh.



Figure A5- 1. Sample of wildlife trap camera photo showing what was recorded in the Westham island study site (Bushnell Apex camera 2020).



Figure A5- 2. Bushnell Strike Force Apex trap camera, broken for unknown reason in the Westham island goose herbivory deterrent study site (2020).

Appendix 6.

Pixel Correction of Drone-based Photos

I used Adobe Photoshop CC to perform colour correction and contrast increasing technique to enhance separation of colours among the ground, *S. pungens*, and other aquatic plant species (Iqbal *et al.* 2010). These allow us to show *S. pungens* in a more easily differentiated green, as compared to the yellow tones of the substrate, and the reddish tones of aquatic vegetation.

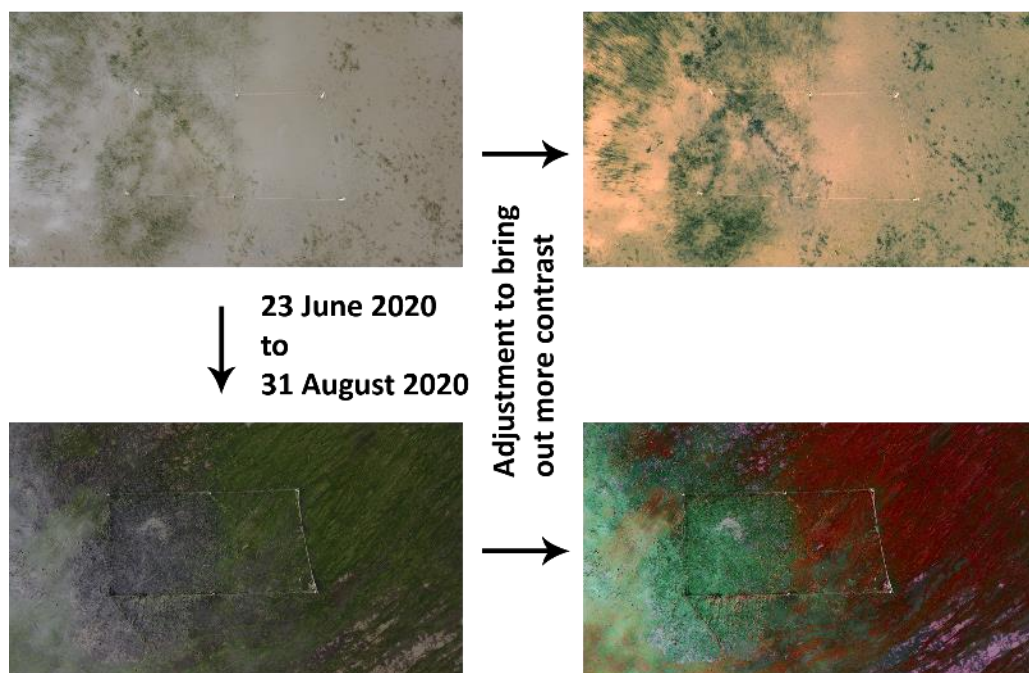


Figure A6- 1. Sample of drone aerial photo showing colour mask and contrast increase to separate different features of the marsh from each other. *S. pungens* tend to be shown in greens, red and black represent aquatic vegetation, and yellows represent mud in the Westham island study site 2020.

Appendix 7.

Visual Surveys of Goose Body Condition

Observational work on my ARP was done using a Sony rx10 IV camera with 24-mm – 600-mm equivalent focal length and a 1.0” sensor. When standardized, abdominal profiles of geese can provide an accurate metric of body fat (Owen 1981; Johnson and Sibly 1993; Féret *et al.* 2005; Madsen and Klaassen 2006; Clausen and Madsen 2014). However, studies varied in the categories used to describe abdominal profiles (Owen 1981; Johnson and Sibly 1993; Féret *et al.* 2005; Madsen and Klaassen 2006). So, I simplified the categories as “thin”, “medium”, or “fat”. Twice a month, from June to October, I observed ten individual CAGO in three locations per day. I then plotted the means per observation day in excel.

Over the course of the study, I noticed a decrease of “fat” CAGO and an increase in both “thin” and “medium” CAGO. This shift appears to coincide with farmers using bird deterrent methods to keep geese away from their crops in the surrounding farmlands of Westham island.

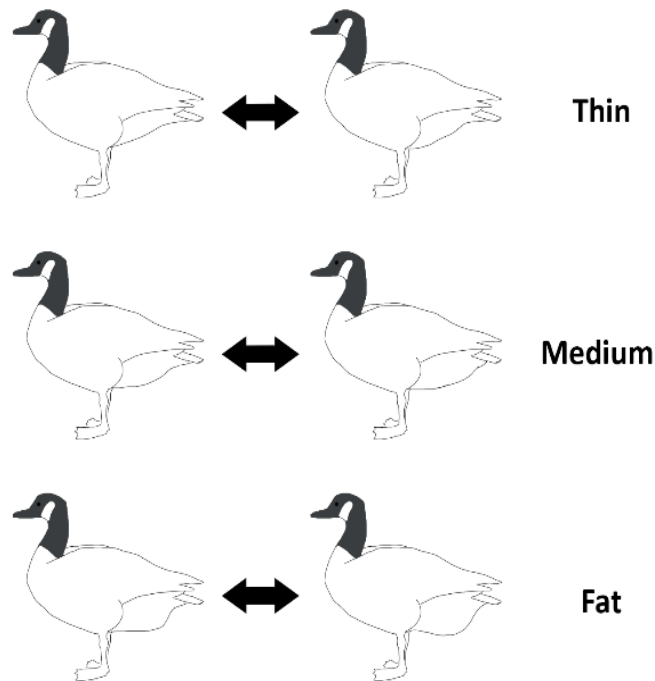


Figure A7- 1. Classes of goose abdominal profile ranges used for this ARP on Westham island. Arrows signify that any geese in between the two templates fall into that respective class.

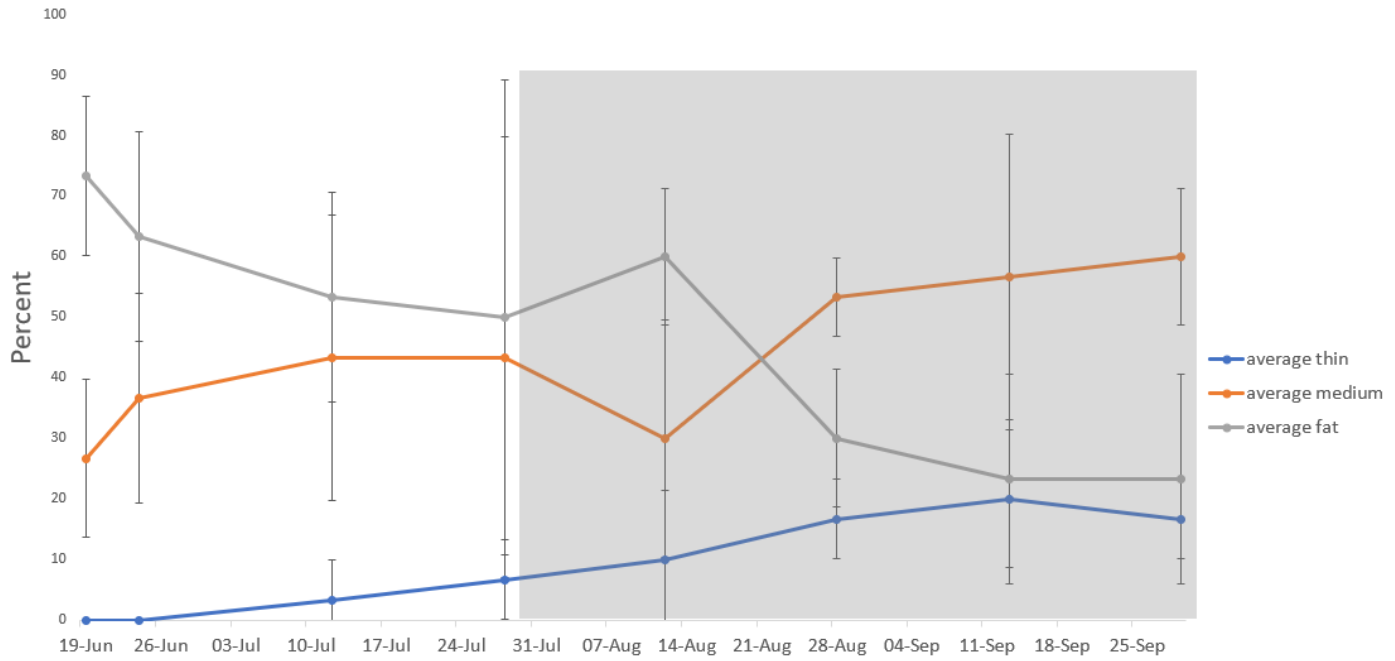


Figure A7- 2. Abdominal goose profiles for CAGO with grey polygon representing when the farmers were actively using goose deterrents. On the Westham Island study site in the Fraser River Estuary (Lower Mainland of BC) June to November 2020.

Appendix 8.

Goose Fecal Matter Surveys

I observed goose fecal matter to identify their diets and serve as another metric to confirm goose presence. Green fecal matter may be associated with agricultural and recreational fields, while black colours may be associated with grazing in tidal marsh (Boyd pers. comm.; Balke, pers. comm.). I categorized each fecal sample as “green”, “black”, or “mixed”. Twice a week, from June to November, I surveyed a randomized 10-m transect and observed goose fecal matter half a metre on each side. Each transect was associated with a study block, with three transects observed per sampling day. During the study period, green and black fecal matter decreased, while mixed fecal matter increased.



Figure A8- 1. Examples of different colours of goose fecal matter taken at the Westham Island study site in the Fraser River Estuary (Lower Mainland of BC) for June to November 2020

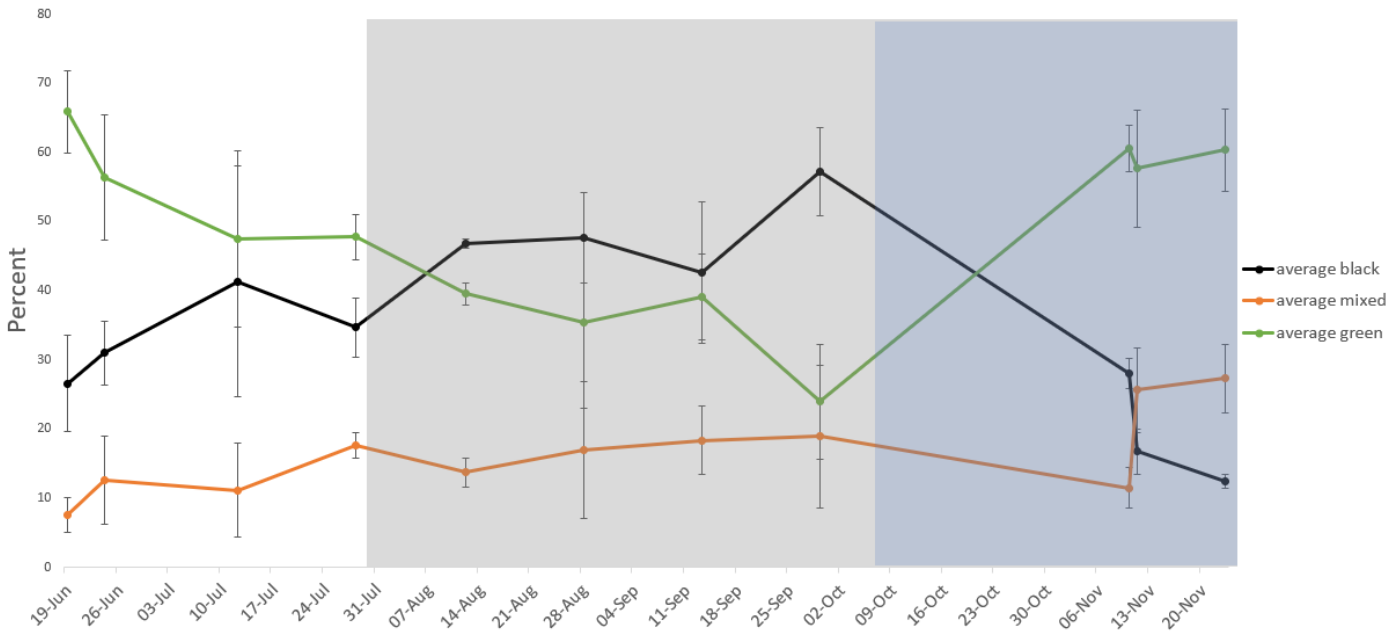


Figure A8- 2. Fecal matter colouration with grey polygon representing when the farmers were actively using goose deterrents, and blue polygon representing arrival of large SNGO flocks. On the Westham Island study site in the Fraser River Estuary (Lower Mainland of BC) June to November 2020.

Appendix 9.

Pilot Studies of Goose Herbivory Deterrents

I assessed the ability of different management techniques to reduce goose herbivory. I focused on using novel barriers to goose herbivory as passive methods, and active methods such as hazing and creating loud noises. The goal was to find herbivory deterrents that can be deployed in a large area and require little maintenance. Results of these pilot projects may guide management prescriptions to ensure the persistence of common three-square bulrush and possibly other tidal marsh vegetation communities. These methods were split into passive and active deterrents.

I used whips, whistles, and birds of prey as active forms of goose herbivory deterrent. Falconry has been used to manage many species of nuisance birds (Baxter and Allan 2006; Belant 2011). Working in conjunction with Pacific Northwest Raptors (PNWR), I observed Harris Hawks (*Parabuteo unicinctus*), Red-tailed Hawks (*Buteo jamaicensis*), and Bald Eagles (*Haliaeetus leucocephalus*) flown at geese on YVR airport. I also used whips and whistles as acoustic deterrents in the WI study site. Goose behaviour was observed for three minutes after the disturbance detected, and was categorized as “no response”, “fight”, or “flight”.

For passive deterrents, I installed different barrier structures in multiple locations in the WI site. Deterrents were made plastic snow fencing secured with wooden stakes in various configurations. I used four configurations consisting of: fencing placed directly on the ground, fencing suspended 1-m in the air parallel with the ground, fencing perpendicular to the ground, and a 2-m by 3-m fully fenced enclosure. I also installed a plastic coyote decoy, but that was lost to the marsh. The original designs included the use of metal chicken wire to reduce plastic waste, they were removed as they may have acted like gill nets and were detrimental to fish populations. I also installed a 10-m x 10-m enclosure using only four PVC poles to determine if a cheaper design would be suitable for large areas. I used photo monitoring and wildlife camera traps to determine the effectiveness of these pilot deterrents. Preliminary observations indicate that both geese species spent more time in the control area, as compared to the area where deterrents were being tested (Appendix 10).



Figure A9- 1. Passive deterrent structures placed at Westham Island Tidal marsh. (A) installing deterrent that directly lies on the ground, (B) Fencing suspended 1 m above vegetation, (C) Fencing parallel above vegetation, (D) Fencing all around vegetation, (E) Large 10 m x 10 m enclosure, (F) Coyote decoy (Photos by G. Gan and D. Mulhert taken on July 2020 on Westham Island tidal marsh)

For active deterrents, geese mostly responded to the presence of predatory birds on the WI tidal-marsh study site. *H. leucocephalus* events were the most effective, consistently causing flight responses in CAGO and SNGO. This matches with the opinions of PNWR staff who say that only *H. leucocephalus* and dog hazing consistently clear SNGO and CAGO (E. Fleming, Pacific Northwest Raptors, pers. comm.).

Table A9-1. Analysis of effectiveness of active deterrents in the Westham Island study site from June to November 2020, with deterrent events on the left panel, and goose responses on the right panel.

Event Number	Event	Goose species	Reaction	Number of flight response	Number of fight response	Number of no response
1	BAEA	CAGO	FLIGHT	10	2	4
2	RTHA	CAGO	no response			
3	COYOTE	CAGO	FLIGHT	% flight	% fight	% no
4	PEFA	CAGO	FIGHT	62.5	12.5	25
5	BAEA	CAGO	FLIGHT			
6	BAEA	CAGO	FLIGHT	% flight caused by BAEA		
7	NOHA	CAGO	no response	50		
8	RTHA	CAGO	no response			
9	BAEA	SNGO	FLIGHT			
10	BAEA	SNGO	FLIGHT			
11	WHIP AND WISTLE	CAGO	FLIGHT			
12	FARM	CAGO	FLIGHT			
13	FARM	CAGO	FLIGHT			
14	WHIP AND WISTLE	CAGO	FIGHT			
15	FARM	CAGO	no response			
16	WHIP AND WISTLE	SNGO	FLIGHT			

Appendix 10.

Wildlife Camera Trap Summary Results

I analyzed all the photos from the wildlife trap cameras based on location and type of wildlife detected. I separated the animals detected into four categories: CAGO, SNGO, unknown geese, and MIXED containing all other herbivorous wildlife captured by the camera. I then calculated where the animals spent most of their time on the study site, by comparing how many sightings occurred at a study location to the total number of sightings for that category. In this case I found that CAGO and SNGO occur more frequently in Blocks 2 and 4. The camera data also suggests the effectivity of the deterrents, as all species occurred more frequently where deterrents were not deployed as compared where deterrents were used.

Table A10-1. Summary of wildlife trap camera data for both exclosure experiments and passive deterrents in the Westham Island study site from June to November 2020.

BLOCK	CAGO	SNGO	unknown	MIXED	% CAGO	% SNGO	% unknown	% MIXED
B1	493	28	0	1072	9.4	2.1	0	32.1
B2	1037	393	2	28	19.8	28.9	100	0.8
B3	593	110	0	104	11.3	8.1	0	3.1
B4	1080	340	0	1103	20.6	25.0	0	33.1
Deterrents	378	7	0	676	7.2	0.5	0	20.3
Deterrent controls	1651	483	0	352	31.6	35.5	0	10.6
Totals	5232	1361	2	3335				