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Masselink, Gerd

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EVALUATION OF SALT MARSH RESTORATION BY MEANS OF SELF-REGULATING TIDAL GATE – AVON ESTUARY, SOUTH DEVON, UK

Gerd Masselink, Mick E. Hanley, Anissa C. Halwyn, Will Blake, Ken Kingston, Thomas Newton, Mike Williams

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ABSTRACT

Salt marshes provide important regulating ecosystem services, including natural flood defence and carbon sequestration, which adds value to restoration and biodiversity offsetting schemes. This study evaluates the success of salt marsh restoration using a Regulated Tidal Exchange (RTE) system in SW England, i.e. a self-regulating tidal gate (SRT), in controlling the partial saline inundation of a 14-ha area of former salt marsh reclaimed for agriculture in 1760. A combination of (a) direct hydrodynamic monitoring of water and sediment flux and (b) repeat surveys to evaluate morphological and ecological (plants and foraminifera) changes over a 5-year period, was implemented immediately following SRT commissioning. Morphological changes were limited to the proximity of the SRT system due to limited sediment influx yielding sedimentation rates that were an order of magnitude below a nearby natural marsh. Ecological change to an ephemeral salt marsh community was only detected after 5 years of inundation cycles, with the delayed response attributed to (a) an initial limited tidal inundation due to conservative SRT settings, followed by (b) excessive inundation due to excessive rainfall and recurring SRT failure in an open position, and (c) a lack of sediment and propagule supply caused by (a) & (b) and the relatively narrow inlet pipe used in the SRT system. While the ecological response under optimum SRT settings was encouraging, the lack of perennial plants and limited foraminifera abundance demonstrated that the marsh was far from reaching natural status. We surmise that this is primarily due to inundation being more rapid than drainage leading to excessive submergence during a tidal cycle. Our study shows that the design of tidal inundation schemes requires a synergistic understanding of core ecological and geomorphological approaches to assess viability and success. We conclude that SRT can be a useful technique for intertidal habitat creation where there are significant site constraints (especially flood risk), but we need to be realistic in our expectations of what it can achieve in terms of delivering a perennial salt marsh community.

1. INTRODUCTION

Sea-level rise and an associated increased frequency and severity of storm surge events (Martin et al., 2011; Zappa et al., 2013) are together challenging the long-held view that so-called 'hard' engineering alone can protect our coasts from flooding. Instead, an integrated strategy involving natural ecosystems is increasingly held to offer the most cost-effective, sustainable, and effective form of coastal defence (Temmerman et al., 2013; Bouma et al., 2014; Hanley et al., 2014). In north-west Europe and North America, managed realignment (MR) or 'de-embankment' schemes (the deliberate flooding of land situated behind coastal defences) are commonly implemented to create new areas of salt marsh, both as compensation for habitat losses elsewhere, and to enhance flood defence and create accommodation space (French, 2006; Spencer and Harvey, 2012; Morris, 2012; Foster et al., 2013; Chang et al., 2016). This is viewed as a desirable outcome, not only to help redress the c. 50% global loss of this habitat (Adam, 2002), but also because salt marshes have a remarkable capacity to attenuate and dissipate wave energy, store flood waters, and so defend in-land areas from the worst excesses of coastal flooding (Gedan et al., 2011; Moller et al., 2014).

Effective intertidal habitat restoration, especially that of salt marsh, has, however, proven difficult to achieve. In their study of 18 MR restorations in the UK, Mossman et al. (2012a) report that managed realignment sites are typified by substantial recruitment failure (bare ground) and, where revegetation has occurred, dominance only by early successional salt marsh communities. Although to some extent these 'failures' represent the relative size and age of managed realignment sites compared with adjacent natural marshes (Wolters et al., 2005), other environmental factors may be important. Sites selected for restoration, usually for opportunistic reasons, are likely to start with physical and biogeochemical conditions very different to natural counterparts (Spencer and Harvey, 2012). Many (typically) former agricultural sites are especially difficult to restore; livestock and farm machinery cause soil compaction, reducing drainage and susceptibility to channel development, and increasing waterlogging potential (Spencer and Harvey, 2012; Chang et al., 2016). Long-term agricultural use also leads to soil shrinkage and consolidation; this reduces surface elevation and increases the amount of time the site spends under water post-breach (Crooks et al., 2002; Spencer and Harvey, 2012).

Elevation within the tidal frame is generally considered to be the pivotal factor determining the success of salt marsh establishment (Adam, 1990; French, 2006; Davy et al., 2011). The duration and frequency of tidal inundation has marked effects on propagule delivery, sedimentation, salinity and soil redox potential, and, therefore, the regeneration potential of any newly arriving species to the site (Mossman et al., 2012a, b; Spencer and Harvey, 2012). Consequently, the most significant barrier to effective salt marsh transition in managed realignment schemes is the difference in elevation between reclaimed land and adjacent, natural salt marsh (Wolters et al., 2005, 2008; Spencer and Harvey, 2012). Despite this recognition, Spencer and Harvey (2012) highlight that, while considerable attention has been devoted to monitoring long-term, post-breach shifts in plant and animal communities, there has been minimal attempt to quantify the concomitant physical-chemical changes that effect ecological transitions. There is also a surprising lack of clear demarcation of the optimum tidal inundation characteristics for salt marsh development in managed realignment schemes; an exception being Ash and Fenn (1997) who concluded, based on the Tollesbury MR scheme, that mudflat, pioneer marsh, and mature marsh is characterised by > 38, 25–38 and < 25 inundations per month, respectively. Additionally, according to Environment Agency (2003), salt marsh habitat develops where there are < 42 inundations per month and where the surface has a small gradient (1–3%).

One frequently used method in managed realignment schemes is to use Regulated Tidal Exchange (RTE) systems which enable habitat creation behind coastal defences, whilst at the same time managing flood risk (Environment Agency, 2003). RTEs are usually situated in a breach in existing seawalls or embankments and utilise structures such as tide-gates and sluices, to tightly control the amount of water entering the restored area (Wolters et al., 2005; Cox et al., 2006). As a result, the relative tidal height can be adjusted to match that experienced by local natural marshes, while allowing plant propagules and sediment to enter; both of which are vital for salt marsh establishment (French, 2006; Mossman et al., 2012b). Buoyancy-controlled systems, or self-regulating tide-gates (SRT), where a float system is adjusted to open the gate until a specified water level is reached, have the further advantage that they can replicate the spring-neap system that drives natural zonation patterns in salt marsh vegetation (Adam, 1990; Ridgway and Williams, 2011). In theory, these designs also ensure maximum control over water levels in the restored area to minimise flood risk to nearby areas (Adnitt et al., 2007). In practice however, the elevation of the tidal frame is not chosen as SRTs are often fitted to existing outfalls and the system may also be prone to mechanical problems (flotsam clogging the mechanism). In summary, the efficacy of SRT in facilitating the development of emergent salt marsh vegetation is largely unknown (cf. Beauchard et al., 2011).

Along with changes in soil structure and site elevation, SRT-imposed modification of tidal regime is one of three key disturbances to natural physical parameters imposed upon managed realignment schemes (Spencer and Harvey, 2012). As a consequence, the potential for development of a functioning salt marsh ecosystem may be reduced and with it the ability of the restoration to deliver the very ecosystem services for which it was implemented (Mossman et al., 2012a; Spencer and Harvey, 2012). To evaluate the ability of SRT to make a consistent and positive contribution to successful salt marsh restoration, it is vital that post-breach plant community development is understood within the context of the tidal environment in which it occurs (Spencer and Harvey, 2012). The principal aim of this study is to evaluate the performance of a managed realignment scheme, involving an SRT system, through addressing the following objectives: (1) quantify the tidal flooding regime; (2) describe the tidal currents and sedimentation processes; (3) describe the morphological changes; (4) describe the ecological changes, specifically in vegetation abundance and type, seed deposition and foraminifera population; and (5) investigate how the ecological transitions are associated with variation in the tidal environment imposed by the SRT system. We will argue that, at least for our study location, a SRT system might not be the best means by which to control the tidal inundation regime for salt marsh restoration.

2. MATERIALS AND METHODS

2.1 Study area

The Avon estuary is located on the south coast of Devon in south-west England (Figure **1**). It is a relatively small estuary with a total surface area of 213.5 ha, of which 146.2 ha are intertidal, an estuarine shoreline length of 19.8 km and a 7.8-km long tidal channel (Davidson, 1991). The estuary has steep-sided margins and is generally considered a ria-type (drowned river) estuary, although it does possess a sand barrier at its mouth (Masselink et al., 2009). The estuary is relatively pristine and the only human modifications comprise several wooden groynes at its mouth (built in the 1930s and now largely obsolete) and the reclamation of a 15-ha salt marsh in the upper estuary more than 100 years ago. It is the restoration of this reclaimed salt marsh, South Efford Marsh (Figure **1**), that is the subject of this study.

Figure **1** here

Ocean tides in the Avon estuary are slightly less than at Devonport (Plymouth; closest Primary Port for tidal data) and are characterised by a mean spring and neap tide range of 4.3 and 2.0 m, respectively (Uncles et al., 2007). The elevation of the high water levels during spring and neap tides at the mouth of the estuary are estimated at 2.5 and 1.2 m ODN (Ordnance Datum Newlyn, which is the mean sea level datum in the UK), respectively. The 1:50 year storm surge height along the south coast of Devon is c. 0.5 m (Lowe and Gregory, 2005).

Freshwater discharge into the Avon estuary is mainly through the Avon River, supplemented with minor contributions from small streams draining the valley slopes. River discharge is measured by an automatic gauging station at Loddiswell located c. 3 km upstream from the tidal limit (Figure **1**). A pronounced seasonal cycle in river discharge is apparent with a monthly-averaged winter discharge of just under 7 m^3/s and a monthly-averaged summer discharge of just over 1 m^3/s (Uncles et al., 2007). Peak flows during winter can exceed 40 m^3/s and minimum flows in summer are often less than 0.5 m^3/s.

Investigation of the tidal dynamics was conducted by Uncles et al. (2007) who deployed a current meter in the upper estuary in the tidal channel near South Efford marsh over a 1-week period during summer and winter in 2006. During summer, peak flood flows (0.5–0.6 m/s) were stronger than during ebb (0.3–0.4 m/s) and the salinity profile was well-mixed during flood, but stratified during ebb. During high river discharge flow in the winter however, flows in the upper estuary were almost exclusively directed seaward with peak velocities of 0.4–0.7 m/s. Significant damping of the tidal wave occurs into the estuary and for a typical ocean tide range of 3.7 m, the tide range at Bantham Harbour and South Efford was 3.0 and 1.5 m, respectively. In addition, distortion of the tidal wave was also evident with a duration of the flood phase at Bantham Harbour of only 4–5 hrs.

2.2 History of South Efford marsh

Around 1760, a large intertidal salt marsh area along the northern margin of the upper Avon estuary, known as South Efford marsh, was reclaimed and converted to pasture through the construction of a surrounding embankment. The embankment was breached in 1943 by a stray bomb and the site reverted to intertidal habitats (mostly mud and sand flats) until the breach was repaired in 1956 and the site returned to agricultural use. In 2011, the Environment Agency (EA) installed a self-regulating tidal gate (SRT) at the southern end of the marsh, utilising the existing outfall with improvements to its headwall, allowing the tide to once more flood the area. Flooding of the marsh by means of an SRT was preferred to breaching the embankment and allowing the breach to evolve naturally, because approval for the restoration scheme was contingent upon not increasing the flood risk to several properties at the back of the north-east corner of the marsh. The SRT has been operational since May

2011 and the EA expected that within 5 years of commissioning the lowest part of this area would revert back to an intertidal salt marsh environment, whilst the upper part remains supratidal grazing marsh. The aim was to generate 7 hectares of new intertidal habitat, including both salt marsh and mudflat.

The LIDAR-derived digital elevation model (DEM) of a 1.2 km x 1.2 km area encompassing South Efford marsh is shown in Figure **1**, as well as an aerial photograph. Most noteworthy is the observation that the elevation of the natural marsh to the south (z = 1.5–1.6 m ODN) is at least half a meter higher than that of the realigned marsh surface (z = 0.9–1.0 m ODN). This is due to the fact that the natural salt marsh continued to accrete after South Efford marsh was reclaimed in 1760, possibly exacerbated by soil compaction. Ignoring compaction, an average accretion rate on the natural salt marsh of at c. 3 mm/year can be deduced, which corresponds to previous estimates made using salt marsh cores from the Avon estuary (Bugler, 2006). The elevation of the embankment around the marsh site is > 3.5 m ODN, which is higher than the highest water level in the estuary at this location. Some indication of the pre-reclamation tidal creek topography is discernible in the DEM (dark blue meandering patterns) and the low area around Easting = 268600 m and Northing = 46800 m is related to a WWII bomb that struck the embankment at this location. The north-east part of South Efford marsh is significantly higher (z = > 1.5 m ODN) than the rest of the marsh. The SRT is located at the south-west end of the marsh and leads straight into the straight ditch that runs along the centre of the marsh.

2.3 Self-regulating tidal gate (SRT)

The tidal dynamics in the realigned marsh, and the ensuing morphological and ecological changes, reported here are controlled by the design and the settings of the SRT and the way through which it controls tidal exchange between the marsh and estuary. Flooding and draining of the realigned marsh occurs by mean of a concrete pipe (diameter = 0.9 m; elevation of base of pipe = 0.39 m ODN) whose connection to the river is controlled by a rotating tide gate to which floats are attached. The timing of opening and closure of the gate depends on the river water level. At low tide, the gate is closed and water leaves the marsh through a side flap if the marsh level exceeds the river level. As the tide rises, the side flap closes under pressure when the river level exceeds the marsh level; the floats cause the SRT to rotate and the aperture begins to align with the pipe, allowing water to enter the marsh. As the tide continues to rise, the SRT rotates until fully open and then gradually closes again until the river reaches a pre-determined level. The SRT is adjusted so that it is fully closed at high tide when the estuary water level. If uncontrolled, would increase flood risk to dwellings. A conservative setting will cause the gate to be closed well before high tide level is reached, while a less conservative setting will result in more delayed gate closure, or no closure at all (e.g., during neap high tide). As the tide falls, the gate will open again and more water may enter the marsh, until the point when the tide drops below the marsh water level and the marsh starts to drain again.

A very important implication of this system is that the duration over which the SRT is open, and therefore the amount of water that will flow through the pipe to flood the marsh, is controlled by how long it takes for the river to reach the pre-set gate-closure level: especially for a conservative gate setting, the pre-set water level for gate closure is reached relatively early during spring tides as the tide rises, whereas the pre-set water level for gate closure is reached relatively late in the rising tide during neap tides. Depending on the gate settings, this can mean that spring tides in the river or estuary generate lower water levels in the marsh than neap tides; in other words, a reversed neap-to-spring tidal variation can be generated in the realigned marsh. Another characteristic of the SRT system is that, frequently, the SRT was not working properly and the gate was open almost continuously. Such malfunctioning results in a progressive increase in water level and salinity in the realigned marsh, up to the point that the marsh ceases to drain. This can result in extended periods of marsh submergence which can have significant implications for the ecological development.

The SRT fitted at South Efford differs from others of a similar design in that, to accommodate the need to open and fully close again over quite a small tidal range, the gate and floats are linked by a mechanism that causes the gate to rotate roughly twice as fast as the float. It is possible to make adjustments to the gate to change to points of opening and closure, but such adjustments are made very infrequently, one of the aims of the design being that of requiring minimal intervention. The South Efford SRT also appears to be more prone to malfunctioning than others of the type. This seems to be as a result of two main factors: (1) the location (facing down the estuary and in a slight backwater) means that debris is prone to collecting around the SRT; and (2) the linkage connecting the rotating gate and the floats means that relatively small pieces of debris can affect the smooth operation of the gate.

2.4 Survey grid and monitoring programme

Pre-breach, a measurement grid was established using laser total station (Figure 2). This measurement grid has as its origin the location of the SRT, and the *x*- and *y*-axis represent the length- and width-axis of the marsh, respectively. A total of 10 across-marsh measurement transects were established (*y* =

25, 50, 100, 150, 200, 300, 400, 500, 600, 700 m) and an additional transect crossing the natural salt marsh to the south of the realigned marsh (y = -60 m). Benchmarks were established along the margin of the site and these are used for re-sectioning the total station during surveys. Ecological monitoring sites were located within this measurement grid and are also indicated in Figure **2**.

Figure **2** here

An extensive data set was collected over a 5-year monitoring period (May 2011 – May 2016) and various parameters were recorded, as discussed below. An overview of these parameters and their sampling frequency is provided in Table **1**.

Table **1** here

2.5 Tidal flooding regime

Water level was recorded at either side of the SRT using pressure sensors and salinity in the realigned marsh was monitored with a conductivity sensor deployed just inside the gate. The sensors were installed and are maintained by the EA, and data collected every 15 min. The water level inside and outside South Efford marsh is referred to as the marsh level and the river level, respectively.

The time series of the river water level was used to compute the inundation characteristics of the natural tidal flat or mudflat (TF; mean elevation z = 1.0 m ODN) and the natural salt marsh (SM; mean elevation z = 1.5 m ODN), and the time series of the marsh water level was used to compute the inundation characteristics of the realigned marsh (RM; mean elevation z = 0.8 m ODN). The following parameters were computed for every month for which reliable water level data were available (56 and 55 months out of 60 months for TF/NM and RM, respectively): (1) the number of over-tides N_{otides}, i.e., tides that inundate the RM, NM and RM surface; (2) the total number of hours that the surface is submerged $T_{sub,tot}$ (3) the maximum continuous period of tidal submergence $T_{sub,max}$; (4) the maximum continuous period of exposure $T_{exp,max}$; (5) the mean water depth over the surface h; and (6) the average rate of the falling tide over the surface dh/dt. Note that these parameters are monthly values.

2.6 Tidal currents and sedimentation

Two self-recording Acoustic Doppler Velocimeters (ADV – Nortek Vector) with external pressure transducer (PT – Druck PTX 1830) and optical backscatterance sensor (OBS – OBS-3 Downing) were deployed from 3 March to 4 May 2012 to record current velocities and suspended sediment concentrations. One of the instruments was deployed in a tidal channel located in the natural salt marsh just outside the SRT linking the Avon estuary with the realigned marsh. The other instrument was mounted above the SRT to record the flows inside the pipe through which water is exchanged between the Avon estuary and the marsh. In the afternoon of 21 March 2012, water and suspended sediment samples were collected from the natural salt marsh through pumping. Suspended sediment concentrations were used to carry out in-situ calibration of the OBS sensors. Combining suspended sediment concentrations with flow velocities enables quantification of suspended sediment fluxes, and from these potential marsh accretion rates could be estimated.

Two squares of 'Astroturf' matting (21 cm x 21 cm) with 1.5 cm plastic tufts (Lambert and Walling, 1987) were secured to the ground surface with a 30 cm steel peg adjacent to each other in the southwestern corner of every 4 m^2 quadrat on the -60, 25m, 50, 100, 150, 200 and 300 m transects. We focussed on the lower marsh area due to its greater dynamicity and sensitivity to hydrodynamic variation (Coulombier et al., 2012). Mats were collected and new ones deployed at the same stations for periods of 141–203 days from 2011 to 2014. Once collected and transported for analysis in separate, sealed plastic bags, one mat (of known mass) from each pair was used to determine dry weight (the second mat was used for studying seed deposition and germination; cf., Section **2.7**). Mats were dried for 48 hours at 55°C and reweighed once cooled to determine mass change (i.e. total weight of deposited sediment during deployment). To account for variation deployment time, sediment accumulation was standardized by expressing as $g/m^2/d^{-1}$ (Reed et al., 1997).

2.7 Morphological change

Using a total station, surveys of the 11 transects across the realigned marsh were conducted annually to record morphological changes in the form of sedimentation and creek development (Figure **2**). These surveys were always conducted concurrent with the vegetation surveys (refer to Section **2.4**). Although the accuracy of the total station is several mm's at the most, the actual survey accuracy of the marsh surface is considerably less due to the disturbance of the marsh by cattle and the presence of vegetation, and is considered several cm's at most. To complement, and concurrent with, the annual total station surveys, a Scan Station 2 terrestrial laser scanner was used to conduct a complete survey of a 100-m radius area in the vicinity of the SRT, with specific focus on any evolution of drainage channels on the realigned marsh. The scanner was installed on top of the embankment at the SRT and acquires a full 360^o span of data with a prescribed horizontal grid resolution of 20mm x 20mm. At the end of the 5-year monitoring period, an Unmanned Aerial Vehicle (UAV) was used to acquire high-resolution aerial photographs of the realigned marsh. The data were acquired at low tide, and, with the aid of a large number of ground control points and Structure-from-Motion algorithms, individual photographs were combined to obtain a fully georeferenced DEM of the marsh.

2.8 Ecological variability

Vegetation surveys were undertaken from 2011 to 2016, using four 4m^2 quadrats randomly positioned along each transect line, ensuring two quadrats lay either side of the central channel (Figure **2**). This arrangement ensured that vegetation across a range of elevations and geomorphological settings was monitored. Surveys were conducted in June 2011 (pre-flooding), October 2011 and then every year in June until 2016. During surveys, the percentage cover of bare ground, dead vegetation and all component species was noted. The species composition in each quadrat was classified according to the NVC Community Type scheme (Table **2**; Rodwell, 1992, 2000); only four of the NVC types were observed more than once during the survey period (MG10 was only observed in one transect during one survey).

Table **2** here

The second 'Astroturf' mat (cf. Section **2.6**) was used to determine the number and species of deposited viable seeds (Goodson et al. 2003). The purpose of this element of the monitoring is to gain insights into whether seeds from the natural salt marsh are imported into the restored salt marsh area. Mats were placed in seed trays (size) filled with potting compost in an unheated, naturally lit greenhouse; a thin layer of vermiculite on each mat prevented desiccation. On germination, seedlings were identified and removed for a maximum of 10-weeks after mat recovery (Goodson et al., 2003).

Foraminifera samples were collected and analysed to complement the vegetation surveys and provide a more comprehensive overview of the ecological change in the realigned marsh. Sediment samples were initially collected at 3-monthly intervals until 2013, but subsequently less frequently. Three samples were collected each time: one from the 25-m transect (closest to the central creek) and two from the 100-m transect (at either side of the central creek). Samples were wet sieved following Gehrels (2002) and Rose Bengal stain was used to discriminate living and dead foraminifera following Walton (1952). From each sample, 5cc was examined under light microscopy with all foraminifera encountered identified to species level and the results are expressed as tests per cc (or per cm^3). A total of 14 species were recorded over the 5-year monitoring period and these were typical of those common to natural salt marsh and estuarine/mudflat assemblages elsewhere in south-west England (Gehrels et al. 2001; Massey et al. 2006; Hart et al. 2014, 2015). On this basis, the foraminifera assemblages were divided into three groups indicating the typical sub-environment with which each community is associated: mudflat, low salt marsh, and high salt marsh (Table **3**). Living foraminifera assemblages closely matched death assemblages at each monitoring station which suggests that the deceased foraminifera are autochthonous populations. Due to seasonal bias in living foraminifera populations (cf. Horton and Edwards, 2003), the entire foraminifera (live and dead) assemblage is considered at each station.

Table **3** here

3 RESULTS

3.1 Tidal flooding regime

Figure **3** shows the 5-year time series of the daily discharge of the River Avon, and the water levels recorded at either side of the self-regulating tidal gate, representing the tidal motion in the estuary and in the realigned marsh. The river discharge is highly seasonal, showing maximum daily discharge during the winter months, peaking at almost 60 m^3/s during the 2012/13 winter. Peak discharge was not exceptional during the 2013/14 winter, but during this period, which was the wettest winter on record (Matthews et al., 2014), river discharge remained persistently high (> 10 m^3/s) for almost the whole winter period. The pressure sensor in the river was installed above MSL (0 m ODN) and only captured the upper half of the tidal curve. Water levels > 2.5 m ODN were experienced during most of the spring tides and water levels > 3 m ODN occurred when spring tides coincided with larger river discharge (e.g., during 2012/13 and especially the 2013/14 winter). As mentioned earlier, water levels in the realigned marsh are much lower than in the estuary and rarely exceeded 1.5 m ODN. The highest marsh water levels coincided with large river discharge during 2013/14.

Figure **3** here

The extent of tidal inundation for the different water levels is illustrated in Figure **4**, which shows the water depth across the realigned marsh for water levels of 1, 1.25 and 1.5 m ODN. For these water levels, 5, 11 and 14 ha of the marsh is submerged, respectively, with maximum water depths across the marsh surface (not the creeks) of 0.2, 0.45 and 0.7 m. The tidal prism associated with marsh levels of 1, 1.25 and 1.5 m ODN is 6.2×10^3, 2.8×10^4, $5.9 \times 10^4 m^3$, respectively. For a marsh level of 0.9 m ODN, the inundation area is < 2 ha (not shown); therefore, for a significant area of the marsh to be flooded, the water level must reach at least 1 m. It is evident from Figure **4** that for water levels higher than 1 m ODN, tidal flooding extends right up to the back of the marsh and could potentially increase flood risk to the properties at the north-east corner of the site.

Figure **4** here

The monthly tidal inundation characteristics for the complete 5-year monitoring period are shown as boxplots in Figure **5**. The number of over-tides per month is largest for the tidal flat (N_{otides} = 56) and smallest for the realigned marsh (N_{otides} = 34), and the water depth over the tidal flat (h = 0.63 m) is also larger than over the realigned marsh (h = 0.16 m). Despite the smaller number of tides and shallower water depths over the realigned marsh, the amount of hours per month that its surface is submerged ($T_{sub,tot}$ = 347 hrs) is larger than for the tidal flat ($T_{sub,tot}$ = 263 hrs). Perhaps more significantly, the maximum continuous period of submergence is also longer for the realigned marsh ($T_{sub,max}$ = 23 hrs) compared to that for the tidal flat ($T_{sub,max}$ = 7 hrs) and for the salt marsh ($T_{sub,max}$ = 5 hrs). Over the 5-year monitoring period, the restored salt marsh experienced continuous submergence for more than 4 days during 9 consecutive months, and during December 2013 and February 2014 the marsh was under water for two continuous periods of more than 11 days. On average, the realigned marsh experiences longer maximum periods of exposure per month ($T_{exp,max}$ = 37 hrs) than the tidal flat ($T_{exp,max}$ = 10 hrs), but shorter than the salt marsh ($T_{exp,max}$ = 98 hrs). However, over the 5-year monitoring period, the realigned marsh was continuously exposed for more than a week during 14 months; the salt marsh was never exposed for that long a period. Finally, the rate of the falling tide over the realigned marsh (dh/dt = 1 mm/min) is considerably slower than for the tidal flat and the salt marsh (dh/dt = 6 mm/min); this is expected to lead to lower flow rates during the ebbing tide over the restored salt marsh, reducing the potential for tidal creek development.

Figure **5** here

The settings of the SRT were modified several times to try and optimise the tidal flooding of the realigned marsh and the SRT also frequently experienced malfunctioning. As a result, the realigned marsh experienced significant temporal variability in the tidal inundation characteristics and several distinct phases can be identified from the water-level time series recorded over the realigned marsh

(Figure 6). The two most extreme situations occurred during phase B (very limited flooding) and phase D (very extensive flooding), and the inundation characteristics that resembled most closely that of the natural salt marsh (blue dashed line in Figure 6) occurred during phase G, at least for N_{otides}, $T_{sub,tot}$ and $T_{exp,max}$ (42 *vs* 45 tides, 215 *vs* 150 hrs and 130 *vs* 95 hrs, respectively). For most of the time, the tidal inundation and exposure on the restored salt marsh most resembled that of the tidal flat (red dashed line in Figure 6).

Figure 6 here

For each year, the water-level time series recorded on the realigned marsh for the period 1 Jan – 1 July was used to compute monthly tidal inundation statistics across the marsh surface. This 6-month period was selected, rather than the full year, because it was felt that this period (end of winter, spring and start of summer) was most relevant for the ecology. Figure 7 shows the spatial distribution in the number of over-tides (tides that flood the realigned marsh surface) for the years 2012, 2015 and 2016 (years 2013 and 2014 are similar to 2015).

Figure 7 here

The tidal flooding of the realigned marsh can be compared with that on the natural salt marsh, and the grey (red) area in Figure 7 represents a flooding frequency of 50–100% of that occurring on the natural salt marsh. The figure strongly suggest that for most of the marsh, the tidal inundation during 2012 was significantly less than that for the natural salt marsh and that only the area adjoining the central tidal creek and the location of where the WWII bomb struck (x = -50 m; y = 400 m) was characterised by a tidal flooding regime similar to that of the natural salt marsh. This was also highlighted in Figure 6, showing prolonged periods of continued exposure during 2012. In contrast, the tidal flooding during 2015 (and 2013 and 2014) was excessive compared to that of the natural salt marsh with almost the complete marsh up to y = 700 m flooded more frequently than the natural salt marsh. This was also evident from Figure 6, showing prolonged periods of continued inundation during the period 2013–2015. Only in the most recent year 2016 does the flooding regime on the realigned marsh resemble that of the natural salt marsh. A larger part of the realigned marsh is flooded regularly (up to y = 700 m), and tidal flooding is only excessive for the region around where the WWII bomb fell.

3.2 Tidal currents and sedimentation

Hydrodynamic measurements demonstrated that the maximum water depth across the adjacent natural salt marsh during spring tides is 1–1.2 m, and that it does not flood during neap tides. Current flows recorded in a small tidal channel in the salt marsh are generally weak; peaking at 0.2 m/s during flood and less than 0.1 m/s during ebb. Suspended sediment concentrations in the water are low, with maximum concentrations of 0.015–0.030 kg/m^3 (or g/l) at the start of the flooding tide, reducing to c. 0.010 kg/m^3 for the remainder of the inundation period. The flow characteristics for the realigned marsh were monitored just inward from the SRT. Here, typical flooding velocities are 1.5–2 m/s. These very localised strong flows only occur for a brief period of time at the start of the flooding tide and are followed by weaker ebb velocities (< 0.5 m/s). Suspended sediment data collected near the inflow pipe was circumspect due to the large amounts of organic material that often enters the realigned marsh during the flooding tide. It is assumed that the sediment concentrations of the water entering the realigned marsh through the pipe are the same as the water over the natural salt marsh.

Using measurements of the suspended sediment concentration and estimates of the tidal prism and inundation area, the potential sedimentation rates in the restored salt marsh can be estimated using the following equation:

$$\Delta z = \frac{1}{P\rho}\frac{NAC}{S}$$

where Δ*z* is sediment accretion rate per year (m/yr), *N* = number of over-tides per year (/yr), *A* = tidal prism (m^3), *C* = average suspended sediment concentration (kg/m^3), *P* = sediment porosity (-), ρ = sediment density (kg/m^3) and *S* = inundation area (m^2). The right term of the equation represents the amount of sediment deposition in kg/yr, and the left term converts this to m/y. This equation assumes that all sediment that enters the marsh during the flooding tide will be deposited with no sediment exiting during the ebbing tide. Considering an average high tide level in the realigned marsh of 1 m, which leads to *A* = 6,000 m^3 and *S* = 50,000 m^2 (Section **3.1**), an average sediment concentration *C* during the flooding tide of 0.02 kg m^{-3}, a sediment density ρ of 2,650 kg m^{-3}, a porosity *P* of 0.6 and 350 tidal cycles per year results in a vertical accretion rate of 0.0005 m yr^{-1}, or 0.5 mm yr^{-1}. This is at least one order of magnitude less than what can be expected on a natural salt marsh (cf. Cundy et al., 2007).

Some indication of sedimentation rates can be estimated from the sedimentation maps deployed in the realigned marsh within 50 m from the SRT and also on the natural salt marsh. Typical values for sedimentation rates on the natural salt marsh and the realigned marsh are 0.010–0.025 and 0.001–0.01 kg/m^2/day, respectively, with typical organic fraction of this material of 0.2 and 0.4 respectively

(White, 2014). These values can be used to estimate vertical accretion rates using the following equation:

$$\Delta z = \frac{NQ(1-O)}{P\rho}$$

where Δz is sediment accretion rate per year (m/yr), N = number of days per year (/yr), Q is the sedimentation rate (kg/m^2/day), O is organic fraction (-), P = porosity (-) and ρ = sediment density (kg/m^3). Application of this equation yields values of Δz for the natural salt marsh of 0.0018 – 0.0046 m/y, or c. 3 mm/y. For the realigned marsh, Δz = 0.0003 – 0.0014 m/y, or c. 0.5 mm/y. Perhaps fortuitously, this value is identical to that derived from the suspended sediment computations.

3.3 Morphological change

Any morphological change was limited to the immediate area around the SRT. The influx of sediment was insufficient to induce vertical accretion of the marsh surface and the flow velocities across the marsh surface were too weak to establish new creeks or modify the existing rather rectilinear drainage system. The planform changes near the SRT recorded using the terrestrial laser scanner data are shown in Figure **8**. The main change is the development of a bend in the central drainage channel and the deposition of a mid-channel 'bar' in the vicinity of the SRT. This seems to have been a steady process over the 5-year monitoring period.

Figure **8** here

The development of the bend is further illustrated by the evolution of transect y = 25 m (Figure **9**), which clearly shows a widening of the drainage channel through 5–6 m erosion of the westward channel bank (from 2011 to 2016), whilst the eastern bank remained stable. This change was very localized, as no significant widening of the channel occurred at any of the other transects (not shown). This erosion of the western bank is considered a direct consequence of the alignment of the inlet / outfall pipe, which directs high velocity flows onto the western bank. Widening of the channel near the SRT is a response to the volume of water flooding and draining the marsh (i.e., the tidal prism) and is accomplished by the high current velocities through the intake/outfall pipe. It appears that the sediment eroded from the bank and the channel is re-deposited on the bank (in the form of a levee) and in the channel (in the form of a mid-channel bar), and is unlikely to contribute significantly to salt marsh accretion.

Figure **9** here

3.4 Ecological variability

3.4.1 Vegetation

All full list of plant species recorded within the flooded area of the site (pre- and post-breach) is given in Table **4**. Species richness was greatest just before breach in June 2011 (37 species) and the site was dominated by plants typical of (wetland) pasture communities. In addition, six common upper salt marsh species (*Atriplex patula, Glaux maritima, Juncus gerardii, Puccinellia maritima, Spergularia maritima* and *Triglochin maritima*) were present and these were associated with saline seepages under the embankment. Although 30 species were recorded a year later, in subsequent years, species richness more than halved compared to pre-breach values and only 13 species remained in 2016; however, these did include two salt marsh species (*Aster tripoium* and *Salicornia europaea*) that were not present in 2011.

Table **4** here

At the community (NVC) level, the vegetation changes in the realigned marsh from before SRT installation in June 2011 to June 2016 can be subdivided into 4 stages (Figure **10**):

- **Pre-breach**: Vegetation was dominated by *Agrostis stolonifera - Alopecurus geniculatus* (NVC MG13) grassland, with small patches of *Juncetum gerardii* SM16 salt marsh near saline seeps. On the adjacent natural salt marsh (-60-m transect), the vegetation was, and remained throughout the study, dominated by *Spartina anglica* SM6 salt marsh.
- **2011-2013**: Modest changes occurred in the realigned marsh over the 2-year period following the breach (Figure **10** – left panel), with most MG13 quadrats unaltered, but a small number replaced by bare ground. The SM16 community extended to 150 m.
- **2013-2015**: By 2015 most vegetation had senesced to be replaced by bare ground (32 of the 40 quadrats surveyed) (Figure **10** – middle panel). The decline was progressive, with bare ground increasing from 13% in 2012, 48% in 2013, 70% in 2014 to 80% in 2015.
- **2015-2016**: Over the 12-months from June 2015 to June 2016, vegetation on the realigned marsh recovered and, whilst the five remaining MG13 quadrats above the flood line remained unaltered, 21 of the 32 bare ground quadrats transitioned to the ephemeral *Spergularia marina-Puccinellia distans* SM23 Community Type (Figure **10** – right panel).

Figure **10** here

3.4.2. Seed deposition

Of the 20 different species (from 1390 individual seedlings) recorded on mats recovered from the restored site, only 8 were salt marsh species and these accounted for only 18% of all seedlings. Of these, *Triglochin maritima* and *Juncus geradii* (both 6% of all seedlings) and *Spergularia marina* (4%) were the most frequently recorded species. There was no increase in seed deposition from time of SRT installation and the relative proportion of salt marsh species did not vary through time (White, 2014). From the mats recovered from the natural salt marsh, 12 different species germinated (from 635 seedlings) and over 97% (9 species) of these were typical salt marsh plant species with *Aster tripolium* dominating the seedling pool (86% of the seedlings).

3.4.3 Foraminifera

Foraminifera first colonised the realigned marsh in May 2013, principally at the station closest to the SRT (y = 25 m). At this location, a typical low salt marsh community, dominated by the agglutinated taxa *Milammina fusca* (43%), *Jadammina macrescens* (21%) and *Trochammina inflata* (11%), became established between May and November 2013, and peaked in the August 2013 samples with a count of 15.5 tests per cc (Figure **11**). This is, however, a very low concentration in comparison to fully established salt marsh communities. As well as a declining foraminiferal stock at this location after the peak in August 2013, there also appears to be a shift to an assemblage more typical of a mudflat community. This is reflected by a decline or absence of the high salt marsh taxa (*Jadammina macrescens* and *Trochammina inflata*), to a typical low salt marsh community dominated by *Miliammina fusca* with a notable presence of calcareous taxa *Elphidium sp.* and *Haynesina germanica* which are typical of mudflat communities. By the final survey in July 2016, when abundance reached its maximum, it appears this low salt marsh to mudflat community was becoming well-established as indicated by the dominance of the low salt marsh species *Miliammina fusca* (35%), the mudflat taxa *Elphidium spp.* (35%) and *Haynesina germanica* (15 %), and the decline of the high salt marsh taxa *Jadammina macrescens (5%)* and *Trocammina inflata* (4%). However, the maximum foraminifera concentration of 27.5 tests per cc indicates communities at this site remain immature.

At y = 100 m there are two sampling locations at either side of the central creek. West of the central creek there is evidence of a typical low salt marsh foraminifera community becoming established with concentrations dominated by *Miliammina fusca* present in all the sampling periods from 2013 to 2016. The greatest concentrations of foraminifera in the marsh occurs at this location in the November 2015 (68.5 tests per cc) and July 2016 (121 tests per cc) samples, indicating that the *Miliammina fusca* dominated low salt marsh community is becoming well established at this location. Additionally, in the November 2015 and July 2016 samples, significant proportions high salt marsh taxa *Jadammina macrescens* (up to 8 %) and *Trocammina inflata* (up to 25%) comprised the assemblages, indicating a subtle transition from low to high salt marsh environments. East of the central creek there was little indication of a foraminiferal community becoming established with few or no foraminifera recorded until November 2015, when concentrations reached a maximum of 4.75 tests per cc. By July 2016 there is evidence of a middle-to-high salt marsh community becoming established with a moderate concentration of foraminifera (52 tests per cc) dominated by *Miliammina fusca* (50%), *Trochammina inflata* (29%) and *Jadammina macrescens* (19%).

Figure **11** here

3.5 Ecological transition and variation in the tidal environment

The spatial distribution in the inundation statistics across the realigned marsh, such as shown in Figure **7**, was applied to all the ecological monitoring locations over the complete 5-year monitoring period. Subsequently, the inundation statistics were pooled for each of the NVC Community Type, and the bare ground category ('bare'), and the results presented in a boxplot (Figure **12**). All values are monthly statistics, but these were computed using 6 months of water-level data over the realigned marsh (January–June).

Figure **12** here

The conditions characterising the 'Bare' vegetation type clearly stands out when compared with the other NVC Community Types; it is associated with the largest number of over-tides (N_{otides} = 52), the largest number of total hours of inundation ($T_{sub,tot}$ = 338 hrs), the longest continuous period of inundation ($T_{sub,max}$ = 8 hrs; also the greatest variability) and also the longest period of exposure ($T_{exp,max}$ = 45 hrs). This vegetation type was particularly ubiquitous over the period 2013–2015. Compared with the other NVC Community Types in the area, the flooding characteristics for 'Bare' are closest to that of the natural *Spartina* (SM6) salt marsh in terms of number of over-tides (N_{otides} = 48, but most similar to the *Juncetum gerardii* (SM16) salt marsh in terms of total hours of inundation ($T_{sub,tot}$ = 328 hrs).

However, the flooding regime of the ‘bare’ category is more extreme than that of SM6 and SM16, and is overall more akin that of the tidal flat (refer to Section **3.1**).

The MG13 Community Type (*Agrostis stolonifera - Alopecurus geniculatus* grassland) is associated with the least amount of tidal inundation: N_{otides} = 10, $T_{sub,tot}$ = 33 hrs, $T_{sub,max}$ = 3 hrs and $T_{exp,max}$ = 171 hrs. MG13 is the original vegetation type, and it seems it can tolerate a significant tidal flooding, as several MG13 plots received more than 20 over-tides and in excess of 100 hrs tidal inundation per month. The SM23 ephemeral *Spergularia marina-Puccinellia distans* salt marsh was the dominant vegetation in the inundated area in 2016. It has tidal inundation characteristics that significantly overlap with that of MG13; however, focussing on the median values, the tidal inundation for SM23 is more frequent and of longer duration than for MG13: N_{otides} = 12 (*versus* 10), $T_{sub,tot}$ = 49 hrs (*versus* 39 hrs), $T_{sub,max}$ = 4 hrs (*versus* 3 hrs).

In summary, there is a clear gradient in terms of increasing tidal inundation intensity from MG13 $\rightarrow$ SM23 $\rightarrow$ SM6 $\rightarrow$ SM16 $\rightarrow$ Bare. There is considerable overlap between the different NVC Communities, but it must be borne in mind that vegetation in the realigned marsh is in transition; therefore, the actual inundation statistics do not necessarily reflect optimal conditions.

4. Discussion

The status of the realigned marsh in June 2016 is illustrated by the aerial photograph shown in Figure **13**. It shows that the majority of the marsh that is regularly inundated by the tide (z < 1.2 m ODN) is characterised by the SM23 NVC Community Type. There is a distinct boundary between SM23 and MG13, both in terms of colour Figure **13** (brown-green versus green) and elevation, as the z = 1.2 m ODN contour line separates these two NVC types very well. Five years post-breach, the realigned marsh at South Efford is now transitioning into salt marsh. However, while the foraminifera assemblage is typical of mid-high-level salt marsh (e.g., Gehrels et al. 2001), the *Spergularia* dominated (NVC SM23) vegetation is at best only an ephemeral salt marsh community.

Figure **13** here

A lack of propagule supply, at least for the first three years after breach, may go some way to explaining why there was no transition to a more typical *Spartina*-dominated (i.e. NVC SM6) perennial plant community. Although we cannot be sure that post-2014 propagules were not entering from the nearby natural marsh, it seems likely that a consequence of the SRT system used at South Efford was that immigration of seeds and root fragments was limited. Indeed, aside from the very small number of (wind dispersed) *Aster tripolium* seeds that germinated on the sediment mats, *Triglochin maritima*, *Juncus gerardii*, and *Spergularia marina* were present inside the embanked area prior to breach. Even with plentiful propagule supply however, the tidal regime at the site would likely have restricted plant establishment.

The limited inundation during the early post-breach phase (2011–2013) was due to the initial setting of the SRT; this would have further restricted the immigration of salt marsh plant propagules and the development of physical-chemical conditions suitable for their establishment. The excessive inundation during 2013–2015 can be attributed to malfunctioning of the SRT when it was blocked with detritus, and extremely wet winters of 2013/14 and 2014/15. During this period the realigned marsh was submerged continuously for several weeks, conditions which probably contributed to the development of the bare surface typical of many MR schemes (Mossman 2012a). Drainage was further hampered by the SRT system which compelled all water to drain through a single circular 0.9-m diameter outflow pipe at an elevation little different to low tide in the adjacent estuary. Only during 2016 did the inundation characteristics on the realigned marsh resemble that of the adjacent natural salt marsh in terms of number of over-tides and duration of tidal inundation. At this time, over most of the realigned marsh, the number of over-tides was between 24 and 48, compared to 48 over-tides on the natural salt marsh and less than 38 and 42 over-tides per month as recommended by Ash and Fenn (1997) and Environment Agency (2003), respectively. Only during 2016 did any typical salt marsh vegetation develop widely across the site and, even then, this was restricted to a plant community dominated by the ephemeral annual *Spergularia marina.* Foraminifera populations remained low (< 130 individuals per cc) compared with natural marshes along the Avon Estuary which generally exceed 1000 individuals per cc (Stubbles, 1999). This indicates that foraminifera communities in the realigned marsh are immature, but their increasing diversity and abundance over the 5-year monitoring period suggests salt marsh species are beginning to thrive.

Given that elevation within the tidal frame is the most important factor determining the success of propagule and salt marsh establishment (Adam, 1990; French, 2006; Davy et al., 2011), our observations at South Efford underscore the challenge in achieving the flooding and drainage regime experienced by natural salt marsh, especially when South Efford is, like many managed realignment sites, of much lower elevation than the adjacent natural marsh (Wolters et al., 2005, 2008; Spencer and Harvey, 2012). In addition to a direct impact on propagule delivery and establishment, prolonged flooding reduces sediment redox potential, while a small tidal prism (depth over the marsh < 0.2 m) limits further the amount of suspended sediment entering the marsh. Together these factors mitigate against the establishment of salt marsh vegetation on managed realignment sites (Mossman et al., 2012a, b; Spencer and Harvey, 2012). Moreover, and unlike natural topographically complex marshes dissected by numerous tidal creeks, South Efford (like many managed realignment sites) is a horizontal surface with limited drainage channels (only one central channel). The only significant morphological change occurred in the vicinity of SRT due to large water flow velocities entering the marsh through the pipe, and there was very limited evidence of an emergent creek network; the flow velocity over the marsh was too low to entrain sediment and create a dendritic creek network that drains to the main drain. More generally, even if all the sediment entering the realigned marsh is deposited on the marsh surface, the vertical accretion rates are estimated to be one order of magnitude less than on the adjacent natural salt marsh. This means that with rising sea level, the realigned marsh will increasingly lag behind the natural salt marsh in terms of its elevation. This will make it increasingly difficult to maintain a 'natural' inundation regime over the restored salt marsh and ultimately the salt marsh restoration effort will fail. Limited sediment influx is probably a characteristic of intertidal habitat schemes involving regulated tidal exchange (including SRT); in fact, Pontee (2014) recommends installation of a RTE to limit siltation rates in intertidal habitat restoration sites, as opposed to breaching.

Concluding remarks

A regulated tidal exchange (RTE) option was implemented at South Efford to enable intertidal habitat creation without increasing flood risk to neighbouring properties, and a self-regulating tide gate (SRT) selected to control the water levels over the realigned marsh. One of the main advantages of RTE is the ability to regulate the tidal water levels; nonetheless, our results suggest that through installation of a SRT, it has not been possible to consistently facilitate a natural tidal inundation regime at South Efford. Apart from the non-trivial issue of optimising the gate settings, frequent malfunctions due to jamming by detritus let in too much, or too little water. Even when the SRT operated as planned, inundation was always quicker than drainage, causing extended periods of submergence that prevented the establishment of vegetation. Slow drainage further impeded the development of tidal creek systems which would enable sediment recycling and accretion fed by creek expansion. Perhaps most significantly, the SRT allowed only small amounts of suspended sediment into the site and sedimentation was insignificant; consequently, a major pre-requisite for natural salt marsh development – vertical accretion – was missing. We conclude therefore that the SRT system used at

South Efford was unable to impose the natural physical parameters required for salt marsh development, and more generally question the ability of SRT to achieve a sustainable and naturally functioning salt marsh at any managed realignment site. It may be possible to achieve a tidal flooding regime conducive to the development of suitable intertidal habitat (mudflat or salt marsh), but facilitating at the same time sustainable vertical accretion rates (similar to the rate of sea-level rise) and the development of a tidal creek network might be over-ambitious. We conclude that SRT can be a useful technique for intertidal habitat creation where there are significant site constraints (especially flood risk), but we need to be realistic in our expectations of what it can achieve in terms of delivering a perennial salt marsh community.

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Table 1 – *Overview of 5-year monitoring programme at South Efford.*

Parameter	Sampling location	Sampling frequency	Sampling period
Water level	In restored salt marsh and in the river	Every 15 min	2011–2016
Salinity	In restored salt marsh	Every 15 min	2011–2016
Tidal currents and suspended sediment concentrations	In restored salt marsh and in natural salt marsh	4 Hz	3 March – 4 May 2012
Suspended sediment concentrations	In natural salt marsh	5-10 min	21 March 2012
Sedimentation and seedling mats	Many locations (> 10) in restored salt marsh	Mats were deployed over several months at a time	2011–2014
Total station surveys	10 transects in restored marsh and 1 transect in natural salt marsh	yearly	2011–2016
Laser scanner surveys	Scan from top of SRT of restored marsh and natural salt marsh	yearly	2011–2016
Unmanned Aerial Survey	Survey of restored marsh and natural salt marsh	once	27 June 2016
Vegetation survey	40 quadrats in restored marsh and 4 quadrats in natural salt marsh	Half-yearly to yearly	2011–2016
Foraminfera sampling	3 samples from restored marsh near SRT site	Three-monthly to yearly	2011–2016

Table 2 – *NVC Community Types observed during the 5-year monitoring period in South Efford marsh.*

Code	NVC Community Type
SM6	*Spartina anglica* salt marsh - distinctive salt marsh community on the seaward fringes of marshes and on creek sides,
SM16	*Juncetum gerardii* salt marsh - characteristic of mid-upper coastal marshes
SM23	*Spergularia marina-Puccinellia distans* salt marsh - characteristic of disturbed situations with soils of variable but generally high salinity (e.g. upper pans) on coastal marshes
MG13	*Agrostis stolonifera - Alopecurus geniculatus* Grassland - Typical of inundation; usually in river flood plains and on the edges of ponds
MG10	*Holcus lanatus - Juncus effusus* rush-pasture - Typically associated with poorly drained permanent pastures
Bare	No vegetation

Table 3 – *Marsh sub-environment classifications based on typical foraminifera assemblages recorded in southwest England (references in text).*

	Foraminifera Community
High salt marsh	*Jadammina macrescens*, *Trochammina inflata*, *Haplophragmoides wilberti*
Low salt marsh	*Miliammina fusca*
Mudflat	*Cibicides lobatulus*, *Elphidium spp.*, *Haynesina germanica, Quinqueloculina spp.*

Table 4 – *Pre- (June 2011) and post-breach changes in plant species composition recorded in fixed 4x4m quadrats positioned along a tidal inundation gradient at the South Efford managed realignment site, Devon, SW England. Species presence in at least one quadrat is denoted by 'P' and species typical of British salt marsh vegetation are highlighted in bold type.*

	Species	Year					
		2011	2012	2013	2014	2015	2016
Grasses	*Agrostis stolonifera*	P	P	P	P	P	P
	Alopecurus geniculatus	P	P	P	P	P	P
	Anthoxanthum odoratum	P					
	Arrenatherum eliatus	P					
	Cyanosurus cristatus	P	P	P	P		
	Elymus repens	P					
	Festuca pratensis	P	P				
	Holcus lanatus	P	P		P		
	Lolium perenne	P	P				
	Poa annua	P					
	Poa pratensis	P					
	Poa trivalis	P					
	Puccinellia maritima	**P**	**P**	**P**	**P**	**P**	**P**
Sedges/Rushes	*Bolboschoenus maritimus*		P	P	P		
	Carex otrubae	P	P	P			P
	Carex ovalis	P					
	Eleocharis palustris	P	P	P			
	Juncus articulatus	P	P	P	P		P
	Juncus bufonius		P				P
	Juncus effusus	P	P	P	P	P	
	Juncus gerardii	**P**	**P**	**P**	**P**	**P**	**P**
Forbs	***Aster tripolium***		**P**		**P**	**P**	**P**
	Atriplex patula	**P**	**P**		**P**	**P**	**P**
	Cardamine pratensis	P	P				
	Cerastium holosteoides	P	P				
	Glaux maritima	**P**	**P**	**P**			
	Leontodon autumnalis	P					
	Leontodon hispidus	P					P
	Plantago major	P	P				
	Plantago media		P				
	Prunella vulgaris		P				
	Ranunculus acris	P					

	Ranunculus sceleratus		P		P		P
	Ranunculus repens	P	P	P	P		
	Rumex acetosa	P					
	Rumex conglomeratus	P					
	Rumex crispus	P					
	Rumex obtusifolius		P	P			
	Salicornia europaea		**P**		**P**	**P**	**P**
	Sonchus arvensis				P		
	Spergularia marina	**P**	**P**	**P**	**P**	**P**	**P**
	Taraxacum officinale	P	P	P	P	P	
	Trifolium dubium	P					
	Trifolium pratense	P					
	Trifolium repens	P	P	P	P		
	Triglochin maritima	**P**	**P**	**P**	**P**		
Total species		**37**	**30**	**17**	**19**	**10**	**13**
Total salt marsh species		**6**	**8**	**5**	**7**	**6**	**6**

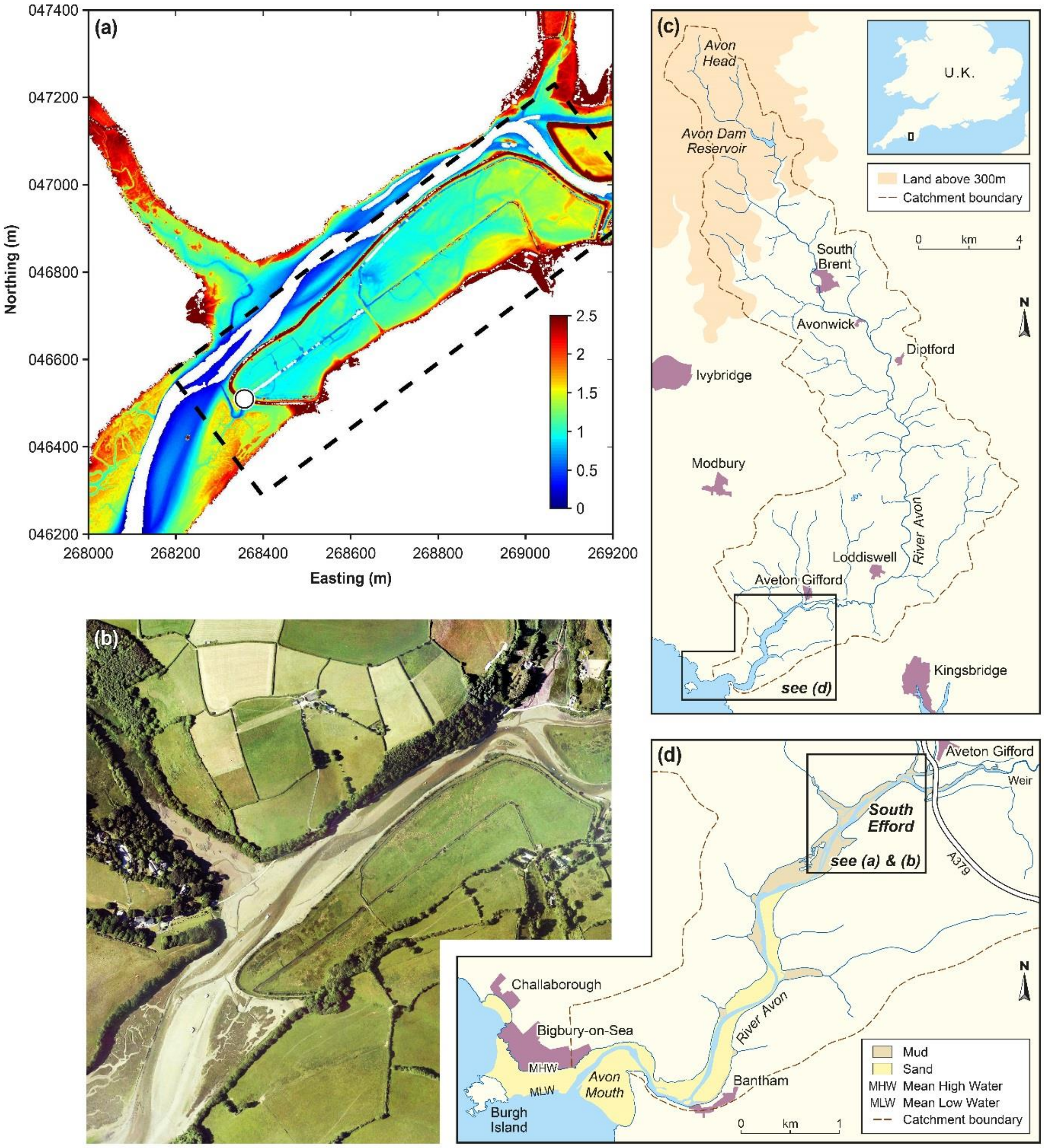

Figure 1 – *Location map of the Avon estuary, and 2010 aerial photograph (photo from GoogleEarth) and digital elevation model (DEM) of South Efford marsh. The DEM is based on LIDAR data provided by the Plymouth Coastal Observatory (PCO), and elevations are in m ODN, which is approximately 0.2 m above mean sea level. The black dashed rectangle represents the realigned marsh area plotted in subsequent figures and is 400 m x 1000 m. The red circle represents the location of the self-regulating tidal gate (SRT) and the colour bar refers to the elevation in m ODN. White regions represent standing water (river section) or elevations > 5 m (valleys sides).*

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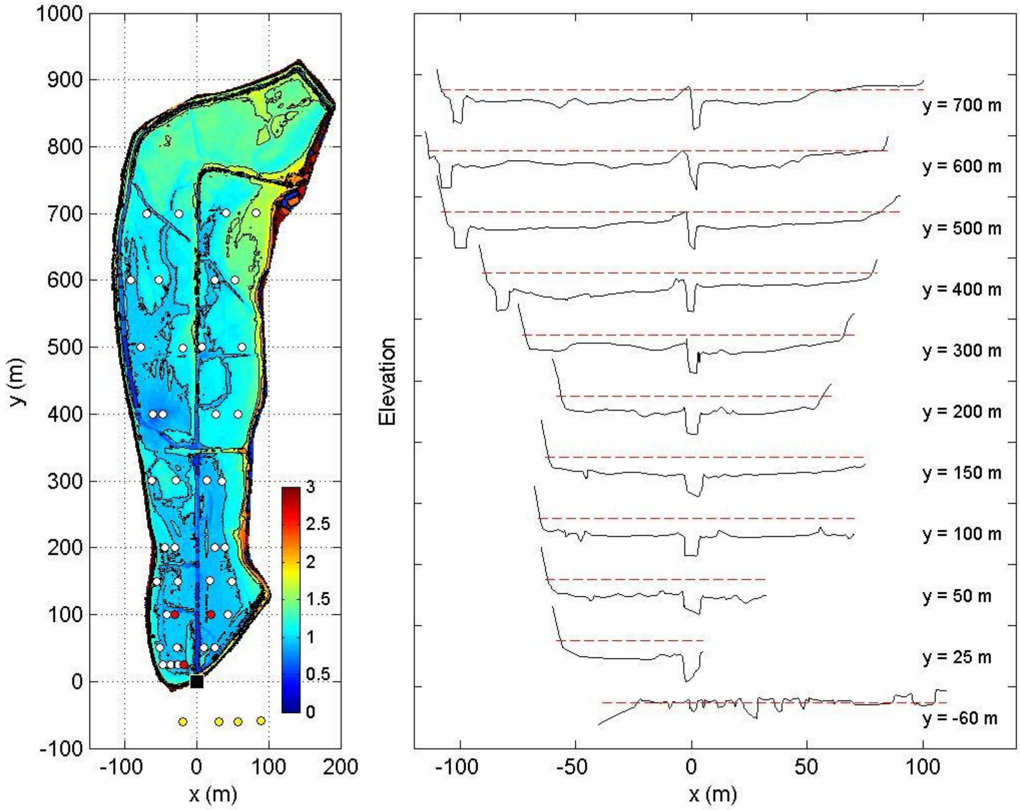

Figure 2 – *Left panel shows digital elevation model of South Efford marsh on local coordinate grid with ecological sample locations (circles). The sample sites on the natural salt marsh are denoted by the yellow circles and the red circles represent the foramifera sample locations. The black square represents the location of the SRT. Right panel shows total station surveys of all across-marsh transects measured during the baseline survey in May 2011. The profiles have been vertically offset by 2 m for ease of comparison and the tick marks on the y-axis represent 1-m intervals. The red dashed horizontal line represents the average level of the natural salt marsh. The contour lines represent 0, 1, 2, 3 and 4 m ODN.*

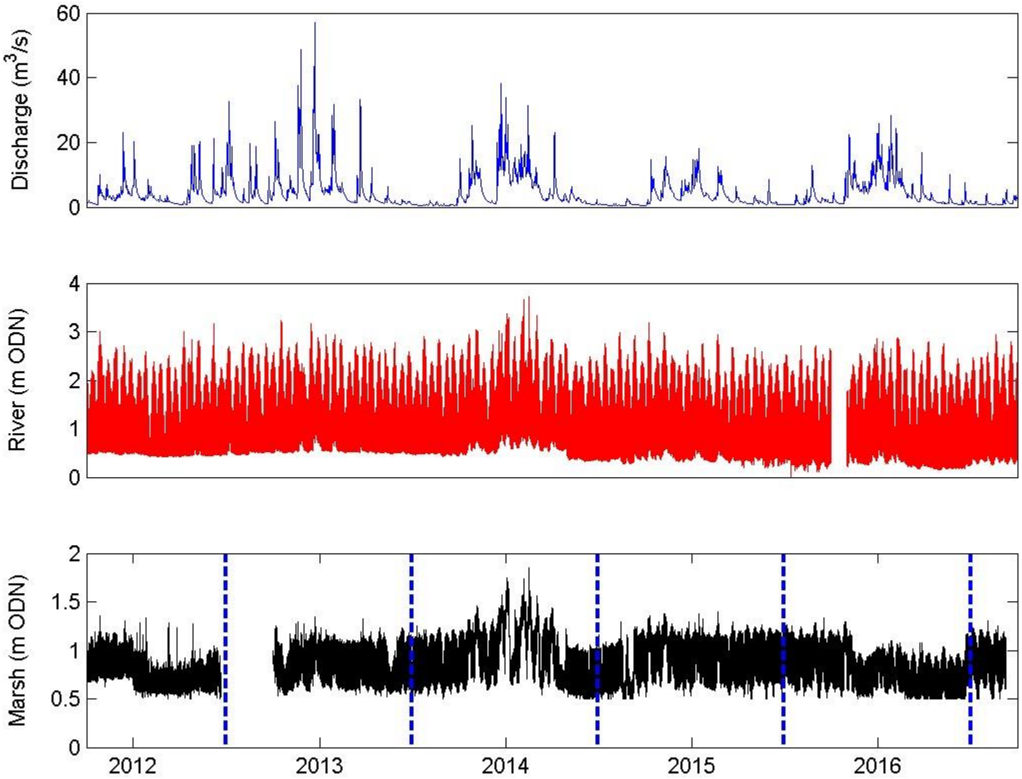

Figure 3 – *Time series of river discharge recorded at Loddiswell (upper panel), water level in the river (middle panel) and the realigned marsh (lower panel). The vertical dashed lines in the lower panel represent 1 July, the date by which the annual ecological surveys were finished.*

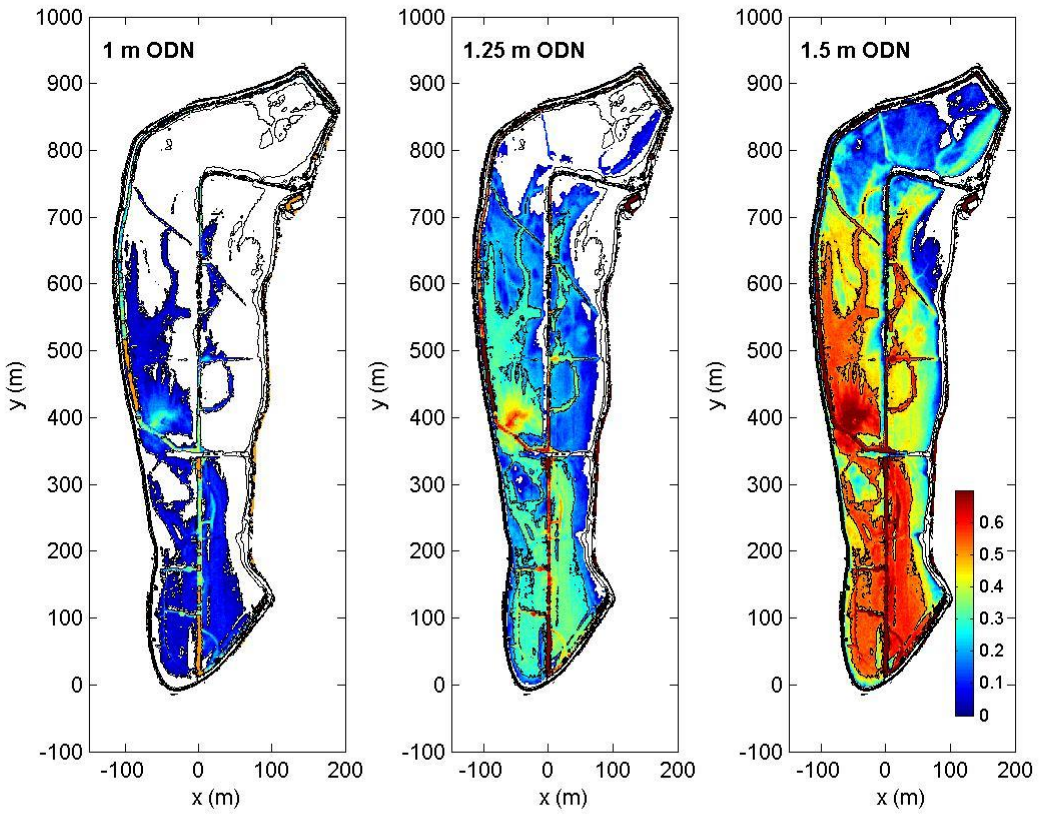

Figure 4 – *Water depth on South Efford marsh for marsh water levels of 1, 1.25 and 1.5 m ODN. The colour bar in the right panel applies to all panels.*

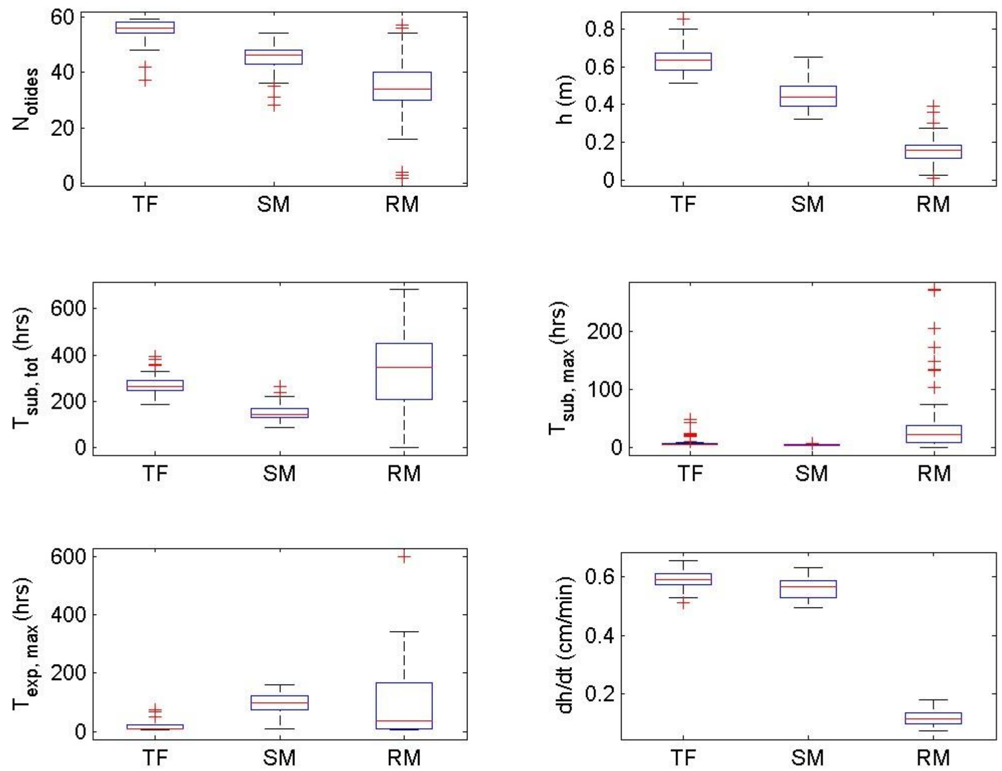

Figure 5 – *Box plots of various monthly tidal parameters computed over the complete 5-year survey period for the natural tidal flat (TF; z = 1 m ODN), natural salt marsh (SM;* z = 1.5 m ODN*) and the realigned marsh (RM; z = 0.9 m ODN):* N_{otides} *= number of over tides;* h *= average water depth;* $T_{sub,tot}$ *= total hours of tidal submergence;* $T_{sub,max}$ *= maximum continuous period of tidal submergence;* $T_{exp,max}$ *= maximum continuous period of exposure;* dh/dt *= average rate of falling tide. On each box, the central mark is the median, the edges of the box are the 25 and 75 percentiles, the whiskers extend to the most extreme data points the algorithm considers to be not outliers (0.7 and 99.3 percentiles), and the outliers are plotted individually*.

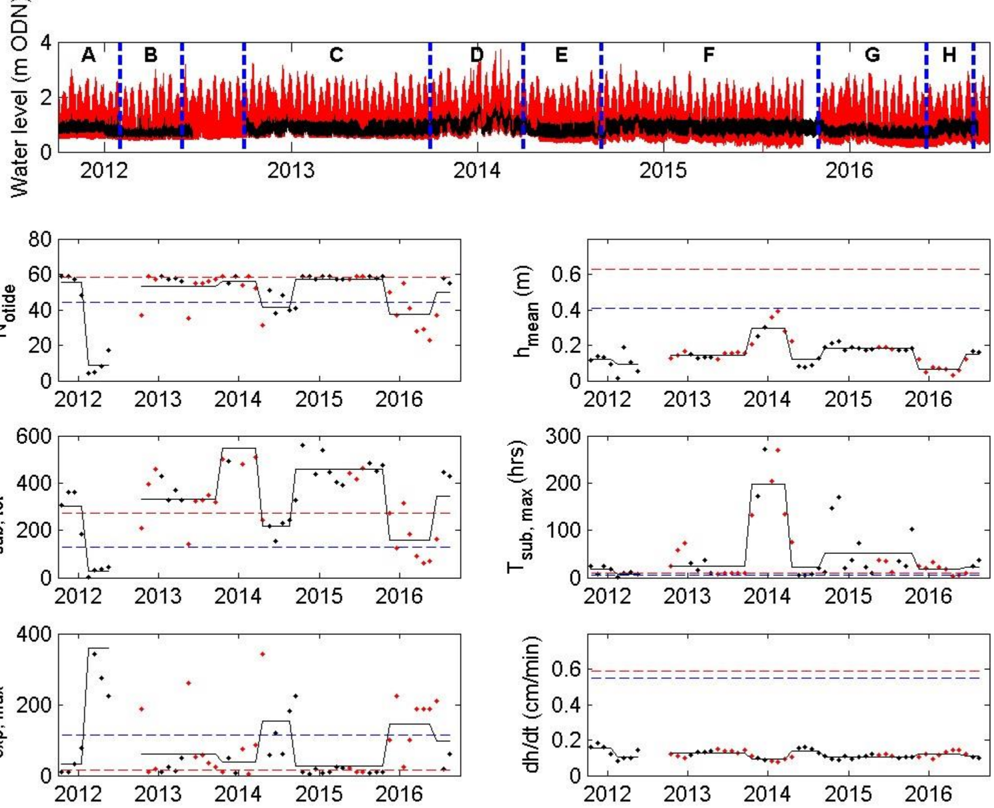

Figure 6 – *Top panel shows time series of water level in the river (red line) and the realigned marsh (black line). Smaller panel show monthly time series of tidal parameters for the realigned marsh compared with the mean value over the 5-year period for the tidal flat (red dashed line) and the salt marsh (blue dashed line). The line in the lower panels represents the mean values for the phases indicated in the top panel and the red symbols represent months with significant malfunctioning of the SRT.*

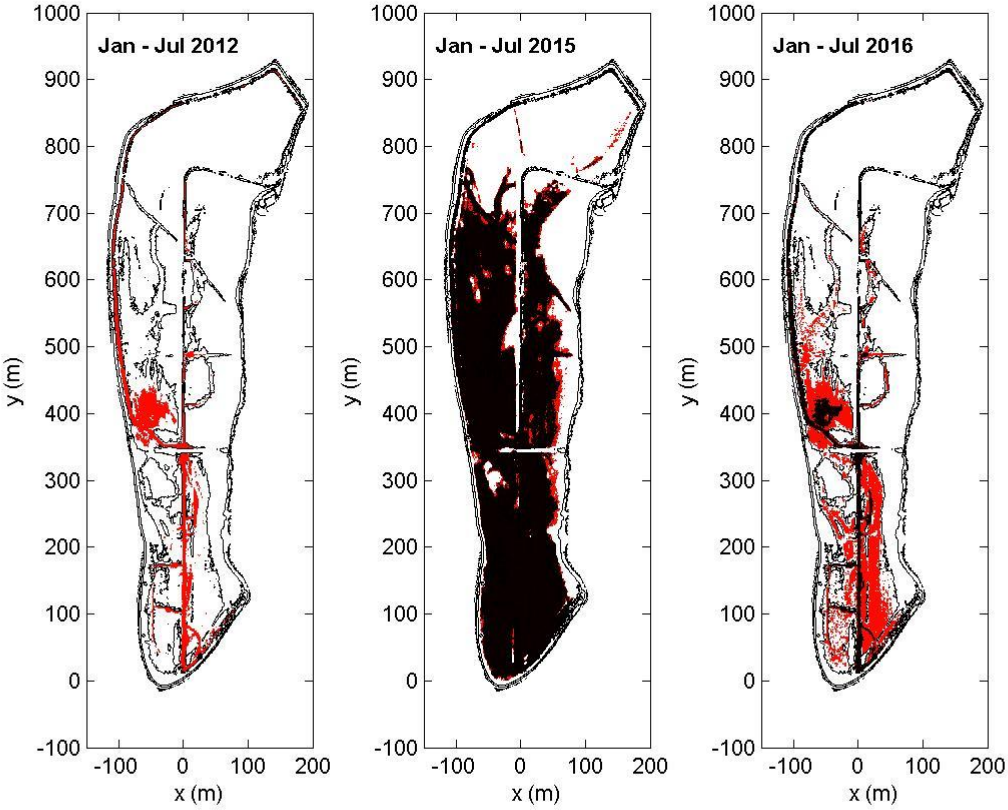

Figure 7 – *Spatial distribution across the realigned marsh of the number of over-tides per month (tides that flood the marsh surface) for three years (2012, 2015 and 2016) and calculated for the 6-month period prior to the ecological survey (January to June). Optimal salt marsh conditions are considered to occur in the grey (red) area, representing 24–48 over-marsh tides per month. White and black areas experience less than 24 or more than 48 over-marsh tides per month, respectively. The natural marsh is, on average, flooded 48 times per month (cf. Figure* 5*). The contour lines represent 0, 1, 2, 3 and 4 m ODN.*

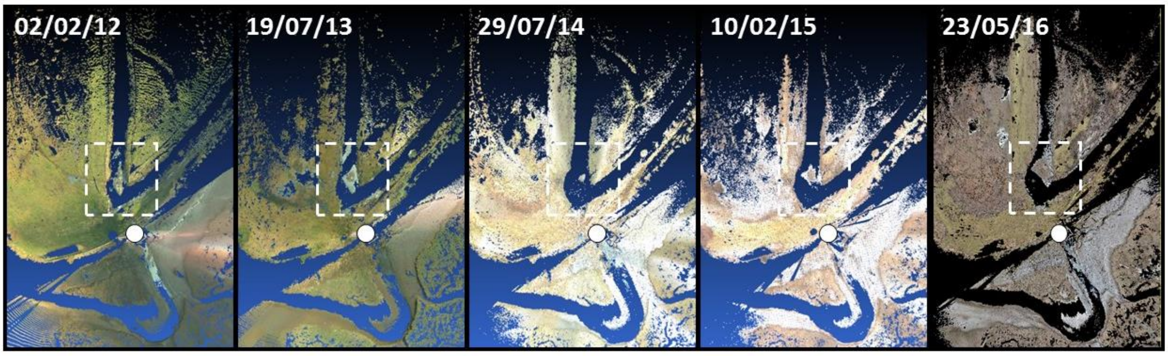

Figure 8 – *Terrestrial laser scans obtained from the top of the tidal gate (denoted by white circle), with the realigned marsh at the top of the image and the natural salt marsh at the bottom. A very modest increase in the curvature of the channel in the realigned marsh can be observed and is being achieved through erosion of the left (west) bank of the channel nearest to the tidal gate (see also Figure 9).*

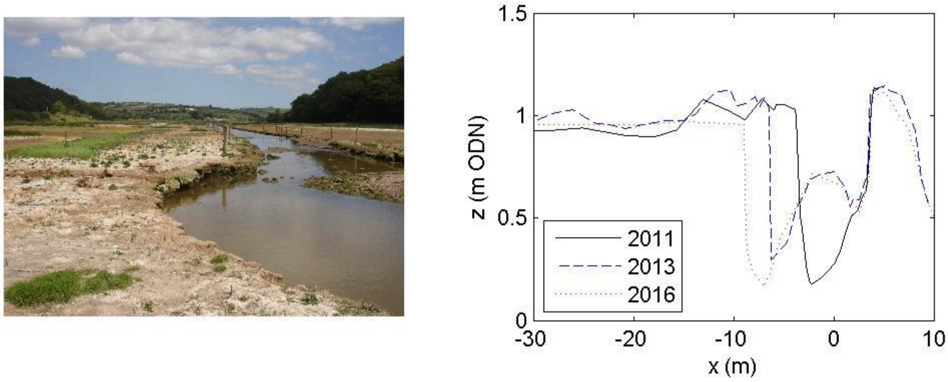

Figure 9 – *Photograph taken from the top of the SRT looking towards the realigned marsh (taken in 2014) and morphological evolution of the transect running across the tidal creek near the SRT (at* y *= 25 m) showing progressive erosion of the western (left) bank.*

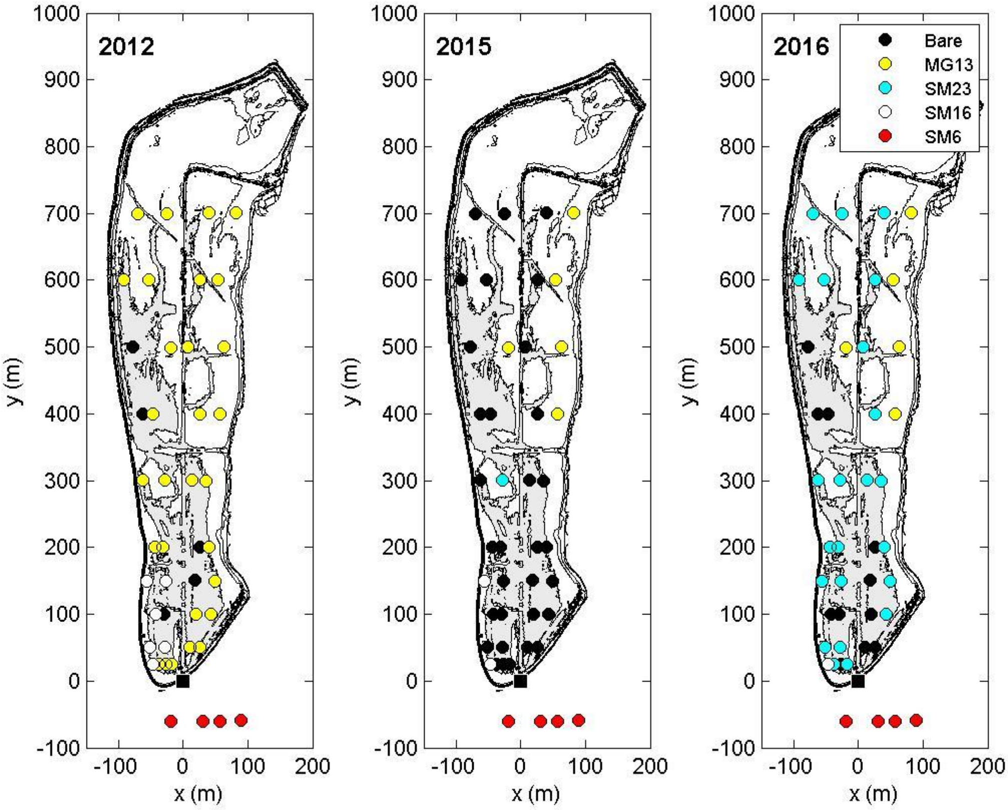

Figure 10 – *Spatial distribution of the NVC Community Type at South Efford for the June surveys in 2012 (also representative of 2011), 2015 (also representative of 2013 and 2014) and 2016. The black square represents the SRT and grey shaded area represents below 1 ODN. The contour lines represent 0, 1, 2, 3 and 4 m ODN. The observed NVC Community Types are: MG13 =* Agrostis stolonifera - Alopecurus geniculatus *Grassland; SM23 =* Spergularia marina-Puccinellia distans *salt marsh; SM16 =* Juncetum gerardii *salt marsh; SM6 =* Spartina anglica *salt marsh.*

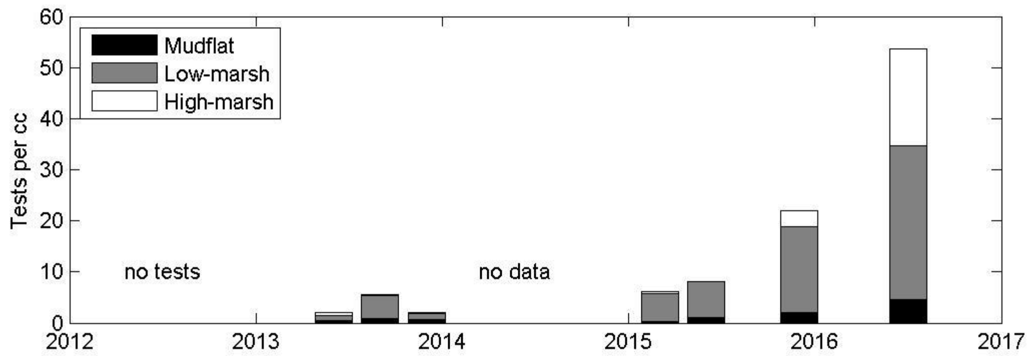

Figure 11 – *Average foraminifera concentrations of mudflat, low-marsh and high-marsh biozones over the monitoring period. The concentrations were averaged for the 3 sample locations (1 at* y *= 25 m; 2 at* y *= 100 m). Biozone classifications are presented in Table **2**.*

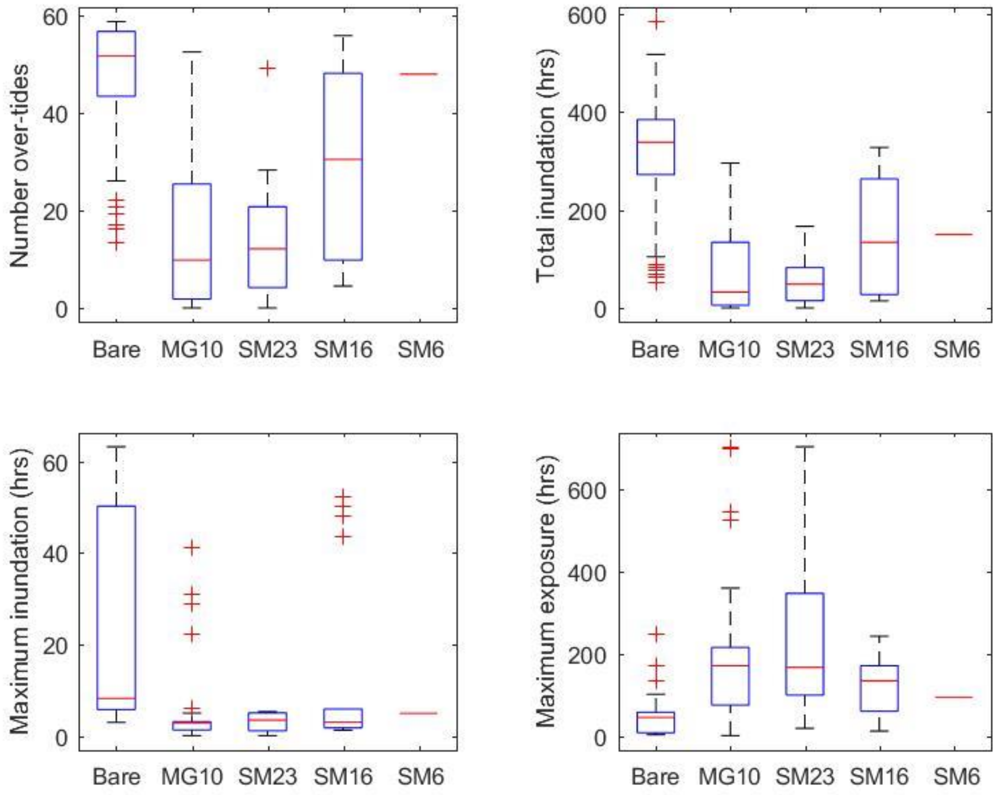

Figure 12 – *Tidal inundation statistics for the different NVC Community Types at South Efford using all ecological monitoring locations over the 5-year monitoring period. The observed NVC Community Types are: MG13 =* Agrostis stolonifera - Alopecurus geniculatus *Grassland; SM23 =* Spergularia marina-Puccinellia distans *salt marsh; SM16 =* Juncetum gerardii *salt marsh; SM6 =* Spartina anglica *salt marsh. On each box, the central mark is the median, the edges of the box are the 25 and 75 percentiles, the whiskers extend to the most extreme data points the algorithm considers to be not outliers (0.7 and 99.3 percentiles), and the outliers are plotted individually.*

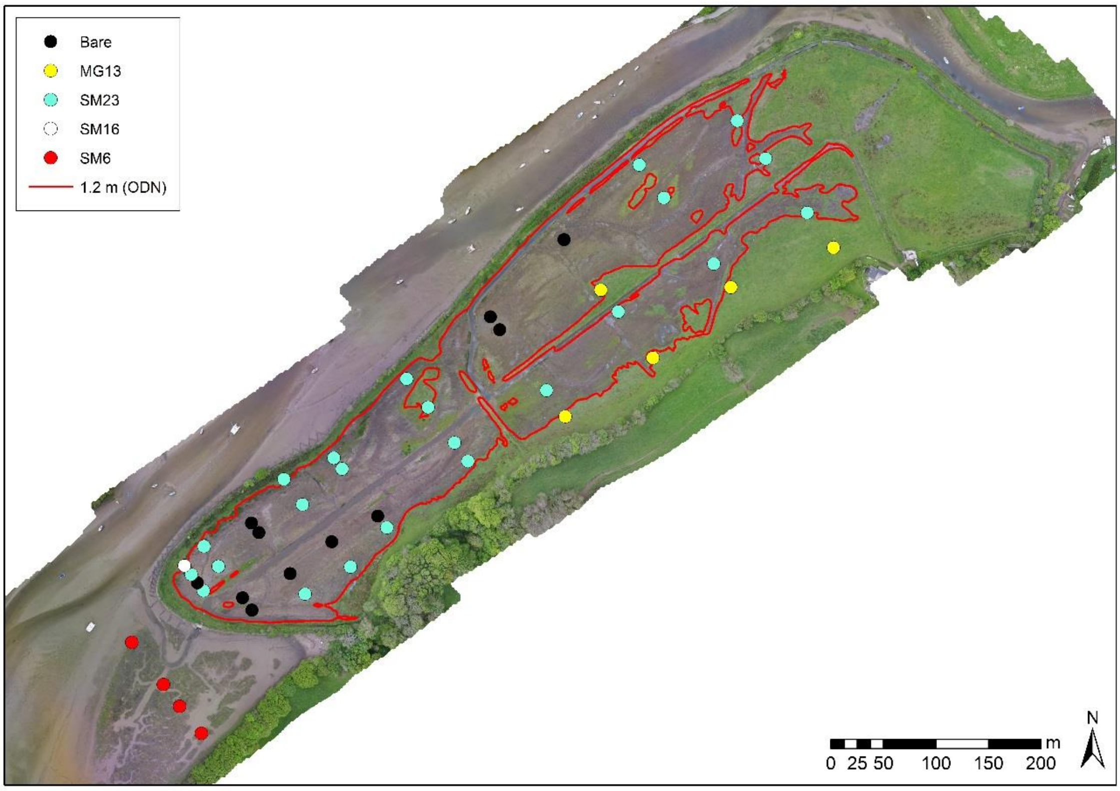

Figure 13 – *Aerial photograph of South Efford marsh obtained with UAV flight in June 2016. The ecological sampling locations and the observed NVC Community Types are indicated, as well as the 1.2 m ODN contour line. NVC types: MG13 =* Agrostis stolonifera - Alopecurus geniculatus *Grassland; SM23 =* Spergularia marina-Puccinellia distans *salt marsh; SM16 =* Juncetum gerardii *salt marsh; SM6 =* Spartina anglica *salt marsh.*

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