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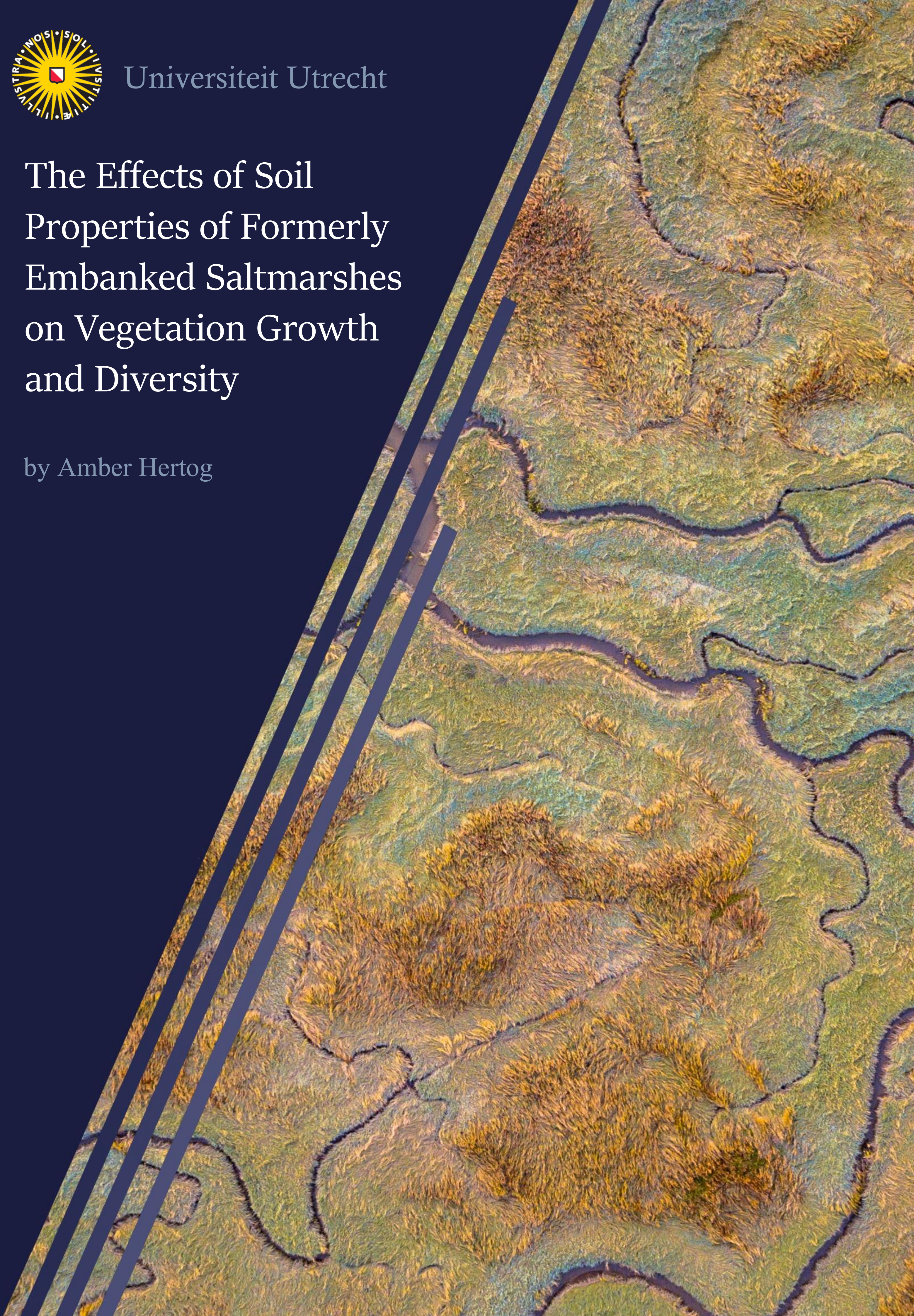
by Amber Hertog



The Effects of Soil Properties of Formerly Embanked Saltmarshes on Vegetation Growth and Diversity

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Abstract

The threat of sea level rise causes many countries to spend time and money on coastal protection. One such protective measure is the de-embankment of former saltmarshes, in order to restore them to functional ecosystems with creeks and vegetation. However, during the time of embankment, these areas were often under agricultural management that turned the land into crop-fields or pastures. The difference in land-use might have had lasting effects on the soil, which could in turn affect the vegetation growth and diversity. This paper contains an overview of observed and possible ways in which the soil properties of formerly embanked saltmarshes affect vegetation growth and diversity. Soil compaction caused by heavy tillage machinery or cattle trampling cannot only make it harder for roots and shoots to penetrate the soil, but it also negatively affects creek formation and drainage of the marsh, which is detrimental to vegetation diversity. This is exacerbated by a low topographic heterogeneity that hinders creek formation. Original research shows that the heterogeneity usually does not increase over time but instead seems to deteriorate. In addition, soil compaction leads to a low marsh surface elevation, which further impedes biodiversity. Excessive fertilizer use before de-embankment might also influence vegetation growth and diversity, as it can lead to high nutrient environments which could favor certain species and induce algal proliferation. Lastly, it may cause the soil of higher elevations to become more acidic. The severity of these factors is likely different pertaining to zonation on the marsh: where the lower marsh elevations likely deal more with factors relating to drainage, higher elevations could be more affected by penetration resistance, high nutrient concentrations and acidification.

Keywords: salt marsh, vegetation growth, vegetation diversity, saltmarsh restoration, agricultural management, soil compaction, fertilizer

Introduction

It is common knowledge that global warming is causing a lot of problems worldwide and it will only continue to do so. Two of these problems are rising sea levels and an increase in storm surges (Oppenheimer *et al.*, 2019; Collins *et al.*, 2019). These threats are especially significant for coastal regions with land below sea level (Oppenheimer *et al.*, 2019). Though coastal protective measures - such as dikes, sluices and dams - have this far been enough to combat most of the effects of climate change in Western Europe, they will not be able to withstand future threats without reinforcement (Vousdoukas *et al.*, 2020). The maintenance has to be kept up indefinitely and usually costs a lot of money, causing people to look into more sustainable forms of coastal protection (Slobbe *et al.*, 2013). This often means reinstating or trying to emulate natural processes, a strategy known as 'building with nature' (De Vriend *et al.*, 2015).

One of these natural forms of coastal protection is the presence of vegetated foreshores, such as saltmarshes. Saltmarshes are areas that appear between the sea and mainland and are defined by the presence of creeks and halophytic vegetation (Adam, 1993). The extent of a marsh is divided into different zones based on elevation and, consequently, proximity to the water (Chirol *et al.*, 2018). The lower marsh, which is closest to the sea, is frequently flooded by the tides and has the most creeks, which facilitate drainage (Pennings & Calloway, 1992; Chirol *et al.*, 2018). Vegetation in this part of European marshes has species like cordgrass (*Spartina*) and sea purslane (*Halimione portulacoides*) (Fig. 1) (Davidson, 2016). The higher marsh is usually only flooded during spring tide or storms and therefore has less need of creeks (Chirol *et al.*, 2018). Here, vegetation consists predominately of grasses and species of rush, such as saltmarsh rush (*Juncus gerardii*) and red fescue (*Festuca rubra*) (Fig. 2) (Davidson, 2016). Between these two zones, there is the middle marsh, which is more like a combination of the two, being less flooded than the low marsh but still more than the higher marsh (Chirol *et al.*, 2018). These different zones each have their own role in the saltmarsh ecosystem. For example, creeks function as nurseries for juvenile fish and the plants higher on the marsh work as flood protection to bird nests (Miller & Simenstad, 1997; Benvenuti *et al.*, 2018).

Figure 1: *The cord-grass (*Spartina anglica*), Wadden Sea near the Eider estuary in 2004. Photo Stefan Nehring. (Gollasch & Nehring, 2004*)

Figure 2: *Close up of saltmarsh rush (*Juncus gerardii*) (Fenwick, 2010)*

Besides promoting biodiversity, saltmarshes also provide coastal protection. Not only do they provide a place where water can go during high tide, the vegetation also acts as a buffer for wave action, decreasing their energy and thus their eroding force (Möller *et al.*, 2014).

However, saltmarshes are disappearing. More than 90% of saltmarshes bordering the North Sea have been lost and embanked in land reclamation schemes (Barkowski, 2009). This land was then often used for agriculture, either as a pasture for cattle to roam or as a crop-field. With the loss of the saltmarsh ecosystem and the threat of sea level rise in mind, projects regarding coastal protection have started to explore the restoration of saltmarshes through a process called 'managed realignment' (Mossman *et al.*, 2012). This is done by de-embanking coastal areas and letting them connect directly to the sea again, after which a new dike is usually constructed behind the marsh (Blackwell *et al.*, 2010). Sometimes, creeks are dug before the point of breaching, to accelerate the restoration process (Teal & Weishar, 2005).

The characteristics of a successfully restored saltmarsh are usually based on their similarity to natural marshes. However, marshes can differ greatly in variables like surface area, age or elevation and are therefore difficult to compare (Wolters *et al.*, 2005). The success of a managed realignment scheme is therefore measured in varying ways, as there are multiple factors that can serve as indicators, such as creek formation, vertical accretion and vegetation cover (Masselink *et al.*, 2017; Staszak & Armitage, 2013). Wolters *et al.* (2005) reasoned that it is important to make a distinction between the structure and functioning of a saltmarsh. They mentioned that though both are important, the functioning is dependent on the structure to work. Different functions require different structural components, which indicates the importance of a highly diverse ecological structure. The restoration success in their study was therefore measured with plant species diversity. Vegetation diversity is not only important for exosystemic functioning but can also affect the way in which the saltmarsh defuses the incoming waves (Möller, 2006). For a managed realignment scheme with coastal protective intentions, it is therefore important that the plant species composition resembles that of a marsh with adequate defensive properties.

Nevertheless, multiple studies show that the plant species diversity of restored marshes is dissimilar to that of natural marshes (Wolters *et al.*, 2005; Garbutt *et al.*, 2006; Barkowski *et al.*, 2009; Spencer *et al.*, 2017). Even many years after de-embankment, this remains true (Garbutt & Wolters, 2008). To provide a better understanding about how to increase the restoration success, this paper will therefore investigate factors that might influence the vegetation growth and diversity in restored saltmarshes. As mentioned above, the areas were often used as pastures or arable land before de-embankment, the management of which might have had a transformative effect on the soil. This study will therefore investigate in which ways the soil properties of formerly embanked saltmarshes affect vegetation growth and diversity. The research will be predominately based on existing literature, which will be expanded upon by an original study about the topographic heterogeneity of restored marshes. I will discuss the specifics of both soil structure and soil chemistry in relation to their effect on vegetation growth and diversity.

1. Soil Structure

Soil structure pertains to the way individual soil particles are grouped together (Finch *et al.*, 2014). It entails several properties, such as soil compaction, porosity, soil strength and topography. Most of these properties are observed to influence vegetation growth (Passioura, 1991; Liu *et al.*, 2020). In this section, I will discuss the ways in which various structural properties can affect the growth, and consequently the biodiversity, of vegetation and how this relates to salt marsh restoration. I will start with soil compaction, followed by topographic heterogeneity.

1.1 Soil compaction

Areas used in managed realignment schemes were often utilized for agricultural purposes before de-embankment (Wolters *et al.*, 2005). Either used as crop-fields or pastures, soil characteristics were very different before flooding and this is reflected in the compaction of the soil (Spencer *et al.*, 2017). Heavy machinery used for crop tillage and the trampling hooves of large cattle put a considerable amount of pressure on the ground, causing pores containing air and water to be compressed (Bauer *et al.*, 2015; Herbin *et al.*, 2010; Shapiro & Elmore, 2017). When this happens, the soil consolidates, which increases soil strength and causes a decrease in volume (Singh *et al.*, 2010; Olsen & Schuster, 1984). This can affect plant growth in several ways, which will be discussed below.

1.1.1 Surface elevation

As mentioned above, soil compaction reduces the volume of the land. This decreases the height of the area, resulting in a relatively low surface elevation in restored saltmarshes

(Crooks *et al.*, 2002). Surface elevation refers to the height of the area above sea level and is often described as one of the most important factors for saltmarsh restoration (Garbutt *et al.*, 2006; Davy *et al.*, 2011). Vegetation is important for the maintenance of, and increase in elevation, as the plants trap sediment between the roots and shoots (Weis, 2016). However, surface elevation in turn has an effect on vegetation, most importantly the biodiversity. For instance, some species cannot survive in frequently flooded soil and need a higher elevation level than species that can (Parrondo *et al.*, 1978). Surface elevation is therefore an important factor in plant species zonation. Accordingly, surface elevation can alter the species composition of a restored saltmarsh relative to a natural one. This was shown by Thom *et al.* (2002), who observed a significant difference in assemblage between a restored and natural marsh due to a 1 m subsidence of the area during the years prior to de-embankment. The species composition resembled only a low marsh system instead of the mid to high marsh systems of the reference marsh. High marsh systems are important for coastal protection against storm surges, which are becoming more frequent due to global warming (Stark *et al.*, 2015; Collins *et al.*, 2019). For a managed realignment scheme with coastal protective intentions, it is therefore important that the surface elevation resembles that of a marsh with adequate coastal protective attributes, and thus a specific plant species composition.

Nevertheless, multiple studies have reported that restored marshes have a surface elevation that is generally lower than natural marshes (Brooks *et al.*, 2015). This difference was shown in a study by Wolters *et al.* (2005), where they examined the variation between the elevation and the tidal range from Mean High Water Neap (MHWN) to Mean High Water Spring (MHWS) for seventy Western Europe-based restoration sites. Not only did they find a significant positive relation between this variation and plant species diversity, but they also observed that for most sites, the height of elevation relative to the MHWN-MHWS range was less than 50 %. The plant diversity of those marshes did not represent the diversity of a natural marsh. A similar situation was documented by Garbutt *et al.* (2006), who found that a realignment site in Essex, U.K., still had a relatively low surface elevation six years after de-embankment. As a result, the marsh had a low species diversity and was dominated by pioneer plant species, which are common at low elevation. In addition, the low elevation resulted in less vegetation, as the lower boundary of vegetation growth is determined by the amount of water in the soil: too much water, and the plants cannot breathe (Silvestri & Marani, 2004).

1.1.2 Root/shoot penetration resistance

Soil compaction reduces the amount of air pockets in the sediment, also known as the porosity of the soil. This also increases the soil strength, which makes it harder for roots to penetrate the soil (Tracy *et al.*, 2011). Though there is little research done on the effect of penetration resistance on common saltmarsh species, there have been studies that investigate its relation to terrestrial plant growth. A study by Valentine *et al.* (2012) about root elongation in Scottish crop-fields, showed that over a period of 48 hours, roots in compacted soil grew an average of 16.0 mm compared to the 48.2 mm in unimpeded soil. They ascribed this difference mostly to the high penetration resistance and low porosity of the compacted soil, suggesting that the lack of available space hindered the elongation. A decrease in elongation is detrimental to a plant's ability to respirate and take up nutrients (Nawaz *et al.*, 2013).

A similar effect can be witnessed in seedling emergence; after all, the same resistance is present during upwards growth as there is for downwards growth (Hyatt *et al.*, 2007). In fact, the effect of soil compaction is generally worse for young plants than established vegetation (Nawaz *et al.*, 2013). A greenhouse experiment by Hyatt *et al.* (2007) showed that when compaction increased, soybean seedling emergence always declined. They also observed a difference between high- and low-vigor seeds, as the emergence of high vigor seeds remained 5 - 40 % higher[1]. The paper noted how this would give high-vigor seeds an advantage over low-vigor seeds. Though there is little research done that suggests seed vigor is a species-specific trait, it

1: *Seed vigor is a term used to describe the quality of a seed. It is measured by a seed's ability to germinate (Hyatt et al., 2007).*

can be linked to an organism's genotype (Sun *et al.*, 2007).

Nevertheless, the consequences of soil compaction do not have the same effect on every species; even though it is generally accepted that relatively high soil strength and low porosity have an adverse effect on root growth, it is important to note that the vegetational response to such circumstances is species specific (Materechera *et al.*, 1991). Materechera *et al.* (1991) discovered a significant relation between root elongation and root diameter. They found that dicotyledonous plant species, who commonly have thicker roots, have an advantage over monocotyledons (see Fig. 3). However, this correlation was questioned by Place *et al.* (2008), who demonstrated how the weeds sicklepod and Palmer amaranth were more effective in penetrating compacted soil than soybeans. Though the study did not further examine what root traits caused this response, they noted that the roots of the Palmer amaranth penetrated the soil more effectively than those of sicklepods and soybeans, though they are known to have a significantly smaller diameter. However, a study by Chen & Weil (2009) established a difference between monocots and dicots based on the root system: they found that tap-rooted species, which are generally dicots, might be more effective in infiltrating the soil than fibrous-rooted species (Chen & Weil, 2009; Sparks *et al.*, 2017). This

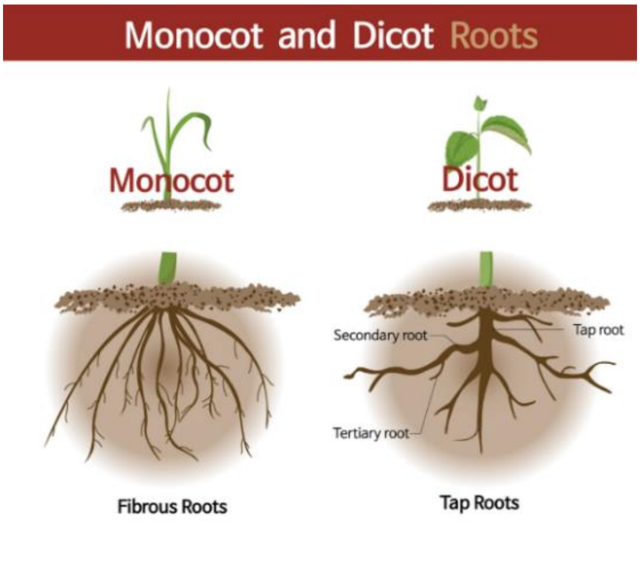

Figure 3: *A visual representation of the differences in monocot and dicot roots. Monocots are known to have fibrous roots, while dicot species usually have tap roots (Jakinboaz, z.d.).*

might explain the discrepancy between Materechera *et al.* (1991) and Place *et al.* (2008), suggesting that it is not the root diameter, but the root system that leads to a bigger advantage. The species used by Place *et al.* (2008) are all dicots, which indicates that the vegetational response to soil compaction also varies amongst dicotyledonous species. A species-specific response such as this might therefore impact the biodiversity.

1.1.3 Soil porosity

Vegetation growth is not only dependent on a plant's ability to photosynthesize, but also to respire through the roots, for which it requires oxygen (Silvestri & Marani, 2004; Tracy *et al.*, 2011). Normally, this oxygen is found in the soil, which is in a permanent gas exchange with the air through pores, but this changes when the soil becomes waterlogged and cannot exchange air particles as easily (Hogarth, 2017). A waterlogged soil can therefore significantly affect vegetation development and survival (Dat *et al.*, 2006). Hypoxia has been known to negatively affect root systems in situations where air pockets take up less than 10 % of the soil (Lynch & Wojciechowski, 2015). Even plant species that are found in wetlands like saltmarshes and have adapted to higher soil moisture contents, can be affected by a frequently flooded soil (Pezeshki, 2001; Ursino *et al.*, 2004). A study by Parrondo *et al.* (1978) found that this effect was species specific: while one common North American saltmarsh grass (*S. alterniflora*) seemed to grow better under high moisture conditions, another (*S. cynosuroides*) was negatively impacted. A similar variation was observed by Crooks *et al.* (2002) for a British marsh, who noticed that the halophytic shrub *A. portucaloides* seemed to favor drained soil, while the grass *P. maritima* was more abundantly found in undrained areas of the saltmarsh. In this way, the soil moisture content can affect vegetation composition in saltmarshes.

Van Putte *et al.* (2020) demonstrated how soil compaction affects the drainage of groundwater in restored tidal marshes. Though water seemed to drain relatively well in the upper layer of the soil, which consists of sediment deposited through the tidal regime, the compacted agricultural soil underneath it acted as an impermeable barrier for water flow. Groundwater levels never dropped lower than 10 cm deep unless the area experienced a prolonged time without inundation. The natural reference marsh drained almost three times as far on average. This lack of drainage in restored marshes reduces the soil's groundwater storage capacity, which consequently impacts the ability of a saltmarsh to act as a flood defense (Tempest *et al.*, 2014). This was also described by Van Putte *et al.* (2020), who observed that the interior groundwater level of the natural marsh decreased by 28 cm on average, whereas it only declined by 8 cm on the natural marsh.

Although hypoxic conditions are commonly caused by flooding and poor drainage, a severe enough soil compaction might exacerbate the lack of oxygen on its own and consequently reduce local vegetation growth (Pedersen *et al.*, 2020; Mossman *et al.*, 2010). An example of this is a restored marsh in Brancaster, U.K., where areas with little to no vegetation cover had a significantly low redox potential, which indicates the low amount of oxygen in the soil (Mossman *et al.*, 2010; Pepper & Gentry, 2015). They described how the plant density within tracks made by heavy machinery was noticeably reduced and explained this by reasoning that the resulting soil compaction contributed to the lack of local drainage. In addition, anaerobic conditions can increase the severity of root diseases, as was shown by Fritz *et al.* (1995), who measured the effect of soil compaction on common root rot in peas.

1.1.4 Creek formation

Creeks and channels play an important role in the drainage system of saltmarshes. Both the drainage of overmarsh tidal floods and subsurface groundwater rely on creeks and their formation is therefore relevant to restoration success (Brooks *et al.*, 2015; Van Putte *et al.*, 2020). Furthermore, vegetation species diversity is higher near creeks (Morzaria-Luna *et al.*, 2004). This is likely because there is better drainage near creeks and more nutrients are available, as creeks are known to transport these (Zedler *et al.*, 1999; Sanderson *et al.*, 2000; Chirol *et al.*, 2018).

As creek development is dependent on the erosion of the sediment, the erodibility of the soil is a factor in the efficiency of the process (Kearney & Fagherazzi, 2016; Brooks *et al.*, 2020). Soil compaction leads to a higher bulk density, which is defined as the dry mass of a volume of soil, and a higher shear strength (Perkins *et al.*, 2013; Crooks & Pye; 2000). This makes it harder for the sediment to erode, which might negatively impact creek formation. A study by Vandenbruwaene *et al.* (2012), which investigated the formation of a channel network in a restored saltmarsh after four years, mentioned how a compacted sediment layer can negatively affect the deepening of channels. As a result, the drainage density was lower than that of the natural reference marsh. This effect was most clearly observed in high elevation zones, where the erosion had to start almost directly on the compacted soil, as there was little to no layer of tidal deposited sediment on top of it, like in lower elevated areas.

1.2 Topographic heterogeneity

Another factor that can influence vegetation composition is the topographic heterogeneity. For saltmarshes, topographic heterogeneity can appear as channels and creeks on a macrolevel, but also as shallow dips and knolls on a microlevel (Morzaria-Luna *et al.*, 2004). This microtopography is a function of the rugosity, the average roughness of a soil, which is often lower in restored marshes due to flattening of the land by former agricultural management (Lawrence *et al.*, 2018; Brooks *et al.*, 2015). Brooks *et al.* (2015) found that areas in a restored marsh with lower topographic heterogeneity were lacking in vegetation coverage and biodiversity. These areas had an elevation level that is normally home to high marsh plant species and positive subsurface oxygen concentrations, which made the low topographic heterogeneity the most obvious reason for the lack of vegetation. Brooks *et al.* (2015) reasoned that water would drain away more easily on a more heterogenous surface with creeks, which might affect the vegetation composition. This explanation is supported by findings of Morzaria-Luna *et al.* (2004), who measured the microtopographic heterogeneity of different areas within a Californian saltmarsh: 25.5 % of areas with creeks had a categorically high microtopographic heterogeneity, compared to 0 % of areas without creeks.

In addition, Xie *et al.*, (2019) found that topographic heterogeneity facilitates seedling establishment on saltmarshes. They discovered a link between marsh zonation and the type of topography that is most beneficial for vegetation establishment. On the high marsh, areas with hummocks were found to trap more seeds. A flattened surface was more likely to dry out, which would increase salinity stress on the seed germination. However, on the low marsh, a flatter topography was deemed better for seedling emergence, as hummocks would retain too much water and reduce the oxygen exchange.

It is worth noting, however, that the relationship between topographic heterogeneity and vegetation does not go one way. The opposite of the abovementioned studies was observed by Stribling *et al.* (2006). They concluded that the presence of vegetation on a saltmarsh increases topographic heterogeneity, instead of the other way around. The two factors seemingly affect each other and an increase in one would likely increase the other.

The box below describes the original research that was done for this paper regarding the development speed of the topographic heterogeneity (Box 1). Most of the restored marshes discussed in the study did not become more heterogenous over the years; in fact, all but one regressed even further. This means the lingering effects of agricultural flattening will likely remain true for a long time.

Box 1: Development of Topographic Heterogeneity

Topographic heterogeneity in saltmarshes is beneficial for multiple factors involving the success of a marsh as a coastal protective measure. Topographic heterogeneity can positively affect saltmarsh drainage, through the formation of creeks (Lawrence *et al.*, 2018); the magnitude of wave attenuation (Moeller *et al.*, 1996); plant species richness (Brooks *et al.*, 2015; Morzaria-Luna *et al.*, 2004); and halophyte establishment (Xie *et al.*, 2019). However, these effects can be impeded on restored saltmarshes, as the topographic heterogeneity is often lower there compared to natural ones (Brooks *et al.*, 2015). To further improve future managed realignment schemes, it is therefore important to know how different the topography of restored marshes is compared to natural ones and if this changes over time. The following paragraphs will therefore try to find out how the topographic heterogeneity in restored saltmarshes changes over the years. Before this, the difference in topographic heterogeneity of restored marshes compared to natural ones will be investigated as well.

Figure B.1: *The location of the three 20 x 20 m squares on the Brancaster marsh. The red numbers indicate proximity to a first order creek, with square 1 being the closest.*

Method

The topographic heterogeneity was measured by calculating the rugosity, which can be described as the 'roughness' of an area. The rugosity was calculated by taking the standard deviation of the elevation of multiple saltmarsh DTM (Digital Terrain Model) files. The RStudio script 'moved' over the sites with a window of 3 x 3 cells, each time measuring the rugosity of the window. This data was later used to get the mean rugosity for each site. All of this was done using a script from RStudio, which can be found in the appendix.

The mean rugosity was measured for fourteen West European sites in total: three natural marshes from the Netherlands (Paulinapolder, Rattekaai and Saeftinghe), one natural marsh from the U.K. (Ramsey), four formerly embanked marshes from the Netherlands (Paardenschor, Rammegors, Schelphoek and Sieperda) and six formerly embanked marshes from the U.K. (Brancaster, Hesketh Out East, Jubilee, Orplands, Ramsey and Trimley). DTMs were collected from the database of the Department for Environment, Food and Rural Affairs or from the NIOZ archive.

In addition to the mean rugosity of the entire site, three 20 x 20 m squares per site were also examined individually. These squares where chosen in areas that had no creeks in them. Sometimes this was not possible because of the high creek density of the site, in which case the area with the least amount of creeks was picked. The squares ranged from close to a main creek (first order creek, according to Hack's stream order) to further away, with square 1 being approximately within 20 m of the creek, square 2 between 20 - 40 m and square 3 between 40 - 60 m (see Fig. B.1).

To measure the possible change in topographic heterogeneity over time, I calculated the mean rugosity of the most recent and oldest DTM that was available for each site and divided the difference in mean rugosity by the difference in years.

Results

Entire sites

When comparing the average mean rugosity of both marsh categories (natural and de-embanked), natural marshes seem to have a higher rugosity than de-embanked marshes (see Fig. B.2): the average rugosity for natural marshes is 0.080 ± 0.036, while the average for de-embanked marshes is 0.044 ± 0.016. However, as can be inferred from the high standard deviation seen in Figure B.2, there is a high variability amongst the values. Three out of the five natural marshes (Paulinapolder, Rattekaai and Ramsey) have a mean rugosity that are more similar to each other, ranging from 0.056 to 0.061. Saeftinghe noticeably differs from this range, with a value of 0.143. This high value could be explained by the high creek density, which also made it harder to extract squares without creeks in them.

The de-embanked marshes are even more variable and range from a minimum of 0.020 (Schelphoek) to a maximum of 0.071 (Orplands). The rest of the values are quite evenly spaced out between them. Orplands had been de-embanked for 24 years by the time the DTM was measured, making it one of the oldest sites in the dataset. Under the assumption that the rugosity increases over the years, it would make sense that the oldest marshes have the highest rugosity. However, there does not seem to be a correlation between the rugosity and the years since de-embankment. In fact, Trimley had been de-embanked for 19 years when it was measured but has the third lowest mean rugosity (0.030).

The lack of correlation between an increase in rugosity and time is even more clear when the speed of change in rugosity of the de-embanked sites are compared. As can be seen in Figure B.3, most of the sites appear to be regressing in rugosity, instead of increasing. Hesketh Out East is the only one with a positively developing rugosity. Paardenschor and Kennet Pans have not been included in this comparison, as I did not have access to older DTMs of these areas.

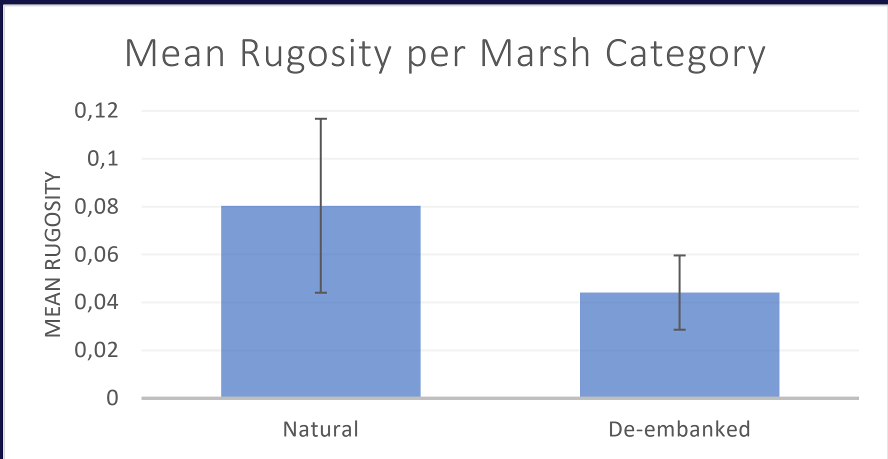

Figure B.2: The average mean rugosity of the natural sites compared to that of the de-embanked sites. This was calculated using data of the entire surface area of the sites. Mean rugosity was measured using a moving window of 3 x 3 cells.

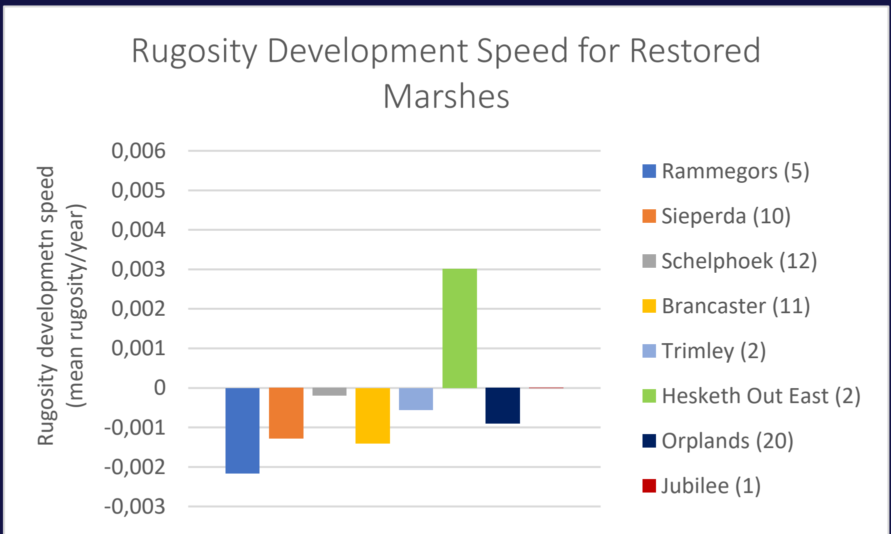

Figure B.3: The rugosity development speed for the restored sites in rugosity per year. The number next to the sites is the number of years between measurements. The rugosity of Hesketh Out East is increasing, while the rugosity of Rammegors, Sieperda, Schelphoek, Trimley and Orplands is decreasing. Jubilee is barely visible, as its value is very close to zero, though it is technically positive.

Squares

There seems to be a slight difference in the mean rugosity of squares on the natural marshes (Fig. B.4). Much like for the entire sites, the standard deviation of the population is quite large. Looking at the data, it is likely Saeftinghe again that causes this high variance, as its values are an order of magnitude higher than the rest of the dataset for square 1 and 3. These squares had some small creeks in them, which is likely the cause for the high values. This hypothesis is supported by the fact that square 2 did not have any creeks in it and has a lower standard deviation of population. If Saeftinghe is removed from the dataset, the values look more similar (Fig. B.5).. Though at first glance they seem to be slightly decreasing in heterogeneity the further they are from the creek, the standard deviation is too high and the error bars overlap too much to consider this significant.

Squares on the restored marshes are lower in topographic heterogeneity compared to those on the natural marshes (Fig. B.6). Much like on natural marshes (if Saeftinghe is not in the dataset), there is little difference in the topographic heterogeneity between squares.

There does not seem to be a trend in the distance to the nearest creek related to the development speed (Fig. B.7a,b,c). Topographic heterogeneity increases as well as decreases for every square, relative to each other. Interestingly, though the entire Trimley marsh seemed to regress in heterogeneity on average, the heterogeneity on the squares is increasing. A similar situation is true for Brancaster, of which square 1, the closest to the creek, has a positive development speed. This means that change in topographic heterogeneity is not necessarily uniform for an entire site, at least not for Trimley and Brancaster.

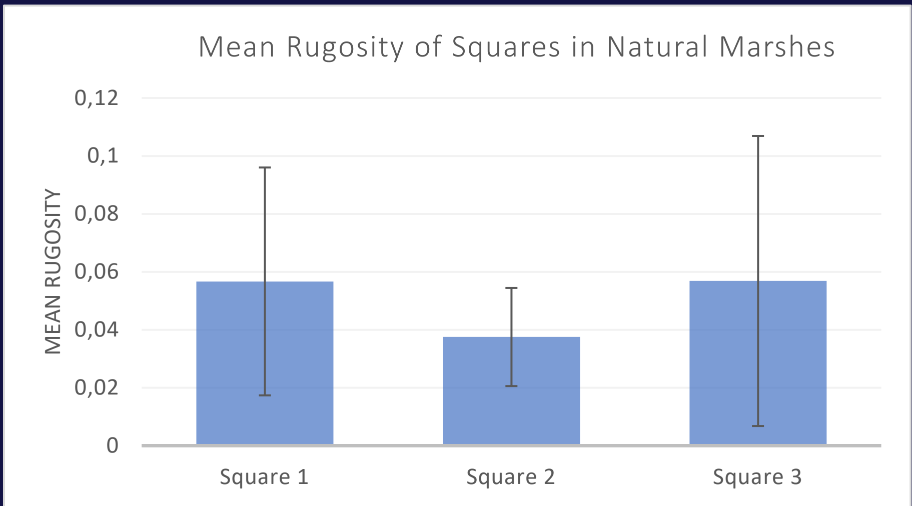

Figure B.4: *The mean rugosity for each square on the natural marshes. Error bars are created using the standard deviation of the population.*

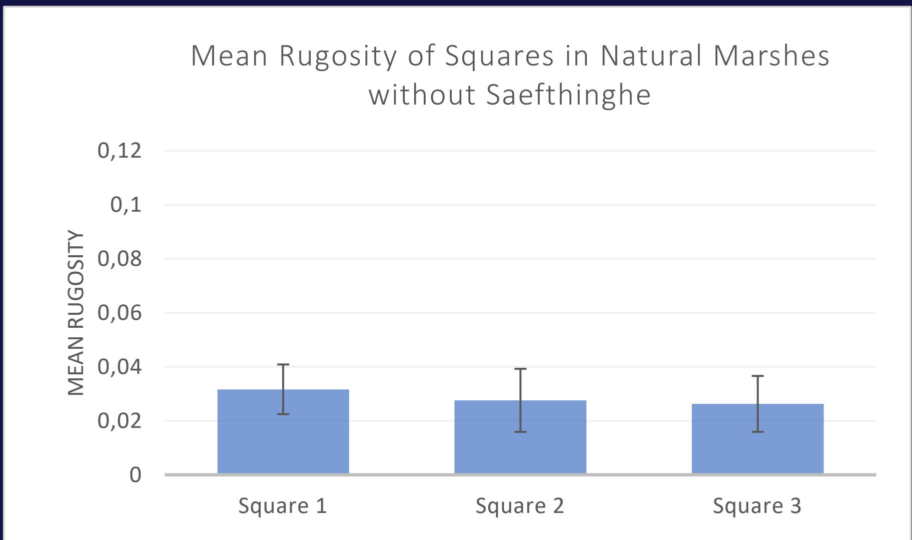

Figure B.5: *The mean rugosity for each square on the natural marshes without the data of Saeftinghe. Error bars are created using the standard deviation of the population.*

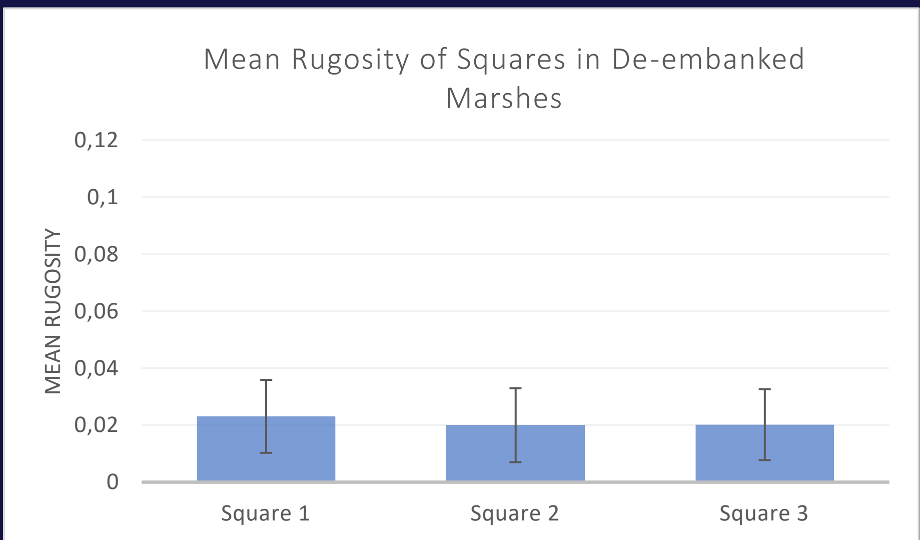

Figure B.6: *The mean rugosity for each square on the restored marshes. Error bars are created using the standard deviation of the population.*

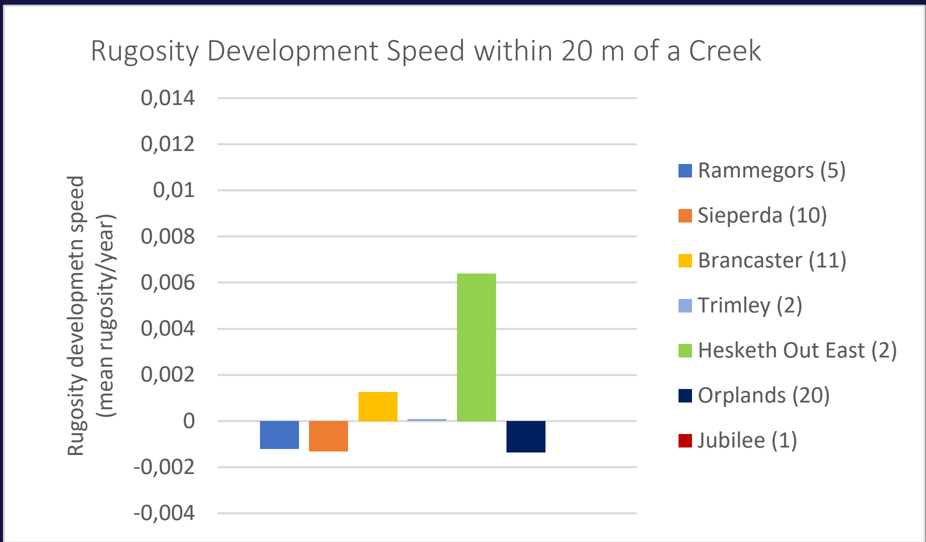

Figure B.7a: *The rugosity development speed in rugosity per year for the square within 20 m of the creek, square 1. The number next to the sites is the number of years between measurements. The rugosity of Brancaster, Trimley and Hesketh Out East is increasing, while the rugosity of Rammegors, Sieperda, and Orplands is decreasing. Jubilee is not visual as its value is zero.*

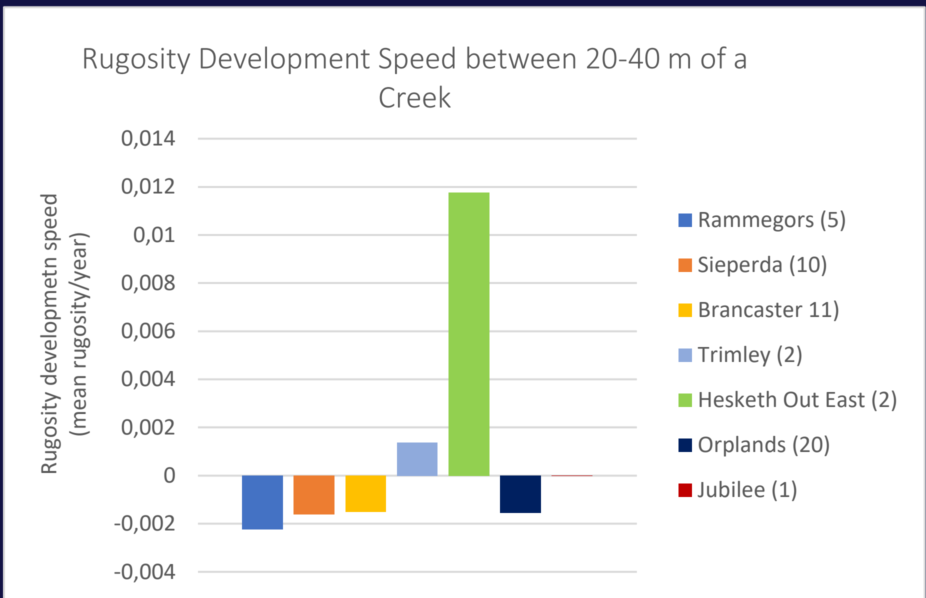

Figure B.7b: *The rugosity development speed in rugosity per year for the square 20 - 40 m away from the creek, square 2. The number next to the sites is the number of years between measurements. The rugosity of Trimley and Hesketh Out East is increasing, while the rugosity of Rammegors, Sieperda, Brancaster and Orplands is decreasing. Jubilee is not visual as its value is very close to zero, though it is technically positive.*

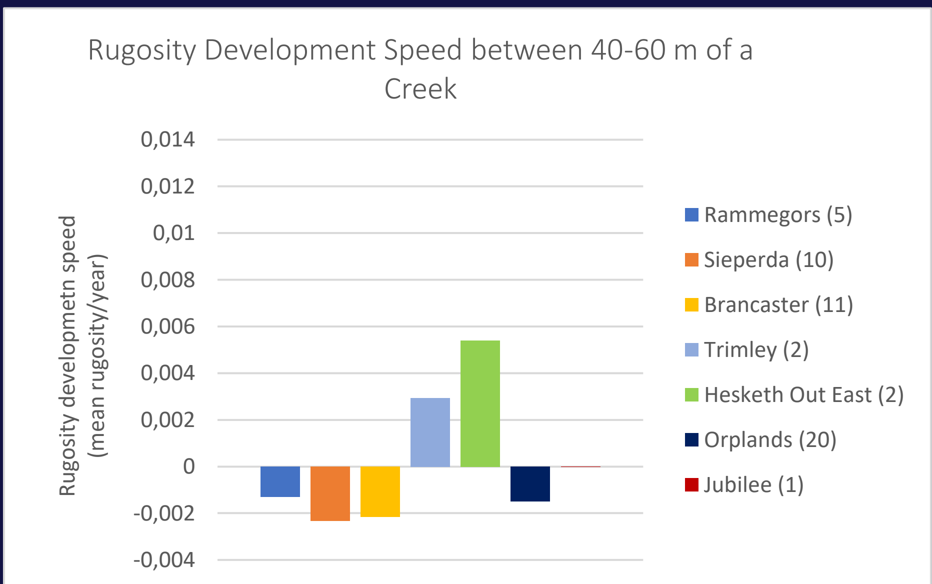

Figure B.7c: *The rugosity development speed in rugosity per year for the square 40 – 60 m away from the creek, square 3. The number next to the sites is the number of years between measurements. The rugosity of Trimley and Hesketh Out East is increasing, while the rugosity of Rammegors, Sieperda, Brancaster and Orplands is decreasing. Jubilee is not visual as its value is very close to zero, though it is technically negative.*

Conclusion

In this study, I asked the question whether the topographic heterogeneity varied between natural and formerly embanked saltmarshes and if it changed over time. There seems to be a difference between the mean rugosity of natural and restored marshes, although both categories have a high variability. While most of the values of the natural sites do seem to have a similar rugosity, the values of the de-embankment sites clearly variate among themselves. This is different from the findings of Lawrence *et al.* (2018): though the values of mean rugosity are similar, their study has less variation. This might be explained by the sample size, as Lawrence *et al.* (2018) investigated more than double the number of sites that were described in this study.

Furthermore, topographic heterogeneity does not seem to change in relation to proximity to a creek. There is a very small difference between square 1 and square 2 & 3, but this is not significant enough to call it a trend. A study with a larger sample size is needed to completely rule out the possibility, however.

Though the topographic heterogeneity does seem to change on restored marshes over the years, there is no clear trend that depicts how it will do so. Some sites – and areas within them – increase in heterogeneity, while others seem to regress. Proximity to the creek does not seem to be a factor that can explain this discrepancy, neither does the age of the marsh. Perhaps it might be influenced by the presence of excavated channels, as there could be a difference in topographic heterogeneity near manmade channels relative to natural creeks. It might also be valuable to investigate if a similar variation in development speed is present on natural marshes, or if this only happens on restored ones.

2. Chemistry

Nutrients play an important role in plant growth. Both macronutrients, like nitrogen and phosphorus, and micronutrients, such as iron and magnesium, are needed for vegetation development and survival (Mahler, 2004). However, not every plant species requires the same amount of nutrients to survive and a change in nutrient distribution might therefore affect vegetation composition (Levine *et al.*, 1998).

The former land management of a restored saltmarsh might have influenced the nutrient availability of the soil. Agricultural land is often fertilized with natural or artificial manure, which is abundant in nutrients. A surplus of fertilizer can lead to eutrophication of neighboring waters, which affects not just vegetation development, but the local ecosystem as a whole (Khan & Mohammad, 2014). However, if fertilizer is not used regularly, the soil can become exhausted and devoid of most nutrients (Gomiero, 2019). In line of this former management, it is important to examine whether these factors have an effect on vegetation growth and diversity in restored saltmarshes.

2.1 Fertilization

Intensive agricultural management can lead to soil exhaustion, as the crops eventually deplete the soil of the nutrients required for plant growth (Gomiero, 2019). To combat this, most farmers use a form of fertilizer, which can be either organic or chemical (Morari *et al.*, 2011). However, the continuous, long-term use of fertilizer can significantly impact the soil, especially when it is used for ecological restoration schemes (Geurts *et al.*, 2011).

In the first place, continuous fertilization can cause a build-up of nitrogen and phosphorus in the soil (Barberis *et al.*, 1996; Geurts *et al.*, 2011). This is especially true for phosphorus, as phosphate does not wash away with groundwater, but rather binds itself to soil particles (Khan & Mohammad, 2014). When the area is flooded, like in restored saltmarshes, the nutrients seep out of the soil or get washed away by erosion, and end up in the water (Geurts *et al.*, 2011; Khan & Mohammad, 2014). More nutrients are initially beneficial for vegetation growth: Darby & Turner (2008) reported that above-ground saltmarsh vegetation biomass increased by 169 ± 40 % in nutrient enriched sites. However, a surplus of nutrients also causes a reduction in root growth, as the plants do not need as many roots to get the required amount of nutrients for development (Valiela *et al.*, 1976; Darby & Turner, 2008). The decrease in the root : shoot ratio not only makes the plants more easy to topple (Deegan *et al.*, 2012), but it also causes the vegetation to be less resistant towards inundation (Wong *et al.*, 2015). In addition, as was shown by Deegan *et al.* (2012), an increase in above-ground biomass on natural marshes in Massachusetts, U.S.A., resulted in more food for denitrifying organisms, which in turn increased the amount of fine organic matter particles in the soil, as the organisms take apart the bigger chunks of dead organic material. The study also reported how the increase in decomposition paired with the loss of root biomass, caused the creek bank to become less stable and crack: large organic matter particles and drainage of macropores decreased as a result of more decomposition and less root biomass, respectively. This resulted in a decrease in the soil's ability to drain and the creek bank consequently became waterlogged. The added weight of the water caused parts of the bank to collapse at low tide. As a result, previously vegetated parts of the observed marsh turned into bare, muddy areas.

Nutrient-enriched sites might also affect the plant species composition of a saltmarsh. Levine *et al.* (1998) showed that competitive dominancy between four common saltmarsh plant species in the U.S.A changed under different nutrient conditions: the species that was the most dominant under limiting nutrient conditions was outcompeted by the others when nutrients were abundant (see Fig. 4). A similar shift in vegetation dominance was witnessed by Emery *et al.* (2001), who suggested nutrient supply likely changed how dominancy was decided among plants, as competing for light could shift to competing for nutrients. However, Johnson *et al.* (2016) found no difference in competitive dominancy between the same species and argued that the previous studies used too much fertilizer in their experiments, which resulted in nutrient levels not representative of real-world conditions. To further emulate natural conditions, the experiment of Johnson *et al.* (2016) also added the nutrients to incoming tidal waters instead of directly fertilizing the soil. Nevertheless, none of these studies were carried out on restored marshes, which might affect the available amount of nutrients and could increase them to levels that do cause a shift in species dominancy.

Another way in which a build-up of nutrients might influence vegetation survival, though indirectly, is the proliferation of algal blooms. Aquatic (macro)algae are benefitted by an increase in nitrogen and phosphorus, and population sizes often increase exponentially (Teichburg *et al.*, 2010). A study by Wasson et al. (2017) in California, U.S.A., described a direct link between the nutrient conditions of a saltmarsh and the exponential growth of macroalgae. Washed up macroalgae, also known as "wrack", were found to decrease vegetation cover, flowering and height, as well as increase vegetation retreat from creek edges. This was mostly due to lack of light and/or oxygen resulting from the agal wrack cover. Little other research is done regarding the effect of exponential algal growth on saltmarsh vegetation, especially restored ones, yet nutrient concentrations of coastal water

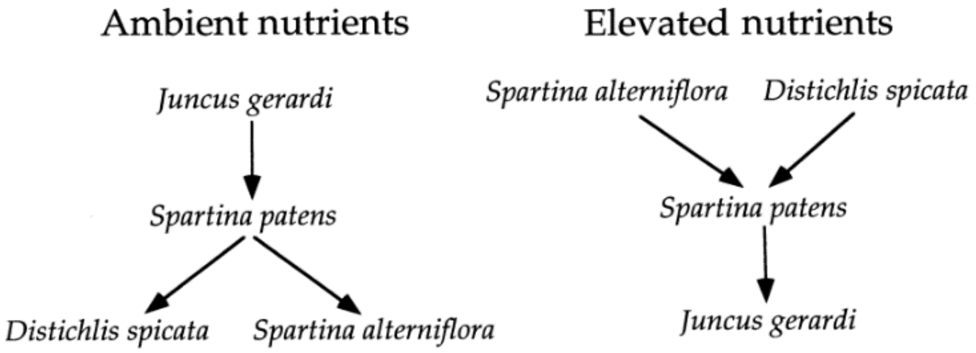

Figure 4: *A visual representation of the change in competitive hierarchy of common North American saltmarsh plant species under ambient and elevated nutrient conditions (Levin et al., 1998).*

have increased worldwide (Rabalais *et al.*, 2009). It can therefore be assumed that similar consequences as described by Wasson *et al.* (2017) might occur elsewhere as well.

2.2 Soil acidification

Saltmarsh soils are normally known to be alkaline, because of the calcium bicarbonate in seawater (Wherry, 1920). A study by Portnoy (1999) showed that when a saltmarsh is embanked and drained, the soil gets more in contact with the air, which can lead to pyrite oxidation. Pyrite oxidation results in the release of sulfate, which causes a decrease in the pH level of the soil: the pH of the studied marsh declined to a level of less than 4. However, when seawater was added to a sediment core of the saltmarsh, the pH increased to about 6 in a few months. This would suggest that acidification of the soil prior to de-embankment has little effect on the soil pH after breaching. However, seawater mostly affects the lower marsh system, as the high marsh is only flooded during storms or springtide (Chirol *et al.*, 2018). This could mean that high marsh soils remain acidic for longer. Excessive use of fertilizer before de-embankment could further exacerbate this situation, as it leads to acidification of the soil in its own right (Wallace, 1994). In fact, the average pH of arable land in Western Europe is 5.2, with pastures going even lower to an average pH of 4.8 (see Fig. 5) (Fabian *et al.*, 2014). More research would have to be done on the pH levels of restored marshes to confirm this hypothesis, however.

Soil acidification has many adverse effects on vegetation growth: dissolution of aluminum ions, which can be toxic to plants and limit root growth, leading to the proliferation of aluminum-tolerant species (Yadav *et al.*, 2020; Matsumoto *et al.*, 2017); increased chances of plant diseases (Li *et al.*, 2017); and decreased abundance and biodiversity of useful microbes (Rousk *et al.*, 2010).

However, little research has been done on the effects of soil acidification on saltmarsh species. A study by Allen *et al.* (1997) did find that a halophytic species (*Schoenoplectus pungens*) found on saltmarshes only grew in a plot with a pH of 8.5, but the experiment was carried out on an inland basin in New Zealand and might therefore not be representative of conditions on a saltmarsh.

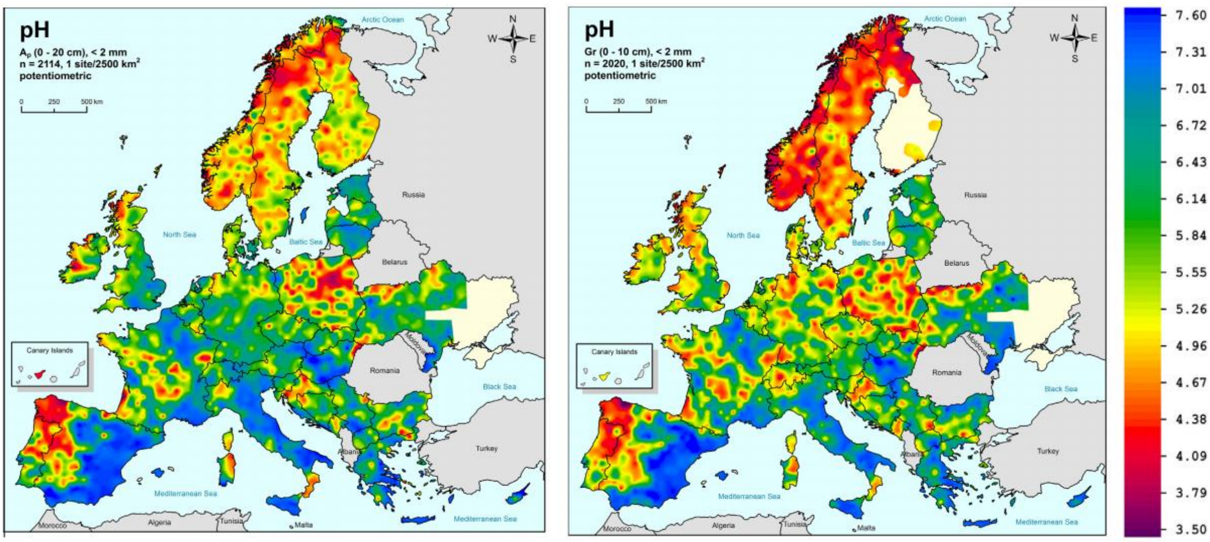

Figure 5: Maps of Western Europe depicting the average pH values of the soil of arable land (left) and pastures (right). The maps were constructed using GEMAS soil samples (Fabian et al., 2014).

Discussion

The purpose of this paper was to provide an overview of soil properties that could influence the vegetation growth and diversity on formerly embanked saltmarshes. The soil properties of de-embanked saltmarshes are not the same as in natural marshes, due to the former agricultural management of the land. The sediment has often been compacted by the weight of large machinery, which can lead to a low surface elevation, a lack of oxygen exchange between the soil and the air, a high penetration resistance and a lack of creek formation. Agricultural soil is also more likely to have been flattened, which causes a lack of topographic heterogeneity that will likely remain for a long time. Excessive use of fertilizer could have had an impact on the chemistry of the soil as well, which might lead to a high nutrient environment and acidification of the soil. The physical and chemical stresses for vegetation that come with these circumstances could explain the different vegetation biodiversity that has been observed in the formerly embanked saltmarshes (Wolters *et al.*, 2005; Garbutt *et al.*, 2006; Barkowski *et al.*, 2009; Spencer *et al.*, 2017).

Effects of low surface elevation in relation to vegetation diversity and climate change

One of the more prevalent effects of soil compaction in restored marshes discussed in this paper is the low surface elevation. Marsh vegetation is normally divided in different zones, which have different properties (such as soil water content and salinity), allowing for a diverse array of species. As the vegetation zonation relies on the surface elevation, a low elevation can consequently cause a decrease in the biodiversity: species that are more common in high marsh zones now have less place to grow and decline in abundance. As a result, pioneer and low marsh species are dominant. Furthermore, a study by Pennings *et al.* (2005) noticed that a low marsh species common in North American saltmarshes (*Spartina alterniflora*) was also able to live in a high marsh environment, suggesting that their growth is not bound by surface elevation. The reason that they are more common on low elevations was explained by competition with high marsh species on higher elevations. However, in their experiment, a high marsh species (*Juncus roemerianus*) was already established before they introduced *Spartina* and while the presence of *Juncus* did hinder *Spartina* growth, the question remains whether *Juncus* would have had the same effect if it was introduced after *Spartina* establishment. If this is not the case, it could indicate that initial dominance of low marsh species earned by early establishment might be less affected by the presence of high marsh species, which would further impede high marsh species abundance.

Not only does a lack of biodiversity directly counteract the goal of ecosystem preservation in saltmarsh restoration, but the species of the high marsh are also important for the attenuation of high storm waves (Stark *et al.*, 2015). As the frequency and intensity of storms are increasing with global warming, the high marsh zone becomes increasingly important, especially when the purpose of the restoration is to strengthen coastal defenses. However, saltmarshes are not exempt from consequences of climate change themselves either: though they seem to be keeping up with rising sea levels normally by retreating more inland, marshes that are barred from the mainland by dikes have no way of doing this (Oppenheimer *et al.*, 2019). This would mean more of the marsh would be frequently flooded and more soil would become waterlogged. As was stated before, this can affect the biodiversity because species vary in their ability to cope with high soil water content. It is therefore important for the biodiversity – as well as the overall exosystemic functions of the marsh – that channels and creek networks develop to stimulate marsh drainage.

Creek formation and topographic heterogeneity are hindered in restored marshes

Soil compaction makes it harder for the soil to erode, and thus hinders the formation of creeks and channels. A layer of relatively easy to erode topsoil is usually deposited through the tides, making creek development more achievable, but this deposition is less apparent at higher, less frequently flooded elevations. There, channel development has to start almost directly on the compacted soil. Still, drainage is negatively affected on the low marsh, as groundwater can still only go as low as the compacted soil, which acts as a barrier (also called an 'aquaclude') (Crooks *et al.*, 2002).

Another factor that seems to influence creek formation is the topographic heterogeneity. Based on the findings discussed in this paper, topographic heterogeneity is both increased by the presence of vegetation, as well as the cause for more vegetation. Its stimulating effect on creek development will positively affect the vegetation, as creeks not only contribute to marsh drainage but also serve as a conduit for nutrients (Sanderson *et al.*, 2000). Nevertheless, topographic heterogeneity is lower on restored marshes than natural ones due to the agricultural flattening of the land during pre-embankment times (Lawrence *et al.*, 2018). It can be inferred from the original study that was detailed in this paper that it can take quite some time before the topographic heterogeneity of a restored marsh resembles that of a natural one, if at all. In fact, most of the studied sites appeared to be decreasing in heterogeneity even further. Whether or not a similar decrease can be witnessed in natural marshes remains unknown and therefore more research needs to be done.

Complications for the high marsh

While lack of creek formation and soil drainage are more pertinent a problem on the lower elevations, higher elevations are not exempt from complications either. As the area is less frequently flooded, there is less need for creeks, but it also means there is less sediment deposition. Root and shoot penetration will likely be harder here, as the compacted soil remains closer to the surface. This is especially hard for young plants, which shoots are not as strong (Nawaz *et al.*, 2013). Seedling emergence might therefore face greater difficulty. As for roots, limited elongation is detrimental to a plant's ability to respirate and obtain nutrients from the soil and could therefore hinder vegetation development (Nawaz *et al.*, 2013). Yet, the abundance of nutrients that is left over from crop fertilization might mitigate this problem, since the plants now have to work less hard to gain the required amount of nutrients. After all, an observed plant response to a high nutrient environment was a decrease in root elongation as well (Darby & Turner, 2008). This may, however, lead to toppling of the plants, as the root : shoot ratio decreases (Deegan *et al.*, 2012). Still, this effect was only observed on a site in Massachusetts, USA, and might therefore not be representative of marshes that are located elsewhere.

Limitations and suggestions for further research

Though the majority of studies discussed in this paper were focused on West European saltmarshes, surrounding the North Sea, lack of previous research caused me to take into account different locations as well. Similar experiments will have to be carried out on West European marshes to find out whether their results are location specific. The same is true for studies about non-saltmarsh plant species, which were used to illustrate how plants might react to different environmental stresses. If anything, this paper can be seen as an overview of possible soil-related issues that could be present on de-embanked saltmarshes, from which ideas for research can be obtained.

The topics that are discussed in this paper do not occur in a vacuum, however. They likely interact with each other, as well as other factors that have not been mentioned because they fell outside of the scope of this review. For instance, the vegetation biodiversity might also be affected by the presence of species neighboring the saltmarsh that can give off seeds (Morzaria-Luna & Zedler, 2007). Furthermore, there might be a difference in the effects of man-made creeks, which are sometimes excavated before de-embankment, and naturally developed channels on vegetation development (Barkowski *et al.*, 2009). Natural channels are usually meandering, whereas manmade ones are often straighter.

It should also be noted that soil compaction and a high nutrient environment are not solely issues on restored marshes. The weight of new layers of sediment causes a natural pressure on the soil, which can lead to compaction (Bartholdy *et al.*, 2009). Nevertheless, soil compaction on restored marshes is likely higher due to the addition of compaction caused by heavy machinery. A similar situation is true for the high nutrient environment: while eutrophication is a global problem that affects both natural and restored marshes, restored marshes can also have a soil that has been fertilized continuously for multiple years, if not decades. This will add even more nutrients, like nitrogen and phosphorus, to the environment. However, it is still unknown how long these added nutrients stay in the system. They might have the most considerable effect shortly after de-embankment, as the tides might eventually drain the marsh from nutrients. This would mean that higher elevations of the marsh that are less frequently flooded will retain the abundance of nutrients for longer. Combined with the effects of soil acidification caused by the embankment and drainage of marshes, which also likely remains longer on higher elevations, the soil chemistry of the high marsh will presumably be different from the low marsh.

Conclusion

The goal of this paper was to give an overview of possible ways in which the soil properties of formerly embanked saltmarshes could affect vegetation diversity and abundance. Soil compaction induced by heavy tillage machinery and cattle trampling has seemingly the biggest impact on vegetation. The soil is harder to penetrate by roots and shoots and is less equipped to drain the marsh, which can cause a hypoxic environment. Creek development is hindered in areas where formation has to start directly on the compacted sediment, and this is exacerbated by the low topographic heterogeneity caused by earlier flattening of the land for crop-fields. The topographic heterogeneity will likely decrease even further over time. Furthermore, surface elevations are generally lower, leaving less room for high marsh species.

Though not much information is known about the effects of earlier fertilization on vegetation of de-embanked saltmarshes, there is reason to believe it could lead to a decrease in the root : shoot ratio, making plants more likely to topple. It might also lead to algal proliferation, which results in an abundance of wrack and is consequently detrimental to plants health. Soil acidification, especially on the higher marsh, can add to this as well.

The abovementioned factors affect the abundance as well as the biodiversity, as some species are more equipped to deal with certain stresses. The intensity of certain factors also seems to be related to the marsh elevation, as the elevations are differently influenced by the tides. A visual representation of this zonation can be found in Fig. 6. The low marsh will likely be less affected by soil acidification or

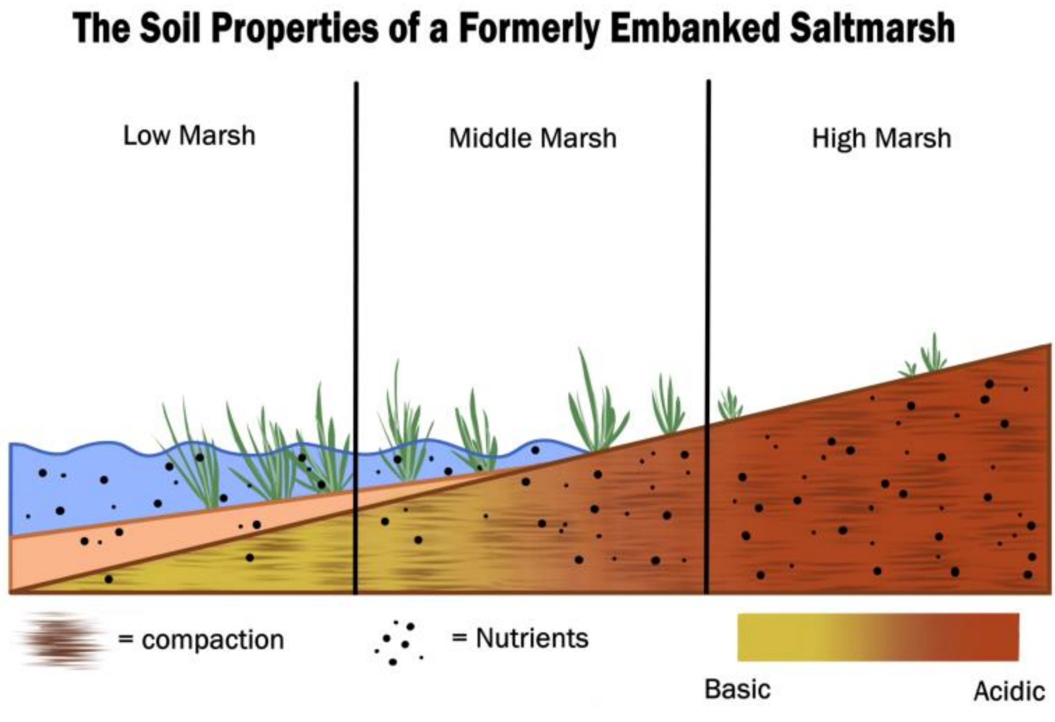

Figure 6: *A visual representation of likely soil circumstances on restored saltmarshes. Compaction is apparent in every part of the sediment, except the topsoil layer on the low and middle marsh. Similarly, lower elevation levels likely have less nutrients in the soil as the seep out to the water.*

compaction, though the latter is still present as an aquaclude underneath the newly deposited topsoil layer. Evidence suggests that any abundance of nutrients will seep out of the soil and might therefore have more effect on the environment than on the high marsh, where the nutrients will likely remain in the soil for longer. The high marsh is also more affected by acidification and compaction, though this only problematic for root/shoot elongation and not for creek formation, as that is less needed on higher elevations. The middle marsh will probably be affected most of all, as the circumstances of the low and high elevations come together here.

Future managed realignment schemes should consider reducing soil compaction as much as they are able to. There have already been cases where the topsoil was removed before breaching the embankment, but this is a costly affair (Groeneveld & Bunje, 2020). However, considering the effects of soil compaction described in this paper, it might be worth the costs. Furthermore, more research should be done concerning the effects of excessive fertilizer use as there is reason to believe this could negatively affect vegetation growth and diversity.

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Appendix

Rstudio script for the mean rugosity

```
# Install and call the required packages
require(rgdal)
require(raster)
library(zoom)
library(oceanmap)
library(rgrass7)
library(sp)
library(tictoc)
library(viridis)

# Choose a color map
colRGB = viridis(1000)

# Load the DTM raster file
M = raster("DTM.tif")

# Calculate slope (in degrees)
slope = terrain(M, opt="slope", unit='degrees')

# Get mean slope for the site
cellStats(slope, stat='mean', na.rm=TRUE)

# Plot the resulting slope map
plot(slope, col=colRGB)

## Calculate rugosity (st. dev. of elevation) using a "moving window" of 3 x 3 cells
RUG <- (focal(M, w = matrix(1,3,3), fun = sd, na.rm=TRUE))
plot(RUG, col=colRGB)

# Get mean rugosity for the site
cellStats(RUG, stat='mean', na.rm=TRUE)
```