A Report for:
Nova Scotia Department of
Transportation and Infrastructure Renewal

Post-Restoration Monitoring (Year 4) of the Cogmagun River Salt Marsh Restoration Project

Prepared by:
Tony M. Bowron, Nancy C. Neatt, Jennifer M. Graham, Dr. Danika van Proosdij, and Dr. Jeremy Lundholm

CBWES Inc.

March 2014
Publication No. 39

Table of Contents

List of Figures

List of Tables

Executive Summary

Restoration of a more natural hydrological regime to the Cogmagun River site was carried out in September 2009. The restoration design, which was developed by CBWES Inc. in collaboration with NS Transportation and Infrastructure Renewal (NSTIR), Ducks Unlimited Canada (DUC) and the property owners (formerly Red Fox Farm, now Abundant Acres), consisted of the creation of a 60 m breach in the dyke and the excavation of a central channel.

A monitoring program consisting of one year pre- and five year post-restoration monitoring was developed for the project. The purpose of the monitoring program was to provide a scientific record of habitat conditions at both the restoration (COG) and reference (COG-R) sites in order to quantify environmental change in response to restoration, to facilitate adaptive management actions if required, and to verify project success (restored marsh exhibits similar physical and biological characteristics and functional processes as the reference site).

The fourth year of post-restoration ecological monitoring was conducted by CBWES during the period of May 2013 through to December 2013, with a winter site visit conducted in February 2014. Data were collected for geospatial attributes, soil and sediments, vegetation, nekton and benthic invertebrates at the restoration site and an adjacent tidal wetland reference site, the results of which are presented in this report and summarized below.

Geospatial Attributes

The revised estimate, based on 2012 (year 3) data, put the total marsh area (including dyke and fringe marsh) at approximately 7.68 ha, and the area of recovering tidal wetland inside the dyke at 4.8 ha (Section 5.0).

Hydrology

<u>Hydroperiod and Tidal Signal</u>:

The monitoring protocol did not include measuring the hydroperiod and tidal signal during the fourth year of post-restoration monitoring. These will be measured during the fifth (2014) monitoring year.

<u>Water Quality</u>

Water quality sampling of the surface floodwaters was conducted concurrently with nekton sampling. All parameters measured were well within the polyhaline range expected for the Bay of Fundy.

Soils and Sediments

<u>Pore Water Salinity</u>:

Salinity levels have increased at the restoration site following restoration, while remaining quite consistent over time at the reference site. Following an initial spike in salinity levels during the first year following restoration, pore water salinity levels and patterns resemble those of the reference site four years post. This would suggest that pore water salinity conditions at the restoration site continues to stabilize (no statistically significant change detected between 2011, 2012 and 2013), and approach parity with those of the reference condition.

<u>Sediment Accretion and Elevation</u>:

By year four, all elevation monitoring stations at COG showed equilibrium to positive changes in surface elevation. Elevation changes continue to be driven by both sediment accretion and below ground processes. Most of the dead plant material that had blanketed much of the back portion of the site had either been exported or decayed *in situ* by year four, enabling new vegetative growth which, in turn, resulted in increased deposition rates. By 2012-13, the rates of change in surface elevation are most comparable to values observed at the reference site and seem to be following the shift in vegetative community structure recorded during this time period. However, below ground processes continue to play a greater role in elevation change at the restoration site.

It appears that year four was a turning point towards soil conditions more comparable to the reference site and may be associated with a shift in vegetative community structure. Year four saw distinctive shift in the vegetation to halophyte dominance, which was reflected in the trends in sediment accretion and elevation.

<u>Soil Characteristics</u>:

Soil analysis was not part of the fourth year post-restoration monitoring activities.

<u>Soil Chemistry (Redox Potential)</u>:

Anaerobic conditions were recorded for all but one location at COG and all locations at COG-R. At depth, the dominant reactions that were occurring at COG were comparable to those at the reference site (nitrate and manganese (IV)). However there was a difference in the surface level reactions with iron (III) at COG compared to nitrate and manganese (IV) at COG-R. Overall, the values for redox potential recorded for both sites indicate moderate anaerobic stress on vegetation and potential for moderate decomposition rates within the soil under anaerobic conditions. The values indicate appropriate drainage and a lack of water logging at the sampling locations, therefore the high anaerobic conditions needed for high sulfide levels to accumulate were not occurring.

Vegetation

There has been a steady increase in salt marsh (halophyte) vegetation at the restoration site. Low marsh communities are dominant in some parts of the site, but there has also been an increase in high marsh vegetation (*Spartina patens* community type). Significant increases in halophytic abundance (matched reference) and richness (exceeded reference) were recorded in year four. The restoration site clearly overlapped with the reference site in low marsh areas, and contained a large amount of *S. patens* (representative of high marsh conditions) usually mixed with *S. alterniflora*. However, the site still lacked the *Juncus gerardii* community present at the reference site. Many species of upland or freshwater vegetation that were present prior to restoration are now absent. Following an initial increase in the amount of unvegetated area immediately following restoration, this has dropped to almost nil by year four, a pattern shared by the amount of dead plant material.

The *Phragmites australis* that was recorded within the restoration site in 2011 was positively identified as the native variety.

Nekton

Species richness and relative abundance continued to increase four years following hydrological restoration. *Fundulus heteroclitus* (mummichog) was the dominant species captured across all years. *Menidia menidia* (Atlantic silverside), a migratory estuarine species, was the second most abundant species captured in both 2012 and 2013. Other higher order species such as *Microgadus tomcod*, *Anguilla rostrata* (American eel) and *Alosa pseudoharengus* (alewife) were also captured in greater numbers in 2013. The presence of these species within the restoration site could be taken as an indicator of recovering/improving tidal wetland habitat conditions.

Benthic and Other Aquatic Invertebrates

Benthic communities at both sites have consistently included a mix of marine estuarine species, amphipods, and freshwater insects. Mean species richness for the reference site has remained relatively constant over time. Diversity has increased at COG following restoration. Mean abundance at both sites varied considerably over the course of the monitoring program, but with the restoration site showing an increasing trend towards similar numbers as the reference four years out. Despite the apparent variation in the benthic community structure between the sites and years, the differences were not statistically significant.

The 2013 aquatic invertebrate samples contained a range of marine/estuarine and freshwater species. Diversity and abundance overall was low to moderate, as in previous years. A statistical comparison of the aquatic invertebrate samples found that although mean abundance and species richness of invertebrates within the restoration site continued to be higher than those from the reference in year four, the difference, unlike in previous years, was no longer statistically significant.

Summary

Habitat conditions at COG prior to restoration resembled in form and function a poorly drained brackish/freshwater environment dominated by areas of standing water (borrow pit), bare ground and a non-halophytic dominated vegetation community structure. Four years following re-establishment of a more natural hydrological regime to COG, the site has experienced a significant decrease in non-halophyte vegetation; an increase in salt marsh indicator species; consolidation of the marsh surface, increased sediment accretion and elevation; further development of the panne and tidal creek network; and continued stabilization of the main tidal channel. Physical and biological conditions at the reference site did not experience similar change over this time period, consistently exhibiting the full range of salt marsh conditions and species typical of a mature salt marsh/tidal wetland habitat.

Based on the documented changes in the physical and biological conditions over the four years following restoration, the site continues to proceed along an acceptable restoration trajectory. The size and placement of the dyke breach and excavated tidal channel in 2009 was sufficient to allow for the restoration of tidal flooding of the former salt marsh system. This re-establishment of regular tidal flow to the site and its reconnection to the broader Cogmagun River system, has initiated the recovery of tidal wetland conditions at the site as well as the availability and accessibility of the site for a range of estuarine species (vegetation, fish, birds).

It was anticipated that the new/restored wetland boundary would eventually encompass an area of tidal wetland between 3.78 ha and 4.8 ha. Based on the 2012 DEM, hydrology, and geo-referenced low-altitude aerial photography, the area flooded by the largest recorded tide was 4.16 ha.

The COG restoration site continues to exhibit considerable evidence of returning to its former tidal wetland condition and that following four years of recovery, the primary tidal wetland parameters are responding in a positive manner. Differences do persist between conditions at the restoration and reference sites; however, it appears that year four was a turning point for the restoration site towards conditions more comparable to the reference site. Significant changes in both the vegetation community structure and sediment conditions (above and below ground processes) that may be directly related to one another were recorded. The fifth year of post-restoration monitoring will enable us to examine this potential relationship further, verify that the site is following an acceptable restoration trajectory and the determination of the area of restored tidal wetland.

Acknowledgements

This project would not have been possible without the agreement, support, and design contributions by the wonderful folks at Red Fox Farm[1] (Jennifer Scott, Carolyn Green and Peter Wallace). Robert Fraser (Ducks Unlimited Canada) was also instrumental in seeing this project realized.

We also wish to thank Greg Baker (Saint Mary's University - SMU); Christa Skinner, Carly Wrathall, and Ben Lemieux; and Patrick Stewart (Envirosphere Consultants Limited) for their assistance with data collection, sample processing, analysis and report preparation.

Financial and in-kind supporters of this project include: Red Fox Farm (now Abundant Acres); NSTIR; Ducks Unlimited Canada; Natural Sciences and Engineering Research Council of Canada (NSERC); Nova Scotia Department of Economic and Rural Development; Human Resources and Skills Development Canada; Applied Geomatics Research Group, Centre of Geographic Sciences; Community-Based Environmental Monitoring Program (SMU), Maritime Provinces Spatial Analysis Research Centre (SMU) and the Intertidal Coastal Sediment Transport Research Unit (SMU).

---

[1] Farm is now owned by Jennifer (Scott) and David Greenberg and the name has changed to Abundant Acres.

1.0 The Cogmagun River Salt Marsh Restoration Project

1.1 Background

CBWES Inc. was commissioned in March 2009 by NSTIR to undertake the design, restoration and monitoring of the Cogmagun River Salt Marsh Restoration Project. The restoration site, formerly a 4.8 ha (12 acre) salt marsh along the Cogmagun River, was dyked in 1991 by DUC in order to create a freshwater pond for waterfowl. High maintenance costs associated with maintaining the dyke and water control structure and the challenge of preventing saltwater intrusion into the pond resulted in the decision by DUC and the property owners (formerly Red Fox Farm) in 2003 to cease all maintenance activities. Through a partnership with NSTIR, it was decided in 2009 to take a more active role in the restoration of the site. The restoration plan was developed by CBWES Inc. in collaboration with NSTIR, DUC and the property owners.

Restoration of a more natural hydrological regime to the site would result in the re-establishment of a tidal wetland (salt marsh) habitat condition similar to that which existed prior to dyking. In addition, the project would:

- Re-establish a more natural connection between the restoration site and the broader Cogmagun tidal river system;
- Restore fish passage to the site and access to the marsh surfaces (an increase in fish habitat);
- Create and enhance bird, waterfowl and wildlife habitat;
- Improve productivity and transport of materials (nutrients); and
- Allow the site to re-establish a natural process of succession.

To accomplish these goals, restoration activities included the following components:

- Removal of a portion (60 m) of the dyke that was constructed to create the impoundment and excavation of a primary tidal channel; and
- Development and implementation of a long-term pre- and post-restoration monitoring program to ensure project success.

Restoration (earthworks) was carried out in September 2009 and consisted of breaching the dyke and excavating a tidal channel, as described in the pre-restoration construction and monitoring report by Bowron et al. (2010). A pre-and post-restoration monitoring program was developed for the project based on the experience with similar projects in the region (Chapter 2). Baseline, pre-restoration monitoring, was conducted by CBWES Inc. during the summer of 2009 prior to restoration (Bowron et al. 2010). The fourth year of the post-restoration monitoring program was carried out by CBWES Inc. during the 2013 field season (May – December), with a structured winter walk conducted in February 2014.

The purpose of the monitoring program, and this years' phase of it, was to:

- Document the efficacy of the compensation being undertaken to restore the Cogmagun site;
- Determine the nature, extent and direction of change, in the physical, chemical and biological indicators being studied, as a result of the restoration activity; and

- Document restoration progress and determine project success (restored marsh exhibits similar physical, chemical and biological characteristics as the reference site), by comparing the post-restoration habitat conditions to those that were present prior to restoration and to those of an adjacent reference site.

The results of the fourth year of post-restoration monitoring are presented in this report.

All aspects of this project were conducted or supervised by CBWES staff and project partners, under contract to NSTIR. Field and laboratory work was carried out by: Tony M. Bowron, Nancy C. Neatt, Jennie M. Graham, Ben Lemieux, Christa Skinner, and Carly Wrathall with CBWES; Dr. Jeremy Lundholm, Dr. Danika van Proosdij, Greg Baker, and Emma Poirier with Saint Mary's University (SMU); and Patrick Stewart and Heather Levy (Envirosphere Consultants Ltd.).

1.2 CBWES Inc.

Since 2005, CBWES has been involved in the restoration and monitoring of ten salt marsh restoration projects within NS in collaboration with NSTIR[2]. These projects, in particular, the design and monitoring activities, have been presented by CBWES staff in poster and oral presentation formats at a number of regional, national and international scientific conferences[3]. Please contact CBWES for more information on these presentations. CBWES is committed to continuing to participate in important events such as these.

CBWES has a strong research partnership with SMU. Through this partnership, a number of undergraduate and graduate level research projects involving the restoration project sites have been supported. As a recognized Industrial Partner with the Natural Sciences and Engineering Research Council of Canada (NSERC), CBWES Inc. received NSERC grants for six of these projects (four undergraduate and two graduate). The resulting theses are available from the SMU library. Summaries of these salt marsh restoration research projects, as well as the non-NSERC funded current and completed projects are provided in Appendix A.

---

[2] Cheverie Creek, Walton River, Lawrencetown Lake, Smith Gut, St. Croix River, Cogmagun River, Antigonish Landing (in collaboration with CBCL Ltd.), Three Fathom Harbour, Tennycape and Morris Island (Bowron et al. 2011a,b; Bowron et al. 2012a,b; Bowron et al. 2013a,b,c; Bowron et al. 2014a,b; CBCL 2011; Neatt et al. 2013; van Proosdij et al. 2010; CBWES reports available for download at www.gov.ns.ca/tran/enviroservices/enviroSaltMarsh.asp)

[3] 6th Annual Atlantic Reclamation Conference (ARC 2013); Coastal and Estuarine Research Federation 22nd International Conference (CERF 2013); 2013 Mid-Atlantic Living Shorelines Summit (RAE 2013); Canadian Land Reclamation 37th National Conference (CLRA 2012); Atlantic Canada Coastal and Estuarine Science Society 2012 (ACCESS 2012); BoFEP's 9th Bay of Fundy Science Workshop (BoFEP 2011); Coastal and Estuarine Research Federation's 21st International Conference (CERF 2011); Restore America's Estuaries 5th National Conference on Coastal and Estuarine Habitat Restoration (RAE 2010); Atlantic Reclamation Conference (ARC 2008; 2009, 2010); Coastal and Estuarine Research Federation's 2009 International Conference (CERF 2009); BoFEP's 8th Bay of Fundy Science Workshop (BoFEP 2009); Canadian Water Resources Association - Maritime Water Resources Symposium (CWRA 2008); Atlantic Canada Coastal and Estuarine Science Societies' 2008 conference (ACCESS 2008); Estuarine Research Federations' 2007 International Conference (ERF 2007); Canadian Land Reclamation Associations National Conference (CLRA 2007); Ecology Action Centre's "Six Years in the Mud – Restoring Maritime Salt Marshes: Lessons Learned and Moving Forward" workshop (EAC 2007).

To date, two peer-reviewed papers have been published focusing on separate restoration projects. One was published in *Restoration Ecology* on the Cheverie Creek Restoration Project titled "*Macro-Tidal Salt Marsh Ecosystem Response to Culvert Expansion*" (Bowron et al. 2011a) and the second appeared in the journal *Ecological Engineering* on the Walton River Restoration Project titled *Ecological Re-engineering of a Freshwater Impoundment for Salt Marsh Restoration in a Hypertidal System* (van Proosdij et al. 2010). A third paper, "*Classification and environmental correlates of tidal wetland vegetation: implications for ecological restoration and monitoring"* is being peer-reviewed for publication in the journal *Estuaries and Coasts* (Porter et al. submitted). A book chapter has also recently been published titled "Chapter 13 – Salt Marsh Tidal Restoration in Canada's Maritime Provinces" in *Tidal Marsh Restoration: A Synthesis of Science and Management* (Roman et al. 2012). Abstracts for each of these publications appear in Appendix A.

2.0 Study Sites and Monitoring Program

2.1 Cogmagun Restoration Site

The Cogmagun River restoration site (COG) is a 6.9 ha (69,000 m^2; 17 acres) tidal wetland along the Cogmagun River in Hants County, NS. In 1991, DUC constructed a protective dyke and water control structure converting 4.8 ha of the site into a freshwater pond (impoundment) for waterfowl and wildlife (Figure 1 to Figure 3). The site was put forth as a potential salt marsh restoration/compensation project in 2009 through a partnership between NSTIR, DUC, Red Fox Farm, and CBWES.

Restoration took place on 22-23 September 2009 and consisted of the excavation of a 60 m breach along the downstream portion (western) of the dyke (Figure 4), the removal of the water control structure and the incorporation of the resulting hole into a tidal creek connecting the existing channel (borrow pit) within the site to the one in the fringe marsh (Figure 5 and Figure 6). Detailed information on the restoration design and construction phases of the project is provided in Bowron et al. (2010).

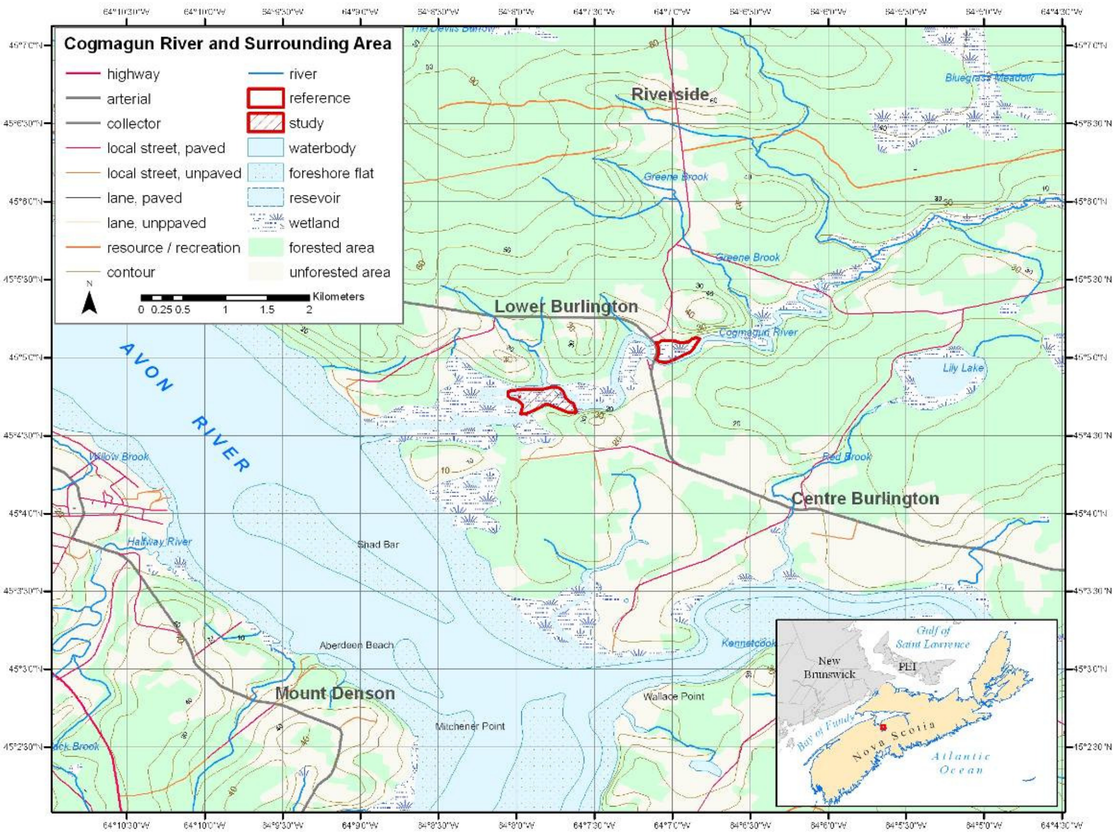


Figure 1 Location of COG within the Southern Bight of the Minas Basin watershed, Bay of Fundy, NS.

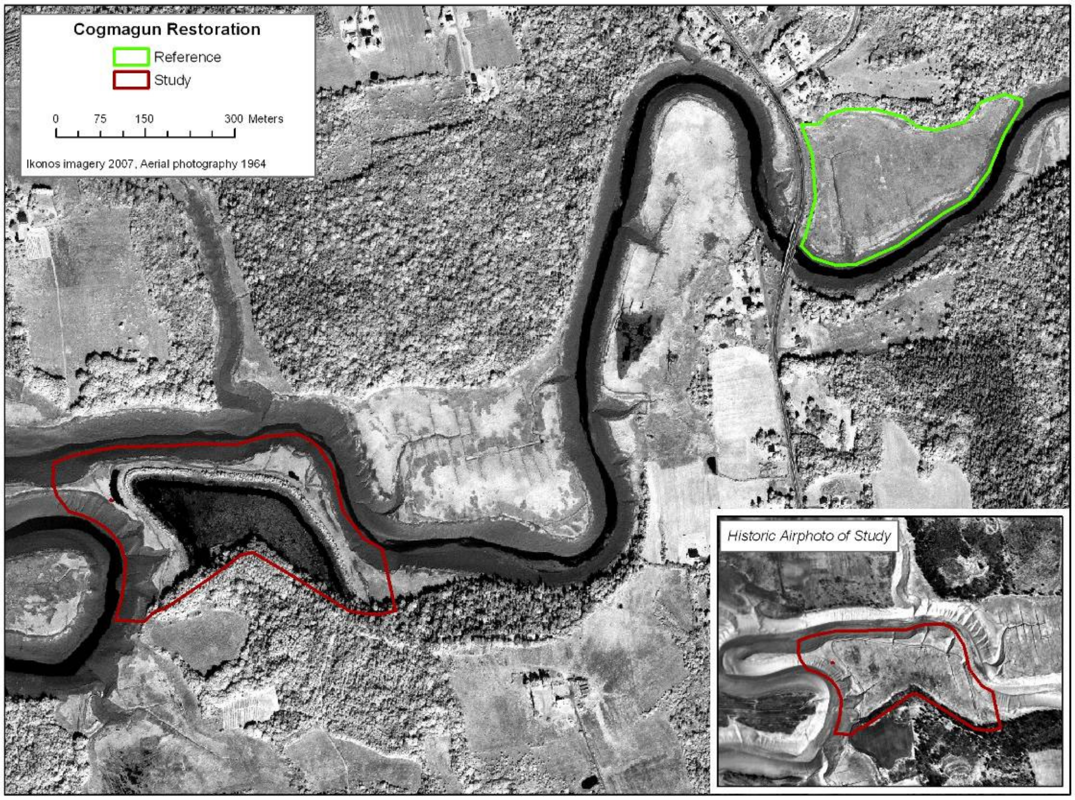


Figure 2 Aerial photograph of restoration site (red) and reference site (green). The inserted image is the restoration site prior to creation of impoundment (1982).

Figure 3 COG following the construction of the dyke and conversion to freshwater impoundment. 1992 aerial photograph.

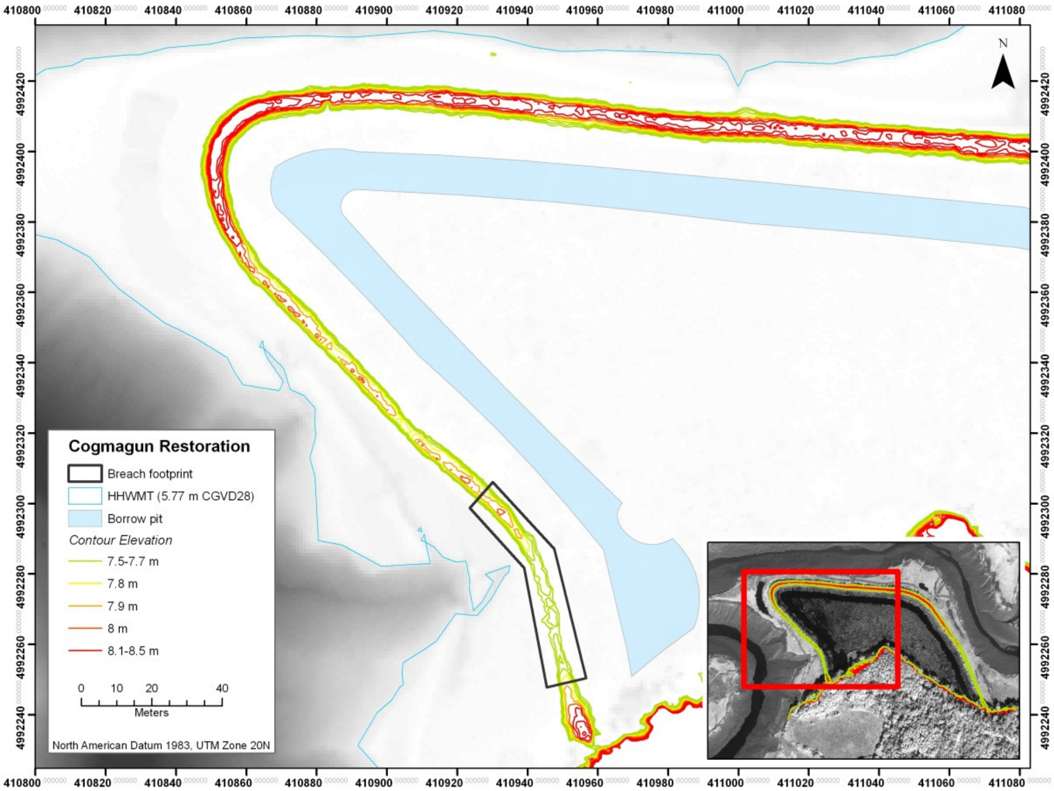


Figure 4 Dyke on the western side of the COG site with location of breach indicated by the dark blue box.

Figure 5 Completed breach (60 m) and tide channel. Photograph by T. Bowron, 24 September 2009.

Figure 6 Tidal channel following completion of restoration work (looking from mouth of channel towards interior of the restoration site). Photograph by J. Graham, 23 September 2009.

Figure 7 Tidal channel four years after restoration (mouth of channel looking towards interior of restoration site). Photograph by T. Bowron, 16 July 2013.

2.2 Cogmagun Reference Site

The reference site (COG-R) for the monitoring program is located on the north side of the Cogmagun River approximately 1.5 km upstream of the restoration site adjacent to where Route 215 crosses the river (Figure 2). The reference marsh is similar in spatial extent (6.1 ha) as the restoration site and displays evidence of an ecological and social history paralleling that of COG. Despite the remnant agriculture ditches, which are also present at COG, COG-R exhibits the typical salt marsh habitat zonation pattern, a well-developed tidal creek, a minimal panne system, and host of plant species common to salt marshes in the Bay of Fundy.

2.3 Monitoring Program

The COG Salt Marsh Restoration Project monitoring program was developed based on experience with similar restoration projects in the region (Bowron et al. 2011a; Neckles et al. 2002; van Proosdij et al. 2010). The monitoring program was used to document the changing habitat conditions following restoration; to evaluate the impacts of restored tidal flow; to indicate whether additional intervention was required; and to determine the ecological benefits of restoration (project success).

The monitoring program utilized a suite of tidal wetland indicators and data collection methods tailored to this project, which sought to characterize a broader range of tidal wetland ecosystem components. These indicators (geospatial attributes, hydrology, soils and sediments, vegetation, fish and invertebrates) are measures of wetland structure and function, and when applied pre- and post-restoration, collectively provide information on ecosystem status and response to restoration. The physical and biological parameters within each of these indicator categories for which data was collected during the 2013 monitoring season are identified in Table 1.

Table 1 The COG Salt Marsh Restoration monitoring program, including core and additional ecological indicators, methodologies, and sampling frequency (annual application indicated by X – all sites; S – COG; R – COG-R; Y – scheduled future sampling).

Category	Parameters	Sampling Method	Annual Sampling Frequency	Monitoring Year					
				Pre (2009)	Post-Restoration (2010-2014)				
					1	2	3	4	5
Hydrology	Tidal signal	Automated water level recorders (5 minute intervals) (Solinst Levelogger (Model 3001)	Minimum 4 week period once per sampling season. 19/8/10 to 4/10/10; 9/11/12 to 5/12/12	S	S		S		Y
	Suspended Sediment Concentration	Teledyne ISCO 6712 Full Size Portable Sampler (SSC)	Only conducted as part of baseline.	S					
	Water Quality	YSI 650 MDS and YSI 600QS sonde Matched with nekton sampling events	30/8/11; 27/10/11 19/9/12; 11/11/12 20/9/13; 7/10/13			S	S	S	Y
Soils & Sediments	Marsh surface elevation	Digital elevation model (DEM). Total Station; Differential GPS; LiDAR	Once per sampling year. COG: 4/10/10; 29/11/10; 9/11/12 COG-R: 16/11/10; 11/11/12	X	X		X		Y
	Interstitial pore water salinity	Sipper & refractometer (2009-2010) FieldScout EC 110 Meter (2011-2012)	Monthly; COG(-R) 28/7/10; 19/8/10; 16/9/10 21/6/11; 21/7/11; 18/8/11; 19/9/11 13/6/12; 11/7/12; 9/8/12; 12/9/12 15&17/7/13; 13&14/8/13; 22&23/8/13; 11/9/13	X	X	X	X	X	Y
	Soil Chemistry (Redox Potential)	Thermo Scientific Orion Star A221 milivolt meter with platinum electrodes & accument calomel reference electrode	23/8/13; 27/9/13					X	Y
	Sediment elevation	Rod Surface Elevation Tables (RSET)	COG: 3 RSET; 23/9/10; 12/10/11; 30/8/12; 22/11/13 COG-R: 3 RSETs; 23/9/10; 12/10/11; 30/8/12; 22/11/13	X	X	X	X	X	Y
	Sediment accretion	Marker horizons (3 per RSET) sampled using a cryogenic corer (Cahoon et	COG(-R): 23/9/10; 12/10/11; 30/8/12; 22/11/13	X	X	X	X	X	Y

Category	Parameters	Sampling Method	Annual Sampling Frequency	Monitoring Year					
				Pre (2009)	Post-Restoration (2010-2014)				
					1	2	3	4	5
		al. 1996).							
	Sediment Characteristics (bulk density, organic matter content, sediment type)	Sediment cores (soil samples) Paired samples: (30 ml syringe with base cut and 5 cm x 15 cm core).	COG-R: 8 samples; 1/9/10; 8/8/12 COG: 8 samples; 1/9/10; 8/8/12	X	X		X		Y
Vegetation	Composition Abundance Height	Point Intercept method (1 m^2 plots)	COG-R: 23 plots; 19/8/10; 12/8/11; 8/8/12; 13&22/8/13 COG: 24 plots; 20/8/10; 11/8/11; 8/8/12; 22/8/13	X	X	X	X	X	Y
	Habitat map	Aerial photograph, DGPS/GIS, Total Station, LiDAR, low-altitude aerial photography	COG: Aerial Photograph: 14/10/10; 21/6/11; 13/9/11; 27/10/12	X	S		S		Y
Nekton	Composition Species richness Density Length	Minnow traps in pannes, tidal creeks and main channel (small fish); beach seine (30 m x 1 m; 6 mm mesh) and fyke net (30 m x 1 m; 6 mm mesh) on marsh surface (all sizes)	9/8/09 30/8/11; 27/10/11 19/9/12; 11/11/12 20/9/13; 7/10/13	S		S	S	S	Y
Benthic and Other Aquatic Invertebrates	Abundance and species richness of benthic invertebrates	6" x 6" Ekman Dredge (0.023 m^2 sediment sample): 4 samples per site from borrow pit, tide channels & pannes	COG(-R): 23/9/10; 19/9/11 23/8/12 11/9/13		X	X	X	X	Y
	Abundance and species richness of aquatic invertebrates	Invertebrate Activity Traps (IAT)	23/7/10; 20/8/10 21/7/11; 18/8/11 26/7/12; 16/8/12 16/7/13; 22/8/13		S	S	S	S	Y
Winter Conditions	Ice/snow conditions	Structured winter walk; photographs along each transect (Appendix B)	15/3/11; 24/2/12; 5/3/13; 21/2/14	X	X	X	X	X	Y

3.0 Methods

Sampling was conducted at both the restoration and reference site using a series of transects (Lines) established as part of the baseline monitoring activities (2009) in a non-biased, systematic sampling design. Five transects were established at COG, 50 m apart (as measured along the dyke), running perpendicular to the main river channel and marked along the upland edge and the dyke with semi-permanent wooden stakes (Figure 8). Five transects were established at COG-R in the same manner (distance measured and transects marked along upland) (Figure 9). A combination of 100 m field tape, compass, Leica TCR-705 Total Station[4] and Trimble G8 GNSS RTK[5] surveying unit (RTK) were employed to produce and digitally map straight, reproducible transects. Data collection was conducted at sampling stations established at equal intervals along transects at each site. The RTK was used to locate the sampling stations at the beginning of each sampling season.

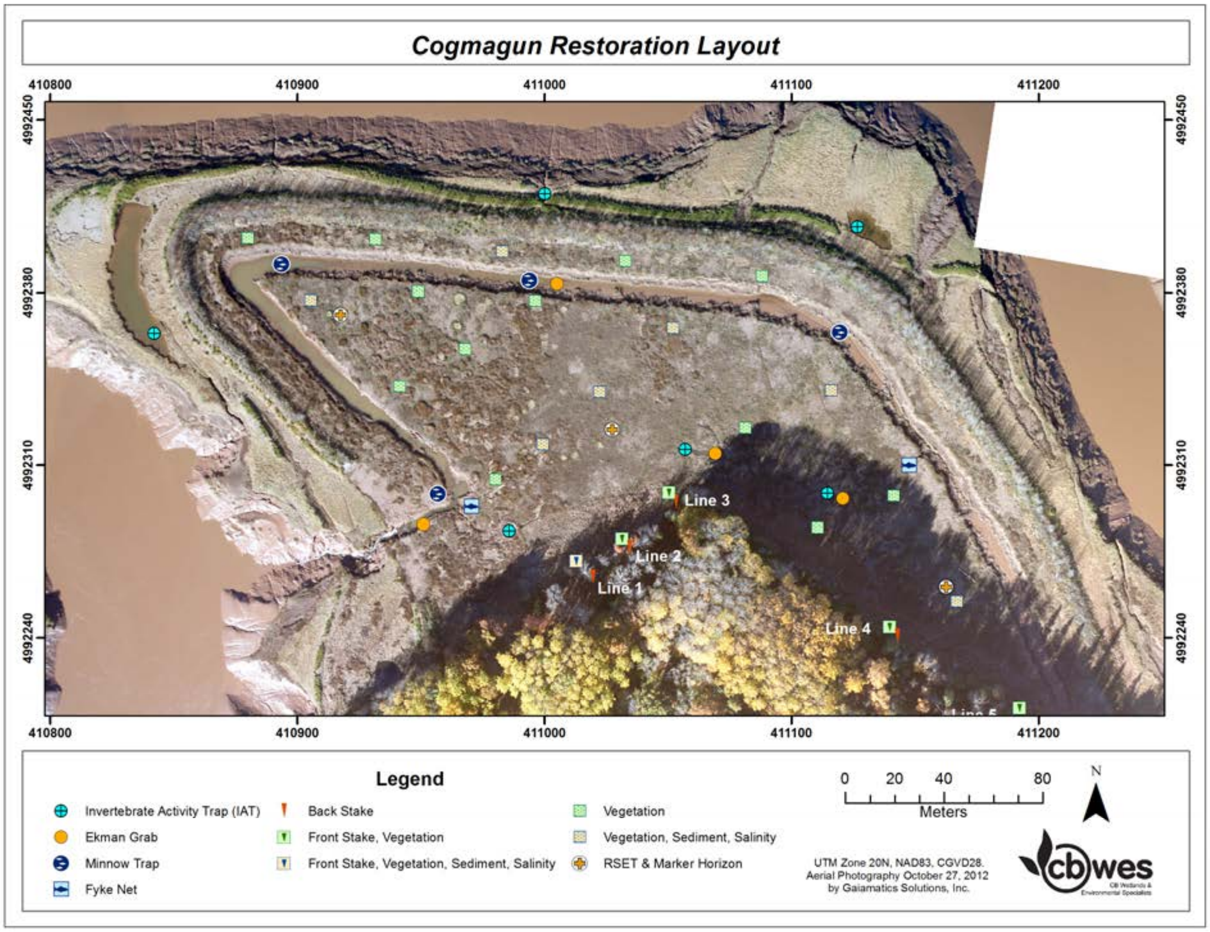


Figure 8 Sampling layout map for COG depicting main marsh features, transects, and sampling locations.

---

[4] www.leica-geosystems.com/corporate/en/lgs_405.htm

[5] www.trimble.com/index.aspx

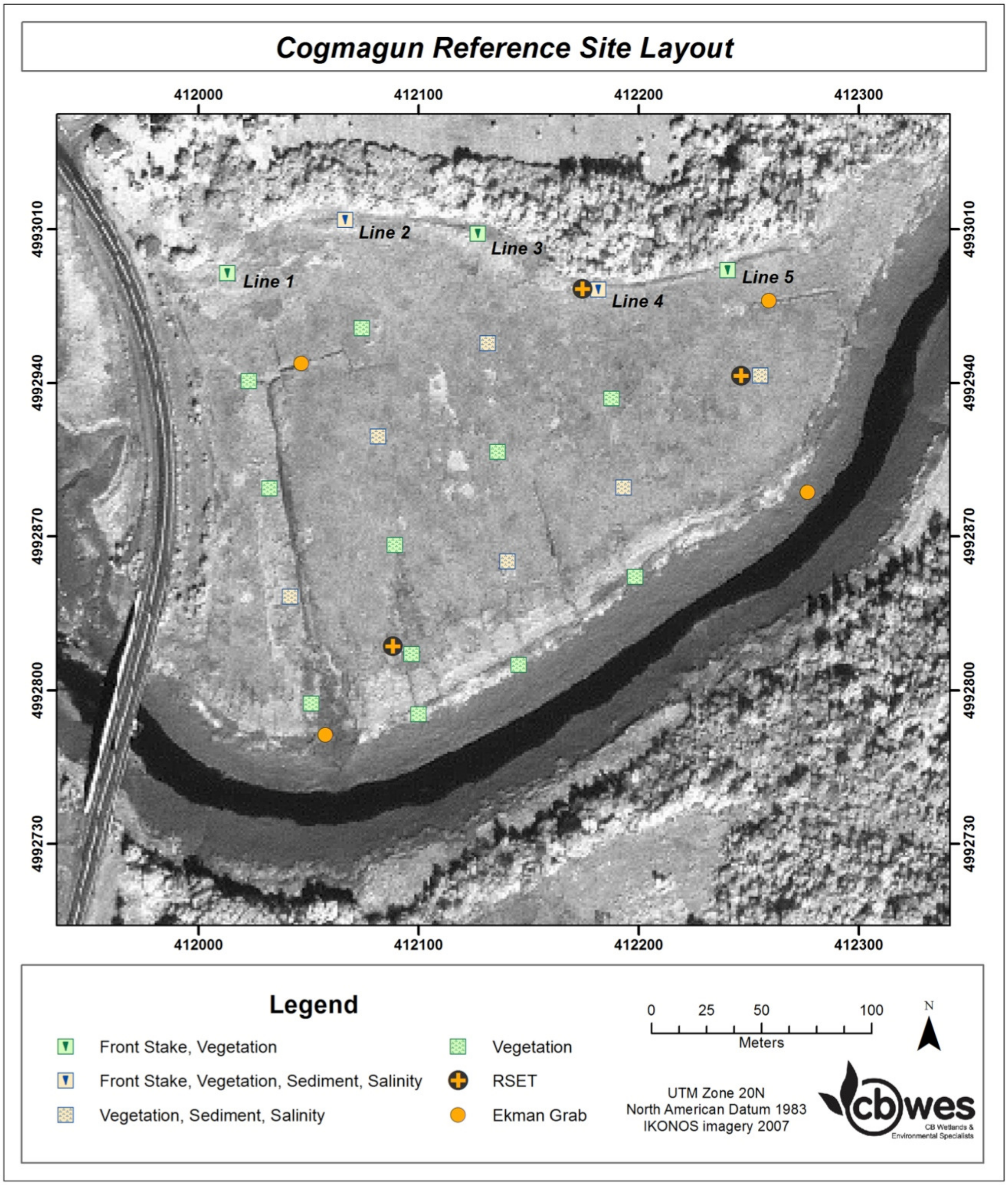


Figure 9 Sampling layout map for COG-R showing the location of key marsh features, transects and sampling stations.

3.1 Habitat Map and Digital Elevation Model (DEM)

The habitat map and DEM for COG and COG-R continue to be developed as conditions at the respective sites change and additional mapping activities are undertaken. Baseline and post-restoration (Year 1) DEM and habitat maps were produced using contour data extracted from

LiDAR[6] collected in 2007, and on-site survey data collected in 2009, and 2010 using a combination of GPS RTK and Total Station, and are presented in Bowron et al. (2010; 2011; 2012b). It was based on the 2009 DEM that the original estimate of restorable area was determined. The habitat map provided a foundation for monitoring activities and a visual record of changes in habitat conditions following restoration.

Habitat maps (surface cover) were also generated to document vegetation community structure and other important habitat features identified from IKONOS imagery, as well as data collected at sampling stations recorded by the Total Station/GPS RTK. For each site the DEM was extracted from an existing LiDAR DEM of the Avon Estuary once the study area had been identified from the IKONOS imagery.

The DEMs, habitat and surface cover maps for both sites were updated based on the 2012 conditions (Bowron et al. 2013c). In addition to the on-site elevation surveys, low-altitude photogrammetry was conducted annually 2010 through 2012. The resulting orthorectified image mosaic was used as the base layer for the maps.

In 2012, additional elevation analysis was carried out to track the development of the primary channel at COG, which has experienced substantial erosion since the return of tidal flow. Elevation profiles were taken along established profiles at the mouth of the channel, where the greatest change had been experienced. Triangular Irregular Networks (TIN) surfaces were created for each year and profiles extracted. While the DEM’s described previously were better suited to capture change over the entire marsh surface due to their ability to maintain global continuity, TIN’s preserve the precision of input data and were more appropriate for working at the creek scale.

Elevation surveys (marsh surface, tidal channel) and low-altitude photogrammetry were not part of the 2013 (fourth year post-restoration) monitoring activities. These will be conducted in 2014 as part of the final year of post-restoration monitoring.

3.2 Hydrology

The fundamental control on the structure and function of salt marsh habitat is flooding with salt water (Mitsch and Gosselink 2007; Neckles and Dionne 2000). It is the hydroperiod (frequency and duration of tidal flooding) of a salt marsh that determines the area of marsh directly available as fish habitat. The hydroperiod of a salt marsh is determined by the tidal signal (pattern of water level change with respect to a reference point) and marsh surface elevation.

When attempting to understand changes in vegetation, water table level can be a valuable parameter to monitor as it provides information on the degree of waterlogging or drainage that is occurring on a marsh (Roman et al. 2001). Surface water quality (salinity, dissolved oxygen, pH and temperature) of flood waters can also influence the diversity, distribution and abundance of plants and animals in a salt marsh.

---

[6] IKONOS imagery provided by SMU. LiDAR data provided by the Applied Geomatics Research Group, Centre of Geographic Sciences.

Hydroperiod and Tidal Signal

The hydroperiod (frequency and duration of tidal flooding) for COG was modeled using the tidal signal data (pattern of water level change with reference to a fixed point) and DEM for the site. The tidal signal within the main river channel and within the restoration site was measured in 2012 using a set of Solinst Model 3001 Levelogger Golds[7] (water elevation and temperature) and a Solinst Barologger (atmospheric pressure and temperature) (Bowron et al. 2013c). The hydroperiod for COG will be updated as part of the fifth year (2014) of post-restoration monitoring.

Water Quality

A YSI 650 MDS sonde was used to measure four physical components of water: temperature (± 0.1 C°), dissolved oxygen (DO) (± 0.1 mg/L), salinity (± 0.1 ppt) and pH. Three water quality readings were taken per nekton sampling event within thirty minutes of peak tide (spring tide). Measurements were taken from the dyke near the breach (Figure 8).

3.3 Soils and Sediments

Monitoring pore water salinity, sediment accretion rates, sediment elevation and soil characteristics can provide insight into the processes controlling vegetation type, cover and productivity and the vertical growth of marsh following restoration (Neckles and Dionne 2000). Soil salinity (interstitial pore water salinity) is one of the main controls on the distribution and abundance of plant species in salt marshes (Niering and Warren 1980). Measuring pore water salinity throughout the early to mid growing season and in conjunction with water table depth monitoring can help explain changes in environmental conditions regulating plant growth, distribution and abundance, and habitat responses to restoration activities.

Accretion of inorganic and organic material deposited onto the marsh surface by floodwaters and vegetation is one of the main processes that allow marshes to build vertically over time, offsetting increased tidal flooding. Failure to keep pace with increased flooding could result in the loss of salt marsh features and functions important to fish (loss of productivity and extent of habitat). Monitoring sediment accretion rates, elevation and determining organic content of marsh soils prior to engaging in restoration activities can reveal insights regarding pre-restoration conditions of the marsh (subsidence due to oxidation of organic matter in sediments) and the process of recovery following restoration.

Marsh soil characteristics are determined by the sediment source and tidal current patterns (Mitsch and Gosselink 1986). As tidal waters flow over the marsh surface, increasing elevation and vegetation slows the water allowing coarse-grained sediment to drop out of suspension close to the main channel edge while finer sediments drop further inland (Redfield 1972; Mitsch and Gosselink 1986). Sediment type and particle size greatly influences soil aeration and drainage (Packham and Willis 1997). Silt, clay and sand are the different soil textures typical of salt marshes. Silt and clay materials tend to retain more salt than sand, and clay is the most absorptive (Mitsch and Gosselink 1986). Clay and silt are expected to dominate high marsh soils, while the low marsh is expected to have a higher proportion of sand (Packham and Willis 1997); however, this will vary depending on the source material. Soil characteristics were sampled in 2009 as part of the baseline study, and again one (2010) and three (2012) years after restoration

---

[7] www.solinst.com/Prod/3001/3001.html

(Bowron et al. 2011c; Bowron et al. 2013c). Soil sampling will also be conducted during the fifth year of post-restoration monitoring (2014).

Restoration of tidal water to a previously tide restricted site has been shown to drastically alter the biogeochemistry of marsh sediments, ultimately affecting vegetation re-colonization (Anisfeld 2012). The site conditions prior to restoration will govern the soil chemistry and long-term success of the restoration project. Reddy and DeLaune (2008) define biogeochemistry as "the study of the exchange or flux of materials between living and nonliving components of the biosphere". The processes that occur within wetlands at the surface or near-surface layers of sediments govern the biogeochemical cycles, productivity of plants, microbial transformations, nutrient availability, pollutant removal, exchange between atmosphere, water and sediment, and sediment transport (Reddy and DeLaune 2008).

Oxidation and reduction reactions represent a transfer of electrons either through donating or accepting an electron respectively. For microbial communities the most preferred electron acceptor is oxygen (Craft 2001; Reddy and DeLaune 2008; Portnoy 1999); however, oxygen found within the soil is rapidly consumed leading to a high electron pressure or reduced state (Colmer and Flowers 2008; Koch and Mendelsshn 1989). Alternative electron acceptors include (in order of decreased energy provided): nitrate (NO_3^-), manganese (IV) oxides (MnO_2), iron (III) oxides ($Fe(OH)_3$), sulfate (SO_4^{2-}), and carbon dioxide (CO_2) (Craft 2001; Reddy and DeLaune 2008). The reduction of these alternative electron acceptors not only reduces the amount of energy accessible to the microbial community, but many produce phytotoxins (e.g. hydrogen sulfide) that are detrimental to vegetation growth (Koch and Mendelsshn 1989).

Redox potential can be used as an indicator for the intensity of anaerobic conditions within the sediments (de la Cruz et al. 1989; Reddy and DeLaune 2008) and represents the dominant oxidation reduction reaction occurring at the time of the measurement (Reddy and DeLaune, 2008; Table 2) Measuring redox potential of soils at representative locations throughout a restoration site reflect the interaction between hydrology, microbial activity, rhizome activity, sediment characteristics and amount of available organic matter and nutrients (Catallo 1999; Reddy and DeLaune 2008).

Table 2 Electron acceptors used by microbial communities and associated range of redox potential (Reddy and DeLaune 2008).

Electron Acceptor	Reduced To	Redox Potential (mV)
Oxygen (O_2)	H_2O	> +300
Nitrate (NO_3^-)	N_2, NH_4^+	+300 to +100
Manganese (Mn^{4+})	Mn^{2+}	+300 to +100
Iron (Fe^{3+})	Fe^{2+}	+100 to -100
Sulfate (SO_4^{2-})	S^{2-}	-100 to -200
Carbon dioxide (CO_2)	CH_4	-200 to -300

Pore Water Salinity

Sampling locations for interstitial pore water salinity were matched with a subset of vegetation sampling stations at both sites (Figure 8 and Figure 9). At each of the eight sampling locations at COG and COG-R, both a shallow (15 cm) and a deep (45 cm) pore water sample were taken during the 2009 and 2010 monitoring years using a soil probe (sipper; Figure 10). The sipper was constructed according to the design specifications outlined in Roman et al. (2001). Samples were taken by sequentially inserting the sipper into the soil to the target depth and drawing out a water sample. Due to the high sediment concentration of water samples collected by the sipper, samples were bottled and returned to the lab and allowed time for the sediment to settle out before a refractometer (nearest 2 ppt) reading was taken. In order to ensure consistent and accurate readings, the sipper sampling method was replaced by a FieldScout EC 110 Meter[8] in 2011. During the 2011 field season, data was collected using both methods for two consecutive sampling events ("double read") in order to ensure consistency and minimize sampling error. Only the FieldScout was used during the 2013 field season. When using the FieldScout, measurements were taken *in situ*, and readings recorded in the field.

Pore water salinity was sampled at COG on 15 July, 13 August, 22 August, and 11 September 2013; and at COG-R on 17 July, 14 August, 23 August, and 11 September 2013. A "between sites" (COG vs. COG-R) comparison of salinity levels was conducted using t-Test: Two-Sample Assuming Unequal Variances. Due to the change in sampling technique in 2011, a "between years" comparison of salinity levels was conducted using the pre (2009) and post (2010) sipper and refractometer data, and between 2011 (year two), 2012 (year three) and 2013 (year four) EC Meter data.

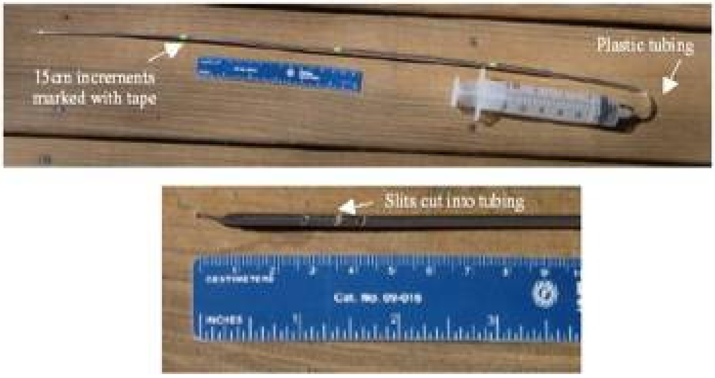


Figure 10 Photograph of a soil probe (sipper) used to take interstitial pore water samples to measure pore water salinity (Roman et al. 2001).

Sediment Accretion and Elevation

Three Rod Surface Elevation Tables (RSET; Cahoon et al. 2002) were installed at both COG and COG-R on 19 August and 11 September 2009 respectively. Geographical coordinates of both sites are presented in Table 3. COG RSET-01 was located near the main drainage channel in an un-vegetated area (Figure 8; Figure 24), whereas both RSET-02 and RSET-03 were located further away from the breach location within the cattail zone (Figure 8; Figure 25; Figure 26). RSETs at the reference site were installed in the low marsh (RSET-1), high marsh/upland boundary (RSET-2) and mid marsh (RSET-3) (Figure 9; Figure 28).

---

[8] http://www.specmeters.com/brands/field-scout/ec110/

Table 3 Coordinates (NAD 83 UTM Zone 20) and elevation of RSET stations installed at COG and COG-R, based on 2012 RTK survey.

RSET code	Location	Easting	Northing	Elevation (m CGVD28)
COG_RSET_01	Low marsh	410 917.33	4 992 371.03	6.678
COG_RSET_02	High marsh	411 027.47	4 992 324.53	6.839
COG_RSET_03	Mid marsh	411 162.57	4 992 260.77	7.087
COG_R_RSET_01	Low marsh	412 088.13	4 992 820.15	7.181
COG_R_RSET_02	High marsh	412 174.35	4 992 982.76	7.197
COG_R_RSET_03	Mid marsh	412 246.48	4 992 943.24	7.087

The 2009 RSET measurements provided the baseline elevation against which annual changes over the next five years would be compared. To determine the change in surface elevation between sampling years (e.g., 2013 and 2012), the difference in elevation at each pin was first calculated by subtracting the value in 2013 from the value in 2012 (Figure 11; Figure 12). It was important that the same point was measured (e.g., same measurement direction and pin position). If the value was negative, the surface has lowered, and if it was positive, the elevation of the surface has increased. A mean was derived from all 36 net change values to give a mean net change in surface elevation in cm per year or in this example, from 2012 to 2013.

Vertical accretion at all sites was measured using feldspar marker horizons and a cryogenic corer as described by Cahoon et al. (1996) (Figure 13; Figure 14). Three 0.5 m^2 marker horizons per RSET station were established at each site. Vertical accretion at each marker was measured annually in conjunction with RSET sampling.

Changes in surface elevation measured by the RSET incorporate both subsurface processes, such as root production, and sediment deposition, whereas sediment accretion measured by the marker horizon cores represents the amount of inorganic and organic material deposited by tidal waters on the marsh surface. Subtraction of the RSET and marker horizon values should provide a measure of the amount of change in surface elevation due to shallow subsidence processes such as root growth, compaction, decomposition and pore water flux (Cahoon et al. 2002). Both surface (e.g., accretion) and subsurface processes will be highly influenced by the elevation of the marsh surface within the tidal frame which affects the frequency and duration of inundation by tidal waters. Sediment accretion will also be affected by other factors such as the proximity to a sediment source, for example, the tidal creek network (van Proosdij et al. 2006).

The RSETs and marker horizons were installed at COG and COG-R in 2009 in advance of restoration. Measurements were taken annually at approximately the same of time year and at low tide to minimize the influence of evapotranspiration (Cahoon et al. 2002; Paquette et al. 2004). The fourth year post-restoration RSET and marker horizon measurements were taken on 22 November 2013. The calculations from the RSETs and marker horizons, as well as the results and discussion were prepared by Dr. Danika van Proosdij (SMU).

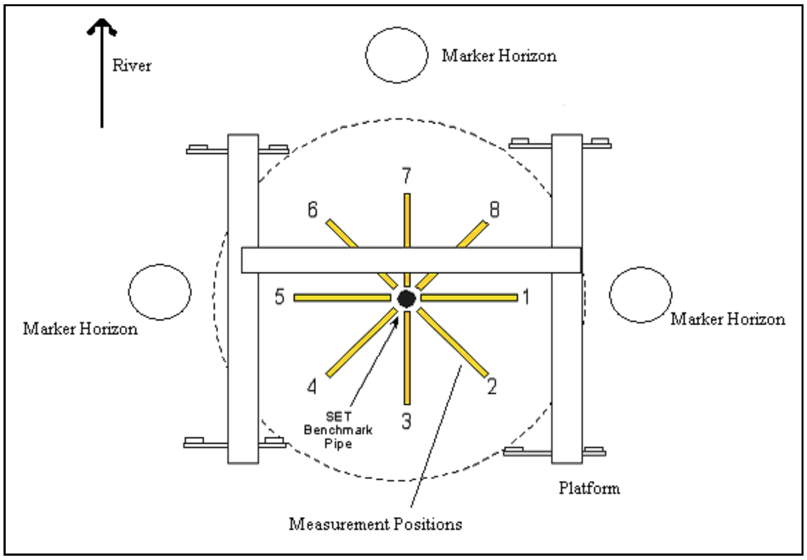


Figure 11 RSET and feldspar marker horizon layout (modified from USGS 2005).

Figure 12 The nine measure pins of the RSET. Photograph by CBWES Inc.

Figure 13 Marker horizon sampling with the cryogenic corer (with stainless steel tubing and copper "bullet") at Lawrencetown Lake Reference Site. Photograph taken by B. Lemieux, December 2011.

Figure 14 Core or "marsh-cicle" on bullet used for marker horizon calculations. The white band is the feldspar "horizon" and the top of core (marsh surface) is to the right. Photograph by CBWES Inc.

Soil Characteristics

Field Methods

Soil samples were not collected in 2013, but are scheduled as part of the year five (2014) post-restoration monitoring activities (Table 1).

Soil Chemistry (Redox Potential)

Redox potential was measured at both sites on 23 August 2014 and 27 September 2014 using a series of platinum electrodes, an accument calomel reference electrode and Thermo Scientific Orion Star A221 Milivolt Meter[9] (Figure 15). Platinum electrodes (probes) were constructed based on the design by Vepraskas and Cox (2002). Probes were calibrated in a mixture of quinhydrone and pH 4.00 buffer before use in the field to ensure accurate readings. Sampling locations (6 per site) were matched with sediment sample locations (Figure 8; Figure 9). A shallow and a deep measurement were taken at each sampling location using two probes inserted into the marsh sediment 2 cm and 20 cm respectively. Probes were deployed 30 minutes before readings were taken to allow the probes to equilibrate. The reference electrode was inserted into the soil close to the probes at the time of measurement. Measurements from the two probes were taken by individually connecting the probe to the Meter and waiting for the Meter to stabilize (max five minutes). Redox potential was determined by adding +244 mV to each field measurement to account for the potential of the reference electrode.

Figure 15 Constructed platinum electrodes (top), accumet calomel reference electrode (bottom) and Thermo Scientific Orion Star A221 Milivolt Meter (Photo: C. Skinner 2013).

3.4 Vegetation

The primary food source in estuaries originates in the vegetation of salt marshes. The majority of this plant material is consumed indirectly as detritus (dead plant material) by decomposers and invertebrate consumers. It is through the production and export of detritus that salt marshes help to sustain commercial and non-commercial fish species by forming the base of coastal food webs. Salt marshes are characterized by their plant communities, with specific plants dominating the different salt marsh zones (low, mid and high). It is the plants of a salt marsh, along with the physical conditions (hydrology, geology and chemical) that create the template for a self-sustaining coastal wetland system and which enable the biological components of the broader ecosystem (invertebrates, fish, birds and animals) to benefit from these habitats.

---

[9] www.coleparmer.com/buy/product/96392-thermo-scientific-orion-star-a221-ph-portable-meter-kit.html

Field Methods

The marsh vegetation community was surveyed at COG on 13 and 22 August 2013, and COG-R on 22 August 2013, using a modified point intercept method (Roman et al. 2002). The point intercept method utilizes permanent 1 m^2 plots positioned at intervals along each transect. Twenty-four plots were established at COG and twenty-three plots at COG-R. Landscape photographs were taken along each transect, as well as close-up photographs of each plot.

Each 1 m^2 plot (quadrat) used was offset 1 m to the left of the transect (facing main tidal channel) and oriented towards the upland end of the transect. The quadrat was divided into a grid of 25 squares (20 cm x 20 cm) and the resulting twenty-five intercept points were used as sampling points. All plant species present in the quadrat were recorded and then a wooden dowel (3 mm in diameter) was held vertical to the first sampling point and lowered through the vegetation to the ground below. Any species that touched the rod (a "hit") were recorded and this was repeated for all twenty-five intercept points. Other categories, such as water, bare ground, rock or debris, were also recorded if hit by the dowel.

Statistical Analysis

Plant species composition was compared between the restoration and reference sites using non-metric multidimensional scaling ordination. Plant species richness, halophytic species and abundance, unvegetated area, and the amount of dead plant material in 1 m^2 plots were compared between the COG and COG-R sites using repeated measures ANOVA. Halophytic species abundance was estimated as the total number of contact points by halophytic species per plot. Because the total number of hits was counted, this can result in a halophytic abundance of greater than 25 (the number of points sampled in each quadrat) when more than one halophytic species were present in the plot.

The following species at these sites were considered halophytes for the purpose of these analyses: *Atriplex glabrisculata, Carex paleacea, Distichlis spicata, Juncus gerardii, Limonium nashii, Potentilla anserina, Puccinelia maritima, Ranunculus cymbalaria, Ruppia maritima, Salicornia europea, Scirpus maritima, Spartina alterniflora, S. patens, S. pectinata, Sueda maritima, Triglochin maritima.* These species are bolded in Table 9 (Section 4.4).

3.5 Nekton

Salt marshes support a wide range and abundance of organisms that swim, collectively referred to as nekton, which include fish and many types of invertebrates. Fish and macrocrustaceans are an important ecological link between the primary producers of the marsh (plants) and near shore fisheries (Neckles and Dionne 2000). Their position in the upper levels of the coastal food webs and their dependence on a wide range of food and habitat resources serve to integrate ecosystem elements, processes and productivity (Kwak and Zedler 1997).

Fish are a challenging group to quantify due to their mobility and temporal variability, as well as the difficulties of sampling in, what can be, a heavily vegetated environment with a varied hydrological regime. Two species commonly found in salt marsh habitats are the mummichog (*Fundulus heteroclitus*) and Atlantic silverside (*Menidia menidia*). The mummichog is a resident species, while the Atlantic silverside moves with the rise and fall of the tide in search for food, and both are prey for larger fish within the tidal rivers and salt marshes during high tide (Gibson 2003). Similar to mummichogs, the *S. alterniflora* dominated low marsh areas of salt marshes is one of the substrates Atlantic silversides use for reproduction (egg attachment) (Fay et al. 1983). Atlantic silversides may also be important exporters of secondary production and biomass from marsh and estuarine systems to offshore areas as they usually die after spawning or during their second winter of life (Fay et al. 1983).

Smaller species accessing the salt pannes and borrow pit were sampled using the minnow traps baited with bread. Traps were set in two pannes located on the fringe marsh, a developing panne within the restoration site and in the borrow pit (Figure 8). Traps were deployed in advance of high tide and retrieved once the tide level had dropped (approximately 3.5 hours).

The fyke net design and methodology was adapted from Dionne et al. (1999). The fyke net was set at low tide and retrieved following high tide once the water level dropped low enough to approach the net while ensuring that the cod end remained submerged. The fyke net, which samples for all species and size ranges of fish utilizing the marsh surface, requires a spring tide to ensure adequate depth (depth of water on marsh surface > 1 m) and duration (approximately 3 hours) of tidal water on the marsh surface. The fyke net was deployed at a different location on the marsh surface each sampling event (Figure 8).

Fish surveys at COG were carried out on 20 September and 7 October 2013. All captured specimens were held in buckets, identified to species using identification guides (Audubon Society 1993; Graff and Middleton 2002; Scott and Scott 1988), counted (to a maximum of 300 per species), and measured for length (15 individuals per species). Photographs, and if necessary a single representative, of unknown species were taken for identification purposes, while all remaining individuals were returned (alive) to the site of capture.

3.6 Benthic and Other Aquatic Invertebrates

Benthic invertebrates, in association with benthic microbial communities, are largely responsible for providing the food resources that help fuel coastal and offshore marine ecosystems. In addition to directly being fish food, these organisms perform the important task of converting the rich productivity of salt marsh vegetation into a form (detritus) that is more palatable to other species such as fish. Benthic marine invertebrates and various aquatic freshwater and saltwater invertebrates such as insect larvae are well-known indicators of changes in hydrology, chemical characteristics and productivity (see the Canadian Aquatic Biomonitoring Network (CABIN) program website for more information on the use of aquatic invertebrates to monitor the health of aquatic ecosystems - www.ec.gc.ca/rcba-cabin/) (Armanini et al. 2012).

Benthic Invertebrates

Field Methods

Benthic invertebrate samples were taken on 11 September 2013 using a standard 6” x 6” Ekman Dredge (15 cm x 15 cm; 0.023 m^2 sediment sample) (Figure 16). Samples were analyzed for biological species composition and abundance. Four samples from the main channel, pannes, creeks and ditches were taken at each site (Figure 8; Figure 9). Each sample was individually bagged, labeled and placed in a cooler containing ice for transport to the laboratory facilities at Envirosphere Consultants Ltd. in Windsor, NS where the samples were sorted and analyzed for species composition and abundance.

Figure 16 Ekman Dredge.

Aquatic Invertebrates

Aquatic invertebrates within the water column of select pannes within and outside (fringe marsh) COG were sampled using Aquatic Invertebrate Activity Traps (IAT) (Figure 17). IAT were submerged and anchored within the water column of the panne being sampled and allowed to passively sample over a single tide cycle (approximately 24 hour period). Four samples were taken on 16 July and 22 August 2013. Traps were set on the dates indicated and then retrieved the following day. No samples were taken at COG-R due to a lack of suitable habitat conditions (no pannes or standing water). Samples were emptied into a 0.5 mm sieve and all captured materials and organisms field-preserved in 70% isopropyl alcohol for transport to the lab for processing.

Laboratory Methods

Preserved Ekman Dredge samples were typically sub-sampled by 25% or 50% before further processing, due to the large bulk of the samples and to maintain a reasonable processing time. IAT traps were not sub-sampled. For sub-sampling, Ekman samples were distributed evenly between four identical rectangular trays to divide the sample into quarters. Each quarter was visually checked for homogeneity and then weighed and adjusted, if necessary, so that each quarter was within 0.5 to 1 g of each other to ensure even distribution of the sediment. The lab technicians then sorted one or two of these quarters depending on the processing time. Following analysis, the counts and biomass for the sub-samples were multiplied by a factor to restore the numbers to an estimate for the sample as a whole and per unit area. Since Ekman samples represent an area of 0.023 m^2, the number of individuals and biomass were multiplied by a factor of 43.5 to give estimates per unit area (per square metre). Sub-sampling affects measures of animal abundance and biomass by increasing variability, and leads to slightly reduced estimates of taxon richness compared to whole samples.

Prior to sorting, samples and sub-samples were rinsed on a 0.5 mm sieve to remove preservative. Samples were examined at 6 - 6.4x magnification on a stereomicroscope, with a final brief check at 16x. Organisms were subsequently stored in labeled vials in 70% isopropyl alcohol. Sorting efficiency for lab personnel is checked periodically by resorting samples, to ensure average recovery levels of about 90% or better. Wet weight biomass (grams per square metre for grabs) was estimated by weighing animals to the nearest milligram at the time of sorting, after blotting to remove surface water.

Organisms were identified to an appropriate taxonomic level, typically to species, using conventional literature for the groups involved (e.g. Barnes 1987; Gosner 1971). Organisms were identified by Patrick Stewart, Renee Raudonis and Heather Levy of Envirosphere Consultants. Species abundance, number of species, and wet weight biomass were estimated from the data. All taxa found, including meiofauna and planktonic taxa (e.g., copepods) as well as fish, were included in estimates of taxon richness and biomass.

Figure 17 Disassembled Invertebrate Activity Trap. Photograph by T. Bowron 2007.

3.7 Structured Winter Walk

On 21 February 2014, a structured winter site-walk was conducted at COG and COG-R. Landscape photographs were taken along each transect from the dyke at COG and the back stakes at COG-R. At COG, the structured walk included the perimeter of the site (dyke), with photographs being taken of key features such as the breach, tide channel, borrow pit, ice, areas of erosion or deposition and other features of interest. At COG-R, the structured walk was completed along the upland edge of the marsh, starting at the first transect, proceeding to the fifth and return via the mid-marsh. Again, landscape photographs were taken along each transect from a reproducible location (back stake) and of all features of interest.

4.0 Results and Discussion

4.1 Habitat Map and Digital Elevation Model (DEM)

Pre- and post-restoration DEMs for COG(-R) based on the third year of post-restoration data were presented in Bowron et al. (2013d). Total marsh area (including dyke and fringe marsh) based on the 2012 data was calculated to be 7.68 ha, and the area inside the dyke at 4.16 ha (Section 5.0; Figure 45). The DEM (Figure 18; Figure 19), surface cover change map (Figure 20), habitat map (Figure 21), and marsh area calculations for COG(-R) will be updated in 2014.

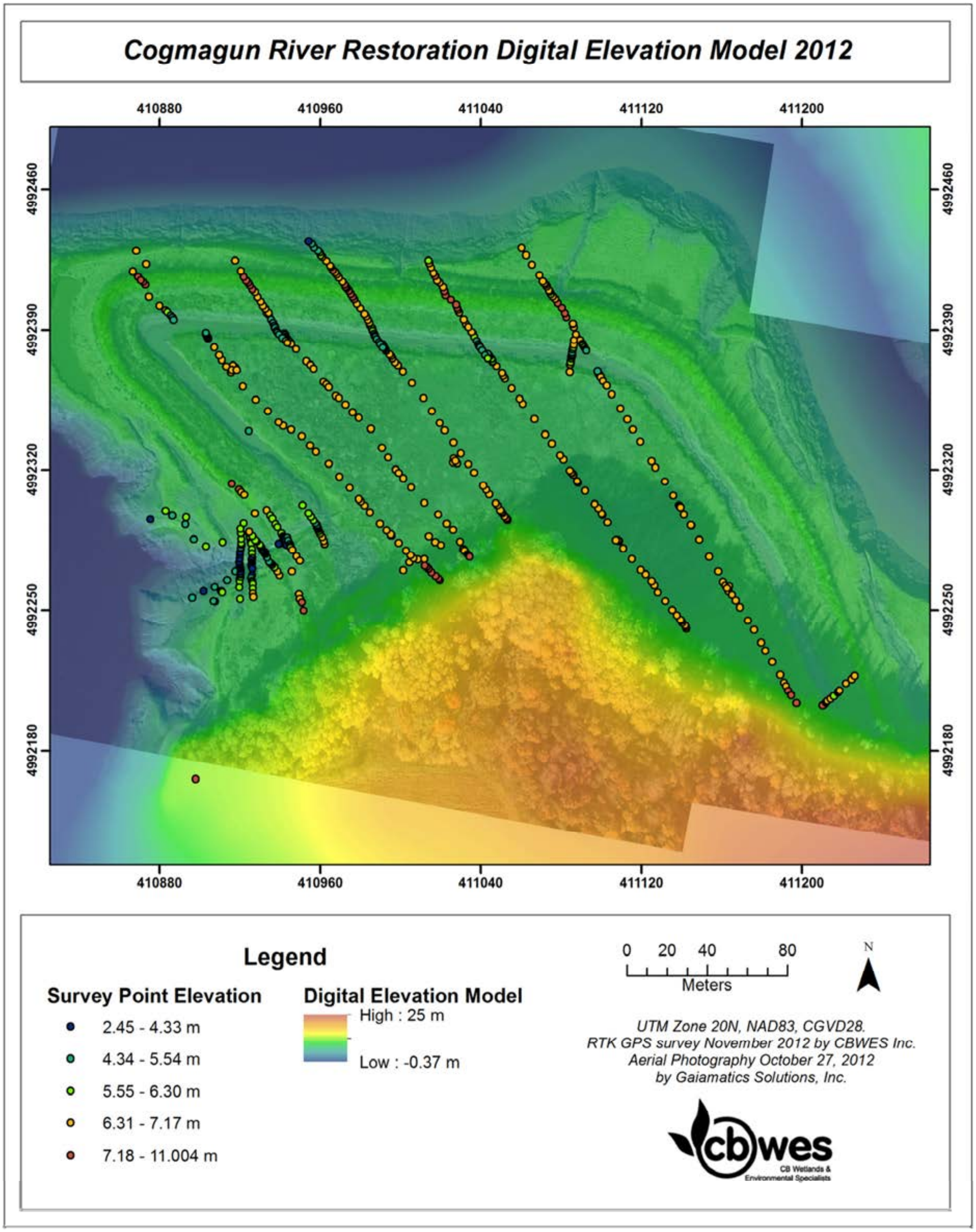


Figure 18 DEM for COG developed using the 2007 LiDAR data, 2012 elevation survey and the 2012 low-altitude photography.

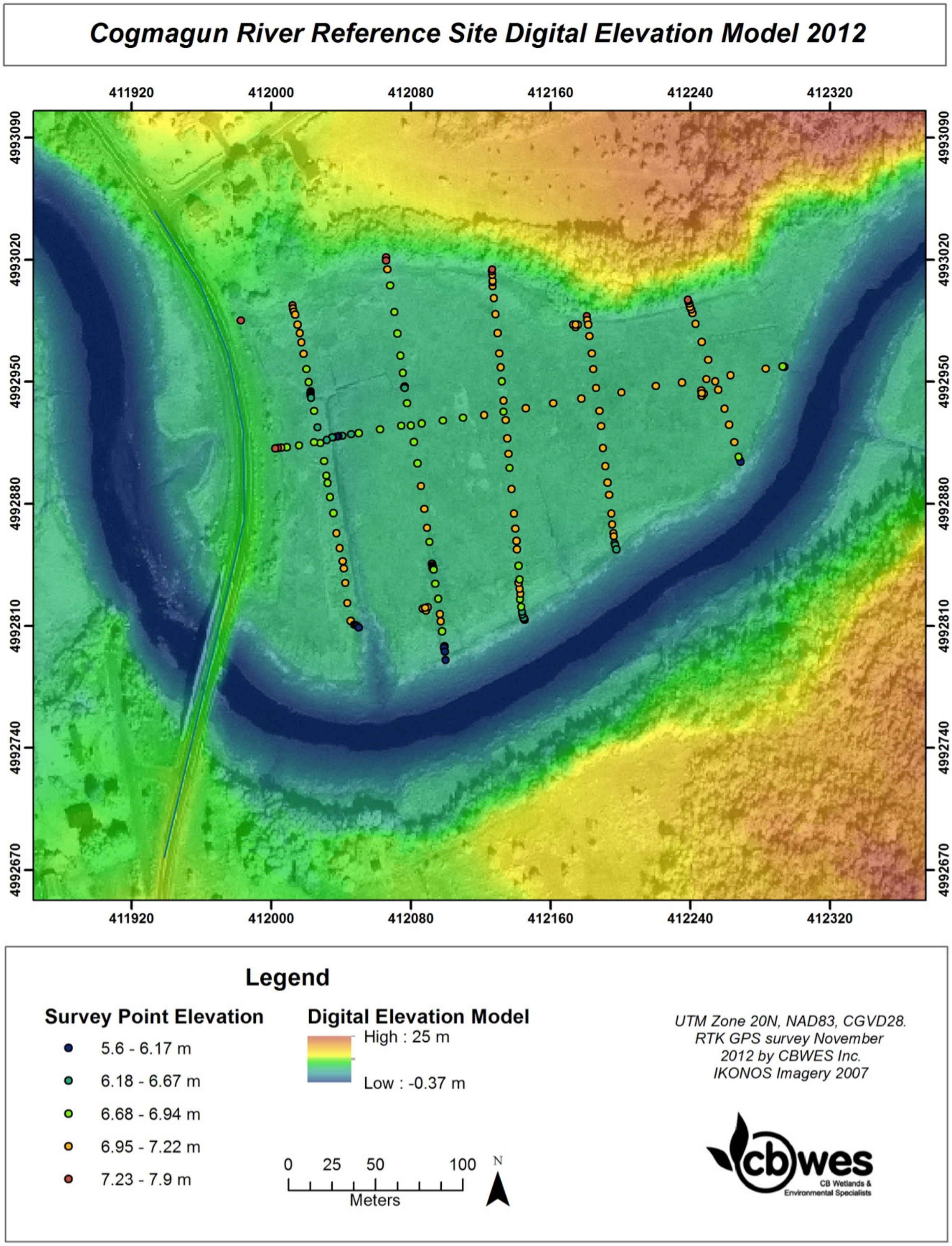


Figure 19 DEM for COG-R developed using the 2007 LiDAR data and 2012 elevation survey.

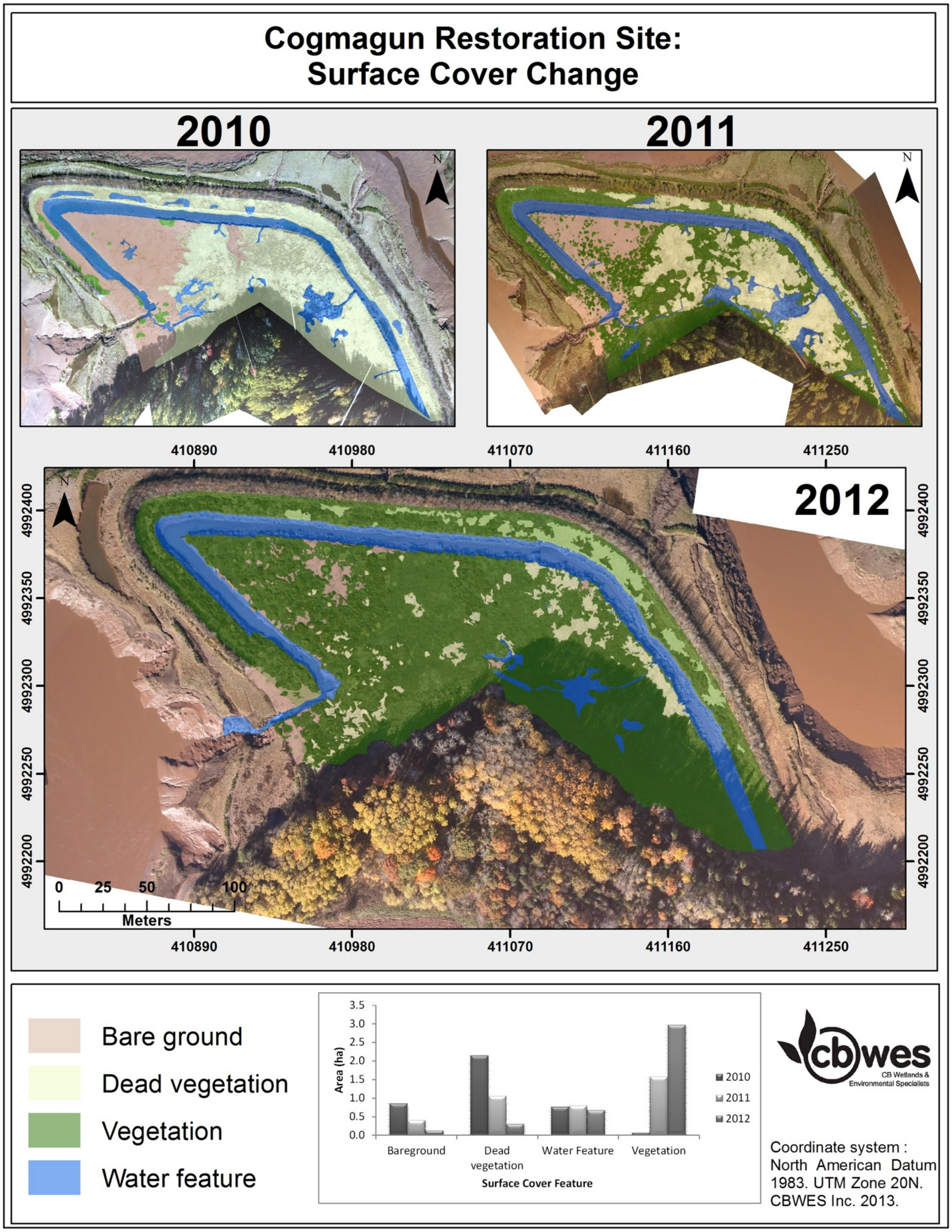
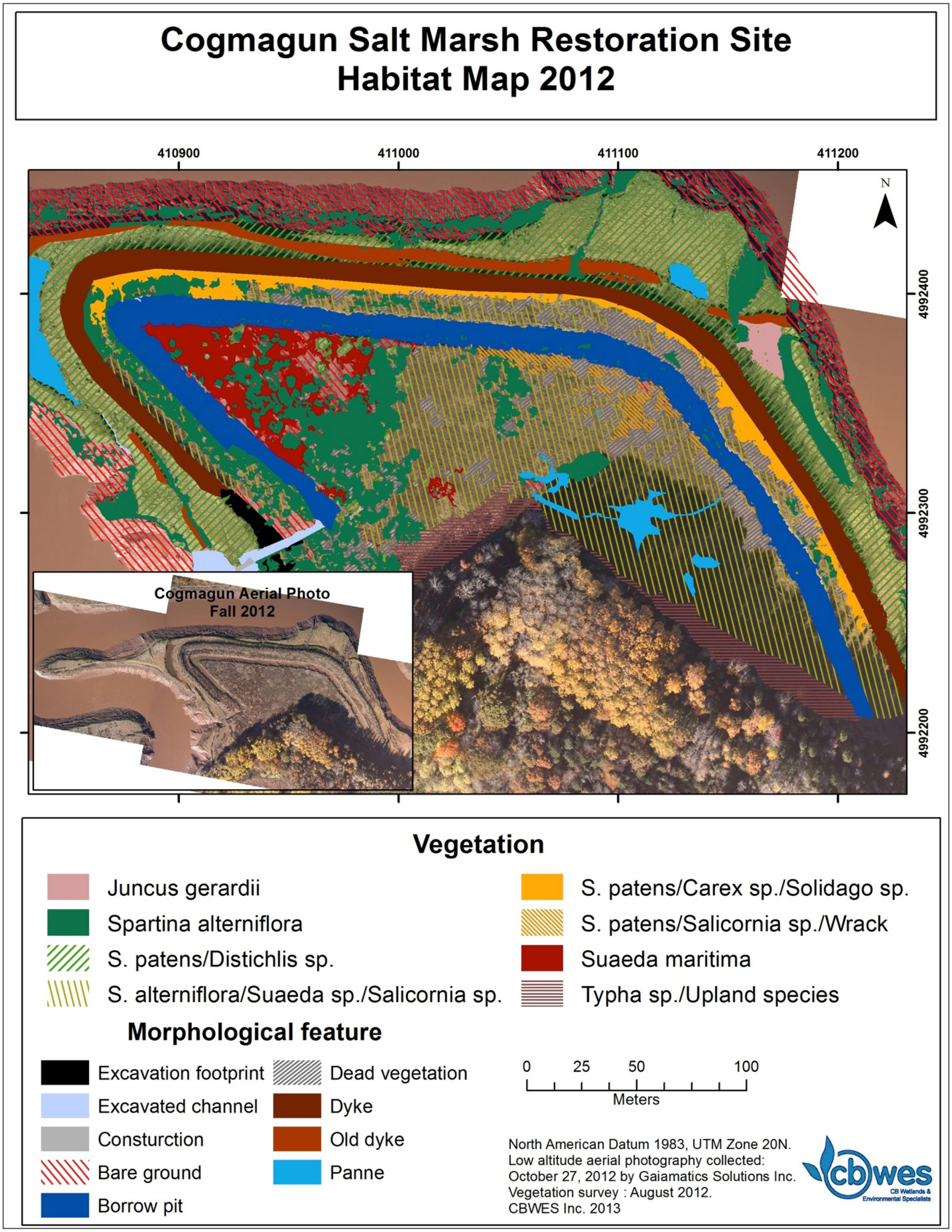


Figure 20 Comparison of the 2010 (post-restoration Year 1), 2011 (Year 2) and 2012 (Year 3) surface cover maps (vegetation community structure) for COG.



Cogmagun Salt Marsh Restoration Site
Habitat Map 2012

North American Datum 1983, UTM Zone 20N.
Low altitude aerial photography collected:
October 27, 2012 by Gaiamatics Solutions Inc.
Vegetation survey : August 2012.
CBWES Inc. 2013

Figure 21 2012 habitat map for COG showing dominant surface cover features and vegetation community assemblages.

4.2 Hydrology

Water Quality

Water quality sampling of the surface floodwaters was conducted concurrently with nekton sampling. Table 4 shows the water quality readings taken at COG for Years 2 through 4 post-restoration. Sampling was conducted at the location of the dyke breach. Despite the limitations of the data set, water quality parameters did not vary greatly between sampling events (months or years), displayed the expected trend of decreasing temperature into the fall months with a corresponding increase in DO. Salinity levels remained well within the polyhaline range of 18 to 30 ppt (Tiner 2005: Odum 1988), and the variation observed was likely the result of freshwater influence. These water quality readings are similar to those recorded for other restoration sites in the region (Bowron et al. 2011a; Bowron et al. 2013 a,b; Neatt et al. 2013. Van Proosdij et al. 2010).

Table 4: Water quality conditions for COG taken during fish survey events (post-restoration years two through four).

Sample Date	Temp (°C)	Sal (ppt)	DO (mg/L)	pH
09/08/11	20.71	26.64	7.54	7.86
19/08/11	20.24	27.54	7.75	7.94
27/10/11	11.22	21.9	9.58	7.98
19/09/12	17.92	25.61	7.74	7.74
11/11/12	8.31	19.79	10.8	8.12
20/09/13	17.18	26.92	7.61	7.31
7/10/13	14.32	26.10	7.15	*

*no data due pH probe malfunction

4.3 Soils and Sediments

Pore Water Salinity

At the restoration site, both shallow and deep salinity readings showed a shift in salinity distributions following restoration. Pre-restoration salinities were lower than those observed at the reference site, with the higher salinity classes missing from the distribution (Figure 22; Figure 23). The year following restoration (2010), pore water salinity levels were higher than pre or reference condition, and had more readings occurring in the higher salinity classes than in subsequent years. This was likely reflective of the hypersaline conditions at the site due to the lack of vegetative cover (e.g., exposed marsh surface/mudflat) and the high influx of new estuarine sediments (see Sediment Accretion and Elevation). The post-restoration Year 2 (2011) and Year 3 (2012) salinity level distribution at COG more closely matched that of the reference condition. By Year 4 (2013), no significant difference was detected between salinity levels at the two sites.

Due to the change in sampling technique made in 2011, a direct comparison of the 2011 through 2013 data to the previous years (2009 & 2010) was not viable. Considered individually, T-tests (95% CI) using only the 2013 (fourth year) data found a significant difference between the shallow and deep readings at both sites (COG-R: t(32)=-5.01, p<0.001); COG: t(35)=-7.18,

$p<0.001$). When compared to each other, no significant differences were found between sites in 2013 (Shallow $t(64)=-0.08$, $p>0.05$; Deep $t(64)=-0.08$, $p>0.05$). When compared to the previous year (2013- 2012), no significant differences were found at either site (COG-R Deep $t(58)=0.02$, $p>0.05$; COG-R Shallow $t(58)=0.02$, $p>0.05$; COG Deep $t(63)=-0.24$, $p>0.05$; COG Shallow $t(64)=0.43$, $p>0.05$). Although the difference was not significant, salinity levels were lower at both the restoration and the reference in 2013. The reduction was recorded for both sites suggesting a system wide occurrence and not a result of the restoration activities.

The similarity in salinity levels between the two sites (no statistically significant differences) since the second year post-restoration (2011) and the similar spatial and temporal trends, would suggest that conditions at the restoration site are stabilizing and reaching parity with the reference condition.

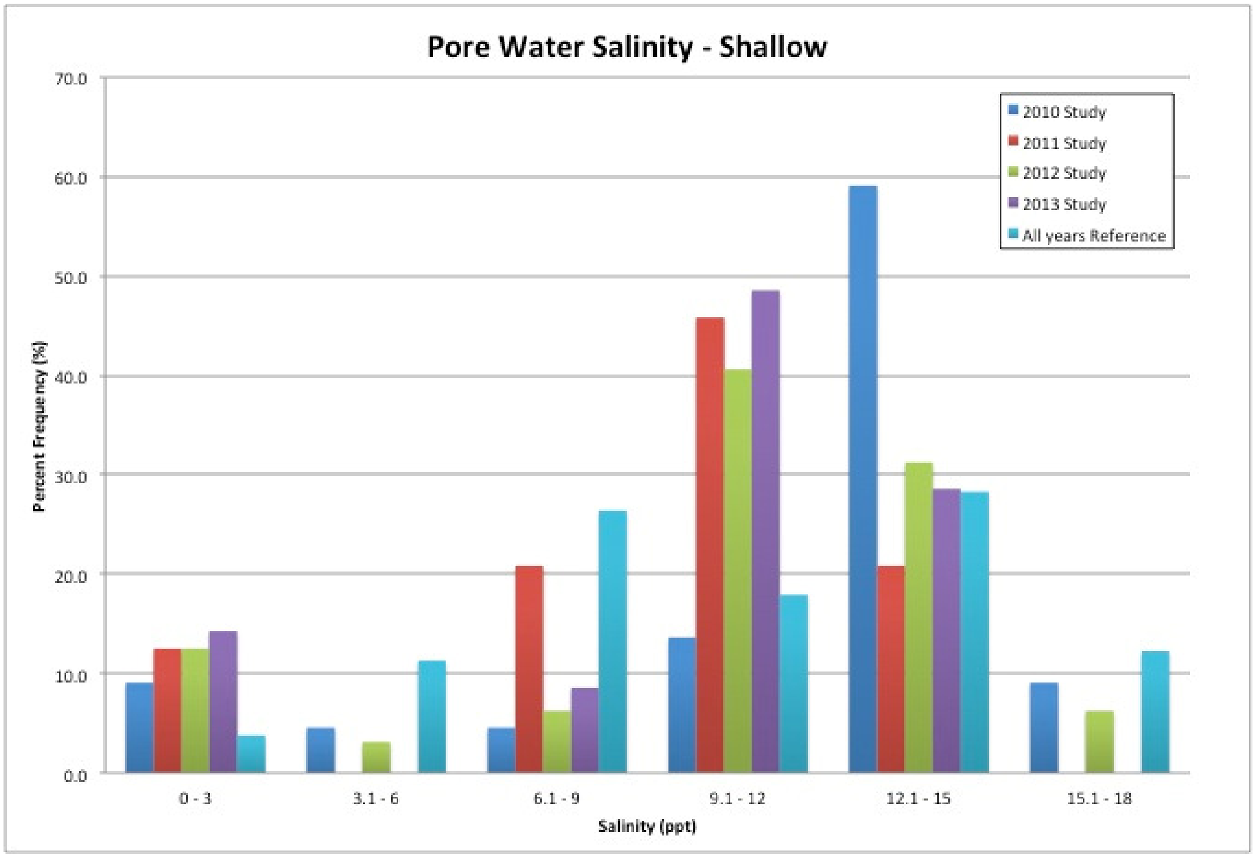


Figure 22 Frequency of pore water salinity values for shallow (15 cm) samples at COG and COG-R.

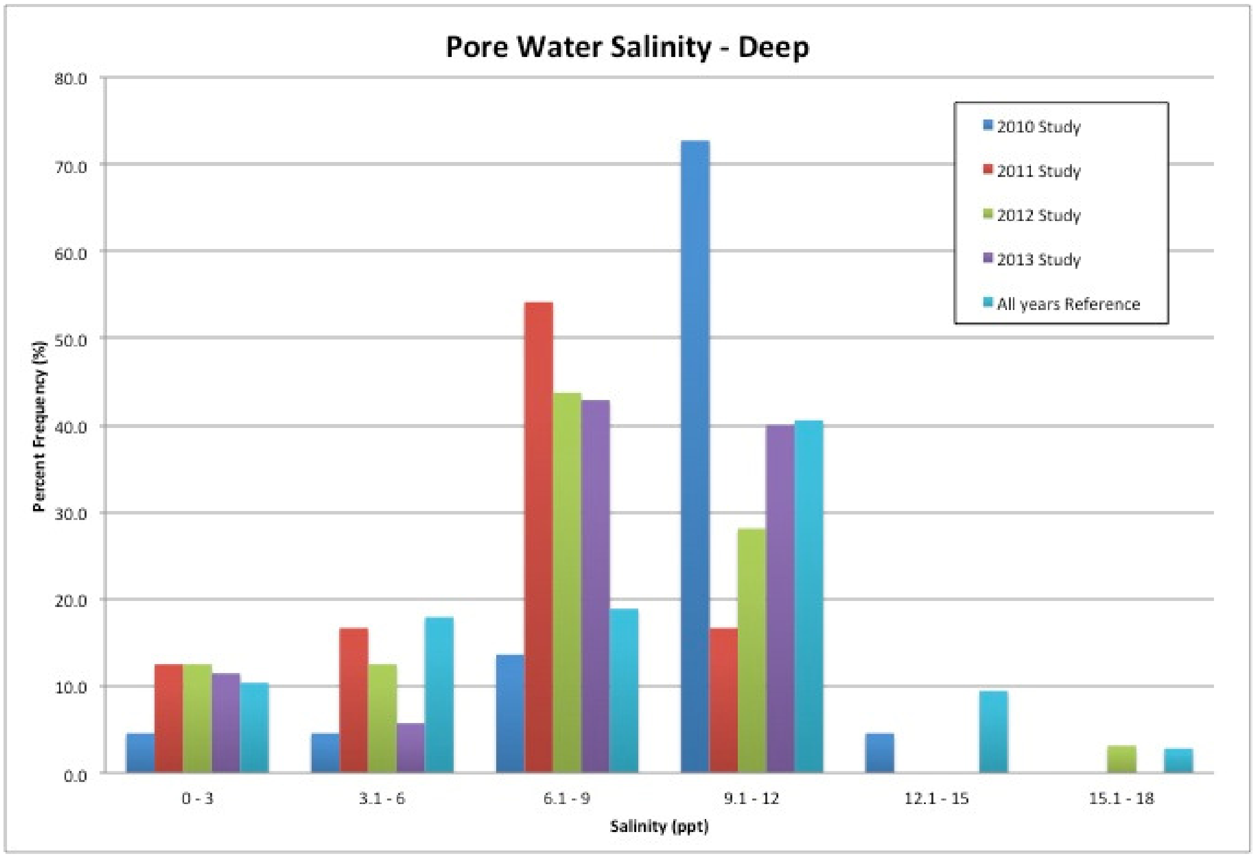


Figure 23 Frequency of pore water salinity values for deep (45 cm) samples at COG and COG-R.

Sediment Accretion and Elevation

One year after the water control structure was removed and dyke breached, all RSETs at COG recorded a decrease in surface elevation ranging from -0.1 (± 0.3) cm at RSET-02 located at the highest elevation to -2.1 (±0.1) cm at RSET-01 (Table 5). This was likely associated with dewatering and draining of the site. This hypothesis was supported by observations of surface compaction from the CBWES field crew. In addition, the highest amounts of sediment accretion (1.43 cm; Table 8) were recorded at RSET-02 and only a thin veneer of sediment was observed over the feldspar horizon at RSET-01 (Table 7). The 2.1 cm decrease in surface elevation was completely associated with subsurface processes. Subsurface processes contributed to decreases of -1.55 cm and -1.31 cm at RSET-02 and RSET-03 respectively.

Two years post-restoration, RSET-01 and RSET-02 recorded positive rates of surface elevation change. The highest (1.07 ± 0.12 cm) was recorded at RSET-01 closest to the borrow pit, followed by 0.1 (± 0.13) cm at RSET-02 (Table 2a). RSET-01 began to see the establishment of low marsh vegetation (Figure 8; Figure 24; Figure 26), which likely contributed to the positive surface elevation change. In addition, it signaled that the substrate was now firm enough for seedling colonization. New shoots of *Spartina alterniflora* and *Sueda maritima* were emerging at RSET-02 (Figure 24g; Figure 8). Salt marsh vegetation plays an important role in sediment trapping and surface consolidation (Friedrichs and Perry 2001; van Proosdij et al. 2006). RSET-03 recorded negative change in elevation (-0.4 ± cm), however, the data was impacted by a white-tailed deer footprint (data from those pins were excluded). Sediment accretion as measured by the marker horizons ranged from 1.32 cm at RSET-01 and 02 and the highest amounts at

RSET-03 (1.76 cm) (Table 5; Figure 26b,e). This station is the lowest of all in elevation relative to datum, which may be contributing to the higher amounts of accretion likely associated with more frequent inundation. Given that these values exceed the rate of surface elevation change, the substrate appears to still be compacting and perhaps decomposing similar to what was seen at the Walton River project site after an equivalent time frame (van Proosdij et al. 2010). This area was also mostly composed of decaying above ground vegetation, which would suggest below ground decomposition (Figure 25e), contributing to just over 2 cm of surface lowering.

Three years post-restoration RSET-01 continued to record a positive surface elevation change (0.83 ± 0.21 cm); however, this was mostly associated with strong sediment accretion values (1.17 cm) and the continued decay of the root mass (Table 5). The increased colonization by vegetation (Figure 26b) likely contributed to increasing sediment deposition. The rate of decay of below ground biomass continued to increase at RSET-02, increasing to a loss of 1.76 cm resulting in a surface lowering of 0.62 cm. The lowest amounts of sediment accretion were recorded at RSET-03 and coupled with decay of below ground biomass lead to surface lowering. In addition, the slowly decaying above ground plant material may be inhibiting or decreasing new vegetative growth. Figure 24L and Figure 26f illustrate that some new growth had appeared. The alternative hypothesis is a dewatering of sediments, decrease in pore space and compaction (Paquette et al. 2004).

By year four, all RSET stations showed positive changes in surface elevation (RSET-01 and 02) or an equilibrium condition (e.g. 0 cm change: RSET-03) (Table 5, Figure 29a). RSET-01 continued to record the greatest change in surface elevation (1.29 ± 0.35 cm) despite having the lowest contribution from sediment accretion (Table 5; Table 8). Sediment accretion was almost double at RSET-03 (2.28 cm; Table 8) which countered the higher rate of collapse in the below ground vegetative base (Table 5). By year four, most of the dead plant material (*Typha angustifolia* mats) had either been exported or decayed *in situ* (Figure 24e; Figure 25e; Figure 26e) potentially facilitating new vegetative growth. This was particularly relevant for RSET-02 and 03.

At COG-R, all RSET stations exhibited markedly lower changes in surface elevation in 2011-2012 compared to previous years (Table 6). RSET-03 even recorded surface lowering (-0.5 ± 0.1) cm. The largest change in surface elevation (0.2 ±0.1 cm) was recorded at RSET-02 within the high marsh (Table 6). Accretion values were notably lower than the range of those recorded at the restoration site (0.10 to 0.70 cm), in the high marsh and low marsh respectively (Table 8). RSET-01 and 03 recorded notable subsidence of the marsh surface of -0.52 and -0.81 cm respectively. This may suggest the influence of surface dewatering or excess evapotranspiration (Paquette et al. 2004). However, higher rates of surface elevation change were recorded at RSET-02 and 03 in 2012-13, ranging from 0.6 ± 0.8 cm (RSET 01) to 2.0 ± 1.3 cm (RSET-02) (Table 6; Figure 29b). Most of this change in surface elevation can be attributed to sediment accretion (0.25 to 1.38 cm) (Table 6) with minor contributions due to below ground production. RSET-02 did, however, record almost 1 cm of increase in surface elevation due to below ground processes (Table 6).

Prior to 2011-12, the main difference between COG and COG-R had been the behavior of the below ground component of the marsh surface. This continued to be the case in 2011-12, with accretion rates at the reference site almost half that recorded at the restoration site (Table 5; Table 6; Table 7; Table 8). In addition, both restoration and reference sites recorded markedly lower rates of accretion when compared with previous years. There were two primary hypotheses for this. Firstly, the lower accretion values during this year may be attributed to the month of data collection. During the 2011-12 season data were collected on 20 August, whereas previously data were collected on 23 September and 12 October. Both of these latter dates would have provided more opportunity for sediment import due to more frequent rain events and storms during that period. Secondly, the higher values within the restoration site were likely associated with increased inundation time, particularly at the far eastern edge of the site, due to the presence of only one opening for tidal water to enter and exit. By 2012-13, the rates of change in surface elevation were most comparable to values observed at the reference site and seemed to be following the shift in vegetative community structure recorded during this time period.

It appears that year four is a turning point towards conditions more comparable to the reference site and may be associated with a shift in vegetative community structure (Section 4.4). Different types of vegetation wrack (mats/piles of dead and decaying plant material) have different potential rates of decomposition. Types of vegetation that contain a lot of lignin as compared to cellulose decompose at a slower rate (Maccubbin and Hodson 1980). This includes species such as *Typha angustifolia* which dominated the site prior to restoration. Cellulose is quickly decomposed by microbial communities in wetland environments as compared to lignin fibers that require aerobic respiration in order to decompose (Reddy and DeLaune 2008). During periods of aerobic respiration, decomposition occurs quickly as compared to anaerobic respiration.

Depending on the amount and type of vegetation dominating the site prior to restoration, it may be critical to remove the vegetation entirely or partially from the site to ensure fast recovery of vegetation post-restoration. The removal is especially important on sites where appropriate hydrology is not going to be restored to the site (i.e. Cogmagun). Without appropriate hydrology, the site is not able to flush the plant debris off of the site and this can lead to mats forming on the marsh surface. The mat creates a barrier to new vegetation growth and can also impede oxygen getting into the soil to assist in the decomposition. Decrease of oxygen and inadequate flushing of the sediments can also lead to high levels of phytotoxins to accumulate (i.e. hydrogen sulfide) that affect the growth of plants (Koch and Mendelsshn 1989). This may explain some of the trends in the RSET data as year four saw a distinctive shift to halophyte dominance (Section 4.4).

COG is behaving very similarly to the Walton River restoration project in many regards. Both sites were former freshwater impoundments located along a tidal river with high suspended sediment concentrations (van Proosdij et al. 2010). Dykes were breached at both sites with minimal new excavation; however, one significant difference is the lack of a second breach at COG at the eastern edge of the former impoundment. This has resulted in marked differences in circulation and likely has contributed to the accumulation of decaying plant matter on the marsh surface in the area of RSET-03, as this material cannot be flushed out into the Bay. The anoxic conditions created by decaying plant matter can suppress new growth and water logged peats can see an increase in sulfate reduction leading to surface lowering (Portnoy 1999). The additional breaches at Walton allowed for sedimentary and nutrient material to flow through the site following the tidal flow of the main river and removing much of the decaying litter (van Proosdij et al. 2010). Although the COG marsh surface sits a bit higher in the tidal frame than Walton, the rates of accretion and change in surface elevation are similar (van Proosdij et al. 2010; Table 3; Table 7). However, ice deposition played a more important role at Walton in earlier years, contributing to higher rates of accretion in selected areas (van Proosdij et al. 2010), whereas there was little evidence of ice deposition at COG.

Table 5 RSET measurements from 2009 to 2013 at the COG.

Cogmagun Restoration				**Net change in elevation between sampling period (cm)**								
RSET-01 high marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	28	-3.9	-2.8	-2.0	-1.5	-1.5	-2.0	-1.7	-0.6	-3.1
mean (cm)	-2.1	3	118	-1.7	-1.6	-1.8	-2.1	-1.8	-1.9	-1.0	-0.9	-0.7
stdev	0.8	5	208	-4.2	-2.0	-2.1	-2.9	-2.5	-3.2	-3.5	-2.9	-2.2
SE	0.1	7	298	-1.7	-1.9	-1.9	-1.7	-1.7	-1.8	-1.9	-1.9	-1.5
2010-2011		1	28	0.6	0.5	0.2	-0.1	-0.2	0.1	-0.3	-0.3	0.5
mean (cm)	1.07	3	118	1.0	1.1	1.0	1.4	1.3	1.4	1.3	1.4	1.3
stdev	0.71	5	208	1.0	1.1	1.2	1.1	1.0	1.1	0.9	0.3	1.0
SE	0.12	7	298	2.1	2.1	2.1	2.2	2.0	2.0	1.8	1.7	1.6
2011-2012		1	28	1.0	0.8	0.9	1.0	0.7	0.5	0.7	0.5	1.0
mean (cm)	0.83	3	118	0.6	0.6	1.1	0.6	0.8	0.8	0.7	0.8	0.9
stdev	0.21	5	208	1.0	1.0	1.0	1.2	1.0	0.8	1.0	1.2	1.1
SE	0.03	7	298	0.6	0.8	0.7	0.8	0.8	0.5	0.5	1.2	0.8
2012-2013		1	28	1.5	0.9	0.7	0.7	0.8	1.3	0.9	0.6	0.9
mean (cm)	1.29	3	118	1.4	1.1	1.0	1.3	1.7	1.9	1.6	1.4	2.0
stdev	0.35	5	208	1.2	0.8	1.4	1.2	1.2	1.5	1.4	1.1	1.4
SE	0.06	7	298	1.2	1.8	1.8	1.5	1.5	1.5	1.3	1.3	1.7
RSET-02 mid marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	230	-0.6	-0.7	-3.3	0.2	-0.1	-1.1	-0.1	0.1	-0.2
mean (cm)	-0.1	3	320	0.9	-0.6	0	0.4	0.2	-0.6	-0.9	-2.2	-1
stdev	1.8	5	50	0.1	0.9	1.1	5.5	1.9	2.8	1.3	0.6	0.3
SE	0.3	7	140	0.5	1.7	1.2	-1	-1.1	-2.7	-2.3	-5.2	-0.5
2010-2011		1	230	-0.3	0.2	0.4	-0.5	-1.5	0.0	0.5	0.9	1.1
mean (cm)	0.1	3	320	0.5	0.3	0.0	-0.1	0.3	1.0	0.2	2.5	1.8
stdev	0.8	5	50	0.3	-0.8	-0.9	-1.1	0.2	0.2	-0.4	0.3	-0.1
SE	0.1	7	140	-0.7	-0.4	-0.1	0.5	0.1	-0.1	-0.4	-0.1	-0.4
2011-2012		1	230	0.2	-0.5	-0.5	-0.5	0.2	0.0	-0.8	-0.9	-0.7
mean (cm)	-0.62	3	320	-1.5	-0.8	-0.1	-0.3	-0.4	-0.4	-0.1	-0.1	-0.1
stdev	0.87	5	50	-0.5	-0.2	0.0	-0.3	-0.5	0.0	-0.5	-0.3	-0.3
SE	0.14	7	140	-0.2	-0.7	-0.1	-1.1	-4.7	-2.3	-1.5	-1.1	-0.7
2012-2013		1	230	0.7	1.4	1.3	0.7	0.3	1.0	2.3	2.1	1.0
mean (cm)	0.63	3	320	0.3	1.4	0.5	-1.1	-1.5	-1.2	-1.8	-1.2	0.3
stdev	1.33	5	50	2.3	1.2	1.0	1.2	1.4	0.0	0.1	-0.5	-1.9
SE	0.22	7	140	0.4	0.2	-0.6	-0.5	2.7	1.8	1.0	2.5	3.8
RSET-03 low marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	26	-0.5	0.1	-0.6	-1.1	-1.2	-0.6	-1.3	0.2	-0.4
mean (cm)	-0.5	3	116	-0.6	0.0	0.3	-1.1	-1.0	-0.3	0.8	0.0	0.5
stdev	0.6	5	206	-0.8	-1.3	-1.0	-1.1	-1.3	-0.8	-0.9	-0.4	-1.0
SE	0.1	7	296	0.3	-1.2	-1.2	-0.7	0.1	-0.5	0.4	0.3	-0.9
2010-2011		1	26	0.3	0.5	-0.3	0.1	0.9	-0.4	-0.4	0.5	0.3
mean (cm)	-0.1	3	116	-0.1	-0.1	0.1	0.1	0.1	0.1	-0.5	-0.7	-1.1
stdev	0.5	5	206	deer footprint			0.0	0.3	-0.1	0.5	0.0	-0.7
SE	0.1	7	296	-0.5	0.3	0.5	0.1	0.0	-0.1	-0.2	-0.4	-1.9
2011-2012		1	26	-0.2	-0.9	0.4	-0.5	-0.4	-0.2	-0.4	-0.6	-0.8
mean (cm)	-0.32	3	116	-1.0	-1.1	0.3	-0.3	0.0	-1.2	-0.8	-0.6	-0.6
stdev	0.85	5	206	2.1	3.0	-0.2	-0.1	-0.5	0.0	-0.2	0.3	0.1
SE	0.14	7	296	-1.2	-0.7	-0.6	-0.5	-1.0	-1.5	-1.1	-0.5	0.0
2012-2013		1	26	-0.1	-0.2	-0.6	0.1	-0.1	0.1	0.6	0.5	0.1
mean (cm)	0.00	3	116	1.1	0.2	-1.1	-0.1	-0.6	0.2	0.4	0.4	0.3
stdev	0.42	5	206	0.3	0.1	0.5	-0.3	0.4	0.1	0.2	0.1	-0.2
SE	0.07	7	296	-0.3	-0.2	-0.7	-0.4	-0.2	0.0	-0.2	-0.6	0.2

Table 6 RSET measurements from 2009-2012 at the COG-R.

Cogmagun Reference				**Net change in elevation between sampling period (cm)**								
RSET-01 low marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	57	0.8	1.0	0.1	1.5	1.5	0.5	2.6	1.1	1.0
mean (cm)	1.3	3	147	1.0	0.9	0.9	0.8	1.1	1.3	1.5	1.0	0.6
stdev	0.7	5	237	0.6	0.9	2.8	2.1	0.8	1.0	0.8	0.8	1.3
SE	0.1	7	327	1.3	0.4	1.8	2.2	1.1	1.1	3.0	1.9	2.1
2010-2011		1	57	0.7	0.5	1.2	0.4	0.2	0.5	1.2	0.9	0.2
mean (cm)	0.8	3	147	1.0	1.4	1.1	1.5	0.9	0.6	0.2	1.4	2.1
stdev	0.5	5	237	0.9	1.7	-0.4	0.7	1.4	0.8	0.4	1.0	0.0
SE	0.1	7	327	0.6	0.7	0.9	1.7	0.5	0.3	0.9	0.3	0.8
2011-2012		1	57	0.2	0.3	0.0	0.1	-0.2	0.4	-0.8	0.5	0.7
mean (cm)	0.2	3	147	0.2	0.1	-0.2	-1.0	-0.4	0.0	0.5	0.0	-1.8
stdev	0.8	5	237	-0.4	-0.2	-0.2	-1.2	-0.4	0.6	1.2	0.6	0.6
SE	0.1	7	327	1.7	0.0	0.2	0.4	2.5	1.5	0.1	0.5	0.3
2012-2013		1	57	0.5	0.5	0.9	0.6	0.8	0.7	2.1	1.3	0.7
mean (cm)	0.6	3	147	0.0	0.0	1.4	2.0	1.7	0.4	0.7	0.8	2.1
stdev	0.8	5	237	0.3	0.5	0.6	1.7	0.2	-0.3	0.1	0.9	1.2
SE	0.1	7	327	0.1	0.6	0.2	0.5	-1.5	-1.8	-0.4	0.1	-0.2
RSET-02 high marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	344	0.9	1.2	0.5	1.0	0.3	0.6	1.1	1.6	0.6
mean (cm)	0.8	3	74	1.1	0.5	0.7	0.8	0.2	0.9	0.9	1.0	0.5
stdev	0.4	5	164	1.3	0.6	0.5	1.0	-0.3	1.2	0.6	0.9	1.1
SE	0.1	7	254	0.9	0.4	0.2	0.0	1.2	0.0	0.6	1.3	1.7
2010-2011		1	344	0.7	0.3	1.7	0.5	0.6	1.3	0.6	-0.2	0.3
mean (cm)	0.8	3	74	0.5	1.0	0.9	0.9	1.2	0.6	-1.2	1.0	1.2
stdev	0.6	5	164	0.9	1.3	-0.1	0.5	1.4	-0.2	0.7	0.7	0.7
SE	0.1	7	254	1.3	2.1	1.7	1.8	0.5	0.9	0.5	1.1	0.7
2011-2012		1	344	0.3	-0.9	-0.4	-0.2	0.0	-0.1	-0.4	0.6	0.1
mean (cm)	0.1	3	74	0.1	0.7	0.2	0.0	0.7	0.0	1.0	0.2	-0.7
stdev	0.8	5	164	1.2	1.3	1.6	1.4	-0.2	0.8	1.0	0.6	-0.6
SE	0.1	7	254	0.2	-0.2	-0.3	-2.6	-0.2	0.2	0.1	-0.2	0.0
2011-2012		1	344	3.2	3.1	2.7	3.0	2.1	1.7	1.8	0.7	1.7
mean (cm)	2.0	3	74	3.2	2.2	3.5	3.5	2.3	3.5	5.2	3.8	4.3
stdev	1.3	5	164	0.0	0.3	1.0	0.6	1.4	0.7	0.7	1.3	1.7
SE	0.2	7	254	1.0	0.6	1.5	4.1	1.1	1.0	1.2	1.8	1.4
RSET-03 mid marsh		Position	Bearing	Pin 1	Pin 2	Pin 3	Pin 4	Pin 5	Pin 6	Pin 7	Pin 8	Pin 9
2009-2010		1	148	1.6	2.2	2.4	2.4	1.9	2.0	2.4	1.9	2.0
mean (cm)	1.7	3	238	1.4	1.1	0.9	2.4	0.8	1.5	2.0	2.3	1.5
stdev	0.5	5	328	2.1	1.9	2.0	1.3	0.8	1.8	1.7	1.6	1.3
SE	0.1	7	58	1.6	2.0	1.4	0.9	1.6	1.7	1.4	0.6	1.0
2010-2011		1	148	1.6	0.6	1.3	1.4	1.4	1.2	1.3	1.8	1.5
mean (cm)	1.4	3	238	1.2	3.1	2.8	0.3	2.2	0.9	0.5	0.9	0.5
stdev	0.9	5	328	0.3	0.6	0.4	1.0	2.2	1.6	3.6	3.5	2.7
SE	0.2	7	58	0.7	1.1	1.9	2.1	1.1	0.4	2.5	0.9	0.9
2011-2012		1	148	-0.3	-0.4	-1.1	-1.1	-1.2	-0.8	-0.7	-1.1	-0.5
mean (cm)	-0.5	3	238	0.1	-1.8	-0.9	0.1	-0.1	0.3	0.4	0.0	0.5
stdev	0.8	5	328	0.2	0.0	0.0	0.1	-1.1	-1.1	-2.6	-1.6	-1.2
SE	0.1	7	58	0.3	-0.8	-0.8	0.1	-0.6	0.2	-2.0	0.1	0.0
2012-2013		1	148	1.3	1.5	1.4	1.1	1.3	0.9	1.0	0.8	0.5
mean (cm)	1.1	3	238	1.2	1.4	1.0	0.6	1.0	1.4	0.9	0.7	0.9
stdev	0.3	5	328	1.3	1.5	1.3	1.5	1.2	1.3	0.9	0.6	0.9
SE	0.1	7	58	1.2	2.2	0.8	0.7	1.2	1.3	1.6	1.2	0.9

Table 7 Net sediment accretion 2009-2013 measured by the marker horizons at COG.

Cogmagun - MH measurements 2012-13				net accretion (cm/yr)			
RSET-01 HM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**
core 1a	4.73	1	Ok	0.00			
core 1b	3.63	1	Ok	0.00			
core 1c	2.70	1	Ok	0.00			
mean	**3.68**			**0.00**	**1.13**	**1.17**	**1.38**
RSET-02 - MM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**
core 2a	6.48	1	Great	1.19			
core 2b	4.83	1	Ok	1.28			
core 2c	4.63	1	Ok	1.81			
mean	**5.31**			**1.43**	**1.31**	**1.14**	**1.43**
RSET -03 LM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**
core 3a	2.90	1	Ok	0.61			
core 3b	8.58	1	Great	0.86			
core 3c	4.23	3	Great	0.89			
mean	**5.23**			**0.79**	**1.75**	**0.41**	**2.28**

Table 8 Net sediment accretion 2009-2013 measured by the marker horizons at COG-R.

Cogmagun Reference - MH measurements 2012-13				net accretion (cm/yr)			
RSET-01 LM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**
core 1a	2.23	1	Great	0.99			
core 1b	2.63	1	Great	0.61			
core 1c	2.55	1	Great	0.63			
mean	**2.47**			**0.74**	**0.78**	**0.70**	**0.25**
RSET-02 - HM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**
core 2a	2.45	1	Great	0.70			
core 2b	3.18	1	Good	1.06			
core 2c	3.78	1	Good	1.35			
mean	**3.13**			**1.04**	**0.91**	**0.10**	**1.08**
RSET -03 MM	**mean (cm)**	**# cores**	**quality**	**2009-10**	**2010-11**	**2011-12**	**2012-13**
core 3a	3.58	1	Great	0.99			
core 3b	3.40	1	Great	1.25			
core 3c	4.68	1	Great	0.93			
mean	**3.88**			**1.05**	**1.42**	**0.28**	**1.13**

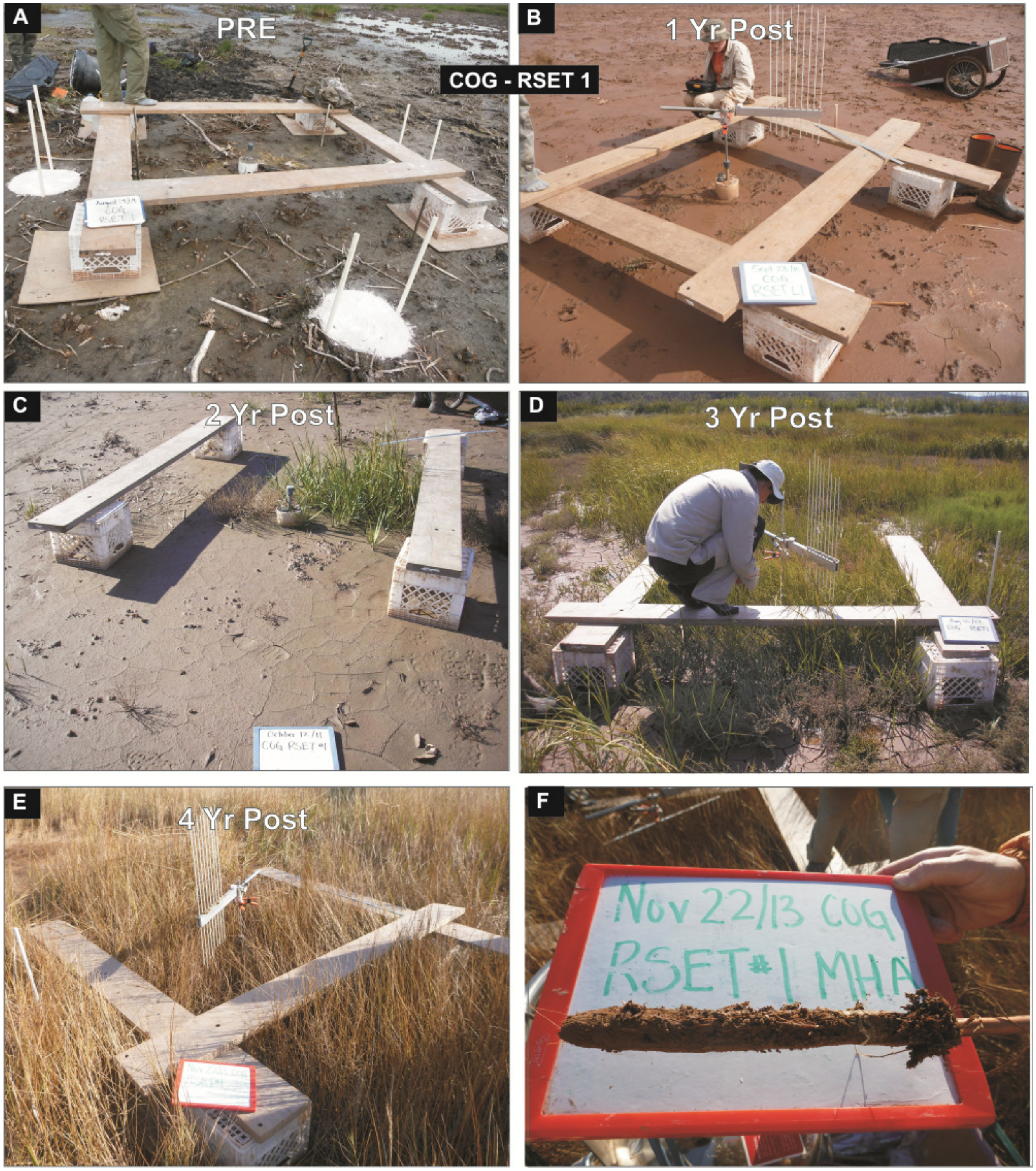


Figure 24 RSET-01 at COG a) Pre (19 August 2009), b) Year 1 Post (23 September 2010), c) Year 2 Post (12 October 2011), d) Year 3 Post (20 August 2012) and e) Year 4 Post (22 November 2013). Photo (f) shows and example of a marker horizon core. Photographs taken by CBWES Inc.

Figure 25 RSET-02 at COG a) Pre (19 August 2009), b) Year 1 Post (23 September 2010), c) Year 2 Post (12 October 2011), d) Year 3 post (20 August 2012) and e) Year 4 post (22 November 2013). Photo (f) shows an example of a marker horizon core. Photographs taken by CBWES Inc..

Figure 26 RSET-03 at COG a) Pre (19 August 2009), b) Year 1 Post (23 September 2010), c) Year 2 Post (12 October 2011), d) Year 3 post (20 August 2012) and e) Yr 4 post (Nov 22, 2013). Photo (f) shows an example of a marker horizon core. Photographs taken by CBWES Inc..

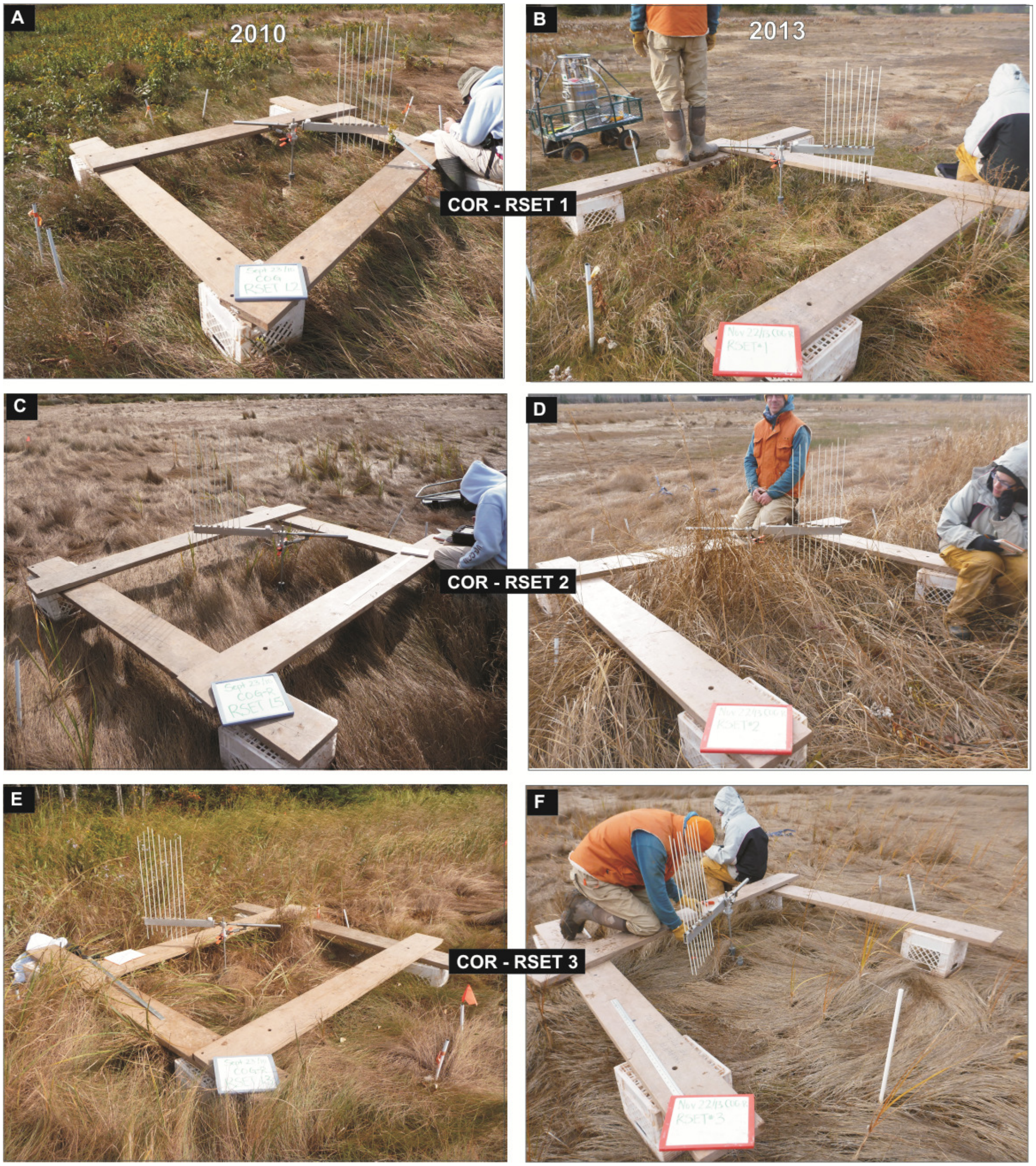


Figure 27 a) RSET-01, b) RSET-02 and RSET-03 at COG-R on 23 September 2010 and 22 November 2013. Photographs taken by CBWES Inc.

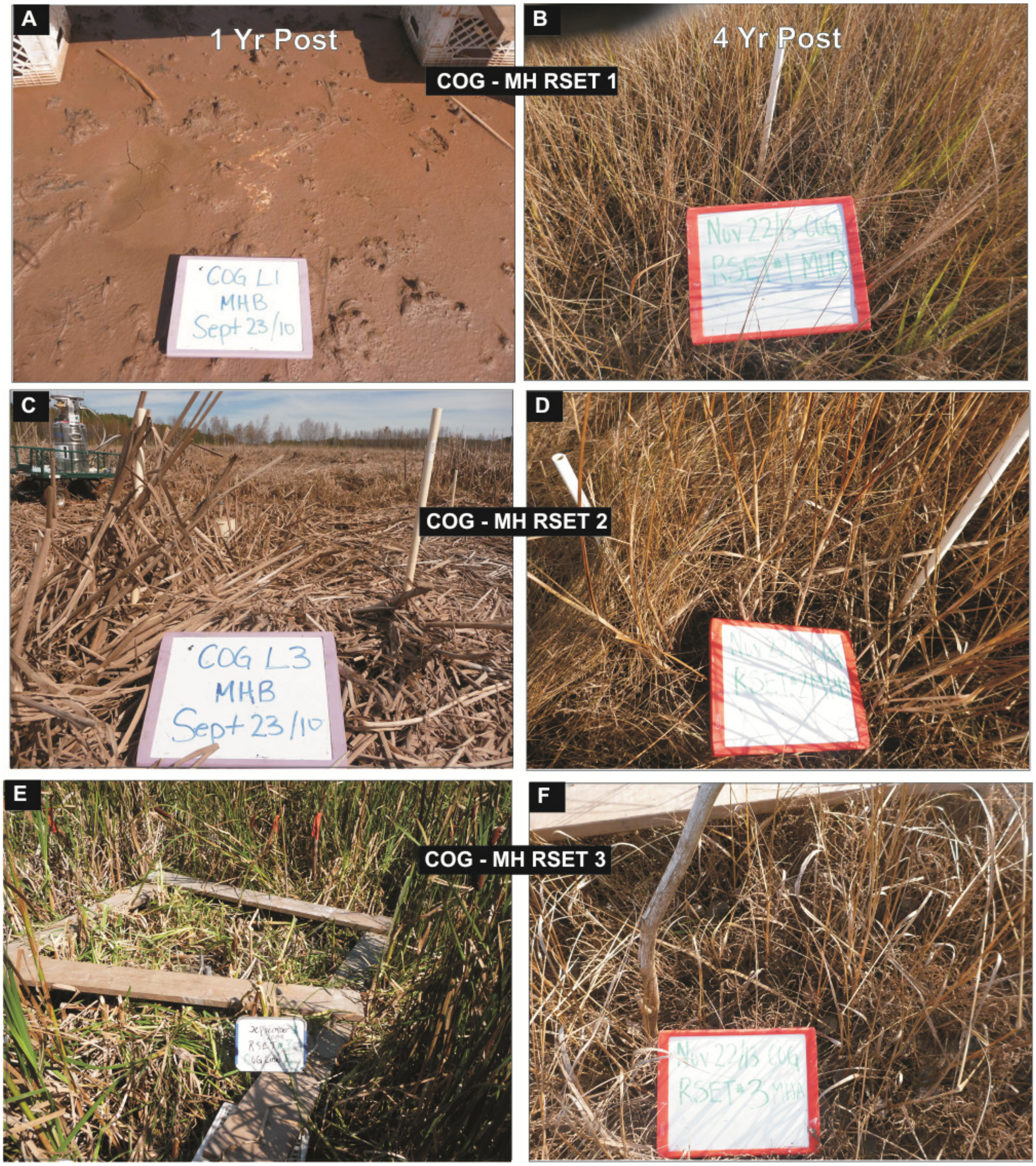


Figure 28 Marker horizon locations at COG in 2010 and 2013 to compare presence and absence of vegetation. Photographs taken by CBWES Inc.

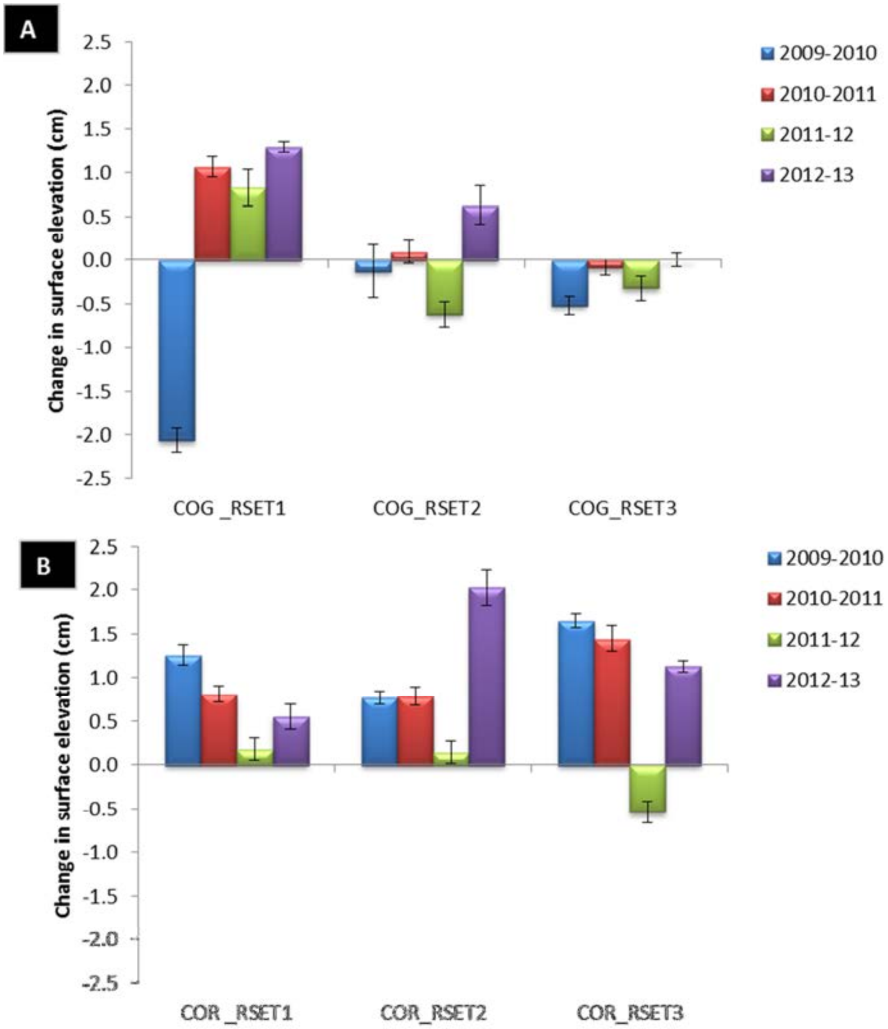


Figure 29 Changes in surface elevation measured by the RSET from 2009 to 2013 at a) COG and b) COG-R.

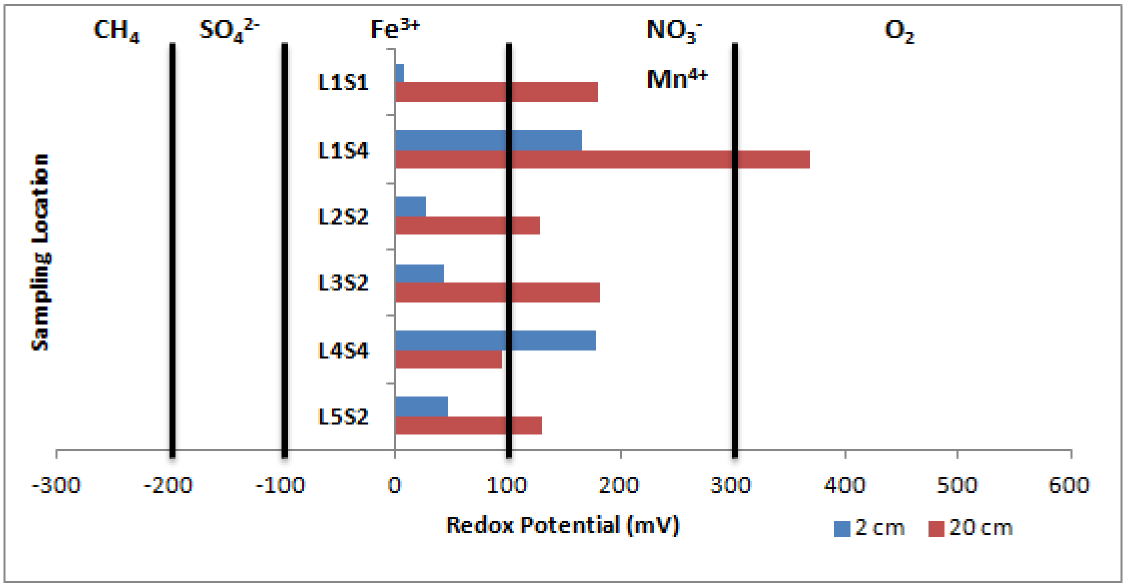
Soil Chemistry (Redox Potential)

COG experienced anaerobic respiration at all locations except L1S4, which experienced aerobic respiration (Figure 30a). Anaerobic conditions were recorded for all sampling locations at the reference site (Figure 30b). The dominant redox reaction occurring at 20 cm depth within the sediment at both the restoration and the reference was nitrate and manganese (IV) reduction. The dominant redox reaction occurring at 2 cm within the restoration site was iron (III) reduction, as compared to nitrate and manganese (IV) reduction at the reference site.

The restoration site was more reduced closer to the surface at all but one sampling location (L4S4). In comparison, shallow conditions at the reference site were relatively evenly divided between highly reduced (L1S4, L2S1 and L4S3) and more oxygenated (L3S2, L4S1 and L5S2). It is interesting to note that COG - L1S4, located very close to the borrow pit, experienced the highest redox potential values, likely due to good drainage. The higher values at the 20 cm zone for both the restoration and reference indicate the potential for oxygen to reach the roots and decrease the overall stress on the vegetation. Overall, the dominant reactions occurring at COG were comparable to those at the reference site.

The values recorded for both sites indicate moderate anaerobic stress on vegetation and potential for moderate decomposition rates within the soil under anaerobic conditions. There is potential for sulfide, a known phytotoxin, to accumulate within the soil, as the soil is heterogeneous and multiple redox reactions can occur at one time. However, sulfate reduction was not the dominant redox reaction at the time of measurement and therefore, high levels are not expected to accumulate. The areas that were experiencing oxygen reduction would allow for the oxidation of the reduced compounds (i.e. sulfide) and decrease the level within the soil. The values indicate appropriate drainage and a lack of water logging at the sampling locations, therefore, the high anaerobic conditions needed for high sulfide levels to accumulate were not occurring.

a)



b)

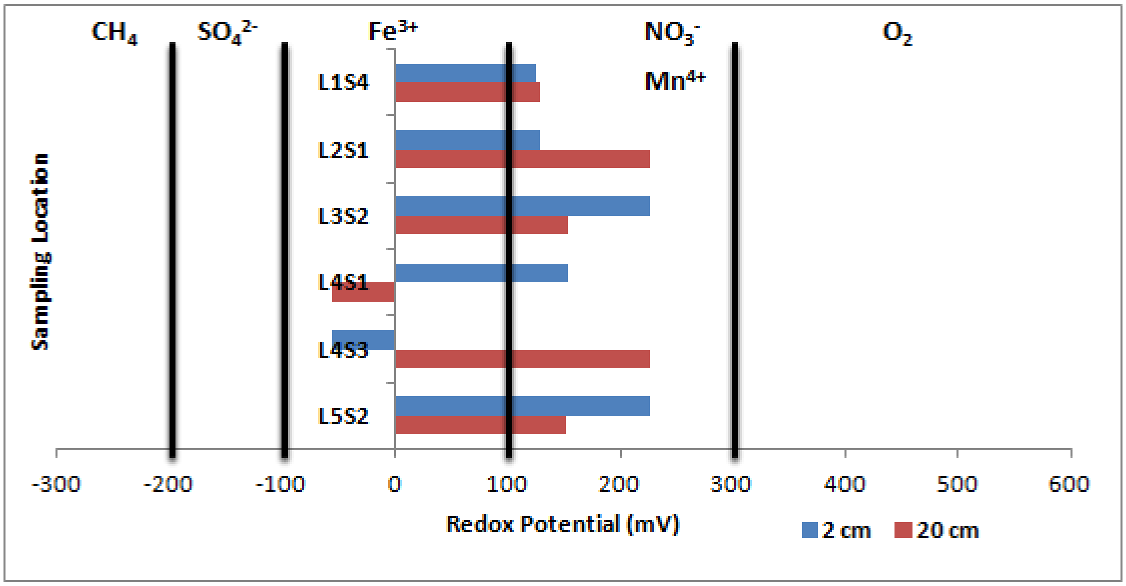


Figure 30 Mean redox potential values in relation to dominant reduction oxidation reaction occurring at 2 cm and 20 cm depth segmented into ranges of redox potential for dominant reduction reactions at: a) COG; and b) COG-R.

4.4 Vegetation

Species Abundance and Frequency Trends

There has been a major shift in vegetation from 2009 through 2013 on the restoration site. Many species of upland or freshwater vegetation that were present in 2009 are now absent (e.g. *Carex stipata*, *Impatiens capensis*, *Lysimachia terrestris* (Table 9). While few halophytes were detected on the site prior to restoration, many are now present, and some have shown significant increases (*Salicornia europea*, *Spartina alterniflora*, *Spartina patens*) (Table 9). Indicators of disturbance (change) caused by dyke breaching (bare ground, unvegetated area) increased in 2010 but have declined since.

Community Patterns

COG vegetation composition has moved closer to COG-R vegetation (Figure 31). Most of the restoration site plots originally appeared on the lower left side of the graph and were characterized mainly by a weedy, terrestrial or freshwater wetland flora, whereas the reference plots (upper right side of graph) had a range of salt marsh vegetation from brackish areas characterized by *Spartina pectinata*, and *Juncus gerardii*, to high marsh (*S. patens*) with only a small amount of low marsh (*S. alterniflora*) (Figure 31). By 2011, most of the restoration site plots had moved on the graph toward the right side, indicating greater abundances of halophytes. By 2012, there were many plots at COG containing low marsh vegetation (mainly *S. alterniflora*), and very similar in composition to reference low marsh plots at the lower right of the graph. In particular, there were many plots at COG containing *S. alterniflora*, and early successional salt marsh annuals (*Atriplex* sp., *Sueda maritima*, *Salicornia europea*). By 2012 there were also a few plots at COG that had relatively high coverage of *S. patens* and this trend continued into 2013 to the point where *S. patens* abundance was similar at the reference and restoration sites. Thus high marsh vegetation is colonizing COG, although the community types at the two sites still differ: the restoration site contains very little of the *Juncus gerardii* community (Porter et al. submitted) that is common at the reference site.

Other Site-level Trends

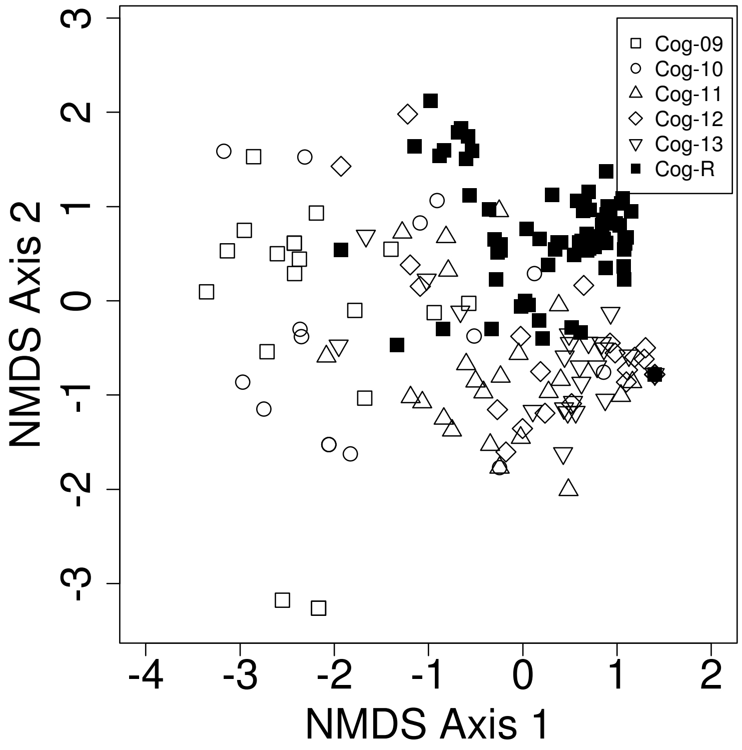
Average number of species per plot has fluctuated somewhat at COG, while remaining stable at COG-R(Figure 32; Table 10). Halophyte richness and abundance have both increased at the restoration site (Figure 33, Figure 34; Table 11, Table 12). Halophytic richness became equivalent to that at the reference site by 2011 and actually exceeded the reference site in 2013 (4 years post-restoration). Halophytic abundance was still significantly lower at COG in 2012, but has caught up by 2013, mainly due to increases in *S. patens* and *Salicornia europea* (Table 9). The amount of unvegetated area increased at COG between 2009 (immediate post-restoration) and 2010 (1 year post) (Figure 35; Table 13), but dropped again in 2011 (2 years post), stayed low in 2012, and dropped to almost nil by 2013 due to heavy colonization and persistence by salt marsh indicator species. The amount of dead plant material showed a similar pattern, peaking at the restoration site the first year following restoration, and subsequently declining, with very little presence of dead material by 2013 (Figure 36; Table 14).

Individual Plots at COG

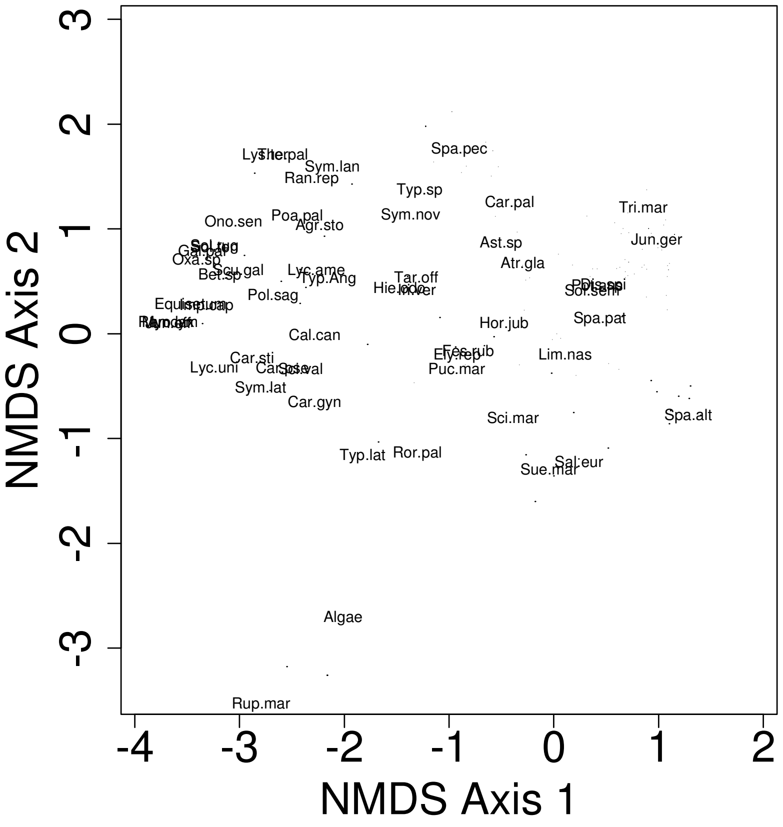
Plot species composition and abundance were relatively consistent at the reference site over the study period, so the focus here is on changes at the restoration site. Many plots were colonized by *S. alterniflora* at the restoration site, although in many locations this did not really begin until 2011 (e.g. COG L1S5, COG L2S4, COG L3S4, COG L2S5). A handful of plots showed a similar trend, but with *S. patens* coming in as well (COG L1S2, COG L3S2, COG L5S4). One plot (COG L1S3, not sampled in 2012) was unvegetated through 2011, but was heavily colonized by *S. alterniflora* and *S. europea* by 2013. One plot containing *Typha* and *S. pectinata* prior to dyke breaching (COG L3S1), saw an increase in *S. pectinata*, but still had some *Typha* in 2013 (?). Two plots at front stake locations (upland edge) originally had grasses, which were later colonized by *Typha*. Many plots that were colonized in 2011 or 2012 by *S. alterniflora* showed later colonization by *S. patens* (e.g. COG L3S2, L3S4, L4S2, L5S5), suggesting that it was more dispersal limited than *S. alterniflora*, at least in this system.

In conclusion, there has been a steady increase in salt marsh vegetation at the restoration site. Low marsh communities are dominant in some parts of the site, but there has been an increase in high marsh vegetation too (*S. patens* community type (Porter et al. submitted). In 2013, the restoration site clearly overlapped with the reference site in low marsh areas, and contained a large amount of *S. patens* (representative of high marsh conditions) usually mixed with *S. alterniflora*, but still lacked the *J. gerardii* community present at the reference site.

a) Plots



b) Species



c) Plot data: 2013 only.

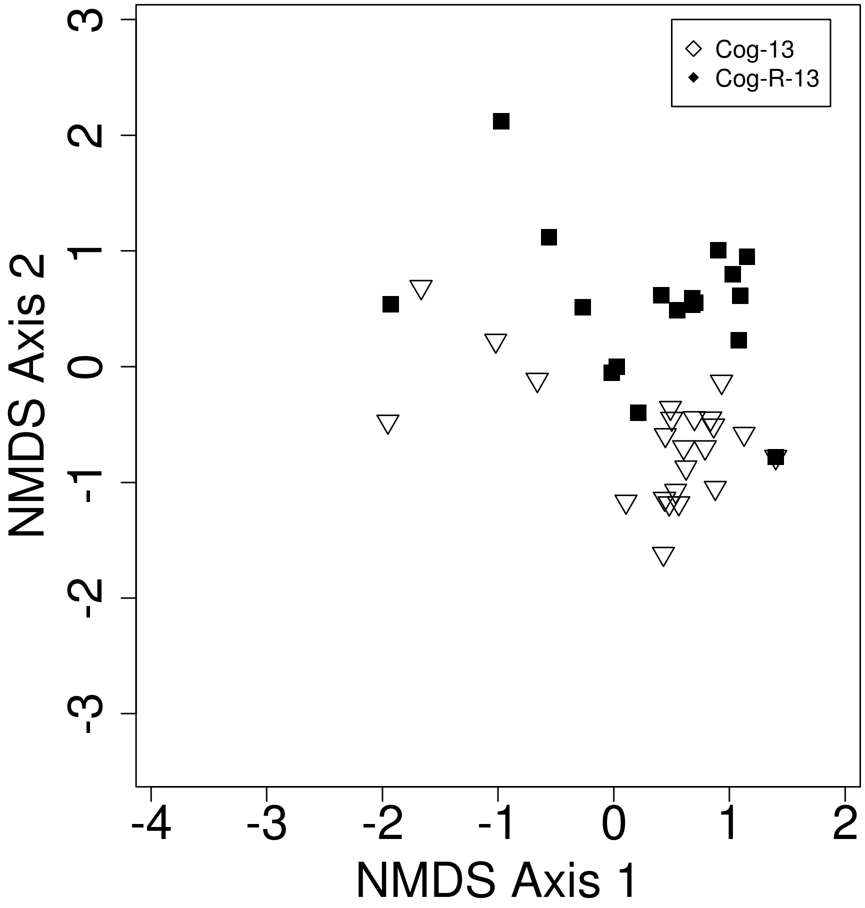


Figure 31 Non-metric multidimensional scaling analysis of plot vegetation (species abundance and composition) for COG and COG-R (stress: 0.09).

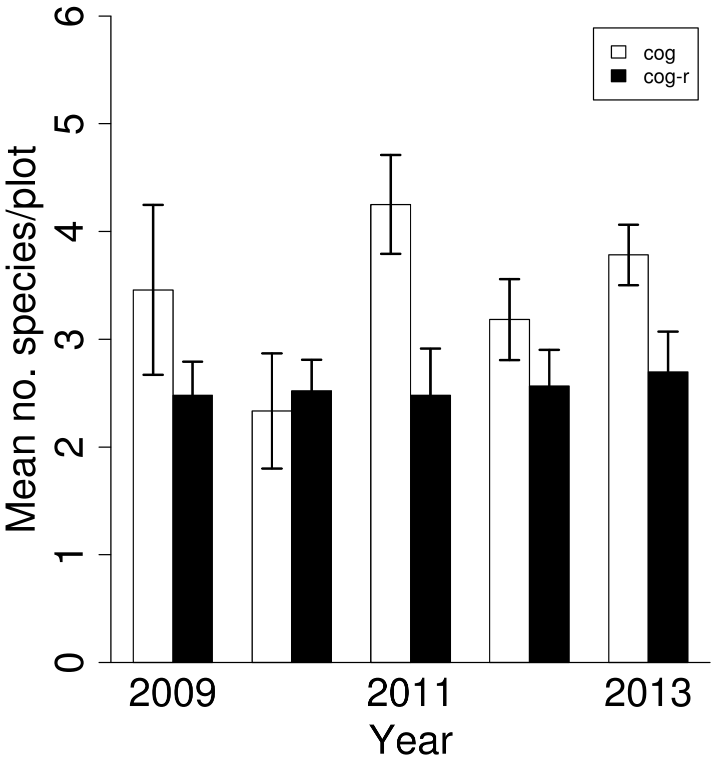


Figure 32 Mean plot species richness at COG and COG-R from 2009-2013.

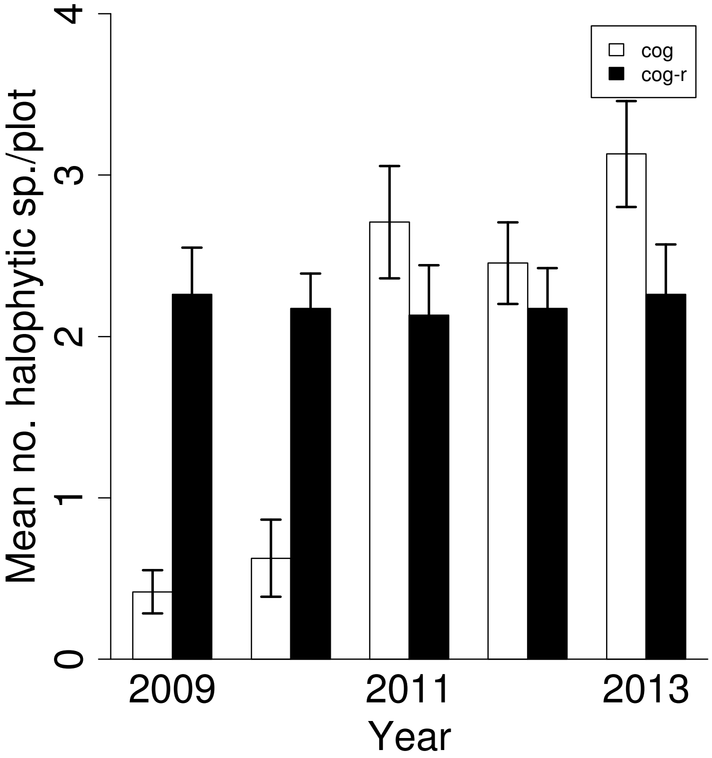


Figure 33 Halophytic species richness at COG and COG-R sites between 2009-2013.

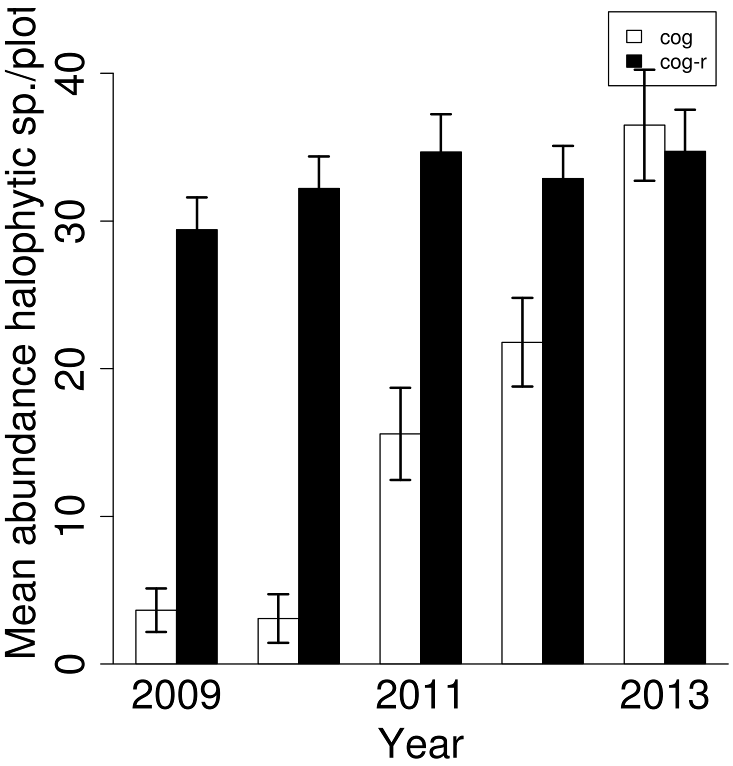


Figure 34 Halophytic species abundance at COG and COG-R between 2009-2013.

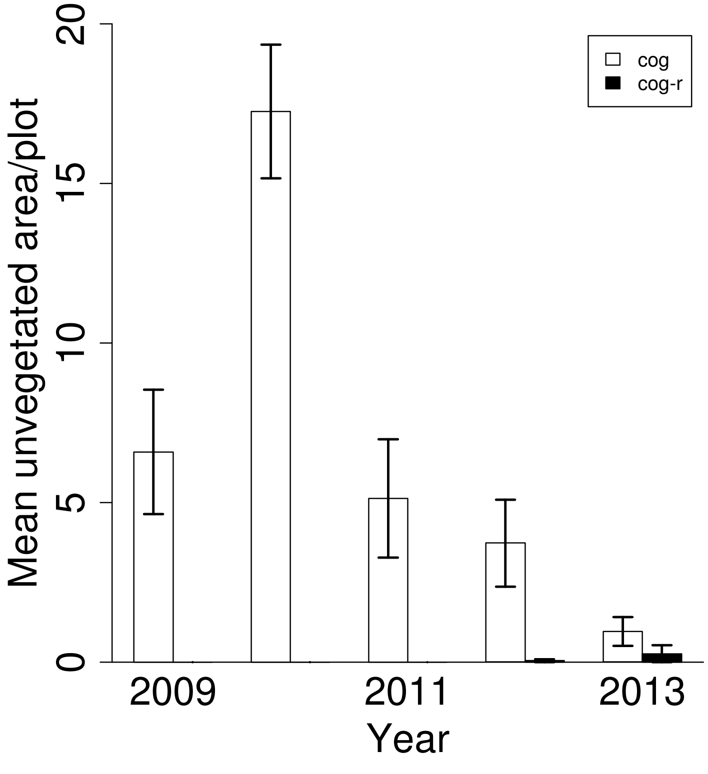


Figure 35 Unvegetated area at COG and COG-R between 2009 and 2013.

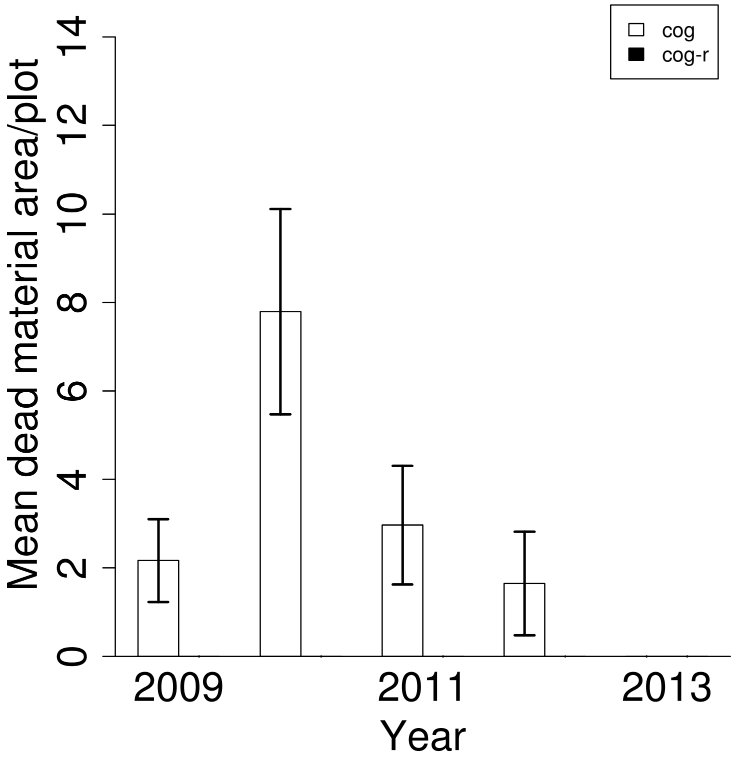


Figure 36 Dead plant material at COG and COG-R between 2009 and 2013

Table 9 Abundances and frequencies of species in the vegetation at COG and COG-R (for COG-R , only 2013 results are shown).

Species names	Abbreviation	COG 2009 means	COG 2009 freq.	COG 2010 means	COG 2010 freq.	COG 2011 means	COG 2011 freq.	COG 2012 means	COG 2012 freq.	COG 2013 means	COG 2013 freq.	COG-R 2013 means	COG-R 2013 freq.
Agrostis stolonifera	Agr.sto	1.62	3	1.42	3	1.5	3	2	3	0.65	2	0	0
Algae	Algae	1.08	3	0.05	2	1.76	4	0	0	0	0	0	0
Aster sp.	Ast.sp	0	0	0	0	0.14	2	0	0	0	0	0	0
Atriplex glabrisculata	**Atr.gla**	**0.14**	**2**	**0.97**	**4**	**0.65**	**6**	**0.32**	**1**	**0**	**0**	**0.35**	**5**
Betula sp.	Bet.sp	0.05	2	0	0	0	0	0	0	0	0	0	0
Calamagrostis canadensis	Cal.can	0.02	2	1.21	3	0.04	1	0.5	2	0.3	1	0	0
Carex gynandra	Car.gyn	0	0	0	0	0.08	1	0	0	0	0	0	0
Carex paleacea	**Car.pal**	**0**	**0**	**0**	**0**	**0**	**0**	**0.01**	**1**	**0.17**	**1**	**1.74**	**3**
Carex pseudo-cyperus	Car.pse	0	0	0.17	1	0	0	0	0	0	0	0	0
Carex stipata	Car.sti	1.29	3	0.75	3	0	0	0	0	0	0	0	0
Taraxacum offinialis	Tar.off	0	0	0	0	0.38	1	0.32	1	0	0	0	0
Distichlis spicata	**Dis.spi**	**0**	**0**	**0**	**0**	**0**	**0**	**0.91**	**1**	**0.57**	**1**	**1.96**	**6**
Elymus repens	Ely.rep	0	0	0.21	1	0	0	0.09	1	0.23	2	0	0
Equisetum sp.	Equisetum	0.38	2	0	0	0.01	1	0	0	0	0	0	0
Festuca rubra	Fes.rub	0.04	1	0	0	0	0	0	0	1.57	3	0.04	1
Gallium palustre	Gal.pal	0.93	4	0	0	0.01	1	0	0	0	0	0	0
Hierochloe odorata	Hie.odo	0.22	3	0.04	1	1.92	4	0.64	2	0.7	2	0.57	2
Hordeum jubatum	Hor.jub	0	0	0	0	0	0	0	0	0	0	0	0
Impatiens capensis	Imp.cap	1.59	4	0	0	0.01	1	0	0	0	0	0	0
Iris versicolor	Iri.ver	0	0	0	0	0	0	0.05	1	0	0	0	0
Juncus effusus	Jun.eff	0.25	1	0	0	0	0	0	0	0	0	0	0
Juncus gerardii	**Jun.ger**	**0.04**	**1**	**0.38**	**2**	**2.17**	**3**	**1.05**	**2**	**1.04**	**2**	**13.04**	**12**
Limonium nashii	**Lim.nas**	**0**	**0**	**0**	**0**	**0.03**	**3**	**0.01**	**1**	**0.11**	**4**	**0.13**	**1**
Lycopus americanus	Lyc.ame	0.05	2	0.01	1	0	0	0	0	0	0	0	0
Lycopus uniflorus	Lyc.uni	0.21	2	0	0	0	0	0	0	0	0	0	0
Lysimachia terrestris	Lys.ter	0.17	1	0.21	1	0	0	0	0	0	0	0	0
Myosotis laxa	Myo.lax	0.21	1	0	0	0	0	0	0	0	0	0	0
Onoclea sensibilis	Ono.sen	0.38	2	0.04	1	0.01	1	0	0	0	0	0	0
Oxalis sp.	Oxa.sp	1.04	3	0	0	0	0	0	0	0	0	0	0
Poa palustris	Poa.pal	0.38	1	1.12	2	0	0	0.82	1	0	0	0	0

Species names	**Abbreviation**	**COG 2009 means**	**COG 2009 freq.**	**COG 2010 means**	**COG 2010 freq.**	**COG 2011 means**	**COG 2011 freq.**	**COG 2012 means**	**COG 2012 freq.**	**COG 2013 means**	**COG 2013 freq.**	**COG-R 2013 means**	**COG-R 2013 freq.**
Polygonum sagittatum	Pol.sag	0.19	3	0	0	0.04	1	0	0	0	0	0	0
Potentilla anserina	**Pot.ans**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
Puccinelia maritima	**Puc.mar**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
Ranunculus cymbalaria	**Ran.cym**	**0.79**	**1**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
Ranunculus repens	Ran.rep	0.5	1	0.05	2	0.12	1	0	0	0	0	0	0
Rorippa palustris	Ror.pal	0	0	0	0	0.08	1	0	0	0	0	0	0
Ruppia maritima	**Rup.mar**	**0.08**	**1**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
Salicornia europea	**Sal.eur**	**0**	**0**	**0.04**	**1**	**1.23**	**15**	**1.56**	**10**	**7.66**	**17**	**0**	**0**
Scirpus cyperinus	Sci.cyp	0.62	1	0.18	3	0	0	0	0	0	0	0	0
Scirpus maritimus	**Sci.mar**	**2.21**	**4**	**0**	**0**	**2.38**	**8**	**1**	**1**	**0.61**	**3**	**0**	**0**
Scirpus validus	Sci.val	3.17	7	0.17	2	0	0	0	0	0	0	0	0
Scutellaria galericulata	Scu.gal	0.5	3	0	0	0	0	0	0	0	0	0	0
Solidago rugosa	Sol.rug	0.12	1	0	0	0	0	0	0	0	0	0	0
Solidago sempervirens	**Sol.sem**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.59**	**6**
Solidago tenuis	Sol.ten	0.04	1	0	0	0	0	0	0	0	0	0	0
Spartina alterniflora	**Spa.alt**	**0**	**0**	**0.39**	**3**	**2.94**	**9**	**9.43**	**17**	**16.36**	**20**	**5.96**	**10**
Spartina patens	**Spa.pat**	**0**	**0**	**0**	**0**	**0.8**	**3**	**2.89**	**9**	**6.43**	**10**	**8.52**	**10**
Spartina pectinata	**Spa.pec**	**0.38**	**1**	**1.04**	**1**	**1**	**2**	**2.09**	**3**	**0.57**	**1**	**2.17**	**2**
Sueda maritima	**Sue.mar**	**0**	**0**	**0.26**	**4**	**4.4**	**16**	**2.52**	**8**	**2.97**	**13**	**0.71**	**2**
Symphotrichum novi-belgii	Sym.nov	0	0	0.1	3	0.25	1	0	0	0.01	1	0	0
Symphotrichum lateriflorus	Sym.lat	0.08	1	0.04	1	0	0	0	0	0	0	0	0
Thelypteris palustris	The.pal	0.01	1	0.08	1	0	0	0	0	0	0	0	0
Triglochin maritima	**Tri.mar**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.13**	**1**
Typha angustifolia	Typ.Ang	7.75	11	0	0	2.38	6	0	0	2.22	3	0.83	1
Typha latifolia	Typ.lat	1.62	3	2.08	10	1.27	7	0.95	1	0.78	1	0	0
Typha sp.	Typ.sp	0	0	0	0	0	0	1.42	3	0	0	0	0

Table 10 Repeated-measures ANOVA comparing overall species richness at COG and COG-R between 2009 and 2013.

Source of variation	Df	Sum Sq	Mean Sq	F	P
Between plots					
Site	1	42.3	42.29	3.162	0.08244
Year	2	161.6	80.81	6.043	0.00487
Residuals	43	575.0	13.37		
Within plots					
Year	4	29.37	7.342	4.409	0.00202
Site x Year	4	25.85	6.462	3.881	0.00478
Residuals	177	294.73	1.665		

Table 11 Repeated-measures ANOVA comparing halophytic species richness at COG and COG-R between 2009 and 2013.

Source of variation	Df	Sum Sq	Mean Sq	F	P
Between plots					
Site	1	7.26	7.261	1.494	0.228
Year	2	22.07	11.033	2.270	0.116
Residuals	43	209.01	4.861		
Within plots					
Year	4	72.67	18.168	20.27	<0.0001
Site x Year	4	73.25	18.311	20.43	<0.0001
Residuals	177	158.63	0.896		

Table 12 Repeated-measures ANOVA comparing halophytic species abundance at COG and COG-R between 2009 and 2013.

Source of variation	Df	Sum Sq	Mean Sq	F	P
Between plots					
Site	1	16623	16623	45.189	<0.0001
Year	2	1050	525	1.427	0.251
Residuals	43	15817	368		
Within plots					
Year	4	11209	2802.2	28.33	< 0.0001
Site x Year	4	7030	1757.4	17.77	<0.0001
Residuals	177	17505	98.9		

Table 13 Repeated-measures ANOVA comparing unvegetated area at COG and COG-R between 2009 and 2013.

Source of variation	Df	Sum Sq	Mean Sq	F	P
Between plots					
Site	1	2656.7	2656.7	53.122	<0.0001
Year	2	249.3	124.7	2.493	0.0946
Residuals	43	2150.5	50.0		
Within					
Year	4	1856	464.1	16.64	<0.0001
Site x Year	4	1888	472.0	16.93	<0.0001
Residuals	177	4935	27.9		

Table 14 Repeated-measures ANOVA comparing unvegetated area at COG and COG-R between 2009 and 2013.

Source of variation	Df	Sum Sq	Mean Sq	F	P
Between plots					
Site	1	2656.7	2656.7	53.122	<0.0001
Year	2	249.3	124.7	2.493	0.0946
Residuals	43	2150.5	50.0		
Within					
Year	4	1856	464.1	16.64	<0.0001
Site x Year	4	1888	472.0	16.93	<0.0001
Residuals	177	4935	27.9		

Phragmites australis

Phragmites australis was first recorded at the restoration site in 2011 and was the subject of a SMU research project in 2012 (abstract provided in Appendix A). The species was not observed to have significantly increased in abundance or range. To determine native versus non-native genotypes, specimens were sent to the Phragmites Research Group at Laval University[10] for identification. Dr. Claude Lavoie concluded (99% confidence) that the *Phragmites* at COG is the native variety. Genetic markers would give 100% identification, but was not recommended unless absolutely necessary.

Although the presence of *Phragmites* within a tidal wetland is an issue, there is little cause for alarm at this stage from the native species. It should continue to be monitored to ensure that its

---

[10] http://phragmites.crad.ulaval.ca/en/

population does not significantly increase. It remains unknown how this plant came to be present at the restoration site, as it is not present on any of the other marshes along the Cogmagun River.

4.5 Nekton

Faunal response to restoration is dependent on access to the shallow intertidal marsh surface and intertidal and subtidal creeks (Able et al. 2008). The response of fish can be rapid following hydrological restoration (Roman et al. 2002; Able et al. 2008, Konisky et al. 2006). As in previous monitoring years, *Fundulus heteroclitus* (mummichog) were observed in large numbers in the fringe marsh pannes and borrow pit channel during low tide, as well as accessing the site on the leading edge of the rising tide. *Mummichog* was consistently the dominant species captured across all years (Table 15) and tends to be a resident salt marsh species typically found in both natural and impacted systems (Konisky et al. 2006).

Species richness continued to increase post-restoration (2013 n=7; 2012 n=6; 2011 n=2), as did relative abundance (2013 - 1364; 2012 – 889; 2011 – 84). The high number of species and abundance post-restoration compared to pre was due to the introduction and placement of the fyke net. However, when only considering the years in which the fyke net was employed, the observed trend of greater species richness and abundance three plus years following hydrological restoration has been seen in other tidal wetland restoration projects in the Gulf of Maine (Konisky et al. 2006. Bowron et al. 2013a). In 2012 the borrow pit near transect three consolidated enough to allow for the fyke net to be deployed within the channel (Figure 37). This represents an increase in fishing effort as positioning of the fyke within the borrow pit meant the net fished a larger portion of the high tide cycle (time) and at greater depth. This increased the likelihood of capturing species and individuals that tend to access the channels and creeks, but not the actual marsh surface during high tides. The fyke was deployed in the same location in 2013 and resulted in an even greater number of species and individuals captured. *Microgadus tomcod* (Atlantic tomcod), *Anguilla rostrata* (American eel), *Alosa pseudolarengus* (gaspereau) and Flounder *sp*. are examples of such species. Each of these species have been encountered at other restoration sites in the Bay of Fundy, primarily in or immediately adjacent to creeks, channels and areas of low elevation (Bowron et al. 2013a; Neatt et al. 2013).

Menidia menidia (Atlantic silverside), a migratory estuarine species, was the second most abundant species captured in both 2012 and 2013. This species has been known to access both the lower elevation portions of tidal wetlands and the vegetated marsh surface during high tide. Unlike the mummichog which thrives in disturbed sites and in marginal habitat conditions, the Atlantic silverside has been observed to be more prevalent on intact, natural tidal wetlands (reference marshes). The presence of species, such as the silverside, tomcod, gaspereau and the adult *Oncorhynchus mykiss (*Rainbow trout) that was caught in 2010, are indicative of higher order species accessing the restoration site during high tide, and could be taken as an indicator of recovering/improving tidal wetland habitat conditions.

Also of note was the presence of the invasive species *Carcinus maenas* (green crab) in fyke net samples (44 individuals in 2012; 28 in 2013). Of all the tidal wetland restoration project sites we have monitored since 2005, the tidal wetland restoration site on the St. Croix River has been the only site that green crab has not been captured during fish sampling activities.

Table 15 Fish species, and abundance, observed during monitoring at COG (Fr – fringe marsh; Re – restored marsh).

Species (common name)	2009 (pre)		2010 (post)		2011 (post)		2012 (post)		2013 (post)	
	Fr	Re	Fr	Re	Fr	Re	Fr	Re	Fr	Re
Menidia menidia (Atlantic silverside)						4		278		457
Microgadus tomcod (Atlantic tomcod)					1			10		10
Oncorhynchus mykiss (Rainbow trout)				1						
Fundulus heteroclitus (Mummichog)		235			40	80	121	568	73	861
Alosa pseudolarengus (alewife; gaspereau)								13		8
Osmerus mordax (Rainbow smelt)								3		
Anguilla rostrata (American eel)								17		26
Flounder (either winter or smooth)										1
Pungitius pungitius (Nine spine stickleback)										1
Species Richness (n=)	-	1	-	1	2	2	1	6	1	7
Abundance (total # of individuals)	-	235	-	1	41	84	121	889	73	1364

Figure 37 Fyke net set within the borrow pit at COG. Photograph by T. Bowron, 27 September 2013.

4.6 Benthic and Other Aquatic Invertebrates

Benthic Invertebrates

Ekman samples for 2013 had a low to moderate diversity of organisms (4 – 15 species); abundances which were high (2958 to 23,403 individuals/m^2); and biomasses which ranged from low to high (2.2 to 302.5 g/m2). Abundances were highest in the borrow pit channel at COG (L3) and in the three tidal channel sampling locations at COG-R (L5, L1, L1HM), all of which had exceptionally high abundances of the estuarine amphipod *Corophium volutator*, which contributed to a higher total abundance at the sites.

The invertebrate community at COG included a range of marine and estuarine species, with the fewest number of species, most of which were not found at any of the other sampling locations, recorded for the sample from the constructed tidal channel (WCS) at COG where the dyke and water control structure originally were located. This sample was the closest to the main river channel and the most turbulent (hydrology) (layout map).

Dominants at the COG site included *Hydrobia totteni* and *Nassarius trivitatus* (the latter at WCS); bivalves (*Macoma balthica* & *Mya arenaria* (WCS))*;* polychaetes (*Nereis diversicolor and N. succinea* (WCS)*, Heteromastus filiformis,* and *Eteone longa;* the amphipods *C. volutator* and *Gammarus mucronatus* (WCS); and freshwater insects, diptera (Chironomid midges & Ceratopogonidae) larvae, as well as aquatic beetles (Water Boatmen, Corixidae) and with oligochaete worms present in some of the samples. Ostracods (small meiofaunal organisms) were also relatively abundant in the salt panne (L5S3).

COG-R samples showed similar community composition and included a mix of marine estuarine species including the clam *Macoma balthica*; the worms *Capitella capitata* and *Nereis diversicolor;* the amphipod *Corophium volutator*; and freshwater insects including biting midge (Ceratopogonidae) larvae, and long-legged flies (Dolichopodidae larvae and pupae). Oligochaetes were also present in varying abundance in all samples, and nematodes (small, meiofaunal animals) were present in samples from the tidal channel and main river channel near transect one (layout figure).

Mean abundance and species richness for COG and COG-R between years was compared. Mean species richness for the reference site has remained relatively constant over time (Figure 38). Diversity (species richness) has increased at COG following restoration (Figure 38). Mean abundance (number of individuals) at both sites varied considerably over the course of the monitoring program, but with similar numbers four years following restoration (Figure 39). Abundances at the restoration increased following restoration (Figure 39).

Although both sites experienced variability in abundance, the variation between samples did not statistically separate the two sites. Benthic invertebrate samples from the two sites were not statistically different (f-critical = 4.17, p -value 0.09; df = 1). The difference between years was not statistically different (f-critical = 2.69; p = 0.20; df = 4), and the interaction between years and samples were not different (f-critical = 2.69; p = 0.06; df = 4). T-tests comparing the samples between the years also showed no separation between the two sites (Table 16).

Table 16 Comparison of benthic invertebrate samples between years between COG and COG-R.

	COG vs COG-R 2009	COG vs Cog-R 2010	COG vs Cog-R 2011	COG vs Cog-R 2012	COG vs Cog-R 2013
p-value	0.255	0.159	0.329	0.435	0.786
t-critical	3.182	3.182	3.182	3.182	2.57
Df	3	3	3	3	5

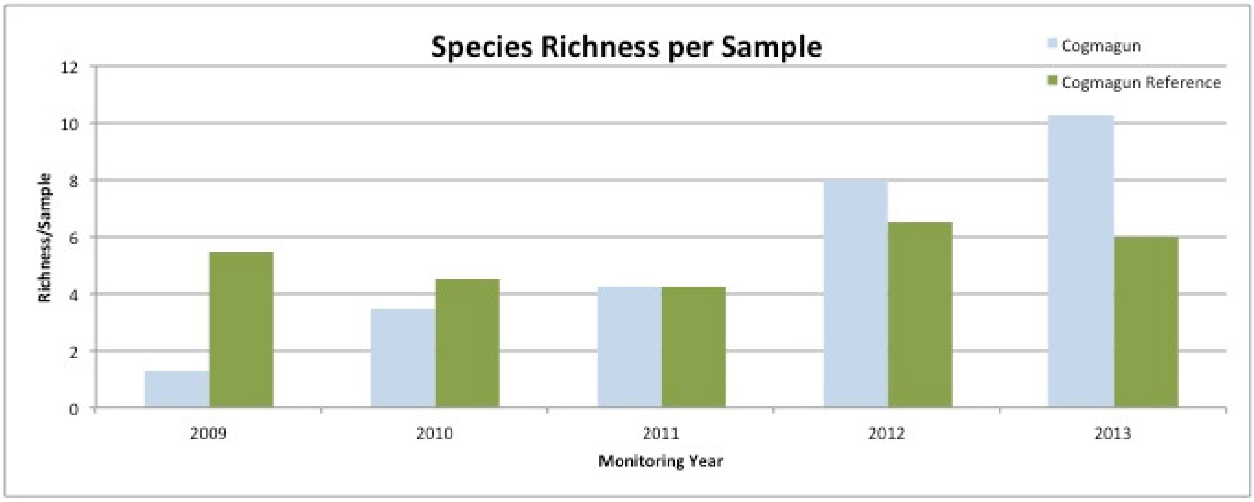


Figure 38 Benthic invertebrate species richness for all years at COG and COG-R.

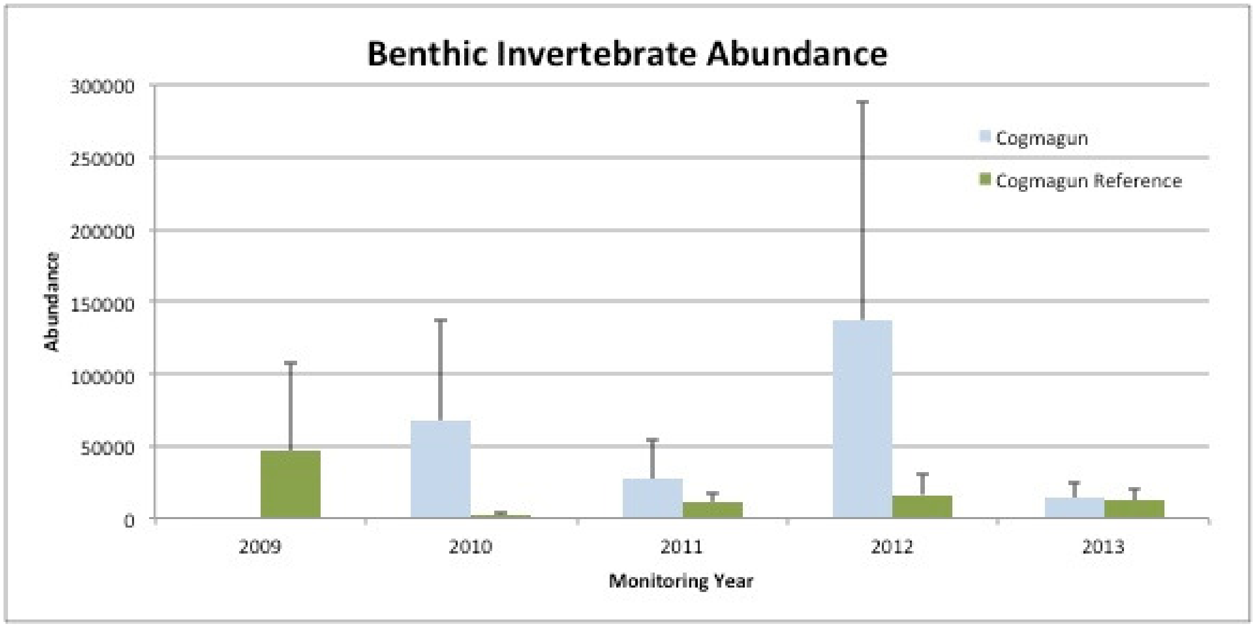


Figure 39 Benthic invertebrate species abundance for all years at COG and COG-R.

Aquatic Invertebrates

The 2013 aquatic invertebrate samples (IATs) from salt pannes located within the restoration site and on the fringe marsh (outside of the dyke) contained a range of marine/estuarine and freshwater animals with estuarine snails (*Hydrobia totteni*), copepods (both harpacticoid and cyclopoid), the estuarine amphipod (*Gammarus mucronatus*) and water boatmen (Corixidae) both common and abundant. Diversity and abundance overall was low to moderate, with 4 – 9 species and 122 – 928 individuals per sample, respectively (Figure 40). Planktonic copepods occurred at all sites and were numerically dominant in three of four fringe samples. The amphipod, *Gammarus mucronatus*, occurred in all but one sample and, as in 2012, was one of the dominant species in the panne near the upland edge of transect three (COG L3 Panne, 17/7/13). The small snail *Hydrobia totteni* was present in six of eight samples and was the dominant species for all samples within the restoration site. Water boatmen (Corixidae) juveniles and adults were present in five samples and were especially abundant in the transect three panne (COG L3).

A comparison of invertebrate samples across all years (2010-2013), found a significant difference in mean abundance of invertebrates within the restoration site and those from the fringe (ANOVA f-crit = 4.94, df = 1, p = 0.07) (Figure 41). Abundance levels within the restoration site were greater than that of the fringe. Mean richness was also determined to be higher within the restoration site (R=7.75 +/-3.6) than the fringe site samples (R=4.8 +/-1.1) (p = 0.002; df = 11, t = 2.20). Although there was a difference between sample locations and years, it was not a statistically significant one. Although not actually statistically significant, the higher productivity numbers for the restored area was not surprising given that mature sites typically are less diverse than younger or disturbed sites (Roxburgh et al. 2004). The proximity to the upland, freshwater influence and differences in fish population (fringe pannes contained large numbers of adult fish) are likely contributing factors to the differences between the two areas.

Although not an invertebrate, it is worth noting the presence of juvenile fish (*Fundulus heteroclitus*) in IAT samples. In 2012, individuals were found in all samples, but were particularly abundant in the July samples from the two pannes inside the restoration site. In 2013 individuals were again captured in the July sample from one of the pannes inside the restoration site, and a single individual in an August sample from the fringe. Although unintentional, the IATs do select for small fish, however, from the actual fish surveys it was found that fish populations were much greater in the fringe pannes and were dominated by adult individuals (section 4.5).

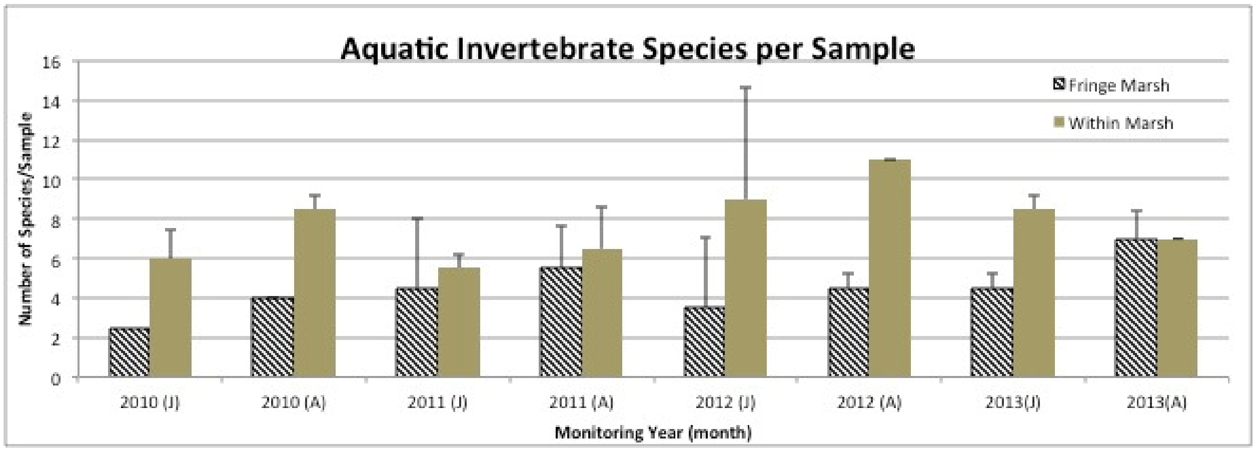


Figure 40 Aquatic invertebrate diversity for all years for samples from the fringe and within the restoration site.

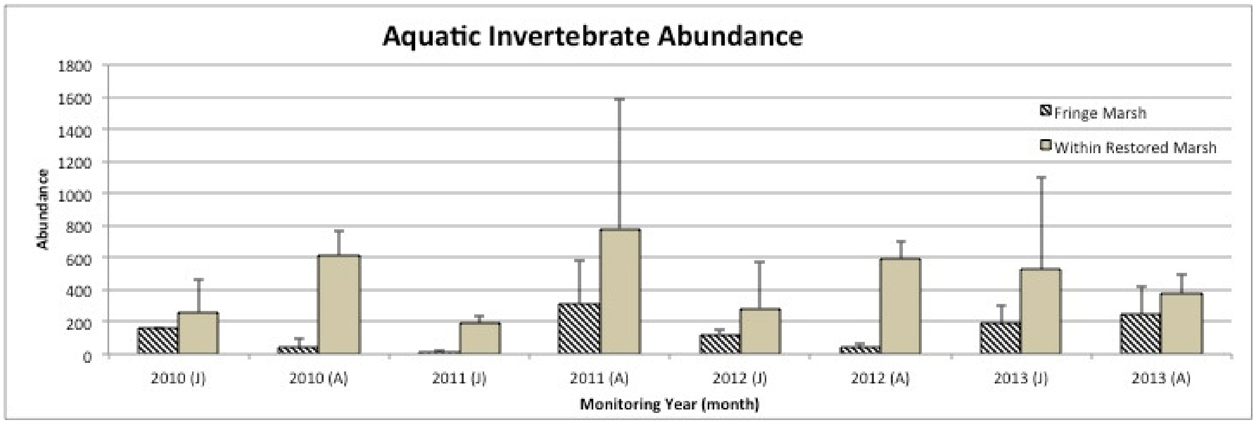


Figure 41 Aquatic invertebrate mean abundance for all years for samples from the fringe and within the restoration site.

4.7 Structured Winter Walk

The period of December 2013 through February 2014 was marked by extended periods of cold temperatures, numerous snowfall events and fast and sea ice formation. Both sites were ice and snow covered (average depth 20 cm). Stranded sea ice (blocks) was present on both sites, however, in greater numbers and size on COG. Sea ice blocks at COG bordered the borrow pit channel and dominated the upstream portion of the site (L4 & L5) (Figure 42). The snow and ice cover would serve to protect the sites from the potentially negative impacts of winter storm events (erosion), while the sea ice would potentially deposit sediment and plant material on the marsh if they remain until spring and melt in-situ (Figure 43).

No significant change (erosion, deposition) in habitat conditions was evident at either COG or COG-R. The main tidal channel at the restoration site continued to experience modification, in part due to winter ice conditions (Figure 44), but not to the extent that occurred during the first year following restoration (Section 4.1).

A selection of landscape photographs from the COG and COG-R winter walk are provided in Appendix B.

Figure 42 Sea ice on the marsh surface at COG. Photograph by T. Bowron 21 February 2014.

Figure 43 Sediment rich ice blocks in the borrow pit channel and marsh surface at COG. Photograph by T. Bowron 21 February 2014.

Figure 44 Fast and sea ice within the constructed tidal channel. Photograph by T. Bowron 21 February 2014.

5.0 Summary and Restored Area

The results of the fourth year of post-restoration (2013) monitoring of the Cogmagun River Restoration Project were presented in this report. The goal of the monitoring program was to provide a scientific record of habitat conditions at both the restoration and reference sites, to document the change in conditions in response to manipulation (in 2009) and to facilitate adaptive management if warranted.

Prior to restoration the site resembled, in form and function, a poorly drained freshwater impoundment that experienced periodic inundation by tidal waters. The decision to discontinue maintenance of the dyke and water control structure resulted in a gradual increase in the frequency of tidal intrusion during spring tide events. The result was a decline in freshwater habitat conditions leading to deteriorated water and soil conditions (increased salinity, high levels of decaying organic matter, anoxic conditions); reduced vegetation cover; and invasion by halophytic species.

As with most salt marsh restoration projects (e.g., Onaindia et al. 2001; Hinkle and Mitsch 2005; Wolters et al. 2005; van Proosdij et al. 2010), the variables driving the ecosystem response are hydrology and surface topography (elevation). The creation of the single dyke breach and excavated tidal channel in September 2009 resulted in the restoration of tidal influence to the former salt marsh system. The size of the breach and channel were found to be sufficient to allow a majority of the marsh surface to flood with tidal waters on spring high tide events. What remains to be determined, however, is how the site will develop based on the single opening rather than the multiple openings that the original design recommended, as was done at the Walton and St. Croix River restoration sites (van Proosdij et al. 2010; Bowron et al. 2014). Due to the difference in the hydrological regime of the site compared to that of the reference and related restoration sites, it has already begun to exhibit distinct differences in sediment and vegetation patterns. This does not necessarily represent a reduced value or functional importance of the site, or a failure of the restoration project.

The re-establishment of regular tidal flow to the site and its reconnection to the Cogmagun River system, has initiated the recovery of tidal wetland conditions at the site, as well as the availability and accessibility of the site for a range of estuarine species, most notably fish. Based on the 2012 (third year post) data, the intermediate estimate of restored (area inside the dyke including the borrow pit) and total marsh area (including dyke and fringe marsh) was approximately 4.16 ha and 7.68 ha, respectively (Section 5.0: Figure 45). The final determination of total restored area will be made following the fifth year of the post-restoration monitoring program.

The data collected for geospatial attributes, hydrology, soils and sediment, and vegetation four years following restoration continued to indicate a positive response to the restoration treatment. Aside from restored tidal flooding of the site, the most rapid change in conditions was recorded for soil conditions (accretion rates, below ground processes) and the vegetation community assemblage. Both indicators showed a strong correlation to one another and together represent a turning point in the systems recovery. Soil conditions four years post-restoration showed a significant shift towards conditions more comparable to the reference site and may be associated with the shift in vegetative community structure that was also recorded. The establishment, and expansion, of halophytic vegetation species following the initial rapid decline in freshwater and terrestrial species continued with both an increase in the number of species and abundance. By the fourth year following restoration, the vegetation community structure at the restoration site clearly overlapped with that of the reference site in low marsh areas, and contained a large amount of *S. patens* (representative of high marsh conditions). However differences continue to persist between the two sites in that a distinct high marsh area has not yet developed (*S. patens* areas heavily mixed with *S. alterniflora*), and the site still lacked the *J. gerardii* community present at the reference site.

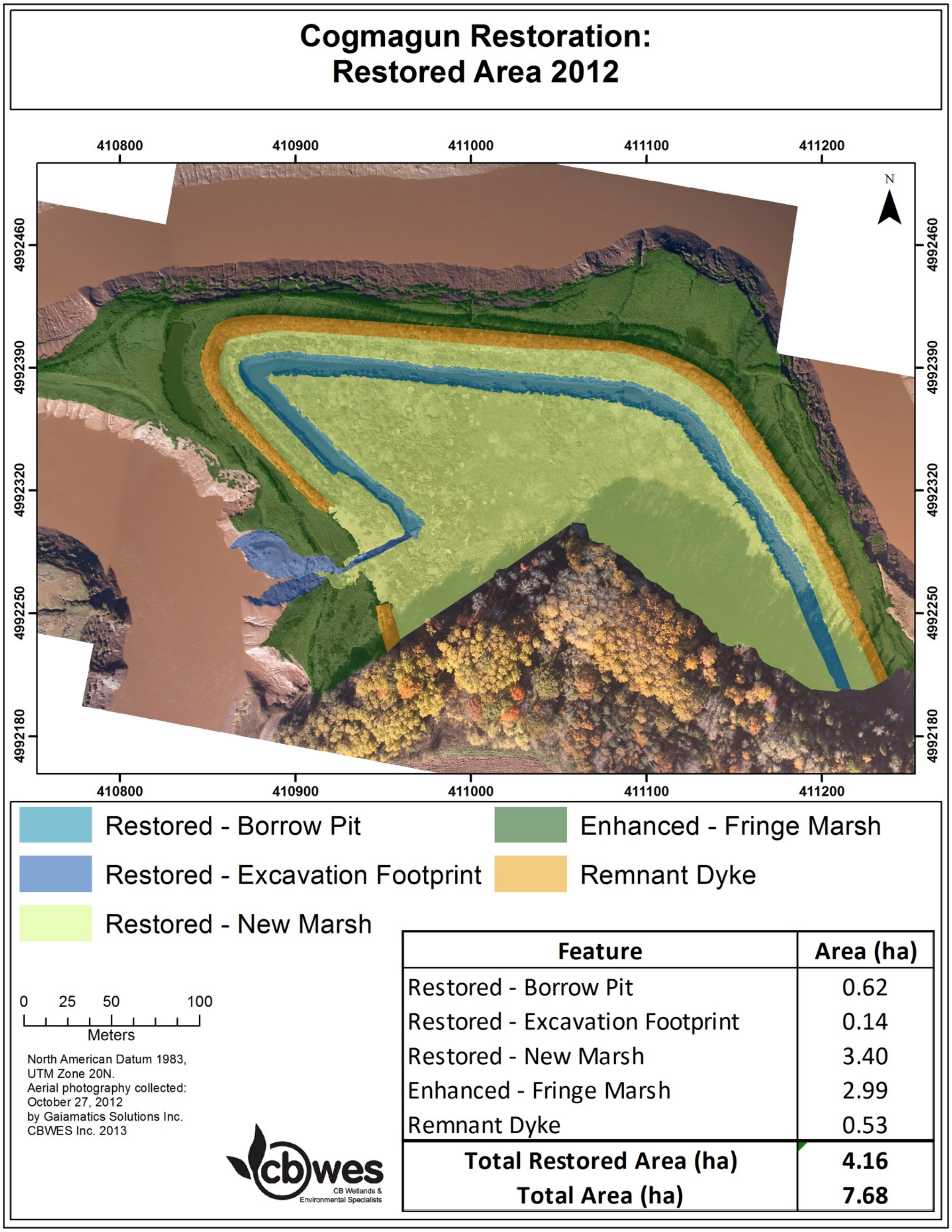
The results of the first four years of post-restoration monitoring of COG supports the growing body of evidence indicating that it is feasible to successfully restore tidally restricted wetland ecosystems. That the removal of even a limited portion of a barrier (single breach in a dyke) and with minimal manipulation of the geomorphology and biota in a macro-tidal environmental, can result in the natural recovery of abiotic and biotic conditions. Furthermore, the results of the first four years of post-restoration monitoring of COG suggest that the success of any tidal wetland restoration project depends, in part, on inclusion of a proper baseline study in order to understand the structure and functioning of the habitat, and careful project design to ensure the correct placement and size of breach(es), culverts or bridges. Ecologically, the appropriate morphological conditions (elevations, sediment supply, hydraulic connectivity) need to be present (naturally or through manipulation) in order for the re-establishment of a natural self-sustaining coastal wetland system to occur. Additionally the presence of local sources of colonists (halophytic plant species) decreases, or eliminates, the requirement for planting, allowing the site to respond to restoration naturally.

Restored Area

The results of the first four years of post-restoration monitoring, as discussed in this report, indicate that the system continues to respond in a positive and acceptable manner to the original intervention, and that no unanticipated or undesirable conditions have emerged during this period.

Based on the 2012 DEM, hydrology, and geo-referenced low-altitude aerial photography, the estimated area of recovering tidal wetland at COG was 4.16 ha (based on area flooded by largest recorded tide) (Figure 45). This area includes the borrow pit and dyke breach footprint. The dyke breach and excavated channel connected to the borrow pit does allow for the regular flooding of the site by tidal water. It is anticipated that the new/restored wetland boundary will eventually encompass an area of restored marsh between 3.78 ha (area inside of the dyke that would be flooded by 90% of tides) and 4.8 ha (maximum area inside the dyke). The total area of restored tidal wetland will be determined based on the elevation, hydrological and aerial photography data collected in 2014 as part of the fifth (final) year of post-restoration monitoring.

In conclusion, the COG restoration site continues to exhibit considerable evidence of returning to its former tidal wetland and salt marsh condition and that following four years of recovery, the primary tidal wetland parameters are responding in a positive manner.



Feature	Area (ha)
Restored - Borrow Pit	0.62
Restored - Excavation Footprint	0.14
Restored - New Marsh	3.40
Enhanced - Fringe Marsh	2.99
Remnant Dyke	0.53
Total Restored Area (ha)	**4.16**
Total Area (ha)	**7.68**

Figure 45 Estimated area of recovering tidal wetland habitat at COG based on 2012 hydrology, elevation and vegetation data.

6.0 Recommendations for Post-Restoration Monitoring

Pre- and post-restoration monitoring is an essential component of any habitat restoration project. Monitoring measures the effectiveness of the restoration effort; provides valuable information on the ecological condition of the restoration and reference sites; and the response of physical and biological elements, as well as the overall system, to the restoration treatment. In this way, a well-developed and implemented monitoring program can inform and support the management of a specific restoration project; identify the need for additional intervention (adaptive management); and help guide future management and restoration efforts throughout the region.

The monitoring program for COG, based on the GPAC Regional Monitoring Protocol, is a six-year program with one year pre- and five years post-restoration. Annual monitoring during the first three years following restoration are critical because it is during these initial years that the greatest and most rapid changes are likely to occur. Monitoring beyond the first three years following restoration allow a greater period of time for change to occur, for the documentation of the longer term, often more gradual, changes in response to restoration and for conditions (e.g. soil salinity, vegetation species composition) to begin to show indications of parity with reference conditions (Able et al. 2008; Burden et al. 2013; Garbutt and Wolters 2008; Mitsch et al. 2012; Neatt et al. 2013). It may also be necessary, as has been observed with the Walton River Restoration Project, that monitoring activities proceed beyond the post five year point, as these types of habitats require longer periods of time to mature.

The fifth (final) year of post-restoration monitoring is scheduled to begin in spring 2014. This will enable the continued documentation of the ecological changes and habitat/species responses to restoration and the determination of project success. The monitoring program, methods and sampling frequencies are outlined in Table 1 (Section 2.3). The results of the fourth year of post-restoration monitoring, as discussed in this report, indicate that the system has responded in a positive and acceptable manner to the original intervention. Based on this, the sampling activities outlined in the monitoring program for the fifth year of post-restoration monitoring are recommended for 2014.

Additionally:
COG specific recommendations:

- Although the *Phragmities australis* within the restoration site has been identified as native, it is recommended that it continue to be monitored to ensure that its population does not begin to significantly increase in abundance or area.

- Detailed mapping of the morphological tidal channel at COG was conducted on a monthly basis between June and November 2010 and 2011 as part of a graduate level research project (Appendix A). It is recommended that detailed mapping of the tidal channel be conducted during the final year of monitoring for comparison.

- Despite the differences in restoration design, the similarities between the COG and Walton River restoration sites should enable the comparison of their restoration trajectories. Such a comparison would significantly advance our knowledge of the response of these systems to restoration efforts and improve our ability to predict rates of habitat (parameter) recovery. A formal comparison of habitat conditions/response to restoration between COG and WAL should be conducted in the final year of the Cogmagun monitoring program (2014).

- The original restoration design for the COG site recommended five dyke breaches and two channels corresponding to the natural drainage basins and creeks within the site. However, in order to secure property owner approval for the project, the final design included only a single breach and tide channel. Although there is no formal component to the monitoring program addressing changes in dyke elevation or breach events, it is recommended that attention be paid to any evidence of erosion or topping of the dyke, changes in marsh surface elevation or vegetation community structure that may indicate the development of additional breaches or channel development or changes in habitat conditions on the dyke.

- It is recommended that redox potential continue to be monitored at both the restoration and reference sites.

Tidal Wetland Restoration Monitoring Recommendations:

- Depending on the amount and type of vegetation dominating a site prior to restoration, it may be advisable to remove the vegetation entirely or partially from the site to ensure fast recovery of vegetation and soil conditions post-restoration. The removal is especially important on sites where appropriate hydrology is not going to be restored to the site (i.e. Cogmagun). Without appropriate hydrology, a site may not be able to flush the plant debris off of the site and this can lead to mats forming on the marsh surface. The mat creates a barrier to new vegetation growth and can also impede oxygen getting into the soil to assist in the decomposition. Decrease of oxygen and inadequate flushing of the sediments can also lead to high levels of phytotoxins to accumulate (i.e. hydrogen sulfide) that negatively affect the growth of plants and thus the rate of recovery of the site (Koch and Mendelsshn 1989).

- Our paper (Porter et al. submitted) classifies tidal wetland habitats based on reference sites, primarily in the Bay of Fundy. With the increasing number of tidal wetland restoration projects and expansion into new regions (e.g. SW Nova Scotia, eastern shore), there is a need for a more comprehensive characterization of reference site vegetation across the province, coupled with quantification of key environmental gradients (salinity, tidal regime etc.). This would allow for a more robust vegetation classification that could be used to set restoration targets, and monitor restoration success using a multiple reference site approach. This would be similar to the Reference Condition Approach (RCA) used for monitoring benthic invertebrate community assemblages as part of the Cheverie Creek and Walton River Salt Marsh Restoration Projects (Armanini et al. 2012; Bowron et al. 2011a; Reynoldson 2005; Reynoldson et al. 1997; van Proosdij et al. 2010; Westhead 2005). If this were undertaken, a further innovation would be to develop a key to allow assignment of any quantitative tidal marsh vegetation sample to a community type.

Reference List

Able, K.W., T.M Grothues, S.M. Hagan, M.E. Kimball, D.M. Nemerson and G.L. Taghon. 2008. Long term response of fishes and other fauna to restoration of former salt hay farms: multiple measures of restoration success. Reviews in Fish Biology and Fisheries 18:65-97

Anisfeld, S.C. 2012. Biogeochemical responses to tidal restoration. In *Tidal Marsh Restoration* eds. Roman, C.T. and Burdick, D.M. pp. 39-58.

Armanini, D.G., Monk, W.A., Carter, L., Cote, D., and Baird, D.J. 2012. Towards generalized reference condition models for environmental assessment: a case study on rivers in Atlantic Canada. Environ Monit Assess. DOI 10.1007/s10661-012-3021-2

Atlantic Canada Coastal and Estuarine Science Society (ACCESS). 2008. ACCESS Conference 2008: Where the People Meet the Ocean: Nearshore Studies. Bedford Institute of Oceanography, May 14-15. Halifax, NS.

Atlantic Canada Coastal and Estuarine Science Society (ACCESS). 2012. ACCESS 2012 Annual General Meeting. May 10 – 13, 2012. Halifax, NS

Audubon Society. 1993. The Audubon Society Field Guide to North American Fishes, Whales & Dolphins. Random House of Canada Ltd., Toronto, ON.

Barnes, R.D. 1987. Invertebrate Zoology. (5th Ed.). CBS College Pub., Toronto.

Bay of Fundy Ecosystem Partnership (BoFEP). 2009. 8th BoFEP Science Workshop "Resource Development and its implication in the Bay of Fundy and Gulf of Maine". May 26-29. Acadia University, Wolfville, NS.

Bay of Fundy Ecosystem Partnership (BoFEP). 2011. Protecting the Watersheds and Estuaries of the Bay of Fundy: Issues, Science and Management. The 9th BoFEP Bay of Fundy Science Workshop. Saint John, New Brunswick. 27-30 September.

Bowron, T.M., N.C. Neatt, J.M. Graham, D. van Proosdij, J. Lundholm, and B. Lemieux. 2014a. Post-Construction Monitoring (Year 4) for the St. Croix River High Salt Marsh and Floodplain Wetland Restoration Project. Report prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 38. Halifax, NS.

Bowron, T.M., N.C. Neatt, J.M. Graham, D. van Proosdij, and J. Lundholm. 2014b. Pre-restoration monitoring of the Morris Island salt marsh restoration project. Report prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 40. Halifax, NS.

Bowron, T.M., N.C. Neatt, J.M. Graham, D. van Proosdij, J. Lundholm, and B. Lemieux. 2013a. Post-Restoration Monitoring (Year 7) of the Cheverie Creek Salt Marsh Restoration Site. Report prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 33. Halifax, NS.

Bowron, T.M., N.C. Neatt, J.M. Graham, D. van Proosdij, J. Lundholm and B. Lemieux. 2013b. Post-Restoration Monitoring (Year 5) of the Lawrencetown Lake Salt Marsh Restoration Project. Report Prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 35. Halifax, NS.

Bowron, T.M., N.C. Neatt, J.M. Graham, D. van Proosdij, J. Lundholm, and B. Lemieux. 2013c. Post-Restoration Monitoring (Year 3) of the Cogmagun River Salt Marsh Restoration Project. Report prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 37. Halifax, NS.

Bowron, T.M., N.C. Neatt, J.M. Graham, D. van Proosdij, J. Lundholm, and B. Lemieux. 2012. Post-Restoration Monitoring (Year Five) of the Smith Gut Salt Marsh Restoration Project. Report prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 29. Halifax, NS.

Bowron, T.M., N.C. Neatt, J.M. Graham, D. van Proosdij, J. Lundholm, and B. Lemieux. 2012b. Pre-Restoration Monitoring (Baseline) of the Three Fathom Harbour Tidal Wetland Restoration Project. Report prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 32. Halifax, NS.

Bowron, T.M.; Neatt, N.C.; van Proosdij, D.; Lundholm, J. and J. Graham. 2011a. Macro-tidal Salt Marsh Ecosystem Response to Culvert Expansion. *Restoration Ecology*. Vol. 19. No. 3.

Bowron, T.M., N.C. Neatt, J.M. Graham and B. Lemieux. 2011b. Tennycape River Tidal Wetland Restoration Project: Feasibility and Design. Report prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Report No. 7. Halifax, NS.

Bowron, T.M., N.C. Neatt, J.M. Graham, D. van Proosdij, and J. Lundholm. 2011c. Post-Restoration Monitoring (Year 1) of the Cogmagun River Salt Marsh Restoration Project. Report prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 27. Halifax, NS.

Bowron, T.M., N.C. Neatt, J.M. Graham, D. van Proosdij, and J. Lundholm. 2010. Pre-Restoration Monitoring and Construction Report for the Cogmagun River Salt Marsh Restoration Project. Report prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 21. Halifax, NS.

Burden, A.; Garbutt, R.A.; Evans, C.D.; Jones, D.L.; Cooper, D.M.. 2013 Carbon sequestration and biogeochemical cycling in a saltmarsh subject to coastal managed realignment. *Estuarine, Coastal & Shelf Science*, 120. 12-20. 10.1016/j.ecss.2013.01.014

Cahoon, D. R., J. C. Lynch, B. C. Perez, B. Segura, R. Holland, C. Stelly, G. Stephenson, and P. Hensel. 2002. A device for high precision measurement of wetland sediment elevation: II. The rod surface elevation table. Journal of Sedimentary Research. Vol. 72, No. 5. pp. 734-739.

Cahoon, D.R., J.C. Lynch, and R.M. Knaus. 1996. Improved cryogenic coring device for sampling wetland soils. Journal of Sedimentary Research 66: 1025 – 1027.

Canadian Land Reclamation Association (CLRA). 2007. Land Reclamation & Environmental Stewardship AGM/Conference. Lord Nelson Hotel, August 25 – 31. Halifax, NS.

Canadian Land Reclamation Association Atlantic Chapter (ARC). 2008. Atlantic Reclamation Conference 2008. Nova Scotia Museum of Industry, October 20 & 21. Stellarton, NS.

Canadian Land Reclamation Association Atlantic Chapter (ARC). 2009. “From the Ground Up” Atlantic Reclamation Conference 2009. November 3 – 5. Halifax, NS.

Canadian Land Reclamation Association Atlantic Chapter (ARC). 2010. Atlantic Reclamation Conference 2010. October 27-29. Halifax, NS.

Canadian Land Reclamation Association Atlantic Chapter (ARC). 2012. 37th National Conference. September 25-28, 2012. Sydney, Nova Scotia.

Canadian Land Reclamation Association Atlantic Chapter (ARC). 2013. Atlantic Reclamation Conference. 3 October. Sackville, New Brunswick.

Canadian Water Resources Association (CWRA). 2008. Maritime Water Resources Symposium: Watershed Health, Planning and Management. Nova Scotia Community College Waterfront Campus. August 21 – 23. Dartmouth, NS.

Catallo, W.J. 1999. Hourly and daily variation of sediment redox potential in tidal wetland sediments. U.S. Geological Survey, Biological Resources Division Biological Science Report USGS/BRD/BSR-1999-0001. 10pp.

CBCL Limited (CBCL). 2011. NSTIR – Antigonish Wetland Compensation Project: Wetland Compensation Proposal & Baseline Inventory Report (Addendum Report). Prepared for the Nova Scotia Department of Transportation and Infrastructure Renewal.

Coastal and Estuarine Research Federation (CERF). 2009. Estuaries and Coasts in a Changing World. International Conference, November 1 – 5. Portland, Oregon.

Coastal and Estuarine Research Federation (CERF). 2011. Societies, Estuaries & Coasts: Adapting to Change. 21st Biennial Conference of the Coastal and Estuarine Research Federation. Daytona Beach, Florida, USA, 6-10 November.

Coastal and Estuarine Research Federation (CERF). 2013. Toward Resilient Coasts and Estuaries, Science for Sustainable Solutions. 3-7 November. San Diego, California.

Colmer, T. D. and Flowers, T. J. 2008. Flooding tolerance in halophytes. New Phytologist, 179: 964-974 doi: 10.1111/j.1469-8137.2008.02483.x

Craft, C. B. 2001. Biology of wetland soils. In J. L. Richardson and M. J. Vepraskas, Eds. *Wetland soils – Genesis, Hydrology, Landscapes and Classification*, pp. 107–135. Lewis Publishers, Boca Raton, London, New York, Washington, DC.

de la Cruz, A.A., Hackney, C.T. and Bhardwaj, N. 1989. Temporal and spatial patterns of redox potential (Eh) in three tidal marsh communities. Wetlands, 9(2): 181-190.

Dionne, M., Short, F.T., and Burdick, D. M. 1999. Fish Utilization of Restored and Reference Salt-Marsh Habitat in the Gulf of Maine. American Fisheries Society Symposium. 22: 384-404.

Ecology Action Centre (EAC). 2007. Six Years in the Mud – Restoring Maritime Salt Marshes: Lessons Learned and Moving Forward. Bedford Institute of Oceanography, February 1 – 2. Halifax, NS.

Estuarine Research Federation (ERF). 2007. 18th International Conference of The Estuarine Research Federation - Estuarine Interactions: Biological-Physical Feedbacks and Adaptations. Providence Place, November 4 – 8. Providence, Rhode Island.

Fay, C.W., R.J. Neves, and G.B. Pardue. 1983. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Mid-Atlantic)—Atlantic silverside. US Fish and Wildlife Service, Divisions of Biological Services, FWS/OBS-82d/11.10. U.S. Army Corps of Engineers, TR EL-82-4. 15pp.

Friedrichs, C. T. and J. E. Perry. 2001. Tidal Saltmarsh Morphodynamics: A Synthesis. Journal of Coastal Research (SI 27), 7–37.

Garbutt, A. and Wolters, M. 2008. The natural regeneration of salt marsh on formerly reclaimed land. Applied Vegetation Science. Vol. 11. Pp. 335-344.

Gibson, M. 2003. Seashores of the Maritimes. Nimbus Publishing Limited, Halifax, Nova Scotia.

Gosner, K.L. 1971. Guide to identification of marine and estuarine invertebrates. J. Wiley and Sons, New York.

Graff, L. and J. Middleton. 2002. Wetlands and Fish: Catch the Link. National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service. Maryland.

Hinkle, R.L., and W.J. Mitsch. 2005. Salt marsh vegetation recovery at salt hay farm wetland restoration sites on Delaware Bay. Ecological Engineering 25:240-251.

Koch, M.S. and Mendelssohn, I.A. 1989. Sulfide as a soil phytotoxin: Differential responses in two marsh species. Journal of Ecology, 77(2): 565-578.

Konisky, R.A, D.M. Burdick, M. Dionne, and H.A. Neckles. 2006. A Regional Assessment of Salt Marsh Restoration and Monitoring in the Gulf of Maine. Restoration Ecology 14(4): 516-525.

Kwak, T. and J.B. Zedler. 1997. Food Web Analysis of Southern California Coastal Wetlands Using Multiple Stable Isotopes. Oecologia 110: 262-277.

Maccubbin, A.E. and Hodson, R.E. 1980. Mineralization of detrital lignocellulose by salt marsh sediment microflora. Appl. Environ. Microbiol., 40(4): 735-740.

Mitsch, W.J. and J.G. Gosselink. 1986. Wetlands. Van Nostrand Reinhold Co., New York, N.Y.

Mitsch, W.J., Gosselink, J.G. 2007. Wetlands. 4th ed. John Wiley & Sons. New Jersey, USA

Mitsch, W.J., L. Zhang, K.C. Stefanik, A.M. Nahlik, C.J. Anderson, B. Bernal, M. Hernandez, and K. Song. 2012. Creating Wetlands: Primary Succession, Water Quality Changes, and Self-Design over 15 Years BioScience. Vol. 62 No. 3. Pp. 237-250.

Neatt, N.C., T.M. Bowron, J.M. Graham, D. van Proosdij, and J. Lundholm. 2013. Post-Construction Monitoring (Year 7) of the Walton River Salt Marsh Restoration Project. Report Prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No. 34. Halifax, NS.

Neckles, H. and M. Dionne. (eds.) 2000. Regional Standards to Identify and Evaluate Tidal Wetland Restoration in the Gulf of Maine. A GPAC Workshop. Wells National Estuarine Research Reserve, Wells, ME.

Neckles, H.A., M. Dionne, D.M. Burdick, C.T. Roman, R. Buchsbaum, and E. Hutchins. 2002. A Monitoring Protocol to Assess Tidal Restoration of Salt Marshes on Local and Regional Scales. Restoration Ecology 10(3): 556 – 563.

Niering, W.A., and R.S. Warren. 1980. Vegetation patterns and processes in New England salt marshes. BioScience 30(5): 301-307.

Odum. W. 1988. Comparitive ecology of tidal freshwater and salt marshes. Annual Review of Ecology and Systematics 19 : 147- 176.

Onaindia, M., I. Albizu, and I. Amezaga. 2001. Effect of time on the natural regeneration of salt marsh. Applied Vegetation Science 4:247-256.

Packham, J.R. and A.J. Willis. 1997. Ecology of Dunes, Salt Marsh and Shingle. Chapman & Hall, New York, NY. pp. 87 – 118.

Paquette, C., Sundberg, K.L. Boumans, R.M.J., and Chmura, G.L. 2004. Changes in saltmarsh surface elevation due to variability in evapotranspiration and tidal flooding. Estuaries 27(1):82-89.

Porter, C., J. Lundholm, T. Bowron, B. Lemieux, D. van Proosdij, N. Neatt, and J. Graham. (submitted). Classification and environmental correlates of tidal wetland vegetation: implications for ecological restoration and monitoring. Estuaries and Coasts.

Portnoy, J.W., 1999. Salt marsh diking and restoration: biogeochemical implications of altered wetland hydrology. *Enviromental Management*. 24 (1), 111–120.

Reddy, K. R. and R. D. DeLaune. 2008. *Biogeochemistry of Wetlands*. Taylor & Francis Group, LLC: Boca Raton, Florida.

Redfield, A.C. 1972. Development of a New England Salt Marsh. Ecological Monographs 42(2): 201 – 237.

Restore America's Estuaries (RAE). 2010. 5th National Conference on Coastal and Estuarine Habitat Restoration - Preparing for Climate Change: Science, Practice and Policy. Galveston, Texas. November 13 – 17.

Restore America's Estuaries (RAE). 2013. Mid-Atlantic Living Shorelines Summit. 9-11 December. Cambridge, Maryland.

Reynoldson, T.B. 2005. Assessment of six saltmarsh sites in the Minas Basin, Bay of Fundy, using the reference condition approach. Field report prepared for CB Wetlands & Environment Specialists.

Reynoldson, T.B., D.R. Rosenberg, K.E. Day, R.H. Norris, V.H. Resh. 1997. The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. Journal of the North American Benthic Society. 16:833-852.

Roman, C.T., and D.M. Burdick. (eds.) 2012. Restoring Tidal Flow to Salt Marshes: A Synthesis of Science and Management. Island Press, Washington, DC.

Roman, C.T., K.B. Raposa, S.C. Adamowicz, M-J. James-Pirri, and J.G. Catena. 2002. Quantifying Vegetation and Nekton Response to Tidal Restoration of a New England Salt Marsh. Restoration Ecology 10(3): 450-460.

Roman, C.T., M-J. James-Pirri, and J.F. Heltshe. 2001. Monitoring Salt Marsh Vegetation: A Protocol for the Long-term Coastal Ecosystem Monitoring Program at Cape Cod National Seashore. http://www.nature.nps.gov/im/monitor/protocoldb.cfm

Roxburgh, S.H., Shea, K. and Wilson, J.B. 2004. The intermediate disturbance hypothesis: patch dynamics and mechanisms of species coexistence. Ecology. 85(2). Pp. 359-371.

Scott, W.B. and M.G. Scott. 1988. Atlantic Fishes of Canada. Can. Bull. Fish. Aquat. Sci. 219:731 p.

Tiner, R.W. 2005. Assessing cumulative loss of wetland functions in the Nanticoke River watershed using enhanced National Wetlands Inventory data. Wetlands 25(2): 405-419.

United States Geological Survey (USGS). 2005. Surface Elevation Table (SET). U.S. Department of the Interior, U.S. Geological Survey, Patuxent Wildlife Research Center. http://www.pwrc.usgs.gov/set/ Last Updated: 23 June 2010.

van Proosdij, D., J. Lundholm, N.C. Neatt, T.M. Bowron, and J.M. Graham. 2010. Ecological Re-engineering of a Freshwater Impoundment for Salt Marsh Restoration in a Hypertidal System. Ecological Engineering 36(10): 1314-1332.

van Proosdij, D., R.G.D. Davidson-Arnott, and J. Ollerhead. 2006. Controls on the spatial patterns of sediment deposition across a macro tidal salt marsh over single tidal cycles. Estuarine, Coastal and Shelf Science 69(1-2): 64-86.

Vepraskas, M.J. and Cox, J.L. 2002. Redox Potential Measurements. NC State University. Accessed July 4, 2013 from http://courses.soil.ncsu.edu/ssc570/redox.pdf

Westhead, M.C. 2005. Investigations of the Reference Condition Approach and intertidal ecology of the Minas Basin, Bay of Fundy, with reference to the impacts of intertidal harvesting. M.Sc. Thesis, Acadia University.

Wolters, M., A. Garbutt, and J.P. Bakker. 2005. Salt marsh restoration: Evaluating the success of de-embankments in North West Europe. Biological Conservation 123: 249-268.

Appendix A - CBWES Supported Student/Research Projects

In addition to the undergraduate and graduate research projects described below, CBWES routinely collaborates with universities, community colleges, local elementary schools, and field naturalist groups to use the restoration sites as outdoor classrooms, provide student volunteers with valuable field experience, and supports student projects by providing research project ideas and access to data, information, expertise and supervision. CBWES has been a recognized NSERC Industrial Partner and multiple NSERC grant recipient since 2009. Through programs such as these, we are able to provide valuable internship opportunities to highly qualified undergraduate and graduate co-operative education students.

Current Projects:

Masters of Applied Science

Department of Environmental Science
Saint Mary's University
Christa Skinner
2013-2015

Temporal and Spatial Patterns of Soil Chemistry and Primary Productivity in a Restored Salt Marsh

Salt marshes are highly productive ecosystems that provide a variety of ecosystem services. These ecosystems have been hindered by human alteration for hundreds of years and now provide opportunities for restoration. Salt marsh restoration of a previously tide-restricted site creates alteration to biogeochemical cycles within the sediment that have implications for vegetation recolonization and expected timeline for marsh development. Restoring appropriate hydrology to the previously tide-restricted site is extremely important to ensure proper drainage and minimize pooling of water on the marsh surface. Biogeochemistry is the study of the exchange of materials between living and nonliving components of the biosphere. Tidal water brings large amounts of salt and sulfate into coastal systems that can influence the productivity of vegetation. The sulfur cycle has been found to be the dominant oxidation-reduction cycle within coastal systems due to the large volume of sulfate available. Sulfate is used as an electron acceptor when conditions are favorable to assist in the oxidation of organic matter and subsequent production of sulfide. High levels of sulfide and salinity have been found to influence the productivity of vegetation within salt marshes and is influenced by the hydrologic network of the site. This study strives to determine the temporal and spatial pattern of sulfide, salinity and redox potential and related primary productivity and hydrology. It is hypothesized that the highest concentrations of sulfide and salinity will be found in areas of poor drainage during neap tides. An understanding of the temporal and spatial patterns of biogeochemical factors within a macro-tidal salt marsh will assist in the development and planning of future restoration projects.

Directed Study

Environmental Science
Saint Mary's University
Carly Wrathall

2014

Identification of the Challenges and Opportunities of Salt Marsh Creation as part of Shoreline Management Strategy

The purpose of this directed studies course is to develop a proof of concept for the use of a created (engineered) salt marsh to reduce erosion along the Lawrencetown Lake section of the Trans Canada Trail. Project will include a thorough review of the scientific and technical literature on the use of engineered salt marshes (living shorelines) to control erosion; the development of a rationale for the use of this technique as part of a shoreline management strategy for Lawrencetown Lake salt marsh restoration project and trail system; a site design proposal (techniques, materials, timelines, cost estimates, etc.); and the presentation of this "proof of concept" to provincial and federal regulatory agencies in order to identify regulatory requirements and obstacles. This course will involve literature review; consultation with restoration practitioners, material supply companies, and provincial & federal regulatory agencies; development of a project design using a geographical information system.

Completed Projects:

Directed Study

Environmental Science
Saint Mary's University
Carly Wrathall
2013

Vegetation Patterns and Primary Productivity of Natural and Restored Bay of Fundy Salt Marshes.

The purpose of this directed studies course is to examine above and below ground productivity of salt marshes in Nova Scotia's Bay of Fundy macro-tidal environment, and the potential for carbon sequestration. This will include examining the physiological, chemical, sedimentological, environmental and anthropogenic factors that could be potentially be influencing productivity. Along with field research and data collection, this course will also include a major research project including statistical analysis of field samples and geospatial analysis using a geographical information system.

Peer-review Publication (submitted)

Caitlin Porter, Jeremy Lundholm, Danika van Proosdij, Tony Bowron, Nancy Neatt, Jennie Graham, Ben Lemieux
Saint Mary's University & CBWES Inc.
2013

Classification and environmental correlates of tidal marsh vegetation in Nova Scotia, Canada.

Tidal wetland vegetation of eastern North America shows conspicuous zonation attributable to biotic interactions between plant species and differential tolerance of salinity and flooding. Tidal wetlands are a conspicuous feature of the coastline in Nova Scotia, and previous descriptions suggest that many of the plant communities are similar to those found in New England, which have been extensively studied. The goal of this study was to perform a numerical classification of tidal wetland vegetation in Nova Scotia, and to determine the relationships between variation in plant species composition and environmental factors. Sampling was conducted from vegetation communities of eight tidal wetland sites along a range of tidal magnitudes (micro- to macro-), designated as reference (intact) sites for paired tidal wetland restoration projects. Cluster analysis revealed seven distinct plant communities related to gradients of inundation duration and salinity. Plant community types were usually dominated by a single graminoid species. Communities detected were similar to those found in New Brunswick Bay of Fundy and Northumberland Strait wetlands, and to those farther south in Maine and New England, but three brackish communities were also identified within this study, of which the Juncus balticus/Festuca rubra and Spartina pectinata communities have not been previously described. Redundancy analysis shows continuous variation among these community types and highlights key environmental variables related to plant community patterns. These analyses provide a baseline for restoration work and identify environmental correlates of plant communities, allowing for better predictions of ecological restoration trajectories in tidal wetlands.

Masters of Applied Science

Department of Geography
Saint Mary's University
Ben Lemieux
NSERC Industrial Postgraduate Scholarship
2010-2012

The influence of drainage network and morphological features on the vegetation recovery pattern of a macro-tidal wetland restoration project.

Almost all life on earth depends on plants for their existence. Plants form the base of most food webs, but they also serve as habitat for many invertebrate, fish, birds and other species. Therefore, any attempt to restore a habitat should primarily aim at restoring vegetation structure. However, in Atlantic Canada there are few salt marsh restoration models or projects for managers to draw upon. This project aims to study the dynamics controlling vegetation community structure, so that a greater understanding of plant propagation patterns can be understood and modeled. The goal is to examine how surface morphology contributes to vegetative re-colonization. Low altitude photometric approaches, such as the use of a helium filled blimp, to document vegetation re-colonization patterns will be used. The contribution that surface features, such as the ponds created at the St. Croix River High Salt Marsh and Floodplain restoration site as well as internal creek structures of the Cogmagun River Salt Marsh restoration site, have on salt marsh propagation will be examined so that a vegetative propagation model can be created. Understanding how marsh morphology changes in time and the response of vegetation to those changes will serve to improve our understanding how habitat restoration is progressing and will further contribute to the continued progression of salt marsh restoration science.

Masters of Applied Science

Department of Geography
Saint Mary's University
Jennie M. Graham
NSERC Industrial Postgraduate Scholarship
2010-2012

Tidal Creek Hydraulic Geometry for Salt Marsh Restoration in the Upper Bay of Fundy

CBWES Inc. has been engaged in tidal wetland restoration and monitoring projects in Nova Scotia since 2005. In 2009, CBWES Inc. developed the project design and undertook restoration at two former tidal wetland systems in the Bay of Fundy; a 8 ha site on the Cogmagun River (COG) and a 19 ha site on the St. Croix River (SC). Both projects involved the breaching of an existing dyke in one or more locations and the excavation and recreation of historical tidal channel networks. The restoration designs put forward the problem of identifying appropriate locations for dyke breaches and excavated tidal channels in order to restore a more natural hydrological regime to the systems including the re-activation of relict creek systems while avoiding excessive erosion. During the restoration design phase of the SC project (Graham et al. 2008) a set of preliminary hydraulic equations were established for the Bay of Fundy region using the methods laid out by Williams et al. (2002). These equations were used to determine width and depth of excavated creeks and were further tested and refined through observations and application to a previously restored salt marsh (Walton River; van Proosdij et al. 2010).
The results of this preliminary work brought up several questions which would be addressed in this research project by:

- Ground-truthing reference marsh systems (i.e. creek widths and depths) to improve the quality of the data set.
- Improving the correlation of hydraulic geometry relationships through the refinement of the existing dataset and the addition of other marsh systems in the region, particularly large pristine marshes.
- Further analyzing the function of channelized versus free flow conditions on creek network development and maintenance and incorporating an analysis of flow velocity within channels using.
- Addressing the importance of additional variables such as location in the tidal frame and depth/width characteristics of the water body that the constructed creek network is entering.
- If possible, examining the impact of large (or multiple) storm events, freshwater runoff, and ice movement on newly constructed creeks which are particularly vulnerable to erosion.

The overall goal for this thesis project will be to produce a GIS-based model and protocol for future use in the design of marsh restoration projects in macrotidal environments.

References

Bowron, T.M., N.C. Neatt, J.M. Graham, J. Lundholm and D. van Proosdij. 2009. Post-Construction Monitoring (Year 3) of the Walton River Salt Marsh Restoration Project. Report Prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Publication No.12

Graham, J.M., D. van Proosdij, N.C. Neatt & T.M. Bowron. 2008. Restoration Design Proposal for the St. Croix River High Salt Marsh and Floodplain Wetland Restoration Project Report Prepared for Nova Scotia Department of Transportation and Infrastructure Renewal. CBWES Inc. Report No.4.

Williams, P.B; Orr, M.K. and N.J. Garrity. 2002. Hydraulic geometry: a geomorphic design tool for tidal marsh channel evolution in wetland restoration projects. Restoration Ecology 10(3): 577-590.

Undergraduate Honours

Environmental Science
Saint Mary's University
Christa Skinner
2012-2013

Analysis of the Relationship Between Vegetative Community Structure and Geodetic Elevation for Salt Marsh Restoration in Hypertidal Systems

Monitoring of salt marsh restoration sites is critical to the success of current and future projects but may also lead to costly projects. The distribution of vegetation across the marsh surface is highly influenced by soil salinity, duration of tidal flooding and competition between plant species. Focus has been placed on vegetation regeneration in post restoration activities and the role vegetation plays in sediment deposition within the Bay of Fundy. The influence that geodetic elevation has on the distribution of vegetation across the marsh has not been studied within restoration salt marshes in the Bay of Fundy. This study analyzes the relationship between vegetation community structure and geodetic elevation within restoration and reference macrotidal salt marshes in the Bay of Fundy.

This reseach was conducted within three newly restored salt marshes (and associated reference site(s)) in the upper Bay of Fundy currently being monitored as a compensation project. Dominant vegetation and geodetic elevation was determined at sampling stations arranged in transects running from the main tidal creek to the upland for each of the study sites in 2010. Five similar salt marsh species were found in both the reference and restoration sites. These include *Carex paleacea, Juncus gerardii, Spartina patens, Spartina pectinata*, and *Spartina alterniflora*. Of these five species, *Juncus gerardii, Spartina pectinata,* and *Spartina alterniflora* were found to have significantly different means and ranges of elevation within the restoration sites as compared to the reference sites. This is due to soil salinity, frequency and duration of inundation, and competition. All of these factors are influenced by geodetic elevation and time since beginning of restoration.

Undergraduate Honours

Environmental Science
Saint Mary's University
Alisha Glogowski
2012-2013

Information From the Wrack: Viability of Halophytic Vegetation within Tidal Wetland Wrack Mats

Nova Scotia's coastal wetlands are under various anthropogenic pressures that can cause destruction or degradation to these ecosystems. Many of these valuable systems have not been protected in the past and have been lost. An important stage in the overall knowledge of coastal wetlands is figuring out how these systems can recolonize without planting. Wrack is understudied in the Minas Basin, Bay of Fundy and determining if there is viable halophytic plant material within the wrack in this area could be a clue to understanding how these systems function. In order to gain a better understanding of the role of wrack mats, 18 samples were analyzed from 6 study areas (3 sample locations per study area). A characterization of the wrack mat was completed and seed material was determined viable. Target species *Spartina patens* and *Spartina alterniflora* did not germinate at all, while target species *Plantago maritima* and *Juncus gerardii* did germinate from seed and rhizome material found within the wrack. This information complements ongoing studies within the Minas Basin, Bay of Fundy, and increases the overall knowledge of relationships between wrack and colonization within coastal wetlands.

Undergraduate Honours

Environmental Science
Saint Mary's University
Alison Bijman
NSERC Industrial Undergraduate Student Research Awards
2011-2012

The Influence of Tidal Creek Networks on Wetland Vegetation Colonization in a Macro-tidal System

Six years of research and experience with restoring Bay of Fundy (Nova Scotia) salt marshes has shown that salt marsh plant species can colonize readily without planting, if the barriers to tidal flow are removed and suitable abiotic conditions (i.e. elevation) are present. Reactivated hybrid creek networks are potentially highly important to the restoration process, as they may represent the primary transport mechanism for seeds and vegetative material for re-colonization. It is unknown how important creeks are for the actual colonization of target species (*Spartina alterniflora; S. patens; Salicornia europaea; Suaeda maritima; Atriplex spp.*). Utilizing the Cogmagun River salt marsh restoration site (Hants County), which was restored in 2009, this research aims to examine if there is a relationship between proximity to creek and colonization rates of common salt marsh species, as well as if seedling coverage of *Suaeda maritima* in the previous year had a relationship with colonization rates of the following year. Colonization rates were positively related to proximity to the main tidal creek for four out of five target species (*S. alterniflora, S. europaea, S. maritima, and Atriplex spp),* and the presence of *S. maritima* in the previous year did increase the colonization rates of newly established communities. These results provide a fine-scale complement to existing and ongoing macro-scale studies and further clarify the relationships between abiotic properties of a recently restored tidal wetland and colonization.

Undergraduate Class Research Project

Department of Biology
Saint Mary's University
by Shawn Adderley, Alison Bijman, Lydia Ephraim, Kristen Gallant, Robert Hicks, Sebastien Letourneau-Paci, Lori Miller, Chantal Pye, Benjamin Royal-Preyra, Shayna Weeks

Edited by Dr. Jeremy Lundholm, Department of Biology/Environmental Science, Saint Mary's University

Phragmites australis at Cogmagun Restoration Site

A population of *Phragmites australis* was discovered at the salt marsh restoration site at Cogmagun Creek in summer 2011. As this species includes native and invasive subspecies, we undertook several analyses to determine a) the extent of colonization at the site; b) whether other nearby sites have also been colonized by *Phragmites*; c) environmental and vegetation characteristics of colonized areas. We found that *Phragmites* has colonized an area of 885 m^2 and has been present for at least two growing seasons (CBWES pers. comm 2011). However, there was no evidence of the species further upstream at the restoration and reference sites, nor on any adjacent marshes.

This population has morphological characteristics suggesting that it belongs to the native subspecies, but several of the measurements overlap with those from other populations from central Nova Scotia known to be non-native. Existing *Phragmites* stands contain a mixture of other species, mostly natives. The presence of many species coexisting within *Phragmites* stands provides more evidence to suggest that the plants at Cogmagun are representatives of the native strain of *Phragmites*, which is known to grow in less dense stands and to coexist with other native species. The elevation range of current populations suggests that much of the restoration site and upstream coastal marshes have similar elevation ranges to the area occupied by current populations, however, soil salinity values suggest that much of the site cannot be colonized by the native subspecies of *Phragmites*. We recommend that the most important next step in assessing the site would include a genetic analysis of the *Phragmites* populations to obtain a definitive genetic identity and to better estimate potential spread on the site.

Based on experiments conducted in other parts of North America, appropriate control measures for non-native *Phragmites* at Cogmagun could include mechanical and/or chemical control.

Undergraduate Honours

Department of Environmental Science
Dalhousie University
Rachel Deloughery
2010

Contribution of seed hydrochory to re-colonization of vegetation in macro-tidal Bay of Fundy salt marsh restoration projects

This project examines the role of seed dispersal *via* water, or hydrochory, in the re-colonization of restored salt marsh vegetation communities. The chosen study sites were macro-tidal coastal wetlands on the Bay of Fundy in Nova Scotia, Canada where CB Wetland and Environmental Specialists have undertaken restoration projects. Actively returning salt water marshes to more natural hydrological regimes through designed and monitored projects is a relatively new practice in Atlantic Canada, but one that is increasingly seen. Research exploring the patterns and mechanisms of initial stages of re-vegetation is limited. This study examined the degree to which hydrochory was occurring, and its contribution to re-colonization by target salt marsh species, on the study sites where tidal flooding was enhanced through construction of breaches in 2009. Using artificial turf traps and seed extraction of collected material, rates and richness of seed dispersal in flooding were assessed. Vegetation surveys measured richness and abundance of emergent vegetation on the sites in August 2010, approximately one-year following restorations. The turf trap and survey data were analysed for overlap of species, relative contributions to target species pool, and similarities in relative abundance at corresponding sample points. Results indicate that hydrochory was contributing to availability of propagules at both sites. Proportions of target species seeds in the turf traps were small or undetected, but this does not necessarily signify a minor effect on above-ground community. Rates and patterns of seed hydrochory, and its relationship to emergent vegetation, are site-specific. Differences in environmental histories, relative locations within the estuary, natural flooding regime dynamics, existing vegetation communities and salinity levels are all possible contributors to the discrepancies seen here.

Undergraduate Honours
Department of Biology
Saint Mary's University
Ben Lemieux
NSERC Industrial Undergraduate Student Research Awards
2009

The influence of soil seed bank on the colonization and restoration of a macro-tidal marsh

The aim of this project was to determine if hydrochory (seed transport by water) was a more likely source of early colonists than the soil seed banks of newly restored salt marshes. The project had two sample sites, St. Croix River and Cogmagun River salt marsh restoration sites. Soil seed banks in this study were defined as viable seeds based in the first 10 cm of soil on the surface of the restoration site. The project aimed to determine the relative contribution of the soil seed bank prior to breaching of the dyke and hydrochory post dyke breach to salt marsh vegetation re-colonization. The soil seed banks of the Cogmagun site and the St. Croix site were both sampled prior to the breaching of the dyke. The soil seed bank was sampled by placing quadrats at pre-determined sample points and sampling the soil using soil cores. This soil was then taken to a greenhouse, allowing any seeds present to grow, and then species and relative seed abundance was determined. The hydrochory traps for the St. Croix site were sampled by placing artificial turf traps at the same locations as the soil seed bank samples post breaching of the dyke. For the Cogmagun traps, due to time constraints with the thesis requirements, artificial turf traps were deployed prior to the dyke breach on an adjacent marsh. This would give a good indication of the potential for seed transport via tidal waters. The traps were deployed for the first spring tide period following the breaching of the dykes, during which time Hurricane Bill passed over Nova Scotia. The storm surge most likely washed away many of the seeds and sediment from the artificial turf traps. The traps were then collected, cold stabilized, and washed on a sieve to collect seeds and sediment which was then sent to the greenhouse for germination.

Preliminary results showed that the dominant plants found in the both the St. Croix artificial turf traps and hydrochory traps were mostly of the *Poacaea* genus. The samples from the Cogmagun soil seed bank were dominated by cattails (*Typha sp.)*. These findings point to the soil seed banks being reflective of the above ground vegetation. The hydrochory traps point to the localized seed transport as species from the St. Croix soil seed bank were dominated by grasses (*Poacaea*). Species for the Cogmagun site are still growing in the greenhouse as they need to flower so that their identification can be complete.

Undergraduate Honours

Department of Biology
Saint Mary's University
Emile Colpron
2008

The avian fauna of restored and natural salt marshes Minas Basin, Bay of Fundy, Nova Scotia

This study focused on the avian fauna of four salt marshes found in the upper Bay of Fundy, on the Minas Basin. The Bay of Fundy salt marshes are important coastal ecosystems for many avian species. They provide breeding and foraging habitat for numerous species of shorebirds, passerines and waterfowl. Many species which breed in the Arctic make use of tidal marshes as well, either for over-wintering, or as stop-over areas to rest and feed during annual migrations (Brawley et al. 1998).

Despite the importance of salt-water marshes for biodiversity conservation, the avian responses to alterations are poorly understood (Benoit and Askins 2002, Shriver et al. 2004, Hanson and Shriver 2006). The loss of salt marshes is especially a threat to salt-marsh specialist species such as the Nelson's sharp-tailed sparrow (*Ammodramus nelsoni*) and the willet (*Tringa semipalmata*). Both Nelson's sharp-tailed sparrow and the willet have been listed as a species at risk by COSEWIC (Committee On the Status of Endangered Wildlife In Canada) in the past due to population declines.

The objectives of this study were to (1) compare the species richness and abundance of avian fauna in restored and natural salt marshes, and (2) to determine the use of restored and natural salt marshes by avian salt marsh specialists.

References:

Benoit, L.K. and R.A. Askins. 2002. Relationship between habitat area and the distribution of tidal marsh birds. *The Wilson Bulletin*. 114(3):314-323.

Brawley, A.H., R.S. Warren and R.A. Askins. 1998. Bird use of restoration and reference marshes within Barn Island Wildlife Management Area, Stonington, Connecticut, USA. *Environmental Management*. 22(4):625-633

Hanson, A.R. and W.G. Shriver. 2006. Breeding birds of Northeast saltmarshes: habitat use and conservation. *Studies in Avian Biology*. 32:141-154.

Shriver, W.G., T.P. Hodgman, J.P. Gibbs and P.D. Vickery. 2004. Landscape context influences salt marsh bird diversity and area requirements in New England. *Biological Conservation*. 119:545-553.

Appendix B - Structured Winter Walk

STRUCTURED WALK PHOTOGRAPHS COG (select images):

Figure 1 Transect one.

Figure 2 Transect three.

Figure 3 Transect five.

Figure 4 Channel and dyke breach. Photograph from the fringe marsh near the mouth of the channel looking towards restoration site.

Figure 5 Borrow pit near transect five. Photograph from the corner of the borrow pit looking towards the upland.

STRUCTURED WALK PHOTOGRAPHS COG-R (select images):

Figure 6 Transect one.

Figure 7 Transect three.

Figure 8 Transect five.