Pre-restoration assessment of carbon at the Lower Otter Restoration Project

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Executive Summary

- The Lower Otter Restoration Project will create approximately 55 hectares of mudflat and saltmarsh by reinstating tidal inundation to reclaimed grazing land in the Lower Otter Valley, a process known as managed realignment.
- Here, we report the findings of a baseline assessment of the carbon stored in pre-restoration vegetation and soils at the Lower Otter site, assess the potential for carbon storage in the managed realignment, and devise a strategy for monitoring that carbon storage.
- The carbon storage and sequestration of the existing vegetation was estimated by assessing areal coverage using a point-based mapping approach and combining these areas with literature values of carbon. It was estimated that the current tree/hedgerow/shrub cover stores ca. 1,200 tonnes of carbon (ca. 4400 t CO_2e) and sequestering a further 20 t carbon annually (74 t CO_2e). Above-ground grass/herbaceous vegetation was estimated to store ca. 80 t C, and sequester ca. 23 t annually
- The total carbon (organic and inorganic) content of soils was measured in ten cores collected from three vegetation-based strata within the Lower Otter site. The carbon contents were combined with measurements of dry bulk density to calculate carbon densities and extended to a site wide estimate based on the areal extent of each stratum. It was estimated that the soils contained ca. 8 to 17 kg carbon per square metre (to 50 cm depth), with a site wide estimate of ca. 8,500 tonnes carbon.
- It was estimated that a total of c. 8,000 – 20,000 tonnes of organic carbon (29,000-74,000 tonnes of CO_2e) could be accumulated (over a period of c. 44-72 years) on the Lower Otter managed realignment site in the sediment that accretes after the restoration.
- A preliminary monitoring scheme based on the pre-breach sampling has been suggested. Practical considerations such as access may mean this scheme needs to be modified, and additional sampling points may be needed to capture vegetation types that develop on the site.

1. Introduction

The Lower Otter Restoration Project will create ca. 55 hectares of mudflat and saltmarsh by restoring tidal inundation to the Lower Otter Valley in a process known as managed realignment.

The primary motivation for the project was to address the challenges of climate change and failing sea defences in the lower River Otter by reconnecting the River Otter with the natural floodplain, creating intertidal saltmarsh, mudflats and freshwater habitats.

In addition to increasing climate resilience by providing improved protection from sea-level rise and storm surge, the project is anticipated to help climate mitigation through carbon sequestration and storage within the restored saltmarsh and mudflats.

1.1. Aims

East Devon Pebblebed Heaths Conservation Trust (EDPHCT) engaged Manchester Metropolitan University to conduct a baseline carbon assessment for the Lower Otter site and devise a strategy for monitoring carbon storage within the Lower Otter managed realignment. Specifically, to:

1. Quantify carbon stocks in existing vegetation on the Lower Otter site, with a focus on trees that will be lost as a result of the project.
2. Quantify the carbon content of soils within the Lower Otter site, stratified by land use and vegetation cover.
3. Devise a strategy for the ongoing monitoring of carbon storage within the Lower Otter managed realignment following flooding of the site.

This report contains four further chapters. Chapters 2 and 3 provide estimates of the carbon stocks of existing above-ground vegetation (Chapter 2) and soils (Chapter 3). Chapter 4 makes estimates of the potential for carbon accumulation following restoration. Chapter 5 makes recommendations for a strategy for ongoing monitoring within the Lower Otter managed realignment.

2. Baseline Assessment – Carbon Stored in Vegetation

2.1 Introduction

The aim of this chapter was to remotely assess the carbon storage and sequestration of existing vegetation of the Lower Otter restoration area, as part of a baseline, pre-restoration assessment. We quantified the area of different vegetated land covers using a point-based land cover mapping approach. We combined the areas of each land cover type with established values of the carbon storage and sequestration for those land covers to estimate the carbon storage and sequestration of vegetation at the Lower Otter prior to restoration, providing a baseline against which future comparisons can be made.

This chapter is structured as follows: Section 2.2 presents an assessment of the land covers currently present at the Lower Otter site. Section 2.3 assesses the potential carbon storage and sequestration of the vegetation present on the site prior to restoration, including a discussion of the values obtained from the literature that were used as multipliers to calculate the carbon storage and sequestration. Section 2.4 identifies some caveats and limitations to the study.

2.2 Lower Otter land cover

Site boundary and preliminary remote investigation

The boundary of the Lower Otter area was manually defined in QGIS from aerial photography and lidar imagery. The boundary (Figure 2.1a) was defined as slightly larger (74 ha) than the site to 1) ensure all areas at and below highest astronomical tides (HAT, required for estimates in Chapter 4) were captured, and 2) to ensure inclusion of vegetation at the boundary that may be altered during site construction.

A preliminary remote investigation of the site was then undertaken, examining the following available data sets:

- high resolution aerial imagery (25 cm) (Getmapping, 2018; Figure 2.1b), which enables visual inspection of the site
- lidar point cloud data (Defra, 2017; Figure 2.1c), which provides information on vegetation height
- woody linear features GIS data (Scholefield et al., 2016a; Figure 2.1d), which provides the distribution of (woody) linear features at the site.

Examining these datasets revealed two clear issues. Firstly, it emphasised the difficulty of remotely distinguishing hedgerows from tree canopy due to apparent unmanaged height and width of hedgerows on site. Secondly, it was apparent that the woody linear features GIS dataset mis-identified other (non-woody) linear features (e.g. a railway line and an aqueduct).

Due to this the primary analysis of land cover was conducted using point-based sampling of aerial imagery (see below) with trees, hedgerows and shrubs grouped as a single tree/hedgerow/shrub land cover class, collectively defined as the area of woody vegetation, including leaves, branches, and trunk, obscuring the ground when observed from above (Grove et al., 2006).

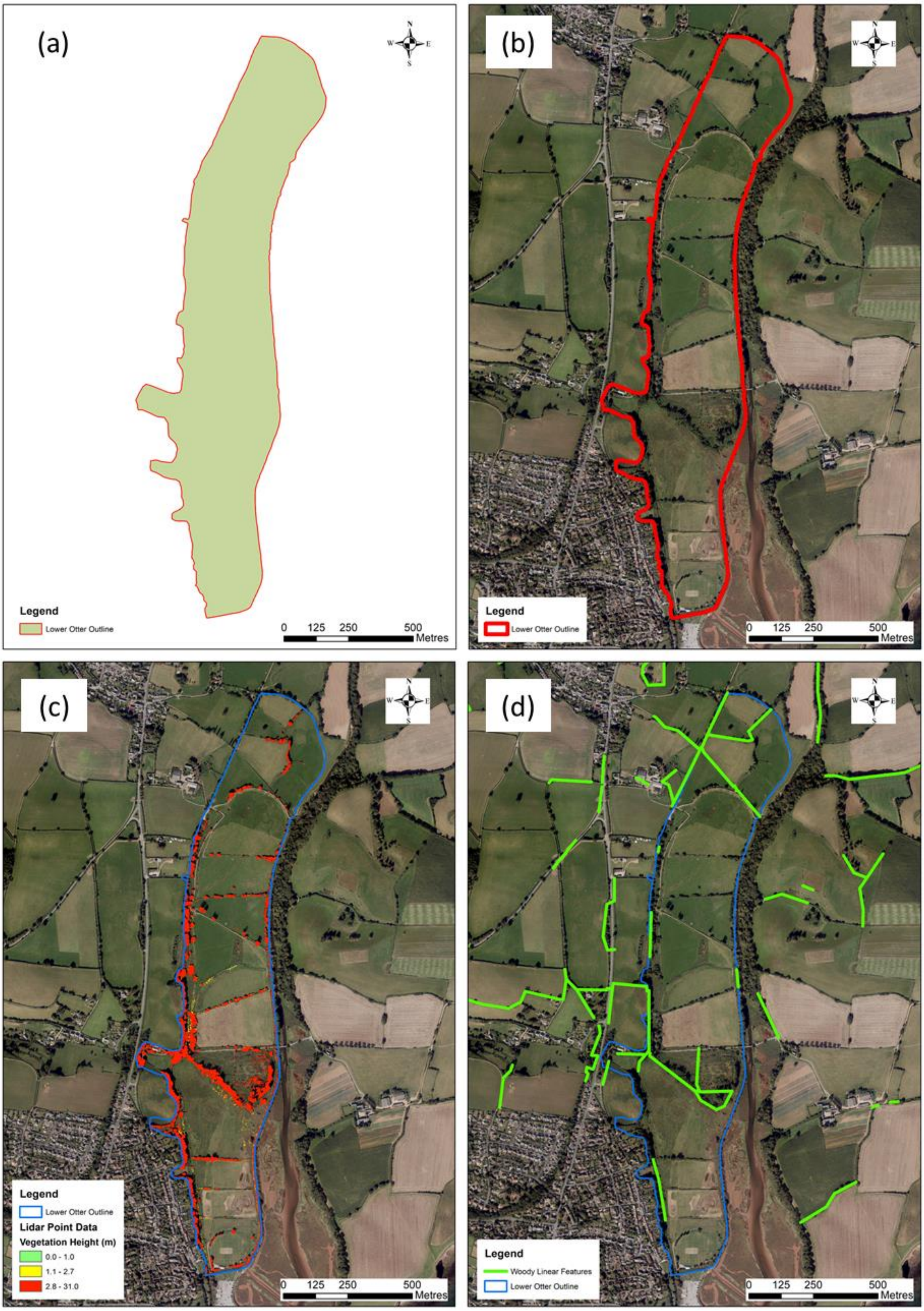

Figure 2.1. Lower Otter boundary and features. ***(a)*** *Lower Otter polygon shapefile representing the site boundary used in analysis (area of 74 ha) and available datasets:* ***(b)*** *High resolution (25cm) aerial imagery of Lower Otter restoration area (within boundary) and the surrounding area (Getmapping, 2020).* ***(c)*** *Classified vegetation height using Lidar point cloud data (Defra 2017) (first of many laser pulses returned) overlayed on aerial imagery of the study site (note: North-west area of the site no data was available) (Defra (2017), contains public sector information licensed under the Open Government Licence v3.0. Note: Lidar imagery was not used in analysis* ***(d)*** *Woody linear feature GIS dataset (Scholefield et al., 2016a; Scholefield et al., 2016b) is an EDINA Environment Digimap supplied service. Note: Dataset was not used in analysis.*

Land cover analysis

The use of point-based sampling approaches to measure land cover, tree canopy and other vegetation cover is well established (Norton-Griffiths, 1988; Nowak et al., 1996), as is the quantification of tree/vegetation carbon storage and sequestration based on this method (Mills et al., 2016; Pasher, McGovern, Khoury, & Duffe, 2014; Strohbach & Haase, 2012). Using this method, random points are generated over aerial imagery and the analyst classifies each point into a cover class (e.g., tree, grass, bare earth). A statistical estimate of the percent land cover of each cover class can then be calculated (Eq. 1) along with the uncertainty or standard error (SE) of each land cover class (Eq. 2):

Land cover percentage: $p = n/N$ **Eq. 1**

Standard error: $\sigma p = \sqrt{}\,(p(1-p))/N$ **Eq. 2**

Where N is the number of sample points, n is the number of sample points assigned to a particular class, p is the land cover percentage, and σ is the population standard deviation (i-Tree, 2011; Mills et al., 2016; Nowak et al., 1996). Percentage land cover is multiplied by the area analysed to determine the total area for each land cover class.

As more points are classified the standard error decreases, and a more precise estimate of land cover is achieved. Due to the manual user identification of land cover for each point, this method has been recognised as more accurate than some other supervised and unsupervised image machine classification methods (Endreny et al., 2017). Once land cover percentages and area statistics are established to a suitable degree of accuracy, carbon multipliers can be used to determine estimated carbon storage and sequestration within a site boundary (Nowak, 2019; see section 2.3 below).

i-Tree Canopy software (https://canopy.itreetools.org) was used as the platform for the point-based classification. i-Tree Canopy is part of suite of peer reviewed i-Tree tools, developed by the US Forest Service and its partners to assess the ecosystem services provided by urban forests (Nowak, 2019). i-Tree Canopy is growing in popularity and has been widely used to estimate tree canopy cover and ecosystem service benefits in the UK, USA, Portugal, Australia, Italy, and Ireland (Buccolieri et al., 2020; Del Moretto, Branca, & Colla, 2018; Doick et al., 2017; Mills et al., 2016; Olivatto, 2019; Preston, 2021).

To undertake the i-Tree Canopy point-based land cover assessment, a polygon shapefile of the Lower Otter study area boundary (Figure 2.1 a) was re-projected to coordinate system World Geodetic System 1984 (WGS84) using ArcGIS 10.7.1, to be compatible with the i-Tree Canopy software and imported into the tool. Six land cover classes were then configured (Table 2.1), the local UK region was defined as South-west and the random point-based survey undertaken.

Table 2.1. Descriptions of land cover classes

Land cover class	Definition
Grass/herbaceous vegetation	Short grasses and herbaceous vegetation or managed grass covered areas
Trees/hedgerows/shrubs	Trees, hedgerows, shrubs, and other woody vegetation
Bare earth	Disturbed ground where the underlying substrate is exposed
Hard surfaces	Man-made surfaces, including tarmac, concrete or other artificial surfaces used to construct roads and pathways
Built structures	Buildings and other man-made structures
Water bodies	Inland water bodies including streams, ponds, and artificially constructed reservoirs

The i-Tree specification recommends a minimum of 500-1000 points are assessed to produce a standard error of approximately ±2% (Doick et al., 2017; i-Tree, 2020a). The number of random points assessed for the Lower Otter land cover assessment was 1,200, achieving a standard error of less than ±1.25%, which corresponds to a 95% confidence interval that the actual land cover percentages deviate less than 2.4% from the estimated values.

Point-based land cover results are shown in Table 2.2 in ranked order. Land cover at Lower Otter was dominated by grass/herbaceous land cover (76.5%), whilst tree/ hedgerow/shrub cover was estimated at 18.25%. Other land cover classes identified were bare earth, hard surfaces, built structures and water bodies, which collectively accounted for c. 5% of the total site area.

Table 2.2. Estimated land cover percentages at Lower Otter

Land cover class	% cover %	Std Error ±	Area ha	Std Error ±
Grass/herbaceous vegetation	76.50	1.22	56.81	0.91
Trees/hedgerows/shrubs	18.25	1.12	13.55	0.83
Bare earth	3.58	0.54	2.66	0.40
Hard surfaces	1.33	0.33	0.99	0.25
Built structures	0.25	0.14	0.19	0.11
Water bodies	0.08	0.08	0.06	0.06

2.3 Carbon storage & sequestration

Carbon multipliers for trees/hedgerow/shrubs and grassland

As noted above, once the area of each land cover type has been determined to a suitable degree of accuracy, appropriate carbon multipliers can be applied to estimate carbon storage and sequestration within the site boundaries (Nowak, 2019). The multipliers considered in this assessment are presented in Table 2.3 and their appropriateness discussed briefly below (with a focus on the carbon storage values as these are of most interest here).

Table 2.3. Carbon storage and sequestration multipliers for trees/hedgerows/shrubs and grass/herbaceous vegetation. Carbon storage values for "line of trees" and all hedgerows are for above ground biomass only. Carbon sequestration in managed hedgerows is assumed to be zero (due to removal of new growth).

Land cover	Width	Height	Carbon multipliers		Ref.
			storage	sequestration	
	m	m	tC/ha	tC/ha/y	
Trees					
Urban trees/shrubs (i-Tree default)			76.85	3.06	Nowak et al. 2020
Woodland (3 m spacing, unthinned)					WCC, 2021
Mixed Broadleaves					
Yield class 8-12, 30 y			144.2-204.0	4.2-4.6	
Yield class 8-12, 100 y			254.1-344.6	0.6-0.9	
Oak					
Yield class 8, 30 y			144.4	2.5	
Yield class 8, 100 y			304.5	1.6	
Line of trees (ash, oak, sycamore)		>6	166	1.38	Robertson et al. 2012
Hedgerow					
Unmanaged, "tall"					Crossland, 2015
Hawthorn	6.0	not reported	93.50	1.79	
Blackthorn	3.5	not reported	131.50	6.00	
Hazel	4.0	not reported	45.08	1.25	
Unmanaged, "gappy"					Robertson et al. 2012
Hawthorn & hazel					
Tall		>3 to 6	45	0.51	
Medium		>2 to 3	22.5	0.26	
Short		2 or less	11.25	0.13	
Minimally managed	4.1±0.21	3.9±0.45	45.8±12.26		Axe, 2015 in Gregg et al. 2021
Laid & flailed (triennial)					Axe et al. 2017
Hawthorn					
Untrimmed (3 y growth)	2.6-2.9	3.5	35.8-44.5		
Trimmed	2.6-2.9	1.9	27.9-32.9		
Hawthorn & blackthorn					
Untrimmed (3 y growth)	4.2	3.5	45.7		
Trimmed	4.2	2.0	35.8		
Coppiced (1 y post coppicing)					Crossland, 2015
Hawthorn	0.7	not reported	25.65		
Blackthorn	0.55	not reported	27.62		
Hazel	1.50	not reported	34.35		
Grass/herbaceous vegetation			1.4		Davies et al. 2011
				0.4	Jo & McPherson, 1995

The default carbon storage multiplier used in the i-Tree Canopy tool is c. 77 $tC.ha^{-1}$ (Nowack, 2020). However, this multiplier is for urban trees, where open-grown and maintained (e.g. trimmed, topped, pollarded) urban trees tend to have lower above ground biomass than forest-grown trees (of the same trunk diameter at breast height) due to different crown shape (Nowak et al. 2013).

The Woodland Carbon Code (WCC, 2021) provides estimates of carbon storage and sequestration over time for trees of different species, planting density, yield class (YC) and under different management regimes. The values presented in Table 2.3 are for un-thinned stands of oak and mixed broadleaf species at 30 years and 100 years (WCC, 2021). Representative yield classes for the Lower Otter site were predicted using the default values in version 4 of the Ecological Site Classification (tree species) tool from Forest Research (http://www.forestdss.org.uk/geoforestdss) for those species identified on the site (as scattered trees or as lines of trees) in the Environment Statement (pedunculate oak YC8; ash YC8; black poplar YC10; aspen, YC12 (Environment Agency, 2022; Appendix E2 and E3-5)). These indicate carbon storage (within the tree biomass – i.e. above and below ground) of c. 144-345 $tC.ha^{-1}$. However, the carbon storage and sequestration of hedgerows and trees outside of woodland may differ significantly from that of trees within dense woodland or forests due to a variety of factors.

A study of carbon storage in arable land reverting to woodland at Rothamsted suggested that narrow strips of woodland may be more efficient at accumulating carbon than blocks of woodland due to greater light interception (Poulton et al. 2003). Using the data from this study, Robertson et al. (2012) estimated carbon storage in the above-ground biomass of a 'gappy' line of broadleaf trees (ash, oak and sycamore; >6m tall) to be c. 166 $tC.ha^{-1}$. Applying a root-to-shoot ratio of c. 0.28:1 (derived from Figure 2 in Poulton et al. 2003 and consistent with other values reported in the literature) would give a total carbon storage within the above and below ground tree biomass of c. 212 $tC.ha^{-1}$.

There are few studies on hedgerow carbon, but those available indicate that unmanaged hedges tend to store more carbon (in above-ground biomass) than those that are heavily managed (Axe et al., 2015 reported in Gregg et al., 2021; Axe et al., 2017; Crossland, 2015), while carbon sequestration rates are assumed to vary depending on hedge height (Robertson et al. 2012). Estimates of above-ground biomass carbon storage in tall, unmanaged or minimally managed hedges range from c. 45 $tC.ha^{-1}$ (hazel or 'gappy' hazel and hawthorn) to c.131 $tC.ha^{-1}$ (blackthorn). These estimates do not include below-ground biomass, where this was not considered in the Robertson et al. (2012) study, and Crossland (2015) applied a root-to-shoot ratio of 0.33:1, increasing the average carbon storage in tall unmanaged hawthorn, blackthorn and hazel hedges from c. 90 $tC.ha^{-1}$ to c. 120 $tC.ha^{-1}$. Estimates of carbon storage in above ground biomass of managed hedges range from c. 26 $tC.ha^{-1}$ (coppiced hawthorn) to c. 46 $tC.ha^{-1}$ (laid and flailed hawthorn and blackthorn, 3 years growth). However, the only study to measure below-ground mass (rather than exclude below-ground biomass or apply an assumed root-to-shoot ratio comparable to woodland trees) was that of Axe et al. (2017) for the laid and flailed hedges. This study found a root-to-shoot ratio 0.94:1, substantially increasing the average carbon storage in hawthorn and blackthorn hedges from c. 42 $tC.ha^{-1}$ to c. 80 $tC.ha^{-1}$ (Axe et al. 2017).

Given the above discussion and uncertainties, we have elected to estimate above ground carbon storage only, using a range of 90±45 $tC.ha^{-1}$. Our best estimate thus corresponds to all trees and hedgerows being represented by tall (grown-out) hedgerow, our lower estimate corresponds to 'gappy' tall hedgerow or managed hedgerow with 3 years growth, and our upper estimate corresponds to c. 60% of the tree/hedgerow/shrub cover being comparable to a 'gappy' line of trees, with the remainder being tall hedgerow. For an indicative assessment of annual carbon sequestration, we have adopted a multiplier of 1.5±1.5 tC.ha-1.y-1, encompassing the range from zero (assumed for managed hedges with removal of biomass) to the average for tall unmanaged hedges.

Low stature vegetation, such as grassland and other herbaceous vegetation, also store and sequester notable amounts of carbon due to their large spatial coverage, high productivity, and increased growth periods, though this is much lower per unit area than trees. There are knowledge gaps with regards to carbon storage and sequestration of grassland biomass because management regimes, including grazing and cutting, remove biomass (and carbon stores) annually. Davies et al. (2011) report herbaceous land cover carbon storage as 1.4 t per hectare, which corresponds with other studies (Golubiewski, 2006; Jo & McPherson, 1995), and carbon sequestration has been reported as 0.4 t per hectare per year (Jo & McPherson, 1995). These multipliers are used to provide an indicator of carbon storage and sequestration of grass/herbaceous land cover at Lower Otter (Table 2.2).

Carbon storage and sequestration

It was estimated that the tree/hedgerow/shrub cover stores approximately 1,041 tonnes of carbon, while grass/herbaceous vegetation stores approximately 79.5 tonnes carbon. In terms of annual sequestration rates, it was estimated that tree/hedgerow/shrub cover sequesters ca. 41.5 tonnes of carbon per year, while grass/herbaceous vegetation sequesters ca. 22.7 tonnes per year (Table 2.4).

Table 2.4. Estimated carbon storage and sequestration of trees/hedgerows/shrubs and above ground biomass in grass/herbaceous vegetation at Lower Otter

	Trees/hedgerows/shrubs				**Grass/herbaceous vegetation**			
	Storage		**Sequestration**		**Storage**		**Sequestration**	
	t	**±**	**t/yr**	**±**	**t**	**±**	**t/yr**	
Carbon	1,219.5	614.3	20.3	20.3	79.5	1.3	22.7	0.4
CO_2	4,468.5	2,250.9	74.5	74.5	291.6	4.7	83.3	1.3

2.4. Caveats and limitations

We have estimated the above-ground carbon storage and sequestration of tree, shrub and hedge cover in the Lower Otter, as well that in the above-ground grass and herbaceous vegetation, to provide an overall estimate of the above-ground carbon storage.

The adopted point-based sampling approach allowed broad estimates of carbon storage and sequestration in the above-ground vegetation (woodland, hedges, and grassland) to be made across the whole site.

The accuracy of the analysis depends the user's personal image interpretation and their ability to classify each point based on the supplied aerial imagery. Whilst the imagery used was relatively high resolution, some errors in interpretation may have occurred due to environmental factors or poor image quality. Other potential for misclassification can include user error, parallax errors (images not taken from directly above), cloud cover, and excessive shadow in a scene. To improve the manual classification of points in this study, contextual information from the surrounding area was taken into consideration by utilising the zoom function in i-Tree Canopy. Additionally, shadows were inspected and used as an indicator to differentiate between trees and other vegetation.

From the remote-sensing techniques we have used, it is not possible to identify the species of individual trees or compositions of hedgerows etc. Species do differ in their carbon storage capacity, so although the carbon multipliers used in the analyses are appropriate for the species identified on site (within the Environment Statement), there will be variations from species-specific analyses. Assessments of the carbon in individual trees that may be removed during construction was not conducted.

Due to the difficulty of distinguishing hedgerows from tree canopy at Lower Otter, carbon storage and sequestration was established for all woody vegetation together, resulting in a relatively large range in estimated carbon storage.

Including below ground carbon storage would increase the overall estimate significantly. Applying typical tree/woodland root-to-shoot ratios would increase the estimated carbon stock by around one third, however root-to-shoot ratios for managed hedges are substantially higher and could approximately double the estimated carbon stock.

3. Baseline Assessment – Soil Carbon

3.1. Introduction

This section presents the baseline assessment for soil carbon within the Lower Otter managed realignment site. Section 3.1 presents the methods employed, including site stratification, field sampling, and laboratory analysis. Section 3.2 presents the results, including an estimate of total soil carbon (within the selected strata) to 50cm depth.

3.2. Methods

Site stratification

Primary stratification of the site was based on the maps of National Vegetation Classification (NVC) communities provided by Clinton Devon Estates (2015a-c). The NVC maps were manually digitised, rescaled and merged to remove overlapping areas using CorelDRAW X7 (v.17.6.0.1021, Corel Corporation). Vegetation communitities and sub-communities were then grouped to produce a simplified map comprising polygons of each vegetation type within the site boundary (Figure 3.1), and the proportion of the total area (74 ha) covered by each type was extracted using the *raster* package in R v4.2.0 (Table 3.1).

Three primary strata were identified as a focus for this analysis: (i) MG10 *Holcus lanatus - Juncus effusus* rush-pasture (Rodwell, 1992); (ii) MG7 *Lolium perenne* leys (Rodwell, 1992); (iii) *Phragmites australis* swamp and reed beds (S4, S26) (Rodwell, 1995). The MG10 and MG7 communities (and sub-communities) were selected as they represent the dominant landcover types across all areas of the Lower Otter site (South Big Marsh, North Big Marsh, North Little Marsh). The *Phragmites* dominated communities (S4 and S26) were selected as *Phragmites* can transport methane from the sediment to the atmosphere (Grünfeld & Brix, 1999) and we anticipate *Phragmites* communities may remain present on site following reinstatement of tidal inundation, particularly in areas with freshwater inputs and lowered salinity.

Collectively, the selected strata (MG10, MG7, S4 and S26) accounted for ca. 69% of the total site area, including 90% of grassland and 71% of swamp and mire communities. The remainder comprised a wide range of NVC communities of comparatively small spatial extent (total 13%), along with areas of water (1.4%) and hardstanding and non-NVC communities (17%).

Table 3.1. Estimated area of the selected strata (MG10, MG7, and S4 and S26). Other vegetation types are given for reference. Area of the site (74 ha) was quantified from the extended boundary described in Section 2.2.

Stratum	**Estimated area**	
	% of site	**ha**
MG10 *Holcus lanatus - Juncus effusus* rush-pasture	47.5	35.3
MG7 *Lolium perenne* leys	16.7	12.4
S4 and S26 *Phragmites australis* swamp and reed beds	4.9	3.6
Other mesotrophic grassland communities (MG)	6.8	5.1
Other swamp communities (S,M)	2.0	1.4
Urtica dioica dominated community (OV24)	0.4	0.3
Woodland communities (W)	3.8	2.8
Water	1.4	1.0
Unmapped areas and non NVC communities and land covers	16.6	12.4

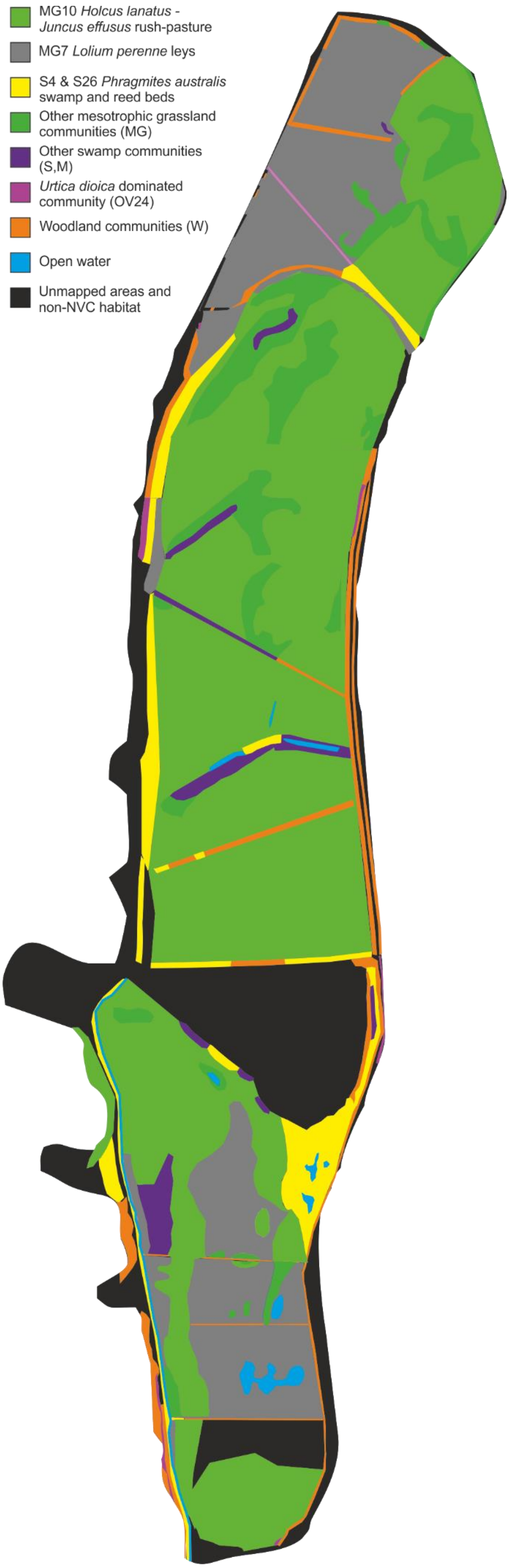

Figure 3.1. Simplified map of NVC vegetation communities within the analytical boundary of the Lower Otter prospective realignment site.

Sample locations and collection

Ten sampling locations were selected on the planned managed realignment, with a reference sample also collected from the adjacent natural marsh (Table 3.2; Figure 3.2).

The sampling locations within the Lower Otter site were determined based on the primary vegetation strata (see above) in combination with management regime (i.e. grazing intensity) and soil conditions (e.g. moistness) at the time of sampling (Table 3.2; Figure 3.3). There was considerable variation between MG10 pastures across the site, with those located in North Little Marsh being notably drier than those located in South Big Marsh, and the central pastures in North Big Marsh being particularly heavily grazed (Table 3.2; Figure 3.3).

All sampling was carried out in April 2021 by Hannah Mossman (MMU) and Kendal Archer (Clinton Devon Estates). A core for soil carbon analysis and a surface soil sample for determination of dry bulk density were collected at each sampling location. Soil cores were collected using an Eijkelkamp soil auger hammered into the ground to a depth of ~60-70 cm or until strong resistance was encountered. Cores were sub-sampled into ~10 cm lengths in the field using a cleaned steel knife and transferred into plastic ziplock bags. Samples of known volume for determination of dry bulk density were collected using a modified 50 ml syringe and stored in a centrifuge tube prior to analysis. All samples were transported back to MMU promptly.

Table 3.2. Vegetation communities, site stratification, and sample locations

Area	NVC community	Field Observation	stratum (sub-stratum)	Sample ID	Latitude	Longitude
South Big Marsh	MG10 *Holcus lanatus* - *Juncus effusus* rush pasture	rush pasture (*Juncus*)	MG10 (rush pasture)	1	50° 38.30’	- 3° 18.84’
				2	50° 38.24’	- 3° 18.85’
	MG7 *Lolium perenne* leys	grazed *Lolium* dominated mesotrophic grassland	MG7 *Lolium* grassland	3	50° 38.17’	- 3° 18.77’
				4	50° 38.10’	- 3° 18.76’
				5	50° 38.06’	- 3° 18.76’
	S4 & S26 *Phragmites australis* swamp & reed beds	*Phragmites* communities	*Phragmites* communities	6	50° 38.17’	- 3° 18.69’
North Big Marsh	MG10 *Holcus lanatus* - *Juncus effusus* rush pasture	heavily grazed mesotrophic grasslands	MG10 (grazed grassland)	7	50° 38.64’	- 3° 18.84’
				8	50° 38.67’	- 3° 18.84’
North Little Marsh	MG10 *Holcus lanatus* - *Juncus effusus* rush pasture	drier rush pasture (*Juncus, Lollium*)	MG10 (dry rush pasture)	9	50° 38.93’	- 3° 18.55’
				10	50° 38.93’	- 3° 18.55’
Saltmarsh			Natural saltmarsh	11	50° 37.98’	- 3° 18.62’

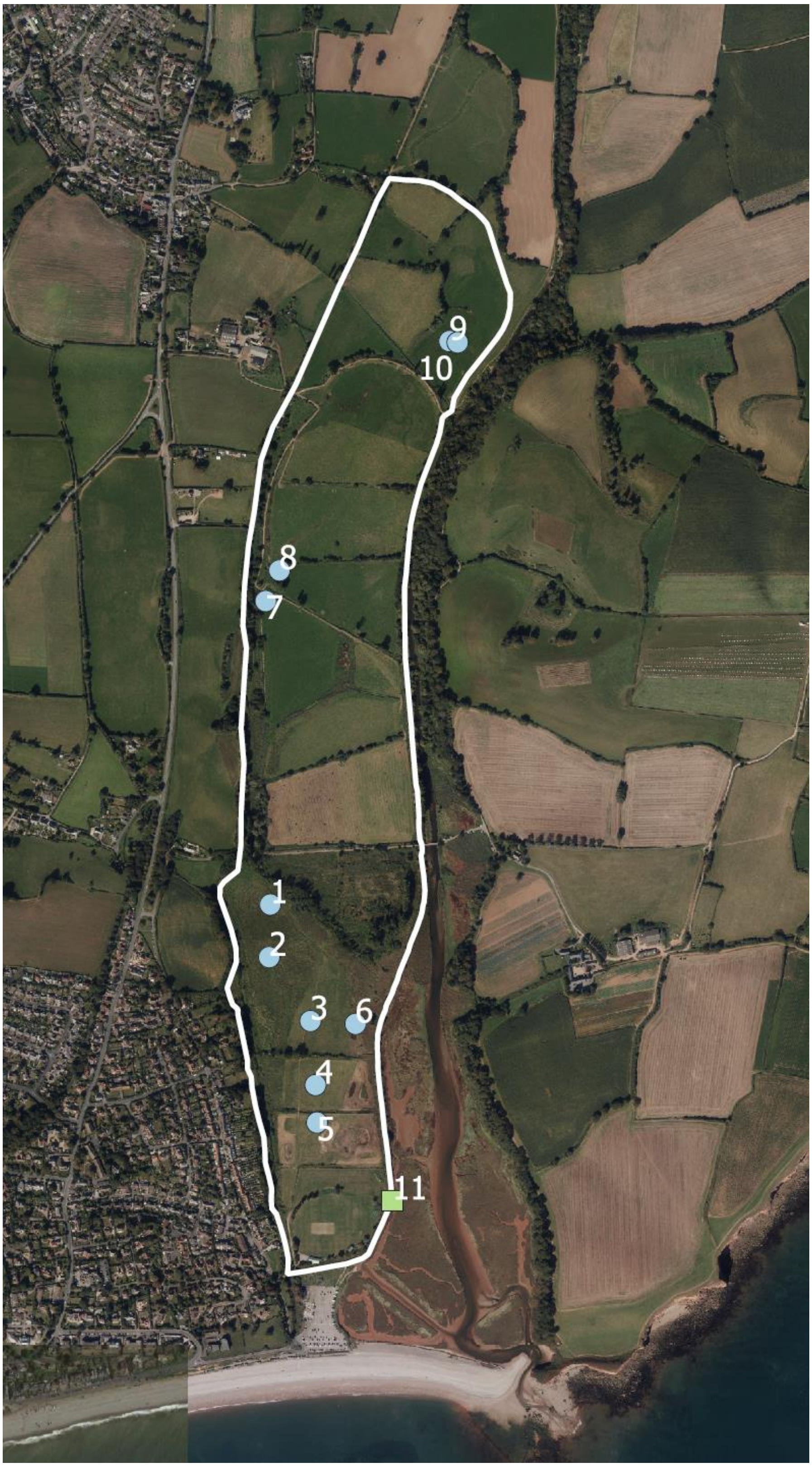

Figure 3.2. Location of samples taken for analysis of carbon content. Blue circles indicate samples taken from within the intended managed realignment, green square indicates sample taken from the adjacent natural saltmarsh. Base map: aerial imagery accessed under licence from Edina (Getmapping, 2018).

Figure 3.3. Photographs of sampling locations (1-10) at Lower Otter prospective realignment site.

Soil carbon analysis

On arrival at MMU, core subsamples were immediately transferred to aluminium trays and dried in an oven at 60°C for approximately 72 hours. After drying, the subsamples were broken up and large items (e.g. stones and twigs) were removed, before homogenising by grinding to a fine powder using a pestle and mortar.

Total carbon (TC) contents were measured on the dried, ground samples by elemental analysis using a Vario EL Cube (Elementar, Germany) instrument. A Certified Reference Material (CRM; Elemental Microanalysis Ltd Soil Standard B2184) was included (in triplicate) in all instrument runs and showed good agreement with certified values (n=6, measured %C = 2.25±0.06, certified %C = 2.31±0.06). One unknown sample (Core 10, 62-72 cm section) was also analysed in triplicate, indicating an analytical precision (relative standard deviation) of ±5% (n=3, %C = 7.58±0.39).

Dry bulk density of the surface soils was determined by drying the samples of known volume to a constant weight at 105°C.

3.3. Results

Soil total carbon content of different land use types

Table 3.3 presents bulk density, total carbon (TC) contents and densities for surface soils (0-10cm) at sampling locations on the Lower Otter prospective realignment and adjacent natural saltmarsh, while Figure 3.4 presents the downcore profiles of soil TC contents.

Soil bulk density varied considerably across the sites, from 0.34 $g.cm^{-3}$ in the wetter rush pastures of the South Big Marsh to 1.1 $g.cm^{-3}$ in the grazed grasslands of the South and North Big Marshes.

The TC content of surface soils ranged from 4.6 to 16.6 weight % (wt%), with the highest values observed in the South Big Marsh rush pasture and *Phragmites* dominated areas, followed by the drier rush pasture of the North Little Marsh, and lower values generally observed in grazed areas.

Downcore profiles of cores #1-6 from the South Big Marsh (Figure 3.4 panels (a)-(c)) all showed higher carbon contents in the surface soils (where plant roots and decaying organic matter are highly concentrated), declining to relatively consistent values of ca. 0.9±0.3 wt% at depths of ca. 20-30cm., and with some indication of a deeper surface layer in the more carbon rich soils of the MG10 rush pasture (Figure 3.3(a)) and *Phragmites* communities (Figure 3.3(c)) compared to the lower carbon soils of the grazed MG7 *Lolium* grassland (Figure 3.3(b)). Downcore profiles in the North Big and Little Marshes showed more variation in carbon content with depth (Figure 3.3(d) and (e)), with two cores showing carbon contents at depth similar to, or in excess of, those observed in the surface soils (North Big Marsh #7; North Little Marsh #10). It is also noted that the downcore profile of carbon content in core #10 from the North Little Marsh most closely resembles that observed in the adjacent natural saltmarsh, declining to minimum values at ca. 25-30cm depth before increasing again below this.

*Table 3.3. Sediment bulk density, carbon content, and carbon density for surface samples from across the Lower Otter prospective realignment. Unless otherwise stated, carbon contents are for 0-10cm depth. *Bulk density for Sample #2. ^Average of 2 measurements.*

Area	Stratum (sub-stratum)	Sample ID	bulk density (g/cm^3)	carbon content (wt %)	carbon density (mg/cm^3)
South Big Marsh	MG10 (rush pasture)	1	0.34*	14.17	47.5
		2	0.34	16.56	55.5
	MG7 *Lolium* grassland	3	1.09	6.23	67.9
		4 (0-7cm)	0.97	5.67	55.2
		5 (0-6 cm)	1.07	4.63	49.4
	Phragmites communities	6	0.47^	15.06	71.5
North Big Marsh	MG10 (grazed grassland)	7 (0-6 cm)	1.12	6.54	73.4
		8 (0-8.5cm)	1.09	6.72	73.0
North Little Marsh	MG10 (dry rush pasture)	9 (0-7cm)	0.76	12.54	95.8
		10 (0-12 cm)	0.86	6.55	56.5
Saltmarsh	Natural saltmarsh	11	0.69	5.47	37.9

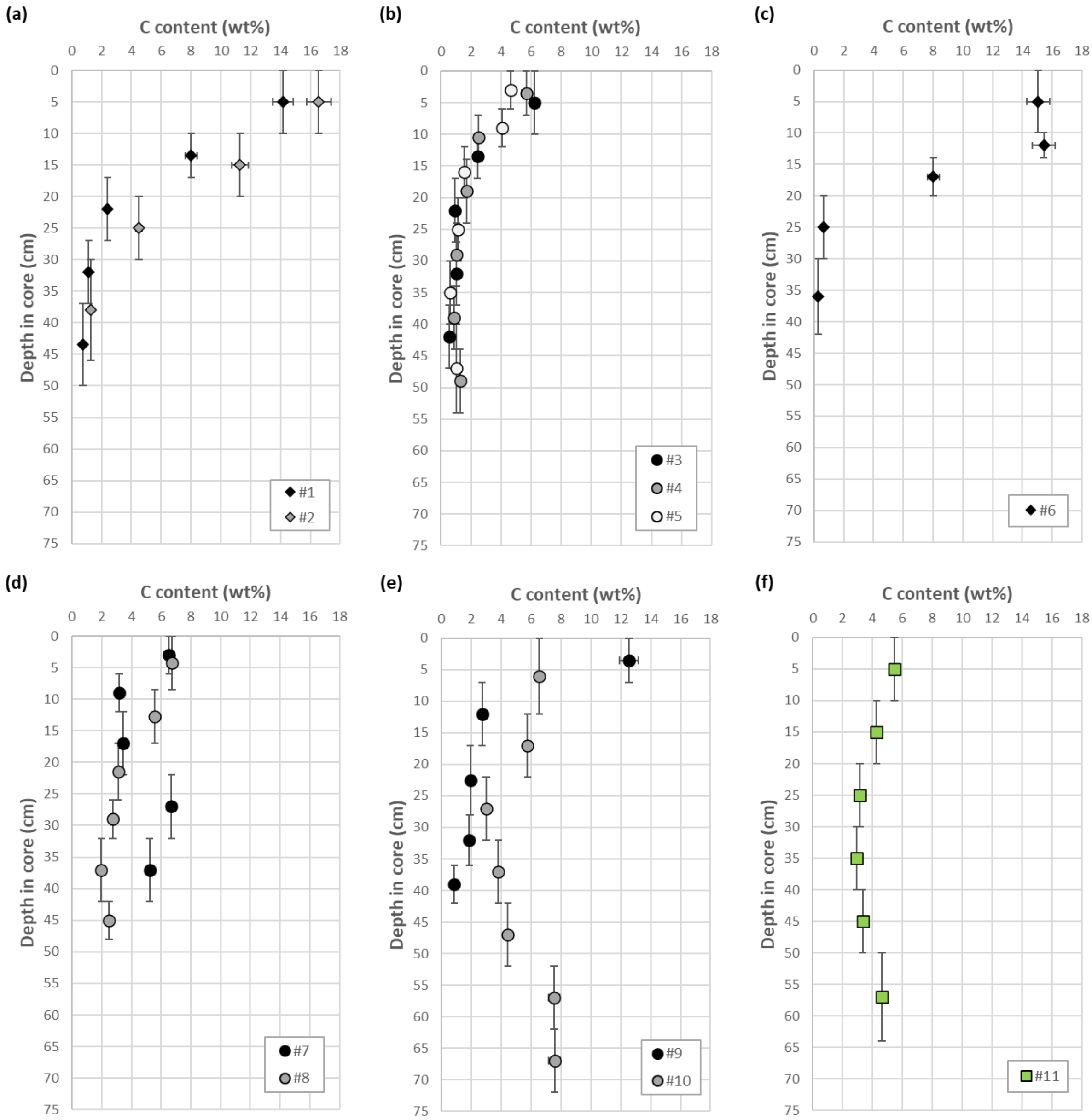

Figure 3.4. Downcore profiles of carbon content (wt% = weight %) at locations within the Lower Otter prospective realignment and adjacent natural saltmarsh. (a) MG10 (rush pasture (b) MG7 Lolium grassland (c) Phragmites communities (d) MG10 (grazed grassland) (e) MG10 (dry rush pasture) (f) Natural saltmarsh.

Estimated carbon stored in soils

A first order estimate of the total carbon within soils across the Lower Otter prospective realignment site is presented in Table 3.4 below. The estimate is based on the primary NVC based stratification of the site (as this is the basis on which the areas covered by each vegetation type has been determined), where we highlight that the apparent differences observed (both in the field, and in bulk density and measured carbon contents) across the MG10 stratum introduces a relatively high degree of uncertainty. Measured carbon contents were converted to carbon densities using the surface dry bulk density measurement for each core, before calculating an average carbon density for each depth interval within each stratum to a total depth of 50cm. An average across all cores collected in MG10 and MG7 grassland was applied to “Other mesotrophic grassland communities

(MG)" stratum, while the carbon densities for the core collected in the *Phragmites* dominated communities was applied to "Other swamp communities (S,M)".

The estimated carbon per square metre of soil to 50cm depth was highest in the MG10 rush pastures (17.3±5.0 kgC m^{-2}), where this reflects the higher carbon contents observed at depth in the northern parts of the site. The lowest estimated carbon per square meter was associated with the *Phragmites* communities (7.9±2.5 kgC m^{-2}), reflecting both the low bulk density of these wetter soils, and the low carbon contents observed below ca. 20-30cm in the southern parts of the site (where this core was collected).

Applying these carbon per unit area estimates to the area covered by each stratum gives an estimate of 8,529±413 tC contained in the upper 50cm of soils within the Lower Otter prospective realignment site (Table 3.4). For comparison, a non-stratified second estimate was also made using the average carbon densities for each depth interval across all cores collected. This gave good agreement to the stratified estimate at 8,630±268 tC (Table 3.4). It is noted that this estimate is for total soil carbon, where this includes both organic and inorganic carbon as carbonates. Given the underlying geology of the Lower Otter floodplain is sandstone, with most soils comprising find sand and silt alluvial deposits (Haycock, 2009), any carbonate contribution is likely relatively low.

During site construction, the organic carbon in any disturbed soils (e.g. during excavation of a creek network) is at risk of remineralisation, which would represent a source of CO_2 to the atmosphere. Following reinstatement of tidal inundation, the carbon contained in these agricultural soils may be stored, or may be vulnerable to remobilisation, depending on site conditions, and should be monitored.

Table 3.4. Average total carbon density by depth interval and total carbon to 50cm depth for soils within the Lower Otter prospective realignment site.

Stratum	**Area**	**Total carbon density of depth interval (mean ± SD, kgC m^{-3})**				**Total carbon to 50 cm depth**	
	ha	**0-10 cm**	**10-20 cm**	**20-30 cm**	**30-50 cm**	**kgC m^{-2}**	**tC**
MG10 *Holcus lanatus* - *Juncus effusus* rush-pasture	35.3	62.5±19.9	39.1±14.6	28.9±22.4	21.0±18.4	17.3±5.0	6,089±310
MG7 *Lolium perenne* leys	12.4	54.0±10.4	20.9±5.0	10.8±1.2	9.2±2.7	10.4±1.3	1,291±69
S4 & S26 *Phragmites australis* swamp & reed beds	3.6	71.5	55.7±25.0	3.1	1.3	7.9±2.5	284±44
Other mesotrophic grassland communities (MG)	5.1	59.4±17.0	31.8±14.6	23.5±20.2	16.6±15.5	14.8±4.3	754±257
Other swamp communities (S,M)	1.4	71.5	55.7±25.0	3.1	1.3	7.9±2.5	110±44
Sum	57.8						8,529±413
All communities	57.8	60.4±16.6	35.8±17.8	21.7±20.2	15.7±15.5	14.9±4.4	8,630±268

4. Estimating the carbon storage potential of the managed realignment

The carbon that can potentially be stored at a managed realignment depends on the volume of sediment that can build up (sediment accumulation) and the volumetric concentration of the carbon in the accumulated sediments (carbon density). Recorded sedimentation rates vary greatly between and within managed realignment sites. Locally within sites, rates of over 100 mm y^{-1} (Medmerry, Dale et al. 2017; Paull Holme Strays, Clapp 2009) have been recorded, while at some regulated tidal exchanges, rates are less than 1 mm y^{-1} (South Efford, Masselink et al. 2017). This variation in sedimentation rate results in substantial differences in the overall rate at which carbon is stored (Mossman et al. 2022). However, restored saltmarshes are ultimately anticipated to accrete to the levels of natural saltmarsh, i.e. mostly between the levels of mean high water neap and mean high water spring tides. So sites with low sedimentation rates might eventually be large carbon stores even if they take a long time to accrete.

Sedimentation rates are determined by a number of factors, including the suspended sediment load in the tidal waters that reach the site, with higher concentrations leading to faster rates of accumulation, and the frequency and duration of tidal inundation. The Lower Otter managed realignment will have relatively free connection with the estuary (i.e. will be breached, not regulated tidal exchange) and so the frequency and duration of tidal inundation will be determined by the elevation of locations within the site relative to the tidal frame. Lower elevations will initially experience more inundation and thus more sediment deposition, but as they increase in elevation sedimentation rates will slow.

In this chapter, we assess the long-term carbon storage potential of the Lower Otter managed realignment site by assessing the potential for total sediment accumulation and coupling this with carbon density values from adjacent natural saltmarsh (Chapter 3) and published values from Steart managed realignment, Somerset (Mossman et al. 2021).

4.1. Methods

Potential sediment accumulation

LiDAR images covering the site (DTM, 1m resolution) taken on 18 September 2020 were obtained from Defra (data.gov.uk) under Open Government Licence, version 3.0. Images were processed in R v4.02 using the package *raster*. Downloaded tiles were first mosaiced to form a single raster image. This raster was clipped by the site boundary (manually created in QGIS), and subsequently clipped to the level of the highest astronomical tides at the closest port, Exmouth Approaches (HAT, 2.36 m ODN, Admiralty tide tables). This resulted in an area of 60.0 ha used in the sedimentation calculations. Note this is larger than the estimated 55 ha of intertidal habitat to be created by the site, with differences resulting from small differences in the boundary of the site.

To quantify the potential for sedimentation, the difference in elevation between the image (current pre-restoration elevations) and the level of HAT was calculated for each pixel in the clipped images, giving the maximum potential depth of sediment that could accumulate at each location. Differences were also calculated between the current elevation and the median elevation of the adjacent natural marsh (1.90 m; Figure 4.1, dashed line), as an alternative for the maximum potential elevation that could be reached. The value for the HAT at Exmouth Approaches (2.36 m ODN) corresponded well with the 97.5th percentile of elevations from natural marsh (i.e. excluding the 5% most extreme upper and lower values; 2.34 m ODN), indicating that Exmouth Approaches is a suitable indicator of tidal levels, despite its distance.

The differences in elevation were then multiplied by the number of pixels in the raster image and the area of each pixel (1 m^2) to give the total volume of sediment that could potentially accumulate. It is important to note that these estimates assume that the site would fill with sediment to a level plain and the volume of drainage channels/creeks has not been excluded, and thus estimates will be in excess of actual accumulation and should be viewed as an upper bound/limit on the potential accumulation.

Sediment volumes were coupled with published values for sedimentation rates to estimate the length of time required to accumulate that volume of sediment. Suspended sediment loads are low in the Otter (Uncles et al. 2002) and so it is anticipated that rates of sediment accumulation may also be low. We therefore used published sedimentation rates from Tollesbury managed realignment, Essex, which are 14.7 mm y^{-1} (Garbutt 2018).

Organic carbon accumulation potential

Analysis of natural saltmarsh and mudflat sediments from around the UK indicates that organic carbon accounts for between ca. 50% and 80% of the total carbon content (Mossman et al. 2021 and unpublished data). Applying this indicative range to the adjacent natural marsh at Lower Otter gives an estimated organic carbon density of ca. 19-30 kg m^{-3}. We have calculated the range in potential organic carbon accumulation (tonnes organic carbon) by multiplying sediment volume by sediment carbon density, using 19 and 30 kg m^{-3} as the lower and upper ranges for sediment carbon density.

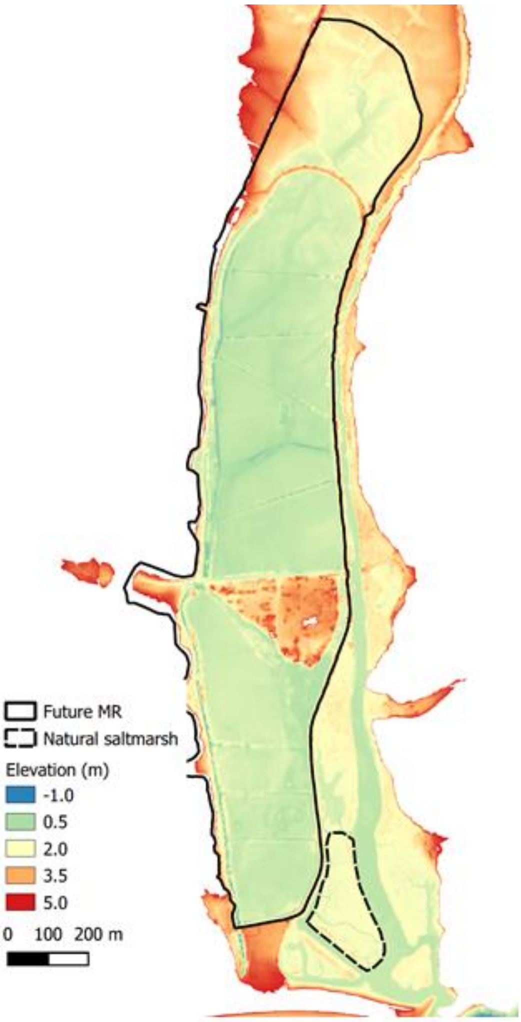

Figure 4.1. Study area boundary and elevation derived from Lidar imagery flown on 18 September 2020. Areas below -1.0 m ODN and above 5 m are masked (white areas).

4.2. Results

The median elevation of the pre-restoration managed realignment was 1.13 m (interquartile range: 0.94-1.40 m), and so most of the site is currently above the level of MHWN (0.96 m) but some way below the level of MHWS (2.16 m) and the median elevation of the adjacent natural saltmarsh (1.90 m) (Figure 4.2a).

The mean difference in elevation between the pre-restoration MR site and the median elevation of the natural marsh is 0.65 m, resulting in a potential sediment accumulation volume of 418,163 m^3, assuming the site fills to a level plain. If the sediment accumulation rate is taken to be 14.7 mm y^{-1} on average across the site, it will take 44 years to accumulate this sediment. If the site were to accumulate to the level plain around HAT (2.36 m), the mean depth of new sediment would be 1.06 m, resulting in a volume of 680,881 m^3. The latter value represents the maximum possible sediment accumulation (at current sea levels).

Sediment accumulation rates would likely be fastest in the central parts of the site that are currently at the lowest elevations. The northern parts of the site will not experience much sedimentation at current sea levels (Figure 4.2b).

If the MR site accretes to the median elevation of the natural marsh, it is estimated to accumulate approximately 8000-12,500 tonnes of organic carbon (Table 4.1). If the maximum possible level is reached (HAT), then approximately 13,00-20,000 tonnes of organic carbon could be accumulated.

Table 4.1. Potential sediment accretion and resulting carbon accumulation in the MR. Sediment accretion estimated assuming the site reached a level plain. Potential organic carbon accumulation is calculated assuming the accreted sediment contains 19 kg m^{-3} (low density) and 30 kg m^{-3}(high density) (values from Mossman et al. 2021 and unpublished data). Sediment accumulation rate was assumed to be 14.7 mm y^{-1} (value for Tollesbury, Garbutt 2018).

Accretion limit	Final marsh plain (m)	Mean depth of new sediment (m)	New sediment volume (m^3)	Potential C_{org} accumulation (tC)	
				Low C density	High C density
Median elevation of natural marsh	1.90	0.65	418,163	7,945	12,545
HAT	2.36	1.06	680,881	12,937	20,426

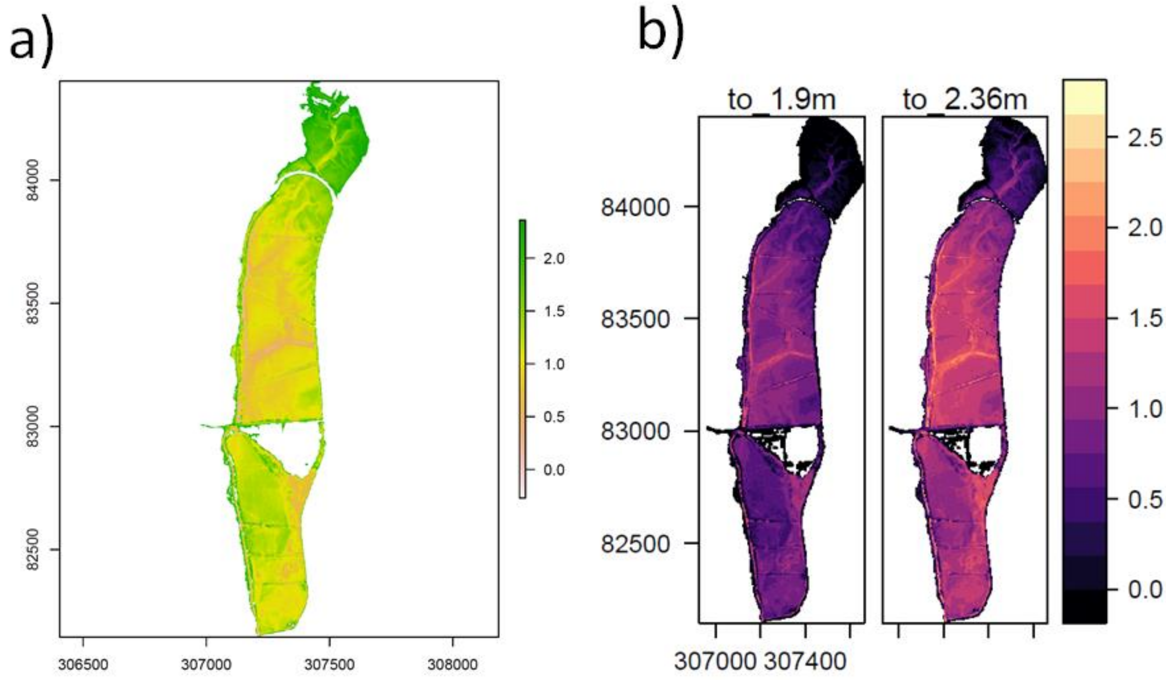

Figure 4.2. a) Area of the site used in analysis of potential sediment accumulation (i.e. area below 2.36 m) showing pre-restoration elevations. b) Potential depth of sediment (m) that could accumulate at each location if the site is assumed to accrete to a level plain at (left) the median elevation of the adjacent natural marsh or (right) the elevation of highest astronomical tide.

4.3 Discussion and limitations

We have estimated that a total of c. 8,000 – 20,000 tonnes (29,000-74,000 tonnes of CO_2e) of organic carbon could be accumulated on the Lower Otter managed realignment site in the sediment that accretes after the restoration, and that it could take approximately 44-72 years to accumulate this sediment. These values are estimated with numerous assumptions, which are discussed below, and as such should be treated with caution.

We assumed a sedimentation rate of 14.7 mm y^{-1} from Tollesbury managed realignment. Published sedimentation rates within other managed realignments range from 4 mm y^{-1} to >200 mm y^{-1} (e.g. Clapp 2009, Dale et al. 2017, Mossman et al. 2021), so the value used is towards the lower end of the potential range. However, the sediment load of the incoming waters is a key factor in the sedimentation rate. The suspended sediment load has previously been measured in the Otter as relatively low at 1.0 mg l^{-1} of suspended particulate matter, and markedly lower than estuaries such as the Humber 27,300 mg l^{-1} (Uncles et al. 2002) and lower than that of the Blackwater Estuary, Essex, where Tollesbury is located (66 mg l^{-1}; Emmerson et al. 1997). Sedimentation rates could therefore be slower at the Lower Otter restoration site.

We have assumed that the restoration site will fill with sediment until it reaches a level plain, i.e. we have not allowed for the volume occupied by creeks or channels and thus these values overestimate the total sediment that will accumulate. In contrast, the estimates are based on the current levels of the natural marsh and of highest astronomical tides, and do not consider sea level rise. Future sea level rises will increase the frequency of inundation and, as long as sediment is available, the area may accrete in pace with sea level rise beyond the currently estimated levels (Schuerch et al. 2018).

Pre-restoration elevations, and therefore changes in elevation due to sedimentation, were obtained from LiDAR. Vertical errors in LiDAR can arise from instrument error (usually c. 0.05 m) and from obscuration of the ground surface by vegetation or ponded water. Trees and hedges would intercept LiDAR beams well above the ground, although these features should normally be removed by processing.

We have used a range in organic carbon density (19-30 kg m^{-3}) based on the measured total carbon density of the adjacent natural marsh and our measurements of organic carbon as a percentage of total carbon from other UK saltmarshes (Mossman et al., 2021 and unpublished data). These are not exhaustive bounds on the range of possible carbon densities and sampling at more sites will give a better idea of the range of carbon values and factors that determine the variation. Preliminary data suggest that more vegetated sites tend to have higher sediment organic carbon densities, and there is variation between vegetation communities (unpublished data), so the colonisation of vegetation at the Lower Otter will be valuable to monitor.

We have estimated the total amount of organic carbon that could be accumulated, but it is also important to consider the source of that carbon when determining the net benefit of a new carbon sink. The organic carbon within saltmarsh sediments comprises a mixture of autochthonous carbon (produced in situ) and allochthonous carbon (transported onto site). Net carbon benefits are assessed relative to a baseline scenario, where only new carbon stores should be accounted for. While all autochthonous carbon can be considered a new sink, a portion of the allochthonous carbon would likely have been stored (in other coastal and marine sediments) in the absence of the Lower Otter Restoration Project and should not therefore be included in any claims regarding the net carbon benefit created by the site. Stable isotopes of carbon (and nitrogen) and the elemental ratio of carbon:nitrogen are often used to infer the source of organic carbon in a mixed sediment.

Likewise, our estimate only considers the organic carbon stock contained with site sediments and does not account for any greenhouse gas emissions (methane or nitrous oxide) from the newly created wetlands. Such emissions would need to be measured or estimated (based on proxy measurements, models, or values published in the academic literature) when determining the net carbon benefit of the site, particularly given the presence of *Phragmites* communities at the site.

5. Future monitoring recommendations

As noted in the introduction, in addition to flood risk benefits, the creation of new saltmarsh and mudflat may also aid climate mitigation through carbon storage benefits. This section presents a strategy for the ongoing monitoring of carbon storage within the Lower Otter managed realignment following flooding of the site in order to allow this potential benefit to be quantified.

The aims of post-breach sampling are:

- To quantify the overall amount and rate of carbon deposition across the site
- To identify differences in carbon deposition between the pre-cursor land use types
- To characterise short and medium term trends in carbon deposition location and rates

The sampling guidance outlined below is built on that in the "Coastal Blue Carbon" handbook (Howard et al., Eds., 2014). This resource is highly recommended as a thorough grounding in blue carbon sample collection and analysis. The key requirements for the sampling strategy are:

- Site stratification to ensure differences in sedimentation and carbon between vegetation types are captured
- Robust estimate of sedimentation across the whole site
- Measurement of organic carbon content of newly deposited sediment and agricultural soils

Each of the strata (vegetation types) identified for pre-breach sampling would ideally be monitored separately, although in due time it is likely that the carbon densities in the new sediment will be driven more strongly by the intertidal vegetation that colonises rather than the pre-breach vegetation.

5.1. Site stratification and sampling locations

We recommend that where possible the pre-breach sampling strategy (Chapter 3) is maintained, to allow changes before and after the breach to be quantified. The current sampling (cores 1-10 on the intended managed realignment) are well dispersed over the dominant pre-restoration vegetation communities (Chapter 3). They are also fairly well stratified by current elevation (Figure 5.1). We would recommend therefore that monitoring continues at these locations where possible. However, changes to the sampling regime may need to be made (e.g. due to access issues), so to facilitate a continued programme of assessment, we have produced a map of discrete elevation strata that can be used to identify alternative areas. The five elevation strata are:

- below the level of mean high-water neap tides (MHWN, 0.96 m at Exmouth Approaches) and thus anticipated to be mudflat in the short term
- between MHWN and half-way between MHWN and mean high-water spring tides (MHWS, 1.56 m), anticipated to be colonised by low to mid saltmarsh vegetation
- between half-way between MHWN and MHWS (1.56 m) and MHWS (2.16 m), anticipated to be colonised by mid to high saltmarsh vegetation
- between MHWS and HAT (2.36 m), anticipated to be colonised by high saltmarsh and transitional vegetation
- above HATs and thus anticipated to remain/be colonised by terrestrial vegetation.

Sampling locations 7 and 8 are at low elevations (below MHWN) and so are anticipated to experience the highest initial rates of sedimentation. It is possible that accessing these sites may become more difficult or unsafe due to excess soft sedimentation. Due to practical and safety issues, there is a tendency for a bias in field sampling of sedimentation rates against areas with the highest rates, e.g. Oosterlee et al. (2020) were forced to abandon sedimentation pins due to high sediment accretion. If sites 7 and 8 are inaccessible, we recommend selecting alterative, safely accessible areas where higher sedimentation rates are expected are considered for monitoring.

Continued sampling of other existing locations may not be possible due to issues with accessibility, practicality or safety, and it is not always possible to predict areas that will become inaccessible. Where such sites are identified prior to breaching, we suggest simply replacing locations with sites with similar existing vegetation and elevation. Where such sites become apparent after breaching, we recommend identifying similar accessible locations, but on installation of the sediment pins (see Section 5.2) the amount of the new sediment already accreted should be quantified. This can be done by taking a core down through the sediment and into the agricultural surface. The agricultural horizon is usually clearly identifiable as a change in texture, colour and there may be a layer of rotting vegetation.

The areas anticipated to be the highest marsh (above MHWS, Figure 5.1 red) have not been captured. Although these areas are not anticipated to experience much sedimentation, it is likely that rarer transitional vegetation communities may colonise these areas and these are communities that are generally less well researched; additional sampling of carbon here may therefore be of interest.

As the original sampling locations on the proposed managed realignment (0-10) are fairly well stratified by elevation, we can anticipate that the vegetation communities that colonise will differ, although due to the current elevation there will be a dominance of low-mid marsh communities (e.g. likely dominated by annuals such as *Salicornia* spp. and *Suaeda maritima*, *Spartina anglica*, *Aster tripolium* and *Puccinellia maritima* (Mossman et al. 2012). Vegetation communities influence the carbon of the sediments beneath them (Ford et al. 2019), and so if different vegetation communities form that are not captured by the current sampling locations, we recommend taking additional samples or re-stratifying by those vegetation communities.

Figure 5.1. Elevation strata for use in designing future sampling. Bands relate to the level of tides at Exmouth Approaches. 0.96 m = mean high water neaps, below which vegetation is not anticipated to colonise without sedimentation, 2.16 m = mean high water springs.

Scaling up assessment of carbon stored in newly laid sediments

Site-wide assessments of sedimentation can be made by quantifying the difference between pre- and post-restoration lidar images, validated by in situ measurements of sedimentation (see Section 5.2). These estimates of sedimentation can be coupled with the average (± variance) carbon density of the new sediments to provide an estimate of site-wide carbon storage. However, if the carbon density is very variable between the different vegetation communities then ideally the average sedimentation and carbon density of the different communities would be combined with the areal coverage of those dominant vegetation communities. It is necessary to have at least three samples of carbon density per strata if uncertainty in within-strata carbon density is to be calculated and propagated into site-wide estimates.

The dominant vegetation communities would normally be defined by the dominant species and we would recommend defining the five or six most common communities (which may include *Phragmites*, *Spartina anglica*, *Puccinellia maritima*, unvegetated). The abundance of vegetation communities are often assessed by quadrat surveys. However, on a site of this size and since it is information on the areal extent that is required for scaling up assessments, we would recommend walk-over Phase 1 type surveys to map the extent of the broad (non-NVC) communities. Classification of vegetation communities can be done from satellite and/or drone aerial imagery, although ground-truthing observations will be needed to calibrate and validate classifications. These remote sensing methods can also only distinguish the broadest communities; field surveys will be needed for any finer classification. As described above, we recommend checking the existing sampling sites against the developing vegetation communities to ensure that vegetation communities are adequately sampled, and adding additional sampling points where necessary.

The frequency of site surveys will reflect the purpose of the analyses and the speed at which the site is changing. For example, if there is interest in studying the evolution of the created habitat then relatively frequency sampling (i.e. every one to two years) would be desirable to monitor changes in vegetation and sedimentation in the early stages of development; sampling may be less frequent when changes in the site are relatively slower. However, if the main purpose of monitoring is to evaluate the carbon stored within the site, then less frequent sampling would suffice, e.g. every 3-5 years in the early years when the changes are likely to be faster.

5.2. Monitoring sediment accretion and erosion

The volume of sediment deposition, in addition to the density of carbon in that sediment, is a key determinant in the carbon accumulation on the managed realignment. The sediment load in the Otter Estuary is low (see Chapter 4) and so we anticipate that sedimentation will be fairly slow. While Lidar provides a cost effective and efficient way of monitoring changes in elevation (i.e. sedimentation rates) across the *whole* site, noise due to instrument error and vegetation are relatively more significant when sedimentation rates are low (i.e. signal is low relative to the noise). Airborne lidar data are available periodically courtesy of the Environment Agency at data.gov.uk, and these flights are useful for looking at longer term (3-5 y) accretion, but are only useful to look at yearly sedimentation where sedimentation rates are very high (i.e. >0.1 m). Difficulties with deriving sedimentation rates from lidar also occur when the vegetation is dense, or where there are large areas of standing water e.g. pools, lagoons or in creeks if lidar data are collected when the tide is in.

Lidar data at a site-scale can be collected with drones but this is relatively expensive and would only be recommended if there was a very long period between airborne flights.

Ground truthing of lidar-derived sedimentation rates is essential everywhere, particularly where rates are low. These ground measurements will be able to detect small changes in elevation and so are critical for quantifying shorter term (weeks to several years) changes, or changes at higher elevations that are not detectable with lidar.

There are a number of ways to measure changes in elevation (accretion and erosion) on the ground. Suitable techniques should allow change to be monitored over periods of weeks to decades and most broadly use a fixed location, embedded ideally prior to restoration, from which changes in elevation are measured. Measures of accretion can be obtained from sediment cores that are taken down through the newly created sediment and into the pre-restoration soils. This horizon between pre and post-breach soil is usually visible on cores as a change in texture, colour or presence of a layer of rotting vegetation, and the depth of newly accreted sediment can thus be measured. However, cores cannot measure erosion, the agricultural horizon may not always be distinct, and compression of the newly accreted sediment can happen during coring. We therefore recommend also making dedicated measurements of elevation change using the techniques described below.

Nolte et al. (2013) provide a review of the pros and cons of each technique for measuring elevation change, and from which Figure 5.2 is taken. The main techniques are described below:

- Horizon markers are a layer of material (e.g. brick dust, sand or often feldspar clay) easily distinguishable from the sediment that is deposited on top. The locations are returned to, and cores taken down through the horizon marker and the depth of new sediment measured. They are simple and cost effective methods, but repeated measurements over time deplete the horizon marker and so cannot be used indefinitely. The markers are vulnerable to washing away with frequent inundation, especially where the area is unvegetated and so they may be less suitable for newly established managed realignments. Sediment erosion cannot be measured with horizon markers.
- Sediment erosion plates are a hard plate (e.g. tiles) are attached to a rod that is driven into the ground. Sediment accretion and erosion can be assessed by measuring the distance from the sediment surface to the top of the plate. They are simple, cost effective methods of measuring accretion and erosion but can be undercut in areas of faster flow.
- Sediment pins are pipes or rods that are driven fairly deep into the ground and a permanent level is marked near the top of the pin. The change in the distance from that level to the surface of the sediment is measured to provide the measure of sedimentation. These are very cost effective (PVC drainage pipe is ideal), so multiple pins can be established and are thus ideal for ground truthing lidar data. A key draw back on the previous two techniques is that sediment pins that are buried can no longer be used, so the depth of the potential sediment needs to be anticipated before installation to ensure the pins are long enough. See http://www.tidalmarshmonitoring.net/monitoring-methods-sediment-pins.php for a useful description of methodology.
- Sediment erosion tables are portable mechanical levelling devices used for measuring relative elevation changes. They always measure the same exact location and so are accurate and precise but are very expensive to install and sedimentation is only measured in the (usually one or two) areas they are installed.

We recommend the use of sediment pins at multiple locations (the expected low sedimentation rates mean that this method should be successful), ideally corresponding to sediment sampling collections (see Section 5.1), combined with analysis of lidar for measuring the sedimentation at the Lower Otter.

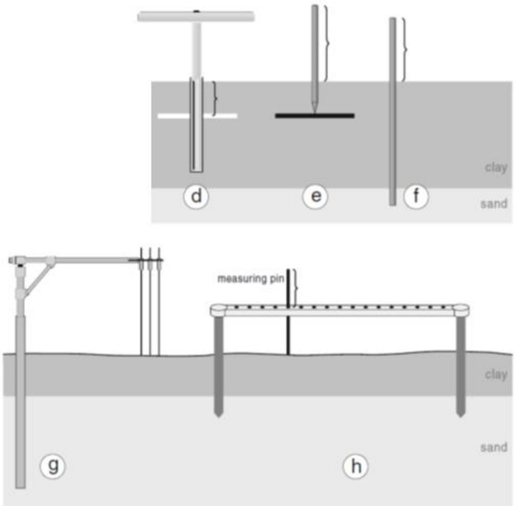

Figure 5.2. Schematic illustration of equipment for (d) marker horizon, (e) sedimentation plate, (f) 'erosion' pin, (g) sediment erosion table (SET), and (h) sediment erosion bar (SEB). Modified from Nolte et al. (2013)

5.3. Analysis of organic carbon content

To quantify how soil carbon content changes with depth (and thus understand changes over time as new sediment is deposited) it is necessary to take a core down through the sediment. However, if only the carbon content of newly deposited surface sediment is of interest, then a small surface same will surfice (e.g. with a trowel or bulb corer). If sampling occurs very reguarly (multiple times a year), then successive surface samples will allow changes in carbon to be tracked. However, coring down through sediment allows these changes to be captured with much less frequent sampling.

There are number of different corers and augers available, and cores can potentially be made using custom modified drain pipes. We used an Eijkelkamp soil auger for pre-breach sampling. The core should be hammered into the ground to a depth of ~60-70 cm, or until strong resistance is encountered, and then the corer should be lifted out taking care to keep the soil sample intact. Factors to consider when taking subsamples of the core are the temporal resolution of changes to be captured (smaller sections gives finer resolution), the volume of sample needed for analysis (~5g is sufficient), the number of samples that can be analysed, and any obvious horizons (subsamples should not cross horizons). Often subsamples ~10 cm in length are adequate to give an idea of changes in sediment carbon. Subsamples should be cut from the core in the field using a cleaned steel knife, and then transferred to plastic zip-locked bags (or a glass jar if isotope analysis is to be conducted and plastic contamination thus needs to be avoided, see below).

Sediment bulk density should be measured in every subsample of the core used for analysis. The key for bulk density measurement is taking a sample of known volume. This can be done by using a modified syringe (with the fine tip cut off). The volume of the sample should always be recorded at the time of sampling as it is needed to calculate bulk density following laboratory analysis.

Soil organic carbon content can be measure using loss on ignition (LOI), or elemental analysis in combination with sample decarbonation. We strongly recommend the latter as LOI is prone to overestimating organic carbon content due to loss of structural water (Hoogsteen et al. 2015).

5.4. Other monitoring

In addition to the monitoring recommended above, the following assessments would provide additional data towards establishing a full carbon budget of the site.

Monitoring gas fluxes from new saltmarsh sediments

Monitoring gas fluxes is important for calculating a full carbon budget. When soils are flooded during wetland creation this can lead to the emission of methane and nitrous oxide, and emissions of these greenhouse gases should be quantified to inclusion in a full carbon budget. There is limited data from the UK for saltmarshes and especially manged realignments, so monitoring has the potential to advance understanding of these fluxes. Measurements can be taken using static chambers developed for use in peatland sites. However, the analysis of gas samples requires specialist equipment and expertise. If gas flux measurements cannot be taken, gas fluxes can be estimated based on the salinity profile of the site. This would require taking salinity or conductivity measurements at locations across the site, and relating them to standard values of gas fluxes (e.g. from Emmer et al. 2015). Further information on gas flux measurements can be found in pages 118-122 of the Blue Carbon Handbook (Howard et al. 2014).

Carbon emissions from construction

If it is desirable to make a robust claim of the net carbon storage benefit of the Lower otter site, it is important to ensure that carbon emissions during site construction are properly accounted for. We therefore recommend asking any contractors to monitor fuel combustion by plant used, and to record the volume of soil excavated and transported. Contractors should also be asked to record any movement of material onto and off the site, and the final destination of that material.

Origin of carbon

Organic carbon can be autochthonous (produced in situ) or allochthonous (transported onto the site). The latter will reflect a mixture of inputs from terrestrial/freshwater (e.g. plant debris, freshwater phytoplankton) and estuarine/marine (e.g. macroalgae wrack, benthic diatoms) origin. Allochthonous organic carbon needs to be accounted for separately for a full carbon budget (and this may be a requirement when generating carbon credits). Methods for accounting for allochthonous carbon are not consistent between carbon accounting methodologies. Some suggest deducting all allochthonous carbon from estimated carbon storage rates, others only suggest deducting mineral-associated carbon, and others allow for an estimate of total or recalcitrant allochthonous carbon. This reflects the uncertainty in determining the fate of allochthonous carbon in the absence of the managed realignment scheme, as some would likely to have been stored in the absence of the scheme. The source of carbon, and thus the likelihood of additionality, can be inferred from measurements of stable isotopes of carbon and nitrogen, the elemental ratio of carbon and nitrogen, and analysis of biomarkers.

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