UNIVERSITY OF SOUTHAMPTON

FACULTY OF NATURAL AND ENVIRONMENTAL SCIENCES

Ocean and Earth Science

Volume 1 of 1

Morphological Evolution of Creek Networks in Restored Coastal Wetland Schemes

by

Clémentine Chirol

Thesis for the degree of Doctor of Philosophy

March 2018

UNIVERSITY OF SOUTHAMPTON

ABSTRACT

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MORPHOLOGICAL EVOLUTION OF CREEK NETWORKS IN RESTORED COASTAL WETLAND SCHEMES

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The restoration of coastal wetlands in the UK and worldwide often involves the excavation of artificial creek networks to encourage hydrological and ecological processes; however the evolution of these artificial creeks post-implementation is insufficiently monitored, and their design still lacks scientific guidance. This thesis analyses the morphological evolution of creeks in restored coastal wetlands in the UK to help inform future design strategies.

A new semi-automated creek parametrisation method was developed for lidar datasets, which is faster and less subjective than manual mapping. Morphological equilibrium was defined for natural saltmarshes; power-law relationships for creek dimensions and distribution at equilibrium were derived from the analysis of 13 mature natural saltmarshes to provide a range of potential end targets for MR creek design. A relationship was found between the overmarsh path length (mean distance to a creek anywhere in the marsh) and the mean site elevation within the tidal frame. This analysis also found that creek morphological equilibrium should be described as a range of potential states rather than as one quantifiable target.

The evolution of 10 MR creeks in their first 2-20 years post-breach was then studied. MR creeks evolved near-linearly towards a larger, more sinuous and better distributed system. These evolution rates towards the proposed equilibrium targets were then related to the initial conditions of the sites: low-lying sites with high accretion rates and large openings had faster-evolving creeks, while high, constrained sites displayed limited creek growth and required more extensive initial creek excavation. The 10 MR schemes and two accidentally realigned sites considered behave as though they should stabilise within 100 years into an alternative equilibrium state to that of natural systems, with a lower density distribution of creeks, mainly concentrated around the breach areas, while the further reaches of the site have fewer channels. Furthermore, the channels have a lower sinuosity due to inherited drainage ditches that remain visible even after 100 years, and a flatter substrate.

MR creek expansion may be hindered by the overcompacted soil and lack of small-scale topography inherited from the previous agricultural land use, which prevents creek incision and preferential deposition around topographic flow paths. Based on these findings, future studies should link the soil geotechnical properties, creek development and biodiversity of MR schemes to better understand creek-forming processes and improve the monitoring, management and design of MR sites.

Table of Contents

Table of Tables

Table of Figures

List of Accompanying Materials

Academic Thesis: Declaration Of Authorship

I, Clémentine Chirol, declare that this thesis “Morphological evolution of creek networks in managed realignment schemes” and the work presented in it are my own and has been generated by me as the result of my own original research.

I confirm that:

1. This work was done wholly or mainly while in candidature for a research degree at this University;
2. Where any part of this thesis has previously been submitted for a degree or any other qualification at this University or any other institution, this has been clearly stated;
3. Where I have consulted the published work of others, this is always clearly attributed;
4. Where I have quoted from the work of others, the source is always given. With the exception of such quotations, this thesis is entirely my own work;
5. I have acknowledged all main sources of help;
6. Where the thesis is based on work done by myself jointly with others, I have made clear exactly what was done by others and what I have contributed myself;
7. Parts of this work have been published as:

Chirol, C., Haigh, I.D., Pontee, N., Thompson, C.E.L. and Gallop, S.L. (2017) Parameterizing tidal creek morphology in mature saltmarshes using semi-automated extraction from lidar Remote Sensing of Environment (In Press).

Signed:

Date: 15/03/2018

Acknowledgements

I am particularly grateful to my supervisors, Dr Ivan Haigh, Dr Shari Gallop, Dr Nigel Pontee and Dr Charlie Thompson for their invaluable scientific guidance, inspiration and moral support throughout this journey. I am also indebted to the Environment Agency's Geomatics team, for the collection and dissemination of lidar data, and for providing me with metadata when possible.

I would like to thank my friends and colleagues Peter Lawrence and Hannah Mossman, for sharing some of their RTK-GPS datasets and their great expertise on saltmarsh ecology and microtopography, as well as for their enthusiasm and warmth during our public engagement projects together. Thanks to Anas Annuar and Matt Wadey for their assistance with the RTK GPS, to the Department of Geography for sharing their RTK GPS equipment and Hachem Kassem for his constant help in the lab, even though that data ultimately couldn't make it into the final thesis.

The field observations conducted during this PhD would not have been possible without the assistance of the Steart Marshes Wildfowl and Wetlands Trust staff and volunteers, and particularly site manager Alys Laver. My field helpers deserve a special mention for their resilience and unshakable good spirits, and for pulling me out of the mud multiple times: Matthew Agius, Aimin Amiruddin, John Andrews, Anas Annuar, Dick Best, David Billett, Nikki Brown, Matthew Cobain, John Davis, Richard Foyle, Shari Gallop, Ivan Haigh, Ali Hendry, Jonathan Hendry, Jamie Hizzett, Nathan Hubot, Renata Khouri, Heiko Loehr, Sian Martin, Kyle Mayers, Laura McMonagle, Ozgun Ozsoy, Alejandro Pinto, Sebastian Pitman, Hagen Radtke, Thomas Stamp, Tyra Standen, Charlie Thompson, Ian Townend, Matt Wadey, Amy Whitington and Jane Zhou.

Outside work, my thanks go to my family, and especially my parents and two siblings for unconditionally supporting my mud-treading career beyond the Channel, my grandmother Christiane for listening to my woes despite being long retired as a psychologist, and my 249 years old adopted brother, a historical Spanish church organ, for making my parents so very proud. I thank the friends back in France who have heroically stuck with me despite the distance and my patchy communication, Iulia, Cecile, Claire and Clelie, as well as my capoeira teacher Mestre Joãozinho and his wife Suzannah for their inspiration and kindness. Finally, my warmest thanks to my friends from the Light Opera Society, the Lopsoc members of 2013-2016, who have been my second family during my time in Southampton. I owe them most of my English speaking skills, pretty much all of my stage confidence (looking at you, Amy), and many precious memories of late night rehearsals, intense performances, creative costume parties, drama, adventures and weddings. Thank you for making the world magical.

Nomenclature

CSA	Cross-Sectional Area
D	Depth
DD	Drainage Density
DSM	Digital Surface Model
DTM	Digital Terrain Model
EA	Environment Agency
GDP	Gross Domestic Product
GPS	Global Positioning System
HAT	Highest Astronomical Tide
HOMW	Hesketh Out Marsh West
LAT	Lowest Astronomical Tide
MAPE	Mean Absolute Percentage Error
MCG	Main Channel Gradient
MCL	Main Channel Length
MHW	Mean High Water
MHWN	Mean High Water Neap
MHWS	Mean High Water Spring
MLWN	Mean Low Water Neap
MLWS	Mean Low Water Spring
MNTR	Mean Neap Tidal Range
MR	Managed Realignment
MSTR	Mean Spring Tidal Range
MWS	Mean Water Spring
NB	Number of creeks
OD	Ordnance Datum
OPL	Overmarsh Path Length
PA	Planform Area
PC	Principal Component
PCA	Principal Component Analysis
QEGD	Quasi-Euclidean Geodesic Distance
RS	Reverse Strahler
RSPB	Royal Society for the Protection of Birds
RTE	Regulated Tidal Exchange
SR	Sinuosity Ratio
STD	Standard Deviation
TCL	Total Channel Length
TP	Tidal Prism
UI	User Input
W/D	Width/Depth

Chapter 1: Introduction

1.1 Background and Justification

Low-lying coastal areas (less than 10 m above present day sea level) are home to 600 million people globally (McGranahan et al. 2007) and constitute strategic economic centres. Coastal areas are particularly vulnerable to climate change-related risks and other pressures such as over-population and pollution. Their management is a multidisciplinary problem, requiring understanding of hydrodynamics, climate science, geomorphological and sedimentological processes, biogeochemistry and ecology, as well as geoengineering, socioeconomics, policy and public perception. While the most immediate and dramatic effect of relative sea-level rise and rapid urban development in coastal areas is the inundation of low-lying coastal areas and islands, another key coastal management challenge is the protection and management of coastal wetlands (FitzGerald et al. 2008). Indeed, coastal populations are growing at three times the global mean rate (Neumann et al. 2015). Losing coastal wetlands has dire consequences as these habitats provide a wide range of benefits (Luisetti et al. 2014), such as reducing coastal flooding and erosion (Kirwan et al. 2016), improving water quality (Nelson et al. 2012), enhancing plant diversity, and trapping carbon at ~55 times the rate of rainforests (Macreadie et al. 2013).

Coastal wetlands are being lost worldwide at an alarming rate, though causes of decline are changing. About 25–50% of the world’s coastal wetlands have been lost to reclamation over the last 150–300 years (Lotze et al. 2006). In Europe, even though the practice of land reclamation has become less and less profitable and has virtually stopped since the 1980s–1990s (Doody 2004; Goeldner-Gianella 2007), the trend of decreasing wetland areas continues. In many countries, including the UK, exact long-term evolution trends are difficult to assess due to the unreliability of historical maps in coastal zones (Baily et al. 2013). Estimates of saltmarsh losses and gains in England and Wales vary from 1 ha gain to 83 ha loss per year between the 1980s and 2009 (Phelan et al. 2011). Current coastal wetland losses are mainly linked to urban and industrial development in the coastal zone, and to coastal erosion and drowning due to relative sea-level rise (Brady et al. 2017). Since coastal wetland decline has dramatic consequences for biodiversity (Atkinson et al. 2004; Doody 2004; Tempest et al. 2014), the European Habitats Directive enforces a “no net loss policy” since 1992 to protect existing habitats and compensate for lost ones (European Commission 1992).

A related coastal management challenge is the long-term resilience of the coast against erosion and flooding, which is threatened by sea-level rise (Temmerman et al. 2013). Rising sea levels increase the coasts' vulnerability to the impact of storm surges and waves (Temmerman et al. 2013; Hinkel et al. 2014). Without a mitigation strategy, coastal flood damage will have severe social and economic consequences, and may affect 0.2–4.6% of the global population by 2100, and cost 0.3–9.3% of the Global Gross Domestic Product (GDP) in 2100 (236–7,309 billion US$ per year according to the global GDP 2014) (Hinkel et al. 2014).

Conventional flood mitigation methods are referred to as "hard defences" and include building seawalls, dykes and embankments (Temmerman et al. 2013). While in some cases hard defences may be appropriate, their use on large portions of urbanised coastlines can increase the risk of catastrophic flooding in case of defence failure (Hinkel et al. 2014). Upgrading hard defences may also be unsustainable, as sea-level rise is expected to continue or accelerate over the coming decades (Haigh et al. 2014); the annual cost of hard flood defence maintenance and upgrades is expected to increase by 150% to 400% depending on future scenarios (Townend et al. 2004). Finally, dikes can hinder the sediment transport processes through which shorelines normally keep up with relative sea levels (Temmerman et al. 2013). Heavily built coastlines also contribute to the loss of coastal wetlands via coastal squeeze in the context of sea-level rise, as these habitats, which normally act as a buffer against tidal and wave energy, are unable to retreat landwards due to the presence of infrastructure. This leads both to a loss of valuable habitat and to the loss of a natural line of defence, making the hard defences costlier to maintain.

There is a strong case for investing in a new coastal defence strategy referred to as "soft defence" (Pethick 2002). This method makes use of the expected resilience of natural coastal habitats against rising sea levels, and aims to restore coastal ecosystems as part of a long-term, cost-effective coastal management strategy that "works with nature" (Temmerman et al. 2013; Esteves et al. 2015). As part of this strategy, and to compensate for historic and more recent losses, coastal wetlands are being artificially recreated in many countries. This is done by reintroducing tidal inundation to previously embanked land, either through Regulated Tidal Exchange (RTE), where tidal levels are controlled through culverts and sluices (Scott et al. 2011), or through the removal or breaching of a flood defence. A new line of defence is generally added landwards to maintain flood protection standards, making these schemes a middle ground between a hard and soft defence strategy (Figure 1.1). The breaching/removal of defences accompanied by the landward realignment of defences is referred to as "managed retreat" in north-western Europe and Australia, or "depolderisation" in France (Goeldner-Gianella 2007), while in the US, those projects are dominantly referred to as "tidal wetland restoration or construction". The current accepted term in the UK is "Managed Realignment" (MR). For improved coherence and since this thesis focuses on UK sites, all coastal wetland recreation projects will be referred to as "MR", even when the term isn't used in the literature referenced. Other, less common definitions of MR, such as removing existing infrastructure to allow for a cliff face to move naturally, are not taken into consideration. Finally, because this thesis focuses on the restoration of self-sustaining tidal flats and saltmarshes, RTE is also not considered.

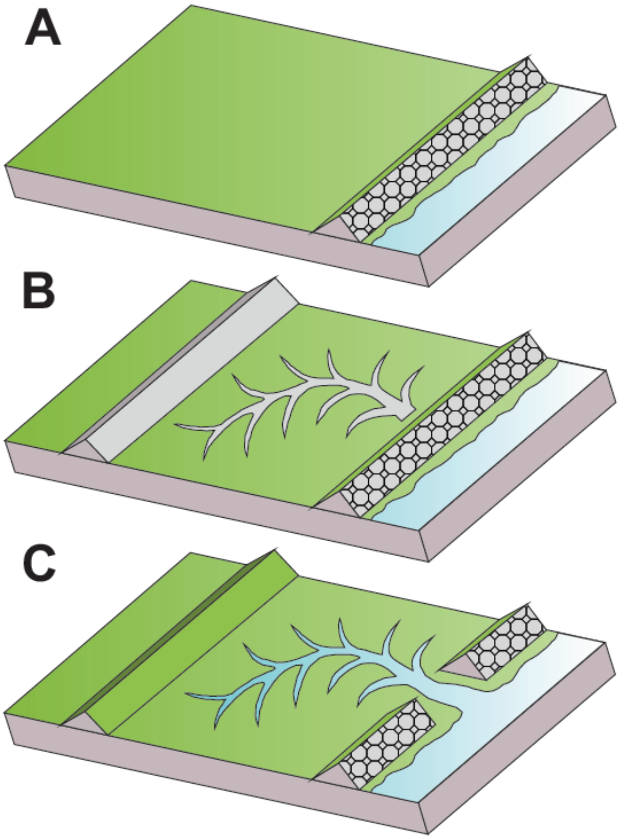


Figure 1.1: Conceptual diagram of a managed realignment site implementation A: Initial site protected by an embankment; B: Line of defence moved further inland and site preparation (elevation distribution, excavation of an artificial creek network); C: breaching of the old defence.

In north-western Europe, over 100 of these so called MR schemes, have been undertaken between 1981 and 2015 (Esteves et al. 2017), including 49 recorded schemes in the UK (ABPmer 2014). The implementation of these schemes is mainly driven by the Habitats Directive adopted in 1992 and transposed into national UK legislation in 1994, which stipulates that national conservation sites, called Natura 2000 sites, be designated by each government and protected following a no net loss policy (Esteves 2014). The "no net loss" policy implies that projects and plans that may damage these conversation sites should be avoided when possible, and that any damage done to these sites should be compensated by the re-creation of similar habitats elsewhere (Rupp 2010). Most tidal flats and saltmarshes in Europe have been designated as Natura 2000 sites (Esteves 2014).

The implementation rate in the UK has increased from 80 ha to 179 ha per year between 2010 and 2016 (Committee on Climate Change 2017). Given the growing rate of MR implementation, good design guidelines are crucial to ensure the quality of the recreated habitats. Indeed, recent studies have expressed concern regarding the efficiency of current schemes, as they often fail to provide the same ecosystem services as their natural counterparts (Esteves 2013; Brady et al. 2017), and do not replicate the plant diversity of natural wetlands (Mossman et al. 2012). If not addressed properly, this will lower their value as wetland loss mitigation schemes. This thesis thus focuses on the design of MR schemes in the UK in particular.

The current consensus is that MR schemes should be designed to encourage the natural physical processes that evolve coastal wetlands towards a mature, stable state (Palmer 2009; Pontee 2015a). This approach is in alignment with previously established concepts for freshwater wetland mitigation: 1) Understand wetland function; 2) Give the system time; and 3) Allow for the self-designing capacity of nature (Mitsch et al. 1996). A key factor is the implementation of an appropriate pre-breach baseline morphology to kick-start the natural evolution of the site (Cooper et al. 2004; Leggett et al. 2004). The number and dimensions of breaches, site elevation and initial creek networks all impact the subsequent evolution of the scheme (Figure 1.1). While some aspects of MR design have been well-researched, like the breach area (Townend 2008), there is a lack of scientific guidance for other morphological characteristics such as creek networks.

Creek networks are increasingly recognised as being crucial to the ecological stability of the wetland. They distribute water, sediment and nutrients during the flood tide, drain the site during the ebb tide, control the distribution of vegetation (Sanderson et al. 1998; Morzaria-Luna et al. 2004; Zheng et al. 2016), influence accretion rates (Reed et al. 1999), and serve as pathways for juvenile fish (Miller et al. 1997). It is thought that excavating initial channels may speed up the development of the creek network, thus helping to support biodiversity and meet target conditions for MR schemes (Crooks et al. 2002; Wallace et al. 2005).

Consequently, recent MR projects often include artificial channel networks, with the plan to accelerate wetland evolution towards a mature state (Figure 1.1B). A large variety of initial designs have been used in the UK over the past 20 years. These design choices range from simple linear distribution channels to more complex configurations mimicking mature network morphologies. The influence of these various design choices on MR schemes' eco-geomorphological evolution, long-term sustainability and delivery of ecosystem services remains a key knowledge gap and should be investigated to provide more systematic design guidelines (Oosterlee et al. 2017). Therefore, it is the focus of this thesis.

The provision of more systematic guidelines requires a thorough investigation of current MR schemes and the evolution of their creeks to learn from practical experience. To encourage natural physical processes, creek design guidelines should also build on previous knowledge of the expected evolution of natural coastal wetlands. Indeed, the easiest and most widely used way to assess whether an artificial habitat is ecologically successful is to compare it with a mature, stable natural habitat of similar conditions (Weinstein et al. 1997; Mossman et al. 2012). This approach implies the existence of an “equilibrium” state of natural saltmarshes, towards which MR schemes are expected to evolve following an asymptotic curve (Weinstein et al. 1997; French 2006). Most design and monitoring of MR schemes worldwide is done following this assumption (Williams et al. 2002; Marani et al. 2003; Zhao et al. 2016), but MR design is not done with a quantifiable morphological target in mind.

Morphological equilibrium implies a degree of stability in a landform under stable input conditions, that is to say an invariable sea-level rise, no change in the sediment budget and no catastrophic “resetting” brought by storms (Steel et al. 1997; Van Goor 2003; Friedrichs et al. 2001; French 2006). From this the notion of dynamic morphological equilibrium can be derived: a system at dynamic equilibrium is one that can keep up with incremental changes in input conditions without significant changes to its morphological characteristics. In saltmarshes, the key morphological characteristics are the surface elevation within the tidal frame and the dimensions and planform of the creek network (French et al. 1992; Allen 2000).

Previous studies on creek networks in stable natural saltmarshes have resulted in the definition of morphological equilibrium relationships linking creek morphological characteristics to the dimensions of the site (Steel 1996; Williams et al. 2002; Marani et al. 2003). By further studying the equilibrium state and the time necessary to reach it, there is potential to use such relationships to establish quantitative or semi-quantitative “target morphologies” for MR schemes, and explore whether the sites are evolving towards this target. Target morphologies could also be used to extrapolate the number of years needed for the site to reach the stable state, as was done in saltmarsh restoration studies in San Francisco Bay (Williams et al. 2002).

The question of the predictive value of those equilibrium relationships remains open, for natural coastal wetlands, but also and especially for MR schemes (Palmer 2009). Indeed, the specific characteristics of MR schemes may lead them to a completely different equilibrium state than that of natural systems: do morphological equilibrium relationships established for natural creek systems constitute valid quantifiable targets for MR schemes? To answer this, further investigation is needed on how initial design and external conditions (tidal forcing, sediment availability, etc.) affect MR creek evolution and to what morphology it is likely to stabilise. Results from such analysis may inform future practice to encourage the evolution of MR schemes towards a stable state that is more likely to support a healthy ecosystem. Deriving evolution rates for MR creeks is also important to predict the time necessary to reach this stable state. This thesis examines these questions and knowledge gaps by developing creek-feature mapping methods, and by applying these to remote sensing data, with a focus on the morphological characteristics and evolution trends of creek networks in MR schemes.

1.2 Aims and objectives

The overall aim of this thesis is to improve understanding of the morphological evolution and expansion of natural and MR creek networks in order to derive implications for future MR practice. This aim will be addressed through the following four Thesis Objectives:

Thesis Objective 1: Provide a comprehensive morphometric analysis of creek systems in natural coastal wetlands in the UK, using remote sensing data and novel parametrisation methods, to define equilibrium characteristics and validate them as potential end targets for MR creek design.

Thesis Objective 2: Undertake a morphometric analysis of creek systems in MR schemes in the UK and of their evolution over the years, in order to estimate creek evolution rates.

Thesis Objective 3: Compare the morphological characteristics of natural and MR creeks to determine whether MR creeks evolve towards more natural systems over time, and whether their evolution rates can be related to initial conditions and design choices.

Thesis Objective 4: Establish a conceptual creek evolution model for MR schemes that highlights the main divergences from natural creek evolution, and derive implications for future practice in order to reduce these divergences in future schemes.

1.3 Thesis structure

This thesis is articulated into 8 chapters. Chapter 2 reviews the state of knowledge in coastal wetland evolution, MR scheme history and design expertise, and highlights the remaining gaps in knowledge. As this thesis focuses on creek networks, the state of knowledge in creek network development, design strategies and monitoring methods are also detailed. Chapter 3 presents the data sources and study sites considered during this thesis. Chapter 4 details the development of a new semi-automated creek parametrisation algorithm, which facilitates the systematic monitoring of creek networks over time. Chapter 5 provides a critical analysis of the morphological equilibrium relationships available in the literature, tested on 13 field-validated mature UK saltmarshes (Thesis Objective 1). Chapter 6 presents the results of a multi-parameter systematic monitoring of 10 MR schemes in the UK, with a focus on the morphometric evolution of creek networks over 3 to 20 years since site opening (Thesis Objective 2). Chapter 7 statistically compares the creek morphology of natural and artificial saltmarshes to determine whether MR creek networks evolve towards morphological equilibrium as defined for mature natural sites (Thesis Objective 3). Chapter 8 draws together the conclusions of Chapters 5, 6 and 7 to propose a synthetic conceptual model of creek network evolution in MR schemes, and how current design strategies may impact this evolution (Thesis Objective 4). The chapter then relates those findings to parallel studies on the ecological evolution of MR schemes to discuss the implications of the observed morphological evolution for the success of MR schemes as healthy ecosystems. The chapter also provides updated implications for future design. Finally, conclusions and recommendations for further research are given in Chapter 9.

Chapter 2: Background of study

2.1 Introduction

The design of MR schemes is a young and rapidly growing field, and should build on knowledge gathered from natural saltmarshes, if they are to reproduce their ecological functions and benefits (Oosterlee et al. 2017). This chapter provides background information on coastal wetlands (mudflats and saltmarshes) and on MR schemes, with a focus on the initiation, evolution and design of creek networks. Section 2.2 defines coastal wetlands and the processes driving their evolution toward equilibrium. Their main ecosystem services, and their vulnerability and potential resilience to rising sea levels, are also explored. Section 2.3 provides further information on the definition and origin of MR, the history of its practice worldwide and in the UK, as well as the current design strategies. The section also critically discusses the notion of success in MR schemes, and highlights the remaining knowledge gaps that may hinder the provision of ecologically successful schemes. Section 2.4 provides detailed information on the relevance of creek networks to coastal wetland function, and reviews the state of knowledge on their initiation, development and monitoring. Remaining knowledge gaps will inform the need for a new, semi-automated monitoring tool for MR creek networks using remote sensing elevation data, the development of which will be detailed in Chapter 4.

2.2 Importance of coastal wetlands

2.2.1 Definitions and notion of equilibrium

Coastal wetlands are low-energy, tidally dominated transitional habitats found between marine and terrestrial systems (Allen et al. 1992). Though coastal wetlands also include seagrass beds and mangroves, this study focuses on tidal flats and saltmarshes because of their prominence in the UK. Tidal flats are characterised by an accumulation of fine sediment (Foster et al. 2013), either sand (sandflat) or, more commonly, mud (mudflat). The upper levels of the tidal flat typically supports salt-tolerant vegetation and form the saltmarsh. The saltmarsh is dissected by a network of tidal channels that diminish landward and form the main interface between sea and land (Allen 2000).

Despite the variety of coastal wetland locations within an estuary (Allen 2000), the dominant processes that drive their vertical evolution are the same: the cross-shore profile of mature coastal wetlands is linked to the astronomical tidal levels. Tidal levels control the amount of water and salinity that the wetland receives each year, and hence the dominant plant species and the resulting saltmarsh zonation (Figure 2.1). Tidal levels are a reflection of the tides periodicity driven by astronomical processes. Semi-diurnal low and high tides are due to the rotation of the Earth on its axis. Spring and Neap tides are due to the alignment of the Earth, Sun and Moon (Full and New moon) roughly every 14 days. The elliptic rotation of the Moon around the Earth creates slightly larger spring tides when the Moon is closer to the Earth roughly every 28 days. The elliptic rotation of the Earth around the Sun also leads to higher tidal levels at the perihelion, when the Earth is closest to the Sun around January every year (Masselink et al. 2014).

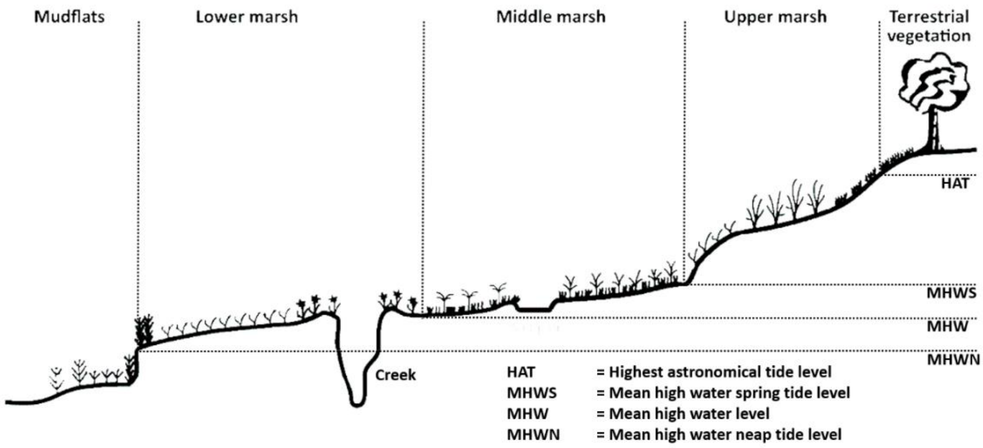


Figure 2.1: Mature tidal flat and saltmarsh zonation within the tidal frame (modified from Foster et al. 2013)

Due to the Coriolis effect, tides rotate around amphidromic points of null tidal amplitude: the tidal range then increases away from these points as it gets influenced by atmospheric pressure fluctuations, wind forces and geomorphology (e.g., water depth, size/shape of the ocean basin). Due to the complex topography of coastal waters, tidal predictions require an empirical approach based on harmonic analysis of local tidal records (Masselink et al. 2014). Tidal levels can then be defined for a specific location based on the mean Spring and Neap tidal events: Mean Low Water Neap (MLWN); Mean High Water Neap (MHWN); Mean Low Water Spring (MLWS) and Mean High Water Spring (MHWS), or based on the maximal and minimal tidal events that can be expected under normal meteorological conditions: Lowest and Highest Astronomical Tide (LAT and HAT).

Coastal wetlands are situated in the coastal zone between HAT and MLW level (Figure 2.1). The transition between saltmarsh and mudflat generally occurs at MHWN (Nottage et al. 2005; Foster et al. 2013), though most of the marsh vegetation is found between mean-Neap tide level and MHWS (Allen et al. 1992). Marsh building from tidal flat to saltmarsh is driven by sediment accretion. The sediment is either brought into the system as suspended particles in the adjacent estuary or connecting river ("allochtonous sedimentation", Figure 2.2), or generated on site by the decomposition of plant matter ("autochtonous sedimentation", Figure 2.2) (French 2006). The availability of sediment in the water column and the productivity of the vegetation on site thus both control the growth rate of the coastal wetland.

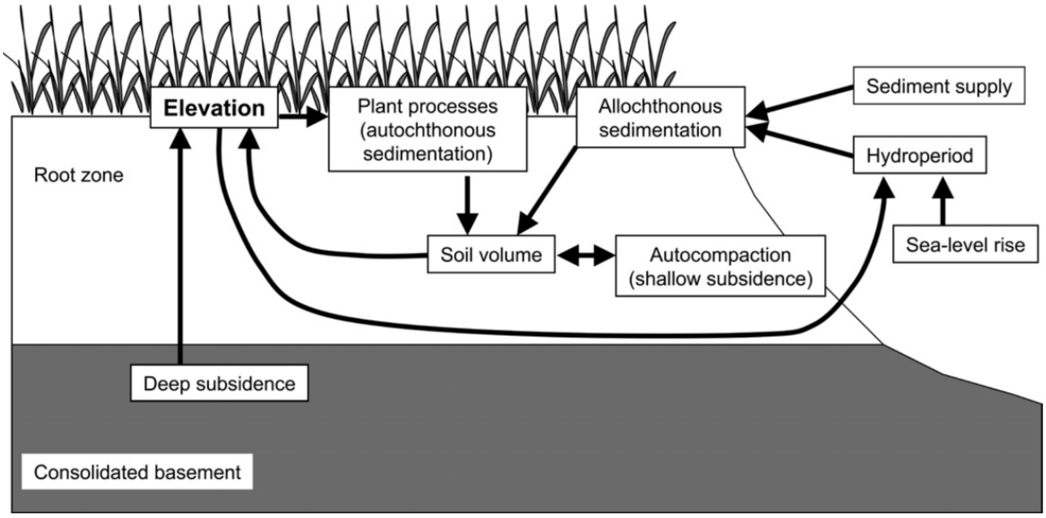


Figure 2.2: Processes of marsh vertical evolution (French 2006)

The hydroperiod (frequency and length of tidal inundation) has a major influence on the sedimentation rate when sediment supply is not a limiting factor (Allen 2000). The hydroperiod is linked to the tidal range, wind-wave climate, relative sea level and storminess. Due to the number of controlling factors, the range of inundation events per year is very broad, spanning from 70 to 415 events based on a study of 13 UK natural mature saltmarshes, using 19 years of tidal level records (Steel 1996). Elevation within the tidal range provides a steadier and more convenient definition of coastal wetlands (Figure 2.1). One last factor of wetland vertical evolution is autocompaction, where a growing sequence of sediment undergoes dewatering and collapses under its own weight (Allen 1999) (Figure 2.2). Autocompaction rates depend on the geotechnical properties of the marsh substrate, which is linked to the geological and anthropological history of the site (Allen 2000).

Saltmarsh elevation evolves following an asymptotic curve until it reaches a stable elevation within the tidal range, typically close to MHWS (Figure 2.3), which corresponds to a state of maturity or morphological equilibrium (Allen 2000). Morphological equilibrium implies a degree of stability in a landform under stable input conditions, that is to say an invariable sea-level rise, no change in the sediment budget and no catastrophic "resetting" brought by storms (Steel et al. 1997; Friedrichs et al. 2001; Van Goor et al. 2003; J. French 2006). A system at dynamic equilibrium may thus be defined as one that can keep up with incremental changes in input conditions without significant changes to its morphological characteristics. In saltmarshes, the key morphological characteristics are the surface elevation within the tidal frame and the dimensions and planform of the creek network (French et al. 1992; Allen 2000). The equilibrium morphology of channels in particular has been described as the “physical expression of the critical erosion threshold”, that is to say an equilibrium state between erosional and depositional processes (Allen et al. 1992). Steel’s model assumes that, as the marsh accretes to keep up with rising sea levels, the planform and volumetric characteristics of the creeks will be maintained.

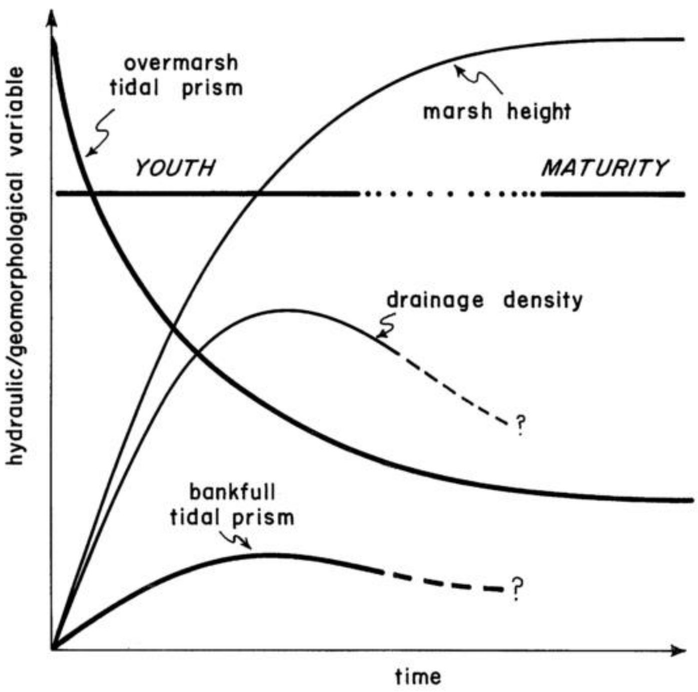


Figure 2.3: A model for the possible hydraulic and geomorphological evolution of salt marshes towards equilibrium in Northwest Europe (Source: Allen 2000). Note the uncertainty associated with the tidal channels drainage density trends once equilibrium is reached.

In addition to hydroperiod and sediment availability, the sediment type has a significant influence on marsh development. More cohesive sediments have a greater resistance to erosion: this depends on grain size, with sandy marshes eroding more easily, but also on the nature of the clay minerals present, which will deflocculate at different flow velocities depending on the strength of their inter-particle bonds (Crooks et al. 2000). The time necessary to reach the mature state varies between wetlands, due to the various controlling factors discussed above. It has been estimated as being between 100 and 300 years for saltmarshes in north Norfolk, Norfolk and Lincolnshire (French et al., 1990 *in* Steel 1996). The equilibrium state is considered to be the same for all saltmarshes, at least on a national scale (Steel et al. 1997; Williams et al. 2002), even though the equilibrium creek extent may vary due to differences in sediment types (Steel, 1996). The equilibrium state is thus an important point of reference to monitor the evolution of MR schemes. However, differences in starting conditions between natural coastal wetlands and MR schemes may lead to different equilibrium states. It should also be kept in mind that different features of coastal wetlands (e.g. marsh elevation and tidal channels) do not evolve at the same rate towards equilibrium (Figure 2.3).

2.2.2 Related ecosystem services

Saltmarshes and tidal flats provide a wide range of benefits, otherwise called ecosystem services (Barbier et al. 2011). They play an important role as biodiversity hotspots (Zedler et al. 2005; Mossman et al. 2012), nurseries for juvenile fish (Luisetti et al. 2014), pollutant filters (Nelson et al. 2012), carbon sinks (Erwin 2009; Ahn et al. 2013; Tempest et al. 2014), natural defences against coastal erosion and flooding (Möller et al. 2002; Wal et al. 2004; Gedan et al. 2011), and support agricultural (King et al. 1995) and recreational activities (Luisetti et al. 2014). These various ecosystem services are detailed below:

2.2.2.1 Biodiversity and fish use

Owing to their nature as transitional zones between sea and land, coastal wetlands host a number of specialist species due to their specific ecological conditions (Nottage et al. 2005; Lefeuvre et al. 2013), including various protected species of plants and birds. Therefore, most tidal flats and saltmarshes in Europe have been designated as national conservation sites (Esteves 2014). The preservation of essential habitats for protected species has driven most US (Williams et al. 2001) and European (Morris 2012) coastal wetland restoration projects. In addition, saltmarshes are used by a number of commercially important fish species as refuge, feeding and nursery areas (Colclough et al. 2010). The monetary value of saltmarshes in terms of food supply is expected to be significant, though difficult to quantify due to insufficient sampling (Luisetti et al. 2014).

2.2.2.2 Pollutant and nutrient filtration

Coastal wetlands, due to their position as buffers between sea and land, play an important role as pollutant filters (Allen 2000; Mudd 2011). Saltmarshes are effective traps for heavy metals due to their high sedimentation rates and specific soil chemistry associated with brackish environments (Chmura et al. 2012). The downside is that the pollutants may be released into the water column if erosion occurs (Allen 2000). Coastal wetlands also act as nutrient filters and can thus protect estuaries from eutrophication: water passing through a saltmarsh is slowed down by the friction effect of the vegetation, which facilitates nutrient uptake by the plants and reduces the nutrient fluxes that reach the wider coastal environment (Barbier et al. 2011; Nelson et al. 2012). The monetary value of this water treatment function is estimated at about 113–583 £/ha/year in terms of nitrate removal service (Shepherd et al. 2007).

2.2.2.3 Protection against coastal erosion and flooding

Wave activity is the main cause of coastal erosion and storm-induced damage in coastal communities (Gedan et al. 2011). Intertidal habitats offset those damages by reducing wave height and energy due to the frictional effects of plants (Cooper 2005; Möller et al. 2014). Wave dissipation is less efficient under high energy conditions due to plant flattening and breakage but remains significant (Möller et al. 2014; Spencer et al. 2016). Marsh vegetation also reduces surface erosion by increasing the shear strength of wetland soils due to the presence of roots and the binding action of organic matter on soil structure (Gedan et al. 2011).

The implementation of wetland habitats as part of a flood defence system could significantly reduce the costs of traditional flood defence implementation and maintenance (Foster et al. 2013). As an example, the value of thin strips of saltmarsh fronting seawalls has been estimated at £400–£600 m^{-1} due to their role in preventing flooding and coastal erosion (King et al. 1995). The efficiency of coastal wetlands at preventing coastal erosion is especially relevant in the context of sea-level rise. Coastal wetlands are important sediment sinks (Morris 2012), which generally allows them to accrete vertically and keep up with rising sea levels (Kirwan et al. 2016). The conditions for this accretion to occur are sufficient sediment supplies and sufficient vegetation production to maintain allochtonous and autochtonous sedimentation (Figure 2.2), as well as limited wave-induced marsh erosion (Fagherazzi et al. 2013). The process of wetlands vertical accretion is crucial in densely populated estuaries, where flood defences and infrastructure often prevents saltmarshes from retreating landward, a phenomenon referred to as coastal squeeze (Pontee 2013; Torio et al. 2013).

2.2.2.4 Carbon sequestration

Saltmarshes are one of the most efficient carbon sinks on the planet, burying carbon at ~55 times the rate of rainforests (Macreadie et al. 2013). This is due on one hand to their high primary productivity, and on the other hand to their role as efficient sediment sinks. The carbon trapped within saltmarshes is brought as dissolved and particulate organic matter by rivers, estuaries and the wider ocean (Moran et al. 2013). Carbon sequestration in saltmarshes also has a greater longevity (millennia) than in rainforests (decades), as the carbon is stored in the sediment rather than in the biomass (Macreadie et al. 2013). Finally, while terrestrial soils eventually become saturated in carbon, coastal wetlands have a capacity for long-term sediment accumulation by accreting vertically as sea levels rise (McLeod et al. 2011).

2.2.2.5 Agricultural and recreational activities

While the value of saltmarshes as agricultural and pastoral lands is estimated to be inferior to its potential value as a flooding and coastal erosion mitigation measure, it still represents a potential alternative use for these habitats, especially since saltmarsh grazing is thought to improve the health of cattle by providing clean, parasite-free feeding ground (King et al. 1995). In addition, coastal wetlands provide benefits to society in the form of recreational activities such as walking and bird-watching. Artificially recreated wetlands also have a unique pedagogic value, distinct from that of natural sites (Chmura et al. 2012). Indeed, due to their history of degradation and restoration, they are an illustration for visitors of the impacts of climate change on the coast, and of resulting mitigation strategies including soft engineering solutions. In that regard, MR could contribute to the economy of rural areas by promoting the growth of a range of new activities including tourism (Chmura et al. 2012; Esteves 2014).

In summary, due to the high environmental, economic, and ecological value of coastal wetlands, their loss will have significant consequences in the UK and worldwide. However most MR schemes tend to focus on a limited numbers of these benefits, mainly biodiversity conservation efforts (Morris 2012). A narrow view of the significance of these schemes can lead to negative press for MR projects. For example, at the recent Steart coastal management project, the member of Parliament for Bridgwater and West Somerset spoke virulently against the project and described it as an “extravagant, ridiculous scheme” and a “bird sanctuary” (Guardian 2014). The article made no mention of the other potential benefits of the schemes such as water treatment, grazing by premium saltmarsh lamb and beef, shelter for fish fry and seabass, carbon storage, biodiversity, amenity value (WWT 2014). Even more critically, the ecosystem services considered will impact how schemes are designed. Lack of focus on fish behaviour within MR schemes has led to inadequate design for juvenile fish, which normally feed in saltmarsh habitats (Colclough et al. 2010). In general, coastal wetlands receive little media attention compared with more charismatic habitats like coral reefs (Duarte et al. 2008). The lack of coverage can be detrimental to public perception of MR schemes and to the acceptance of new schemes. In order to improve public acceptance as well as future design, those different ecosystem services and related morphological requirements should be publicised more widely.

2.2.3 **Vulnerability and long-term resilience to a changing climate**

Saltmarshes and tidal flats are found on all continents, except Antarctica, at mid to high latitudes (Johnson et al. 2016) (Figure 2.4). The current extent and rate of loss for tidal flats and saltmarshes in the UK and worldwide is difficult to quantify due to lack of monitoring (Allen 2000; Foster et al. 2013), especially in Africa, the Neotropics and Oceania (Davidson 2014). Indeed, saltmarshes are mainly documented in temperate regions, and recent maps probably underestimate the extent of saltmarshes in the tropics (Friess et al. 2012).

Figure 2.4: Qualitative measure of saltmarsh abundance by marine ecoregion, based on point estimates of saltmarsh locations scaled by the coastline length (Hansen et al. 2015)

A widespread but poorly justified estimation states that the world has lost 50% of its wetlands since historical times. The claim originates from a report on the loss of marshes and swamps in seven US states between 1850 and 1953 (Davidson 2014), and has been widely extrapolated in subsequent publications (e.g. Winkler et al. 1985; Maltby et al. 2011). Hence, careful attention must be given to the origin of data when reviewing trends of coastal wetland extent over time.

Current estimations, made from species richness and primary productivity records from Europe, Australia and North America, point to a global pattern of coastal wetland decline of 25–50% over the last 150–300 years (Lotze et al. 2006). This fits with more recent findings of a 46–50% global loss of coastal wetlands since before 1900 AD, based on the average results of 18 reports (Davidson 2014). Despite increased awareness of their importance, this rate of loss has accelerated 4.2 times during the 20th and early 21st centuries (Davidson 2014). It is now estimated that 1% of the 1990 global coastal wetland extent is being lost each year (Duarte et al. 2008).

Coastal wetlands are found all around the UK (Figure 2.5). The best estimate of current tidal flat and saltmarsh extent in the UK is 278,831 ha and 46,850-49,193 ha respectively, based on OS maps (Foster et al. 2013). Current evolution trends are uncertain due to the various mapping methodologies used and the lack of comprehensive national-scale surveys (Phelan et al. 2011). Regional-scale studies indicate stark differences in saltmarsh extent trends across the UK: as an example, consistent loss of saltmarshes has been recorded in the south and south-east of England, while saltmarsh areas have increased in north-west England (Foster et al. 2013). A commonly used estimation of saltmarsh loss rate for England is 100 ha per year (Pye et al., 1993 *in* Phelan et al. 2011), but this may be an overestimation. Indeed, a more recent estimation conducted by the Environment Agency found a median loss rate of 40–50 ha per year, with national estimates ranging as far as 83 ha loss per year to 1 ha gain per year (Phelan et al. 2011). Establishing a more accurate rate of saltmarsh change for the UK would require further nationwide surveys. Similarly, losses of mudflats due to sea-level rise have been estimated at about 230 ha per year in England and Wales (Lee 2001; Foster et al. 2013), but the calculation is broad and based on many assumptions. The same study estimated mean yearly national saltmarsh losses of 140 ha: assuming similar detection errors for mudflats and saltmarshes, and using the -1 to 83 ha range discussed above, mudflat losses in England and Wales can be tentatively estimated at 89 to 173 ha per year.

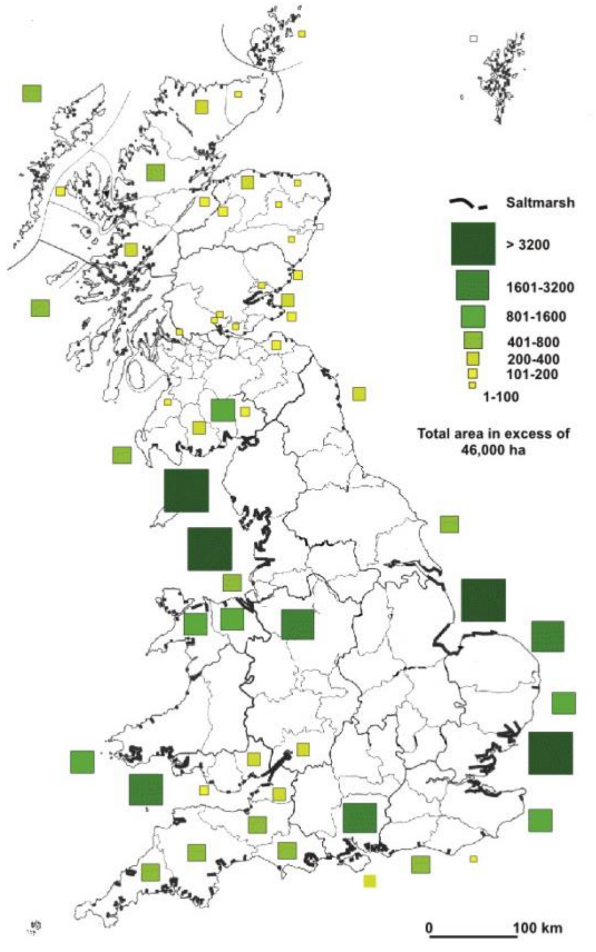


Figure 2.5: Estimated distribution of saltmarshes in the UK (area measurements based on 1:50000 OS maps for each county administrative unit in ha) (modified from Doody 2013)

Natural erosion of coastal wetlands occurs when the sediment supplies cannot keep up with the relative sea-level rise (Ganju et al. 2013), and can also be triggered by an excessive exposure to wave action (Fagherazzi et al. 2013). Coastal wetlands usually respond to this erosion by migrating landward (Boorman 1999); however in areas constrained by embankments or infrastructure, this migration is blocked, leading to contraction of saltmarsh surfaces (Pontee 2013; Torio et al. 2013). This effect, called coastal squeeze, is most pronounced in densely populated countries like the Netherlands and the UK, while the US and Canada have more areas where the saltmarshes are free to migrate inland. A more prominent factor of coastal wetland loss is direct human impact, where habitats are destroyed for agricultural land reclamation, waste disposal or industrial development (Day et al. 2008; Foster et al. 2013).

There is a concern that current rates of coastal wetland loss in the UK and worldwide are bound to accelerate in the context of climate change. A number of climate change-related forcings can impact the functioning of coastal wetlands, including: sea level rise; changes in storm frequency and intensity; changes in freshwater, sediment, and nutrient inputs (Day et al. 2008). These factors have complex interactions, and their respective roles on the projected future of coastal wetlands in the UK and worldwide remains an open question (Day et al. 2008). Current observed rates of sea-level rise over the last few decades range between 2.8 and 3.6 mm/year, and are expected to accelerate (Church et al. 2013; Haigh et al. 2014). Rising sea levels will increase the frequency of extreme sea level events (Nicholls et al. 2008). A 1 m rise in relative sea level could threaten up to 70% of the world's coastal wetlands (Nicholls 2004), though it should be noted that this figure is based on the assumption of continued, unmitigated anthropogenic pressures. In the coming decades, human attitude to coastal management is expected to have a much greater impact on coastal wetland sustainability than sea-level rise. Indeed, given the right conditions, coastal wetlands display a strong resilience to changing sea levels.

Various ecogeomorphic interactions allow marshes to adapt to rising sea levels, for instance by increasing their plant growth rate (Kirwan et al. 2013, 2016). Studies show that saltmarshes are able to keep up with predicted rates of sea-level rise, if sediment inputs and organic soil formation are sufficient (Day et al. 1999) (Figure 2.6). Similarly, both anecdotal evidence and true-to-scale storm surge laboratory experiments suggest that saltmarshes are very resilient to storm conditions (Spencer et al. 2016). Overall, recent studies suggest that the vulnerability of saltmarshes to sea-level rise is overestimated (Kirwan et al. 2016): they are resilient provided that human influence does not limit sediment input or inland retreat.

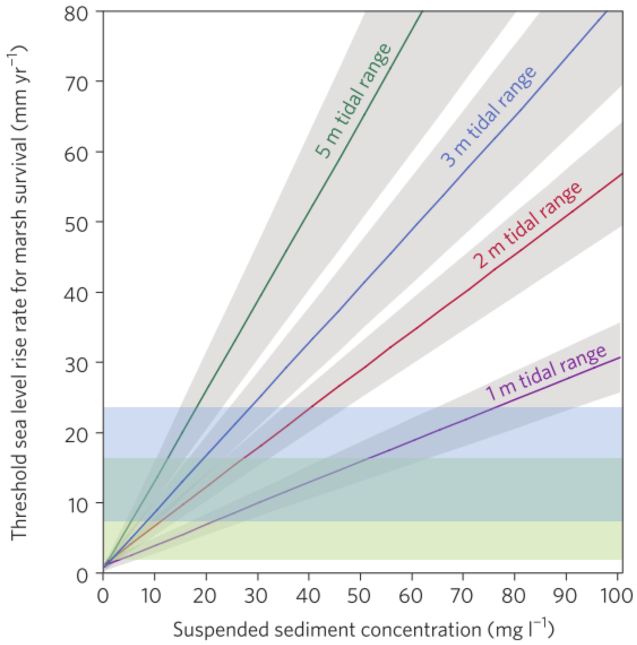


Figure 2.6: Threshold survival curves of saltmarshes depending on predicted rates of sea-level rise (pale green for process-based model projections by IPCC AR5, pale blue for semi-empirical projections and dark green for overlap between the two), tidal range and suspended sediment concentration (Source: Kirwan et al. 2016)

Anthropogenic pressures are most likely to impact the sustainability of coastal wetlands. Human activity worldwide tends to simultaneously increase sediment transport in rivers via enhanced soil erosion, and decrease sediment load into the ocean due to damming (Syvitski et al. 2005). Depleting sediment supply is expected to be the most serious threat to long-term coastal wetland survival by hindering marsh vertical accretion (Morris 2013; Ratliff et al. 2015). The problem is particularly relevant for mineralogenic saltmarshes like those of Northwestern Europe, where the marshes' potential to accrete vertically and keep up with rising sea levels is highly dependent on sediment supplies from outside the system (Allen 2000; French 2006). A related limiting factor for coastal wetland growth is the availability of seeds, which allows tidal flats to be colonised by vegetation and transition into saltmarshes, and which depends on the proximity of neighbouring marshes (Friess et al. 2012). The destruction of coastal wetlands can lead to increased patchiness, greater distance between habitats, and poorer water-borne seed transfer from marsh to marsh (Wolters et al. 2005).

Changes in the chemistry of coastal wetlands, linked to climate change and human influences such as agriculture, can impact primary productivity and marsh survival. Indeed, organic matter in saltmarshes contributes both to marsh building and to better soil stability. Rising levels of atmospheric CO_2 could significantly increase the resilience of saltmarshes to sea-level rise by enhancing the productivity of saltmarsh plants and the production of organic soil (Ratliff et al.

2015). Conversely, eutrophication caused by run-off of fertilisers from agricultural lands into water bodies could cause saltmarsh loss (Canfield et al. 2010; Deegan et al. 2012). Indeed, overly nutrient-enriched saltmarsh plants tend to have lower root to shoot ratios as there is less need to gather nutrients from the soil: this decreases the stabilising action of roots along the creek edges, and leads to bank collapse, creek widening and the conversion of part of the saltmarsh into tidal flat (Deegan et al. 2012). More efficient organic matter decomposition also leads to finer-grained, less porous sediment, encouraging water retention and decreasing creek bank stability. The different ways in which human influence can directly or indirectly alter coastal wetlands are summarised in Lee et al. (2006).

This section demonstrated that current shrinking of coastal wetlands in the UK and worldwide is due mostly to anthropologic pressures such as habitat destruction and coastal squeeze, as unconstrained coastal wetlands display a natural resilience to changing sea levels. This highlights the need for coastal wetland restoration schemes, but it also shows that MR is a promising strategy if properly designed. Indeed, provided that they are given favourable growth conditions (sediment availability in particular), coastal wetlands are capable of keeping up with predicted rises in sea level, and of providing long-term coastal protection.

2.3 Managed realignment schemes

2.3.1 History and definitions

In an attempt to mitigate for coastal wetland losses and preserve the ecosystem services described in Section 2.2.2, intertidal habitat recreation through the landward realignment of flood defences has been attempted in many countries, such as the US (Williams et al. 2001), Australia (Rogers et al. 2014), Germany (Rupp-Armstrong et al. 2007), Belgium and the Netherlands (Van den Bergh et al. 2005), and the UK (Esteves 2014). MR consists of removing or deliberately breaching flood defences to allow seawater to inundate previously reclaimed land (generally agricultural) (Figure 1.1), and restore natural processes of inundation, erosion and accretion (Crooks et al. 2002; Esteves et al. 2014). New, more easily maintained defences are frequently built on the landward side of the scheme to maintain flood protection standards (Esteves 2014). The implementation rate of MR schemes is on the rise worldwide (Oosterlee et al. 2017). In order to highlight the recent advances and main gaps in knowledge in MR design, this section provides a brief history of coastal wetland recreation projects in the UK and abroad since the 1970s.

Legislated saltmarsh restoration and reconstruction started in the US in 1972 to reverse environmental damage to the coastal wetlands fringing San Francisco Bay (Williams et al. 2001), 90% of which had been destroyed since the 1850s for agricultural and commercial development (Williams et al. 2004). The history of San Francisco Bay MR projects is instructive due to the various implementation methodologies tested, depending on individual site objectives and financial resources (Williams et al. 2001). In the early 1970s, sites were prepared by planting saltmarsh vegetation before breaching. Physical criteria like marsh elevation and tidal circulation were considered unimportant and were poorly designed for. As a result, some sites were too high or their breach area too small to receive sufficient tidal flooding, and over 90% of wetland vegetation plantings died (Williams et al. 2001).

Further design strategies were tested in the 1980s with more focus on geomorphology. Attempts were made at reproducing the elevation, shape and tidal channels of a mature saltmarsh (Williams et al. 2001). Other projects ignored the natural morphology of saltmarshes altogether and artificially created tidal conditions that would favour specific protected species, generally through control gates and weirs (Williams et al. 2001). However, both of these methods are costly and fail to encourage natural physical processes. As a result, only 50% of those early projects were deemed successful based on vegetation growth monitoring (Mitsch et al. 1996).

In northwest Europe, over 100 schemes have been undertaken or are being planned, corresponding to 9,350 ha of recreated coastal wetland (Oosterlee et al. 2017). 49 MR schemes have been implemented in the UK as of 2014 according to the ABPmer Online Marine Registry (ABPmer 2014, Figure 2.7).This represents a stark reversal of the previous reclamation policy, which led to 1.5 Mha of coastal land being reclaimed in northwest Europe since the 11th century, primarily for agricultural purposes and to protect coastal infrastructure by multiplying the lines of defence (Goeldner-Gianella 2007). Land reclamation for agricultural purposes and flood protection is estimated to have stopped in Europe around the 1970s-2000s. This reversal was facilitated by agricultural overproduction, reducing the economic interest of reclaiming new lands (Goeldner-Gianella 2007). In addition, growing ecological concern over the loss of important coastal ecosystems led to environmental regulations being passed by the European Union: in particular, the Habitats Directive adopted in 1992 and transposed into national UK legislation in 1994 stipulates that national “Natura 2000” conservation sites should be designated by each government and protected following a no net loss policy (Esteves 2014). The law prevents projects or plans that are expected to have an adverse effect on Natura 2000 sites to go ahead, except in cases of “imperative reasons of overriding public interest, including those of a social or economic nature”, and where no alternative, less damaging solutions have been found (European

Commission 1992; Defra 2012). In those cases compensatory measures must be taken to ensure the continuity of "no net loss", often in the form of MR implementation elsewhere (Crooks et al. 2001; Defra 2012).

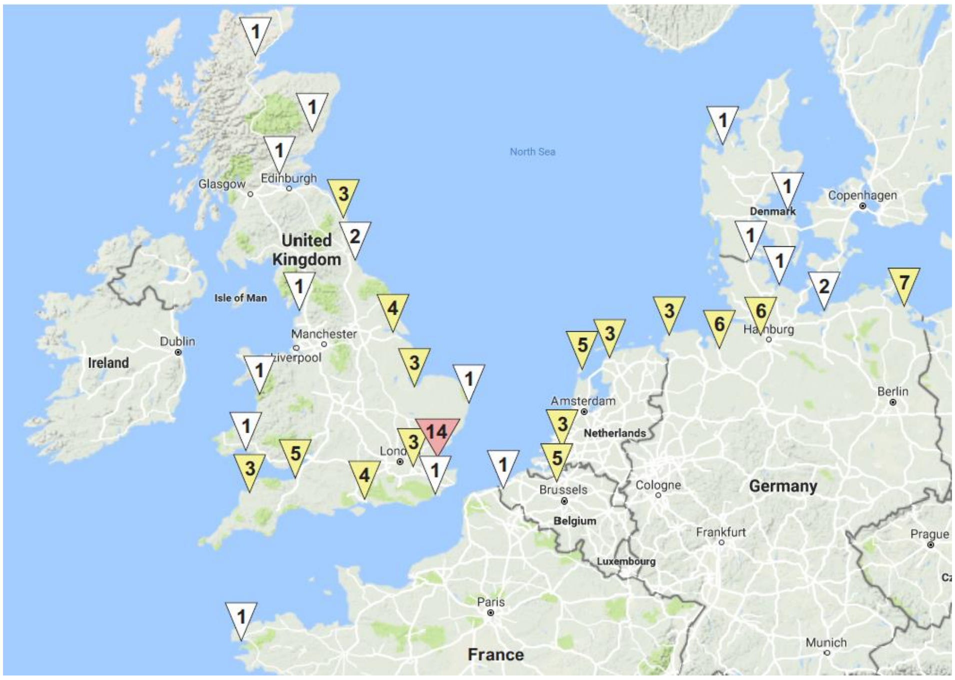


Figure 2.7: Distribution of MR schemes implemented as of 2014 in Northwest Europe. White: <3 MR schemes; yellow: <10; red: >10 (based on ABPmer 2014).

Most tidal flats and saltmarshes in Europe have been designated as Natura 2000 sites (Esteves 2014). However, as highlighted in Section 2.2.3, the exact yearly rates of coastal wetland losses are difficult to quantify: in this thesis they are tentatively estimated at 88–256 ha/year (for tidal flats and saltmarshes combined). The implementation of MR schemes in the UK has led to the creation of about 400 ha of intertidal habitats between 2010 and 2015 (80 ha per year) (Committee on Climate Change 2017). The trend is towards an accelerated implementation, with 179 ha created between 2015 and 2016 (Committee on Climate Change 2017). On paper, this seems to fulfill the no net loss policy. However concerns remain that coastal wetlands are still being lost faster in the UK than they are being replaced by MR, at least on a regional scale (Foster et al. 2013). There is also rising concern that the MR compensatory habitats will not be of equivalent quality to the lost Natura 2000 sites (Mossman et al. 2012; Brady et al. 2017).

MR strategy in the UK is driven by two main considerations: the first factor is the European Habitats Directive which prescribes a "no net loss" policy: habitats lost through coastal squeeze or industrial development should be compensated by equivalent habitats (Esteves 2014); the second factor is the rising cost of flood defence maintenance in the face of rising sea levels, making the "hold the line" coastal management approach less and less sustainable (Lee 2001; Ledoux et al.

2005). The UK is the first country in Europe to integrate MR within a broader long-term coastal protection scheme (Pethick 2002; Goeldner-Gianella 2007).

2.3.2 **Current design strategies**

There are 49 schemes recorded in the UK as of 2014 (ABPmer 2014), the oldest of which being the <1 ha Northey Island pilot project breached in 1991 (Pethick 2002). Implementation strategies in the UK then evolved from experimental schemes with a breach and a very simplified creek network excavated from a drainage ditch, as in Tollesbury in 1995 (Garbutt et al. 2006), to larger and highly engineered projects such as Steart (Burgess et al. 2013). Most of the current MR know-how stems from restoration projects in the US (ABP Southampton 1998; Williams et al. 2004), observations of sites of accidental breaching in Essex and Suffolk in the UK (Burd 1995), or from recent MR practice in the UK (ABP Southampton 1998; Leggett et al. 2004; Nottage et al. 2005). In the rest of Europe, MR is not as explicitly embedded into coastal management policy as it is in the UK (de la Vega-Leinert et al. 2012): as a result most of the available literature focuses on the overall feasibility of MR (Goeldner-Gianella et al. 2013; Meur-Férec et al. 2013; de la Vega-Leinert et al. 2017), or investigates isolated case studies (Almeida et al. 2014; Smolders et al. 2015; Chang et al. 2016; De Martis et al. 2016), rather than attempting to establish a general scheme design. Thus, the provision of guidelines to systematise the design and implementation process appears to come mainly from the US and UK.

2.3.2.1 MR rationale

MR schemes should be cost-effective and sustainable both technically and environmentally (Ledoux et al. 2005). The current rationale behind MR is to provide a baseline morphology that encourages natural physical processes so the site may evolve towards a state of ecological functioning equal to that of natural marshes (Oosterlee et al. 2017), rather than recreate “instant” mature wetland topography (Williams et al. 2004). The logic is a middle ground between unmanaged retreat, where the breach is left to occur naturally and where no preliminary profiling occurs (Pethick 2002), and more interventionist approaches such as regulated tidal exchange (RTE), where tidal levels are controlled through culverts and sluices (Scott et al. 2011). RTE schemes are better sheltered from wave impact and encourage sediment deposition (Esteves 2014); however they are more expensive than MR, and their long-term ecological value has been called into question in comparison with that of restored natural systems (Williams et al. 2004).

While most MR schemes aim for the restoration of saltmarsh habitats, some are compensatory schemes for tidal flats: design advice will differ depending on the desired habitats. In general, it is considered that creating tidal flat habitats requires less active management than saltmarsh habitats and that having a low starting elevation is sufficient (Burd 1995). This view has been challenged by examples of recent MR schemes in the UK that silted up rapidly into saltmarshes due to high water turbidity (Mazik et al. 2010), thereby missing the project's objectives in terms of habitat compensation. Nevertheless, most design guidelines found in the literature focus on saltmarsh restoration, as it is considered to be the mature state into which a tidal flat evolves.

2.3.2.2 MR planning

In the UK, complex legislative considerations lead to the decision to implement MR at a particular location, in agreement with Natural England and various other stakeholders (Pontee, 2017, pers. com.); these considerations are beyond the scope of this thesis. However, some circumstances of MR planning can have a significant impact on site design. MR planning can be separated into three categories: strategic, opportunistic and legislative (Leggett et al. 2004):

- *Strategic MR* identifies potential sites to meet objectives set by non-statutory coastal management plans such as Shoreline Management Plans (SMPs) and Coastal Habitat Management Plans (CHaMPs) (Leggett et al. 2004). Strategic MR is an ideal scenario from an ecomorphological standpoint because it gives the opportunity to select a site with the best available characteristics to support a saltmarsh.
- *Opportunistic MR* occurs when a particular site is made available by a landowner, or when realigning is expected to reduce flood defence maintenance costs at a specific location (Leggett et al. 2004).
- *Legislative MR* occurs when mitigation or compensation is needed to comply with statutory Habitat Directives. This scenario situates itself between strategic and opportunistic MR in terms of flexibility of site selection. Indeed, while the MR site is not preselected, it is considered preferable to provide compensation space close to the degraded habitat.

The ecomorphological considerations that will determine the approval of a site for MR implementation, and the main steps that can be undertaken to improve the site suitability to saltmarsh development, are summarised in Figure 2.8. The flowchart applies to either strategic, opportunistic or legislative MR. Opportunistic MR sites may have to undergo more site preparation than strategic ones, as they were not necessarily chosen for their initial similarity to saltmarsh habitats.

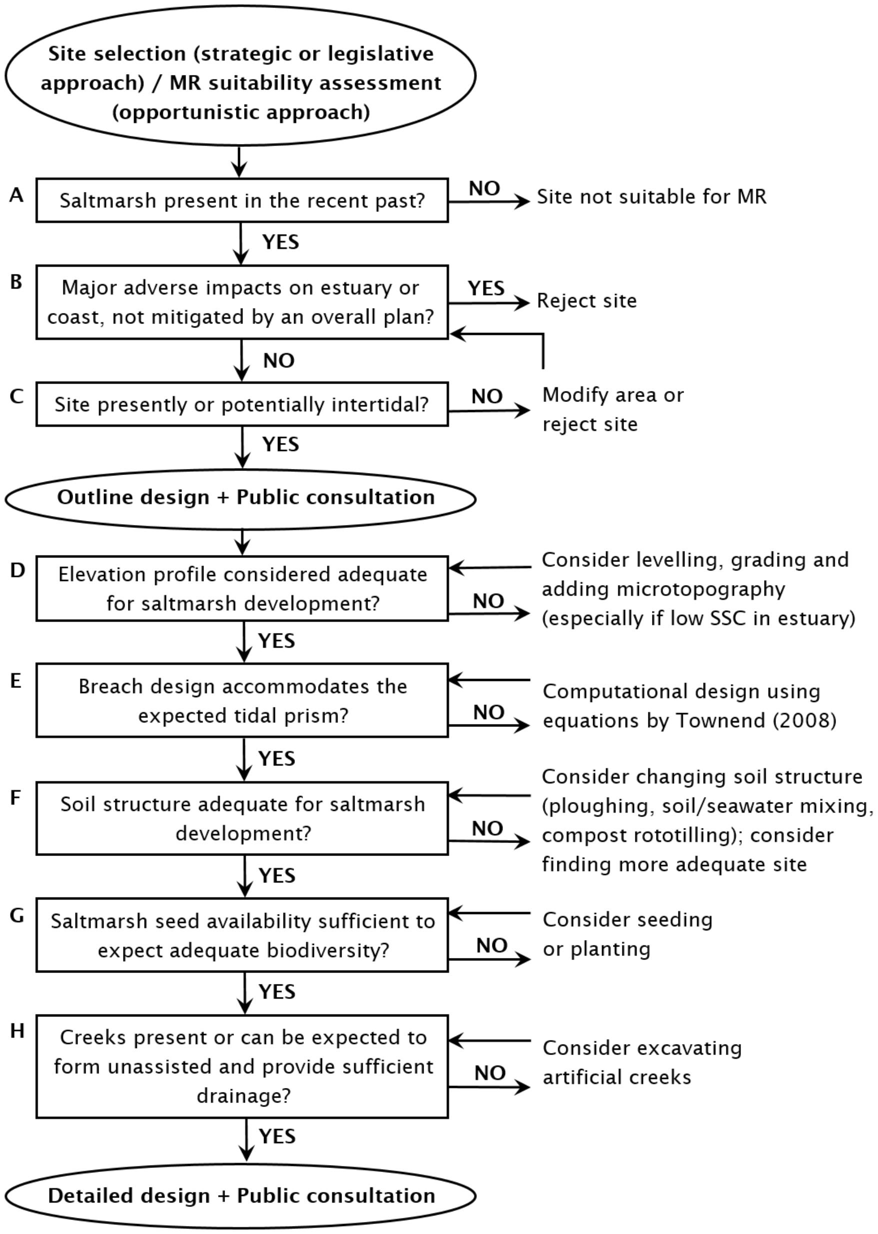


Figure 2.8: Ecomorphological planning and design of MR, not taking into account socioeconomic considerations. A-C: considerations for MR site selection or suitability assessment; D-H: considerations for MR design. SSC= Suspended Sediment Concentration. Compiled based on information from Burd (1995); Leggett et al. (2004); Nottage et al. (2005); O'Brien et al. (2006); Mossman et al. (2012); Brooks et al. (2015)

The previous history of the site and of the wider estuary is expected to play a crucial role in assessing MR suitability (Figure 2.8A-C). Indeed, previously embanked saltmarshes are more likely to have the soil structure and chemistry suitable for the re-establishment of a saltmarsh (Burd 1995). Conversely, an estuary that doesn't naturally support saltmarshes is unlikely to be suitable for MR and is one of the first factors that can lead to the rejection of a potential MR scheme (Figure 2.8A). The far-field effects of MR implementation on the wider estuary should also be considered (Figure 2.8), especially in the case of large schemes which can significantly increase the tidal prism of the estuary and impact flow velocities and sediment transport (Pontee 2015a).

Guidelines for the design phase, excluding creek guidelines which will be detailed in their own section, can be broadly separated into "Elevation profile", "Breach design" and "Other factors":

2.3.2.3 Elevation profile design

Despite recent interest in the role of vegetation for marsh building and soil stabilisation (Kirwan et al. 2013), the provision of a good physical template is still seen as the major deciding factor of MR scheme evolution (Blott et al. 2004; Williams et al. 2004; Nottage et al. 2005) (Figure 2.8D). Indeed, the distribution of plant species in natural coastal wetlands depends on the elevation and on the proximity to tidal creeks (Zedler et al. 1999). In most schemes, the initial site elevation is likely to be lower in a MR scheme than in a natural mature saltmarsh as the site has been cut off from the sediment sources that usually drive its vertical accretion (Burd 1995), and may require engineering work before breaching. The implementation of a gentle slope in the marsh elevation profile is also advised to promote a greater diversity of habitats (Burd 1995). The slope should be angled seaward and towards the nearest tidal channels to optimise drainage (Williams et al. 2004). More recent studies stress the need to add microtopography in order to provide more niche habitats for plants (Brooks et al. 2015).

2.3.2.4 Breach design

A lot of research has gone into designing the breach profile, as it constitutes the main point of exchange between the MR scheme and the wider estuary (Burd 1995; Townend 2008; Hood 2015) and should be able to accommodate the expected tidal prism (Figure 2.8E). Some MR schemes in the UK have opted for complete removal of the seawall to leave the site open to tidal and wave energy, increase the connectivity to the fronting tidal flats, and thus increase the likelihood of forming tidal flat habitats (Mazik et al. 2010). However breaching is generally the preferred option as bank removal is expensive (Pontee et al. 2006; Mazik et al. 2010).

The breach or breaches should be connected to existing channels on the fronting marsh or tidal flat: a side-effect is that a portion of the fronting habitat will be lost as the channel widens post-breach (Williams et al. 2004). Empirical logarithmic relationships have been found for historically storm-breached sites between breach width and tidal prism, the volume of water exchanged with a site in a single tidal event (Burd 1995). This implies that the width of the breach is very sensitive to the potential volume of water entering and leaving the site: careful consideration should be taken of the site volume to avoid excessive erosion in the breach area.

However, this breach design method was later found to have limited practical use due to its potential for error and the inability to account for site-specific characteristics; consequently an alternative method was developed by Townend (2008). In this method, the breach profile is found through an iterative process performed at regular time-step intervals of mean Spring tidal height. The definition of the equilibrium cross-sectional breach dimensions at each tidal stage depends on the basic flow conditions, on the limiting erosion threshold and on the site hypsometry. The lower limit of all cross-sections provides an output cross-section that can accommodate the whole tidal prism, and thus gives the expected breach dimensions at equilibrium. This method is being increasingly used in recent MR projects (Wright et al. 2011; Pontee 2015a).

2.3.2.5 Soil structure, sediment and seed availability

The first two subsections ensure that the morphological template provided will provide the tidal levels and inundation times necessary to the development of saltmarsh ecosystems. Whether or not a diverse and stable saltmarsh will actually occupy the site will depend on a number of additional factors, including soil structure, sediment fluxes and seed availability.

Poor soil structure is increasingly considered to hinder the successful development of MR schemes (Figure 2.8F). Even in MR sites with a history of tidal inundation, recent studies suggest that decades of embankment and agricultural use negatively impact the soil structure, leading to over-consolidated sediment through dewatering, mineralisation of organic carbon, shrinkage of clays and reduction of microporosity (Brooks et al. 2015; Spencer et al. 2017). This is likely to lead to poor drainage and poor circulation of nutrients and contaminants, with negative consequences for the marsh biodiversity (Spencer et al. 2017). This problem is expected to be less prominent in rapidly accreting sites, making sediment availability a major factor of MR success (Burd 1995). Indeed no evidence of soil structure disturbance was found in the first 30 cm of century-old accidentally breached sites soils, probably due to the deposition of new sediment not affected by dewatering (Spencer et al. 2017). Other parameters such as nutrient supply, water quality and early plant community structure can also impact MR evolution (Burd 1995; Pontee 2003).

Regarding plant colonisation, natural establishment is the preferred approach due to the high cost of seeding and planting (Nottage et al. 2005), and due to the high mortality rates of plantings in trial sites in San Francisco Bay (Williams et al. 2001). Plant colonisation is expected to occur naturally if seed sources are sufficient (Williams et al. 2004). However, as mentioned in Section 2.2.3, limited seed availability can be expected in some UK MR schemes, due to a lack of neighbouring marshes and to the constrained nature of most schemes. Therefore, some guidelines advise planting to accelerate plant colonisation and avoid the establishment of invasive species or the overrepresentation of pioneers (Figure 2.8G) (Burd 1995; Mossman et al. 2012).

The recommendations presented in this section provide advice to design a “generic” coastal wetland according to current scientific knowledge. However, due to the great diversity of coastal wetland shapes and functions, general guidelines may be insufficient to ensure that a specific site will evolve into a healthy habitat. In general, the main advice is to try and imitate the physical characteristics of a nearby natural saltmarsh (Burd 1995). Consequently, design guidebooks (e.g. Burd 1995; Williams et al. 2004; Nottage et al. 2005) stress the need for baseline surveys and quantified objectives at the beginning of each scheme in terms of expected changes in fauna and flora, expected proportion of saltmarshes and mudflats, expected changes within the breach and creek network etc. The realisation of these objectives should then be verified through long-term monitoring. Design strategies should be updated by drawing on practical experience following each scheme (Pontee 2015a). The next section will discuss to what extent these recommendations are being applied in the UK, and what remaining knowledge gaps need to be filled.

2.3.3 **Notion of success and challenges**

Section 2.3.1 explored the different factors motivating MR implementation. These depend on the country and stakeholders, and on the context that led to MR implementation (strategic, opportunistic or legislative) as explained in Section 2.3.2. The definition of success will thus differ based on the initial objectives of each scheme. However, a common denominator is the aspiration to recreate functioning coastal wetland habitats in order to preserve/replace significant ecosystem services cited in Section 2.2.2, which are threatened by human development and climate change. This thesis bases itself principally on ecomorphological rather than socioeconomic definitions of success, as focusing on habitat restoration makes it easier to provide general success criteria. An early definition of wetland restoration success is “one that consistently and reliably provides food and shelter for birds and animals whose existence may be otherwise threatened” (Haltiner et al. 1987). However, the success rate of MR schemes in the UK has been called into question in the recent literature (Brady et al. 2017). This section discusses the notion of ecomorphological success for MR schemes, as well as the factors that currently hinder the evaluation of MR success or failure.

The most common way of assessing MR ecomorphological success is to monitor the fauna and flora in comparison with that of nearby natural mature saltmarshes, or with that of the lost habitat that the MR scheme means to provide compensation for (Brady et al. 2017). Simple indicators include plants present, animals witnessed and percentage vegetation cover (Mitsch et al. 1996). However, saltmarsh species typically live in narrow environmental bands, leaving little tolerance when designing the MR template (Haltiner et al. 1987). This lack of leeway led to frequent failure to deliver stated objectives in terms of vegetation cover in the early US schemes (Race 1985). In the UK, recent studies also suggest that saltmarshes recreated through MR fail to replicate the plant community composition of nearby natural marshes (Mossman et al. 2012). The mitigated track record has led to criticism against MR and the “no net loss” strategy as a whole, as it risks legitimising habitat destruction without delivering equivalent recreated habitats (Brady et al. 2017). However, it is unclear whether remaining differences between natural mature coastal wetlands and MR schemes are due to inadequate design or to a premature assessment of success: establishment of natural saltmarsh vegetation may take around 100 years (Garbutt et al. 2008), well beyond current monitoring efforts, which generally range from 3 to 5 years after the breach (Mitsch et al. 1996; Esteves 2013), and rarely cover more than 10 years (Brady et al. 2017).

Consequently, while MR schemes are promising for delivering ecosystem services, and as a flood mitigation strategy in the context of rising sea levels, their long-term efficiency is uncertain. This is complicated further by the vague definition of objectives for what constitutes “successful habitat compensation” before implementing a scheme (Esteves 2013; Brady et al. 2017). Quantitative scheme objectives are not always available in reports (ABP mer 2011a, 2010), or may only provide target areas of mudflats and saltmarshes with no precision on their expected biodiversity or ecomorphological functioning (ABP mer 2011b; Pendle 2013). Also, while some form of MR monitoring is routinely conducted by the consulting or governmental bodies linked to the project design and management, their results can only be accessed as semi-quantitative reports that often fail to address the initial success criteria in terms of hectares of recreated mudflats and saltmarshes (ABP mer 2011b; Pendle 2013). The detailed monitoring data are generally unavailable (Morris et al. 2014). Insufficient dissemination of the monitoring data hinders the development of more in-depth or multidisciplinary academic studies of MR evolution and efficient learning from past practice. Finally, the scope of monitoring remains limited: post breach monitoring generally focuses on flora and fauna, with little funding available for geomorphological or hydrological monitoring, and no hindcasting of models.

In summary, MR as a long-term coastal management method is well embedded in UK legislation and guidance. Nevertheless, three main types of constraints arise from this review that can impede the functioning of the recreated coastal wetland and the success of MR. These are:

1) **Design limits**: the pre-defined objectives of the MR project in terms of physical and ecological characteristics are often poorly defined. This should be improved through a more holistic understanding of the physical, chemical and biological processes in coastal

wetlands, in order to make better informed decisions on whether and to what extent vulnerable sites can be compensated for (Wolters et al. 2005).

2) **Scientific limits:** a holistic approach to MR planning and design requires improved understanding of the processes at play in natural and recreated coastal wetlands, including hydrodynamics, surface and sub-surface processes, interactions between flora, fauna and the soil, and how these various processes impact the morphological evolution of coastal wetlands and their long-term resilience. One significant knowledge gap is the design of creek networks and their expected evolution in restored habitats, as will be explored in the next section.

3) **Monitoring limits:** Monitoring of MR schemes post-breach is either lacking, conducted over insufficient time-scales or not easily accessible. This further complicates the notion of MR success as the data available is insufficient to judge the functioning of the recreated wetland. Lack of monitoring data also hinders scientific interpretation of past MR schemes needed to learn from experience and develop a more holistic understanding of how these schemes function (Spencer et al. 2012). As outlined earlier, geomorphological monitoring of MR schemes in particular is lacking. This thesis will develop an easy to reproduce methodology to collect remote sensing data of creek evolution across the UK, in order to provide much needed datasets on MR creek morphometry and to facilitate future monitoring schemes.

Following current philosophies of “working with nature” and restoring natural physical processes (Brown et al. 2014; Pontee 2015a), a successful scheme should be one that develops the same morphological, hydrodynamic and ecological characteristics as a natural system. The next section will give more details on a major feature of saltmarshes that is still poorly understood despite being at the core of a coastal wetland’s functioning: the creek network.

2.4 Creek network design

Creek networks play a crucial role as an interface between land and sea within a coastal wetland. Due to their structural complexity, they remain a key knowledge gap in MR design. Old guidelines have been provided based on early MR schemes in the US (Haltiner et al. 1987; Burd 1995), but the extent to which these are being applied, or indeed their relevance to UK sites, has yet to be explored. Finally, in some cases socioeconomic considerations have led MR designers to opt for non-natural creek network shapes: the consequences of this choice on the ecomorphological functioning of the site remain uncertain. This section reviews the currently accepted processes driving creek network evolution in natural saltmarshes, before discussing the current design strategies adopted to encourage that evolution in UK MR schemes.

2.4.1 **Conceptual evolution model and equilibrium morphology**

Creek networks consist of an intricate system of bifurcating channels (Coco et al. 2013) in shallow coastal environments, that drive the exchange of sediment and water between estuaries and wetlands. Their stability depends on a delicate balance between sedimentary processes and hydrodynamics (Coco et al. 2013). A single creek system is characterised by an outlet, or entry channel, which constitutes the main point of entry and drainage of the tide. The "area of influence" that the creek system can potentially drain, and where sediment exchange is expected to occur, is referred to as the catchment area, as a parallel to fluvial drainage basins (Steel 1996).

Creek network development was first thought to be driven, in a similar way to fluvial channels, by the erosive action of a unidirectional flow, with the draining of the ebb tide as the main driver (Lohani et al. 2006). However, tide and wave energy dissipation on the flood tide also plays a significant role (Pethick 1992; Lohani et al. 2006): caution should be applied when comparing tidal creeks to fluvial systems because the flow affecting tidal creeks is bidirectional. Also, in natural systems only a fraction of the tide is distributed through the creek network, while the rest enters through over-ground flow over the marsh surface. Therefore, a distinction is made between the undermarsh tidal prism — the volume of water that the creek network can contain, and the overmarsh tidal prism — the volume of water the saltmarsh can contain between the marsh level and HAT (Vandenbruwaene et al. 2012).

According to Steel et al. (1997)'s evolution model, the development of a creek network in natural coastal wetlands occurs in four main stages (Figure 2.9). Stage A sees the tide-controlled initiation of the creek network on the unvegetated tidal flat. Due to the limited volume of these early channels, most of the water exchange occurs as overmarsh tidal prism (marked O in (Figure 2.9). In stage B, vegetation colonisation triggers a transition from tidal flat to saltmarsh and fixes the creek network (Temmerman et al. 2005; Vandenbruwaene et al. 2013). In stage C, the marsh accretes vertically in response to the low energy hydrodynamic conditions and the trapping action of the vegetation, encouraging the deposition of organic and inorganic sediment (D'Alpaos et al., 2007; Vandenbruwaene et al. 2013). The elevation gradient enhances water energy within the channels, causing their deepening and the appearance of higher order creeks by headward erosion (D'Alpaos et al. 2005). This causes a positive feedback loop: the increasing creek volume leads to a larger undermarsh tidal prism (marked U in Figure 2.9), while the overmarsh tidal prism diminishes as a result of vertical accretion, so most of the tidal energy is forced through the creek system. In stage D, the elevation is such that only the larger tidal events affect the upper marsh. The drainage density decreases and the higher order creeks are abandoned. The creek network is then considered to be at dynamic equilibrium with the tidal range: in case of sea-level rise, the saltmarsh can be expected to return to stage C and accrete vertically in response to the increased tidal prism (Steel et al. 1997).

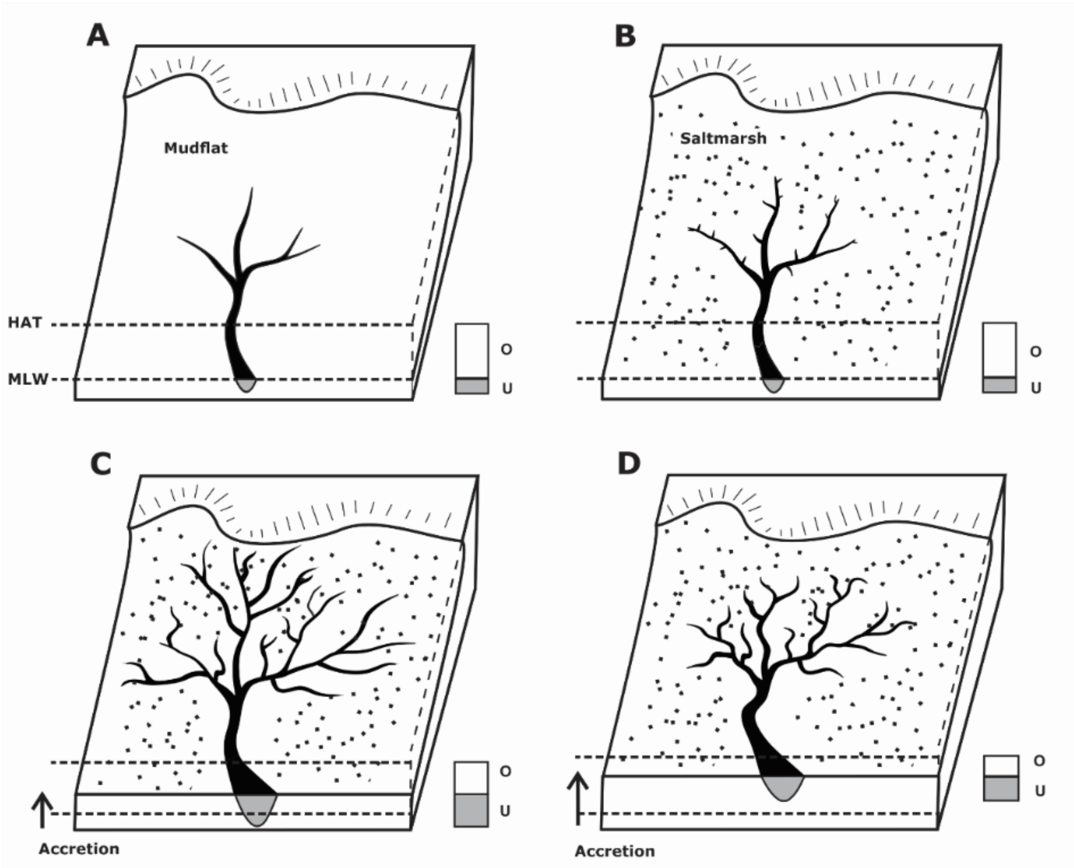


Figure 2.9: *Conceptual model showing the initiation (A) and development (B-C) of a tidal creek network towards a state of dynamic equilibrium (D) in response to tidal forcing, as the wetland accretes vertically and transitions from tidal flat to saltmarsh. Modified from Steel et al. (1997). HAT: Highest Astronomical Tide. MLW: Mean Low Water. O: Overmarsh tidal prism. U: Undermarsh tidal prism.*

The evolution rate and equilibrium morphology of creek networks will depend on local conditions such as antecedent geology, sediment type, vegetation cover, tidal and wave regime, and frequency of drainage pattern 'resetting' by storm events (Hughes 2012). Creeks are expected to undergo significant evolution towards their equilibrium state within 4 to 13 years (Williams et al. 2002), and some recorded mature creek systems are as young as 30-50 years old (Steel 1996). Despite the natural variability of creek network shapes, previous studies have established general trends of creek morphometry for marshes of various sizes; this has led to the definition of a number of morphological equilibrium relationships for natural mature creek networks (Steel 1996; Williams et al. 2002; Marani et al. 2003). Logarithmic relationships are typically used to relate planform or cross-sectional parameters of the creek network to the total size of the saltmarsh (Figure 2.10). Whether and to what extent these relationships are also applicable to MR sites will be explored in this thesis.

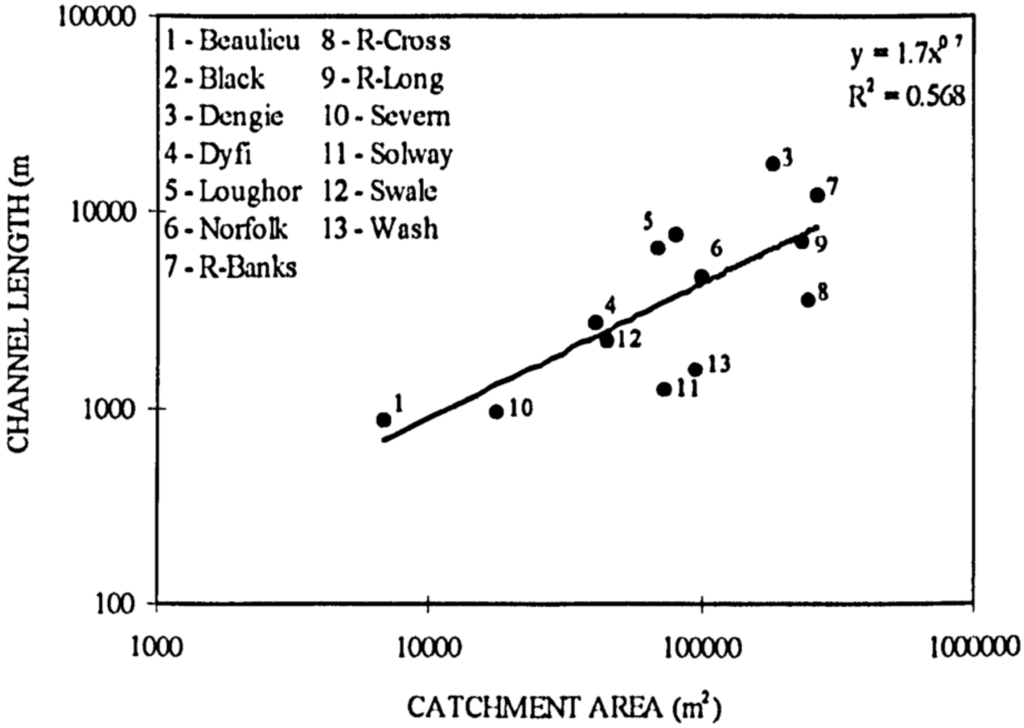


Figure 2.10: Example of a morphological equilibrium relationship relating the creek network morphometric parameters (here: total channel length) to the saltmarsh's dimensions (here: catchment area) (Source: Steel 1996)

The total creek system is thus proportional to the marsh area. Furthermore, the dimensions of individual creek segments are related to their position within the creek system. The composition and branching complexity of the creek network can be expressed quantitatively in terms of stream order, the most commonly used being the Strahler order. In this system, all terminal segments are assigned the order 1; when two segments of the same order meet, their confluence stream is one order higher, and so on (Strahler 1957) (Figure 2.11). The Strahler ordering system's utility is based on the premise that the order number is proportional to the catchment area and to the creek network dimensions (Strahler 1957), in agreement with the laws of drainage composition established by Horton (1945) on fluvial networks. This fits with findings from subsequent studies on intertidal creek network composition (Steel 1996; Zeff 1999), which have shown an exponential decrease in creek number and increase in creek length with increasing Strahler order (Figure 2.12).

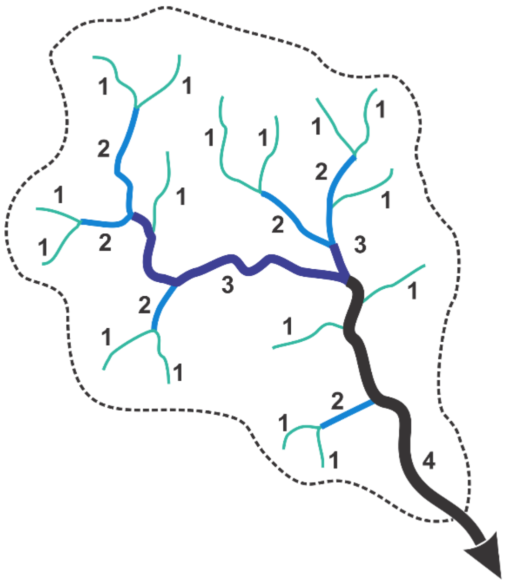


Figure 2.11: Illustration of Strahler ordering system (modified from Strahler 1957)

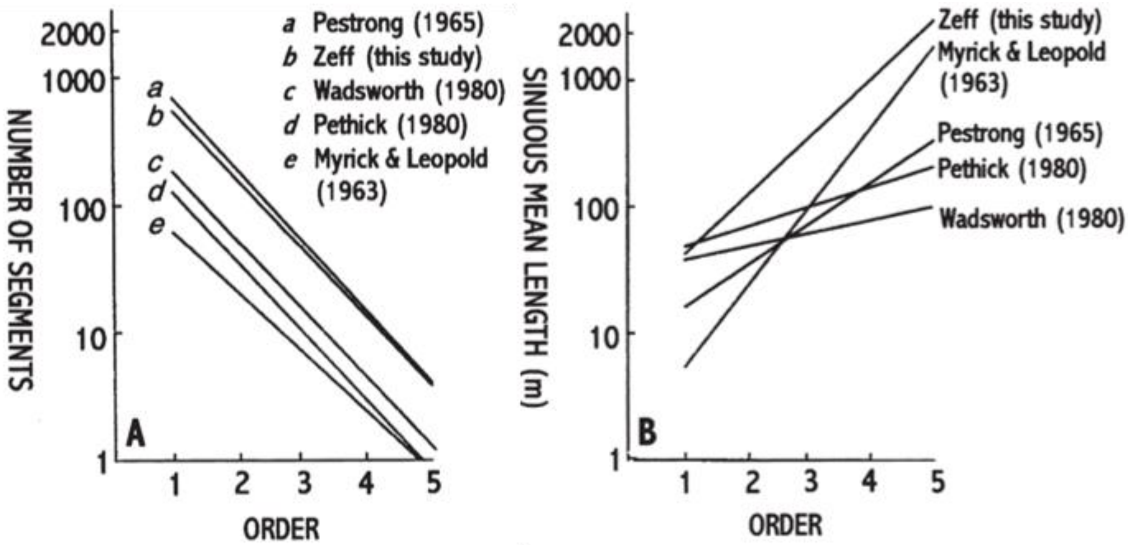


Figure 2.12: Horton's law of stream numbers (A) and stream lengths (B) derived from five studies on tidal marsh drainage networks in the US and UK (Zeff 1999).

Compared to the conceptual evolution model, the starting elevation of MR sites will likely differ from that of a natural stage A (tidal flat) (Figure 2.9A): it may be lower due to a lack of post-reclamation sedimentation, or conversely higher in the case of a recently reclaimed saltmarsh. There will be a phase of terrestrial vegetation die-off following the reintroduction of tidal influence if the terrestrial vegetation has been kept in place (Bowron et al. 2011). Seasonal variations in salinity, water content and bulk density of the sediment will be reintroduced along with tidal influence (Kadiri et al. 2011), which may lead to their destabilisation, and to erosion and slumping. The presence of relic features, such as agricultural drainage trenches, can also influence the development of a creek network by constraining it to a particular morphology (Crooks et al.

2002). The latter factor is expected to concern most MR schemes as they dominantly occur in reclaimed agricultural lands. The presence of a breached seawall in most MR sites is also expected to affect the flow regime within the saltmarsh by constraining it to the creek system (Hood 2014).

For all of these reasons, the development of creek networks in MR schemes is likely to differ from that of a natural system, and a separate conceptual evolution model will be needed. Whether or not MR creek networks also follow Horton’s laws of stream numbers and lengths has yet to be determined. New morphometric parameters of MR creek network evolution, using a reliable creek extraction tool, must be extracted to establish these differences. Finally, in order to advise the design of artificial creeks in MR schemes, more information is needed on the initiation and early development of creeks in natural saltmarshes.

2.4.2 Creek initiation and development processes

Since early creek development is difficult to observe in the natural environment, laboratory and numerical models are the main available tools to investigate the processes at play. Physical or numerical models make it possible to monitor a simplified system at a superior resolution and frequency than could be achieved with field observations alone (Vlaswinkel et al. 2011). Models give control over the external conditions impacting the study site, such as the tidal regime or the initial channel geometry (Tambroni et al. 2005). They are therefore useful to explore the morphodynamic processes that control creek growth. However, they still need to be validated with field data, especially in the case of complex systems like creek networks (Stammerman 2013; Zhou et al. 2014).

Tidal creek formation has been investigated in laboratory tank experiments (Tesser et al. 2007). Compared to numerical modelling, laboratory experiments have the advantage of reproducing physical processes that cannot be simulated through equations, such as the erosion of the banks, and are thus less simplified (Tesser et al. 2007; Kleinhans et al. 2012; Zhou et al. 2014). The main limitation of laboratory models is the scaling effect. Since the modelled creek network develops in a much smaller basin than its real-world counterpart, and over a much shorter timescale, the forces involved will not be identical (Heller 2011). The scaling effect is hardest to avoid in the case of sediment transport modelling: even if similar width/depth ratios are preserved, the grains used will be too large compared to the size of marsh. The hydraulic energy of the system is then too low to allow for sediment motion to occur, which hinders creek formation processes. Such inaccuracies can be partially compensated by other distortions, for instance choosing a sediment of lesser density as substitute (Tambroni et al. 2005). Common substitutes are lightweight plastic grains, but they have further limitations, such as failing to represent sediment suspension or cohesiveness (Tambroni et al. 2005; Tesser et al. 2007; Iwasaki et al. 2012).

Even considering these limitations, experiments have succeeded in reproducing a morphology similar to that of natural creek systems, with exponential widening of channels in the seaward direction (Stefanon et al. 2010). However, the channels obtained are overly wide and shallow (Stefanon et al. 2010; Iwasaki et al. 2012). This exclusively erosive behaviour is partly due to the absence of sediment input in most physical models: this model cannot accurately represent creek network growth in infilling basins where the sediment inputs supersede sea-level rise (Stefanon et al. 2010). The absence of vegetation and cohesive sediment also lowers the stability of the creek banks and contribute to their widening.

Creek formation according to such models is mainly due to headward erosion into the marsh surface once a critical bottom shear stress is exceeded. Maximal shear stress is expected to occur during the ebb velocity maximum, when the water level falls below marsh level and the flow focuses in the channels (Stefanon et al. 2010). This process occurs intermittently in the marsh, during the higher (Spring) tidal events that can overtop the creek edge (Fagherazzi et al. 2001). Despite the low frequency of tidal events that allow erosion to occur, headward erosion can be rapid in natural saltmarshes, with reported rates of headcut migration of 5–7 m/year under normal conditions (Steers 1960 *in* French et al. 1992), and up to 400-500 m/year in response to catastrophic changes in tidal prism due to breaching (Knighton 1992; Symonds et al. 2007).

Similar to laboratory experiments, numerical models allow researchers to target specific influencing factors in order to investigate their impact on the whole system. Here, rather than physically reproducing the channel network development processes at a smaller scale, and dealing with scaling problems, morphodynamic processes are represented by equations. Changes to the baseline state of the marsh are reproduced by iteratively applying these equations to represent variations over time. The equations stem from either simplified hydrodynamic equations (Rinaldo et al. 1999) or empirical equations derived from field observations and statistically correlated to environmental controls (Fagherazzi et al. 2012).

Some numerical models of creek development use a simplified coastal morphology (Marciano et al. 2005; Iwasaki et al. 2012); others base themselves on tidal channel networks extracted from digital terrain maps. The creek evolution predicted by the model can then be compared with the equilibrium state observed on site (D'Alpaos et al. 2007). Model results demonstrate the importance of the initial morphology: the modelled creeks tend to closely resemble the ones observed in real life, even when other input parameters (tide and sediment characteristics) are changed. Indeed the topography controls the inundation time, the flow routes, and the spatial distribution of bottom shear stress, hence where channel incision will occur (D'Alpaos et al. 2007).

The effect of vegetation on creek development has also been investigated with numerical models, highlighting their erosive effect under certain circumstances. (Temmerman et al. 2005; Schwarz et al. 2014). It is generally accepted that vegetation stabilises the substrate, directly through the binding properties of the roots and indirectly through the flow reduction caused by the stems and leaves (Marani et al. 2006; Gedan et al. 2011). However, if the elevation is such that the vegetation is not entirely submerged at high water, dense vegetation patches can obstruct the flow and redirect it to channels that will thus be incised more efficiently (Temmerman et al. 2007; Schwarz et al. 2014).

The role of the initial topography is thus predominant for creek initiation and development, as it controls preferential flow paths over the marsh during both flood and ebb tide. Areas of higher flow will tend to erode into channels, and the surrounding areas, characterised by weaker flows, will accrete into levees if the sediment input permits it (Stumpf 1983; D'Alpaos et al. 2007). Similarly, the shape of the initial creek system determines the distribution of shear stress, which will be higher at the channel bends (D'Alpaos et al. 2007). Vegetation growth is then expected to occur preferentially near the creeks which provide drainage and seed transport (Zedler et al. 1999). Plants contribute to sediment deposition near the creek banks through the trapping action of stems and leaves, while focusing the flow and causing erosion around them. Plants therefore act as a positive feedback to the hydrodynamically-driven creek network development.

Other factors can affect the stability of the creek banks and the availability of sediments: surface water runoff (but in the sites considered in this thesis, saltmarshes connected to a freshwater outlet were avoided), rainfall, which can erode the creek bank and redistribute sediment within the marsh (Murphy 2006), and bioturbation by fauna (Perillo 2003, Fagherazzi 2011). Finally, the circulation of water through the marsh and the capacity of creeks to form via erosion will be influenced by the soil structure (Figure 2.13). There is previous evidence that, in restored coastal wetlands, the pre-restoration disturbances have led to a reduction in microporosity and in pore network connectivity (Spencer 2016, 2017), which hinders subsurface water circulation and root penetration.

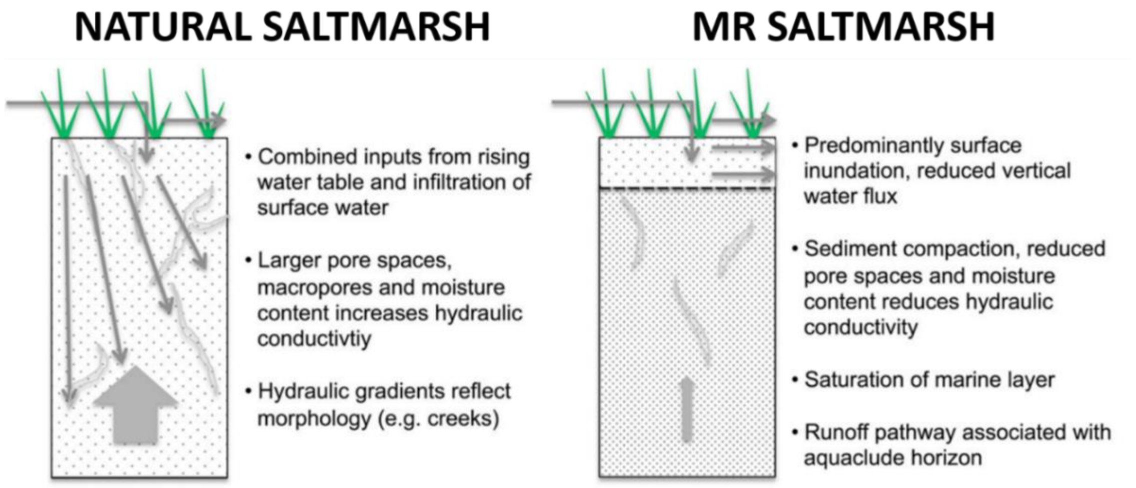


Figure 2.13: Conceptual model of drainage patterns in MR marshes and natural saltmarshes. Block arrows represent subsurface (groundwater) inputs, with thinner arrows representing impeded flows. Line arrows represent surface flooding inputs. Modified from Tempest et al. (2014)

In summary, despite starting out with different initial hypotheses and limitations, such as lack of vegetation or lack of depositional processes, models generally tell similar stories of creek network growth. It is therefore argued that the underlying mechanism remains the same whether the marsh is accretional or erosional: new creeks are formed in locations where bed shear stress exceeds a critical threshold for sediment motion, either through erosion along the path of greater flow or through deposition on the banks where the flow is weaker (Friedrichs 1995; Stefanon et al. 2010), causing more sediment transport along the channels and the formation of levees on the banks (Stumpf 1983). Vegetation provides a positive feedback mechanism: their growth is expected to occur in higher densities near the creeks (Morzaria-Luna et al. 2004), where they contribute to sediment deposition and levee building. The figure below provides a conceptual model of the expected morphodynamic interactions between initial site topography, plant growth and creek development (Figure 2.14). Soil structure and sub-surface processes also play a significant role on creek formation and stability, but require further research before they can be incorporated into the conceptual evolution model.

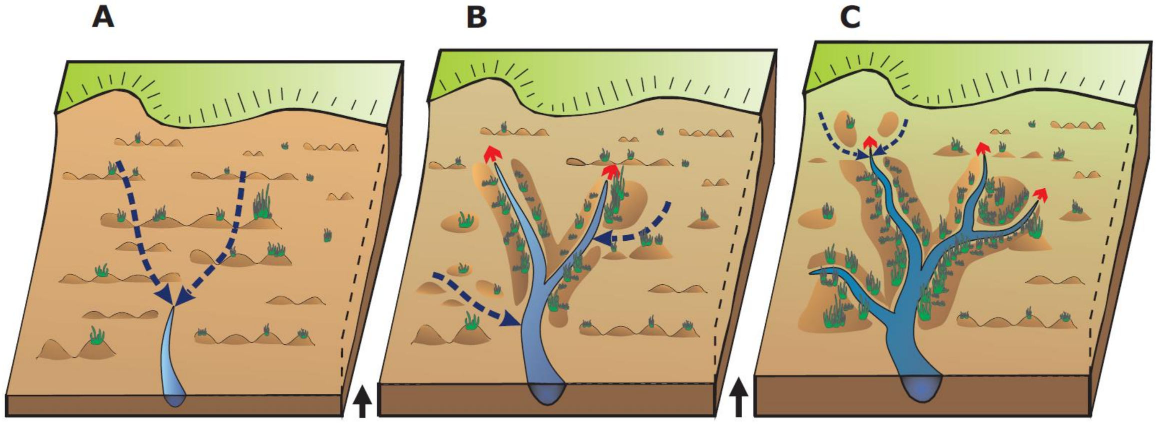


Figure 2.14: Conceptual model of creek formation processes during the mudflat-saltmarsh transition phase. A: Initial site topography and vegetation clumps lead to preferential flow paths of increased shear stress (dashed blue arrows) and to the initiation of a creek network. B: As the mudflat accretes vertically (black arrow), scouring along the flow paths and deposition in areas of low shear stress lead to the expansion of the creek network; feedback processes occur in the form of levee building and vegetation growth near the creeks, further contributing to sediment trapping; increased flow velocity within the creek system leads to channel down-cutting and headward erosion (short red arrows). C: complexification of the creek network as the site transitions into a vegetated saltmarsh; increase in sinuosity due to higher shear stress at the bends, leading, in some circumstances, to the cutting of new channels. Compiled from information in Stumpf (1983); D'Alpaos et al. (2005); Temmerman et al. (2005); D'Alpaos et al. (2007) and Fagherazzi et al. (2012).

A more detailed conceptual creek evolution model has been proposed based on model results and field observations. However, numerical and physical models cannot yet reproduce the whole range of processes that control creek evolution, so monitoring the evolution of creek networks in natural and artificial saltmarshes is critical to improve the state of knowledge. In that regard, MR schemes provide an open laboratory to study early creek development. The thesis will provide new MR creek monitoring data in an attempt to statistically link them to environmental controls and to initial design conditions, in order to lay the ground for a new conceptual model for MR creek development.

2.4.3 Design of creek networks in managed realignment schemes

Creek networks are considered crucial to the continued health of MR saltmarshes (Burd 1995; Nottage et al. 2005). Though letting creek networks form spontaneously in MR schemes is presented as a potential design technique by some authors (Nottage et al. 2005), unassisted evolution is expected to take "many decades" (Williams et al. 2004), with potential negative impacts on biodiversity. Furthermore, whether MR creeks can develop to a similar equilibrium morphology and distribution as natural saltmarshes is debatable. MR schemes generally occur in agricultural lands, where the compacted ground could slow down or even prevent the spontaneous initiation of a creek network, especially if the elevation of the site is high within the tidal range (Williams et al. 2004).

Due to these concerns, excavation of an initial creek system is being undertaken more and more frequently in MR schemes in the UK. The working hypothesis is that excavating artificial creeks will jump-start creek development towards the morphological equilibrium state described in Section 2.4.1, and ensure that appropriate habitats are created. In order to provide self-sustaining mitigation habitats, restored coastal wetlands should be designed so the site elevation and creeks will be able to reach a state of dynamic morphological equilibrium where they can keep up with changing input conditions, especially sea-level fluctuations (Pontee 2015a). Thus, in order for a MR creek design to be considered cost-efficient, it needs to kick-start creek development so that evolution towards equilibrium occurs faster than if no creek excavation had taken place. Ideally, MR creek evolution should occur within a similar timescale as for immature natural systems, or even faster if the MR creeks are excavated close to their equilibrium morphologies. However, there is no decisive evidence to determine whether and to what extent this strategy accelerates the development of an active creek network.

Three main strategies are currently being used in the UK to provide drainage to MR schemes. Strategy 1 consists in providing conditions resembling that of an immature tidal flat where a creek network is more likely to spontaneously develop. Strategy 2 attempts to reproduce a mature creek network morphology, generally by deepening a relic creek network predating the reclamation of the site. Strategy 3 consists in excavating an initial template of channels in the absence of a relic system of creeks. In the latter strategy, the design of the initial template should be informed either by knowledge of creek-forming processes or by practical experience. Experimental MR schemes from San Francisco Bay in the 1970–1980s have led to a number of guidelines (Table 2.1) in order to provide an initial template that encourages the unassisted evolution of MR creeks. A lot of these guidelines are now dated, and their continued relevance to recent MR schemes in the UK will be investigated in Chapter 8 of this thesis.

These guidelines cover most aspects of creek morphological design. However, they are based mainly on experimental MR schemes in the US, and their efficiency at promoting creek growth and delivering functioning coastal wetland habitats remains to be established. The extent to which these guidelines are being applied to current MR projects in the UK is also unclear. Indeed, several factors may prevent MR creek guidelines from being applied, leading to a creek morphology that significantly differs from natural systems. Initial channel designs may be modified due to site constraints (presence of archaeological sites or pipelines), or to deliver additional ecosystem services such as amenity value (Pontee 2015a), flood protection and water storage space for storm surges (Manson et al. 2012), or safe grazing space for sheep and cattle (Burgess et al. 2013). The creek network morphology may also be simplified to reduce implementation costs, most commonly by adjusting pre-existing agricultural drainage ditches. For those reasons, design strategy 3 is the most widely applied in terms of potential design, and will be disproportionately represented in this study.

Table 2.1: Summary of existing creek network guidelines for MR schemes

Factor	**Guidelines**
Initial marsh elevation	Initial marsh elevation should be about 0.5 m below MHWS to promote depositional processes (Burd 1995). Slope should be added to the marsh surface to encourage drainage (Williams et al. 2004).
Total creek volume	The creek volume should be at equilibrium with the potential tidal prism to prevent sediment infill, erosion or poor water circulation (Haltiner et al. 1987). Equilibrium relationships relate creek volume to the total discharge (Haltiner et al. 1987) and outlet cross-sectional area (Steel 1996).
Channel length	Channel length per creek should be a function of the creek order following Horton's power laws of channel length decrease with increasing Strahler order (Coats et al. 1995).
Channel cross-sectional dimensions	Channel width and depth should obey hydraulic geometry relationships, and have similar width/depth ratios as natural systems (Zeff 1999).
Bifurcation ratios	Bifurcation ratios should be approximately 3.5 as in natural systems (Coats et al. 1995).
Junction angle	Slough junctions should be around 120 degrees, and channel junctions with sloughs at around 90 degrees to imitate natural systems (Haltiner et al. 1987).
Sinuosity ratio	The mean sinuosity ratio should range between 1.1 and 2.0 for Strahler orders 3-5 to imitate natural systems. Smaller channels will tend to be straighter (Coats et al. 1995).
Drainage densities and distribution	Drainage density should be similar to that of nearby reference saltmarshes (Williams et al. 2004). Maximum distance between creeks should be about 30 m (Haltiner et al. 1987).
Inherited structures	Inherited structures like drainage ditches should ideally be infilled so as to prevent overly straight channels (Williams et al. 2004).

2.4.4 Creek network detection and parametrisation

A key challenge of saltmarsh morphological studies is mapping complex features over large areas. In MR schemes in particular, precise surface elevation monitoring should ideally be conducted on site using high resolution equipment (Nottage et al. 2005). However, due to lack of field surveying, creek network monitoring still relies heavily on remote sensing data, such as aerial photography or elevation maps from lidar (Light Detection and Ranging). Creek networks are still primarily extracted from these maps through manual digitisation, then corrected with field surveys. That method is too time-consuming, expensive and subjective for large-scale, long-term monitoring (Mason et al. 2004).

Automating creek extraction methods have been explored in various studies. A common method is the eight direction (D8) flow accumulation model (Ozdemir et al. 2009; Passalacqua et al. 2010; Lang et al. 2012). This hydrodynamic detection tool produces a one-pixel wide flow path, the size of which depends on the resolution of the dataset, corresponding to the channel centreline, but does not detect the creek planform area. In addition, as tidal channel development does not depend solely on the runoff measured by flow accumulation, but also on other topographic and erodibility factors, this makes the D8 method at best a first approximation of a creek network (James et al. 2010). This is especially true for artificial creeks, which are primarily shaped by human intervention rather than in response to hydrodynamic forces. Therefore, the creek network in MR schemes is better defined by its artificially chosen morphological template rather than by the flow conditions.

Alternative semi-automated creek extraction methods using lidar rely on elevation and slope or curvature thresholds (threshold methods), determined through sensitivity tests (Fagherazzi et al. 1999; Lohani et al. 2006; Mason et al. 2006; Lashermes et al. 2007). However, due to the complexity of tidal channels, no entirely automated method has been developed, and manual corrections are necessary. A recent method (Liu et al. 2015) comes close to full automation by enhancing and extracting Gaussian-shaped cross-sections from lidar, corresponding to channels. Though the method seems promising for natural saltmarshes, initial testing conducted as part of this research showed that it may not apply to artificially excavated creek networks in MR sites as they can assume a variety of shapes other than Gaussian (i.e. rectangle, terraces or multimodal curve).

Although some previous studies have manually extracted creek cross-sections to infer volumetric parameters (Mason et al. 2006), none of the currently proposed algorithms include a fully automated parameter extraction routine. This makes them poorly adapted to morphological interpretation, and does not exploit the full potential of lidar data. In order to fulfill the objectives of this thesis, a new algorithm is needed to map the creek network using lidar data and provide a comprehensive list of parameters, such as length, width, depth, cross-sectional area, sinuosity, junction angle, bifurcation ratio, and drainage efficiency for all creeks. The newly-developed algorithm should be automated, or semi-automated with minimal user inputs, in order to reduce the subjectivity and lack of replicability that characterise manual mapping techniques.

2.5 Summary

Temperate coastal wetlands such as saltmarshes and tidal flats are valuable habitats that provide a range of ecosystem services, including cost-effective coastal defence. Furthermore, under the “no net loss” policy in place in most European countries, wetland losses due to coastal squeeze and urban development must be compensated for. In recent years, the rate of compensation habitats implemented through MR has kept pace with estimated national rates of wetland loss. However, there are growing concerns that MR schemes do not provide habitats of equivalent quality to that of the lost natural wetlands. The success rate of MR schemes in the UK, wherein a successful scheme is defined as one that develops the same morphological, hydrodynamic and ecological characteristics as a natural system, is difficult to assess due to the limited scope of monitoring programs: they are limited to 5–10 years post-breach and the monitoring does not generally include geomorphology and creeks. As creek networks are known to play a significant role on the ecological functioning of coastal wetlands, the design and evolution of creek networks in UK MR schemes is a crucial knowledge gap that this thesis proposes to address.

This thesis aims to improve understanding of the morphological evolution and expansion of natural and MR creek networks in order to derive implications for future design. This aim can be separated into four sub-objectives:

i. First, the equilibrium morphology of creek networks in natural coastal wetlands needs to be better defined using newly developed monitoring tools. Indeed, the latest estimations of equilibrium relationships for UK saltmarshes date back to 1996 and do not make use of recent monitoring techniques such as lidar. This knowledge gap will be filled by Thesis Objective 1: to provide an exhaustive morphometric analysis of creek systems in natural coastal wetlands in the UK, using remote sensing data and novel parametrisation methods, to define equilibrium characteristics that will be used as an end target for the design of MR creeks.

ii. Second, the lack of data available in the scientific literature for MR creek evolution needs to be addressed. The lack of MR creek monitoring makes it difficult to learn from experience: indeed, various creek designs have been tested in the UK between 1995 and 2014, but their impact on creek evolution are uncertain. This knowledge gap will be filled by Thesis Objective 2: to provide a morphometric analysis of creek systems in 10 MR schemes in the UK over the years, in order to infer creek evolution rates.

iii. Third, it is currently unclear whether MR creek networks in the UK evolve towards the equilibrium shape of natural systems, or whether they remain significantly different after 5 to 20 years of evolution. This knowledge gap will be filled by Thesis Objective 3: to compare the morphological characteristics of natural and MR creeks to determine whether MR creeks evolve towards more natural systems over time, and whether their evolution rates, inferred from the monitoring of creek morphometric parameters undertaken in Thesis Objective 2, can be related to initial conditions and design choices.

iv. Fourth, it is uncertain whether current creek design guidelines, most of which come from experimental US sites from the 1970s–1980s, are adapted to UK sites, properly applied and sufficient to encourage creek development. This knowledge gap will be filled by Thesis Objective 4: to establish a conceptual creek evolution model for MR schemes that highlights the main divergences from natural creek evolution, and propose measures to reduce these divergences in future schemes.

Chapter 3: Data sources and study sites

3.1 Introduction

This chapter introduces the data sources and study sites that will be used throughout the thesis. Section 3.2 presents the lidar and tidal data as well as their sources of uncertainty. Section 3.3 introduces the 13 natural mature saltmarshes used as a reference for MR schemes. The creek morphological characteristics of these sites will be used to validate the morphological equilibrium relationships found by Steel (1996), to be used as end targets for MR creek design. Section 3.4 introduces the 10 MR sites considered for this study, as well as their design and implementation context. Finally, Section 3.5 introduces 2 accidentally realigned sites: these sites, opened to tidal influence in 1897 and 1921 due to storm surges, provide insight into the possible long-term evolution of MR schemes. Descriptions and justification of datasets and methods specific to individual result chapters are described in their respective chapters.

3.2 Data sources

Two main types of datasets were used to assess the evolution of tidal creek networks in 13 mature coastal wetlands, 10 MR sites and 2 accidentally breached sites distributed around England and Wales (Figure 3.1): tidal data and lidar data. These are described in the sub-sections below.

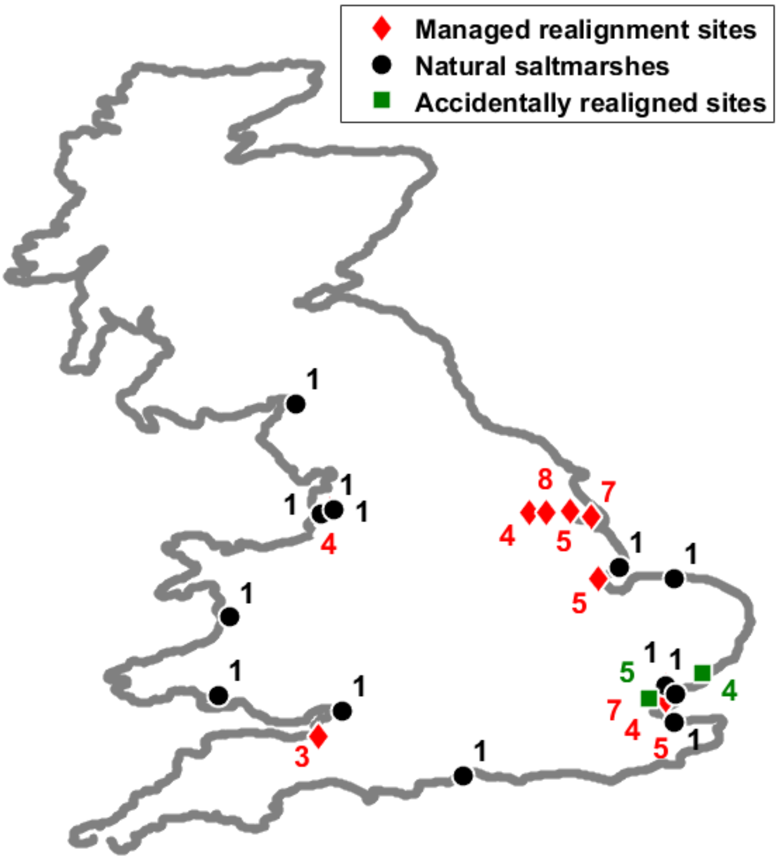


Figure 3.1: Location of the natural saltmarshes and MR sites for which tidal and lidar data were collected (numbers = number of lidar datasets suitable for each site)

3.2.1 **Lidar data**

Lidar is an airborne sensor that measures distances by transmitting a laser pulse and calculating the return time of the reflected beam. Lidar is particularly efficient for the geomorphological monitoring of coastal environments (Lohani et al. 2006; Mason et al. 2006). The use of lidar is increasing due to its accessibility in many countries, high coverage (e.g., all of the UK territory), high horizontal resolution (up to 0.25 m horizontally), and the possibility to compare datasets over several years to infer evolution rates.

The lidar data used in this thesis were collected by the Environment Agency (EA) and Natural Resources Wales, and accessed via the survey open data (https://data.gov.uk/) and Lle download portals (http://lle.gov.wales/Catalogue/Item/LidarCompositeDataset/), respectively. EA obtained datasets are interpolated from a point cloud into a raster using a nearest neighbour interpolation to Euclidean distance method. Contrary to other interpolation methods, nearest neighbor preserves variations in the data such as small (~1 m) channels which would otherwise be smoothed over (Lohani et al. 2001), making this method particularly suited to representing creek networks (Rapinel et al. 2015). The freely available data comes in two forms: a raw Digital Surface Model (DSM) interpolated from the points of first laser beam return, and a Digital Terrain Model (DTM) where the above-ground features such as buildings and vegetation have been filtered out to obtain a bare-Earth model using EA proprietary algorithms (Liu 2008).

The vertical resolution of lidar data is estimated at 0.15 m or lower based on GPS ground-truthing surveys, as described in the EA's lidar quality control reports (Environment Agency 2016). Horizontal resolutions vary between 0.25 m and 2 m. Due to the fractal behaviour of creek networks (Vlaswinkel et al. 2011) their morphological complexity increases with the horizontal resolution, until the limiting case of the "Koch's Snowflake" is reached, an object of finite area but infinite perimeter (Zhou 2011). Thus, in order to compare the morphological evolution of the site at different years, all datasets need to be at the same resolution. The most common available horizontal resolution, 1 m, is therefore used as a standard for this thesis. Where necessary, datasets have been down sampled to 1 m by taking the mean of the higher resolution cells. Further quality control was undertaken to justify carrying out a multi-annual monitoring of coastal wetlands using lidar:

- Tests of the repeatability of lidar data collection over different years;
- Investigations of the potential systematic bias caused by the presence of low-lying saltmarsh vegetation in the DSM and DTM;
- Comparison of the respective efficiency of the DSM and DTM at detecting the marsh surface

3.2.1.1 Repeatability of lidar

The repeatability of lidar data was tested for 5 different datasets collected near HOMW in 2007, 2009, 2010, 2011 and 2014. Elevation values were found using the nearest neighbor interpolation method along a low-lying unmoving feature, in this case a road (Appendix A1). The largest vertical repeatability error estimated from the mean standard deviation for the five points is +/-0.06 m, which is less than the 0.15 m vertical resolution of lidar.

3.2.1.2 Effect of vegetation

The effects of saltmarsh vegetation can lead to an overestimation of the marsh surface by lidar and a systematic bias in the dataset. This effect needs to be quantified. Also, the capacity of the EA proprietary vegetation removal algorithms at minimising this bias needs investigating. Real-Time Kinematic (RTK) GPS data were collected with an accuracy <2 cm and a precision <1.5 cm in

August 2015 in a MR scheme at Tollesbury and in an adjacent natural mature saltmarsh (Figure 3.2). The marsh surface elevation data relative to ODm were collected across a 50x50 m section following the systematic grid sampling method described by Brooks et al. (2015) and expanded by Lawrence (2018). 36 focal sampling points are selected every 10 m, with 4 additional sampling points at 1 m N, S, E and W of each focal points. For 8 randomly selected focal points, 20 additional points are taken at 2 and 3 m N, S, E and W of each focal points. Finally, 25 additional points are selected within the grid at 5 m N, S, E and W from the focal points.

The method is used to study ground heterogeneity over length scales ranging from 1 to 80 m, and is well applicable to lidar ground-truthing. The closest in time lidar data available are from the 22nd February 2015. Contrary to the GPS data lidar may detect clumps of vegetation. Some marsh accretion may have occurred between February and August 2015; however, with accretion rates in Tollesbury MR of 2.3 cm/yr to 2.89 cm/yr estimated by Garbutt et al. (2006) and the present study, the difference is unlikely to be detected by RTK GPS or lidar. Most of the differences between the two datasets should thus be due to the vegetation present in February 2015.

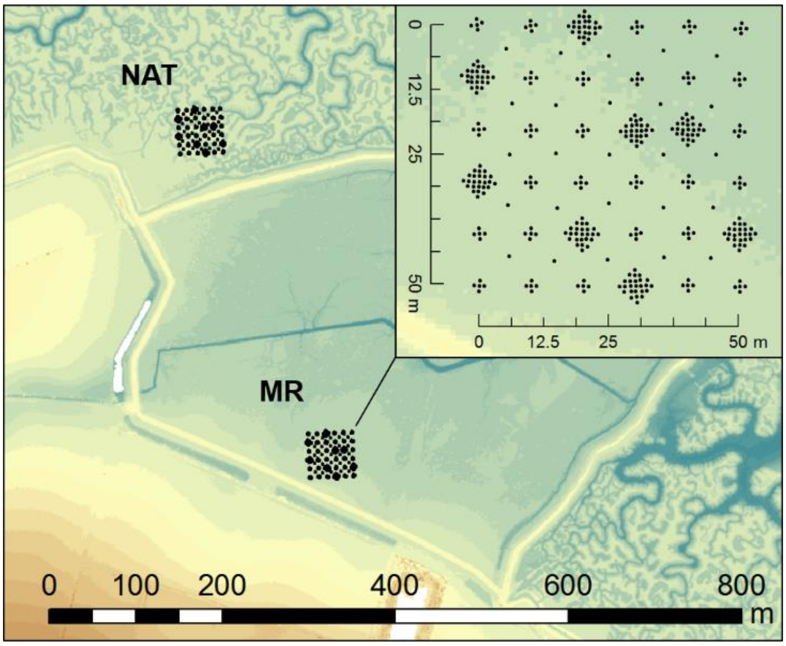


Figure 3.2*: Layout of RTK-GPS points collected in August 2015 in a natural marsh (NAT) and in the Tollesbury MR schemes (MR), overlain over lidar DSM collected in February 2015. RTK GPS dataset curtesy of Peter Lawrence.*

Most data points (97% using the DSM and 96% using the DTM) fall within the limit of agreement (Figure 3.3). A positive systematic bias of 0.09 m was found between the lidar and GPS data, for both the DSM and the DTM: lidar tends to overestimate the elevation values by 0.09 m. This value fits with the expected elevation of dense saltmarsh canopy in the UK (Neumeier 2005). The persistence of this positive bias in the DTM shows that the EA vegetation removal algorithms fail to systematically remove the vegetation cover. This is expected when the vegetation cover is lower than the vertical resolution of lidar; however previous studies have found lidar DTM to record dense clumps of vegetation as high as 2 m (Pontee 2014c). The Root Mean Square Error of the lidar versus GPS elevation data is higher for the natural saltmarsh (0.13 m) than for the MR scheme (0.07 m), probably due to the higher ground heterogeneity (Brooks et al. 2015) and higher vegetation diversity found in natural saltmarshes (Mossman et al. 2012).

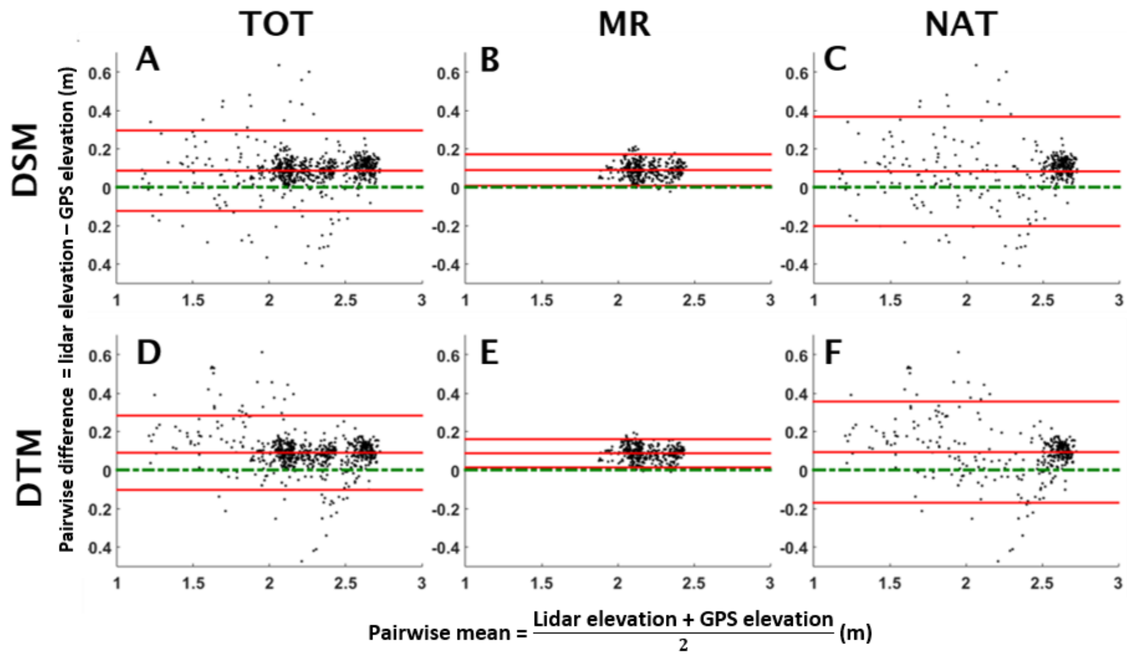


Figure 3.3: Differences between lidar (A-C: DSM and D-F: DTM) and RTK-GPS data at a natural (NAT) and artificial (MR) marsh in Tollesbury, and at the sum of both locations (TOT). The three red lines show the limits of agreement (2xSTD) and the mean value of the differences. The dashed green line shows the ideal mean difference if there is no bias between the two methods.

Finally, some fluctuation can be expected in the mean vegetation cover between summer and winter (Boorman, 1997). This should be visible as fluctuations in the mean marsh elevation data from year to year. If interannual fluctuations in the mean marsh elevation data are small, and if the elevation changes are linear, then it can be assumed that the changes are due to accretion rather than vegetation growth and die-back. Elevation changes data as detected by lidar should also be confronted with field monitoring data of accretion rates when available in the literature.

3.2.1.3 DSM/DTM comparison

Compared to the uncorrected DSM, the EA proprietary DTM generation algorithms aim to remove first return points corresponding to vegetation or standing water, and to replace them with points of latter arrival corresponding to the ground. In order to explore the efficiency of those algorithms, the DTM was subtracted from the raw DSM in HOMW, were trees were present in 2009, for 5 different years (Appendix B1). Most of the differences between the models occur along and inside the creek network, where the elevation is higher in the DSM than in the DTM. Some of the larger positive values correspond to trees, isolated or in a line along one of the main channels. According to the standard deviation values, discrepancies of about 0.26 m can be expected between the DSM and the DTM.

However, overall the amount of correction differs greatly from one year to the other: it is suspected that the algorithm itself gets updated over time, but that the newest algorithm is not systematically applied to the older datasets, leading to inconsistencies. Some tiles in the 2007 dataset also seem to have undergone no correction at all, showing that the algorithms are not always systematically applied, in accordance with previous findings (Leung 2017).

Uncertainties concerning the functioning and consistency of application of EA algorithms make the DTM datasets unreliable for monitoring the morphological evolution of coastal wetlands. Furthermore, since the DTM datasets do not remove the systematic bias caused by low saltmarsh vegetation as seen in Subsection 3.2.1, this study uses the raw DSM. This approach assumes that the low vegetation cover characteristic of saltmarsh areas is unlikely to mask the creek network or significantly affect the detection of creek edges, as based on aerial photography and field observations, plants develop on the creek banks but rarely within the creeks themselves. This choice might lead to underestimation of the channel depth, if the laser is reflected by residual water within the creek during low tide. It could also lead to overestimation of the saltmarsh elevation if the vegetation cover is detected as the ground level, and could limit the monitoring of accretion rates when performing the MR evolution analysis over several years (Wang et al. 2009; Hladik et al. 2013).

Some open-source algorithms exist attempt to correct vegetation from lidar data, but 1) they rely on precise knowledge of the vegetation distribution on site, obtained through regular field surveying, which defeats the purpose of lidar as a quick and cheap monitoring method over large areas, and 2) the correction factors based on local dominant vegetation height creates unrealistic “steps” in the dataset, which may complicate the detection of creek networks (Hladik et al. 2013). Overall, vegetation removal has been a major issue in previous lidar-based saltmarsh monitoring studies (Wang et al. 2009): even state of the art lidar sensors fail to penetrate the saltmarsh canopy (Hladik et al. 2012). This study provides an opportunity to estimate the efficiency of the uncorrected DSM at detecting evolving creek systems.

3.2.2 **Tidal data**

In order to establish tidal forcing parameters, mean predicted tidal levels were obtained for 582 standard (tidal data tabulated) and secondary (tidal data calculated from the standard ports) ports, as provided by the Admiralty Tide Tables 2014. Visual comparisons found a good agreement between the predicted and actual tidal levels at the 6 standard ports closest to the studied MR sites (data not shown). The mean tidal levels at the 13 natural saltmarshes and 10 MR sites were then interpolated from the weighted mean of the surrounding ports' values (up to 30 km away). The tidal asymmetry was approximated for each port by the ratio of the mean flood and ebb periods, taken as the average of three representative Spring and three representative Neap tidal cycles respectively, then interpolated for each site in the same way as the tidal levels. The effect of sea-level rise was considered negligible in this study which spans a maximum of 20 years of creek evolution: since observed rates of sea-level rise around the English Channel in the 20th century range between 0.8 and 2.3 mm/yr (Haigh et al. 2009), the change in tidal elevation over 20 years is unlikely to exceed 0.05m, which lies below the vertical resolution of lidar.

The chosen method of estimating Neap and Spring tidal levels provides a higher level of confidence than that of most other studies of MR scheme evolution, which usually take the tidal values from the nearest port (Pontee et al. 2006). However, the weighted mean interpolation does not take into consideration the shape of the coastline, which could be a problem in complex areas with several small estuaries.

Due to the sheltered locations of the MR schemes considered within their estuaries, wave energy is expected to be negligible and is not considered in this study. The effect of locally generated wind waves, even though they probably have an impact on the redistribution of sediment within the marsh, especially in the larger schemes, is also neglected for lack of data. This omission is justified for two reasons: 1) this study considers the total import/export of sediment in and out of the marshes based on overall changes in marsh elevation: the exact behaviour of the sediment within the site is of lesser relevance; 2) previous studies have found waves to have a significant impact on the erosion of marsh edges (Feagin et al. 2009; Tonelli et al. 2010), while the saltmarsh surface is resilient to waves generated during storm surges (Spencer et al. 2016): since MR schemes tend to be still partially sheltered by embankments, they have small to non-existent areas of exposed marsh edge, and can be considered resilient to the effect of waves.

3.2.3 Sediment characteristics

The sediment characteristics available for this study include the mean grain size in the wider estuary, measured in µm, as well as the water turbidity in the wider estuary, measured in ftu (Formazin Turbidity Units). Assuming that the mean grain size in the estuary is representative of what deposits within the marsh, this gives an approximation of the site mean grain size, and so of the cohesivity of the sediment. The water turbidity informs the sediment exchanges and accretion rates within the site, and is particularly relevant to MR schemes as they are expected to act as sediment sinks due to being partially sheltered by flood defences (Pontee 2014a). The two datasets were obtained from the Water Quality Archive (WIMS) Open Data Portal (http://environment.data.gov.uk/water-quality/index.html), then interpolated for each MR site using a weighted mean interpolation.

3.3 Reference sites

There are 13 natural saltmarshes considered for this study as reference sites, covering a wide range of locations across England and Wales and various environmental settings (Figure 3.4). The sites were first selected by Steel (1996), who followed several criteria to ensure they be representative of natural mature saltmarshes. In particular, the creek system should be unaffected by human activity and should not receive any terrestrial discharge. The history of the saltmarshes was also explored by Steel (1996) to verify their stability. As of 1996, their ages were estimated to range from 30 years, to more than 2000 years (Table 3.1). In an attempt to restrict the study to saltmarshes at morphological equilibrium, marshes displaying frontal erosion were generally avoided, though some erosion was observed at Grange, Hen Hafod and Newton Arlosh. At each site, a single creek network was identified by Steel (1996) and morphological parameters extracted from aerial photographs and field surveys. More details are provided below for each site. The information on marsh age and development is based on sequential aerial photography analysis performed by Steel (1996). The grain size information is taken from Steel (1996). Recent aerial photography is provided when available, otherwise the lidar data is used to visualise the layout of the sites.

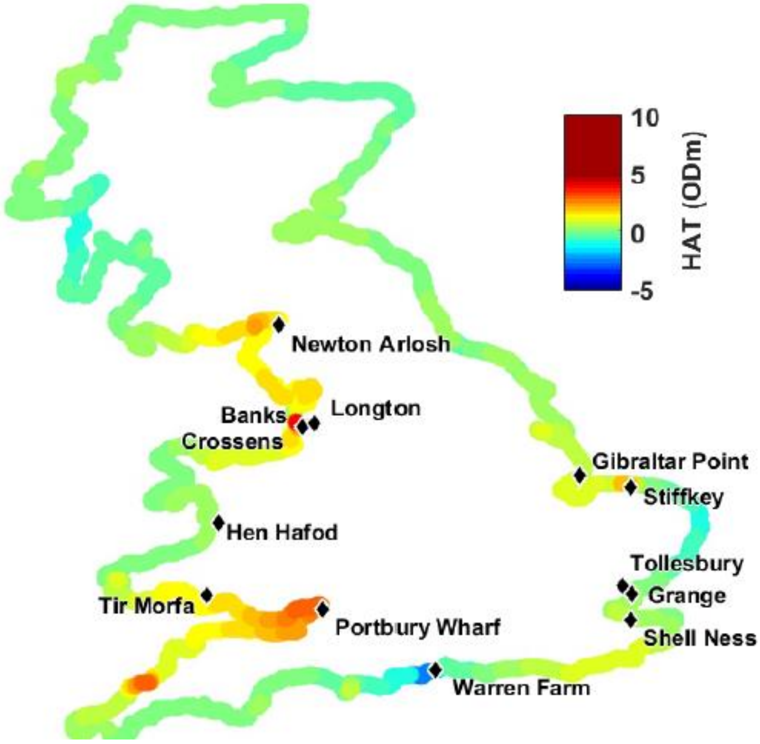


Figure 3.4: Location of the 13 mature natural saltmarshes selected for this study. Colorbar shows HAT values along the British coastline established through linear interpolation of the Admiralty Tide data (Admiralty Tide Table, 2014). Red lines show the catchment area contours of each creek system considered.

Table 3.1*: Location and physical characteristics of the 13 British natural saltmarshes considered in this study. The OS grid reference corresponds the entry channel outlet as defined in Section 2.4.1.*

Marsh name / Location	**O.S. grid reference**	**Age (updated from Steel, 1996)**	**Most recent suitable lidar flight**	**Height above MWS (m)**	**Tidal range (m)**	**Site size (km^2)**
Warren Farm / Beaulieu	SZ4225097275	~110	2014	2.27	1.98	0.006
Tollesbury / Blackwater	TL9609112003	>500	2008	1.40	4.32	0.081
Grange / Dengie	TM0384802264	~110	2014	1.81	3.89	0.169
Hen Hafod / Dyfi	SN6488694902	~70	2015	1.24	4.30	0.040
Tir Morfa / Loughor	SS5308697743	~65	2010	3.45	8.00	0.069
Stiffkey / North Norfolk	TF9751844481	>2000	2014	1.76	6.21	0.073
Banks / Ribble	SD3693822645	~120	2014	3.62	9.55	0.260
Crossens / Ribble	SD3502421112	~50	2014	3.61	9.95	0.243
Longton / Ribble	SD4465526146	>220	2014	5.18	7.74	0.227
Portbury Wharf / Severn	ST4862577476	~70	2009	5.79	11.97	0.016
Newton Arlosh / Solway Firth	NY1936156388	>170	2013	4.03	8.57	0.067
Shell Ness / Swale	TR0461167413	~70	2014	2.47	5.01	0.045
Gibraltar Point / Wash	TF5581657517	~50	2014	3.11	6.11	0.088

3.3.1 Warren Farm / Beaulieu

Warren Farm is located on the western side of the Beaulieu estuary mouth, in an estuarine back-barrier setting. The environment is microtidal with an interpolated tidal range of 2.0 m. The saltmarsh started developing around 1907. The area of interest is 0.6 ha in size and is intersected by a dense creek network, which has undergone little morphological change since 1948 (Figure 3.5). The dominant sediment types on the creek banks are clay and silt, with about 46 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was an intertidal mudflat.

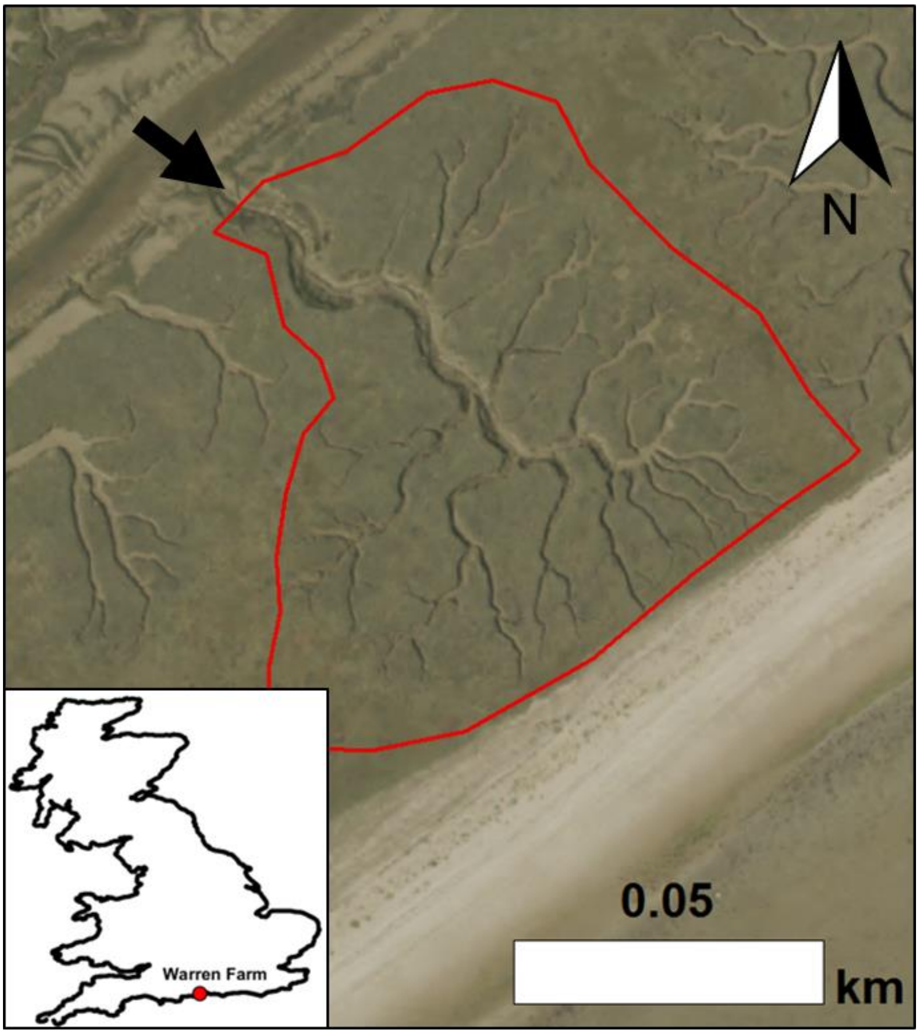


Figure 3.5: Aerial photograph of Warren Farm (20 cm resolution date of flight 21/07/2013, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.2 Tollesbury / Blackwater

Tollesbury is located close to the mouth of the Blackwater estuary. The environment is macrotidal with an interpolated tidal range of 4.3 m. The saltmarsh extent has been stable since 1838 and is thought to be over 500 years old. The area of interest is 8.1 ha in size and is intersected by a dense and sinuous creek network, with large basins at the head of several creeks (Figure 3.6). The dominant sediment types on the creek banks are clay and silt. The sediment is cohesive, with about 65 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was an intertidal mudflat.

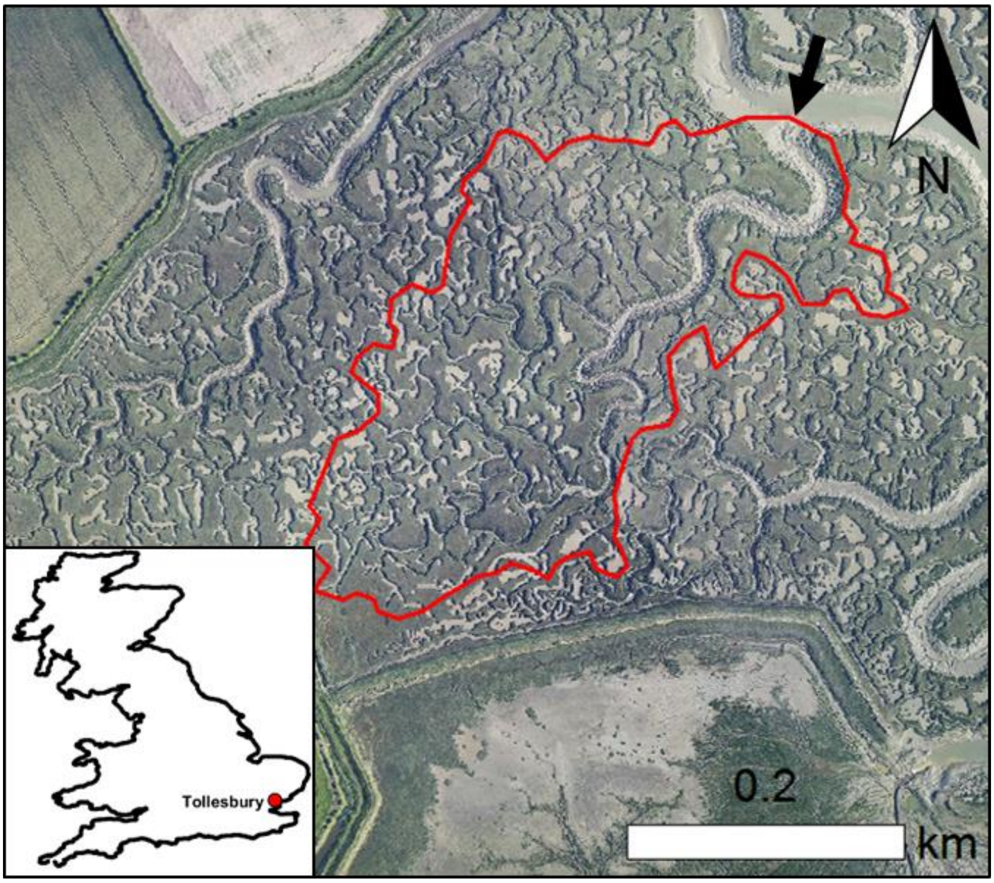


Figure 3.6: Aerial photography of Tollesbury (20 cm resolution composite image from three flights on the 15/07/2011, 15/08/2011 and 14/09/2011, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.3 Grange / Dengie

Grange is located midway along the Dengie Peninsula. The environment is macrotidal with an interpolated tidal range of 3.9 m. The saltmarsh started developing around 1925. The area of interest is 16.9 ha in size and is intersected by a dense creek network of low sinuosity (Figure 3.7). The dominant sediment types on the creek banks are clay and silt, with about 30% of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was a sandflat.

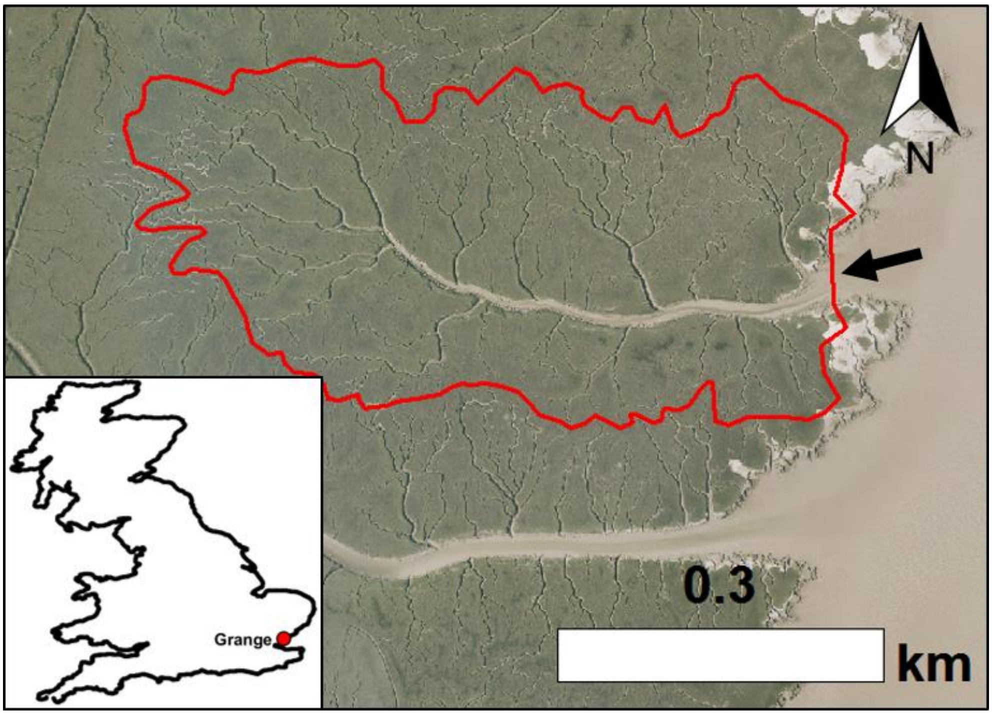


Figure 3.7: Aerial photography of Grange (20 cm resolution composite image from two flights on the 25/06/2013 and 15/07/2013, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.4 Hen Hafod/Dyfi

Hen Hafod is located on the southern side of the Dyfi Peninsula, on the west coast of Wales. The environment is macrotidal with an interpolated tidal range of 4.3 m. The saltmarsh started developing around 1900 and has undergone little visible change since 1977. The area of interest is 4.0 ha in size and is intersected by a creek network of fairly low density (Figure 3.8). The dominant sediment types on the creek banks are silt and sand, with about 25% of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was a sandflat.

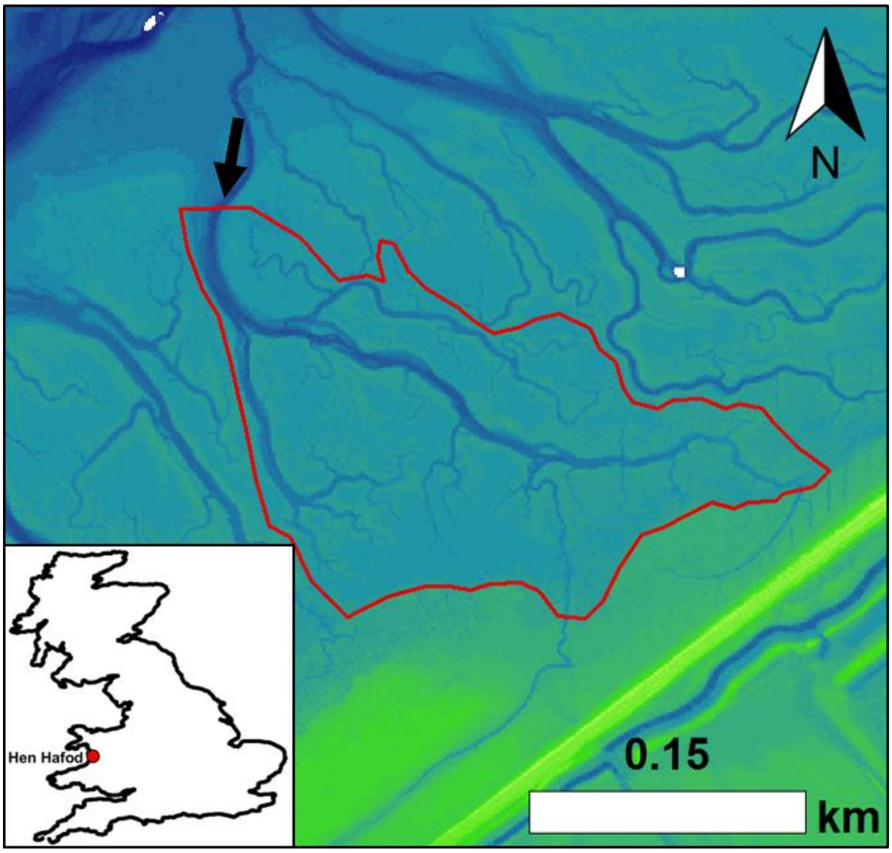


Figure 3.8: lidar map of Hen Hafod (horizontal resolution 1 m, vertical resolution 0.15 m, date of flight 07/01/2015, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.5 Tir Morfa / Loughor

Tir Morfa is an estuarine fringing marsh located on the northern side of the Loughor estuary. The environment is hypertidal with an interpolated tidal range of 8.0 m. The saltmarsh started developing around 1948 and has undergone little visible change since 1969. The area of interest is 6.9 ha in size and is intersected by a dense creek network (Figure 3.9). The dominant sediment types on the creek banks are silt and sand, with about 26 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was a sandflat.

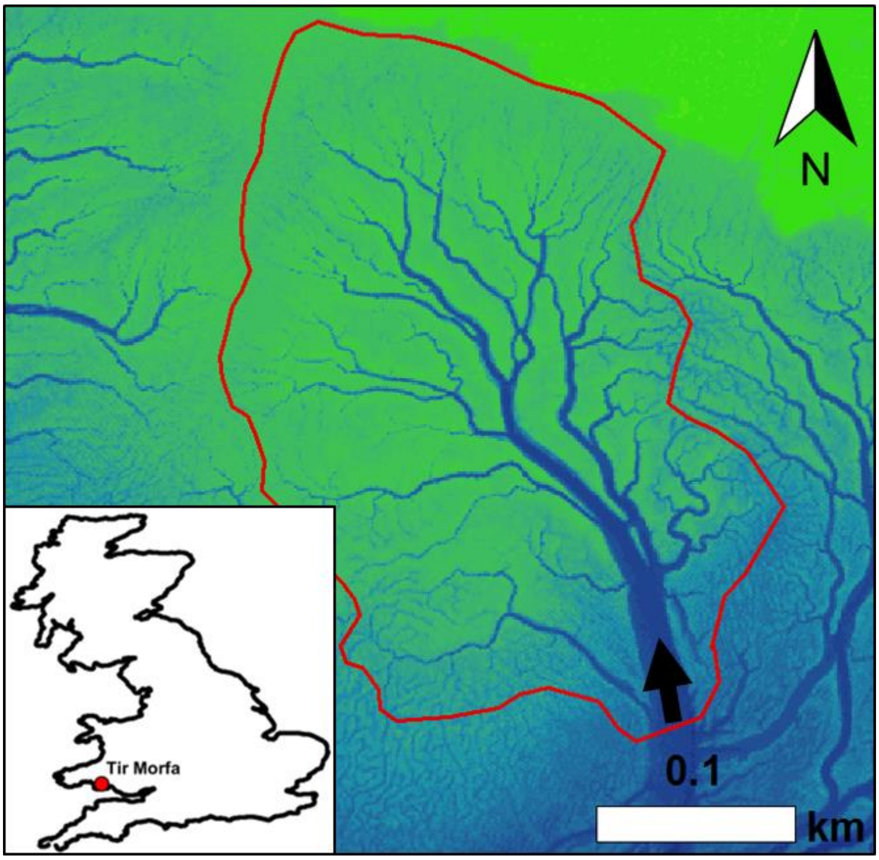


Figure 3.9: lidar map of Tir Morfa (horizontal resolution 1 m, vertical resolution 0.15 m, date of flight 07/01/2015, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.6 Stiffkey / North Norfolk

Stiffkey is an open-coast marsh located in north Norfolk. The environment is hypertidal with an interpolated tidal range of 6.2 m. The saltmarsh started developing around the early-mid Holocene, over 2000 years ago, and the creek network has undergone little visible change over the last 170 years. The area of interest is 7.3 ha in size and is intersected by a dense creek network (Figure 3.10). The dominant sediment types on the creek banks are silt and clay, with about 21 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was a sandflat.

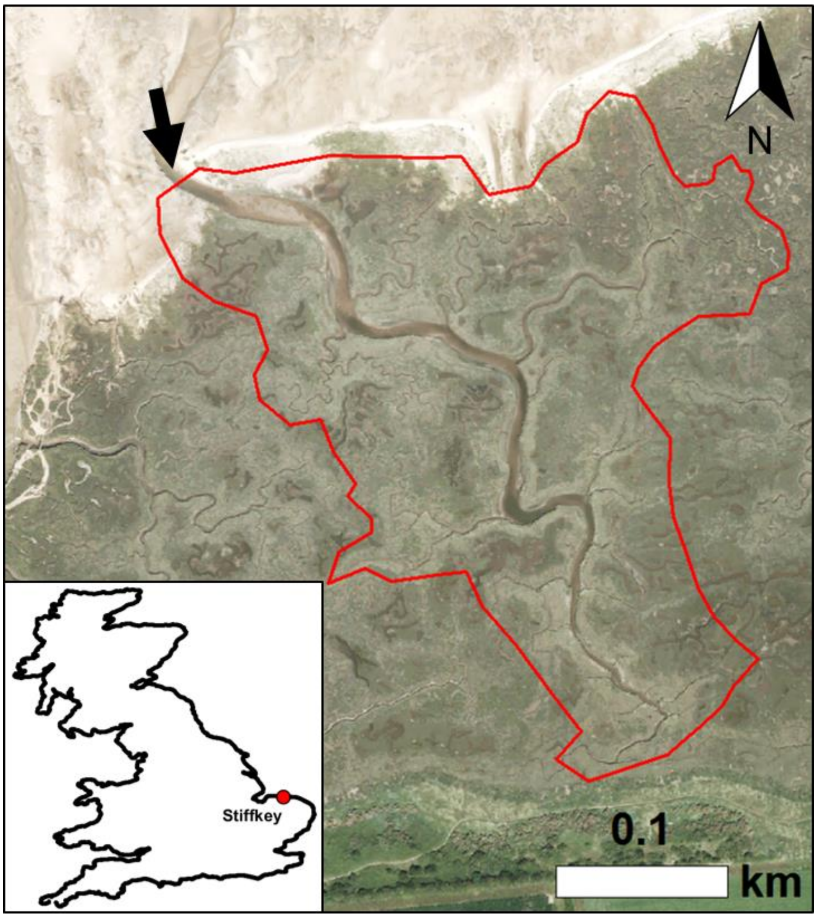


Figure 3.10: Aerial photography of Stiffkey (20 cm resolution composite image from two flights on the 12/06/2014 and 31/10/2014, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.7 Banks / Ribble

Banks is an estuarine fringing marsh located on the southern shore of the Ribble estuary. The environment is hypertidal with an interpolated tidal range of 9.5 m. The saltmarsh started developing around 1895 and has undergone little visible change since 1979. The area of interest is 26.0 ha in size and is intersected by a creek network of low sinuosity (Figure 3.11). The dominant sediment types on the creek banks are silt and sand, with about 29 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was a sandflat.

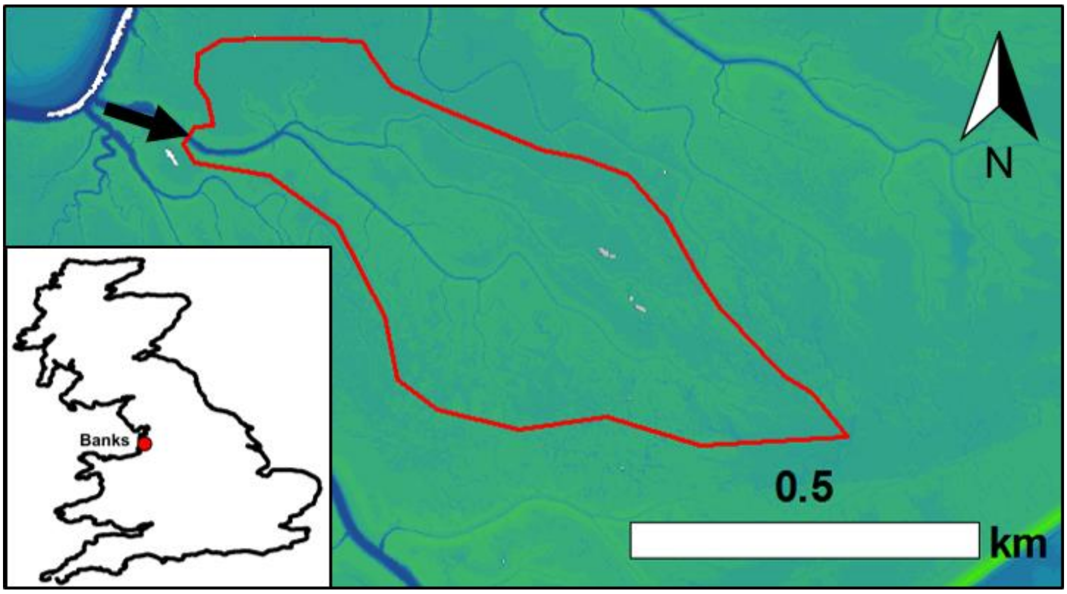


Figure 3.11: lidar map of Banks (horizontal resolution 1 m, vertical resolution 0.15 m, date of flight 18/12/2014, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.8 Crossens / Ribble

Crossens is an estuarine fringing marsh located on the southern shore of the Ribble estuary. The environment is hypertidal with an interpolated tidal range of 10.0 m. The saltmarsh started developing after 1974 and the creek network has undergone little visible change since 1979. The area of interest is 24.3 ha in size and is intersected by a poorly distributed creek network (Figure 3.12). The dominant sediment types on the creek banks are silt and sand, with about 15 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was characterised by a coarse-grained sandflat.

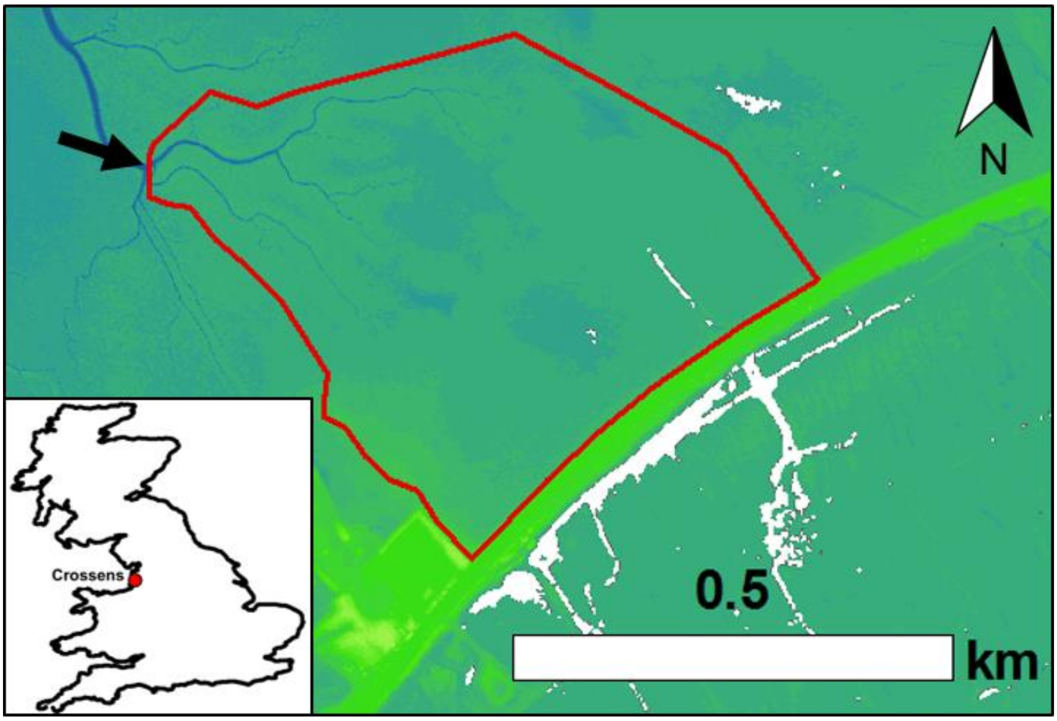


Figure 3.12: lidar map of Crossens (horizontal resolution 1 m, vertical resolution 0.15 m, date of flight 18/12/2014, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.9 Longton / Ribble

Longton is an estuarine fringing marsh located on the southern shore of the Ribble estuary. The environment is hypertidal with an interpolated tidal range of 7.7 m. The saltmarsh started developing around 1912 and has undergone little visible change since 1988. The area of interest is 22.7 ha in size and is intersected by a dense creek network (Figure 3.13). The dominant sediment types on the creek banks are silt and sand, with about 25 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was a sandflat.

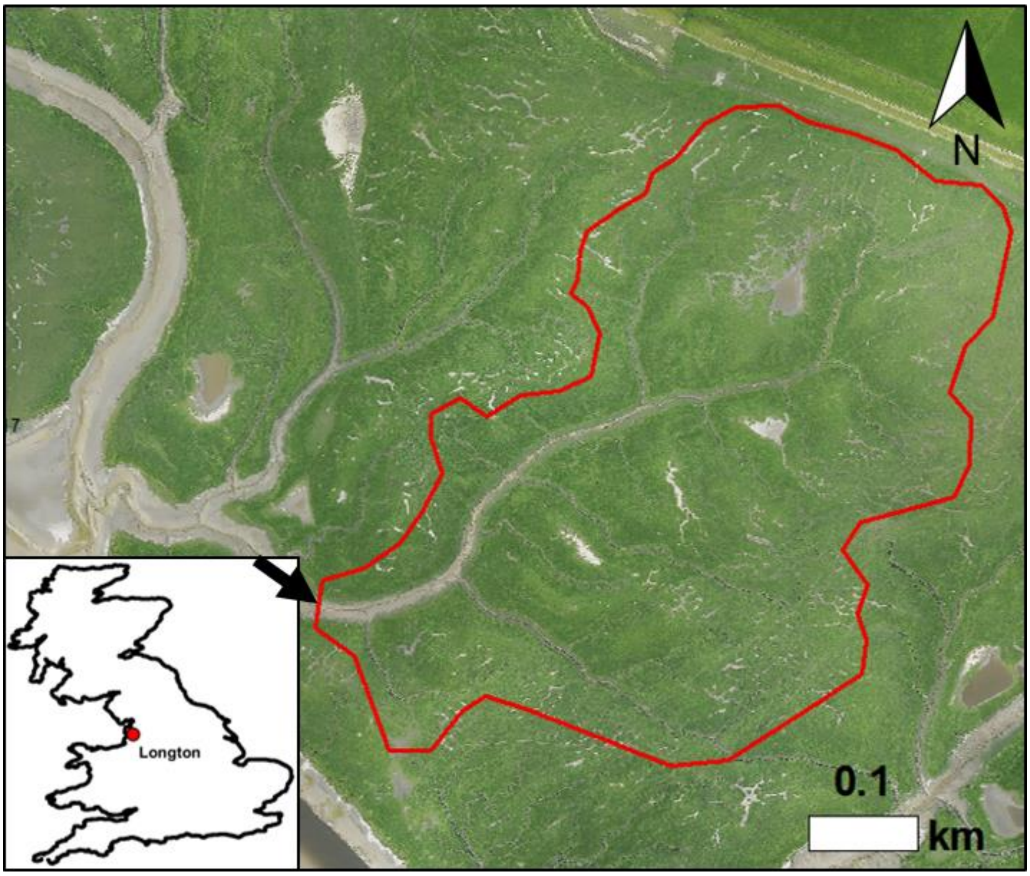


Figure 3.13: Aerial photography of Longton (10 cm resolution, date of flight 26/05/2007, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.10 Portbury Wharf / Severn

Portbury Wharf is an estuarine fringing marsh located in the Severn estuary. The environment is hypertidal with an interpolated tidal range of 12.0 m. The saltmarsh started developing around 1948 and has undergone little visible change since 1969. The area of interest is 1.6 ha in size and is intersected by a dense creek network (Figure 3.14). The dominant sediment types on the creek banks are silt and clay, with about 25 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was a mudflat.

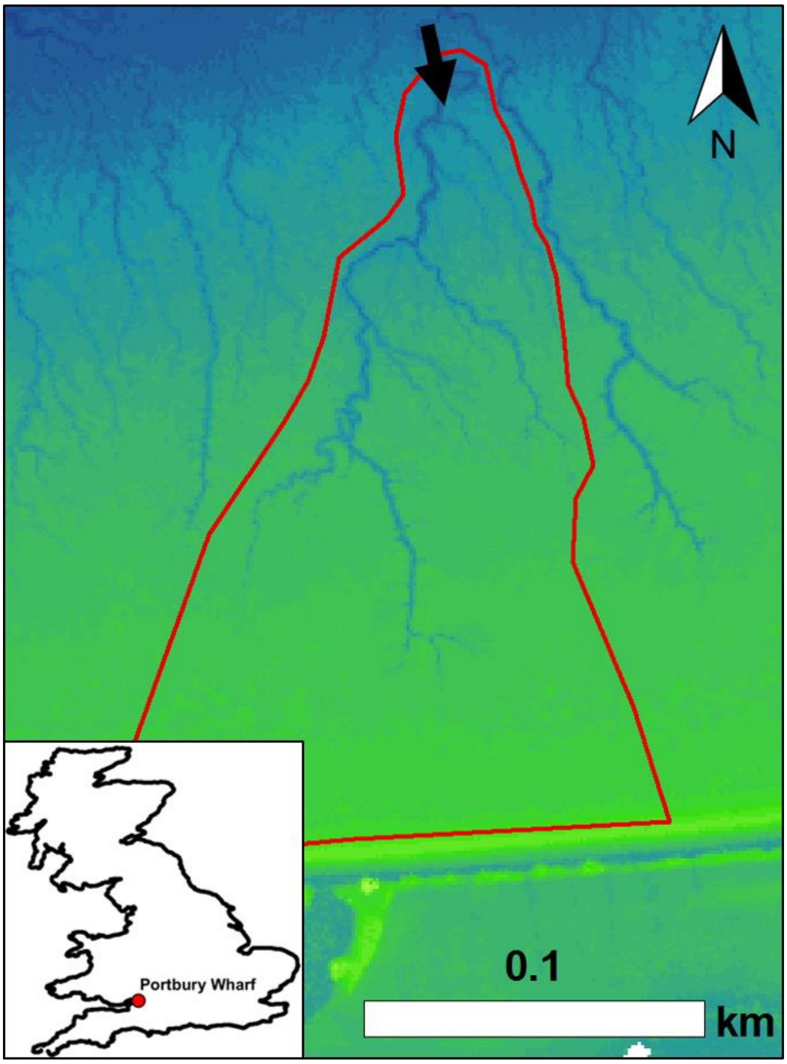


Figure 3.14: lidar map of Portbury Wharf (horizontal resolution 1 m, vertical resolution 0.15 m, date of flight 11/03/2009, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.11 Newton Arlosh / Solway Firth

Newton Arlosh is an embayment marsh located within Moricambe Bay, on the English side of the Solway. The environment is hypertidal with an interpolated tidal range of 8.6 m. The saltmarsh started developing around 1866. The area of interest is 6.7 ha in size and is intersected by a dense creek network (Figure 3.15). The dominant sediment types on the creek banks are silt and sand, with about 3 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was a tidal flat dominated by silt and sand.

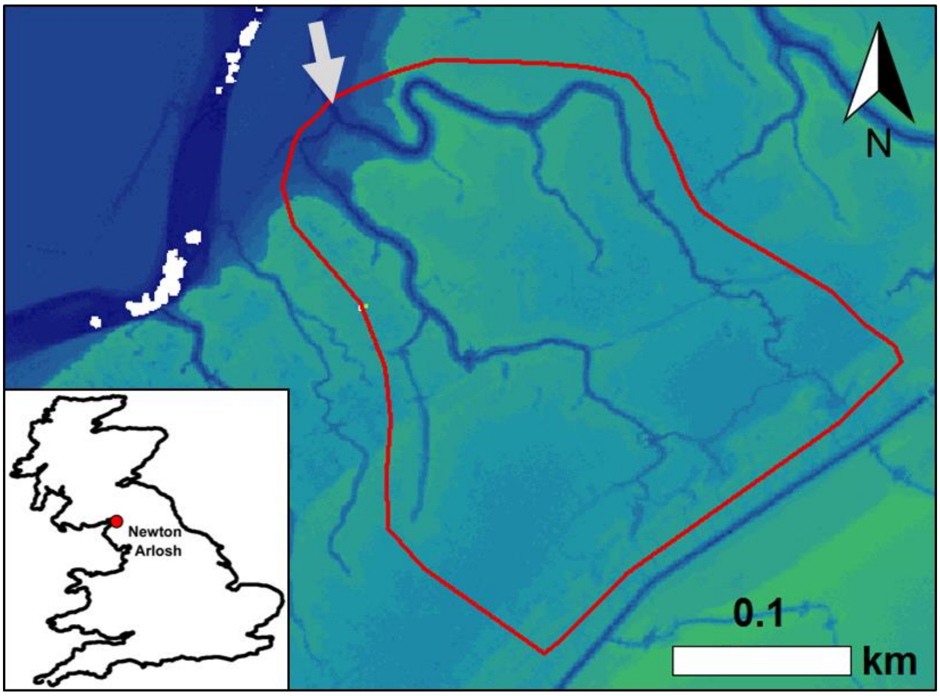


Figure 3.15: lidar map of Newton Arlosh (horizontal resolution 1 m, vertical resolution 0.15 m, date of flight 11/03/2009, Environment Agency). Outlet shown by grey arrow. MR extent considered shown by red line.

3.3.12 Shell Ness / Swale

Shell Ness is an estuarine back-barrier marsh located at the mouth of the Swale estuary. The environment is macrotidal with an interpolated tidal range of 5.0 m. The saltmarsh started developing around 1920. The area of interest is 4.5 ha in size and is intersected by a dense creek network (Figure 3.16). The dominant sediment types on the creek banks are silt and clay, with about 59 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was a mudflat.

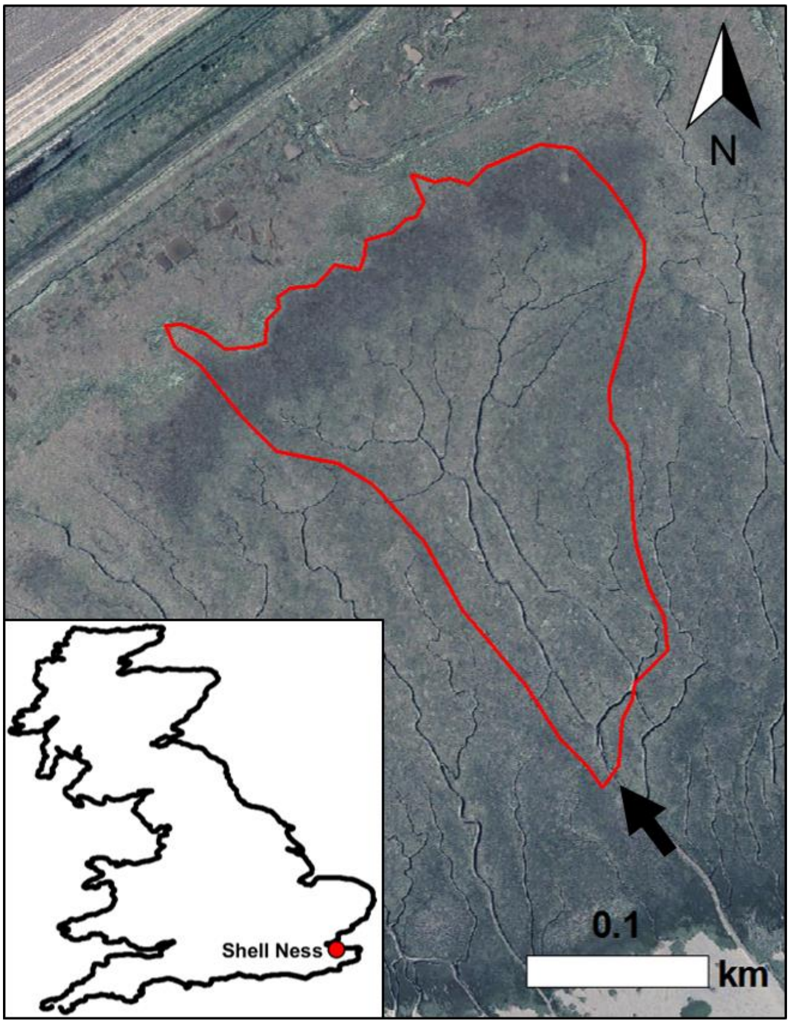


Figure 3.16: Aerial photography of Shell Ness (20 cm resolution, date of flight 13/08/2017, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.3.13 Gibraltar Point / Wash

Gibraltar Point is a back-barrier marsh located on the north-east side of the Wash. The environment is hypertidal with an interpolated tidal range of 6.1 m. The saltmarsh started developing between 1946 and 1960. The area of interest is 8.8 ha in size and is intersected by a dense creek network (Figure 3.17). The dominant sediment types on the creek banks are silt and clay, with about 20 % of the sediment fraction under 20 µm (Steel, 1996). Before colonisation by saltmarsh plants, the area was characterised by a coarse-grained sandflat.

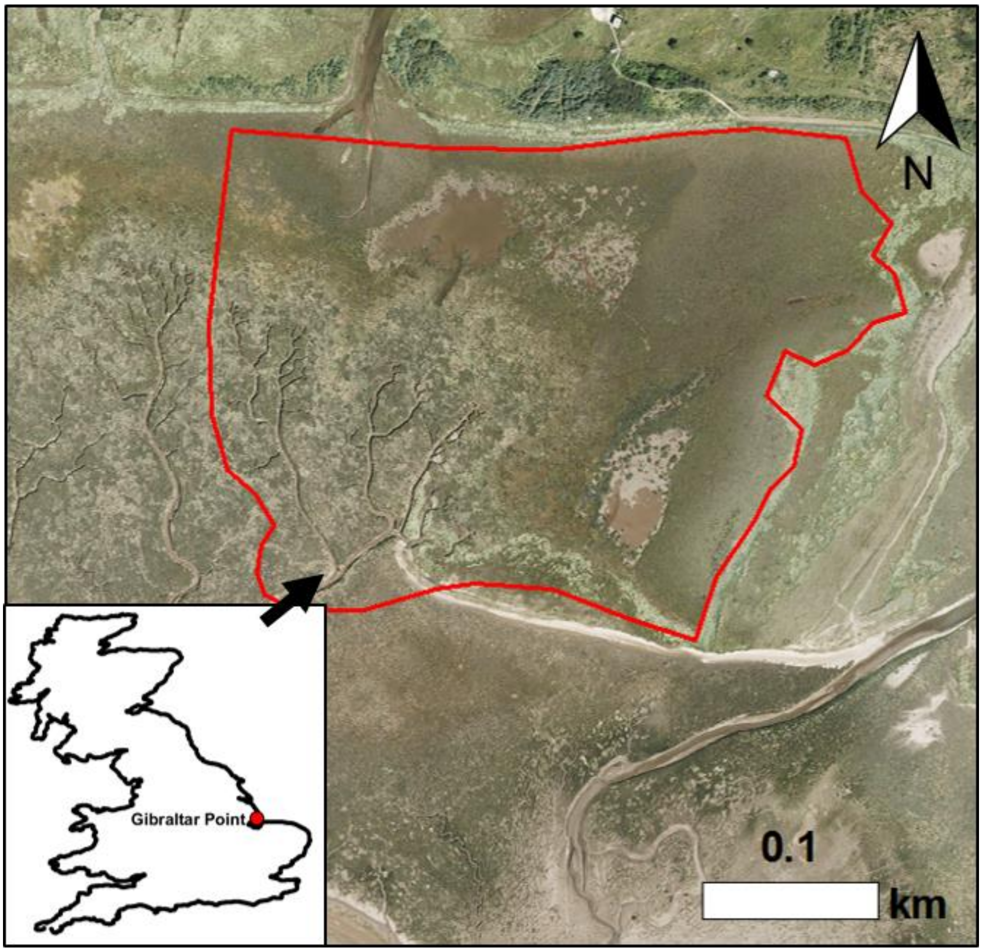


Figure 3.17: Aerial photography of Gibraltar Point (20 cm resolution composite image from two flights on the 12/06/2014 and 31/10/2014, Environment Agency). Outlet shown by black arrow. MR extent considered shown by red line.

3.4 MR sites

This study uses 10 MR schemes, implemented between 1995 and 2014 around the coast of England (Figure 3.18). Sites were selected based on four criteria. The first selection criterion was data availability (number of lidar datasets and case studies available). The second criterion was the sites' settings, which should capture a range of initial external conditions (i.e. tidal range, size, implementation date, position within the estuary, suspended sediment concentration entering the site, land use history and sediment properties). The third criterion was the general scheme design (i.e. number and size of breaches, initial site elevation within the tidal frame, targeted habitats). Finally, the fourth criterion was the creek network design: absence of initial creeks (strategy 1), excavation of a creek system from a natural template (strategy 2), or excavation of artificial creeks in the absence of a natural template (strategy 3). A complete list of the initial conditions, design choices and available datasets for each MR scheme is compiled in Appendix C1 and summarised in Table 3.2. Further details on the initial conditions, design choices and initial goals in terms of habitats for the 10 MR schemes were provided previously in Section 2.4.3.

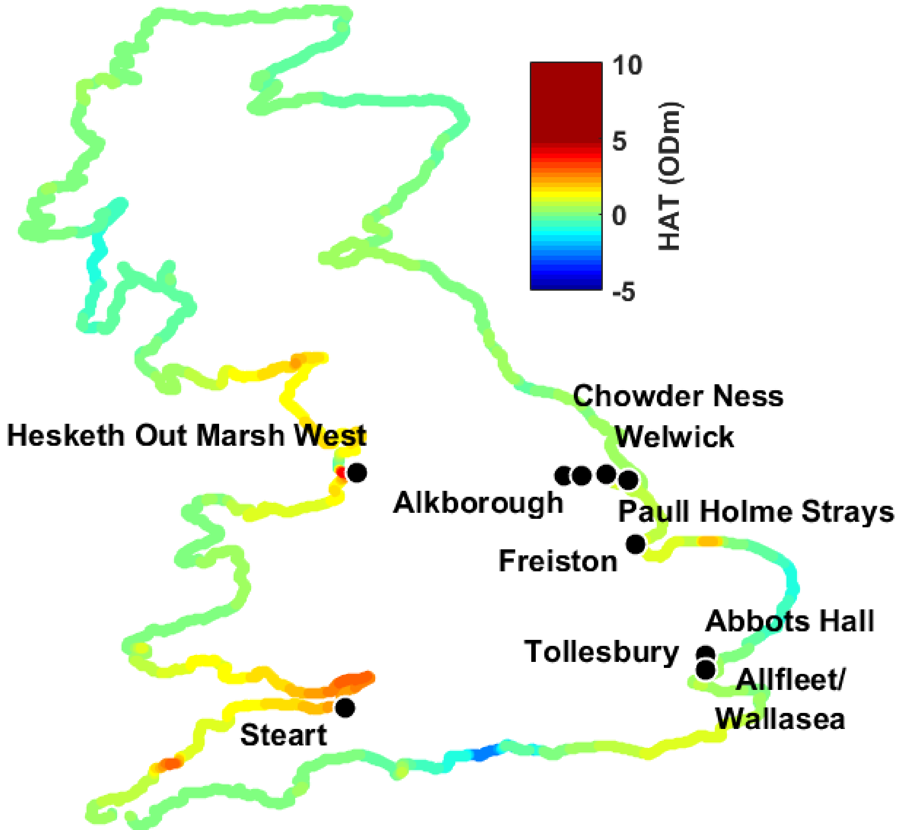


Figure 3.18: Location of the 10 MR schemes selected for this chapter. The colour shows the highest astronomical tide along the British coastline established through linear interpolation of the Admiralty Tide data (Admiralty Tide Table, 2014)

Table 3.2: Design details for all MR schemes considered (Source: ABPmer 2014). Creek design strategy refers to the 3 design strategies identified in Section 2.4.3:

	Estuary	Site opening date	Lidar datasets suitable for use	Height above MWS (m)	Tidal range (m)	Site size (km^2)	Time embanked (years)	Creek design (See 2.4.3)
Abbots Hall	**Blackwater**	2002	2002; 2008; 2013; 2014; 2015	1.39	4.1	0.25	>200	3
Alkborough	**Humber**	2006	2007;2010; 2012; 2015	2.33	5.8	3.61	>56	3
Allfleet	**Crouch**	2006	2007; 2011; 2013; 2015	0.96	4.7	0.97	>400	3
Chowder Ness	**Humber**	2006	2007; 2009; 2010; 2011; 2012; 2013; 2015; 2016	2.33	6.6	0.1	97	1
Freiston	**The Wash**	2002	2002; 2009; 2011; 2013; 2014	2.55	6.7	0.7	19	3
HOMW	**Ribble**	2008	2009; 2010; 2011; 2014	4.21	8.6	1.59	28	2
Paull Holme Strays	**Humber**	2003	2007; 2010; 2012; 2013; 2014	2.86	6.5	0.8	50	3
Steart	**Parrett River**	2014	2014; 2015; 2016	5.51	11	2.1	100	3
Tollesbury	**Blackwater**	1995	2002; 2009; 2012; 2013; 2014; 2015; 2016	0.86	4.3	0.19	150	3
Welwick	**Humber**	2006	2007; 2009; 2010; 2011; 2012; 2013; 2014	2.43	5.8	0.59	35	1

There are 4 to 8 individual lidar datasets collected for each site, ranging from less than 2 years pre-breach until 5 to 20 years post-breach (Table 3.2). The exception is Steart, which was breached in 2014 and has only 3 lidar datasets available. Also, at Paull Holme Strays and Tollesbury no lidar dataset of adequate resolution could be found within the first 2 years post-breach, thus data availability is skewed towards the later years of scheme evolution. Lidar data availability tends to decrease going back in time, which complicates the study of the early evolution of older sites. Furthermore, changes in lidar technology means that older datasets may be noisier and less reliable. In general, lidar datasets of 2 m horizontal resolution were omitted due to increased noise and reduced reliability, however, an exception was made for Allfleet in 2007 in order to get some idea of the early stage of this scheme, and because the data was less noisy than the 2002 Freiston dataset and so considered more reliable.

Mean tidal levels and tidal asymmetry values for each scheme were obtained by finding the weighted mean of the surrounding (up to 30 km away) main and secondary ports values, as explained in Section 3.2.2. The tidal ranges of the sites considered range from 4.3 m to 11.0 m (Table 3.2). A description of 10 MR examples and their creek design strategies is given in the next subsections in order of implementation year, based on information found on online databases, case study reports and insight from consultants. Sites were selected based on the amount of data and reports available, and on the variety of design strategies applied to be representative of the design trends in the UK. These schemes will be studied further in Chapters 6 and 7 of this thesis.

3.4.1 Tollesbury

Tollesbury, breached in August 1995, is situated in the Blackwater estuary. Prior to realignment, it had been embanked for over 150 years and used as arable farmland (Garbutt et al. 2006). Tollesbury is an experimental MR scheme commissioned by Defra's Flood and Coastal Defence research programme, and builds on previous work on the terrestrial ecology of the site, making it an example of opportunistic MR planning (Reading et al. 2008). It is one of the pioneer MR schemes in the UK, and as such does not have targeted extents of recreated mudflat and saltmarsh habitats. It is connected to a larger system of mature saltmarshes via a single 60 m wide breach (red in Figure 3.19). Aside from one channel cut from a pre-existing drainage ditch, no site profiling was undertaken, corresponding to creek design strategy 3.

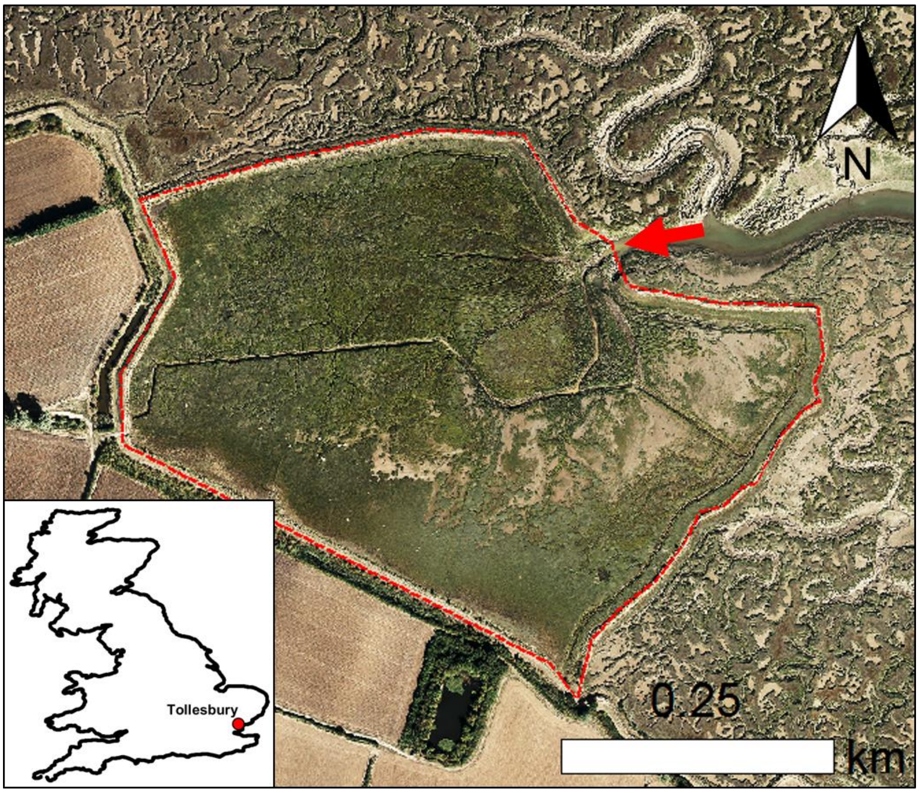


Figure 3.19: Aerial photograph of Tollesbury (20 cm resolution composite image from two flights on the 06/05/2016 and the 23/09/2016, Environment Agency). Breach area shown by red arrow. MR extent considered shown by dashed red line.

3.4.2 Abbots Hall

Abbots Hall, implemented in October 2002, is situated in Salcott creek in the Blackwater estuary. The site was embanked for over 200 years before realignment, and was used as grazing land, then as arable land between 1943 and 1970 (Brooks et al. 2015). The scheme is an example of strategic MR planning, implemented to create new intertidal habitat and provide natural flood protection (ABP mer 2011a). The scheme was opened to tidal inundation via 5 breaches, but the area of focus for this study is the portion of the scheme linked to the largest breach (red in Figure 3.20). The scheme is situated on a steeper slope than most MR sites, which lessened the necessity to build new flood defences: only 400 m of new embankment have been implemented for 49 ha of intertidal habitat. Artificial channels of 3–4 m width were dug, as well as lagoons that remain flooded at low tide to provide habitats for fish (Scott et al. 2011). The site is also part of a wider farmland (ABP mer 2011a). This artificial-looking creek system, designed to provide additional ecosystem benefits (fish use), corresponds to strategy 3.

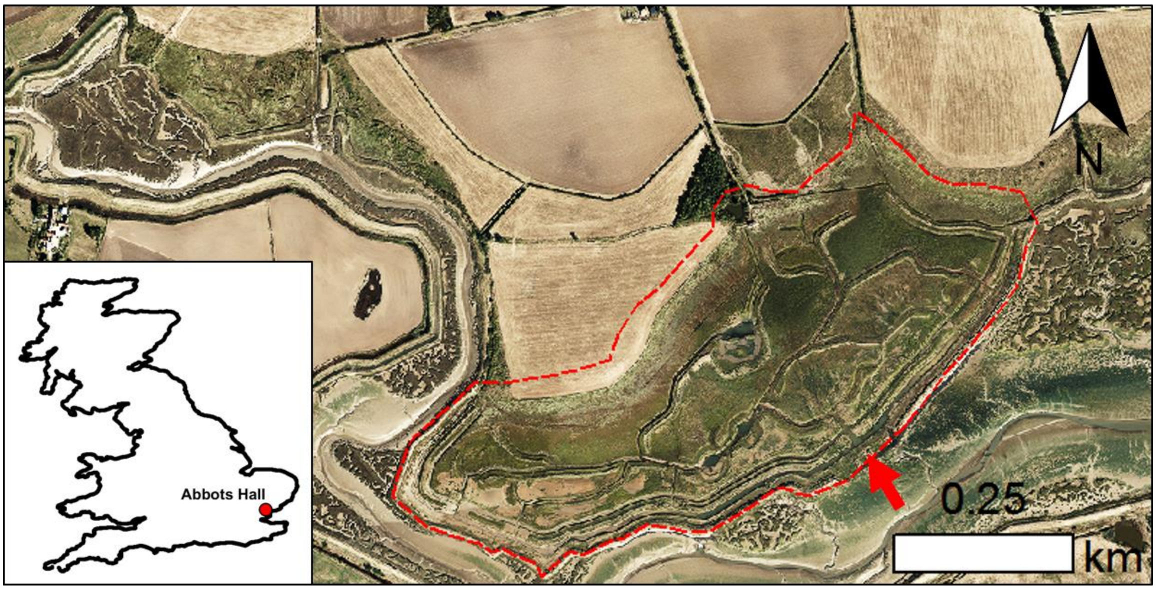


Figure 3.20: Aerial photograph of Abbots Hall (20 cm resolution composite image from two flights on the 06/05/2016 and the 23/09/2016, Environment Agency). Largest breach area shown by red arrow. MR extent considered shown by dashed black line.

3.4.3 Freiston

Freiston, opened in 2002, is situated in the Wash. The site was previously embanked for 19 years and used as arable land by HM Prison Service (ABP mer 2011c). The site was then acquired by the Royal Society for the Protection of Birds (RSPB) after the stretch of seawall protecting it was found too expensive to maintain (Nottage et al. 2005): Freiston is therefore driven by opportunistic MR planning, and aims to recreate saltmarsh habitat (ABP mer 2011c; Friess et al. 2014). The project includes three 50 m- wide breaches associated with 2 m wide, 1 m deep excavated channels (red in Figure 3.21). Site preparation included the grazing of the site, infill of some of the field drains, and the excavation of 1,200 m of artificial creek system (two channels leading from each breach), corresponding to design strategy 3 (Nottage et al. 2005). Outside the site, 50 m of the external primary creek network was deepened to facilitate drainage (Nottage et al. 2005).

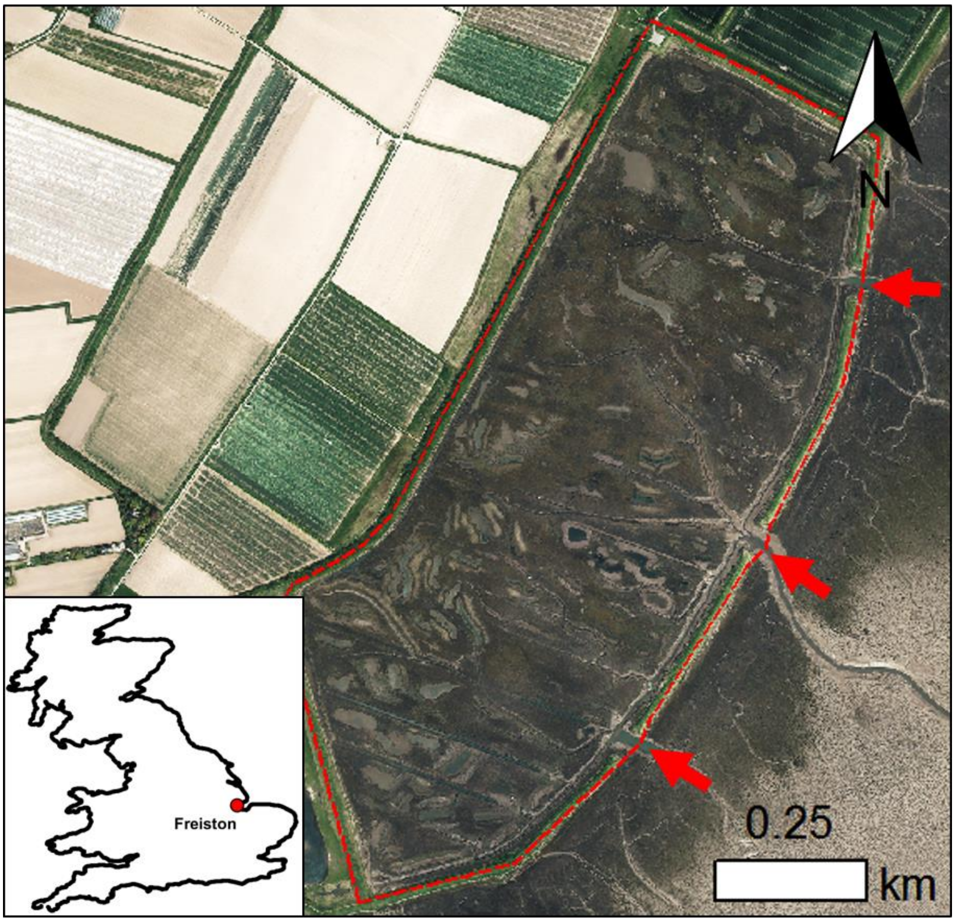


Figure 3.21: Aerial photography of Freiston (20 cm resolution composite image from two flights on the 06/05/2016 and the 23/09/2016, Environment Agency). Breaches shown by red arrows. MR extent considered shown by dashed red line.

3.4.4 Paull Holme Strays

Paull Holme Strays, opened in September 2003, is situated within the Humber estuary. It had been reclaimed in the 1950s for agricultural purposes (Edwards et al. 2006). The scheme is an example of opportunistic MR planning: realignment was chosen as a cheaper and more sustainable alternative to raising the previous flood defence, which was considered insufficient to protect nearby roads and gas infrastructure (Edwards et al. 2006). The main objectives were flood protection, but also the compensation of habitats loss within the same estuary during floodwall stabilisation works (Edwards et al. 2006). The site was expected to provide 28 ha of middle to upper saltmarsh, 15 ha of lower saltmarsh and 32 ha of mudflats (Edwards et al. 2006). Paull Holme Strays is linked to the Humber estuary by two breaches, which are 150 m and 50 m wide (red in Figure 3.22). New creeks were excavated by deepening previous field drains, corresponding to design strategy 3.

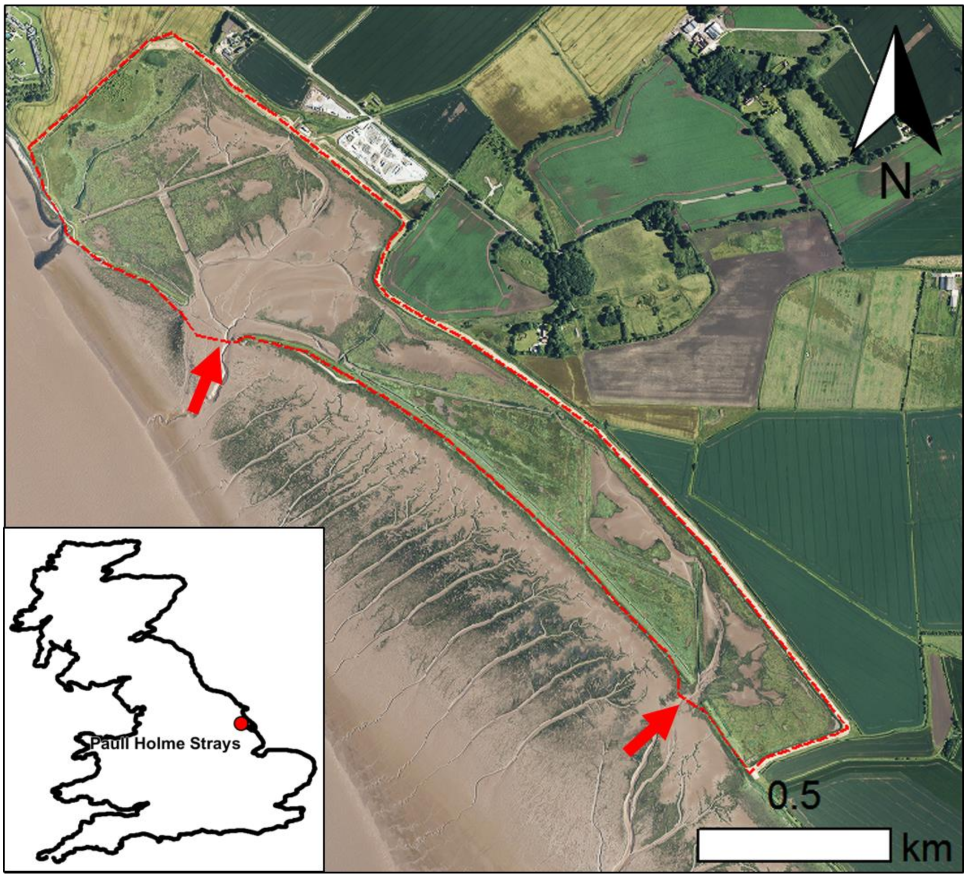


Figure 3.22: Aerial photograph of Paull Holme Strays (20 cm resolution composite image from two flights on the 07/05/2016 and the 23/08/2016, Environment Agency). Breaches shown by red arrows. MR extent considered shown by dashed red line.

3.4.5 Alkborough

Alkborough, breached in September 2006 is the largest MR scheme on the Humber estuary (Manson et al. 2012). Before realignment, this site was embanked in phases over many years until the 1950s. Alkborough is an example of strategic MR planning: its main objective is to provide flood storage space to reduce tidal levels in the upper estuary (Manson et al. 2012), in order to decrease the flood risk in Hull (Medlock et al. 2013). The project aimed to create 170 ha mudflat and saltmarsh, with a further 230 ha serving as water storage for extreme surge events (Manson et al. 2012). Due to the constraint of maintaining navigability in the Humber Estuary, this was achieved by lowering the flood defences over 1.5 km along the scheme to allow water to overtop during higher tidal events, and by adding a single 20 m wide armoured breach, with a sill set above MHWN at 2.8 mODN (Manson et al. 2012, red in Figure 3.23). No leveling of the site was undertaken, leaving the land levels at 2–4 mODN. Drainage is provided by a single straight distribution channel connected to the breach (Pontee 2015a): the simplistic design fits with design strategy 3.

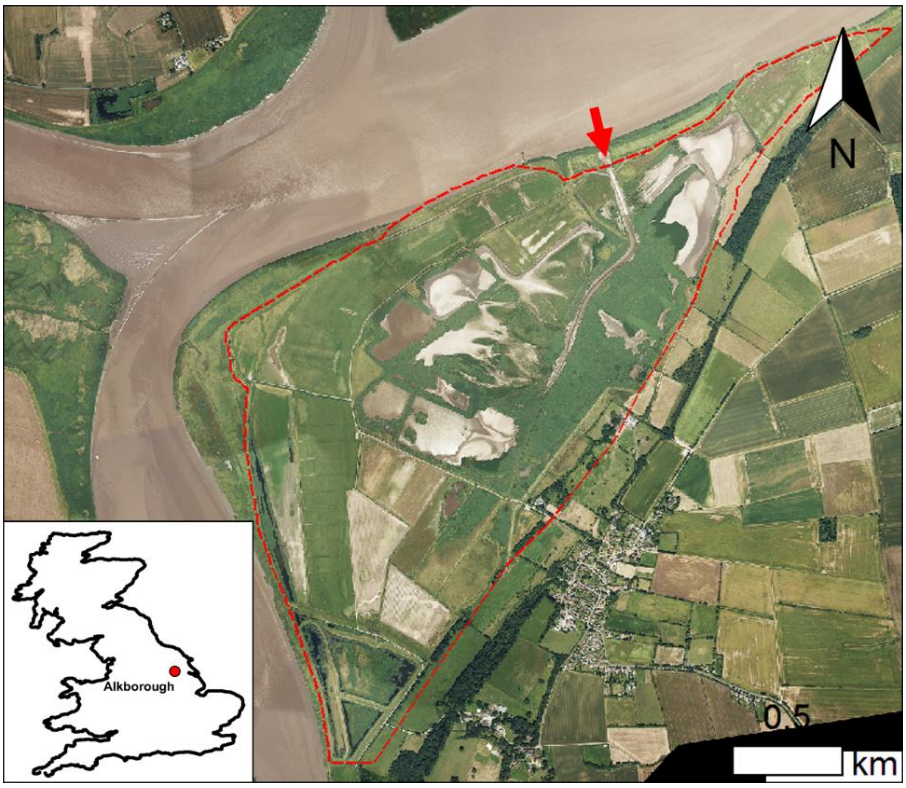


Figure 3.23: Aerial photograph of Alkborough (20 cm resolution composite image from two flights on the 07/05/2016 and the 23/08/2016, Environment Agency). Breach area shown by red arrow. MR extent considered shown by dashed red line.

3.4.6 Allfleet/Wallasea

Allfleet/Wallasea, breached in 2006, is situated in the Crouch estuary and is the oldest historically embanked scheme among the selected study sites, having been embanked and used as arable land for over 400 years. The scheme is driven by legislative considerations: it aims to compensate for habitat loss due to port development in the Medway and Fagbury Estuary (ABP mer 2010), by producing 60 ha of mudflat, 20 ha of saline lagoons and 25 ha of saltmarsh (Pendle et al. 2013). Allfleet is a step in a larger project to turn the entire island into coastal habitats (Medlock et al. 2013). The site is linked to the estuary by 6 breaches between 60 and 210 m wide. Only the 5 breaches connected to the two largest sections of the MR scheme are considered (red in Figure 3.24), since the section connected to the 6th breach is small and disconnected from the rest of the scheme. The breaches were made over-wide and sufficiently deep to ensure stability, with particular attention paid to creating a ‘regime’ shape with the site hypsometry (Pendle et al. 2013). Site levelling included the raising of a 45m-wide strip seaward of the new seawall to provide potential saltmarsh habitats, and the creation of 7 island features within the site; the rest of the scheme was left at an elevation halfway between MLW and MHWN to provide mudflat habitats (ABP mer 2010). Drainage on site is provided by field drains and borrow dykes left in place, and by the excavation of one additional channel, corresponding to design strategy 3.

Figure 3.24: Aerial photograph of Allfleet/Wallasea (20 cm resolution composite image from two flights on the 06/05/2016 and the 23/09/2016, Environment Agency). Breaches shown by red arrows. MR extent considered shown by dashed red line.

3.4.7 Chowder Ness.

Chowder Ness, breached in July 2006, is situated in the Humber estuary. The site was previously embanked for 97 years and used as arable land. It is the smallest of the MR schemes considered, and aims to recreate 0.8 ha of saltmarsh and 10.5 ha of mudflat, as a compensation for port development, making it part of the legislative MR planning category. The site was lowered to increase the proportion of land below MHWN (ABP mer 2011b). Also, in order to leave the site open to tidal and wave energy and encourage mudflat development, 570 m of the old flood defence was removed (Figure 3.25), leaving only 200 m of the old line of defence. No initial creeks were excavated, corresponding to design strategy 1.

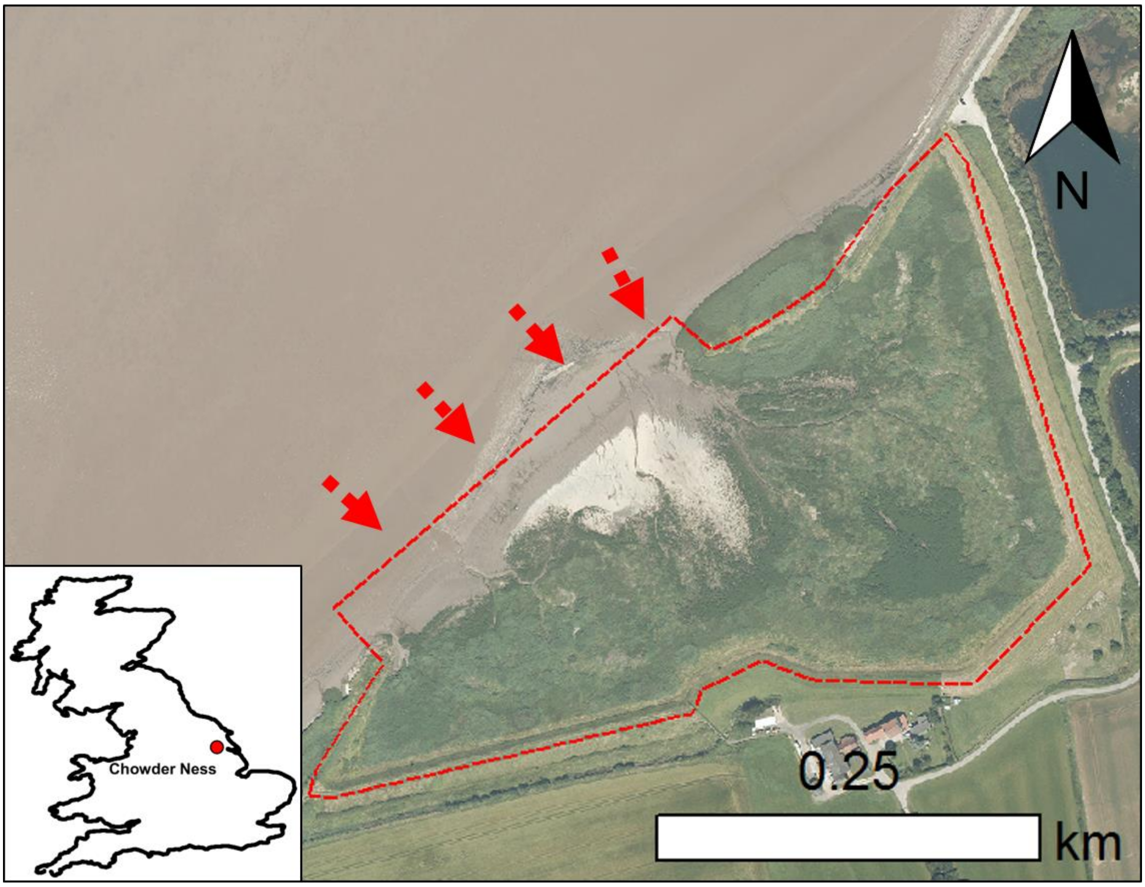


Figure 3.25: Aerial photograph of Chowder Ness (20 cm resolution composite image from two flights on the 05/07/2016 and the 23/08/2016, Environment Agency). Bank removal area shown with dashed red arrows.

3.4.8 Welwick

Welwick, opened in June 2006, is situated in the Humber estuary. The site was previously embanked for about 35 years and used as arable land (Pontee et al. 2006). The aim of the site was to provide 15–38 ha of mudflat and 12–28 ha of saltmarsh, as a compensation for port development, making it part of the legislative MR planning category (Pontee et al. 2006). In order to encourage mudflat formation, over 1,400 m of the previous embankment was removed to increase wave energy and prevent vegetation colonisation. Two breaches through the fronting saltmarsh were added (ABP mer 2011d) (red in Figure 3.26). Reprofiling of the site was undertaken to provide a gentle slope from the fronting mudflats to the highland (ABP mer 2011d). No initial creek system was dug, corresponding to creek design strategy 1.

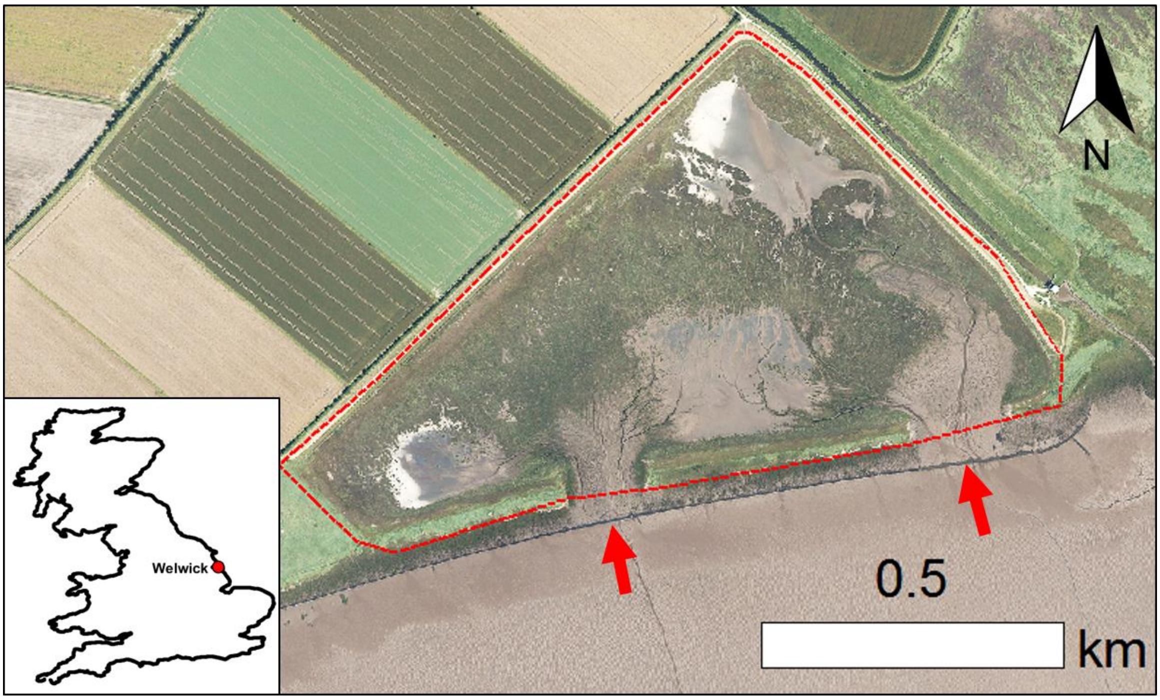


Figure 3.26: Aerial photograph of Welwick (20 cm resolution composite image from two flights on the 07/05/2016 and the 23/08/2016, Environment Agency). Breaches in red. MR extent considered shown by dashed red line.

3.4.9 Hesketh Out Marsh West (HOMW)

Hesketh Out Marsh West (HOMW), opened in September 2008, is situated within the Ribble estuary. The site had previously been embanked in the early 1980s for agricultural use (Tovey et al. 2009). The scheme is an example of legislative MR planning, aiming to create new intertidal habitats to compensate for losses related to flood alleviation work at Morecambe Bay, without increasing the flood risk in the surrounding area (Tovey et al. 2009). Due to its recent history as a saltmarsh, remnants of a previous, mature creek system were still visible in aerial photography at the time of scheme design, and those creeks were used as a template to excavate 15 km of shallow proto creeks linked to the Ribble estuary via 4 breaches (Tovey et al. 2009) (red in Figure 3.27). This corresponds to creek design strategy 2, with elements of design strategy 3, since some artificial features were added to improve the value of the site as a bird-watching area: 11 saline lagoons were excavated at the head of the creeks, within view of the nearby footpaths (Tovey et al. 2009).

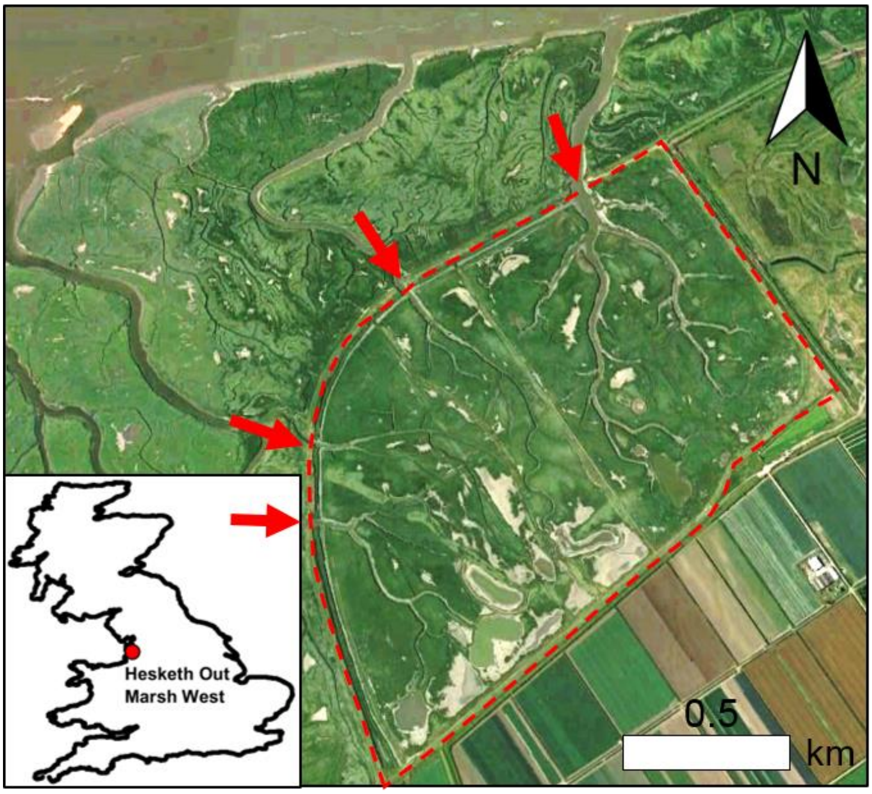


Figure 3.27: Satellite image of Hesketh Out Marsh West (Date: 17/07/2017, Source: Google Earth). Breaches shown by red arrows. MR extent considered shown by dashed red line.

3.4.10 Steart

Steart, opened in September 2014, is situated in the Parrett River in the Severn Estuary. Prior to MR implementation, the site was embanked since at least 1817 and used as agricultural land (Burgess et al. 2013). Steart is an example of strategic MR planning: it aims to help meet Habitats Directives, while providing compensation for historic habitat losses and improving flood protection in the peninsula. Steart is the largest scheme in this study with 400 ha of recreated habitat, about 300 of which is intertidal. Despite its large size, the scheme contains a single breach (red in Figure 3.28) to minimise the impact of the site on the bathymetry of the surrounding estuary (Pontee 2015a). A creek system was excavated to provide drainage and promote the creation of sustainable mudflats. Its curved herring-bone shape was dictated by a requirement to provide safe grazing space for sheep and cattle (Pontee 2015a). Ponds were also added to the head of channels to provide feeding grounds for birds and facilitate bird-watching. Mounts were added to the initially flat topography to provide niche habitats. This corresponds to design strategy 3, which was chosen over design strategy 2: an alternative suggested design featured a more natural-looking dendritic creek system connected to three breaches, but was rejected due to its potential impact on the Parrett River bathymetry (Pontee 2015b).

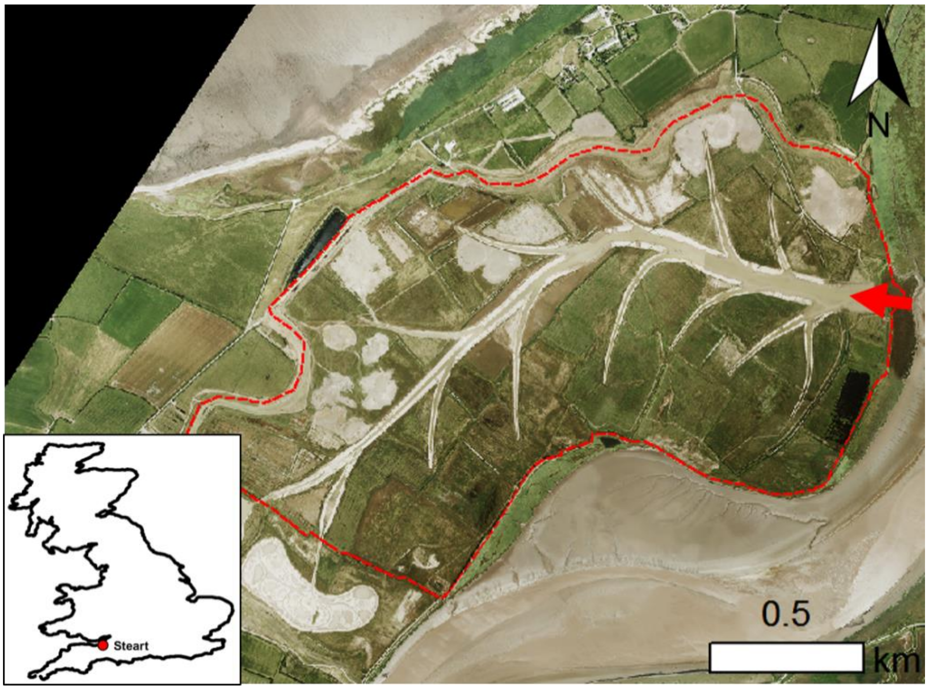


Figure 3.28: Aerial photograph of Steart (10 cm resolution image, date of flight 10/07/2014, Environment Agency). Breach area shown by red arrow (site breached in early September 2014). MR extent considered shown by dashed red line.

3.5 Accidentally realigned sites

One limit of comparing the evolution of creek networks in MR schemes in the UK is that most schemes are relatively young. In order to infer the long-term evolution of MR creeks, two accidentally realigned (AR) sites were selected based on their size (>50 ha) and their age since accidental breaching (95-110 years). Those sites were opened to tidal inundation during storm surges in 1897 and 1921 (Table 3.3). The use of acidentally breached sites as reference for the long-term evolution of MR schemes has been used extensively in the literature: before the development of the more process-based methods described in Section 2.3.2, naturally occurring breaches were used as an empirical reference to find the dimensions of MR breaches depending on the site's catchment area (Townend 2008). AR sites have also been used to infer the biodiversity found within restored coastal wetlands up to 100 years after implementation (Mossman et al. 2012). The assumption that the creek networks of these AR sites will be representative of the creeks' morphology and extent in MR schemes after about 100 years is thus coherent with the literature. The two AR sites considered should most resemble MR sites corresponding to design strategy 2, due to the presence of relic drainage ditches.

Table 3.3: Design details for all AR schemes considered (Source: ABPmer 2014). Creek design strategy refers to the 3 design strategies identified in Section 2.4.3:

	Location	Accidental breaching date	Lidar datasets suitable for use	Height above MWS (m)	Tidal range (m)	Site size (km^2)	Time embanked (years)
Brandy Hole	**River Crouch**	1897	2003; 2009; 2013; 2015; 2017	1.40	5.0	0.76	>123
Foulton Hall	**Hamford Water**	1897-1921	2009; 2013; 2015; 2017	1.69	3.7	0.57	>123

3.5.1 Brandy Hole

Brandy Hole is situated within the River Crouch in Essex. The environment is macrotidal with an interpolated tidal range of 5.0 m. The marsh was enclosed by seawalls some time before 1774 (Burd 1992). The site was breached by a storm surge in 1897, after an embankment period of over 123 years. The area of interest is 76 ha in size and dissected by a dense grid of old drainage ditches (Figure 3.29). The mean grain size of the suspended sediment in the nearby estuary is 0.14 µm, corresponding to a mud-dominated environment. Before the accidental breach, half of the site was pasture and the other half arable (Mossman et al., 2012).

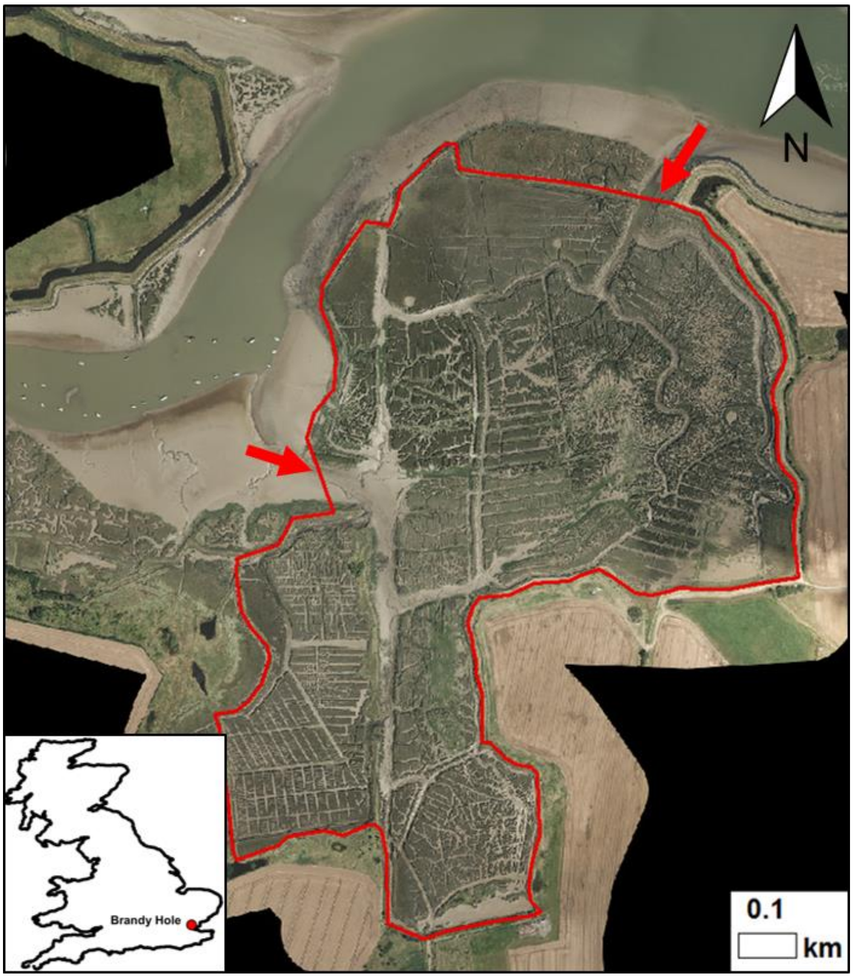


Figure 3.29*: Aerial photograph of Brandy Hole (20 cm resolution composite image from two flights on the 06/05/2016 and 23/09/2016, Environment Agency). Breaches shown by red arrows. AR extent considered shown by red line.*

3.5.2 Foulton Hall

Foulton Hall is situated on the coast of Hamford Water in Essex. The environment is mesotidal with an interpolated tidal range of 3.7 m. The marsh was enclosed by seawalls some time before 1774 (Burd 1992). The site was then breached by storm surges in two phases in 1897 and 1921, after an embankment period of over 123 years. The area of interest is 57 ha in size, and is dissected by a dense creek network partly inherited from old drainage ditches (Figure 3.30). The mean grain size of the suspended sediment in the nearby estuary is 3.38 µm, corresponding to an environment dominated by silt and clay. Before the accidental breach, the site was used as grazing land for horses or as a mowing marsh for hay (Burd 1992).

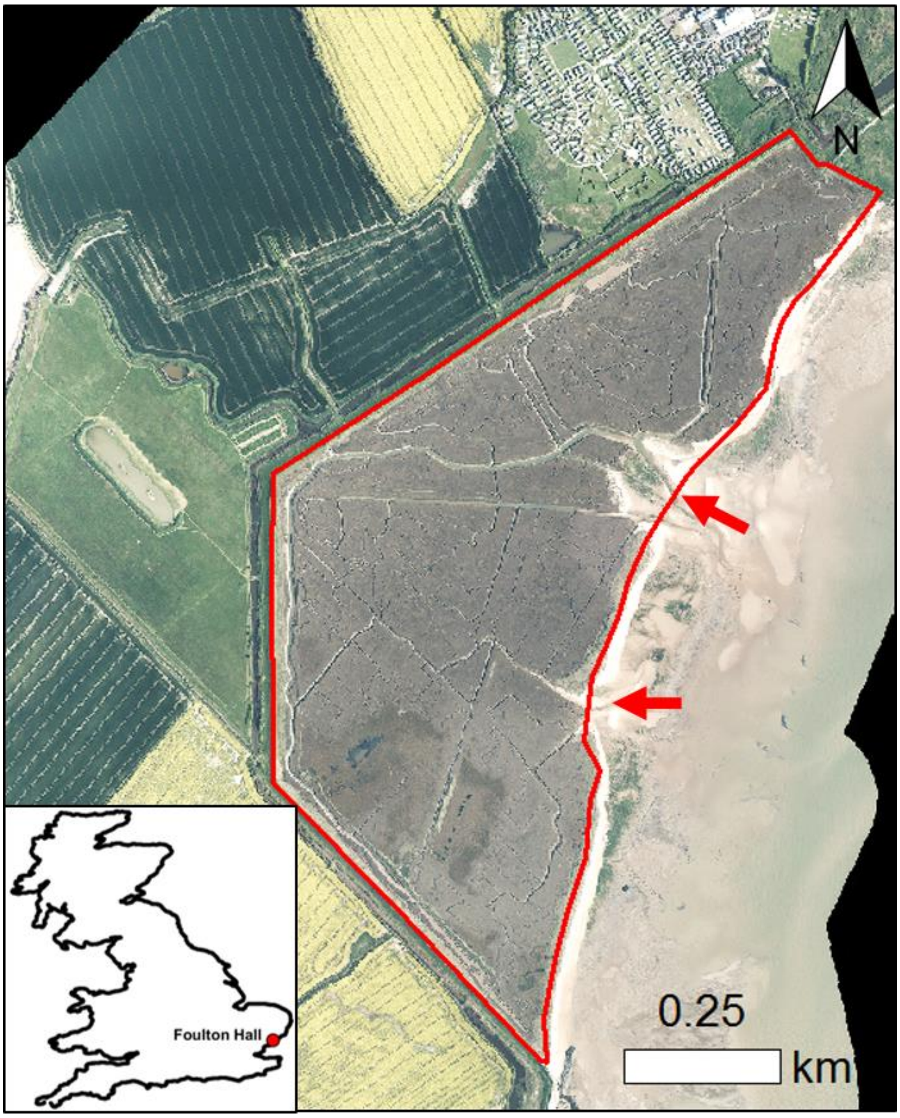


Figure 3.30: Aerial photograph of Foulton Hall (20 cm resolution composite image from two flights on the 06/05/2016 and 23/09/2016, Environment Agency). Breaches shown by red arrows. AR extent considered shown by red line.

3.6 Summary

This chapter has outlined the data sources and study sites used for this thesis. The lidar data collected by the EA and Natural Resources Wales provide morphological information over all of the UK coast and can capture the evolution of MR sites over the years by comparing several datasets taken at the same location. However, the temporal frequency of data acquisition is lower in the earlier years. Furthermore, the horizontal resolution of 1 m means that some creeks may be missed compared to field-based creek surveying. There is also an uncertainty linked to the possible detection of low-lying vegetation by the laser beam instead of the ground. Despite these limits, lidar remains the most efficient tool for large-scale, systematic morphological monitoring of coastal areas. It is also suspected that the presence of vegetation should not significantly impede the detection of creek areas, a hypothesis that will be tested further in Chapters 6 and 7. The tidal data for each site are obtained from a weighted mean interpolation of the tidal levels at standard and secondary ports as predicted by the Admiralty Tide Tables 2014.

These datasets are applied to 13 natural mature saltmarshes (England and Wales), 10 MR schemes (England) and 2 AR sites (England). The natural mature saltmarshes provide the expected equilibrium range of creek networks, and will be used to infer end targets for MR creek design, as well as to verify whether MR creeks are evolving towards the morphology expected of natural mature systems. The 10 MR sites are used as case studies to monitor how MR creeks have been evolving since implementation, depending on their initial conditions and design strategies. Finally, the 2 AR sites have been added to the analysis to provide some insight into the possible long-term evolution of MR creeks: can they be expected to reach the morphological equilibrium characteristics of natural systems after about 100 years of evolution?

Chapter 4: Method development

4.1 Introduction

The state of knowledge in creek network mapping from lidar was explored in Section 2.4.4. In summary, none of the currently available algorithms manage to automatically extract the morphological parameters of an entire creek system, which limits a detailed analysis of creek evolution. Conventional methods such as manual mapping from aerial photography and field surveys are time-consuming, and do not provide systematic volumetric data. In order to address this lack of monitoring tool and exploit the wealth of information provided by lidar, a new semi-automated creek parametrisation algorithm was developed, utilising Matlab R2015a. Section 4.2 describes the preprocessing steps performed on the lidar data before the algorithm can be used. Section 4.3 explains how the catchment areas of the natural saltmarshes were delimited. Finally, the main processing steps performed by the algorithm and the output morphological parameters are detailed in Section 4.4. Description and justification of the methods specific to individual result chapters are described in their respective chapters.

4.2 Preprocessing

Preprocessing of lidar data using ArcGIS 10.2.2 involves merging mosaics into a single dataset, georeferencing to the UK National Grid, interpolating data gaps to the values of the nearest neighbours according to Euclidean distance (lidar data are generally collected at low tide when most creeks are drained, but remnant water within ponds for instance may lead to gaps in the dataset), cropping to the saltmarsh area, and extracting elevation and slope maps. Visual observation found creek edges to be more visible on the slope map than on the curvature map for the selected datasets, so the slope was chosen as a threshold parameter unlike previous studies (Fagherazzi et al. 1999). The chosen preprocessing steps are minimalistic to provide monitoring tools that are easily reusable by MR designers for future schemes. The creek parameters are detected from freely available lidar DSM that have undergone minimal preprocessing, as is likely to be the case for most MR monitoring work performed by the EA and contracted consulting companies. The two outputs of this preprocessing phase, an elevation map and a slope map in degrees, both at a horizontal resolution of 1 m, are converted into text files and exported to Matlab for the processing phase.

4.3 Catchment area delimitation

In Section 2.4.1, the catchment area was described as the "area of influence" of the creek system, initially defined for fluvial drainage basins (Steel 1996). However, unlike fluvial drainage basins, saltmarsh catchment boundaries cannot be directly inferred from drainage divides in the elevation map, as drainage directions vary both in space and time due to the bidirectional tidal flow (Steel 1996; Marani et al. 2003). A hydrodynamic approach based on the estimation of water surface topography found drainage divides in saltmarshes to be equidistant between channels (Marani et al. 2003). The half-way distance between the creek system and the adjacent creeks thus constitutes a good approximation of the catchment area boundaries. This method also fits with Steel's (1996) catchment delimitation criteria, which facilitates the comparison of results between the two studies.

The seaward limit of each natural saltmarsh was defined as the mouth of the entry channel at the marsh/mudflat boundary, which serves as outlet for the considered creek network. The landward limit was defined as the local HAT level, corresponding to the limit between the tidally influenced saltmarsh and the land.

4.4 Workflow

This section details the workflow of the herein developed algorithm. The Matlab script takes as input the two text files created by the pre-processing phase (Section 4.4) containing elevation and slope maps at 1 m resolution. It comprises three main phases: 1) creek detection; 2) creek path repair; and 3) morphological parameter extraction (Figure 4.1). The parameters extracted per creek segment include creek order (topology of the creek system), sinuous creek length, sinuosity ratio, junction angle, and the width, depth and cross-sectional area taken at a cross-section across the middle of each segment (Figure 4.1). In addition, the following parameters are extracted for the whole system: total channel length, total creek area, total creek volume, and the distribution of the creek over the site. The parameter extraction method will be further explained in Sections 4.4.3 and 4.4.4.

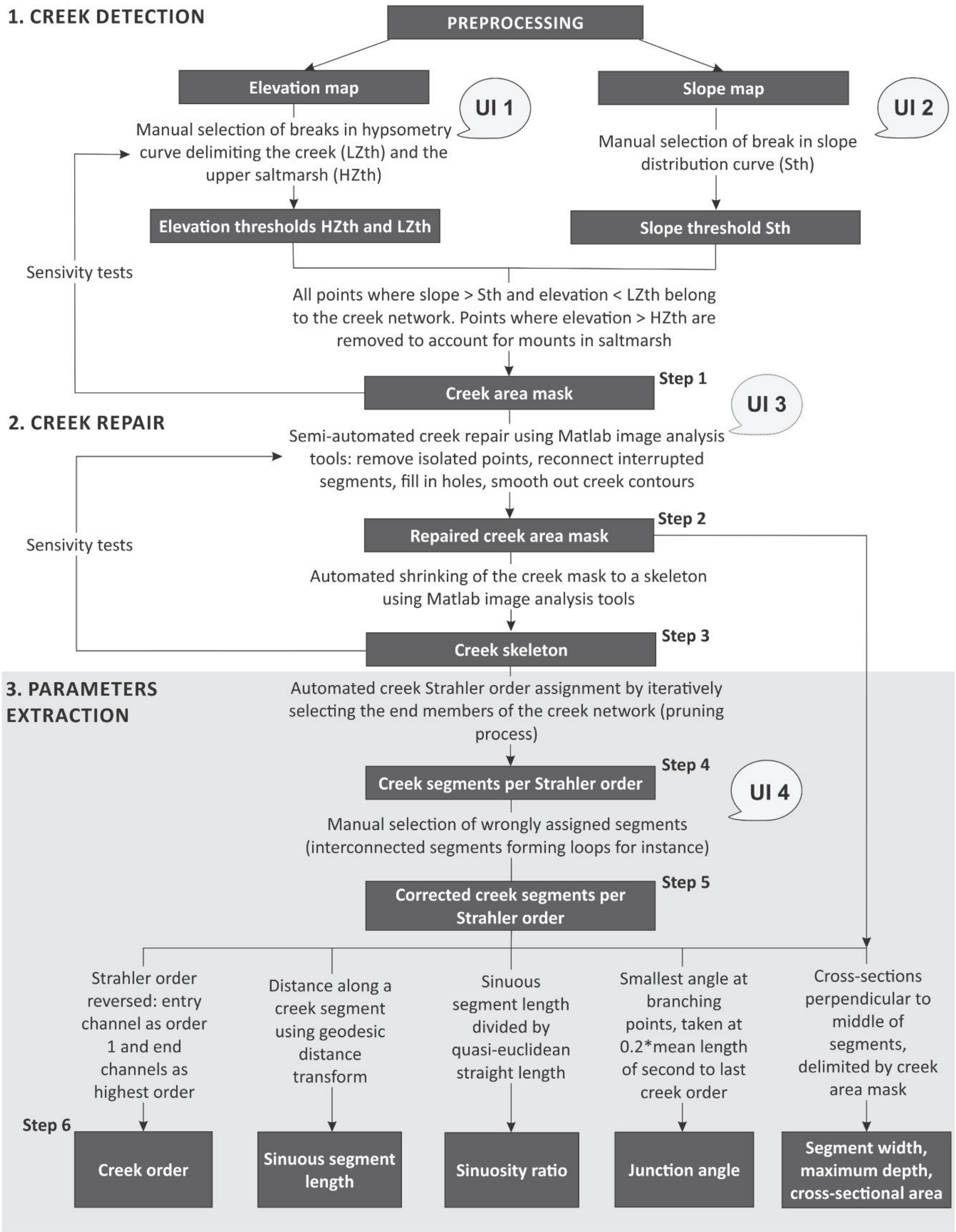


Figure 4.1: The creek parametrisation algorithm workflow comprises 6 processing steps and can be broken down into three phases: creek detection (step 1); creek repair (steps 2 and 3) and parameter extraction (steps 4 to 6). The steps where user inputs (UIs) are necessary are marked as UI 1 to 4.

4.4.1 **Creek detection**

First, a mask of low elevation and high slope is created using elevation and slope thresholds selected near the break points of hypsometry and slope distribution curves (Step 1, User Input (UI) 1 and 2, Figure 4.2). All points below the elevation threshold LZth correspond to the bottom of channels, and all points above the slope threshold Sth correspond to channel edges. In order to avoid detecting mounts or the edges of a flood defence, all points higher than the upper marsh elevation threshold HZth are excluded from the results. LZth, HZth and Sth are then refined by manual trial and error adjustments to obtain a creek layout visually similar to that observable in the lidar map, output as a logical mask (Figure 4.2). The 1 m horizontal and 0.15 m vertical resolution of the dataset used herein means that the smallest channels that can reliably be identified would be 2 m long, 2 m wide and 0.3 m deep. This means that small, developing creeks may not be detected. On the other hand, this ensures consistency in the detection of creeks compared to manual mapping.

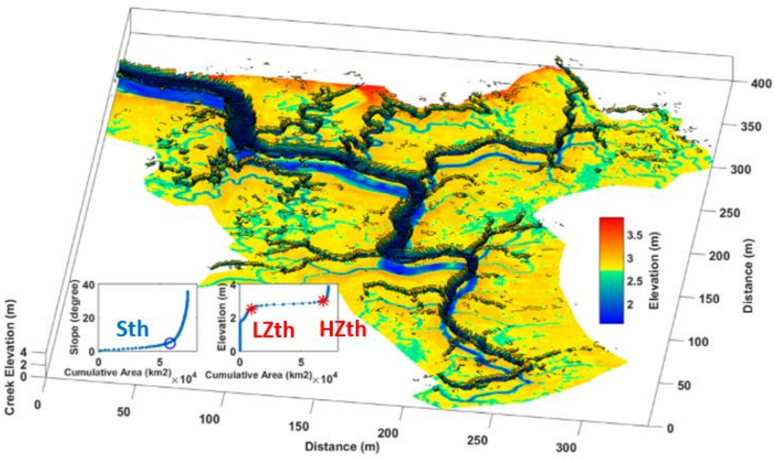


Figure 4.2: Creek elevation and slope threshold detection at Stiffkey Marsh (step 1). Sth: blue circle. HZth, LZth: red asterisks. The logical mask detected with those thresholds is displayed in black over the lidar elevation map.

4.4.2 **Creek repair**

After the initial creek detection phase, the noise from the raw creek mask (Figure 4.3A) is filtered by removing all connected elements smaller than a number of pixels defined by the user (UI 3, Figure 4.1). Fragmented terminal channels are then reconnected to the creek system (Step 2, Figure 4.1): the unconnected objects are selected in ascending stream order, and the shortest Euclidean distance to the rest of the network is chosen as the repair path, with the option of adding a distance threshold above which no reconnection occurs (Figure 4.3B). This method can be a source of error as the repair path is only 1 pixel wide, meaning channel width can be underestimated within the reconnected segments Figure 4.3C). This error is considered negligible due to the small contribution of terminal channels to the creek volume (e.g. 5% for Stiffkey Marsh) compared with their large contribution to the total channel length (33% for Stiffkey): accurate extraction of the length is therefore considered more important than the width of terminal channels. The generation of straight repair paths may also underestimate channel sinuosity, but here again the error is considered negligible since terminal channels have been found by previous studies to have a sinuosity ratio close to 1 (Steel 1996).

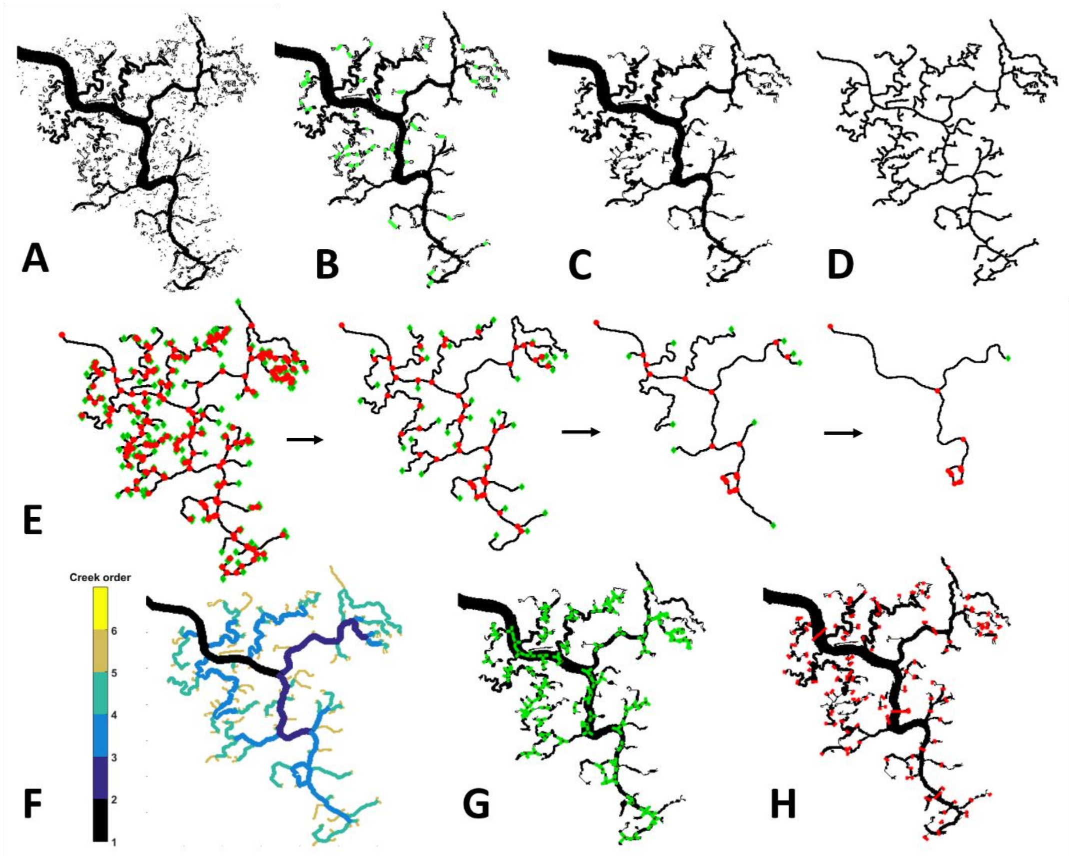


Figure 4.3: Creek network repair and parametrisation steps applied to Stiffkey Marsh. A: disconnected, noisy creek network mask extracted using elevation and slope thresholds (Step 1). B: creek segments reconnection using shortest Euclidean distance, with repair paths shown in green (Step 2). C: repaired creek mask (Step 2). D: creek skeleton (Step 3). E: pruning process used to assign a Strahler order to the creek segments (Step 4). Green diamonds correspond to end points and red dots to branch points. Terminal segments removed on first iteration are assigned the Strahler order 1, etc. In the case of breach segments and interconnected segments, the Strahler order is assigned manually (Step 5, UI 4). F: Reverse Strahler order (Step 6). G: junction angle, defined as the minimum of the three angles at each channel junction, measured at 0.2 times the mean length of second Strahler order creeks, shown in green (Step 6). H: Creek cross-section across the middle of each segment and delimited by the creek mask, used to calculate the channel width, depth and cross-sectional area, shown in red (Step 6).

The reconnection algorithm is only efficient if the creek network is the only significant low-elevation feature: otherwise elements such as ponds or agricultural trenches could be erroneously connected to the creek network. Such features can be removed from the logical mask by adding a feature size filter, which requires further sensitivity tests to ascertain the size of the unwanted features. Variations in pixel size will also influence the accuracy of this phase: a greater pixel size decreases the resolution of the dataset and thus increases the probability of neighbouring creeks getting fused together during the detection and repair process.

Other creek repair operations include infilling holes in the creek mask, and spur removal to smooth creek contours. Holes can occur in the dataset between the edges and the bottom of channels, where the elevation is too high and the slope too low to be detected by the threshold parameters. They are removed by filling in the inner pixels of the connected elements. Jagged creek contours (spurs) can be detected as individual creeks during the morphological thinning step described in the next paragraph: consequently smoothing creek contours removes spurious small channels without significantly changing the shape of the network.

The creek mask gives the total area and, combined with the elevation data from lidar, the volume of the channel network measured from the top of the creek edges (Figure 4.3C). The latter constitutes the potential semi-diurnal tidal prism as defined by Steel (1996). Once the creek area mask is complete, morphological thinning is applied to shrink the creek network to a skeleton corresponding to the centerline of the channels (Step 3, Figure 4.1). Though different from the thalweg, since the skeleton will be traced in the middle of each creek without necessarily matching the maximal depth point, this method gives an accurate representation of the shape and length of each creek segment (Figure 4.3D).

4.4.3 Creek parametrisation

Once the creek network shape and skeleton have been defined, the following steps of the algorithm provide a suite of morphometric parameters. The composition and complexity of the creek network can be expressed quantitatively in terms of stream Strahler order as defined in Section 2.4.1. Creek order can be automatically computed by using a pruning process and assigning a Strahler order iteratively to all creek segments (Step 4, Figure 4.4E). The creek system outlet is distinguished from the other end points by finding the maximal mean depth over a 10 pixels window, and is assigned the highest Strahler order at the end of the pruning process.

While fully automatic in dendritic systems, the pruning process is halted by the presence of interconnected creeks (Figure 4.4E). Previous studies have circumvented this issue by only selecting creek systems that do not branch out in the downstream direction (Novakowski et al. 2004), but the new algorithm must be able to monitor MR creek networks, which are typically highly interconnected (Pontee 2003). A new user input is therefore necessary: erroneous creeks are assigned their correct order manually using an interactive user interface (Step 5, UI 4, Figure 4.1, Figure 4.4) to get the final creek order map (Figure 4.3F).

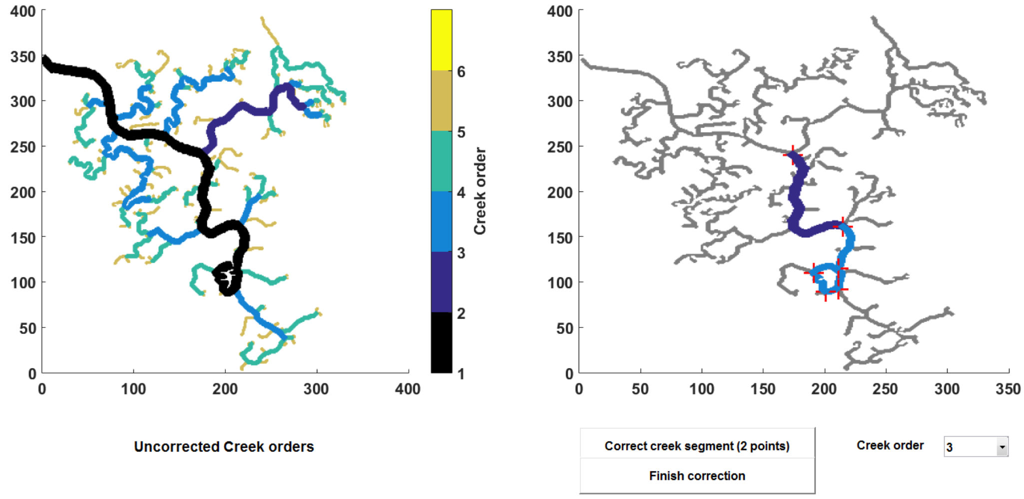


Figure 4.4: Creek order correction using an interactive user interface (Step 5, UI 4, Figure 4.1). The correct order is selected in a drop-down menu. Segments are then corrected by clicking on their two end points in any order (selected end points displayed as red plus signs on the right-hand map)

For each segment, the following parameters are automatically extracted (Step 6, Figure 4.1): (1) the channel length — the quasi-euclidean geodesic distance (QEGD) measured within an area constrained by the segment skeleton mask using equations [1] and [2] (Matlab function bwdistgeodesic, in which x1, x2, y1 and y2 represent the coordinates of the segment's end points within the matrix);

$$\mathrm{QEGD} = |x1 - x2| + \left(\sqrt{2} - 1\right) * |y1 - y2| \text{ if } |x1 - x2| > |y1 - y2| \qquad \textbf{[1]}$$

$$\mathrm{QEGD} = \left(\sqrt{2} - 1\right) * |x1 - x2| + |y1 - y2| \text{ if } |x1 - x2| < |y1 - y2| \qquad \textbf{[2]}$$

(2) the sinuosity ratio - the sinuous distance divided by the straight quasi-euclidean distance; and (3) the junction angle — defined as the minimum of the three existing angles at each channel junction, measured at 0.2 times the mean length of second Strahler order creeks following Steel's (1996) method (Figure 4.3G). Furthermore, a cross-section of the creek mask obtained in Step 2 (Figure 4.1) is taken across the middle of each segment to obtain the channel width, maximal detected depth in the cross-section, and cross-sectional area (Figure 4.3H). The main channel length is measured using the geodesic distance function along the longest channel in each creek network, connecting the mouth to the furthest terminal channel. The cross-sectional

characteristics of the creek mouth or breach area, delimited by the creek tops, can also be extracted by manually selecting a cross-section of the creek mask.

The Strahler ordering system's main limitation for saltmarsh restoration studies is that the order of the entry channel appears to change over time as new, smaller channels develop (Weishar et al. 2005). Referring to the entry channel as the first order channel avoids such confusion. Therefore, the Strahler order of every creek segment is reversed: the entry channel becomes the first order, and the terminal channels the highest order (Figure 4.3F). The Reverse Strahler (RS) order first was proposed to study dendritic outgrowth at different stages of development (Uylings et al. 1975). This approach is well suited to the analysis of MR creek evolution, where the highest order is expected to increase over time as the network expands.

The time required to obtain the creek maps and parameter tables for a 500,000-pixel dataset containing a complex sixth order creek network is only 80 seconds on a standard desktop computer, making this a much faster and less labour-intensive method than manual digitisation from aerial photographs (Mason et al. 2006). This also facilitates running a large number of sensitivity tests for each site, using different thresholds to optimise the output creek network. Another advantage is that consistent morphological criteria are used to detect creeks, rather than subjective visual interpretation: manual extraction can be a source of error in studies that do not ground-truth their extracted creeks with field surveys, as creeks of an order of one centimeter depth can be clearly visible in aerial photography, despite having no significant influence on water distribution and drainage.

4.4.4 **Drainage efficiency**

Though giving information on the mature size and volume of creek networks in relation to catchment area, the existing morphological equilibrium relationships described so far give no information regarding the distribution of the creek system across the site (

Table *4.1*). In order to address this gap in knowledge, it will be necessary to develop a new morphological equilibrium relationship using a proxy of the creek network's spatial distribution. This proxy should be representative of the creek network's efficiency at draining the site during the ebb and at distributing water and sediment during the flood tide.

Table 4.1: List of morphological equilibrium relationships of the creek network in natural saltmarshes established by Steel (1996)

Parameters (y vs. x)	
Total channel length (m) vs. catchment area (m^2)	$y=1.7x^{0.7}$
Maximum creek mouth width (m) vs. potential semi-diurnal tidal prism (m^3)	$y=0.28x^{0.4}$
Main creek length (m) vs. catchment area (m^2)	$y=1.5x^{0.5}$
Maximum creek mouth width (m) vs. total channel length (m)	$y=0.22x^{0.5}$
Mean cross-sectional area of main creek (m^2) vs. total channel length (m)	$y=0.04x^{0.7}$
Mean cross-sectional area of main creek (m^2) vs. catchment area (m^2)	$y=0.01x^{0.6}$
Mouth cross-sectional area of main creek (m^2) vs. potential semi-diurnal tidal prism (m^3)	$y=0.02x^{0.7}$

A commonly used proxy for creek distribution across the site is the drainage density, defined by Steel (1996) as the ratio of total channel length versus catchment area. This one-dimensional metric gives some information on the creek density, but does not account for spatial distribution (Marani et al. 2003). A more accurate representation of the creek network drainage efficiency is the unchanneled length, the hillslope distance of all pixels to the nearest creek (Lohani et al. 2006; D'Alpaos et al. 2007; Kearney et al. 2016). Flat distance was used in this study instead of 3D distance over the sloping surface to reduce computation time, and because the elevation gradient is generally very low in intertidal wetlands. This parameter is a morphological approximation of the hydrodynamic-based measures of flow paths (Marani et al. 2003), and is more useful than drainage density for describing ecosystems, as vegetation distribution is correlated with the distance to the creek network (Temmerman et al. 2005).

The unchanneled length is plotted as a semi-logarithmic exceedance probability distribution curve showing the distance to the creek network for each pixel in the DSM (Figure 4.5A). The drainage efficiency of the creek network was also expressed quantitatively using the Overmarsh Path Length (OPL), expressed as the slope of the first 50 % values of the exceedance probability distribution in meters (Figure 4.5B). OPL gives the average distance that needs to be crossed within a marsh before encountering a creek: the lower this parameter, the better irrigated the site by a dense and well-distributed creek system (Marani et al. 2003), while a marsh characterised by a large OPL is "emptier".

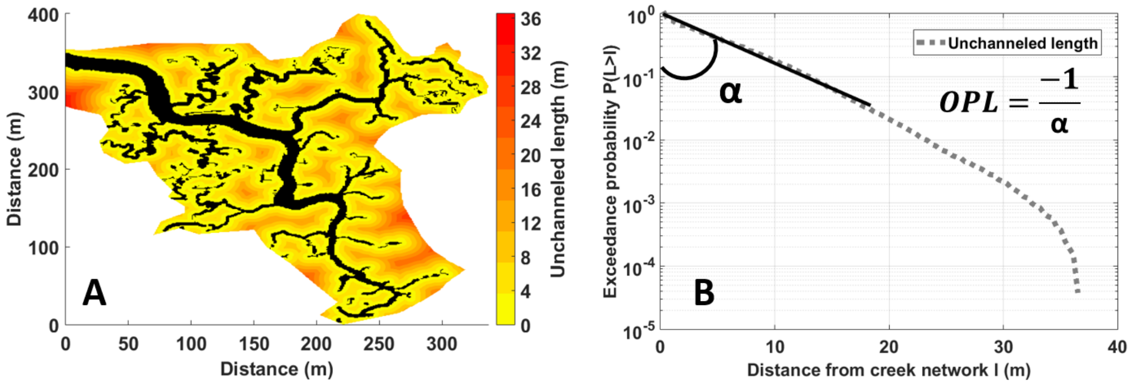


Figure 4.5: Illustration of the unchanneled length distribution (A-B) and overmarsh path length OPL (B) at Stiffkey, expressed as the slope of the first 50 % values of the exceedance probability distribution in meters.

4.5 Summary

The semi-automated creek parametrisation algorithm presented in this section is intended to improve the accuracy and usability of monitoring data for creek networks in natural and artificial coastal wetlands. Creek networks are difficult to map manually due to their complex structures, and recent algorithms are poorly suited to the morphological interpretation of creek networks because they do not assign creek order nor extract creek parameters.

A semi-automated algorithm was developed based on the threshold method, which allows for intuitive creek detection criteria, in accordance with most previous studies: a creek network is defined as a connected feature which lies lower than the rest of the saltmarsh, and whose edges are delimited by a steeper slope. Those criteria are strictly based on morphology rather than as a function of runoff like the D8 flow accumulation method (James et al. 2010) and are thus well suited to the study of both natural and artificial creek networks.

The algorithm is faster and less subjective than manual mapping, and interactive interfaces make it easy to use by researchers and stakeholders. Furthermore, the short running time at the pixel size used for the study allows for a number of sensitivity tests to refine the threshold parameters. The algorithm is currently being improved in collaboration with CH2M/Jacobs to improve its usability and generate impact.

The algorithm gives as outputs a number of morphometric parameters for the creek network (Table 4.2), following previous recommendation guidelines for creek design (Zeff 1999). The rest of this thesis will exploit these parameters to analyse creek networks in natural and artificial saltmarshes. Chapter 5 compares results from this algorithm with that of field-validated manual digitisation in natural mature saltmarshes in the UK. Chapter 6 monitors the evolution of creek networks in MR sites over the years since their implementation and estimate the rates of change for each parameter. Finally, Chapter 7 compares creek networks from natural saltmarshes and from MR sites, in order to estimate the efficiency of their initial design at promoting creek growth towards a natural-looking system.

Table 4.2: Summary of outputs from semi-automated creek parametrisation algorithm

Variable (Symbol when applicable)	**Unit**
General site variables	
Mean elevation above MHW	m
Catchment area	m^2
Mean marsh gradient	%
Creek Planimetric variables	
Creek planform area	m^2
Total channel length	m
Drainage density –total channel length per km^2	km/km^2
Overmarsh Path Length (OPL) –mean distance to creeks	m
Highest order of creek system	*no unit*
Number of creeks per order (N_i)	n
Bifurcation ratio per order (N_{i+1}/N_i)	*no unit*
Main sinous channel length	m
Main straight channel length	m
Main channel sinuosity ratio	*no unit*
Mean sinous creek length per order	m
Mean straight creek length per order	m
Mean sinuosity ratio per order	*no unit*
Mean junction angle per order	^{o}
Creek Cross-sectional variables	
Total creek volume (measured from creek top)	m^3
Mouth top width	m
Mouth depth	m
Mouth cross-sectional area	m^2
Mean creek top width per order	m
Mean creek depth per order	m
Mean creek cross-sectional area per order	m^2
Mean creek width/depth ratio (W/D) per order	*no unit*
Area/depth^2 ratio –Mean W/D	*no unit*
Main channel gradient	%

Chapter 5: Parametrising tidal creek morphology in mature saltmarshes

This chapter is adapted from the following publication:

Chirol, C., Haigh, I.D., Pontee, N., Thompson, C.E.L. and Gallop, S.L. (2018) Parametrizing tidal creek morphology in mature saltmarshes using semi-automated extraction from lidar, *Remote Sensing of Environment*, 209, pp. 291-311. DOI: https://doi.org/10.1016/j.rse.2017.11.012

5.1 Introduction

MR is the dominant mitigation method to counter for the loss of coastal wetland habitats worldwide, and is being integrated in long-term plans for sustainable coastal management in the UK. However, some aspects of MR scheme design rely mostly on practical experience from consulting bodies, and lack guidelines inferred from scientific evidence. Creek network design is the most prominent knowledge gap due to the complexity and variability of creek systems in natural saltmarshes. Creek networks are crucial to the ecological functioning of coastal wetlands, and should be investigated further to deliver better MR schemes. This chapter aims to provide tools to monitor the evolution of creek networks towards morphological equilibrium, and to validate these tools through a comparison with creek morphometry data collected in the field.

Improved understanding of the evolution of MR creek networks can only happen through systematic monitoring of existing schemes. A new parametrisation algorithm has been developed in Chapter 4 to facilitate this process. The creek mapping method builds on conventional elevation and slope threshold methods (Fagherazzi et al. 1999). The morphology of mature creek systems in the UK has been extensively investigated by Steel (1996) using a combination of manual creek mapping from aerial photography and field surveying. Morphological equilibrium relationships were established, relating creek length and volume parameters to the dimensions of the catchment area. However, the predictive value of these relationships has yet to be investigated. Their applicability to lidar-extracted datasets should also be verified as more and more large-scale monitoring surveys are currently undertaken using lidar (Gesch 2009; Klemas 2013). Finally, the currently available morphological equilibrium relationships do not account for creek distribution over the site: this is a problem since creeks in MR schemes are usually dug as a simplified network of oversized channels, (Pontee 2015a), which could lead to a poorer creek distribution.

This chapter addresses Thesis Objective 1: to provide a comprehensive morphometric analysis of creek systems in natural coastal wetlands in the UK, using state of the art datasets and data extraction methods, to define equilibrium characteristics that will be used as an end target for MR creeks. To that end, there are three sub-objectives as follows:

(1) To test the efficiency of a newly developed semi-automated creek parametrisation algorithm at extracting creek morphology in natural saltmarshes;

(2) To assess the applicability and predictive value of existing creek network morphological equilibrium relationships using parameters extracted using in sub-objective (1); and

(3) To establish a new equilibrium relationship linking the creek network distribution within the site to initial morphological conditions and tidal forcing, to be used as a MR design tool.

The structure of this chapter is as follows. Section 5.2 describes the methods specific to this chapter, Section 5.3 lists the results for each sub-objective, Section 5.4 discusses the implications of the results, and Section 5.5 provides a summary of the key findings.

5.2 Methods

5.2.1 **Comparison of field-validated manually extracted creeks and semi-automatically extracted creeks from lidar**

In order to fulfill the first sub-objective, the new algorithm described in Chapter 4 was used to extract creek networks from 1 m resolution lidar data collected between 2008 and 2015 at each of the 13 sites (Table 5.1). The extracted creek morphological parameters, as well as key environmental controls found in the Admiralty Tide Tables (MSTR and tidal asymmetry) or in the literature (Marsh age and percentage of fine sediments (Steel 1996)) are presented in Table 5.2.

The newly extracted creeks were compared with datasets collected in 1996 through manual mapping from aerial photography and field surveying (Steel 1996). To that end, two approaches were used: first, a visual comparison of the creeks' shape and extent by superimposing the newly extracted data on Steel's creek network maps, in order to discuss the potential errors of acquisition or omission linked to both methods; second, a statistical comparison of the morphometric parameters was performed to determine the difference and agreement between the two methods. Bland and Altman diagrams (Bland et al. 1986) were used to determine whether the agreement between the two pairs of readings was acceptable, detect outliers and visualise systematic biases in the results (Watson et al. 2010).

Table 5.1: List of morphological parameters considered in this chapter. The list is a subsection of morphological parameters from Table 4.2, after removal of redundant parameters such as straight main channel length/sinuous channel length/main channel sinuosity ratio. Environmental controls found in the Admiralty Tide Tables or from (Steel 1996) were also added.

Variable (Symbol when applicable)	Unit
Planimetric variables	
Catchment area	m^2/km^2/ha
Total channel length	m/km
Drainage density –total channel length per km^2	km/km^2
Overmarsh Path Length (OPL) –mean distance to creeks	m
Number of creeks	n
Mean channel length for each order	m
Main Channel length	m
Mean sinuosity ratio	
Junction angle	°
Cross-sectional variables	
Total creek volume (measured from creek top)	m^3
Mean creek cross-sectional area for each order	m^2
Creek mouth cross-sectional area	m^2
Mean creek width for each order	m
Main channel width	m
Creek mouth width	m
Mean creek depth for each order	m
Main channel depth	m
Creek mouth depth	m
Width/depth ratio –shape of the creek (W/D)	
Area/depth^2 ratio –Mean W/D	
Main channel gradient	
Environmental controls	
Approximate marsh age	years
Mean marsh elevation above mean spring sea level	m
MSTR (Mean spring tidal range)	m
Tidal asymmetry (flood duration/ebb duration)	
<20µm sediment %	%

5.2.2 Validation of existing creek network morphological equilibrium relationships

In order to fulfill the second sub-objective, the newly obtained parameters were assess the applicability and predictive value of existing creek network morphological equilibrium relationships. To that end, two statistical methods were used. The determination coefficient R^2 represents the scatter of points about the best fit regression line between the two variables studied, and thus the scatter of actual values around the morphological equilibrium lines. The mean absolute percentage error (MAPE) is an unbiased measure of the predictive value of a model used in hydrological studies (Wang et al. 2009). The MAPE provides a term by term comparison of the predicted and actual values of the studied variable, and is commonly defined as in equation 3 (X_m is the measured parameter at equilibrium, X_p the parameter at equilibrium as predicted by the power law relationship being tested, and n is the number of sites):

$$\mathrm{MAPE} = \frac{1}{n}\sum_{i=1}^{n}\left|\frac{X_p(i)-X_m(i)}{X_m(i)}\right| * 100 \qquad \textbf{[3]}$$

The creek network morphological equilibrium relationships considered are taken from the UK reference sites studied by Steel (1996), but also from other studies in San Francisco Bay (Williams et al. 2002) and in Venice Lagoon (Marani et al. 2003) to test their potential applicability beyond the national scale. The determination coefficient R^2 and the MAPE will be used to verify whether these morphological relationships are effective in quantitatively predicting the final shape of the creek network based on the initial conditions. This will inform to what extent these relationships can be reliably used in MR design.

5.2.3 Linking creek network drainage efficiency to environmental controls

In order to fulfill the third sub-objective, a new morphological equilibrium relationship was developed using OPL, a proxy of creek distribution across the site. OPL represents the "emptiness" of the site, or the mean distance that needs to be crossed within the marsh before encountering a creek. OPL is relevant to the marsh ecological functioning, so it is of interest to investigate whether it reaches an equilibrium state with the dimensions of the site, with the sediment properties or with the tidal forcings.

OPL was related to the initial morphological and sedimentological conditions and to the tidal forcings through a multiple linear correlation. In order to select, among the available variables, which accounted for most of the variability in the dataset, a principal component analysis (PCA) was performed. PCA reduces the dimensionality of the dataset while retaining as much of the variation as possible (Jolliffe 2002). The variables are transformed into a new set of uncorrelated principal components (PC), where the first few PCs contain most of the variation in the original variables. One limit of PCA is that it assumes all variables are normally distributed, a condition not necesseraly met in this study. However, useful descriptive information can still be inferred from PCA in terms of classification of intercorrelated parameters even if the data are not normally distributed (Jolliffe 2002).

The choice of variables included in the PCA was guided mainly by the need to produce a tool to help future MR design. This justifies the use of a limited number of parameters: designers tend to have limited access to information on the vegetation distribution or sediment properties, especially since those properties may change post-implementation with new sediment being brought to the system. Another limitation of PCA is the assumption that the variables selected fully represent the statistical variation of the dataset (Steel 1996; Jolliffe 2002). Since this study operates with a limited set of parameters, the interpretations inferred should be treated with caution. However graphical observation of the PCs can provide some indication of the relative importance of each considered variable.

The selected parameters were separated into 5 groups (Table 5.2): morphological markers of creek development already used in equilibrium relationships; markers of creek drainage efficiency; tidal forcings; markers of creek maturity corresponding to phase D of marsh development (Figure 2.9) where the system is at equilibrium with a low channel gradient and a sinuous creek network; and markers of sediment stability given by the percentage of cohesive sediment. All variables were standardised by substracting the mean and dividing by the standard deviation in order to account for the effects of scale and different units used. The parameters of each group were expected to be intercorrelated and hence to remain clustered in the PCs. Of particular interest was to which group the drainage efficiency (group 2) would be most closely correlated: to other morphological characteristics, to the tidal forcings, to the maturity of the system or to the sediment stability?

Table 5.2: List of parameters selected for PCA due to their relevance to creek design and evolution.

Parameters selected for PCA	**Relevance to creek design**
Catchment area (m^2) Main channel cross-sectional area (m^2) Main channel depth (m) Main Channel length (m) Main channel width (m) Total channel length (m) Total creek volume (potential tidal prism) (m^3)	Morphological markers of creek development already used in equilibrium relationships (group 1)
Drainage density (m/m^2) Overmarsh path length (OPL) (m)	Markers of creek drainage efficiency (group 2)
Mean marsh elevation above mean spring sea level (m) MSTR (Mean spring tidal range) (m) Tidal asymmetry (flood duration/ebb duration) ()	Tidal forcings (group 3)
Main channel gradient (^{o}) Marsh age (years) Mean sinuosity ratio ()	Markers of creek maturity (group 4)
<20µm sediment % ()	Marker of sediment stability (group 5)

The first PCs that account together for > 75% of the dataset variance are considered sufficiently representative of the variability of the dataset (Stevens, 1986 *in* Steel, 1996). The classification into independent PCs helped to pick variables for the multiple linear regression, and to propose a morphological equilibrium relationship using the drainage efficiency represented by OPL. The Matlab function fitlm was used to create a linear regression model, and the function step to automatically add or remove variables to optimise the fit. This iterative process is efficient for the small number of variables considered here. The best fits gave out new morphological equilibrium relationships to fulfill the third sub-objective.

5.3 Results

5.3.1 Creek network parametrisation algorithm performance compared with field-based survey

The first sub-objective is to evaluate the performance of the semi-automated creek parametrisation algorithm. The creek networks yielded by the algorithm were overlain over those extracted manually by Steel (1996) for visual comparison (Figures 5.1-5.4). The algorithm successfully captured the variety in width and sinuosity of the low RS order channels. For example, the entry channel of the Tollesbury marsh is noticeably wider and more sinuous than the others Figure 5.4B), while the Banks marsh displays a thin entry channel compared to the other sites (Figure 5.1B).

Despite the good visual agreement between the two outputs, some differences remain. Fusing of high RS order channels caused an underestimation of the number of creeks and an overestimation of the creek width in the most complex systems (See Grange marsh, Figure 5.2B). Fusing also happened in the smallest saltmarsh in the dataset where the 1m lidar resolution is insufficient to capture the correct creek distribution: at Warren Farm (catchment area of 0.006 km^2 Table 3.1), the headmost creeks were detected as one clump (Figure 5.1B), leading to an underestimation of the number of creeks in this area (Figure 5.1C). The same can be seen for Gibraltar Point (Figure 5.1B-C). In other places, the creek network repair algorithm produced very thin channels (visible for Tollesbury in Figure 5.4B) to limit channel interconnection, as explained in Subsection 4.4.2. Many of the high RS order channels drawn by Steel (1996) were not detected by the algorithm, especially in complex systems like Grange Marsh Figure 5.2A-C). However, the shape of the larger channels and the extent of the creek network match well with Steel's results.

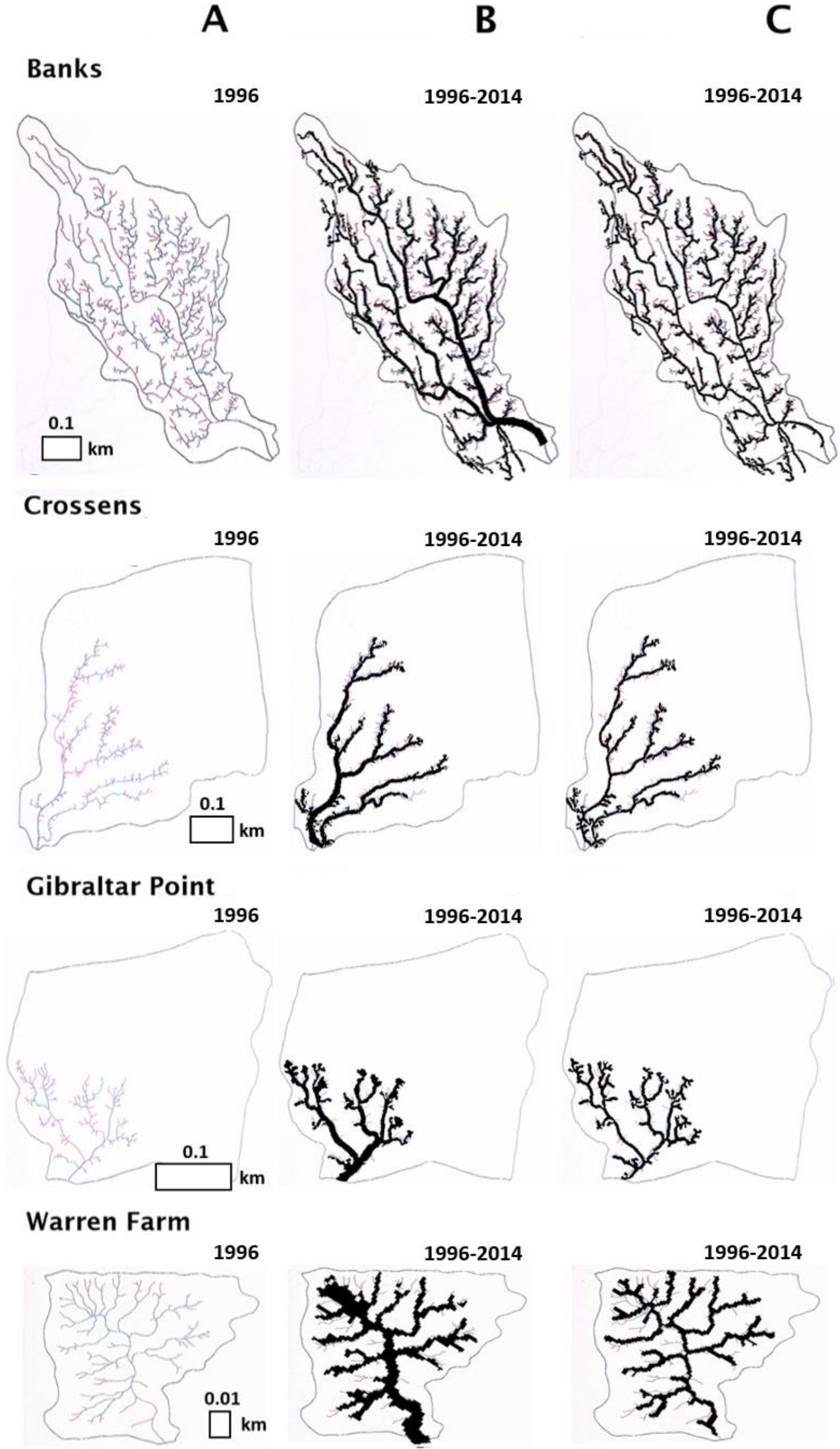


Figure 5.1: Creek area and skeleton extraction results from Banks. Crossens, Gibraltar Point and Warren Farm, compared with Steel's channel extraction results (1996). A: Creek network skeleton manually extracted by Steel. B: creek mask. C: creek skeleton (B and C overlain over Steel's manual extraction results).

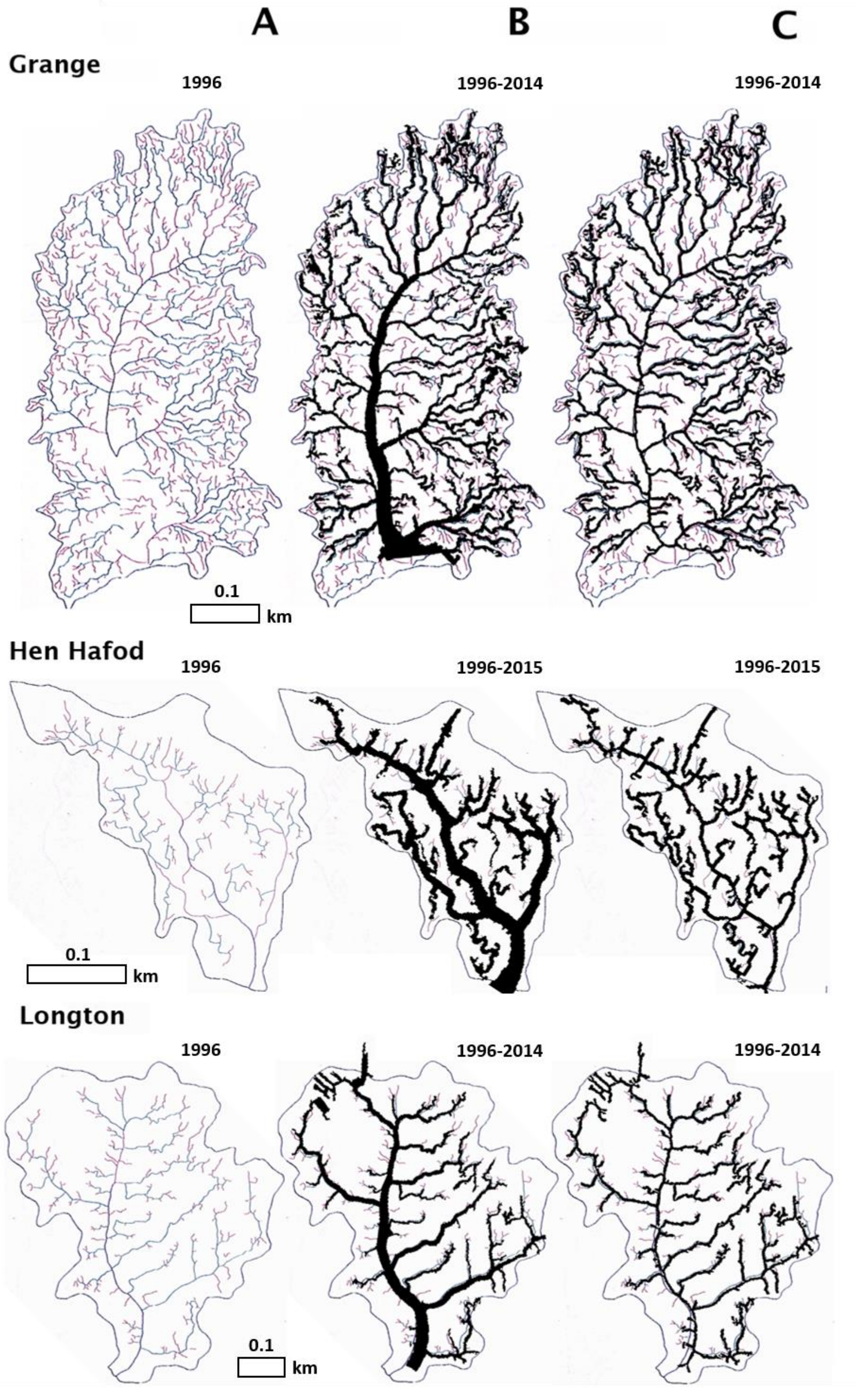


Figure 5.2: Creek area and skeleton extraction results from Grange, Hen Hafod and Longton, compared with Steel's channel extraction results (1996). A: Creek network skeleton manually extracted by Steel. B: creek mask. C: creek skeleton (B and C overlain over Steel's manual extraction results).

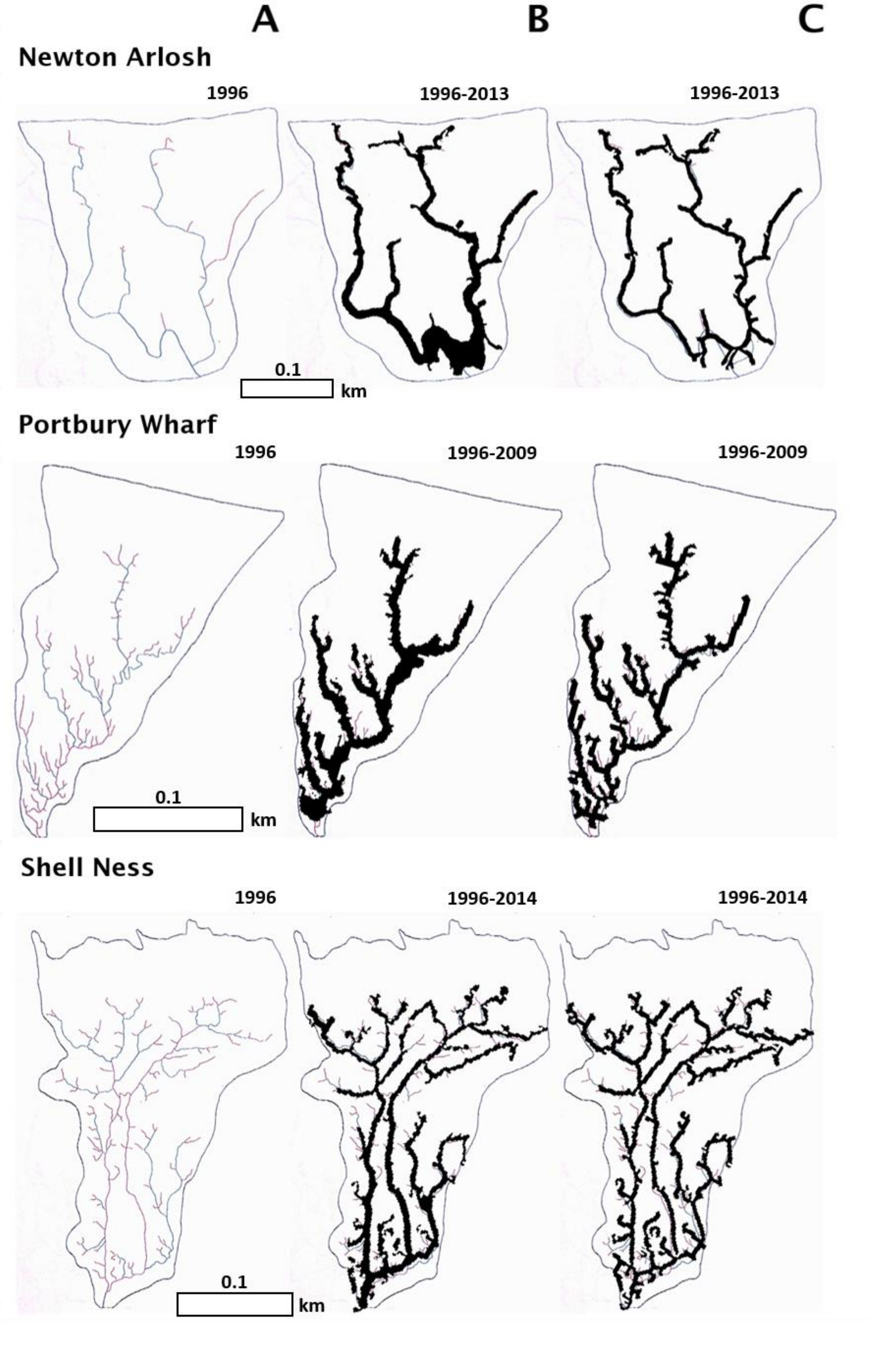


Figure 5.3: Creek area and skeleton extraction results from Newton Arlosh, Portbury Wharf and Shell Ness, compared with Steel's channel extraction results (1996). A: Creek network skeleton manually extracted by Steel. B: creek mask. C: creek skeleton (B and C overlain over Steel's manual extraction results).

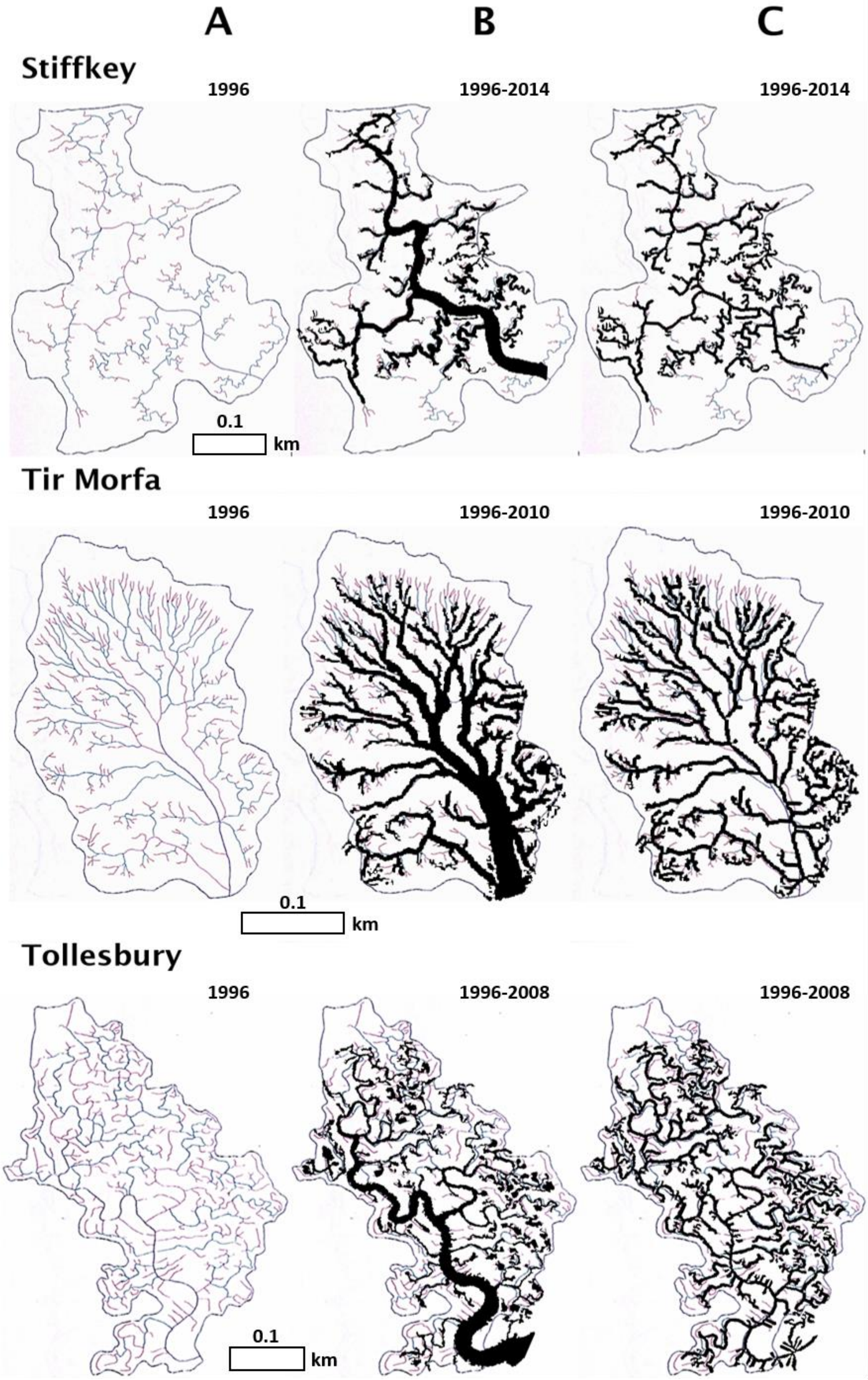


Figure 5.4: Creek area and skeleton extraction results from Stiffkey, Tir Morfa and Tollesbury, compared with Steel's channel extraction results (1996). A: Creek network skeleton manually extracted by Steel. B: creek mask. C: creek skeleton (B and C overlain over Steel's manual extraction results).

All segments were given a RS order for better representation of the creek network branching complexity (Figure 5.5). Mean morphological parameters were extracted for each RS order and plotted against Steel's (1996) results (Figure 5.6). The number of creek segments increases exponentially and their length, depth and cross-sectional area decrease exponentially with RS order (Figure 5.6A-D-F). The differences with Steel's results (1996) are under an order of magnitude and confirm the good visual agreement between results. The sinuosity ratio yields similar results to Steel's (Figure 5.6E): the first RS order channels display a range of sinuosity values between 1 and 1.6, and creeks become more uniformly straight with increased RS order. The mean bifurcation ratio of the higher RS order channels ranges between 3.0 and 5.2, with a mean value of 3.70 (Figure 5.6C), meaning that this algorithm detected on average 3.7 times more creeks for each new RS order.

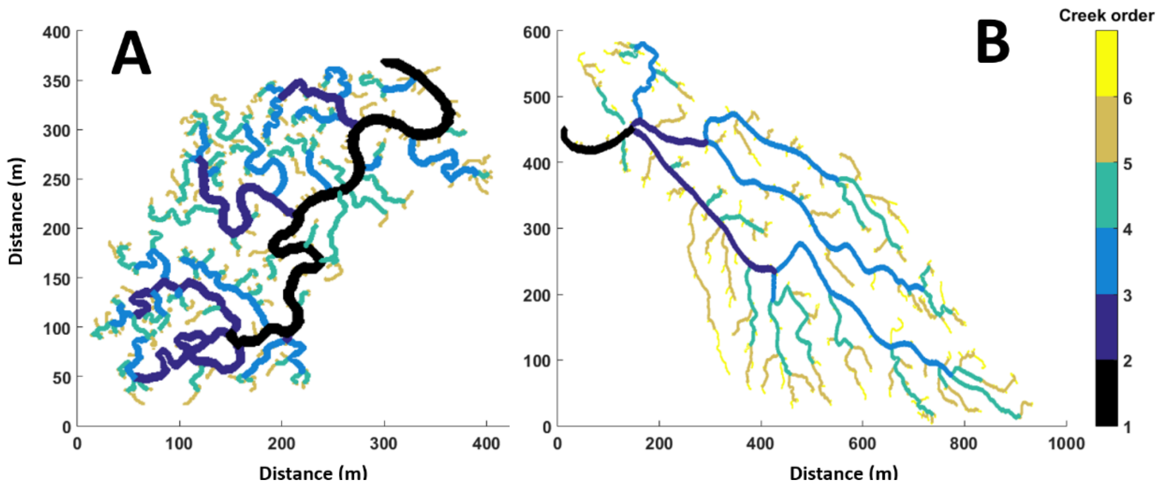


Figure 5.5: Reverse Strahler (RS) order of creek segments in Tollesbury Marsh (A) and Banks Marsh (B).

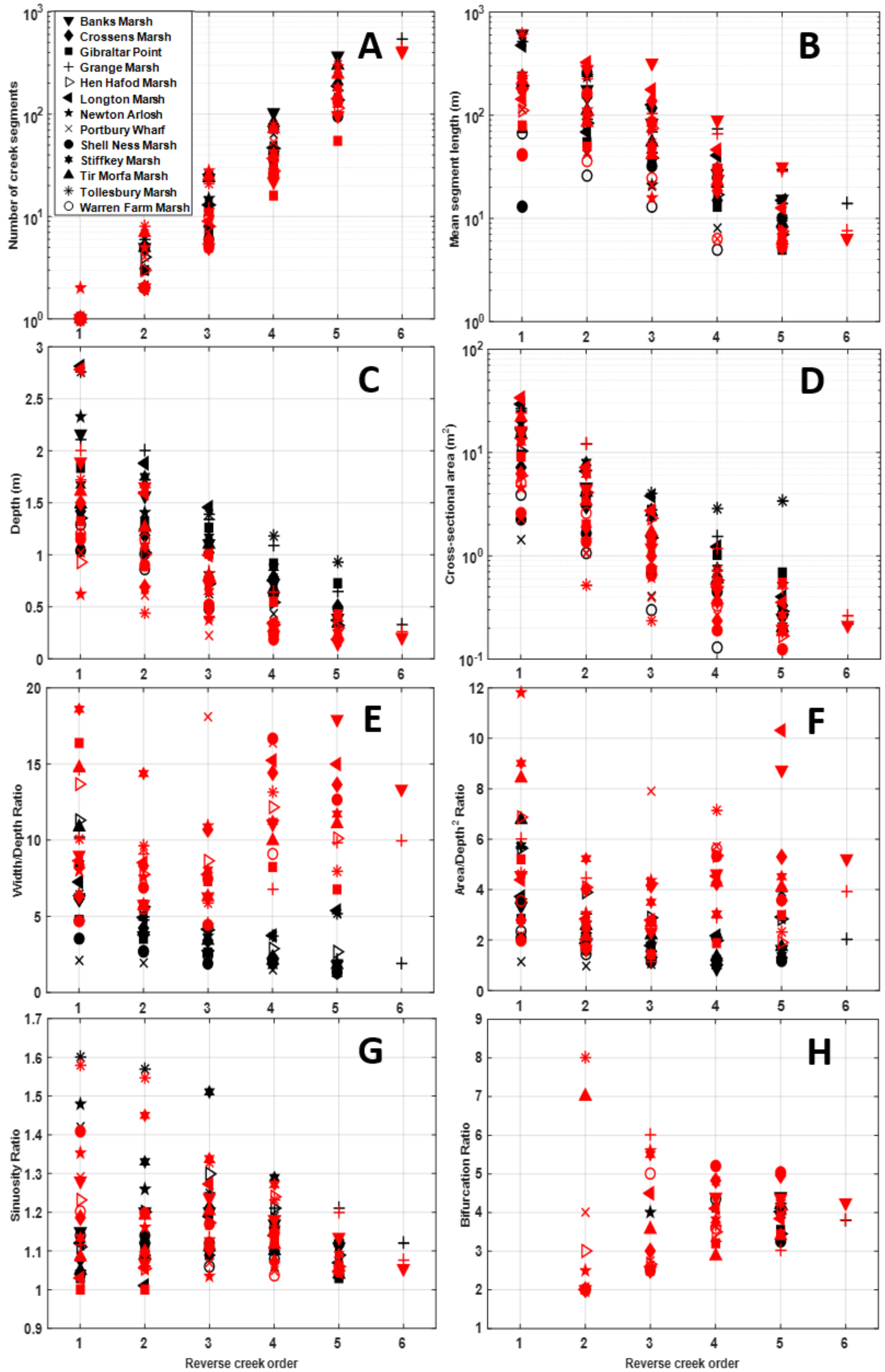


Figure 5.6: Visual comparison of creek morphological characteristics per reverse Strahler order as identified by Steel (1996) in black and by the semi-automated extraction algorithm in red. A: number of creeks; B: mean length; C: creek depth; D: cross-sectional area; E: width/depth ratio from top of creek; F: area/depth^2 ratio (mean width/depth ratio); G: sinuosity ratio; H: bifurcation ratio.

In order to assess the agreement between the two pairs of readings and detect outliers, the differences were plotted on a Bland and Altman diagram (Figure 5.7). Most of the differences between parameters lie within the limits of agreement of 2 STD. The errors of omission in creek number compared with Steel's results generally increase for higher RS orders Figure 5.7A), especially for complex or small systems where the resolution of the lidar data is insufficient to detect close creeks, like Grange, Gibraltar Point and Warren Farm. Interestingly, Figure 5.7A shows errors of addition for the high RS order creeks at Banks, Shell Ness and Tollesbury that almost reach the limit of agreement: it is unclear whether this difference is due to errors in Steel's manual creek mapping due to the complexity of these systems, errors of creek detection in the algorithm, or the potential evolution of the three marshes and their creek networks since 1996.

The Bland and Altman diagram also helps to visualise systematic biases in the results yielded by the lidar creek extraction method compared with field-validated manual extraction. No visible bias was found in the detection of mean creek length (Figure 5.7B) and junction angle (Figure 5.7D). Data from both methods were in good agreement for the creek numbers (negative mean difference of 7 creeks with the algorithm, Figure 5.7A), the cross-sectional areas (mean negative difference of 1.1 m^2 Figure 5.7F) and the bifurcation ratio (mean positive difference of 0.3 Figure 5.7C). The depth of channels measured using lidar were about 0.4 m shallower than Steel's field validated results (Figure 5.7H), probably due to the presence of residual water at the bottom of creeks: this is a limitation of using near infrared lidar data which cannot penetrate water (Brzank et al. 2008) . As a result, and because creek width is overestimated when adjacent creeks are detected as one channel due to the resolution of the dataset, leading to a mean difference of 2 m (Figure 5.7G), the width/depth ratio is overestimated by 6.3 by the extraction algorithm compared to Steel's results (Figure 5.7I). Even though the values fell within those expected of intertidal creek networks, between 5 and 34 (Zeff 1999), no correlation could be found between the width/depth ratio and the RS order. However, in the case of the cross-sectional area and the mean width value (given by the area/depth), the depth underestimation had a much lower impact, and the results are close to Steel's (1996), with higher values for first RS order channels, and a positive mean difference of 1.9 for the mean width/depth ratio (Figure 5.7J).

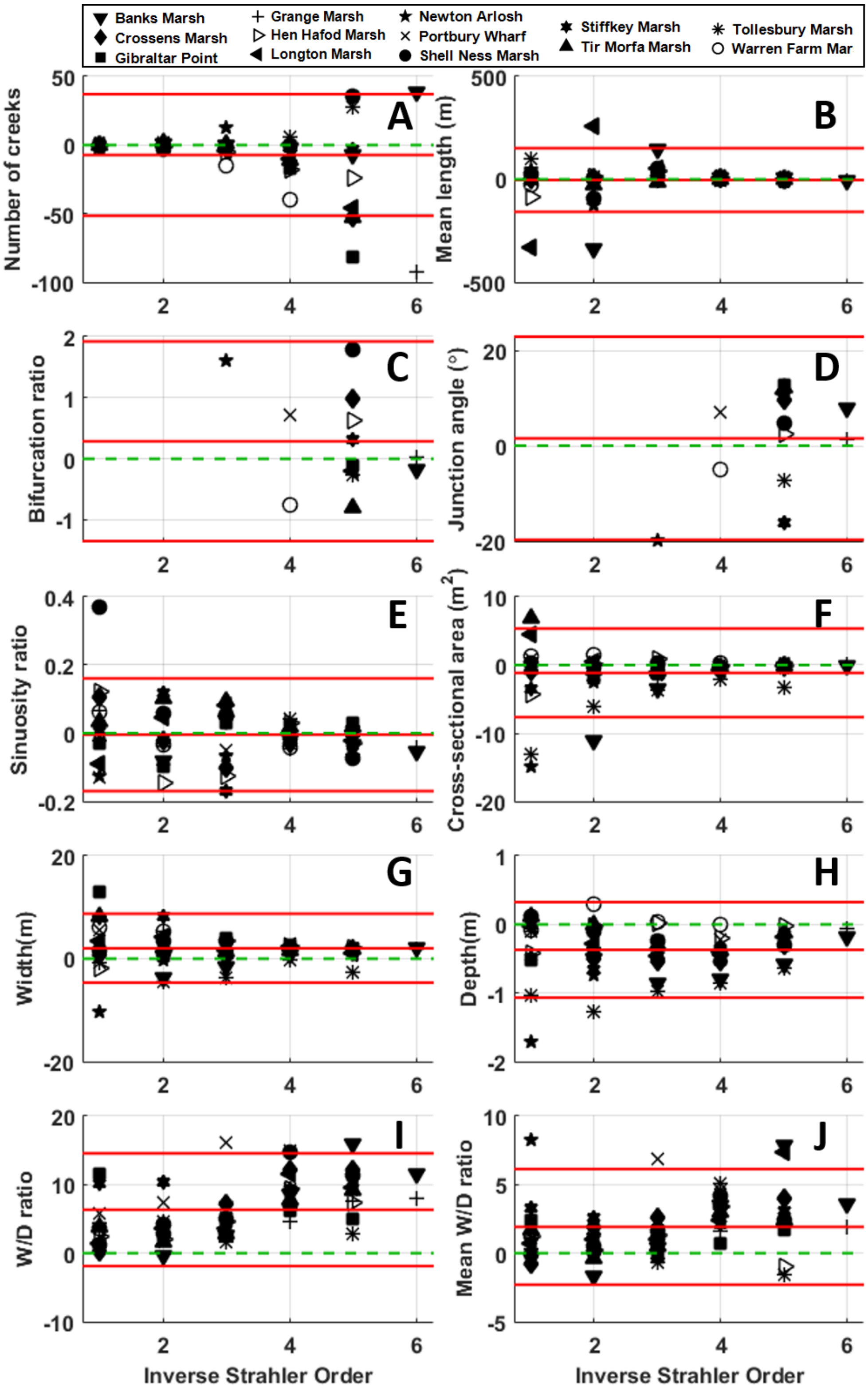


*Figure 5.7: Morphological parameter differences for each reverse Strahler order (lidar extraction results minus Steel results (1996)). A: number of creeks; B: mean length; C: bifurcation ratio; D: junction angle; E: sinuosity ratio; F: cross-sectional area; G: creek width; H: creek depth; I: width/depth ratio from top of creek; J: area/depth^2 ratio (mean width/depth ratio). Red lines show the limits of agreement (2*STD) surrounding the mean value of the differences. The dashed green line shows the ideal mean difference of 0 if there is no bias between the two methods.*

Outliers in creek length for the first and second RS orders can generally be explained by the differences in detected creek branching. Indeed, the bifurcation ratio varies between -1 and 2 depending on the creek extraction method chosen (Figure 5.7C): those variations in detected branching change the distribution of creek orders, with a potential knock-on effect on the rest of the creek system (Figure 5.8). A good example of this is the detected creek length at Longton marsh (Figure 5.7B), where the first RS order length is significantly underestimated while the second RS order creek length is overestimated: subtle differences in the detection of high RS order creeks led to vastly different characteristics of the first RS order entry channel. A similar problem is likely to occur when considering the evolution of creek networks over the years, as will be discussed in Chapter 6. However, this problem does not affect the characteristics of the whole system such as the total channel length, drainage density, total creek volume and OPL.

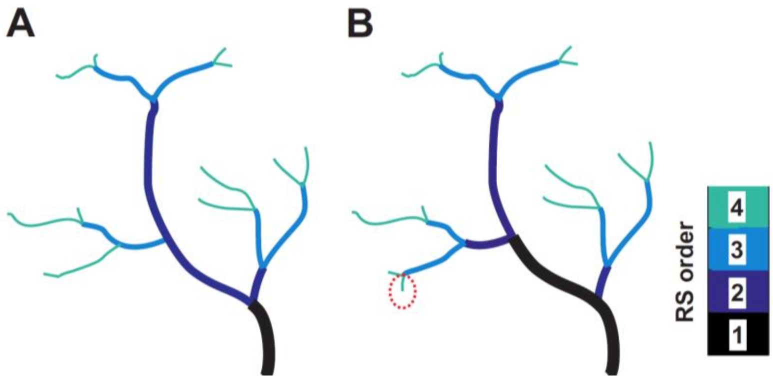


Figure 5.8: Illustration of how subtle differences in creek detection can significantly impact branching and creek order throughout the system (valid for both Strahler and Reverse Strahler ordering). Detection method A led to one 4^{th} Reverse Strahler (RS) order creek to be omitted (red dashed circle) compared with method B, with a knock-on effect on the rest of the creek system. The detected entry channel (RS order 1) is twice as long and sinuous in B than in A.

The first sub-objective was to evaluate the performance of the semi-automated creek parametrisation algorithm. Overall, there is a good agreement between the field-validated creek morphological parameters and those extracted by the algorithm. Most of the differences are linked to limits in ground detection by lidar, due to the presence of remnant water within creeks or of low-lying dense vegetation. Small differences in creek detection can have a knock-on effect on creek ordering, but the general creek characteristics such as total channel length, volume and OPL remain unaffected.

5.3.2 Applicability of existing morphological relationships

The second sub-objective is to assess the applicability and predictive value of existing creek network morphological equilibrium relationships. The relationships currently used to guide MR design (Steel 1996; Williams et al. 2002; Marani et al. 2003) were re-tested using the parameters obtained from the semi-automated extraction algorithm. Similar relationships were obtained, with correlation coefficients R > 0.60 and determination coefficients R^2 comprised between 0.38 and 0.79, all significant at a 0.05 significance level (Table 5.3, Figure 5.9). The largest discrepancies between Steel (1996)'s and the present study's equilibrium relationships' coefficients were those relating the creek mouth width to the tidal prism and to the total channel length. The algorithm underestimates the total channel length and overestimates channel width as explained in Section 5.3.1.

Table 5.3: Determination coefficients of creek morphological equilibrium established by Steel (1996) and the algorithm used for the present study.

Parameters (y vs. x)	**Steel's relationship (Steel, 1996)**	**R^2 (Steel, 1996)**	**Creek extraction relationship (algorithm)**	**R^2 p-value (algorithm)**	**Main reason for difference between the two relationships**
Total channel length (m) vs. catchment area (m^2)	$y=1.7x^{0.7}$	0.57	$y=1.5x^{0.7}$	$R^2=0.59$ $p<0.01$	Underestimation of total channel length in algorithm results (underrepresentation of small channels due to lidar resolution)
Maximum creek mouth width (m) vs. potential semi-diurnal tidal prism (m^3)	$y=0.28x^{0.4}$	0.69	$y=0.7x^{0.4}$	$R^2=0.49$ $p=0.01$	Overestimation of creek width in algorithm results (fusing of close channels into wider ones) and underestimation of creek depth (standing water in creeks)
Main creek length (m) vs. catchment area (m^2)	$y=1.5x^{0.5}$	0.70	$y=1.6x^{0.5}$	$R^2=0.71$ $p<0.01$	Underestimation of main creek length in algorithm results (underrepresentation of small channels due to lidar resolution)
Maximum creek mouth width (m) vs. total channel length (m)	$y=0.22x^{0.5}$	0.46	$y=0.61x^{0.4}$	$R^2=0.38$ $p=0.03$	Overestimation of creek width in algorithm results (fusing of close channels into wider ones); underestimation of main creek length (underrepresentation of small channels due to lidar resolution)
Mean cross-sectional area of main creek (m^2) vs. total channel length (m)	$y=0.04x^{0.7}$	0.46	$y=0.03x^{0.7}$	$R^2=0.61$ $p<0.01$	Underestimation of main creek length in algorithm results (underrepresentation of small channels due to lidar resolution)
Mean cross-sectional area of main creek (m^2) vs. catchment area (m^2)	$y=0.01x^{0.6}$	0.48	$y=0.03x^{0.5}$	$R^2=0.41$ $p=0.02$	Overestimation of creek width in algorithm results (fusing of close channels into wider ones) and underestimation of creek depth
Mouth cross-sectional area of main creek (m^2) vs. potential semi-diurnal tidal prism (m^3)	$y=0.02x^{0.7}$	0.80	$y=0.04x^{0.7}$	$R^2=0.79$ $p<0.01$	Overestimation of creek width in algorithm results (fusing of close channels into wider ones) and underestimation of creek depth

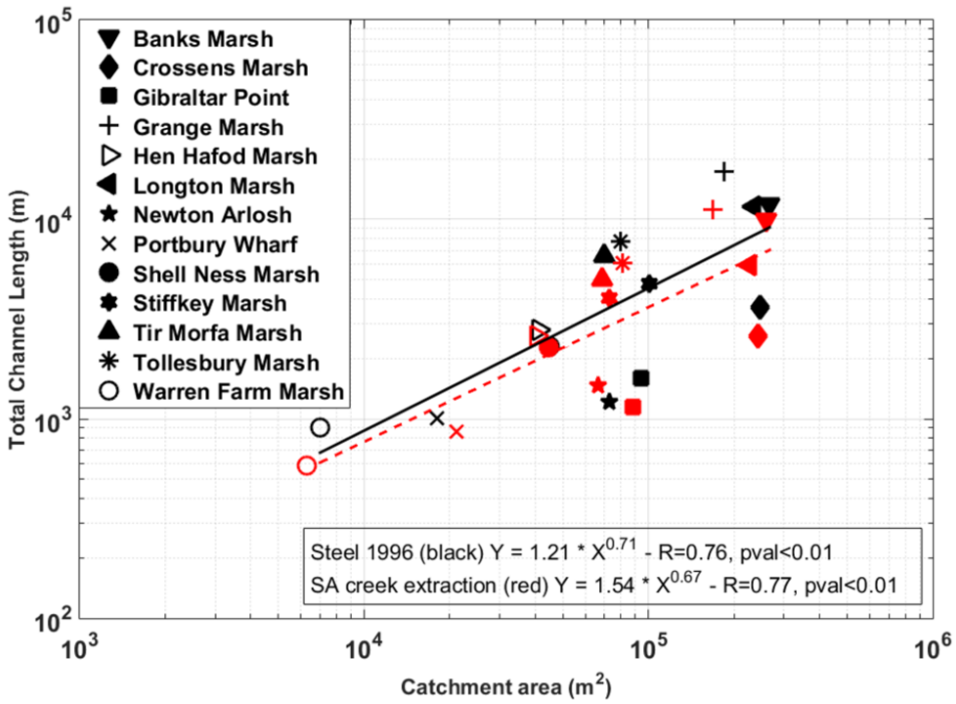


Figure 5.9: Correlation between the total channel length (m) and the catchment area (m^2) using Steel's data (1996) in black and the results from the semi-automated (SA) creek extraction algorithm in red.

Predictive values, approximated by the mean absolute percentage error MAPE, (Table 5.4), are over 25% for all morphological relationships tested. MAPE is higher for the total channel length, because the number of channels detected depends on the resolution of the dataset, and on the channel width due to the extraction error discussed in section 5.3.1. Yet, even though Steel's (1996) relationships predict higher depth values and lower width values compared to the lidar results, both projections lie within the spread of the measured data (Figure 5.10C-D). The main and total channel length are well represented by both relationships (Figure 5.10A-B). Finally, MAPE is higher for the cross-sectional versus main channel catchment area than versus tidal prism or total channel length. This suggests that catchment area is an imperfect predictor of the main channel cross-sectional area at equilibrium. In general, the low predictive values of the morphological equilibrium relationships show that they are not very effective in quantitatively predicting the final shape of the creek network based on the initial conditions. Implications for MR design will be outlined in the next section.

Table 5.4: Predictive value of the 19 morphological equilibrium relationships considered (numbered [4]-[22]) as calculated by the MAPE. Relationships are marked [P] (present study) and [B] (Steel 1996) for British saltmarshes, [V] for Venice lagoon saltmarshes (Marani et al. 2003) and [SF] for San Francisco Bay saltmarshes (Williams et al. 2002): the latter study expresses the catchment area in ha instead of m^2.

Parameters (y vs. x)	**Relationship**	**Origin**	**MAPE (%)**
Total channel length (m) vs. catchment area (m^2)	$y=1.7x^{0.7}$ [4]	[P]	86
	$y=1.5x^{0.7}$ [5]	[B]	54
	$y=0.02x$ [6]	[V]	58
Main creek length (m) vs. catchment area (m^2)	$y=1.5x^{0.5}$ [7]	[P]	38
	$y=1.6x^{0.5}$ [8]	[B]	34
Maximum creek mouth width (m) vs. potential semi-diurnal tidal prism (m^3)	$y=0.28x^{0.4}$ [9]	[P]	53
	$y=0.7x^{0.4}$ [10]	[B]	34
Maximum creek mouth width (m) vs. catchment area (m^2)	$y=3.44x^{0.552}$ [11]	[SF]	53
Maximum creek mouth width (m) vs. total channel length (m)	$y=0.22x^{0.5}$ [12]	[P]	39
	$y=0.61x^{0.4}$ [13]	[B]	37
Main channel depth (m) vs. catchment area (ha)	$y=0.91x^{0.21}$ [14]	[P]	25
	$y=1.31x^{0.202}$ [15]	[SF]	53
Mean cross-sectional area of main creek (m^2) vs. total channel length (m)	$y=0.04x^{0.66}$ [16]	[P]	31
	$y=0.03x^{0.7}$ [17]	[B]	30
Mean cross-sectional area of main creek (m^2) vs. catchment area (m^2)	$y=0.01x^{0.6}$ [18]	[P]	60
	$y=0.03x^{0.5}$ [19]	[B]	59
Mean cross-sectional area of main creek (m^2) vs. catchment area (ha)	$y=2.4x^{0.772}$ [20]	[SF]	85
Mouth cross-sectional area (m^2) vs. potential semi-diurnal tidal prism (m^3)	$y=0.02x^{0.7}$ [21]	[P]	60
	$y=0.04x^{0.66}$ [22]	[B]	30

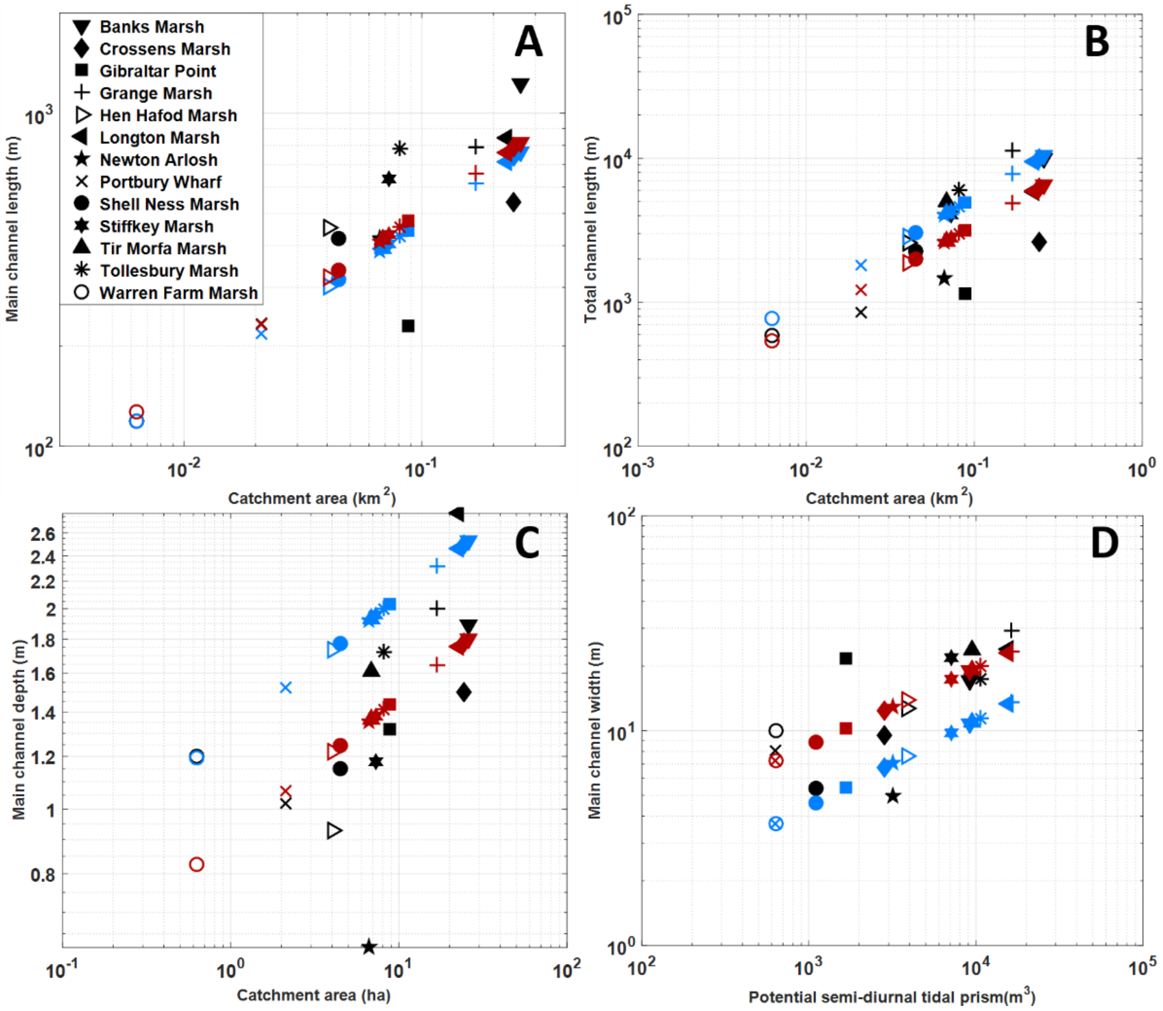


Figure 5.10: Visual comparison of the actual versus predicted data using Steel's (1996) relationships (light blue) and those established from the algorithm on the same coastal saltmarshes (dark red). Algorithm-extracted data in black. A: Main channel length versus catchment area. B: Total channel length versus catchment area. C: Main channel depth versus catchment area (Williams et al's (2002) relationships express the catchment area in ha and not m^2). D: Main channel width versus potential semi-diurnal tidal prism.

The second sub-objective was to assess the applicability and predictive value of existing creek network morphological equilibrium relationships. A mean prediction error greater than 25% was found for all morphological relationships tested. The low predictive value highlights the inherent variability of saltmarsh creek network shapes. Morphological equilibrium should thus be described as a range of potential states rather than as one quantifiable target.

5.3.3 Linking creek network drainage efficiency to environmental controls

The third sub-objective is to establish a new equilibrium relationship linking the creek network distribution within the site to initial morphological conditions and tidal forcing, to be used as a MR design tool. A new equilibrium relationship was defined relating OPL, as defined in Section 4.4.4, to initial morphological conditions and tidal forcings. A PCA was then applied to relate this new parameter to other creek morphological markers and environmental controls (Table 5.2). The aim of the PCA was to guide the selection of variables for a multiple linear regression to define a new morphological equilibrium relationship using OPL.

An exponential decay of the unchanneled length exceedance probability distribution was observed, in accordance with results from Marani et al. (2003). Faster collapsing curves correspond to better drained saltmarshes, which have lower values of OPL and a better creek distribution (Figure 5.11). There was no sign of proportionality between OPL and the catchment area or tidal prism. For example, Warren Farm and Grange had similar distance distributions to the creek network despite their size difference (Table 3.1, Figure 5.11A). The maximum unchanneled length remained below 100 m and OPL between 3 and 25 m, except for Crossens and Gibraltar Point, which display a poor creek distribution as illustrated by the comparison with Grange (Figure 5.11B-C). Crossens and Gibraltar Point are both younger than the other saltmarshes studied by Steel (1996) (Table 3.1), but age is not enough to explain their poor creek extent. Indeed four sites, Tir Morfa, Hen Hafod, Portbury Wharf and Shell Ness, were estimated by Steel (1996) to be 50 years old or less when their creek network was first mapped (Table 3.1), and do not display the poor distribution of creeks visible at Crossens and Gibraltar Point (Figure 5.11A). The limited creek development at those two locations is more plausibly tied to the sediment characteristics (coarse-grained sand) of the tidal flats: the high substrate permeability hindered the initiation of a creek network (Steel 1996), which normally fixates the creek template before vegetation colonisation (Figure 2.9A). Crossens and Gibraltar Point were considered to be at a different equilibrium state than the rest of the sites and removed from the PCA.

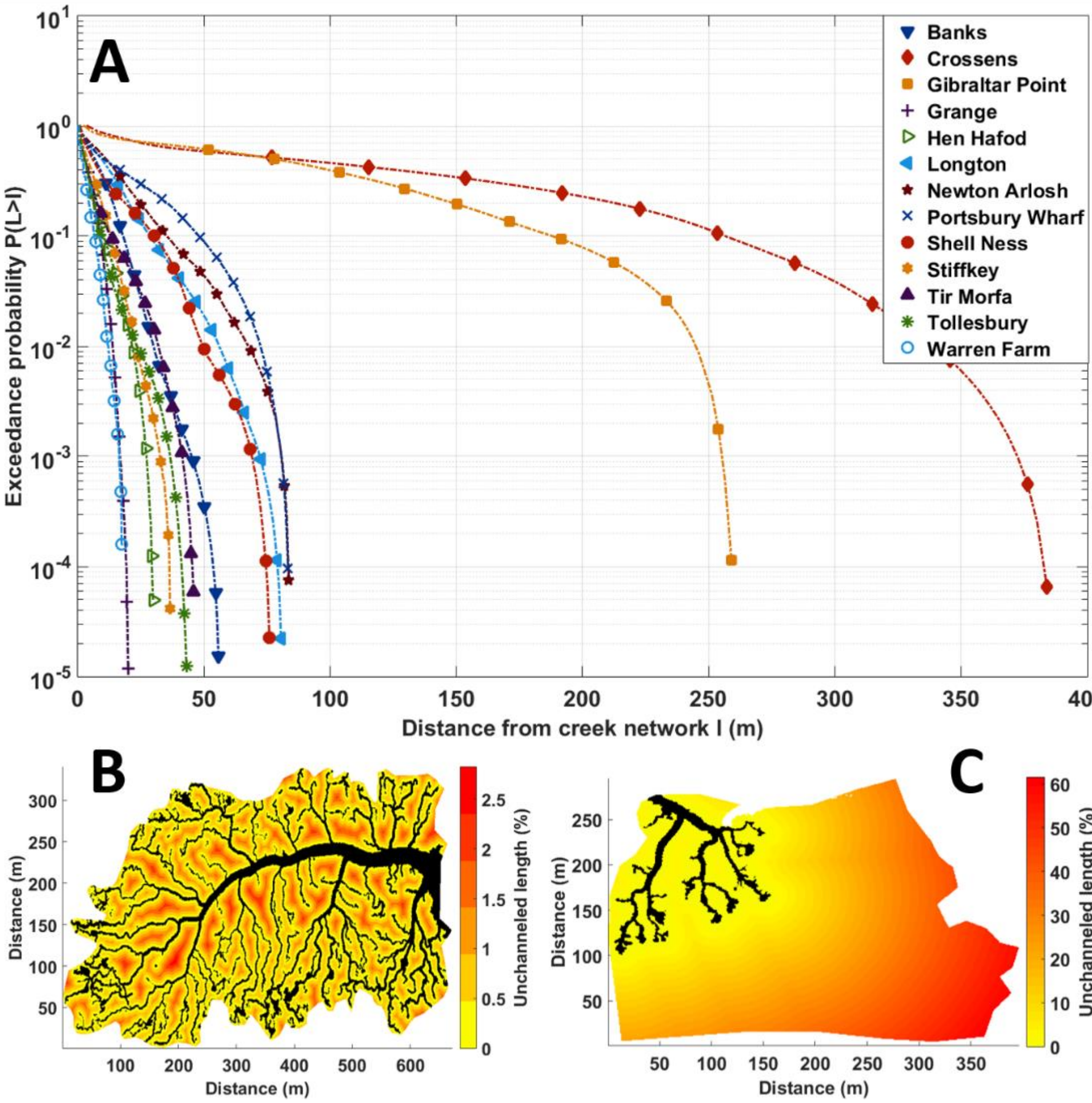


Figure 5.11: A: Unchanneled length exceedance probability distribution for all sites in meters (Slope gives OPL in meters). B and C: Unchanneled length exceedance probability distribution for Grange (B) and Gibraltar Point (C) expressed as percentage of the major axis length parallel to the main channel.

The PCA provided the contribution of each selected variable to the first three PCs, which cumulatively accounted for 78% of the total variability of the dataset (Table 5.5). The results are also displayed in a three-dimensional biplot to visually interpret the relationships between the variables (Figure 5.12). Due to missing parameters relevant to saltmarsh development (vegetation, flow velocity, number of flooding events per year, sub-surface sediment properties), PCs were hard to interpret beyond the third component. While those missing parameters are likely to play a significant role in creek development, and should be incorporated in further studies, previous modelling studies have concluded that the geomorphology and elevation within the tidal frame are the primary factors of creek growth in restored saltmarshes (D'Alpaos et al.

2007), as described in Section 2.4.2, thus justifying the choice of parameters in the present study. Input variables were split into the three PCs based on their highest absolute loadings.

Table 5.5: Loadings for the first three principal components, explaining over 75% of the total variation, following PCA applied on standardized variables for all sites except Crossens and Gibraltar Point. Groups of correlated variables are shown in blue for PC1, red for PC2 and black for PC3.

Principal Components	**1**	**2**	**3**
Catchment area	0.353	0.141	0.030
Main channel cross-sectional area	0.355	-0.002	0.181
Main channel depth	0.332	0.031	0.295
Main Channel length	0.348	0.038	-0.139
Main channel width	0.332	-0.124	0.141
Total channel length	0.354	-0.063	0.083
Total creek volume (potential tidal prism)	0.373	-0.037	0.112
Drainage density	-0.078	-0.433	0.261
Overmarsh path length (OPL)	-0.162	0.419	0.027
Mean marsh elevation above MWS	-0.015	0.467	0.189
MSTR (Mean spring tidal range)	0.016	0.454	-0.063
Tidal asymmetry	-0.216	-0.297	0.254
Main channel gradient	-0.239	0.073	0.443
Marsh age	0.059	-0.143	-0.423
Mean sinuosity ratio	0.025	-0.204	-0.400
<20µm sediment %	-0.054	-0.114	0.335
Explained variation	40.046	24.602	13.350
Cumulative explained variation	42.631	64.647	77.997

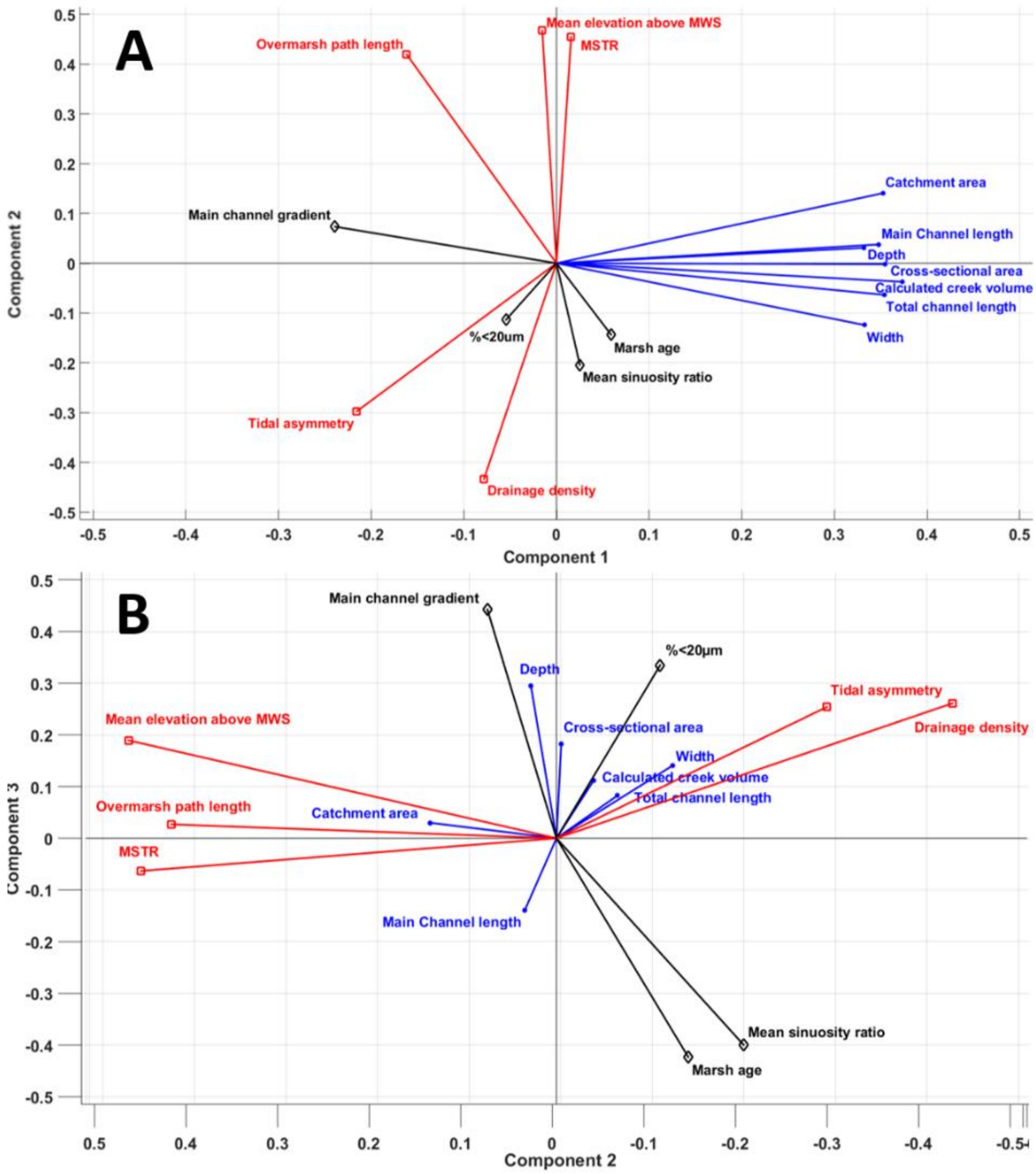


Figure 5.12: Biplot of normalized variables for all sites except Crossens and Gibraltar Point, with the first three PCs as axes (Matlab function biplot). A: contributions of the variables to PC1 (blue dots) and PC2 (red squares). B: contributions of the variables to PC2 and PC3 (black diamonds).

Principal Component 1 is dominated by size-related variables, as is typical of a first PCA component (Steel 1996). Group 1 variables identified in Table 5.2 are all positively correlated (Figure 5.12A) in agreement with traditional morphological equilibrium relationships for creek networks. They display similar loading values between 0.33 and 0.37 (Table 5.5), suggesting that they all have similar weights as a proxy of the creek network size.

Principal Component 2 relates the tidal forcings (group 2) and the markers of drainage efficiency (group 3). This suggests that the drainage efficiency of a saltmarsh at equilibrium depends more on the tidal forcings than on morphological features such as the dimensions of the main channel (group 1). A well distributed creek network, characterised by a low OPL and a high drainage density, is associated with a low MSTR, a high tidal asymmetry indicating a higher flood duration and so a greater ebb dominance in terms of velocity, which is promoted by low MSTR (Fortunato et al. 2005), and critically by a low elevation above MWS. Indeed, the higher the site within the tidal frame, the fewer flooding events will reach the saltmarsh, reducing both hydrodynamic energy and sediment supply. This can hinder creek network development or even cause its contraction as seen in the conceptual model (Steel 1996), thus reducing the drainage efficiency (Figure 2.9).

Finally, Principal Component 3 relates the creek maturity variables (group 4) with the sediment stability (group 5). Higher concentration of cohesive material restricts the growth of the creek system by increasing the stability of the channels against erosion (Steel 1996), which leads to lower creek sinuosity even in older marshes. This seems an accurate representation of controls on channel sinuosity, if simplistic: in saltmarshes, the channel sinuosity will also depend on flow conditions and vegetation cover (Zeff 1999).

The third sub-objective was to establish a new equilibrium relationship linking the creek network distribution within the site to initial morphological conditions and tidal forcing, to be used as a MR design tool. The PCA facilitated the selection of variables for a multiple linear regression and allowed the definition of morphological equilibrium relationships using OPL. The best relationship found relates OPL with two independent variables: the elevation within the tidal frame and the mouth cross-sectional area (equation [23], Figure 5.13A). The MAPE is 50.6% and the determination coefficient R^2=0.80 (Figure 5.13B). Another statistically significant relationship was found between OPL and the elevation within the tidal frame (equation [24], Figure 5.13).

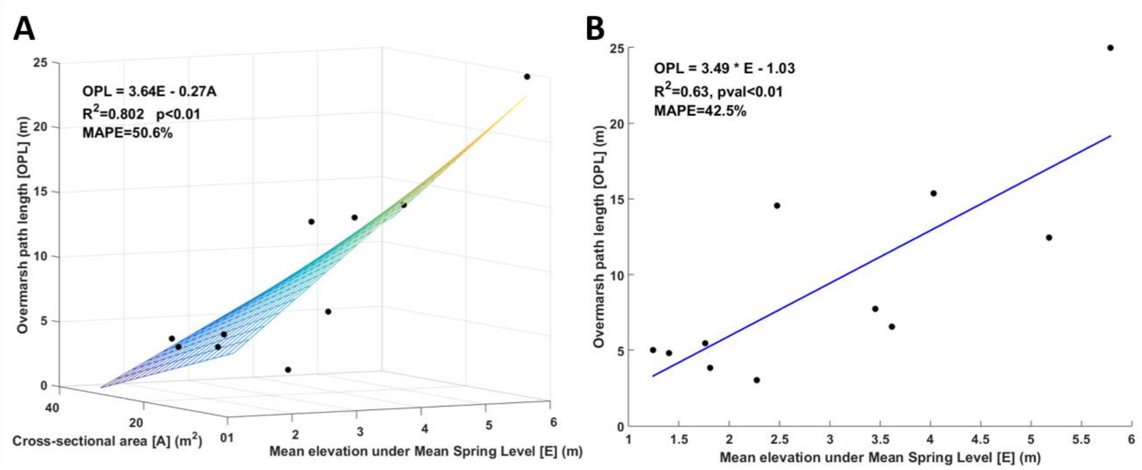


Figure 5.13: A: Morphological relationship relating the overmarsh path length (OPL) to the mean marsh elevation relative to mean spring level (E) and to the cross-sectional area of the entry channel mouth or breach (A). Equation [23] OPL = 3.64 E – 0.27 A. B: Morphological relationship relating the overmarsh path length (OPL) to the mean marsh elevation relative to mean spring level (E). Equation [24] OPL = 3.49 E -1.03.

5.4 Discussion

The first sub-objective was to test the efficiency of the newly developed creek parametrisation algorithm using lidar data, by comparing results with those from field-validated studies. (Mason et al. 2006). The method yielded accurate creek network shapes for all 13 studied sites (Figures 5.1-5.4). The preprocessing phase was minimal, which avoided loss of data through smoothing. Most of the errors observed, such as the underestimation of channel depth, could be retraced to data collection, such as the reflection or absorption of the laser beam by residual water in the channels (Mason et al. 2006). With a running time of a few seconds to a few minutes, the algorithm proves to be very time-efficient compared with the time-consuming and subjective process of manual mapping.

The creek extraction algorithm is also the first to yield as output a comprehensive suite of morphological parameters for each creek order. An exponential increase in creek number and decrease in creek length was found with increasing RS order (Figure 5.6A-B), meaning that there are exponentially more small shallow creeks than large deep ones in a creek system. This finding is in agreement with the laws of drainage composition established by Horton (1945) on fluvial networks, and with previous studies on intertidal creek network compositions (Steel 1996; Zeff 1999). Consequently, this algorithm can be used to monitor saltmarsh restoration sites to verify the creek system's compliance with natural drainage compositions, and to monitor rapidly evolving natural systems under the influence for instance of subsidence or sea-level rise. Lidar is expected to become more and more efficient as a monitoring tool as the frequency, resolution and accuracy of data collection increases. It should however be noted that subtle variations in creek branching, either linked to the detection method used or to the natural evolution of the creek system over time, can change the distribution of creek orders and significantly impact the detected parameters for the entry channel and other low RS order creeks. Caution should be applied when interpreting order-specific parameters in rapidly evolving creek systems. This problem does not affect the characteristics of the whole system such as its total channel length, drainage density, total creek volume and OPL.

Other automated methods have been developed recently based on an eight direction enhancement and detection of Gaussian shapes to extract creek networks (Liu et al. 2015) . Their results outperformed the traditional threshold methods in the case of a tidal creek network covering an area of 5,000 km^2, for creek networks as small as 5 m wide. However, the applicability of this method to artificial creek networks, which are not always characterised by Gaussian-shaped cross-section profiles like natural channels, is unclear. An interesting extension of this work would be to apply their method to the 13 British saltmarshes to test whether the error in channel width extraction observed in the algorithm-generated results is reduced (Figure 5.7G). A notable advantage of the present study’s algorithm is that, while creek detection relies on traditional threshold methods, the more novel creek parametrisation phase can be applied to the output of other creek extraction techniques, as long as they provide a connected creek mask.

The second sub-objective was to determine the applicability of existing morphological equilibrium relationships to lidar-extracted data. In Steel’s (1996) study, the creeks mapped were validated during field surveys as being all deeper than 0.2 m and longer than 2 m. Given the 0.15 m vertical and 1 m horizontal resolution of the available lidar datasets, the smallest features that could be reliably identified were channels of 2 m length, 2 m width, and 0.3 m depth. Furthermore, channels less than 2 m apart often appeared fused in the lidar data. These resolution limits were identified as the cause for the small errors of omission observed when comparing our results with Steel's (1996) for both creek number and length (Figures 5.1-5.4). Yet, despite those omissions, the algorithm accurately detected the creek system composition, even though the development phase of the creek network may be underestimated due to the resolution used.

The systematic depth underestimation was interpreted as an error inherent to lidar data collection, as the laser does not penetrate water (Brzank et al. 2008). The width of creeks was overestimated when adjacent creeks were fused due to the resolution of the dataset, causing the width/depth ratio to be systematically overestimated (Figure 5.7G). Using the area/depth^2 ratio as an approximation of the mean segment width lowered the impact of width overestimation and gave closer results of mean width/depth ratio to those found by Steel (1996) (Figure 5.7I-J). As some MR schemes guidelines recommend reproducing the width/depth ratios of similar natural systems (Zeff 1999), when monitoring the channel shape the mean width/depth ratio given by area/depth^2 should be used to lessen the error associated with creek width detection in 1 m resolution lidar data.

The relationships were also qualitatively in accordance with those established in previous studies using aerial photography and field surveys in the Venice lagoon (Marani et al. 2003) and San Francisco Bay (Williams et al. 2002). However, the MAPE was over 25% for all morphological relationships tested. This is due to the inherent variability of creek network shapes, and implies that morphological relationships are not effective in quantitatively predicting the final shape of the creek network based on the initial conditions. They can, however, provide semi-quantitative trends suggesting how design choices may accelerate or hinder creek evolution towards an equilibrium range. In Chapter 7 of this thesis, these relationships will be used to determine how close to morphological equilibrium creek networks in current MR schemes have gotten after up to 20 years of evolution in order to discuss the efficiency of their design.

Considering this limitation, the predictive value of the morphological relationships did not vary significantly with data collection site: the British saltmarshes studied display tidal ranges ranging from mesotidal to hypertidal, and environments ranging from estuarine back-barrier to open coast saltmarshes (Steel 1996). By contrast, the creek networks analysed to obtain equation [6] in Table 5.4 were developed in the microtidal environment of the Venice Lagoon (Marani et al. 2003), and those studied to obtain equations [11], [15] and [20] in Table 5.4 developed in San Francisco Bay under a micro- to mesotidal regime (Williams et al. 2002). The results tend to confirm the hypothesis that the trends indicated by those relationships are site-independent (Coats et al. 1995; Marani et al. 2003). This provides further evidence that these relationships can be used for saltmarsh restoration projects across the world.

The morphological relationships selected to study MR creek network evolution towards equilibrium in Chapter 7 are the ones with a prediction error under 40 %, except for relationships involving the channel width, as this parameter is extracted with the least confidence by the algorithm (Figure 5.7G). This corresponds to equations [8] (main creek length versus catchment area), [14] (maximum creek mouth depth versus catchment area), and [22] (Mouth cross-sectional area of main creek versus potential semi-diurnal tidal prism) in Table 5.4. The total channel length versus catchment area (equation [5], Table 5.4) will also be used despite its weak predictive value because it is expected to be very sensitive to creek network development (Hampshire 2011) .

The third sub-objective was to establish new equilibrium relationships linking the drainage efficiency of the creek network to initial morphological conditions and tidal forcing. OPL, by giving the mean distance to the creeks within the saltmarsh, was found to be a relevant descriptor of drainage efficiency. The studied natural saltmarshes at equilibrium displayed no correlation between OPL and the catchment area, which fits with findings from Marani et al. (2003). OPL values range between 3 and 25 m, except for the two anomalous sites Crossens and Gibraltar Point. This characteristic is important for plant diversity, as tidal channels have an influence on vegetation within 20 m away from the channel, with a greater plant diversity found within the first 10 m (Sanderson et al. 2000). OPL values significantly over 25 m could be detrimental to saltmarsh plant diversity, and OPL should thus be treated as a critical design factor for MR sites, in accordance with Haltiner et al. (1987)'s creek distribution design guidelines (Table 2.1).

In an attempt to guide future design, a new equilibrium relationship was found relating OPL with two independent variables: the average elevation within the tidal frame and the mouth cross-sectional area (equation [23], Figure 5.13). This is in accordance with previous studies stating that marsh elevation within the tidal frame is the main control of the tidal forcing parameters (Reed et al. 1999) and of the sedimentation rates (French 2006). In terms of MR design, this relationship indicates that the drainage efficiency of the creek network at equilibrium is higher when the scheme is more open to tidal influence (low within the tidal frame with a large site opening); this also fits with the PCA results which found the drainage efficiency to be dominantly correlated to the tidal forcings (Figure 5.12A). The MAPE values for the two equations are lower than for equation [5], and with R^2 value of 0.63 and 0.80 respectively, they are considered suitable tools to assist in the monitoring and design of creek networks in MR schemes. Preliminary testing found equation [24] to be more convenient than [23] for MR creek evolution monitoring: indeed, the mouth cross-sectional area in MR schemes generally corresponds to a breach in a seawall, with much greater values than a natural channel outlet, making the creek evolution in relation to equation [23] difficult to interpret.

MR scheme designers need advice on creek network design, which the morphological relationships validated in this analysis can provide. However, the applicability of this chapter's findings is limited by the relatively small number of sites (13) considered. Indeed, the study was limited to sites where morphological parameters had already been extracted by a field-validated study; the dataset collected by Steel is the most complete in that regard. Still, the relationships were similar to those found in previous studies, which analysed 12 marshes in San Francisco Bay and 20 marshes in Venice Lagoon, respectively (Williams et al. 2002; Marani et al. 2003), suggesting an applicability of the equilibrium relationships beyond the British saltmarshes considered here.

5.5 Summary

The new semi-automated parametrisation algorithm successfully extracted morphological characteristics of creek networks for 13 British saltmarshes. The efficiency of existing morphological relationships has been tested for lidar-extracted data. A mean prediction error greater than 25% was found for all morphological relationships tested, highlighting the inherent variability of saltmarsh creek network shapes, and that morphological equilibrium should be described as a range of potential states rather than as one quantifiable target. Morphological relationships can be used to monitor MR creek evolution towards this equilibrium range. In addition to existing equilibrium relationships monitoring the channel length, depth and cross-sectional area, the overmarsh path length (OPL) is a valid proxy of the creek network's maturation stage. OPL represents the spatial distribution and drainage efficiency of the creek network, and is a relevant addition to the planimetric parameters already in use to determine morphological equilibrium. The following relationships will be used for further analysis of artificial creek networks in MR schemes:

Total channel length (m) vs. catchment area (m^2):	$y=1.5x^{0.7}$	**[5]**
Main channel length (m) vs. catchment area (m^2):	$y=1.6x^{0.5}$	**[8]**
Main channel depth (m) vs. catchment area (ha):	$y=0.91x^{0.21}$	**[14]**
Mouth cross-sectional area (m^2) vs. tidal prism (m^3)	$y=0.04x^{0.66}$	**[16]**
OPL (m) vs. mean marsh elevation above MWS (m)	$y=3.49x-1.03$	**[24]**

The next chapter will apply the methods and relationships validated in this chapter to monitor the evolution of creek networks in MR schemes, to address Thesis Objective 2.

Chapter 6: Parametrising tidal creek morphology in managed realignment schemes

6.1 Introduction

Despite the growing number of MR schemes in the UK, case studies of MR creek network evolution remain sparse. Yet, given the crucial role played by creeks for sediment exchanges and plant growth, differences in biodiversity found between natural systems and MR schemes could be linked to differences in the creek morphology. However, in order to compare creek and ecological evolution of MR schemes, preliminary knowledge of MR creek composition and growth rates is necessary. To improve the state of knowledge and lay the ground for such multidisciplinary studies, this chapter will conduct a systematic monitoring of morphological changes occurring within 10 UK MR creek systems, described in Section 3.4. The study of two century-old AR sites, described in Section 3.5, provide a window of observation into the longterm evolution of creek systems in realigned marshes.

Previous monitoring of MR schemes has often been limited to bird and vegetation surveys, and little information has been gathered on their morphodynamic evolution (Esteves 2013). This gap is in part due to the complexity of mapping creek systems, as discussed in Section 2.4.4. The methodology developed in Chapter 4 and tested on natural creek systems in Chapter 5 facilitates this monitoring phase and allows for a comprehensive monitoring of MR creeks. Another limit to the morphodynamic monitoring of MR schemes is the incapacity of lidar data to penetrate low-lying saltmarsh vegetation, as discussed in Section 3.2.1. Since current vegetation removal algorithms do not reliably detect the bare ground, distinguishing between plant growth/die-off and sedimentological processes comes down to post-processing interpretation. The known morphodynamic processes at play in creek initiation and growth are summarised in Section 2.4.2. To what extent these processes can be discerned from interannual lidar datasets, or masked by the effect of vegetation growth, is still unclear.

This chapter addresses Thesis Objective 2: to undertake a morphometric analysis of creek systems in MR schemes in the UK and of their evolution over the years, in order to estimate creek evolution rates. To that end, there are four sub-objectives as follows:

(1) To monitor the changes in MR marsh elevation distribution and creek network morphological characteristics over 5 years of evolution or more;

(2) To estimate whether inter-annual creek evolution trends can be inferred from various morphological parameters;

(3) To discuss the reliability of these creek evolution trends by identifying the respective roles of vegetation, vertical accretion and creek-forming processes on lidar-detected marsh elevation changes;

(4) To analyse the evolution trends occurring during the last 10 years in century-old, accidentally breached sites to get some insight into the potential long-term evolution of MR creeks.

The structure of this chapter is as follows. Section 6.2 provides the methods specific to this chapter. Section 6.3 lists the results for each sub-objective. Section 6.4 discusses the implications of the results and their limitations, and Section 6.5 provides a summary of the key findings.

6.2 Methods

6.2.1 Catchment area delimitation

The boundaries of MR schemes are defined slightly differently than those of the natural coastal wetlands defined in Section 4.3, as they are typically constrained at least partially by seawalls. However the dominant principles remain the same. The MR landward limit is still defined by the local HAT level, corresponding to the limit between the tidally influenced saltmarsh and the land (Steel 1996). This limit occurs either within the naturally-occurring slope (Figure 6.1A) or at the flood defence (Figure 6.1B), whichever is lowest. The seaward limit is defined as the mouth of the entry channel that serves as the outlet for all the considered creek networks, which generally coincides with one or several breaches (Figure 6.1). The area comprised within those limits is referred to as the catchment area in the rest of the thesis.

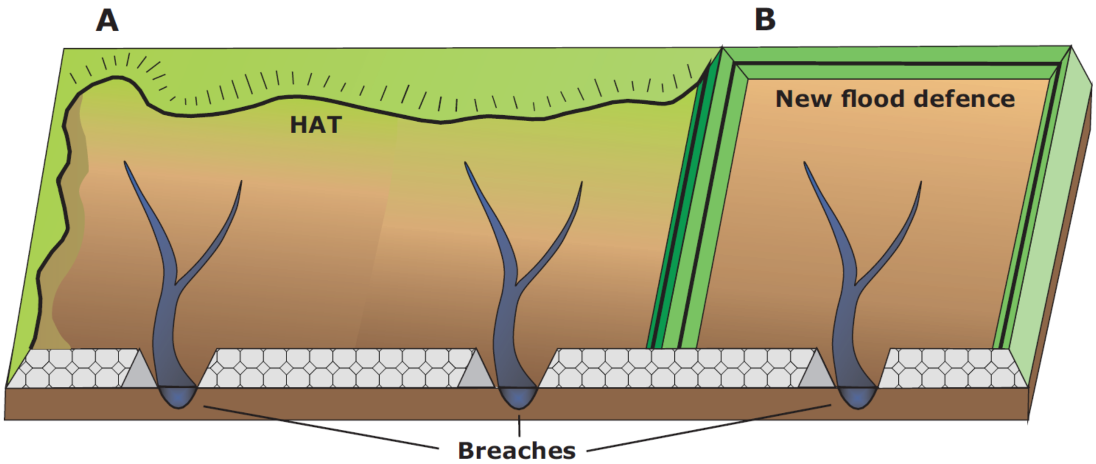


Figure 6.1: Schematic view of a MR scheme and its catchment boundaries (thick black line: HAT level). A: MR catchment area delimited by the natural topography landward (no new defence). B: MR catchment area constrained by flood defences.

Furthermore, while the saltmarshes studied in Chapter 5 have only one outlet, MR schemes are often characterised by multiple creek systems, each linked to a different breach (Figure 6.1A). Previous studies have avoided this issue by splitting MR sites into separate catchment areas for each breach (Hampshire 2011), defining boundaries as the half-way distance between each creek network following Steel's definition (Steel 1996). However, this method contains uncertainties because creeks inherited from drainage ditches often link different breaches together, as is the case at Freiston, Allfleet and HOMW (Figures 3.21, 3.24 and 3.27, respectively).

Alternatively, MR schemes can be compared to a "marsh island", defined as a saltmarsh area that remains surrounded by water during low tide. A marsh island may encompass several creek systems, and yields similar power law relationships between total channel length and catchment area than for a single creek system watershed (Novakowski et al. 2004). A marsh island can be "dike adjacent", as is the case for MR schemes (**Error! Reference source not found.**A), and obey the same rules (Hood 2007). The applicability of the morphological equilibrium relationships explored in Chapter 5 was tested for a natural marsh island near the Stiffkey natural marsh, and for a dike-adjacent marsh island, containing Stiffkey and delimited southwards by a footpath and terrestrial vegetation (Figure 6.2). North Norfolk marshes were selected as well-known mature saltmarsh habitats (Steel 1996).

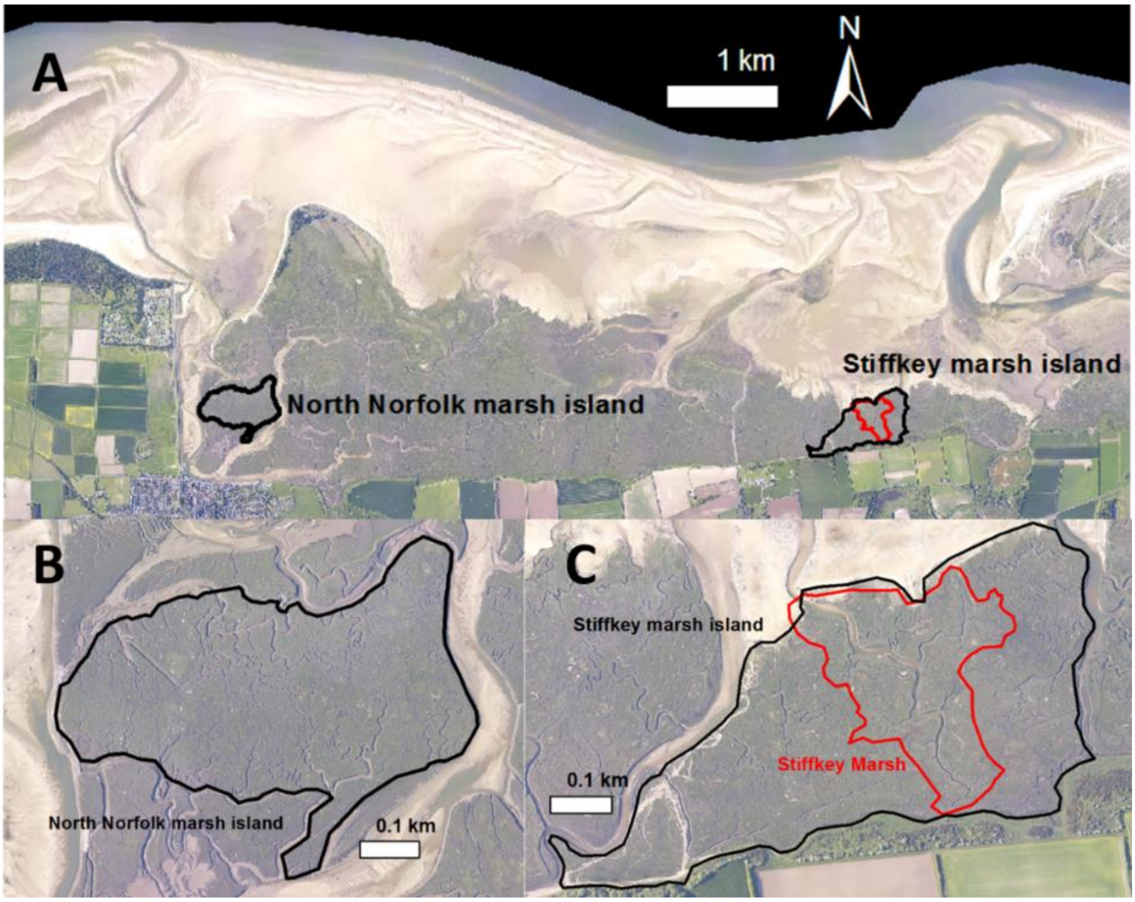


Figure 6.2: A: Location of marsh islands (black) near Stiffkey (red). B: close-up of North-Norfolk marsh island. C: close-up of Stiffkey dike-adjacent marsh island. Photo source: 20 cm resolution composite image from two flights on the 06/05/2016 and the 23/09/2016, Environment Agency.

Stiffkey Marsh, North Norfolk marsh island and Stiffkey dike-adjacent marsh island yielded similar unchanneled length distributions and similar values of OPL (Figure 6.3). They also fit on the same equilibrium lines for the total channel length (Figure 6.4A) and OPL (Figure 6.4B). This indicates that single creek systems and marsh islands, dike-adjacent or not, follow the same morphometric relationships at equilibrium. It is then possible to compare the creek parameters of MR schemes with those of Steel's natural sites. A key issue is to decide whether to apply the equilibrium relationships to the main channel length and mouth cross-sectional area of all outlets, or of the largest outlet only. Preliminary testing performed by this study suggests that the catchment area of marsh islands is at equilibrium with the largest main channel length rather than with the sum of all main channels (Figure 6.4C). Conversely, the undermarsh tidal prism is expected to be at equilibrium with the sum cross-sectional area of all outlets, as it represents the total water exchange (Figure 6.4D).

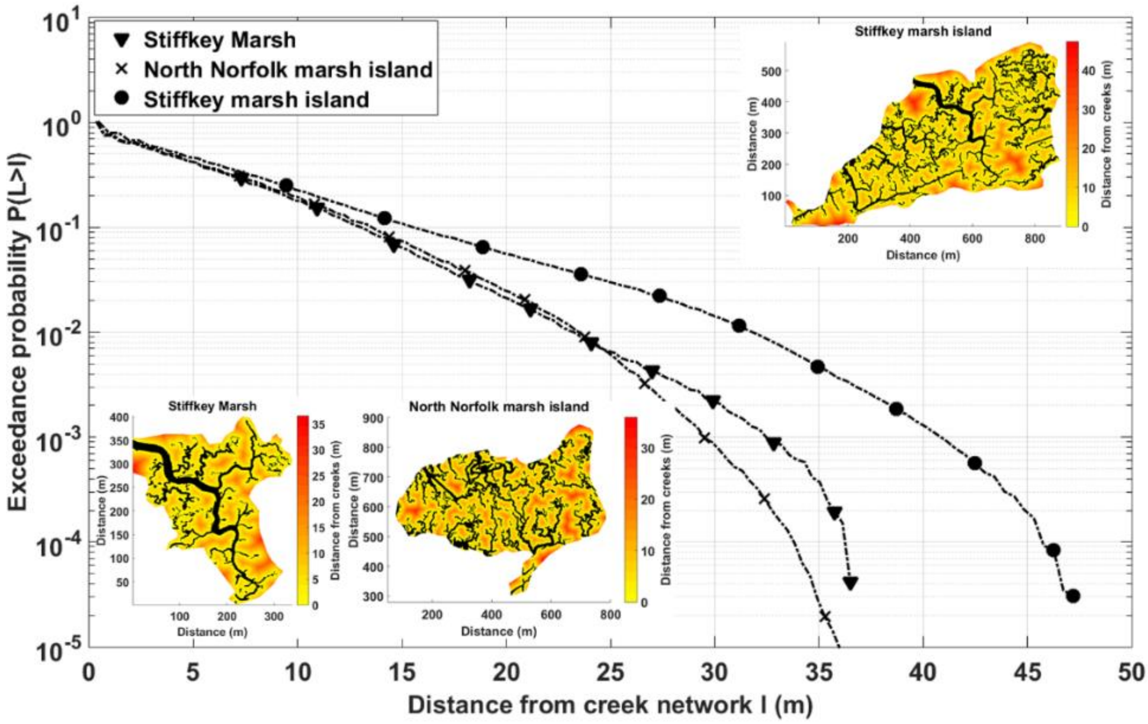


Figure 6.3: Distance from creek distribution at Stiffkey Marsh (Steel 1996), North Norfolk marsh island and Stiffkey dike-adjacent marsh island.

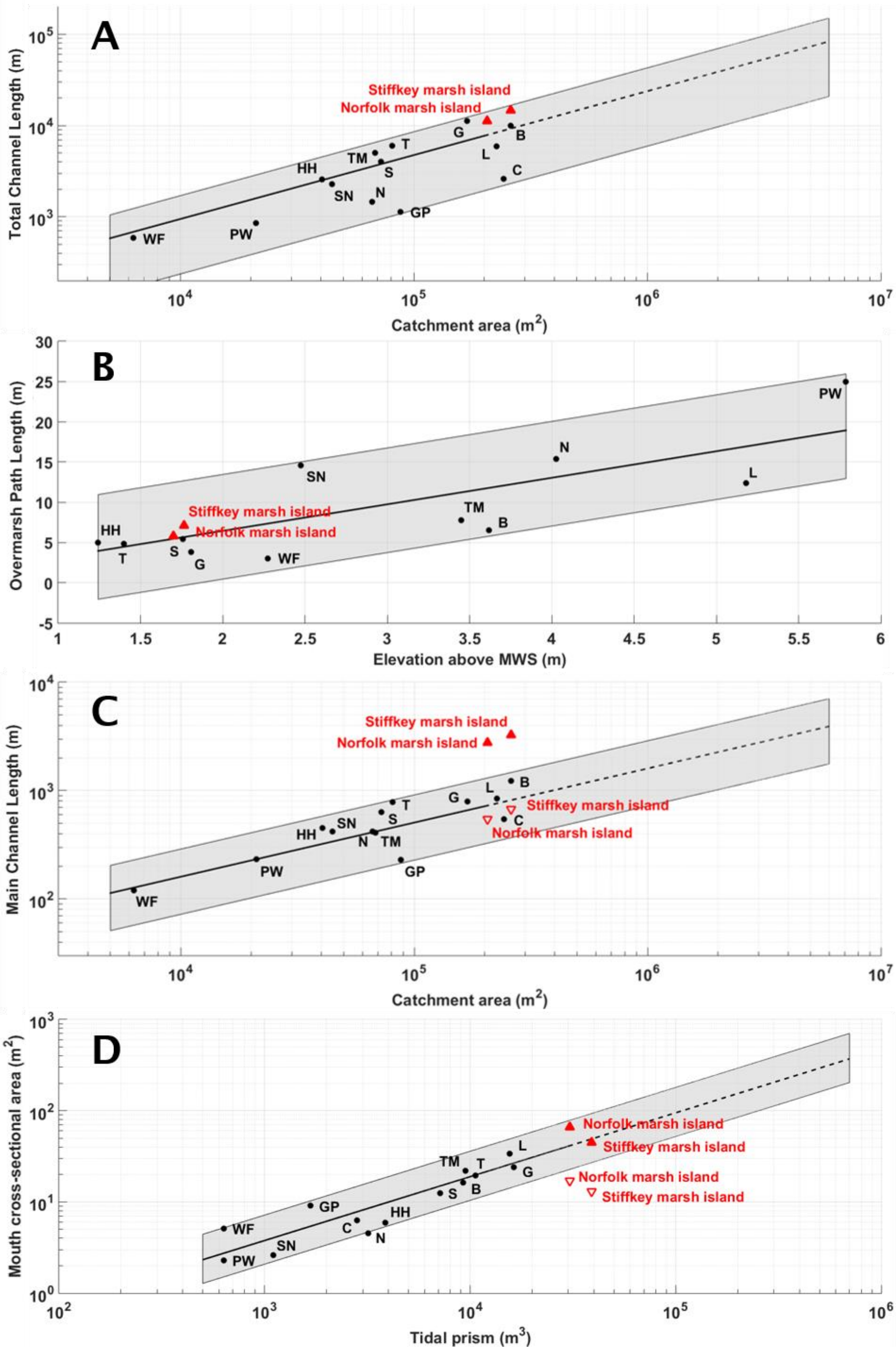


Figure 6.4: Applicability of morphological equilibrium relationships to marsh islands (dike-adjacent or not). A: Total channel length vs catchment area. B: OPL vs elevation above MWS. C: Main channel length vs catchement area (filled triangles = sum length of all entry channels; empty triangles = length of the largest channel). D: Mouth cross-sectional area vs tidal prism (filled triangles: sum of cross-sectional areas for all outlets; empty triangles: cross-sectional area of the largest outlet). B=Banks; C=Crossens; GP=Gibraltar Point; G= Grange; HH=Hen Hafod; L=Longton; N=Newton Arlosh; PW=Portbury Wharf; SN=Shell Ness; S=Stiffkey; TM=Tir Morfa; T=Tollesbury; WF=Warren Farm

The boundary conditions of the MR sites considered will thus be defined as follows: the landward and lateral limits are marked by the HAT level, occurring either at the flood defence or within a naturally occurring slope towards the terrestrial environment. The seaward limit is located at the breach, breaches or at the location of dike removal. The area constrained within these limits may contain several creek systems linked to several breaches, and so should behave like a dike-adjacent marsh island.

6.2.2 MR creek parametrisation

In order to fulfill the first sub-objective, lidar data were processed as detailed in Chapter 4 to monitor the changes in MR marsh elevation distribution and creek network morphological characteristics post-implementation. Elevation and slope thresholds were found through sensitivity tests, made possible by the short (seconds to minutes) computational running time of the algorithm. The elevation thresholds LZth and HZth were selected for the initial year then increased over the years following the accretion rate, to ensure a more coherent and less subjective detection of creek evolution than manual mapping. The following parameters were computed for each site:

(1) changes in elevation distribution over the years (in the absence of vegetation data, the potential mudflats, low marsh and high marsh are shown strictly as a function of marsh elevation within the tidal range: the saltmarsh – mudflat transition generally occurs at MHWN, while MHW marks the transition between the lower marsh, where most creeks are expected to form, and the middle marsh (Figure 2.1), shown as maps and hypsometry curves;

(2) the width, depth and cross-sectional area of the largest breach for each available year, measured by applying the same cross-section line to each year near the largest MR breach, and delimited by the creek network mask;

(3) the RS order of each creek segment in the first and last available year;

(4) the morphometry parameters per RS order plotted against the 95% spread of natural creek parameters for each available year; and

(5) the unchanneled length distribution and creek extent evolution maps for each available year, which are used to calculate OPL, as described in Section 4.4.4.

6.2.3 MR and AR creek evolution trends

In order to fulfill the second and fourth sub-objectives, a linear fit was tested at a 95 % confidence interval to infer creek evolution trends for each MR and AR site, for all normalised parameters listed in Table 6.1. To ensure their relevance as proxies for morphodynamic processes, the parameters selected are either those already used in morphological equilibrium relationships (TCL, OPL, mouth CSA, creek volume, mouth depth), or parameters that were used as markers of creek extent and maturity in the PCA in Chapter 5 (creek magnitude, planform area, main channel gradient and sinuosity ratio). The main channel is taken as the longest channel connected to the largest outlet, and the mouth cross-sectional area as the sum of all outlets following observations of marsh islands from Section 6.2.1.

An analysis of residuals was also performed to check whether the evolution trends exceed the inter-annual variations of the creeks, and so whether the studied creek parameters are following a trend rather than fluctuating around a mean value. The evolution trends were normalised by their initial parameter to allow for comparison between sites of various sizes. The creek evolution rates were then expressed as the slope of the linear fit taken from the normalised evolution trend in %/yr. The only exception is OPL for which, because it decreases over time in an expanding creek network, the evolution rate was given in -%/yr so positive evolution rates correspond to a more rapidly developing creek system.

Table 6.1: List of parameters for which evolution rates are inferred

Creek parameters for which evolution rates are inferred	**Symbol**
Mean elevation above MWS (%/yr)	MWS
Drainage density (%/yr)	DD
Overmarsh path length(OPL) (-%/yr)	OPL
Main channel length (%/yr)	MCL
Total channel length (%/yr)	TCL
Number of creeks (%/yr)	NB
Total mouth cross-sectional area (%/yr)	CSA
Main channel mouth depth (%/yr)	D
Planform area (%/yr)	PA
Undermarsh tidal prism (creek volume) (%/yr)	TP
Sinuosity ratio (%/yr)	SR
Main channel gradient (%/yr)	MCG

6.2.4 Relation to creek-forming processes

Finally, in order to fulfil the third sub-objective, the elevation changes within the site were investigated in more detail and in relation to the distance to the creeks. As discussed in Section 2.4.2, morphodynamic processes at play in creek evolution are due to local bed shear stress exceeding a sediment resuspension threshold. This leads to erosional processes like channel down-cutting, creek widening and headward erosion of new channels into the marsh substrate. Depositional processes along the paths of weaker flow also lead to creek growth by building up the surrounding levees. As a result a positive correlation is expected between accretion rates and distance to the creek (Stumpf 1983).

However, in MR creeks, which result from human excavation rather than hydrodynamic processes, the mechanisms of creek growth are less clear. Elevation changes along creeks as detected by lidar can capture some of these processes, but they are likely to be mixed with other non-geomorphic processes such as saltmarsh vegetation growth or terrestrial vegetation die-off following flooding by estuarine water. Reactivation of MR site features like drainage ditches can also occur. Thus, to better distinguish between those processes, and in the absence of systematic aerial photography collection for each lidar flight, the datasets were separated into:

(1) **Positive elevation changes (elevation gains):** these changes will be due to either sediment deposition leading to levee building or plant growth. While the two factors cannot be easily distinguished using lidar, they should be positively correlated: sediment deposition is encouraged by the trapping action of plants, which grow in higher densities near creeks (Christiansen et al. 2000). This action is thought to be significant up to 20 m away from the creeks (Stumpf 1983). Another potential factor of elevation gain is human activity, in the rare case where further site reprofiling has occurred after site implementation. Those can be identified visually as forming distinct objects (clear contours) rather than a cloud of points.

(2) **Negative elevation changes (elevation loss):** the lowering of the initial site surface after MR implementation implies either vegetation die-off, creek-forming erosional processes (channel down-cutting, headward erosion, bank slumping), or down-cutting and reactivation of other features like drainage ditches. Vegetation die-off is likely to be more dispersed than creek erosion, which will display characteristic dendritic or linear patterns. The effect of channel down-cutting can be reduced by removing the initial creek mask from the dataset, leaving only the down-cutting of reactivated features to interpret: these are likely to be straighter than the headward erosion patterns, as they occur within

drainage ditches. Another potential factor of elevation loss is human activity, as discussed in the previous paragraph. Of particular interest for this study is the occurrence of headward erosion, which shows a capacity of the creek network to expand into the substrate. This process is important in MR schemes characterised by a high initial elevation and low sediment fluxes into the scheme, as these conditions will hinder the deposition-driven growth of creek networks.

Maps of positive and negative elevation changes between first and last year of lidar data collection were produced for each site. Linear correlations between elevation changes (positive and negative) and distance to the creeks were then tested at a 95 % confidence interval. The distance to the creeks was computed using the Matlab function *bwdist*, which finds, for each grid cell, the Euclidean distance to the nearest pixel belonging to the creek mask. The initial creek mask was compared to the elevation losses to make erosive processes following site implementation more visible. The latest creek mask was compared to the elevation gains to capture the effects of creek proximity on levee building or vegetation growth.

Finally, linear correlations were tested at a 95 % confidence interval between the initial marsh elevation and the marsh elevation changes outside the creek network. This analysis aims to verify the common observation in natural saltmarshes that accretion rates remain dominantly driven by the elevation of the site within the tidal frame (Friedrichs et al. 2001). MR schemes are expected to obey the same principles (Leung 2017). However, some differences may occur because they are constrained by flood defences, which can lead to a bigger water column above the site at each tidal event, a longer residency time, and so enhanced accretion over the whole site. For this analysis the creek mask was removed from the marsh elevation changes because creek-specific processes like channel down-cutting could skew the data.

6.2.5 Estimation of uncertainty

The sources of uncertainty in this chapter can be separated into: (1) data acquisition uncertainty; and (2) data processing uncertainty. The data acquisition uncertainty is linked to lidar technology, which implies that younger datasets are likely to be more accurate than older datasets. The effect of vegetation detection by lidar on creek evolution interpretation will also be discussed by fulfilling the third sub-objective of this chapter.

The data processing uncertainty is due to potential errors in the detection of creek networks and the extraction of morphological parameters by the algorithm developed in Chapter 4. The uncertainty linked to creek detection can be quantified using the standard deviation of morphological parameters values when the elevation thresholds vary by +/- 0.15 m (Figure 6.5). This uncertainty was tested for HOMW, one of the largest and more complex creek networks considered, for 4 lidar datasets taken between 2009 and 2014 (Table 6.2). For some parameters (mouth cross-sectional area, main and total channel length, creek planform area or OPL), the standard deviation is proportional to the MR size. For these parameters, the creek detection uncertainty is defined as a percentage of the mean parameter for each site. Other parameters (channel mouth depth and sinuosity ratio) are independent from the site size, and the standard deviation can be directly used as the uncertainty.

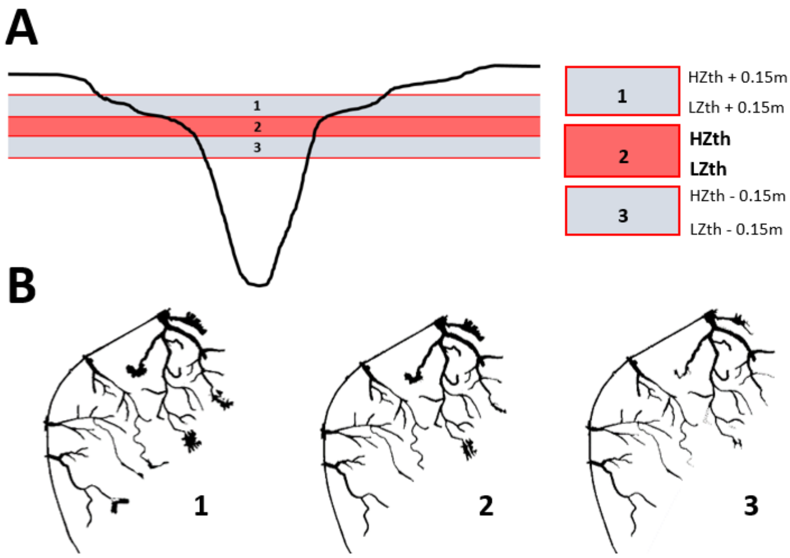


Figure 6.5: Sensitivity tests for creek detection at Hesketh Out Marsh West (HOMW). A) Expected variation in creek detection when the elevation thresholds are changed by +/- 0.15 m; B) Effect on HOMW creek detection in 2010

Table 6.2: Mean standard deviation of HOMW morphological parameters detected by the algorithm when the elevation thresholds are changed by +/- 0.15 m:

Parameter (tested on HOMW 2009-2014)	**Mean STD (% of mean value)**
Mean elevation above MWS (m)	N/A
Drainage density (km/km^2)	0.5 (5%)
Overmarsh path length (m)	5.08 (11%)
Main channel length (m)	36.8 (3%)
Total channel length (m)	800 (5%)
Number of creeks (no unit)	21.2 (9%)
Total mouth cross-sectional area (m^2)	4.98 (4%)
Main channel mouth depth (m)	0.33 (12%)
Planform area (m^2)	3.49*10^4 (20%)
Undermarsh tidal prism (creek volume) (m^3)	2.02*10^4 (17%)
Sinuosity ratio (no unit)	0.07 (6%)
Main channel gradient (^{o})	0.05 (4%)

6.3 Results

The evolution rates observed through this multi-year lidar analysis are detailed and compared with monitoring results of MR schemes from the wider literature. Section 6.3.1 details the creek evolution for each scheme. Section 6.3.2 infers general trends of creek evolution and discusses their robustness. Finally, Section 6.3.3 analyses the potential morphodynamic processes that caused those evolution rates to discuss the reliability of lidar at detecting creek-forming processes.

6.3.1 MR monitoring results

The first sub-objective is to monitor the changes in elevation distribution and in creek network morphological characteristics within MR sites over the available timescales. The proportions of mudflat, low marsh and higher marsh are provided in Table 6.3 for the oldest and most recent available years of each MR scheme, based on their elevation within the tidal range rather than on their vegetation assemblages. Most MR schemes lie within the same elevation range as that expected from natural mature saltmarshes (Figure 6.6A-B; Table 6.3). Only Abbots Hall, Allfleet and Tollesbury remain mudflat dominated and do not significantly accrete after breaching (Figure 6.6B-C; Table 6.3), probably due to lower suspended sediment concentrations in the Crouch and Blackwater estuary (Appendix C1). Most of Chowder Ness is at the elevation of a middle marsh in 2007, but rapidly accretes to the elevation of a high marsh. All other sites start out and remain high marsh-dominated. The percentage of high marsh increases over time for all sites except Allfleet and Abbots Hall.

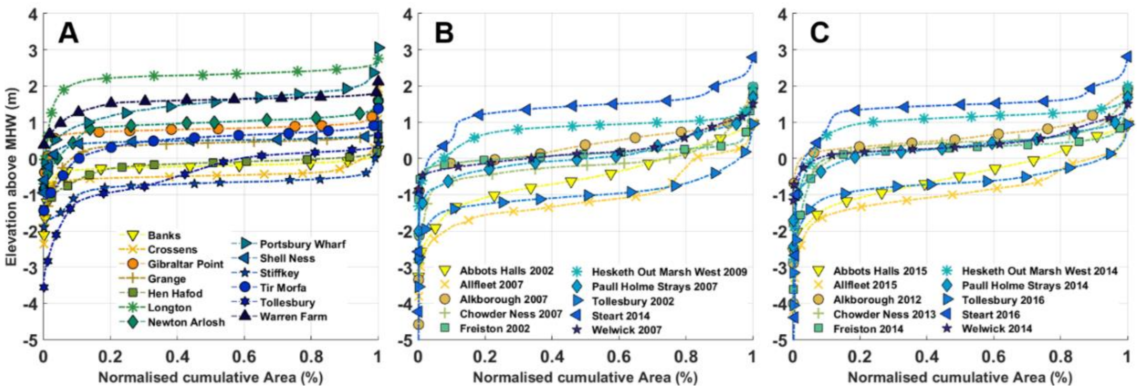


Figure 6.6: Hypsometry evolution for all sites. A: 13 mature natural saltmarshes; B: 10 MR sites close to the year of breaching; C: 10 MR sites after evolution

Table 6.3: Evolution of tidal flat, low marsh and higher marsh mean elevations and proportions in 10 MR schemes according to elevation data within the tidal frame (tidal flat-low marsh transition = MHWN; low-high marsh transition = MHW)

Site name and year of breach	Years since breach	% of mudflat	Mean elev (ODm)	% of low marsh	Mean elev (ODm)	% of high marsh	Mean elev (ODm)
Abbots Hall	0	58.53	1.31	14.20	2.12	27.27	2.76
2002	12	54.82	1.33	17.96	2.12	27.22	2.77
Alkborough	1	0.31	0.42	21.91	3.04	77.78	3.71
2006	8	0.34	0.79	1.97	2.89	97.69	3.80
Allfleet	1	77.84	1.00	4.40	2.18	17.76	2.63
2006	8	77.83	1.23	10.27	2.22	11.89	2.61
Chowder Ness	1	4.21	1.29	68.66	2.58	27.13	3.42
2006	9	2.56	1.17	4.86	2.54	92.58	3.64
Freiston	0	0.00	1.83	22.55	2.74	77.45	3.10
2002	11	3.82	1.12	5.53	2.54	90.65	3.23
Hesketh Out Marsh W	1	0.45	1.89	6.36	2.89	93.19	4.06
2008	6	1.55	1.64	4.14	2.73	94.31	4.30
Paull Holme Strays	4	5.26	1.67	41.86	2.47	52.88	3.19
2003	11	2.80	1.60	16.48	2.41	80.72	3.14
Steart	0	0.19	0.43	2.83	3.99	96.96	5.84
2014	2	0.76	0.25	2.39	3.90	96.86	5.87
Tollesbury	7	88.81	1.26	6.81	2.12	4.37	2.61
1995	21	75.96	1.52	15.81	2.11	8.23	2.58
Welwick	1	0.01	1.65	32.87	2.22	67.12	2.76
2006	8	0.24	1.62	7.24	2.16	92.53	2.82

With the exception of the strong elevation gradients at Abbots Hall and Allfleet, the hypsometry curves have similar structures for most available MR schemes, with a flat marsh similar to that expected from natural systems (Figure 6.6A-B). Marsh build-up is accompanied by creek expansion: the maximum RS creek order, total creek length and number of creeks increase for all sites, while OPL decreases, indicating a better creek distribution (Table 6.4**Error! Reference source not found.**). However, even after years of evolution, significant differences remain between MR and natural creeks: detailed results of site hypsometry, creek extent and creek morphological parameters are given in Appendices F, G, H, I, J and K, and will be discussed in the following sub-sections for each of the 10 MR sites.

Table 6.4: Morphological characteristics of the creek network for each MR scheme and each available year

Site name and year of breach	Number of years since breach	Max order	Number of creeks	Total channel length (m)	Mean bifurcation ratio	Junction angle	Sinuosity ratio	Mean cross-sectional area	Mean W/D ratio	Creek volume (m3)	OPL (m)
Abbots Hall 2002	0	4	58	5680	2.8	87	1.08	2.7	1.6	18999	26.3
	6	5	83	6122	2.2	86	1.08	1.0	4.2	6812	22.3
	11	5	89	6557	2.5	95	1.08	3.5	3.6	17343	16.4
	12	5	94	6708	2.2	92	1.09	3.0	3.2	17663	17.0
	13	5	89	6634	2.1	91	1.18	3.5	2.6	18297	16.9
Alkborough 2006	1	3	31	2951	5.0	92	1.05	4.2	34.2	20712	774.2
	4	3	23	2056	4.7	90	1.07	0.7	13.0	1891	796.3
	6	4	50	3565	3.4	100	1.09	3.4	18.3	12679	768.7
	9	4	52	4672	5.2	75	1.09	5.3	8.1	17565	540.2
Wallasea (Allfleet's Marsh) 2006	1	3	48	2546	3.0	89	1.04	3.7	3.0	8927	116.4
	5	4	175	10777	3.2	88	1.09	6.0	5.5	49468	54.7
	7	4	174	9064	3.3	90	1.08	6.5	6.5	53767	58.4
	9	4	200	9730	3.4	90	1.11	6.4	5.0	44302	56.1
Chowder Ness 2006	1	2	11	579	4.5	88	1.09	5.5	6.3	2999	101.2
	3	3	27	829	3.3	90	1.08	1.1	3.7	872	106.4
	4	3	23	759	2.9	102	1.07	2.2	5.1	1204	69.6
	5	3	29	1078	3.5	83	1.09	1.3	8.6	1450	50.5
	6	3	50	1489	4.4	90	1.09	1.7	7.7	2641	29.6
	7	3	25	920	2.5	96	1.11	0.6	3.5	531	40.1
	9	4	55	1369	2.9	91	1.08	1.7	5.7	1939	33.1
	10	3	60	1730	3.9	90	1.08	0.8	3.1	1104	30.1
Freiston 2002	0	4	74	4920	3.3	89	1.05	2.8	8.2	10620	110.1
	7	4	101	6522	3.1	96	1.05	8.5	3.4	30541	57.3
	9	4	125	8356	3.1	95	1.07	9.7	4.0	36860	50.3
	11	4	129	8386	3.1	89	1.06	9.5	4.5	42271	49.9
	12	4	116	8076	3.1	95	1.07	12.5	4.3	52622	43.4
Hesketh Out Marsh W 2008	1	4	151	13890	5.0	79	1.09	32.9	13.8	120644	53.2
	2	4	155	14596	5.4	78	1.09	24.9	9.0	126438	46.1
	3	5	239	16410	3.9	81	1.12	41.5	8.6	118021	36.3
	6	5	411	18692	4.9	82	1.11	39.3	5.7	140374	43.5
Paull Holme Strays 2003	4	4	96	7914	4.7	78	1.09	5.5	9.8	28346	57.3
	7	4	106	9051	3.1	75	1.08	3.6	10.2	20655	58.4
	9	4	184	11858	3.3	84	1.15	3.9	8.0	29851	29.0
	10	4	173	12985	3.5	83	1.08	3.8	7.2	31182	24.9
	11	4	187	12364	3.7	91	1.14	4.9	7.4	44429	25.2
Steart 2014	0	4	77	9320	3.9	92	1.05	36.7	15.1	118256	84.6
	1	4	53	8770	3.7	79	1.06	70.0	9.8	230353	85.1
	2	4	62	9288	4.0	90	1.06	70.0	12.4	177490	82.4
Tollesbury 1995	7	4	49	2265	2.8	93	1.12	1.8	2.5	3110	53.1
	14	4	50	2357	3.5	97	1.07	3.7	1.6	4689	56.2
	17	4	62	2549	3.8	93	1.07	3.4	1.4	5154	49.2
	20	4	60	2550	4.1	97	1.06	3.0	1.4	4646	50.1
	21	4	64	2679	4.1	95	1.06	4.1	1.2	5255	46.9
	22	4	76	2880	4.4	83	1.06	4.3	1.3	5543	46.8
	23	4	101	3339	4.3	92	1.07	1.4	0.9	4556	37.4
Welwick 2006	1	3	37	632	3.7	98	1.06	0.5	11.9	463	261.8
	3	4	52	885	2.7	93	1.08	0.8	21.8	1545	201.6
	4	4	51	710	2.6	95	1.09	0.5	18.9	1199	183.3
	5	4	67	1054	3.0	97	1.10	1.0	11.4	2659	126.7
	6	4	83	1436	3.2	94	1.06	1.0	6.4	2866	122.5
	7	4	97	1920	3.6	97	1.10	1.6	7.9	3474	119.4
	8	4	99	1452	3.6	89	1.14	1.6	7.6	6066	116.0

6.3.1.1 Abbots Hall (Blackwater estuary, breached in 2002)

Abbots Hall has undergone limited morphological changes between 2002 and 2015. The initial elevation has lowered in patches in the lower sections of the site (Figure 6.7), which may be due to vegetation die-off after the flooding of the site. The creek network has remained mostly stable aside from a widening and deepening of the entry channel (Figure 6.7). The maximal creek order increased from 3 to 4 between 2002 and 2008, but this is due to the excavation of the breach, not yet implemented in the 2002 dataset (Table 6.4). The breach area leading to the entry channel remains stable (Appendix G1).

The creek system has partially filled in with sediment, as shown by the upward trend of the creek hypsometry curve's first inflexion point over the years (Appendix F1). This agrees with previous studies that found the site to act as a sediment sink (ABP mer 2011a). There is no evidence of erosion within the creeks. The creek morphology resembles that of natural systems: the sinuosity, junction angle, cross-sectional area and mean creek volume all fall within the expected range (Appendix I1). Still, significant differences remain. Because of its interconnected nature, Abbots Hall contains more low RS order channels despite having only one breach area. The high RS order creeks are fewer, longer and have a lower bifurcation ratio than natural creeks. At all orders, creeks are wider and deeper than their natural counterparts. The low mean W/D values suggest a narrower, deeper shape than semi-circular. The anomalous sinuosity value for creek order 4 in 2015 is likely due to an interconnected segment looping around itself.

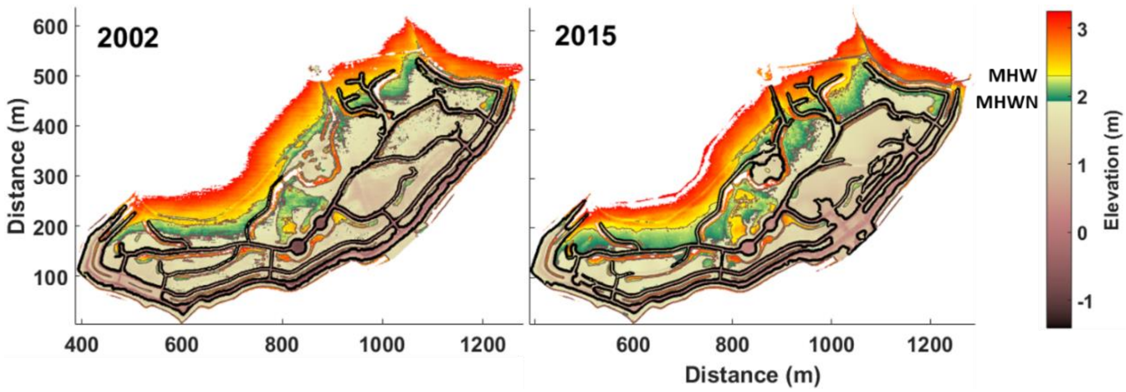


Figure 6.7: Changes in elevation distribution within the tidal frame for Abbots Hall between 2002 and 2015 (vertical resolution 15cm, horizontal resolution 1m). Creek extent as detected by the parametrisation algorithm shown in black.

6.3.1.2 Alkborough (Humber estuary, breached in 2006)

Changes in elevation distribution and creek morphology can be observed at Alkborough. There is a 20% increase in the proportion of high marsh between 2007 and 2015 (Table 6.3). The increase in elevation occurs mainly in the lower sections of the site, close to the entry channel (Figure 6.8): field observations relate this increase to accretion rates (Manson et al. 2012), exacerbated locally by reed-bed growth close to the main channel of up to 2 m between 2005 and 2012 (Pontee 2014c). While the riveted breach area maintains at a constant width (Appendix G2), the entry channel (RS order 1) widens and deepens, and its cross-sectional area doubles between 2007 and 2015 (Appendix H3 and H4).

The initial creek network shrinks between 2007 and 2010 before expanding via reactivation of drainage ditches (Appendix I2). The creek morphometry data from 2010 is anomalous due to the presence of standing water at the MHW level at the time of lidar acquisition (Appendix F2) but does not affect the overall creek evolution trend. The appearance of new creeks feeding into the main channel leads to an increase of the maximum RS order from 3 to 4 between 2010 and 2012 (Table 6.4). Despite this expansion, the creek network remains significantly sparser than a natural system, with OPL values exceeding 500 m (Table 6.4) due to the large initial areas empty of creeks (Figure 6.8). The sinuosity ratio is also lower than that of natural systems, especially in the high RS order channels (Appendix I2). Those creeks, inherited from drainage ditches, have a low cross-sectional area and low mean W/D ratios marking a deep, narrow shape.

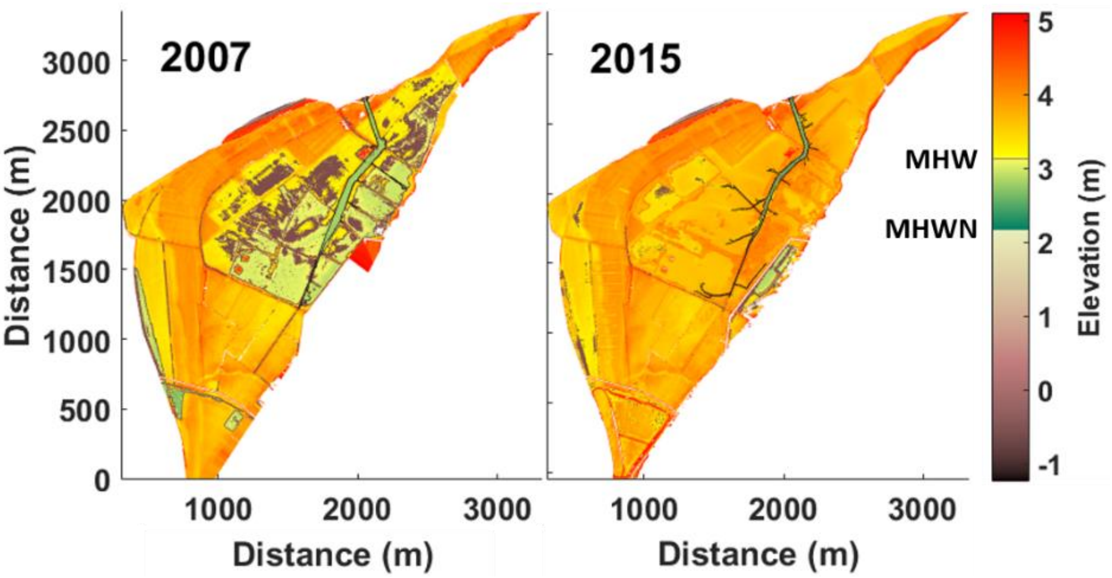


Figure 6.8: Changes in elevation distribution within the tidal frame for Alkborough between 2007 and 2015. Creek extent shown in black.

6.3.1.3 Allfleet/Wallasea (Crouch estuary, breached in 2006)

Changes in elevation distribution and creek morphology can be observed at Allfleet. The majority of the site starts out at the elevation of a tidal flat (Figure 6.9), in compliance with the site habitat mitigation objectives. An increase in elevation of about 3 cm/year can be observed between 2007 and 2015, which fits with previously measured accretion rates of the order of 3–5 cm/year and is attributed to sediment import into the site (ABP mer 2010). The breaches and entry channels remain stable: some sediment infill can be observed rather than deepening and widening (Appendix G3).

The creek system developed significantly between 2007 and 2014, as shown by the increase in channels' length, sinuosity, cross-sectional area and volume (Figure 6.9, Appendix I3). The maximal creek RS order has increased from 3 to 4 (Table 6.4). The creek network expands by building on previous drainage ditches and by differential accretion within the dominantly mudflat environment. Except for the 4th RS order creeks, which tend to be overly straight, too deep and have lower mean W/D values than their natural counterparts, creeks at Allfleet have a morphology similar to natural systems (Appendix I3).

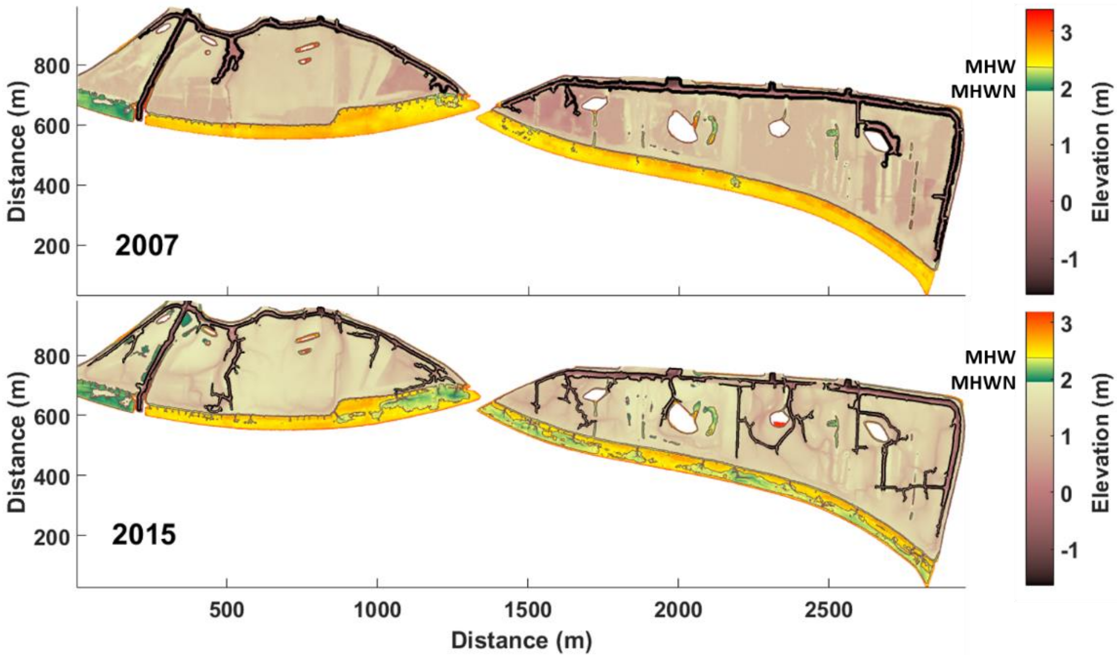


Figure 6.9: Changes in elevation distribution within the tidal frame for Allfleet (Wallasea) between 2007 (vertical resolution 0.15 m, horizontal resolution 2 m) and 2015 (vertical resolution 0.15 m, horizontal resolution 1 m). Creek extent in black.

6.3.1.4 Chowder Ness (Humber estuary, opened in 2006)

Rapid changes in elevation and creek morphology can be observed at Chowder Ness. By 2007, most of the site lies above MHWN (Figure 6.10), even though the initial reprofiling of the site aimed to provide mudflat habitats (ABP mer 2011b). As of 2016, over 90% of the site has reached the elevation of a middle to high marsh (Table 6.3). The bank removal at Chowder Ness provides a large opening that accommodates several competing entry channels. The initial entry channel visible in 2007 fills in and shrinks between 2007 and 2016 (Appendix G4). In parallel, a new entry channel forms from 2010 and branches out to cover most of the catchment area, causing a decrease in OPL from 100m to 30m (Table 6.4). The development of creeks builds on the initial topography and is accretion-driven.

Despite the rapid changes, the creek network remains significantly different and more dynamic than a natural system. Though the high elevation and presence of vegetation by 2016 (Figure 3.25) should stabilise the creek banks, the creek network displays stark inter-annual changes: the maximal RS creek order fluctuates between 3 and 4 from year to year (Table 6.4), and the creek morphometric parameters have a larger spread than other sites (Appendix I4). This is likely due to high siltation rates and the absence of a pre-excavated entry channel to focus the flow. The low undermarsh tidal prism means that the flow going through the channels is too limited to cause down-cutting or prevent sediment infill. Consequently, the creeks' depth and cross-sectional area are lower than in natural systems (Appendix I4).

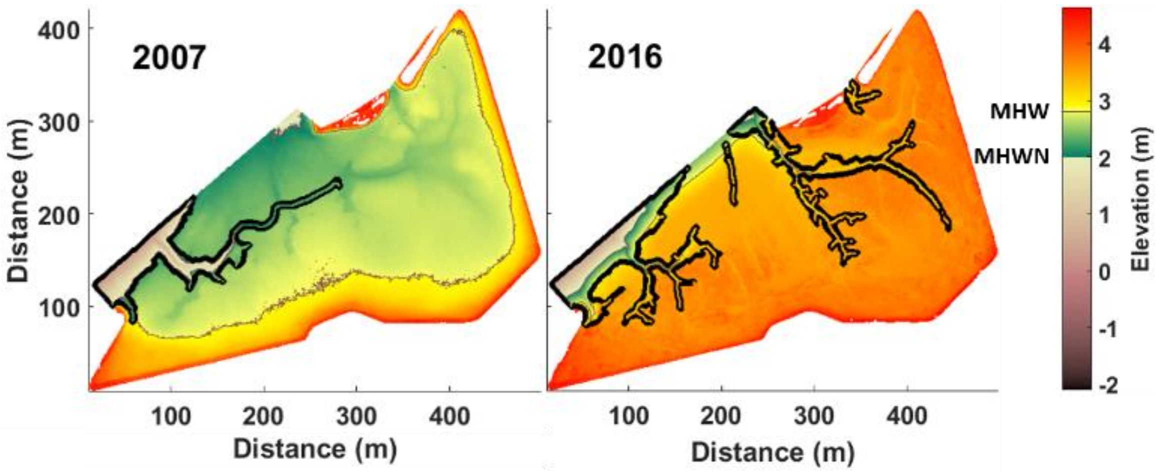


Figure 6.10: Changes in elevation distribution within the tidal frame for Chowder Ness between 2007 and 2015. Creek extent shown in black.

6.3.1.5 Freiston (The Wash, breached in 2002)

Changes in elevation distribution and creek morphology can be observed at Freiston. Between 2002 and 2014 the elevation increases to that of a middle to high marsh (Table 6.3). By 2014, almost all potential low marsh habitats have disappeared, due to both an elevation increase of the marsh and the down-cutting of channels below MHWN levels (Figure 6.11). The three breaches have undergone significant erosion since the site opening: the cross-sectional area of the largest breach increases tenfold between 2002 and 2009 before stabilising (Appendix G5), in agreement with field measurements (Symonds 2006).

The OPL values decrease over the years, indicative of a better distribution of the creek network (Table 6.4). Creek extension seems to be driven mostly by headward erosion (Figure 6.11). The creek morphometry generally fits with that of natural systems, though significant differences remain: the newly formed, high RS order creeks have lower W/D ratios, corresponding to deep narrow channels (leptokurtic shape) (Appendix I5), while in natural systems, the higher RS order creeks tend to become flatter in shape. The creeks at Freiston are also excessively straight and have low bifurcation ratios (Appendix I5). Finally, human intervention has influenced the creek network after the breach: some creeks have been artificially excavated or infilled between 2013 and 2014 (Lawrence 2018).

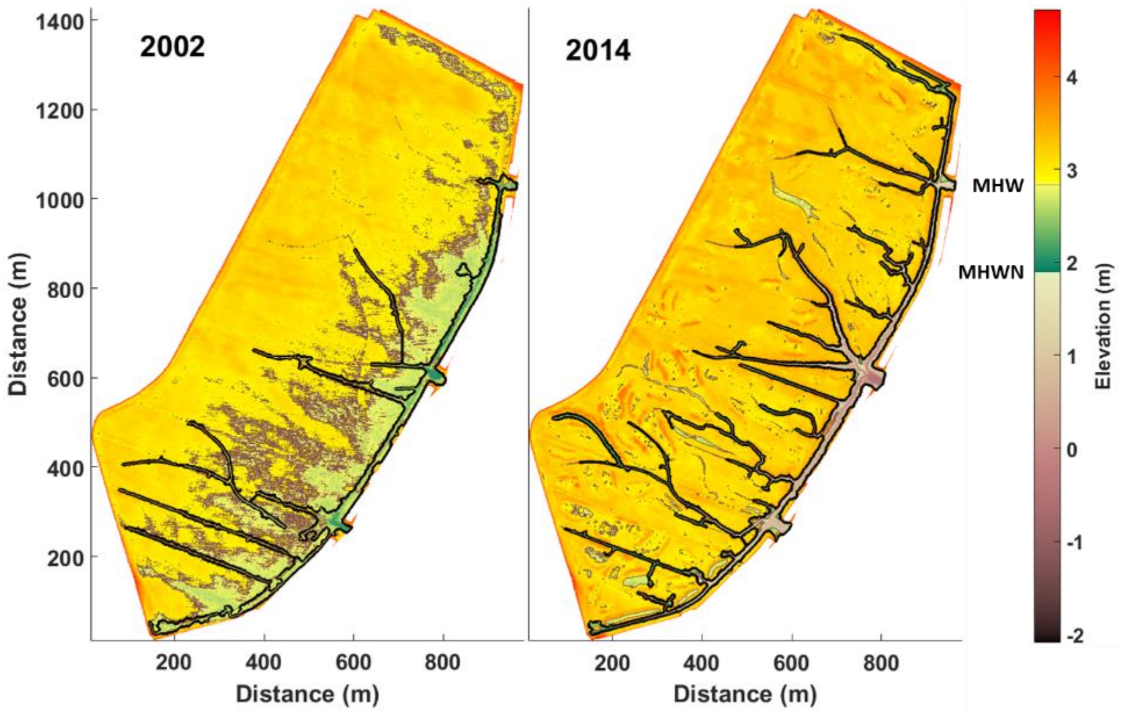


Figure 6.11: Changes in elevation distribution within the tidal frame for Freiston between 2002 and 2016. Creek extent shown in black.

6.3.1.6 Hesketh Out Marsh West (HOMW) (Ribble estuary, breached in 2008)

Changes in elevation distribution and creek morphology are observed at HOMW. The elevation starts out as dominantly that of a high marsh and shows a continuous increasing trend (Table 6.3). The ponds headward of creeks get infilled between 2011 and 2014, as shown by the decrease in maximum cumulative area of the creek network hypsometry curves (Appendix F6). Erosion occurs within the lower portions of the creeks, especially the one connected to the largest breach, which deepens to below MHWN levels (Figure 6.12). New creeks branch out from the initial template between 2009 and 2014, seemingly led by headward erosion (Figure 6.12).

Due to being excavated from a previous mature creek template, out of all studied schemes HOMW's creeks visually look most like a natural system: sinuosity values fall within the expected range, which few other schemes achieve (Appendix I6). However other parameters are significantly different from those found in natural sites: HOMW contains fewer creeks per RS order, and the creeks are wider and deeper than natural ones, especially near the breach area.

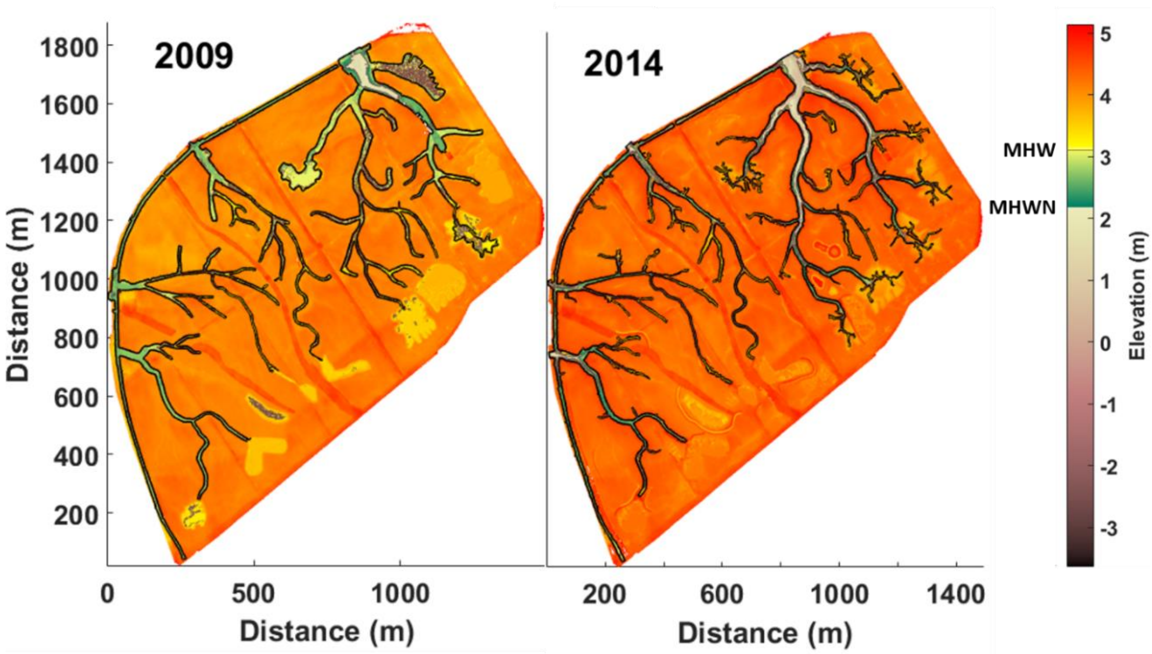


Figure 6.12: Changes in elevation distribution within the tidal frame for HOMW between 2009 and 2014 (vertical resolution 0.15 m, horizontal resolution 1 m). Creek extent as detected by the parametrisation algorithm shown in black.

6.3.1.7 Paull Holme Strays (Humber estuary, breached in 2003)

Changes in elevation distribution and creek morphology can be observed at Paull Holme Strays. The site elevation leads to an increase in the proportion of potential middle to high marsh (Figure 6.13). This fits with a previous study (Mazik et al. 2007) which found high accretion rates within the site, causing pioneer vegetation to colonise designated mudflat areas by 2006. Accretion rates are expected to cause saltmarsh development over most of the site (Brown et al. 2007). The largest breach remains stable over the timescale considered (Appendix G7). The creek area however increases steadily over the years, leading to a decrease in OPL (Table 6.4). Some headward erosion occurs in the creek system connected to the second, smaller breach, lowering elevation to below MHW within the new channel (Figure 6.13).

Overall, the creek parameters are within the spread of natural creek morphometry (Appendix I7), with some exceptions. The highly interconnected system leads to a wide spread of sinuosity ratio values. Most channels are very straight due to being inherited from drainage ditches, but their interconnection makes them appear kinked, which leads to anomalously high sinuosity ratios. This will be discussed further in the discussion section of this chapter. The fourth RS order creeks tend to have overly leptokurtic shapes, while the entry channel is very wide but shallower than most natural entry channels (Appendix I7).

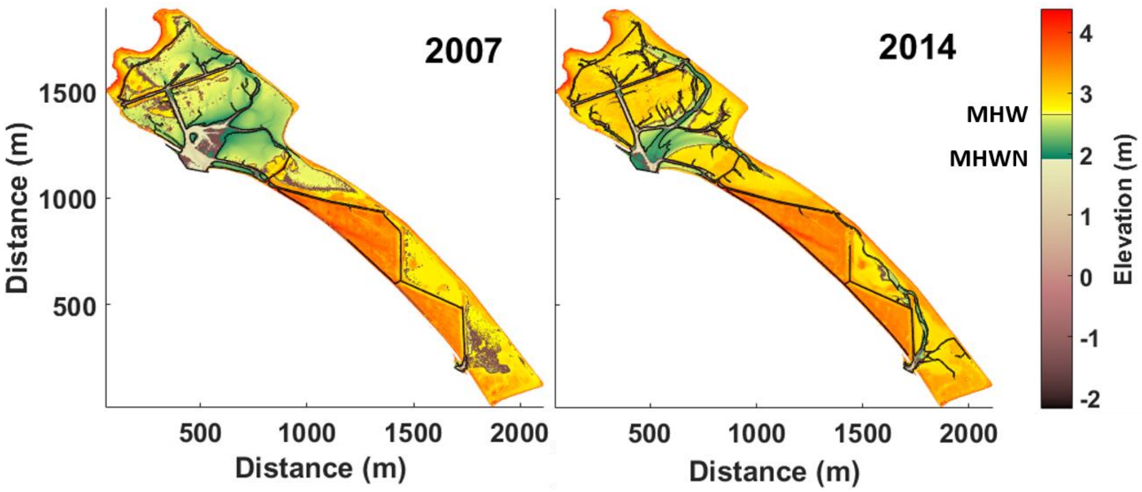


Figure 6.13: Changes in elevation distribution within the tidal frame for Paull Holme Strays between 2007 and 2014 (vertical resolution 0.15 m, horizontal resolution 1 m). Creek extent as detected by the parametrisation algorithm shown in black.

6.3.1.8 Steart (Parrett River, Severn estuary, breached in 2014)

Changes in elevation distribution and creek morphology can be observed at Steart, but are difficult to interpret after only 2 years of evolution. Steart lies very high in the tidal frame and starts out at the elevation of a high marsh (Table 6.3). The breach and entry channel have undergone significant erosion (Figure 6.14): the Steart entry channel was built overly shallow for the predicted tidal prism (Pontee, 2016, pers. com.), and expected to deepen in response to the hypertidal conditions. As a result, within 2 years the channels' depth and cross-sectional area have increased above the values expected of natural systems, while the mean W/D ratio has dropped to expected levels (Appendix I8:). The cross-sectional shape of the Steart channels thus corresponds to an enlargement of a natural systems' shape. The high tidal range and high marsh elevation, causing a high elevation gradient between Steart and the River Parrett MLWS level also cause rapid erosion in the entry channel.

The creek network is significantly different from a natural system: the creeks are oversized, too long and straight. Field observations showed the high RS order creeks infilled with sediment over the first two months following breaching. The rapid accretion is either due to the reworking of sediment from the eroding breach area, or to sediment import from the wider estuary, which is characterised by turbidity values of 4-16 g/L (Appendix C1). Aside from that infill, the creek network has not changed significantly in 2 years, except for limited headward erosion (0.4 m/year according to preliminary field measurements conducted by this study), bank slumping, and shallow creek formation in the headward ponds which have not yet been picked up by lidar.

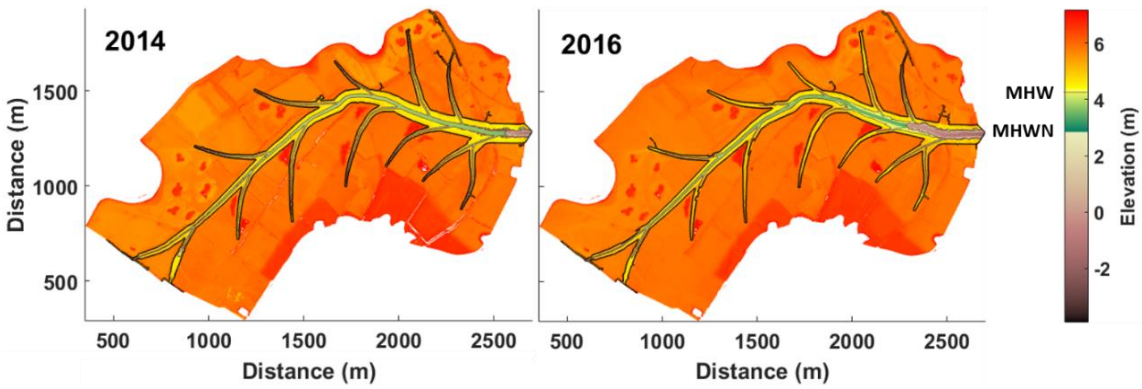


Figure 6.14: Changes in elevation distribution within the tidal frame for Steart between 2007 and 2014 (vertical resolution 0.15 m, horizontal resolution 1 m). Creek extent as detected by the parametrisation algorithm shown in black.

6.3.1.9 Tollebsury (Blackwater estuary, breached in 1995)

Changes in elevation distribution, and to a lesser extent in creek morphology can be observed at Tollesbury. This site has remained dominantly at the elevation of a tidal flat from 2002 until 2016, though the mean elevation has increased by 26 cm in the tidal flat area (Table 6.3), corresponding to a mean rising rate of 1.9 cm/yr. This resembles results from previous studies, which have measured linear accretion rates of 2.3 cm/yr between 1995 and 2001 (Garbutt et al. 2006). Additionally, areas initially above MHW show no significant elevation increase over the time considered, as found by previous field monitoring (Garbutt et al. 2006; Hughes et al. 2009).

The creeks do not evolve significantly from their initial template until 2009 (Appendix I9), which fits with findings from Brown et al. (2007). Creek extension also seems dominantly driven by depositional processes (Figure 6.15). The creek network remains significantly different from a natural system. The creeks, inherited from agricultural ditches, are straighter and deeper than natural channels, with a more leptokurtic shape as shown by the low mean W/D ratios (Appendix I9). The creeks at Tollesbury also have excessively low bifurcation ratios, which, added to the limited evolution of the creek system, points to insufficient branching of the channels.

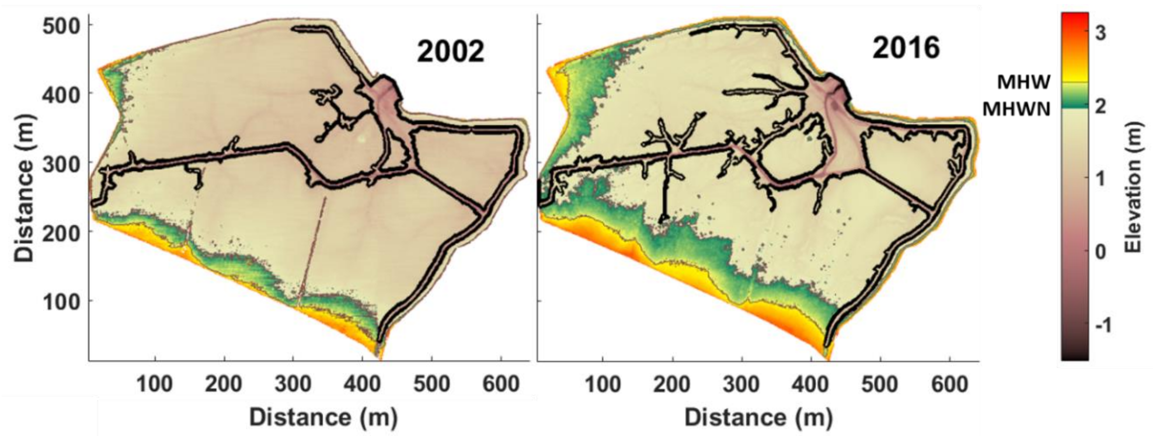


Figure 6.15: Changes in elevation distribution within the tidal frame for Tollesbury between 2002 and 2016 (vertical resolution 0.15 m, horizontal resolution 1 m). Creek extent as detected by the parametrisation algorithm shown in black.

6.3.1.10 Welwick (Humber estuary, breached in 2006)

Rapid changes in elevation and creek morphology can be observed at Welwick. Even though part of the site was lowered to increase the proportion of mudflat habitat (ABP mer 2011d), the site accretes rapidly and most of it reaches the elevation of a middle marsh or higher by 2014 (Table 6.3). The proportions of potential habitats derived in 2007 are different from those found by previous studies, possibly partly due to accretion between 2006 and 2007, but mainly to differences in the spring and neap tidal levels chosen: Pontee et al. (2006) use the tidal data measured at Grimsby, while this study uses the weighted mean of values from several ports.

Despite having no initial template, the creek network develops rapidly to resemble the morphology of a natural system. Headward erosion of channels can be observed between 2007 and 2014, causing part of the higher marsh from 2007 to erode below MHW levels (Figure 6.16). The creek area increases while OPL decreases (Table 6.4). The creeks' depth, cross-sectional area and volume also increase over the years and reach the expected range (Appendix I10). The bifurcation ratio values are lower than in natural systems, aside from the anomalous value in 2007, before the creeks could develop (Appendix I10). The sinuosity ratio values are within the expected range for the first and second RS order creeks, while the terminal creeks tend to be straighter and more leptokurtic as they erode headward. These differences can be interpreted as a sign of creek network immaturity.

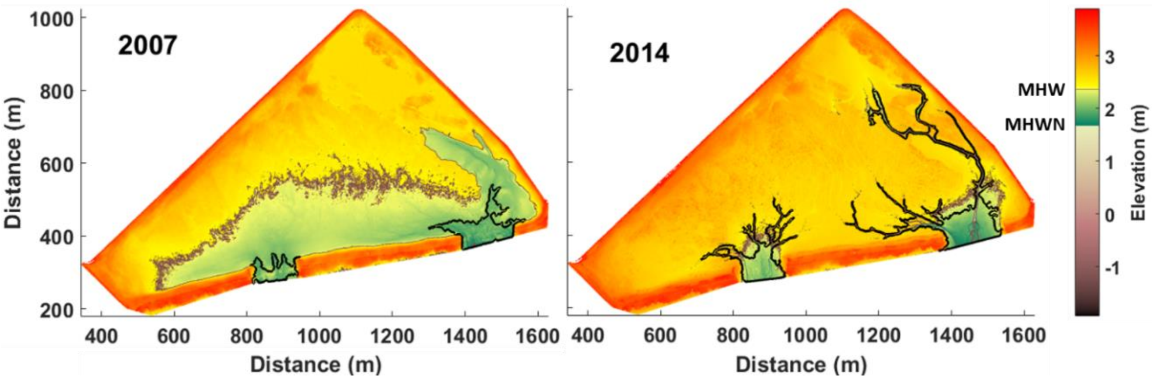


Figure 6.16: Changes in elevation distribution within the tidal frame for Welwick between 2007 and 2014 (vertical resolution 0.15 m, horizontal resolution 1 m). Creek extent as detected by the parametrisation algorithm shown in black.

The first sub-objective was to monitor the changes in elevation distribution and in creek network morphological characteristics within MR sites over the available timescales. The MR schemes observed act as sediment sinks, with a steady increase in elevation shown by both the lidar data and by the field monitoring of accretion rates found in the literature. While Steart, which is only observed over 2 years and 3 datasets, does not display any clear change in creek extent, significant creek growth is observed in the other schemes that have undergone 5 years of post-breach evolution or more. MR creeks tend to become larger, deeper, more sinuous and better distributed. Differences remain however with natural creek systems, especially in the creek segments inherited from drainage systems, which remain overly straight, deep and narrow over the timescales considered.

6.3.2 Definition of linear creek evolution trends

The second sub-objective is to explore potential linear evolution trends for MR creek parameters over the years for all parameters listed in Table 6.1. Evolution rates are given by the slope of the linear fits (Figures 6.17-6.18), and provided in Table 6.5. Less than half of the linear fits obtained are significant at a 95 % confidence interval: the evolution rates per year are a first approximation only. Indeed, for some parameters, over long enough timescales, a decaying trend may be more appropriate than a linear trend to describe parameter evolution towards equilibrium (See conceptual evolution model). For instance, the logarithmic fit of OPL evolution over time yields equal or higher R^2 values than the linear fit for 6 of the 10 sites: Abbots Hall, Allfleet, Chowder Ness, Freiston, HOMW and Welwick. Similarly, the elevation evolution at all sites except Chowder Ness, which is still undergoing significant accretion, and Steart, the youngest site, produces equal or better R^2 values for the logarithmic fit than for the linear fit. Nevertheless, at the timescale considered, the linear evolution rate remains a useful simplified representation of the creek network behaviour, provided that the linear trend supersedes the inter-annual variability of the creek parameters. The linear trend is greater than 50% of the residuals in 93% of cases, and greater than 95% of the residuals in 84% of cases (Table 6.5).

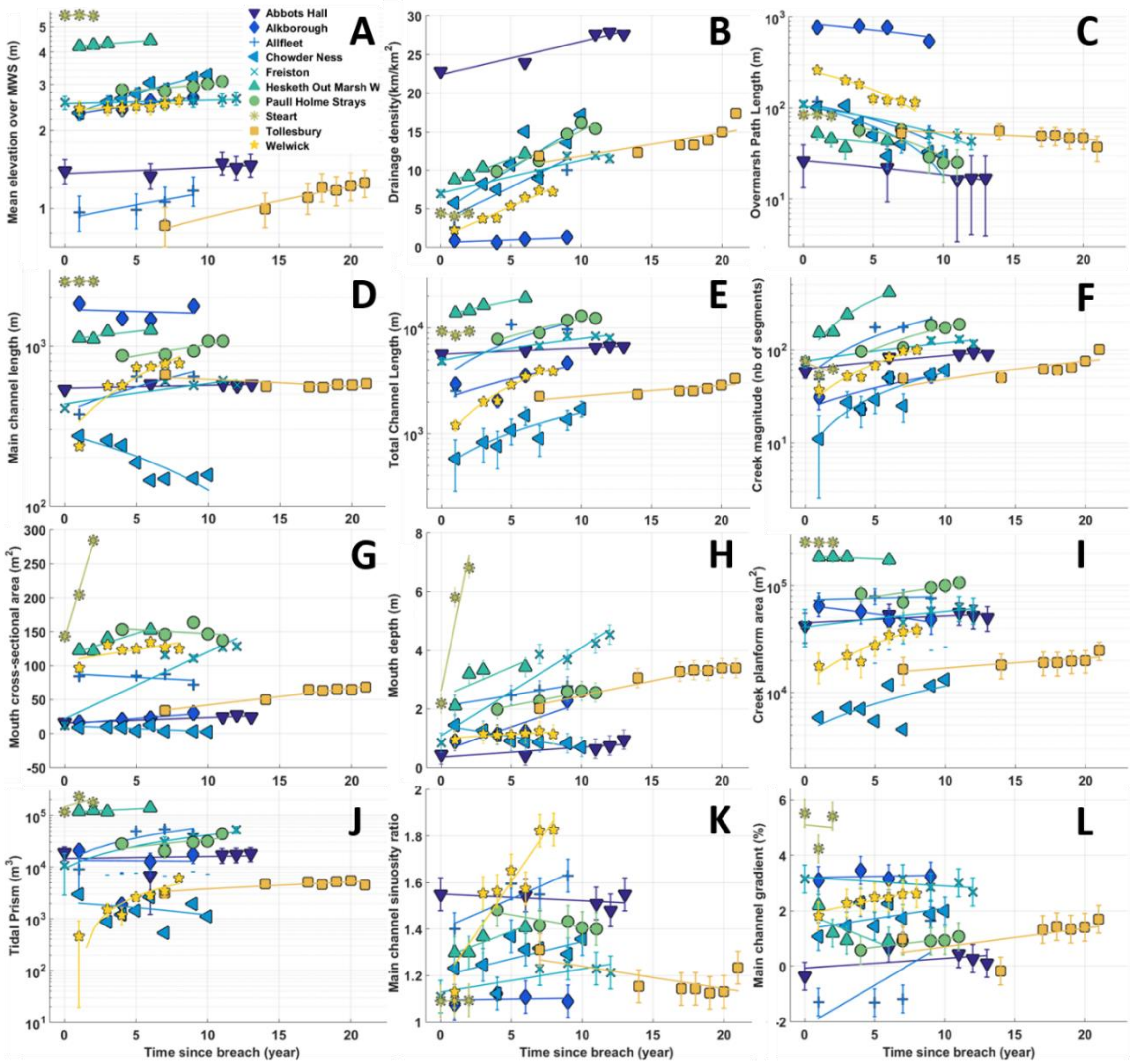


Figure 6.17: Evolution trends for all sites, with the parameter detection uncertainty shown as error bars (note that the error bars are exaggerated in the lower values due to the logarithmic scale used to visualise sites of varying sizes on the same graph). Solid lines correspond to best linear fit. A: Mean elevation above MWS (m/yr); B: Drainage density (km/km^2/yr); C: Overmarsh path length (m/yr); D: Main channel length (m/yr); E: Total channel length (m/yr); F: Number of creeks (nb/yr); G: Total mouth cross-sectional area (m^2/yr); H: Main channel mouth depth (m/yr); I: Planform area (m^2/yr); J: Undermarsh tidal prism (creek volume) (m^3/yr); K: Sinuosity ratio (/yr); L: Main channel gradient (^{o}/yr)

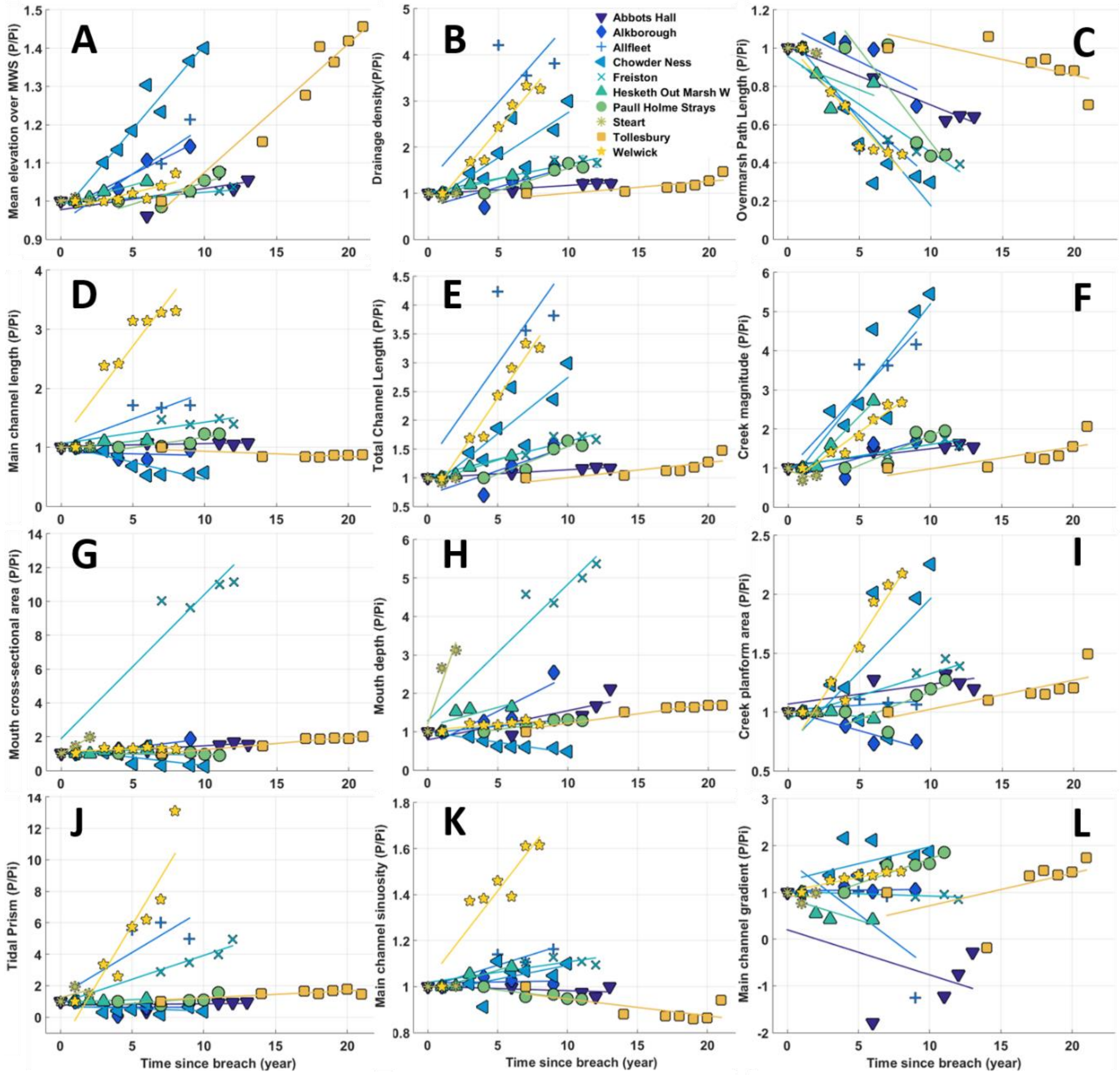


Figure 6.18: Normalised evolution trends of MR mean elevation and creek parameters (Parameter/Initial parameter) since site implementation. Solid lines correspond to best linear fit. A: Mean elevation above MWS (%/yr); B: Drainage density (%/yr); C: Overmarsh path length (%/yr); D: Main channel length (%/yr); E: Total channel length (%/yr); F: Number of creeks (%/yr); G: Total mouth cross-sectional area (%/yr); H: Main channel mouth depth (%/yr); I: Planform area (%/yr); J: Undermarsh tidal prism (creek volume) (%/yr); K: Sinuosity ratio (%/yr); L: Main channel gradient (%/yr)

Table 6.5: MR characteristics evolution rates calculated by applying equation [22] to the best linear fit for all sites. Significant correlations at a 0.05 significance level are shown in bold. Linear trends greater than the 95% spread of the residuals are highlighted in light green, 68% in light yellow, 50% in dark yellow. Trends inferior to the mean of the residuals are highlighted in light red.

	Abbots Hall	**Alkborough**	**Allfleet**	**Chowder Ness**	**Freiston**	**HOMW**	**Paull Holme Strays**	**Steart**	**Tollesbury**	**Welwick**
Mean elevation over MWS in cm/yr (%/yr)	0.78 (0.56)	**4.43 (1.90)**	2.42 (2.51)	**10.30 (4.42)**	0.76 (0.30)	**4.33 (1.03)**	**3.26 (1.14)**	-0.17 (-0.03)	**2.89 (3.36)**	**2.23 (0.92)**
Drainage density in km/km2/yr (%/yr)	**0.43 (1.89)**	0.07 (8.60)	0.90 (34.48)	**1.13 (19.67)**	**0.43 (6.16)**	**0.68 (7.76)**	**0.95 (9.65)**	--0.01 (0.29)	**0.31 (2.59)**	**0.80 (36.25)**
Overmarsh path length in -m/yr (-%/yr)	**0.80 (3.02)**	28.33 (3.66)	7.62 (6.55)	**9.12 (9.01)**	**5.54 (5.03)**	1.58 (2.96)	**5.59 (9.75)**	1.13 (1.33)	0.87 (1.64)	**21.88 (8.36)**
Main channel length in m/yr (%/yr)	2.41 (0.45)	-9.80 (-0.53)	33.64 (8.98)	**-15.70 (-5.74)**	**14.81 (3.63)**	29.58 (2.64)	30.49 (3.49)	8.09 (0.32)	-5.88 (-0.89)	**75.11 (31.98)**
Total channel length in m/yr (%/yr)	**79.53 (1.40)**	247.00 (8.37)	880.16 (34.57)	**113.20 (19.56)**	**301.29 (6.18)**	**1069.51 (7.70)**	**758.98 (9.59)**	-15.84 (-0.17)	**59.33 (2.62)**	**434.63 (36.17)**
Number of creeks in yr^{-1} (%/yr)	**2.50 (4.31)**	3.26 (10.53)	18.79 (39.14)	**5.07 (46.11)**	**4.31 (5.83)**	**55.43 (36.71)**	**14.67 (15.28)**	-7.50 (-9.74)	2.74 (5.60)	**9.77 (26.40)**
Mouth cross-sectional area in m2/yr (%/yr)	**0.80 (4.96)**	**1.68 (10.31)**	-1.11 (-1.32)	-0.89 (-9.66)	**9.86 (85.60)**	6.59 (5.40)	-1.24 (-0.81)	**70.64 (49.30)**	**2.50 (7.32)**	3.30 (3.41)
Mouth depth in m/yr (%/yr)	0.03 (7.60)	0.17 (18.37)	**0.08 (3.61)**	**-0.08 (-5.41)**	**0.30 (35.57)**	0.21 (9.84)	0.09 (4.66)	2.31 (105.92)	**0.10 (4.95)**	**0.03 (2.95)**
Planform area in m2/yr (%/yr)	706 (1.68)	-2201 (-3.41)	570 (0.79)	729 (12.44)	**1639 (3.80)**	-2268 (-1.23)	3898 (4.64)	**-1244 (-0.49)**	**408 (2.46)**	**3366 (18.93)**
Tidal prism in m3/yr (%/yr)	144 (0.76)	-2 (-0.28)	4880 (54.67)	-93 (-3.10)	**3229 (30.41)**	3812 (3.16)	1967 (6.94)	29611 (25.04)	**138 (4.43)**	**702 (151.66)**
Main channel sinuosity in yr^{-1} (%/yr)	-<0.01- (0.19)	<0.01 (0.09)	0.03 (1.89)	0.01 (1.20)	0.01 (0.89)	0.02 (1.82)	**-0.01 (-0.72)**	<0.01 (0.11)	<0.01 (-0.73)	**0.09 (7.84)**
Main channel gradient in o/yr (%/yr)	-0.04 (-9.64)	<0.01 (0.31)	-0.30 (-22.96)	0.08 (7.23)	-0.03 (-0.98)	-0.20 (-9.31)	**0.06 (10.55)**	-0.05 (-0.88)	0.07 (6.93)	**0.11 (5.94)**

There are some occurrences where the linear trends of creek morphological parameters evolution over time are inferior to 68% (dark yellow in Table 6.5) or even 50% of the spread of residuals (red in Table 6.5). In those cases, the inter-annual variations are considered to supersede the linear trend: the creek parameters fluctuate around a mean value rather than evolve towards an equilibrium state. One example is Abbots Hall, where the creek volume (TP) has remained virtually constant since its implementation, but has undergone some phases of sediment infill, leading to fluctuations in volume (Figure 6.17G).

The creek network at Alkborough has been through phases of shrinkage and expansion since 2007 as described in Section 6.3.1.2, resulting in significant fluctuations of MCL (Figure 6.17G). TP, MCS and MCG at Alkborough also display higher inter-annual variabilities due to the episode of infill and shrinking that occurred in the entry channel. Those fluctuations could also be due to remnant water being detected at the bottom of channels in the 2010 dataset, and to the effect of reed bed growth between 2005 and 2012, as discussed in Section 6.3.1.2.

At Steart, due to the shorter monitoring period, the evolution trends are smaller than the residuals for MWS, DD, TCL and MCG. Early behaviour of the site points to an infill of the side creeks (number of creeks and planform area are decreasing) and an increase in the size of the entry channel (evolution rate is largest for CSA and D), which fits with field observations and with lidar data observation in Section 6.3.1.8.

A high inter-annual variability is found for TP at Chowder Ness, probably linked to the instability of the creek network, caused by the bank removal and the absence of pre-excavated entry channels to focus the flow and fix an initial template. The mouth cross-sectional area at Paul Holme Strays is mostly stable over the years (Appendix G7), so the inter-annual variations are comparatively larger.

The second sub-objective of this chapter was to explore potential linear evolution trends for MR creek parameters over the years. The parameters considered (mean marsh elevation, creek length, volume, cross-sectional area, sinuosity, and distribution across the site) evolve following a trend for most sites rather than fluctuate around a mean value. While most of these trends are linear, some parameters like OPL and marsh elevation start to display a decaying trend after about 5 years of evolution. Those creek evolution trends can be used to determine whether MR creeks are evolving towards the morphological equilibrium range expected of natural systems.

6.3.3 Relation to creek-forming morphodynamic processes

The third sub-objective is to test the reliability of the creek evolution trends by identifying the respective roles of vegetation, vertical accretion and creek-forming processes on lidar-detected marsh elevation changes. As expected from saltmarsh environments, the elevation changes within the marsh are correlated with the site elevation for each site (Figure 6.19A-B, Appendix K1). A significant linear correlation is found between 95% of the elevation change data plotted against the initial site elevation, with p-values <0.01 and a goodness of fit R^2 >=0.48, for all sites except Abbots Hall, which underwent limited elevation change since its implementation. A split occurs in the Alkborough dataset (Appendix K2), probably linked to the initial terraced topography of the site, with jumps in elevation occurring across the entry channel (Figure 6.8).

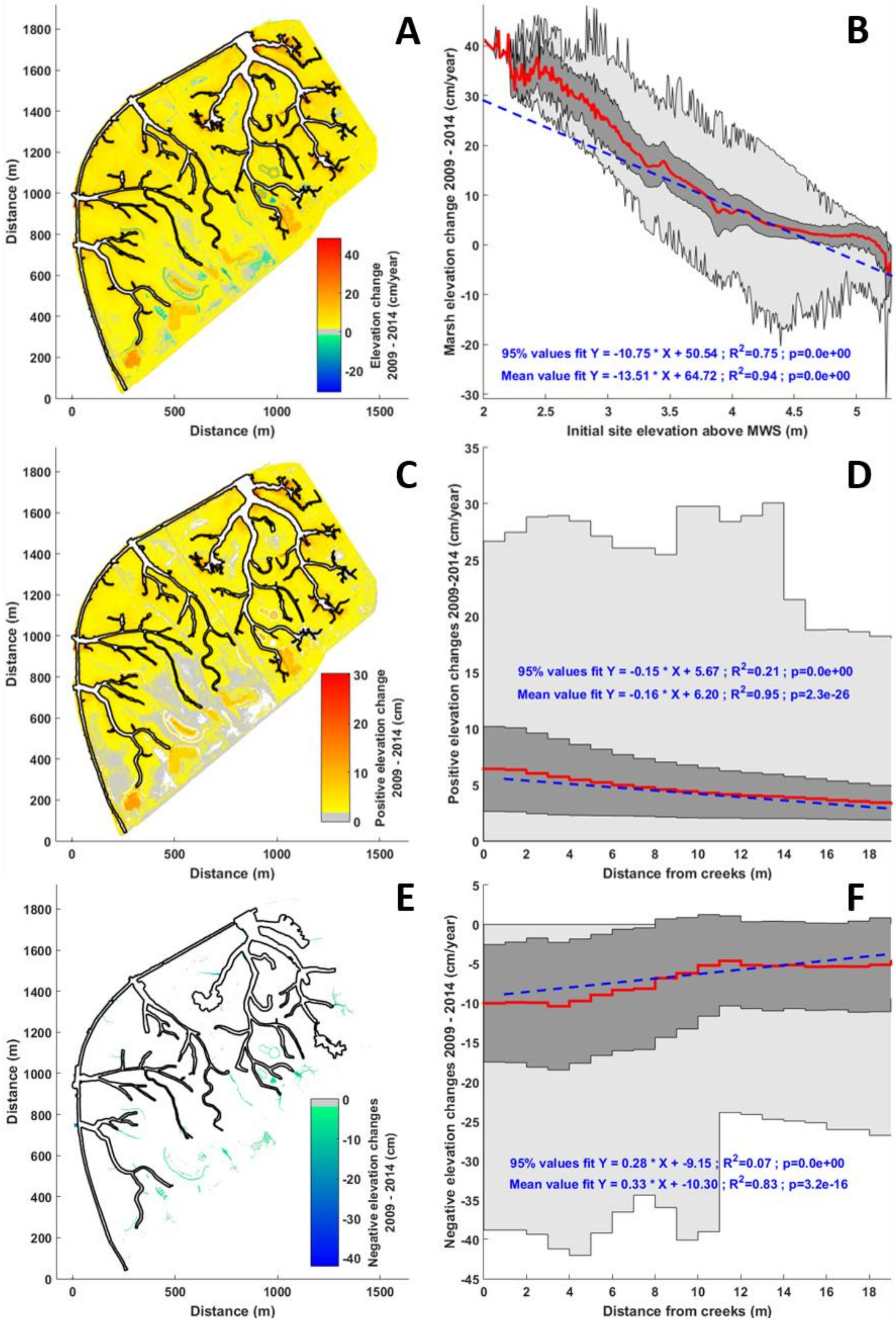


Figure 6.19: Linear correlation tests for the 95% data spread and for the mean values between (A) marsh elevation changes versus site elevation; (B) marsh elevation gains versus distance up to 20 m to the final creek network; and (C) marsh elevation losses versus distance up to 20 m to the initial creek network. Red line: mean value of elevation change; dark grey envelope: 95% spread around the mean value; light grey envelope: total data spread. The marsh elevation changes correspond to the last versus first lidar dataset, divided by the number of years between the two.

For most schemes, a significant negative relationship is also found between the mean elevation gains and the distance to creeks with p-values<0.01 and R^2>=0.63 (Figure 6.19C-D). Elevation gains since site implementation are higher near the creeks and decline up to 20 m away. This is a sign of either plant growth or levee building occurring around MR creeks, as would be expected from natural systems. A poor fit is found for Alkborough (Appendix K2), probably because the tributary channels, being drainage ditches reactivated between 2012 and 2015 (Appendix I2), are still too young to have significantly influenced sedimentation or plant growth. Similarly, at Steart the poor fit is likely due to the short time elapsed since site opening (Appendix K8).

At Alkborough, Allfleet, Freiston, HOMW, Paull Holme Strays and Welwick, a significant positive relationship is found between the mean elevation losses and the distance to creeks with p-values<0.01 and R^2>=0.54 (Appendices K2, K3, K5, K6, K7 and K10). Visual observation of the elevation loss maps indicates that most of the site lowering occurs as sinuous linear features extending from the initial creek system, which is characteristic of channel erosion rather than vegetation die-off. Some vegetation die-off may have occurred between the two breaches at Paull Holme Strays, where more dispersed patterns of surface lowering are visible, but the most significant change is the extension of a new channel from the second breach area (Appendix K7). The negative trends can then be due to the following two processes: first, the down-cutting of pre-excavated features not connected to the initial creek system. Features like drainage ditches could have been infilled with water or soft sediment that got flushed out once the connection with the rest of the creek was established. This is probably the case at Alkborough, Allfleet and at Paull Holme Strays and Freiston, where many drainage ditches can be found. Additionally, the visible erosive processes are probably exaggerated at Freiston and Allfleet: since their initial datasets are noisier and of lower resolution respectively, the initial creek network's extent is likely to be underestimated. As a result, some of the erosion detected corresponds to the down-cutting of existing channels rather than the headward erosion of new creeks.

At Allfleet, due to the combined effect of initial creek underestimation, down-cutting of non-connected features and high accretion rates, it is considered unlikely that headward erosion plays a dominant role in creek growth. The role of headward erosion of new creeks into the marsh substrate appears more prominent at Freiston, HOMW, Welwick and Paull Holme Strays. At HOMW, incision of new creeks by headward erosion is facilitated by the deposition of new sediment within the head ponds, but some incision also occurs within the marsh substrate.

At Abbots Hall, Chowder Ness, Steart and Tollesbury, no significant relation could be found between elevation loss and distance from creeks. The absence of erosive processes is either due to high accretion rates leading to a dominantly depositional environment, like at Chowder Ness and Tollesbury, to an overcompacted marsh substrate, or to a site too high within the tidal frame for the flow-induced bed shear stress to exceed the surface shear strength and cause headward erosion, like at Abbots Hall. Both processes can coexist. Tollesbury is characterised by high accretion rates (Brown et al. 2007) and a very strong substrate resistant to erosion (Watts et al. 2003). At Steart, the time of monitoring is probably too short to accurately interpret results, even though visual observation has found its substrate to be highly compacted and resistant to erosion.

The third sub-objective was to test the reliability of the creek evolution trends by identifying the respective roles of vegetation, vertical accretion and creek-forming processes on lidar-detected marsh elevation changes. The initial elevation of the site is a significant driver of accretion rates. The proximity to creeks has also been found to have an impact on the elevation gains of most schemes up to 20 m away from the creeks. Levee building and/or growth of denser vegetation occurs preferentially near the creeks. While all creek networks have grown in some way, less than half of the sites display channel erosion into the substrate.

6.3.4 Evolution rates of century old AR sites

The fourth sub-objective is to analyse the evolution trends occurring during the last 10 years in century-old, accidentally breached sites to get some insight into the potential long-term evolution of MR creeks. The changes occurring within two AR sites 95 to 110 years after storm breaching are investigated using the same methods as those of sub-objective 2. Both sites can be considered to be an equivalent of century-old MR sites with an embankment period of over 100 years, and with different land uses before breaching: Brandy Hole was half-pasture and half-arable, while Foulton Hall was used as a grazing and mowing marsh.

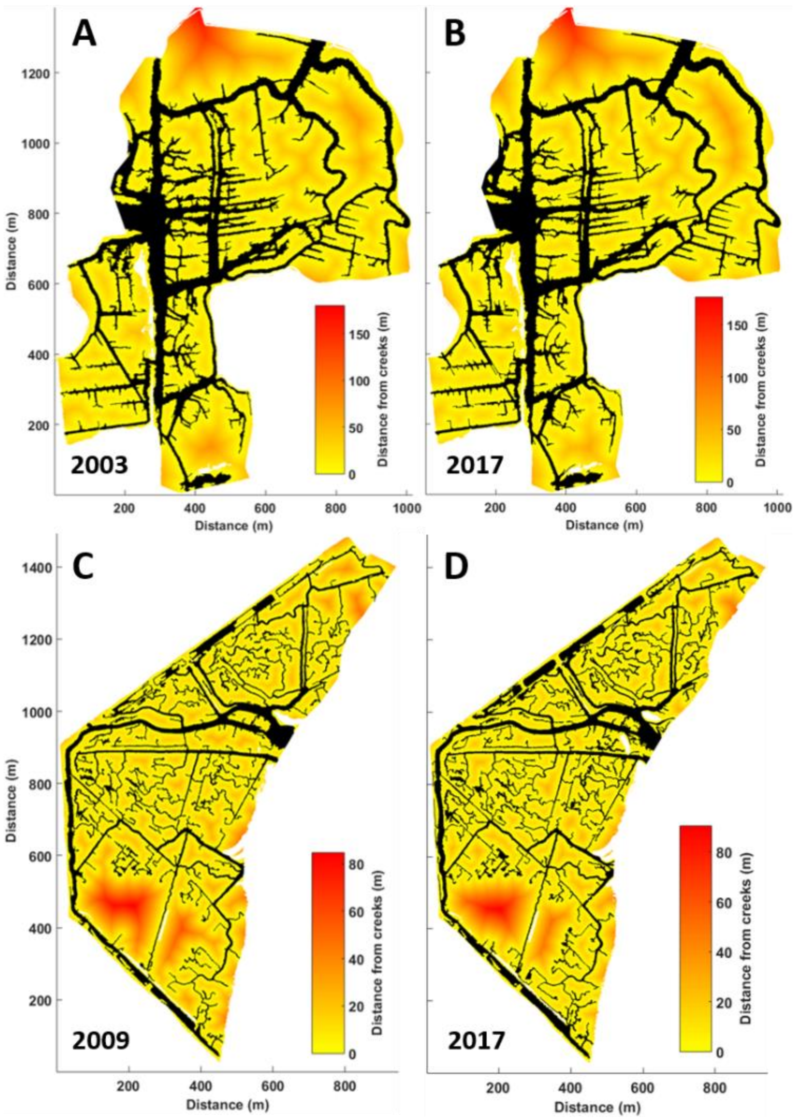


Figure 6.20: Changes in creek distribution across two accidentally realigned sites: Brandy Hole (A-B) and Foulton Hall (C-D) occurring between 2003-2017 and 2009-2017 respectively.

Table 6.6: MR characteristics evolution rates calculated by applying equation [22] to the best linear fit for two AR sites. Significant correlations at a 0.05 significance level are shown in bold. Linear trends greater than the 95% spread of the residuals are highlighted in light green, 68% in light yellow, 50% in dark yellow. Trends inferior to the mean of the residuals are highlighted in light red.

	Brandy Hole	**Foulton Hall**
Mean elevation over MWS in cm/yr (%/yr)	0.11 (0.08)	0.86 (0.53)
Drainage density in km/km^2/yr (%/yr)	-0.10 (-0.64)	0.11 (0.37)
Overmarsh path length in -m/yr (-%/yr)	0.07 (0.32)	0.05 (0.38)
Main channel length in m/yr (%/yr)	3.08 (0.33)	1.72 (0.27)
Total channel length in m/yr (%/yr)	-74.53 (-0.64)	62.31 (0.37)
Number of creeks in yr^{-1} (%/yr)	-5.99 (-0.79)	**10.96 (1.48)**
Mouth cross-sectional area in m2/yr (%/yr)	-5.53 (-1.47)	1.71 (2.99)
Mouth depth in m/yr (%/yr)	-0.06 (-1.47)	0.028 (2.88)
Planform area in m^2/yr (%/yr)	-361.87 (-0.18)	1238.56 (1.19)
Tidal prism in m^3/yr (%/yr)	264.95 (0.21)	-1666.95 (-3.78)
Main channel sinuosity in yr^{-1} (%/yr)	<0.01 (0.06)	<-0.01 (-0.57)
Main channel gradient in o/yr (%/yr)	0.05 (3.41)	-0.03 (-3.30)

Visually, few changes are seen for the two AR sites over the time period considered (Figure 6.20), and the evolution rates are generally lower than those calculated for the 10 MR sites (Tables 6.5 and 6.6). The creek networks at these two sites seem better distributed than in the more immature MR sites, with a dense network of creeks and OPL values of 22.0 m for Brandy Hole in 2017 and 12.7 m in 2017 in Foulton Hall (Appendix D1). Nevertheless, compared to the creek distribution of the natural sites studied in Chapter 5, a few empty patches remain. Furthermore, the straight interconnected channels inherited from the old drainage ditches remain clearly visible. These results appear to suggest that, 100 years after the breach, MR sites are likely to reach a stable state characterised by a dense network of creeks, but with visible remnants of the initial topography of the site.

6.4 Discussion

6.4.1 General evolution trends in MR schemes

This chapter provides the largest monitoring scheme of MR creeks to date, in terms of number of study sites, monitoring time and number of parameters considered. Indeed, 10 MR schemes in the UK have been analysed over 2 to 20 years of evolution through the systematic extraction of 24 morphometric creek parameters, as listed in Table 4.2. The second largest study to date monitors the dimensions of the entry channel (3 morphological parameters: width, depth, cross-sectional area of the outlet) in 3 restored coastal wetlands over 4 to 13 years (Williams et al. 2002).

Despite the variability of designs, creek networks in MR schemes display some similar morphometric characteristics. The low RS order creeks (entry channels and adjacent creeks) are generally built oversized. This design decision simplifies the excavation phase, ensures the stability of the breach area (Pendle et al. 2013), and is expected to allow the creeks cross-section to adopt more natural shapes over time via sedimentation (Zeff 1999; Pontee 2015a). It is also expected that, in a site constrained by flood defences like most MR schemes, the flow focused through the breach area and entry channel will be greater, requiring a larger breach area.

A number of changes have been observed within the creek networks of the 10 MR schemes. Overall, creek networks tend to become more complex over time, as expected from the conceptual evolution model for natural marshes. The number of high RS order creeks increases. The maximal RS order increases for half of the schemes, indicating a higher level of branching, and the OPL values decrease for all sites, indicating a better distribution of the creeks over the marsh. The parameters evolve following a trend rather than fluctuate around a mean value (Table 6.5). It is therefore suggested that they are evolving towards a state of equilibrium with the site elevation and the tidal forcings. The mean elevation increases following a regular trend, which is interpreted as a dominantly accretional behaviour in MR schemes, in agreement with previous studies (Mazik et al. 2010; Morris 2013).

While a linear trend is used here as a first approximation of parameters evolution, some creek parameters, like OPL and the site elevation within the tidal range, are expected to evolve following a decaying trend when considered over longer timescales. Indeed, elevation tends to plateau at a value near MHWS (Allen et al. 1992). OPL evolution is expected to first occur rapidly when the creek system grows in the lower sections of the site, near the breach area, and then slow down as the creek grows into the higher regions, where tidal energy and sediment import are lower. This behaviour is already visible at Abbots Hall, Allfleet, Chowder Ness, Freiston and Welwick, where the goodness of fit is better for the logarithmic than linear fit. Similarly, the changes in elevation at all sites except Chowder Ness, which is still undergoing significant accretion, and Steart, the youngest site, display equal or better goodness of fit for the logarithmic fit than for the linear fit.

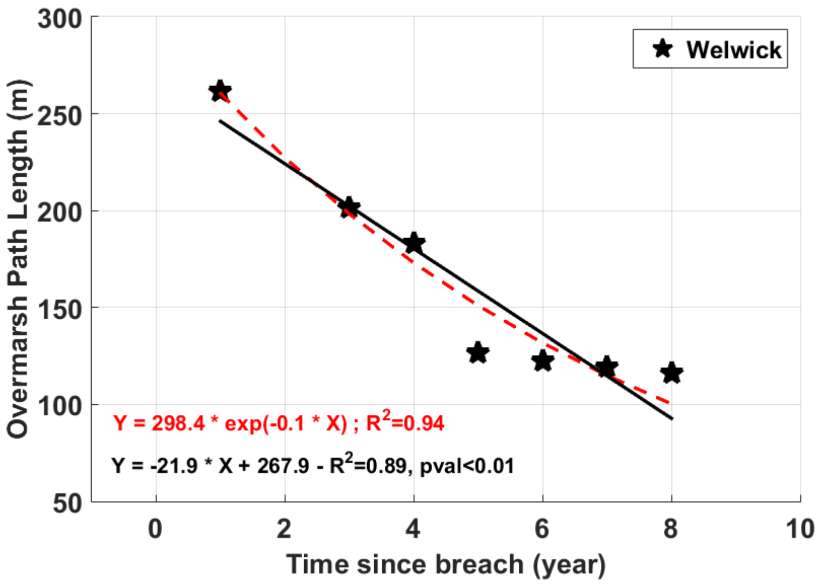


Figure 6.21: Linear (black) and exponential decay (red) fits for the overmarsh path length evolution at the Welwick MR site, showing a decaying trend over 8 years

This tentative observation is strengthened by the analysis of 2 AR sites, for which the creek extent has not changed significantly in the last decade (Figure 6.20), and for which evolution rates are low for all morphological parameters (Table 6.6). This seems to indicate that these sites have reached a state of equilibrium with their input conditions. Whether the AR equilibrium is equivalent to the natural equilibrium will be explored in Chapter 7.

While not as important a driver as the initial elevation of the site, the proximity to the creek system has an influence on accretion (Figure 6.19). As has been extensively covered in both natural and realigned coastal wetlands (Crooks et al. 2002; Fagherazzi et al. 2012; Oosterlee et al. 2017), the proximity to creeks has been found to have an impact on the elevation gains of most schemes up to 20 m away from the creeks. Levee building and/or growth of denser vegetation occurs preferentially near the creeks, in agreement with observations made in natural wetlands (Stumpf 1983; Reed et al. 1999).

While all creek networks have grown in some way, less than half of the sites display channel erosion into the substrate: Freiston, HOMW, Welwick and Paull Holme Strays. Their main channel length increases by 15 m/year (Freiston) to 75 m/year (Welwick), exceeding rates of natural creeks headcut migration measured in Norfolk saltmarshes (Steers 1960 *in* French et al. 1992). Interestingly, the main channel length evolution rate within Freiston is the same as that measured outside the site before MR implementation, suggesting that this scheme is successful at encouraging creek growth at the same pace as in natural, undisturbed mudflats (Symonds et al. 2007). Yet those rates remain below headward erosion rates recorded in disturbed natural creek systems, in response to a catastrophic increase in tidal prism due to breaching, which can be as high as 400-500 m/year (Knighton 1992; Symonds et al. 2007).

The 4 sites that display channel erosion into the substrate are morphologically dissimilar (various initial elevations and slopes, Figure 6.6) and do not display the highest tidal ranges in the dataset. Consequently, something beyond the morphological and hydrodynamic conditions prevents the unassisted incision of channels within the 6 other schemes. A possible cause is the compaction of the MR soils following their reclamation (Crooks et al. 2002; Pethick 2002). Indeed, Freiston, HOMW, Welwick and Paull Holme Strays had all been embanked for less than 50 years before realignment (Table 3.2). The sites may thus have a soil structure and chemistry closer to that of a saltmarsh, which would facilitate the natural development of creeks, as theorised by Burd (1995). A similar conclusion was reached by a previous study of Freiston and Tollesbury: Freiston, more recently embanked, had creeks develop into the agricultural soil while creek formation at Tollesbury was limited to the newly deposited sediment (Brown et al. 2007).

Other sites with a rapid creek development include Chowder Ness and Allfleet. Both sites are characterised by a low initial elevation and large breach areas or bank removal, encouraging sediment exchange and vertical accretion. The creek planform area increases more rapidly at Chowder Ness, probably due to the higher accretion rate and the lower marsh gradient, which allows new soft sediment where creeks can develop to occupy a greater portion of the scheme. However, the creek volume at Allfleet increases rapidly while the creeks at Chowder Ness remain shallow and morphologically dynamic, with more significant shifts in creek position. At Allfleet, a more fixed template was obtained by excavating initial creeks from field drains and digging oversized channels within the breach areas, resulting in a larger undermarsh tidal prism. At Chowder Ness, the lack of a clear channel outlet is expected to result in water exchange occurring preferentially through the marsh edge independently of the creek system, thus reducing creek-forming processes (French et al. 1992).

6.4.2 Limitations and future work

The multi-year analysis of creek evolution performed using lidar data in this chapter provides a quantitative and systematic assessment of MR creek morphometry. While results generally agree with field observations carried out by previous case studies as cited in Section 6.3.1, some anomalous values remain, which can be traced back to the limits of the available datasets and creek parametrisation techniques. A summary of these limits is given below.

The analysis of potential habitats in MR schemes (Table 6.3) is based solely on elevation data within the tidal range as detected by lidar. In reality, suitability for habitat creation depends on other aspects such as the slope, proximity to other marshes, soil properties and water quality (Parker et al. 2004). The tidal and wave action will also play a role in enabling or preventing plant colonisation. At Chowder Ness, despite the lidar data detecting a site elevation of a middle to high marsh since 2007, field observations have found the MR scheme to be dominated by tidal flats until 2012 (Pendle et al. 2013). This can be linked to the openness of the site to hydrodynamic action. After 2012, the site got colonised by vegetation due to continued accretion and is now evolving into a high marsh (Elliott et al. 2016).

Equating the elevation changes in lidar data with bare ground surface changes and with sediment accretion rates is a simplistic hypothesis that does not account for the effect of vegetation growth. Current vegetation correction algorithms used to produce DTMs are ill-suited to coastal wetlands where the vegetation cover is lower than the 0.15 m vertical resolution of lidar, as discussed in Section 3.2.1. As a result, this effect cannot be easily removed, and should instead be kept in mind during data interpretation. The best way around this issue is to compare lidar monitoring results with data collected in the field, such as accretion stakes or surface elevation tables from past studies (Wilson et al. 2014; Burgess et al. 2015). Independent, field-based topography surveys from past studies gave out consistent results with the ones from this study as shown in Section 6.3.1. Elevation changes have also been visually interpreted in terms of creek morphodynamic processes in Section 6.3.3 to mitigate this effect. There can also be discrepancies between bare ground surface changes and sediment accretion rates due to site-specific sub-surface processes like sediment compaction or below-ground accumulation of organic matter (Cahoon et al. 2000), which this remote-sensing based study cannot capture. The incorporation of these sub-surface processes for further studies would require either field data to estimates average yearly rates of compaction and organic matter production, which could then be used as correction factors when inferring the accretion rates.

Other anomalous results can be traced back to the creek parametrisation methods chosen, which are well adapted to mature natural creek systems, but less to evolving artificial ones. For instance, anomalously high sinuosity ratios are found for some very straight channels at Paull Holme Strays due to its especially interconnected creek structure inherited from perpendicular drainage ditches (Appendix I7). Similarly, strong dissimilitude between order-specific parameters from one year to another can be retraced to the chosen ordering system. Subtle changes in the distribution of high RS order creeks can lead to vastly different measured entry channel characteristics by changing the branching pattern (Figure 5.8). It may be necessary to develop other definitions of sinuosity and creek order that would be better suited to evolving artificial systems for further studies, as will be discussed in Section 9.2.

Finally, the limited number of datasets available complicates the estimation of creek evolution trends, especially in the early years of older MR schemes, where lidar data are sparser and noisier. The opposite problem arises for the recent scheme Steart, for which only 3 data points covering 2 years of development are available at the time of study, and for which linear evolution trends cannot be confidently inferred. The time needed to reliably assess MR evolution, be it for the creek network or other factors such as the biodiversity, is a debated issue and is probably underestimated by most studies. While most of the changes are expected to occur within the first two years, especially in the entry channel, it is generally considered that at least 5 years is needed for a MR scheme to evolve towards a mature state (Zhao et al. 2016). This is an underestimation as creek development in restored saltmarshes has been observed to take over 13 years (Williams et al. 2002), yet those values will be used in this thesis as a first approximation of the necessary MR evolution time, for practical reasons of data availability. The development of creeks between an early stage (<= 2 years), and a later stage (> 5 years), should be explored to link more accurately the evolution rates to the initial conditions and the design choices. Future comparative studies of coastal wetland evolution should be done over similar time periods, which should be made easier by the increasing frequency of lidar data collection in the UK since 2009.

6.5 Summary

Morphological changes occurring within 10 UK MR schemes were investigated with a focus on creek network morphometry in order to fulfil Thesis Objective 2: Undertake a morphometric analysis of creek systems in MR schemes in the UK and of their evolution over the years, in order to estimate creek evolution rates. Morphological data were extracted from lidar datasets for different periods using the creek parametrisation algorithm developed in Chapter 4. This study is the first to systematically record changes in the creek networks of a large number of MR sites and to look for common trends. Following implementation, MR creeks expanded by increasing the number of high RS order creeks and the level of branching, improving creek distribution through the site (lower OPL values). Over the timescales of 2 to 20 years considered, the scheme accretion rate and the creek growth follow linear trends that supersede inter-annual fluctuations. Those evolution trends could be a sign that MR creeks evolve towards a state of dynamic equilibrium with the tidal forcings, as would be expected of immature natural systems. The analysis of two AR sites suggests that equilibrium can be expected to be reached within 100 years or less. Significant differences nonetheless remain between natural and MR creeks. The latter are designed with a sparser network of oversized and overly straight creeks. Subsequent creek growth may curb these characteristics for the overall creek system, but the initial template is unlikely to be modified significantly. Indeed, of all the sites considered, only Freiston, HOMW, Welwick and Paull Holme Strays display signs of erosion into the substrate close to the creek network. Their main channel extension rate is equal or faster than the headcut migration rate of creeks in natural mudflats under normal conditions, but remains lower than observed headward erosion rates in disturbed creek systems following a sudden increase in tidal prism. These 4 sites have in common large channel outlets (cross-sectional area larger than 170 m^2) and a site embankment period of 50 years or less before realignment. The balance between hydrodynamic forces and the bed shear stress is expected to play a major role in the capacity of the creek system to develop unassisted. The next chapter's focus will be to determine whether MR creeks evolve towards the equilibrium state of natural systems, and which initial design choices or external forcings may encourage or impede this evolution.

Chapter 7: Do MR creeks evolve to resemble natural mature systems?

7.1 Introduction

Because creek networks are recognised as crucial to the continued health of MR saltmarshes, various design strategies have been tested in the UK to kick-start the development of creeks in MR schemes. Section 3.4 details 10 examples of MR creek design tested between 1995 and 2014. However, there is no evidence that the creeks resulting from those design strategies consistently evolve towards the morphology of mature natural creeks. The timescale necessary for MR creeks to reach the equilibrium state is also uncertain, as is the impact of to the initial design choices and external forcings in encouraging or impeding creek growth. For this project, the equilibrium state of creek networks is defined as the state where their planform and volumetric characteristics are maintained under stable input conditions, and where the system can keep up with incremental changes in input conditions without significant changes to the creeks' morphological characteristics. Due to the natural variability of creek network shapes highlighted in Chapter 5, equilibrium is defined as a range of potential states rather than as one quantifiable target. This chapter aims to estimate whether and how quickly existing MR creeks grow towards the equilibrium range expected of natural mature systems, and to investigate the potential drivers of creek evolution.

Results from Chapter 5 found that currently existing morphological equilibrium relationships are applicable to creek networks extracted from 1 m resolution lidar data, despite the lower resolution compared to field-validated manual creek mapping from aerial photography. These relationships provide a range of equilibrium for different creek parameters, which can be used as semi-quantitative end targets for MR creek design. Results from Chapter 6 then showed that, after implementation, MR creeks go through a phase of expansion (higher level of branching, larger volume, better sinuosity and better distribution through the site) that can be approximated by a number of linear evolution rates. The analysis of two AR sites also suggested that the creek morphology of these artificial systems should stabilise within 100 years. By bringing together the results from these two chapters, the capacity of current MR creeks to reach the equilibrium range characteristic of natural marshes, and the time needed to do so, is investigated.

This chapter addresses Thesis Objective 3: to compare the morphological characteristics of natural and MR creeks to determine whether MR creeks evolve towards more natural systems over time, and whether their evolution rates can be related to initial conditions and design choices. To that end, there are three sub-objectives as follows:

1) To compare the creek networks of natural and artificial coastal wetlands, and verify whether they should be considered as two statistically distinct groups;

2) To determine whether the MR creek parameters evolve towards morphological equilibrium, and to estimate the time to equilibrium;

3) To investigate the potential of initial design choices and external conditions to encourage or impede the evolution of creek networks towards equilibrium

The structure of this chapter is as follows. Section 7.2 provides the data and methods specific to this chapter. Section 7.3 lists the results for each sub-objective. Section 7.4 discusses the implications of the results and their limitations, and Section 7.5 provides a summary of the key findings.

7.2 Data and methods

7.2.1 Datasets used

In this chapter, the datasets from Chapter 5 (13 natural mature saltmarshes) and Chapter 6 (10 MR schemes) are combined to compare natural and artificial creek systems (Figure 7.1). Gibraltar Point and Crossens were excluded from the natural mature creeks dataset due to their poorly distributed creek systems as described in Section 5.3.3. MR schemes Paull Holme Strays, Tollesbury and Steart were also excluded from the natural-MR creek comparison because the available lidar data do not capture both the early (<= 2 years) and late (>= 5 years) stages, and so their evolution from an initial to a final state cannot be reliably estimated. Therefore 11 natural and 7 MR sites are compared to infer general conclusions on the evolution of MR creeks compared to natural systems (first sub-objective). Nevertheless, it remains of interest to investigate the current creek evolution trends, towards or away from equilibrium, for all 10 MR sites, to try and infer the time necessary to reach equilibrium (second sub-objective). Those observations will also help to discuss the consequences of various MR design choices in Chapter 8. Finally, the 2 AR sites from Chapter 6 are used to inform the expected morphology of MR creeks 100 years after site opening.

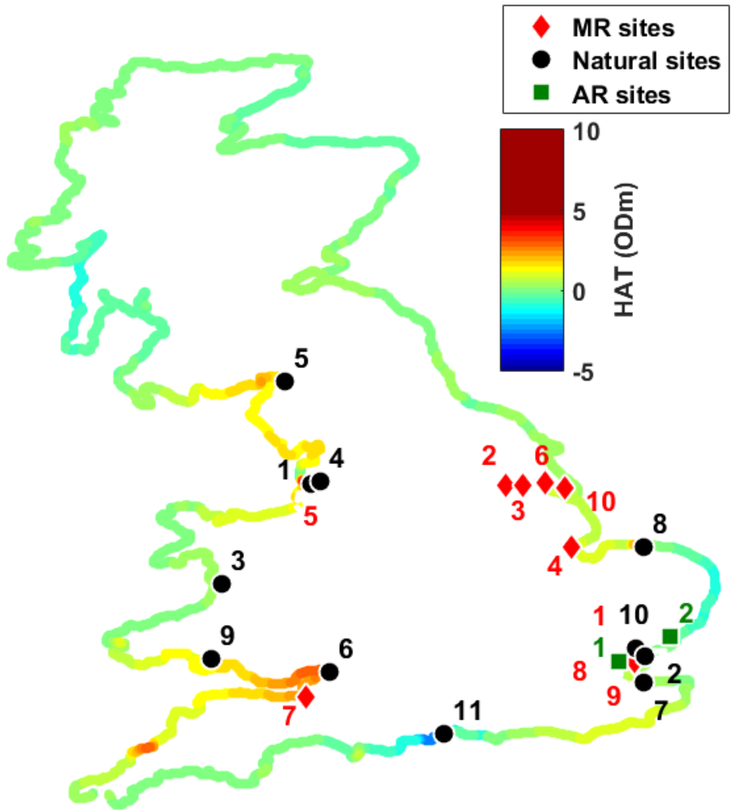


Figure 7.1: List of natural saltmarshes and managed realignment schemes used in this chapter. MR sites: 1=Abbots Hall; 2=Alkborough; 3=Chowder Ness; 4=Freiston; 5=Hesketh Out Marsh West; 6=Paull Holme Strays; 7=Steart; 8=Tollesbury; 9=Wallasea/Allfleet; 10=Welwick. Natural sites: 1=Banks; 2= Grange; 3=Hen Hafod; 4=Longton; 5=Newton Arlosh; 6=Portbury Wharf; 7=Shell Ness; 8=Stiffkey; 9=Tir Morfa; 10=Tollesbury; 11=Warren Farm. AR sites: 1= Brandy Hole; 2= Foulton Hall.

7.2.2 Comparison of initial and final state of MR schemes with natural creeks

In order to fulfill the first sub-objective, creek networks in MR schemes at their early stage of evolution (< 2 years), at their later stage of evolution (> 5 years), and natural creek systems were compared. This was done using a non-parametric Kruskal-Wallis (KW) test of ranks at a 95 % confidence interval to test whether they statistically belong to the same overall distribution. Kruskal-Wallis is used to compare non-normal distributions of different sizes, and is well suited to the statistical analysis of channel morphology (Opdyke et al. 2006). The creek parameters extracted from the 2 AR sites are also plotted over the natural and MR creek distributions, to verify to which group they are the closest.

In addition, the small scale (<10 m) topography of MR schemes and natural saltmarshes are compared. Indeed, small-scale topography is expected to play a dominant role on the ecological functioning of coastal wetlands (Callaway 2005) and on the development of channels (D’Alpaos et al. 2007). The ground heterogeneity was extracted from 365 data points over a 50x50 m^2 window in the lidar data for each year. The relative positions of the sampling points follow a systematic grid sampling method previously tested in both natural and MR saltmarshes (Brooks et al. 2015; Lawrence 2018) (Figure 3.2), as detailed in Section 3.2.1.

The analysis was repeated within the 7 MR schemes, 2 AR sites and 11 natural sites at regular intervals of 70 to 400 m depending on the size of the saltmarsh considered, in order to capture the entire area (Figure 7.2). The window of minimum standard deviation for elevation was then selected in order to avoid the largest channels, which are expected to positively skew the ground heterogeneity values, especially in MR schemes where the channels are generally oversized. Building on results from Brooks et al. (2015), which found the starkest differences in ground heterogeneity between natural and MR saltmarshes to occur between locations up to 10 m apart, the ground heterogeneity was defined as the mean elevation difference between points up to 10 m apart. The ground heterogeneity of natural and artificial marshes were then compared with a Kruskal-Wallis test at a 95 % confidence interval.

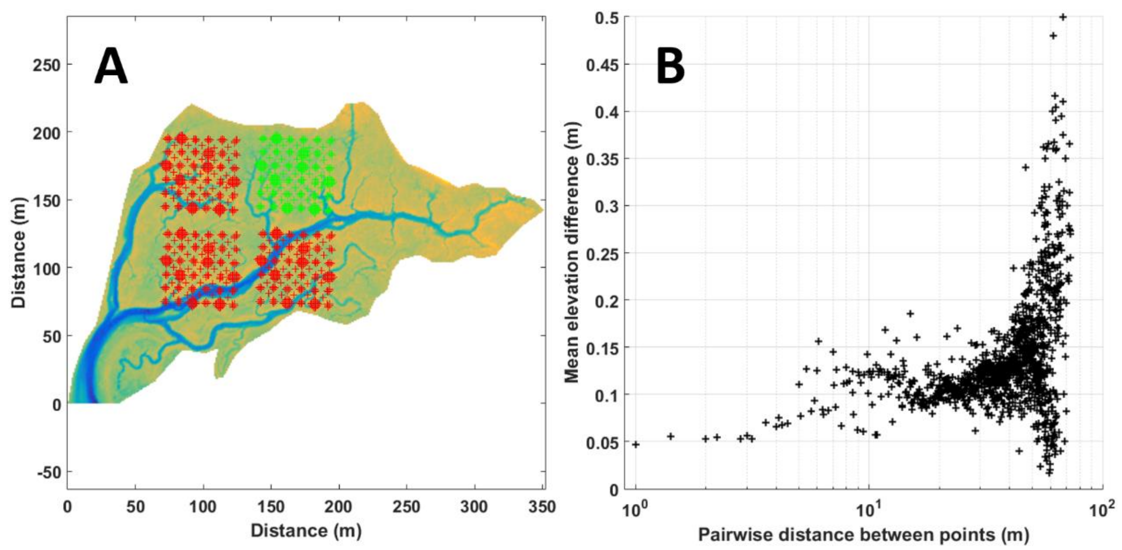


Figure 7.2: Ground heterogeneity calculation at Hen Hafod (natural saltmarsh); A: 50x50 m^2 windows of data sampling applied every 70 m within the site. The window with the smallest elevation standard deviation is the least influenced by channels and is thus selected to give the marsh topography. B: Scatter plot of mean elevation difference per pairwise difference between points. The mean value of elevation difference for points <10 m apart gives the ground heterogeneity.

7.2.3 Creek growth extrapolation towards equilibrium

In order to fulfill the second sub-objective, the evolution of all 10 MR schemes towards a mature state was investigated using the morphological equilibrium relationships validated in Section 5.3.2. The position of the 2 AR sites in 2017, about 100 years after accidental breaching, relatively to the natural equilibrium range, is also investigated. The list of equations is as follows, given along with their predictive value estimated by the mean absolute percentage error (MAPE):

Total channel length (m) vs. catchment area (m^2): $y=1.5x^{0.7}$ (MAPE=54%) **[5]**

Main channel length (m) vs. catchment area (m^2): $y=1.6x^{0.5}$ (MAPE=34%) **[8]**

Main channel depth (m) vs. catchment area (ha): $y=0.91x^{0.21}$ (MAPE=25%) **[14]**

Mouth cross-sectional area (m^2) vs. tidal prism (m^3) $y=0.04x^{0.66}$ (MAPE=30%) **[16]**

OPL (m) vs. mean marsh elevation above MWS (m) $y=3.49x-1.03$ (MAPE=39%) **[24]**

Due to the low predictive values of the relationships, equilibrium is defined in this section as a range rather than an equilibrium state, as discussed in Section 5.4. The time needed to reach equilibrium is extrapolated from the linear evolution rates estimated in Section 6.3.2. Evolution trends greater than at least the mean plus the standard deviation of the residuals are considered a good first approximation of the growth trend.

7.2.4 Relationships between initial conditions and creek evolution rates

In order to fulfill the third sub-objective, inter-correlations between the starting conditions, the design choices and the evolution rates of each MR creek system, as calculated in Chapter 6, were investigated. This part of the analysis excluded Steart because the site is too recent for any significant creek evolution to have occurred. MR creeks were ranked from slower to faster evolution towards equilibrium using pie plots. The mouth cross-sectional area and undermarsh tidal prism (creek volume) were chosen to represent the volumetric changes within the creek network, while the number of creeks, total channel length and planform area represent the planimetric changes occurring within the creek. Finally, the OPL and sinuosity ratio represent the creek spatial distribution changes. Together these parameters give an approximation of the total evolution rate of each creek. The goal is to compare which group of variables evolve faster for each site, and whether this can be linked to the design choices.

General correlations between the creek evolution rates, initial conditions and design choices were explored using PCA (see Section 5.2.3 for explanation) and correlation matrixes. The parameters for this analysis were separated into 4 groups depending on their relevance to creek evolution (Table 7.1). All variables were standardised by subtracting the mean and dividing by the standard deviation in order to account for the effects of scale and different units (length, area, volume, time). The parameters chosen are not exhaustive of the factors influencing creek evolution, and depend on data availability in open databases and in the literature. Each group of data is described below:

Table 7.1: List of parameters selected for PCA and correlation analysis due to their relevance to creek design and evolution.

Parameters selected for PCA	Relevance to creek evolution
MSTR (m)	Tidal forcings (group 1)
HAT (m)	
Tidal asymmetry (ratio)	
Time embanked (years)	Initial MR morphology (group 2)
Catchment area (m^2)	
Initial elevation above MWS (m)	
Elevation gradient (ratio)	
Mean local slope (°)	
Ground heterogeneity (m)	
Largest initial outlet area (m^2)	Initial breach and creek design (group 3)
Largest initial outlet depth (m)	
Largest initial outlet width (m)	
Total initial outlet area (m^2)	
Total initial breach width (m)	
Initial TP/Catchment area (m^3/m^2)	
Initial PA/Catchment area (m^2/m^2)	
Initial TCL/ Catchment area (m/m^2)	
Initial OPL (m)	
Mean grain size in nearby estuary (µm)	Sediment characteristics (group 4)
Turbidity in nearby estuary (ftu)	
Mean elevation over MWS (%/yr) -MWS	Marsh elevation and creek evolution rates (group 5)
Drainage density (%/yr) -DD	
Overmarsh path length (-%/yr) -OPL	
Main channel length (%/yr) -MCL	
Total channel length (%/yr) -TCL	
Number of creeks (%/yr) -NB	
Mouth cross-sectional area (%/yr) -CSA	
Mouth depth (%/yr) -D	
Planform area (%/yr) -PA	
Tidal prism (%/yr) -TP	
Main channel sinuosity (%/yr) -MCS	
Main channel gradient (%/yr) -MCG	

The tidal forcings (group 1), consists of MSTR, HAT and tidal asymmetry. They were obtained from the Admiralty Tide Tables for all MR schemes as explained in Section 3.2.2. The initial MR morphology variables (group 2) include the time embanked, size, mean elevation, elevation gradient between the breach area and the landward reaches of the site, mean local slope corresponding to the first derivative of the elevation map, and ground heterogeneity as shown in Figure 7.2 for each scheme (Appendix L1). The initial breach and creek design variables (group 3) were extracted from the lidar data using the creek parametrisation algorithm, and encompass the dimensions of the breaches and the initial size and distribution of the creek network. The sediment characteristics (group 4) available for this study include the mean grain size in the wider estuary, measured in µm, as well as the water turbidity in the wider estuary, measured in ftu (Formazin Turbidity Units), as explained in Section 3.2.3. Finally, the marsh elevation and creek evolution rates (group 5) are taken from Table 6.5. Linear evolution trends that fail to exceed the 68% spread of the residuals (light red and dark yellow in Table 6.5) are considered a poor representation of the evolution trend and brought to zero.

In the absence of systematic data collection of shear strength, bulk density or soil porosity in MR schemes, the number of years for which the site was embanked prior to breaching (time embanked) may give some indication of the compaction of the sediment. Indeed, changes to sediment structure due to compaction can be expected for sites that have been embanked longer and used as arable land (Spencer et al. 2012). Previous land use also has an impact on the biogeochemical properties and functioning of the sediment (Spencer et al. 2012), but such considerations are beyond the scope of this study. Time embanked also depends on the human history of the region. Sites characterised by a lower tidal range require lower defences against Spring tides, and so may have been reclaimed earlier.

7.3 Results

7.3.1 MR creek morphological changes after 5 years

The first sub-objective is to compare the characteristics of pre- and post-evolution MR schemes with those of natural mature saltmarshes. For most of the morphological parameters considered, including creek number, length, volume and bifurcation ratio per RS order, MR creek networks are not statistically different from natural systems (Figure 7.3). However, some differences exist between natural and immature MR sites (< 2 years after implementation). MR creeks are wider and less sinuous, and the high RS order creeks have a lower WD ratio, so the narrower and deeper shape is probably due to the reactivation of drainage ditches during the design phase. In general, the branching complexity of the creeks is lower than in natural systems, with maximum RS orders ranging from 2 to 4, versus 3 to 5 for the natural creek networks considered. In summary, young MR sites tend to have a simplified template of wider, deeper and straighter channels, with less branching out and fewer small creeks. The small creeks that do exist are generally inherited from old drainage ditches, which gives them a deeper, narrower shape than natural creeks.

As the creek networks evolve following site implementation, they become larger and more complex. In their final state (> 5 years of evolution), the branching complexity of MR creeks increased to a maximum RS order of 5 across the 10 MR schemes considered because of the growth of smaller channels (Figure 7.4). MR creek sinuosity also increases until it becomes statistically undistinguishable from natural systems, for all RS orders except RS order 3. Two mechanisms are at play here: (1) in the entry channel (RS order 1), the sinuosity is increased by a few sites like Alkborough, Allfleet and Welwick, in which the main channel extends landwards and leads to a higher sinuosity (Appendices H2, H3 and H10); (2) in the smaller, high RS order channels, the sinuosity increases as new, more naturally-shaped creeks form. Pre-excavated creeks do not evolve much from their initial template, as observed in Section 6.3.1, which explains why intermediate creeks of RS order 3 remain overly straight.

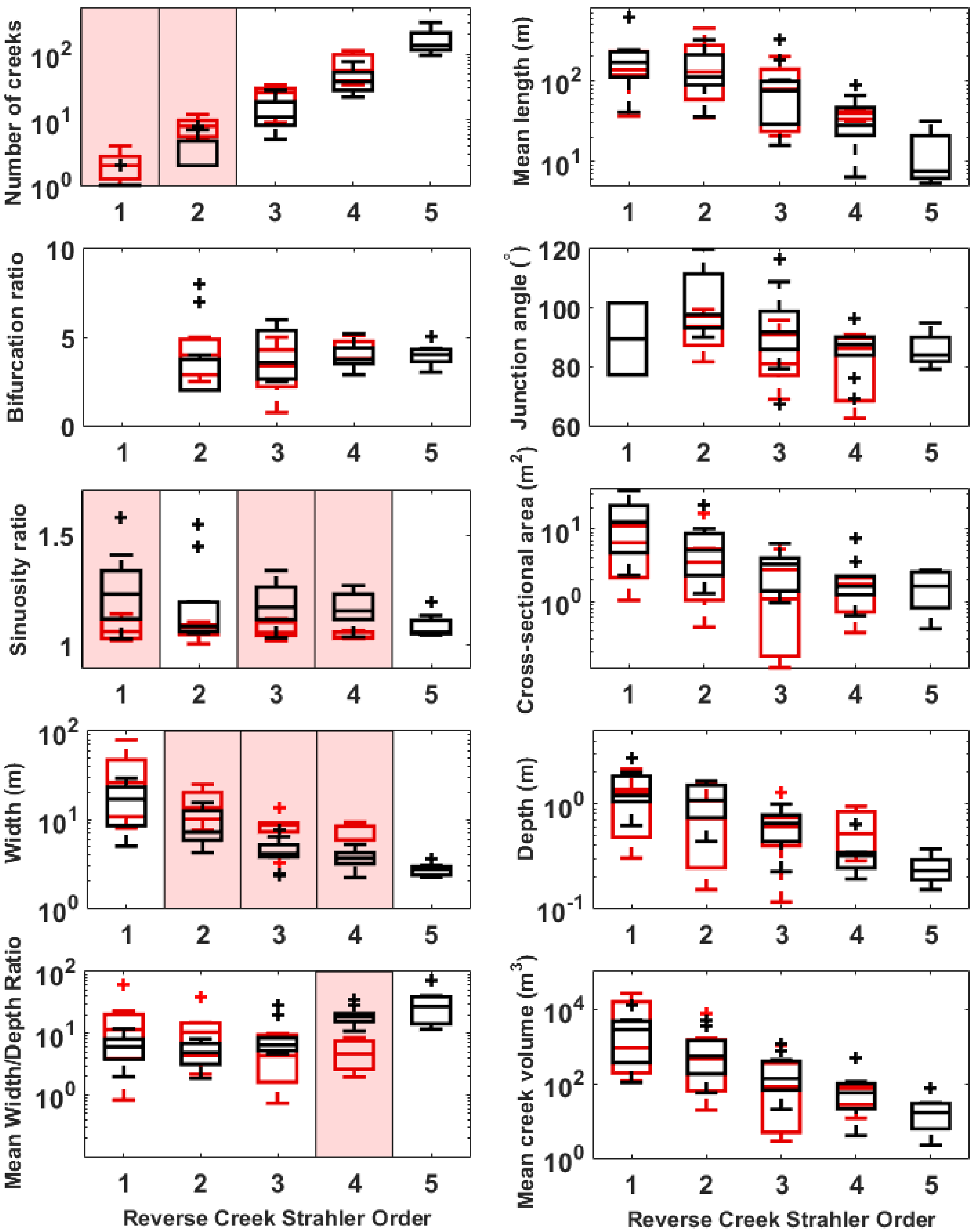


Figure 7.3: Comparison of morphological creek parameters per RS order for 7 initial MR schemes in red (older dataset available, < 2 years after implementation) and 11 natural saltmarshes in black. Statistically different groups according to Kruskal-Wallis test at a 95 % confidence interval highlighted in pink.

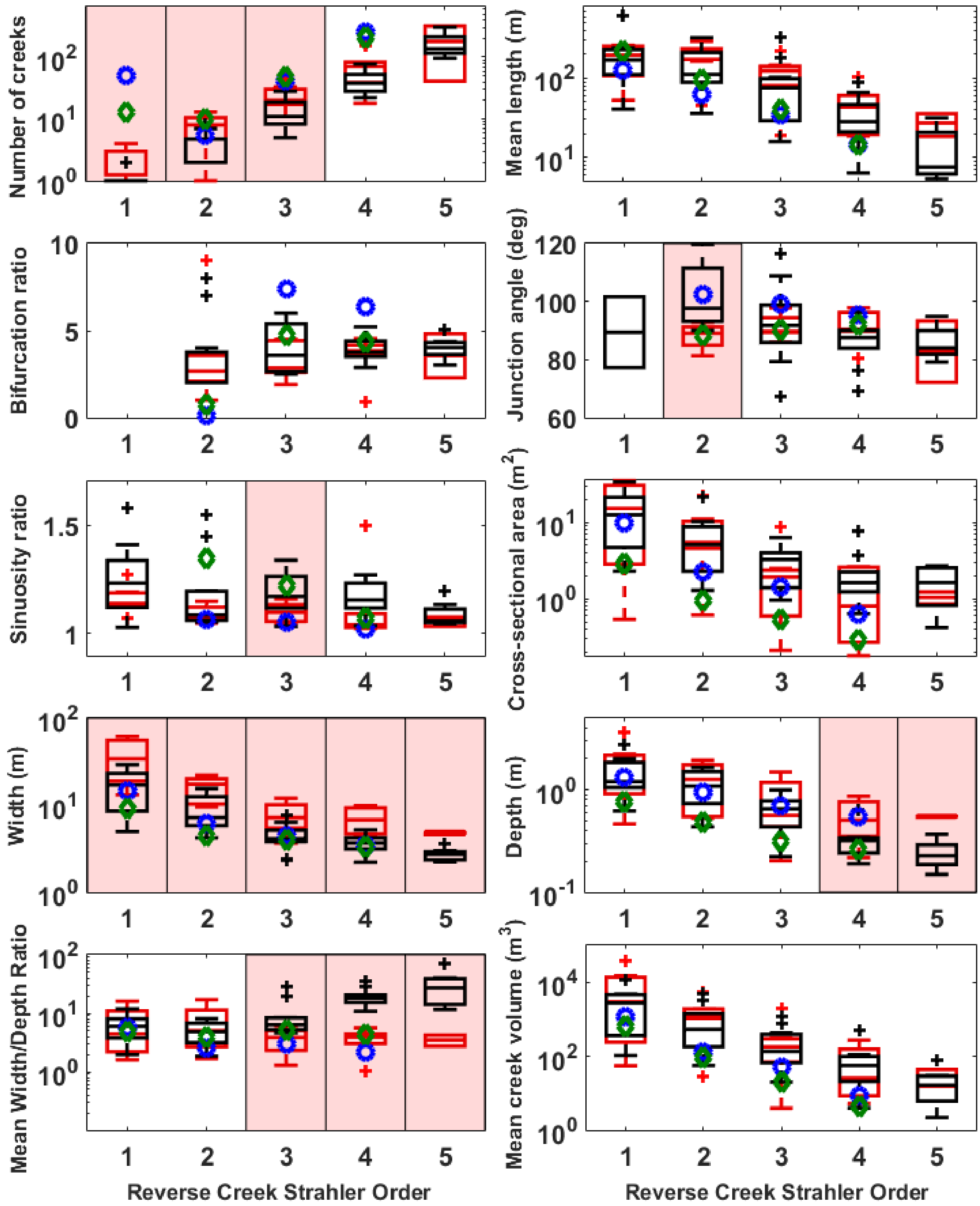


Figure 7.4: Comparison of morphological creek parameters per RS order for 7 final MR schemes in red (most recent dataset available, > 5 years after implementation), 11 natural saltmarshes in black, statistically different groups according to Kruskal-Wallis test at a 95 % confidence interval highlighted in pink. The two AR sites are plotted as blue circles (Brandy Hole) and green diamonds (Foulton Hall).

For other morphological parameters, MR creeks grow apart from their natural counterparts over the years. First, RS order 2 creeks display a lower junction angle. The wider breach areas allow for a cluster of entry channels to form at low angle with each other in some MR sites such as Chowder Ness, Tollesbury and Welwick (Appendices H4, H9 and H10). In addition, MR creeks start out overly wide and keep widening, especially around the entry channels where the erosive action is greatest. As a result the new terminal creeks being formed at RS order 4 and 5 are too deep and wide. The depth exaggeration is greater than the width exaggeration, leading to a comparatively narrow cross-sectional shape (Figure 7.5). This is the case at Alkborough, where most of the newest channels form through the reactivation of preexisting ditches (Figure 6.8). Another possibility is that artificial creeks are excavated with ditch-like cross-sections by designers, but this is not obvious from the W/D ratios at HOMW and Steart (Appendices H7 and H9). Finally, the exacerbation of the leptokurtic shape in high RS order MR creeks after 5 years of evolution could be due to the overcompaction of MR scheme soils, by allowing down-cutting of channels but hindering bank slumping.

The creek morphological parameters of the two AR sites are very different, suggesting that MR sites can reach a variety of equilibrium morphologies depending on their initial conditions. Both systems have a large number of creeks, but the creeks at Foulton Hall are highly sinuous while those at Brandy Hole have low sinuosity ratios. Foulton Hall also displays lower values of creek depth and cross-sectional area than Brandy Hole. For both sites, the creek width falls within the range of the natural sites. AR creeks of RS order 3 and 4 also tend to have lower W/D ratios than natural sites.

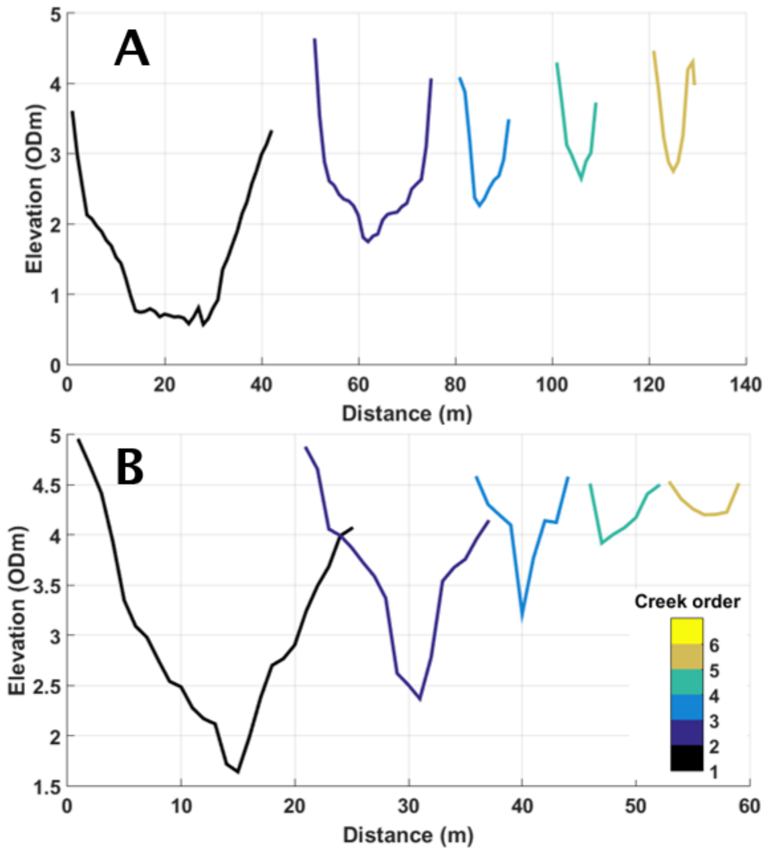


Figure 7.5: Representative creek cross-section per RS order for a MR scheme (A: HOMW 2014) and a natural mature saltmarsh (B: Longton)

A more general representation of the creek network evolution in MR schemes is shown in Figure 7.6. After > 5 years of evolution, MR creek networks reach statistically similar dimensions (planform area and volume standardised by catchment area) as their natural counterparts. However, they are not as dense nor evenly distributed through the site. MR sites have lower drainage densities and are “emptier” of creeks. The two AR sites display a larger planform area than both natural and MR sites, which in the case of Brandy Hole is correlated with a larger creek volume. These two sites also have a better distribution of creeks than the MR sites, but neither the OPL nor the drainage density fall within the range of the natural sites. Foulton Hall displays a better drainage density and a better distribution of creeks than Brandy Hole.

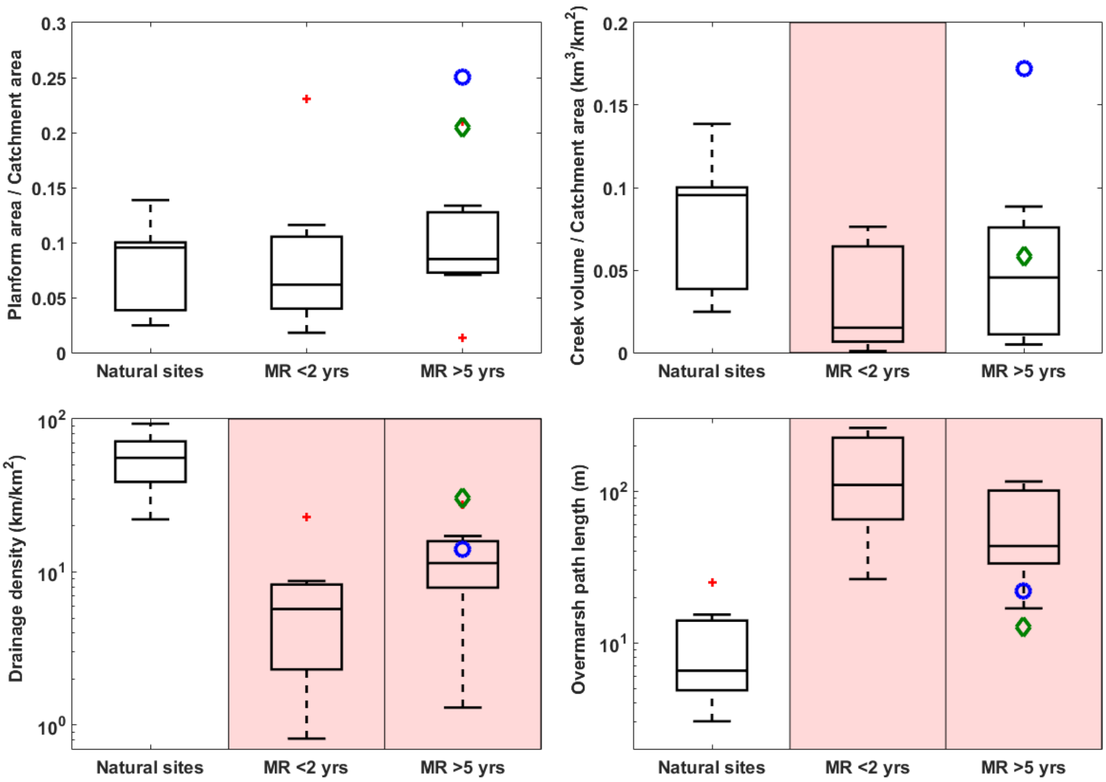


Figure 7.6: Comparison of morphological parameters of the total creek system for 11 natural and 7 MR sites; statistically different groups from natural sites according to Kruskal-Wallis test at a 95 % confidence interval highlighted in pink. The two AR sites are plotted as blue circles (Brandy Hole) and green diamonds (Foulton Hall).

MR sites also have a lower small-scale ground heterogeneity at both stages of evolution compared to natural saltmarshes (Figure 7.7). The ground heterogeneity does not change significantly after >5 years of evolution, except at Abbots Hall, where the minimal ground heterogeneity for points <10 m apart changes from 0.26 m to 0.10 m between 2002 and 2015. The cause for this change is unclear. Abbots Hall has undergone limited accretion since its implementation, so it has not been infilled by a flat layer of soft sediment. The anomalously high heterogeneity detected in 2002, before the breach, may be due to the presence of terrestrial vegetation which would have subsequently died off. The ground heterogeneity measured in the two AR sites falls within the range of the MR sites, and is lower than that of the natural sites.

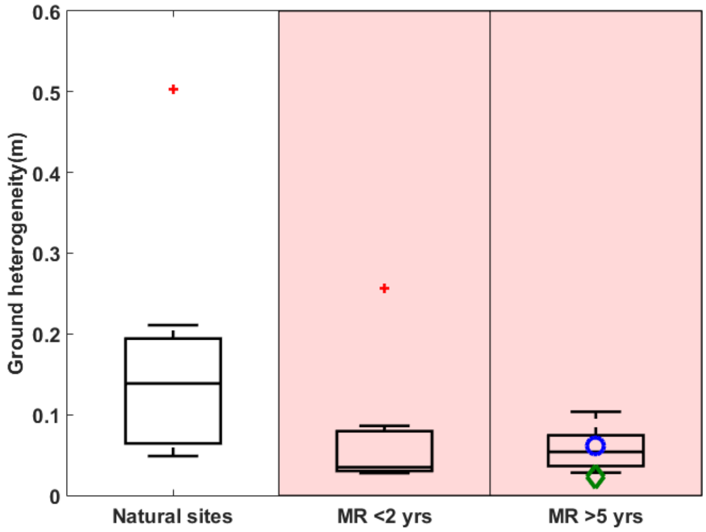


Figure 7.7: Comparison of (minimal) ground heterogeneity at locations <=10 m apart for 11 natural and 7 MR sites. Statistically different groups from natural sites according to Kruskal-Wallis test at a 95 % confidence interval highlighted in pink. The two AR sites are plotted as blue circles (Brandy Hole) and green diamonds (Foulton Hall).

The first sub-objective was to compare the characteristics of pre- and post-evolution MR schemes with those of natural mature saltmarshes. For some creek parameters, such as the total channel length and the depth of the creek segments, MR schemes >5 years after implementation and natural systems are not statistically different. However, MR creeks remain overly straight and are poorly distributed compared to natural sites. The ground heterogeneity of MR sites is also significantly lower than that of natural systems. The two century-old AR sites considered fall within the MR range rather than the natural range for both the creek distribution and the ground heterogeneity, suggesting that these morphological differences are potentially long-lasting in MR schemes.

7.3.2 MR creek evolution towards equilibrium

The second sub-objective is to determine whether the MR creek parameters evolve towards morphological equilibrium, and to estimate the time to equilibrium. The creek morphological parameters are plotted against the equilibrium lines to examine their respective evolution trends and extrapolate the time needed to reach the equilibrium range. After 5 years of evolution or longer, most MR schemes fall within the range of predicted equilibrium parameters for main channel length, total channel length and main channel mouth depth, or can be expected to reach that range within 10 years (Figures 7.8-7.10). A few MR sites fail to reach their expected equilibrium ranges within the timescales considered: 1 site (Alkborough) does not reach the equilibrium range for the total creek length; 3 sites (Steart, Freiston and Tollesbury) fail to stay within the equilibrium range for the main channel mouth depth.

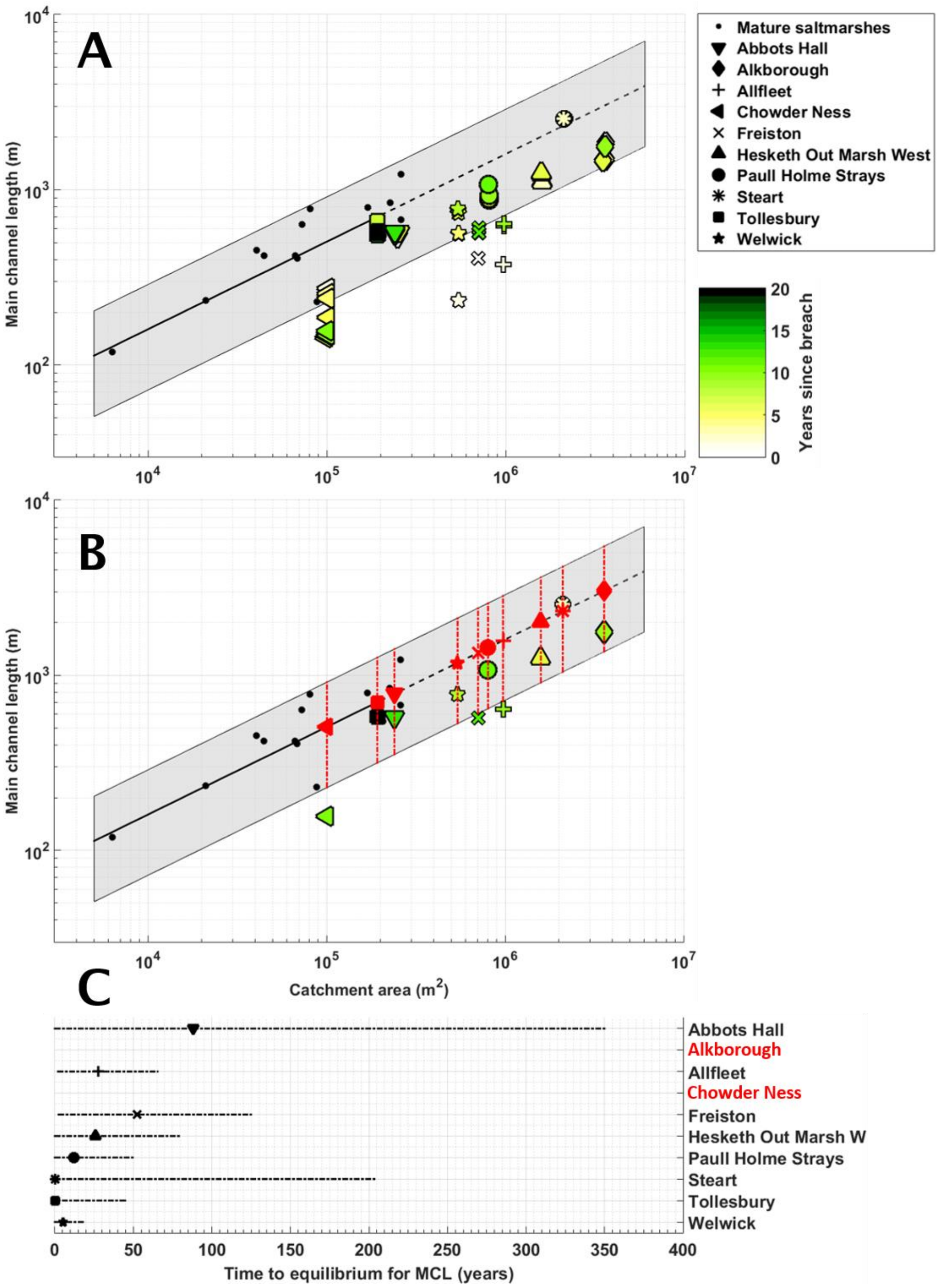


Figure 7.8: A: Evolution of MR schemes' main channel length versus catchment area towards the equilibrium range (grey shading); B: projected equilibrium range for each MR scheme plotted in red; C: time necessary to reach the equilibrium range extrapolated from linear evolution rates for each scheme except Alkborough (linear evolution trend inferior to the mean of the residuals) and Chowder Ness (evolves away from equilibrium). Error bars at A and B smaller than the markers.

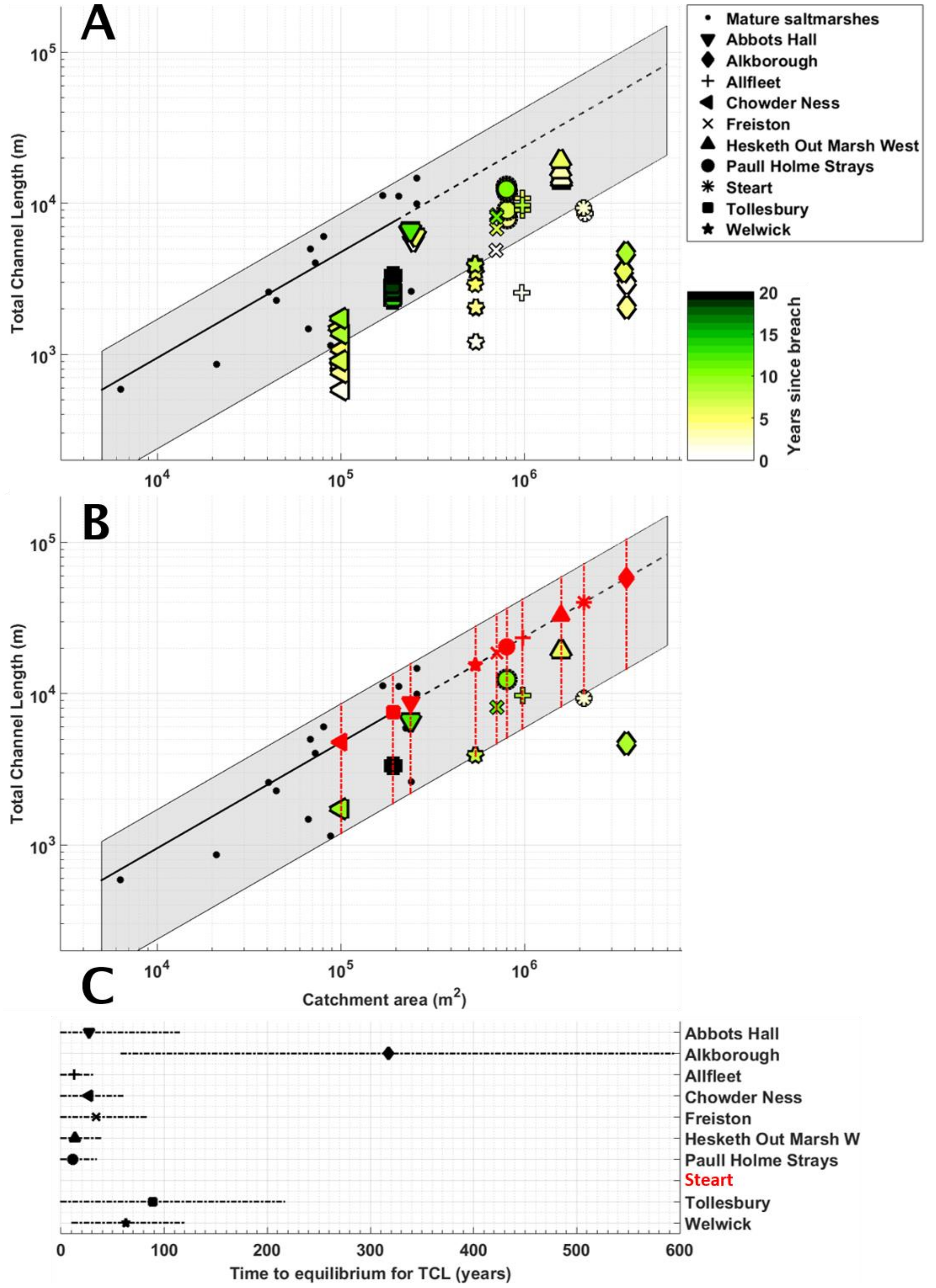


Figure 7.9: A: Evolution of MR schemes' total channel length versus catchment area towards the equilibrium range (grey shading); B: projected equilibrium range for each MR scheme plotted in red; C: time necessary to reach the equilibrium range extrapolated from linear evolution rates for each scheme except Steart (linear evolution trend inferior to the mean of the residuals). Error bars at A and B smaller than the markers.

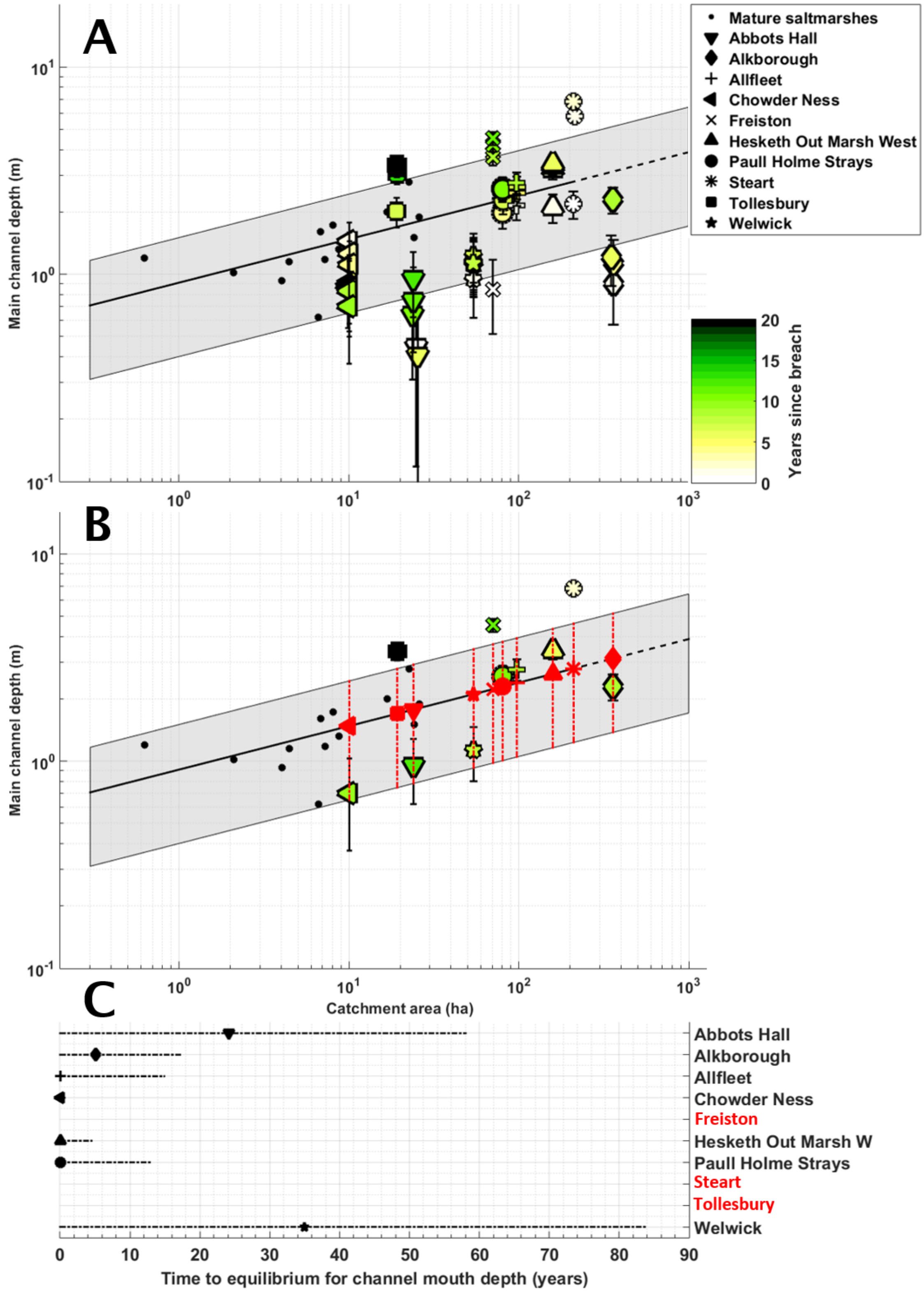


Figure 7.10: Evolution of MR schemes' main channel mouth depth versus catchment area towards the equilibrium range (grey shading); B: projected equilibrium range for each MR scheme plotted in red; C: time necessary to reach the equilibrium range extrapolated from linear evolution rates for each scheme except Freiston, Steart and Tollesbury (evolve away from equilibrium). Error bars are exaggerated by the logarithmic scale.

Alkborough has been designed with too few channels for its catchment area. The total channel length starts far below the equilibrium range and, assuming a continuous linear increase, will take 60 years to reach the equilibrium range, and 320 years to reach its ideal equilibrium state (Figure 7.9C). The slow evolution may also be due to Alkborough's armored breach area, which limits water exchange and creek development (Appendix C1). The main channel length evolution at Chowder Ness is difficult to interpret due to the concomitant shrinking of the initial main channel and expansion of a new one (Figure 6.10).

At Steart, Freiston and Tollesbury, the main channel mouth depth surpasses the equilibrium range over the timescales considered (Figure 7.10A). In some sites this can be explained by the tidal range. Steart lies close to MHWS level (Burgess et al. 2013) and is hypertidal, consequently rapid erosion is occurring within the entry channel to accommodate the elevation gradient with the Parrett River MLWS level, about 11 m below. However, at Freiston and Tollesbury, the tidal range and mean elevation above MWS are lower than at HOMW, so other processes must play a role.

Conversely, while the main channel mouth depth at Chowder Ness still lies within the equilibrium range at the time of study, it is evolving away from equilibrium (Figure 7.10) while a new main channel starts forming. Despite its mean elevation above MHWN, the site seems to behave like a tidal flat with shallow, unstable creeks corresponding to stage A of creek development in Figure 2.9. According to aerial photographs, vegetation colonisation has occurred at Chowder Ness between 2012 and 2015, in what can be interpreted as a mudflat-saltmarsh transition period. More monitoring will be needed to determine whether this vegetation will contribute to the stabilisation of channels and facilitate their deepening to equilibrium levels in future years.

The relation between the creek volume and the outlet area is harder to interpret. 3 MR sites (Abbots Hall, Alkborough and Chowder Ness) fluctuate around the equilibrium line and do not display clear linear trends. The other 6 MR sites (Allfleet, Freiston, Paull Holme Strays, Steart, Tollesbury and Welwick) all lie above the equilibrium range (Figure 7.11). The latter sites evolve towards equilibrium by increasing their creek volume to accommodate the amount of water being exchanged through the outlets. Assuming that the outlets' cross-sectional areas stabilise while the tidal prism keeps evolving linearly, reaching the equilibrium would take a few years for Steart, over 20 years for Allfleet, Freiston, HOMW and Paull Holme Strays, and over 100 years for Tollesbury and Welwick.

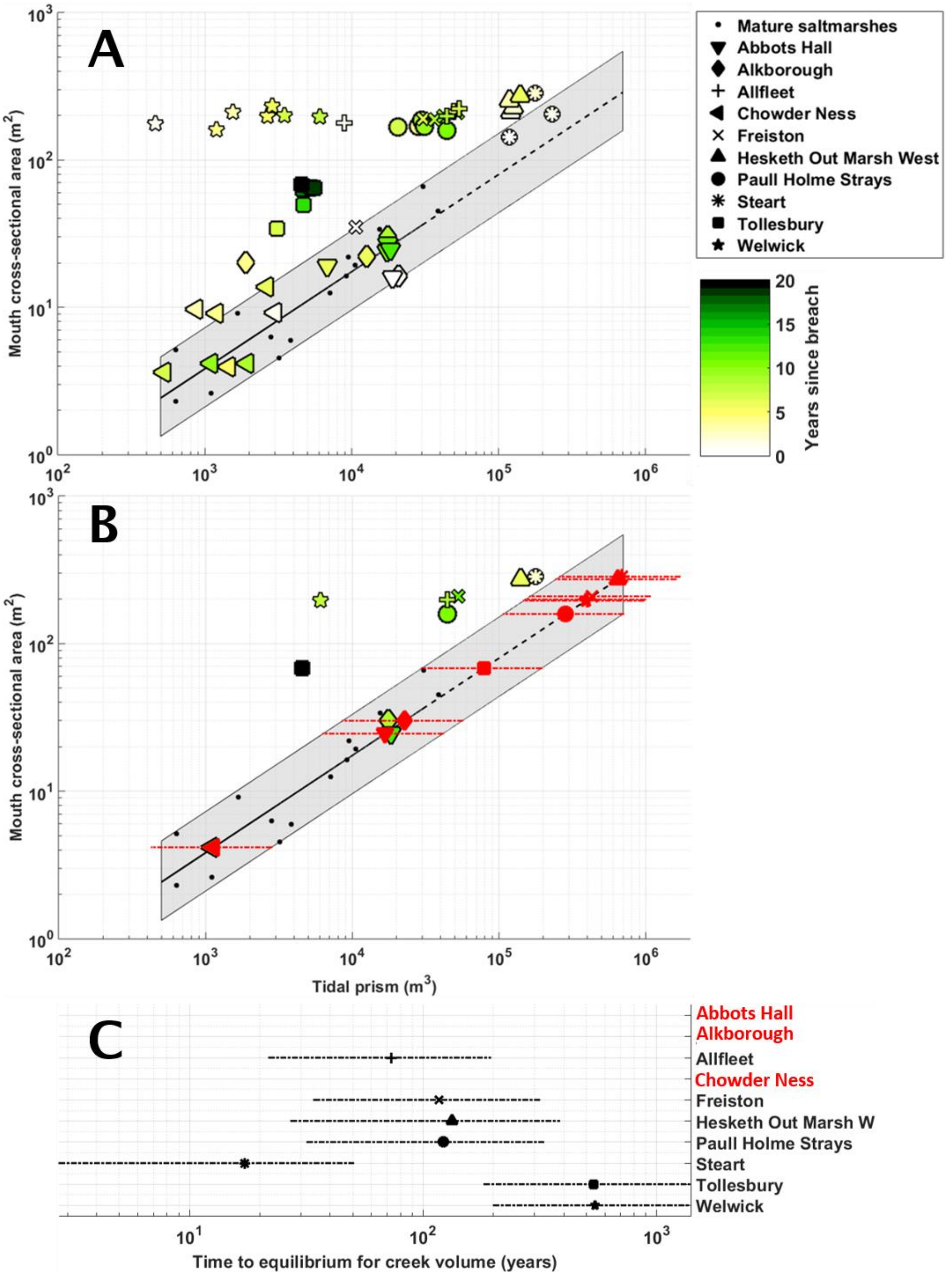


Figure 7.11: A: Evolution of MR schemes' mouth cross-sectional area (CSA) versus tidal prism (TP) towards the equilibrium range (grey shading); B: projected equilibrium range for each MR scheme plotted in red, assuming CSA stops evolving and TP keeps evolving linearly; C: time necessary to reach the equilibrium range extrapolated from linear evolution rates for each scheme except Abbots Hall, Alkborough and Chowder Ness (linear evolution trend fails to exceed the 68% spread of the residuals). Error bars at A and B smaller than the markers.

All 10 MR sites evolve towards a better distribution of their creek network across the site, as shown by the decreasing values of OPL (Figure 7.12A). Closest sites from the equilibrium range are Chowder Ness (rapid accretion in a relatively small site which facilitates creek formation) and Paull Holme Strays (due to the reactivation of old drainage ditches which expand the creek network). OPL decreases at HOMW due to the infill of head-channel ponds and creation of new creeks into the softer sediment. Nevertheless, OPL is the only parameter among those considered where none of the MR schemes has yet reached the range of equilibrium (Figure 7.12B). 6 of those sites are expected to reach that stage within 10 years or less (Abbots Hall, Allfleet, Chowder Ness, Freiston, Paull Holme Strays and Welwick) when assuming a continuous linear trend of OPL evolution (Figure 7.12C). HOMW is expected to reach the equilibrium range after 12 years, Alkborough after 19 years, Tollesbury after 27 years and Steart after 51 years (Figure 7.12C).

However, as discussed in Chapter 6, over longer timescales than those considered in this thesis, OPL and the site elevation are more likely to evolve following a logarithmic curve rather than a linear one. This behaviour is already visible for 5 of the 10 MR sites: at Abbots Hall, Allfleet, Chowder Ness, Freiston and Welwick, the logarithmic fit of OPL evolution over time yields higher R^2 values than the linear fit. Similarly, the marsh elevation evolution rates give out equal or better R^2 values for the logarithmic fit than for the linear fit, for 8 sites (all except Chowder Ness, which is still undergoing significant accretion, and Steart, the youngest scheme). Assuming decaying trends for marsh accretion and OPL, only Chowder Ness (15-71 years) and Paull Holme Strays (11-84 years) can be expected to reach the equilibrium range in under a century (Figure 7.13B). Tollesbury and Alkborough are expected to not reach the equilibrium range even after 500 years.

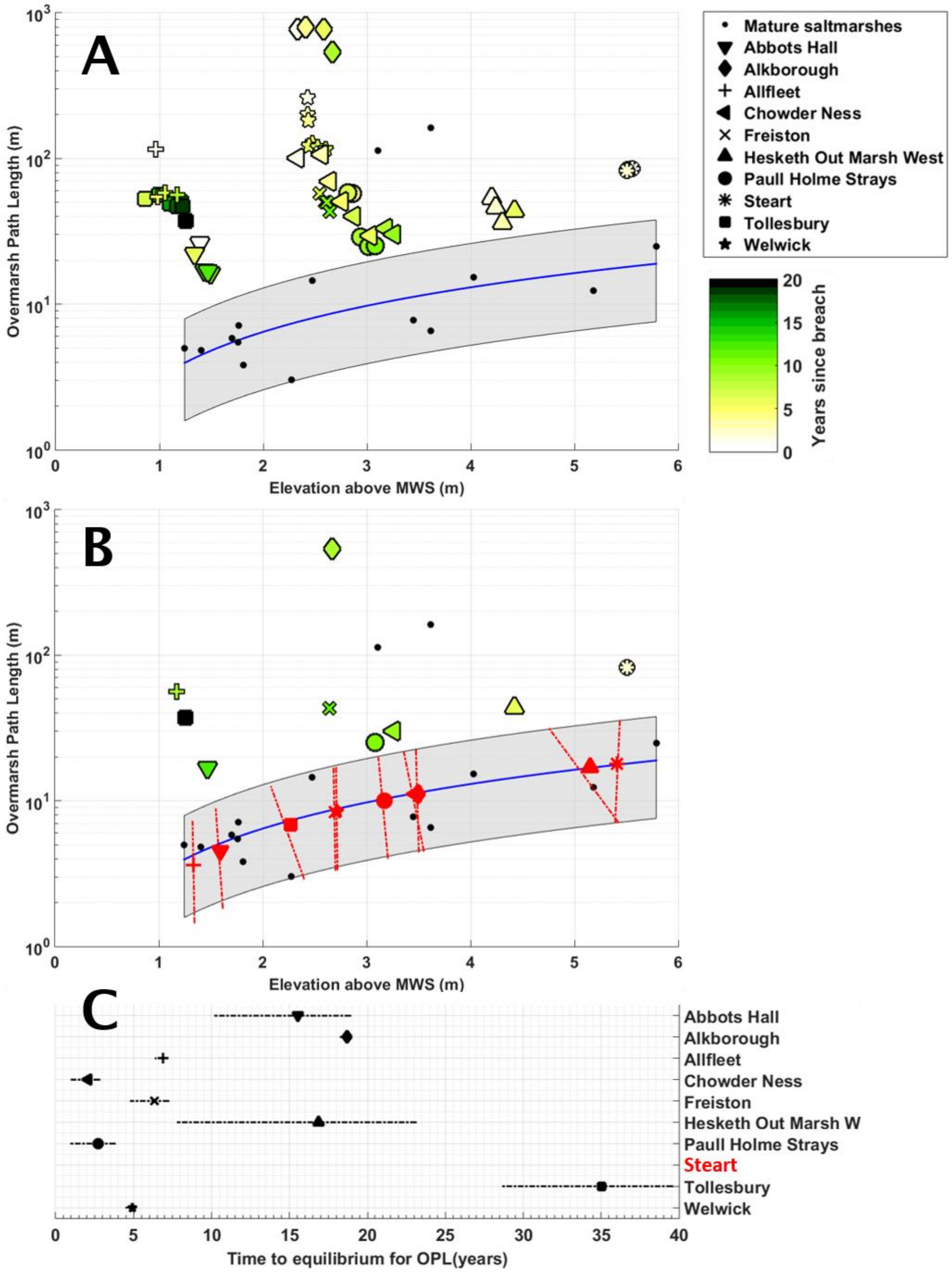


Figure 7.12: A: Evolution of MR schemes' OPL versus elevation above MWS towards the equilibrium range (grey shading); B: projected equilibrium range for each MR scheme plotted in red, assuming a linear evolution of both marsh accretion and OPL; C: time necessary to reach the equilibrium range extrapolated from linear evolution rates for each scheme except Steart (too young to infer evolution rates for OPL). Error bars at A and B smaller than the markers.

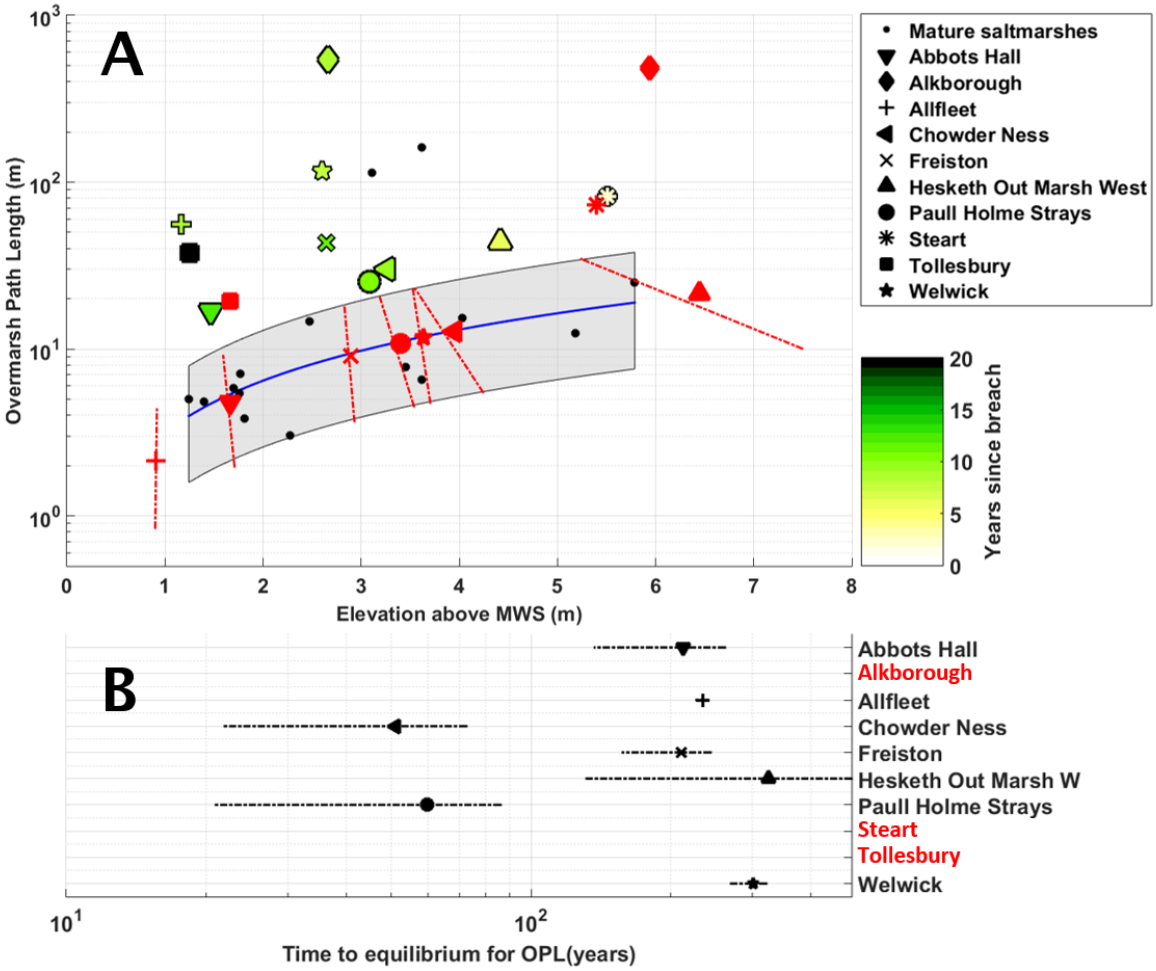


Figure 7.13: A: projected equilibrium range for each MR scheme plotted in red, assuming a decaying evolution trend for both marsh accretion and OPL; B: time necessary to reach the equilibrium range extrapolated from decaying evolution trends for each scheme except Steart (too young to infer evolution rates for OPL), Alkborough and Tollesbury (fail to reach the equilibrium range within the 500 years range considered). Error bars at A smaller than the markers.

In order to inform the potential long-term behaviour of MR sites, the creek morphology at the two AR sites Brandy Hole and Foulton Hall in 2017 is compared with the natural equilibrium range for the five equilibrium relationships used above (Figure 7.14). The main channel length and depth, as well as the total channel length, fall within the natural equilibrium range for both AR sites. Brandy Hole is above the natural equilibrium range for mouth cross-sectional area while Foulton Hall is close to the upper limit. Finally, both AR sites are above the natural equilibrium range for the overmarsh path length, though Foulton Hall has a better distribution of creeks out of the two sites.

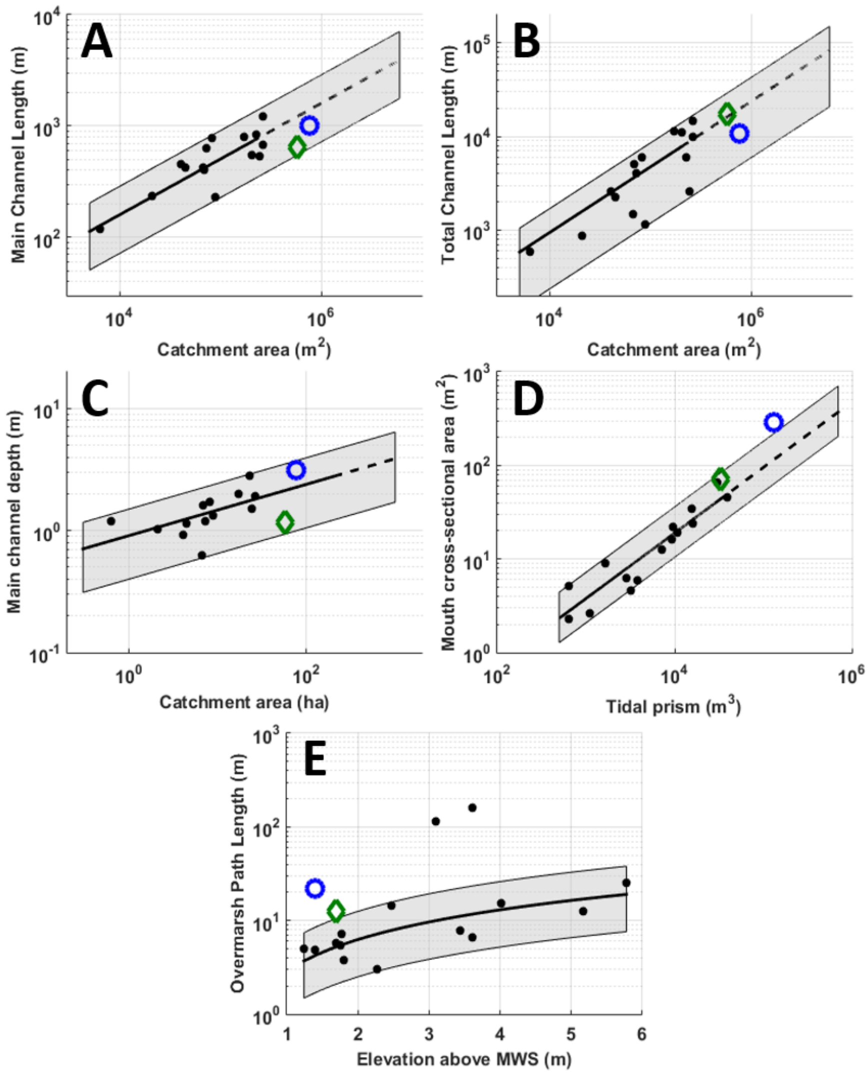


Figure 7.14: A: main channel length versus catchment area of the two AR sites (blue circle: Brandy Hole, green diamond: Foulton Hall) in comparison with the natural equilibrium range (grey shading); B: Total channel length versus catchment area; C: Main channel depth versus catchment area; D: Mouth cross-sectional area versus tidal prism; E: OPL versus elevation above MWS.

The second sub-objective was to determine whether the MR creek parameters evolve towards morphological equilibrium, and to estimate the time to equilibrium. For some creek parameters, such as the total channel length and the depth of the creek segments, the majority of MR sites considered reach the natural equilibrium range within the first 5 years after implementation. However, extrapolation of the creek distribution evolution rates suggest that MR schemes may take over 100 years to reach the natural equilibrium range. The observation of two AR sites suggest that they may instead stabilise at an alternative equilibrium state characterised by a poorer distribution of creeks.

7.3.3 Impact of initial conditions on MR creek evolution rates

The third sub-objective is to investigate the potential of initial design choices and external conditions to encourage or impede the evolution of creek networks towards equilibrium. To that end, the creek evolution rates obtained in Chapter 6 are compared with the schemes' starting conditions and design choices. Figure 7.15 shows the distribution of the studied MR schemes across the UK and their evolution rates. No obvious correlation between the creek evolution rate and the MR scheme location in the UK, or within the estuary can be found. However, the distribution of MR schemes is uneven, with an overrepresentation of the Humber estuary (4 MR sites) and no MR scheme on the south coast.

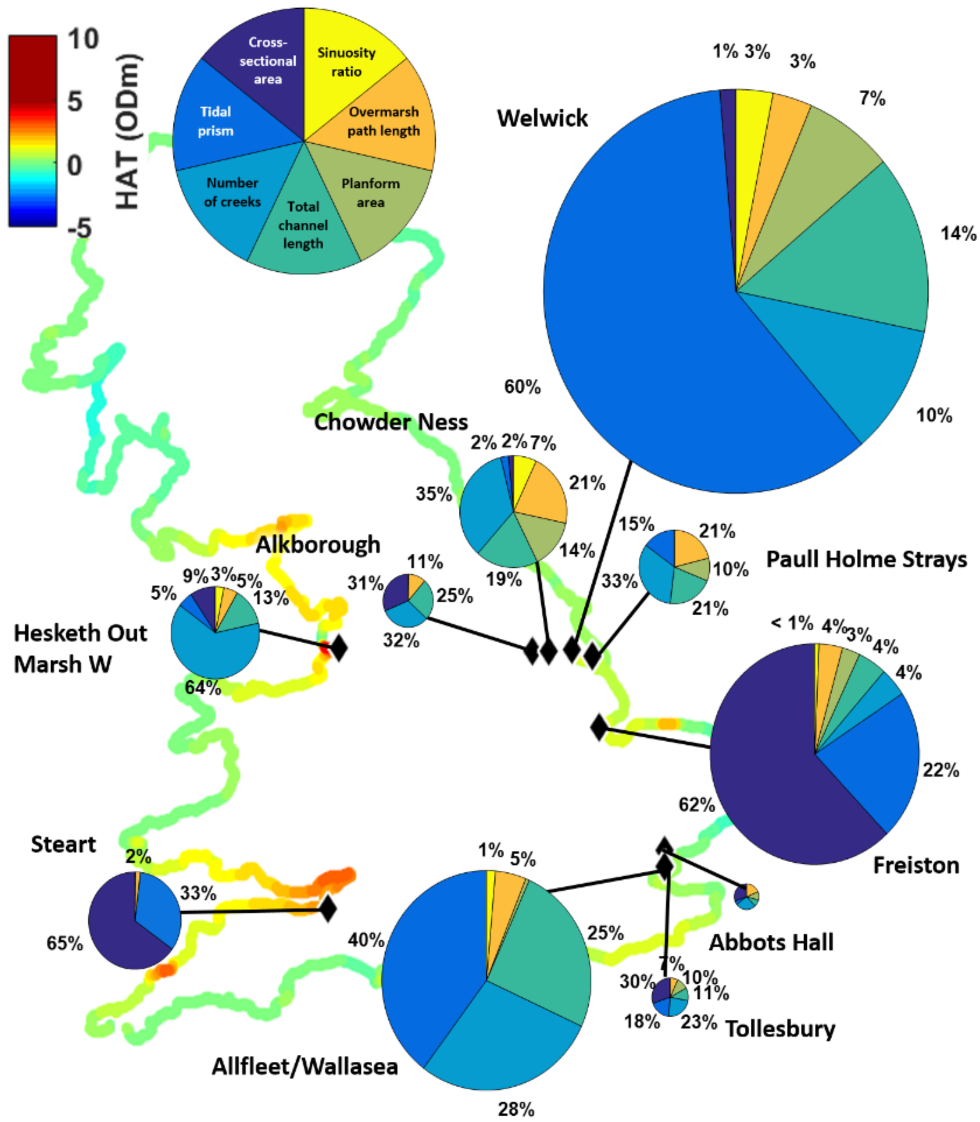


Figure 7.15: Normalised MR creek evolution rates per year represented as pie plots across the UK. Negative evolution rates have been brought to zero. The radius of the pie plots represent the sum of the evolution rates considered.

The 10 MR sites are then ranked by ascending total creek evolution rate (Figure 7.16), in an attempt to link creek evolution to the particular context of each scheme. MR schemes can be categorised based on their creek-forming processes: schemes where creek growth is dominated by erosional processes, deposition processes, or where creek growth is negligible. The three groups are detailed below:

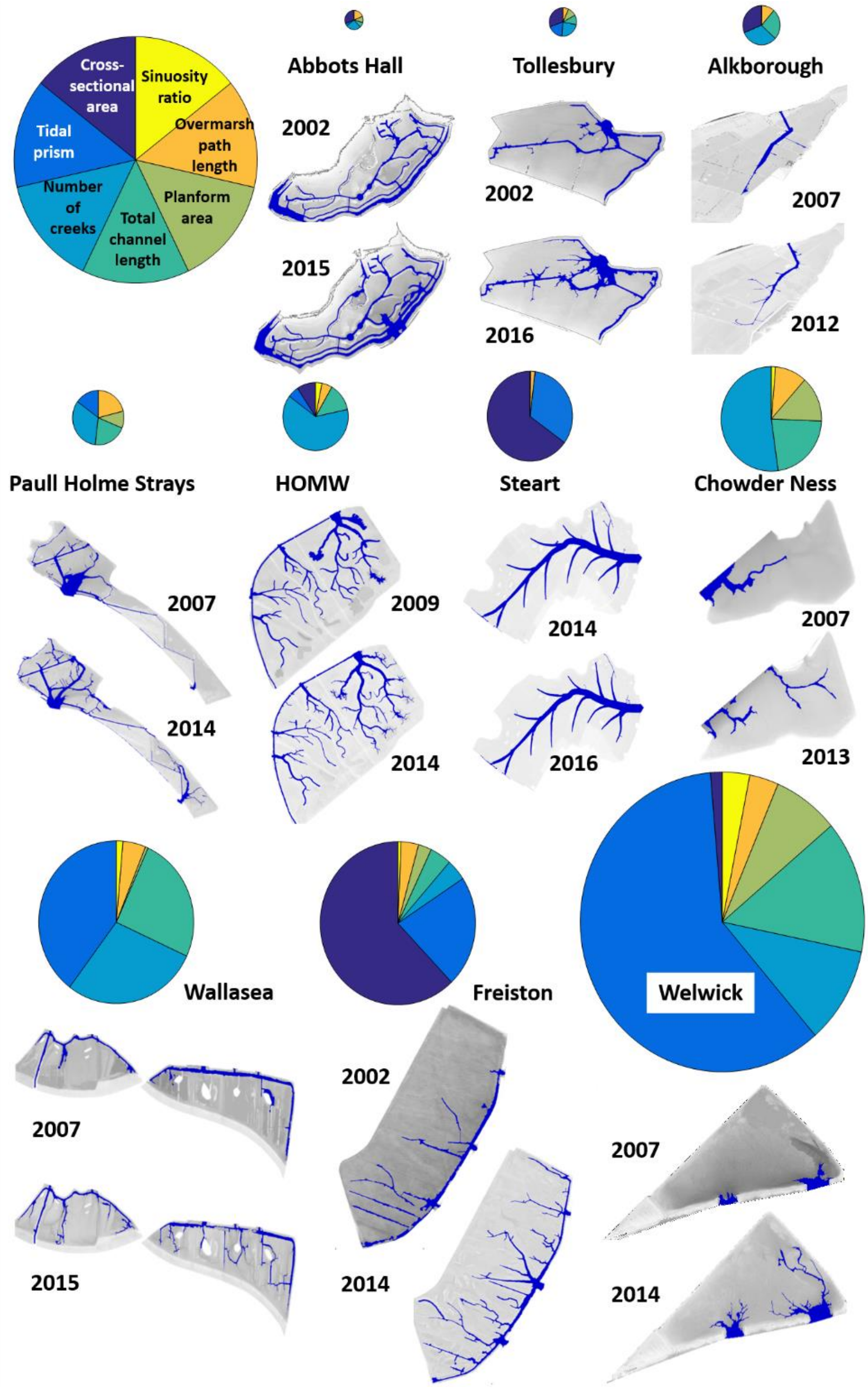


Figure 7.16: MR schemes ranked by total normalised evolution rates (creek extent for the first and last year of monitoring shown in blue). The evolution rates pie plots are invisible at this scale for the 2 AR sites, indicating a negligible evolution compared to the younger breached sites.

Erosional processes: at MR sites Freiston, Welwick and Steart, over 50% of creek evolution contribution comes from volumetric parameters, such as the tidal prism and the mouth cross-sectional area. At Steart and Freiston, this behavior is mainly due to the rapid erosion within the breach area directly after implementation (Symonds 2006; Pontee 2015a). The preponderance of erosive processes at Steart is also due to the fact that the scheme hasn't had enough time for the creeks to develop via accretional processes. At Welwick, mixed processes are at play: the creek network expands through headward erosion, but also through heterogeneous deposition that accentuates and deepens initial elevation heterogeneities (Figure 6.16).

Deposition processes: at MR sites Allfleet, Chowder Ness, Paull Holme Strays and HOMW, over 50% of the creek evolution comes from planimetric parameters such as the number of creeks, total channel length and planform area. The dominant process of creek formation could then be heterogeneous sediment deposition that exacerbates existing topography heterogeneity and deepens them into creeks by raising the surrounding banks. Here again the group is diverse. At Allfleet, the significant increase in creek volume is due to the reactivation of old agricultural ditches, building on an initial template with large and deep entry channels. Chowder Ness's creek system by contrast remains shallow and unstable because it has no initial template. The growth at HOMW is driven dominantly by the infill of headward ponds and the formation of new small creeks within the softer sediment, but some headward erosion of new creeks can also be observed (Figure 6.12). The creek expansion at Paull Holme Strays is also driven by a combination of heterogeneous deposition and headward erosion.

Poorly expanding creek systems: MR sites Abbots Hall, Tollesbury and Alkborough show limited creek evolution. At Alkborough, this is due to the small and constrained breach area and an insufficient initial creek template, which limits water exchange between the site and the wider estuary. At Abbots Hall, the sediment shear strength is twice as high as in a nearby natural marsh, flooding is less frequent in the higher regions and water velocities are lower (Atkinson et al. 2001; Wallace et al. 2005). At Tollesbury, observations suggest that the creeks could not develop into the substrate due to the previous land history as agricultural soil (Brown et al. 2007). Note that the two AR sites display even less evolution than these 3 sites.

The fastest evolving MR creeks are those that have a larger contribution of the volumetric parameters, namely the cross-sectional area at the breach and the creek volume. Erosive processes near the breach occur earlier than the deposition-driven processes of creek expansion due to the high energy of the breach area, making that portion of the site highly reactive. This is the case at Freiston, where the breach area has increased dramatically after implementation, and at Steart, where no creek expansion has yet taken place, but where dramatic changes in the breach area have occurred. The evolution rates at Steart are thus more descriptive of the breach area than of the creek network, and Steart is removed from the PCA.

The PCA provided the contribution of each selected variable to the first 4 PCs, which cumulatively accounted for over 75% of the total variability of the dataset (Table 7.2). The results are also displayed in a three-dimensional biplot to allow visual interpretation of the relationships between the variables (Figure 7.17). Due to missing parameters relevant to saltmarsh development (vegetation, flow velocity, number of flooding events per year), PCs were hard to interpret beyond the fourth component. Input variables were separated into the 4 PCs based on their highest absolute loadings.

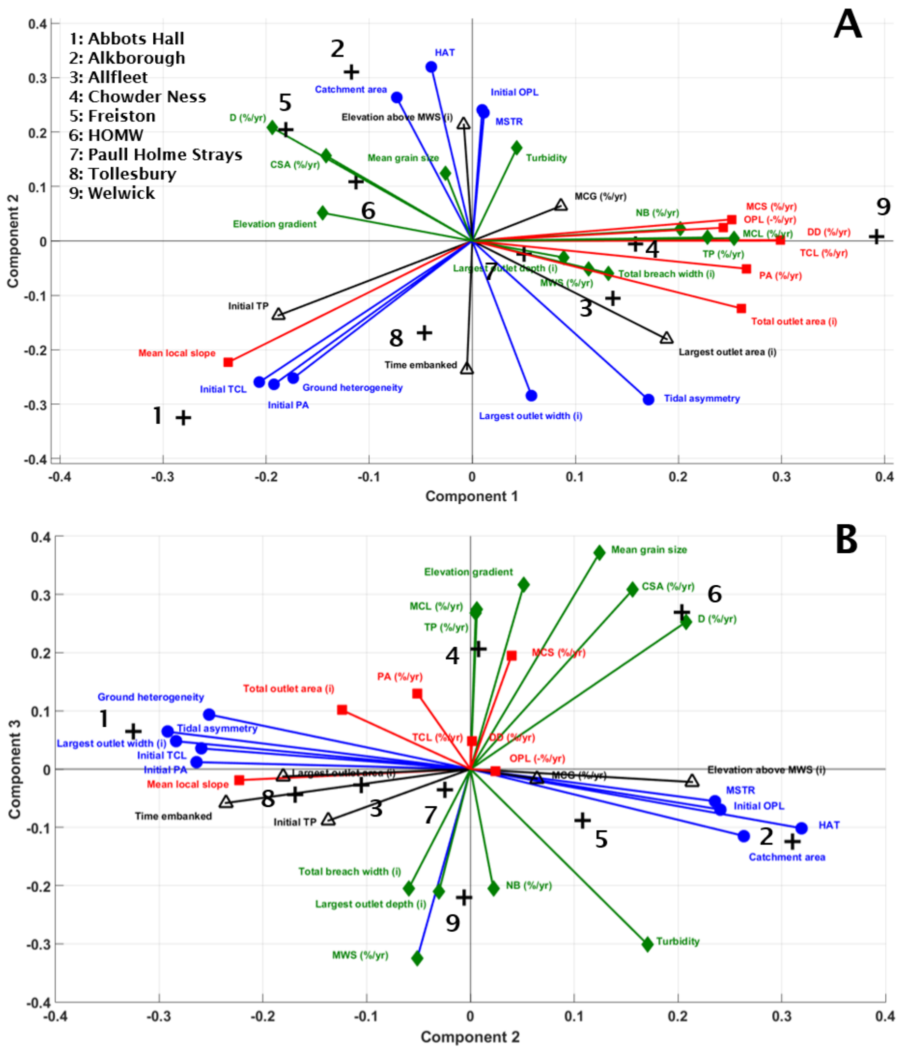


Figure 7.17: Biplot of normalised variables for all MR sites, with the first three PCs as axes (Matlab function biplot). A: contributions of the variables to PC1 (red squares) and PC2 (blue dots). B: contributions of the variables to PC2 and PC3 (green diamonds). Contributions of variables to PC4 shown as black empty triangles. Position of each MR scheme within the biplot shown as black crosses.

Table 7.2: Loadings for the first 4 principal components, explaining over 75% of the total variation, following PCA applied on standardized variables for all MR sites. Groups of correlated variables are shown in red for PC1, blue for PC2, green for PC3 and black for PC4.

Principal Components	**1**	**2**	**3**	**4**
MSTR	0.011	**0.235**	-0.055	-0.304
HAT	-0.040	**0.319**	-0.102	-0.186
Tidal asymmetry	0.171	**-0.292**	0.064	-0.070
Time embanked	-0.005	-0.236	-0.058	**0.364**
Catchment area	-0.073	**0.263**	-0.115	0.221
Elevation above MWS (initial)	-0.008	0.214	-0.022	**-0.324**
Elevation gradient	-0.145	0.051	**0.317**	-0.040
Mean local slope	**-0.237**	-0.223	-0.019	0.045
Ground heterogeneity	-0.174	**-0.252**	0.094	0.128
Turbidity	0.043	0.171	**-0.302**	-0.006
Mean grain size	-0.026	0.124	**0.371**	-0.116
Largest outlet area (initial)	0.189	-0.181	-0.013	**-0.206**
Largest outlet depth (initial)	0.088	-0.030	**-0.211**	-0.037
Largest outlet width (initial)	0.057	**-0.284**	0.048	-0.203
Total outlet area (initial)	**0.261**	-0.124	0.101	-0.082
Total breach width (initial)	0.132	-0.060	**-0.205**	-0.130
Initial TP	-0.188	-0.137	-0.089	**-0.230**
Initial PA	-0.193	**-0.264**	0.012	-0.105
Initial TCL	-0.207	**-0.260**	0.035	-0.171
Initial OPL	0.010	**0.241**	-0.069	0.254
Mean elevation over MWS (%/yr) -MWS	0.113	-0.051	**-0.325**	0.055
Drainage density (%/yr) -DD	**0.298**	<0.001	0.048	0.191
Overmarsh path length (-%/yr) -OPL	**0.243**	0.024	-0.003	-0.154
Main channel length (%/yr) -MCL	0.228	0.006	**0.274**	0.071
Total channel length (%/yr) -TCL	**0.299**	0.002	0.048	0.190
Number of creeks (%/yr) -NB	0.202	0.022	**-0.205**	-0.009
Mouth cross-sectional area (%/yr) -CSA	-0.142	0.156	**0.308**	-0.032
Mouth depth (%/yr) -D	-0.194	0.208	**0.253**	0.052
Planform area (%/yr) -PA	**0.266**	-0.051	0.130	-0.195
Tidal prism (%/yr) -TP	0.254	0.005	**0.268**	0.094
Main channel sinuosity (%/yr) -MCS	**0.252**	0.039	0.194	0.025
Main channel gradient (%/yr) -MCG	0.086	0.064	-0.017	**-0.355**
Explained variation	28.581	23.874	13.159	11.120
Cumulative explained variation	27.205	52.455	65.614	76.734

PC1 relates most MR creek planimetric evolution parameters together. DD, OPL, TCL, PA and MCS are all positively correlated with the total initial outlet area. Creeks that expand rapidly tend to also see their creek distribution and their sinuosity improve, in accordance with the conceptual model in Figure 2.14. This is linked to a large initial cross-sectional area of the outlets, which facilitates the water exchange necessary to creek-forming processes. The creek parameters are also inversely correlated to the mean local slope. On the biplot, this parameter is correlated with the initial TP, TCL and PA (Figure 7.17A). A high mean local slope indicates a higher initial concentration of creeks as they constitute the main factor of topography in a marsh. Creek growth is faster when the site is initially empty of creeks, as this leaves more space in the lower regions of the site for new creeks to form. PC1 is strongly influenced by Welwick on one end (large outlet, no initial creek system and rapid creek evolution), and Abbots Hall and Alkborough on the other end (slow creek evolution, small breach area for Alkborough and extensive initial creek template for Abbots Hall).

PC2 relates the initial MR creek design (PA and OPL) to the ground heterogeneity, catchment area, outlet width and tidal forcings (MSTR, HAT and tidal asymmetry; a visual correlation is also found with the initial elevation above MWS and the mean local slope on the biplot at Figure 7.17B). PC2 opposes Alkborough, a large scheme lying high within the tidal range, with a small breach width and a high initial OPL, with Abbots Hall and Tollesbury, which lie lower within the tidal range, are smaller and have a comparatively larger initial creek system. Ground heterogeneity is proportional to the initial creek template and mean local slope. Creeks are the main source of surface heterogeneity in MR schemes. Finally, tidal asymmetry is inversely correlated to the catchment area, HAT, MSTR and elevation above MWS. This correlation may reflect the influence of Alkborough, which has both the lowest tidal asymmetry and the highest catchment area of the dataset (Figure 7.18A). Another possibility is that the tidal asymmetry is related to the tidal levels (Figure 7.18B): a high tidal range promotes flood dominance (Fortunato et al. 2005), hence a shorter flood, so a lower value of the tidal asymmetry (calculated herein as the ratio of flood duration over ebb duration).

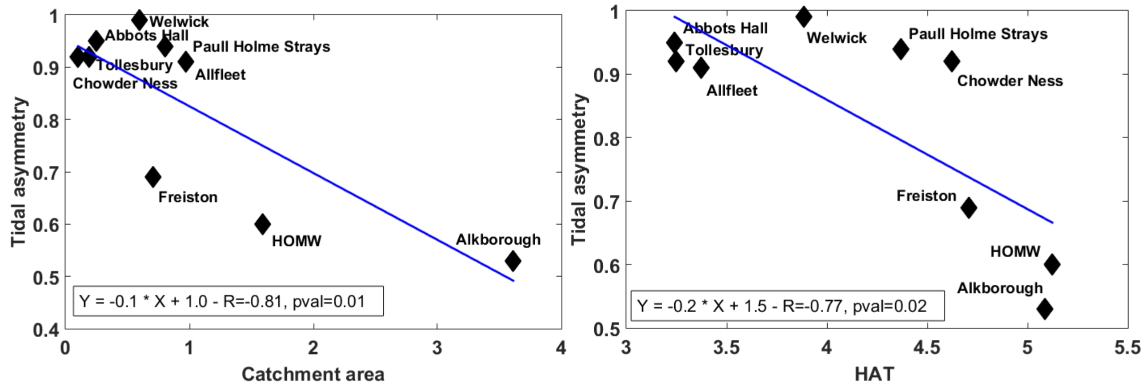


Figure 7.18: Correlation between tidal asymmetry and catchment area (A) and HAT (B)

PC3 relates the mean grain size and turbidity within the nearby estuary, elevation gradient, largest outlet depth and total breach width with accretion rates and with creek evolution parameters including MCL, NB, D, CSA and TP. PC3 opposes accretion-driven sites like Chowder Ness and to a lesser extent HOMW to erosion-driven sites like Welwick and Freiston (Figure 7.17B). A high accretion rate due to a high sediment availability in the nearby estuary and a large site opening, because they facilitate sediment import, encourage the initiation of new shallow creeks (greater number of creeks). Conversely, a high elevation gradient between the landward reaches of the site and the outlet, and a rapid deepening and widening of the scheme's outlets, both encourage the lengthening of the entry channel and the undercutting of creeks, increasing the creek network volume. A higher mean grain size would also mean that the sediment is less cohesive and thus easier to erode after deposition.

PC4 inversely relates the time embanked to the elevation above MWS, initial largest outlet area, initial creek volume and MCG, and is also visually correlated to MSTR. MR sites that were embanked earlier historically tend to lie lower within the tidal range, and have lower values of MSTR: this is either due to compaction, lack of sediment import and drying during the embanked phase, which reduces the level of reclaimed land in comparison to estuarine levels, or to the fact that the low tidal range made these schemes easier to defend with seawalls. Higher-lying schemes tend to be designed with a higher initial creek volume. A higher tidal range also indicates that the MR scheme lies significantly above the MLWS level in the wider estuary, which explains the faster evolution of MCG to accommodate this elevation gradient.

The third sub-objective was to investigate the potential of initial design choices and external conditions to encourage or impede the evolution of creek networks towards equilibrium. PCA results suggest that a lower initial marsh elevation, larger outlets, higher sediment availability and higher accretion rates all encourage creek development in MR schemes. These criteria will be analysed further in the discussion.

7.4 Discussion

7.4.1 Evolution of MR sites depending on starting conditions and design

After implementation, MR creek networks go through a phase of near-linear growth of varying intensity towards equilibrium (16 to 331% of total evolution per year, including planimetric and volumetric parameters). This phase lasts for >5 years and, for the sites studied, shows no clear sign of plateauing. The exceptions are the elevation of the site within the tidal range and OPL. Their evolution follows logarithmic trends more closely than linear ones. Indeed, while low sections of the site close to the breach get inundated often, leading to a high initial development of creeks, the higher and more landward reaches of these schemes are affected by fewer flooding events, which should result in a decaying trend of OPL evolution over longer timescales. For the same reasons, accretion rates are also expected to slow down as the site becomes closer to MHWS level.

Taken separately, some parameters indicate a good agreement between MR and natural mature creeks. When considering the creek system's length and depth, the majority of MR sites considered reach the equilibrium range within the first 5 years after implementation. This is due to either an initially excavated creek network which lies close to equilibrium (HOMW, Abbots Hall) or to a rapid creek evolution following implementation (Chowder Ness, Welwick). In some cases, the evolution of the creek network is encouraged by the reactivation of agricultural ditches into channels (Paull Holme Strays, Allfleet, Freiston and Alkborough). Rapid creek development in reconstructed saltmarshes has been observed in other studies, especially if artificial channels were excavated to kick-start creek growth (Wallace et al. 2005; Vandenbruwaene et al. 2012).

Other aspects of MR creek morphology display starker distinctions from natural systems. All considered MR schemes except for Abbots Hall (limited accretion and erosion), Chowder Ness (shallow creeks) and Alkborough (breach area armoured to prevent erosion) display an outlet area too high for their creek volume. Those sites, assuming a linear yearly increase in creek volume, may take over 100 years to reach the equilibrium state of natural systems. They are more likely to stabilise at an alternate equilibrium state, above the natural equilibrium range. Indeed, most MR schemes are partially constrained by remnant flood defences, which force a greater flow through the breach areas and promote larger entry channel dimensions. Also, most MR sites fail to incise new creeks into the substrate, which slows down the increase of the overall creek volume. Finally, none of the OPL values are at equilibrium with the site elevation: MR sites are emptier of creeks than their natural counterparts. Even Abbots Hall and HOMW creeks are too sparse despite having the most extensively excavated channels.

As a result, the structures of natural and artificial creek systems are likely to remain fundamentally different within the next 100 years. Artificial creek systems are generally characterised by a smaller quantity of oversized channels that cluster near the oversized breaches, leaving the furthest sections of MR schemes poorly drained. This configuration explains why MR creeks have a similar volume to natural creek systems but lower values of drainage density and excessive values of OPL (Figure 7.6). These characteristics are also observed within century-old AR sites, which have not reached the OPL values characteristic of natural saltmarsh creeks at equilibrium, and show little sign of evolution, suggesting that they have stabilised at a different equilibrium specific to restored saltmarshes. Those oversized, poorly distributed creeks linked to a large outlet could have an influence on the tidal asymmetry within the site, by changing the speed at which water is distributed during the flood and drained during the ebb.

The comparative analysis of MR creek evolution rates and the PCA results highlight potential drivers of creek evolution, and help explain how they differ from natural systems. PC1 relates the outlet dimensions to the planform growth of the creeks and to the initial creek distribution: while an extensive initial creek template may kick-start creek development, a large outlet permitting water exchange with the wider estuary is more important to allow a dynamic system to develop. This trend supports the current MR design philosophy that encouraging natural processes will be more efficient than providing a mature-looking morphological template. However, caution should be applied. Since breaches are designed because of specific site characteristics, the breach size may be a surrogate for another driver like the catchment area or tidal range. However, no positive correlation was found between the outlet area and these parameters. Instead, current designs have tended to make lower sites more open to tidal influence to encourage the development of mudflats rather than saltmarshes, as was done for MR sites Chowder Ness and Welwick. This design approach leads to more rapid creek expansion, but is expensive and provides less protection against tidal and wave energy to the hinterland, and is thus not chosen very often unless mudflats habitats must be compensated for.

PC2 shows that the initial creek distribution in MR schemes is proportional to the catchment area of the site. Larger sites, like Alkborough, tend to be emptier because excavating a dense creek network is more expensive for larger sites. Practical considerations supersede concerns about the potential ecological consequences of a restored marsh being empty of creeks. New design guidelines may be needed to mitigate for this effect, as will be explored in Chapter 8. Another common design strategy is highlighted by PC4. Sites that lie higher within the tidal range tend to have a higher initial creek volume and length. This design choice seems sensible since high schemes have fewer options to develop their creeks due to the lower potential for accretion. Finally, PC3 opposes erosion-dominated MR sites like Welwick and Freiston to accretion-dominated MR sites like Chowder Ness and HOMW. The distribution matches with results from the comparative analysis of creek evolution rates (Figure 7.16). PC3 confirms that both erosive and accretional processes are important to creek development. In the absence vertical accretion like at Abbots Hall, creeks cannot expand further into the site. Conversely, if only accretional processes occur, creeks may remain overly shallow like at Chowder Ness.

In MR schemes especially, deposition-based creek expansion does not seem sufficient to form a creek system with equivalent morphological characteristics to a natural one, because most MR creeks fail to incise the creeks without an initial template to build from. This is the case in Chowder Ness, in Abbots Hall and in Allfleet, where the bulk density is greater than in nearby natural saltmarshes (Kadiri et al. 2011; Brooks et al. 2015). Similarly, even after 15 years of implementation, Tollesbury has maintained soil properties of an agricultural soil in terms of below-ground biomass and organic matter content (Burden et al. 2013), shear strength and resistance to erosion (Reading et al. 2008). Indeed, the period of embankment alters the sediment properties and tends to reduce the porosity of the substrate and the size of macropores (Tempest et al. 2014), with negative consequences for drainage and thus creek development (Figure 2.13). This fits with observations made in Chapter 6 that creek incision into the substrate is more efficient when the embankment period is shorter.

MR schemes and century old AR sites also display significantly lower values of small-scale topography, which fits with previous field observations and could be another factor preventing creek development (Brooks et al. 2015). Indeed, ground heterogeneity usually drives preferential flow directions that subsequently develop into creeks (Figure 2.14). Thus, lack of sedimentation, excessive ground compaction during the embankment period and lack of topographic heterogeneity are likely to negatively influence creek development in MR schemes (Atkinson et al. 2001; Mossman et al. 2012). There is a strong case for investigating the relations between soil properties and creek network development in MR schemes. Interestingly, out of the two AR sites, Foulton Hall, which was used as a grazing and mowing marsh, is closer to the natural morphological equilibrium range than Brandy Hole, which was used partially as arable land. The role of previous land use on subsequent soil structure perturbation and on creek development after breaching should also be investigated.

This problem mirrors a trend observed in the vegetation development in MR schemes. Vegetation cover or number of species gives a misleadingly optimistic view of the health of the ecosystem, while the diversity remains significantly lower than in natural sites (Mossman et al. 2012). It has been estimated that vegetation communities and soil properties such as organic carbon content could take approximately 100 years to resemble that of natural systems (Crooks et al. 2002; Burden et al. 2013), or even more (Garbutt et al. 2008). Similarly, estimations of OPL evolution suggest that it may take current MR schemes over 100 years to reach the natural equilibrium range, and that they may instead stabilise at a creek morphology characterised by a poorer creek distribution. Results from this chapter agree with the general observation that MR schemes in the UK are not currently reaching a state of ecological equilibrium, and that this difference may be due to the lack of small-scale topography, sparsity of creeks and their inability to develop unassisted due to altered sediment properties (Mossman et al. 2012; Chang et al. 2016). The relationship between creek distribution and other indicators of coastal wetland health (such as plant diversity), should be the focus of further studies to formally establish this link. Other possible factors restricting the successful development of saltmarsh plant communities include soil compaction and dewatering within the reclaimed sites, soil chemistry changes compared to natural saltmarshes (Reed et al. 1999; Mossman et al. 2012; Brooks et al. 2015), and greater hydroperiod due to the constrained nature of MR schemes.

7.4.2 **Limitations**

The Achilles' heel of attempting to interpret the efficiency of creek design is that it implies making predictions on future trends and thus extrapolating from a small dataset. The extrapolation is particularly problematic when applied to creek networks, because of their natural variability and because the processes leading to their growth are not linear. As observed in Section 7.3.2, vastly different estimations of time to equilibrium are obtained for OPL depending on whether a linear or logarithmic trend was used. Therefore the times to equilibrium proposed in this section are tentative.

Conclusive data have been found showing that the ground heterogeneity of MR schemes is lower than that of natural saltmarshes, which fits with previous studies (Brooks et al. 2015), and may constitute a significant obstacle to creek growth in MR schemes. However, the topographic differences between natural and MR marshes may be dampened by the lidar resolution used. Features smaller than 2 m long, 2 m wide and 0.30 m deep are likely to be smoothed over within the lidar cells as discussed in Chapter 4. This should affect natural sites disproportionately since the microtopography and the density of small channels is higher in those sites. By contrast, in MR schemes visual observation shows that their surface is flatter except for large features (oversized channels, mounds built as bird habitats, drainage ditches, etc.). As those features are less likely to be smoothed over, the topography of the MR schemes is artificially brought closer to that, underestimated, of the natural marshes. A topography analysis using higher resolution lidar datasets (0.25 m for instance), preferably validated by field surveying, would provide further information on the relative topography of natural and MR sites. The presence of vegetation in the DSM may also affect the detected ground heterogeneity.

Finally, as this study is one of the first large-scale comparisons of natural and MR creek networks, some important background information on the design and properties of the schemes are still missing from the literature. In particular, sedimentological properties within the site itself rather than in the nearby estuary, or the geotechnical properties were not available for all sites. Only the embankment period was systematically available as a proxy of site compaction. This factor is thought to play a role in enabling or preventing creek incision into the marsh. However, the erodibility of the sediment is not simply a function of the time spent cut off from tidal influence, but also of the sediment properties including grain size, water content and soil organic matter (Watts et al. 2003; Burden et al. 2013). There is a need for more systematic monitoring of MR schemes geotechnical properties to get a better idea of the drivers and dampeners of creek growth. Furthermore, due to limits in the number of lidar data available and of background information for the MR sites, 4 of the studied sites are located in the Humber estuary, which is probably skewing the results. Future studies should try and get a better repartition of MR sites if the data is available.

7.5 Summary

In this chapter, the evolution of creek morphometric parameters of 10 UK MR schemes was compared with a stable state established for mature coastal wetlands. 5 years after implementation, most creek parameters in MR sites, such as total creek volume, main and total channel length, have evolved to resemble the stable state of natural systems. MR creek networks also evolve to follow Horton's laws of stream numbers and lengths, with an exponential increase in creek number and decrease in creek length with increasing RS order. However, some MR creek parameters remain significantly different from those of natural systems. MR creek outlets remain larger than in natural systems due to the presence of embankments that focus the tidal flow through the breach and entry channel as undermarsh tidal prism. Furthermore, MR schemes remain significantly emptier of creeks, even in century-old accidentally realigned sites. This suggests that restored coastal wetlands have the tendency to stabilise at a different equilibrium state than that of natural systems, and that this state is characterised by a poorer distribution of creeks. Some of the current creek design strategies are more efficient than others at promoting creek growth. Lower, more open sites that have spent less time embanked and so may have undergone less compaction, are better at encouraging creek expansion. Pre-existing features such as drainage ditches can be reactivated but will result in unnatural, overly straight, narrow and deep channels. In general, while MR schemes succeed in encouraging some creek development, their capacity to excavate their own creeks is overestimated and is unlikely to result in equivalent distributions to that of natural systems. The poor distribution of creeks is suspected to have adverse effects on the ecological functioning of MR schemes. Consequently, a new conceptual evolution model should be defined for MR creeks to improve understanding of these systems and inform future design.

Chapter 8: Implications for MR scheme design

8.1 Introduction

The overreaching aim of this research was to improve understanding of creek network development in MR sites in comparison with creek systems in mature natural saltmarshes, in order to distil implications for future practice. The research fits within the greater study of "working with nature" approaches to coastal management in the face of climate change (Brown et al. 2014; Esteves et al. 2015). This aim has been addressed by undertaking three main stages of research, each addressing a specific objective. The objective of the first stage was to provide a pluri-parameter morphometric analysis of 13 UK saltmarsh creek systems, using remote sensing datasets and novel data extraction methods, in order to define equilibrium characteristics to use as end targets for MR creeks. The objective of the second stage was to undertake a morphometric analysis of 10 UK MR creek and infer evolution rates over the years. Finally, the objective of the third stage was to compare the morphological characteristics of natural and MR creeks to determine whether they develop towards more natural systems over time, and whether their evolution rates can be related to initial conditions and design choices.

Together, these results indicate that while current MR design strategies allow for some creek development, they do not result in similar creek structures to those found in natural mature saltmarshes within the timescale considered (3 to 20 years). Extrapolation of OPL evolution rates and the observation of century-old accidentally realigned sites suggest that MR creeks may not reach the distribution of a natural mature system even after 100 years, and could instead stabilise at a different, MR-specific equilibrium morphology. It is therefore unlikely that artificial creek networks provide the same ecological function as their natural counterparts. Improved design guidelines would be required to ensure that future schemes evolve towards more natural morphologies. A new conceptual model for creek evolution specific to MR schemes is thus necessary to derive implications for future practice.

This section fulfils Thesis Objective 4: to establish a conceptual creek evolution model for MR schemes that highlights the main divergences from natural creek evolution and derive implications for future practice. To that end, there are three sub-objectives as follows:

1) To provide a list of the recurring divergences between MR and natural creek systems;

2) To develop a conceptual creek evolution model for MR schemes;

3) To derive implications for future practice that will encourage the development of MR creeks towards the natural morphological equilibrium range in future schemes

The structure of this chapter is as follows. Section 8.2 addresses sub-objective 1 by drawing together results from Chapter 6 and Chapter 7 to infer general observations on the design, implementation and subsequent evolution of MR creeks in the UK. The main commonly observed divergences from natural creeks systems are enumerated. Section 8.3 develops a conceptual creek evolution model for MR schemes to address sub-objective 2. Section 8.4 derive implications for future practice that will encourage the development of creeks towards the morphological equilibrium range of natural saltmarshes in future schemes to address sub-objective 3. Section 8.5 discusses how practical considerations may in some cases limit the applicability of design guidelines.

8.2 Recurring divergences between natural and MR creeks

This section builds particularly on results from Thesis Objective 2 and 3: to provide an exhaustive morphometric analysis of MR creek systems, and to determine whether MR creeks develop towards more natural systems over time, and whether their evolution rates can be related to initial conditions and design choices. The morphology of MR creek systems was found to be highly variable and dependent on the initial conditions of each scheme. However some general observations have been made for all schemes, which help distinguish MR schemes from natural sites. These characteristics are described in more detail below and are illustrated when relevant with examples from site observations at recent MR schemes like Steart and Hesketh Out Marsh East. The potential implications of these morphological particularities for the ecological functioning of the sites are also discussed.

The main characteristics of MR creek morphology that distinguish them from natural systems are summed up into 6 main divergences (summed up in Table 8.1). In the sub-sections below, the following elements are provided for each divergence: *proof of recurrence* (based on results from Chapter 6 and Chapter 7); *cause* of the divergence (what initial difference from a natural system has driven the divergence); *consequences for the morphological evolution of the creek system* (observed and expected); *expected ecological consequences*; and *potential mitigation measures*.

Table 8.1: Summary of creek morphological differences between natural and MR coastal wetlands, their implications for the site functioning and potential mitigation measures for future schemes

MR creek morphology divergence from natural systems	**Consequences for creek morphological evolution and ecological functioning**	**Implications for future practice**
1: MR creeks are oversized, wider and deeper than natural systems	This facilitates the creation of mudflat habitats and provides space for creek reprofiling. Faster flood could homogenise inundation duration through the site and provide fewer niche habitats.	• Research needed on the effects of creek dimensions on hydroperiod and habitat provision • Increase channel branching to provide more smaller creeks
2: MR creeks are inherited from drainage ditches, making them too straight and leptokurtic	The creek system cannot adjust its shape to a more natural, sinuous one, leading to poorer creek distribution, with potential negative impacts on biodiversity and fish use.	• Fill in existing drainage systems • Provide some sinuosity (1.1-2) to initial creek system. • Improve soil erodibility to promote unassisted creek shape adjustment (see 6).
3: MR creeks form in clusters at small junction angles near the breach area	Creek clustering near the breach leads to too much energy dissipation near the breach and not enough further into the site, with negative ecological consequences	• Research needed on the effect of energy dissipation from branching on MR creek growth • Excavate one entry channel to encourage water transport and creek growth further inland
4: MR sites are emptier of creeks than natural sites	The equilibrium OPL is unlikely to be reached in under 100 years. Poor creek distribution is suspected to be an important factor for lower plant diversity.	• The target OPL should be 25-30 m. Initial OPL needed will depend on initial site elevation, soil properties and accretion rates • Improve soil erodibility to promote unassisted creek growth (see 6)
5: MR sites are flatter than natural sites	The lack of preferential flow paths hinders creek growth. The lower habitat diversity lowers biodiversity.	• Add microtopography during implementation phase. Goal: 0.1-0.2 m difference in height between points up to 10 m apart to imitate natural sites • Initiate flow paths with microcreeks
6: MR creeks are rarely able to erode into the substrate	The overcompaction of MR soils is suspected to hinder creek growth, root penetration and the development of perennial species that would stabilise the creek banks.	• Prefer locations with short embankment history for realignment • Research needed on whether soil geotechnical properties can be restored to that of natural marshes: most efficient method currently reported is kelp compost rototilling

8.2.1 Divergence 1: MR creeks are oversized, wider and deeper than natural systems

Proof of recurrence: The comparison of natural and MR creeks in Figure 7.4 found the MR creek width to be greater for all RS orders and the MR creek depth to be greater for the highest RS orders when compared with natural mature systems. In Abbots Hall, HOMW and Steart, where the most extensive artificial creek systems have been excavated, the oversizing of creeks is particularly visible. By contrast, Chowder Ness, implemented without creeks and without an entry channel, has developed an undersized creek system with narrow and shallow channels.

Cause: The overly wide creek systems are a relic of the initial design: creeks are built overly wide in a lot of schemes to give channels the space to reach the correct equilibrium depth through sediment infill in the landward reaches of the site (Haltiner et al. 1987). Indeed, the tidal flow is higher at the mouth and then dissipates landward, leading to sediment deposition and a narrowing of the channels. As a result natural tidal channels display a funnel shape and widen exponentially in the downstream direction (Lanzoni et al. 2015). Oversized channels are also excavated in the hope of providing additional compensatory mudflats (Pontee 2015a), since most MR schemes have been found to rapidly accrete into saltmarsh habitats (Morris 2013). Chapter 6 shows that in many MR sites, including Freiston, HOMW and Steart, only the channels are at the elevation of a mudflat within the tidal frame (Figures 5.11, 5.12 and 5.14).

Consequence for the morphological evolution of the creek system: Rapid infill of creeks has been observed at Steart within the first 2 years of evolution post-breach (Figure 8.1), with some slumping of the creeks banks, which have contributed to giving the creeks' cross-section a more natural-looking shape. However, the width of creeks remains excessive in almost all sites. The breach areas undergo erosion and so become even larger. Consequently the equilibrium state of the outlet cross-sectional area versus creek volume may be higher in most MR schemes than in natural saltmarshes. Wider creeks do not systematically promote the faster development of tributaries, despite larger potential volumes of water and sediment being exchanged through the creeks: Freiston, HOMW and Abbots Hall rank 2nd, 5th and 10th respectively out of 10 MR schemes for overall creek development (Figure 7.16). Other limiting factors may supersede the effect of creek width, such as the tidal range, the availability of suspended sediment and the erodibility of the MR site substrate.

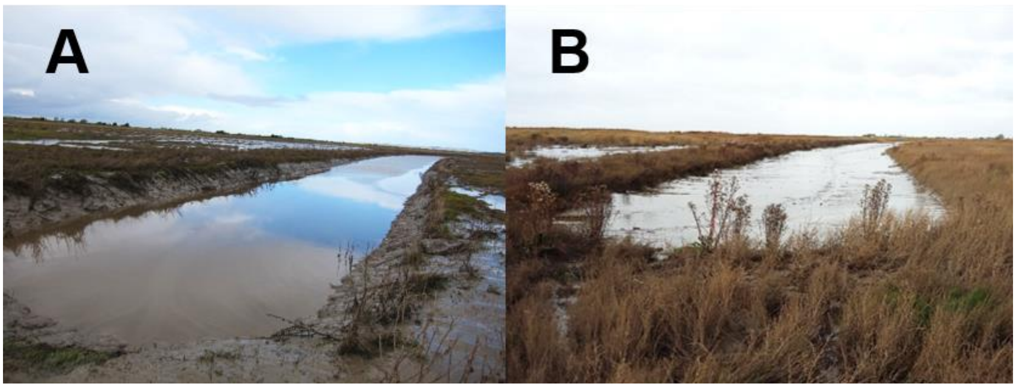


Figure 8.1: Infill of large creek at Steart between 23/02/2015 (A: photo taken by Anas Annuar) and 16/11/2016 (B: photo taken by Matthey Agius)

Expected ecological consequences: More research is needed to assess this effect, but the tidal regime can be expected to differ from that of natural wetlands. The larger creeks should promote a faster flood, and a more rapid infill of the site when water overtops the creeks, in contrast with the more gradual distribution of flood water expected from a shallower, more extensive creek system. Added to the flat topography and the constrained nature of most MR schemes, this effect could lead to fewer variations in inundation time throughout the site, akin to a tub filling up. A consequence would be fewer niche habitats for plants, which depend on specific redox and salinity conditions (Doherty et al. 2015). Interestingly, the creek configuration at Abbots Hall, while it does not promoting plant biodiversity (Brooks et al. 2015), has a positive impact on juvenile fish use (Colclough et al. 2005). In natural saltmarshes, fish are expected to enter the marsh through the channels rather than through the marsh edge (Miller et al. 1997), so wider entry channels may facilitate access to the MR scheme. However, connection of creeks to lagoons that remain flooded during low tide, like those present at Abbotts Hall, have a clearer positive impact on fish use (Colclough et al. 2005; Cavraro et al. 2017).

Implications for future practice: The design of creeks will always to some extent depend on the design of the breach area, as one of the creeks' primary function is to distribute the water going through the outlet (Pethick 1992). The constrained nature of most MR schemes is likely to lead to wider and deeper entry channels to accommodate the greater undermarsh tidal prism. However, a higher level of branching may help to reduce the channel dimensions by spreading that volume across a greater number of initial creeks, in order to mitigate the potential homogenising effect on flood duration. The width of creeks should diminish exponentially after each branching following Pethick (1992).

8.2.2 Divergence 2: MR creeks are often straighter than natural creeks and have a more leptokurtic shape

Proof of recurrence: The comparison of natural and MR creeks in Figure 7.4 found the MR creek sinuosity for RS order 3 be lower than for natural mature systems. Straighter channels are particularly visible in MR schemes inherited from drainage channels like Alkborough, Freiston, and Tollesbury. At Steart, the excavated creek template is also significantly less sinuous than a mature system. Also, the high RS order creeks tend to have a more leptokurtic shape, as illustrated in Figure 7.4.

Cause: Most MR creeks are excavated at least partially from pre-existing drainage ditches, which makes them abnormally straight. The drainage ditches also lead to deeper, narrower shapes in the high RS order channels. Indeed, as the creek system grows it connects remnant ditches, which act as newly formed high RS order creeks, but have a very different morphology. In natural saltmarsh creeks, the wide and shallow channel shape is due to consolidation processes of cohesive sediment, which leads to increasing resistance to erosion with depth (Fagherazzi et al. 2001). As a result, natural creeks tend to widen rather than deepen.

Consequence for the morphological evolution of the creeks: Model results indicate that higher values of shear stress occur at the channel bends, driving the incision of new creeks (D'Alpaos et al. 2007). Consequently, low sinuosity values can have a negative impact on creek growth. Previous studies of unmanaged realignment sites show that, even after over 100 years of restoration, creek networks comprising of a dense system of reactivated drainage ditches remain less sinuous than natural creeks (MacDonald et al. 2010). Also, straight channels automatically result in poorer distribution than sinuous creeks of the same length because they occupy less space (Haltiner et al. 1987; Marani et al. 2003), so this potentially has negative impacts on creek growth and density.

Expected ecological consequences: Water exchange between the marsh and the creek network is facilitated by creek meandering as it provides a larger area of contact (Coats et al. 1995) and reduces OPL values (Kearney et al. 2016). Sinuous creeks also provide more complex habitats (Williams et al. 2004). Overly straight channels could have a negative impact on biodiversity and fish use. The steep-sided, deep and narrow shape may also hamper the migration of juvenile fish from the creek to the marsh: according to previous studies fish communities will prefer depositional creeks with gentle slopes to creeks with steeper, erosional banks (Williams et al. 1999).

Implications for future practice: Excavating creeks at sinuosity values within the range of natural mature creeks (1.1–2.0 for Strahler orders 3-5), as suggested by previous guidelines (Coats et al. 1995), and with gentle bank slopes to reduce bank slumping and encourage drainage (Burgess et al. 2015), would probably have a positive influence on the ecological functioning of the site. The sinuosity of tidal channels has been found to depend on the tidal range and on the catchment area slope (Finotello 2017), so the sinuosity of the excavated creeks should depend on site-specific conditions. However, excavating sinuous creeks is not enough to promote plant biodiversity if the geotechnical properties of the soil and the topography of the marsh surface do not allow for the formation of niche habitats and for root penetration, as probably happened at Abbots Hall (Brooks et al. 2015). In the logic of facilitating natural processes, allowing creeks to dynamically adapt to changing conditions is more important than excavating a mature-looking template. In sites where high accretion rates are expected, it may be more efficient to excavate shallow sinuous protocreeks surrounded by sufficient microtopographic heterogeneity to focus the flow, and to ensure that the soil is not too consolidated for erosion to occur. The latter factor will require more research into the soil structure and geotechnical properties of natural and restored coastal wetlands.

8.2.3 Divergence 3: MR creeks commonly have smaller junction angles than natural creeks

Proof of recurrence: The comparison of natural and MR creeks in Figure 7.4 found the MR junction angle for RS order 2 to be significantly lower than in natural mature systems. This effect is most visible at Chowder Ness, Tollesbury and Welwick, which have a large breach or bank removal area and a limited initial drainage system.

Cause: A cluster of creeks forms near the breach area, leading to lower junction angles for RS order 2 creeks in sites characterised by a limited or absent initial drainage system, and a wide outlet. Tidal energy may then be dissipated close to the breach due to the lack of a clear channel or preferential flow path to follow, leading to a cluster of smaller creeks. In the natural environment (Kearney et al. 2016) and in laboratory experiments (Stefanon et al. 2010), this behaviour is mostly observed in non-cohesive, non-vegetated sediment where channel formation is less efficient.

Consequence for the morphological evolution of the creeks: Branching is linked to energy dissipation in both river and tidal systems (Rodriguez-Iturbe et al. 1992; van der Wegen et al. 2008). Too much branching occurring near the breach could weaken the flow upstream, hindering the development of creeks further inland and leading to poorly distributed creek systems with larger empty, undrained areas.

Expected ecological consequences: The lower regions of the site will be occupied by a dense network of channels, which could lead to excessive values of salinity, water content and redox dissuasive to plants. The higher regions are at risk of remaining empty of creeks, which is likely to negatively impact plant diversity by providing less habitat niches, and to negatively impact fish use, as their access to the middle and high marsh will be limited.

Implications for future practice: Further research needs to be carried out regarding tidal energy dissipation within MR schemes and its influence on creek distribution before proper guidelines can be proposed. However, it could help to excavate an entry channel into a better defined shape than at Paull Holme Strays or Welwick, to encourage tidal flow to reach further into the site before getting dissipated by branching channels. Some initial channels occurring in mudflats are already well distributed through the available space, justifying this choice (Figure 8.2). Steart is a recent example where this strategy has been applied, though the entry channel is significantly less sinuous than a natural channel.

Figure 8.2: Channel formation on a mudflat outside Steart, Parret River, 23/02/2016 (Photo taken by John Davis)

8.2.4 Divergence 4: MR sites are emptier of creeks than natural saltmarshes

Proof of recurrence: This characteristic is common to all MR schemes: while natural creeks in the UK can be expected to have OPL values of 3 to 25 m, artificial creek systems in MR schemes have OPL values ranging from 25 m (Abbots Hall) to several hundred meters (Alkborough) (Figure 7.12). Natural and MR sites have significantly different distributions of both OPL and ground heterogeneity. Century-old accidentally realigned sites also display values of OPL above the equilibrium range of natural systems (Figure 7.14).

Cause: Excavating an initial creek system that covers the entire scheme is considered to be prohibitively expensive, especially in a large site. Therefore, current MR creek design involves the excavation of a simplified template of creeks, with few creeks of high RS order, which is expected to then develop over the years into a more natural shape and distribution (Pontee 2015a).

Consequence for the morphological evolution of the creeks: OPL values decrease linearly and often get halved after 5 years of evolution, proving that current design choices do allow some creek development. However, the linear trends observed are likely to turn into decaying trends over longer timescales. Indeed, while creeks develop preferentially in lower areas with higher inundation frequencies, higher regions are likely to remain empty of creeks. Assuming a decaying trend of site accretion and OPL evolution, all the studied MR schemes except Chowder Ness (small site and large opening) and Paull Holme Strays (good initial OPL and creek extension through both substrate erosion and depositional processes) are expected to take over 100 years to reach the same creek distribution as natural mature systems. This is the main and most consistent difference between natural and artificial creeks: MR creeks are characterised by a poorer distribution across the wetland.

Expected ecological consequences: The poor creek distribution in MR schemes is suspected to be one factor of their poor biodiversity, even after over a decade of evolution. Lack of drainage can lead to low sediment redox potentials adverse to the colonisation of certain saltmarsh plants (Mossman et al. 2012). Once again, this effect may be secondary to the impact of the soil properties, which explains why Abbots Hall fairs poorly in terms of biodiversity despite having the denser creek system (O'Brien et al. 2006; Brooks et al. 2015).

Implications for future practice: Creek distribution should be improved upon, either by excavating more creeks or preferably by encouraging unassisted creek growth. The goal OPL based on natural creek observation should be 25-30 m, but this goal may be difficult to achieve in the implementation phase due to material cut-fill balance constraints and implementation costs, especially in large MR sites. However, as some unassisted creek development can be expected, slightly lower values of initial OPL may be appropriate if the soil properties and tidal forcings allow for creek incision to occur. For instance, recent MR projects at Wallasea tried to achieve a distance between creeks of no more than 70 m (Wright et al. 2009). This means that measures must be taken to encourage natural creek development.

Creeks form through a mix of preferential deposition and erosional processes. In areas of high suspended sediment concentration, a low initial elevation will encourage creek growth by increasing the number of tidal events per year. In areas of high elevation, erosional processes need to be encouraged instead. Note that two of the studied natural wetlands (Crossens and Gibraltar Point) also displayed an underdeveloped creek system due to their coarser grained and less cohesive sediment, suggesting that sediment characteristics and/or geotechnical properties may also impact OPL evolution rates. Adding head ponds encourages the branching out of new channels into soft sediment in HOMW. Overall, the practice of kick-starting creek growth is still experimental: some recent design have added short creek segments or "grips" to guide the formation of tributaries (Figure 8.3). Their efficiency is still uncertain and will probably depend on expected rates of headward erosion to extend them (see Divergence 6). The addition of small-scale topography (i.e. small channels/levees) to help focus the flow into protocreeks, as discussed below, would be more judicious, being more similar to what is observed in natural systems.

Figure 8.3: Channel "grip" at Hesketh Out Marsh East, August 2017. Lidar image curtesy of Richard Shirres.

8.2.5 Divergence 5: MR sites are flatter than natural saltmarshes

Proof of recurrence: Natural and restored coastal wetlands, even after 100 years, have significantly different distributions of ground heterogeneity: 0.02–0.1 m difference in height between points up to 10 m apart for MR schemes and 0.1–0.2 m or higher for natural saltmarshes (Figure 7.7).

Cause: Since most MR schemes were previously used as agricultural land, they display very little small-scale topography. Features currently created, like at Steart, include lagoons several meters wide to provide fill material and bird feeding areas within view of visitors. Shallow scrapes are also commonly added, and more recently mounds of the order of 10-20 cm have been created at Hesketh Out Marsh East (Pontee, pers. com.). Steart is one of the flatter schemes identified with ground heterogeneities values of 0.02 m, so the features excavated there fail to solve the problem, and the site still appears very flat (Figure 8.4).

Figure 8.4: Oversized creek surrounded by empty, flat space in Steart, 23/05/2016, photo taken by Charlie Thompson

Consequence for the morphological evolution of the creeks: The formation of preferential flow paths based on initial ground heterogeneities encourages better creek distribution as described in Section 2.4.2, so a flat MR scheme may hinder creek development from its initial template and slow the OPL reduction rates. Preferential flow paths also provide paths of lesser friction through which water and sediment can be transported further inland, so their absence may play a role in the premature energy dissipation observed in MR schemes, leading to clusters of channels near the breach (Divergence 3).

Expected ecological consequences: Local topography encourages plant growth by providing niche habitats with better drainage at various locations of the site. Consequently, lack of microtopography is thought to be a major dampener of plant diversity in MR schemes (Brooks et al. 2015).

Implications for future practice: Future schemes should aim for a mean elevation difference of 0.1-0.2 m or higher for points up to 10 m apart, as in natural saltmarshes. Further research should look into the small-scale elevation heterogeneity of mudflats to advise the design of lower-lying MR schemes. This is especially important since MR substrate is not expected to be as mobile as that of a natural mudflat and saltmarsh, as explained below.

8.2.6 Divergence 6: MR creeks are rarely capable of expanding via headward erosion into the substrate

Proof of recurrence: Less than half of the considered sites (Freiston, HOMW, Welwick and Paull Holme Strays), those that have been embanked for 50 years or less, display clear signs of headward erosion of creeks into the substrate. Field observations at Steart show some bank slumping towards the head of one channel at a slow rate (0.4 m/year) despite the high tidal range within the Severn estuary.

Cause: Erosion of sediment in a natural marsh is linked to a balance between sediment shear strength and erosion shear stress caused by the tidal forcings (Watts et al., 2003). In natural saltmarshes, headward erosion is expected to be a major process of creek development (Hughes 2012), corresponding to phase C of the conceptual evolution model (Figure 2.9). Headward erosion rates have been measured at 5-7 m/year in undisturbed conditions (Steers 1960 *in* French et al. 1992), and can be as high as 400-500 m/year in disturbed systems (Knighton 1992; Symonds et al. 2007). The absence of erosion means either a low hydrodynamic energy or a high sediment shear strength. The initial elevation of the site (most MR sites start out at the elevation of a saltmarsh) could play a role by reducing the number and duration of tidal events, but some very high sites like HOMW display signs of headward erosion. Other factors, such as the soil structure of MR schemes, must play a role in dampening creek-forming erosional processes. Indeed, it has been observed that the substrate in MR soil is usually overconsolidated, which makes it harder for root systems to form (Spencer et al. 2012; Tempest et al. 2014). MR schemes where headward erosion occurs were embanked for 50 years or less. The hypothesis resulting from this thesis is that the compaction of the soil during the phase of agricultural use hinders creek incision into the substrate.

Consequence for the morphological evolution of the creeks: Two main creek-forming processes were identified in Section 2.4.2, namely: 1) depositional processes (heterogeneous deposition leading to preferential flow paths); and 2) erosional processes (headward erosion and incision of tributary creeks into channel bends). Depositional processes are likely to dominate in sites with low initial elevation and high accretion rates such as Chowder Ness and Allfleet. Pre-excavated creeks linked to large outlets like in Allfleet help to kick-start this process by providing clear flow paths. In Chowder Ness, while a creek network develops by building on the initial topography, the lack of pre-excavated outlets limits the volume of water forced through the channels at each tide, and the creek network remains very shallow. In higher sites and/or with low sediment input, and so where depositional processes play a smaller role, the dampening of erosional processes significantly hinders creek growth, as observed in Abbots Hall. This problem was identified early on by Burd (1995), who suggested lowering the site mean elevation to promote depositional processes.

Expected ecological consequences: It remains an open question whether low-lying MR schemes with high sedimentation rates can evolve into functioning mature saltmarshes regardless of the initial substrate properties. At Chowder Ness, the creek system is still shallow and unstable, but may evolve towards equilibrium creek cross-sections after vegetation has developed into the newly deposited sediment. However, if the new vegetation displays a poor biodiversity, like that observed at several other MR schemes (Mossman et al. 2012), the poorer root penetration will lead to a less efficient stabilisation of the creek banks and less efficient flow rerouting. In MR sites that display low accretion rates, either due to their initial elevation or to low sediment availability, soil overcompaction is almost guaranteed to negatively impact plant diversity through waterlogging and poor root penetration.

Implications for future practice: Further study of MR creek headward erosion, done in tandem with sedimentological and geotechnical monitoring would be needed to confirm this tentative observation from lidar data. However there is a growing consensus that MR soils are fundamentally different from saltmarsh soils, and that this could negatively impact saltmarsh development. As such, soil reworking in future MR schemes is recommended to break down the hardened soil layer and facilitate creek incision. Previous soil treatment testings include rototilling kelp compost into the soil, which improved organic matter content and soil moisture, lowered bulk density, and significantly improved plant survival rates (O'Brien et al. 2006). Soil reworking could also add heterogeneities to the substrate and thus mitigate for Divergence 5. However, alteration of the soil structure during the embankment period may be irreversible (Crooks et al. 2000; Tempest et al. 2014). Another option would be total or partial site lowering to encourage depositional processes. Headponds are often excavated in MR schemes, and bear some similarity to those naturally occurring within eroding saltmarsh creeks (Hughes et al. 2009), so could be considered a good design choice to encourage channel development. Preliminary field observations at Steart have found that those headponds contribute to ebb flow re-routing and so the capacity of new channels to form (Figure 8.5).

Figure 8.5: Ebb flow routing through pre-excavated channel head tributary at Steart (Photo taken by Anas Annuar, 23/02/2015). This has continued to expand, leading to secondary channel formation into the adjacent marsh.

8.3 Conceptual model for MR creek evolution

The previous section has highlighted how current MR creek design strategies lead to significant divergences between natural and MR creeks, as summarised in Table 8.1. Those differences do not diminish significantly after 5 years of evolution. Extrapolations of OPL evolution trends suggest that they are unlikely to ever reach the creek distribution characteristic of natural saltmarshes.

Current guidelines provide advice on creek dimensions and morphology (Coats et al. 1995). The observed differences between natural and artificial creek networks occur either because currently available guidelines are not all applied, or because the guidelines themselves are not sufficient to provide good creek morphologies. This is explored by providing a summary of current creek guidelines in the literature (see Table 2.1) and verifying to what extent, in the studied MR creeks, these guidelines were applied (Table 8.2).

Table 8.2: Application of existing creek network guidelines for the construction phase in current UK MR schemes

Factor	**Guidelines on construction**	**Are the guidelines applied in the MR sites considered?**
Initial marsh elevation	Initial marsh elevation should be about MHWS-0.50 m (MHWS) to promote depositional processes (Burd 1995). Slope should be added to the marsh surface to encourage drainage (Williams et al. 2004)	**YES** MR scheme implemented below MHWS elevation (Appendix C1). Slope at Abbotts Hall and Welwick
Total creek volume	The creek volume should be at equilibrium with the potential tidal prism to prevent sediment infill, erosion or poor water circulation (Haltiner et al. 1987). Equilibrium relationships relate creek volume to the total discharge (Haltiner et al. 1987) and outlet cross-sectional area (Steel 1996).	**NO** the MR creek volume is within the expected range for a given catchment area (Figure 7.6), but the outlet cross-sectional area is above the equilibrium range (Figure 7.11) as the flow is focused through a breach constrained by embankments
Channel length	Channel length per creek should be a function of the creek order following Horton's power laws of channel length decrease with increasing Strahler order (Coats et al. 1995).	**YES** the overall creek system is too sparse, with not enough creeks, but the distribution of creek mean length per order falls within the expected range for natural systems (Figure 7.4)
Channel cross-sectional dimensions	Channel width and depth should obey hydraulic geometry relationships, and have similar width/depth ratios as natural systems (Zeff 1999).	**YES** for the larger, low RS order channels (Figure 7.4 **NO** for the high RS order creeks (Figure 7.4) because of the reactivation of drainage ditches
Bifurcation ratios	Bifurcation ratios should be approximately 3.5 as in natural systems (Coats et al. 1995).	**YES** (Figure 7.4)
Junction angle	Slough junctions should be about 120 degrees, and channel junctions with sloughs at about 90 degrees to imitate natural systems (Haltiner et al. 1987).	**YES** during the design phase (Figure 7.3) **BUT** the subsequent development of the breach area leads to clusters of new creeks in the lower sections, at lower junction angle (Figure 7.4)
Sinuosity ratio	The mean sinuosity ratio should range between 1.1 and 2.0 for Strahler orders 3-5 to imitate natural systems. Smaller channels will tend to be straighter (Coats et al. 1995).	**NO** during the design phase due to the reactivation of drainage ditches (Figure 7.3): the effect remains prominent even after more sinuous new high RS order creeks form
Drainage densities and distribution	Drainage density should be similar to that of nearby reference saltmarshes (Williams et al. 2004). Maximum distance between creeks should be about 30 m (Haltiner et al. 1987).	**NO:** the drainage density and creek distribution as shown by OPL remain too low after > 5 years of evolution (Figure 7.6)
Inherited structures	Inherited structures like drainage ditches should ideally be infilled so as to prevent overly straight channels (Williams et al. 2004).	**NO** in most of the sites considered, but this is improving: drainage ditches have been infilled partially at HOMW, and completely at recent sites like Steart and Hesketh Out Marsh East

Overall there is a good application of available creek guidelines in UK MR schemes. The main source of differences between recommendations and end-product is the reactivation of drainage ditches, which leads to overly straight channels of unnatural, deep and narrow shapes. Another source is the large and high-energy breach area, where a cluster of creeks tends to develop, leading to low junction angles near the breach and to a creek volume out of equilibrium with the outlet cross-sectional area. Finally, not enough creeks are excavated overall, leading to poor creek distribution. This can be summed up into a simplified conceptual model for MR creek evolution (Figure 8.6). The model is explained below and related to observations in the 10 MR schemes monitored:

Stage 1 represents the initial design, and as such bears the fewest similarities to Steel et al's model (1997). MR schemes are not necessarily initiated at the elevation of a mudflat, and the excavated creeks are often straighter and wider than naturally occurring creeks. After 2-3 years, the breach area adjusts rapidly to the tidal regime post-implementation (visible at Freiston, HOMW and Steart), corresponding to Stage 2. While the breach and entry channel deepen, the furthest reaches of the creek system infill with sediment due to the lower tidal energy there (visible at HOMW and Steart). This adjustment period to tidal influence is also accompanied by the die-off of terrestrial vegetation and the colonisation by saltmarsh plants once the elevation permits (within the first 2 years at Steart, 5 years after implementation at Chowder Ness). Stage 2 is followed by a period of creek growth not unlike phase C of Steel et al.'s model (1997): the creek growth phase corresponds to Stage 3. Creeks grow preferentially in the lower regions and in headward ponds where soft sediment has deposited: the majority of the MR creeks considered fail to incise into the agricultural soil to expand beyond their initial excavated template. After Stage 3, most of the creek expansion has taken place, and both accretion rates and OPL evolution rates start following a decaying trend (visible after 5 to 15 years at Abbots Hall, Allfleet, Freiston and Welwick). Other sites characterised by high accretion rates, like Tollesbury and Chowder Ness, may take longer to reach their maximal creek extent (> 20 years). Decaying OPL evolution rates observed in Chapter 6 suggest that creek growth beyond Stage 3 will be limited and that the landward, higher reaches of the sites will remain empty of creeks. Based on these decaying trends, on the negligible creek evolution rates occurring within two century-old AR sites, and assuming a similarity with Steel and Pye’s model (1997), it can be assumed that MR creeks will reach a stage of creek stabilisation and/or contraction (Stage 4), though this hasn’t been observed in the MR dataset due to the young age of the schemes considered. Overall, MR creeks behave as though they will develop into a MR-specific equilibrium state, characterised by a smaller number of oversized creeks, and with most of the branching and energy dissipation occurring near the breach area while the higher, landward sections remain empty. In the next section, implications for future practice will be tentatively proposed to inform the design of future MR creeks.

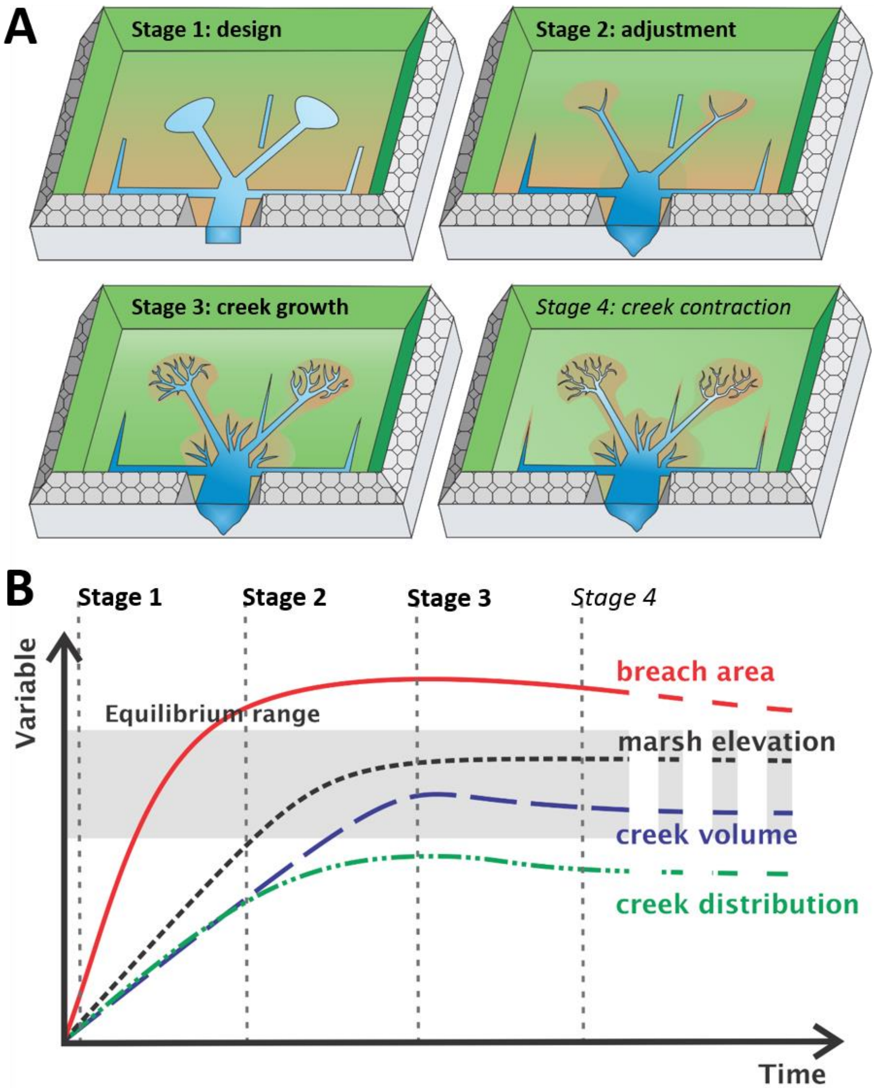


Figure 8.6: A: Conceptual morphological evolution model of MR creek networks based on observations at 10 MR schemes in the UK. B: Evolution of key variables as marsh elevation increases to equilibrium with the tidal range. The breach cross-sectional area, if built too small at Stage 1, increases rapidly and stabilises above the equilibrium range due to the flow being forced through the breach by the remaining embankments (Stage 2). Deepening of the entry channel leads to a rapid increase in creek volume, but the absence of sufficient flow routing landwards leads to premature energy dissipation in a cluster of smaller creeks near the breach (Stage 3). At that stage, the number of creeks increases in the lower regions, in headward ponds and through reactivation of drainage ditches, the creek distribution remains below the natural equilibrium range (high OPL) because the creek system cannot expand significantly beyond its initial template. Creeks inherited from drainage ditches remain overly straight and show no sign of expansion. Based on Steel et al.'s model (1997) and on the observations of 2 century-old AR sites, the final stage should see the creek network stabilise or shrink as the number of tidal events reaching the site diminishes (expected Stage 4).

8.4 Implications for future practice

The differences between natural and MR coastal wetlands as highlighted by the conceptual evolution model have two main causes. The first source of differences is the presence of the old embankment in most schemes, which constrains the flow through the outlet, leading to larger cross-sectional areas (Figure 8.6) and probably causing MR schemes to remain flooded for longer. As the hydroperiod controls vegetation survival, this can have an impact on the ecological evolution of the site. A longer hydroperiod would lead to the site being exposed to wind waves for longer, which may lead to enhanced erosion of the creek banks (Tonelli et al. 2010). The second source of differences is the greater creek sparsity and lower ground heterogeneity in MR sites compared to natural sites, with expected negative impacts on plant diversity (O'Brien et al. 2006; Mossman et al. 2012; Spencer et al. 2012; Brooks et al. 2015).

While creek sparsity is a problem common to most MR schemes, simply excavating more creeks will not be sufficient to promote better biodiversity. Abbots Hall contains a denser system of creeks than most other sites, with OPL values ranging between 17 and 26 m (Table 6.4), but still displays poor plant biodiversity (Brooks et al. 2015). The problem is suspected of being exacerbated by the altered soils inherited from the time those sites spent embanked and used as pasture or agricultural lands. The sediments have been altered and compacted thus preventing direct creek incision, but also drainage, nutrient transport and root growth (Spencer et al. 2012; Tempest et al. 2014). This can have indirect negative impacts on creek growth by hindering the biophysical feedback processes developed in Section 2.4.2. Other ecosystem services usually associated with coastal wetlands could be impacted in turn, potentially reducing the value of MR schemes if not addressed properly. The protection of the new embankment against erosion through the energy dissipation action of plants within the MR site could be less efficient (Möller et al. 2014). The potential impact on carbon storage is uncertain: on the one hand the longer hydroperiod may make MR schemes more efficient sediment sinks and improve the storage of imported carbon; on the other hand, low biodiversity may lead to poorer local organic matter production and storage (Ahn et al. 2013).

Further design recommendations should thus be added to the current ones, focusing on soil properties as well as on initial morphology. Figure 8.7 provides a tentative conceptual model for future designs. The design depends on whether it is feasible to lower the initial elevation to that of a mudflat and to provide a wide breach area/bank removal (this depends on other constraints and on Townend (2008) model for breach design). In low-lying sites with high sediment availability, the focus should be on facilitating natural creek-forming processes from a simplified template. Facilitation procedures include preparing the soil to reduce its compaction, adding small-scale topography, and excavating a short, sinuous and branching entry channel to help direct the flow to the landward reaches of the site (Figure 8.7A-C).

In sites that lie high within the tidal frame and/or have a low sediment availability, the creek network cannot be expected to form unassisted due to the limited accretion rates within the site, and a more extensive initial template is necessary. The goal OPL should be 25-30 m, but this might pose practical challenges as explained further in Section 8.5. Even in high-lying sites, some creek development can be expected, so OPL values of about 50-70 m should be enough, especially if the shear stress/shear strength ratio allows for creek-forming erosional processes. Adding head ponds could encourage sediment deposition into which new channels can branch out (Figure 8.7B-D). Sinuosity can also increase shear stress at the bends and promote incision of new channels (occurs in one place at HOMW). The design of creeks will still be experimental: different ways of kick-starting a creek or treating the substrate should be tested depending on the means of each particular site. Experimental creek starters like grips, protocreeks or preferential flowpaths through microtopography should be placed at the channel bends so higher flow energy goes through them and they are less likely to be abandoned (Figure 8.7C).

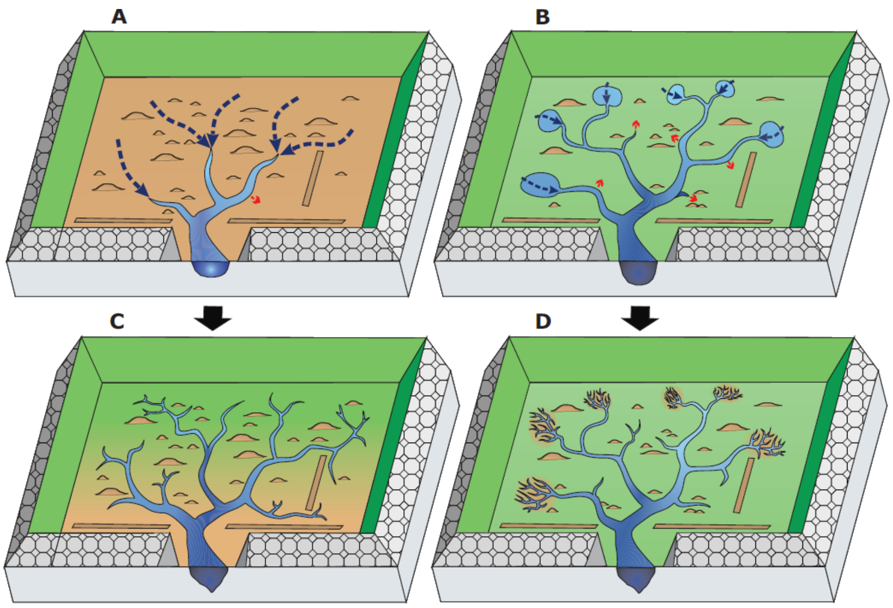


Figure 8.7: Conceptual model for future MR schemes design (A-B) and their expected evolution (C-D). A: initial elevation of a mudflat, with high enough sediment concentrations for deposition-driven creek growth to occur. Minimal sinuous creek network with small-scale topography to focus flow and reduced soil compaction should promote creek growth. B: initial elevation of a saltmarsh or low expected accretion rates: more extensive creek system necessary. Headponds and bends in the creeks can promote branching out of new creeks. Also add small-scale topography and reduce soil compaction to promote creek growth. In both scenarios, relic drainage ditches should be infilled.

8.5 Addressing practical limits to MR implementation

This research explores design options for MR schemes based on morphodynamic and ecological considerations. However, as has been hinted at throughout this thesis, design of MR schemes also depends on practical considerations. The characteristics of MR schemes (tidal range, size of the site, characteristics of the sediment) depend on site availability. Indeed, while most design guide books available assume "strategic" MR, almost half of the MR schemes in the UK fall in the category of "opportunistic" MR (Leggett et al. 2004). A list of potential sites for wetland restoration in Europe has been proposed by previous studies (Schleupner et al. 2013), but the probability of finding a perfectly suitable site for a given project is low. Limiting factors include economic and social factors such as proximity of the site to urban areas, and opposition from the stakeholders (Pontee 2014b). Other factors of success include public acceptance of the project. The example of Steart shows that long planning (over 10 years) and regular communication with local communities, directly with the site managers rather than through the media, has helped public opinion.

Consequently, the design of creek networks will have to take into account a number of practical considerations, such as avoiding pipes and cables, or in some cases providing safe access for farm animals. Crucially, creek excavation should not be prohibitively expensive, and should maintain a cut-fill balance, that is to say a balance between the sediment excavated on site (breach area, creeks, lagoons, site lowering) and the sediment used to build new features (new flood defences, mounds). Those constraints make the excavation of creeks at the right OPL values while maintaining the Horton's laws of creek numbers problematic to apply in large schemes: hundreds of first RS order creeks would have to be excavated to replicate a mature system's morphology. Practical considerations are probably the main reason why creek density guidelines established early in the history of MR practice (Coats et al. 1995) are still poorly applied.

To circumvent this issue, there is a need to increase the current toolkit of creek initiation options so designers can adapt to the particular constraints of each site. Design options will also be easier to defend if they provide additional benefits (niche habitats, local community engagement, etc.). The following bullet points list a few practical options to encourage better creek distribution:

- Headponds are increasingly popular as a design method: they resemble naturally-occurring ponds at the head of eroding creek systems. They are likely to promote headward erosion by providing an extended source of ebb flow (Figure 8.5), and the deposition of soft sediment promotes dense creek branching (Figure 8.7D), which could constitute a good interface for juvenile fish to access the saltmarsh. Finally, headponds in the landward reaches of MR schemes provide bird-watching spots within view of footpaths that are typically installed on the top of the new embankments.

- Modelling results have found the initial topography to be highly deterministic of the creek network morphology (D'Alpaos et al. 2007). Shaping the scheme microtopography to create preferential flow paths could constrain creek growth into the desired final shape. The cut-fill balance would be easier to maintain with this method since the heterogeneities would be both positive and negative. This design option also agrees with recommendations to provide more niche habitats at different elevations for plants: this is considered more efficient than a smooth slope from the breach to the landward reaches.
- Various types of protocreeks have been tested in the US and UK. Bales of hay have been placed along desired flow paths to locally prevent sedimentation: when the hay eventually disperses, it leaves behind unvegetated troughs that are expected to deepen into creeks (ABP Southampton 1998). No study could be found testing the efficiency of this method. The excavated creeks at Hesketh Out Marsh East, which opened in August 2017, include short grips that aim to initiate the branching out of new channels (Figure 8.3), but similarly the efficiency of such a method is uncertain. Another option would be to hand-dig shallow creeks after the initial excavation phase to kick-start a denser system. This method could be cost-effective if it were done by volunteers, following the model of volunteer planting and wildlife flushing before site implementation, as happened at Steart. This approach could help to engage with the local community.

8.6 Summary

Several key divergences of MR creeks from natural systems have been highlighted. A new conceptual model has been proposed to represent creek design and evolution in current MR schemes in the UK. MR creeks are built oversized and connected to a breach area constrained by embankments. This configuration forces most of the tidal prism through the breach, and leads to a rapid readjustment of the entry channel to accommodate the undermarsh flow. Despite existing recommendations to infill drainage ditches, in a lot of current schemes these features are kept or get reactivated post-implementation, leading to a creek system made of overly straight and leptokurtic segments. The formation of new creeks is generally limited to the lower regions of the site near the breach, where tidal energy and hydroperiod are greatest, and to other low-lying features of rapid sediment deposition like headward ponds. This stage of creek growth bears similarities to stage C of Steel et al.'s (1997) conceptual creek evolution model, but is more limited in extent. The lack of microtopography and altered soil properties in MR schemes due to the previous land use probably plays a role in hindering creek expansion past the initial excavated template. Future schemes should either excavate more creeks, or preferably provide more favourable initial conditions of creek development, such as less compacted soils to favour creek incision and more initial topography variations to favour the initiation of preferential flow paths and heterogeneous accretion.

Chapter 9: Conclusions and further research

9.1 Conclusions

The overall aim of this thesis was to improve understanding of the morphological evolution and expansion of natural and MR creek networks in order to derive implications for future MR practice. This was addressed through 4 Thesis Objectives. The key conclusions from each of these 4 objectives are discussed in the following 4 paragraphs, respectively.

Thesis Objective 1 was to provide a comprehensive morphometric analysis of creek systems in natural coastal wetlands in the UK, using remote sensing datasets and novel data extraction methods, to define equilibrium characteristics that can be used as an end target for MR creek design. After defining the main processes of creek initiation and development in natural coastal wetlands in Chapter 2, a novel semi-automated algorithm was developed in Chapter 4 to map and parametrise complex creek systems. Chapter 5 applied this algorithm to 13 natural mature coastal wetlands in the UK, in order to test the applicability of currently existing morphological equilibrium relationships to creek networks extracted from 1 m resolution lidar data. The method yielded consistent results with that of field-validated manual mapping from aerial photography for the 13 sites, and was significantly faster and less subjective to implement. The creek parameters obtained via this method successfully reproduced the morphological equilibrium relationships used in the literature as the best available empirical model of creek evolution towards a mature state. In addition, a new relationship relating the creek distribution within the site to the site elevation within the tidal frame and the dimensions of the outlet was developed. This relationship gives the expected sparsity of creeks, that is to say the mean distance to the nearest creek at all points within the marsh. A creek sparsity above equilibrium values is suspected to have negative consequences for the ecological functioning of the site. As such the new relationship developed in Chapter 5 provides a valuable end target for MR creek distribution.

Thesis Objective 2 was to undertake a morphometric analysis of creek systems in MR schemes in the UK and of their evolution over the years, in order to estimate creek evolution rates. The morphological evolution of 10 UK MR schemes was investigated in Chapter 6 using 1 m resolution lidar datasets. Between 3 and 8 individual datasets were available for each site, covering between 2 and 20 years of evolution since the site opening, making this study the largest systematic analysis of MR creeks development to date. In addition to data availability, the schemes were chosen to encompass a wide range of locations, external conditions such as tidal range, and design choices. Results showed that the 10 studied MR creeks evolved following a near-linear evolution trend since the scheme implementation towards larger, more complex and better distributed systems. That evolution was mainly driven by depositional processes, because MR schemes typically act as sediment sinks, and because the incision of new channels into the substrate is hindered for most MR schemes, likely due to a higher compaction of the previously reclaimed land. Indeed, only the sites that had been removed from tidal influence for the shortest amount of time (<= 50 years) displayed clear signs of erosion into the substrate. Aside from the very energetic breach area, the creeks' shapes do not evolve from their initial template and remain less sinuous than natural creeks. This could be due to the overly straight design of the initial template: the usual process of sinuosity accentuation through erosion at the bend of the creeks cannot take place. Another possibility is that the soil is too compacted and too flat to allow for much erosion to occur outside the breach area, leading to a frozen template.

Thesis Objective 3 was to compare the morphological characteristics of natural and MR creeks to determine whether MR creeks evolve towards more natural systems over time, and whether their evolution rates can be related to initial conditions and design choices. Chapter 7 showed that, after 5 years of evolution and more, the total channel length and other morphological characteristics of MR schemes fall within the range of equilibrium values. However, the outlet dimensions and the creek distribution remain significantly different from those of natural mature systems. Rapid changes occur within the entry channel and around the breach area, where the hydrodynamic energy is greatest due to the focusing action of the breaches. As a result the MR entry channels, already built oversized, tend to become larger rather than shrink towards a more natural shape, and the dimensions of the creek networks' outlets exceed the equilibrium range. Furthermore, MR schemes remain significantly emptier of creeks than natural sites, even when they reach the expected total creek length and volume, because those creeks are oversized and concentrated around the breach area, leaving the further reaches of the site undrained. Some creek characteristics such as OPL are expected to evolve following a decaying trend, and indeed the creek evolution rates measured within two century-old accidentally realigned sites are negligible, with OPL values higher than the natural equilibrium range. Consequently MR schemes are unlikely to reach the equilibrium range for creek distribution unassisted, even after 100 years. The main drivers of creek evolution identified by PCA are the dimensions of the outlet, which determine how much of the tidal prism is forced through the creek system to encourage creek-forming processes.

Thesis Objective 4 was to establish a conceptual creek evolution model for MR schemes that highlights the main divergences from natural creek evolution, and to derive implications for

future practice in order to reduce these divergences in future schemes. Results from Chapter 8 showed an overall good application of the available creek guidelines, which aim to reproduce natural creek shapes, to current UK MR schemes. The main source of differences between recommendations and end-product is the reactivation of drainage ditches, which leads to overly straight channels of unnatural, deep and narrow shapes. Another source of difference is the large and high-energy breach area, where a cluster of creeks tends to develop. Finally, not enough creeks are excavated overall, leading to poor creek distribution. Those differences are not enough to explain why MR creeks are not more reactive and do not grow to occupy a greater proportion of the MR scheme. Instead, lower ground heterogeneity and overconsolidation of the MR soils due to their previous agricultural use are expected to play an important role in hindering creek development. Substrate preparation before site opening to reduce the compaction of surface sediments could promote better creek development as well as better root infiltration. Adding small-scale topography strategically to focus the flow and increase channel growth rate would also encourage natural development. The state of knowledge, and the great variability of MR schemes' context and objectives make it arduous to provide systematic guidelines for creek design. However, the observations made in this thesis provide direction for further experimentation with creek design in future schemes.

In conclusion, this thesis has linked traditional methods of estimating creek networks equilibrium morphologies, through empirical power-law relationships, with novel creek parametrisation methods, opening the way for more systematic and repeatable monitoring of natural and MR creeks. The creek volumetric characteristics of 10 UK MR schemes implemented between 1995 and 2014 have been compared with the characteristics of 13 mature natural creek systems and with those of 2 accidentally realigned sites. MR creeks follow near-linear evolution trends towards larger and better distributed systems, showing a good potential for creek development. Most MR creeks considered reach the equilibrium range for the total length and volume in under 20 years, but their distribution across the site remains poor. MR creeks are oversized and develop in clusters in the lower regions near the breaches, leaving the higher, landward regions empty. Low initial elevation, a large enough connection to the wider estuary and high sediment availability are critical to encourage creek growth, but further research should also explore the role of initial microtopography and bed shear stress on creek development in realigned agricultural lands. The main findings are given in the following bullet points:

- Creek systems from both natural and restored coastal wetlands can be mapped efficiently and semi-automatically using a novel Matlab parametrisation tool on lidar data. This method of analysis is faster and less subjective than manual mapping, and allows for the extraction of a large number of creek morphological parameters including creek length, order, sinuosity, cross-sectional area, volume and distribution.

- Morphological equilibrium relationships for creek parameters previously established in the literature have low predictive values due to the natural variability of creek systems, and should be used as an equilibrium range rather than a single-line equilibrium state.
- In addition to the relationships previously established in the literature, a morphological equilibrium relationship exists between the overmarsh path length (mean distance to a creek anywhere in the marsh), and the mean site elevation within the tidal frame. This relationship provides a range of creek distribution values in natural systems, which can be used as a semi-quantitative target for MR creek design.
- MR creeks in their first 2-20 years post-breach evolve near-linearly towards larger, more sinuous and better distributed systems of larger volume. The equilibrium range is reached for most sites after 5 years for the total channel length, main channel length and breach depth, but the MR schemes remain too empty of creeks compared with natural systems.
- The MR schemes and AR sites considered behave as though they should stabilise within 100 years into an alternative equilibrium state, different from that of natural systems. This alternative equilibrium is characterised by a poorer distribution of creeks, mainly cluttered around the breach areas leaving the further reaches of the site empty, a lower sinuosity due to inherited drainage ditches that remain visible even after 100 years, and a flatter substrate.
- Creeks were found to rarely extend past their initial templates in restored coastal wetlands, especially if the site had been embanked for over 50 years before realignment. Little erosion could be observed within the MR sites' substrate, and century-old accidentally breached sites with a long history of embankment still display visible remnant structures of drainage ditches. This points to a strong heredity of the initial topography even after 100 years. The initial design of MR schemes and of their excavated creeks is thus expected to have long-lasting consequences.
- Other studies have pointed out that the sediment in restored coastal wetlands is highly compacted due to the embankment phase, with heretofore poorly researched alterations to the soil structure. Further studies should look into the previous land use and altered soil structure as probable factors dampening the extension of MR creeks past their initial template.

9.2 Further Research

The research conducted herein stresses the importance of coastal wetlands and the need to compensate for lost habitats, highlights the fundamental divergences between natural and MR creek development, and discusses the potential impact of these divergences on ecological functioning. A number of suggestions for further studies can be inferred from these conclusions. These can be grouped into three main axes of research: 1) to improve currently available creek monitoring tools to better capture rapidly evolving creek systems; 2) to improve understanding of the sedimentological processes and biogeochemical feedbacks that drive creek expansion, especially in MR schemes where the previous land use may affect those processes; and 3) to explore the potential of creek growth as a geomorphological proxy of saltmarsh alteration (drowning or retreat) in the context of climate change. These are detailed below:

9.2.1 Improve creek monitoring tools

The creek parametrisation algorithm developed for this research project could be improved to be better adapted to the monitoring of rapidly evolving creek systems within natural and realigned coastal wetlands. Many recent studies have addressed the problem of creek network mapping from lidar data; however existing algorithms are not yet adapted to geomorphological interpretation. A computing research project would give the opportunity to enhance the geomorphology-adapted algorithm proposed herein by combining it with other lidar interpretation methods. For instance, a vegetation detection function could be implemented to help correct accretion values by exploiting the red and near-infrared wavelengths available from recent lidar technology (Collin et al. 2010). The creek detection with threshold method should also be confronted with creek detection using the Gaussian shapes (Liu et al. 2015) to verify the potential applicability of the latter method to MR schemes, and whether it allows for a more accurate detection of creek edges.

Furthermore, significant differences between natural and MR creeks make the latter systems more challenging to map and parametrise. First, because many MR creeks build on previously existing features like drainage ditches, they are highly interconnected. Therefore creek ordering (Strahler or Reverse Strahler) is challenging to apply in its current form. A new ordering system, for instance based on the volumetric properties of each creek segment, may need to be defined to better capture the evolution of MR creeks and facilitate geomorphological interpretation. Second, a different sinuosity ratio extraction method may be needed for artificial systems, by taking into consideration “kinked” creeks in a reticulate system of perpendicular straight channels, for instance by using the mean radius of curvature (Marani et al. 2002) rather than the ratio of sinuous length to straight length used herein.

Finally, in order to define the state of dynamic morphological equilibrium, this thesis relies on relationships established for small saltmarshes characterised by a single creek system linked to a single outlet. MR schemes, by contrast, often have a number of interconnected outlets, which makes it difficult to study individual creek systems separately. While this thesis provided arguments that single-system saltmarshes and dike-adjacent marsh islands obey the same morphological rules, in accordance with previous studies (Novakowski et al. 2004; Hood 2007), it would be worthwhile to formally establish this equivalence by producing morphological equilibrium relationships from a whole dataset of marsh islands. These relationships might provide better targets for MR schemes' creek morphology, and thus better monitoring tools.

9.2.2 Improve understanding of creek forming processes

Results from this thesis suggest that creek expansion is hindered in MR schemes. Further research is needed to understand the underlying processes at play. Such a study should be based primarily on field surveying, in order to collect more data on the hydrodynamic processes (not just tidal levels but hydroperiod, flow velocity and wave action) and on the sedimentological and geotechnical properties of MR schemes (granulometry, bed shear strength, suspended sediment concentration) to better interpret the morphological changes occurring within MR creeks. It is suggested in this thesis that the underdevelopment of creek networks in MR schemes is linked to the overconsolidation of the agricultural soil. This hypothesis should be tested by conducting a formal study relating sediment shear strength, shear stress and creek development patterns in MR schemes, natural mudflats and immature natural saltmarshes (Stages A to C of Steel et al.'s model (1997), Figure 2.9), in order to verify whether agricultural soil hinders erosional processes of creek development. Another hypothesis of this thesis is that lack of small-scale topography hinders creek development by failing to provide preferential flow paths where the shear stress will be locally higher. There should also be feedback processes with plant colonisation, which depends on creek proximity and topography, and in turn affects the stability of the creeks and accentuates ground heterogeneity through the trapping action of stems and leaves. A multidisciplinary study of these feedbacks processes would provide more insight into the functioning of MR schemes, and help improve future design.

However, there is a limit to how much field surveying can be undertaken, so other ways of conducting detailed analysis of creek network evolution within MR schemes should be explored.

Another direction this research could take would be to run hydro- and morpho-dynamic evolution models to explore a wider range of synthetic MR designs (i.e. with different number and sizes of breaches, tidal channel configurations and forms) and for a range of different external conditions like tidal range, sea-level rise, suspended sediment availability and soil properties. The model would be calibrated using data collected at existing MR schemes, in the field and through a remote-sensing analysis such as those conducted in this thesis, and would provide an opportunity to test a greater number of design options for future schemes than what can be explored from case studies alone.

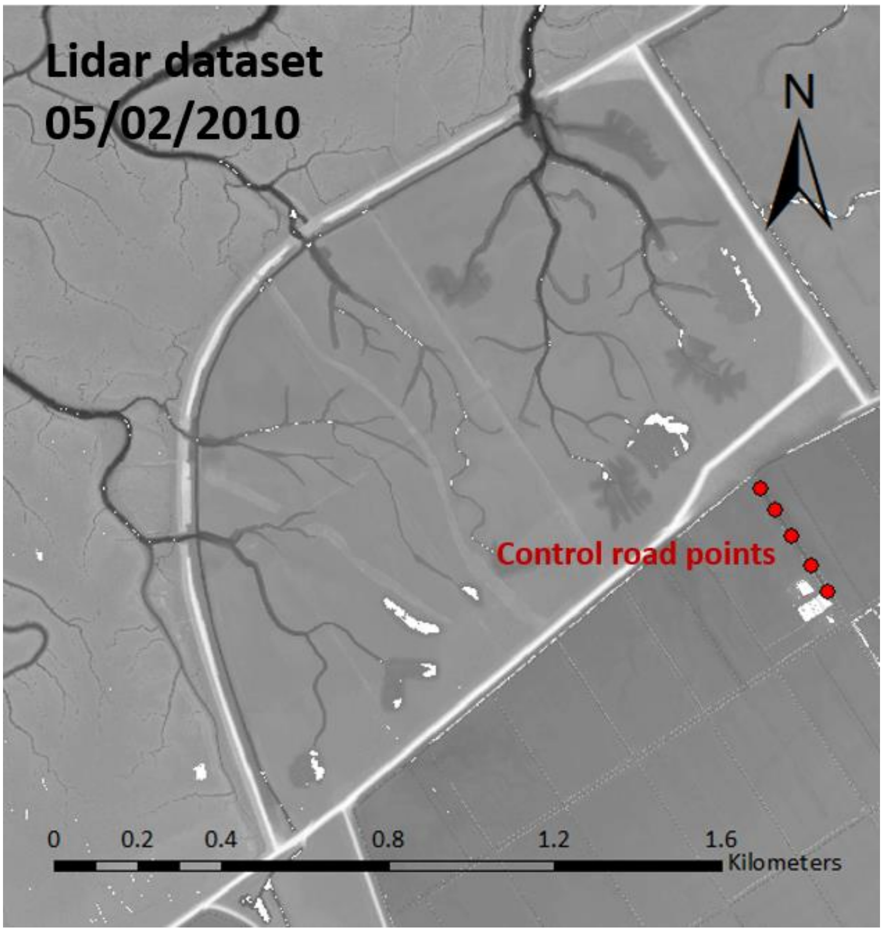
9.2.3 Explore the potential of creek growth as a geomorphological proxy of a saltmarsh drowning or retreat

The tools developed as part for this thesis make it possible to conduct a systematic and repeatable analysis of creek network development. These tools could easily be applied to the monitoring of rapidly evolving coastal wetland systems in a changing climate. Indeed, significant morphological changes are expected to occur within the creek network as the saltmarsh drowns (Schepers et al. 2017) or retreats landwards (Kirwan et al. 2016) in response to relative sea-level rise. Changes in the creek network structure is already used as a proxy of saltmarsh degradation and drowning (Schepers et al. 2017). Due to the dramatic rate of headward extension observed in some natural systems (Knighton 1992), creek growth should also be considered as a potential proxy of landward retreat. This proxy would provide a geomorphological counterpart to the biological proxies of landward saltmarsh retreat currently used, such as changes in vegetation cover (Kirwan et al. 2016).

Finally, while this thesis has focused on temperate high latitude coastal wetlands such as those found in the UK, it is crucial to also monitor lower latitude areas where saltmarshes and mangroves interact. Indeed, due to the competition between the two types of wetlands, these regions are particularly sensitive to changes in temperature and in relative sea level (McKee et al. 2012), and thus form important sentinels of climate change. The creek parametrisation tools developed in this thesis could help to monitor areas of saltmarshes that are susceptible of being encroached by mangroves in the poleward limits of mangrove distribution, provided that regular lidar data is available.

Appendix A : Vertical repeatability test

The repeatability of lidar data was tested for 5 different datasets collected near HOMW in 2007, 2009, 2010, 2011 and 2014. Elevation values were found using the nearest neighbour interpolation method along a road. The vertical repeatability error between years is estimated from the mean standard deviation for the five points.



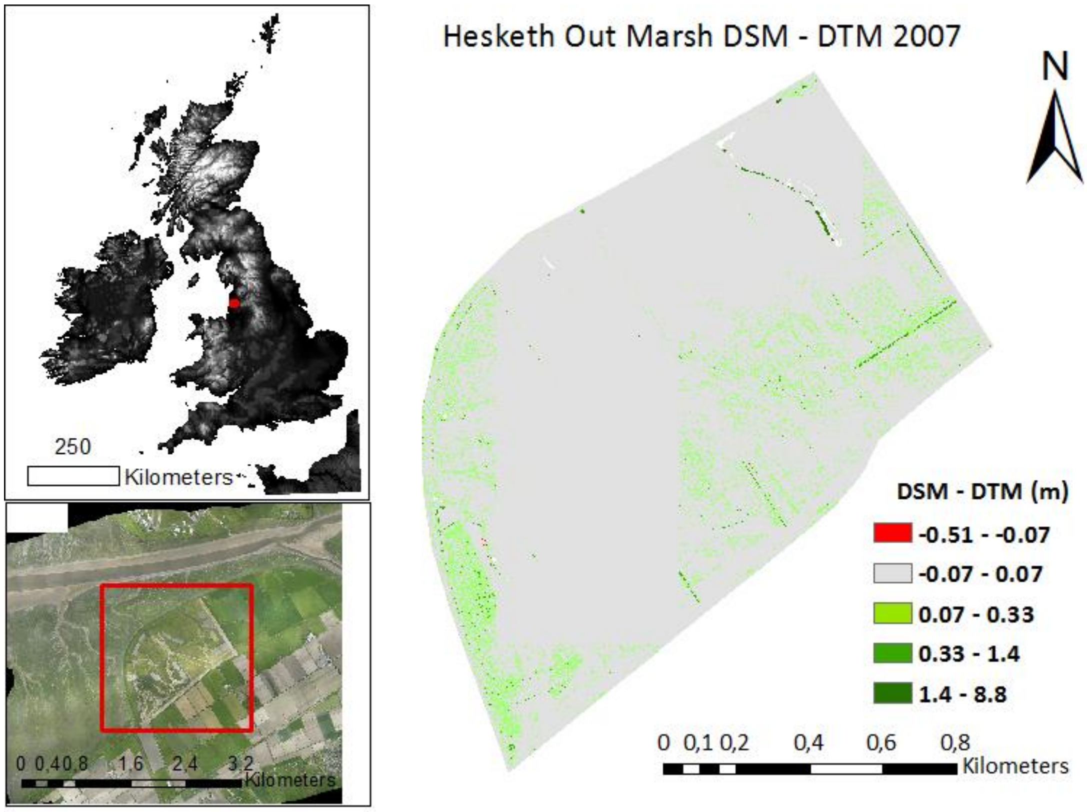
Appendix A1: Location of the 5 points chosen along a road to determine the interannual variability of lidar data at HOMW

Appendix A2: Coordinates and interpolated elevation at the 5 selected road points for the 5 years considered

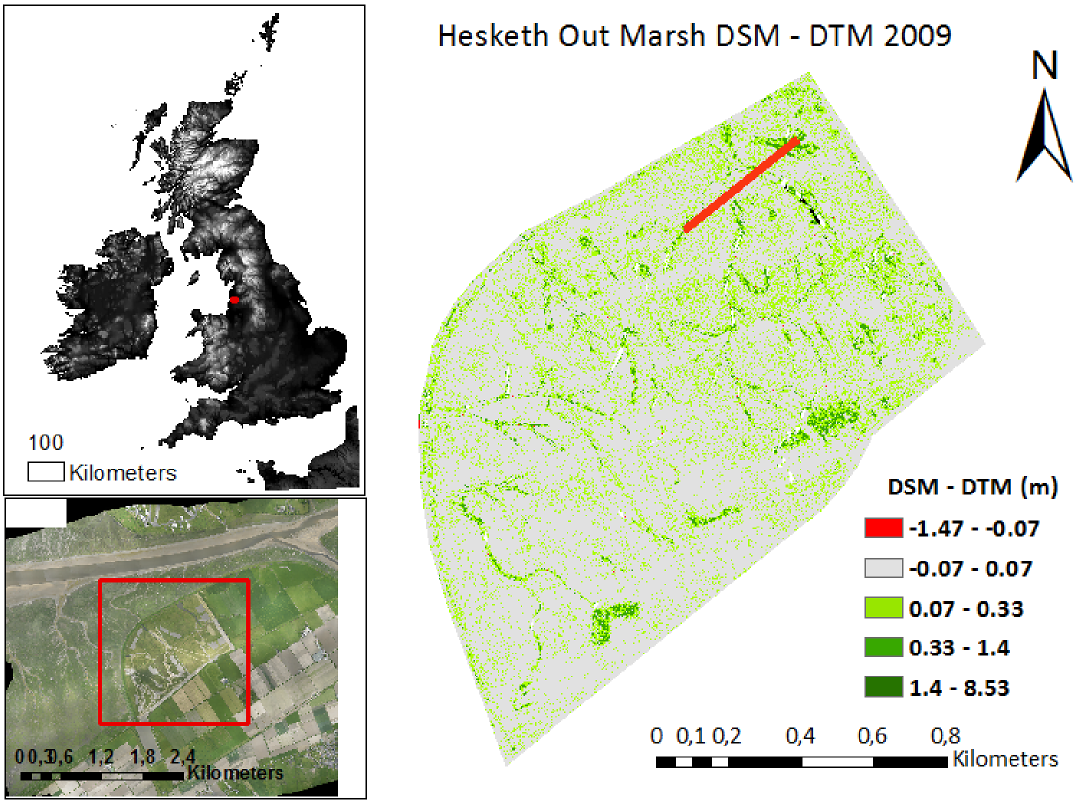
	X position	Y position	Elevation (m)					
			2007	**2009**	**2010**	**2011**	**2014**	**STD**
Road 1	342208.1	424964.0	3.45	3.45	3.47	3.55	3.56	0.05
Road 2	342334.4	424770.4	3.67	3.67	3.71	3.74	3.70	0.03
Road 3	342173.8	425017.3	3.43	3.44	3.43	3.49	3.48	0.03
Road 4	342248.9	424902.8	3.60	3.55	3.63	3.65	3.72	0.06
Road 5	342296.4	424830.3	3.68	3.63	3.69	3.72	3.66	0.03

Appendix B : DSM/DTM comparison

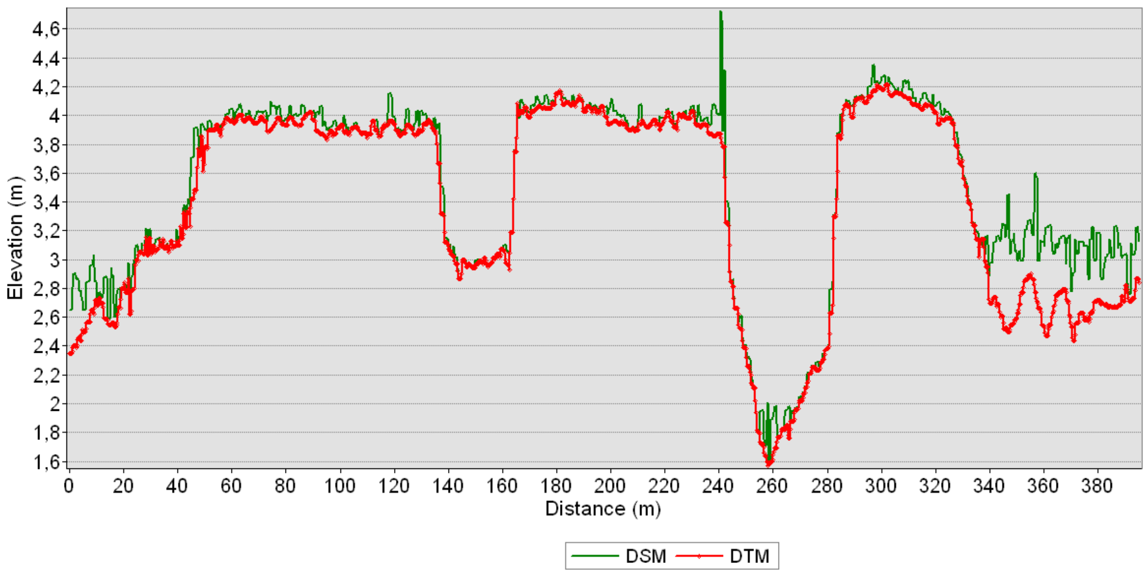
This appendix compares the DSM and DTM outputs at HOMW for 5 years. The DTM was subtracted from the raw DSM for each lidar dataset in order to explore the efficiency of the EA proprietary algorithms at consistently removing the vegetation and detecting the bare ground. Cross-sections are provided where large discrepancies between DSM and DTM are observed.



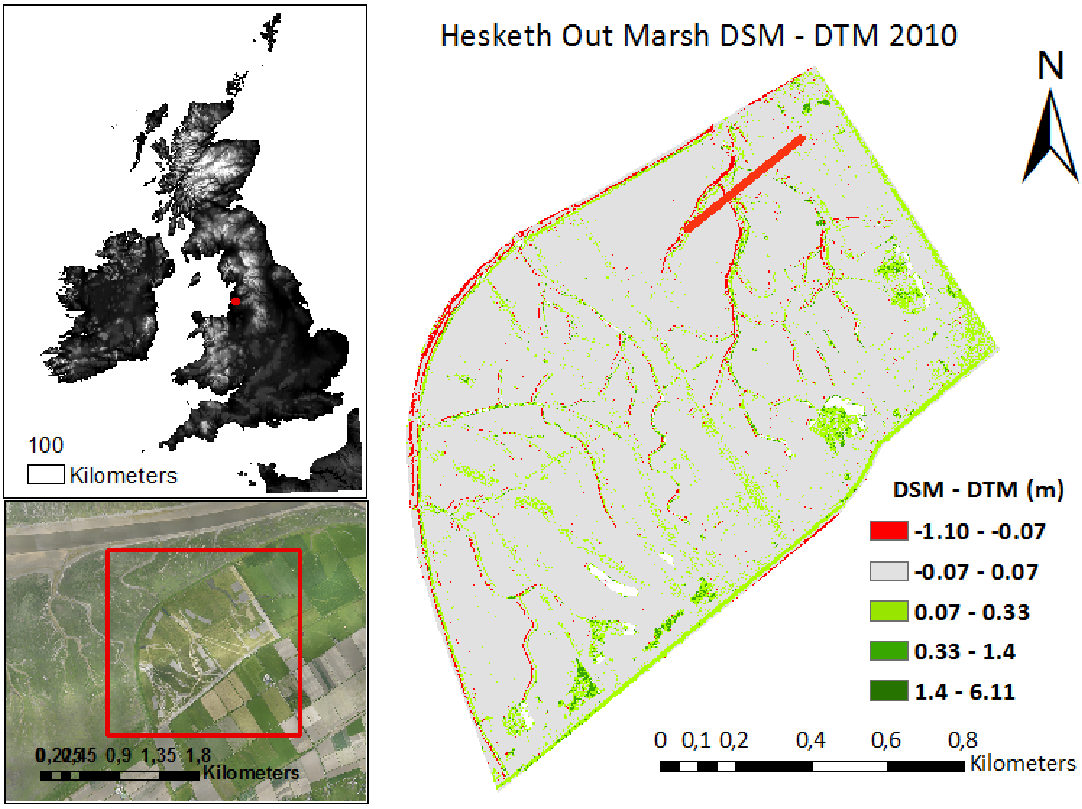
Appendix B1: DSM – DTM substraction for HOMW 2007



Appendix B2: DSM – DTM substraction for HOMW 2009

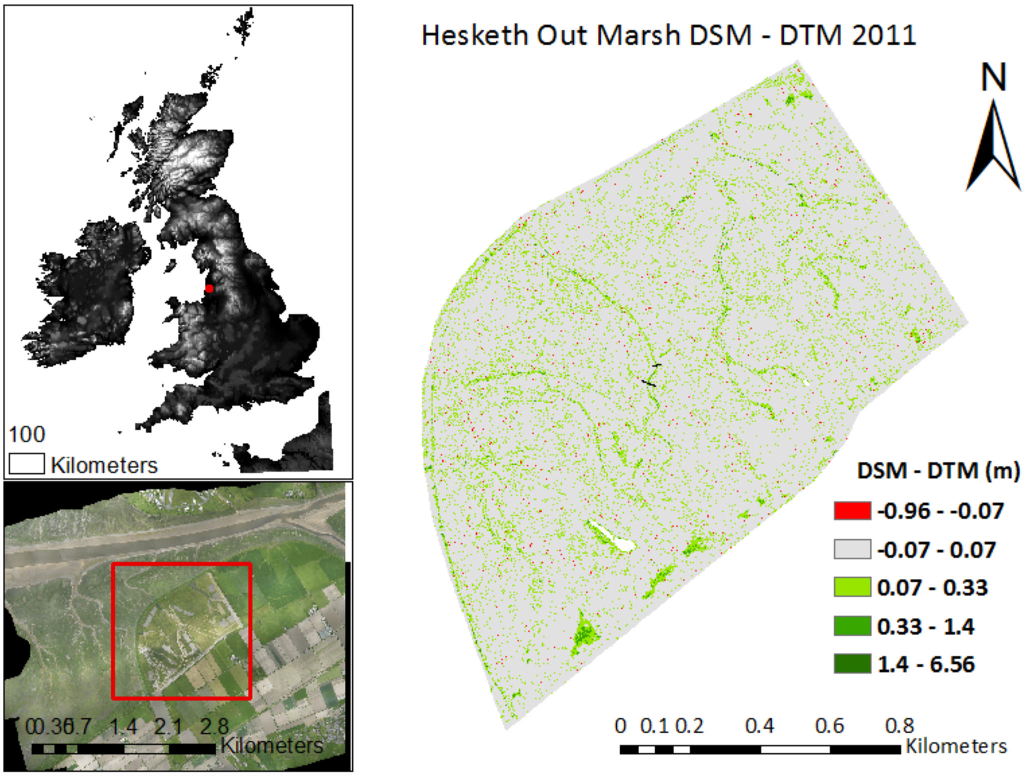


Appendix B3: Cross-section at HOMW 2009 (green: DSM; red: DTM)

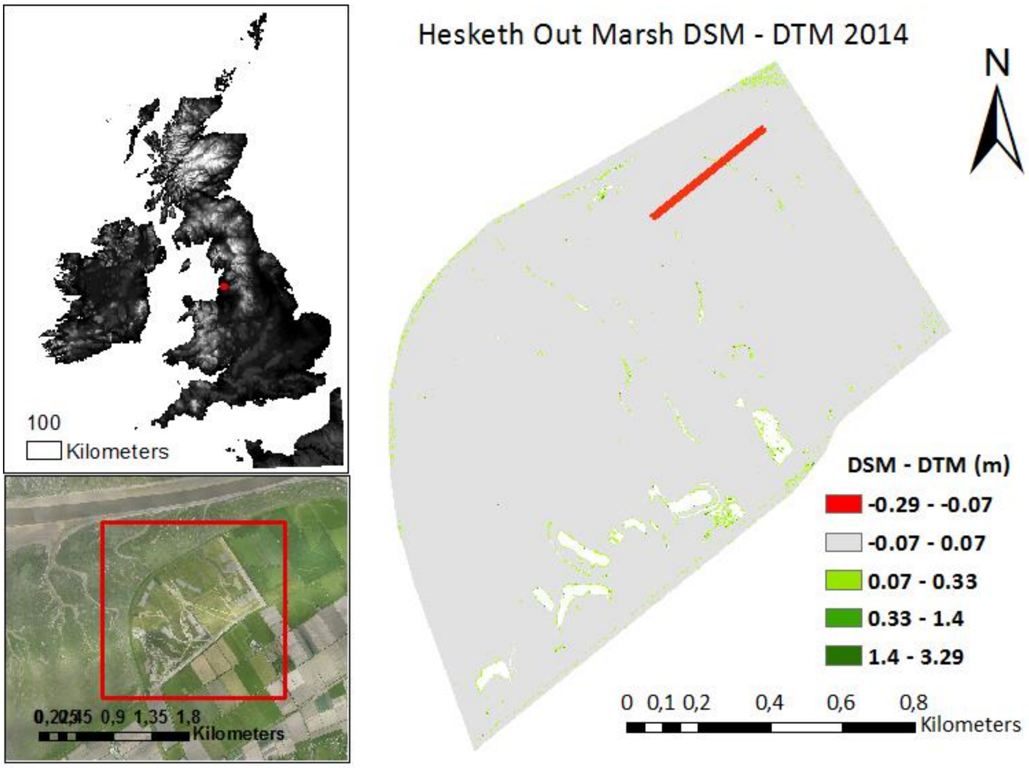


Appendix B4: DSM – DTM substraction for HOMW 2010

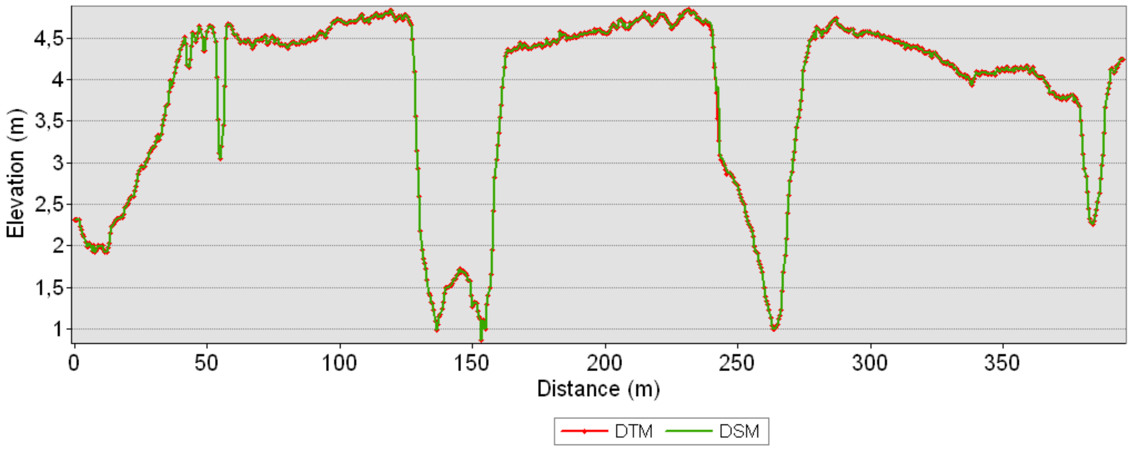
Appendix B5: Cross-section at HOMW 2010 (green: DSM; red: DTM)



Appendix B6: DSM – DTM substraction for HOMW 2011



Appendix B7: DSM – DTM substraction for HOMW 2014



Appendix B8: Cross-section at HOMW 2014 (green: DSM; red: DTM)

Appendix B9: Statistical comparison of DSM and DTM elevation range at HOMW

DSM – DTM (m)	26/05/2007	10/03/2009	05/02/2010	28/01/2011	05/11/2014
Max	8.80	8.53	6.11	6.56	3.29
Min	-0.51	-1.47	-1.1	-0.96	-0.29
mean	0.02	0.05	0.02	0.02	0.00
STD	0.13	0.11	0.08	0.07	0.02

Observations: Most of the corrections occur along and inside the creek network, where the elevation is higher in the DSM. The aim of the EA algorithm is to remove first arrival data corresponding to vegetation or standing water, and to replace it with a point of latter arrival corresponding to the ground. As a result all anomalies in the DSM-DTM dataset should be positive.

The larger positive values (dark green), of up to 1m, correspond to trees. Anomalous negative points are generally isolated and occur randomly in the site, except for the 2010 dataset where they occur almost systematically along the western side of channels, probably due to the reflection of the laser off the slope. According to the standard deviation values, a discrepancy of about 26cm can be expected between the DSM and the DTM.

The amount of correction differs greatly from one year to the other; this is unlikely to be due to seasonal changes in vegetation since saltmarsh vegetation stops growing but does not shrink during winter. Some tiles in the 2007 dataset also seem to have undergone no correction at all. Due to the lack of consistency in the vegetation removal algorithm results, the uncorrected DSM dataset will be used for the rest of the analysis.

Appendix C : MR initial conditions and datasets

This appendix lists the initial conditions, design choices and available lidar datasets for each MR scheme considered. The table contains for each scheme: start date, latitude/longitude, estuary, studied catchment area, height above MWS, HAT, Mean High Water Spring, Mean Low Water Spring, Mean Spring Tidal Range, Mean High Water Neap, Mean Low Water Neap, Mean Neap Tidal Range, Nearby suspended sediment concentration (with references), Time embanked (with references), Sediment type (with references), Policy context, Site aims, Number of breaches (if MR), Initial creek system, Design details, Other references, available lidar data at 1 m resolution (Source: http://environment.data.gov.uk/ds/survey/index.jsp#/survey)

Appendix C1: MR initial conditions and design choices

Project name	Start date	Lat/ long	Estuary	Catchment area (km^2)	Height above MWS (m)	HAT (ODm)	MHWS(ODm)	MLWS (ODm)
Abbots Hall	2002	51.7846 0.8455	Salcott creek, Blackwater estuary	0.2492	1.3907	3.2412	2.6735	-1.435
Alkborough	2006	53.6914 -0.6773	Humber estuary	3.6115	2.333	5.0885	4.1358	-1.69
Wallasea (Allfleet's Marsh)	2006	51.6163 0.8364	Crouch	0.9725	0.9635	3.3739	2.8134	-1.905
Chowder Ness	2006	53.6916 -0.4815	Humber estuary	0.1009	2.3293	4.6212	3.7296	-2.874
Freiston	2002	52.9646 0.0923	The Wash	0.7049	2.5485	4.7062	3.8267	-2.891
Hesketh Out Marsh West	2008	53.7203 -2.8876	Ribble	1.5897	4.2063	5.1234	4.1234	-4.432
Paull Holme Strays	2003	53.7082 -0.2193	Humber estuary	0.8046	2.8617	4.3691	3.4864	-2.994
Steart	2014	51.1983 -3.0506	Parrett River, Severn estuary	2.1	5.513	7.159	5.877	-5.168
Tollesbury	1995	51.7673 0.8402	Blackwater estuary	0.192	0.86	3.249	2.68	-1.641
Welwick	2006	53.6471 0.0095	Humber estuary	0.594	2.428	3.887	3.127	-2.675

Sediment type	**Time embanked**	**Nearby suspended sediment concentration**	**MNTR (m)**	**MLWN (ODm)**	**MHWN (ODm)**	**MSTR (m)**
Sampled marshes at Tollesbury, Northey Island and North Fambridge all have fine sediment, mean size 7-16 micron: 11.5 (Crooks and Pye, 2000)	250 years (+/-50 years, "Reclaimed in the 18th century", Brooks et al., 2015)	Blackwater estuary suspended solids measured at Orplands tidal exchanges: 34-103 mg/l, mean 60-66 mg/l (Emmerson et al., 1997)	3.3067	-1.365	1.941	4.1088
"Coarse to fine silts" Manson & Pinnington 2012, 8 to 62 um, mean grain size 35um?	51 years (+/- 2.5 years, "floodbank […] in the mid-1950s", Edwards & Winn 2006)	1 to 10 g/l (Measured at Blacktoft Jetty, Uncles 2004)	2.936	-0.757	2.1786	5.8262
35-38% clay, 47-52% silt (Kadiri et al., 2011): mean grain size 30um	>400 years (ABPmer 2011)	60mg/l measured at Crecksea (dry weight) (Sheldon, 1968)	3.3136	-1.35	1.9636	4.7186
Estuary near Paull Holme Strays: slightly sandy mud with median grain size 8-28um (Mazik et al., 2010). 10um	97 years (ABPmer 2011)	1 to 10 g/l (Measured at Blacktoft Jetty, Uncles 2004)	3.3081	-1.297	2.0114	6.6037
Sandier than the rest of the Humber. Mean grain size 12.21 um (Rotman, 2008)	19 years (ABPmer 2011)	About 10mg/l within the embayment (higher when storms) (Ni, 2014); 200mg/l general levels in the Wash (Collins et al., 2009)	3.1795	-1.273	1.9062	6.7177
51 um (11-13% clay, 67-75% silt, 25-33% sand according to samples taken on the backmarsh of Banks and Longton Marshes, Steel 1996)	26 years (+/- 2.5 years, Reclaimed in the early 1980s, Tovey et al., 2009)	Order of 100mg/l (Burton, 1995)	5.0048	-2.819	2.1857	8.5552
Mixture of sand and silt/clay: mean grain size 6.4PHI = 11.8um (Swig, 2009)	50 years (Edwards & Winn 2006)	350-650 mg/l (Clapp, 2009)	3.2703	-1.337	1.9337	6.4807
Mean grain size ~10um (Fieldwork at Steart, 2014-2016)	197 years ("The entire estuary has been embanked since at least 1817, Burgess et al., 2013)	4-16g/l in the region of the Parrett Estuary (Darbyshire and West, 1993 in Manning et al., 2010)	5.48	-2.622	2.859	11.04
Sampled marshes at Tollesbury, Northey Island and North Fambridge all have fine sediment, mean size 7-16 micron: 11.5 (Crooks and Pye, 2000)	150 years (Wolters et al., 2005)	Blackwater estuary suspended solids measured at Orplands tidal exchanges: 34-103 mg/l, mean 60-66 mg/l (Emmerson et al., 1997)	3.321	-1.363	1.959	4.32
Clay-silt, median grain size 7-44 um (Andrews 2008); 25um?	35 years (ABPmer 2011)	1 to 10 g/l (Measured at Blacktoft Jetty, Uncles 2004)	2.862	-1.175	1.687	5.802

Number of breaches (if MR)	Sites aims	Policy context
5 breaches, one 100 m wide (1.5 mOD = about MHWN), the others 10-20 m wide. 3-4m wide channels cut through the breaches	49ha of the 85ha scheme is intertidal (ABPmer White Papers 2011)	Strategic MR planning, implemented to create new intertidal habitat and provide natural flood protection
1 strongly reveted breach, 20m with sill set at 2.8 mODN. Defences lowered over 1 500 m	Site intended to create 170ha mudflat and saltmarsh; remaining 230 ha to serve as water storage for extreme surge events (Manson et al., 2012)	Strategic MR planning, with main objective the provision of flood storage space to reduce tidal levels in the upper estuary
6 breaches of between 60 and 210m wide (total width of all breaches = 590m).	Scheme aims to produce 60ha of mudflat, 20ha of saline lagoons and 25ha of saltmarsh (Pendle et al. 2013)	Legislative MR planning: scheme aims to compensate for habitat loss due to port development in the Medway and Fagbury Estuary
Banked realignment	Scheme aims to recreate 0.8 ha of saltmarsh and 10.5 ha of mudflat (ABPmer White Papers 2011)	Legislative MR planning: scheme aims to compensate for port development
This project included three breaches: initially 50m wide with 2m wide x 1m deep channels within. These expanded rapidly	No objective found	Opportunistic MR planning: site was purchased by the Royal Society for the Protection of Birds (RSPB) after the stretch of seawall protecting it was found too expensive to maintain
Four breaches (80, 48, 80 & 40m wide)	Compensation for 13ha of intertidal habitat	Legislative MR planning, aiming to compensate for losses related to flood alleviation work at Morecambe Bay, without increasing the flood risk in the surrounding area (Tovey et al. 2009)
2 breaches (150 and 50m wide)	Expected habitat: Upper saltmarsh 3ha; middle saltmarsh 25ha; lower saltmarsh 15ha; mudflat 32ha (Edwards & Winn 2006)	Opportunistic MR planning: cheaper and more sustainable than raising the previous flood defence, which was insufficient to protect nearby roads and gas infrastructure. Also: compensation of habitats loss within the same estuary during floodwall stabilisation works (Edwards et al. 2006)
1 breach	No objective found	Strategic MR planning: it aims to help meet Habitats Directives, while providing compensation for historic habitat losses and improving flood protection in the peninsula.
1 breach: 60m	No objective found	Experimental site
2 breaches through fronting saltmarsh + defence removal over 1400 m	15-38ha of mudflat and 12-28ha of saltmarsh (Pontee et al. 2006)	Legislative MR planning: scheme aims to compensate for port development

Design details	Initial creek system
Two stage realignment with initial regulated tidal exchange (1996) over 20 ha and open sea all breaching in 2002	Artificial channels of 3–4m wide channels were dug, as well as lagoons that remain flooded at low tide to provide habitats for fish. Design strategy 3.
No leveling of the site was undertaken, leaving the land levels at 2-4 mODN.	Drainage is provided by a single straight distribution channel connected to the breach. Design strategy 3.
Breaches over-wide and sufficiently deep to ensure stability. 550,000m^3 sediment recharge landwards to create 45m-wide strip of saltmarsh; creation of 7 islands within the site; the rest halfway between MLW and MHWN to provide mudflat habitats (ABPmer White Papers 2010)	Existing field drains and borrow dykes left in place - 1 new channel cut. Design strategy 3.
570 m of the old flood defence was removed, leaving only 200 m of the old line of defence	No initial creeks were excavated. Design strategy 1.
In 2000, 1,100m of remnant landward sea wall enforced, new 500m cross wall built. Prior to breaching, vegetation on the 66ha site was cut, baled and removed, field drains were infilled	1,200m of artificial creek system were dug (two channels leading from each breach). Outside the site, 50m of the external primary creek network deepened. Design strategy 3.
-	15km of shallow proto creeks and 11 saline lagoons excavated. Design strategy 2 to 3.
Two breaches: 150 m and 50 m wide; approx. 2,300 m of embankment remains along frontage (Environment Agency, 2012).	Artificial creeks excavated by deepening previous field drains. Design strategy 3.
Ponds were also added to the head of channels to provide feeding grounds for birds and facilitate bird-watching. Mounts were added to the initially flat topography to provide niche habitats.	Fishbone shaped creek network to provide drainage while allowing safe grazing of the site. Design strategy 3.
No site reprofiling was undertaken	One channel cut from a pre-existing drainage ditch, corresponding to creek design strategy 3.
Reprofiling to create a gentle slope from the fronting mudflats to the rear of the site. New defences built to a height of 6.1m ODN	No initial creeks were excavated. Design strategy 1.

Available lidar data at 1m resolution	Other references
2002; 2008; 2013; 2014; 2015	Blott and Pye, 2004; Shepherd, 2007; ABPmer White Papers 2011; Scott et al., 2011; Nikolaou, 2012; Pendle, 2013; Brooks et al., 2015
2007;2010; 2012; 2015	Manson et al., 2012; Costa, 2013; Medlock et al., 2013; Morris, 2013; Pendle et al., 2013; Luisetti et al., 2014; Pontee (in Esteves), 2014; Pontee, 2015
2011; 2013; 2015	ABPmer White Papers 2010; Dixon, 2008; Nikolaou, 2012; Medlock et al., 2013; Pendle, 2013
2007; 2009; 2010; 2011; 2012; 2013; 2015; 2016	ABPmer White Papers 2011; Luisetti, 2014 Costa, 2013; Morris, 2013; Pendle, 2013
2002; 2009; 2011; 2013; 2014	Friess, 2013; Friess, 2012; Frost, 2005; Hampshire, 2011; Nikolaou, 2012; Nottage et al., 2005; Rotman, 2008; Symonds, 2006; Symonds, 2007
2009; 2010; 2011; 2014	Hampshire, 2011; Nikolaou, 2012; Pontee (in Esteves), 2014; Tovey et al., 2009
2007; 2010; 2012; 2013; 2014	Clapp, 2009; Costa, 2013; Edwards et al., 2006; Luisetti, 2014; Mazik, 2010; Morris, 2013; Nikolaou, 2012; Pendle, 2013; Pontee, 2006
2014; 2015; 2016	Burgess et al., 2013; Hampshire, 2011; Townsend, 2014; Wright, 2011
2002; 2009; 2012; 2013; 2014; 2015; 2016	Atkinson, 2004; Frost, 2005; Garbutt et al., 2006; Luisetti, 2014; Nikolaou, 2012; Paramor, 2004; Pendle, 2013; Reading, 2006; Shepherd, 2007; Steel, 1997 Watts et al., 2003
2007; 2009; 2010; 2011; 2012; 2013; 2014	ABPmer White Papers, 2011; Costa, 2013; Frost, 2005; Hampshire, 2011; Luisetti, 2014; Morris, 2013; Nikolaou, 2012; Pontee (in Esteves), 2014; Pontee, 2006

Appendix D : MR creek morphometric parameters

This appendix lists all results from the creek network morphometric analysis performed at 10 MR schemes in the UK, utilising the semi-automated creek parametrisation algorithm for each available lidar dataset.

Appendix D1: Morphological characteristics of the creek network for 10 MR sites

Site	Year	RS order ()	Number of creeks per order (N_i)	Total Length (m)	Mean Length (m)	Bifurcation N_i/N_{i+1}	Sinuosity Ratio ()	Mean junction angle (°)	Mean channel width (m)	Mean channel depth (m)	Mean cross-sectional area (m)	A/D ()	W/D ()
Abbots Hall	2015	5	41	1456.82	35.53	2.28	1.08	93.29	5.00	0.54	1.23	2.29	9.28
	2015	4	18	1082.99	60.17	0.90	1.50	90.52	9.89	0.76	2.59	3.39	12.94
	2015	3	20	2549.70	127.48	2.86	1.16	95.34	8.74	1.24	1.99	1.60	7.03
	2015	2	7	1387.14	198.16	2.33	1.05	83.66	9.52	1.81	5.48	3.03	5.27
	2015	1	3	157.38	52.46		1.14		37.64	2.01	6.43	3.21	18.77
	2014	5	53	1652.97	31.19	3.31	1.06	92.50	5.44	0.43	0.99	2.33	12.76
	2014	4	16	853.68	53.35	1.00	1.14	95.12	9.72	0.74	2.37	3.18	13.07
	2014	3	16	2724.10	170.26	2.67	1.07	92.33	9.46	1.30	2.71	2.08	7.28
	2014	2	6	1323.31	220.55	2.00	1.05	87.05	8.27	1.64	4.50	2.74	5.03
	2014	1	3	153.45	51.15		1.14		36.23	1.23	4.37	3.55	29.45
	2013	5	47	1630.83	34.70	3.92	1.07	89.35	6.30	0.62	1.26	2.03	10.16
	2013	4	12	764.74	63.73	0.57	1.13	90.18	12.18	0.98	4.01	4.11	12.49
	2013	3	21	2857.55	136.07	3.50	1.17	98.81	8.03	1.10	3.08	2.79	7.27
	2013	2	6	1190.91	198.49	2.00	1.04	101.49	13.35	1.63	2.23	1.37	8.19
	2013	1	3	112.53	37.51		1.01		21.10	0.96	6.87	7.16	21.98
	2008	5	50	1411.48	28.23	3.33	1.06	70.86	5.87	0.48	1.07	2.23	12.26
	2008	4	15	1617.89	107.86	1.36	1.10	85.09	8.36	0.53	1.01	1.92	15.85
	2008	3	11	2156.76	196.07	2.75	1.16	89.62	12.83	0.74	1.61	2.17	17.26
	2008	2	4	807.31	201.83	1.33	1.02	97.07	8.26	0.09	0.05	0.62	93.18
	2008	1	3	128.61	42.87		1.05		26.90	0.62	1.18	1.90	43.23
	2002	4	34	1433.29	42.16	3.78	1.07	90.86	5.87	0.95	1.75	1.85	6.20
	2002	3	9	1154.83	128.31	0.75	1.12	77.08	9.16	1.30	2.74	2.10	7.04
	2002	2	12	2485.97	207.16	4.00	1.11	93.65	10.54	1.52	5.03	3.30	6.92
	2002	1	3	606.81	202.27		1.03		7.99	1.27	1.35	1.06	6.28
Alkborough	2015	4	39	1839.48	47.17	3.55	1.05	90.25	6.31	0.42	0.18	0.43	15.17
	2015	3	11	1601.27	145.57	11.00	1.05	91.55	3.65	0.46	0.47	1.03	7.96
	2015	2	1	293.02	293.02	1.00	1.07	92.79	20.26	0.59	4.48	7.65	34.64
	2015	1	1	938.62	938.62		1.18		34.04	1.00	16.16	16.16	34.04
	2012	4	37	1046.84	28.29	4.11	1.05	92.54	7.40	0.48	0.31	0.64	15.50
	2012	3	9	968.79	107.64	3.00	1.08	84.05	6.90	0.34	1.58	4.60	20.11
	2012	2	3	632.68	210.89	3.00	1.06	93.13	6.83	0.22	0.28	1.29	31.77
	2012	1	1	917.00	917.00		1.18		34.48	0.47	11.62	24.72	73.37
	2010	3	19	615.00	32.37	6.33	1.03	90.47	9.97	0.55	0.61	1.10	18.10
	2010	2	3	418.26	139.42	3.00	1.04	108.90	20.57	0.14	0.24	1.66	143.52
	2010	1	1	1023.00	1023.00		1.13		31.93	0.23	1.39	5.94	136.59
	2007	3	25	695.37	27.81	5.00	1.04	69.00	8.61	0.49	0.18	0.36	17.60
	2007	2	5	526.28	105.26	5.00	1.01	81.67	13.60	0.15	0.88	5.90	91.04
	2007	1	1	1729.32	1729.32		1.09		53.24	0.43	11.67	27.01	123.23
Allfleet	2015	4	150	2780.37	18.54	4.55	1.02	89.78	4.25	0.35	0.53	1.51	12.13
	2015	3	33	3077.62	93.26	2.54	1.14	89.40	7.20	0.56	1.93	3.43	12.84
	2015	2	13	3088.48	237.58	3.25	1.13	91.32	11.69	1.26	8.07	6.40	9.27
	2015	1	4	783.17	195.79		1.14		15.19	1.86	15.30	8.24	8.18
	2013	4	129	2747.07	21.30	4.45	1.03	83.75	5.75	0.47	1.30	2.78	12.29
	2013	3	29	2742.12	94.56	2.42	1.09	92.99	6.20	0.45	1.62	3.63	13.86
	2013	2	12	3480.46	290.04	3.00	1.12	94.53	13.46	1.42	12.86	9.09	9.51
	2013	1	4	94.84	23.71		1.05		14.95	1.36	10.11	7.46	11.03
	2011	4	128	3118.22	24.36	4.00	1.04	82.39	4.88	0.43	0.83	1.93	11.41
	2011	3	32	3450.74	107.84	2.91	1.11	89.50	7.06	0.61	2.43	3.99	11.59
	2011	2	11	2958.90	268.99	2.75	1.14	93.14	10.17	1.26	7.27	5.76	8.05
	2011	1	4	1249.21	312.30		1.07		14.90	1.47	13.60	9.24	10.12
	2007	3	34	700.48	20.60	3.40	1.02	90.92	3.23	0.72	1.17	1.63	4.49
	2007	2	10	1295.07	129.51	2.50	1.09	87.17	7.61	1.01	3.49	3.44	7.50
	2007	1	4	550.48	137.62		1.02		9.39	1.37	6.51	4.74	6.84
Chowder Ness	2016	3	46	857.84	18.65	4.18	1.04	89.47	5.15	0.20	0.21	1.04	25.31
	2016	2	11	501.17	45.56	3.67	1.08	90.81	9.82	0.53	0.61	1.15	18.39
	2016	1	3	371.26	123.75		1.13		13.18	0.89	1.66	1.87	14.86
	2015	4	40	510.29	12.76	4.00	1.05	88.18	5.49	0.27	0.10	0.36	19.99
	2015	3	10	348.17	34.82	3.33	1.05	99.31	8.16	0.36	2.10	5.83	22.67
	2015	2	3	289.29	96.43	1.50	1.19	84.21	10.78	0.67	1.94	2.87	16.01
	2015	1	2	221.19	110.60		1.03		23.87	1.63	2.70	1.65	14.64
	2013	3	17	327.89	19.29	3.40	1.04	103.72	4.68	0.39	0.45	1.17	12.08
	2013	2	5	328.68	65.74	1.67	1.08	87.95	5.86	0.41	0.47	1.14	14.21
	2013	1	3	245.61	81.87		1.20		7.10	0.44	0.93	2.11	16.15
	2012	3	39	474.69	12.17	4.33	1.03	89.79	4.77	0.13	0.10	0.76	37.11
	2012	2	9	484.86	53.87	4.50	1.08	91.13	8.05	0.22	0.51	2.35	36.92
	2012	1	2	529.65	264.83		1.15		33.47	0.82	4.43	5.38	40.65
	2011	3	22	324.53	14.75	4.40	1.05	75.18	4.81	0.27	0.35	1.30	17.94
	2011	2	5	569.34	113.87	2.50	1.19	90.00	7.76	0.36	1.71	4.72	21.44
	2011	1	2	183.79	91.90		1.04		16.04	0.51	1.98	3.92	31.77
	2010	3	16	370.81	23.18	3.20	1.08	92.95	4.95	0.24	0.26	1.11	21.04
	2010	2	5	284.68	56.94	2.50	1.09	110.40	15.95	0.53	2.45	4.58	29.84
	2010	1	2	103.02	51.51		1.03		20.61	1.43	3.98	2.79	14.45
	2009	3	20	393.12	19.66	4.00	1.03	84.15	7.12	0.36	0.45	1.25	19.97
	2009	2	5	179.21	35.84	2.50	1.06	95.03	7.55	0.26	0.44	1.68	29.06
	2009	1	2	256.33	128.17		1.16		21.27	1.47	2.41	1.64	14.43
	2007	2	9	312.15	34.68	4.50	1.05	87.61	9.97	0.45	1.52	3.36	21.99
	2007	1	2	266.58	133.29		1.14		26.17	1.36	9.47	6.96	19.24
Fre	2014	4	83	3250.65	39.16	3.77	1.04	97.89	4.68	0.60	1.07	1.80	7.84
	2014	3	22	2731.56	124.16	2.75	1.09	97.81	6.39	0.99	2.50	2.53	6.47

	2014	2	8	1309.71	163.71	2.67	1.07	89.08	17.39	1.60	11.07	6.90	10.84
	2014	1	3	783.91	261.30		1.07		30.05	2.22	35.46	15.95	13.52
	2013	4	89	3084.46	34.66	3.30	1.03	79.55	4.49	0.50	0.92	1.83	8.93
	2013	3	27	3203.29	118.64	2.70	1.08	94.36	7.17	0.92	2.39	2.59	7.76
	2013	2	10	1268.01	126.80	3.33	1.08	93.77	11.13	1.47	7.02	4.78	7.57
	2013	1	3	829.90	276.63		1.07		28.81	1.82	27.56	15.12	15.81
	2011	4	89	3352.75	37.67	3.71	1.03	85.38	4.92	0.59	1.18	1.98	8.28
	2011	3	24	3058.01	127.42	2.67	1.12	100.99	6.15	0.91	2.33	2.57	6.77
	2011	2	9	1327.82	147.54	3.00	1.07	97.92	8.85	1.38	5.47	3.96	6.40
	2011	1	3	617.92	205.97		1.07		27.91	2.08	29.97	14.38	13.40
	2009	4	73	2361.08	32.34	3.65	1.03	103.02	4.34	0.42	0.48	1.15	10.37
	2009	3	20	2842.54	142.13	4.00	1.07	97.29	6.54	0.99	2.50	2.52	6.60
	2009	2	5	559.03	111.81	1.67	1.03	87.46	11.63	1.62	5.78	3.57	7.19
	2009	1	3	759.09	253.03		1.06		27.92	2.02	25.13	12.44	13.82
	2002	4	56	1754.03	31.32	5.09	1.03	86.25	5.86	0.28	0.37	1.31	20.72
	2002	3	11	1541.23	140.11	2.20	1.06	83.52	7.29	0.40	1.00	2.52	18.43
	2002	2	5	1507.08	301.42	2.50	1.08	97.78	21.53	0.72	5.38	7.48	29.93
	2002	1	2	72.84	36.42		1.04		14.84	0.62	4.49	7.19	23.77
Hesketh Out Marsh W	2014	5	318	5844.64	18.38	4.82	1.03	72.10	4.59	0.56	0.85	1.53	8.21
	2014	4	66	6788.97	102.86	3.88	1.09	80.40	7.27	0.87	2.63	3.04	8.40
	2014	3	17	3710.87	218.29	1.89	1.11	86.05	12.06	1.49	8.59	5.76	8.07
	2014	2	9	2109.88	234.43	9.00	1.11	88.74	22.07	1.94	22.35	11.53	11.39
	2014	1	1	237.15	237.15		1.19		113.64	3.64	162.18	44.53	31.20
	2011	5	179	5863.73	32.76	3.89	1.04	72.54	7.76	0.44	1.43	3.22	17.48
	2011	4	46	5817.19	126.46	4.18	1.10	80.46	10.37	0.54	2.43	4.50	19.21
	2011	3	11	3267.11	297.01	5.50	1.13	84.64	16.61	1.10	8.88	8.09	15.14
	2011	2	2	1220.75	610.37	2.00	1.17	84.68	20.24	1.83	19.89	10.90	11.09
	2011	1	1	241.15	241.15		1.18		100.66	3.51	175.09	49.93	28.70
	2010	4	115	4779.30	41.56	3.83	1.04	62.35	9.50	0.45	1.82	4.07	21.24
	2010	3	30	4917.96	163.93	3.33	1.11	78.90	13.23	0.59	3.69	6.26	22.44
	2010	2	9	4677.57	519.73	9.00	1.08	92.86	27.13	1.40	17.68	12.61	19.35
	2010	1	1	220.76	220.76		1.15		46.24	3.27	76.44	23.39	14.15
	2009	4	113	4449.55	39.38	3.77	1.04	62.53	9.28	0.52	2.24	4.34	17.95
	2009	3	30	6042.06	201.40	4.29	1.10	78.38	13.60	0.73	5.32	7.24	18.53
	2009	2	7	3155.94	450.85	7.00	1.09	95.53	24.97	1.09	16.64	15.24	22.86
	2009	1	1	242.15	242.15		1.13		79.31	2.16	107.44	49.77	36.74
Paull Holme Strays	2014	4	135	3813.54	28.25	3.86	1.03	90.93	5.18	0.30	0.27	0.91	17.32
	2014	3	35	5107.24	145.92	2.50	1.11	89.63	10.88	0.63	3.40	5.37	17.21
	2014	2	14	2099.30	149.95	4.67	1.15	92.38	14.00	0.99	5.87	5.92	14.13
	2014	1	3	1343.51	447.84		1.28		18.11	0.92	10.22	11.14	19.76
	2013	4	125	3834.00	30.67	3.47	1.04	87.98	5.64	0.31	0.55	1.80	18.39
	2013	3	36	6577.34	182.70	4.00	1.09	76.53	9.30	0.51	2.01	3.90	18.08
	2013	2	9	1668.24	185.36	3.00	1.09	85.39	13.26	1.07	5.55	5.19	12.40
	2013	1	3	905.10	301.70		1.10		20.17	0.83	7.29	8.81	24.40
	2012	4	132	3816.91	28.92	4.00	1.03	77.81	5.56	0.29	0.61	2.11	19.06
	2012	3	33	4300.43	130.32	2.20	1.07	84.99	8.01	0.48	2.03	4.26	16.79
	2012	2	15	2707.65	180.51	3.75	1.35	88.59	14.28	0.86	3.29	3.83	16.65
	2012	1	4	1032.82	258.20		1.15		35.91	0.92	9.54	10.33	38.86
	2010	4	79	2891.26	36.60	4.39	1.04	66.87	7.12	0.25	0.63	2.50	28.21
	2010	3	18	4123.08	229.06	3.60	1.16	74.32	7.46	0.50	1.58	3.15	14.85
	2010	2	5	1158.33	231.67	1.25	1.03	85.29	10.96	1.07	5.47	5.12	10.24
	2010	1	4	878.48	219.62		1.09		25.82	0.58	6.78	11.63	44.28
	2007	4	68	2104.32	30.95	3.58	1.04	82.10	5.97	0.25	0.55	2.17	23.68
	2007	3	19	3413.89	179.68	2.38	1.06	73.62	9.45	0.71	2.79	3.93	13.32
	2007	2	8	1866.09	233.26	8.00	1.14	77.77	31.30	0.84	5.81	6.94	37.38
	2007	1	1	529.97	529.97		1.12		28.46	0.87	12.92	14.87	32.75
Steart	2016	4	49	4272.66	87.20	4.90	1.06	94.91	13.08	0.69	3.89	5.68	19.07
	2016	3	10	2746.90	274.69	5.00	1.09	79.06	22.26	1.26	13.31	10.57	17.68
	2016	2	2	2084.64	1042.32	2.00	1.07	101.90	54.17	1.23	39.94	32.56	44.16
	2016	1	1	184.00	184.00		1.00		101.00	5.99	222.89	37.24	16.87
	2015	4	40	3505.05	87.63	3.64	1.03	83.96	12.28	0.85	6.18	7.26	14.44
	2015	3	11	2978.96	270.81	5.50	1.09	63.77	24.06	1.44	17.89	12.41	16.69
	2015	2	2	2040.56	1020.28	2.00	1.06	89.25	70.58	1.78	51.58	28.98	39.65
	2015	1	1	245.57	245.57		1.05		101.02	5.97	204.27	34.22	16.92
	2014	4	54	3429.59	63.51	3.18	1.03	89.85	11.00	0.91	6.06	6.69	12.14
	2014	3	17	3069.72	180.57	3.40	1.07	78.38	18.09	0.90	12.94	14.45	20.20
	2014	2	5	2598.27	519.65	5.00	1.05	101.16	32.09	1.33	12.33	9.29	24.17
	2014	1	1	222.77	222.77		1.04		100.58	1.96	115.32	58.84	51.32
Tollesbury	2016	4	73	1306.44	17.90	3.48	1.03	100.32	5.06	0.61	0.69	1.13	8.33
	2016	3	21	964.03	45.91	3.50	1.03	87.50	5.49	0.92	0.90	0.98	5.96
	2016	2	6	1002.32	167.05	6.00	1.15	88.21	17.76	2.07	2.69	1.30	8.56
	2016	1	1	66.67	66.67		1.07		27.31	2.81	1.41	0.50	9.71
	2015	4	55	1194.91	21.73	4.23	1.04	67.54	5.06	0.63	0.61	0.96	7.98
	2015	3	13	883.32	67.95	1.86	1.07	100.99	5.07	1.23	1.48	1.20	4.11
	2015	2	7	719.39	102.77	7.00	1.09	79.10	10.88	1.76	3.54	2.02	6.20
	2015	1	1	81.94	81.94		1.03		30.87	2.62	11.71	4.46	11.77
	2014	4	44	788.21	17.91	3.67	1.04	85.86	4.34	0.65	0.54	0.82	6.64
	2014	3	12	884.32	73.69	1.71	1.04	103.53	5.03	1.07	1.05	0.98	4.69
	2014	2	7	908.85	129.84	7.00	1.14	94.22	13.25	1.97	3.02	1.54	6.74
	2014	1	1	97.25	97.25		1.03		34.50	2.49	11.84	4.76	13.88
	2013	4	41	1111.15	27.10	3.73	1.03	83.08	3.78	1.03	0.68	0.66	3.68
	2013	3	11	827.03	75.18	1.57	1.07	118.77	5.98	0.74	1.60	2.15	8.04
	2013	2	7	543.81	77.69	7.00	1.06	89.06	14.56	1.85	3.99	2.16	7.89
	2013	1	1	68.08	68.08		1.09		47.51	2.50	5.92	2.36	18.97
	2012	4	42	1049.07	24.98	3.23	1.03	93.60	4.45	0.69	0.56	0.81	6.41
	2012	3	13	527.20	40.55	2.17	1.05	85.01	6.56	1.08	1.91	1.77	6.07
	2012	2	6	927.38	154.56	6.00	1.10	101.22	11.75	1.85	3.45	1.87	6.36
	2012	1	1	45.70	45.70		1.08		31.30	1.99	7.68	3.85	15.71
	2009	4	34	874.25	25.71	3.40	1.05	102.73	4.43	0.77	0.91	1.19	5.78
	2009	3	10	1125.77	112.58	2.00	1.09	85.55	6.88	1.25	1.93	1.55	5.52
	2009	2	5	343.50	68.70	5.00	1.08	101.25	17.20	1.67	4.73	2.83	10.28
	2009	1	1	13.66	13.66		1.04		8.94	1.85	7.23	3.91	4.84
	2002	4	34	925.70	27.23	4.25	1.04	79.97	4.57	0.44	0.85	1.96	10.48
	2002	3	8	781.41	97.68	1.60	1.06	103.44	7.15	0.78	1.53	1.96	9.15
	2002	2	5	481.66	96.33	2.50	1.12	95.59	9.10	0.99	1.85	1.86	9.15
	2002	1	2	75.87	37.94		1.26		25.42	1.69	3.01	1.78	15.01
Welwick	2014	4	75	1451.95	19.36	4.17	1.03	96.26	9.31	0.22	0.27	1.23	42.69
	2014	3	18	1374.62	76.37	4.50	1.10	89.71	10.56	0.43	0.96	2.22	24.36
	2014	2	4	882.20	220.55	2.00	1.15	81.23	19.52	0.53	4.81	9.08	36.83
	2014	1	2	204.55	102.28		1.27		61.90	0.47	0.54	1.16	133.13
	2013	4	76	1920.45	25.27	5.07	1.05	78.97	7.57	0.23	0.34	1.50	33.42
	2013	3	15	1427.69	95.18	3.75	1.13	97.13	7.84	0.44	0.71	1.63	17.90
	2013	2	4	484.11	121.03	2.00	1.07	116.14	23.50	0.70	2.83	4.02	33.46
	2013	1	2	172.34	86.17		1.15		60.11	0.41	2.52	6.22	148.43
	2012	4	62	1436.26	23.17	4.43	1.04	89.15	9.78	0.21	0.19	0.87	45.63

	2012	3	14	967.76	69.13	2.80	1.09	82.37	6.45	0.31	0.51	1.62	20.63
	2012	2	5	879.72	175.94	2.50	1.12	110.87	19.81	0.58	2.10	3.62	34.16
	2012	1	2	213.00	106.50		1.00		26.83	0.35	1.24	3.53	76.67
	2011	4	49	1053.84	21.51	4.08	1.04	93.43	7.50	0.16	0.39	2.38	45.57
	2011	3	12	1023.44	85.29	3.00	1.11	93.21	7.41	0.25	0.58	2.28	29.14
	2011	2	4	697.74	174.44	2.00	1.11	103.92	15.60	0.38	2.17	5.72	41.05
	2011	1	2	141.54	70.77		1.10		34.89	0.37	0.97	2.64	95.58
	2010	4	34	710.49	20.90	3.09	1.02	96.08	5.98	0.19	0.23	1.20	31.52
	2010	3	11	707.24	64.29	2.75	1.10	105.10	6.98	0.16	0.22	1.38	42.83
	2010	2	4	551.90	137.98	2.00	1.18	82.85	22.53	0.18	1.56	8.62	124.32
	2010	1	2	83.00	41.50		1.06		38.50	0.12	0.20	1.60	314.29
	2009	4	36	885.41	24.59	3.60	1.05	97.01	6.29	0.17	0.20	1.17	36.10
	2009	3	10	636.54	63.65	2.50	1.11	91.10	9.61	0.24	0.62	2.54	39.69
	2009	2	4	406.50	101.63	2.00	1.13	89.77	29.74	0.31	2.35	7.53	95.16
	2009	1	2	91.00	45.50		1.04		10.00	0.06	0.16	2.75	166.67
	2007	3	27	631.69	23.40	3.38	1.06	95.76	8.86	0.11	0.12	1.07	77.79
	2007	2	8	345.95	43.24	4.00	1.06	99.45	16.38	0.17	0.44	2.56	94.59
	2007	1	2	224.00	112.00		1.06		29.07	0.30	1.04	3.45	96.57

Appendix D2: Morphological characteristics of the creek network for each MR scheme and each available year (continued)

Site	Chowder Ness		Allfleet				Alkborough				Abbots Hall				
Year	2015	2016	2007	2011	2013	2015	2007	2010	2012	2015	2002	2008	2013	2014	2015
Drainage density (km/km^2)	13.57	17.19	2.62	11.03	9.29	9.99	0.82	0.57	1.02	1.30	22.80	23.96	27.64	27.89	27.65
Total Channel Length (m)	1368.94	1730.26	2546.03	10777.06	9064.50	9729.64	2950.97	2056.26	3565.31	4672.39	5680.90	6122.05	6556.56	6707.51	6634.02
Catchment area (km^2)	0.10	0.10	0.97	0.98	0.98	0.97	3.61	3.61	3.50	3.59	0.25	0.26	0.24	0.24	0.24
Number of creeks (n)	55	60	48	175	174	200	31	23	50	52	58	83	89	94	89
Main channel length (largest breach) (m)	148.72	156.13	374.63	642.41	625.13	642.03	1848.91	1491.19	1460.56	1772.81	534.47	575.91	575.47	560.31	573.47
Main channel length (all breaches) (m)	423.19	480.28	857.19	1962.53	1941.13	1994.03	1848.91	1491.19	1460.56	1772.81	534.47	575.91	575.47	560.31	573.47
Main channel sinuosity ratio ()	1.29	1.36	1.40	1.60	1.55	1.63	1.08	1.12	1.11	1.09	1.55	1.54	1.51	1.48	1.55
Main channel gradient (%)	1.88	1.99	-1.30	-1.32	-1.18	1.63	3.09	3.46	3.16	3.24	-0.36	0.65	0.44	0.27	0.10
Tidal Prism (m^3)	1938.64	1104.54	8926.56	49468.10	53767.35	44302.13	20762.25	1891.30	12679.29	17565.59	18998.84	6812.05	17343.61	17662.99	18293.67
Mouth cross sectional area (largest breach) (m^2)	2.89	2.24	84.36	85.23	87.05	72.24	16.26	20.11	22.10	30.01	16.03	19.08	24.26	27.15	24.53
Mouth cross sectional area (sum of all breaches) (m^2)	4.16	4.16	179.47	214.16	222.99	199.25	16.26	20.11	22.10	30.01	16.03	19.08	24.26	27.15	24.53
Mouth creek depth (m)	0.83	0.70	2.15	2.47	2.63	2.77	0.90	1.14	1.21	2.29	0.45	0.41	0.64	0.75	0.95
Mouth creek width (m)	10.63	7.81	44.18	48.17	50.16	43.10	38.01	38.01	41.62	34.41	77.10	83.49	90.55	91.97	84.90
Mean elevation above MWS (m)	3.18	3.26	0.96	0.99	1.06	1.17	2.33	2.41	2.58	2.67	1.39	1.34	1.49	1.43	1.47
Overmarsh Path Length (m)	33.06	30.10	116.36	54.68	58.45	56.15	774.16	796.28	768.72	540.24	26.35	22.26	16.35	17.01	16.89
Mean elevation under HAT (m)	3.61	3.69	1.42	1.44	1.51	1.62	3.56	3.63	3.80	3.89	2.01	1.95	2.11	2.05	2.09
Mean slope under HAT (%)	2.77	2.44	4.05	3.84	3.74	3.25	2.69	3.16	3.26	2.65	6.83	4.65	5.01	5.06	5.15
Cth (%)	2.92	2.59	8.56	8.35	8.25	3.25	4.02	4.02	4.02	4.02	5.58	3.40	3.76	3.81	3.90
HZth (m)	3.46	3.54	0.96	0.98	1.05	1.16	3.21	3.28	3.46	3.54	1.57	1.52	1.68	1.62	1.65
LZth (m)			0.17	0.19	0.26	0.37	2.88	2.95	3.13	3.21	0.96	0.90	1.06	1.00	1.03
Planform area (m^2)	11530	13215	72184	79968	77820	76597	64561	57116	47185	48327	42026	53701	55541	52351	50368

Tollesbury	Steart			Paull Holme Strays					Hesketh Out Marsh West				Freiston										
2016	2014	2015	2016	2007	2010	2012	2013	2014	2009	2010	2011	2014	2002	2009	2011	2013	2014	2007	2009	2010	2011	2012	2013
17.35	4.44	3.96	4.41	9.84	11.18	14.73	16.16	15.39	8.74	9.19	10.35	12.05	6.92	9.60	11.84	11.89	11.44	5.73	8.20	7.50	10.68	15.06	8.95
3339.46	9320.36	8524.57	9288.20	7914.28	9051.15	11857.80	12984.68	12363.58	13889.71	14595.60	16409.92	19117.22	4875.18	6771.85	8356.50	8385.65	8075.82	578.73	828.66	758.51	1077.67	1489.20	902.17
0.19	2.10	2.15	2.10	0.80	0.81	0.81	0.80	0.80	1.59	1.59	1.59	1.59	0.70	0.71	0.71	0.71	0.71	0.10	0.10	0.10	0.10	0.10	0.10
101	77	53	62	96	106	184	173	187	151	155	239	411	74	101	125	129	116	11	27	23	29	50	25
580.00	2527.66	2542.38	2543.72	873.56	888.47	932.03	1072.53	1070.53	1120.34	1106.28	1230.19	1253.69	408.13	597.94	564.50	606.41	568.88	273.47	256.47	239.34	187.03	143.75	147.41
580.00	2527.66	2542.38	2543.72	1200.84	1329.31	1501.34	1874.44	1860.31	3760.72	3867.94	4224.31	3957.25	1128.22	1456.13	1568.88	1721.31	1668.63	273.47	317.47	434.94	436.13	466.06	439.19
1.23	1.09	1.09	1.09	1.48	1.42	1.43	1.40	1.40	1.30	1.30	1.37	1.41	1.11	1.23	1.25	1.23	1.21	1.23	1.25	1.12	1.37	1.31	1.32
1.69	5.51	4.23	5.42	0.57	0.91	0.91	0.92	1.06	2.19	1.19	0.93	0.91	3.14	3.16	2.84	3.00	2.67	1.06	1.45	2.29	1.45	2.25	1.72
4556.24	118255.97	230352.69	177489.53	28345.96	20654.78	29850.61	31181.94	44428.62	120644.17	126437.47	118021.35	140373.80	10619.64	30541.19	36859.57	42270.93	52622.06	2999.48	872.32	1204.11	1449.95	2641.08	530.71
68.15	240.26	248.77	291.99	152.90	145.60	163.24	146.49	137.27	122.09	121.45	140.32	152.67	11.52	115.63	110.75	126.79	128.45	9.22	9.74	9.03	3.65	12.96	2.81
68.15	240.26	248.77	291.99	170.22	167.84	186.81	169.34	158.87	211.38	228.83	253.44	273.40	35.03	190.30	188.51	199.48	209.26	9.22	9.74	9.11	3.95	13.78	3.63
3.39	6.40	7.14	7.26	1.99	2.27	2.58	2.61	2.56	2.10	3.20	3.32	3.43	0.85	3.87	3.68	4.23	4.53	1.45	1.26	1.11	0.91	0.88	0.86
45.34	101.02	99.13	99.13	149.31	140.77	149.31	137.62	136.69	98.11	98.11	81.99	81.12	26.42	41.59	42.94	44.29	44.29	19.21	27.02	24.21	7.81	56.22	3.61
1.25	5.51	5.56	5.51	2.86	2.82	2.94	3.01	3.08	4.21	4.24	4.31	4.42	2.55	2.54	2.60	2.62	2.64	2.33	2.56	2.64	2.76	3.04	2.87
37.40	84.64	85.15	82.40	57.33	58.36	29.02	24.92	25.24	53.23	46.10	36.28	43.51	110.09	57.34	50.32	49.91	43.37	101.22	106.41	69.59	50.50	29.60	40.08
1.77	5.87	5.91	5.86	3.11	3.06	3.18	3.26	3.33	4.05	4.09	4.16	4.27	3.02	3.01	3.07	3.08	3.11	2.76	2.99	3.07	3.19	3.46	3.30
3.40	2.84	2.69	1.71	2.96	2.32	3.01	3.22	3.28	4.77	2.16	2.50	4.44	3.28	3.28	3.42	4.35	4.82	2.87	2.91	2.84	3.15	2.19	3.00
6.42	8.19	8.04	7.06			3.65	3.87	3.80	11.74	9.13	9.47	9.10	3.45	3.45	3.59	4.52	4.99	3.02	3.07	2.99	3.30	2.35	3.15
1.67	5.80	5.85	5.80	2.70	2.68	2.80	2.88	2.95	4.03	4.06	4.13	4.24	2.97	2.97	3.02	3.04	3.07	2.61	2.84	2.92	3.04	3.31	3.15
1.25	5.28	5.33	5.28	2.00	1.98	2.10	2.18	2.25	3.33	3.37	3.44	3.55	2.55	2.55	2.60	2.62	2.65	0.00	0.00	0.00	0.00	0.00	0.00
24787	253896	252646	251422	84001	69716	95979	100495	106740	184429	183394	185100	173500	43137	45000	57477	62549	60001	5859	7197	7059	5459	11790	4559

Welwick												
2007	2009	2010	2011	2012	2013	2014	2002	2009	2012	2013	2014	2015
2.21	3.72	3.77	5.37	6.44	7.38	7.21	11.82	12.24	13.25	13.27	13.93	14.98
1201.64	2019.45	2052.64	2916.56	3496.74	4004.59	3913.32	2264.64	2357.18	2549.35	2550.08	2678.63	2879.57
0.54	0.54	0.54	0.54	0.54	0.54	0.54	0.19	0.19	0.19	0.19	0.19	0.19
37	52	51	67	83	97	99	49	50	62	60	64	76
234.88	559.59	566.19	737.69	738.66	771.94	778.03	660.91	556.03	555.38	549.03	574.16	571.38
404.91	753.34	833.84	1006.34	1034.25	1090.69	1095.88	660.91	556.03	555.38	549.03	574.16	571.38
1.13	1.55	1.56	1.65	1.57	1.82	1.83	1.31	1.15	1.14	1.14	1.13	1.13
1.81	2.27	2.36	2.49	2.47	2.60	2.62	0.97	-0.18	1.32	1.42	1.33	1.40
463.15	1544.65	1198.99	2658.83	2866.25	3474.49	6066.43	3109.73	4688.62	5153.96	4645.56	5254.51	5542.62
96.85	130.27	123.24	124.94	134.92	127.17	124.59	34.12	49.79	64.29	63.02	65.34	64.32
176.63	210.99	161.30	198.59	231.92	200.25	195.70	34.12	49.79	64.29	63.02	65.34	64.32
0.95	1.15	1.11	1.11	1.17	1.24	1.13	2.01	3.05	3.27	3.32	3.30	3.39
144.81	147.96	147.96	152.09	155.50	158.40	159.36	41.11	42.52	45.34	45.34	45.34	45.34
2.43	2.43	2.44	2.48	2.44	2.52	2.60	0.86	0.99	1.10	1.21	1.17	1.22
261.75	201.62	183.34	126.73	122.50	119.42	116.00	53.07	56.25	49.16	50.06	46.91	46.80
2.65	2.65	2.66	2.70	2.67	2.75	2.83	1.38	1.51	1.62	1.73	1.69	1.74
1.82	1.94	1.87	2.03	2.08	2.17	2.34	3.94	3.35	3.56	3.67	3.72	3.69
3.44	3.56	3.50	3.65	3.71	3.80	3.97	6.97	6.38	6.58	6.69	6.75	6.72
2.44	2.44	2.45	2.49	2.46	2.54	2.62	1.28	1.41	1.52	1.63	1.59	1.64
1.97	1.97	1.98	2.02	1.99	2.07	2.15	0.86	0.99	1.10	1.21	1.17	1.22
17782	22331	19477	27531	34443	36970	38679	16602	18240	19223	19098	19838	20018

Appendix E : MR and AR evolution rates fits and R^2 values

This appendix gives the equations, R^2 and p-values for the best linear and exponential fits of the evolution of creek morphological parameters over the years following breaching for 10 MR sites and 2 AR sites.

Appendix E1: Equations, R^2 and p-values for the best linear and exponential fits of the evolution of creek morphological parameters over the years following breaching for 10 MR sites and 2 AR sites.

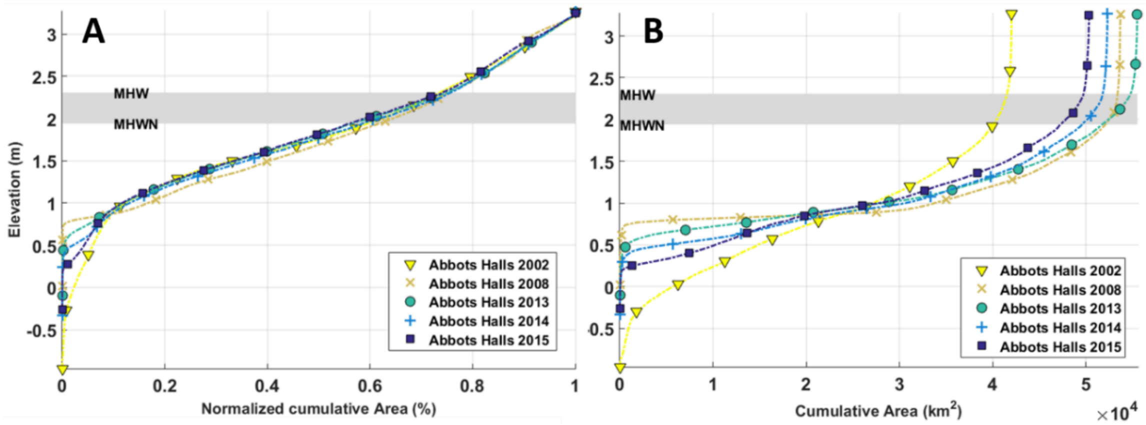
	Mean elevation over MWS vs time linear fit	Mean elevation over MWS vs time exp fit	Drainage density vs time linear fit	Drainage density vs time exp fit	Overmarsh path length vs time linear fit	Overmarsh path length vs time exp fit
Abbots Hall	Y = 0.01 * X + 1.98 - R^2=0.45, pval=0.21	Y = 1.98 * EXP(<0.01 * X) ; R^2=0.45	Y = 0.43 * X + 22.37 - R^2=0.93, pval=0.01	Y = 22.40 * EXP(0.02 * X) ; R^2=0.94	Y = -0.80 * X + 26.46 - R^2=0.96, pval<0.01	Y = 26.64 * EXP(-0.04 * X) ; R^2=0.96
Alkborough	Y = 0.04 * X + 3.50 - R^2=0.94, pval=0.03	Y = 3.50 * EXP(0.01 * X) ; R^2=0.94	Y = 0.07 * X + 0.58 - R^2=0.58, pval=0.24	Y = 0.58 * EXP(0.09 * X) ; R^2=0.65	Y = -28.33 * X + 861.50 - R^2=0.63, pval=0.21	Y = 859.39 * EXP(-0.04 * X) ; R^2=0.58
Allfleet	Y = 0.02 * X + 1.36 - R^2=0.80, pval=0.10	Y = 1.36 * EXP(0.02 * X) ; R^2=0.82	Y = 0.90 * X + 3.27 - R^2=0.65, pval=0.19	Y = 4.69 * EXP(0.10 * X) ; R^2=0.55	Y = -7.62 * X + 113.33 - R^2=0.75, pval=0.13	Y = 124.75 * EXP(-0.11 * X) ; R^2=0.85
Chowder Ness	Y = 0.10 * X + 2.68 - R^2=0.95, pval<0.01	Y = 2.72 * EXP(0.03 * X) ; R^2=0.94	Y = 1.13 * X + 4.52 - R^2=0.72, pval=0.01	Y = 5.85 * EXP(0.10 * X) ; R^2=0.71	Y = -9.12 * X + 108.88 - R^2=0.77, pval<0.01	Y = 130.99 * EXP(-0.16 * X) ; R^2=0.82
Freiston	Y = 0.01 * X + 3.00 - R^2=0.68, pval=0.09	Y = 3.00 * EXP(<0.01 * X) ; R^2=0.68	Y = 0.43 * X + 7.01 - R^2=0.91, pval=0.01	Y = 7.23 * EXP(0.04 * X) ; R^2=0.89	Y = -5.53 * X + 105.36 - R^2=0.94, pval=0.01	Y = 108.81 * EXP(-0.08 * X) ; R^2=0.98
Hesketh Out Marsh West	Y = 0.04 * X + 4.01 - R^2=0.99, pval=0.01	Y = 4.01 * EXP(0.01 * X) ; R^2=0.99	Y = 0.68 * X + 8.05 - R^2=0.98, pval=0.01	Y = 8.26 * EXP(0.06 * X) ; R^2=0.98	Y = -1.57 * X + 49.50 - R^2=0.24, pval=0.51	Y = 50.27 * EXP(-0.04 * X) ; R^2=0.26
Paull Holme Strays	Y = 0.03 * X + 2.92 - R^2=0.70, pval=0.08	Y = 2.93 * EXP(0.01 * X) ; R^2=0.71	Y = 0.95 * X + 5.67 - R^2=0.90, pval=0.01	Y = 7.18 * EXP(0.07 * X) ; R^2=0.90	Y = -5.59 * X + 84.79 - R^2=0.80, pval=0.04	Y = 101.33 * EXP(-0.12 * X) ; R^2=0.75
Steart	Y = <-0.01 * X + 5.88 - R^2<0.01, pval=0.96	Y = 5.88 * EXP(<-0.01 * X) ; R^2=0.00	Y = -0.01 * X + 4.28 - R^2<0.01, pval=0.97	Y = 4.28 * EXP(<-0.00 * X) ; R^2<0.01	Y = -1.12 * X + 85.19 - R^2=0.59, pval=0.44	Y = 85.18 * EXP(-0.01 * X) ; R^2=0.58
Tollesbury	Y = 0.03 * X + 1.15 - R^2=0.95, pval<0.01	Y = 1.19 * EXP(0.02 * X) ; R^2=0.96	Y = 0.31 * X + 8.77 - R^2=0.61, pval=0.04	Y = 9.03 * EXP(0.03 * X) ; R^2=0.66	Y = -0.87 * X + 62.96 - R^2=0.49, pval=0.08	Y = 63.31 * EXP(-0.02 * X) ; R^2=0.46
Welwick	Y = 0.02 * X + 2.59 - R^2=0.67, pval=0.02	Y = 2.59 * EXP(0.01 * X) ; R^2=0.68	Y = 0.80 * X + 1.26 - R^2=0.95, pval<0.01	Y = 2.30 * EXP(0.16 * X) ; R^2=0.92	Y = -21.88 * X + 267.89 - R^2=0.89, pval<0.01	Y = 298.41 * EXP(-0.14 * X) ; R^2=0.94
Brandy Hole	Y = <0.01 * X + 1.26 - R=0.33, pval=0.58	Y = <0.1 * X + 1.3 ; R^2=0.11	Y = -0.10 * X + 25.81 - R=-0.71, pval=0.18	Y = -0.1 * X + 25.8 ; R^2=0.50	Y = -0.07 * X + 29.32 - R=-0.24, pval=0.70	Y = -0.1 * X + 29.3 ; R^2=0.06
Foulton Hall	Y = 0.01 * X + 0.84 - R=0.62, pval=0.38	Y = <0.1 * X + 0.8 ; R^2=0.38	Y = 0.11 * X + 19.36 - R=0.69, pval=0.31	Y = 0.1 * X + 19.4 ; R^2=0.48	Y = -0.05 * X + 15.30 - R=-0.09, pval=0.91	Y = <-0.1 * X + 15.3 ; R^2=0.01

Mouth depth vs time linear fit	Mouth cross-sectional area vs time exp fit	Mouth cross-sectional area vs time linear fit	Number of creeks vs time exp fit	Number of creeks vs time linear fit	Total channel length vs time exp fit	Total channel length vs time linear fit	Main channel length vs time exp fit	Main channel length vs time linear fit
Y = 0.03 * X + 0.35 - R^2=0.68, pval=0.08	Y = 15.75 * EXP(0.04 * X) ; R^2=0.91	Y = 0.80 * X + 15.53 - R^2=0.90, pval=0.01	Y = 62.99 * EXP(0.03 * X) ; R^2=0.86	Y = 2.50 * X + 61.61 - R^2=0.90, pval=0.01	Y = 5682.17 * EXP(0.01 * X) ; R^2=0.98	Y = 79.34 * X + 5673.79 - R^2=0.98, pval<0.01	Y = 544.18 * EXP(<0.01 * X) ; R^2=0.53	Y = 2.39 * X + 543.89 - R^2=0.53, pval=0.16
Y = 0.17 * X + 0.56 - R^2=0.81, pval=0.10	Y = 14.56 * EXP(0.08 * X) ; R^2=0.98	Y = 1.68 * X + 13.74 - R^2=0.95, pval=0.03	Y = 24.47 * EXP(0.09 * X) ; R^2=0.62	Y = 3.26 * X + 22.68 - R^2=0.59, pval=0.23	Y = 2097.23 * EXP(0.08 * X) ; R^2=0.65	Y = 246.90 * X + 2076.72 - R^2=0.57, pval=0.24	Y = 1698.03 * EXP(-0.01 * X) ; R^2=0.03	Y = -9.85 * X + 1692.63 - R^2=0.03, pval=0.83
Y = 0.08 * X + 2.08 - R^2=1.00, pval=0.00	Y = 88.19 * EXP(-0.01 * X) ; R^2=0.30	Y = -1.11 * X + 88.32 - R^2=0.32, pval=0.44	Y = 73.91 * EXP(0.12 * X) ; R^2=0.77	Y = 18.79 * X + 45.93 - R^2=0.88, pval=0.06	Y = 4580.18 * EXP(0.10 * X) ; R^2=0.54	Y = 880.14 * X + 3188.55 - R^2=0.65, pval=0.19	Y = 413.54 * EXP(0.06 * X) ; R^2=0.71	Y = 33.65 * X + 385.97 - R^2=0.77, pval=0.12
Y = -0.08 * X + 1.44 - R^2=0.89, pval<0.01	Y = 12.13 * EXP(-0.12 * X) ; R^2=0.37	Y = -0.89 * X + 11.58 - R^2=0.43, pval=0.08	Y = 14.45 * EXP(0.14 * X) ; R^2=0.76	Y = 5.07 * X + 6.47 - R^2=0.76, pval<0.01	Y = 587.36 * EXP(0.10 * X) ; R^2=0.73	Y = 113.21 * X + 454.98 - R^2=0.73, pval=0.01	Y = 304.64 * EXP(-0.09 * X) ; R^2=0.82	Y = -15.69 * X + 282.30 - R^2=0.77, pval<0.01
Y = 0.30 * X + 1.09 - R^2=0.93, pval=0.01	Y = 41.47 * EXP(0.10 * X) ; R^2=0.79	Y = 9.86 * X + 21.68 - R^2=0.91, pval=0.01	Y = 77.57 * EXP(0.04 * X) ; R^2=0.82	Y = 4.32 * X + 75.33 - R^2=0.85, pval=0.03	Y = 5095.98 * EXP(0.04 * X) ; R^2=0.89	Y = 301.06 * X + 4944.72 - R^2=0.91, pval=0.01	Y = 441.72 * EXP(0.03 * X) ; R^2=0.73	Y = 14.82 * X + 433.59 - R^2=0.76, pval=0.05
Y = 0.21 * X + 2.39 - R^2=0.52, pval=0.28	Y = 115.90 * EXP(0.05 * X) ; R^2=0.88	Y = 6.60 * X + 114.34 - R^2=0.89, pval=0.06	Y = 117.45 * EXP(0.21 * X) ; R^2=0.98	Y = 55.43 * X + 72.71 - R^2=0.97, pval=0.02	Y = 13117.56 * EXP(0.06 * X) ; R^2=0.98	Y = 1069.76 * X + 12793.83 - R^2=0.98, pval=0.01	Y = 1093.00 * EXP(0.02 * X) ; R^2=0.72	Y = 29.58 * X + 1088.89 - R^2=0.72, pval=0.15
Y = 0.09 * X + 1.64 - R^2=0.92, pval=0.01	Y = 159.34 * EXP(-0.01 * X) ; R^2=0.12	Y = -1.24 * X + 159.29 - R^2=0.13, pval=0.56	Y = 59.89 * EXP(0.11 * X) ; R^2=0.84	Y = 14.67 * X + 28.92 - R^2=0.84, pval=0.03	Y = 5814.36 * EXP(0.07 * X) ; R^2=0.90	Y = 758.94 * X + 4611.00 - R^2=0.90, pval=0.01	Y = 734.20 * EXP(0.03 * X) ; R^2=0.77	Y = 30.47 * X + 717.55 - R^2=0.75, pval=0.06
Y = 2.31 * X + 2.62 - R^2=0.90, pval=0.20	Y = 70.64 * X + 140.15 - R^2=0.99, pval=0.05	Y = 144.40 * EXP(0.34 * X) ; R^2=1.00	Y = 72.33 * EXP(-0.13 * X) ; R^2=0.42	Y = -7.50 * X + 71.50 - R^2=0.38, pval=0.58	Y = 9060.94 * EXP(<-0.01 * X) ; R^2<0.00	Y = -16.08 * X + 9060.46 - R^2<0.00, pval=0.98	Y = 2529.90 * EXP(<0.01 * X) ; R^2=0.81	Y = 8.03 * X + 2529.89 - R^2=0.81, pval=0.29
Y = 0.10 * X + 1.45 - R^2=0.93, pval<0.01	Y = 26.32 * EXP(0.05 * X) ; R^2=0.93	Y = 2.50 * X + 17.05 - R^2=0.96, pval<0.01	Y = 24.54 * EXP(0.06 * X) ; R^2=0.63	Y = 2.74 * X + 20.51 - R^2=0.54, pval=0.06	Y = 1729.75 * EXP(0.03 * X) ; R^2=0.66	Y = 59.29 * X + 1677.28 - R^2=0.62, pval=0.04	Y = 684.52 * EXP(-0.01 * X) ; R^2=0.56	Y = -5.89 * X + 675.75 - R^2=0.55, pval=0.06
Y = 0.03 * X + 0.99 - R^2=0.56, pval=0.05	Y = 108.62 * EXP(0.03 * X) ; R^2=0.40	Y = 3.30 * X + 107.12 - R^2=0.42, pval=0.12	Y = 32.22 * EXP(0.15 * X) ; R^2=0.96	Y = 9.77 * X + 21.99 - R^2=0.94, pval<0.01	Y = 1249.29 * EXP(0.16 * X) ; R^2=0.92	Y = 434.59 * X + 689.85 - R^2=0.95, pval<0.01	Y = 357.25 * EXP(0.11 * X) ; R^2=0.77	Y = 75.12 * X + 261.84 - R^2=0.85, pval<0.01
Y = -0.06 * X + 10.45 - R=-0.65, pval=0.23	Y = -5.5 * X + 945.1 ; R^2=0.74	Y = -5.53 * X + 945.12 - R=-0.86, pval=0.06	Y = -6.0 * X + 1402.7 ; R^2=0.67	Y = -5.99 * X + 1402.71 - R=-0.82, pval=0.09	Y = -74.5 * X + 19666.0 ; R^2=0.50	Y = -74.53 * X + 19666.00 - R=-0.71, pval=0.18	Y = 3.1 * X + 622.0 ; R^2=0.57	Y = 3.08 * X + 622.00 - R=0.75, pval=0.14
Y = 0.03 * X + -1.34 - R=0.47, pval=0.53	Y = 1.7 * X + -88.7 ; R^2=0.42	Y = 1.71 * X + -88.73 - R=0.65, pval=0.35	Y = 11.0 * X + -218.3 ; R^2=0.93	Y = 10.96 * X + -218.29 - R=0.97, pval=0.03	Y = 62.3 * X + 11053.0 ; R^2=0.48	Y = 62.31 * X + 11053.03 - R=0.69, pval=0.31	Y = 1.7 * X + 475.6 ; R^2=0.86	Y = 1.72 * X + 475.57 - R=0.93, pval=0.07

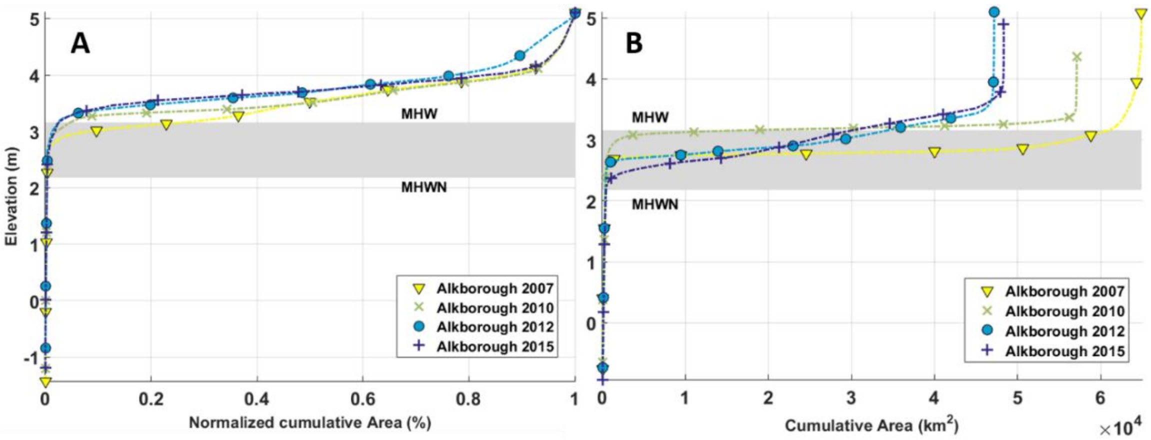
Main channel gradient vs time exp fit	Main channel gradient vs time linear fit	Main channel sinuosity vs time exp fit	Main channel sinuosity vs time linear fit	Tidal prism vs time exp fit	Tidal prism vs time linear fit	Planform area vs time exp fit	Planform area vs time linear fit	Mouth depth vs time exp fit
Y = 0.11 * EXP(0.09 * X) ; R^2=0.15	Y = 0.04 * X + -0.08 - R^2=0.30, pval=0.34	Y = 1.55 * EXP(<-0.01 * X) ; R^2=0.27	Y = <-0.01 * X + 1.55 - R^2=0.27, pval=0.37	Y = 14359.63 * EXP(0.01 * X) ; R^2=0.03	Y = 144.13 * X + 14611.55 - R^2=0.02, pval=0.81	Y = 45197.60 * EXP(0.01 * X) ; R^2=0.51	Y = 705.31 * X + 44872.81 - R^2=0.53, pval=0.16	Y = 0.33 * EXP(0.07 * X) ; R^2=0.78
Y = 3.19 * EXP(<0.01 * X) ; R^2=0.04	Y = 0.01 * X + 3.19 - R^2=0.04, pval=0.80	Y = 1.09 * EXP(<0.01 * X) ; R^2=0.03	Y = <0.01 * X + 1.09 - R^2=0.03, pval=0.84	Y = 13711.12 * EXP(-0.01 * X) ; R^2<0.01	Y = -58.78 * X + 13518.53 - R^2<0.01, pval=0.98	Y = 66431.18 * EXP(-0.04 * X) ; R^2=0.85	Y = -2201.97 * X + 65307.10 - R^2=0.83, pval=0.09	Y = 0.65 * EXP(0.14 * X) ; R^2=0.91
Y = <0.01 * EXP(12.35 * X) ; R^2=0.23	Y = 0.30 * X + -2.18 - R^2=0.49, pval=0.30	Y = 1.40 * EXP(0.02 * X) ; R^2=0.79	Y = 0.03 * X + 1.40 - R^2=0.80, pval=0.11	Y = 21305.74 * EXP(0.10 * X) ; R^2=0.53	Y = 4880.14 * X + 12275.26 - R^2=0.66, pval=0.19	Y = 73599.64 * EXP(0.01 * X) ; R^2=0.35	Y = 571.64 * X + 73498.21 - R^2=0.35, pval=0.41	Y = 2.10 * EXP(0.03 * X) ; R^2=1.00
Y = 1.40 * EXP(0.04 * X) ; R^2=0.27	Y = 0.08 * X + 1.33 - R^2=0.30, pval=0.16	Y = 1.20 * EXP(0.01 * X) ; R^2=0.32	Y = 0.01 * X + 1.20 - R^2=0.32, pval=0.15	Y = 2321.94 * EXP(-0.07 * X) ; R^2=0.13	Y = -92.84 * X + 2114.81 - R^2=0.10, pval=0.43	Y = 4673.74 * EXP(0.10 * X) ; R^2=0.48	Y = 728.71 * X + 4234.49 - R^2=0.44, pval=0.07	Y = 1.54 * EXP(-0.08 * X) ; R^2=0.93
Y = 3.20 * EXP(-0.01 * X) ; R^2=0.50	Y = -0.03 * X + 3.20 - R^2=0.51, pval=0.18	Y = 1.13 * EXP(0.01 * X) ; R^2=0.69	Y = 0.01 * X + 1.13 - R^2=0.71, pval=0.07	Y = 12399.07 * EXP(0.12 * X) ; R^2=0.98	Y = 3229.57 * X + 9392.03 - R^2=0.97, pval<0.01	Y = 40766.52 * EXP(0.03 * X) ; R^2=0.79	Y = 1637.28 * X + 40862.04 - R^2=0.76, pval=0.05	Y = 1.56 * EXP(0.09 * X) ; R^2=0.85
Y = 2.47 * EXP(-0.25 * X) ; R^2=0.68	Y = -0.20 * X + 1.92 - R^2=0.53, pval=0.27	Y = 1.27 * EXP(0.02 * X) ; R^2=0.88	Y = 0.02 * X + 1.27 - R^2=0.88, pval=0.06	Y = 115244.81 * EXP(0.03 * X) ; R^2=0.69	Y = 3813.97 * X + 114927.29 - R^2=0.68, pval=0.17	Y = 188490.93 * EXP(-0.01 * X) ; R^2=0.80	Y = -2268.00 * X + 188409.75 - R^2=0.81, pval=0.10	Y = 2.49 * EXP(0.06 * X) ; R^2=0.48
Y = 0.48 * EXP(0.07 * X) ; R^2=0.84	Y = 0.06 * X + 0.38 - R^2=0.87, pval=0.02	Y = 1.52 * EXP(-0.01 * X) ; R^2=0.80	Y = -0.01 * X + 1.51 - R^2=0.80, pval=0.04	Y = 15572.69 * EXP(0.08 * X) ; R^2=0.48	Y = 1966.54 * X + 14766.78 - R^2=0.40, pval=0.25	Y = 61588.78 * EXP(0.05 * X) ; R^2=0.58	Y = 3898.79 * X + 59416.10 - R^2=0.54, pval=0.16	Y = 1.74 * EXP(0.04 * X) ; R^2=0.90
Y = 5.11 * EXP(-0.01 * X) ; R^2=0.01	Y = -0.05 * X + 5.10 - R^2<0.01, pval=0.96	Y = 1.09 * EXP(<0.01 * X) ; R^2=0.69	Y = <0.01 * X + 1.09 - R^2=0.69, pval=0.37	Y = 150665.25 * EXP(0.15 * X) ; R^2=0.24	Y = 29616.78 * X + 145749.28 - R^2=0.28, pval=0.65	Y = 253892.69 * EXP(-0.00 * X) ; R^2=1.00	Y = -1237.00 * X + 253891.67 - R^2=1.00, pval=0.00	Y = 3.05 * EXP(0.43 * X) ; R^2=0.82
Y = 0.15 * EXP(0.11 * X) ; R^2=0.39	Y = 0.07 * X + 0.02 - R^2=0.27, pval=0.23	Y = 1.35 * EXP(-0.01 * X) ; R^2=0.45	Y = -0.01 * X + 1.33 - R^2=0.44, pval=0.11	Y = 2853.03 * EXP(0.03 * X) ; R^2=0.65	Y = 137.72 * X + 2425.08 - R^2=0.69, pval=0.02	Y = 13331.01 * EXP(0.02 * X) ; R^2=0.63	Y = 408.79 * X + 12912.40 - R^2=0.60, pval=0.04	Y = 1.77 * EXP(0.03 * X) ; R^2=0.89
Y = 1.91 * EXP(0.04 * X) ; R^2=0.85	Y = 0.11 * X + 1.85 - R^2=0.88, pval<0.01	Y = 1.21 * EXP(0.06 * X) ; R^2=0.81	Y = 0.09 * X + 1.16 - R^2=0.84, pval<0.01	Y = 412.37 * EXP(0.33 * X) ; R^2=0.94	Y = 702.41 * X + -801.31 - R^2=0.84, pval<0.01	Y = 14657.71 * EXP(0.13 * X) ; R^2=0.91	Y = 3365.92 * X + 11824.54 - R^2=0.89, pval<0.01	Y = 1.00 * EXP(0.02 * X) ; R^2=0.54
Y = <0.1 * X + -3.6 ; R^2=0.59	Y = 0.05 * X + -3.61 - R=0.77, pval=0.13	Y = <0.1 * X + 1.0 ; R^2=0.01	Y = <0.01 * X + 1.03 - R=0.10, pval=0.88	Y = 264.9 * X + 101596.5 ; R^2=0.08	Y = 264.95 * X + 101596.53 - R=0.29, pval=0.64	Y = -361.9 * X + 235516.7 ; R^2=0.52	Y = -361.87 * X + 235516.71 - R=-0.72, pval=0.17	Y = -0.1 * X + 10.4 ; R^2=0.43
Y = <-0.1 * X + 3.9 ; R^2=0.27	Y = -0.03 * X + 3.90 - R=-0.52, pval=0.48	Y = <-0.1 * X + 2.1 ; R^2=0.43	Y = -0.01 * X + 2.08 - R=-0.66, pval=0.34	Y = -1666.9 * X + 190579.7 ; R^2=0.72	Y = -1666.95 * X + 190579.69 - R=-0.85, pval=0.15	Y = 1238.6 * X + -3577.3 ; R^2=0.47	Y = 1238.56 * X + -3577.29 - R=0.69, pval=0.31	Y = 0.0 * X + -1.3 ; R^2=0.22

Appendix F : MR Hypsometry curves

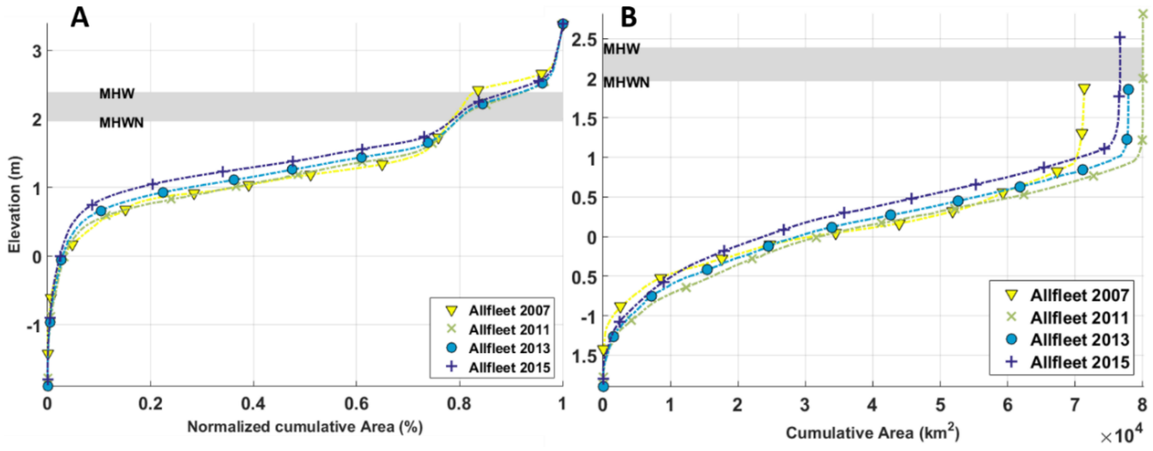
This appendix provides hypsometry curves (for entire site and for creek network) for all 10 MR sites and for all available years.



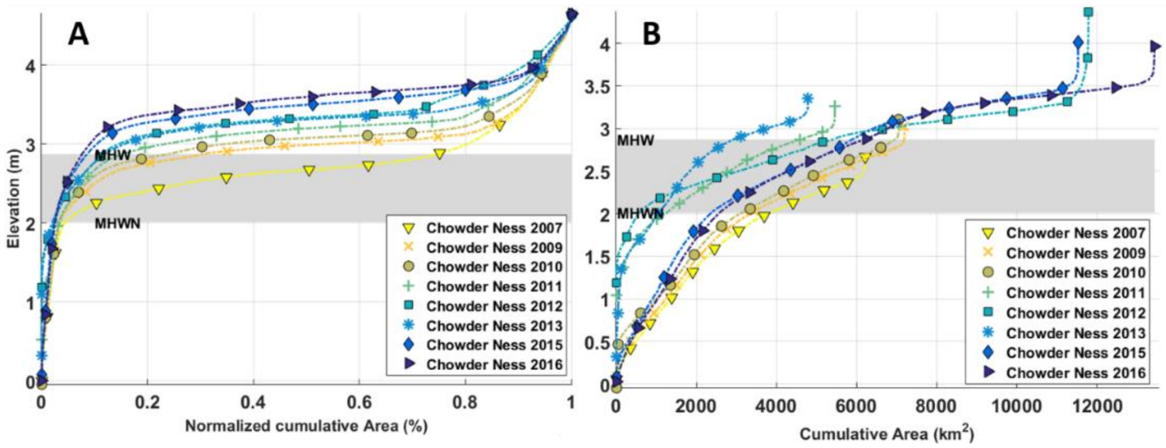
Appendix F1: Abbots Hall hypsometry curves for all available years. A: for the entire site; B: for the creek network as detected by the creek extraction algorithm



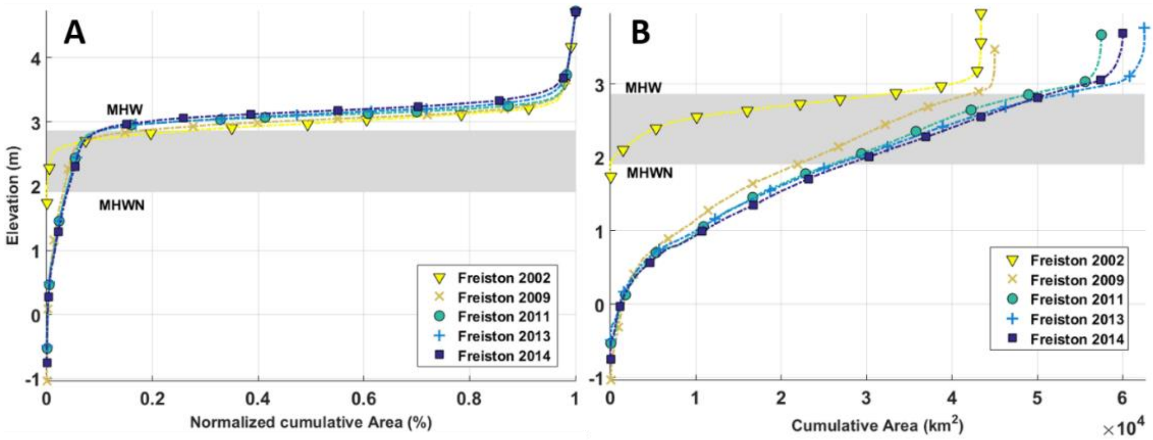
Appendix F2: Same as Appendix F1 but for Alkborough



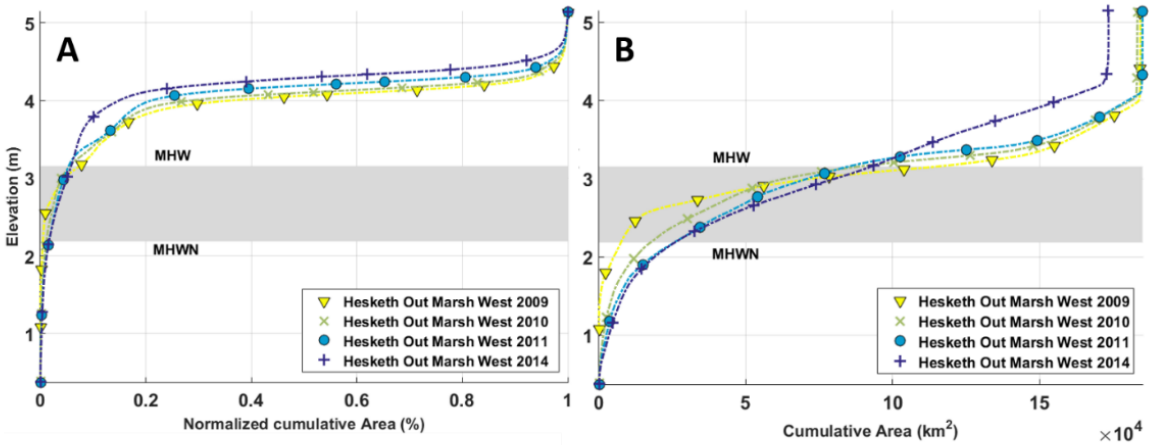
Appendix F3: As Appendix F1 but for Allfleet



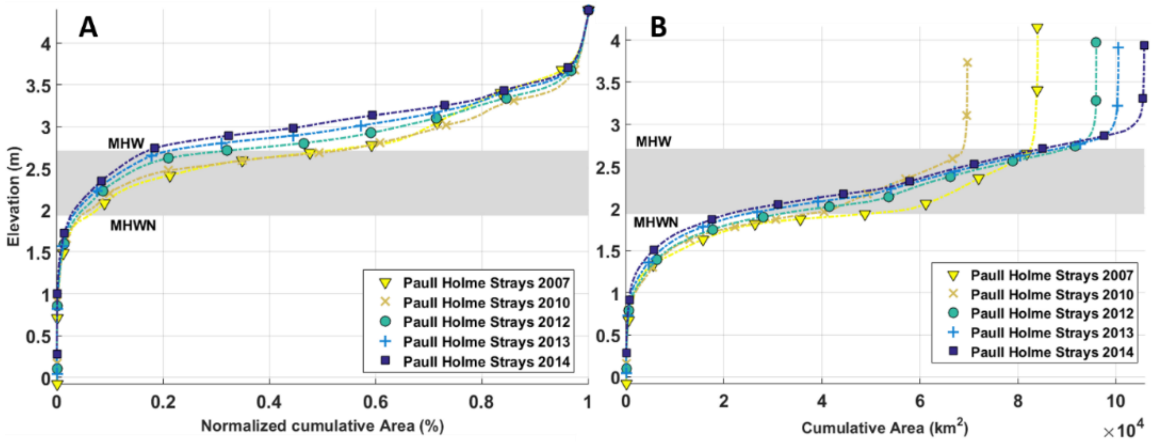
Appendix F4: As Appendix F1 but for Chowder Ness



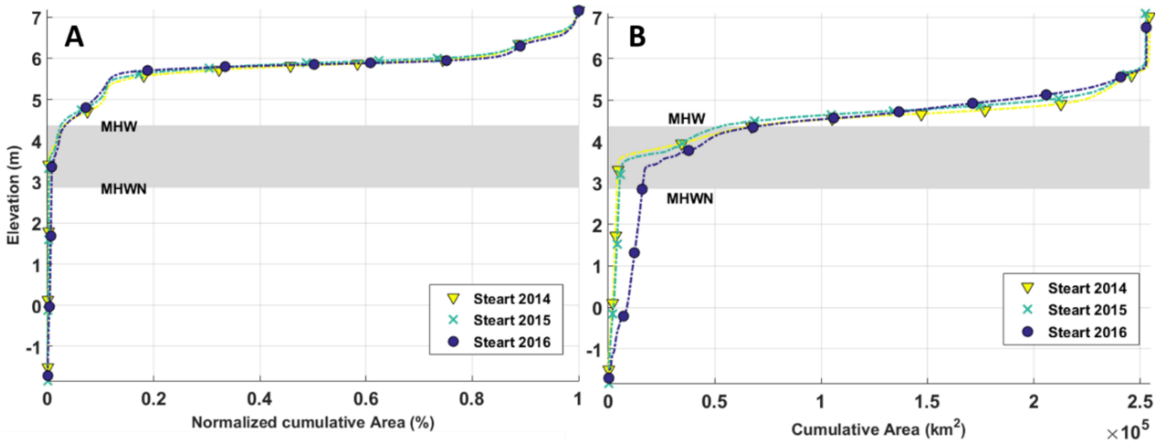
Appendix F5: As Appendix F1 but for Freiston



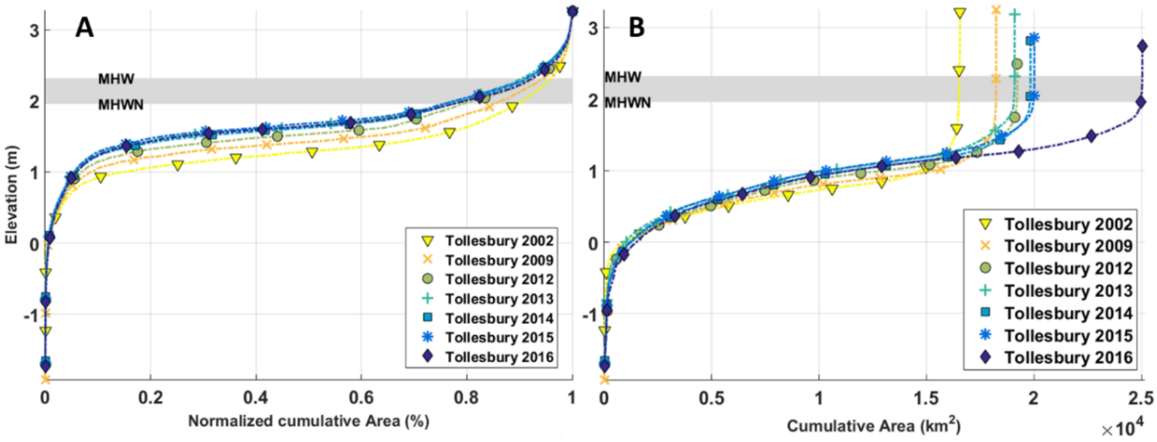
Appendix F6: As Appendix F1 but for Hesketh Out Marsh West



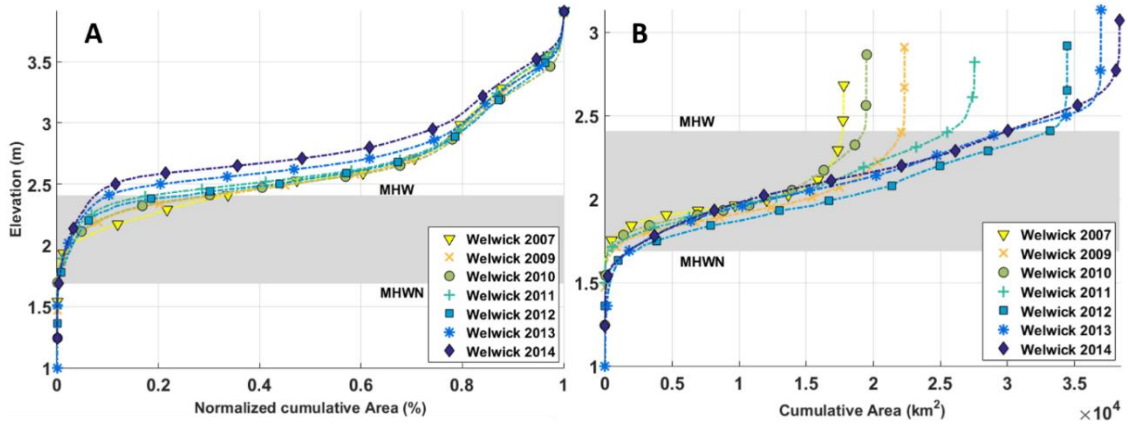
Appendix F7: As Appendix F1 but for Paull Holme Strays



Appendix F8: As Appendix F1 but for Steart



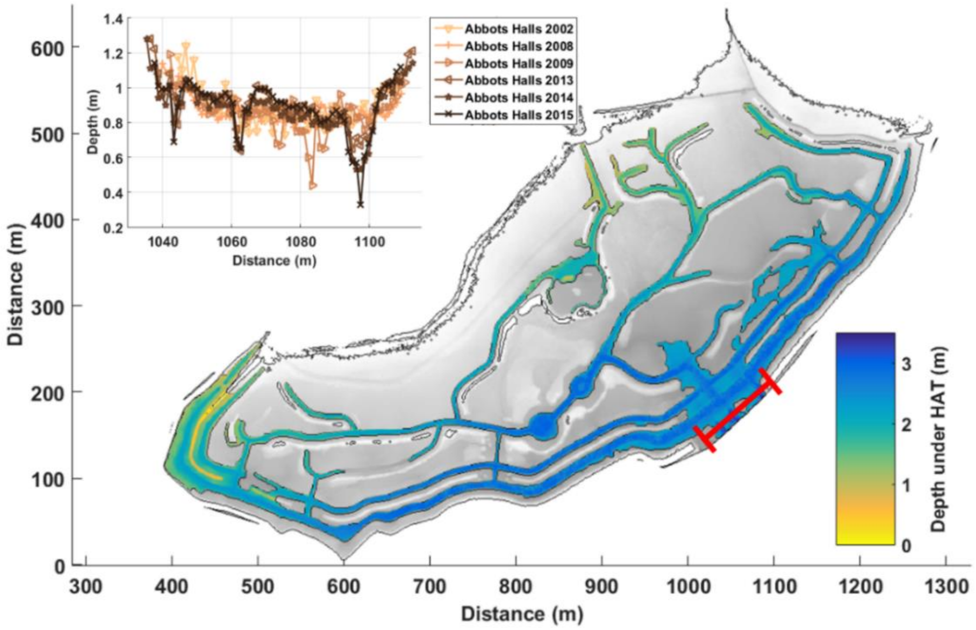
Appendix F9: As Appendix F1 but for Tollesbury



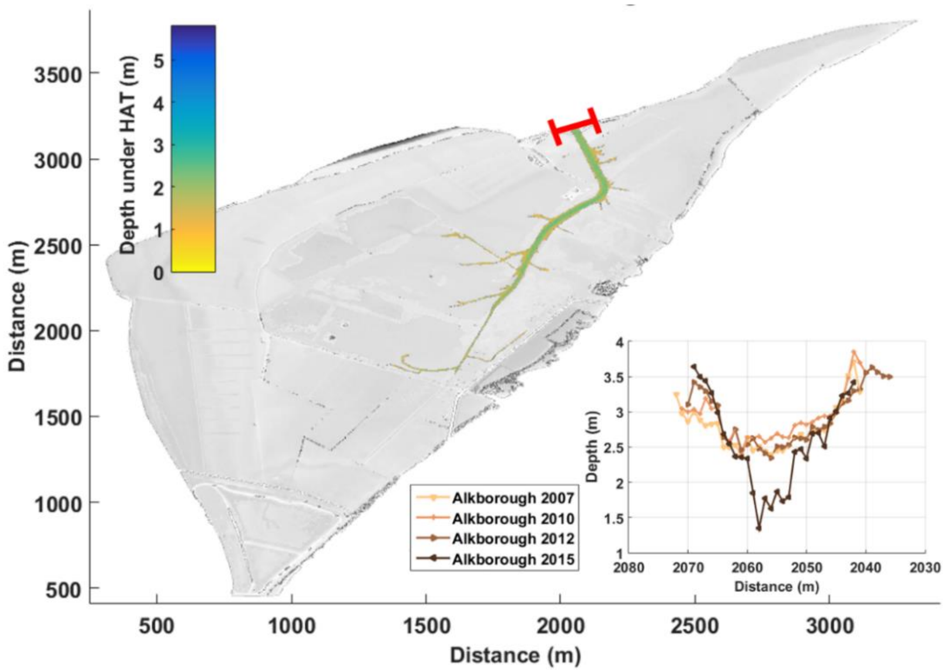
Appendix F10: As Appendix F1 but for Welwick

Appendix G : MR Entry channel cross-sections

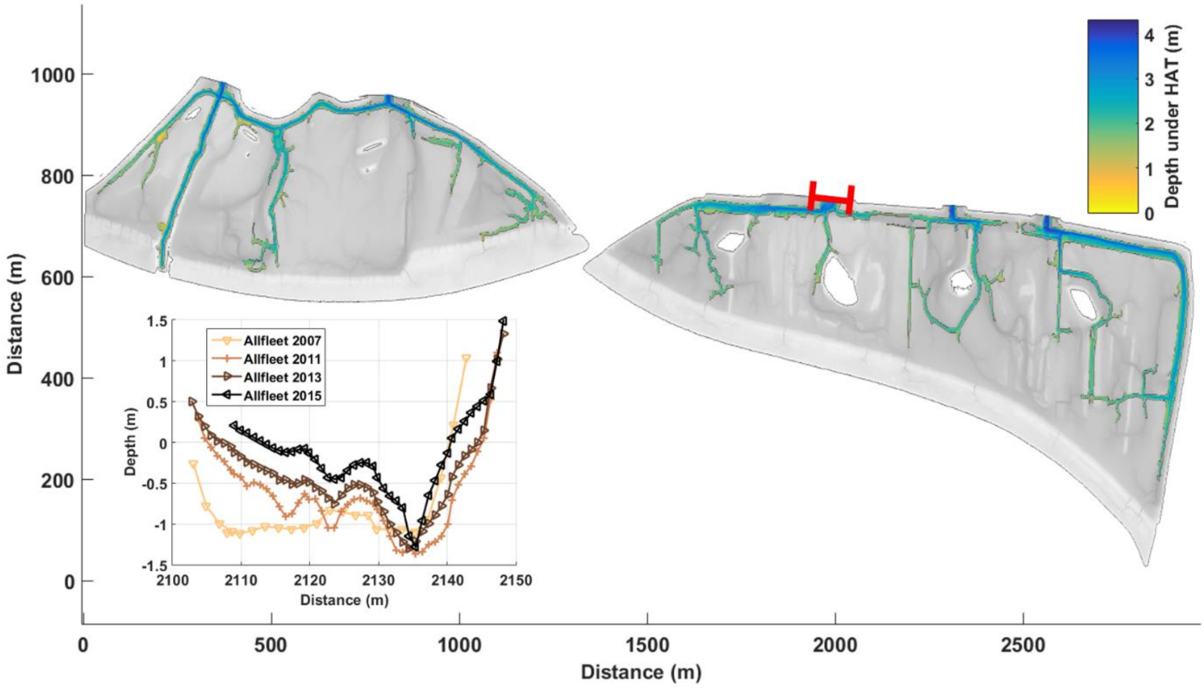
This appendix provides cross-sections of the largest entry channel mouth for all available years for the 10 MR schemes.



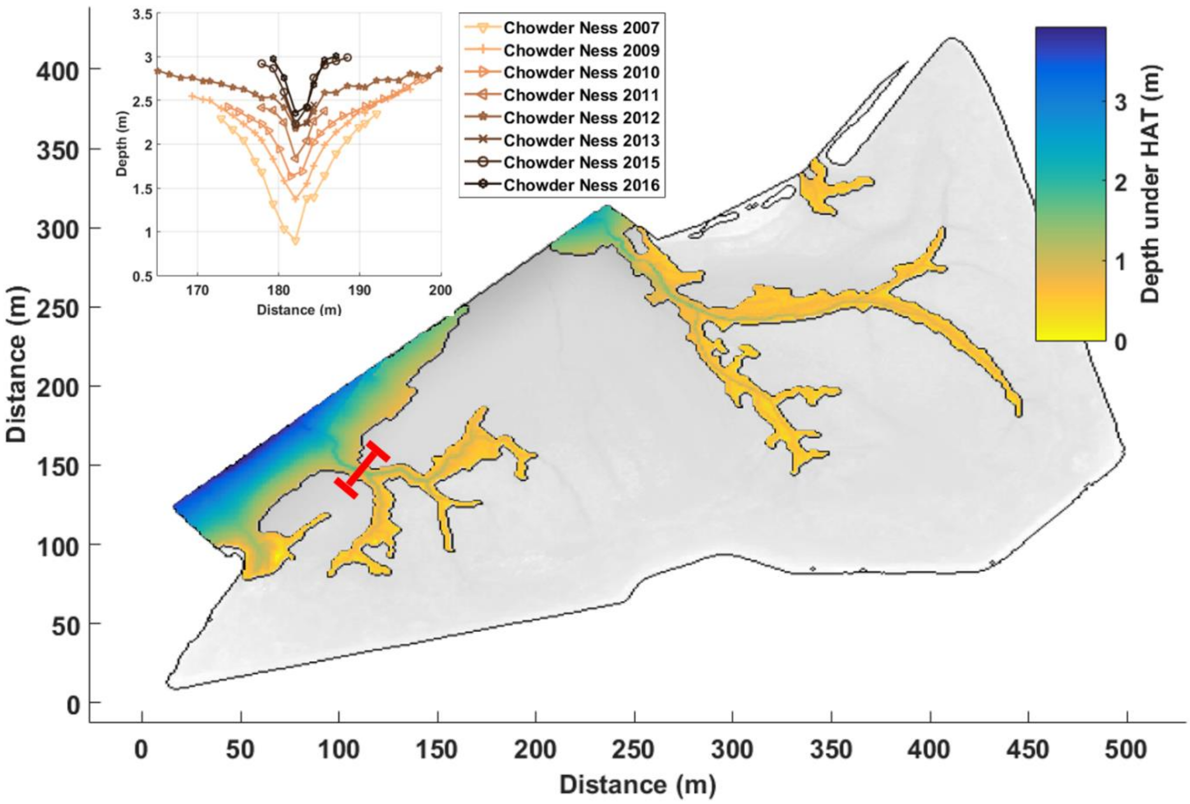
Appendix G1: Cross-sections of the largest entry channel mouth for Abbots Hall for all available years (marsh in greyscale, creek extent for the last available year in colour, entry channel cross-section in red)



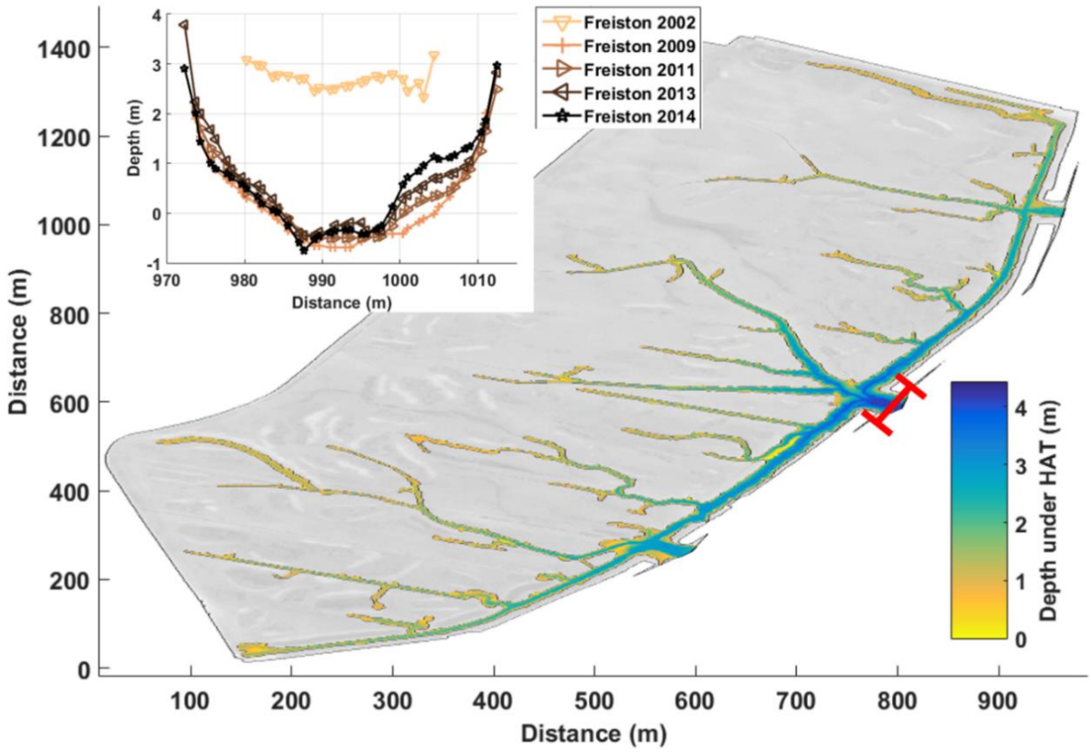
Appendix G2: As Appendix G1 but for Alkborough



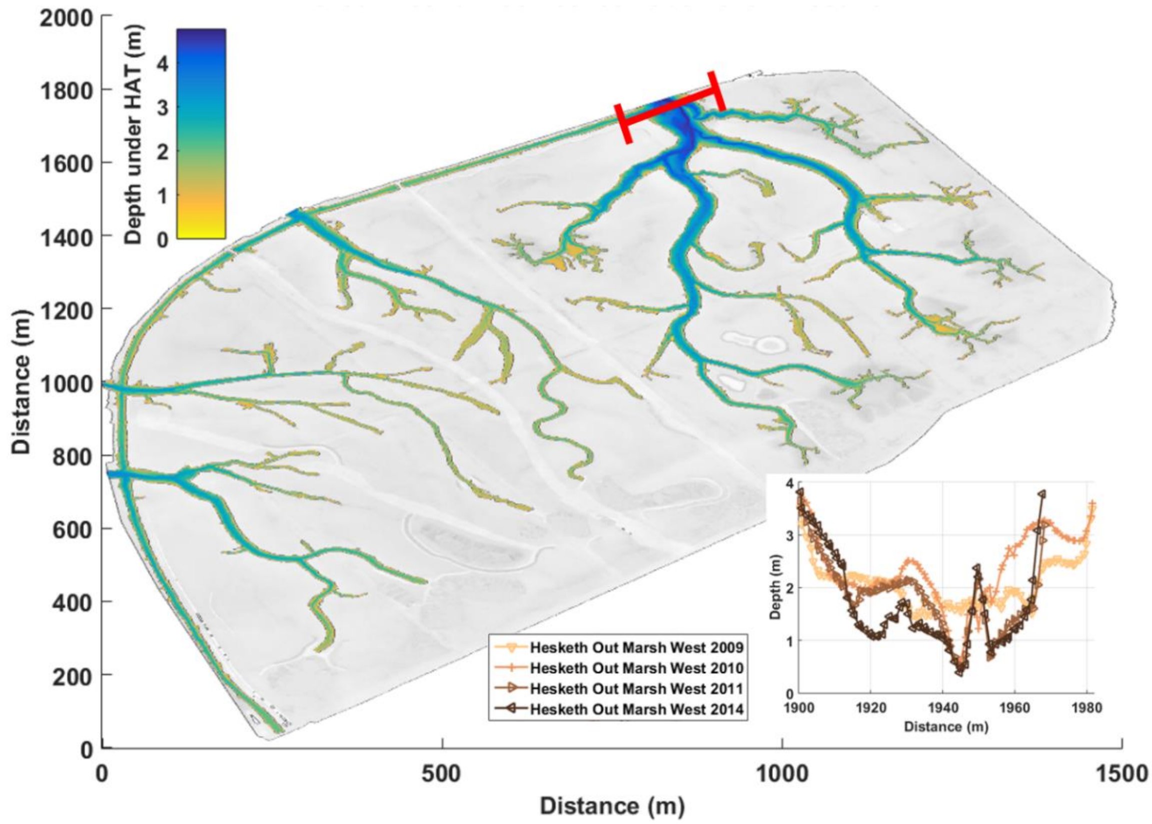
Appendix G3: As Appendix G1 but for Allfleet



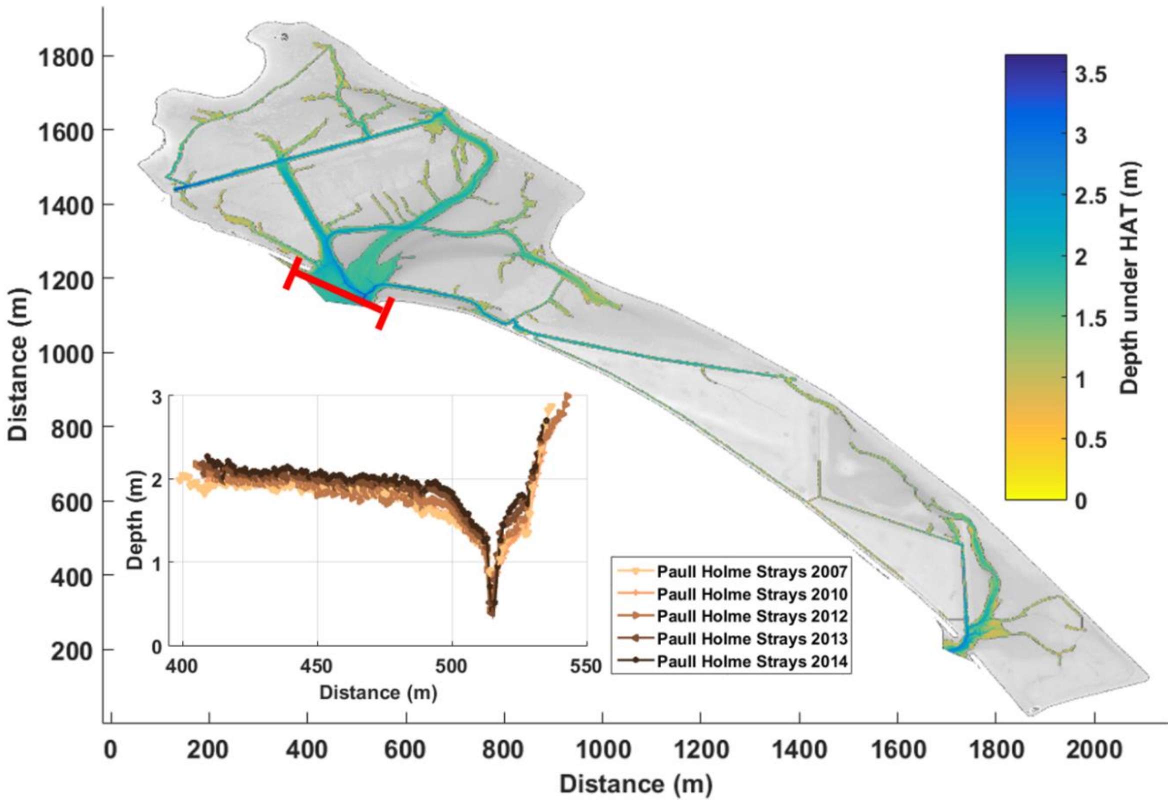
Appendix G4: As Appendix G1 but for Chowder Ness



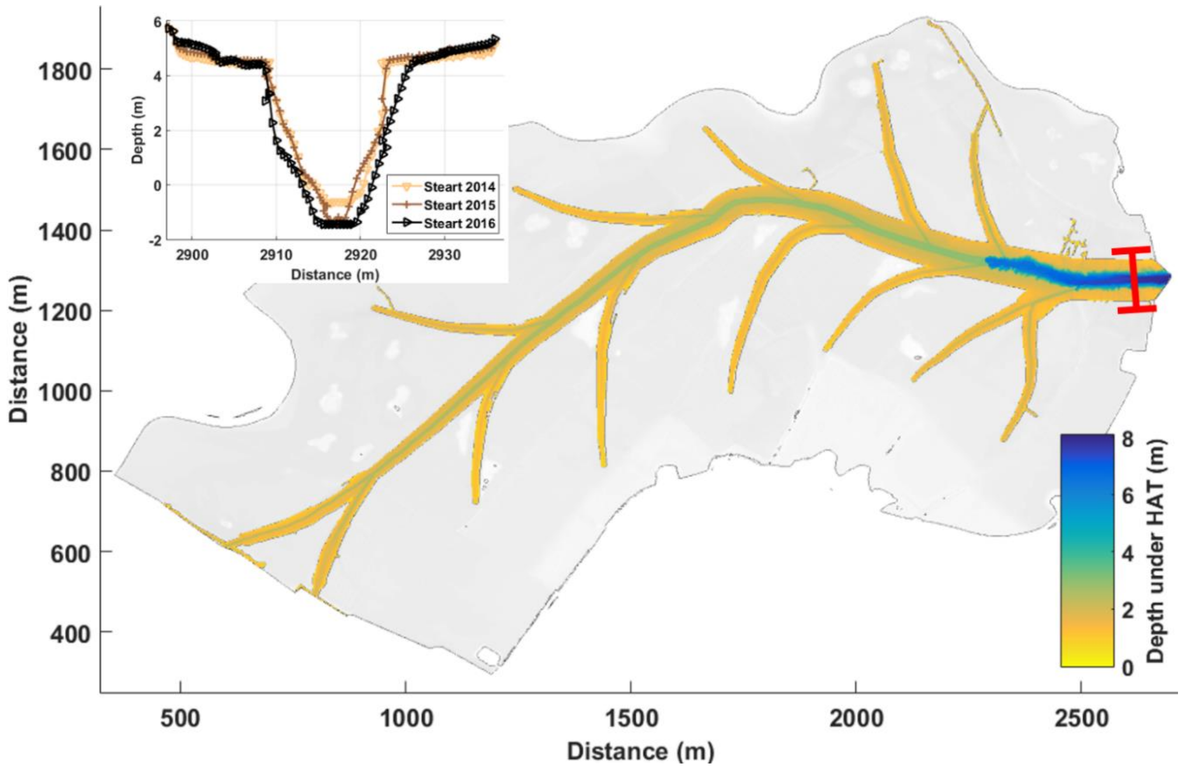
Appendix G5: As Appendix G1 but for Freiston



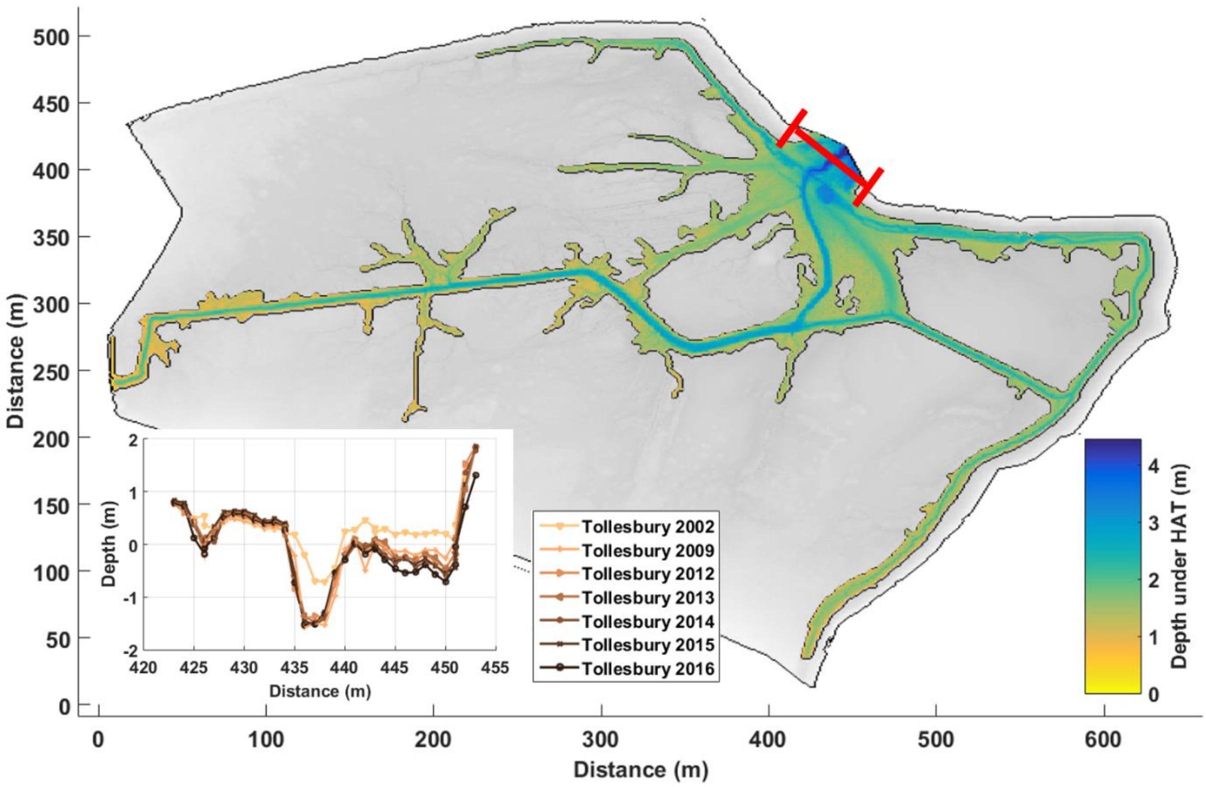
Appendix G6: As Appendix G1 but for Hesketh Out Marsh West



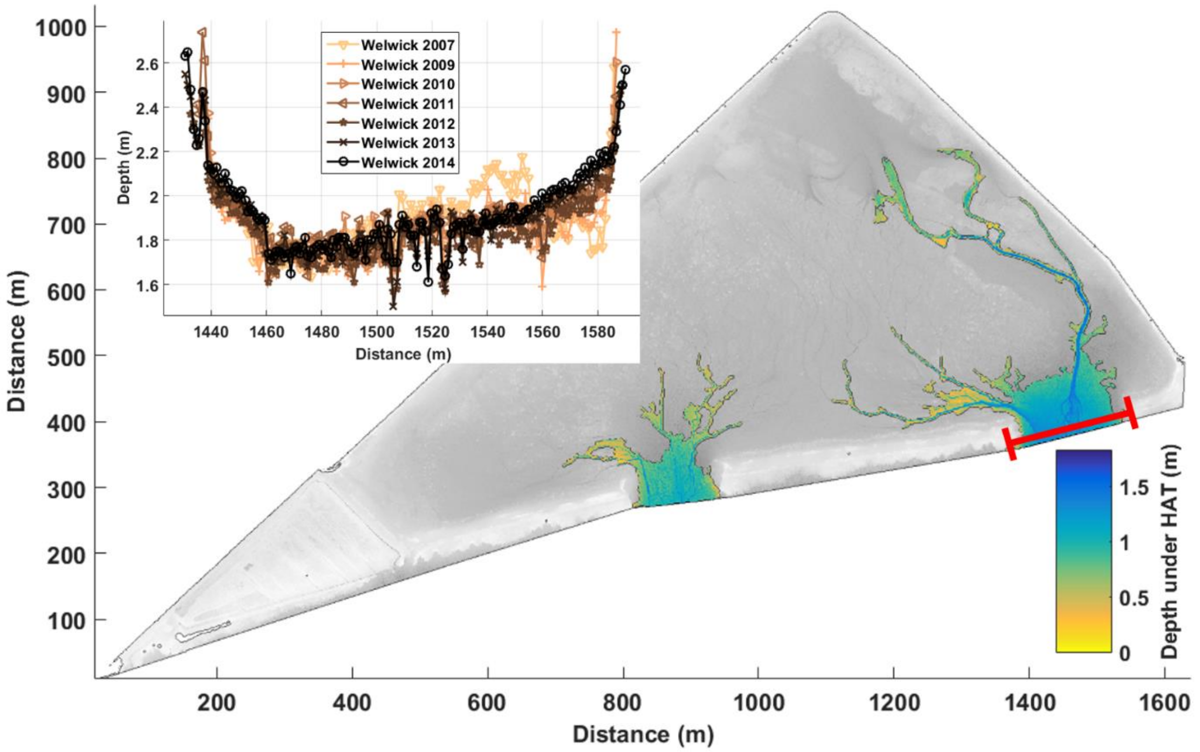
Appendix G7: As Appendix G1 but for Paull Holme Strays



Appendix G8: As Appendix G1 but for Steart



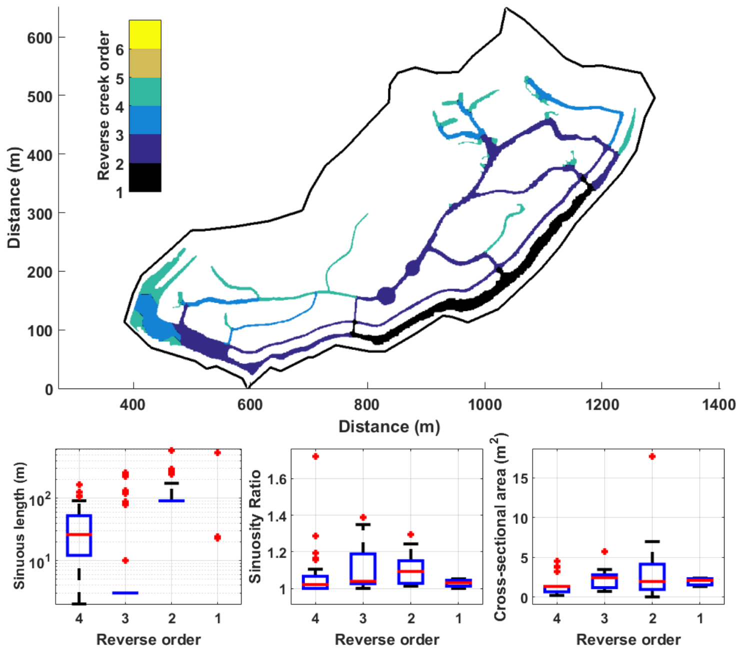
Appendix G9: As Appendix G1 but for Tollesbury



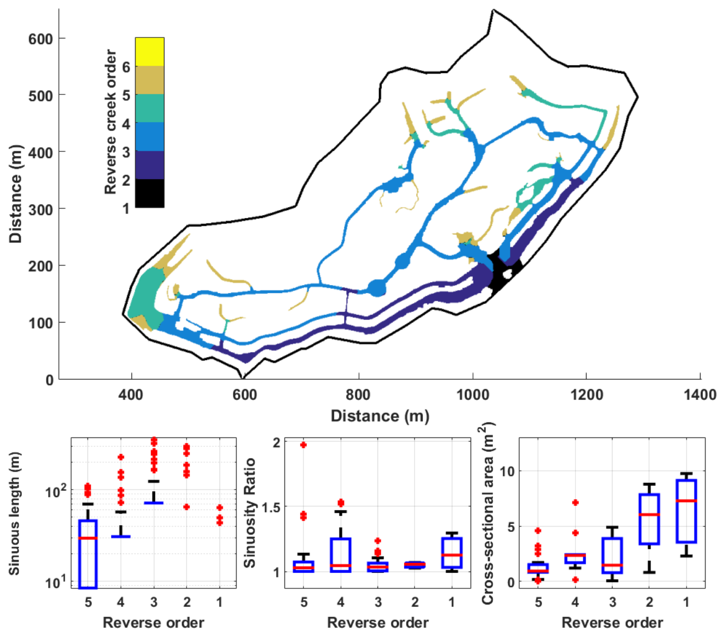
Appendix G10: As Appendix G1 but for Welwick

Appendix H : MR creek RS order distribution

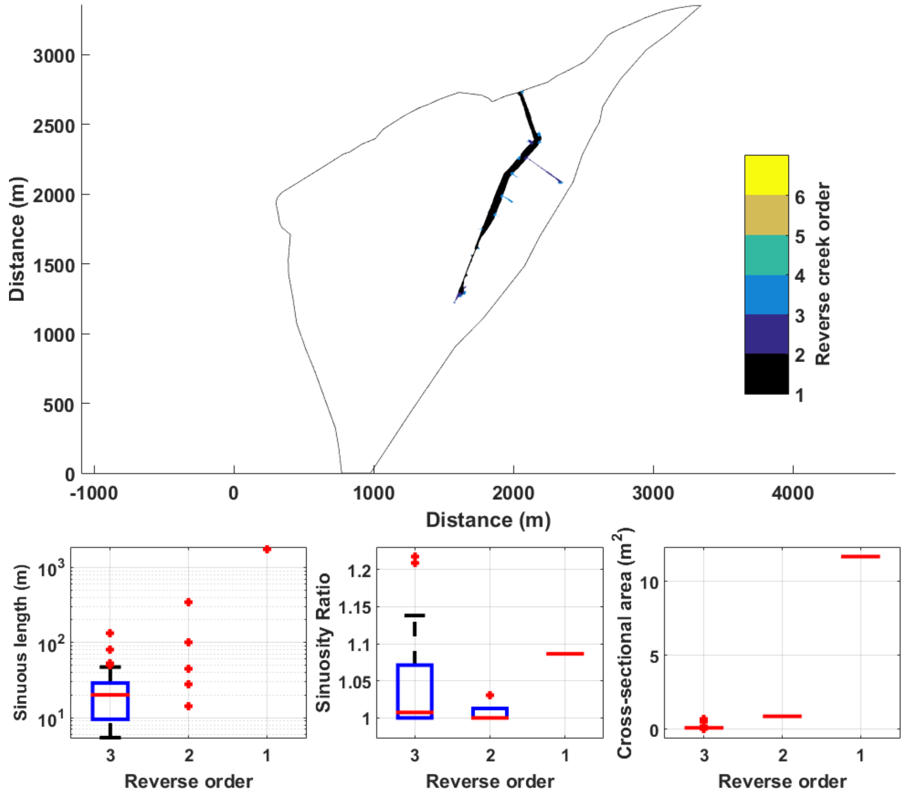
This appendix provides the distribution of RS order creeks within the 10 MR schemes considered at the first and last year of analysis, and gives the corresponding values of sinuous channel length, sinuosity ratio and cross-sectional area per RS order.



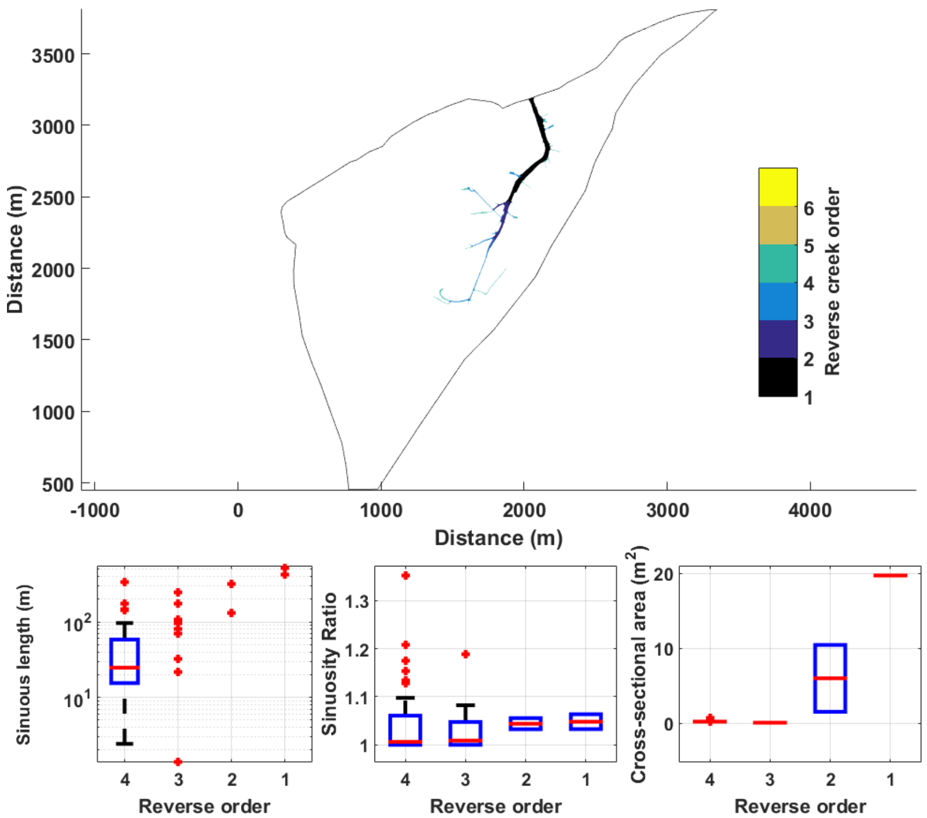
Appendix H1: Creek Reverse Strahler order evolution and distribution of sinuous channel length, sinuosity ratio and cross-sectional area per order for Abbots Hall 2002



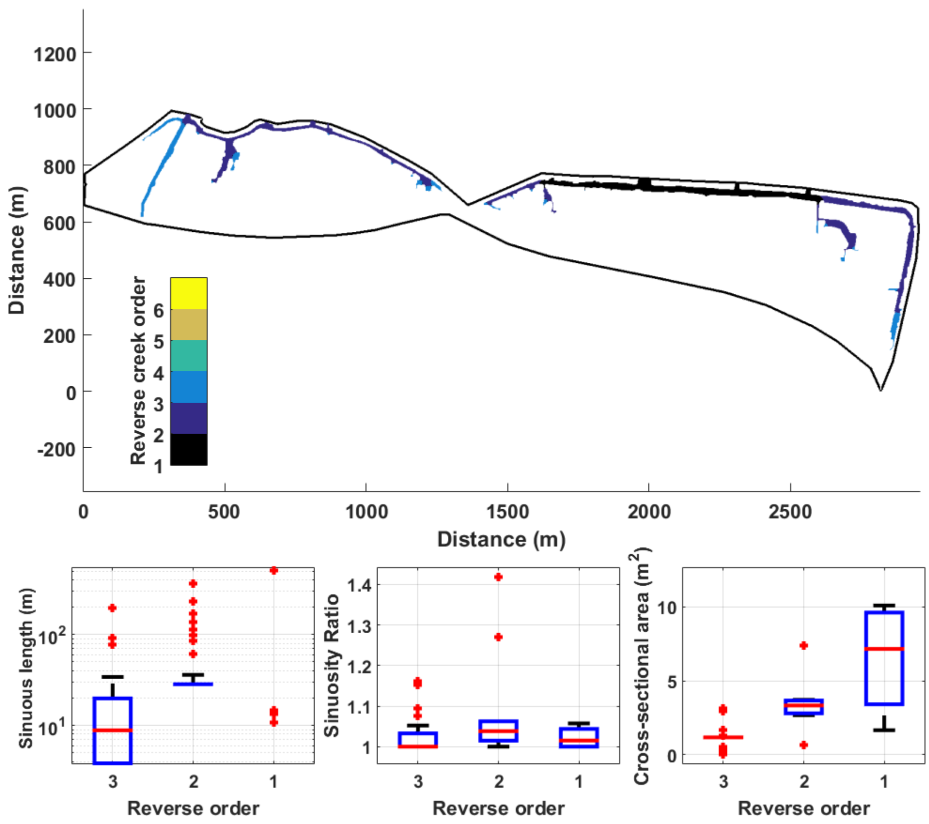
Appendix H2: As Appendix G1 but for Abbots Hall 2015



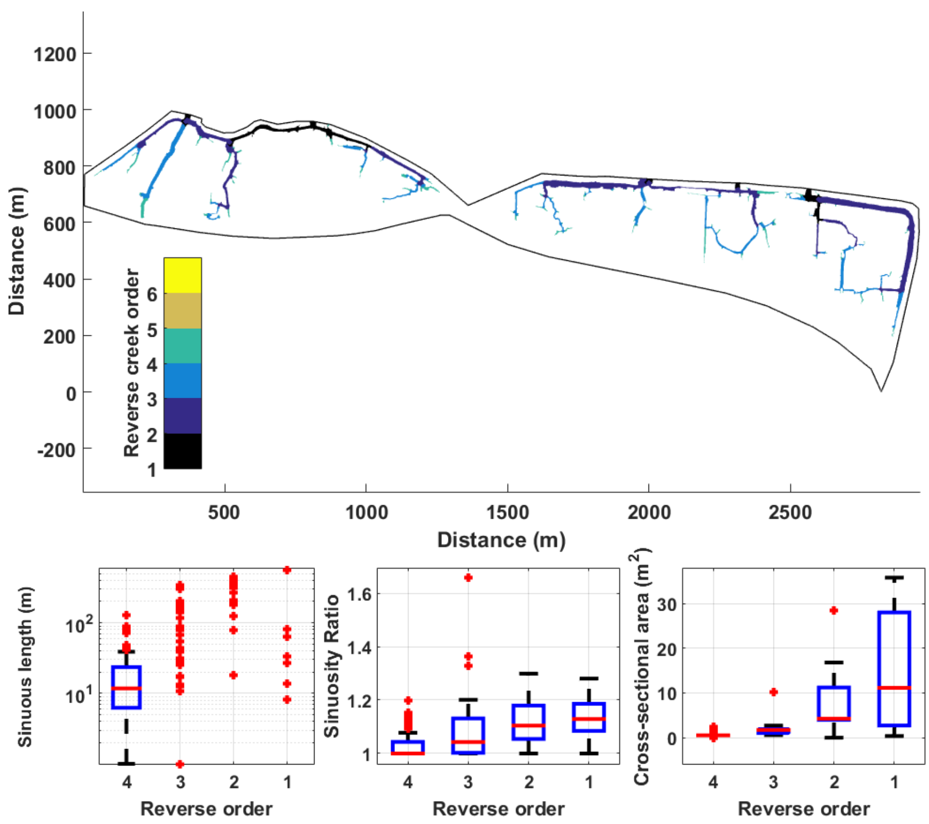
Appendix H3: As Appendix H1 but for Alkborough 2007



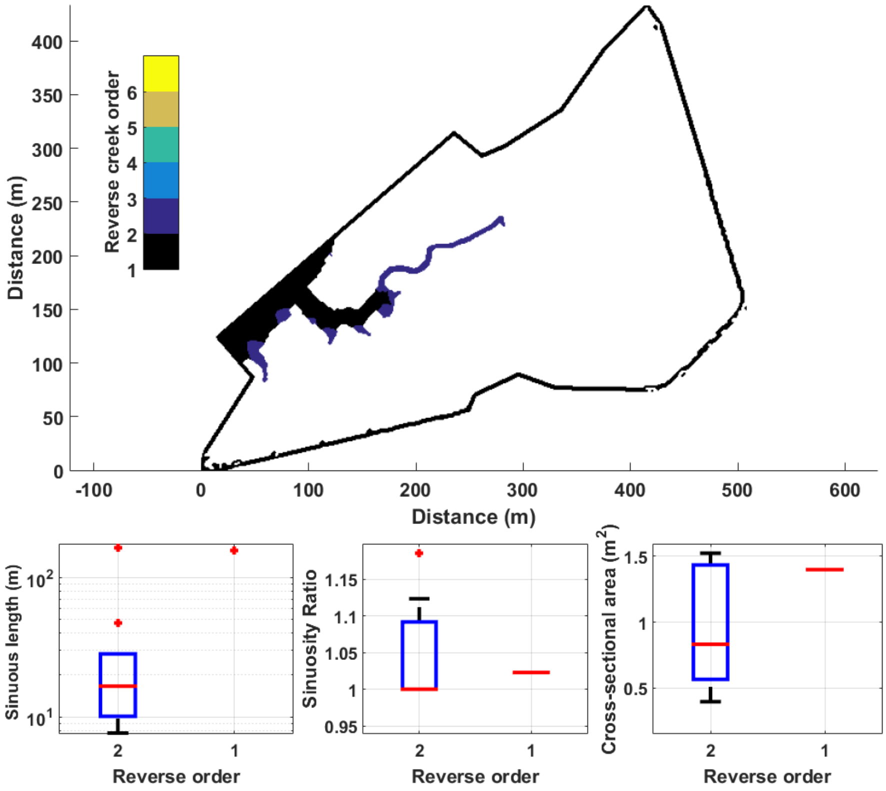
Appendix H4: As Appendix H1 but for Alkborough 2015



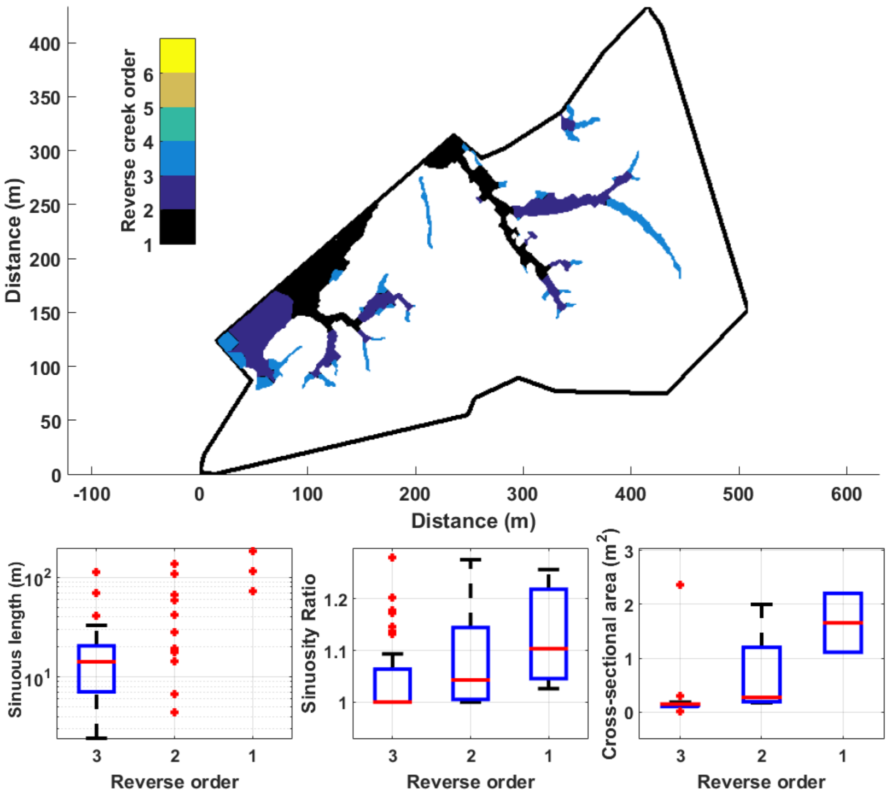
Appendix H5: As Appendix H1 but for Allfleet 2007



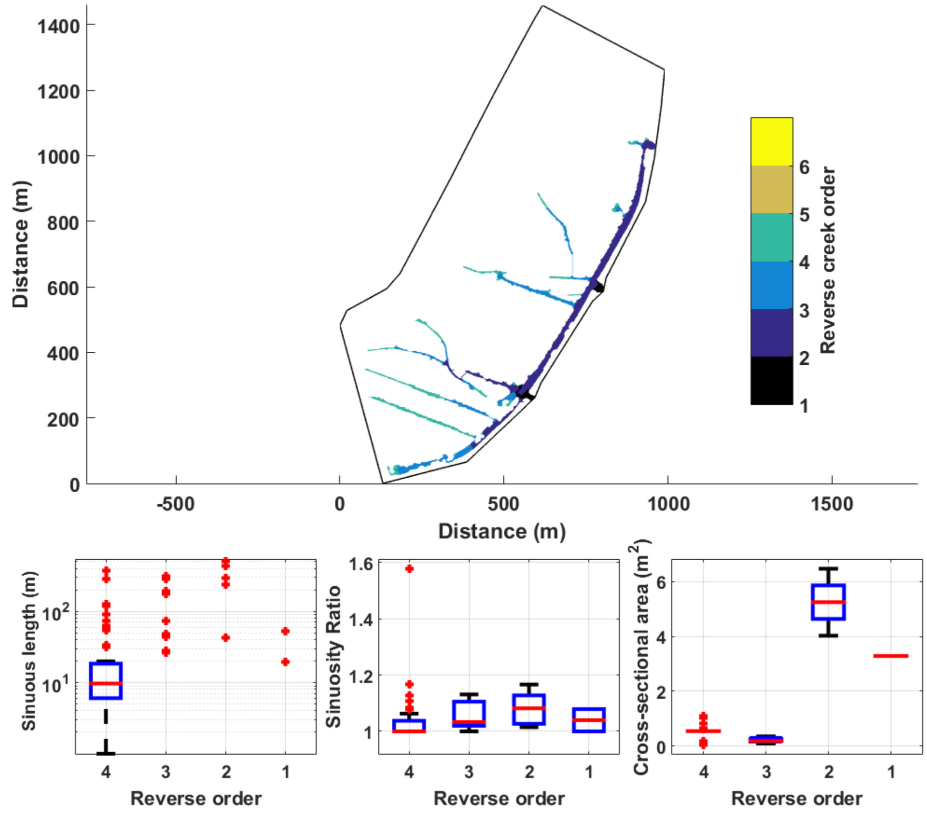
Appendix H6: As Appendix H1 but for Allfleet 2015



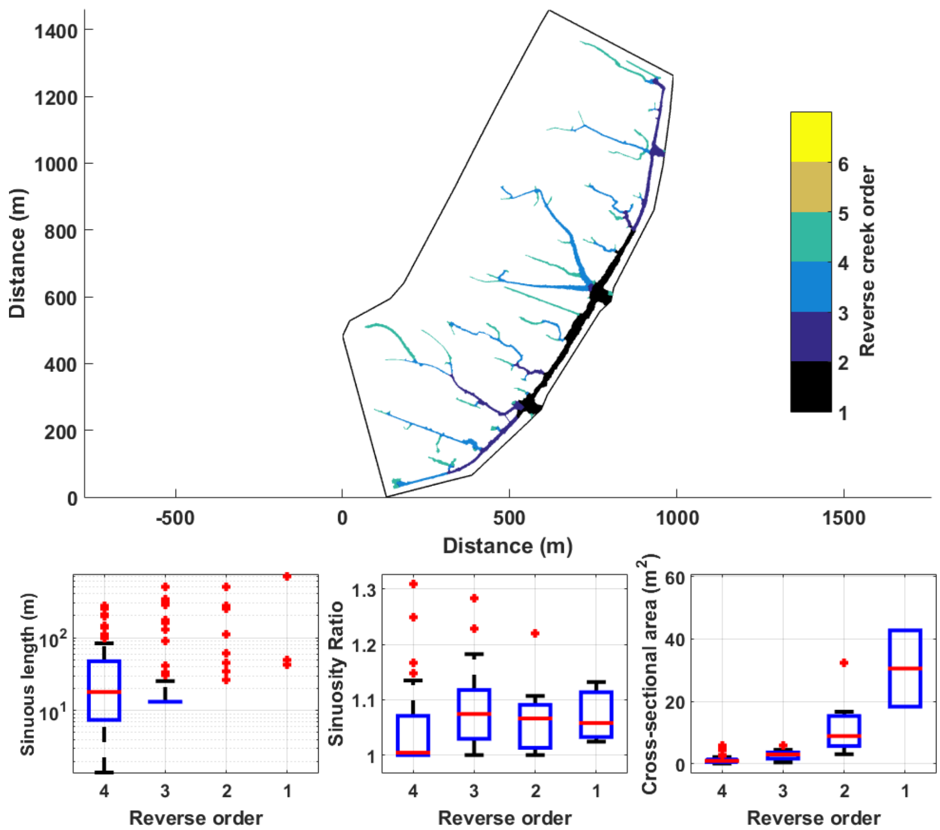
Appendix H7: As Appendix H1 but for Chowder Ness 2007



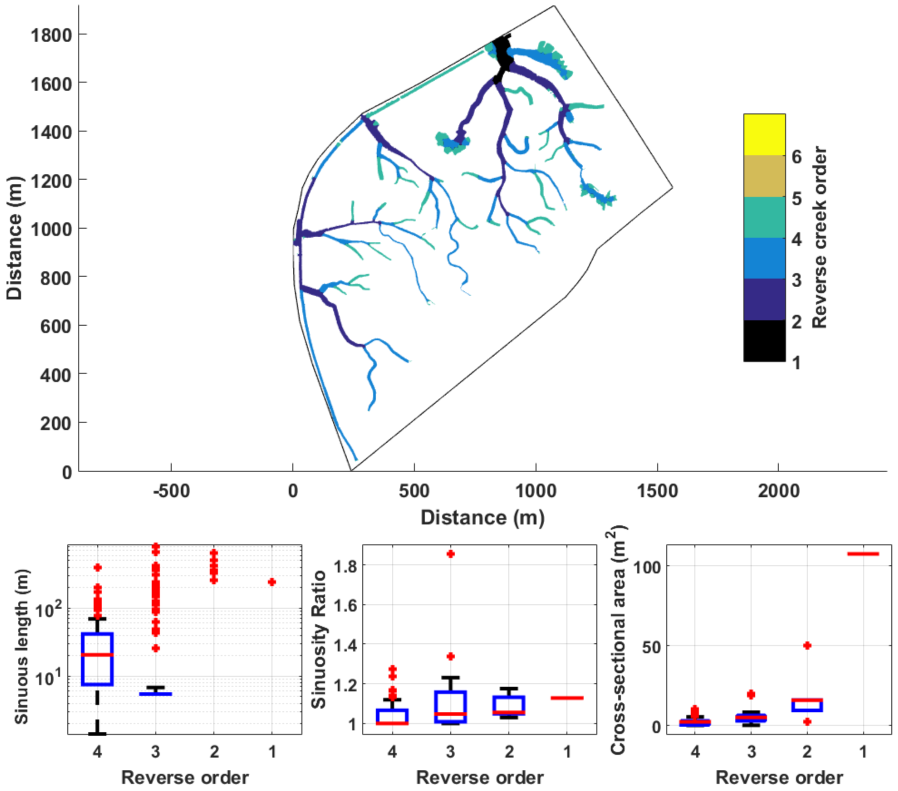
Appendix H8: As Appendix H1 but for Chowder Ness 2016



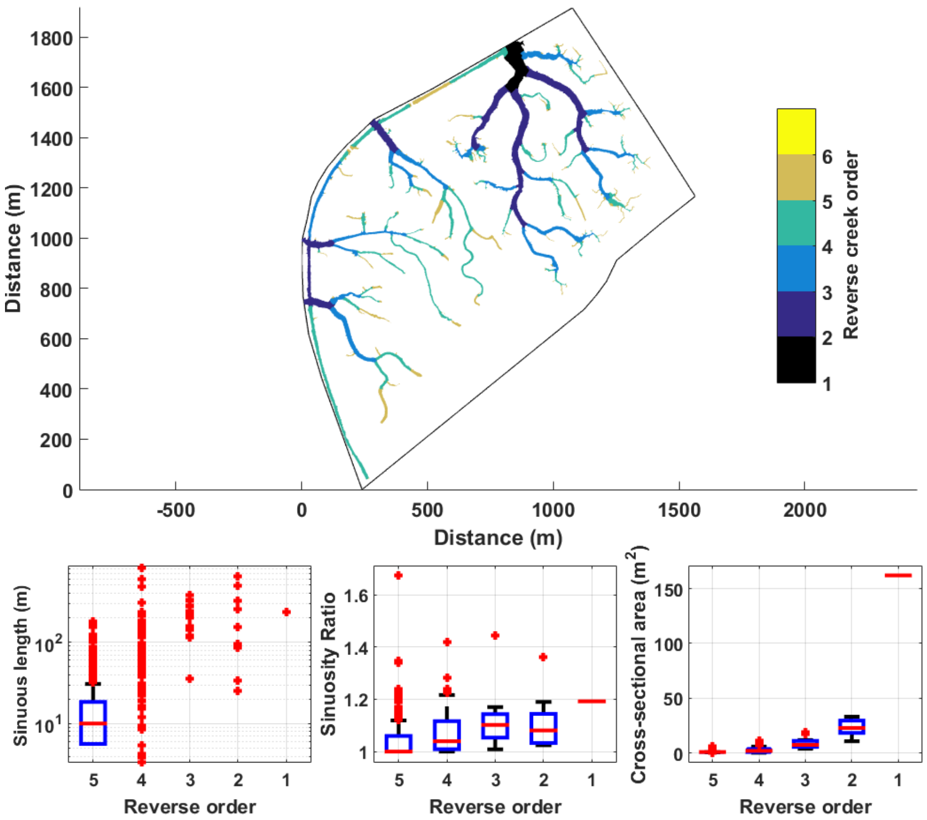
Appendix H9: As Appendix H1 but for Freiston 2002



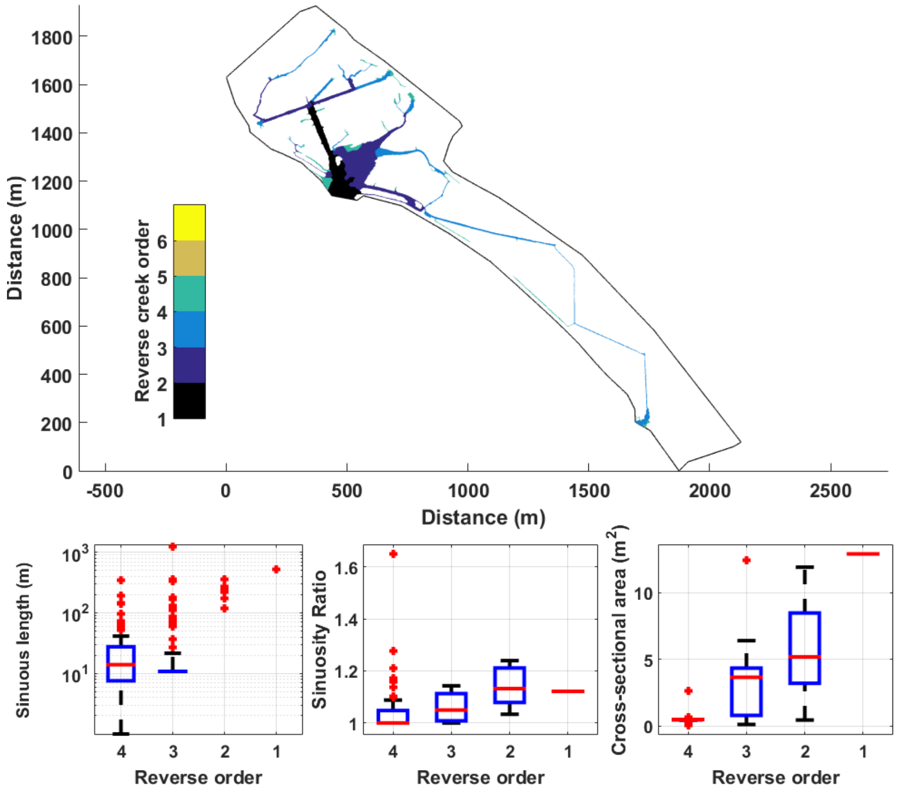
Appendix H10: As Appendix H1 but for Freiston 2014



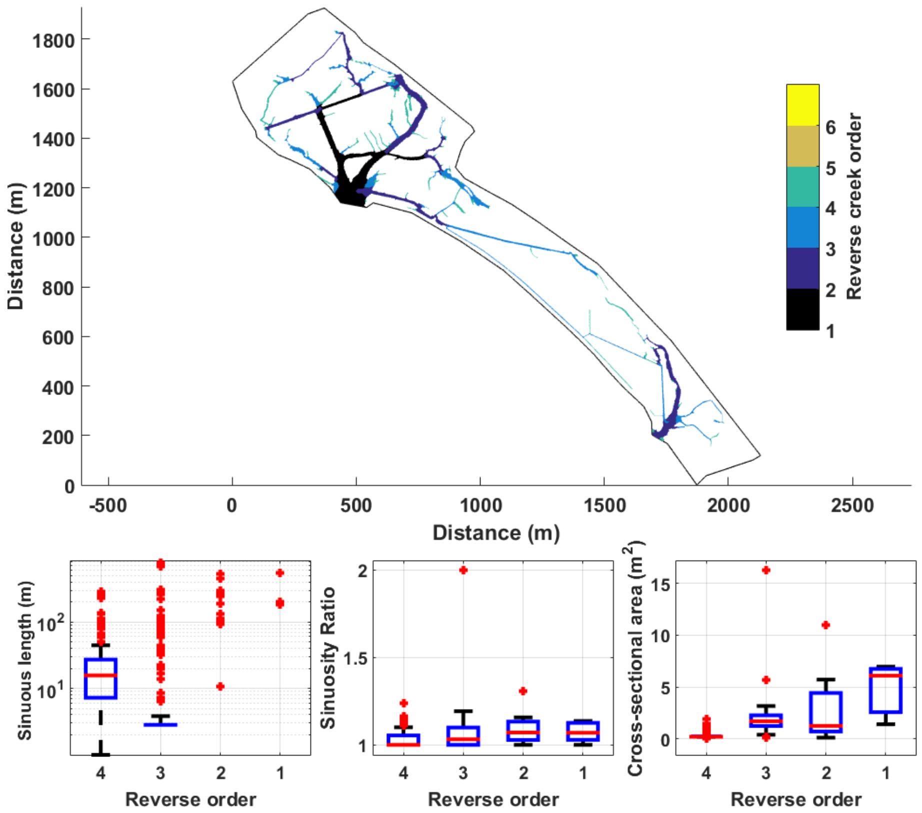
Appendix H11: As Appendix H1 but for Hesketh Out Marsh West 2009



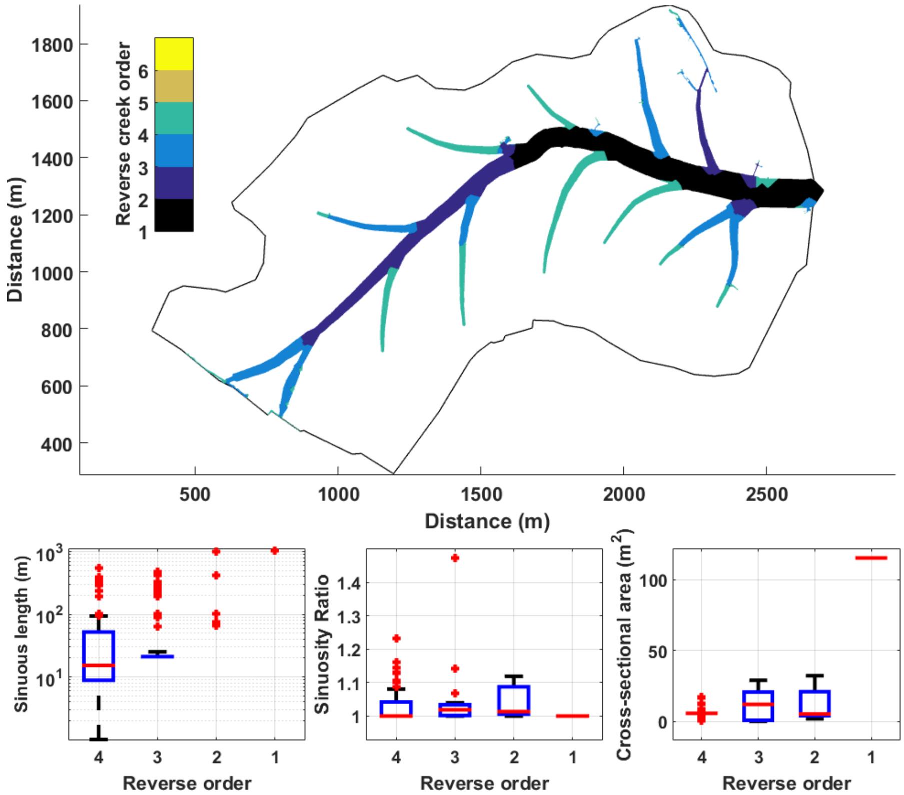
Appendix H12: As Appendix H1 but for Hesketh Out Marsh West 2014



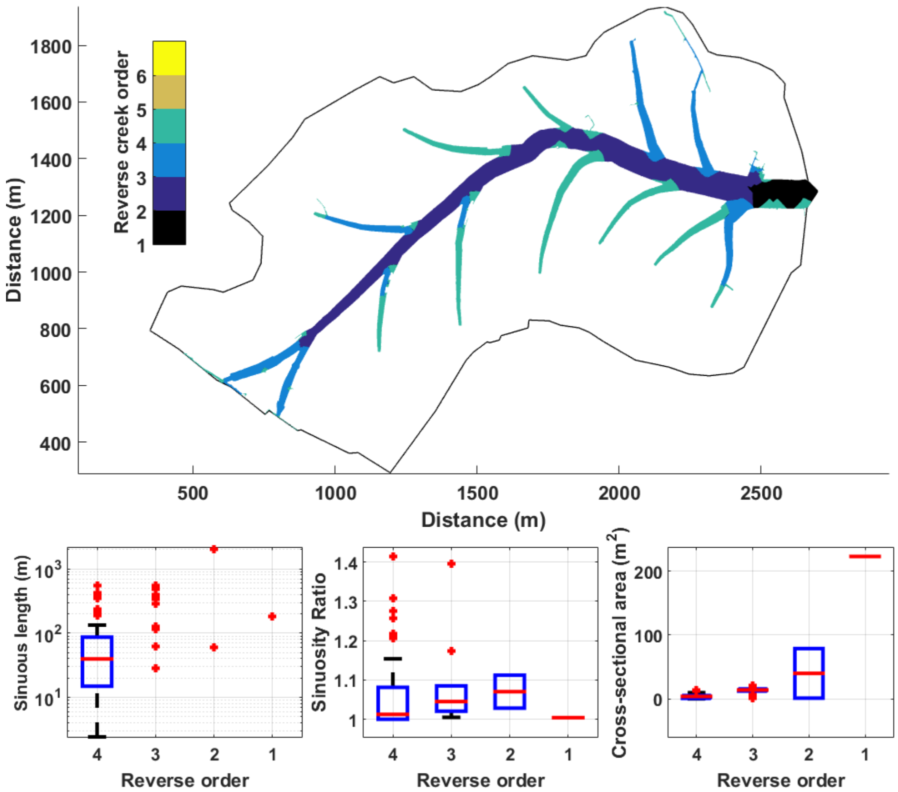
Appendix H13: As Appendix H1 but for Paull Holme Strays 2007



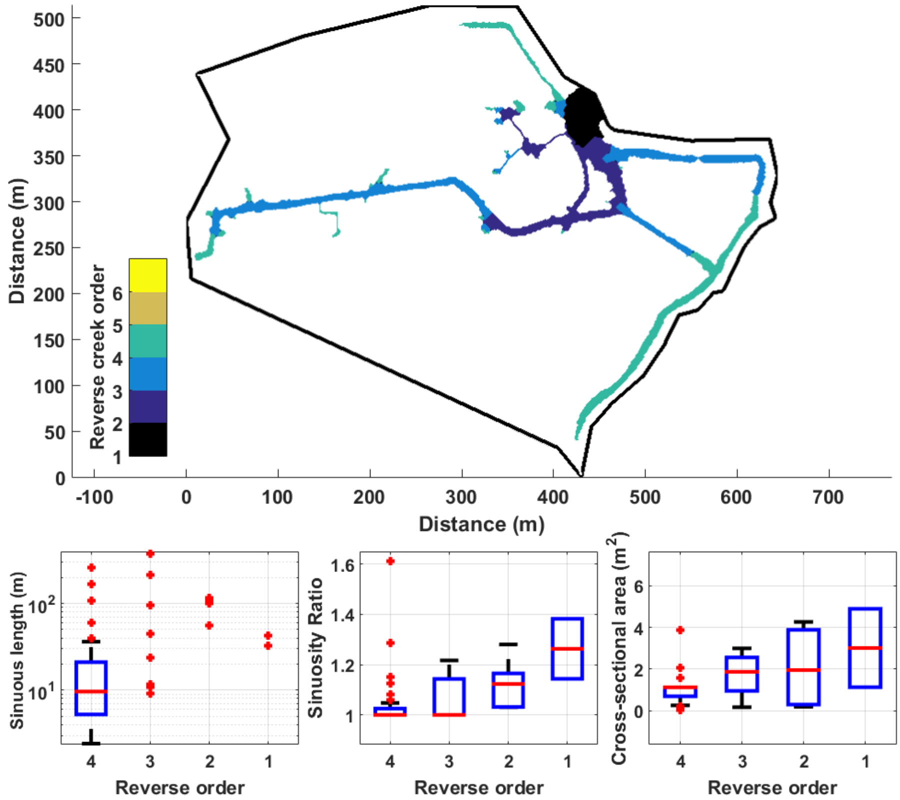
Appendix H14: As Appendix H1 but for Paull Holme Strays 2014



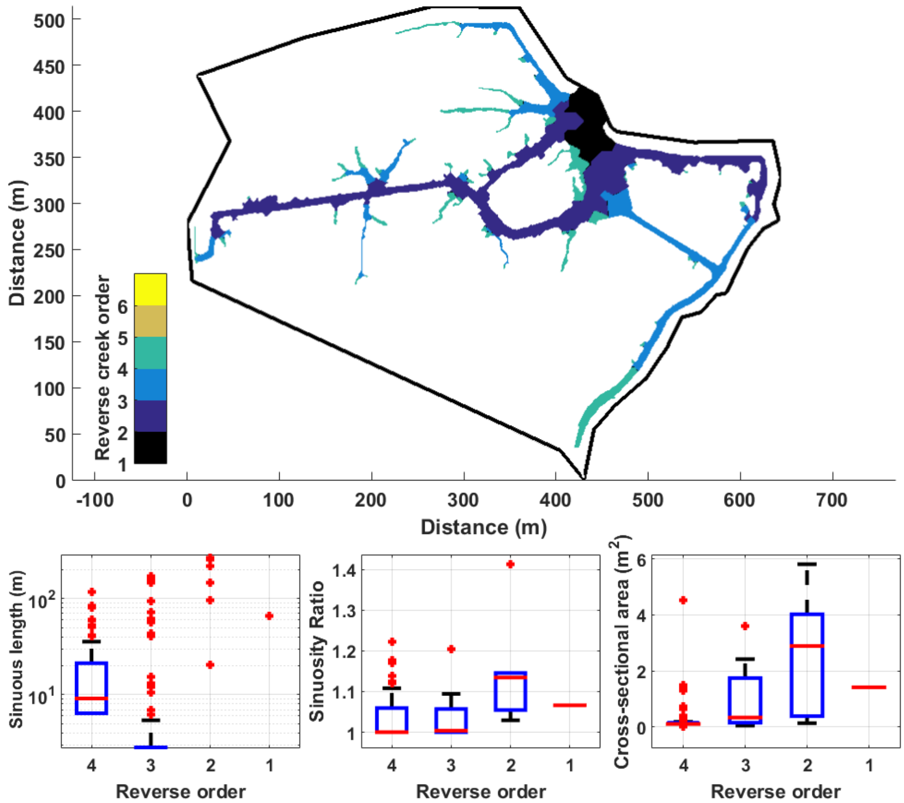
Appendix H15: As Appendix H1 but for Steart 2014



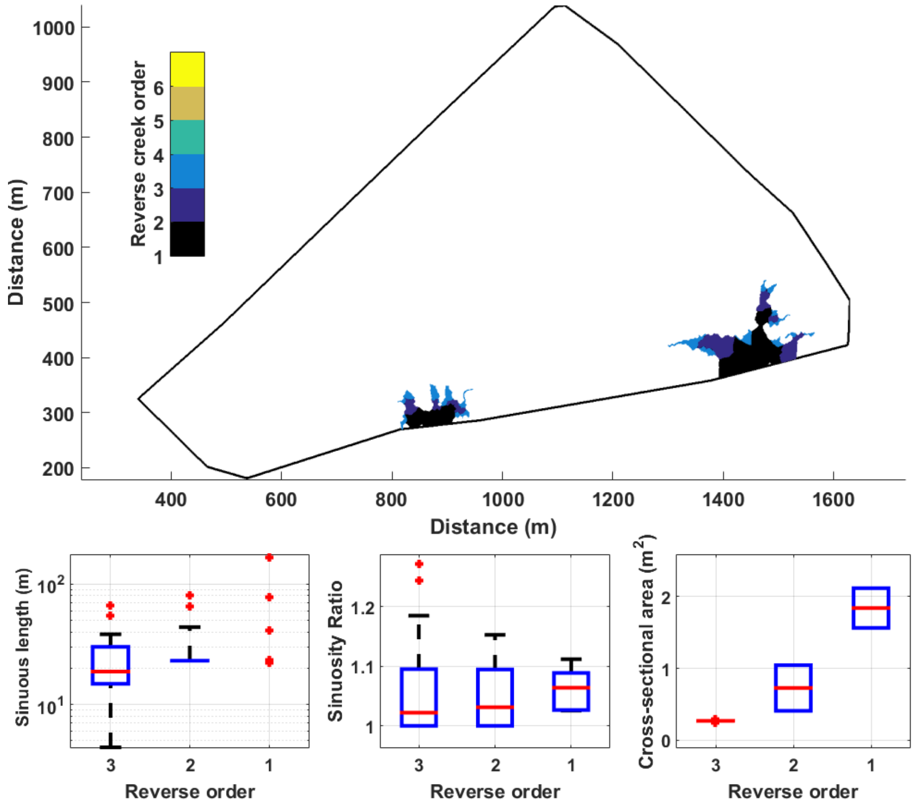
Appendix H16: As Appendix H1 but for Steart 2016



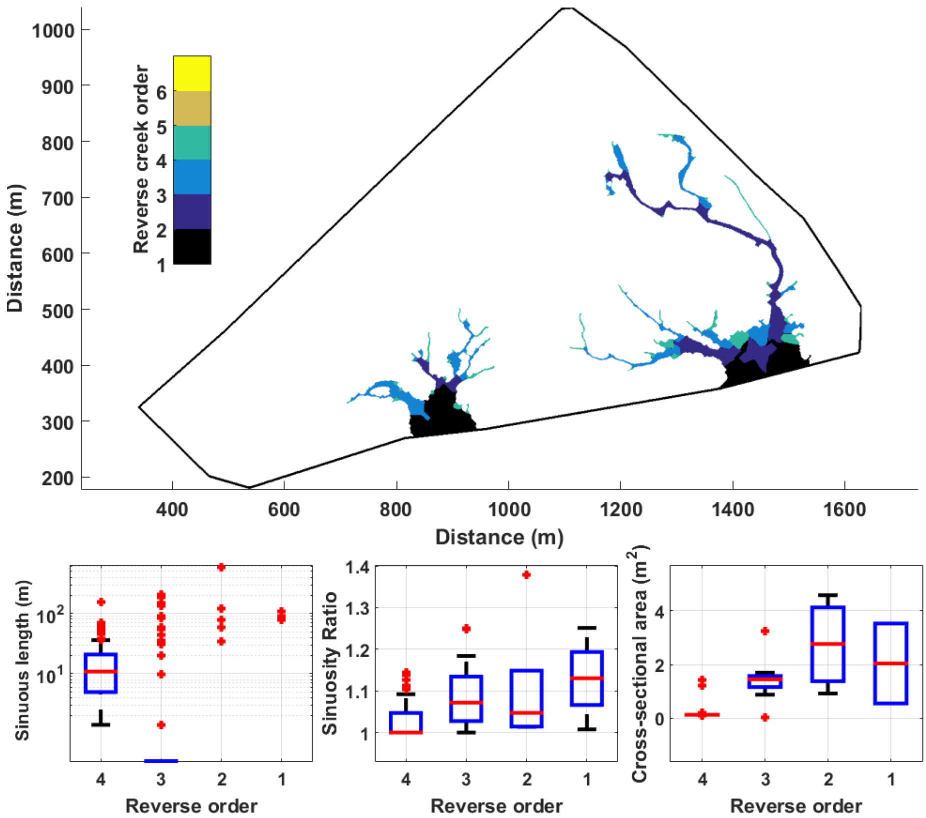
Appendix H17: As Appendix H1 but for Tollesbury 2002



Appendix H18: As Appendix H1 but for Tollesbury 2016



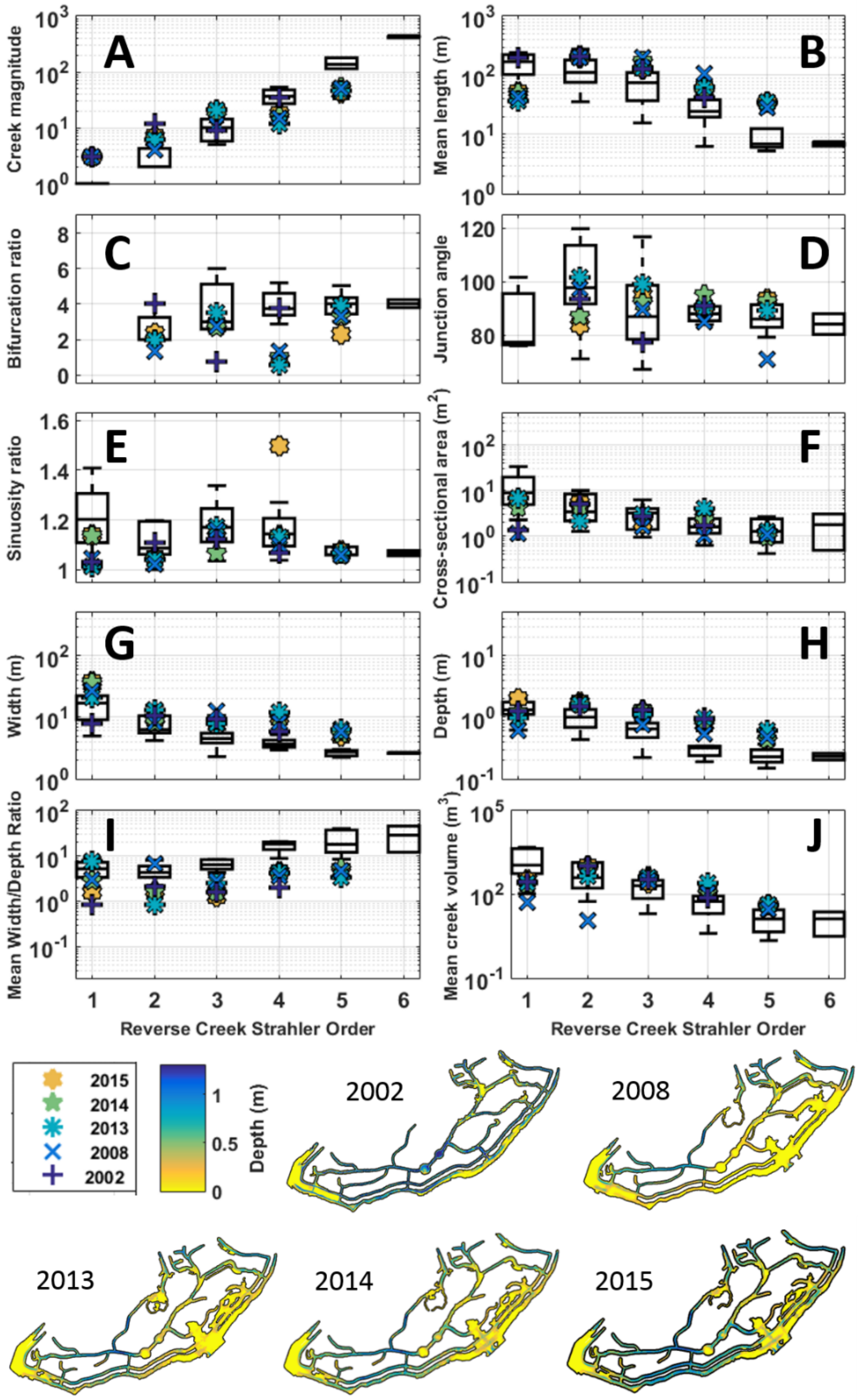
Appendix H19: As Appendix H1 but for Welwick 2007



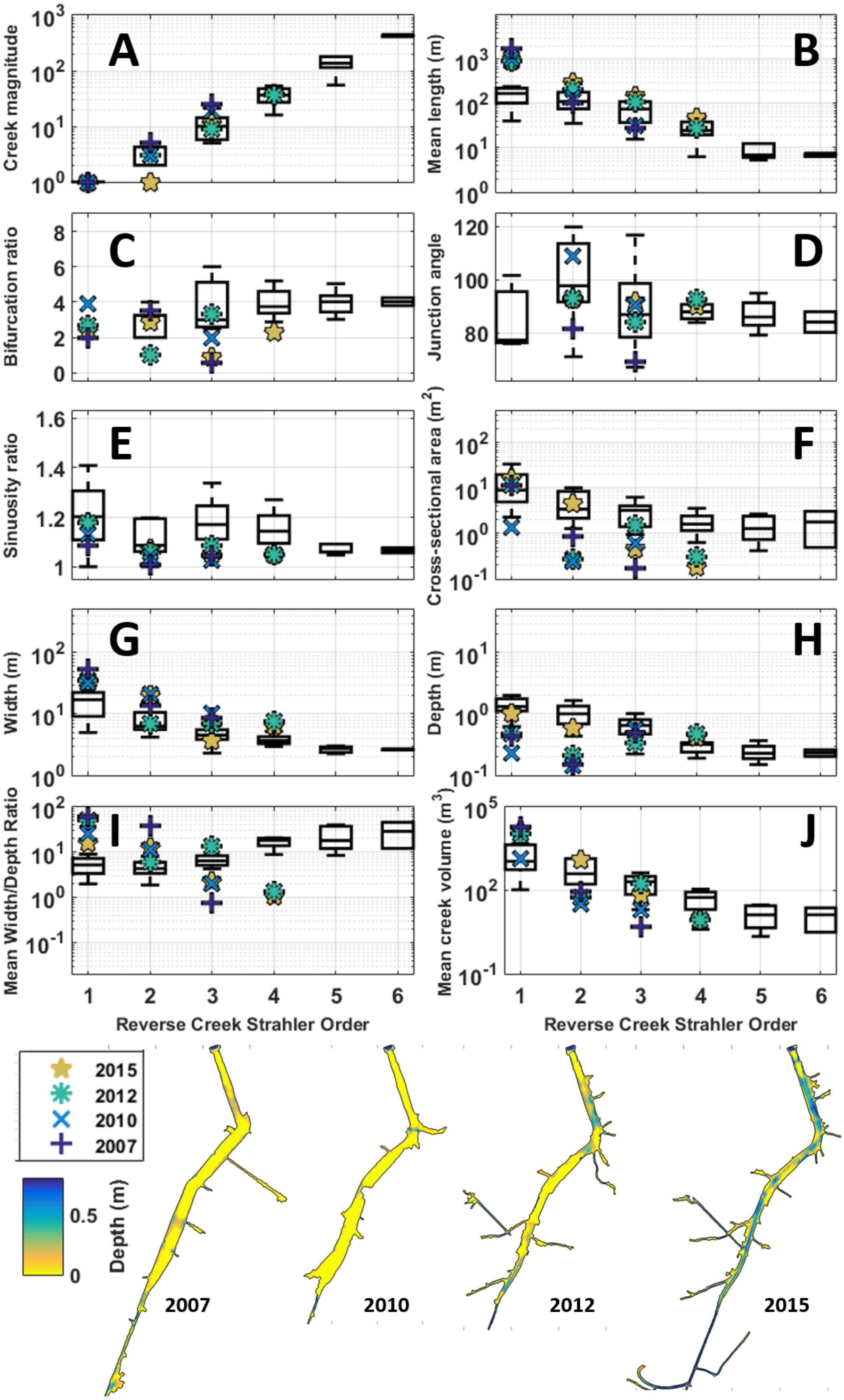
Appendix H 20: As Appendix H1 but for Welwick 2014

Appendix I : MR *vs.* natural creek morphometry

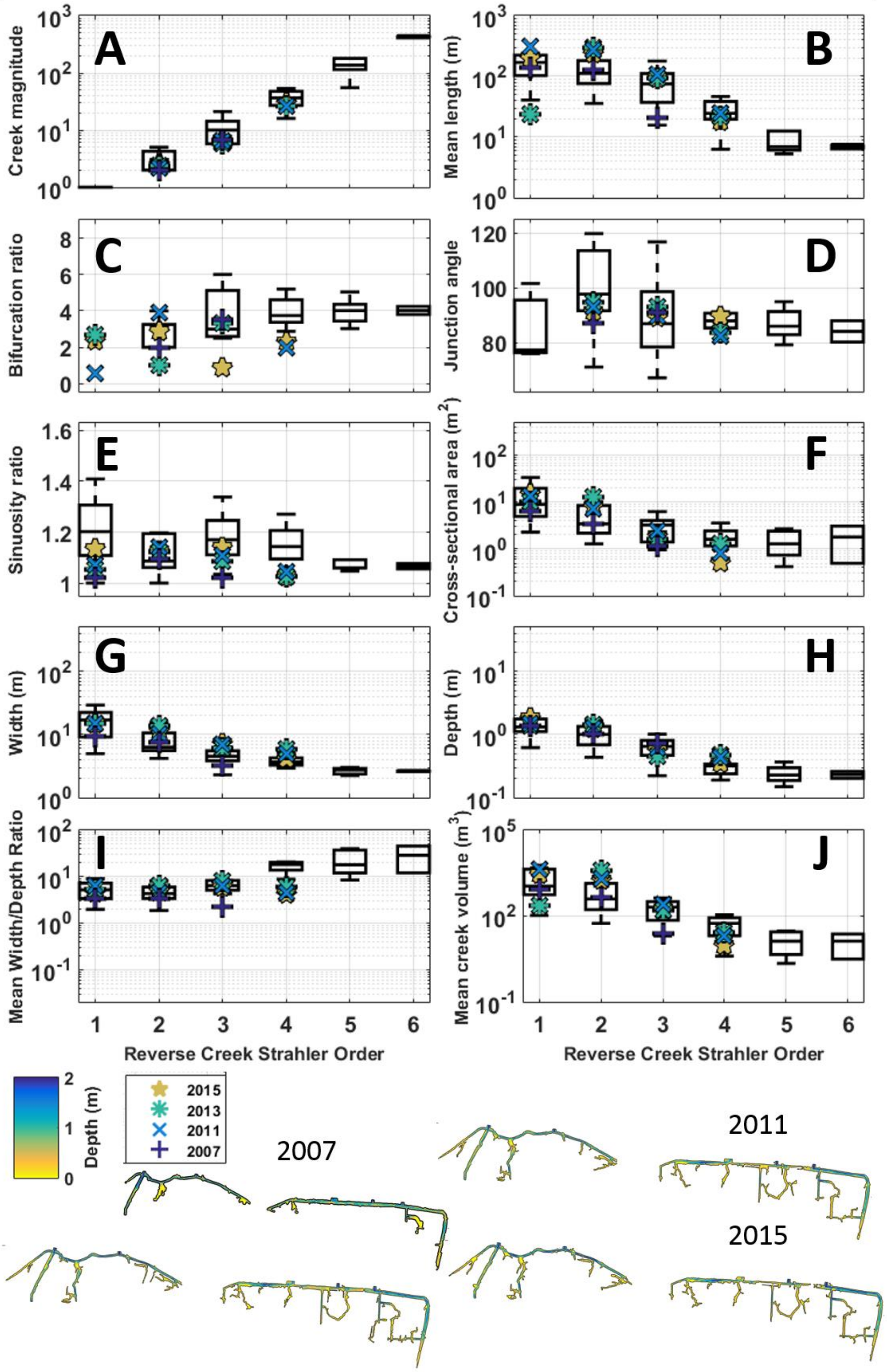
This appendix shows the evolution of MR creek morphometric parameters per RS order over the years, plotted against the 95% spread of natural creek parameters.



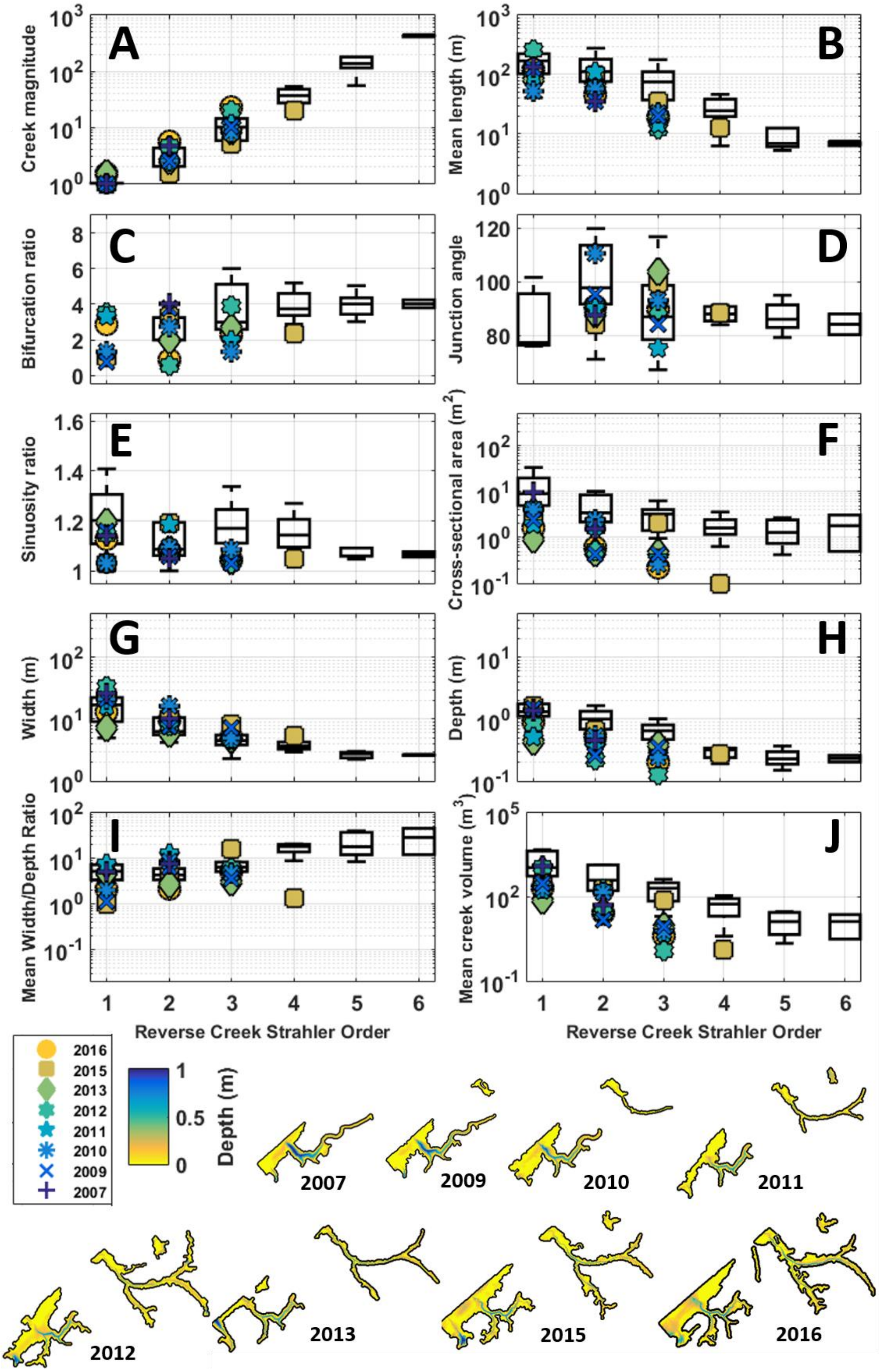
Appendix I1: Morphometry parameters per Reverse Strahler order plotted against the 95% spread of natural creek parameters and creek extent for each available year for Abbots Hall



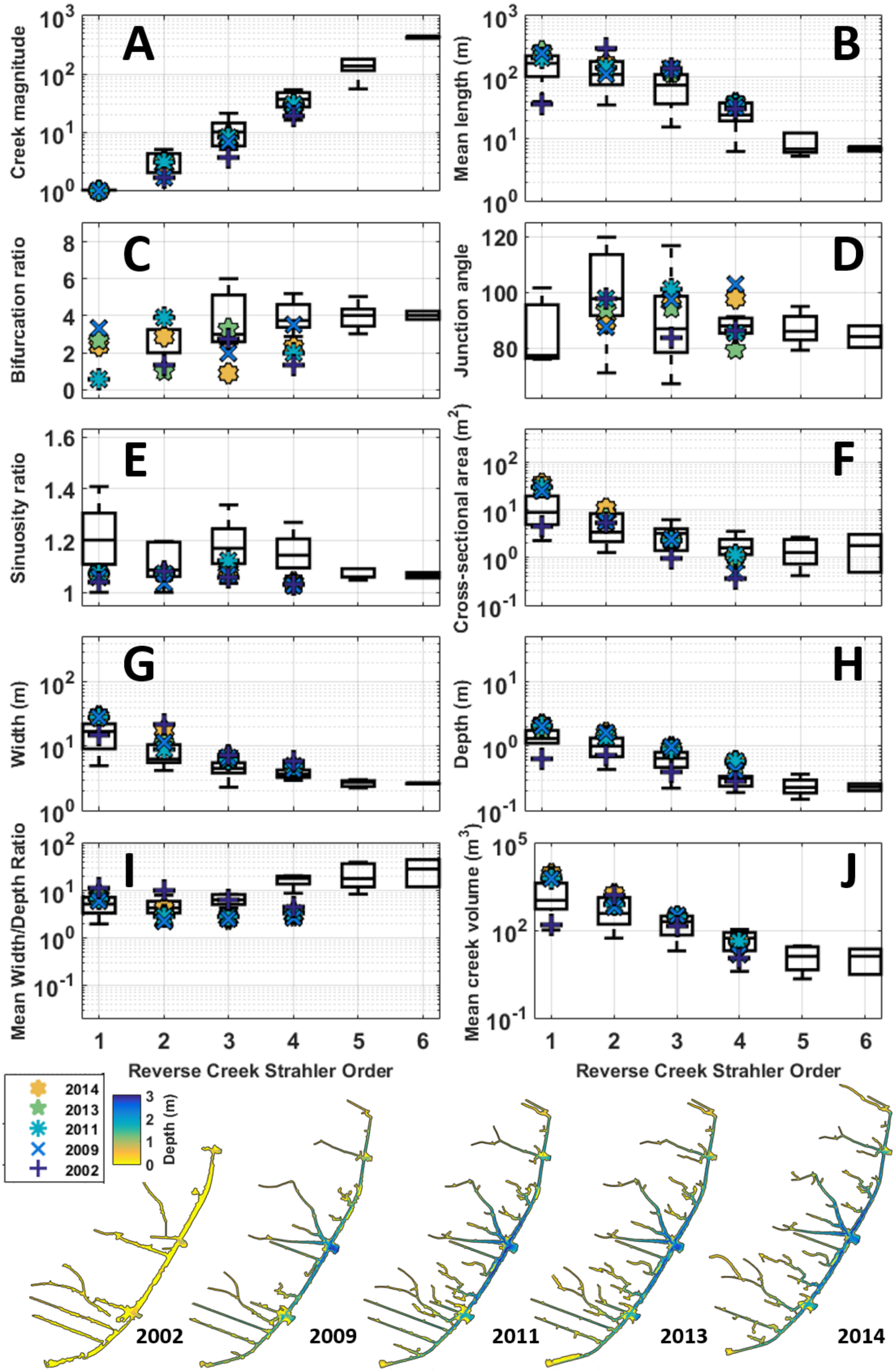
Appendix I2: As Appendix I1 but for Alkborough



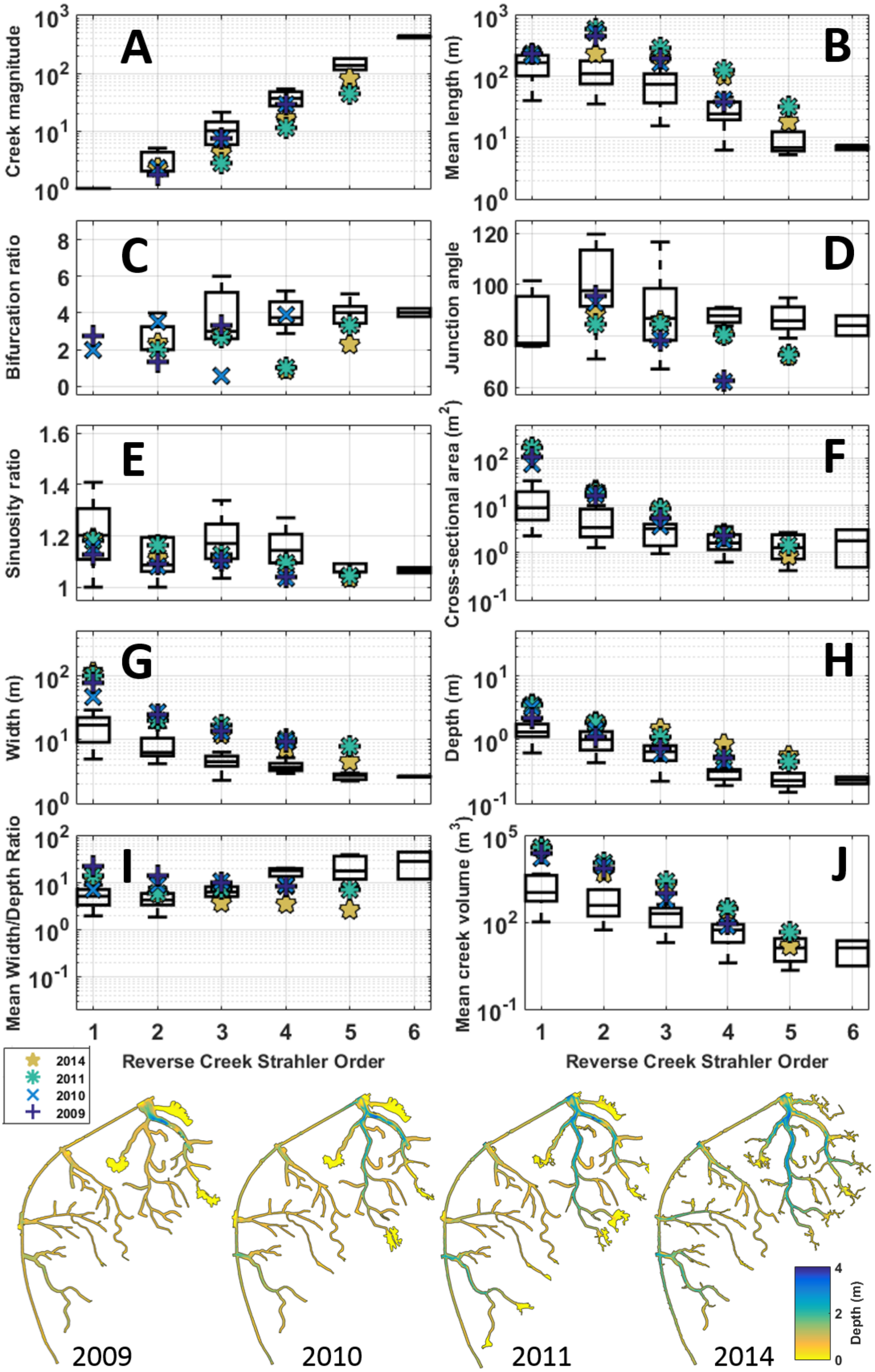
Appendix I3: As Appendix I1 but for Allfleet



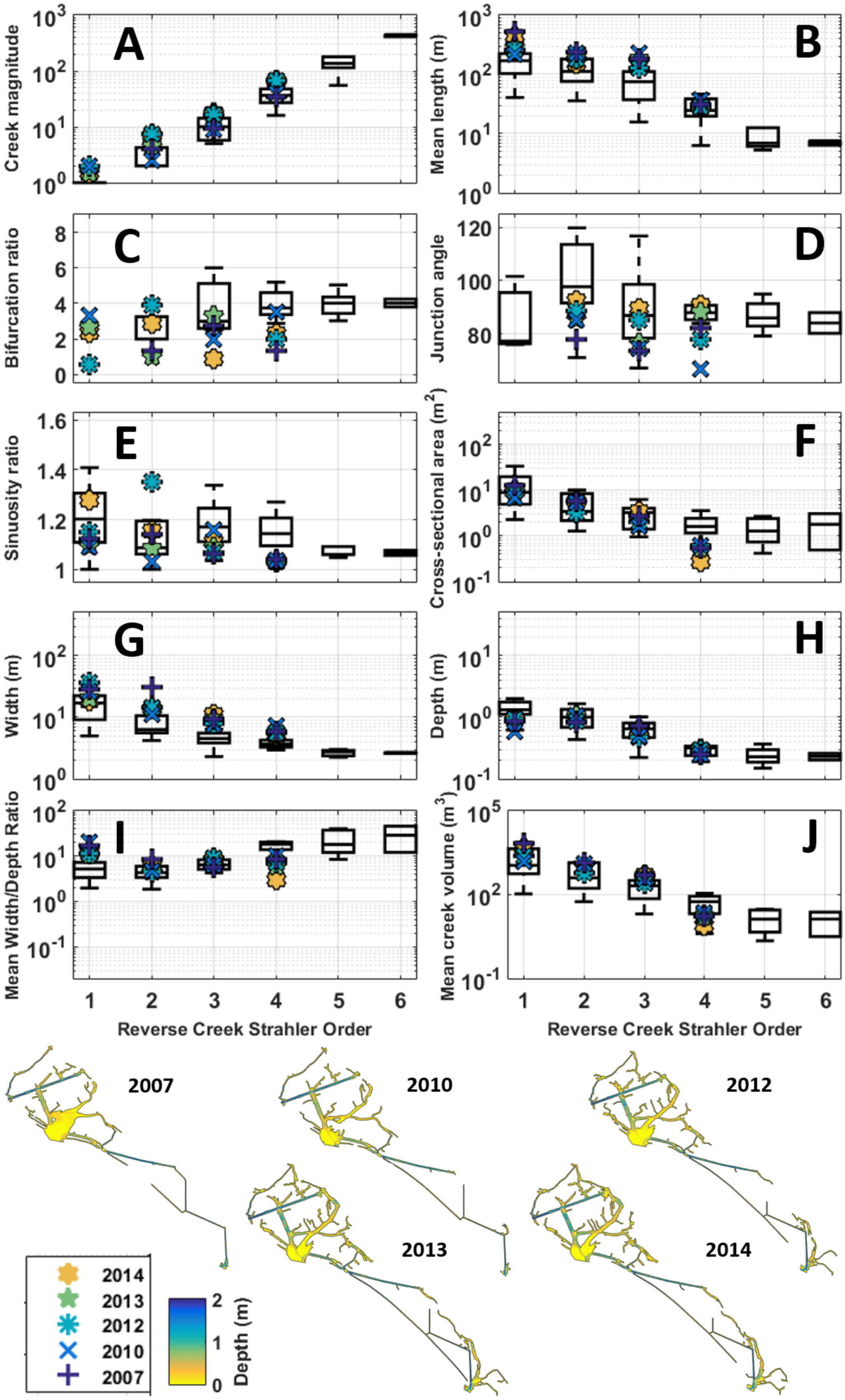
Appendix I4: As Appendix I1 but for Chowder Ness



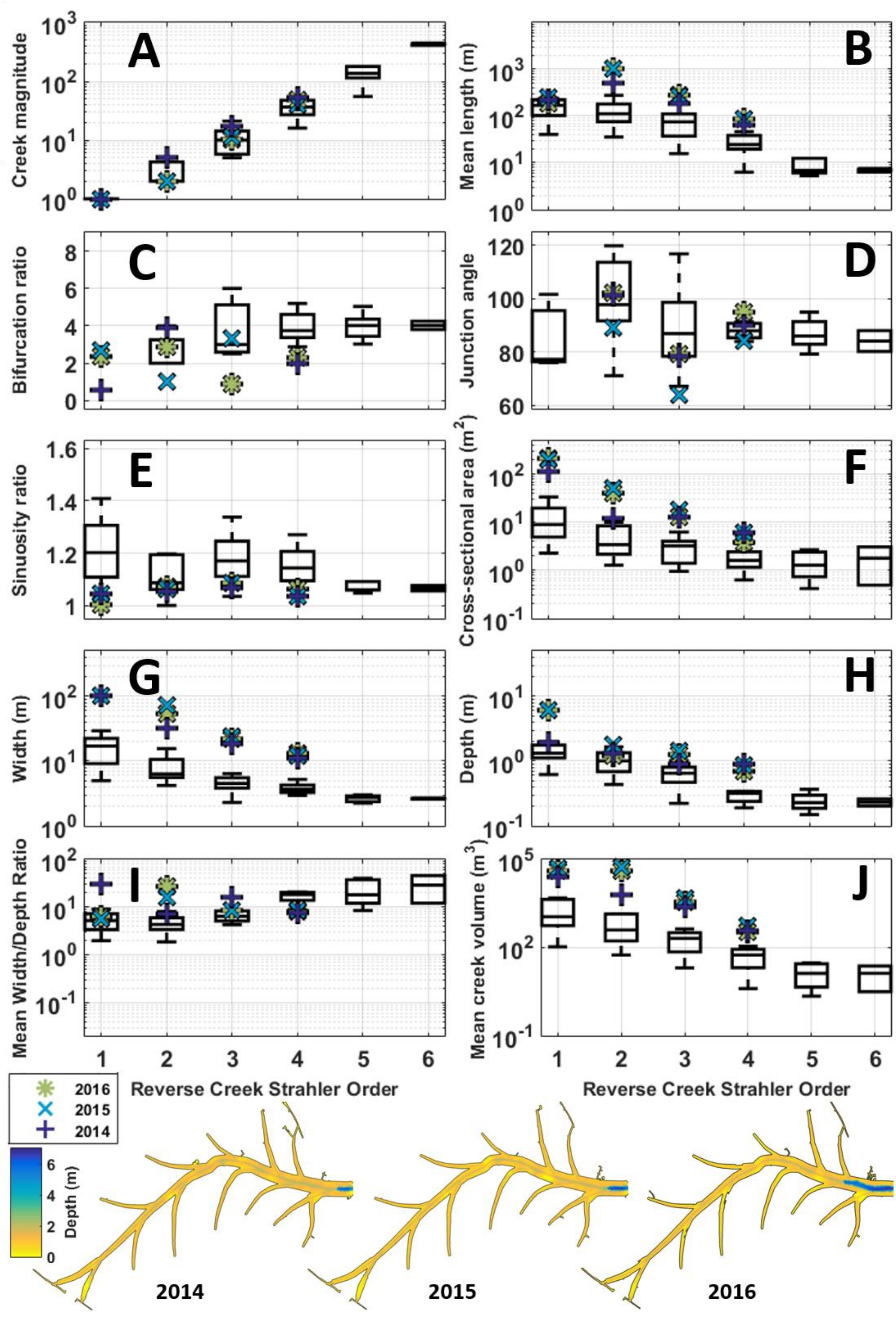
Appendix I5: As Appendix I1 but for Freiston



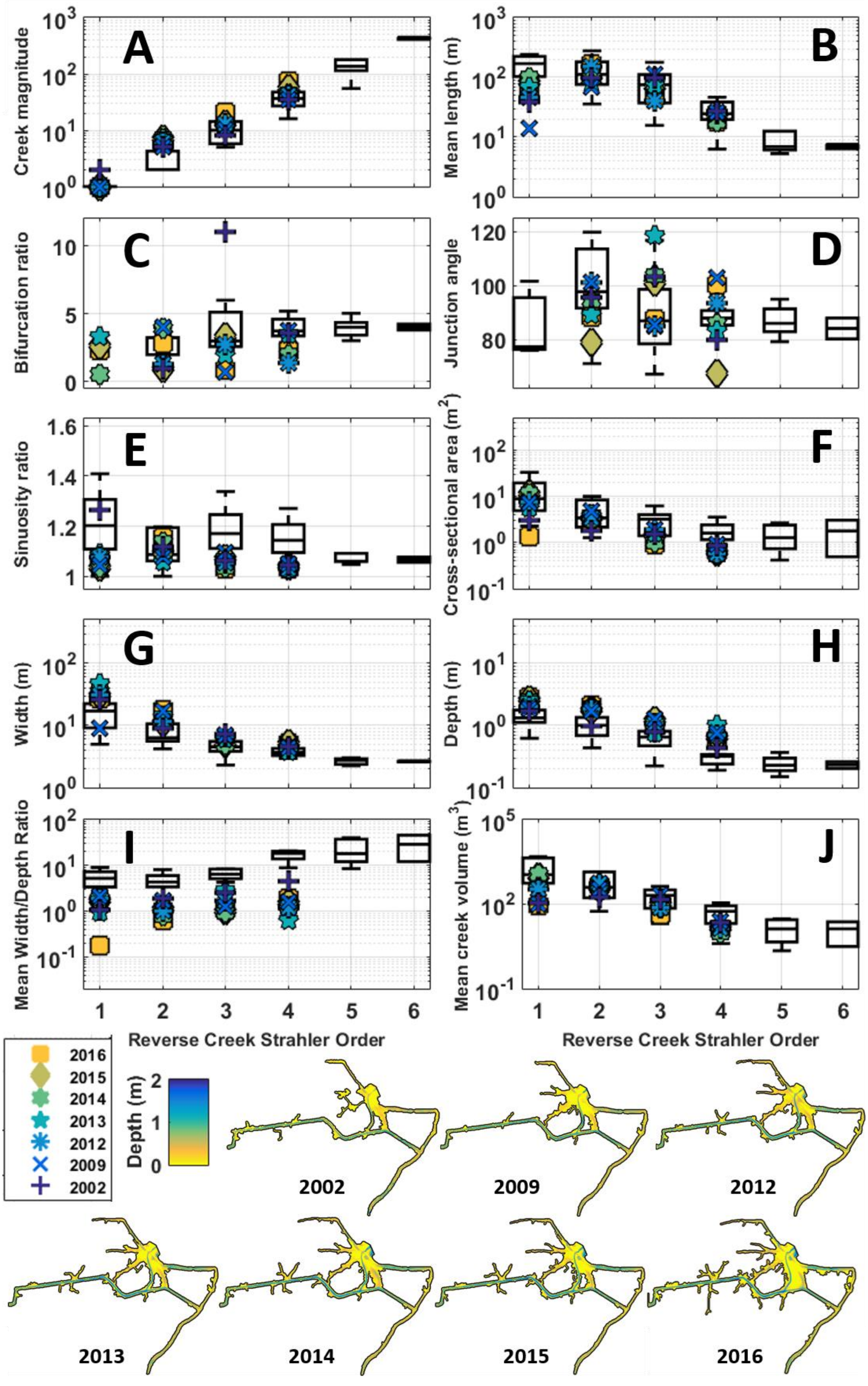
Appendix I6: As Appendix I1 but for Hesketh Out Marsh West



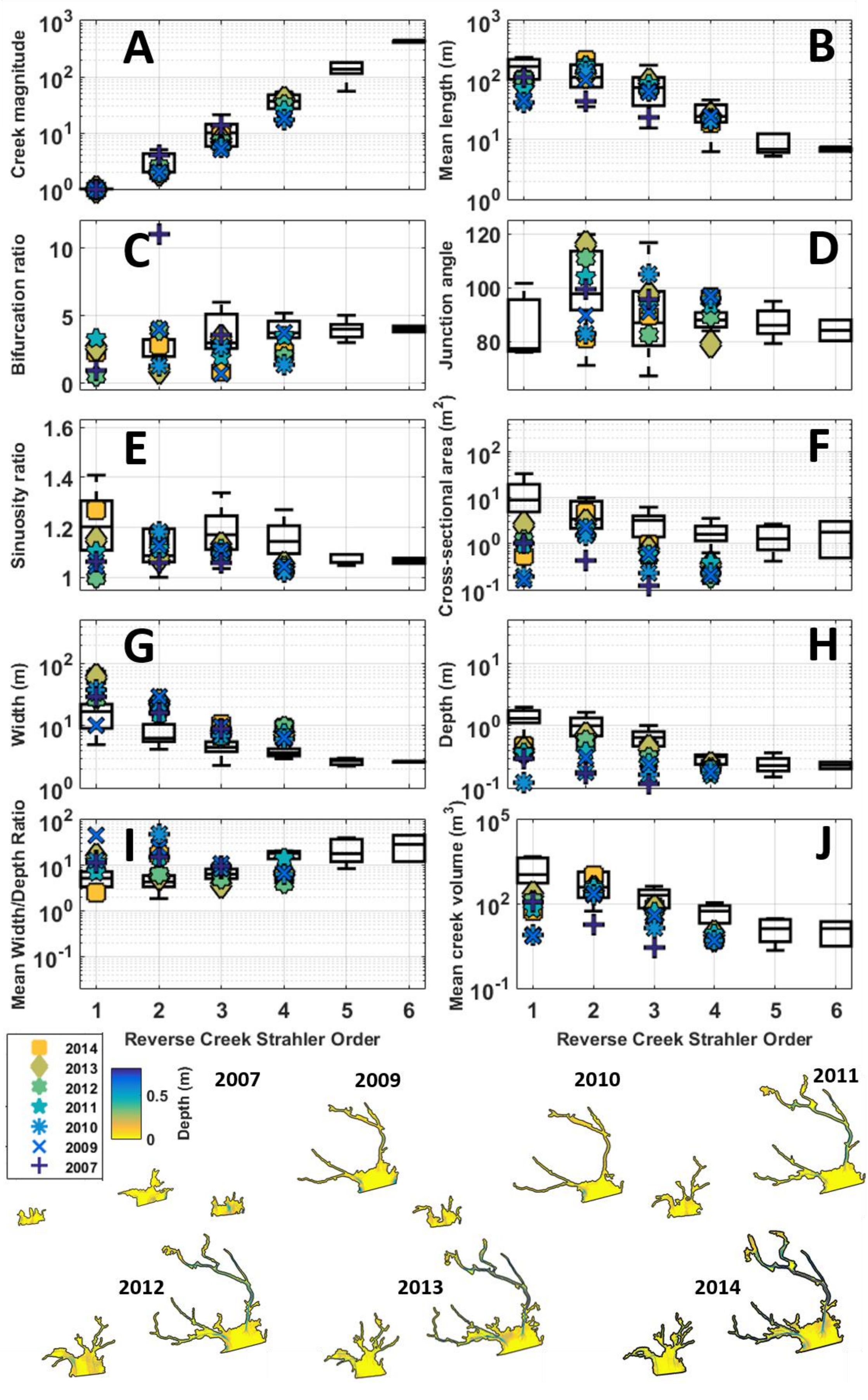
Appendix I7: As Appendix I1 but for Paull Holme Strays



Appendix I8: As Appendix I1 but for Steart



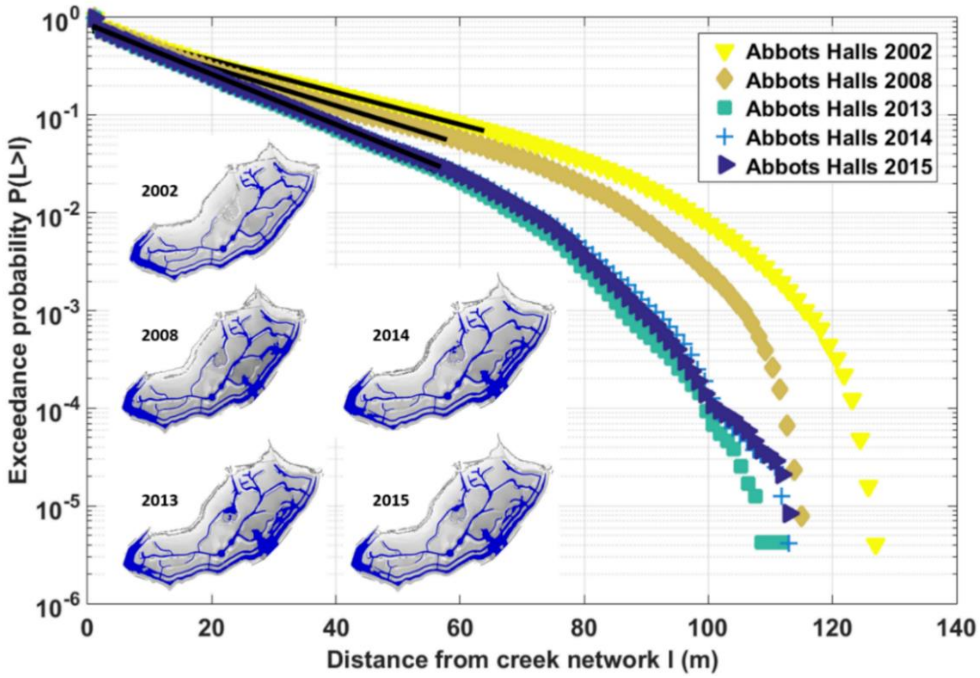
Appendix I9: As Appendix I1 but for Tollesbury



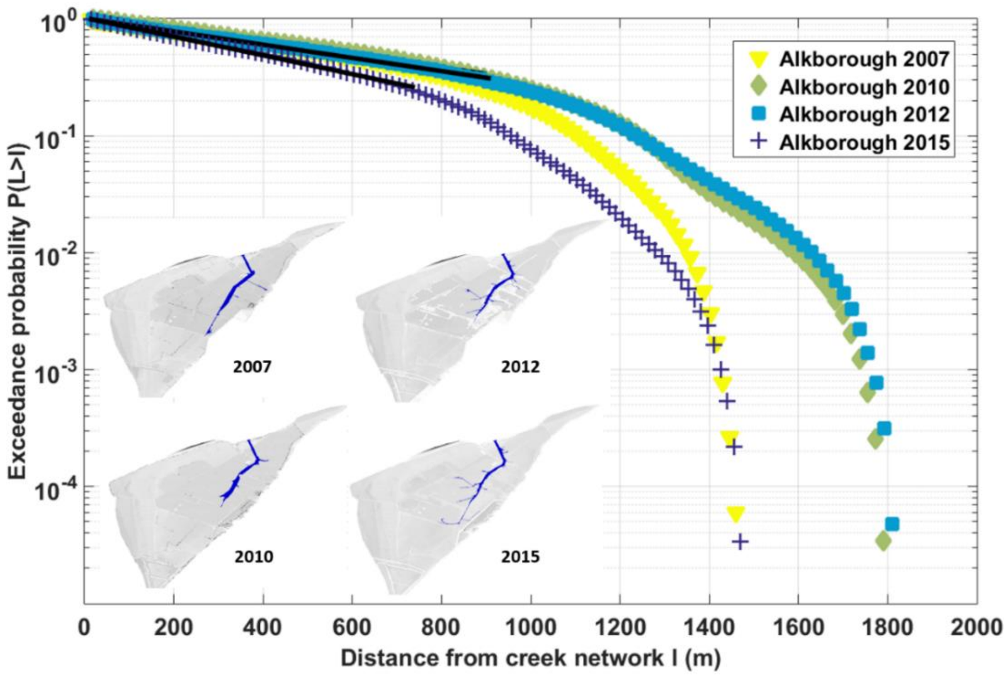
Appendix I10: As Appendix I1 but for Welwick

Appendix J : MR overmarsh path length

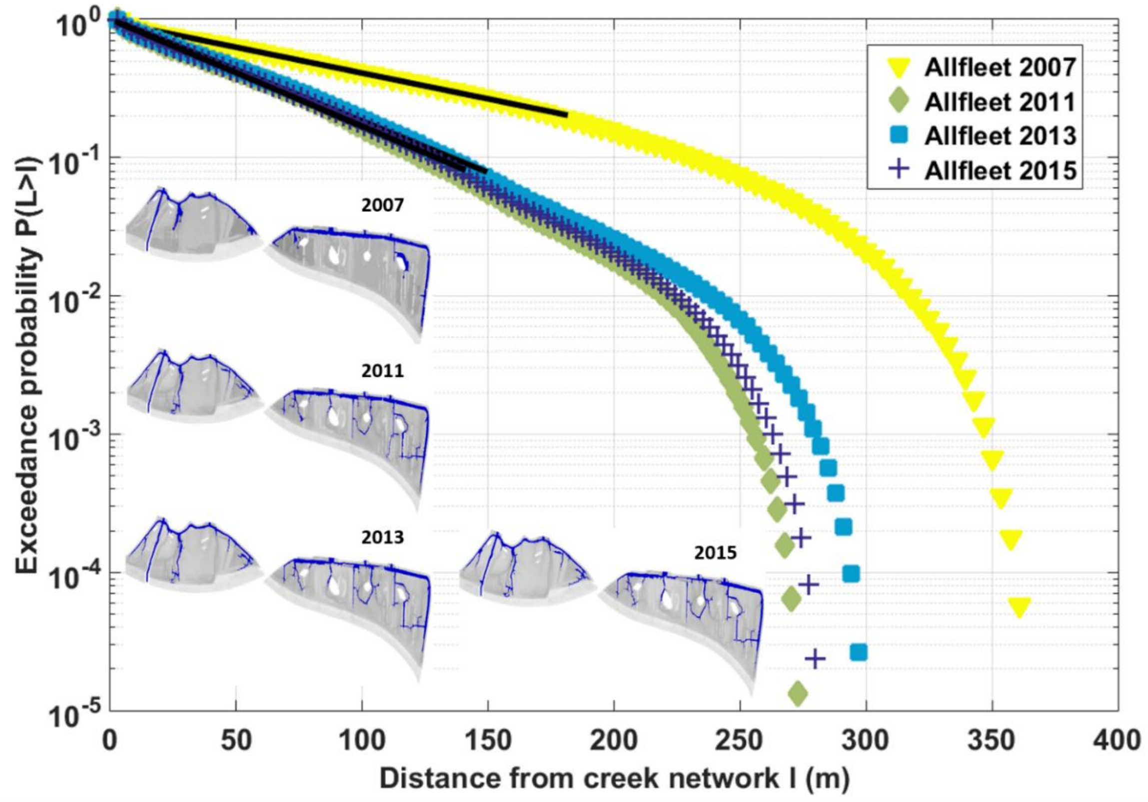
This appendix shows the evolution of the MR creek extent, unchanneled length and overmarsh path length over the years for all sites.



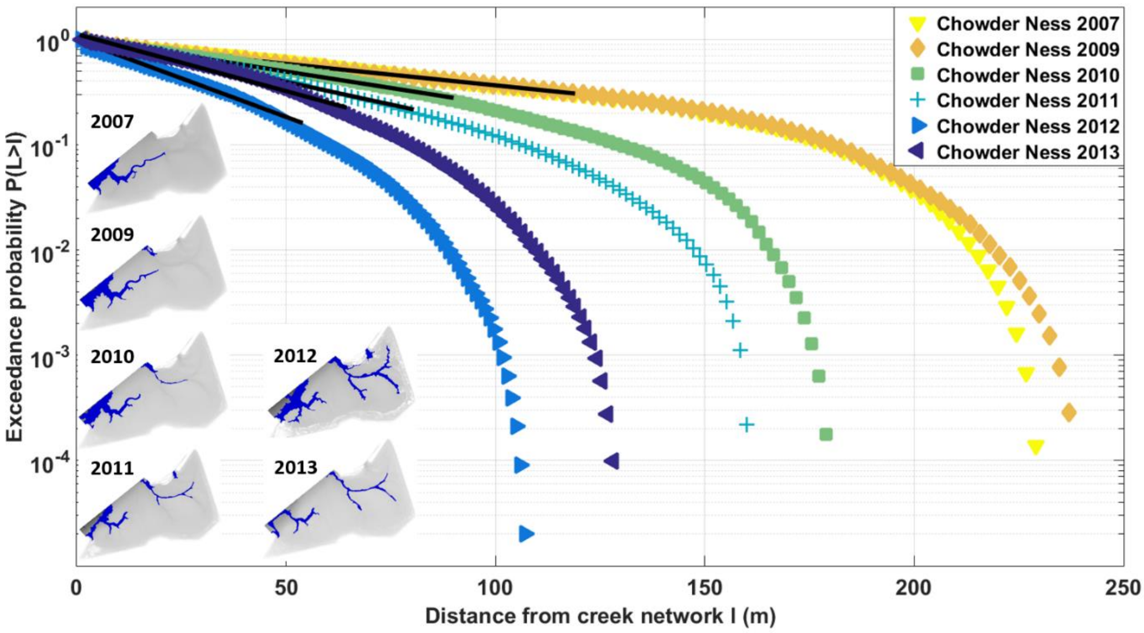
Appendix J1: Creek extent (shown in dark blue on the maps), unchanneled length distribution and overmarsh path length evolution for each available year for Abbots Hall



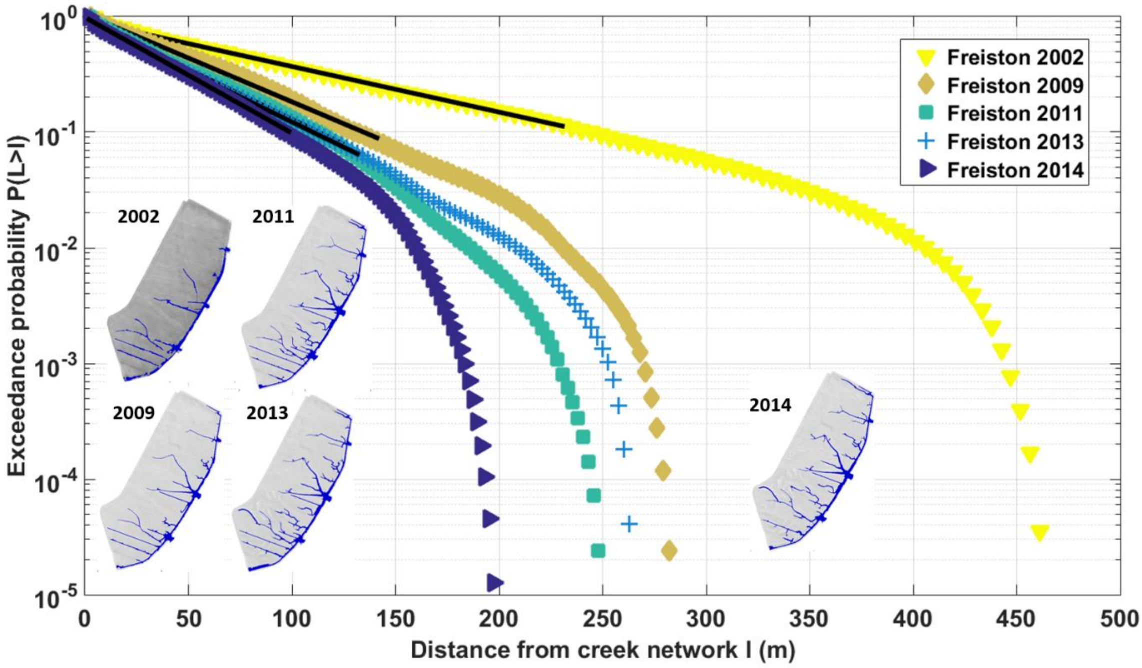
Appendix J2: As Appendix J1 but for Alkborough



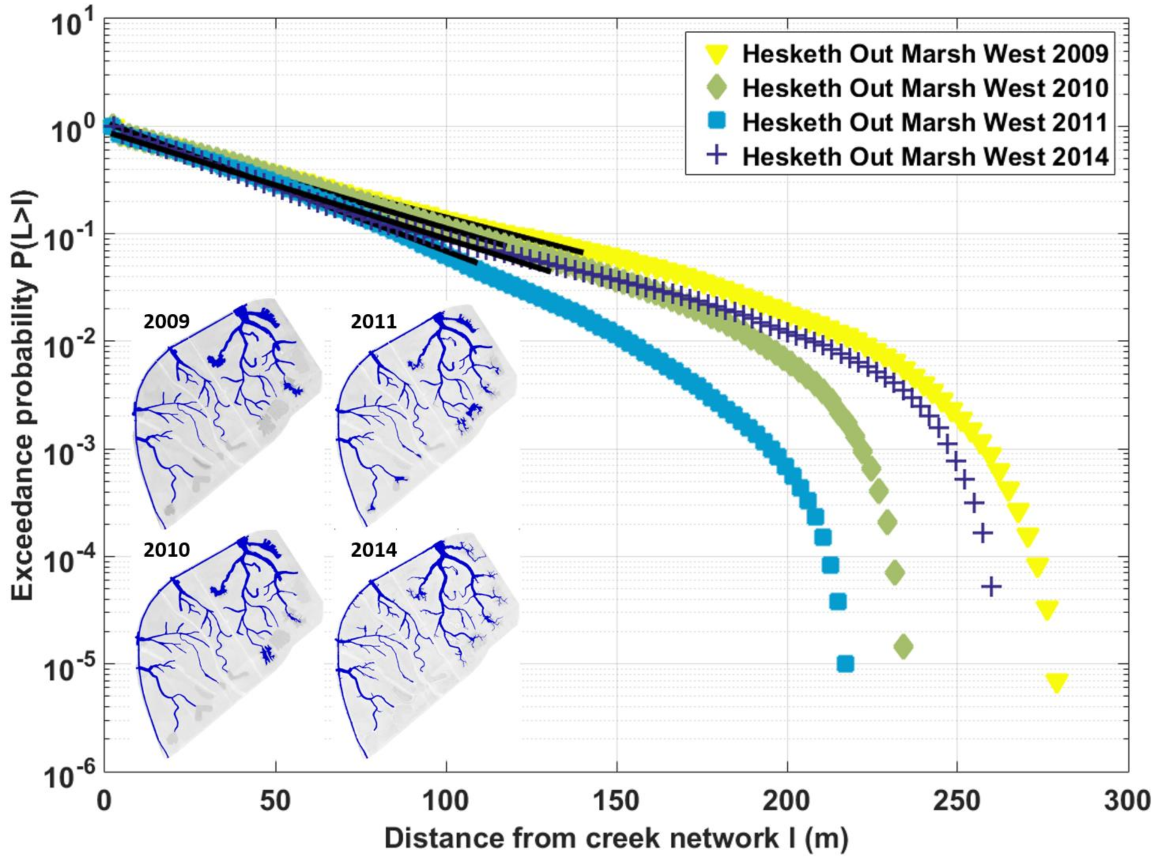
Appendix J3: As Appendix J1 but for Allfleet



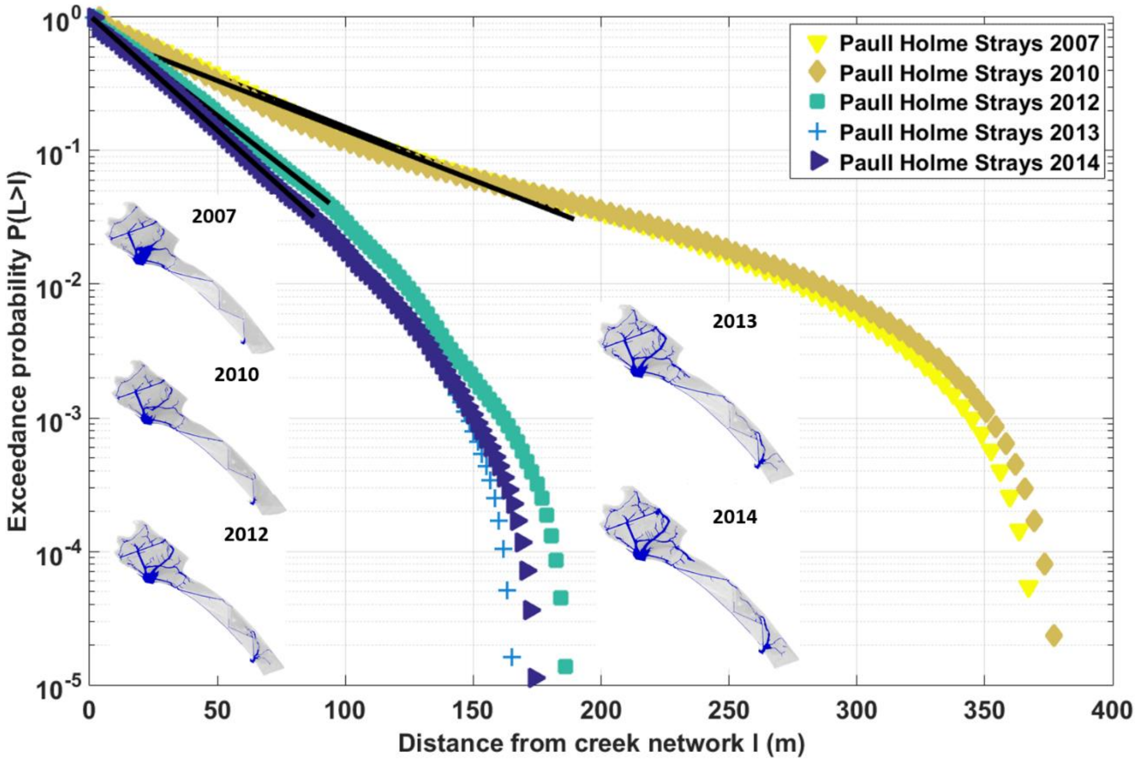
Appendix J4: As Appendix J1 but for Chowder Ness



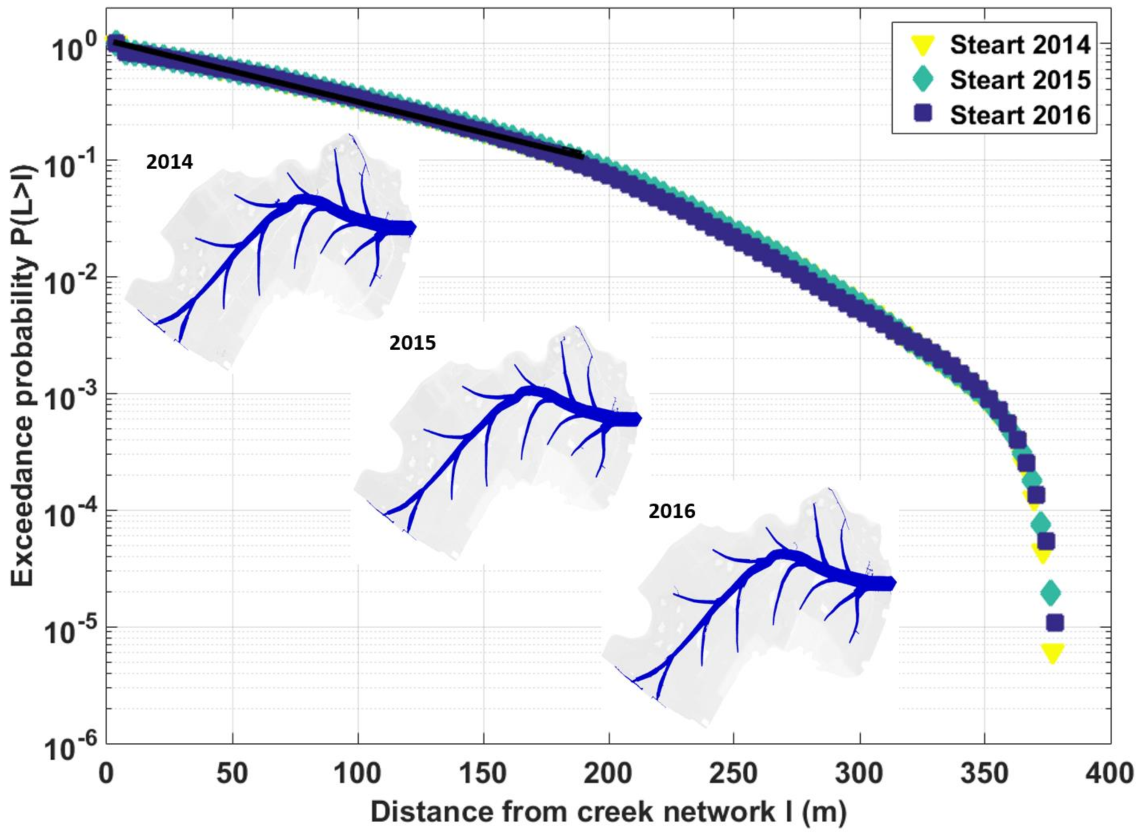
Appendix J5: As Appendix J1 but for Freiston



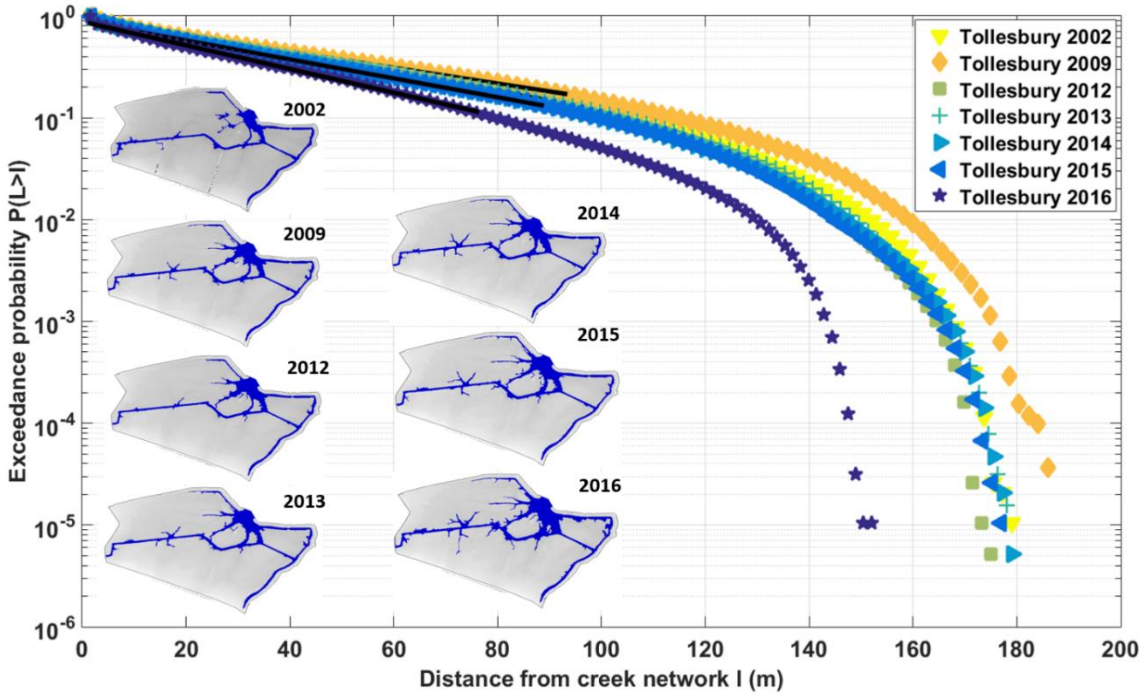
Appendix J6: As Appendix J1 but for Hesketh Out Marsh West



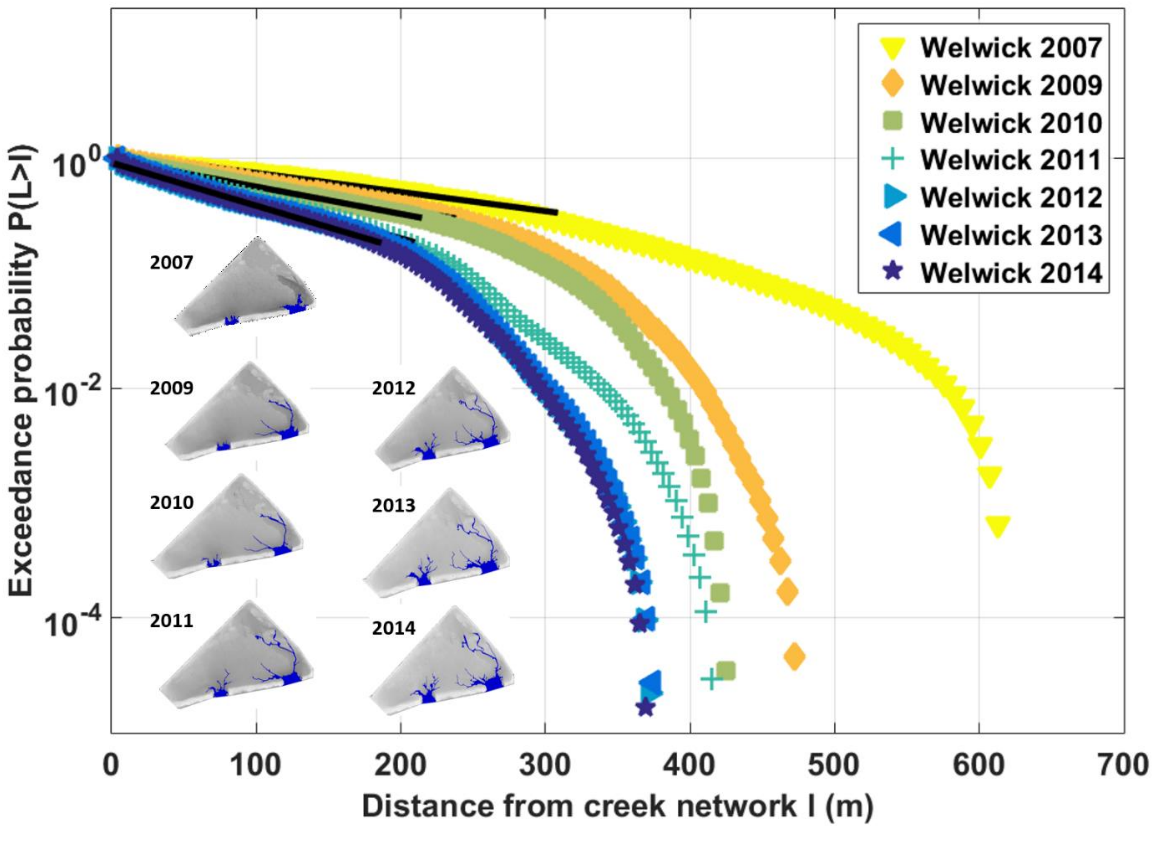
Appendix J7: As Appendix J1 but for Paull Holme Strays



Appendix J8: As Appendix J1 but for Steart



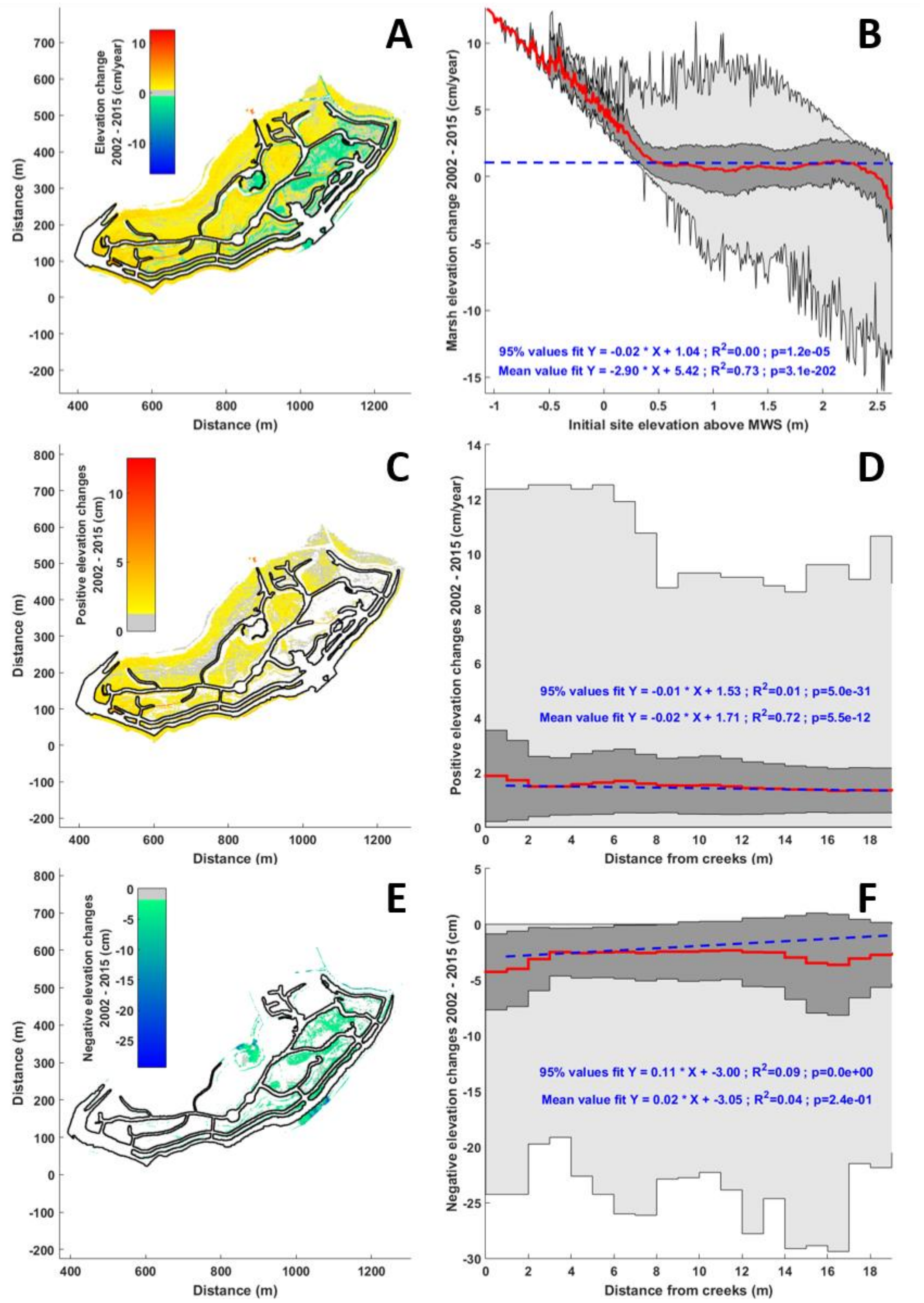
Appendix J9: As Appendix J1 but for Tollesbury



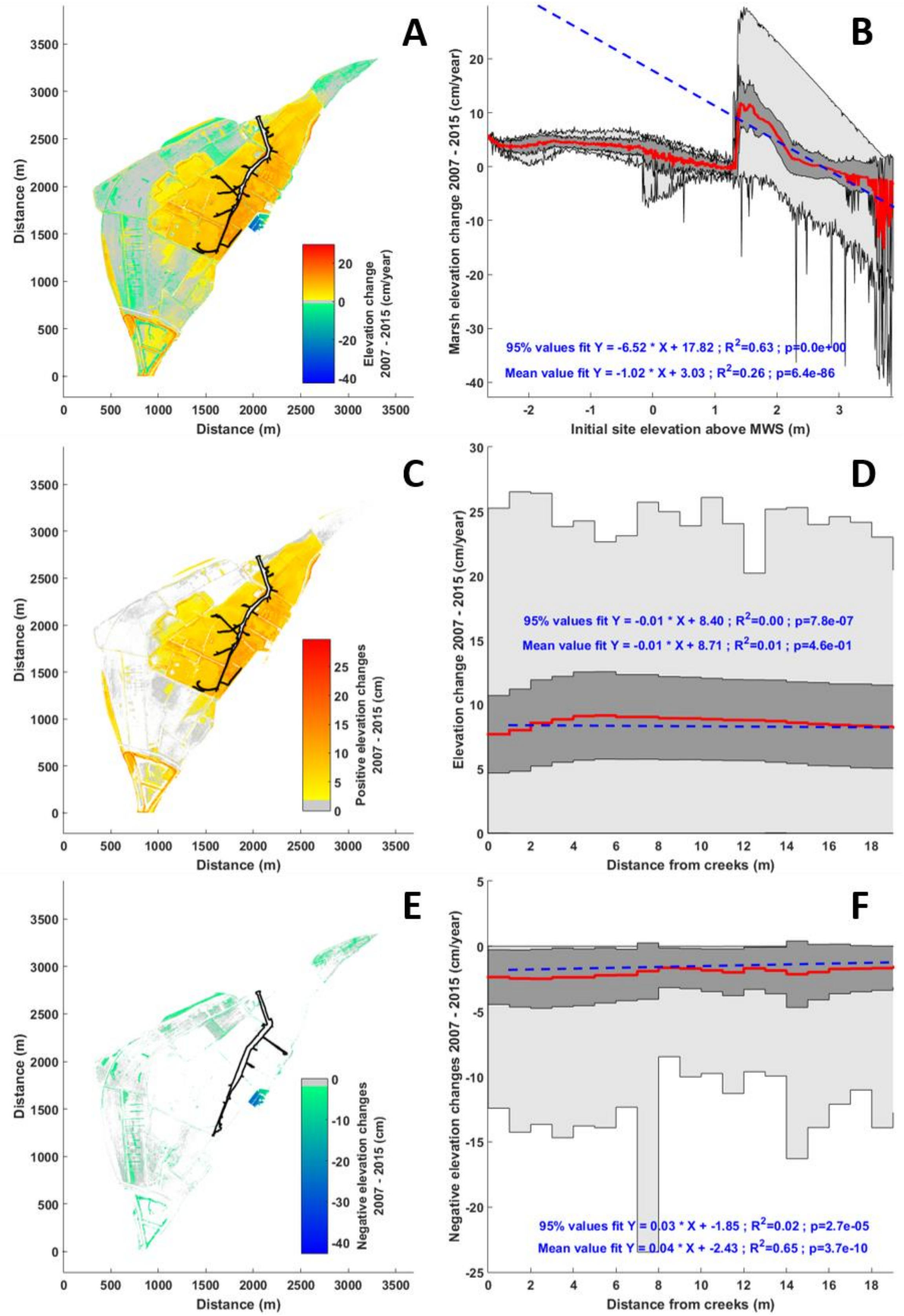
Appendix J10: As Appendix J1 but for Welwick

Appendix K : MR creek forming processes

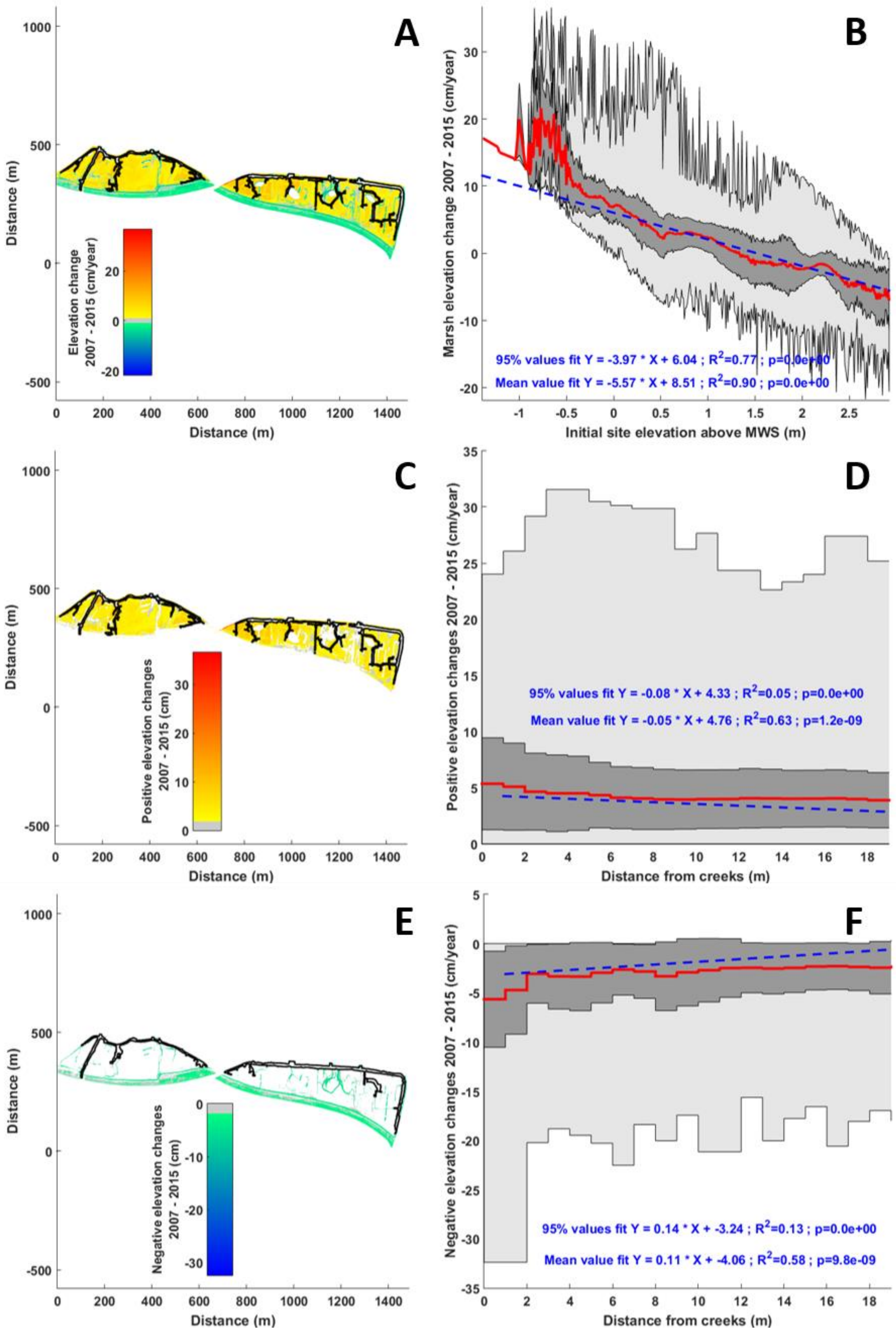
This appendix shows the elevation gains and losses of all MR sites between the first and the last year considered, and correlates them to the initial site elevation and distance to creeks to analyse creek forming processes. The effect of creek proximity on the creek network is evaluated up to 20 m away from the creeks to reduce the impact of site elevation or of multiple creek influence on the results. Positive elevation changes are correlated to the most recent creek extent and negative elevation changes to the initial creek extent.



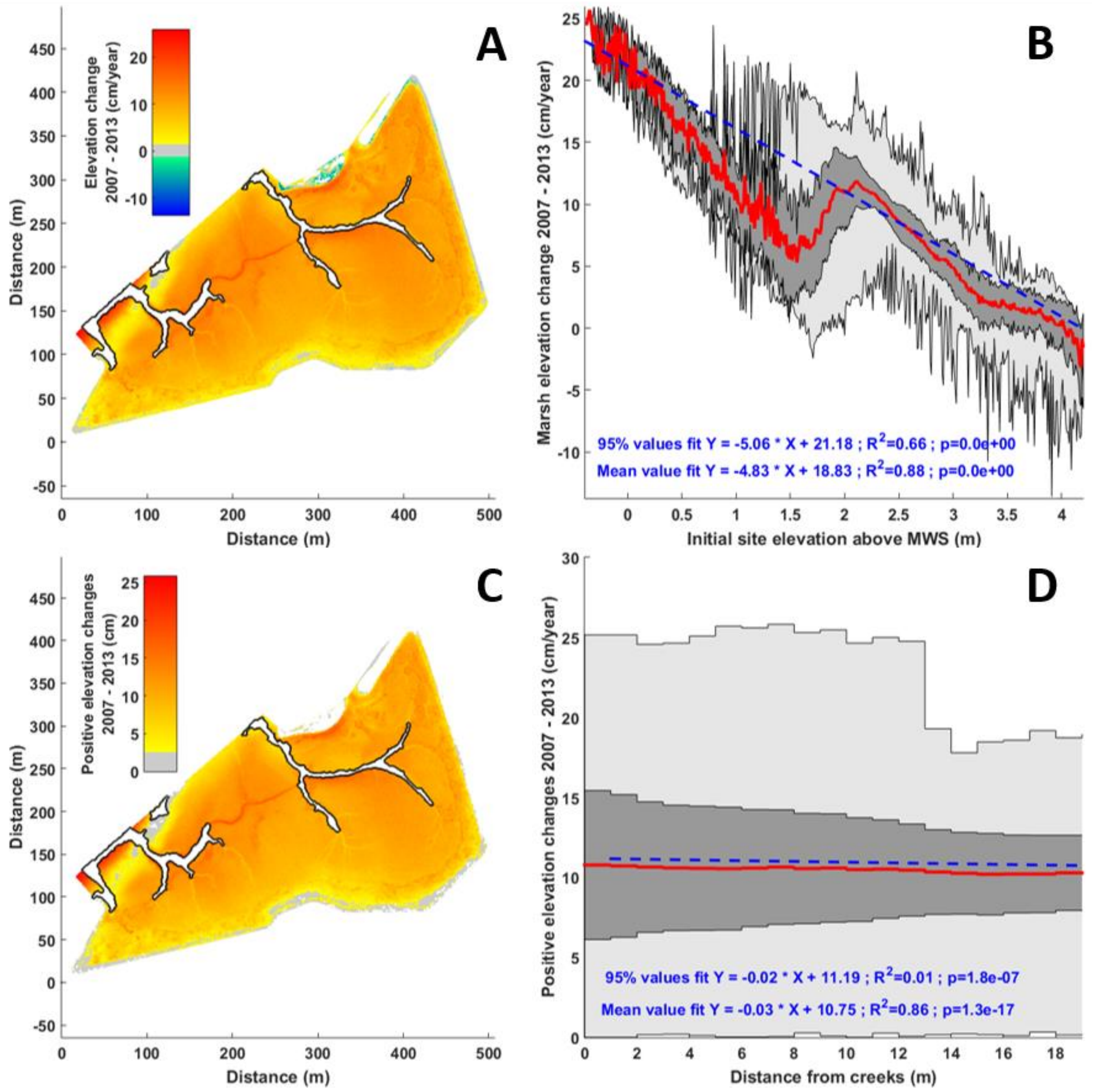
Appendix K1: Linear correlation tests at Abbots Hall for the 95% data spread and for the mean values between (A) marsh elevation changes versus site elevation; (B) marsh elevation gains versus distance up to 20 m to the final creek network; and (C) marsh elevation losses versus distance up to 20 m to the initial creek network. Red line: mean value of elevation change; dark grey envelope: 95% spread around the mean value; light grey envelope: total data spread. The marsh elevation changes correspond to the last versus first lidar dataset, divided by the number of years between the two.



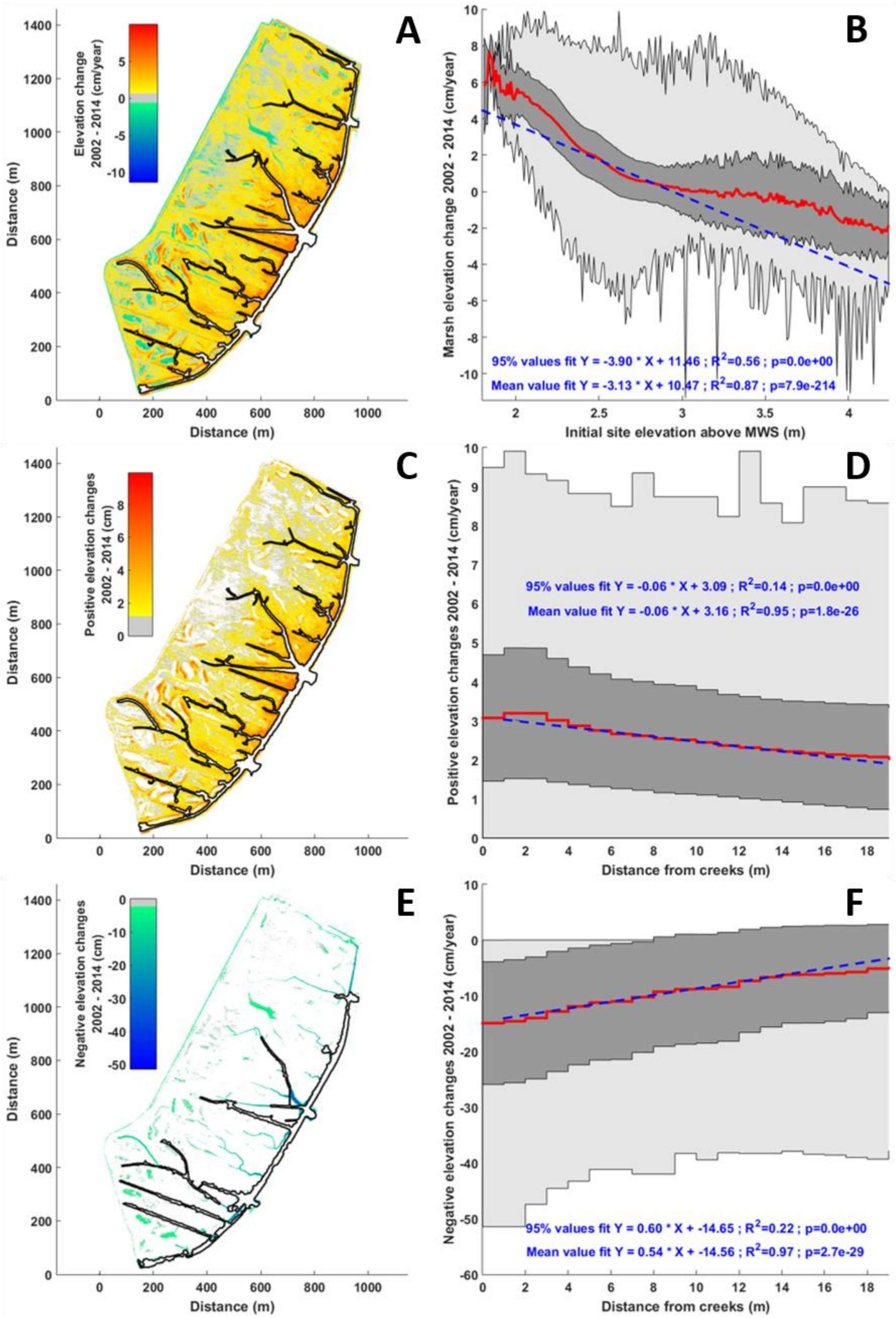
Appendix K2: As Appendix K1 but for Alkborough



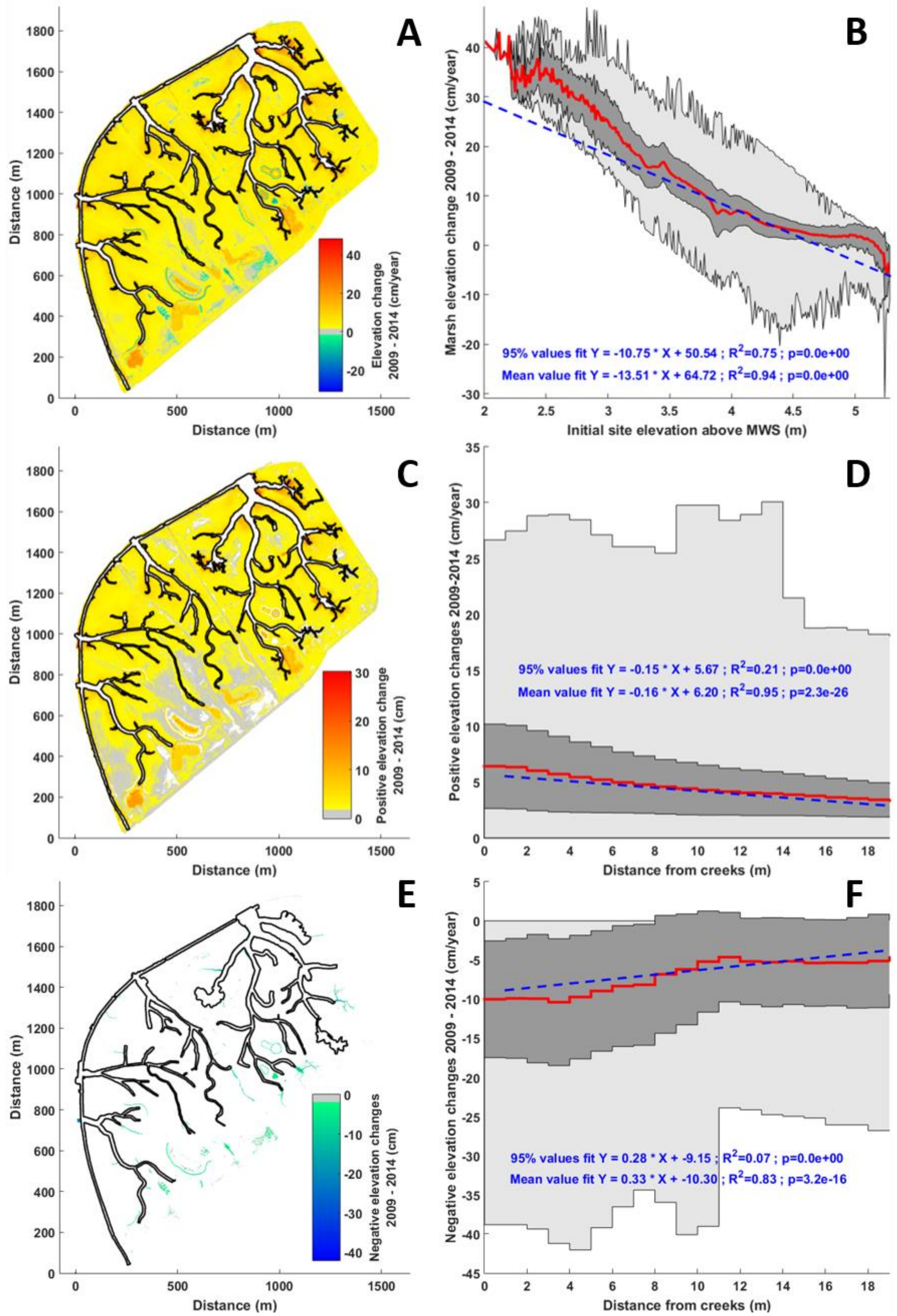
Appendix K3: As Appendix K1 but for Allfleet



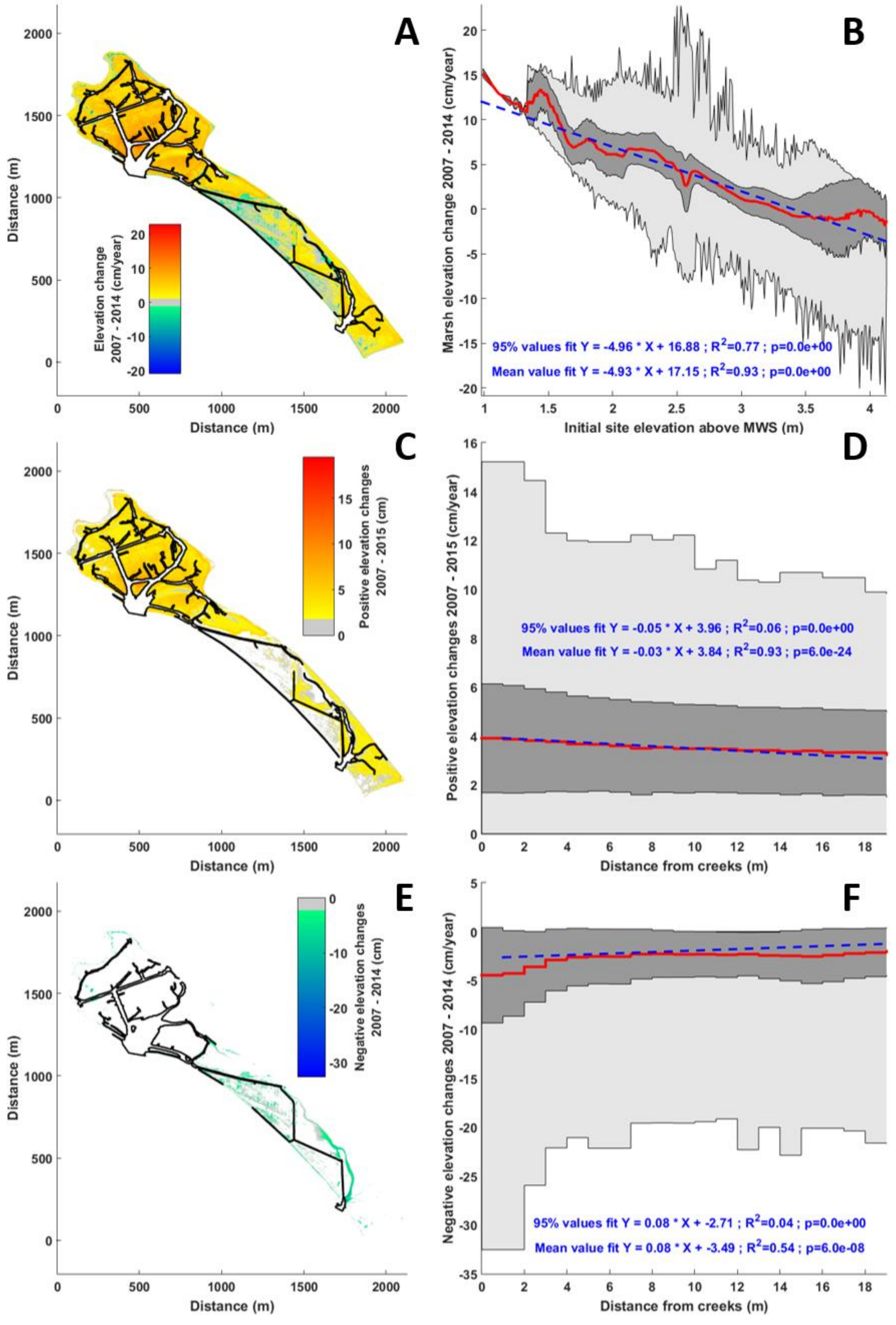
Appendix K4: As Appendix K1 but for Chowder Ness (no visible elevation loss)



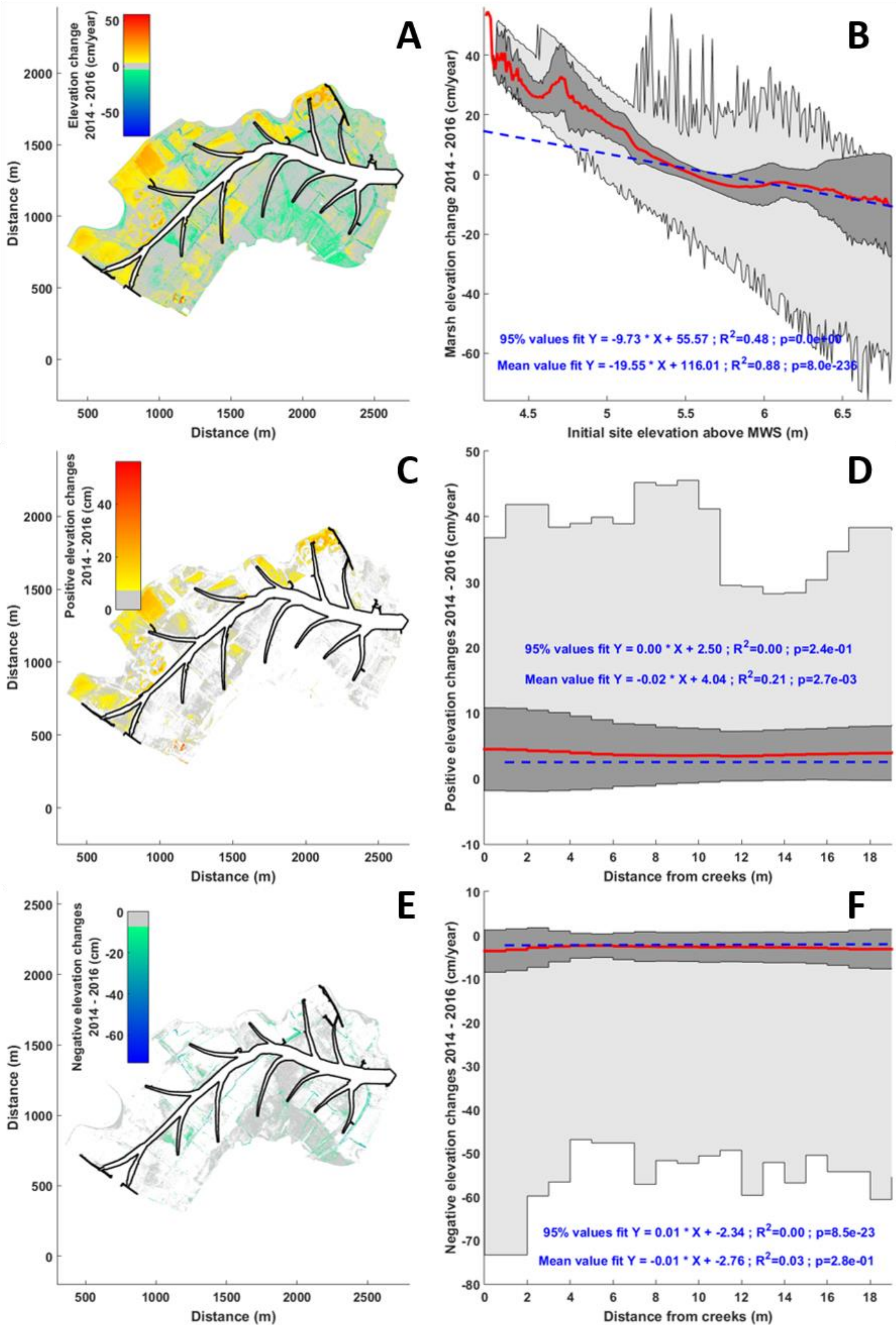
Appendix K5: As Appendix K1 but for Freiston



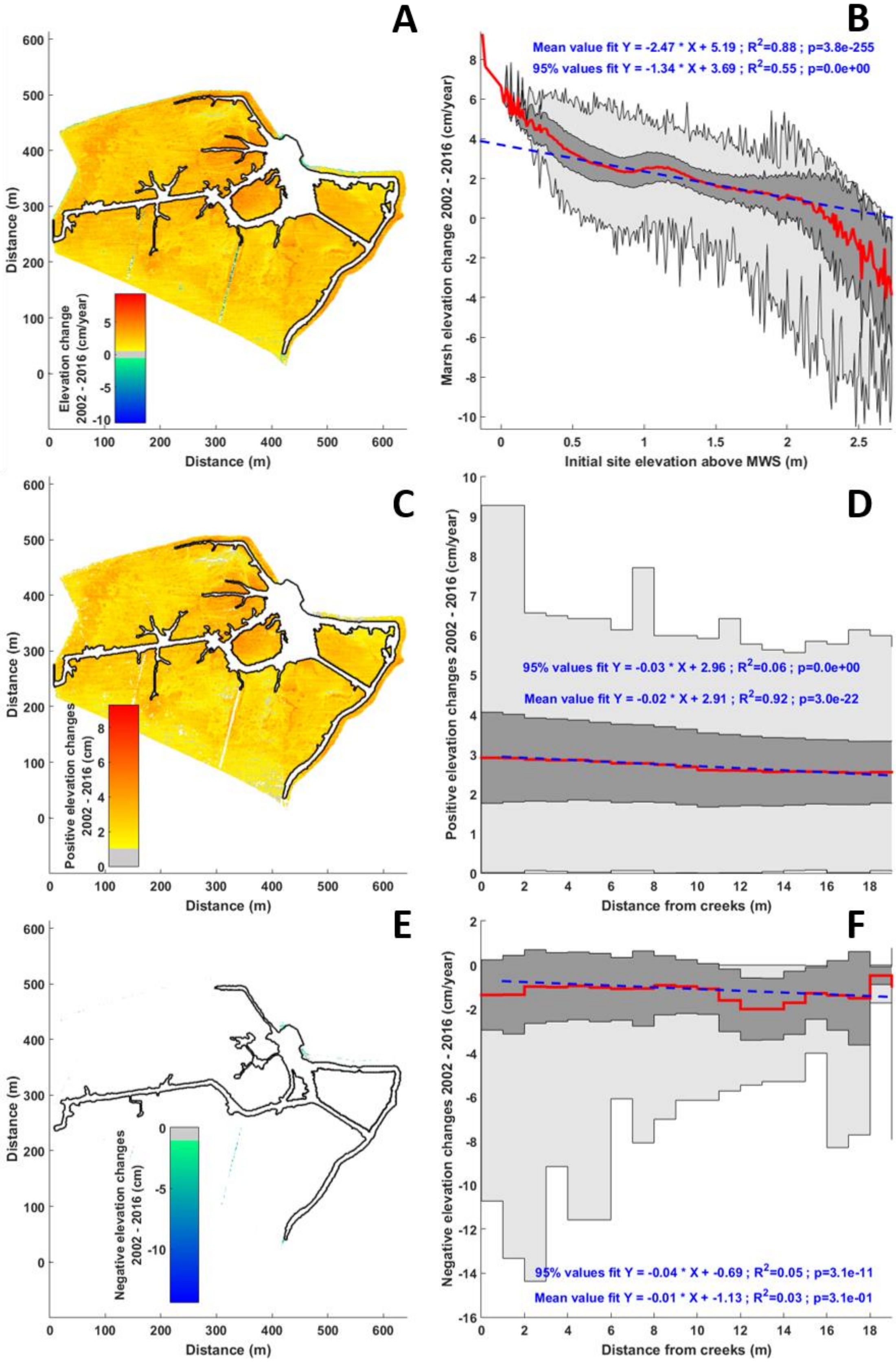
Appendix K6: As Appendix K1 but for Hesketh Out Marsh West



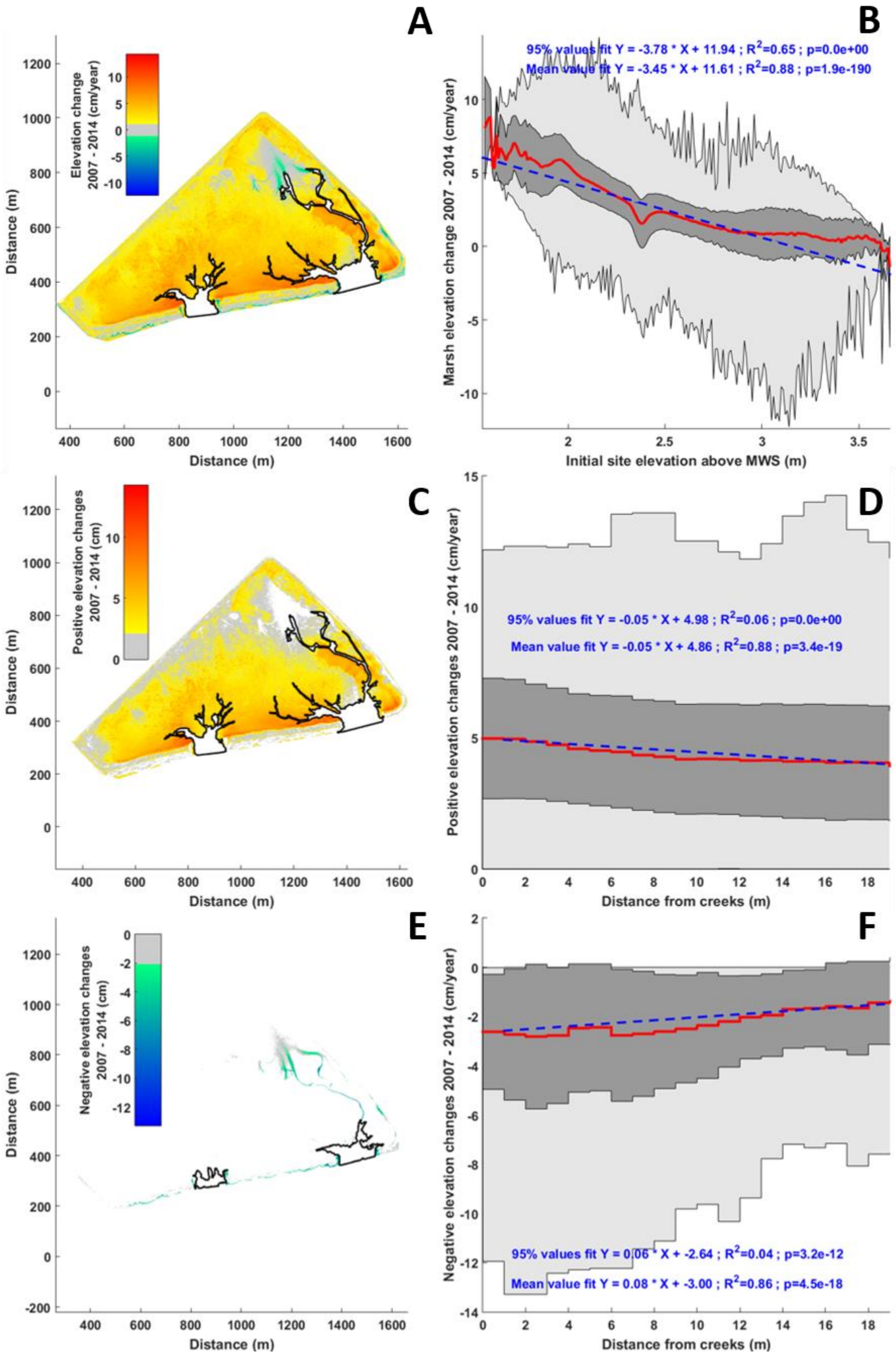
Appendix K7: As Appendix K1 but for Paull Holme Strays



Appendix K8: As Appendix K1 but for Steart



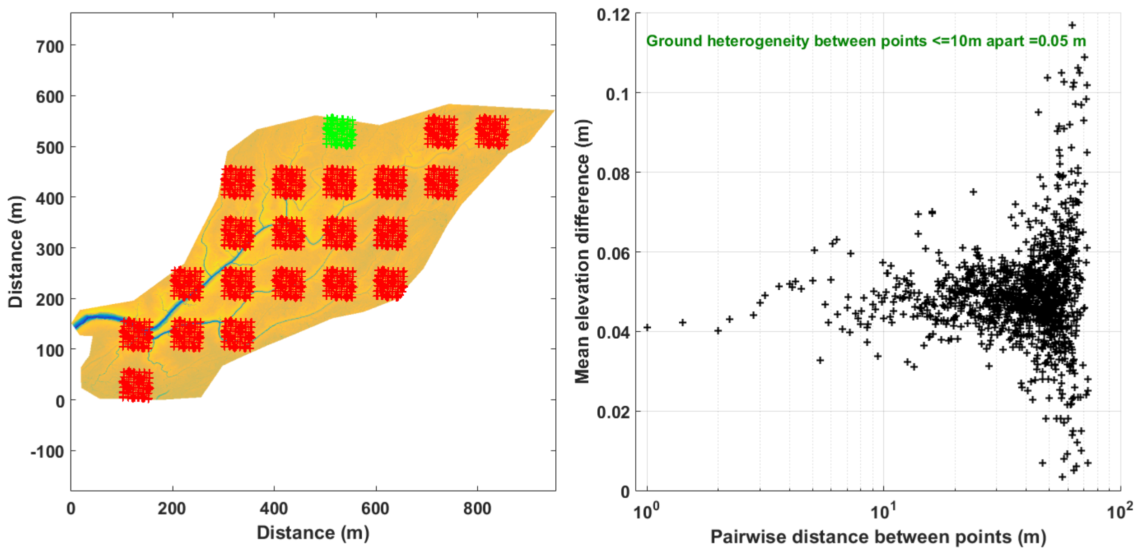
Appendix K9: As Appendix K1 but for Tollesbury



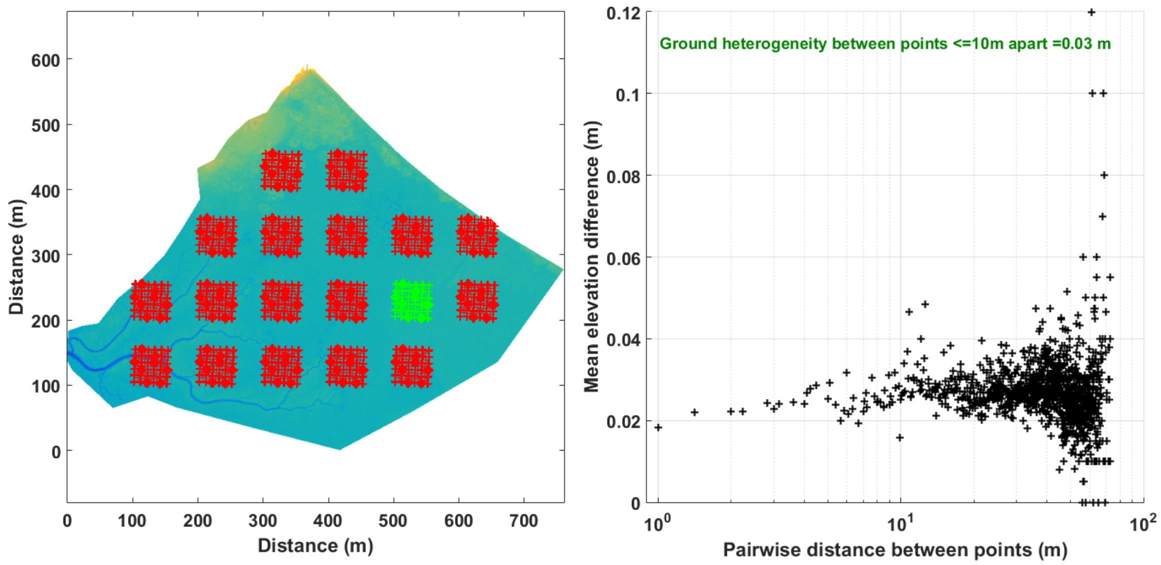
Appendix K10: As Appendix K1 but for Welwick

Appendix L : Ground heterogeneity of MR schemes and natural marshes

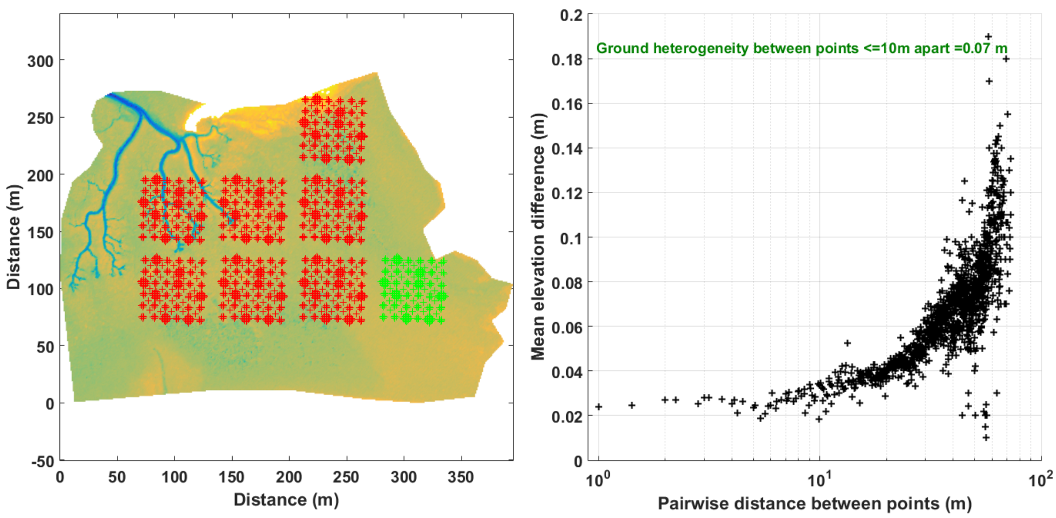
This appendix provides the ground heterogeneity (cm- to dm-scale topography) extracted from 360 DSM lidar data points over a 50x50 m^2 window following a method by Brooks et al. (2015) within 10 MR schemes and 13 natural saltmarshes. The analysis was repeated at regular intervals of 70 to 400 m depending on the size of the saltmarsh considered, in order to capture the entire marsh area. The window of minimum standard deviation for elevation was then selected in order to avoid the largest channels. The ground heterogeneity was then defined as the mean elevation difference between points up to 10 m apart.



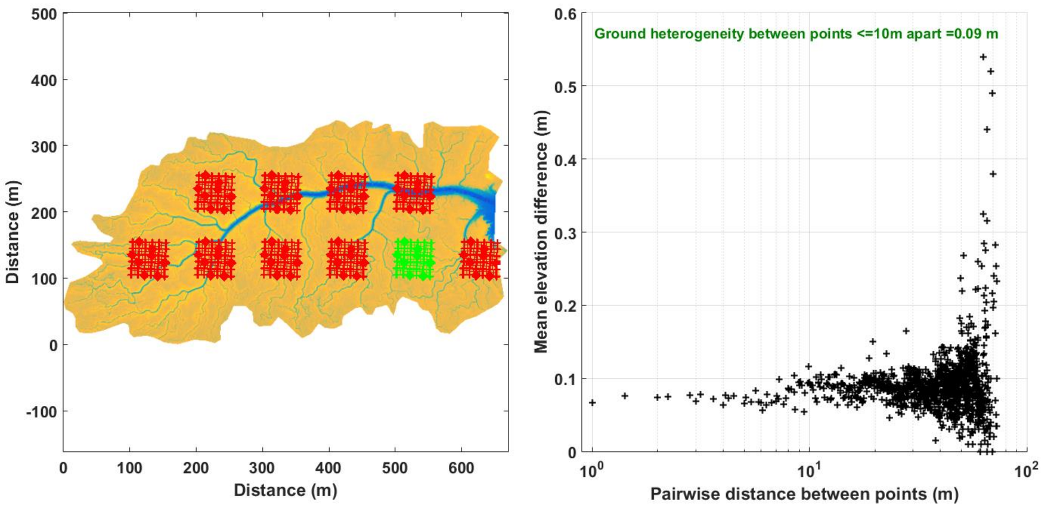
Appendix L1: Ground heterogeneity calculation at Banks (natural saltmarsh); A: 50x50 m^2 windows of data sampling applied every 70 m within the site. The window with the smallest elevation standard deviation is the least influenced by channels and is thus selected to give the marsh topography. The scatter plot shows the distribution mean elevation difference per pairwise difference between points. The mean value of elevation difference for points <10 m apart gives the ground heterogeneity.



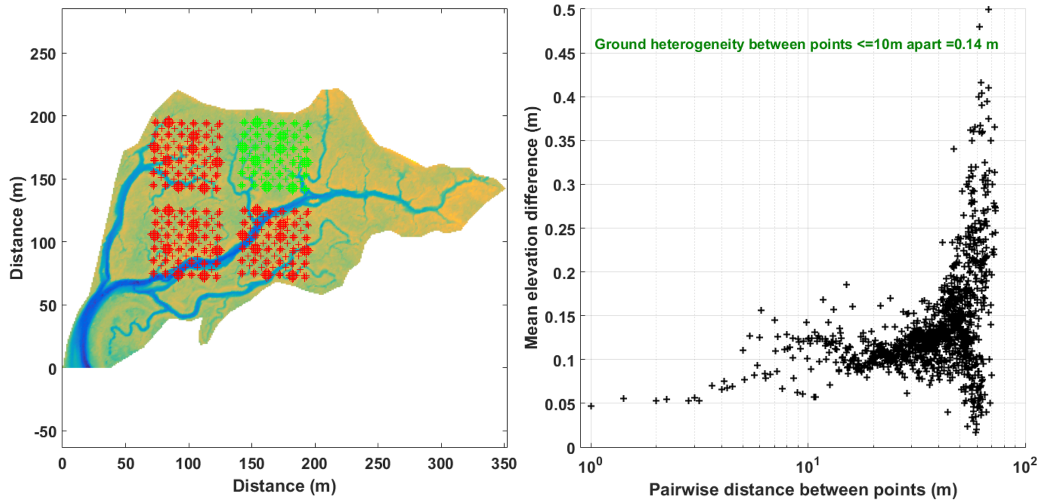
Appendix L2: As Appendix L1 but for Crossens (natural saltmarsh)



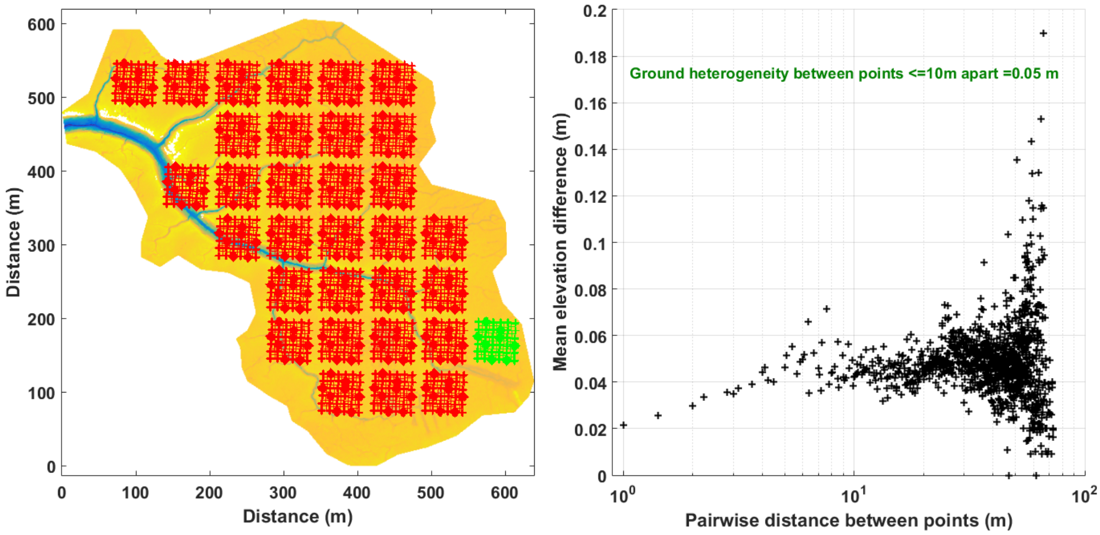
Appendix L3: As Appendix L1 but for Gibraltar Point (natural saltmarsh)



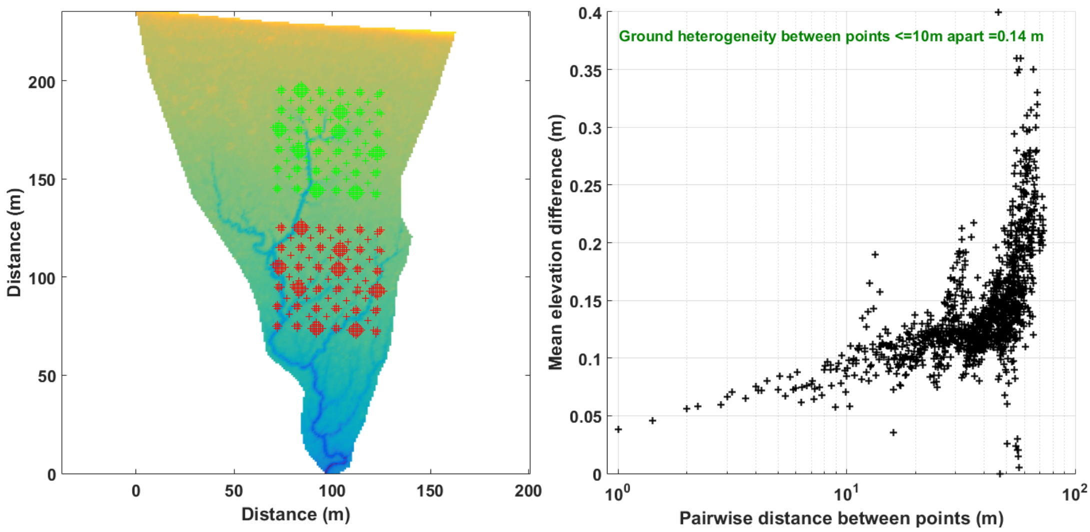
Appendix L4: As Appendix L1 but for Grange (natural saltmarsh)



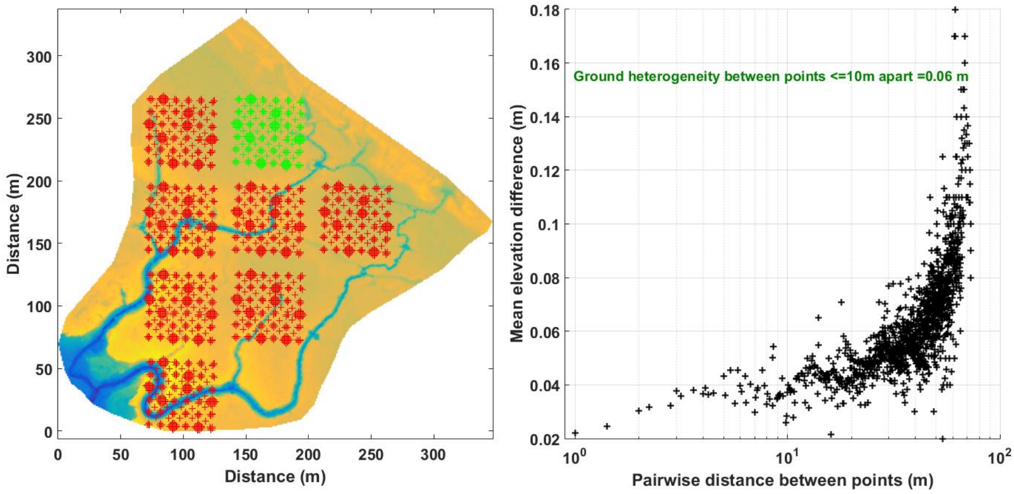
Appendix L5: As Appendix L1 but for Hen Hafod (natural saltmarsh)



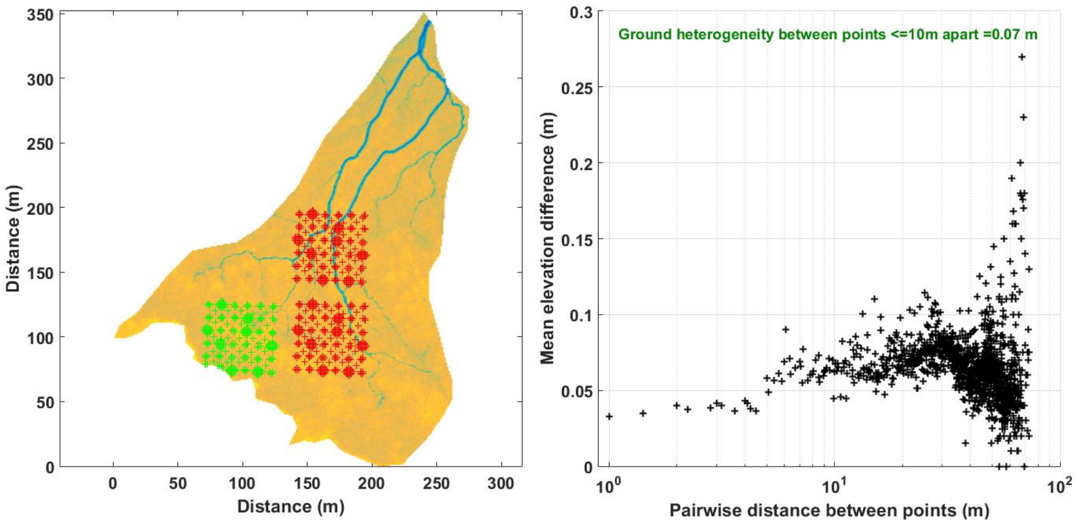
Appendix L6: As Appendix L1 but for Longton (natural saltmarsh)



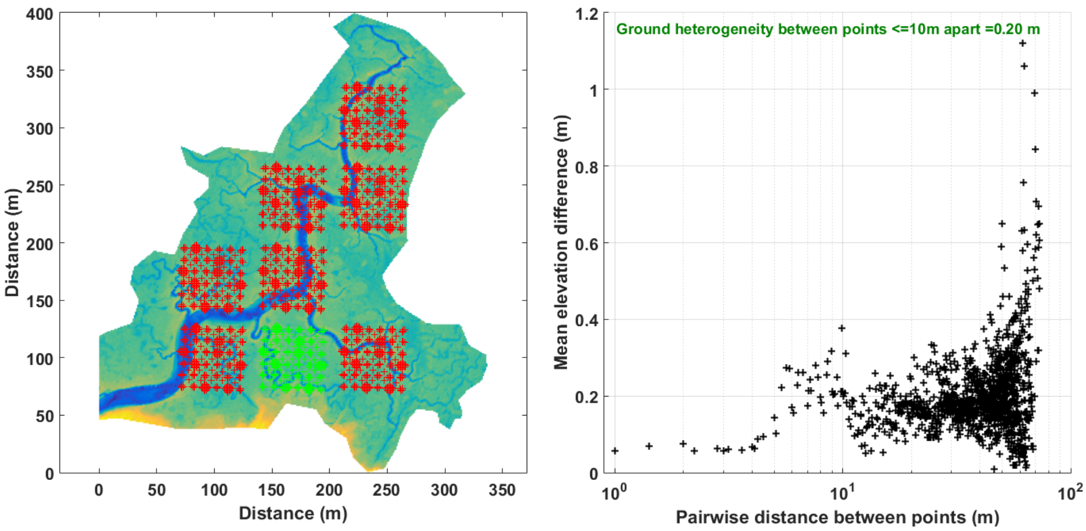
Appendix L7: As Appendix L1 but for Portbury Wharf (natural saltmarsh)



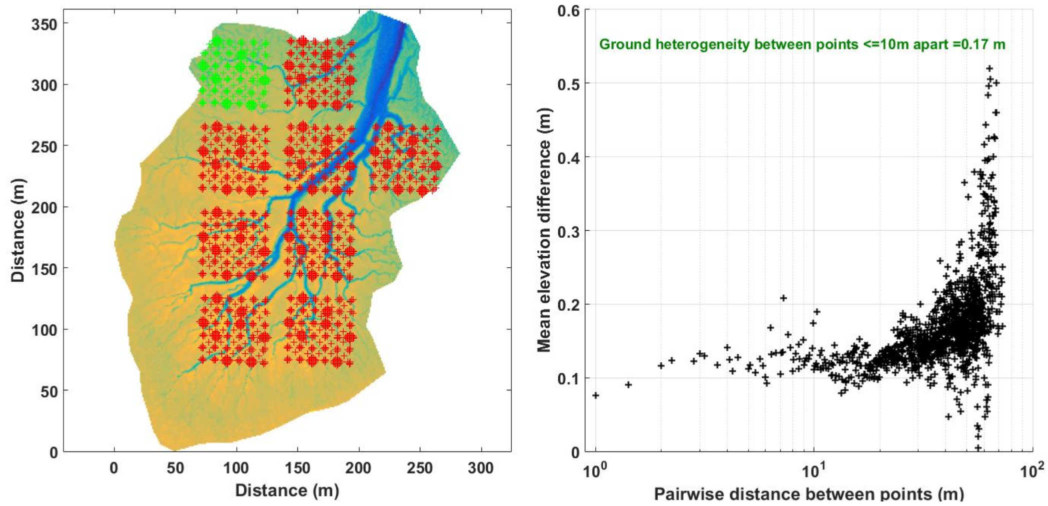
Appendix L8: As Appendix L1 but for Newton Arlosh (natural saltmarsh)



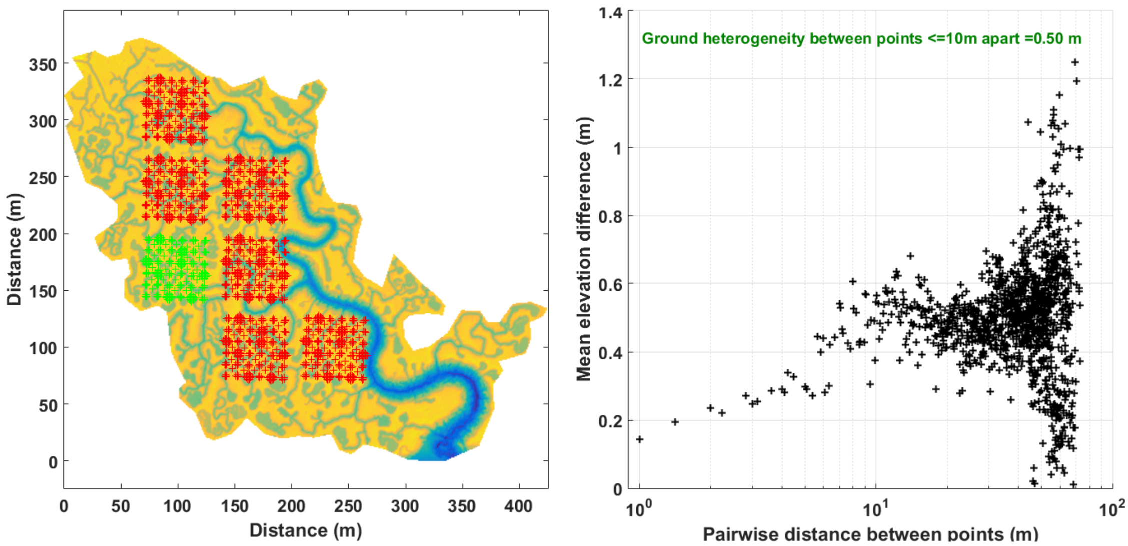
Appendix L9: As Appendix L1 but for Shell Ness (natural saltmarsh)



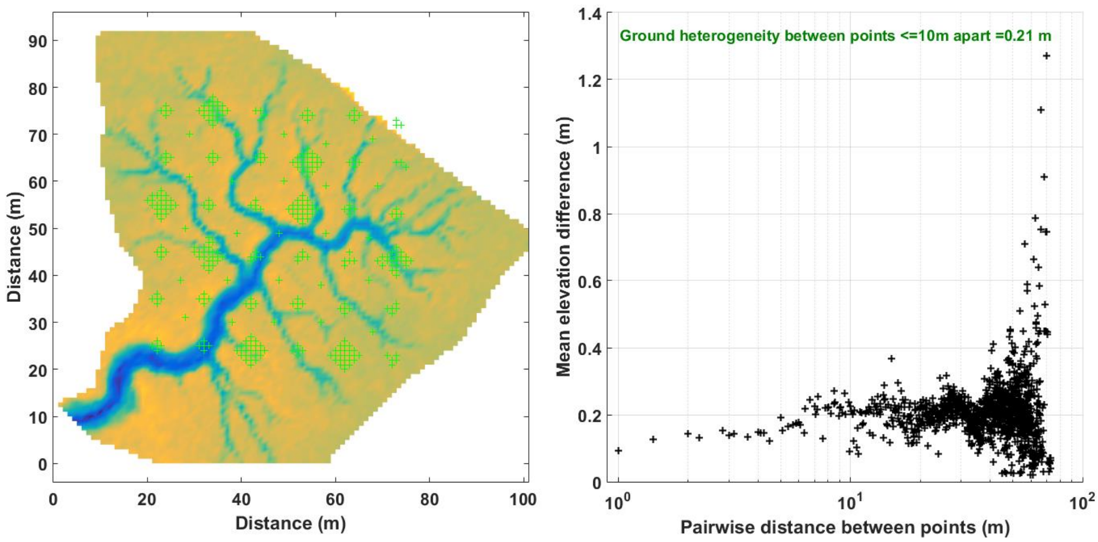
Appendix L10: As Appendix L1 but for Stiffkey (natural saltmarsh)



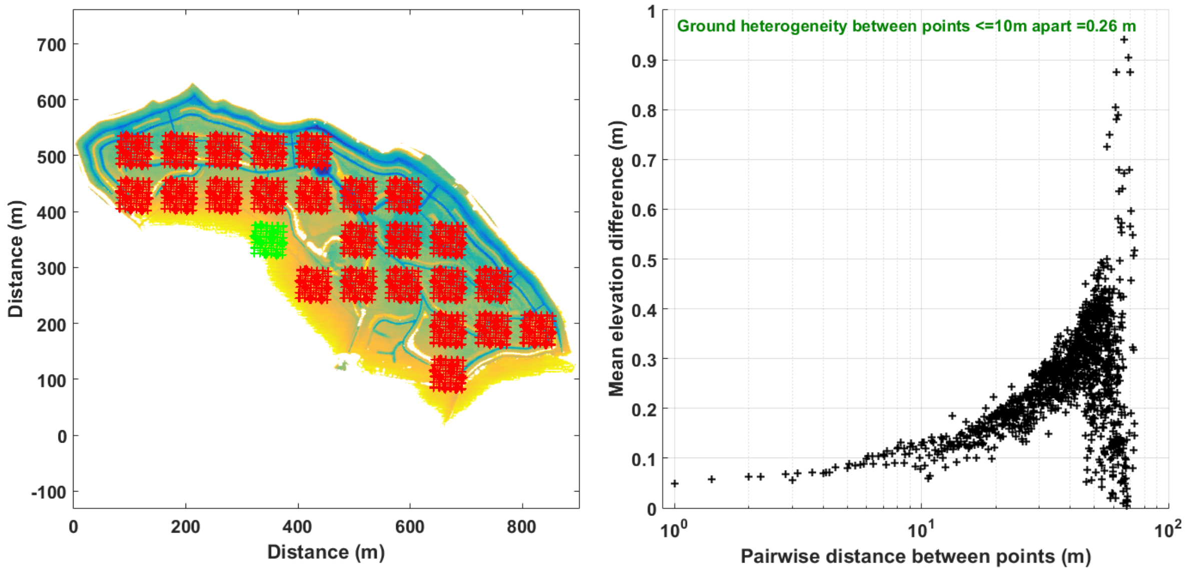
Appendix L11: As Appendix L1 but for Tir Morfa (natural saltmarsh)



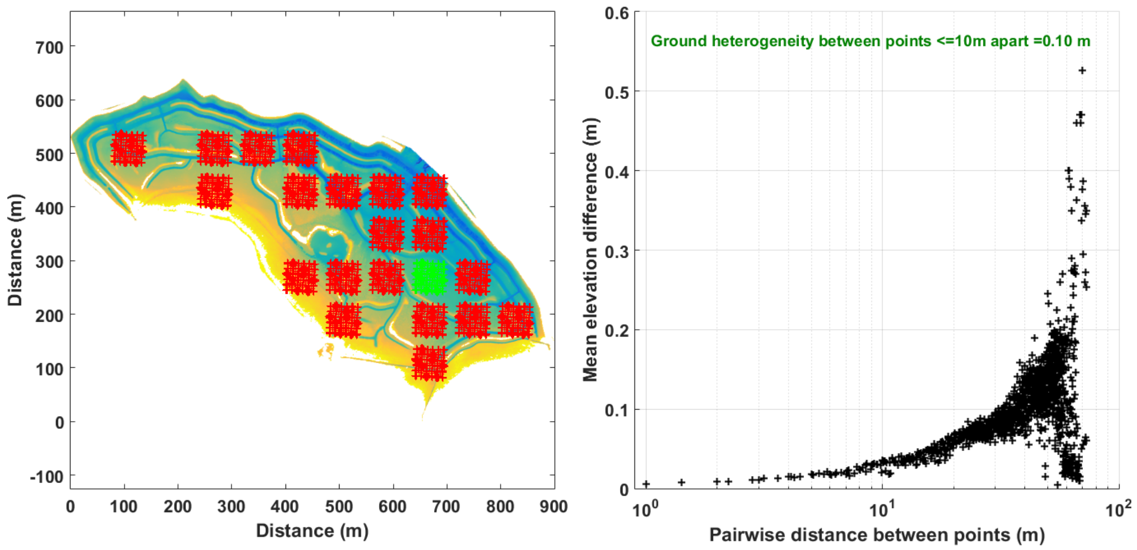
Appendix L12: As Appendix L1 but for Tollesbury (natural saltmarsh)



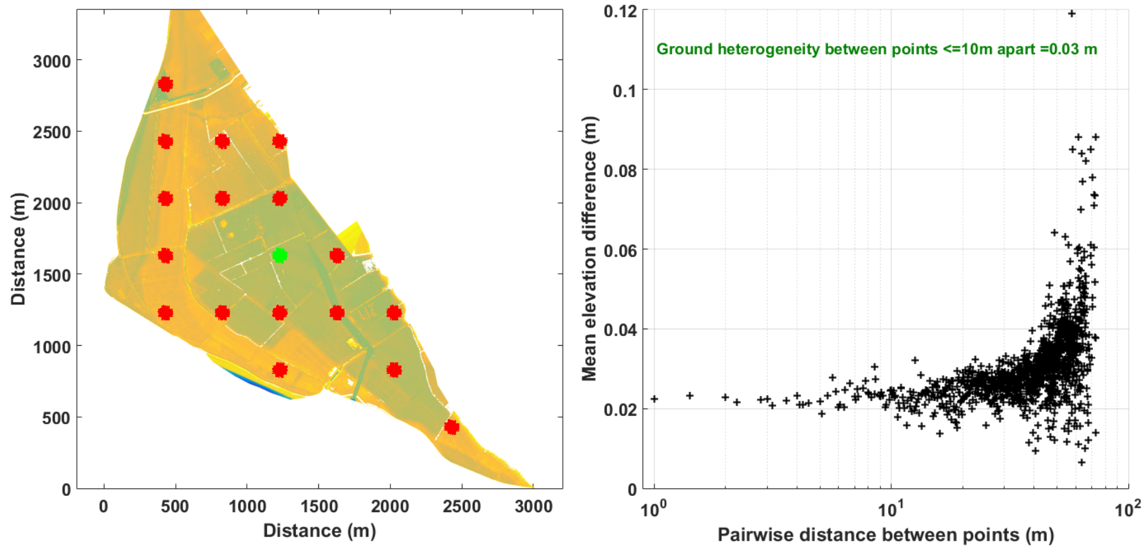
Appendix L13: As Appendix L1 but for Warren Farm (natural saltmarsh)



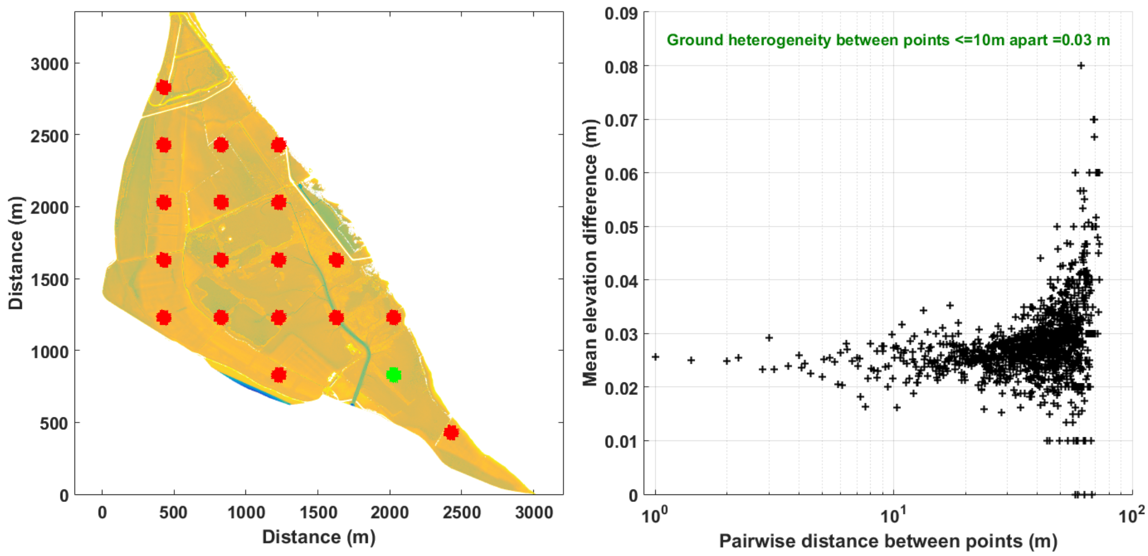
Appendix L14: As Appendix L1 but for Abbots Hall 2002 (MR site)



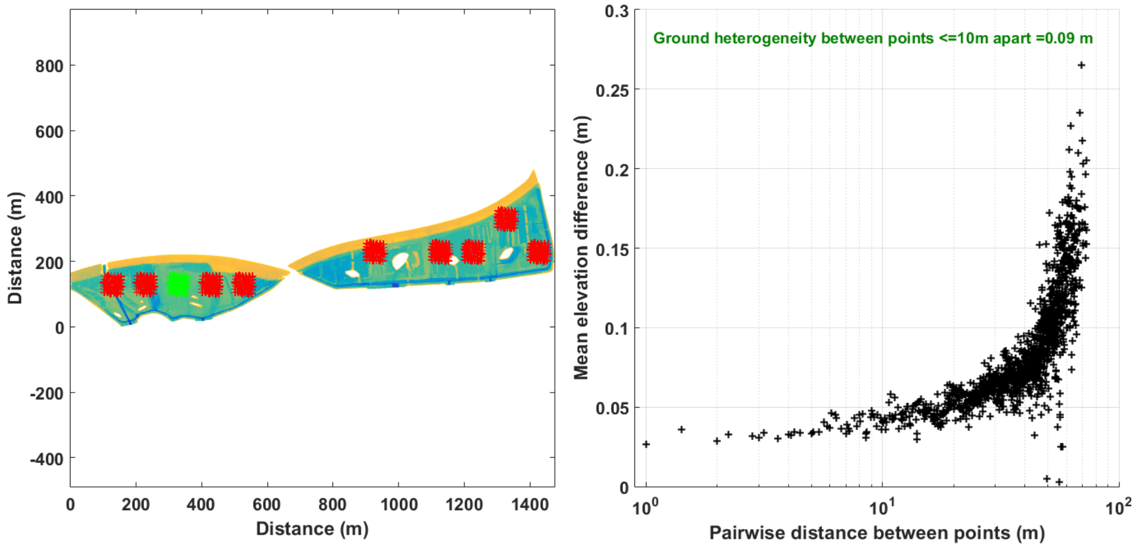
Appendix L15: As Appendix K1 but for Abbots Hall 2015 (MR site)



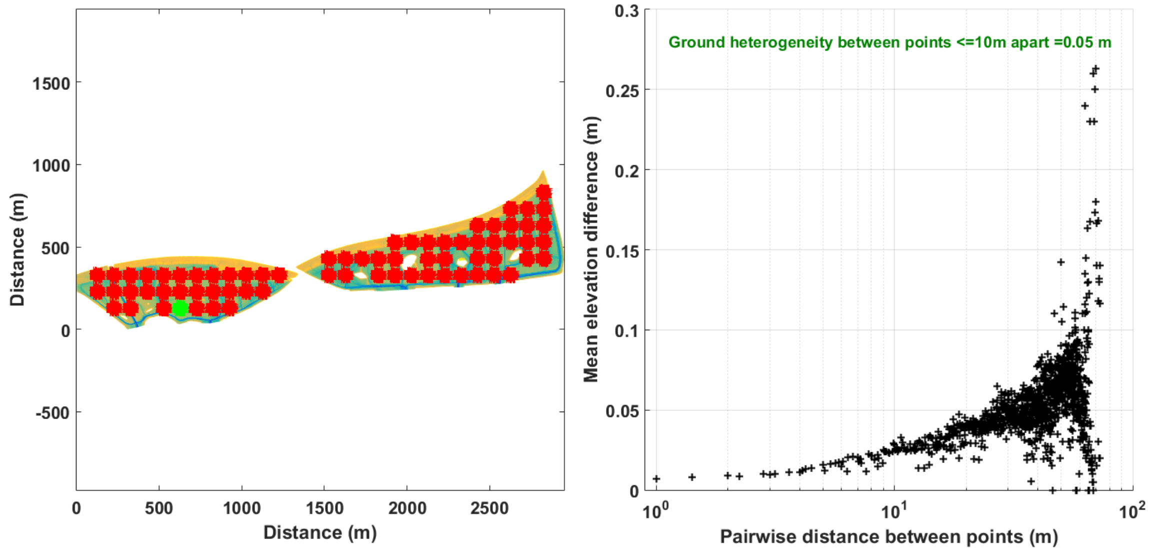
Appendix L16: As Appendix L1 but for Alkborough 2007 (MR site)



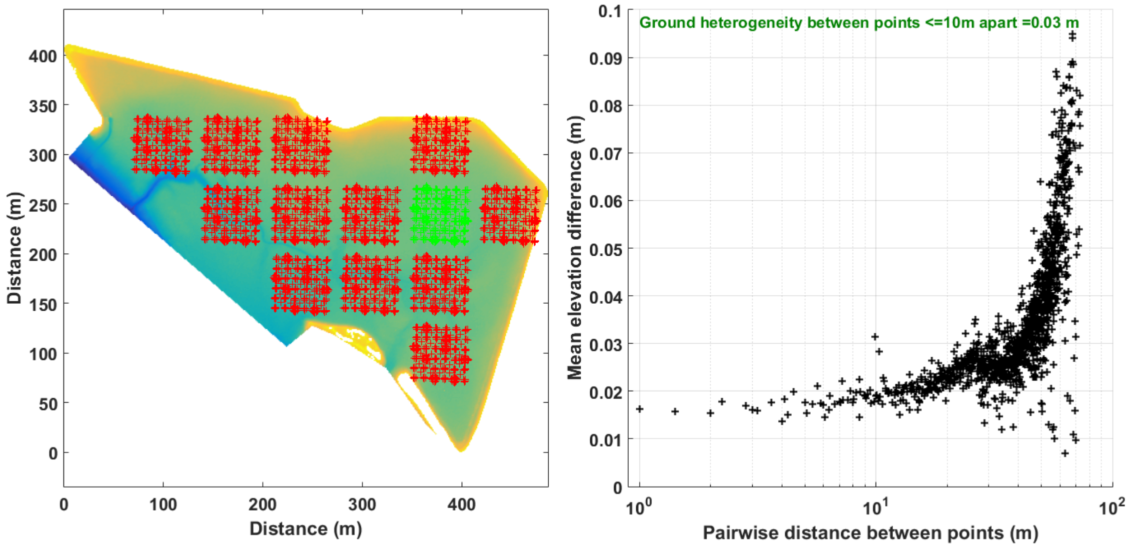
Appendix L17: As Appendix L1 but for Alkborough 2015 (MR site)



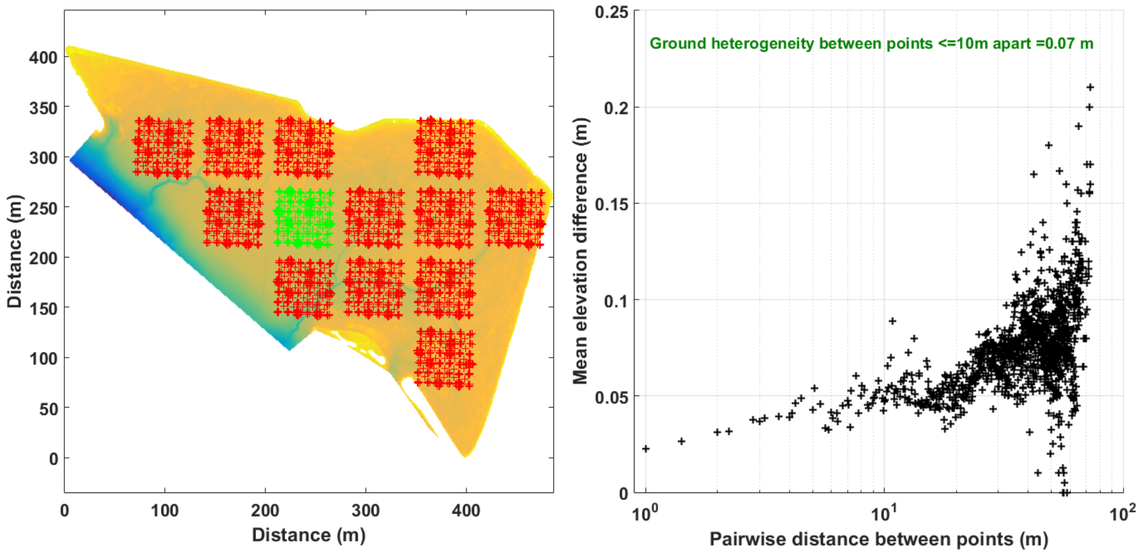
Appendix L18: As Appendix L1 but for Allfleet 2007 (MR site)



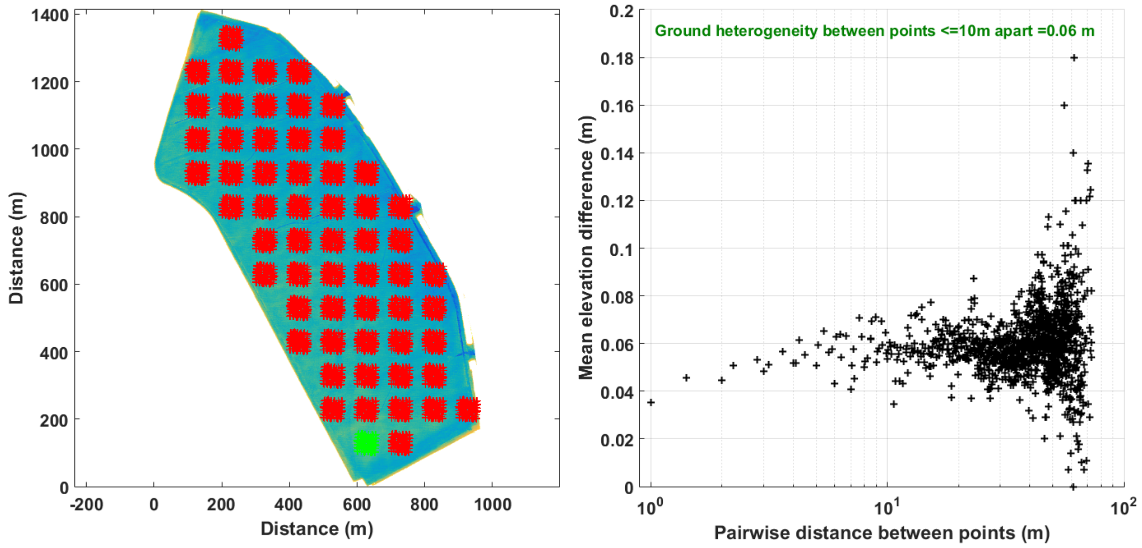
Appendix L19: As Appendix L1 but for Allfleet 2015 (MR site)



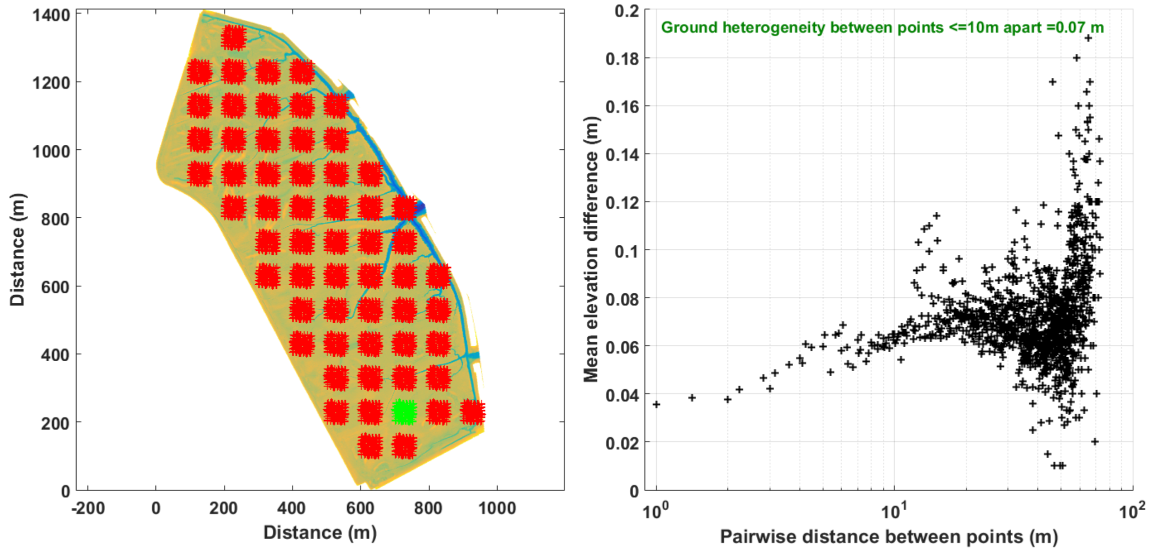
Appendix L20: As Appendix L1 but for Chowder Ness 2007 (MR site)



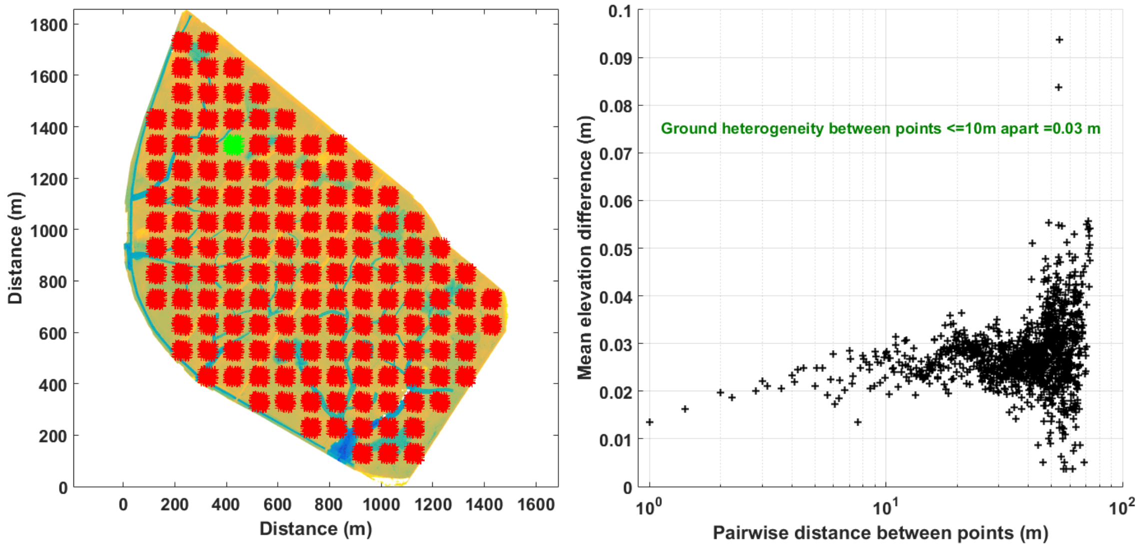
Appendix L21: As Appendix K1 but for Chowder Ness 2016 (MR site)



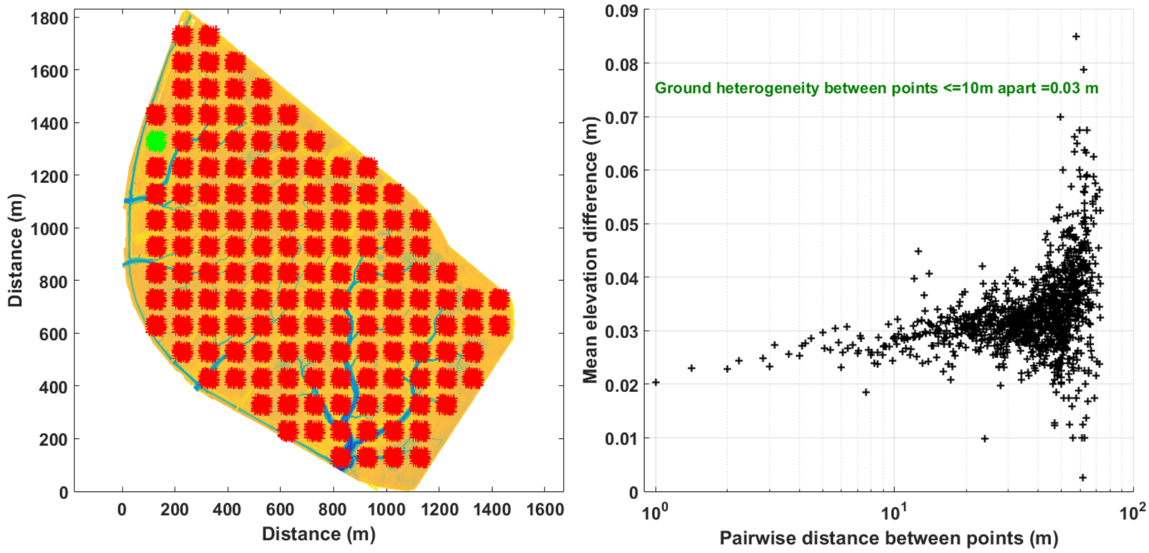
Appendix L22: As Appendix L1 but for Freiston 2002 (MR site)



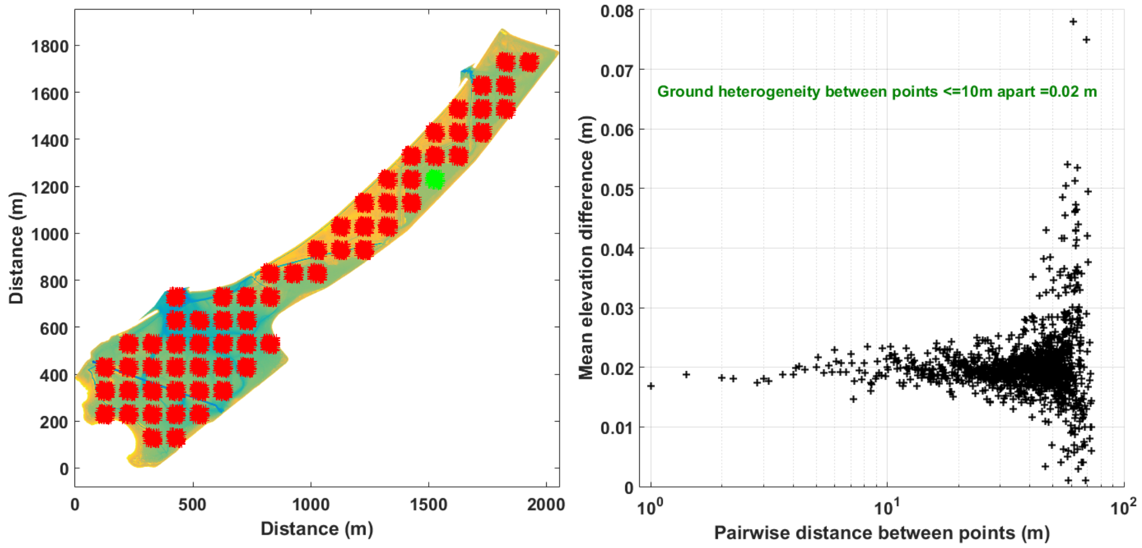
Appendix L23: As Appendix L1 but for Freiston 2014 (MR site)



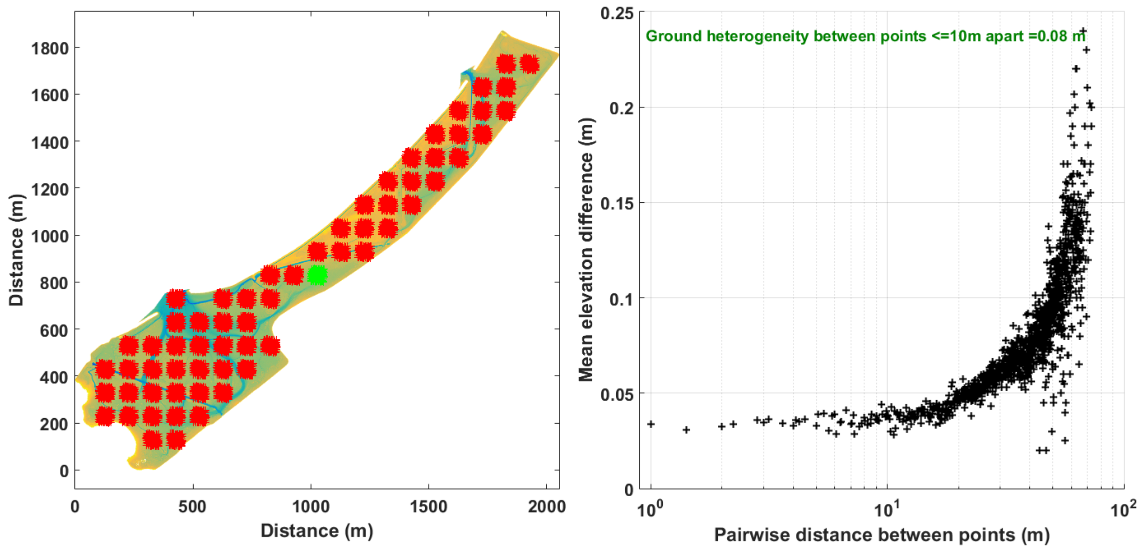
Appendix L24: As Appendix L1 but for Hesketh Out Marsh West 2009 (MR site)



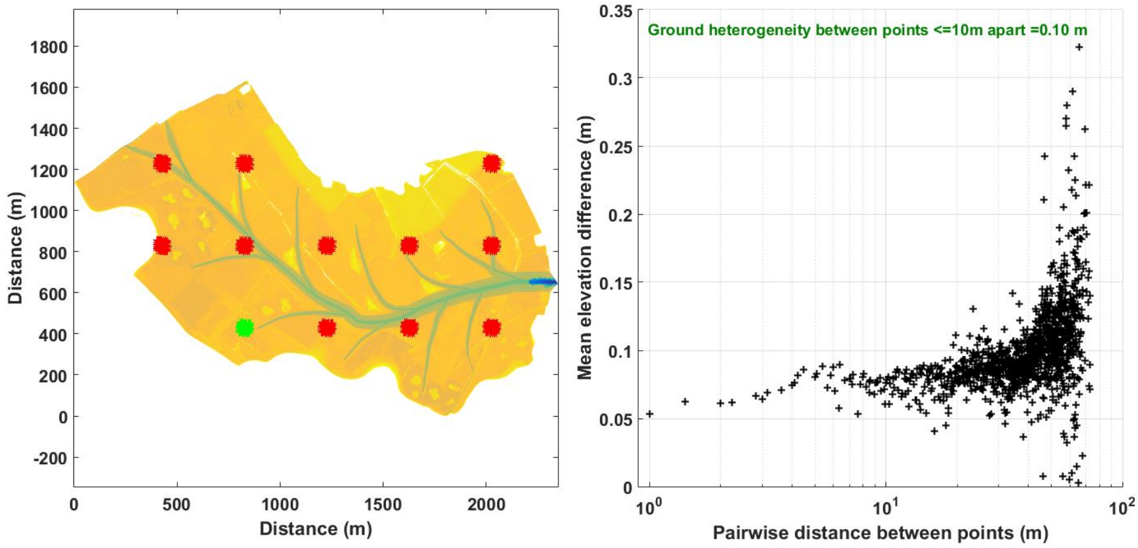
Appendix L25: As Appendix L1 but for Hesketh Out Marsh West 2014 (MR site)



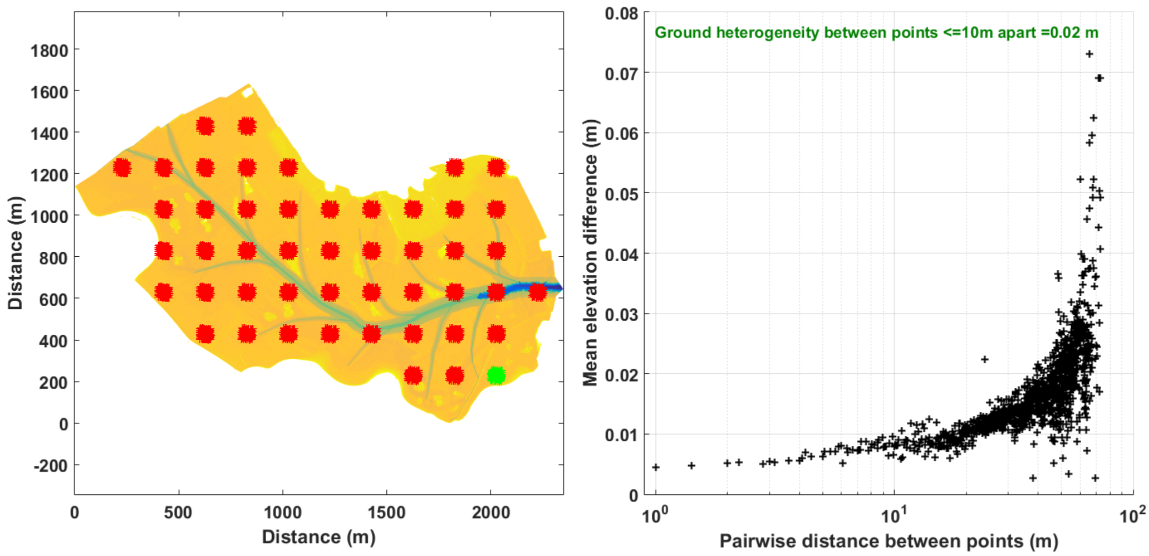
Appendix L26: As Appendix K1 but for Paull Holme Strays 2007 (MR site)



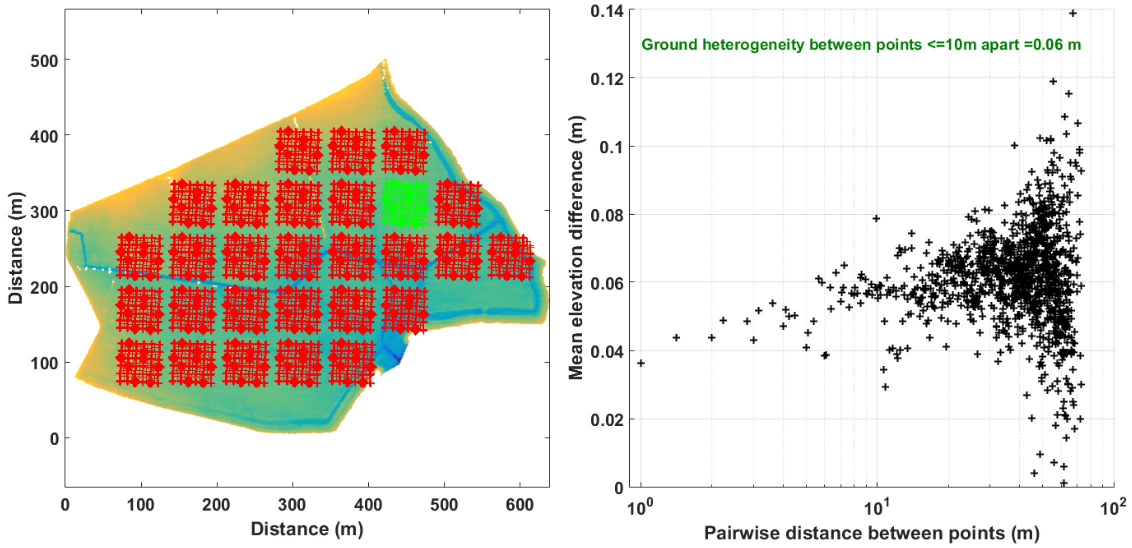
Appendix L27: As Appendix L1 but for Paull Holme Strays 2015 (MR site)



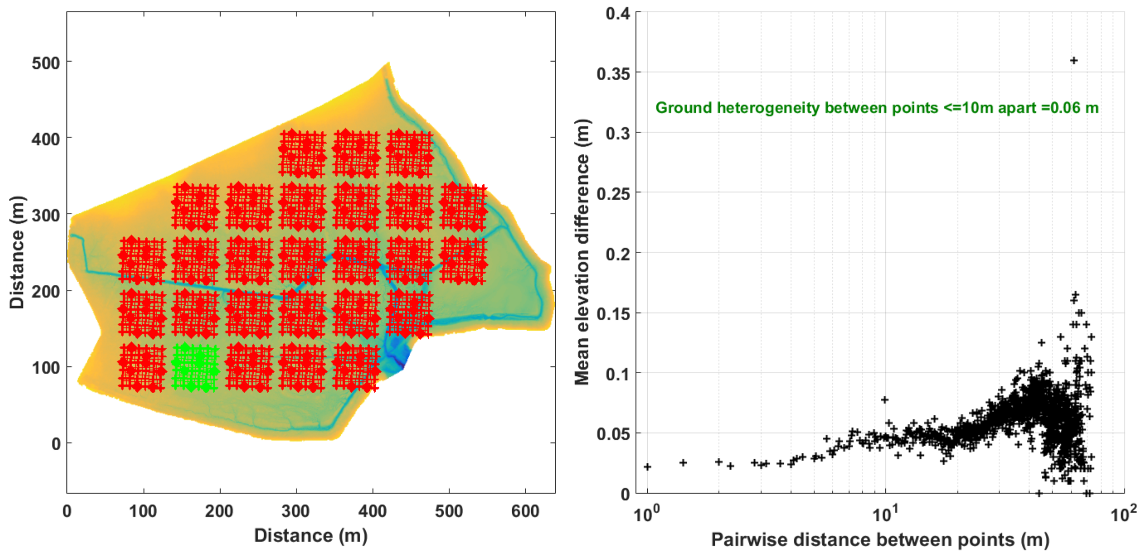
Appendix L28: As Appendix L1 but for Steart 2014 (MR site)



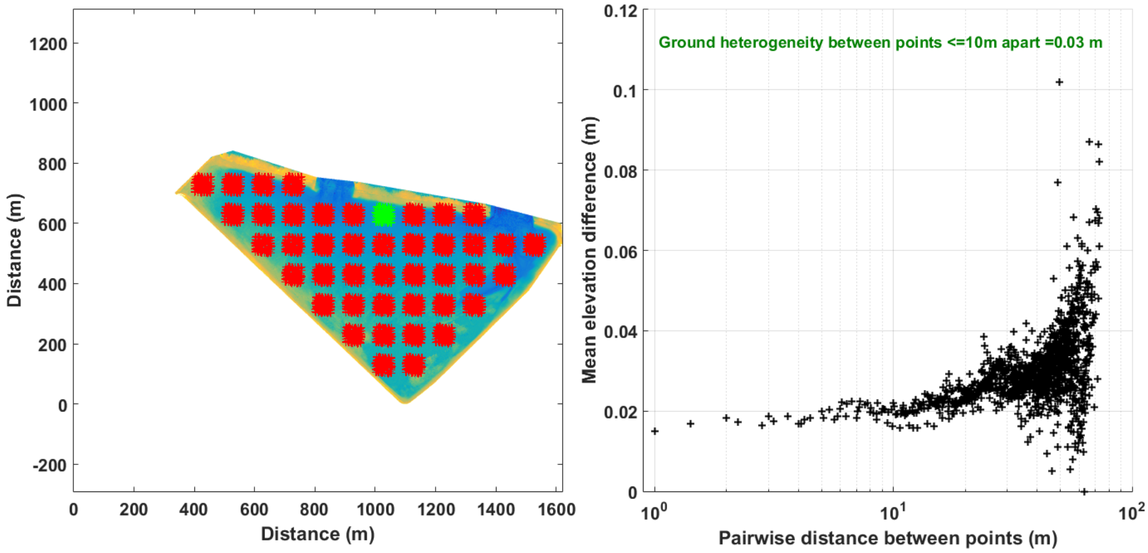
Appendix L29: As Appendix L1 but for Steart 2016 (MR site)



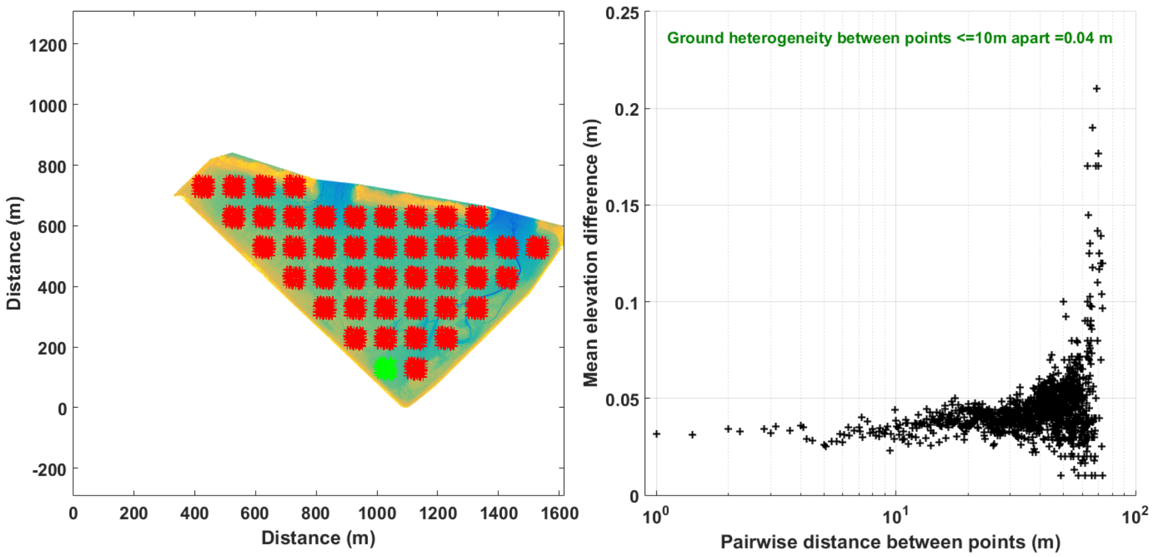
Appendix L30: As Appendix L1 but for Tollesbury 2002 (MR site)



Appendix L31: As Appendix L1 but for Tollesbury 2016 (MR site)



Appendix L32: As Appendix L1 but for Welwick 2007 (MR site)



Appendix L33: As Appendix L1 but for Welwick 2014 (MR site)

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