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Sediment supply explains long-term and large-scale patterns in saltmarsh lateral expansion and erosion

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Key Points:

- Sea level rise alone does not explain marsh lateral changes over the past 150 years
- Sediment flux is by far the strongest indicator of long-term lateral changes in saltmarsh extent.
- Small increases in fetch length may boost marsh expansion through stimulating wind-driven sediment transport onto marshes

Abstract

Salt marshes often undergo rapid changes in lateral extent, the causes of which lack common explanation. We combine hydrological, sedimentological and climatological data with analysis of historical maps and photographs to show that long-term patterns of lateral marsh change can be explained by large-scale variation in sediment supply and its wave-driven transport. Over 150 years, northern marshes in Great Britain expanded while most southern marshes eroded. The cause for this pattern was a north to south reduction in sediment flux and fetch-driven wave sediment resuspension and transport. Our study provides long-term and large-scale evidence that sediment supply is a critical regulator of lateral marsh dynamics. Current global declines in sediment flux to the coast are likely to diminish the resilience of salt marshes and other sedimentary ecosystems to sea level rise. Managing sediment supply is not common-place, but may be critical to mitigating coastal impacts from climate change.

Plain Language Summary

Salt marshes are valuable ecosystems for human societies, and are especially vulnerable to losses caused by human activity and climate change. Little is known about how the size of marshes has changed in response to disturbance over large- and long-term scales. We used historical maps and aerial photographs to capture 150 years of change in marsh area extent in 25 estuaries and ~100 marshes across Great Britain. We then related the rates of marsh change to existing data on hydrology, biology, climate, sediment supply and other variables, to find out which elements best explained patterns of erosion and expansion for the period between 1967 and 2016. We found a shift from long-term marsh erosion in the south-east, to long-term marsh expansion in the north-west of Great Britain, and that this pattern was explained by a south-to-north gradient of increasing sediment flux into marshes and wave fetch lengths which

helps transport sediment onto marshes. Our study demonstrates how sediment supply should be monitored and managed to preserve saltmarsh extent into the future.

1 Introduction

The threat of sea level rise has dominated theoretical and empirical saltmarsh research for more than thirty years, from concerns that over 90% of global marshes could drown by 2100 (Crosby et al., 2016; Spencer et al., 2016; Horton et al., 2018; Valiela et al., 2018). Recent results show that marshes are adept at keeping pace with sea level rise by growing vertically when sediment is available to settle onto the marsh surface (Kirwan et al., 2016a); an irony, given that fear of marsh loss by drowning has had an overriding influence on conservation policy since the 1970s (Hatvany et al., 2015). Despite the vertical resilience to sea level rise, there are many documented cases from Europe, North America and Asia where marshes have undergone extensive lateral changes in cover, expanding or eroding hundreds of metres in just a few years (Yang et al., 2001; Lotze et al., 2006; Fagherazzi et al., 2013; Gunnell et al., 2013; Leonardi et al., 2016). This study heeds the call to investigate the drivers causing lateral marsh change (Kirwan et al., 2016a; Kirwan et al., 2016b; Schuerch et al., 2018), shifting the current emphasis away from a predominant focus on vertical growth dynamics alone (Mariotti & Fagherazzi, 2010; Kirwan et al., 2016b). The causes for lateral marsh change need to be understood if natural coastal protection by marshes is to be effectively managed (Bouma et al., 2014; Kirwan et al., 2016b; Ganju, 2019).

Marsh loss by lateral retreat is thought to be the consequence of wind-wave attack (Mariotti & Fagherazzi, 2010, 2013; Marani et al., 2011; Mariotti & Carr, 2014). Sea level rise and increased severity of storm and river flooding collectively act to raise water depths and wave/current scour over tidal flats, thereby increasing the likelihood of initiating lateral marsh erosion (Mariotti & Fagherazzi, 2010; Mariotti & Carr, 2014; Hu et al., 2015b). Previous studies have indicated that sediment supply from marine or riverine sources can diminish this erosion risk when the replenishment of sediment is sufficiently large to cause tidal flats to elevate through accretion. For example, marshes in the macrotidal Bay of Fundy, Canada, are resilient to erosion because new sources of sediment from ice rafting are transported to the saltmarsh edge by large-amplitude tides (van Proosdij et al., 2006). In contrast, some marshes in the microtidal Venice Lagoon, Italy, are erosion-prone because of low river sediment supply, as well as limited tide-driven sediment mobilisation and transport (Day et al., 1999; Marani et al., 2007; Fagherazzi et al., 2013). Along sediment-starved coastlines, erosion of adjacent tidal flats can provide a local sediment source for marsh accretion (Schuerch et al., 2019) even if tidal flat loss eventually exposes marshes to long-term lateral erosion (Bouma et al., 2016). Marsh change is also associated with human activity. Land reclamation has reduced the extent of marshes globally (Gedan et al., 2009), while the introduction of invasive marsh building plants (*Spartina* species) has expanded marshes (Ranwell, 1967; Gedan et al., 2009). Large fluctuations in marsh cover have also been linked to changes in hydrology and sediment transport driven by coastal development and land-use change (Yang et al., 2001).

While numerical models have pioneered the mechanistic understanding of lateral marsh dynamics (Mariotti & Fagherazzi, 2010; Mariotti & Carr, 2014; Hu et al., 2015a; D'Alpaos & Marani, 2016; Kirwan et al., 2016b; Schuerch et al., 2018), empirical evidence has lagged behind and been limited to process-based studies (Feagin et al., 2009; Francalanci et al., 2013), isolated sites (Chauhan, 2009; Gunnell et al., 2013; McLoughlin et al., 2014) and single explanatory drivers of change (Weston, 2013; Gabler et al., 2017). We aimed to change this situation. Here we ask which key climate, biotic, and anthropogenic drivers best explain long-term (150-year), large-scale (across Great Britain) lateral marsh change.

2 Methods

2.1 Study sites

We measured change in saltmarsh extent for 25 estuaries and embayments located in 6 regions across Great Britain (GB): the Solway, Morecambe and Cardigan regions located along the west coast, in the Irish Sea; and the Wash, Essex-Kent and Solent regions along the east/south-east, in the North Sea and English Channel (Figure 1). In total, these estuaries occupied around 19,000 ha of salt marsh (~40% of the total marsh area in GB) (Phelan et al., 2011; Haynes, 2016). Estuaries were shallow, generally well-mixed with semidiurnal meso- to macro-tidal ranges. Flood-dominance was common along the west coast, the Wash region and many of the Essex-Kent regions, whereas in the Solent region, all the estuaries were ebb-dominant (Manning & Whitehouse, 2012). Typical estuary morphology ranged from bar-built to embayment/coastal plains (Pye & Blott, 2014). Relative sea level rise (RSLR) generally increases along an axis from the north-west to the south-east due to isostatic adjustment of the British Isles following deglaciation at the end of the Last Glacial Maximum (Bradley et al., 2009). Along a similar axis, tidal amplitude and estuary depth generally decrease, and sediment type changes from sand- to silt/clay-dominance (Goudie, 2013). All regions have historically seen some sea wall construction, with extensive stepwise reclamation occurring in the Wash and the Essex-Kent regions (Davidson et al., 1991). Fluvial suspended sediment supply to the coastline across the UK has been historically low (Worrall et al., 2013).

2.2 Change in saltmarsh extent

We quantified saltmarsh area for the entirety of each estuary approximately every 30 years between 1846 and 2016 using a combination of Ordnance Survey (OS) maps and aerial photographs. OS maps were accessed via the EDINA Digimap Resource Centre. Survey dates of maps were taken from Oliver (2013) and used as timestamps. For the Cardigan regions, aerial photographs were taken from the Royal Commission on Ancient Historical Monuments Wales. Photographs were scanned and georeferenced onto OS 1:25,000 rasters in the British National Grid projection. Pixel size corresponded to ca. 0.25×0.25 m in the field. Marsh extent measurements for the Solent and Essex-Kent regions, originally delineated from aerial photographs, were taken from Baily and Pearson (2007) and Cooper et al. (2001) respectively.

Marsh extent from OS maps and aerial photographs were delineated manually at a scale of 1:7,500 by placing vertices along the marsh edge approximately every 5 m. To account for boundary precision of the seaward marsh edge, visual comparisons between our georeferenced images to reference shapefiles (Phelan et al., 2011; Haynes, 2016) was done to ensure accuracy of the georeferencing procedure. We also looked for site-specific literature to verify whether observations of significant change in marsh extent could be considered ‘real’ or were likely caused by differences in map surveyors’ interpretation of where the marsh edge lay (see supporting information, Table S1). In the case of the Wash, large areas of marshland were reclaimed over the study period. To account for this, we calculated the area of reclaimed land and subtracted it from the marsh extent in the previous map revision. The new value was included as an additional measurement of marsh area between map revisions. See supporting information, Text S1, for methods used to calculate an error term for each measure of marsh area.

A linear rate of saltmarsh change per year was calculated for each estuary and used as the response variable in statistical modelling. Due to the highly non-linear change of marsh extent in the Wash region, an average rate of marsh change was calculated following each reclamation phase. See supporting information, Table S2, for the rates of marsh change, and the dates over which this rate was taken. We also contrasted observed rates of lateral marsh change with published empirical measurements of vertical accretion on the nearby marsh surface. All accretion rates were measured in the low-marsh zone using Caesium radio-isotope dating, Sediment Elevation Tables or Marker Horizons (see references in Table 1). See supporting information, Table S2, for dates over which accretion rates were measured.

2.3 Predictor variables of lateral marsh change

For each estuary, we collated data on key hydrological, sedimentological and climatological variables known to structure saltmarsh extent within estuaries. Annual net sediment flux per unit area of marsh was calculated by using the ratio of vegetated and unvegetated surfaces within each estuarine marsh complex (UVVR), which has been shown to be a proxy for external sediment supply (Ganju et al., 2017). See supporting information, Text S2, for information on how net sediment flux was calculated and validated. Estimated bedload sediment flux volume (in or out of the estuary) were taken from HR Wallingford (2002), Brown and Davies (2010), Halcrow (2010), and NFDC (2017). Due to differences in the precision of modelled bedload sediment flux estimates between studies, all values were rounded to their nearest 10^{th} value, representing a magnitude flux either into (positive) or out (negative) at the estuary mouth. Long-term tide gauge records were used to calculate the rate of RSLR for each estuary. Trends of RSLR are linear rates calculated from monthly-averaged records with a minimum 30-year timespan (NOAA, 2019). Where nearby tide gauges were unavailable, we took the average RSLR rate from two nearest equidistant stations. Admiralty Tide Tables were used to determine the mean tidal range of each estuary, taken from Manning and Whitehouse (2012). Frequency of storm events were calculated using daily averaged wind speed data from the UK Met Office Integrated Data Archive System (Met Office, 2012). Stations were selected based on their proximity to each estuary. The temporal range for each station varied considerably, although at most limited to between 1957 and 2016. As a consequence, some stations nearby had low number of samples and were rejected for further analysis. The final representation of stations was limited to one per region, and storm events recorded by that station were assumed to be representative of all estuaries for the respective region. Prior to analysis, wind speed data was screened for quality and completeness (see Watson et al. (2015) for method). Frequency of storm events were then estimated from annual datasets as a count above an absolute threshold of 23 ms^{-1} ('strong gale' on the Beaufort scale), and rate of change in number of events per year was used in the statistical analysis. Prevailing wind directions within 10-degree compass bearing intervals of each station were also used to calculate fetch length of each estuary (the distance over which wave-generating winds blow). The Waves Toolbox for ArcGIS 10.1 (Rohweder et al., 2012) with an 'SPM-Restricted' method was used to calculate fetch length every 200 m along the seaward marsh edge of each estuary (using a national marsh shapefile taken from Phelan et al. (2011) and Haynes (2016)). The median fetch lengths for each estuary were recorded. Rate change in river flood frequency events were calculated using number of Peaks-Over-Threshold per water year data provided by the National River Flow Archive (Robson and Reed, 1999). Predictor variables, and the timescale over which they were measured, are noted in supporting information, Table S2. Dates of *Spartina townsendii* and *Spartina anglica* (henceforth *Spartina spp*). Colonisation (Figure 2; grey shading) were taken from Goodman et al. (1959), Hubbard and Stebbings (1967), and Harwood and Scott (1999). Information on significant infrastructure projects (Figure 2; arrows) were taken from Kestner (1962), Marshall (1962), and Burd (1992) for the Solway, Wash and Essex-Kent regions.

2.4 Statistical treatment

All statistical analyses were implemented in R. Predictor variables were checked for outliers, and log- or cube-transformed to meet assumptions of normality and equal variance. Predictor variables were also checked for collinearity, and dropped if Variance Inflation Factors exceeded 3 (Zuur et al., 2009). To identify groupings across our study sites, we used pairwise Euclidean distances between all 25 estuaries and found 6 clearly defined regions (Figure 1). We then used region as a random variable to test for spatial autocorrelation, but did not find a significant effect. A Stepwise Linear Regression model was therefore used to select the minimal adequate model. See supporting information, Text S3, for details on the full statistical analysis used.

4 Results and Discussion

Our analysis of marsh extent change revealed a stronger tendency for seaward lateral marsh expansion than for marsh erosion. Five of the six regions increased in marsh cover by 29% to 158% between 1846 and 2016 (Figures 2a–e and 2h) and marshes overall expanded by 11%. South-east Britain was the only region to consistently lose marsh cover (Figures 2f and 2g). The largest lateral expansion occurred in the south, where Solent marshes had grown 307% by the 1970s before declining to their current levels; 29% greater than in 1868 (Figures 2d and 2h). The north-eastern Wash region lost large areas of salt marsh on four occasions due to land reclamation (Figure 2e; arrows), however new marshes always expanded laterally on the seaward side of walls, leading to a 52% overall increase in marsh area.

Effects of *Spartina* colonisation on long-term marsh change appeared to be limited. In estuaries where marsh areal extent had been increasing, trends of marsh expansion generally preceded the arrival of invasive *Spartina* (Figures 2a–c; grey shading), with the exception of the Solent region (Figures 2d and 2h; grey shading), where *Spartina* invasion has been substantial (Hubbard, 1965). Causes for erosion post-1970 in the Solent are unclear (Baily & Pearson, 2007), however studies have reported marsh loss through lateral marsh erosion which indicates losses may be related to dynamics at the salt marsh- tidal flat interface (Johnson, 2000). In the Essex-Kent region, eroding marshes saw a prolonged period of little marsh change between 1900 and 1970 during which *Spartina* was first recorded and several sea walls were breached by storms (Figures 2f and 2g; grey bars and white arrows). Overall, coastal works also had little effect on long-term marsh change. In the Wash and Solway regions, marshes expanded despite losses through reclamation (Figure 2e; black arrows) and canalisation (Figure 2a; grey arrows) respectively. The prevailing hydrological and sedimentological environment appeared to be conducive to achieving a new dynamic equilibrium in marsh extent (Kestner, 1975). Both the effects of the introduction of *Spartina* and coastal works appear to have only temporarily offset a long-term trend of marsh decline. We therefore conclude that long-term patterns of marsh lateral change were not driven by direct human impact alone.

We next considered which key drivers were responsible for lateral marsh change for the period between 1967 and 2016. Results from a Stepwise Linear Regression model showed that sediment flux per unit area and median fetch length in combination best explained (62% of variation) the rate of marsh lateral change in estuaries across Great Britain (Table S3). Marshes shifted from eroding to expanding when sediment flux and fetch length concurrently increased

(Figure 3). Bedload sediment flux was retained in the best fit model, but was not significant (Table S3).

From a range of key hydrological, sedimentological and climatological variables known to influence lateral marsh dynamics, we find that sediment supply plays a crucial role in explaining large-scale, long-term trends of lateral marsh change (Figure 3a). Whilst increases in fetch length are typically associated with marsh loss rather than expansion (Callaghan et al., 2010), the relatively sheltered meso- to macro-tidal estuaries in our study had small fetch lengths (averaging 1.2 km) compared to the ~10 km threshold fetch lengths needed to trigger runaway marsh erosion along microtidal U.S. coastlines (Mariotti & Fagherazzi, 2013). Since wave action is also responsible for sediment resuspension and transport (Green & Coco, 2014), it is likely that moderate increases in fetch length enhances sediment transport to the coast, thereby facilitating marsh accretion (Figure 3b) as observed along other macro-tidal coastlines (Pringle, 1995; van de Groot et al., 2011). Across Great Britain, marshes with larger wave fetch lengths also tended to have longer foreshore widths (Taylor et al., 2002). The presence of a wide foreshore can attenuate incoming waves, reducing the potential for marsh edge erosion (Bouma et al., 2014 and references therein). Additional field-based measurements would be required to ascertain whether a shift from marsh erosion to expansion across Great Britain is primarily influenced by increased wave-driven sediment transport to the coast, or greater wave-protection from wider foreshores. Nevertheless, our results provide empirical support for large-scale and long-term shifts in the lateral extent of marshes driven by sediment supply and transport, in agreement with numerical models (Mariotti & Fagherazzi, 2010).

Global declines in sediment supply to the coast could lead to large-scale marsh loss through lateral erosion, as observed along the eastern U.S. coast (Weston, 2013). A spatial shift over the 1967-2013 period, from marsh complexes with a positive sediment flux to marshes that have been exporting sediment (Figure 3a) implies there might have been differences in sediment availability across Great Britain. There is no evidence that fluvial suspended sediment flux to the UK coast has changed since 1974 (Worrall et al., 2013) and there is also no indication that marine sediment sources have depleted over the past 50 years (HR Wallingford, 2002; Halcrow, 2010; NFDC, 2017). Intertidal flats, which can provide a local sediment source for marsh accretion (Mariotti & Carr, 2014; Schuerch et al., 2019), have reduced in size across GB since 1843 (Taylor et al., 2004; Pontee, 2011). More severe reductions in tidal flat widths along south and eastern England (Taylor et al., 2004) may have impaired their capacity to supply marshes with enough sediment to keep pace with sea level rise, exposing the marsh edge to long-term lateral erosion (Figure 2d and 2f-h). Estuaries with a greater capacity for sediment remobilisation and transport by wave action (Figure 3b) may have allowed marshes to continue to expand at the expense of tidal flat erosion (Figure 2a-c and 2e). Without increases in sediment supply to the coast, trends of lateral marsh erosion are likely to continue (Figure 2d and 2f-h) and may reverse trends of marsh expansion currently observed in the northern regions of Great Britain (Figure 2a-c and 2e).

Given that studies of marsh stability have tended to focus on whether or not vertical growth is equal to or greater than local sea level rise (Crosby et al., 2016; Kirwan et al., 2016a; Spencer et al., 2016; Horton et al., 2018; Schuerch et al., 2018; Valiela et al., 2018), we also compared our rates of lateral marsh change with the rates of vertical marsh accretion (references within Table 1) versus RSLR for each region. We found that all marshes had a positive accretion balance (Table 1). Marshes can erode at their flanks, but still accrete with RSLR, because lateral erosion provides a sediment source for vertical accretion (Mariotti & Carr, 2014). Coupled lateral and vertical marsh dynamics may therefore better predict saltmarsh resilience than comparing marsh vertical growth against RSLR alone (Mariotti & Carr, 2014; Gonneea et al., 2019; Kirwan et al. 2016a; Kirwan et al. 2016b).

Schemes involving managed realignment of the coastline with engineering solutions to control sediment supply and tidal inundation can be used to build large-scale and long-term marsh resilience in historically eroding systems including San Francisco Bay, U.S.A. (Stralberg et al., 2011), and the Scheldt estuary, Netherlands (Vandenbruwaene et al., 2011). Despite such large investments into the restoration of saltmarsh flood protection, the monitoring of short-term sediment dynamics at the marsh edges (Bouma et al., 2016) and profile changes of tidal flats (Taylor et al., 2004; Pontee, 2011; Murray et al., 2014) is rarely done. This hampers the ability to predict whether marsh restoration schemes are likely to succeed or fail. Having shed light on the key drivers of long-term saltmarsh lateral change, researchers should now capitalise on advances in satellite remote sensing (Dorji et al., 2016) and novel and cheap instruments to quantify the short-term sediment dynamics at the coast (Hu et al., 2015c) to evaluate coastal resilience against human- and environment-induced change at a global scale. The evidence presented here contributes to an emerging emphasis on investigating the causes for spatial shifts in coastal systems, including mudflats (Murray et al., 2014), seagrass beds (Suykerbuyk et al., 2015) and mangroves (Gabler et al., 2017). Though important, a shift away from a focus on sea level rise alone to consider also the influences of other anthropogenic and macroclimatic drivers of coastline change should be a priority.

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The GIS layers showing change in marsh extent and the variables used in the statistical analysis for this study are accessible via the Environmental Information Data Centre repository (DOI: 10.5285/03b62fd0-41e2-4355-9a06-1697117f0717).

The authors declare no conflict of interest.

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Table 1. Rates of lateral and vertical marsh change per region. Mean ± S.E values per region for marsh lateral expansion rates, and the rate of marsh vertical accretion, minus the rate of relative sea level rise, to give the 'accretion balance'. Measures of vertical marsh accretion rates were unavailable for the Morecambe region.

Region	Lateral expansion (ha yr^{-1})	Accretion balance (mm yr^{-1})
Solway	0.88 ± 1.17	15.41 ± 14.53 (Marshall, 1962)
Morecambe	2.94 ± 0.37	n.a.
Cardigan	2.31 ± 1.37	8.25 ± 4.06 (Kestner, 1975)
Wash	1.27 ± 0.00	46.17 ± 26.87 (Shi, 1993)
Essex-Kent	-6.42 ± 3.55	3.20 ± 3.56 (Cundy & Croudace, 1996)
Solent	-3.59 ± 1.65	2.91 ± 0.84 (van der Wal & Pye, 2004)

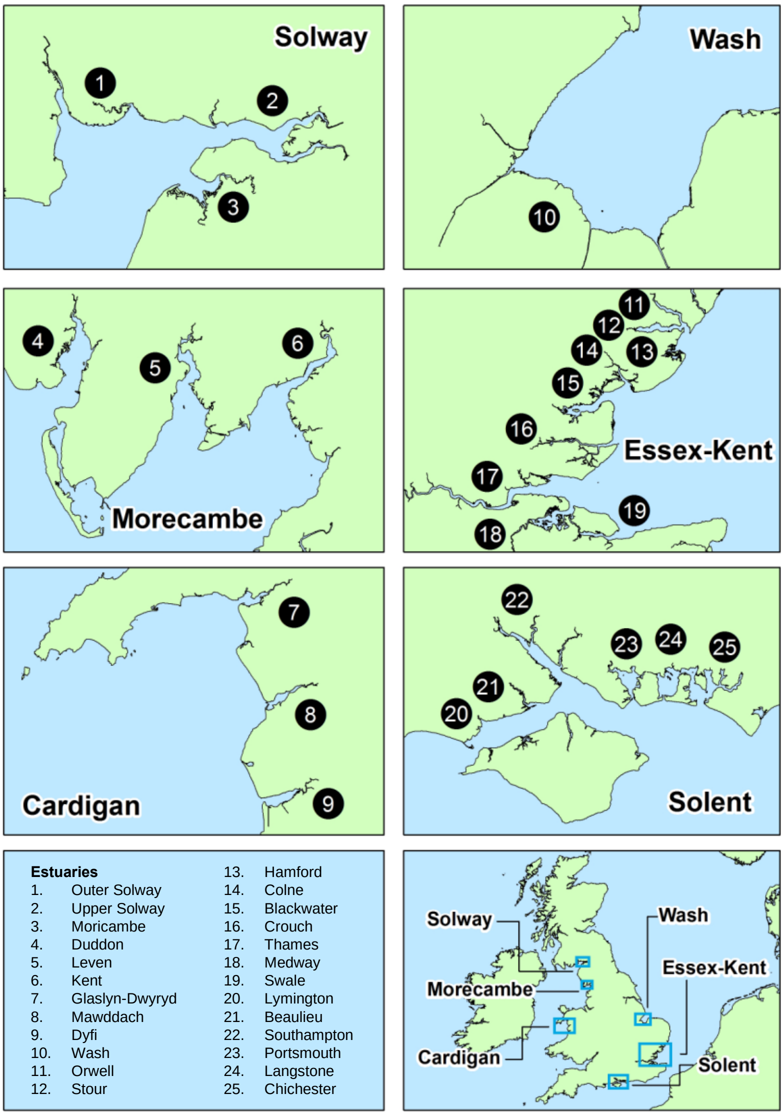

Figure 1. Estuaries examined within each region. A total of 25 estuaries separted into 6 regions across Great Britain.

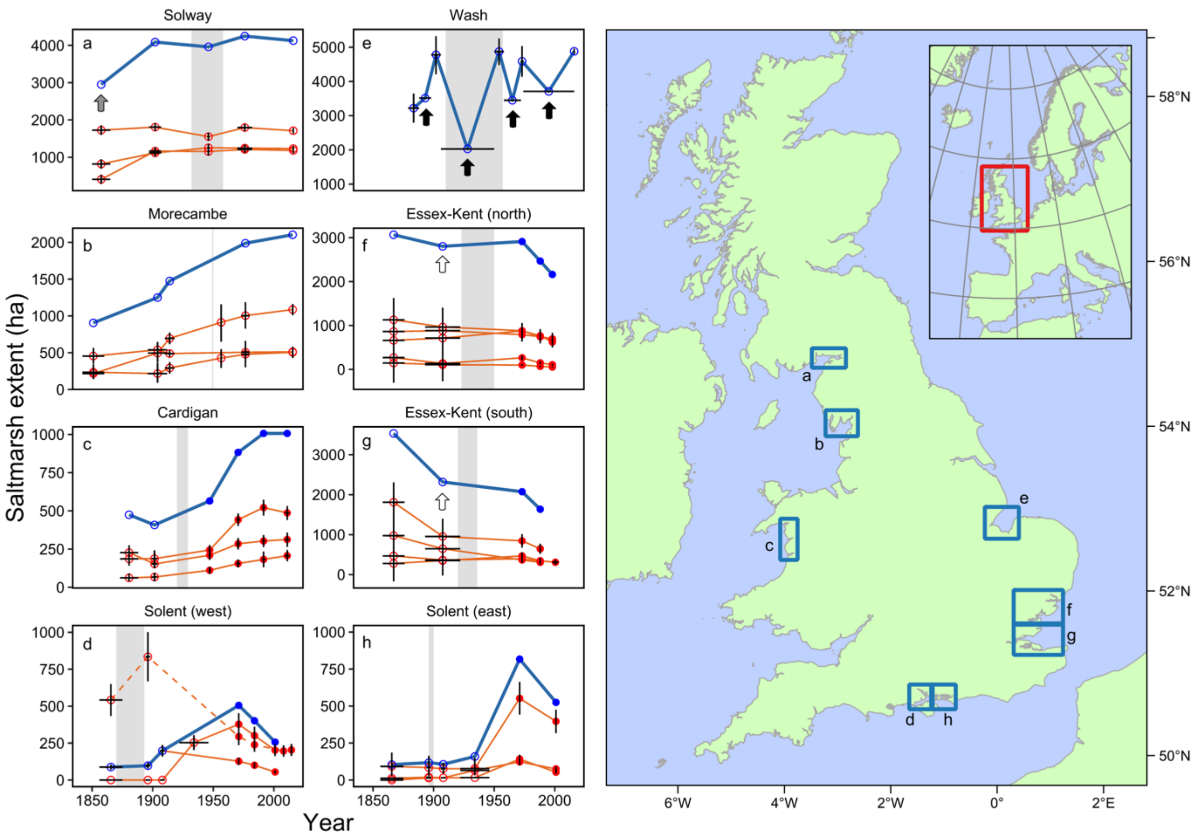

Figure 2. Change in estuarine-scale marsh extent across Great Britain. Regional- (blue line) and estuarine-scale (orange line) change in areal extent of salt marshes between 1856 and 2016 from photographs (filled circles) or maps (hollow circles). Arrows indicate occurrences of embankment (solid arrow), canalisation (grey arrow) or collapse of sea walls after storms (hollow arrow). Grey shading indicates *Spartina spp.* Colonisation in each region. Vertical error bars indicate 95% confidence intervals in marsh area extent. Horizontal lines indicate the dates over which surveys of marsh extent were carried out. Essex-Kent and Solent regions have been subdivided for ease of presentation. Regional-scale marsh change (blue line) only includes marsh extent measures for all estuary in a given region and year. Marsh change in Southampton estuary (panel d: dashed line) was excluded from the regional-scale marsh change line due to paucity of contiguous cover in saltmarsh extent across multiple years.

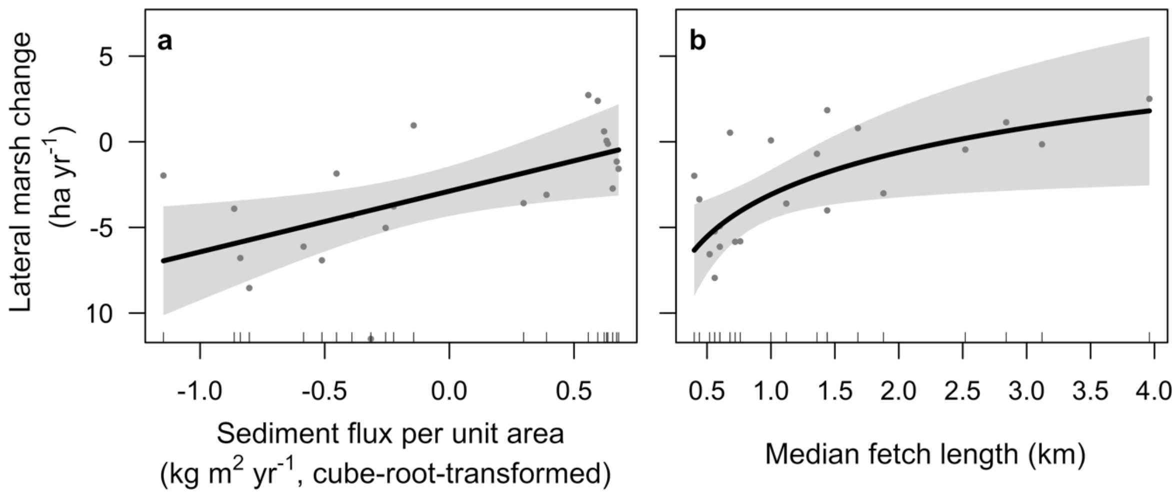

Figure 3. Relationships of estuarine-scale lateral marsh change with two significant predictor variables identified from a best-fit linear regression model on data for 1967 to 2016 (n = 22): **a** sediment flux per unit area and **b** median fetch length. Data points represent distribution of standardised partial residuals. Solid lines represent model-fit though the data, bounded by 95% confidence intervals (solid grey shading). Tick marks along the bottom of each plot denote deciles of the distribution of each predictor value.

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Supporting Information for

Sediment supply explains long-term and large-scale patterns in saltmarsh lateral expansion and erosion

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Introduction

The following supporting information contains details on how error resulting from digitising saltmarsh extent across Great Britain was calculated (Text S1), how net sediment fluxes values across marsh complexes were calculated and validated from the unvegetated/vegetated ratio (Text S2), and a report on the full statistical treatment applied to our data (Text S3). Tables S1 to S3 provide further details on literature that corroborates marsh extent change detected from the GIS study, the variables used to explain change in marsh extent, and the results of a Stepwise Linear Regression model used to identify which key variables explain marsh change respectively.

Region	Area of rapid change	Description and reference
Solway	Rapid expansion and erosion phases of all major marshes in each estuary	OS maps record the expansion and erosion of these larger marshes throughout the Solway region, with a net increase in marsh extent. A study by Marshall (1962) reaches a similar conclusion from interpreting maps (1856-1864) and aerial photographs (1946). Marshall (1962) also investigated aspects of saltmarsh morphology (accretion rates, areas of erosion and expansion and channel configuration) to corroborate their findings and conclude that there is close agreement between phases of expansion and erosion observed in the field and change recorded from maps. Rapid expansion of Caerlaverock is also described by Bridson (1980) from maps dating back to 1654, in agreement with Marshall (1962). Firth et al. (2000) and CCO (2011) describe the rapid expansion and erosion of different parts of Rockcliffe marsh (Inner Solway) and the marshes in Moricambe Bay since 1776 and 1864 from literature searches and site visits, which we observe in the OS maps.
Morecambe	Rapid erosion and expansion phases in all estuaries	OS maps record the overall expansion of marshes throughout the Morecambe Bay region, with individual marshes undergoing extensive erosion and accretion phases. Of the most drastic change, phases of erosion in Silverdale marsh have been documented by Pringle (1995) from repeat transect measurements between 1983 and 1992, which is out of phase with marsh expansion at Grange-over-Sands on the opposite side of the estuary, described in field surveys by Gray (1972). All estuaries in Morecambe Bay region are considered dynamic and have experienced rapid changes in saltmarsh extent determined from site visits, historical records and modelling data (CH2M HILL, 2013a, 2013b, 2013c; Dixon-Gough, 2006). These observations support marsh change observed in OS maps.
Cardigan	Rapid expansion of marshes in the outer estuary	OS maps reveal that marshes throughout the Cardigan Bay region have expanded gradually, with more rapid rates of expansion around the 1950s. Field sketches made by Yapp (1917) include detailed vegetation surveys of the Dyfi estuary in the mid 1910s which show a similar extent of saltmarshes to the OS maps. Rapid expansion of the marshes in the outer estuary in the late 1940s is documented by Chater & Jones (1957) from site surveys. Both studies are in agreement with observations of marsh change from OS maps.
Wash	Rapid expansion following reclamation	OS maps document the step-wise loss of marshlands to reclamation, followed by phases of new marsh growth in front of seawalls throughout the Wash embayment. Kestner (1962) reconstructed past area cover of salt marsh extent for the Wash by knowing the dates when seawalls were constructed. Kestner (1975) later described the mechanism by which sediment deposits in front of the sea wall, allowing marshes to rapidly colonise and expand. These phases of reclamation and new marsh growth are in agreement with marsh change determined from OS maps.
Essex-Kent	Gradual erosion across all estuaries, with some areas of rapid marsh expansion	OS maps record the gradual erosion of marshes across Essex-Kent, with small areas of rapid expansion or erosion subject to embankment/deembankment. Burd (1992) report in detail the numerous areas of embankment and deembankment from history books, parish and estate records, property deeds, maps and historical surveys from the 17th century onwards. Areas of reclaimed and deembanked marshland is reported by Wolters et al. (2005). Burd (1992) also report that the government at the time were aware of marsh erosion and more frequent flooding due to land subsidence and rising sea levels. Kirby (2013) and Spearman et al. (2014) also document marsh decline from historical maps and illustrations in the northern and southern parts of the region respectively.
Solent	Rapid expansion across all estuaries	OS maps record the expansion of marshes across the Solent region. Baily & Inkpen (2013) account for the rapid expansion because of the colonisation and spread of the pioneer-marsh hybrid *Spartina townsendii*, and later, fertile allotetraploid species *Spartina anglica*, onto tidal flats across the region. Baily & Inkpen (2013) support this argument by referring to a number of articles published during this period that document the nature and spread of *Spartina spp.* across the region. Baily & Inkpen (2013) did find issues with the accuracy of maps compared to aerial photographs, however much of the error was due to not knowing the date a map was surveyed (only when it was published) when comparing to an aerial photograph, and revision errors where marshes were copied over to successive map editions thus not representing the true marsh extent of that year. Both error terms were accounted for in our study and described in Text S1.

Table S1. Literature searched to determine whether marsh change from maps can be considered 'real'.

	Lateral expansion (ha yr^{-1})	Vertical accretion (mm yr^{-1}) ± S.D.	Relative sea level rise (mm yr^{-1})	Net sediment flux (kg m^{-2} yr^{-1})	Suspended sediment conc. (mg l^{-1})
				(2006-2009)	(2000)
Solway	(1970-2016)	(1961)	(1960-2015)		
Outer Solway	0.14	21.43 ± 10.56	2.28 ± 0.65	0.313	404.22
Upper Solway	2.22	11.79 ± 18.12	2.28 ± 0.65	0.302	404.22
Moricambe	0.27	25.40 ± 0.00	2.28 ± 0.65	0.281	404.22
Morecambe	(1967-2010)		(1960-2015)		
Duddon	2.84	NA	2.34 ± 0.68	0.211	291.49
Leven	3.35	NA	2.34 ± 0.68	0.173	313.42
Kent	2.62	NA	2.34 ± 0.68	0.251	313.42
Cardigan	(1969-2013)		(1938-2016)		
Glaslyn-Dwyryd	1.56	NA	2.27 ± 0.30	0.212	89.37
Mawddach	1.48	NA	2.27 ± 0.30	0.155	NA
Dyfi	3.89	10.29 ± 4.06 (1988-1989)	2.27 ± 0.30	0.239	111.03
Wash	(1972-2016)		(1955-2016)		
Wash	1.27	48.00 ± 26.87 (1956-1962)	1.83 ± 0.38	0.257	295.97
Essex-Kent	(1973-1998)		(1933-2016)		
Orwell	-1.85	NA	1.89 ± 0.27	-0.003	64.46
Stour	-6.39	NA	1.89 ± 0.27	-0.017	64.46
Hamford	-9.98	NA	1.89 ± 0.27	-0.517	59.51
Colne	-3.81	NA	1.89 ± 0.27	0.059	78.69
Blackwater	-7.99	4.84 ± 5.00 (1963-1998)	1.22 ± 0.19	-0.200	122.68
Crouch	-6.50	6.70 ± 0.00 (1897-1994)	1.22 ± 0.19	-0.011	106.40
Thames	-3.93	3.93 ± 1.38 (1963-1998)	1.22 ± 0.19	-0.060	NA
Medway	-13.22	4.05 ± 2.05 (1989)	1.22 ± 0.19	-0.031	86.05
Swale	-4.11	NA	1.22 ± 0.19	0.026	NA
Solent	(1971-2001)		(1961-2016)		
Lymington	-4.86	5.2 ± 0.00 (1893-1995)	1.62 ± 0.46	-0.134	42.54
Beaulieu	-2.44	3.3 ± 0.00 (1893-1995)	1.62 ± 0.46	-0.224	54.12
Southampton	-4.31	4.8 ± 0.14 (1870-1995)	1.62 ± 0.46	-0.093	102.87
Portsmouth	-2.72	NA	1.62 ± 0.46	-1.509	83.39
Langstone	-1.45	1.5 ± 0.00 (1907-1995)	1.62 ± 0.46	-0.643	79.57
Chichester	-5.77	NA	1.62 ± 0.46	-0.590	78.56

Table S2. Site characteristics for all 25 estuaries divided into 6 regions. Parentheses indicate the timescales over which rates were measured, or in which empirical data was used to derive values. These dates are representative of either the entire Great Britain, a whole region, or a specific estuary.

	Bedload sediment flux ($m^3 yr^{-1}$)	Wind storm frequency ($n yr^{-1}$)	River flood frequency ($n yr^{-1}$)	Median fetch length (m)	Tidal range (m)
				(2006-2009)	(2006-2009)
Solway	(2010)	(1961-2008)			
Outer Solway	10,000,000	0.02	1.34 (1979-2014)	3,120	5.92
Upper Solway	100,000	0.02	-1.36 (1963-2014)	2,520	5.92
Moricambe	0	0.02	0.00	1,880	5.92
Morecambe	(2010)	(1957-2015)			
Duddon	10,000	0.01	0.79 (1968-2014)	1,000	6.12
Leven	1,000,000	-0.15	0.44 (1939-2014)	1,440	6.36
Kent	10,000,000	-0.15	0.28 (1968-2014)	3,960	6.36
Cardigan	(2010)	(1957-2015)			
Glaslyn-Dwyryd	NA	0.09	0.34 (1961-2014)	520	3.04
Mawddach	NA	0.09	NA	1,040	2.94
Dyfi	-1,000,000	0.09	0.02 (1962-2013)	1,360	2.90
Wash	(2002)	(1969-2015)			
Wash	10,000,000	-0.01	0.04 (1939-1996)	2,840	4.42
Essex-Kent	(2002)	(1957-2015)			
Orwell	10,000	0.04	0.11 (1964-1996)	400	2.64
Stour	10,000	0.04	-0.13 (1928-1992)	600	2.64
Hamford	1,000	0.04	0.00	560	2.64
Colne	10,000	0.04	0.07 (1959-2014)	720	3.24
Blackwater	10,000	0.04	0.10 (1932-1968)	520	3.48
Crouch	1,000,000	-0.50	0.21 (1976-2014)	560	3.76
Thames	-1,000,000	-0.50	0.27 (1883-2014)	600	5.00
Medway	100,000	-0.13	-0.02 (1956-2014)	600	4.08
Swale	10,000	-0.13	0.00	760	3.90
Solent	(1998)	(1957-2015)			
Lymington	0	0.06	0.57 (1960-2014)	1,440	1.56
Beaulieu	0	0.06	NA	240	2.26
Southampton	10,000	0.06	0.05 (1972-2014)	440	2.72
Portsmouth	1,000	0.06	0.26 (1951-2014)	680	2.82
Langstone	1,000	0.06	0.11 (1979-2014)	1,680	2.98
Chichester	1,000	0.06	0.04 (1967-2014)	1,120	2.98

Table S2. *Continued*.

Model variables	Estimate	SE	*t*-Value	*P* value	R^2
Best model fit (AIC = 116.79, F = 9.95, *df* = 3, 18, $P < 0.001$***, $R^2 = 0.62$)					
Net sediment flux (kg m^{-2} yr^{-1})	3.547	1.332	2.662	0.016*	0.48
Median fetch length (km)	3.549	1.323	2.682	0.015*	0.46
Bedload sediment flux (m^3 yr^{-1})	-0.014	0.010	-1.495	0.152 n.s.	0.06

$P < 0.05$*, $P < 0.01$**, $P < 0.001$***, n.s. = $P > 0.05$.

Table S3. Predictor variables explaining lateral saltmarsh change (1967-2016) identified by best fit models (Stepwise Linear Regression) for 25 estuaries across Great Britain.

Text S1. Estimating error in saltmarsh area cover.

We calculated change in the areal extent of salt marshes across Great Britain using maps and aerial photographs. Analysis was done in ArcGIS 10.1. In order to quantify an error term associated with these measurements, we calculated the Root Mean Squared Error (RMSE) which describes the average deviation of observed points from their true positions (Wernette et al., 2017). Four independent RMSE sources are associated with geographical data: displacement of the basemap, to which historical maps and aerial photographs are referenced, from its 'true' location on the Earth's surface ($RMSE_B$); distortions in historical maps and aerial photographs that introduce error when georeferencing to a basemap ($RMSE_G$); interpreter error when digitising the salt marsh at a given scale ($RMSE_I$), and; errors introduced by the cartographer when presenting spatial data on a map (not relevant for aerial photographs) ($RMSE_M$). Because each error source is independent, they can be added for a total error estimate. To determine distances, in metres, below which 95% of the positional errors in delineated salt marsh edges are expected to fall, FGDC (1998) recommend the added RMSE values are multiplied by 1.7308 in order to calculate $RMSE_{95}$, given as:

$$RMSE_{95} = 1.7308\left(\sqrt{\left(RMSE_B{}^2 + RMSE_G{}^2 + RMSE_I{}^2 + RMSE_M{}^2\right)}\right)$$

Maps produced between 1842 and 1952 (Six-inch County Series Edition) the Ordnance Survey (OS), the national mapping agency of the UK, were produced using ground surveys. Demarcating the seaward limit of the salt marsh accurately is dependent on the cartographer's capacity to survey difficult-to-reach or dangerous areas, and distinguish the edge of the marsh which is often 'fuzzy' (due to patchy growth of plants) (Baily & Collier, 2010; Baily, 2011; Baily & Inkpen, 2013). OS standards on the quality and accuracy of saltmarsh surveying were not stringent (Baily & Inkpen, 2013) therefore the marsh edge is sometimes represented as a stamped symbol without a clearly defined margin (Baily & Inkpen, 2013). OS maps produced after 1952 (National Series Edition maps) were compiled using a combination of ground surveys and aerial photographs. Delineating the marsh edge from aerial photographs accurately depends on surveyor capacity to correctly distinguish plants from other features (such as macroalgae patches) as well as the quality of the aerial image. There is no specific guidance set by the OS on demarcating the marsh edge from aerial photographs (OS, pers. comm., 2018). Baily and Inkpen (2013) assessed how successful OS ground-surveys were at determining the marsh edge by comparing maps with aerial photographs captured near the map publication date. Where maps were surveyed at similar times to when images were taken, both media were in close agreement.

A value for the positional error of digitised marsh edge (map or photo) from the true position ($RMSE_I$) was not given by Baily and Inkpen (2013). To calculate $RMSE_I$, we selected an example marsh boundary (a 5 km section saltmarsh edge in the Wash) to digitise at very high resolution (vertice placed every metre) at high magnification to capture the 'true' marsh edge from an OS map.

Resolution of the map was then scaled to 1:7500, and the marsh edge was digitised once again to capture the 'interpreted' marsh edge. Distance from the 'interpreted' line to the 'true' line was calculated every 20 metres along perpendicular lines from the 'true' line. This is the same procedure used when assessing interpreter error for maps and aerial photographs. $RMSE_I$ is given as:

$$RMSE_I = \sqrt{\left(\frac{\sum d^2}{n}\right)}$$

Where:

d is the distance between the 'true' and 'interpreted' marsh edge

n is the number of distance measurements.

An additional error term, associated with maps produced from ground surveys only, is the interpretation of the surveyor of where the marsh edge lies which is then reproduced on a map as a line or stamp ($RMSE_M$). Given that marsh edges from maps and photos have been shown to be in close agreement (Baily & Inkpen, 2013), $RMSE_M$, is assumed to be of the same magnitude as $RMSE_I$:

$$RMSE_M = RMSE_I$$

Both $RMSE_M$ and $RMSE_I$ should be included for estimates of marsh extent taken from maps that have drawn from ground surveys.

$RMSE_G$ in maps and aerial photographs can arise during and after the survey. For maps, inaccuracies arise when noting positions from traditional trigonometry surveys or modern Geographical Positioning Systems. After publication, historical maps can distort over time through shrinkage and stretching before digitisation occurred. For aerial photographs, tilt, pitch and yaw of the aeroplane will affect the angle at which images were taken. Unevenness of the topography being captured will also cause distortions to the image. After acquisition, both the original film and reprints can distort over time once produced. These issues reduce the accuracy of features on maps and aerial photographs once images are georeferenced. Georeferencing distortion can be calculated by the distance from which the source deviates from a reference position (Jongepier et al., 2016) as follows:

$$RMSE_G = \sqrt{\frac{\left(\sum V_{xy}{}^2\right)}{n-2}}$$

Where *n* is the number of points and V_{xy}, is a displacement vector made up of vector distances v_x and v_y (in metres) between the distorted points and the reference positions. Both V_x and V_y are calculated as follows:

$$V_{xy} = \sqrt{\left(v_x{}^2 + v_y{}^2\right)}$$

$RMSE_G$ was calculated for all maps and photographs to OS 1:2500 basemaps as reference, using MapAnalyst (Jenny & Hurni, 2011). 12 well-distributed control points were identified in both the source and OS 1:2500 maps and the $RMSE_G$ between them was calculated using a Helmert transformation (Jongepier et al., 2016). Measurements of marsh extent for the Essex-Kent and Solent regions were taken from Cooper et al. (2001) and Baily & Pearson (2007). Cooper et al. (2001) does not report $RMSE_G$ for their survey, however Baily and Pearson (2007) report a precision value of between ± 3 and 5 m. An average $RMSE_G$ of 4 metres was taken for their survey and applied to Essex-Kent and Cardigan Bay regions where aerial photography was used to delineate marsh extent.

The OS 1:2500 maps used as a reference for measuring distortions in older maps and aerial photographs in this study are themselves subject to some positional error between the ‘real life’ position and that recorded on the map known as $RMSE_B$. For the OS 1:2500, $RMSE_B$ of 1.1 metres has been calculated (HMLR, 2016). $RMSE_{95}$ is a linear measure (units in metres). In order to express $RMSE_{95}$ for areal measures, we constructed a buffer area around the inner and outer circumference of each marsh where the width was $RMSE_{95}$ calculated for each source (Wernette et al., 2017). The buffer area indicates the minimum and maximum size of the marsh to provide an error term for each extent measurement, representing a 95% confidence interval. Calculating a buffer area was not possible for values taken from Cooper et al. (2001) and Baily and Pearson (2007), and no error term is reported by these authors. A marsh buffer area was therefore estimated for each study. The buffer area was estimated by resampling marsh extent from aerial photographs of Cardigan Bay, but at the image scales used by Cooper et al. (2001) and Baily and Pearson (2007) (1:5,000 and 1:10,000 respectively). $RMSE_{95}$ was recalculated, then the percentage difference in area extent between delineated marsh extent and maximum/minimum area buffers were calculated for each scale. Marsh extent was found to vary by ±18.4 and ±20.0% at scales of 1:5,000 and 1:10,000 respectively. These error margins were applied to the values of marsh extent taken from Cooper et al. (2001) and Baily and Pearson (2007) as the $RMSE_{95}$ error term. The final error term for marsh area extent can be considered conservative, because accuracy in delineating the marsh edge in many cases will be much higher. For example, where the back of the marsh is bounded by a clearly-defined seawall that can be mapped to a high degree of accuracy.

After the OS produced first edition maps (County and National Series), revisions were soon needed to keep maps up-to-date in a rapidly developing landscape. However, revisions did not always include complete re-surveys of an area. Revisions tended to be made only for areas heavily used by people, whilst less important features were simply copied over from the previous edition known as ‘partial-revisions’ (Baily & Inkpen, 2013). Salt marshes were not always resurveyed during map revisions, and when revisions occurred, the specific area that had been revised was not always recorded (Baily, 2011). In our study, revision error was accounted for by comparing map revisions against first editions of each marsh in each estuary. On the assumption that the marsh boundary is likely to change during a ~30 year period, marshes that had near-identical boundaries in both first and revised editions were considered copied, so areal extent was not calculated.

All RMSE values are contained within the attributes table of the marsh extent change GIS layer accessible via the Environmental Information Data Centre repository (DOI: 10.5285/03b62fd0-41e2-4355-9a06-1697117f0717).

Text S2. Calculating and validating net sediment flux.

Sediment supply is a key predictor of long-term marsh stability (e.g. Kirwan et al., 2016), yet empirical measurements of sediment flux across marshes are sparse (Ganju et al., 2015). Recent work has shown that the ratio of unvegetated surfaces such as tidal channels, saltpans and marsh edges (sites of sediment erosion) to vegetated marsh areas (sites of sediment accretion) can act as a proxy for external sediment supply (Ganju et al., 2017). We calculated the UVVR values for all marshes in our study and used regression fits from (Ganju et al., 2017) to derive measures of net sediment flux. We also validated the flux rates against estimated measues of suspended sediment concentration for UK estuaries.

We used a combined Great Britain-wide saltmarsh extent shapefile (collated by the UK Environment Agency [EA] and the Scottish National Heritage [SNH]) to calculate the UVVR for each estuary. Both the EA and SNH shapefiles represent the vegetated portions of marshes across Great Britain and distinguish vegetated marshes from tidal channels and salt pans. The EA captured colour aerial images with 10 cm resolution for the UK coastline between 2006 and 2009. Images were georeferenced with root mean square error ranging from 10 cm to 1 m. Marsh delineation was done manually, and digitally using various feature-identification techniques. Creeks less than 1.5 m wide and marshes less than 5m^2 were overlooked. In cases where there was low confidence in mapping results, site visits were made to ground-truth the digitised marsh surface (Phelan et al., 2011). The SNH mapped salt marshes larger than 3 ha from colour aerial photographs captured between 2003 and 2009 at a 1:4,000 scale across the Scottish coastline. All creeks, salt pans, and other marsh features were mapped when above the mapping resolution. Marsh edges were compared to field surveys to ensure accuracy (Haynes, 2016).

In a GIS, we used both saltmarsh extent shapefiles to calculate the area of vegetated portions for each marsh complex within our target estuary. We then applied a workflow of ArcGIS 10.1 tools to outline the overall marsh complex, thereby effectively separating tidal channel and salt pan features from the vegetated marsh surface. The original shapefile was then subtracted from this 'boundary' layer to calculate the area of unvegetated portions within the marsh complex (e.g. Figure S1). UVVR was then calculated (A_{UV}/A_V) for all 25 estuaries. We then regressed the values of UVVR and net sediment flux reported in Ganju et al. (2017) to give us an equation for predicting net sediment flux ($y = -0.855 \ln x + 0.330$). We fitted the values of UVVR calculated for each marsh complex in our study into the regression equation to derive measures of net sediment flux.

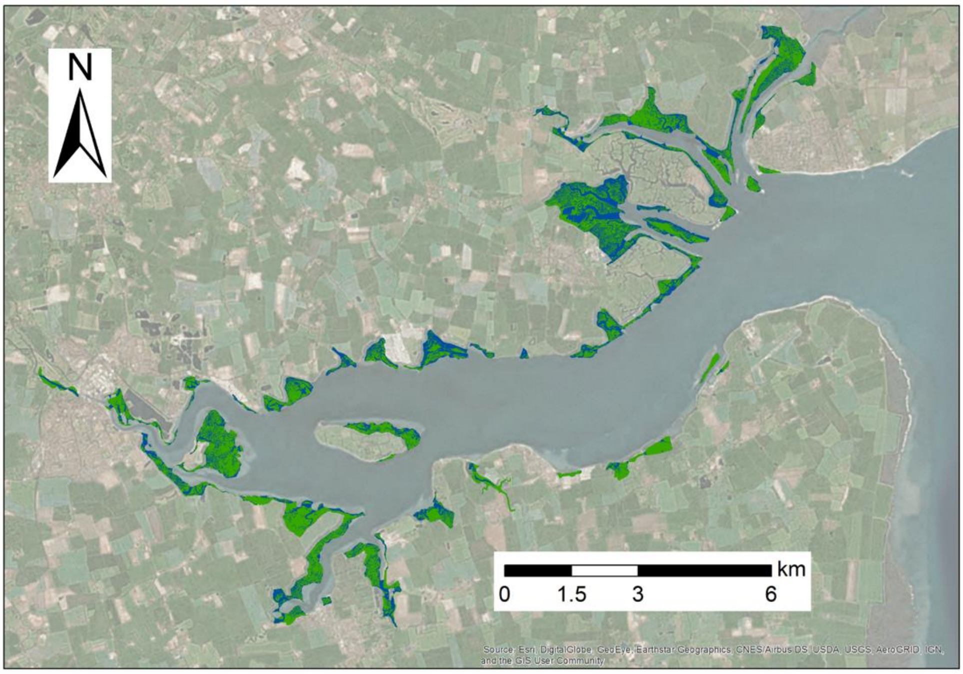


Figure S1. Example of how vegetated and unvegetated portions of a marsh are identified for the Blackwater estuary, south-east Great Britain. Unvegetated areas are shown in blue, vegetated surfaces are shown in green. Saltmarsh extent was taken from the UK Environment Agency, and the marsh boundary was determined using a series of polygon processing tools in ArcGIS 10.1. Imagery from the ArcGIS World Imagery Basemap.

To validate our use of sediment flux estimated from UVVR as a predictor of lateral marsh change, we correlated sediment flux values against an estimated measure of estuarine maximum static time- and depth-averaged fine cohesive suspended sediment concentration (SSC_E) (Manning & Whitehouse, 2012). SSC_E is indicative of sediment supply, and has been shown to broadly represent real conditions in validation studies (Prandle et al., 2005, see Figure 4). We used a Pearson correlation to find that SSC and sediment flux were significantly positively correlated ($R = 0.68$, $p < 0.001$) (Figure S2). We therefore consider our use of net sediment flux a suitable indicator of external sediment supply. SSC_E values for each estuary are shown in Table S2.

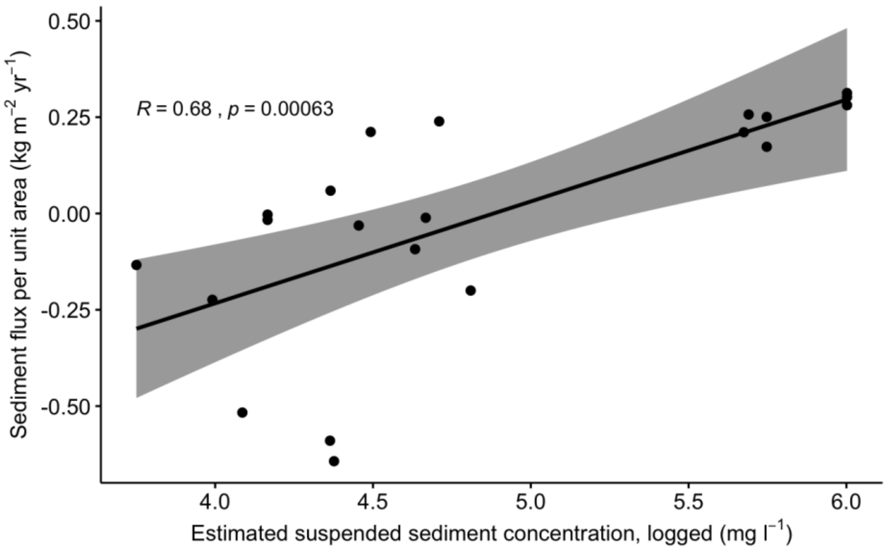


Figure S2. Pearson correlation between sediment flux and estimated suspended sediment concentration ($R = 0.68$, $p < 0.001$).

Text S3. Data analysis and model selection.

In this section, we present the statistical analysis used to determine which environmental drivers best describe rate of saltmarsh change across Great Britain. All statistical analyses are carried out using the 'R' software package. The data used in the statistical analysis is accessible via the Environmental Information Data Centre repository (DOI: 10.5285/03b62fd0-41e2-4355-9a06-1697117f0717).

Prior to selecting a statistical model, we tested the necessary assumptions of each statistical model. We begin by loading the dataset, graphics package ggplot2, and some additional functions held in the `additional_functions.R` file:

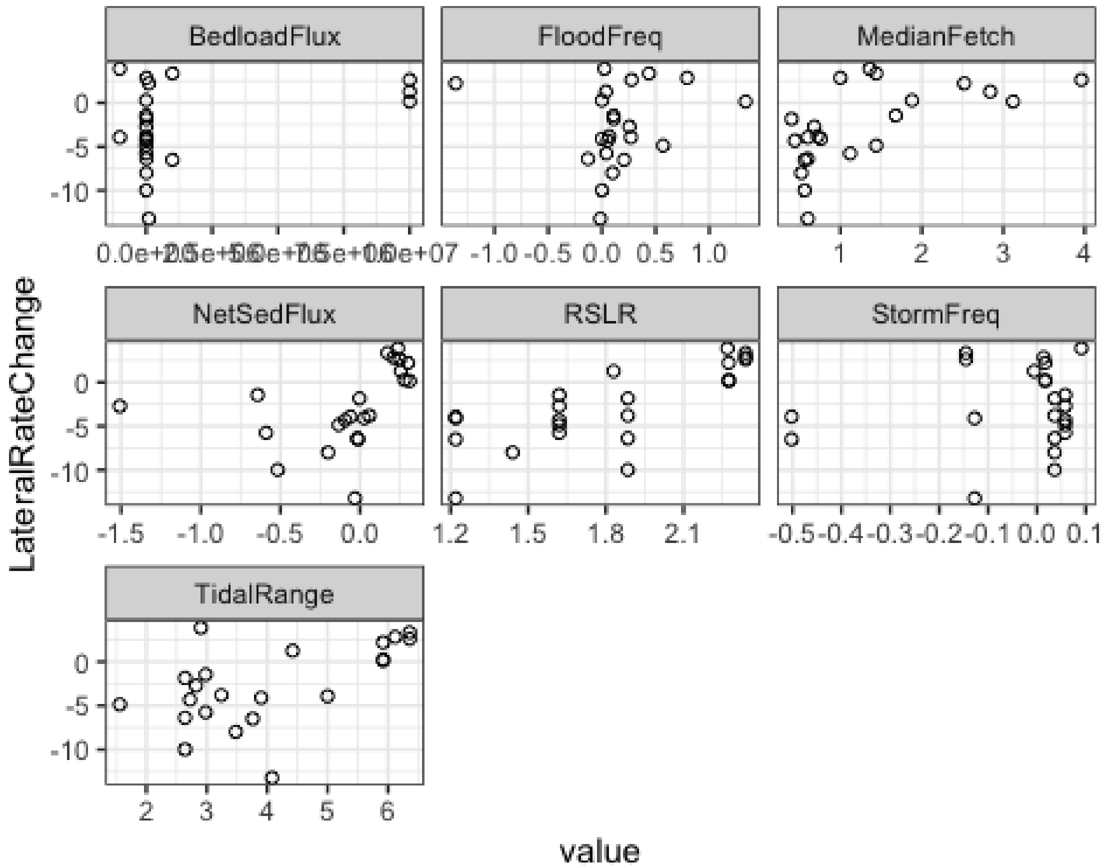
```
library(ggplot2)
source("/Users/Home/Code/additional_functions.R")
marshes<-read.csv("/Users/Home/Data/PredictorVariables.csv",header=T)
```

There are 4 cases in the dataset where data on river flood frequency change and bedload sediment flux were unavailable of a given estuary. We therefore subset the dataset by removing NAs:

```
marshes_RM<-marshes[complete.cases(marshes),]
```

Using the subset dataset without NAs (`marshes_RM`), we begin our data exploration by investigating how each predictor variable relates to rate of saltmarsh change. To make this easier, we create a new object containing the response and predictor variables only:

```
library(dplyr)
library(tidyr)

marshcheck<-marshes_RM[-c(1:3)]

marshcheck %>%
  gather(-LateralRateChange,key="var",value="value") %>%
  ggplot(aes(x=value,y=LateralRateChange))+
  geom_point(shape=1)+
  facet_wrap(~var,scales="free_x")+
  theme_bw()
```

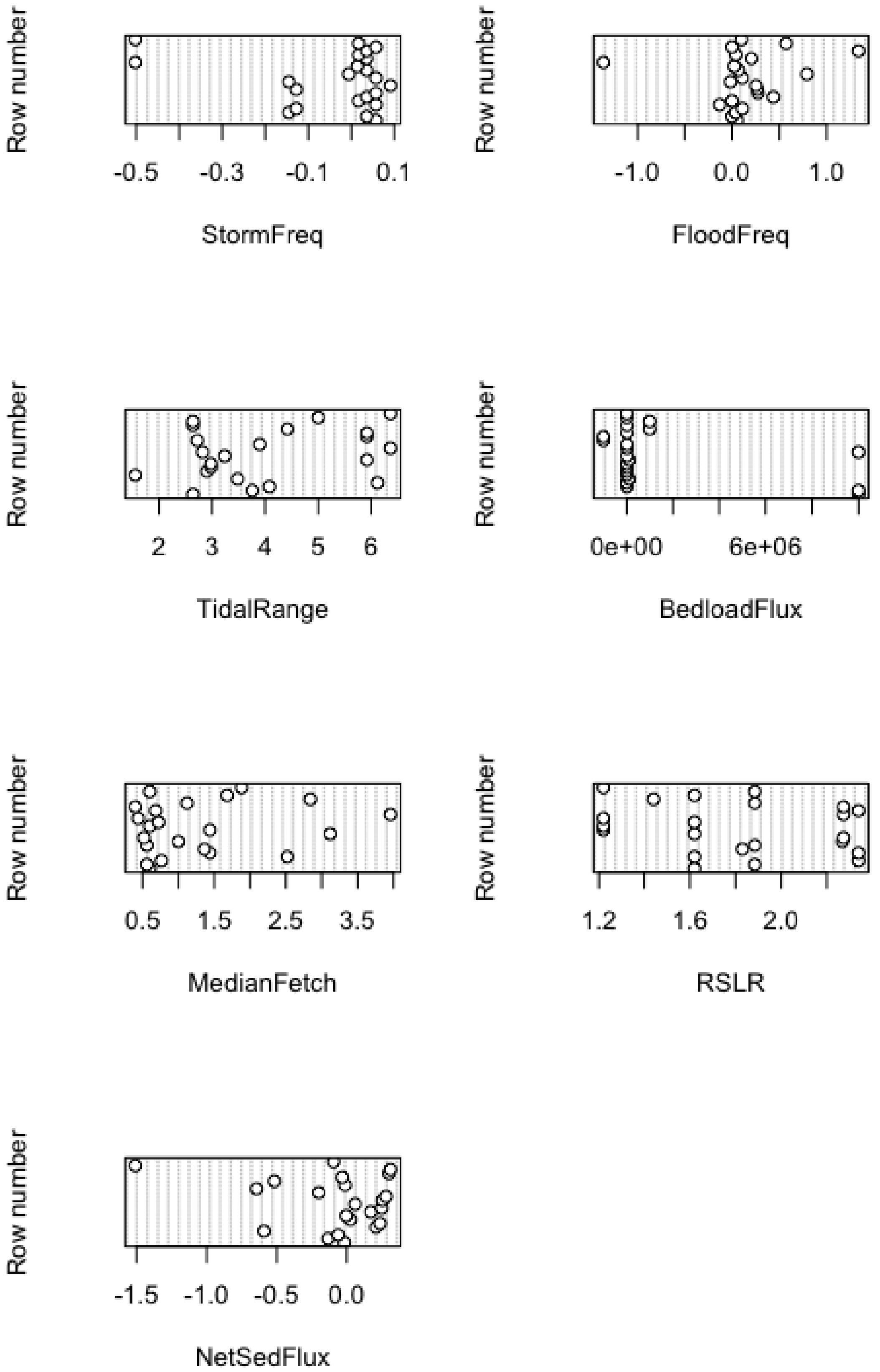


It appears that increases in fetch distance, sediment flux per unit area in/out of the marsh, relative sea level rise rate, and tidal range, may all be associated with shifts from marsh erosion to expansion. The other predictor variables do not appear to have a relationship with lateral marsh change.

We used Cleveland dotplots to identify any extreme outliers in our dataset. Outliers may have a significant impact on the results. The data is organised along the y-axis only by row name (i.e. the order in which it was entered into the dataframe):

```
par(mfrow=c(2,2))

lapply(X=c("StormFreq","FloodFreq","TidalRange","BedloadFlux","MedianFetch
```

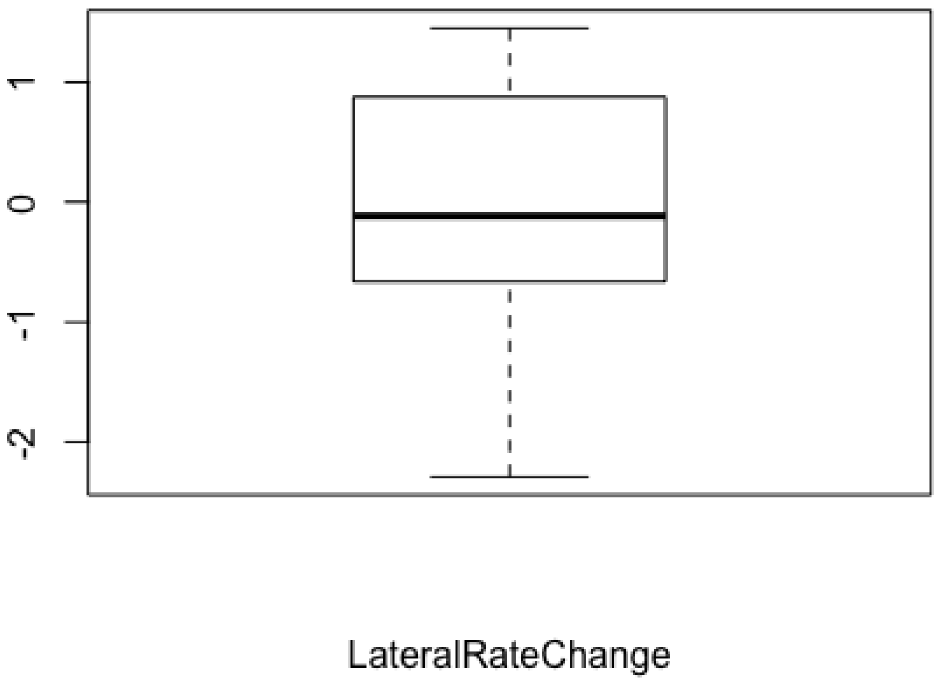
```
","RSLR","NetSedFlux"), FUN=function(s)
  dotchart(sample(marshcheck[, s]), xlab=s,ylab="Row number"))
```


Row number
-0.5 -0.3 -0.1 0.1
StormFreq
Row number
-1.0 0.0 1.0
FloodFreq
Row number
2 3 4 5 6
TidalRange
Row number
0e+00 6e+06
BedloadFlux
Row number
0.5 1.5 2.5 3.5
MedianFetch
Row number
1.2 1.6 2.0
RSLR
Row number
-1.5 -1.0 -0.5 0.0
NetSedFlux

Large variation from the centre of data clusters suggests the presence of outliers. *StormFreq*, *FloodFreq*, *BedloadFlux*, and *NetSedFlux* however none appears to be unprecedented. These are not unprecedented amounts, but may affect our model results. We will consider their effect when testing the final model assumptions.

To check whether the response variable has a normal distribution, we build a boxplot, and scale the y axis to make their ranges comparable:

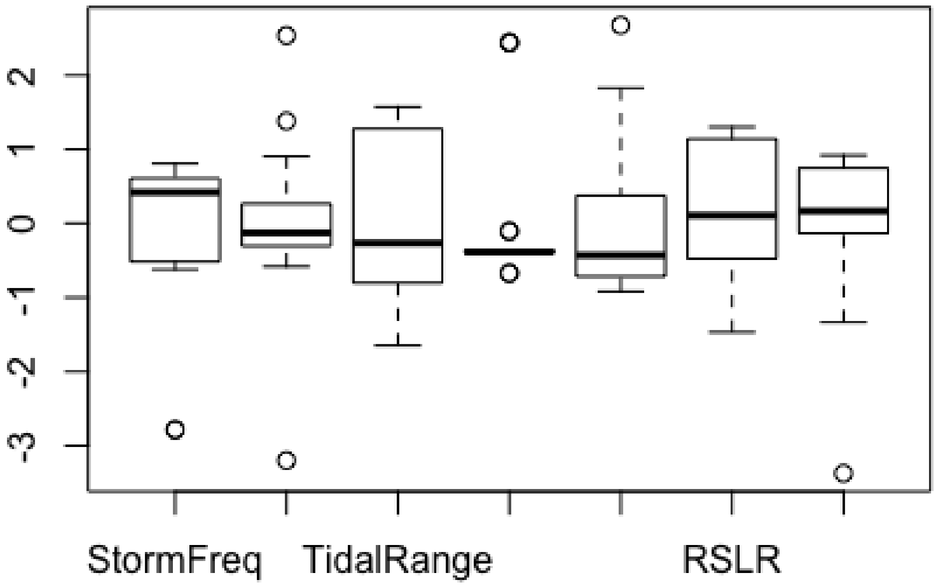
```
boxplot(scale(marshes_RM$LateralRateChange),
              xlab="LateralRateChange")
```




```
shapiro.test(marshes_RM$LateralRateChange)

##
##  Shapiro-Wilk normality test
##
## data:  marshes_RM$LateralRateChange
## W = 0.96174, p-value = 0.5253
```

There is no evidence of skew. We next consider the distribution of all predictor variables:

```
boxplot(scale(marshes_RM[,5:11]))
```

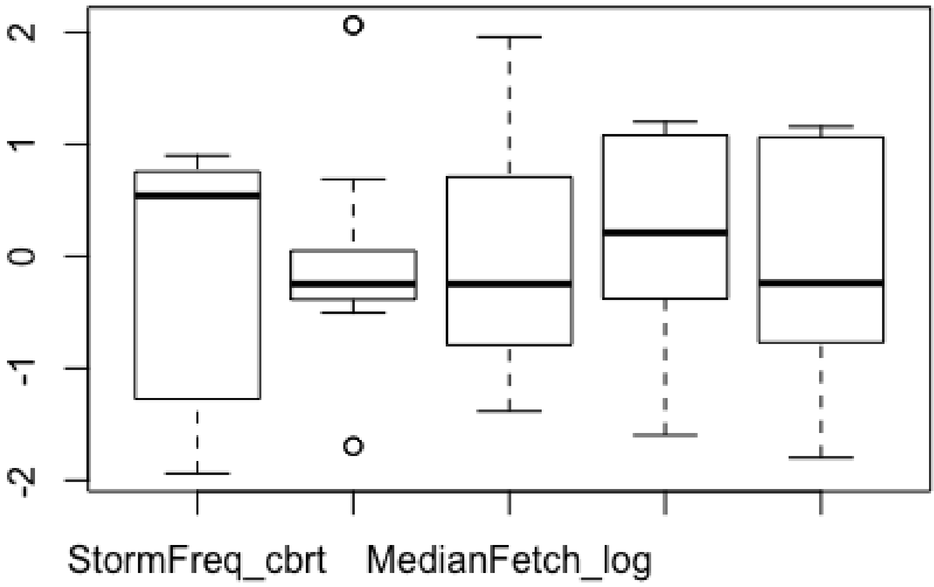


There is some evidence of skew in the predictor variables, notably for `StormFreq`, `BedloadFlux`, `MedianFetch`, and `RSLR`. Some of these are likely influenced by outliers in the data. Transformations using log- and cube-root- (capable of transforming both negative and positive values. See the `Math.cbrt` function in the `additional_functions.R` file) may produce more normal populations suitable for parametric modelling. We apply the transformations and add these to our dataframe:

```
marshes_RM$StormFreq_cbrt<-Math.cbrt(marshes_RM$StormFreq)
marshes_RM$BedloadFlux_cbrt<-Math.cbrt(marshes_RM$BedloadFlux)
marshes_RM$MedianFetch_log<-log(marshes_RM$MedianFetch)
marshes_RM$RSLR_log<-log(marshes_RM$RSLR)
marshes_RM$NetSedFlux_cbrt<-Math.cbrt(marshes_RM$NetSedFlux)
```

We next inspect the distribution of the transformed predictor variables:

```
boxplot(scale(marshes_RM[,12:16]))
```



The transformed predictor variables now have a more symmetric distribution. We consider the distribution of our data suitable for parametric modelling.

Prior to using a parametric model to determine which suite of variables best explains rate of saltmarsh areal extent change, we need to check for high collinearity between predictor variables, and reduce it if necessary. We can examine the Variance Inflation Factor associated with each predictor variable (see `corvif` function in `additional_functions.R` file) to assess how much variance of an estimated regression coefficient increases if variables are correlated. Values over 3 are a cause for concern (Zuur et al., 2009):

```
corvif(marshes_RM[,c("StormFreq_cbrt","FloodFreq","TidalRange","BedloadFlu
x_cbrt","MedianFetch_log","RSLR_log","NetSedFlux_cbrt")])
```

```
## Correlations of the variables
##
##                 StormFreq_cbrt   FloodFreq TidalRange BedloadFlux_cbrt
## StormFreq_cbrt      1.00000000 -0.06642290 -0.4471729       -0.2730783
## FloodFreq          -0.06642290  1.00000000  0.1015293        0.2630995
## TidalRange         -0.44717292  0.10152926  1.0000000        0.4555946
## BedloadFlux_cbrt   -0.27307833  0.26309947  0.4555946        1.0000000
## MedianFetch_log    -0.01892536  0.07428223  0.5719886        0.5589768
## RSLR_log            0.41207815  0.06286752  0.4446737        0.2756686
## NetSedFlux_cbrt    -0.21050716  0.02345964  0.7079982        0.4041456
##                 MedianFetch_log   RSLR_log NetSedFlux_cbrt
## StormFreq_cbrt      -0.01892536 0.41207815     -0.21050716
## FloodFreq            0.07428223 0.06286752      0.02345964
## TidalRange           0.57198855 0.44467372      0.70799815
```

```
## BedloadFlux_cbrt      0.55897675 0.27566864      0.40414564
## MedianFetch_log       1.00000000 0.59019663      0.58825538
## RSLR_log              0.59019663 1.00000000      0.60255965
## NetSedFlux_cbrt       0.58825538 0.60255965      1.00000000
##
##
## Variance inflation factors
```

```
## Warning in summary.lm(object): essentially perfect fit: summary may be
## unreliable
```

```
##                       GVIF
## StormFreq_cbrt    3.519783
## FloodFreq         1.099187
## TidalRange        3.490519
## BedloadFlux_cbrt  1.751466
## MedianFetch_log   2.334006
## RSLR_log          4.365753
## NetSedFlux_cbrt   2.883363
```

There is high collinearity caused by `RSLR_log` (VIF = 4.37). We test whether collinearity has dropped to acceptable levels after excluding `RSLR_log`:

```
corvif(marshes_RM[,c("StormFreq_cbrt","FloodFreq","TidalRange","BedloadFlu
x_cbrt","MedianFetch_log","NetSedFlux_cbrt")])
```

```
## Correlations of the variables
##
##                    StormFreq_cbrt   FloodFreq TidalRange BedloadFlux_cbrt
## StormFreq_cbrt         1.00000000 -0.06642290 -0.4471729       -0.2730783
## FloodFreq             -0.06642290  1.00000000  0.1015293        0.2630995
## TidalRange            -0.44717292  0.10152926  1.0000000        0.4555946
## BedloadFlux_cbrt      -0.27307833  0.26309947  0.4555946        1.0000000
## MedianFetch_log       -0.01892536  0.07428223  0.5719886        0.5589768
## NetSedFlux_cbrt       -0.21050716  0.02345964  0.7079982        0.4041456
##                    MedianFetch_log NetSedFlux_cbrt
## StormFreq_cbrt         -0.01892536     -0.21050716
## FloodFreq               0.07428223      0.02345964
## TidalRange              0.57198855      0.70799815
## BedloadFlux_cbrt        0.55897675      0.40414564
## MedianFetch_log         1.00000000      0.58825538
## NetSedFlux_cbrt         0.58825538      1.00000000
##
##
## Variance inflation factors
```

```
## Warning in summary.lm(object): essentially perfect fit: summary may be
## unreliable
```

```
##                       GVIF
## StormFreq_cbrt    1.502641
## FloodFreq         1.093159
## TidalRange        2.869967
## BedloadFlux_cbrt  1.739354
## MedianFetch_log   2.257387
## NetSedFlux_cbrt   2.257476
```

All VIF scores are below 3, so we therefore proceed with investigating which predictor variables best explain lateral marsh change using General Linear Models (GLMs). Before proceeding with model selection, we need to account for any spatial autocorrelation that might invalidate our model. We use hierarchical clustering to identify groups in our data.

Accounting for spatial autocorrelation can dramatically improve model performance, and help to avoid biased estimates of Type I error. We will examine whether there are groupings between our estuaries, based on their pairwise Euclidean distances. If so, this will form a random factor in the model structure.

We start by creating the matrix of Euclidean distances between estuaries:

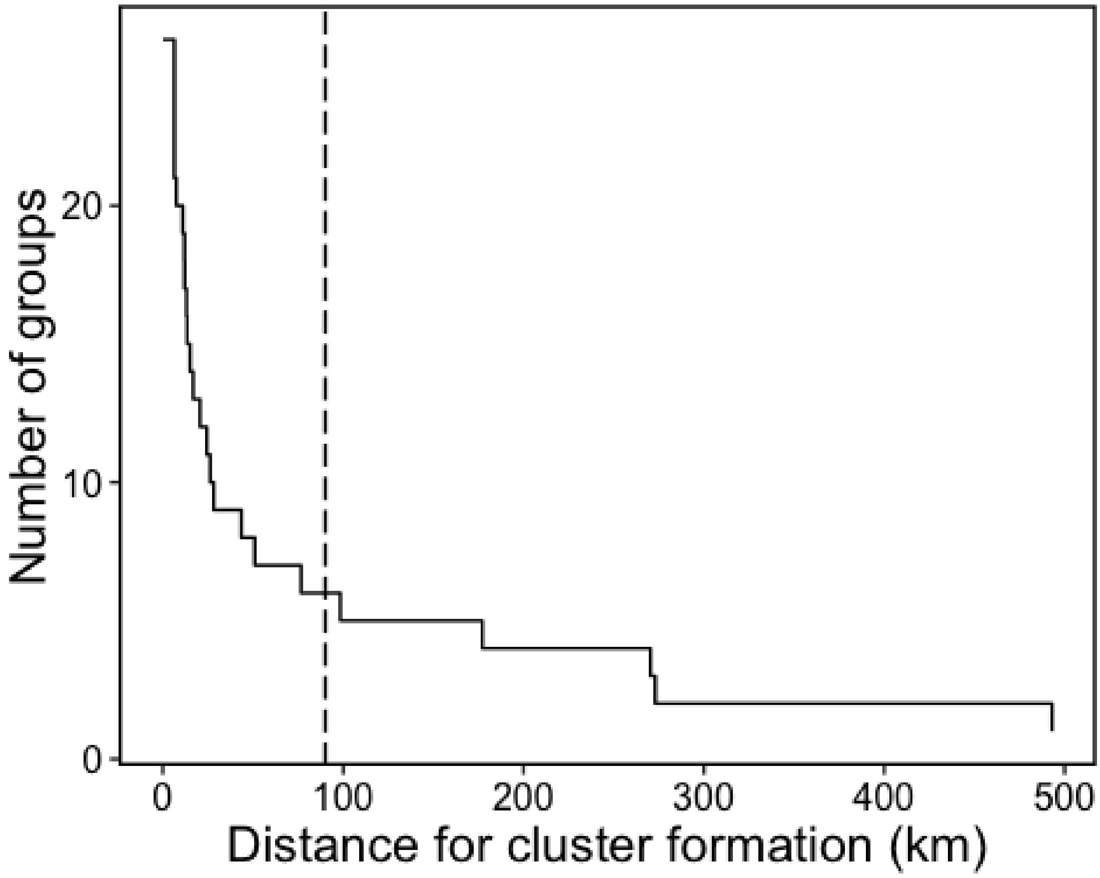
```
library(gmt)

distance_estuary_matrix=matrix(NA,length(marshes_RM$Estuary),length(marshe
s_RM$Estuary))
for(est1 in 1:length(marshes_RM$Estuary)){
   for(est2 in est1:length(marshes_RM$Estuary)){

distance_estuary_matrix[est1,est2]=geodist(marshes_RM$Latitude[est1],marsh
es_RM$Longitude[est1],
marshes_RM$Latitude[est2],marshes_RM$Longitude[est2], units="km")
   }}
distance_estuary_matrix=as.dist(t(distance_estuary_matrix))
full=hclust(distance_estuary_matrix,method="complete")
```

We can extract the cophenetic distance between groups to select a value for the inflection point in intra-estuary group variance and use Elbow plots to validate our selection. First, we prepare our data:

```
dist_clust<-data.frame(data.frame(unique(as.numeric(cophenetic(full))))[or
der((unique(as.numeric(cophenetic(full)))),decreasing=T),])
names(dist_clust)<-"distance"
dist_clust$group<-seq.int(nrow(dist_clust))
add_zero<-c(0,26)
dist_clust<-rbind(dist_clust,add_zero)
```

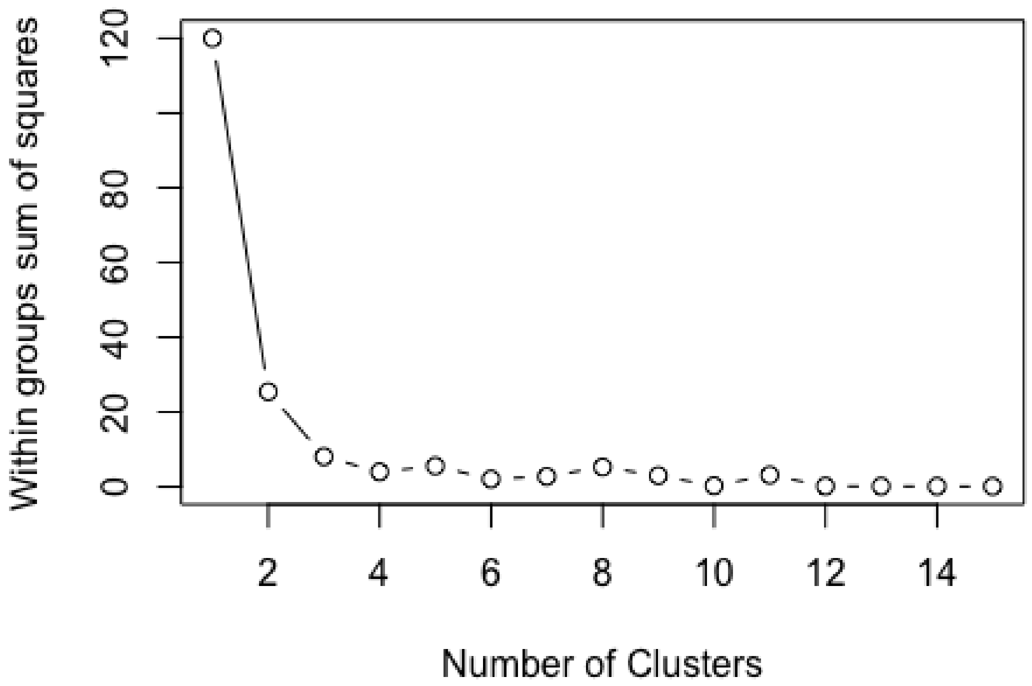
Now we can plot the number of groups against distance to form the clusters:

```
ggplot(dist_clust,aes(distance,group))+
  geom_step()+
  geom_vline(xintercept=90,linetype="longdash")+
  xlab("Distance for cluster formation (km)")+
  ylab("Number of groups")
```



Number of groups reduces exponentially with distance, and we see an 'evening out' at around 90 km, suggesting this is a suitable inflection point to distinguish our groups. This ends up producing 6 distinct hierarchical clusters. We validate our selection of 6 clusters using Elbow plots with k-means clustering method:

```
wss<-(nrow(subset(marshes_RM,select=c(Longitude,Latitude)))-1)*sum(apply(s
ubset(marshes_RM,select=c(Longitude,Latitude)),2,var))
for (i in 2:15) wss[i] <- sum(kmeans(subset(marshes_RM,select=c(Longitude,
Latitude)),
                                     centers=i)$withinss)

plot(1:15, wss, type="b", xlab="Number of Clusters",
     ylab="Within groups sum of squares")
```



Sum of squares within groups flattens out at 6 groups. This is in agreement with the previous plot, so we can justify grouping our estuaries into 6 hierarchical clusters. We now cut the hierarchical clustering analysis tree at the 90 inflection point to form our 6 groups and bind as a new column to our dataframe:

```
group=cutree(full, h=90)
marshes_RM=cbind(marshes_RM,group) # column bind the dataset and the previ
ously determined grouping
marshes_RM$group<-as.factor(marshes_RM$group)
```

If we inspect the dataframe marshes, we see that group identity is the same as the region in which each estuary occurs. We will now build a linear model and determine whether inclusion of region as a random factor improves the model. Before selecting an appropriate GLM, we consider whether inclusion of location (represented by `group` from the hierarchical clustering analysis) as a random effect term significantly improves a maximal model fit using a Restricted Maximum Likelihood (REML) approach (Zuur et al., 2009). We construct two models, one with a random effect, and one without. Anova tables can be used to see if there is a significant difference between the models. Since the model is using REML, we need to adapt the significance level using the L Ratio (Zuur et al., 2009):

```
library(nlme)

m1<-gls(LateralRateChange
        ~ 1 + StormFreq_cbrt+FloodFreq+TidalRange+BedloadFlux_cbrt+MedianF
```

```
etch_log+NetSedFlux_cbrt,
        method = "REML",
        control="optim",
        data=marshes_RM)

m2<-lme(LateralRateChange
        ~ 1 + StormFreq_cbrt+FloodFreq+TidalRange+BedloadFlux_cbrt+MedianF
etch_log+NetSedFlux_cbrt,
        random = ~1|group,
        method = "REML",
        control="optim",
        data=marshes_RM)

anova(m1,m2)
```

```
##    Model df      AIC      BIC    logLik   Test  L.Ratio p-value
## m1     1  8 117.1529 122.8173 -50.57646
## m2     2  9 117.3425 123.7150 -49.67127 1 vs 2 1.810397  0.1785
```

```
0.5*(1-pchisq(1.810397,1))
```

```
## [1] 0.08923031
```

Though close (p=0.089), there is no significant difference between the models. We can therefore use the more parsimonious model (`m1`, without accounting for spatial groups).

Stepwise Linear Regression can be used to identify which suite of predictor variables best explain the response variable, based on whichever models produce the lowest AIC scores. We apply a Stepwise Linear Regression to our data, using a forwards-and-backwards selection criterion to drop terms based on AIC scores. We switch from `gls` to `lm` to do this:

```
m3<-lm(LateralRateChange
       ~ StormFreq_cbrt+FloodFreq+TidalRange+BedloadFlux_cbrt+MedianFetch_
log+NetSedFlux_cbrt,
       data=marshes_RM)

step(m3,direction="both")
```

```
## Start:  AIC=56.25
## LateralRateChange ~ StormFreq_cbrt + FloodFreq + TidalRange +
##     BedloadFlux_cbrt + MedianFetch_log + NetSedFlux_cbrt
##
##                    Df Sum of Sq    RSS    AIC
## - FloodFreq         1     4.115 154.22 54.842
## - TidalRange        1     4.605 154.71 54.911
## - StormFreq_cbrt    1     9.586 159.69 55.609
## <none>                          150.11 56.247
## - BedloadFlux_cbrt  1    16.489 166.59 56.539
## - NetSedFlux_cbrt   1    37.316 187.42 59.131
## - MedianFetch_log   1    39.266 189.37 59.359
##
## Step:  AIC=54.84
## LateralRateChange ~ StormFreq_cbrt + TidalRange + BedloadFlux_cbrt +
```

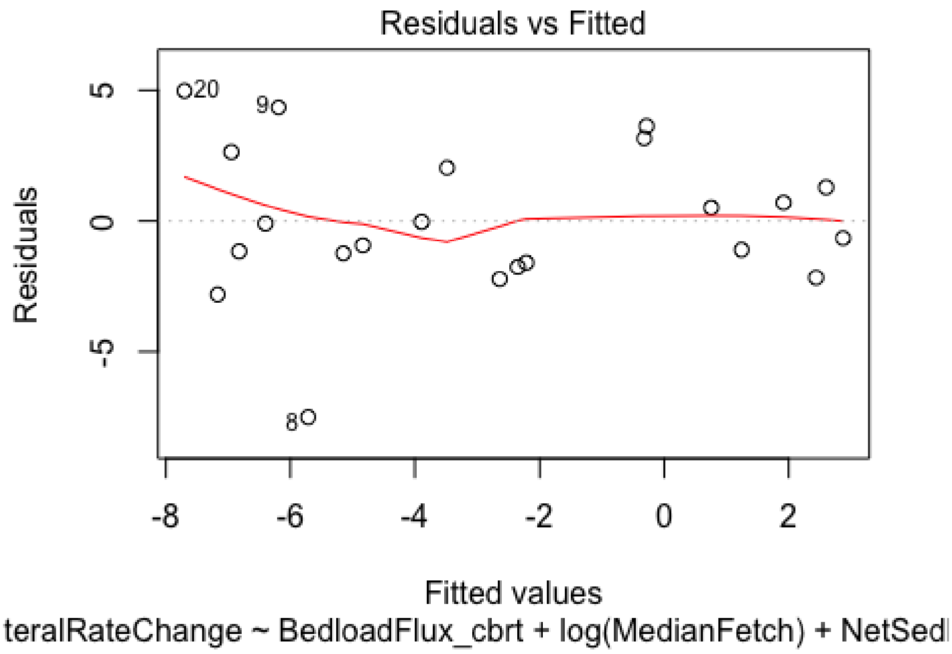
```
##     MedianFetch_log + NetSedFlux_cbrt
##
##                     Df Sum of Sq    RSS    AIC
## - TidalRange         1     5.309 159.53 53.586
## - StormFreq_cbrt     1    10.152 164.37 54.244
## - BedloadFlux_cbrt   1    13.370 167.59 54.671
## <none>                           154.22 54.842
## + FloodFreq          1     4.115 150.11 56.247
## - NetSedFlux_cbrt    1    35.490 189.71 57.398
## - MedianFetch_log    1    37.604 191.82 57.642
##
## Step:  AIC=53.59
## LateralRateChange ~ StormFreq_cbrt + BedloadFlux_cbrt + MedianFetch_log
+
##     NetSedFlux_cbrt
##
##                     Df Sum of Sq    RSS    AIC
## - StormFreq_cbrt     1     5.686 165.22 52.357
## - BedloadFlux_cbrt   1    12.994 172.52 53.309
## <none>                           159.53 53.586
## + TidalRange         1     5.309 154.22 54.842
## + FloodFreq          1     4.820 154.71 54.911
## - MedianFetch_log    1    52.575 212.10 57.853
## - NetSedFlux_cbrt    1    70.467 230.00 59.635
##
## Step:  AIC=52.36
## LateralRateChange ~ BedloadFlux_cbrt + MedianFetch_log + NetSedFlux_cbr
t
##
##                     Df Sum of Sq    RSS    AIC
## <none>                           165.22 52.357
## - BedloadFlux_cbrt   1    20.510 185.73 52.931
## + StormFreq_cbrt     1     5.686 159.53 53.586
## + FloodFreq          1     4.912 160.30 53.693
## + TidalRange         1     0.843 164.37 54.244
## - NetSedFlux_cbrt    1    65.049 230.26 57.660
## - MedianFetch_log    1    66.010 231.23 57.752

##
## Call:
## lm(formula = LateralRateChange ~ BedloadFlux_cbrt + MedianFetch_log +
##     NetSedFlux_cbrt, data = marshes_RM)
##
## Coefficients:
##      (Intercept)  BedloadFlux_cbrt   MedianFetch_log   NetSedFlux_cbrt
##         -2.12177          -0.01429           3.54861           3.54688
```

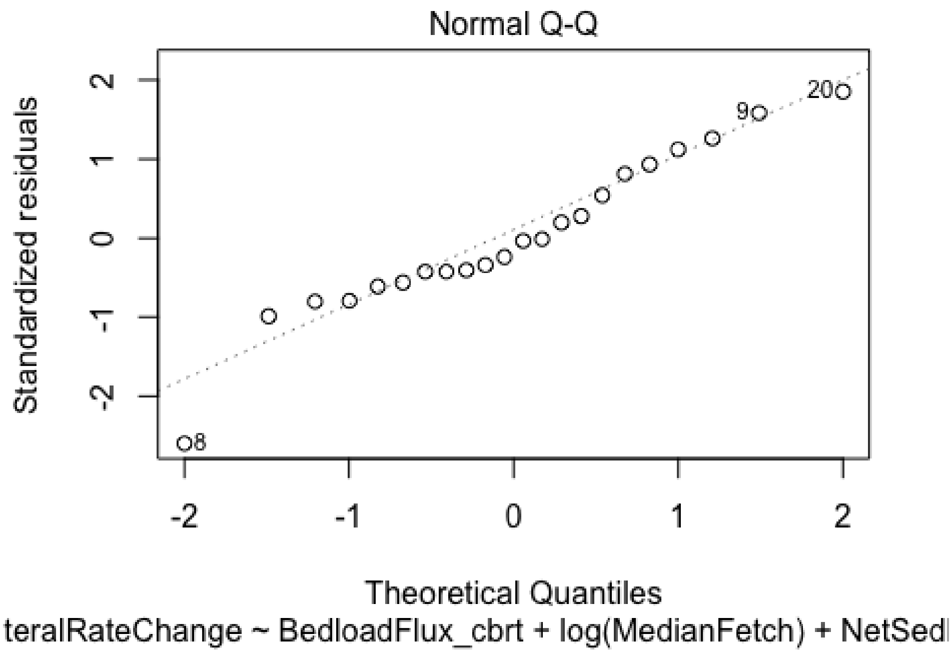
The predictor variables that should be retained in the minimal adequate model are *BedloadFlux_cbrt*, *MedianFetch_log*, and *NetSedFlux_cbrt*. We assign the predictor variables to a minimal adequate model:

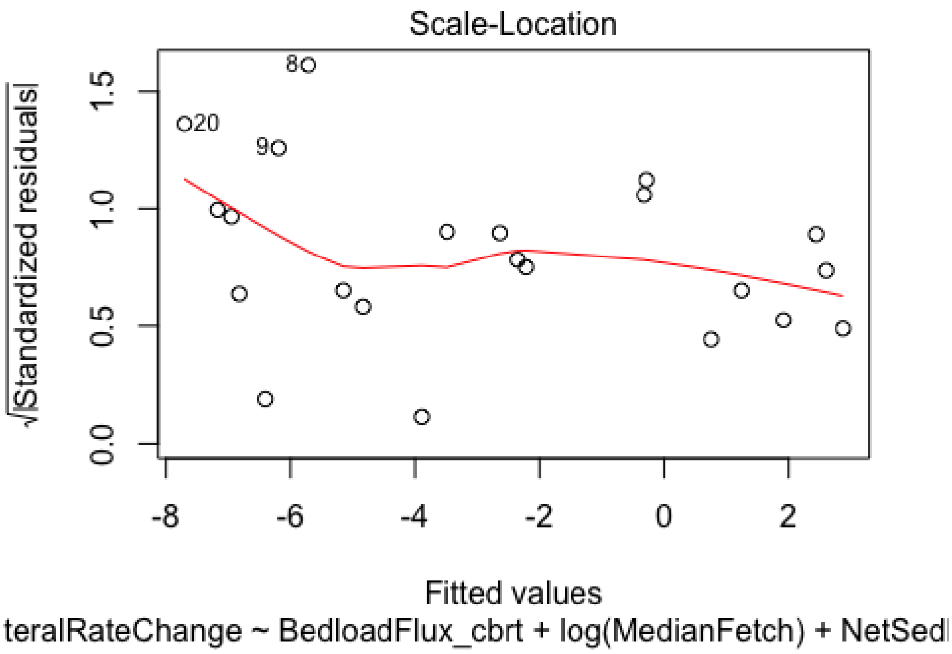
```
m4<-lm(LateralRateChange
       ~ BedloadFlux_cbrt+log(MedianFetch)+NetSedFlux_cbrt,
       data=marshes_RM)
```

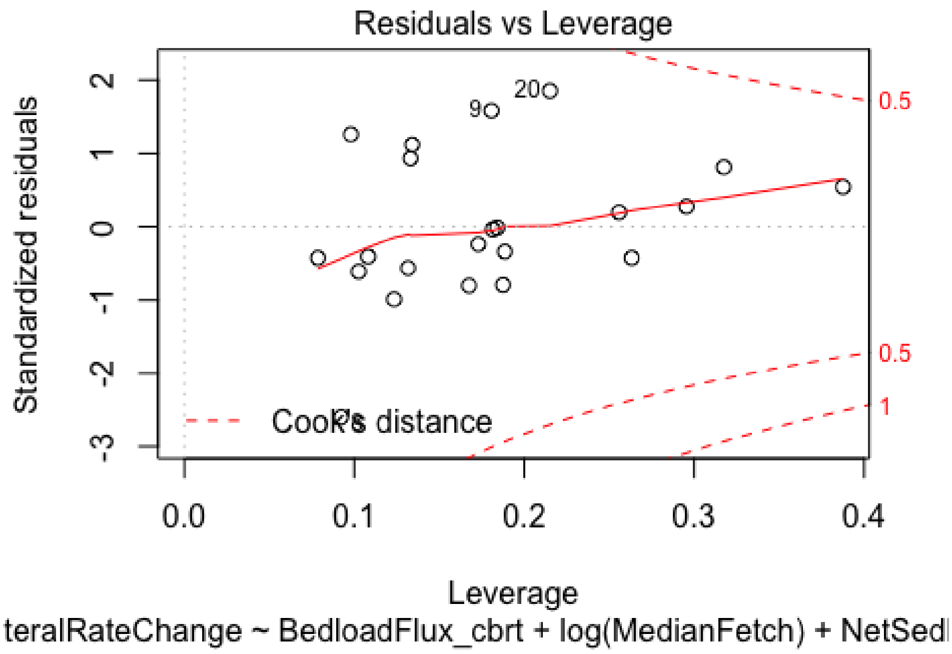
We check for heteroscedacity and bias in the model residuals, to check whether assumption of Heterogeneity of Variance have been violated:

```r
plot(m4)
```









There are no major issues with the model assumptions. We can now report the model results. We use anova tables to identify significant factors in our model using 'Type I' sums of squares, and consider which terms of the model are significant, the AIC score, and use the *relaimpo* package to calculate relative importance for the linear model using R^2 partitioned by averaging over orders:

```
library(relaimpo)
anova(m4)
```

```
## Analysis of Variance Table
##
## Response: LateralRateChange
##                   Df  Sum Sq Mean Sq F value    Pr(>F)
## BedloadFlux_cbrt  1  22.987  22.987  2.5044 0.1309427
## log(MedianFetch)  1 185.898 185.898 20.2533 0.0002768 ***
## NetSedFlux_cbrt   1  65.049  65.049  7.0870 0.0158758 *
## Residuals        18 165.216   9.179
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
```

```
summary(m4)
```

```
##
## Call:
## lm(formula = LateralRateChange ~ BedloadFlux_cbrt + log(MedianFetch) +
##     NetSedFlux_cbrt, data = marshes_RM)
##
## Residuals:
##     Min      1Q  Median      3Q     Max
## -7.5047 -1.5047 -0.3771  1.8486  4.9780
##
## Coefficients:
##                   Estimate Std. Error t value Pr(>|t|)
## (Intercept)      -2.121770   0.758995  -2.795   0.0120 *
## BedloadFlux_cbrt -0.014293   0.009561  -1.495   0.1523
## log(MedianFetch)  3.548614   1.323257   2.682   0.0152 *
## NetSedFlux_cbrt   3.546880   1.332338   2.662   0.0159 *
## ---
## Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1
##
## Residual standard error: 3.03 on 18 degrees of freedom
## Multiple R-squared:  0.6238, Adjusted R-squared:  0.5611
## F-statistic: 9.948 on 3 and 18 DF,  p-value: 0.0004326
```

```
AIC(m4)
```

```
## [1] 116.7899
```

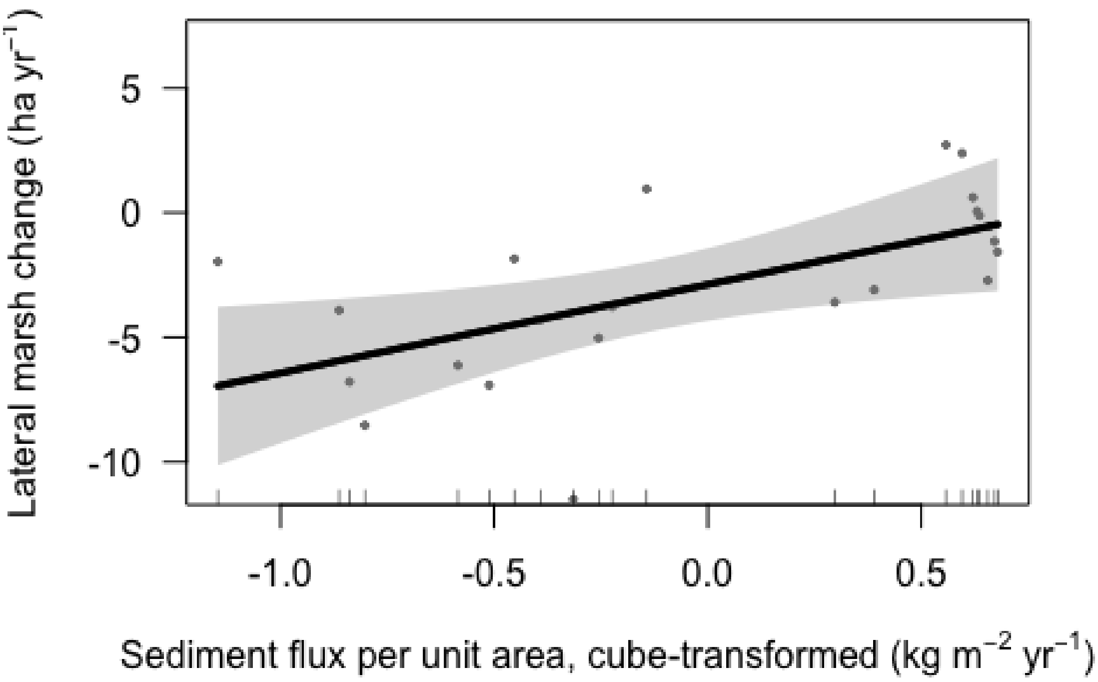
```
calc.relimp(m4,type=c("lmg"),rela=T)
```

```
## Response variable: LateralRateChange
## Total response variance: 20.91194
## Analysis based on 22 observations
##
## 3 Regressors:
## BedloadFlux_cbrt log(MedianFetch) NetSedFlux_cbrt
## Proportion of variance explained by model: 62.38%
## Metrics are normalized to sum to 100% (rela=TRUE).
##
## Relative importance metrics:
##
##                         lmg
## BedloadFlux_cbrt 0.06176616
## log(MedianFetch) 0.45986451
```

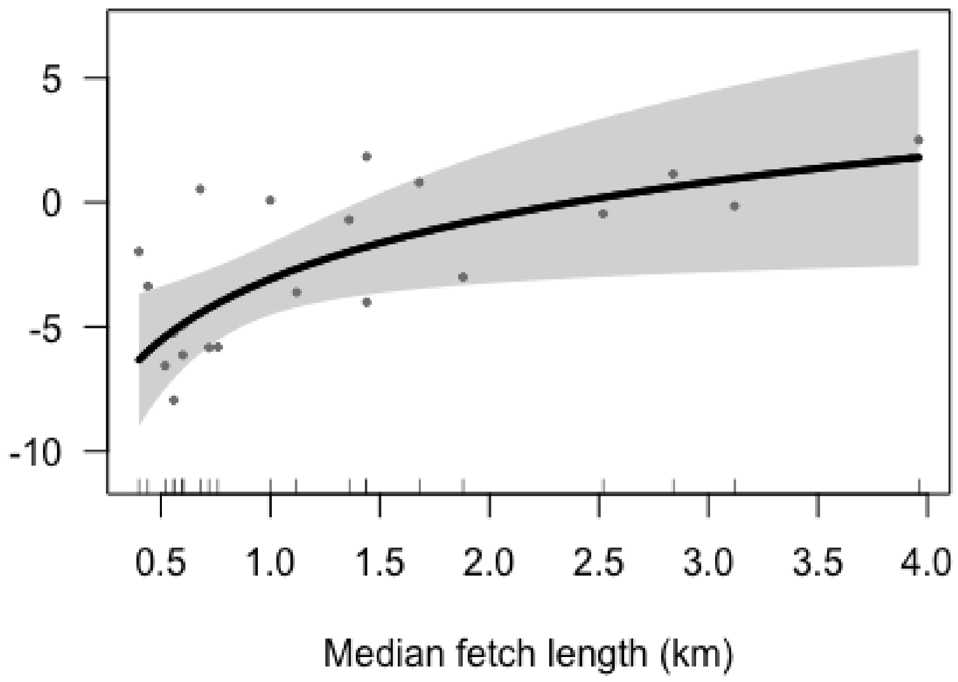
```
## NetSedFlux_cbrt  0.47836933
##
## Average coefficients for different model sizes:
##
##                           1X          2Xs         3Xs
## BedloadFlux_cbrt 0.01246703 -0.007295296 -0.01429271
## log(MedianFetch) 4.44960924  3.960577731  3.54861396
## NetSedFlux_cbrt  5.08045110  4.288503610  3.54687976
```

`MedianFetch_log` and `NetSedFlux_cbrt` are both significant terms, whereas `BedloadFlux_cbrt` is not. The AIC score of the minimal adequate model is 116.79. The proportion of variance explained by the model is 62.38%, and of that variance, MedianFetch_log and NetSedFlux_cbrt account for nearly half each (46% and 48% respectively). Finally, visualise the significant terms of the regression model using the `visreg` package. Marshes shift from a trend of expansion to erosion in response to increased fetch length and sediment supply:

```
library(visreg)

visreg(m4,"NetSedFlux_cbrt",
       ylim=c(-11,7),
       scale="response",
       partial=T,
       rug=T,
       line=list(col="black"),
       xlab=expression("Sediment flux per unit area, cube-transformed (kg
m"^-2*" yr"^-1*")"),
       ylab=expression("Lateral marsh change (ha yr"^-1*")"))
```




```
visreg(m4,"MedianFetch",
       ylim=c(-11,7),
       scale="response",
       partial=T,
       rug=T,
       line=list(col="black"),
       xlab="Median fetch length (km)",
       ylab="")
```



References

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