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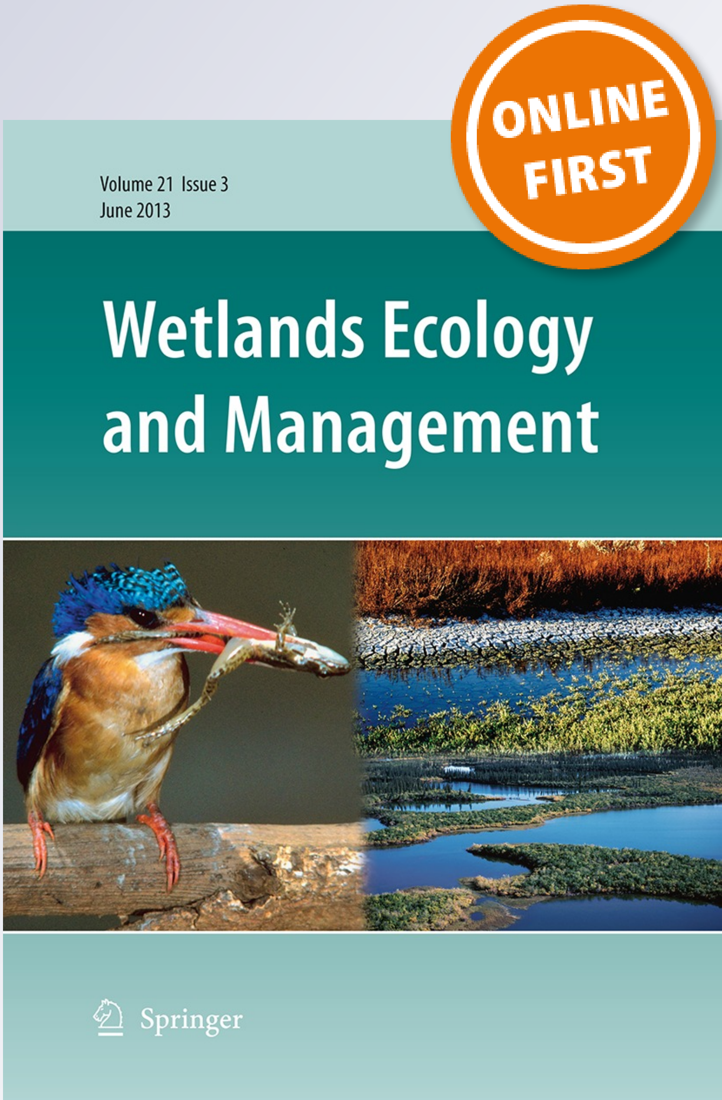

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ORIGINAL PAPER

Use of GIS and high resolution LiDAR in salt marsh restoration site suitability assessments in the upper Bay of Fundy, Canada

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Abstract Salt marshes exhibit striking vegetation zonation corresponding to spatially variable elevation gradients which dictate their frequency of inundation by the tides. The salt marshes in the upper Bay of Fundy, a dynamic hypertidal system, are of considerable interest due to increasing recognition of salt marsh ecosystem values and the extent of prior conversion of salt marshes to agricultural lands, much of which are no longer in use. To determine the suitability of two potential restoration sites at Beausejour Marsh in New Brunswick, Canada, geomatics technologies and techniques were used to assess vegetation and elevation patterns in an adjacent reference salt marsh and the proposed restoration sites. Light detection and ranging digital elevation models (DEMs) were created for the reference marsh and the restoration sites in both the spring (leaf-off) and late summer (leaf-on, maximum biomass) periods. Aerial photographs and Quickbird multispectral imagery were used to visually interpret vegetation zones on the reference marsh and were field validated using vegetation characteristics from quadrats referenced with differential GPS. Elevation limits of the salt marsh vegetation zones were extracted from the DEM of the reference marsh and applied to the DEM of the restoration sites to determine the percentage area of each site that would be immediately suitable for new salt marsh growth. Of the two restoration sites assessed, one had experienced significant subsidence since dyking; only about 40 % of the site area was determined to be of sufficient elevation for immediate vegetation colonization. The second site, while more than 88 % suitable, would require the installation of a large dyke on the landward side of the restoration site to prevent flooding of adjacent lands. This study provides essential high resolution elevation and vegetation zonation data for use in restoration site assessments, and highlights the usefulness of applied geomatics in the salt marsh restoration planning process.

Keywords Salt marsh · Wetland · *Spartina* · Restoration · LiDAR · Geomatics · Vegetation zonation · Bay of Fundy

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Introduction

Salt marshes commonly occur along the margins of temperate estuaries and are dominated by halophytic vegetation species that vary in their tolerance to tidal inundation (Allen 2000). They are among the most productive ecosystems in the world (Gordon et al. 1985; Simas et al. 2000) and serve important functions

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in flood mitigation, moderation of climate change (via carbon sequestration) (Connor et al. 2001; Mcleod et al. 2011), filtration of pollutants and sediments (Niering and Warren 1980), and provision of habitat for wildlife (Gordon et al. 1985).

Salt marsh zonation

Three inter-related physical factors that primarily determine salt marsh vegetation distribution are:

(1) land elevation (Bertness and Ellison 1987; Wolters et al. 2005; Mudd et al. 2004; Rosso et al. 2005),
(2) inundation frequency and duration (Eleuterius and Eleuterius 1979; Bertness 1991; Allen 2000; Bockelmann et al. 2002; Bernhardt and Marcus 2003; Genc et al. 2004; Olsen et al. 2005; Friess et al. 2012), and
(3) magnitude of tidal inundation (Gordon et al. 1985; Ollerhead et al. 2003).

The same type and even similar species of vegetation are found within salt marsh communities worldwide (Adams 1963). Not surprisingly, physical factors influencing marsh development and spatial patterns in vegetation are common among sites. Frequency and duration of tidal inundation decreases with increasing elevation of land and typically results in the vertical zonation of salt marsh vegetation (Bertness 1991; Silvestri et al. 2005; Blott and Pye 2004). The ability to predict where salt marsh vegetation species are most likely to colonize and grow thus depends on understanding the relationship between the elevation of the landscape, local tidal characteristics, and the growth limitations and interactions of salt marsh plant species (Weinstein et al. 2001; Weinstein and Weishar 2002; Blott and Pye 2004; Spencer and Harvey 2012).

Extensive salt marsh growth and striking zonation in the distribution of vegetation species can be found in coastal environments with large tidal ranges (>4 m) (Ganong 1903; Gordon et al. 1985; Davidson-Arnott et al. 2002; Olsen et al. 2005; Millard et al. 2007; van Proosdij et al. 2010). In the Bay of Fundy and throughout the eastern US and Canada, the low marsh vegetation zone is dominated by *Spartina alterniflora* and grades into high marsh, which is dominated by *Spartina patens* (Gordon et al. 1985; Bertness 1991; Davidson-Arnott et al. 2002; Byers and Chmura 2008). Flats of sand and mud which lie below the lowest limit of the marsh are inundated too often and for too long for salt marsh vegetation to establish. The high marsh zone experiences much less inundation than the low marsh zone and is more species diverse (Adams 1963; Eleuterius and Eleuterius 1979; Niering and Warren 1980; Bertness 1991; Byers and Chmura 2008). In an unrestricted tidal environment, a salt marsh would gradually grade into upland species, however, in many locations, including the upper Bay of Fundy, salt marshes have been dyked, resulting in steep banks (dykes) that restrict the landward growth of salt marshes.

Salt marsh reclamation and restoration

In the Bay of Fundy, The United Kingdom, eastern US, France, Denmark, The Netherlands and other regions, salt marshes have been severely reduced in area due to centuries of farming and other coastal development activities (Gordon et al. 1985; Weinstein et al. 2001; Konisky and Burdick 2004; Friess et al. 2012). Most commonly, salt marshes have been embanked or dyked in order to restrict tidal flooding, usually in an effort to increase the amount of agricultural land (Davidson-Arnott et al. 2002). In the Bay of Fundy such activities commenced in the seventeenth century when the Acadian people began dyking and draining coastal marshes (Desplanque and Mossman 2004). It is estimated that more than 300 km of the original high marsh in the Bay of Fundy were dyked (approximately 85 % of total) with the most extensive alterations occurring in the Cumberland Basin of the upper Bay of Fundy (approximately 150 km) (Gordon et al. 1985; Desplanque and Mossman 2004). Many of the farms developed on reclaimed lands in the Cumberland Basin currently lie fallow or are underutilized. During the last decade there has been growing interest in restoring salt marsh ecosystems for both conservation purposes (Ducks Unlimited 2004; Hansen and Torrent-Ellis 2004) and for mitigating past or future conversion of salt marsh to coastal developments (Ducks Unlimited 2010).

While efforts to restore salt marsh habitat have increased worldwide (Blott and Pye 2004; Ducks Unlimited 2010; van Proosdij et al. 2010; Bowron et al. 2011), there have been few or limited attempts to predict or assess the ability of a restoration site to be restored (Weinstein et al. 2001; Konisky and Burdick 2004; Blott and Pye 2004). Many restorations that have occurred have been due to accidental breach, while some tidal restrictions (e.g. dykes, culverts) have been purposefully removed, with the assumption that the structure and function of a natural salt marsh would resume (Neckles et al. 2002; van Proosdij et al. 2010; Spencer and Harvey 2012). In many cases, restoration outcomes have not been modeled before a deliberate breach, in large part due to lack of high resolution elevation data for sites of interest.

A major challenge faced in many salt marsh restoration initiatives is subsidence of the reclaimed land, the elevation of which can become too low for salt marsh vegetation to establish (Weinstein et al. 2001; Weinstein and Weishar 2002; Blott and Pye 2004; Byers and Chmura 2008; Spencer and Harvey 2012). Prevention of tidal inundation through the use of dykes severely reduces halophytic biomass production (below and above ground), and when decaying biomass compacts, the elevation of the land subsides (Byers and Chmura 2008; Spencer and Harvey 2012). Elevation is also affected by dykes and other tidal barriers which effectively reduce sediment accretion behind the physical structures (Weinstein and Weishar 2002).

For the most part, the high suspended sediment concentration of the upper Bay of Fundy has lead to sedimentation rates that have allowed marshes to keep pace with relative sea level rise in the region (Gordon et al. 1985; Allen 2000; Chmura et al. 2001; van Proosdij et al. 2006). Salt marshes grow upwards, while nearby areas no longer subject to inundation (dyked) remain at lower elevation. The elevation differences are several decimeters to meters, and vary from site to site in the Bay of Fundy (Desplanque and Mossman 2004). Weinstein and Weishar (2002) found that the longer a site has been restricted from tidal inundation, the greater the difference between the elevation of the restoration site and the seaward marshes. In the upper Bay of Fundy, most of the dyked marshes are now well below the highest level reached by the tides (Gordon et al. 1985; Desplanque and Mossman 2004). When a site has subsided beyond the lower limit of salt marsh plant colonization, pools of standing water form; such areas cannot support salt marsh vegetation when tidal inundation is restored. However, frequent flooding by the tides or by engineered tidings could, over time, lead to increased sedimentation and eventually to sufficient elevation to support the colonization of salt marsh vegetation.

Tools for assessing salt marsh landscapes and restoration site suitability

Site restoration studies should consider the combined use of GIS, remote sensing, GPS and Light detection and ranging (LiDAR) technologies and techniques (Blott and Pye 2004; Collin et al. 2010; Friess et al. 2012). Remotely sensed data can provide a synoptic view of large areas and locations, including areas such as salt marshes that are difficult to access (Connor et al. 2001; Blott and Pye 2004; Montané and Torres 2006; Sadro et al. 2007; Millard et al. 2009; Millette et al. 2010; Chassereau et al. 2011; Moeslund et al. 2011; Timm and McGarigal 2012). They also allow spatio-temporal change analysis as surveys can be repeated over time (Millette et al. 2010).

Before salt marsh restoration activity is undertaken, a thorough analysis of the feasibility of site restoration should be performed. Several authors note the importance of formally planning a restoration and ensuring that the site chosen will allow the return of vital processes and species within an acceptable time frame (Weinstein et al. 2001; Weinstein and Weishar 2002; Neckles et al. 2002; Spencer and Harvey 2012). The use of traditional surveying equipment or GPS in restoration planning for large sites is costly and time consuming (Sallenger et al. 2003). In addition, many inter-tidal areas are difficult to survey due to poor accessibility and daily inundation. Air photos may be sufficient for assessing salt marsh vegetation if available at sufficiently high resolution (Zharikov et al. 2005; Friess et al. 2012), however, in areas where multiple air photos are needed, mosaicking and colour balancing of air photos is required and can be difficult. Satellite imagery provides a view of a larger area, but often at lower resolution (Millette et al. 2010), and can also provide additional spectral information. For example, different wavelengths can be combined to allow for visualization not available using aerial photographs which are usually collected in the visual spectrum only (Smith et al. 1998; King 2012). Seasonal changes in vegetation are also important to consider with any imagery type (Smith et al. 1998).

Fine-scale elevation features of landforms can be assessed using airborne LiDAR technology due to the high resolution and high accuracy of the digital elevation models (DEMs) produced. LiDAR has been used for a range of wetland research activities, including measuring change in salt marsh distribution

(Rosso et al. 2005), geomorphology (Morris et al. 2005; Millette et al. 2010), plant species distribution (Sadro et al. 2007) and assessing vegetation height in wetlands (Genc et al. 2004; Hopkinson et al. 2006; Millard et al. 2008). Collin et al. (2010) showed that bathymetric LiDAR can discriminate between high and low marshes although they did not use LiDAR to quantify the elevation ranges of these zones. Recently, several authors have assessed the accuracy of LiDAR DEMs within salt marshes, with results emphasizing the importance of processing techniques, and the limits of LiDAR in penetrating the vegetation canopy in dense grasses (Montané and Torres 2006; Millard et al. 2008; Chassereau et al. 2011; Schmid et al. 2011). The study presented here demonstrates how LiDAR coupled with other geomatics tools can be used to accurately assess elevation in salt marshes and adjacent restoration sites for the purposes of salt marsh restoration planning.

Study aim

The primary aim was to predict the suitability of proposed restoration sites in the upper Bay of Fundy for re-colonization of salt marsh vegetation. The approach involved applying a range of geomatics tools and techniques, coupled with field validation of data, to quantitatively assess fine-scale landscape features of a reference salt marsh and two potential restoration sites (currently unused agricultural land).

Study area

The study was conducted in the upper reaches of the Bay of Fundy, on the east coast of Canada (Fig. 1). The Bay of Fundy is a large, hypertidal system with semi-diurnal tides and the highest recorded tidal range in the world (>16 m in its upper reaches).

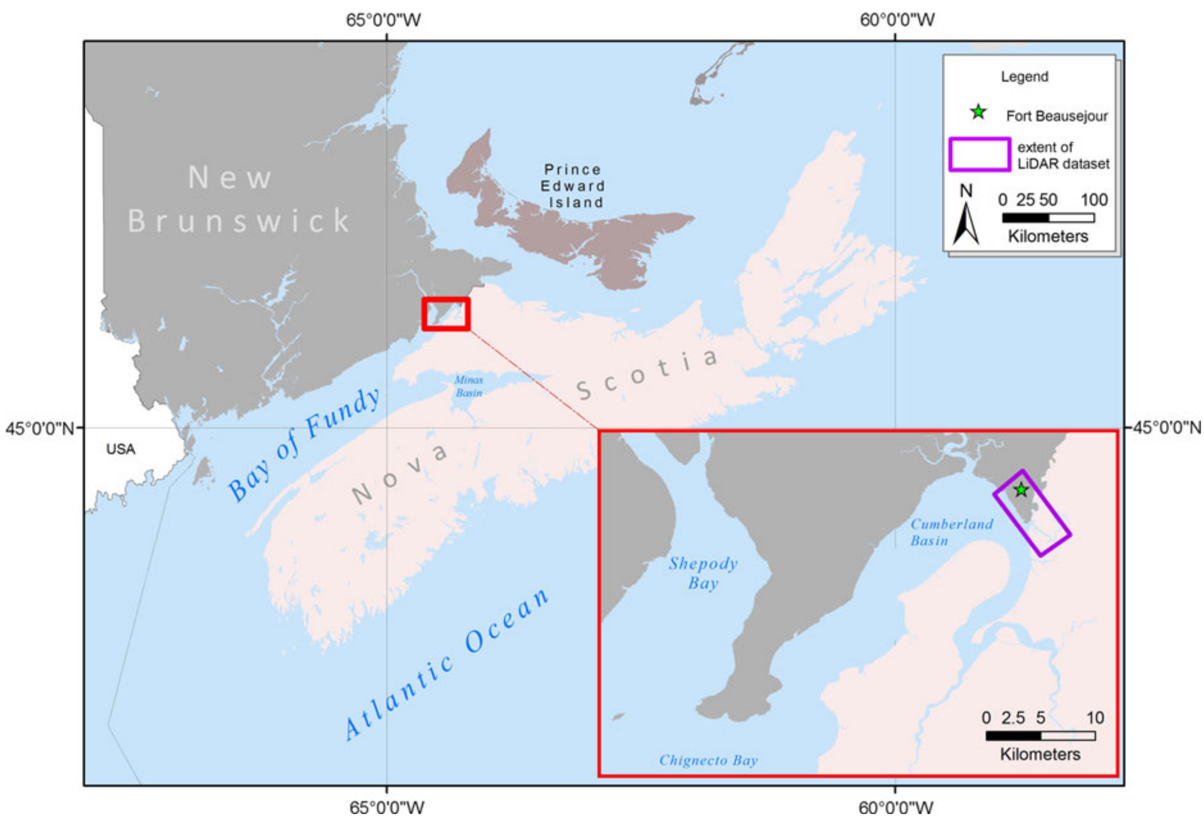

Fig. 1 Map of the Maritime Provinces and Bay of Fundy, with the *inset* showing Chignecto-, Shepody Bays, Cumberland Basin, Fort Beausejour (*star*) and the study area (*rectangle* in the *lower right inset box*)

The study area is at the head of the Cumberland Basin near the Missaguash River which forms the political border between the provinces of Nova Scotia and New Brunswick. The two proposed restoration sites border the reference salt marsh and are located within two kilometers of each other (Fig. 2).

Site description

Restoration sites under consideration were chosen based on their location, ease of access and seaward dyke condition (i.e. dyke requires significant upkeep) (Fig. 2). Both sites are part of a much larger expanse of salt marsh that was converted to farmland in the seventeenth century (Gordon et al. 1985). Restoration Site 1 sits directly west of the reference marsh and, at the time of the study, was completely enclosed by dykes with the seaward dykes being the oldest. The New Brunswick Department of Agriculture installed the landward dykes in 2006 when the seaward dykes began to erode and fail. As upkeep on these outer dykes has ceased, this site will over time become inundated by tidal waters, and may experience significant erosion given its exposure to extreme tidal conditions (range up to 16 m) and weather related events (e.g. storms).

Restoration Site 2 is approximately 200 m landward of the reference marsh and borders the Missaguash River. This site was selected for consideration largely due to bank erosion and the difficulty and expense in maintaining the riverside dykes. An engineered breach in the dyke would allow tidal inundation to resume to the interior area but would

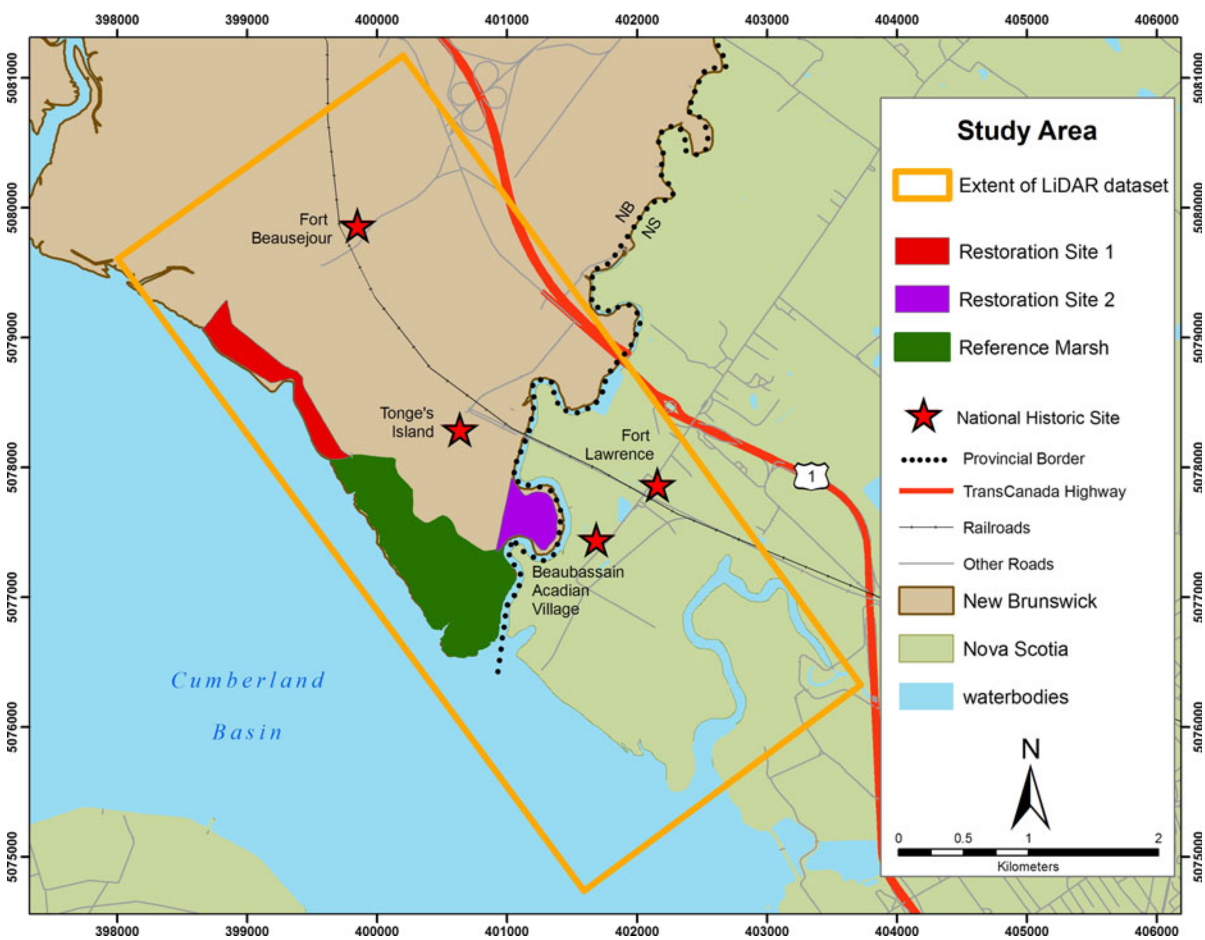

Fig. 2 Study area showing the two restoration sites, the reference marsh and the extent of the LiDAR dataset. *Coordinates* referenced to North American Datum of 1983, UTM Zone 20N

require regular reinforcement of existing dykes. As it is located slightly inland (Fig. 2), some relief from exposure is provided; however, if restored to salt marsh, a landward dyke would be required to protect adjacent farmland from seawater intrusion.

The reference marsh (approximately 0.9 km^2) is considered a stable salt marsh (Ollerhead et al. 2003) and experiences similar geophysical conditions (e.g. climate, soil, exposure) to the two restoration sites (Fig. 2). Historically, there was some dyking and ditching on the reference marsh, but during the last century, tidal flow has been un-restricted. A few remnant dykes (less than 0.5 m above the marsh platform) and ditches (less than 0.5 m deep) remain on the marsh platform. Open water areas on the marsh platform represent less than 10 % of the reference marsh surface.

Data acquisition and processing

LiDAR

An Optech ALTM 3100 LiDAR system was used to collect data in May 2006, prior to vegetation emergence on the reference salt marsh, and in August 2006, at maximum vegetation biomass. Technical specifications of the acquisition are found in Table 1. For both of the datasets, some flight line matching was required prior to processing and was carried out using TerraSolid software. A classification algorithm (TerraScan) was run on the raw LiDAR point cloud data and the classified point data (Table 2). Manual classification (fencing) was required in some areas due to the misclassification of the tops of dykes. These should have been included in the ground class but were often classified as either non-ground or building classes, which if left uncorrected would result in significant errors in the DEM. The resulting data were interpolated using a triangulated irregular network algorithm to produce a raster grid with 1 m *XY* resolution.

Table 1 LiDAR scan parameters

Survey dates	12 May 2006, 16 August 2006
Sensor	Optech ALTM 3100
Aircraft	Cessna Skymaster C-GPZL
GPS base station	NSHPN #213249
Cross track point spacing (m)	0.5
Long track point spacing (m)	0.5
Laser frequency (kHz)	50
Scanner rate (Hz)	28
Scan angle (°)	18
Beam divergence	Narrow
Flight line overlap (%)	50
Flying speed (knots)	110
Flying height (m above ground level)	1500

Table 2 LiDAR dataset ground/non-ground separation filtering parameters (used in TerraScan)

Parameters	Accuracy result (mm)
Standard deviation of *X* quality	6.7
Standard deviation of *Y* quality	4.6
Standard deviation of *Z* quality	14.8
Position quality	8.3
Height quality	14.8
Position + height quality	17.1

Acquiring two datasets, one during early spring (leaf-off conditions) and one during full biomass, allowed production of accurate bare earth and canopy height models (CHMs, Hopkinson et al. 2006; Millard et al. 2008). While some vegetation did remain on the high marsh, most of the creeks and low marsh had little to no vegetation due to ice shear over the winter. In addition to the traditional LiDAR accuracy assessment on hard surfaces, a separate accuracy assessment was carried out using GPS elevations on the reference marsh surface. In areas of dense vegetation, LiDAR does not always fully penetrate the canopy to the ground (Hopkinson et al. 2006; Sadro et al. 2007; Millette et al. 2010; Chassereau et al. 2011), however, careful consideration was taken to ensure ground classification accuracy on the marsh and in the channels. Using the methods described in Millard et al. (2008), the relationship between differential GPS (DGPS) and field measured vegetation height was used to produce a more accurate DEM and CHM than can be obtained using tradition DEM/DSM separation. This resulted in acceptable accuracy both on hard surfaces and within vegetated areas on the marsh. The bare earth DEM was produced through the interpolation of points that were classified as "ground" whereas the digital surface model was produced through the interpolation of points that were classified as "ground", "vegetation", "buildings" or any permanent feature above the ground.

Normal procedure when comparing data referenced to different vertical datum requires all datasets to be converted to the same datum. When collected, LiDAR data elevation values are referenced to an ellipsoid (GRS80). Tidal height values are often referenced to Chart Datum (CD). CD-geoidal separation values exist for specific locations and were obtained from the Canadian Hydrographic Service for Joggins, Nova Scotia. To allow for direct comparison between the various datasets, LiDAR data, GPS data and tidal heights were converted to orthometric heights (Canadian Geodetic Vertical Datum 1928, CGVD28), as in Webster et al. (2003). CGVD28 is the most recent and most commonly used vertical datum in Canada. At Joggins (approximately 22 km down-Bay), the Mean Water Level (MWL) is +0.11 m referenced to CGVD28 and +6.61 m referenced to CD.

GPS and vegetation quadrats

Leica dual frequency survey grade GPS equipment was used to validate both of the LiDAR datasets (survey specifications found in Table 3), as mentioned above. Several hundred validation points were collected on hard, flat surfaces on dates coinciding with both of the flights. The elevation values from these locations were compared to the elevation values of the bare earth DEMs to determine the error in the LiDAR datasets.

DGPS was also used to record the location of vegetation quadrats (1 m^2) used in imagery interpretation. Using a stratified sampling design, 76 quadrats throughout the reference marsh were examined for vegetation characteristics, with up to four repeat assessments at some locations for revalidation at different times during the marsh growth season. Vegetation data from the quadrats were collected on 12–14 August 2006 and coincided with the full biomass LiDAR data acquisition (12 August 2006). The data were stored in a GIS database and included vegetation species identification, percentage cover of each species and vegetation height, as well as the distribution of species in the surrounding area. Independent GPS validation points within the marsh ($n = 76$) were also collected to aid imagery interpretation and accuracy assessment.

Table 3 Differential GPS base station parameters used for LiDAR data acquisition and DEM validation

System	Leica 850 DGPS
Location	Amherst, NS NSHPN #213249
Latitude	45°48′14.586602″
Longitude	64°12′19.108740″
Elevation	−3.4904
Antenna height	1.826 m

Imagery

Two types of imagery were used. A Quickbird satellite image (60 cm pansharpened resolution) was acquired in December 2006 and orthorectified using the LiDAR DEM. Several aerial photographs (10 cm ground resolution, acquired May 2005), purchased from the Nova Scotia Geomatics Centre, were orthorectified and mosaicked, and then visually interpreted for vegetation distribution across the marsh.

Manual feature extraction ("heads-up digitizing") was performed using on-screen digitizing techniques based on spectral reflectance of vegetation, relative position, textural patterns and elevation characteristics (topographic forms visible in images [not LiDAR] and relative location of topographic features) (King 2012). Statistical classification routines (Supervised and Unsupervised classification) were also attempted in order to automate the feature extraction process, however, the resultant accuracy of these techniques was too low (<70 %) due to difficulties in separating different vegetation classes to an acceptable level based on spectral reflectance alone (Millette et al. 2010; Friess et al. 2012; King 2012). Overlaying the various GPS field data plots on the imagery allowed the dominant, secondary and tertiary species (by percent cover) to be determined for each polygon. Where validation points did not exist, the colour, texture and relative location in the imagery were used to estimate the spatial extent of different assemblages of species. The main images used for this task were the aerial photographs as the Quickbird imagery was lower resolution (60 cm as compared to 10 cm aerial photographs) and had been acquired in the winter when remaining vegetation (reduced due to dieback) was more difficult to distinguish. However, the Quickbird imagery was useful for identifying salt pannes and differentiating areas of vegetation (darkly coloured) from areas of standing water, which also appear dark in summer images. The LiDAR intensity data was also used to differentiate open standing water from salt pannes with vegetation, and from areas of dry vegetation. Although LiDAR intensity cannot be used to quantitatively differentiate vegetation, due to an inherent error added to the signal for range normalization (Garroway et al. 2011), LiDAR intensity can be used qualitatively. Wetter areas and areas with standing water will have lower LiDAR intensity values than dry areas but open water will result in very low values or even no LiDAR return due to absorption of infrared energy by the water. Therefore, open water will appear very dark (lower intensity) and vegetated areas will appear much brighter (higher intensity), depending on their wetness.

In areas where vegetation types were difficult to determine, based on imagery and validation plots, additional field surveys were carried out to determine the species composition and relative abundance. These surveys identified the boundaries of vegetation zones using GPS where these boundaries were uncertain in imagery. Independent vegetation spot-checks were used to perform image interpretation accuracy assessment. The final resulting image interpretation resulted in >89 % accuracy based on dominant species identification.

Modelling

Vegetation limits

Various species combinations were identified in the vegetation surveys, and/or interpreted through image analysis, and were designated to one of six salt marsh vegetation "zone" (Table 4), based on the zones (high and low) commonly adopted in other studies (Adams 1963; Gordon et al. 1985; Bertness and Ellison 1987; Bertness 1991; Pennings and Callaway 1992; Davidson-Arnott et al. 2002; Roberts and Robertson 2005; Byers and Chmura 2008). The low zone was dominated by tall *S. alterniflora* and the high zone by *S. patens*. The high zone was subdivided into five subzones:

(1) *S. patens* dominant,
(2) *S. patens* and *S. alterniflora* (short) mixed,
(3) salt pannes (where standing water exists and *S. patens* and *S. alterniflora* are mixed),
(4) *Juncus gerardii* dominant, and
(5) disturbed areas where historic and degraded dykes exist.

These subzones were based on distinct vegetation assemblages observed in the field and are thought to exist due to differences in physical factors other than elevation (for example, historic influence of ditches or dykes, or standing water). Other studies of marshes in North America describe similar assemblages and physical characteristics. Several authors (Adams 1963; Niering and Warren 1980; Millette et al. 2010) report salt panne communities and mixed communities where standing water exists. Bertness (1991) determined that *S. alterniflora* was excluded from areas dominated by *J. gerardii* and *S. patens* due to competition, unless other physical stressors (e.g. salinity) allow it an advantage. It is well known that *S. patens* has limited ability to withstand high saline conditions (Roberts and Robertson 2005; Bertness and Ellison 1987; Moeslund et al. 2011). Bertness and Ellison (1987) noted two height classes of *S. alterniflora*: a tall class that grew at a significantly lower elevation, and a shorter class that grew in the high marsh zone, often inter-mixed with *S. patens*.

Zonal elevation statistics (minimum, mean, maximum, range, standard deviation) were calculated for zones determined through polygons produced from image interpretation. The elevation "limit" (mean ± standard deviation) of each vegetation zone was computed, as in Olsen and Ollerhead (2005) and Byers and Chmura (2008). It is important to note that the values reported for the means and limits have taken into account error of the measurement device (i.e. LiDAR) within vegetated areas. Although LiDAR is reported by the industry to be vertically accurate within 15 cm, others have found significant over estimation of ground measurements due to poor penetration through dense vegetation (Montané and Torres 2006; Millette et al. 2010; Chassereau et al. 2011). Our results indicated a uniform offset of 0.3 m in low- and high marsh zones; this offset was accounted for in the calculations of vegetation zone limits.

Assessing suitability of restoration sites

The ability of vegetation to form within restoration sites is largely dependent on the elevation of the land and patterns in tidal inundation (Weinstein et al. 2001;

Table 4 Salt marsh vegetation species and associated zones

Zones	Subzones	Species
Low marsh		*Spartina alterniflora*
High marsh	SPAL/SPPA mixed	*Spartina alterniflora* and *S. patens*
	SPPA dominated	Any of: *Spartina patens*, *Distichlis spicata*, *Juncus gerardii*, *Hordeum jubatum*, *Limonium carolinium*, *Plantago maritima*, *Puccinellia maritima*, *Solidago sempervirens*, *Triglochin maritima*—but not *Spartina alterniflora*
	Salt panne	*Spartina alterniflora* and/or *S. patens* and standing water
	JUGE dominated	Dominated by *Juncus gerardii* but may also find: *Spartina patens*, *Solidago sempervirens*, *Triglochin maritima*
	Disturbed	Any of: *Achillea millefolium*, *Elymus trachycaulus*, *Agropyron* sp., *Spartina pectinata*, *Rubus* spp.—but not *Spartina alterniflora*

Weinstein and Weishar 2002; Blott and Pye 2004; Byers and Chmura 2008; Spencer and Harvey 2012). The relationship between elevation and vegetation characteristics at the Beausejour reference marsh was used to determine the suitability of the sites considered for restoration. Using the vegetation elevation zone limits derived from the reference marsh, the DEM of each restoration site was reclassified using ArcGIS to predict the extent of land that would be immediately suitable for salt marsh growth after reintroduction of tidal waters, assuming no change in elevation due to dyke breaching. It was assumed restoration site areas that fall within the marsh elevation zone limits (derived from the reference marsh) would be suitable for the growth of vegetation types associated with specific elevation zones (Table 4; Fig. 6). It is important to note that high exposure and wave action on the seaward edge of the reference marsh may have reduced the lower limit of the elevational range of *S. alterniflora*. If exposure of one or both of the restoration sites, following full or partial removal of barriers, is less severe than on the reference salt marsh, then *S. alterniflora* may be able to grow at lower elevations.

Results

LiDAR validation

On hard surfaces, the assessment of the bare earth LiDAR DEM when compared to the DGPS elevations resulted in a uniform offset of 1.0 m (±0.02 m standard deviation). Following a correction of 1 m in the DEM, the accuracy was ±0.02 m. This is significantly better than the industry reported vertical accuracy of LiDAR of ±0.15 m.

As low- and high marsh zones have characteristically different vegetation height and density, accuracy assessment was performed within each of the marsh zones. As reported in other studies (Montané and Torres 2006; Sadro et al. 2007; Millette et al. 2010; Chassereau et al. 2011), we found that the LiDAR was unable to fully penetrate the vegetation, however, the overestimation of surface height of the bare earth DEM was similar for low- (0.32 m) and high marsh (0.30 m) (Table 5). This is likely due to the time of year the data was acquired (lowest level of biomass) and the processing techniques used. Based on these

Table 5 Error in LiDAR elevation for low and high marsh zones, as compared to DGPS elevation measurements

Zones	Mean LiDAR elevation error (m)	Standard deviation of LiDAR elevation error (m)	Mean vegetation height (m)	Standard deviation of vegetation height (m)
Low	0.32	0.077	0.45	0.79
High	0.30	0.09	0.23	0.146

results, a uniform error of +0.3 m was accounted for in the DEM and zone elevation limits.

Image interpretation

Image interpretation of the 10 cm resolution aerial photographs and 60 cm resolution Quickbird image resulted in over 1,200 polygons with 35 unique combinations of dominant, secondary and tertiary species (Fig. 3). The polygons ranged from 1 to 40,000 m^2.

The majority of the reference marsh was dominated by *S. patens*, a high-marsh zone species, inter-mixed sporadically with the high-marsh species listed in Table 4. Image interpretation was used to map the distribution of vegetation by dominant species (Fig. 4) and by vegetation zone type (Fig. 5). High exposure and erosion has resulted in an exposed 1 m high bank along the seaward edge of the reference marsh, thus potentially limiting the extent of low marsh. The creeks and low marsh areas are colonized by *S. alterniflora*. In the eastern part of the marsh, nearest the Missaguash River, there are several remnant dykes occupied by *Spartina pectinata* and upland species. These remnant dykes are less than half a meter above the high marsh zone and are easily identifiable in the LiDAR due to a stark difference in elevation as well as vegetation height. A large area of *J. gerardii* also exists on the eastern part of the marsh near the remnant dykes. In the western section of the marsh, large areas of standing water can be identified, as well as a series of abandoned man-made ditches, which also hold standing water.

LiDAR and vegetation zone limits

From the zone limits computed based on LiDAR elevation, it is clear that the low- and high marshes can be separated based on elevation, as has been shown by others (Blott and Pye 2004; Byers and Chmura 2008). Overlap in elevation within the high marsh subzones was expected as these are distinguished by vegetation type or physical differences in their environment (e.g. standing water), not by elevation alone (Fig. 6). As also shown by Niering and Warren (1980), Gordon et al. (1985), Konisky and Burdick (2004), Roman et al. (2005), and Byers and Chmura (2008), *J. gerardii* was found at higher elevations within the high marsh zone than most other high marsh species, with the exception of *S. pectinata* and *Elymus trachycaulus*, both of which were observed in historically disturbed areas (e.g. previously dyked). On the neighbouring John Lusby Marsh, Byers and Chmura (2008) reported *S. patens* growth between 0 and 1.4 m above the Mean High Water Level. We found similar separation of the low- (*S. alterniflora* dominated) and high marshes (*S. patens* dominated) zones at the Mean High Water Level (Fig. 6). In a comparative study of several marshes in the Bay of Fundy, Olsen and Ollerhead (2005) found that *S. alterniflora* and *S. patens* occurred at elevations (referenced to CGVD28) of 4.95 and 5.6 m, respectively, which corresponds closely to our findings of 4.8 and 5.4 m. Similar patterns have been reported for salt marshes located elsewhere along the east coast of North America (Adams 1963; Eleuterius and Eleuterius 1979; Bertness and Ellison 1987), although those studies report a greater percent contribution of *S. alterniflora* than has been found in the Bay of Fundy. This can be explained by differences in tidal range. In areas where tidal range is small, *S. alterniflora* occupies a larger portion of the tidal range (Gordon et al. 1985). In this study, and in Gordon et al. (1985), *S. alterniflora* grew within an elevation range of about 2 m, but this represents less than 20 % of the mean tidal range in Cumberland Basin.

Comparing vegetation elevations derived from GPS and LiDAR

Since GPS elevations were recorded at each vegetation plot, we can use these to compute salt marsh elevation limits (Fig. 7) and compare these with the LiDAR derived elevation limits. Interestingly, GPS data show the low marsh zone to have a much larger range in elevation than that shown by the LiDAR. The upper limit is much higher with the values derived from GPS, resulting in elevation overlap between the *Spartina* species. This difference is likely due to sample size, where in the *S. alterniflora* dominant zone only 14 GPS points [quadrats] were used in calculations versus in the *S. altnerniflora* polygons used. GPS surveys typically have small sample sizes due to the time involved in collecting the GPS points manually. They are thus subject to greater error on estimates of mean elevation. Another potential factor with small GPS sample sizes is measurement error. The larger

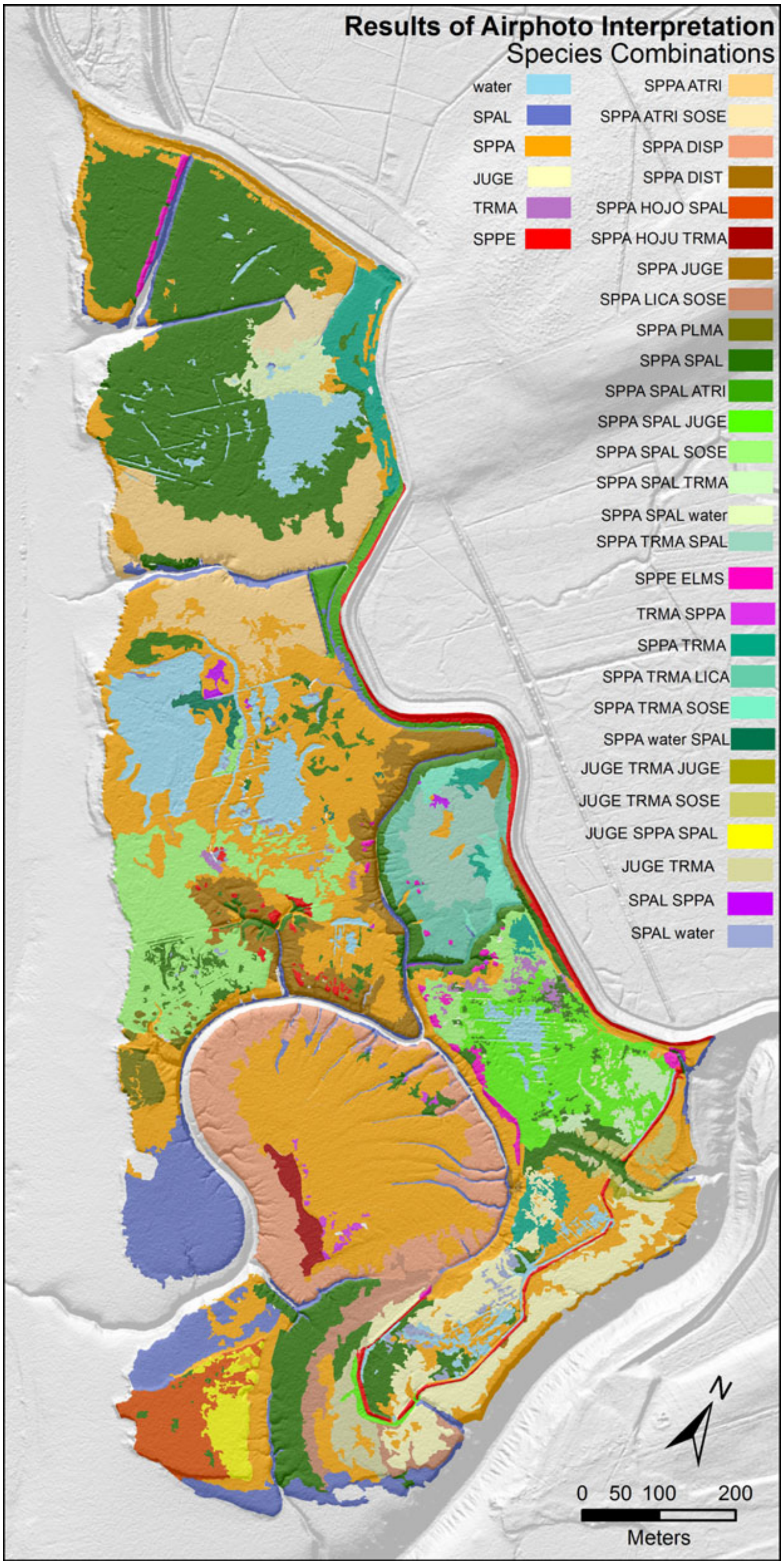

Fig. 3 Vegetation on the reference marsh determined from air photo and satellite imagery interpretation with the aid of GPS control points and associated 1 m^2 *quadrats* (species identification). *Colours* represent species and species combinations (see USDA codes in Appendix)

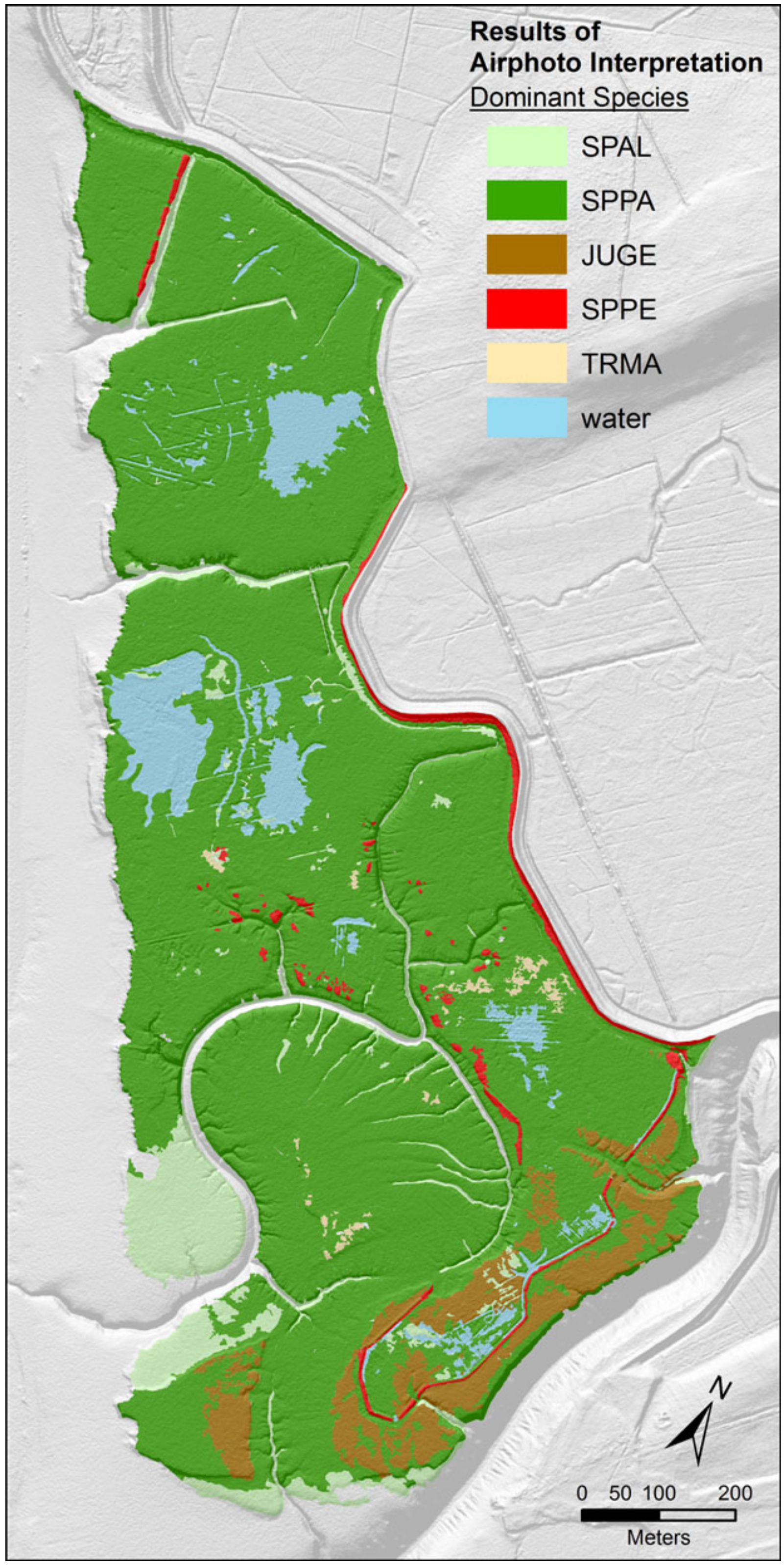

Fig. 4 Distribution of dominant vegetation species in the reference marsh, as determined through validation *quadrats*, and air photo and imagery interpretation. The dominant species overall is *Spartina patens* (SPPA). Species codes are shown in Appendix

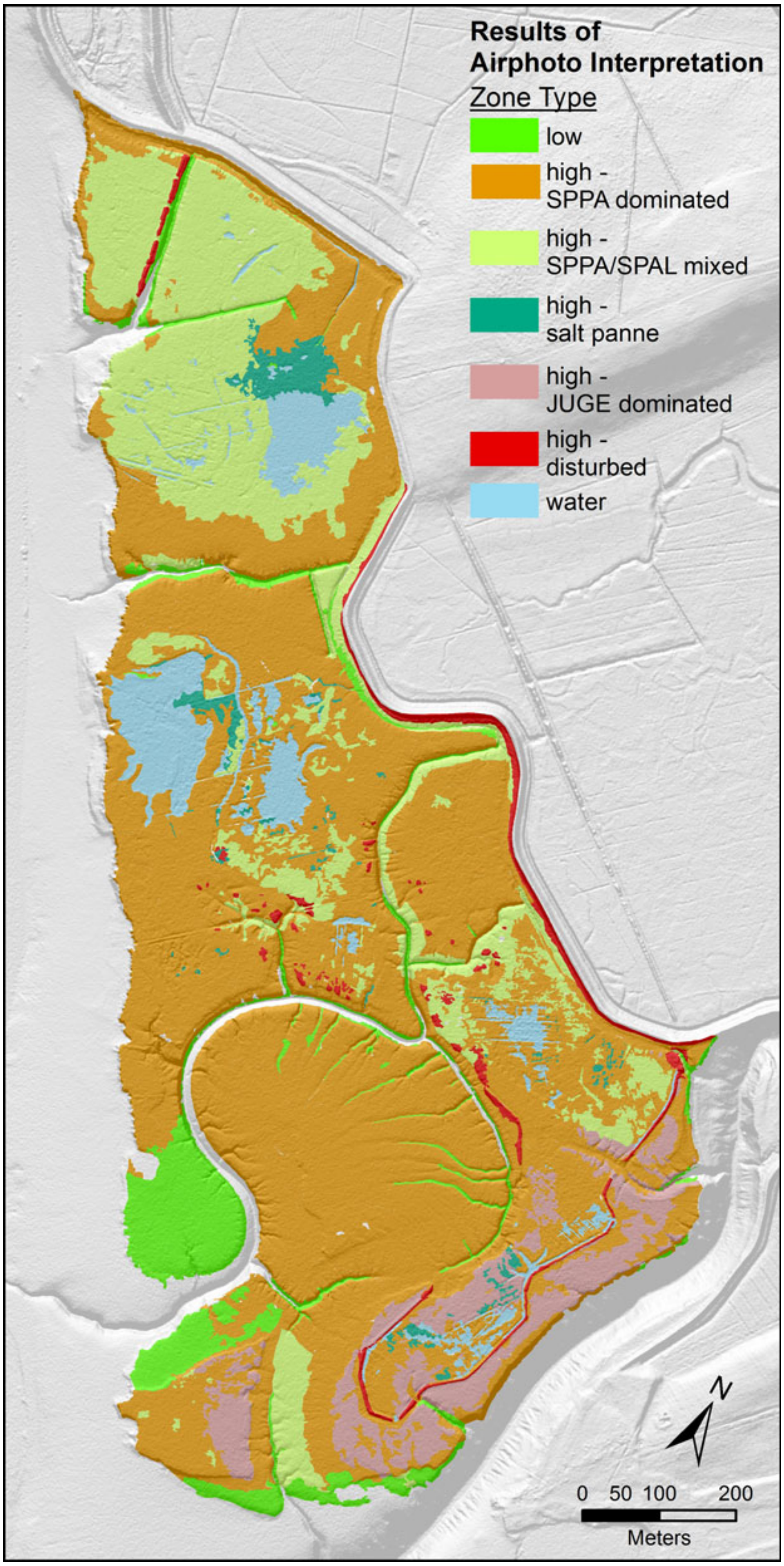

Fig. 5 Distribution of salt marsh vegetation zones in the reference marsh, showing *low* and *high zones* including five high *subzones* (*S. patens* dominated, SPPA/SPAL mixed, salt panne, *Juncus* dominated, and disturbed), as determined through validation *quadrats*, and imagery interpretation (aerial photographs, Quickbird and LiDAR intensity, not derived from elevation limits)

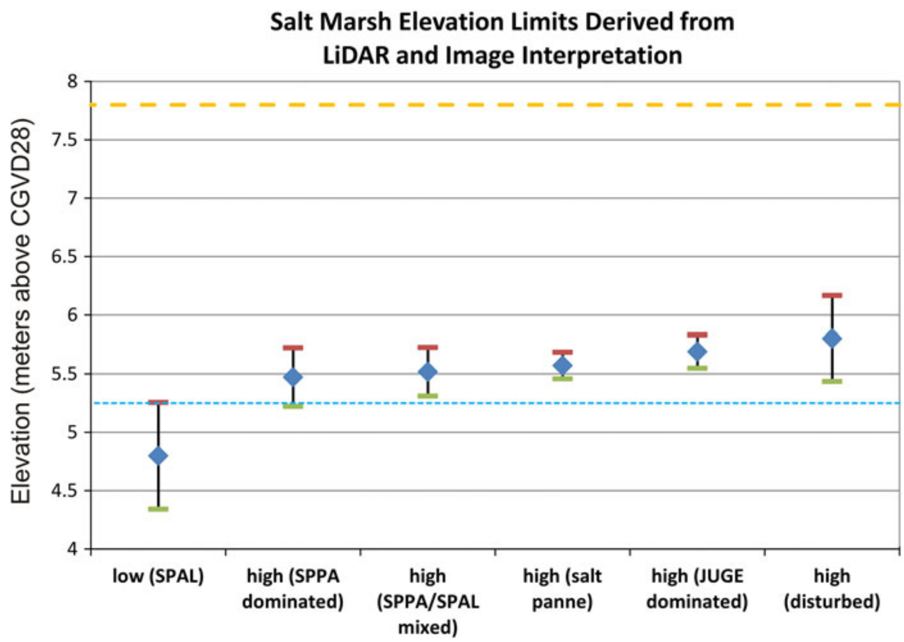

Fig. 6 Plot of elevation limits (mean ± 1 standard deviation) for each salt marsh zone derived from the LiDAR DEM and vegetation polygons interpreted from imagery. Elevations are referenced to CGVD28, where Mean Water Level (MWL) is approximately 0.11 m (Chart Datum MWL = 6.61 m). The *lower dotted line* represents the Mean High Water level and the *high dotted line* represents High High Water Large Tide

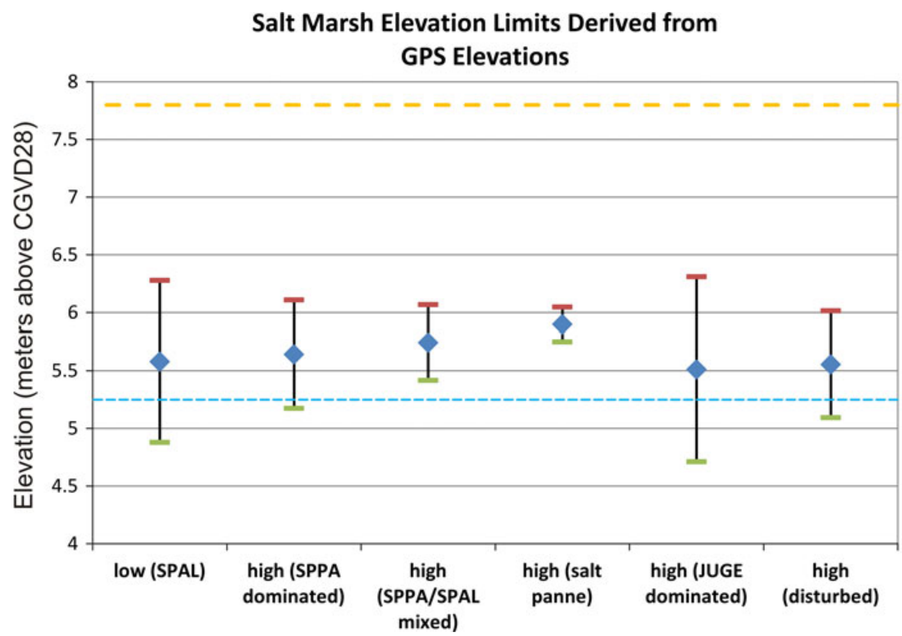

Fig. 7 Plot of elevation limits (mean ± 1 standard deviation) for each salt marsh zone derived from the GPS validation plots. Elevations are referenced to CGVD28, where Mean Water Level (MWL) is approximately 0.11 m (Chart Datum MWL = 6.61 m). The *lower dotted line* represents the Mean High Water level and the *high dotted line* represents High High Water Large Tide

sample size for LiDAR minimizes the effect of extreme or outlier values.

The two methods resulted in similar elevation limits for all high marsh subzones, except for the disturbed subzone where LiDAR showed a higher mean elevation. This may be due to LiDAR penetration issues in the vegetation canopy as this zone was inhabited by taller vegetation (>1 m). The influence of vegetation was removed from the DEM based on measured values of vegetation height found in the field (discussed in "LiDAR validation" section). However, we collected very few data on vegetation height for the "disturbed" subzone ($n = 3$ plots) and therefore the vegetation height measurements may not have been representative, resulting in increased error in the LiDAR DEM for this zone.

Modeling restoration site suitability

Modeling of the restoration sites, using vegetation elevation limits, allowed us to determine if any areas of the site were too low or too high (above level of tidal inundation) for immediate colonization of marsh vegetation (Figs. 8, 9). An estimate of the area and percentage area in each restoration site that was considered suitable for salt marsh vegetation is shown in Table 6. Approximately 47 % of Restoration Site 1 was lower in elevation than the lowest limit of salt marsh vegetation in the reference marsh, indicating that it is unlikely that salt marsh vegetation could, at least initially, colonize the entire site (Fig. 8). Significant sediment accretion would need to occur for the elevation of this site to be sufficient for extensive salt marsh growth. Restoration Site 2 had the greatest area of land immediately suitable for restoration, with 89 % of its total area already at an elevation suitable for marsh growth (Fig. 9). The elevation data indicate that, for both restoration sites, *S. alterniflora* would dominate newly established marsh vegetation.

Discussion

A growing recognition of the ecosystem values of salt marshes has prompted conservation efforts to restore previously existing salt marsh areas (Weinstein et al. 2001; Davidson-Arnott et al. 2002; Blott and Pye 2004; Byers and Chmura 2008; van Proosdij et al. 2010; Friess et al. 2012; Spencer and Harvey 2012). When selecting sites for restoration, it is desirable to undertake an analysis of the suitability of potential sites for re-establishment of marsh vegetation. As shown here and elsewhere (Blott and Pye 2004; van Proosdij et al. 2010), site assessment can be aided by geomatics tools, in particular LiDAR, which allows the collection of high resolution elevation data. Remote sensing offers many advantages over traditional methods (e.g. surveying with a Total Station or GPS). It allows data to be quickly collected over large areas, and in sensitive locations or areas that are difficult to access, such as mudflats and salt marshes (Sallenger et al. 2003). Remote sensing also allows for spatio-temporal change analysis as large amounts of data can be repeatedly collected over time (Millette et al. 2010). Our study also showed that LiDAR-multispectral combination has a distinct advantage over GPS surveys of salt marsh vegetation in that LiDAR allows for a more representative characterization of the marsh surface and a reduction in the effects of any measurement outliers.

Using a combination of high resolution LiDAR elevation data, remotely sensed imagery, and high precision DGPS, salt marsh vegetation zones can be mapped and the relationship between the elevation of the land, tidal characteristics and vegetation species can be assessed on a marsh-by-marsh basis. This information can then be used to determine site suitability for the re-establishment of salt marsh

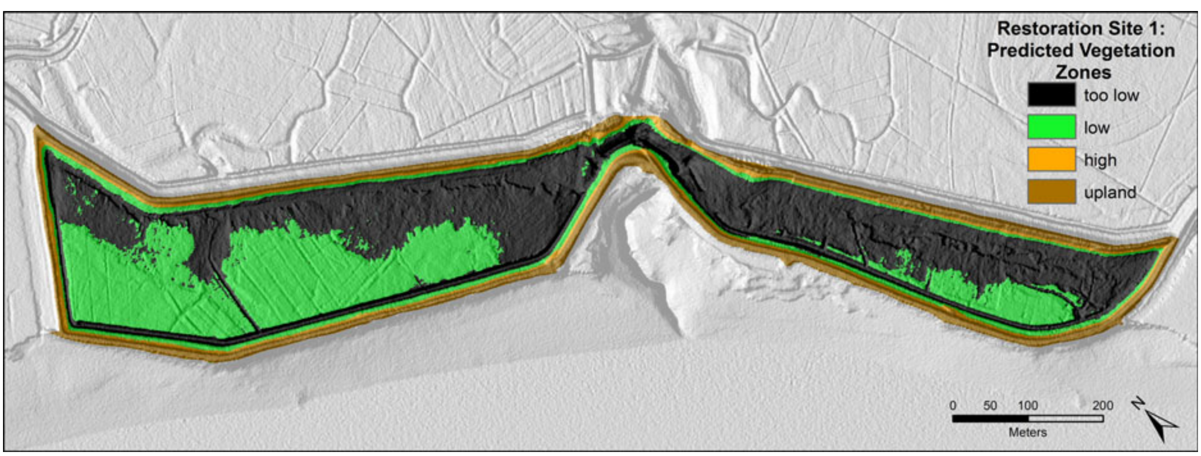

Fig. 8 Map of predicted salt marsh vegetation in *Site 1*. Note the "double dykes" visible in the LiDAR data. The seaward dyke is the original, now failing dyke and the landward dyke is the new dyke recently installed to protect adjacent farmland from flooding

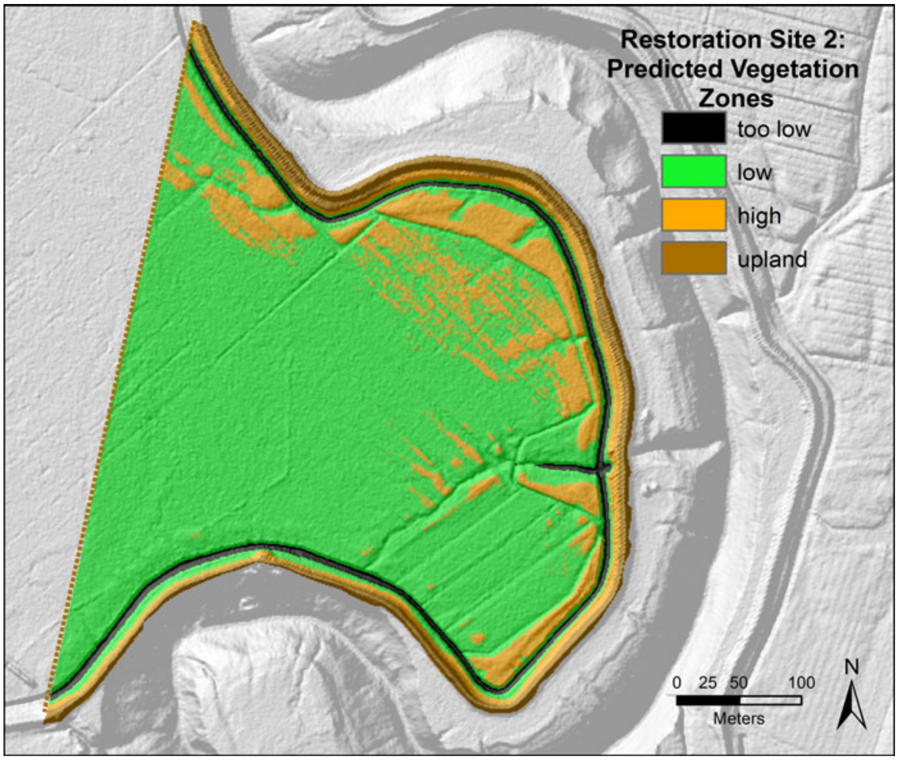

Fig. 9 Map of predicted salt marsh vegetation in *Site 2*. Note the computer-simulated dyke running southwest–northeast, along the river edge. This dyke was simulated in order to model flooding of the restoration site and not adjacent farmland. If a restoration were attempted at this site, a similar physical barrier would be required

Table 6 Restoration site suitability for marsh vegetation colonization

	Area of each site that is suitable for salt marsh vegetation after initial breach		Percentage of restoration area suitable for salt marsh vegetation (low plus high) (%)
	Area (m^2)	Area (%)	
Restoration Site 1			
Too low	113328	46.8	39.3
Low	82108	33.9	
High	13174	5.4	
Upland	33645	13.9	
Restoration Site 2			
Too low	6332	4.0	88.9
Low	111314	70.9	
High	28221	18.0	
Upland	11176	7.1	

Note that only low and high marsh zone areas can be immediately colonized

vegetation. It is preferable that the reference marsh be similar to the restoration site(s) in size, geomorphology, tidal range, landscape position and adjacent land use (Neckles et al. 2002). Understanding the relationship between elevation and salt marsh vegetation of the reference site provides insight into the likely suitability of a particular restoration site before any actual breaching of barriers (Neckles et al. 2002; Blott and Pye 2004).

The work conducted in this study confirmed that vegetation zonation in the Cumberland Basin is highly dependent upon elevation, as found in many other areas (e.g. Bertness and Ellison 1987; Donnelly and Bertness 1991; Bernhardt and Marcus 2003; Genc et al. 2004; Wolters et al. 2005; Rosso et al. 2005; Silvestri et al. 2005). One of the two potential restoration sites (Site 1) was not found to be immediately suitable for colonization by salt marsh plants. Following the conversion of marsh land to agricultural lands in the 1600s, the reclaimed land at this site experienced a high degree of subsidence, as interpreted through the LiDAR elevation measurements. When these measurements were made, much of this site was below the lowest elevation of salt marsh vegetation in the adjacent reference marsh. This restoration site is currently the focus of a restoration experiment involving three engineered breaches in the outer dyke wall (breached in October 2010). The site is being monitored, and sediment accretion processes examined, in advance of expected colonization by *S. alterniflora*.

Restoration Site 2 is less exposed (borders a river) and offers considerable area (89 %) that is of suitable elevation for the immediate establishment of salt marsh vegetation. While the latter site is the preferred option for salt marsh restoration, any breach in the dyke will need to be carefully engineered in order to avoid unwanted, large scale sediment movement and structural changes to the river channel. Effects on the hydrodynamics and hydraulics of the adjacent river will need to be investigated prior to any breaching. A large landward dyke will be required to prevent tidal inundation on adjacent farmlands.

Another factor to consider is that, when tidal inundation resumes, the natural links between tidal basins and marshes may not reform on their own. In the development of a natural marsh, creek networks form first, followed by marsh plant colonization and stabilization of sediments, allowing creek banks to form and more complex creek networks to develop (Weinstein et al. 2001). In potential restoration sites that are fully vegetated by land plants, stable sediments already exist and tides (even large tides) may not be able to erode channels that allow drainage of developing salt marshes (Weinstein et al. 2001). Post-restoration monitoring is essential and can be undertaken using remote sensing methods (Byers and Chmura 2008; van Proosdij et al. 2010). For example, repeated LiDAR data acquisitions would enable assessment of spatio-temporal changes in elevation within the restoration site and adjacent channels. Spectral imagery could also be acquired for mapping the development and distribution of species but would need to be coupled with field validation, as in this study.

LiDAR has many applications beyond the creation of high resolution elevation models (Richardson et al. 2010; Garroway et al. 2011). Several authors (Genc et al. 2004; Hopkinson et al. 2006; Millard et al. 2008) have used LiDAR to calculate vegetation height in wetlands which can then be used in flood models or in estimates of above ground biomass. LiDAR also provides values of return intensity which may be correlated with wetness and soil moisture (Challis et al. 2011; Garroway et al. 2011). Coupling LiDAR with other remotely sensed imagery, such as multispectral (as in this study) or radar imagery, can provide a unique combination of information about the nature of species assemblages and zonation patterns that is impossible to obtain at the same level of resolution and coverage using field methods. Due to the small footprint of airborne LiDAR acquisitions, LiDAR may be prohibitively expensive for very large study areas (Chmura 2011), however, the vegetation structure and topographic information that can be derived from it are unique to this technology and can lead to new understandings of processes that cannot be obtained with other earth observation or field methods.

Conclusion

A combination of geomatics technologies and techniques, coupled with field validation, were used to map the elevation and vegetation of a reference salt marsh in Cumberland Basin. Based on vegetation zonation and elevation data, marked differences in the suitability of two adjacent salt marsh restoration sites were revealed. This study highlights the importance of mapping the elevation of proposed restoration sites and determining vegetation zonation patterns of a reference marsh before restoration activities are undertaken. As vegetation patterns and relationships observed in Cumberland Basin are similar to those found worldwide, the same methods can be applied elsewhere to determine restoration site suitability before a purposeful breach.

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Appendix: USDA species codes

ACMI	*Achillea millefolium*
ATPR	*Atriplex prostrata*
DISP	*Distichlis spicata*
GLMA	*Glaux maritima*
ELMS	*Elymus trachycaulus*
HOJU	*Hodeum jubatum*
JUGE	*Juncus gerardii*
LICA	*Limonium carolinium*
PLMA	*Plantago maritima*
PUMA	*Puccinellia maritima*
SOSE	*Solidago sempervirens*
SPAL	*Spartina alterniflora*
SPPA	*Spartina patens*
SPPE	*Spartina pectinata*
SUMA	*Suaeda maritima*
TRMA	*Triglochin maritima*

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