Landscape and ecological modelling: Development of a plant community prediction tool for Estonian coastal wetlands

Raymond David Ward

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Abstract

Estonian coastal wetlands are of international importance as they support characteristic biological diversity. Their limited extent and distribution mean that these wetlands are of high conservation concern, and as such have been identified as a priority in the European Union Habitats Directive. These wetlands are typified by a flat, extensive landscape, situated between the micro-tidal (<0.02m), brackish Baltic Sea and a forest interior. Due to the low relief these wetlands may be under threat from sea level rise. This research consisted of four studies:

(i) to determine and quantify the relationship between a range of coastal wetland plant community types, elevation and edaphic conditions. Results demonstrated that plant community distribution was significantly affected by micro-topography and edaphic variability. The majority of the plant communities were discernible in the field by elevation alone and elevation was found to be the factor that could distinguish the greatest number of plant communities. (ii) to determine an appropriate method of interpolating LiDAR elevation data and assess the use of LiDAR data in creating a static correlative model to determine plant community type based on elevation. Results showed that with dGPS calibration the model could accurately predict plant community location. Validation of the model in two further sites showed that the correlative model was able to predict plant community with almost perfect (κ 0.81) and moderate agreement (κ 0.53) dependent on the site. (iii) to determine sediment accretion rates to complete the dynamic model by analysing the level of radionuclides, ^{137}Cs and ^{210}Pb, in discrete core sections. Results showed that during periods of greater storminess sediment accretion increased almost threefold. These sensitivity data were included in the dynamic correlative model. (iv) to assess the effects of sea level rise on plant communities in Estonian coastal wetlands under five sea level scenarios, two accretion rate scenarios and factoring in isostatic uplift rates. Results showed that local sea level will rise in some sites and decrease in others dependent on location and SLR scenario.

This study has indicated that in many instances Estonian coastal wetlands will increase in extent in the future due to high rates of sediment accretion, particularly in a scenario with more frequent storms, and isostatic uplift. The study has shown that following validation, calibration and sensitivity analysis LiDAR data can be used to accurately predict plant community type in microtopographical ecosystems. The model developed in this study of Estonian coastal wetlands is likely to be transferable to other appropriate habitats such as tidal, estuarine, and floodplains wetlands.

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Declaration

I declare that the research contained in this thesis, unless otherwise formally indicated within the text, is the original work of the author. The thesis has not been previously submitted to this or any other university for a degree, and does not incorporate any material already submitted for a degree.

Signed:

Dated:

1 Introduction

1.1 Research context

At present biodiversity loss is a worldwide concern due to impacts from a variety of anthropogenic factors (Krebs, 2001). Amongst the most serious threats is climate change (IPCC, 2007). In coastal areas the biggest threat from climate change in many areas is sea level rise (Morris *et al*., 2005). Estonia has a long coastline of 3794 km and many coastal areas are low lying, and hence they are potentially at risk from sea level rise. Estonian coastal wetlands, which are classified as boreal Baltic coastal wetlands according to the EU Habitats Directive (1992), occur in sheltered bays and coastlines and are characterised by low relief, often not exceeding a maximum elevation of 2m above mean sea level (Truus, 1999). The low relief of Estonian coastal wetlands means that these are the coastal zones of Estonia most likely to be under threat from sea level rise (Kont *et al*., 2008). Sea level rise is likely to be somewhat offset by post glacial isostatic uplift and sediment accretion in many Estonian coastal wetland sites (Suursaar & Kullas, 2009) although no sediment accretion data are available.

Estonian coastal wetland systems are a priority habitat, according to the EU Habitats Directive (1992). They consist of coastal wet grasslands, swamp vegetation on the seaward edge, and scrub vegetation on the landward side. Estonian coastal wetlands are formed and maintained by isostatic uplift and management, usually in the form of low intensity grazing or mowing (Jutila, 2001). In spite of their ecological importance Estonian coastal wetlands have been subjected to habitat degradation in the form of agricultural intensification during Soviet times and subsequent abandonment following the collapse of the Soviet Union (Puurmann & Ratas, 1998; Busmanis *et al*., 2001). Recent efforts have been made to reintroduce low intensity management in many of these semi natural Estonian coastal wetlands, with considerable success (Rannap *et al*., 2004; Berg, 2009).

The Estonian coastal wetland habitat mosaic supports characteristic plant species and provides a habitat for a wide range of migratory and breeding bird species (Puurmann & Randla, 1999) as well as a variety of plant species on the edge of their ranges (Paal, 1998). However, there is little information available as to the current location and extent of the plant communities within Estonian coastal wetland landscapes.

1.2 Research aim

The aims of this research were: (i) to generate a predictive model of plant communities in Estonian coastal wetlands using field-based ecological and geomatic techniques and LiDAR data, and (ii) to apply this model to assess the impact of sea level rise (SLR) on the distribution of plant communities within three Estonian coastal wetland sites.

The objectives of the study were: (i) to determine and characterise the relationship between a range of coastal wetland plant community types, elevation and edaphic conditions; (ii) to assess the use of LiDAR data to develop a model to determine plant community distribution based on fine scale elevation data; (iii) to conduct a sensitivity analysis of the plant community model using ground recorded dGPS elevation data and sediment accretion rates and (iv) to assess the effects of SLR on the plant communities of Estonian coastal wetlands under five sea level scenarios and two accretion rate scenarios and factoring in isostatic uplift rates

These data and results will be useful for land managers and stakeholders of Estonian coastal wetlands in assessing the threat level of sea level rise and in planning any future response. Further to this, the model could be transferred to other important international sites for the assessment of open wetland landscapes such as salt marshes and floodplain meadows.

1.3 Research approach

This research was focussed on a study of the plant communities in Estonian coastal wetlands, in particular to develop a model to assess their present extent using a variety of field based techniques and geomatic data, and to further use the plant community model to assess the impacts of sea level rise.

Field studies were based in three Estonian coastal wetlands in north west and west Estonia. The field sites were chosen as they were representative of semi-natural Baltic coastal wetlands in Estonia with swamp vegetation, coastal grassland, and the rarely inundated scrub and developing woodland vegetation. Field data were collected in one Estonian coastal wetland site in the summer of 2009 in order to assess and quantify the relationship between plant community type and elevation and edaphic variables. A model was developed combining plant community ecology with a GIS and geomatics unifying approach. Previous studies have used a similar approach in salt marshes (Morris *et al*., 2005) or salt meadows (Moeslund *et al*., 2011), using remotely sensed altimetry data to determine plant communities. These studies have been able to identify broadly different plant communities in tidal areas. However, in micro-tidal areas such as the eastern Baltic (~0.02m) there are very small differences in elevation above mean sea level (m.s.l.) between the plant communities. Previous models would therefore not be able to distinguish between different plant communities in Estonian coastal wetlands.

Sediment core data were taken from two Estonian coastal wetland sites over the 2010 field season in order to assess past sediment accretion rates using radionuclide dating and because accretion data were not available. The sediment accretion data were extrapolated and used as an integral dataset in a predictive plant community model taking into account isostatic uplift and eustatic sea level rise linked to climate change.

1.4 Thesis structure

Chapter 1: This chapter provides the research context, explains the aims and thesis structure.

Chapter 2: This chapter provides a review of the current and past research that has been conducted in Estonian coastal wetlands and similar habitats. In particular focus is placed upon literature regarding ecological, geomatic and sediment dating techniques in appropriate habitats. It illustrates the applicability of geomatic techniques in ecological studies.

Chapter 3: This chapter provides an introduction to the characteristics of the study sites. It also provides details of general methodological techniques that are used in the analysis chapters 4, 5, 6, and 7.

Chapter 4: Chapter 4 contains details of the study assessing the relationship between a variety of environmental variables and the plant communities in Estonian coastal wetlands. This chapter provides a quantification of the relationship between elevation and plant communities. This relationship has not previously been quantified although a variety of authors have suggested that it exists. The quantification of the elevation – plant community relationship was necessary for the development of the predictive plant community model developed in chapter 5.

Chapter 5: Chapter 5 uses remotely sensed elevation data to develop a predictive plant community model based on the data derived from chapter 4. The methodology included a calibration and adjustment of the remotely sensed elevation data. A dGPS approach was used to capture accurate field derived elevation data and provided a correction value for the model. The corrected remotely sensed elevation data were categorised according to the elevation preferences of each plant community type. The resultant predictive plant community model was tested in two non-contiguous coastal wetlands. The model formed the platform for a predictive model to assess future plant community changes with regards to anthropogenically induced sea level rise (slr). The methodology developed from chapters 4 and 5 forms the basis for a new rapid landscape assessment tool.

Chapter 6: Chapter 6 analyses past sediment accretion rates and explains variation in sediment accretion over time. The sediment accretion rates were derived from an assessment of artificial (^{137}Cs) and natural (^{210}Pb) radionuclides in discrete sections of sediment cores. ^{137}Cs derived sediment accretion rates were used as a validation of the ^{210}Pb derived sediment accretion rates. These data were essential for any model predicting plant community type based on the elevation relationship in sea level rise scenarios and there are no sediment accretion rate data available for Estonian coastal wetlands.

Chapter 7: Chapter 7 uses the model developed in chapters 4, 5 and 6 as a basis to predict the location and extent of plant communities in three Estonian coastal wetland sites in 2099. A variety of scenarios were used taking into account: four IPCC (2007) slr scenarios and one scenario with no change in eustatic sea level, post glacial isostatic uplift, and sediment accretion (extrapolated from chapter 6). The results provide important information for land managers to assess how best to deal with potential changes to the plant communities, which provide an internationally important habitat for both breeding and migratory birds.

Chapter 8: This chapter discusses the methodological considerations in the study, including limitations in the model and how these can be overcome in the future. The chapter also discusses further applications of the model in other appropriate open environments.

Chapter 9: This chapter presents the key findings of the research project and provides recommendations for further study.

1.5 Nomenclature

Geology: Geology follows Raukas, A. and Tavast, E. (1994)

Plants: Scientific names follow Stace, C. (2010)

Animals: Scientific names follow Burfield, I. and van Bommel, F. (2004)

Units: SI units are used but GIS distance units are in kilometres and accretion and isostatic uplift rates are in millimetres.

2 Literature review

2.1 Preamble

This chapter is divided into three main sections: (i) an introduction to coastal wetlands, with particular reference to those in Estonia and the plant communities that occur there, (ii) a discussion of the main literature regarding the environmental variables that affect plant communities in coastal wetlands with particular reference to Estonia, and (iii) a critical analysis of vegetation studies with particular reference to the recent use of GIS and remote sensing for modelling plant communities.

2.2 Coastal wetlands

Wetlands have a global distribution (figure 2.1) and have a variety of definitions. One of the most widely accepted definitions is that of the International Union for the Convention of Nature and Natural Resources (IUCN) "areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish, or salt including areas of marine water, the depth of which at low tide does not exceed 6 meters" (Davies et al., 2004).

Figure 2.1: The global distribution of wetlands (Wetlands International, 2007).

Wetlands are often located at the margins of water bodies and terrestrial ecosystems and hence this has been taken as evidence by some scientists that wetlands are merely extensions of the adjacent terrestrial or aquatic ecosystem (Mitsch & Gosselink, 2000). However, wetland scientists take the view that wetlands possess some properties that are distinct from both terrestrial and aquatic ecosystems such as edaphic factors and hydrology. It is these distinct properties which will be discussed in this literature review and will be used to develop the plant community model which is the subject of this thesis.

Coastal wetlands are found in all continents and at all latitudes and include salt marshes, mangroves, tidal flats, sea grasses and coastal grasslands and coastal swamps (Wolanski *et al*., 2009). Wetlands are characterised by presenting high water tables either at or above the surface, or within the root zone. They often have different soil conditions from adjacent uplands, and contain vegetation which is tolerant to prolonged inundation (hydrophytes) (Mitsch & Gosselink, 2000).

In a European context coastal wetlands have been divided into: estuaries, mudflats and sandflats, coastal lagoons, large shallow inlets and bays, boreal Baltic coastal meadows, and boreal Baltic narrow inlets by Annex 1 of the EU Habitats Directive (1992). However, these are designations set by policy makers and often these designations are overly broad and can cover a variety of different habitats. A further discussion of habitat classification systems will be discussed later in this chapter. All of the aforementioned European coastal wetlands are typified by low relief, with micro-topographical variations, and are often found along protected shorelines such as in harbours, estuaries and behind sea defences (Perillo *et al*., 2009). However, relief can be more varied in sites with a greater tidal range. The typically low relief is a result of both coastal wetland evolution and geology. The relationship between relief and sea level has been discussed by a variety of authors for estuaries (Allen, 2000; Allen & Haslett, 2007; Kirwan & Murray, 2007), mudflats and sandflats (Dalrymple *et al*., 1992; Dyer *et al*., 2000), and coastal lagoons (Nichols, 1989; Ward & Ashley, 1989). However, there have been no studies conducted on the relationship between sea level and relief on boreal Baltic coastal meadows.

2.2.1 Boreal Baltic coastal wetlands

Boreal Baltic coastal wetlands are generally not natural landscape features. They occur on uplifting and sheltered coastlines in Estonia and to a lesser extent Sweden, Finland and Latvia (EU Habitats Directive, 1992). It has been suggested by a variety of authors that Boreal Baltic coastal wetlands have typically been formed by grazing or mowing of areas which prevents succession to coarser vegetation following uplift from the sea (Tyler, 1969; Wallentinus, 1973; Jutila, 1999). Some are formed through human activities such as forest clearance and are then maintained by mowing or grazing (Puurmann & Ratas, 1998; Puurmann & Randla, 1999; Truus, 1999). Although management is a major factor in the development of Boreal Baltic coastal wetlands there are, however, a variety of other factors which determine both the relief and vegetation. Boreal Baltic coastal wetlands are characterised by periodic flooding or a high water table (Tyler, 1971b) by the brackish Baltic Sea. Boreal Baltic coastal wetlands support a range of different plant communities such as swamp vegetation, wet grassland and scrub. The plant species present are a mixture of grasses, broad leaved herbs, sedges and rushes in addition to some typical salt marsh plants (e.g. *Glaux maritima*, *Juncus gerardii*, *Salicornia europaea* and *Suaeda maritima*) (Tyler, 1969; Wallentinus, 1973;. Rebassoo, 1975; Burnside *et al*., 2007). Boreal Baltic coastal wetlands have a high conservation value and contain large numbers of rare and threatened plant species. Moreover, these systems also contain plant species that are typical to both grasslands and coastal wetlands. The functioning and composition of these coastal wetlands are controlled by a variety of different factors such as: hydrology, micro-topography, nutrient availability, salinity, sediment accretion, and management; many of which have not been fully studied. These factors and the literature available are discussed in detail in this chapter.

2.2.2 *Coastal wetland plant communities*

The coastal wetland environment, being neither terrestrial nor aquatic, is physiologically harsh due to stresses in anoxia, salinity and water fluctuations (Mitsch & Gosselink, 2000). Anoxia, salinity and water fluctuations require a variety of physiological and ecological adaptations, which in turn affect the structure of the plant community.

Coastal wetland plant communities are subject to a variety of controlling factors (Keddy, 2000; Mitsch & Gosselink, 2000). Physical factors that influence wetland plant community structure are: the hydrological period, hydrological gradient (Tyler, 1971a), salinity gradient (Tyler, 1971b), nutrient availability (Tyler, 1971c), pH, soil type (Mitsch & Gosselink, 2000) and micro-topography (Vivian-Smith, 1997).

Figure 2.2: A simplified view of vegetation zonation in Estonian coastal wetlands using the Paal (1998) classification (Berg, 2009).

There are four main Boreal Baltic coastal wetland classifications: Tyler (1969), Rebassoo (1975), Paal (1998) and Burnside *et al*. (2007). The Paal (1998) classification offers the simplest classification. Paal (1998) highlighted three main plant community zones that exist in Baltic coastal wetlands (figure 2.2). These zones are: sublittoral (typically submersed), eulittoral (periodically submersed) and epilittoral (rarely inundated, only at extreme high water levels such as during storm surges) and are reflected by distinct plant communities. The simplistic plant community classification suggested by Paal (1998) groups plant communities which can be seen to be quite distinct in the field. The simplification of the Paal (1998) classification into three zones comes partly from the dominant thinking in coastal wetland ecology in Europe which assumes regular tidal inundation and is partly as a result of the large scale of his classification, which was designed to cover all plant communities throughout the whole of Estonia.

The classifications devised by Tyler (1969) and Rebassoo (1975) are also based on the principle of sublittoral, eulittoral and epilittoral zones although both of these classifications suggest several plant communities within each of these zones. The Tyler (1969) classification was developed in Swedish boreal Baltic coastal wetlands. It has been noted by previous authors that there are regional variations in the plant communities in boreal Baltic coastal wetlands (Tyler, 1969; Rebassoo, 1975), which limits the use of the Tyler (1969) classification for use outside Sweden. Rebassoo (1975) dealt with this problem for Estonian sea shore plant communities by developing his own classification in Estonia. However, this broad classification was developed for all plant communities that occur on the sea coasts of Estonia. This classification contains 14 main communities in coastal meadows, each of which is subdivided into a several regional varieties, many of which only occur on single islands (Rebassoo, 1975) (Appendix I). This makes the Rebassoo (1975) classification cumbersome to use. As mentioned previously the Paal (1998), Tyler (1969) and the Rebassoo (1975) classifications make considerable mention of zonation of the plant communities. However, this idea of zonation of the plant communities in boreal Baltic coastal wetlands is an oversimplification. Burnside *et al*. (2007) developed a vegetation classification specifically for Estonian coastal wetlands. The Burnside *et al*. (2007) classification was the first to make mention of the mosaic nature of the plant communities within the landscape.

Burnside *et al.* (2007) identified 11 different coastal plant communities for Estonian coastal wetlands and identified indicator species for the seven main plant communities, two of which exist at a higher elevation than Paal's (1998) epilittoral zone. The Burnside *et al.* (2007; 2008) studies also investigated edaphic differences between the plant communities. Burnside *et al* (2007) showed that grazing also had a strong influence on vegetation type but suggested that other factors such as edaphic factors and micro-topography determined wetland plant community.

Tyler (1971b) noted a distinct gradient in Estonian coastal wetland plant communities related to the hydrological and salinity gradients. Tyler (1971b) also identified 5 plant communities associated with the duration and frequency of inundation including a further community which is located inland of Paal's (1998) epilittoral zone. However, whilst both the Tyler (1971b) and the Burnside *et al*. (2007; 2008) studies noted elevation differences between plant communities no work has been done to evaluate and quantify this relationship more precisely in boreal Baltic coastal wetlands.

2.2.3 Coastal wetland vegetation

The 7 plant communities identified by Burnside *et al*. (2007) classify the vegetation in Estonian coastal wetlands as Clubrush Swamp (CS), Reed Swamp (RS), Lower Shore (LS), Upper Shore (US), Open Pioneer (OP), Tall Grass (TG), and Scrub and developing Woodland (SW). The CS and RS plant communities are almost continuously submerged and are predominantly composed of vegetation such as *Schoenoplectus lacustris*, *Bolboschoenus maritimus* (CS), and *Phragmites australis* (RS). These plant communities typically occur on the seaward edge of Estonian coastal wetlands although *P. australis* can be found encroaching into less frequently inundated areas following the cessation of management (Berg *et al*., 2011). The LS and US plant communities are characterised by *Glaux maritima*, *Juncus gerardii* and *Festuca rubra* (Burnside *et al*., 2007). A study by Berg (2009) showed that both the LS and US plant communities are less frequently inundated than the CS and RS plant communities. The OP plant community is the rarest of the plant communities described by Burnside *et al*. (2007). OP has sparse vegetation, typically consisting of halophytes such as *Suaeda maritima* and *Salicornia europaea* and typically occurs in the upper reaches of Estonian coastal wetlands. The TG and SW plant communities are more species rich than the CS, RS, LS and US plant communities. A study by Berg *et al*. (2011) found that over 27 months the TG plant community was not inundated once whilst over this same period the LS plant community was inundated four times for periods of up to several weeks. The SW plant community is typically found on the margins of Estonian coastal wetlands although some isolated patches occur in the middle of the wetland (Burnside *et al*., 2007).

The differences in salinity and hydrology throughout Estonian coastal wetlands affect plant species composition. There are few plant species that have a high tolerance to salt or inundation (Gray, 1992). In non halophyte plants soil salinity can cause injury, inhibit seed germination and limit vegetative and reproductive growth (Flowers, 2005). It can also alter plant morphology in a similar way to flooding and can lead to eventual plant death. In combination with flooding, salinity can cause a greater limitation on the plant growth and survival than either of these two factors alone. Plants have adopted a variety of ecological and physical adaptations to deal with these environmental stresses.

Figure 2.3: Species response to an environmental gradient. The upper graph shows a single species response to an environmental gradient. The lower graph shows different responses of differing species to a single environmental gradient. (Lee, 2006).

Plant survival strategies can affect inter and intraspecific relationships between plant species (Jefferies *et al*., 1979; Rorison *et al*., 1986). Each species has an upper and lower limit to tolerances of environmental factors (figure 2.3) such as light (Walters & Reich, 1996), pH (Grime & Lloyd, 1973), soil moisture (Werner & Platt, 1976), soil water table depth (Araya *et al*., 2010a), salinity (Greiner La Peyre *et al*., 2001), nutrients (Rorison, 1987) and disturbance (Verkaar, 1987). These tolerances limit a species' optimum range and this ultimately determines, along with competition, which environments a species will be present in. Previous authors have suggested that hydrology, salinity, nutrient availability, and management are the main limiting factors in Baltic coastal wetlands (Tyler, 1971(a); Tyler, 1971(c); Wallentinus, 1973; Rebassoo, 1975; Burnside *et al*., 2007). However, these previous authors have looked at all of these factors in isolation.

2.3 Factors affecting coastal wetlands

2.3.1 Coastal wetland management

Wetlands have been used by humans for thousands of years. Many coastal wetlands have historically been used for grazing and to a lesser extent for hay making (Roman *et al*., 1984; Tiner, 1993; Kiehl *et al*., 1996; Puurmann & Ratas, 1998). Baltic coastal wetlands are ancient features and previously had great economic importance as pasture and for hay production (Puurmann & Randla, 1999; Rannap *et al*., 2004). In Estonia between 1950 and 1991, with the installation of Soviet collective farming, herd sizes increased in many coastal wetlands often leading to over exploitation of the wetland resource (Puurman & Ratas 1998). Small-scale haymaking was reduced in these areas due to centralisation around the Kolkhoz (Collectivised farms) and mechanisation (Rannap *et al.*, 2004). Other coastal wetlands were abandoned causing them to revert to *P. australis* stands in the wetter areas and scrub woodland vegetation in the drier areas (Burnside *et al*., 2007).

Grazing management selectively removes some species and avoids others leaving low growing lateral spreading plants and halting succession to coarser plant species, increasing species richness (Jerling & Andersson, 1982; Bakker, 1985; Coughenour, 1985; Andresen *et al*., 1990; Jutila, 1997; Jutila, 1999). This allows less competitive species, slower growing perennials and species of smaller stature to survive (Benstead *et al.*, 1997; Joyce & Wade, 1998a). Mowing consists of the cutting and subsequent removal of vegetation for animal fodder. In the past mowing was an economically important activity, nowadays it is often used as an alternative to grazing management. Mowing effectively lowers sward height allowing low growing species to compete for light (Fynn *et al*., 2004).

Table 2.1: Change in Estonian coastal wetland management over time (Kaisel *et al.*, 2004).

Year	Scale of management
1957-1960	90% of coastal wetlands were grazed or mown
1982-1987	65% of coastal wetlands remained in use (mostly grazed)
1992-1995	32% of coastal wetlands remained in use
2001-2003	35% of coastal wetlands remained in use (minor recovery due to farmers being compensated for grazing/mowing coastal wetlands)

After independence in 1991 there was a significant decline in Estonian agriculture and consequently a decrease in the grazing and cutting (table 2.1) of the coastal wetlands, which led to large areas being overgrown by *P. australis*. This expansion was assisted, in part, by high nutrient levels due to eutrophication of the Baltic Sea. In recent years, a return to management in the form of grazing/mowing has limited biomass and returned some coastal wetlands to their former mosaic structure of plant communities resulting in an increase in species richness (Rannap *et al*., 2004). The effects of management on boreal Baltic coastal wetlands has been covered by a variety of authors (Wallentinus, 1973; Dijkema, 1990; Grace & Jutila, 1999; Jutila, 2001; Rannap *et al*., 2004; Berg, 2009; Berg, 2011) and management is essential in maintaining the open landscape typical to these wetlands. However, in managed boreal Baltic coastal wetlands it is necessary to determine which other factors influence the location of the plant communities.

2.3.2 Hydrology

Hydrology is a major controlling factor affecting the biological and physical characteristics of coastal wetlands (Mitsch & Gosselink, 2000). Small differences in water table levels have been shown to structure wetland plant communities in both field (Silvertown *et al*., 1999) and laboratory experiments (Araya *et al*., 2010b). The hydrological regime of coastal wetlands is controlled by several environmental factors including precipitation, extent and duration of flooding, topographic variability, ground water levels, soil type and evaporation rates (RSPB *et al.*, 1997). In tidal coastal wetlands, the tidal regime affects vegetation in a variety of ways: by controlling wave impact, flooding frequency and duration, and sediment accretion rates (Woodroffe, 2003). There are two recognised tidal systems affecting coastal wetlands (Orme, 1990): micro-tidal (less than 2m tidal range) and macro-tidal (over 2m tidal range). In tidal coastal wetland systems two vegetation zones are typically found: the low marsh and high marsh. The low marsh is flooded almost daily whereas the high marsh can experience at least 10 days of continuous exposure to the atmosphere (Mitsch & Gosselink, 2000). In micro-tidal systems salt and sea water input is more strongly influenced by atmospheric pressure changes and storm surges (Suursaar *et al*., 2003). Unlike tidally influenced changes in sea level, storm surges and changes in atmospheric pressure can increase or decrease sea level for periods lasting several days. A decrease of 1millibar (mb) in atmospheric pressure will raise sea level by 0.01m and wind driven storm surges can raise seas level by over 5m (Masselink, 2008). Micro-tidal coastal wetlands also typically have a flat relief with a low elevation above m.s.l. and large areas can be inundated during storm surges (Friedrichs & Perry, 2001).

In Estonian coastal wetlands, the tidal regime (amplitudes of M_2 and K_1 waves caused by the gravitational attraction of the Moon and variations in the declination of the Moon and Sun respectively) is negligible at <0.02m (Keruss & Sennikovs, 1999; Suursaar *et al*., 2001b) and hence atmospheric pressure and storm surges are the predominant factors that change sea level. Atmospheric pressure causes more regular variations in Baltic Sea level than storm surges (Finnish Meteorological Institute, 2011). Average yearly atmospheric pressure in the Baltic is 1013 millibars (mbars) and air pressure normally varies between 950 mbars and 1050 mbars over the year and hence the expected variation in sea level due to air pressure is between +0.63m and -0.37m (Swedish Meteorological and Hydrological Institute, 2010). However, in Estonian coastal wetlands as with other coastal wetlands greater and more rapid changes in sea level occur due to combined low pressure and storm surges (Swedish Meteorological and Hydrological Institute, 2010). Any rise in water levels is exacerbated within bays and inlets, where most Estonian coastal wetlands occur, due to a funnelling effect (Leatherman, 1984; Suursaar *et al*., 2003). In the micro-tidal water body of the Baltic Sea a record sea level occurred during the 2005 winter storm, Gudrun, of 2.75m in Parnu Bay, Estonia (Suursaar *et al*., 2006). The low relief of Estonian coastal wetlands, typically <2m above mean sea level, means that all the plant communities were inundated. It must also be noted that most years there is little variation in sea level during the winter due to ice cover of the Baltic Sea.

In the majority of cases coastal wetland hydrology is dominated by sea water (Mitsch & Gosselink, 2000). However, precipitation and groundwater can also influence the water regime. This is particularly important for vegetation during periods of low sea level. Seasonal precipitation can, in some coastal wetlands, be the main controlling hydrological factor although this occurs in areas with very high seasonal rainfall (Bildstein *et al*., 1991). However, this is not likely to be the case in Estonia which had a mean annual precipitation of 742mm/yr for the period 2007-2010, a large amount of which falls as snow (Estonian Meteorological and Hydrological Institute, 2011). Whilst snow melt provides a considerable input of freshwater to all Baltic coastal wetlands, this typically occurs, in west Estonia, towards the end of March (Ratas & Nilson, 1997). The vegetation growing season does not start until the middle of May and hence the influx of freshwater to the wetland system has little effect with respect to vegetation (Ratas & Nilson, 1997).

Groundwater has a variety of influences on hydrology in coastal wetlands as it can maintain soil moisture levels and act as a reservoir to supply water to vegetation during periods without inundation (Nuttle & Harvey, 1995). This is particularly important in Baltic coastal wetlands where sea water levels can be low for extended periods, such as occur when high pressure systems sit over the Baltic Sea. Ground water can also absorb excess surface water in periods of inundation although interactions between ground water and wetland biota are dependent on soil permeability (White, 2006). The depth of the water table is an important factor in determining vegetation species composition. Anoxic soil conditions in the root zone prevent the abundance of non hydrophytic species (Richardson & Vepraskas, 2001). Tyler (1971b) in a study of hydrology in Swedish Baltic coastal wetlands suggested that ground water is driven by sea water levels and suggested that the location of the plant community zones within the wetland were determined by sea level. This theory put forward by Tyler (1971b) is an oversimplification of the hydrological system in Baltic coastal wetlands, as this literature review shows there are a variety of factors that affect the hydrology of coastal wetlands. However, a variety of studies have made this same suggestion with regards to Baltic coastal wetlands (Wallentinus, 1973; Rebassoo, 1975; Jerling, 1981; Puurmann & Ratas, 1998; Puurmann & Randla, 1999; Burnside *et al*., 2007; Berg *et al*. 2011) and whilst it may be an oversimplification of the hydrology, it appears to partly explain the location of the plant communities. Soils in Baltic coastal wetlands are permeable, being composed mainly of silt and sand (Puurmann & Ratas, 1998). However, as previous studies have shown soil texture, whilst an important component, is not the sole factor influencing soil hydrology (Heiskanen, 1993; Tamminen, 1998). Soil aeration can also influence soil hydrology, well aerated soils are often more permeable than less well aerated soils due to an increase in both matrix and macropore flow (Heiskanen *et al*., 2007; Mäkitalo *et al*., 2008; Unguraşu & Ungureanu, 2011). Berg *et al*. (2011) suggested that sea water penetrates the interior of Estonian coastal wetlands through both overland flow and throughflow. This suggests a mechanism by which sea water could influence the hydrology of the different plant communities which are located in a mosaic rather than zonal structure. The Tyler (1971b) study investigated the relationship between hydrology and plant communities in Baltic coastal wetlands and while she suggested that this was related to elevation above mean sea level there was no quantification of the relationship between elevation and plant community type. A pilot study was conducted by Burnside *et al*. (2008) in order determine whether there were any elevation differences between any of the plant communities in Estonian coastal wetlands. The Burnside *et al*. (2008) study used a limited amount of data points to record elevation and the data points were recorded using a transect method and hence only captured elevation data in adjacent plant community patches. Also, the Burnside *et al*. (2008) study neglected to record elevation data in the Reed Swamp plant community which covers extensive areas and is rarely absent in Estonian coastal wetlands. However, the results of the Burnside *et al*. (2008) study suggest that there is a relationship between elevation, hydrology and plant community type.

2.3.3 Salinity

Salinity is an important factor influencing coastal wetland plant community location. Salt levels are often linked to hydrology and typically controlled by the frequency and duration of sea water flooding, precipitation/evaporation, soil permeability, vegetation, water table levels and freshwater inflow (Mitsch & Gosselink, 2000). In Atlantic salt marshes it has been shown that soils that are regularly flooded tend to have the same salinity as the sea water, whereas further inland there is a trend of increasing salt levels up to the elevation which is reached by the highest spring tides, above this elevation salinity rapidly drops (Wiegert & Freeman, 1990). This has been suggested to be related to frequent tidal inundation maintaining salinity at or below that of sea strength in the lowest elevations. As inundation frequency decreases in higher elevation zones, in poorly drained soils evaporation causes an increase in the salinity of the water deposited (Wiegert & Freeman, 1990). However, the Baltic Sea is a shallow semi enclosed brackish body of water with an average salinity level of 10.6-11.4 mS/cm on the west coast of Estonia (Puurmann & Ratas, 1998), much lower than that of the Atlantic average, which is 53mS/cm (Krom, 2008). In addition the micro-tidal regime (<0.02m tidal range), of the eastern Baltic Sea means that storm surges and atmospheric pressure changes are the main causes of the inland movement of saline water (Tyler, 1971b) and not the more regular and often more frequent tidal inundation as is found in tidal salt marshes. Although the salinity of the Baltic Sea is much lower than that of the Atlantic and inundation periods are less regular and frequent, the gradient of salinity across Baltic coastal wetlands has been found to be similar to that found by Wiegert & Freeman (1990) in salt marshes (Tyler, 1971b; Puurmann & Ratas, 1998). Although as a result of the lower salinity of the Baltic Sea the salinity of the soils was also found to be lower in Baltic coastal wetlands than Atlantic salt marshes and hence this in turn provides a less preferential habitat for halophytes. A study by Burnside *et al*. (2008) in Estonian coastal wetlands found quite different results from both the Tyler (1971b) and the Puurmann & Ratas (1998) studies with regards to elevation and vegetation and the salinity gradient although this is likely to be due to prolonged low water levels and a very hot period which occurred during this study (Joyce, *pers. comm.*, 2009). Burnside *et al*. (2008) recorded considerably higher salinity levels in Open Pioneer patches, 5.5 mS/cm, than in all the other plant communities, and more than double that of the next highest, the typically inundated Clubrush Swamp community. The Open Pioneer plant community has been described previously by other authors as salt patches (Wallentinus, 1973) and *Salicornietum europaeae* (Rebassoo, 1975) with regards to the vegetation. However, other than the pilot study by Burnside *et al*. (2008) no attempt has been made to describe the environmental variables that occur within this plant community.

Silt and clay soils can have low permeability and therefore also tend to retain more salt than more permeable substrates such as sand (Wang *et al*., 2007a). The high sand content that occurs in Estonian coastal wetland soils could explain why they have considerably lower salinity than the surrounding brackish Baltic Sea (Puurman & Randla, 1998). However, it does not explain the high salinity of the Open Pioneer patches. In areas with high salinity, vegetation tends to be quite sparse as few species can tolerate saline conditions for extended periods (Grime *et al*., 1988). In turn decreased vegetation cover can increase evaporation rates and hence this can rapidly increase soil salinity. The Open Pioneer patches are typified by sparse halophytic vegetation (Wallentinus, 1973; Rebassoo, 1975; Burnside *et al*., 2007). The sparse vegetation and resultant high evaporation rates could, in part, explain why the salinity is so much higher in these patches, although this does not take into account the much lower transpiration rates that occur with sparse vegetation (Michener *et al*., 1997). What is lacking in the literature is a holistic assessment of the main environmental factors determining the plant communities in Estonian coastal wetlands.

2.3.4 Nutrient availability

Nutrients are a major factor determining productivity and species richness of plant communities (Grime, 1979; Fitter, 1997; Foster & Gross, 1998; Critchley *et al*., 2002; Taiz & Zeiger, 2002). Nutrients are carried into wetlands by precipitation, flooding, groundwater and surface water, and can be removed from the system by water outflow. The frequency, depth and duration of the hydrological inputs/outputs affect the biochemistry of soils (Mitsch & Gosselink, 2000), which is a major factor in determining vegetation composition.

Coastal wetlands with silt or clay substrates retain incoming nutrients better than wetlands with sandy substrates partly due to the lower surface tension capillary forces. Smart & Barko (1978) found that plant growth rates were of an order of magnitude greater on clay than on sand for four different salt marsh plants, *Spartina alterniflora*, *S. foliosa*, *S. patens* and *Distichilis spicata*, due to greater nutrient retention in the soil. Additionally nutrient levels in the soil can be increased through the breakdown of plant litter (Collins & Kuehl, 2001) although the amount of nutrients in the soil from this source depends on a variety of factors such as soil hydrology (Tobias & Neubauer, 2009), plant type (Moore & Bellamy, 1974), temperature (Tate, 1987), and pH (Neue, 1985; Oades, 1988). Therefore, there are a variety of factors that affect nutrient levels in coastal wetland soils and since these factors are likely to vary within a site, nutrient levels are also likely to vary (Keddy *et al*., 2000). Several studies have shown that indeed nutrient levels in wetlands do vary between distinct vegetation zones (Levine *et al*., 1998; Emery, *et al*., 2001; Bertness *et al*., 2001; Silliman & Bertness, 2004). However, no studies have been conducted comparing nutrient levels in different plant communities in Baltic coastal wetlands.

In the last decades of the Soviet Union (1970's and 1980's) an increase in the production/use of sewage discharge, fertilizers and the increased farm sizes led to large scale eutrophication of the Baltic region (Larsson *et al*., 1985). This in turn caused an increase in the inflow of nutrients to coastal areas leading to greater biomass in Baltic coastal wetlands in the form of large stands of *Phragmites australis.* These *P. australis* stands expanded into areas formerly covered by wet grassland species (Elmgren, 1989). *P. australis* expansion into other plant communities has been shown to decrease biodiversity in Estonian coastal wetlands (Berg *et al*., 2011) although management can halt the expansion (Rannap *et al*., 2004; Berg, 2009).

2.3.5 Accretion rates and storm events

A major factor affecting the development of wetlands is sediment accretion and hence any future modelling of wetland development must include some estimation of accretion rates. Accretion rates in wetlands are dependent on sediment supply. There are two sources of sediment in wetlands, allochthonous and autochthonous (Neubauer, 2008). In coastal wetlands allochthonous sources are usually deposited as suspended sediment in sea water during high tide or atmospherically induced inundation. Autochthonous sources are derived from the accumulation of animal or plant detritus (Orme, 1990). Autochthonous sources can be very high particularly in frequently inundated areas where anoxic conditions limit decomposition of organic matter, although they are usually not as important as allochthonous sources in coastal wetlands (Mitsch & Gosselink, 2000).

Coastal wetlands are usually considered to be depositional areas and wetland growth is often led by an increase in allochthonous sediment (Allen & Pye, 1992). Factors affecting deposition and hence wetland accretion are available sediment type, tides/inundation, storm events, vegetation and relative sea level (Allen & Pye, 1992; Kearney *et al*., 1994; Harff & Meyer, 2011). In coastal wetlands, sediment accretion raises the level of the land and hence alters the hydrological regime by decreasing the duration and depth of inundation, which in turn influences vegetation (Scott Warren & Niering, 1993). Vegetation has also been found to increase accretion rates by trapping sediment from allochthonous sources (Boorman *et al*., 1998; D'Alpaos *et al*., 2009). Ranwell (1972) found that, in areas with *Spartina anglica*, sediment deposition increased as the plant impedes the movement of water and stabilises sediments. However, Brown *et al*. (1998) found that

Spartina anglica had little or no effect in promoting marsh accretion, yet they did find that it had a significant effect as a sediment stabiliser during periods of erosion. However, a study by Feagin *et al*. (2009) has shown that sediment type and not vegetation is the primary variable influencing coastal wetland edge erosion. A review by Gedan *et al*. (2011) has attempted to explain the complex relationship between sediment stabilisation and vegetation in coastal wetlands. This review found that in most cases vegetation does stabilise soils in coastal wetlands but that gaps remain in knowledge about context dependent aspects of shoreline protection due to the complexity of coastal wetland systems

In macro-tidal coastal wetlands, sediment deposition usually occurs during slack high tide where water movement is limited and sediment can settle out of suspension (Davidson-Arnott *et al.*, 2002). In micro-tidal coastal wetlands, like those found around the Baltic Sea, deposition often occurs during high water levels caused by changes in atmospheric pressure (Ratas & Nilson, 1997). The flood deposits consist of both coarse sediments and finer particles which contribute to wetland accretion. Accretion rates vary depending on the frequency and duration of flooding, available suspended sediment, vegetation type and sediment type.

Using radiometric dating methods, Cundy & Croudace (1995a) found that a salt marsh was accreting in the micro tidal area of Poole Harbour, England, at a rate of between 4.2mm and 4.7mm/year. Roman *et al.* (1997) found accretion rates in the micro tidal area of the Nauset Inlet, New England, USA, to be on average between 2.6-4.2mm/year but were found to be as high as 24mm in some years that included several major coastal storms. Storms can also be erosional in character as is shown by the Baltic winter storm, Gudrun, in 2005 which caused extensive coastal erosion throughout the Baltic (Haanpaa *et al.*, 2007). Saaramaa Island and the Parnu area were the worst affected areas in Estonia and some unprotected coastlines receded by tens of metres inland (Kont *et al.*, 2006). However, no work has been undertaken to assess sediment deposition during the same periods on sheltered bays and inlets such as where Estonian coastal wetlands occur. The northern and eastern Baltic Sea experiences a negligibly small tidal range. However, as mentioned earlier, Baltic coastal wetlands do experience inundation by the sea due to low atmospheric pressure and storm surges and hence sediment accretion from allochthonous sources is likely to be factor in Baltic coastal wetland development although there are no data available.

Several studies have suggested that sea level rise will have a positive effect on sediment accretion and can even be the driving force behind coastal wetland development (Gehrels & Leatherman, 1989; Scott Warren & Niering, 1993). Reed (1995) agrees that sea level rise can increase allochthonous sediment accretion but states that organic matter accumulation is likely to decrease during periods of extended flooding due to lower net plant production. A study by Morris *et al*., (2002) also agrees that sea level rise increases sediment accretion. However, Morris *et al*. (2002) state that beyond the optimal rate of sea level rise with respect to sediment accretion, sea level rise will in fact lead to wetland loss due to the plant community being unable to sustain an elevation that is within its tolerance range. A variety of other authors have suggested that this provides a long term threat to a variety of coastal wetlands such as estuarine salt marsh (Clifton & Hamilton, 1982; Cundy & Croudace, 1995b; Cundy & Croudace, 1996), coastal salt marsh (Callaway *et. al*. 1996) and subsiding salt marsh (Roman *et al*., 1997).

Most studies suggest that coastal wetland development in the northern and eastern Baltic area is predominantly controlled by post glacial isostatic uplift (Tyler, 1969; Puurmann & Ratas, 1998; Johansson *et al*., 2004; Kont *et al*., 2003; Kont *et al*., 2007). The land is rising at a rate of 2mm/year in west Estonia up to 8mm/ year in the north of the Bay of Bothnia (Vallner *et al*., 1988; Ekman & Mäkinen, 1996; Eronen *et al*., 2001), although a more recent study using long term GPS data suggests that these rates are slightly higher in southern areas and lower in northern areas of the Baltic region (Lidberg *et al*., 2007). The studies that have investigated the effects of sea level rise on Baltic coastal wetlands have neglected sediment accretion rates in their calculations and hence are likely to be inaccurate.

If an accurate assessment of sea level rise is to be made in Baltic coastal wetlands, it is important to make some assessment of the rate of sediment accretion. There are several methods for analysing sediment accretion rates: radiometric dating techniques (Appleby & Oldfield, 1978), laying marker horizons such as brick dust or aluminium filings (Richard, 1978), or measuring from graded stakes (Ranwell, 1964). The methodologies using marker horizons or measuring from graded stakes are useful for long term studies over several years although they are subject to errors such as washing out of marker horizons and atypical sediment accretion in the vicinity of the stake, caused by the stake itself (Armentano & Woodwell, 1975). However, radionuclide dating is free from such errors and can calculate accretion rates retrospectively over much longer periods. The measurement of radioactive decay provides the basis for a wide range of dating techniques within the sciences. Within geomorphology this has been used extensively to date older sediments (thousands of years) using ^{14}C (Shennan *et al*., 1995) however, coastal sedimentary depositional processes occur over considerably shorter time scales requiring a different methodology. For more recent sedimentary processes, radioisotopes with shorter half lives than ^{14}C are required. The naturally occurring radioisotope ^{210}Pb, which is part of the ^{238}U decay series has a half life of 22.26 years (Appleby & Oldfield, 1978). This much shorter half-life makes this radioisotope ideal for conducting analysis into recent sedimentary processes. ^{210}Pb has been extensively used as a dating method for floodplain, coastal, lake and marine sediments (Chanton *et al*., 1983; Crusius & Anderson, 1991; Cundy & Croudace, 1996; He & Walling, 1996; Plater *et al*., 1999; Craft & Casey, 2000; Cundy *et al*., 2003; Teasdale *et al*., 2011).

^{210}Pb is one of many radionuclides that constitute the ^{238}U decay series. The decay of the parent isotope ^{226}Ra (half-life = 1600 years), which is present in bedrock via a series of relatively short lived daughter isotopes, results in the formation of ^{210}Pb in secular equilibrium with ^{226}Ra. Therefore within this closed system ^{210}Pb is referred to as being the supported component ($^{210}Pb_{supported}$). Disequilibrium in the system between ^{210}Pb and ^{226}Ra occurs in open systems as a result of the loss and subsequent decay of the intermediate gaseous isotope ^{222}Rn (half-life 3.8days) from the solid matrix either via diffusion through the soils/rocks or due to recoil on the ejection of an alpha particle. One of the remaining daughter isotopes of the decay that stays in the system is ^{214}Pb, which can be used as a proxy to measure the $^{210}Pb_{supported}$, hence that which is derived from autochthonous sources. Following diffusion into the atmosphere ^{222}Rn decays to ^{210}Pb via a series of decay reactions. The ^{210}Pb is then rapidly adsorbed onto atmospheric aerosols and eventually redeposited onto the surface of the Earth. Average residence times for atmospherically derived ^{210}Pb in the northern hemisphere range from between 5-10 days (Wise, 1980). Several studies have shown an inverse correlation between ^{210}Pb concentration and average rainfall (Peirson *et al*., 1966; Fujinami, 1996; Beks *et al*., 1998). This can affect the supply of ^{210}Pb to the sediment and hence require a different dating model for sediment geochronology.

The determination of unsupported (derived from an allochthonous source) gamma ^{210}Pb activity for each horizontal sediment profile slice is commonly achieved by using an ultra low background alpha or gamma ray spectrometer to record the total activity of ^{210}Pb and ^{214}Pb. Zaborska *et al*. (2007), in a comparison of both the alpha and gamma spectrometry techniques used in ^{210}Pb geochronology, found that there were no benefits between using either alpha or gamma spectrometry and therefore either can be used. Following the determination of $^{210}Pb_{excess}$, it is necessary to use a calculation to determine accretion rates. Three commonly used methods are CF:CS (Constant Flux: Constant Sedimentation), CIC (Constant Initial Concentration) and CRS (Constant Rate of Supply). The CIC methodology is compromised by variable accretion rates (Noller, 2000), as are likely to occur in Baltic coastal wetlands due to irregular inundation.

The principal basis for the CF:CS model is the measurement of mean sediment accretion over time. This is calculated using a logarithmic plot of the exponential radioactive decay curve for $^{210}Pb_{excess}$ derived from a constant supply of $^{210}Pb_{excess}$ to a steadily accreting sediment profile (Appleby & Oldfield, 1992). Mean sediment accretion is estimated using the best-fit straight line equation to describe radioactive decay with depth (Yeager *et al*., 2004).

In many depositional wetlands the rate of sediment accretion changes over time (Mitsch & Gosselink, 2000). The CRS model can be applied to take into account differing rates of sediment accretion over time (Cundy & Croudace, 1996). The principal assumption of the CRS model is that of a constant rate of supply of ^{210}Pb to the sediment surface over time. This results in the initial $^{210}Pb_{excess}$ activity having an inverse relationship with sediment accretion rates as a consequence of $^{210}Pb_{excess}$ being diluted by increased sediment accretion. The CRS model calculates sediment age at a given depth by comparing the ratio of $^{210}Pb_{excess}$ (the unsupported component) below a certain depth interval to the $^{210}Pb_{supported}$ (background activity) throughout the sediment core as derived from the activity of ^{214}Pb, which is in secular equilibrium with $^{210}Pb_{supported}$ (Appleby & Oldfield, 1992). Calculated sediment ages at discrete depths down the core provide data for the age/depth curves. From these age/depth curves sediment accretion rates can be estimated at discrete periods in the past. This methodology has been used in a variety of studies where variable sediment accretion rates over time have been recorded (Allen *et al*., 1993; Cundy & Croudace, 1996; He & Walling, 1996; Dyer *et al*., 2002; Teasdale *et al*., 2011).

Previous authors have shown that both the CF:CS and CRS methods are able to accurately calculate sediment accretion rates (Chanton *et al*., 1983; Crusius & Anderson, 1991; Cundy & Croudace, 1996; He & Walling, 1996). However, Appleby & Oldfield (1992) suggest that independent verification of sediment accretion rates must be undertaken in order to prove their validity. ^{137}Cs is a useful dating tool for independent verification of the ^{210}Pb geochronological methods (Kirchner & Ehlers, 1998). Caesium-137 (^{137}Cs) is an artificial radio-isotope produced solely from nuclear fission, weapons generation processes and discharges from accidents at nuclear power generation facilities. Large scale introduction of ^{137}Cs to the environment occurred in the early 1950's to 1963 when a variety of nations conducted testing of the first above ground high yield thermonuclear weapons. This culminated in a marked maxima in weapons derived atmospheric fallout immediately prior to the above ground Nuclear Weapons Test Ban Treaty between the UK, USA and USSR in October, 1963 (Kirchner & Ehlers, 1998). Further atmospherically derived ^{137}Cs was deposited over large areas of the northern hemisphere in 1986 as a result of the Chernobyl nuclear power reactor incident. The main peaks in ^{137}Cs to sediments in north east Estonia are from the 1986 Chernobyl incident, with secondary peaks in activity related to the pre 1963 nuclear weapons testing (Realo *et al*. 1995). The known dates of peak ^{137}Cs deposition can be used to date specific sediment horizons and therefore independently verify ^{210}Pb sediment accretion estimates.

2.3.6 Micro-topography

There have been a wide range of biogeographical studies showing the relationship between varying topographies and flora/fauna. These have their origins in the studies of Humboldt and Candolle in the early 19th century, followed up by C. Hart Merriam in the late 19th century. Merriam's (1890) specific field studies were conducted in south western North America but set the context a framework for a number of later studies in a variety of ecosystems. Merriam (1890) showed that elevation changes in vegetation type are generally equivalent to latitudinal vegetation changes (figure 2.4).

Figure 2.4: Merriam's life zone concept diagram (Merriam, 1890).

Changes in topography have been shown to increase biodiversity through an increase in environmental heterogeneity (MacArthur & MacArthur 1961; Huston, 1994).

Zedler & Zedler (1969) showed that changes in plant communities occurred even with fine scale changes in topography of up to 1.5m. There are now a variety of studies that have looked at small scale changes in topography or micro-topography and hence a variety of definitions. This study will use the definition suggested by Moser *et al*. (2007) which states that micro-topography "describes soil surface variation within an elevation range from roughly one centimetre to as much as one meter".

A variety of studies have shown that micro-topography affects wetland plant community composition by influencing a variety of other environmental factors such as: soil moisture (Moran *et al.*, 2008), water table depth (Prach, 1992; Moser *et al*., 2007), soil nutrients (Kojima, 1994; Moser *et al*., 2008), sediment accretion (Werner & Zedler, 2002), substrate variability (Cardinale *et al.*, 2002), resource specialisation (Vivian-Smith, 1997), and increased surface area (Peach & Zedler, 2006). A variety of studies have also shown that micro-topographic diversity leads to an increase in plant biodiversity due to the creation of a variety of microhabitats (Vivian-Smith, 1997; Cardinale *et al.*, 2002; Werner & Zedler, 2002; Morzaria-Luna *et al*., 2004; Peach & Zedler, 2006).

Vivian-Smith (1997) found that differences in elevations of 0.01-0.03m had an effect on wetland plant species composition. In her study it was suggested that micro-topography not only affected the spatial pattern of inundation but was also responsible for temporal variations in water levels, with increased inundation periods in hollows and decreased inundation periods on hummocks. Toogood *et al*. (2008) have shown that vegetation responses to alterations in the hydrological regime can be fairly rapid in wet grasslands, as a decrease in wetness led to an increase in species diversity, an increase in the abundance of competitive species and a decrease in typical wetland plants. Roy *et al.* (1999) showed that micro-topography had an effect on the survival of black spruce seedlings. The seeds were more likely to germinate and survive on hummocks than in hollows. This was shown to be due to less saturated, better aerated, and warmer substrate on the hummocks. Roy *et al.* (1999) also showed that levels of foliar nitrogen (N) were inadequate for seedling germination in depressions whereas on hummocks, levels were sufficient and caused no limitations for seedlings in forested wetlands and hence that micro-topography can also influence nutrient levels. Martin *et al.* (2007) found that micro-topographical changes were sufficient to alter plant species composition in the Pampa Ondulada floodplain in Argentina and suggested this was due to poor drainage of the clay rich soils which in turn affected the hydrology. It has also been suggested that micro topography can increase water retention as water is stored in micro depressions (Moser *et al.*, 2007). Whilst there have been many studies on the influence of micro-topography on plant communities in coastal salt marshes, estuaries and floodplain grasslands there has been only one pilot study, by Burnside *et al*. (2008), investigating this relationship in Baltic coastal wetlands.

The results of the Burnside *et al*. (2008) study suggested that the wetland plant communities were influenced by micro-topography with particular reference to the elevation above m.s.l.. However, the study used a belt transect method, which provided elevation and plant community data for only adjacent plant community patches, and covered a limited number of adjacent plant communities that occur in Estonian coastal wetlands. The majority of the aforementioned studies regarding micro-topography have suggested that while a variety of factors influence plant communities in wetlands, micro-topography due to its relationship with hydrology is the main influence (Prach, 1992; Vivian-Smith, 1997; Zedler *et al*., 1999; Cardinale *et al.*, 2002; Werner & Zedler, 2002; Morzaria-Luna *et al*., 2004; Peach & Zedler, 2006; Martin *et al*., 2007; Moser *et al*., 2007; Moran *et al.*, 2008). Therefore micro-topography or elevation could provide a single easily measurable factor which can be used to predict plant community type. In order to use micro-topography to predict plant community type it is necessary to quantify the relationship between micro-topography, edaphic variables and plant community and this quantification is thus far absent from previous studies. This study addresses this issue.

2.3.7 Sea level rise and associated factors derived from climate change

It has been widely accepted by the scientific community that global climate is undergoing change (IPCC, 2007; McKitrick & Michaels, 2007; Santer *et al*., 2007; Willet *et al*., 2007; Zhang *et al*., 2007; Lockwood & Froehlich, 2008). Climate change is predicted to have its greatest effect in high northern latitudes (IPCC, 2007). According to the Intergovernmental Panel for Climate Change (IPCC) (2007) 11 of the years between 1995 and 2006 were among the 12 warmest for global surface temperatures since records began in 1850. The greatest temperature increases have been recorded in the high northern latitudes. This has been particularly evident in Arctic areas where average temperatures have risen at nearly double the global average rate of 0.3ºC/decade. A rise in average global temperatures has been suggested by a variety of authors to lead to several other environmental changes such as: rising global sea level (IPCC, 2007; Rahmstorf, 2007; Horton *et al*., 2008; Pfeffer *et al*. 2008; Vermeer & Rahmstorf, 2009; Grinsted *et al*., 2010; Jevrejeva *et al*., 2010) changes in average precipitation (IPCC, 2007; Sillmann & Roeckner, 2008; Sheffield & Wood 2008; Gorman & Schneider, 2009) and an increase in the frequency, strength, and duration of severe weather events (i.e. increased storminess) (Bengtsson *et al*., 2007; IPCC, 2007; Emanuel *et al*., 2008; Füssel, 2008).

It has been suggested by several authors that sea level rise and increased storminess will affect the distribution of ecological communities within coastal wetlands through increased inundation periods and increased sediment accretion or erosion respectively (Saenger, 2006; Hopkinson *et al*., 2008; Charles & Dukes, 2009; Kirwan & Temmerman, 2009; Nicholls & Cazenave, 2010; Day *et al*., 2011). A variety of previous studies have shown that the relationships between environmental gradients, such as inundation; substrate; salinity; elevation above mean sea level, and ecological communities are of sufficient statistical significance that they may be used as statistical indicators of sea level (Whiting & McIntire, 1985; Palmer & Abbott, 1986; Nelson & Kashima, 1993). Further to this a variety of authors have used this relationship to assess past sea levels using microfossils such as diatoms (Zong & Horton, 1999; Szkornik *et al*., 2006), foraminifera (Gehrels & Newman, 2004; Gehrels *et al*., 2001; Leorri *et al*., 2008) and testate amoebae (Gehrels *et al*., 2001; Charman *et al*., 2002) using an ecological transfer function method, although these methods are subject to errors particularly over shorter time periods and in micro-topographical areas such as are used in this study (Sawai *et al*., 2004).

A few studies made an attempt at quantifying the predicted change in the area and distribution of plant communities within coastal wetlands linked to sea level rise (Chust *et al*., 2008; Poulter & Halpin, 2008; Moeslund *et al*., 2010; Geselbracht *et al*., 2011). All of the aforementioned studies used the relationship between elevation above mean sea level and plant community to map the present day plant communities using remotely sensed elevation data. However, the remotely sensed elevation data used in those studies can give a false elevation reading in vegetation if not corrected (Sadro *et al*., 2007), this will be discussed further in chapter 5. The elevation inaccuracies present in model simulations of the impacts of sea level rise on plant communities in coastal wetlands can cause inaccuracies in plant community modelling. The inaccuracies are likely to be much greater in areas with lower relief. A comparison of the studies by Chust *et al*. (2008), Poulter & Haplin (2008), Moeslund *et al*. (2010), and Geselbracht *et al*. (2011) shows that only the last study took into account sediment accretion in the model. This factor can substantially influence future local sea level and wetland development as discussed in detail within the sediment accretion section of this chapter.

There has been very little work undertaken regarding the effects of sea level rise on plant communities in Baltic coastal wetlands. In Estonia, a study by Kont *et al.* (2008) used an arbitrary sea level rise figure of 1.0m over the next century (or 10mm/year), although this is not corroborated by IPCC (2007) estimates, which are no higher than 0.59m over the next century. According to preliminary estimates by Kont *et al.* (2008) using this 10mm/year sea level rise, all reed beds and 80% of the coastal wet grasslands of Hiiumaa (2nd largest island of Estonia situated in the northwest of the country) will be threatened by sea level rise. The Kont *et al*. (2008) model is based on an arbitrary sea level rise figure and does not include sediment accretion, and employs coarse elevation data. Therefore the results of the Kont *et al*. (2008) study are likely to be of limited use. In order to accurately predict the effects of sea level rise on plant communities in Estonian coastal wetlands, a more robust methodology must be developed with improved elevation accuracy of the plant community model and the inclusion of sediment accretion data.

2.4 Ecological studies and geomatics

2.4.1 Vegetation studies

Plant community classification is a well established aspect of ecology (Tansley, 1920; Mueller-Dombois & Ellenberg, 1974; Zonneveld, 1983; Crawley, 1997; Ewald, 2003). Vegetation can be used as a bio-indicator that can highlight environmental gradients (Ellenberg, 1979, Diekmann, 2003; Kollmann & Fischer 2003; Piernik, 2003) such as light (canopy species), hydrology (Mulamoottil *et al*., 1996), pH (soil or water), nitrogen (a general indicator of soil fertility) and salinity (Mitloehner & Koepp, 2007). Plant communities can also be used as an indicator of climatic variables due to the occurrence of similar biological spectra in different regions (Raunkiaer, 1937), and can also be used to elucidate management status/ disturbance/ abandonment (Burnside *et al*., 2007). Plants are often used as bio-indicators due to the fact that they are relatively simple to assess and can give a good general overview of the ecosystem. Vegetation can also be used to indicate micro-topographical changes in the landscape (Vivian-Smith, 1997; Peach & Zedler, 2006; Moser *et al*., 2007) and conversely micro-topography and other environmental variables can be used to predict plant communities. Data concerning the location and extent of plant communities over large scales is often costly and time consuming to acquire. This problem has been in part addressed through the increased use of remotely sensed data and Geographic Information Systems, both of which are now used in a variety of disciplines including meteorology, climate studies, geology and ecology (Jensen, 2007).

2.4.2 Geographic Information Systems and remote sensing

A Geographic Information System (GIS) is a software system for storing, analysing and displaying spatial data from the real world (Longley *et al*., 2010). With the advent and rapid development of GIS software and the large amount of remotely sensed data available, these tools have been increasingly used for predictive plant community mapping (Burnside & Waite, 2011). Remotely sensed data has a range of different definitions, the one used in this study includes both passive and active data recorded from a platform (i.e. helicopter, fixed wing aircraft, satellite). There are a range of studies that use passive multispectral remotely sensed data to identify plant communities (Townsend & Walsh, 2001; Brown *et al*., 2006; Balzarolo *et al*., 2009; Berni *et al*., 2009; Hamada *et al*., 2011). These studies used a methodology based on identifying specific reflectance values in different wavelengths of distinct vegetation by performing some form of classification either unsupervised or supervised (Jones & Vaughn, 2010). The unsupervised classification classes pixels according to their similarity in feature distance using a variety of different algorithms. However a major drawback to this methodology is that the classes do not necessarily relate to different plant community classes on the ground, although this methodology does provide a good overview of spectral differences over the whole dataset (Jones & Vaughn, 2010). Another form of classification is the supervised classification which uses training sample areas to direct the classification process. Training sample areas can relate to plant communities or habitat data and are used to classify across the image (Hamada *et al*., 2011). These training pixels can be used to provide an accurate prediction of the location of different plant communities. Congalton (1991) compared supervised and unsupervised classification methods and suggested that while the supervised method more accurately identified land use the methodology provided a large amount of unclassified data, a drawback that is not present in the unsupervised classification method. However, if the full suite of plant communities can be identified in the field and all training sample areas delimited, this is likely to reduce the amount of unclassified data. The use of multispectral data, in conjunction with GIS, has been shown to be very good at mapping existing plant communities (Rutchley & Vilcheck, 1994; Aschbacher *et al*., 1995; Harvey & Hill, 2001; Tucker *et al*., 2005). However, predictive vegetation modelling often relies on other data sources, such as LiDAR data (Chust *et al*., 2008; Moeslund *et al*., 2010).

2.4.3 Use of LiDAR for plant community mapping

Light Detection and Ranging (LiDAR) data are collected by laser transmitters and receivers which compute distances by measuring the travel time of light from the transmitter to the ground and back to the receiver. LiDAR devices are often mounted on aircraft and used in conjunction with kinematic differential GPS (dGPS) systems and inertial measurement units (IMU) mounted on aircraft that accurately measure the position, speed, rate of acceleration, pitch roll and yaw of the aircraft. The dGPS and IMU information with the LiDAR can then provide detailed and accurate ground elevation data over large areas very rapidly (Adams & Chandler, 2002) (figure 2.5).

Accurate elevation data can be captured using conventional dGPS rover units in field surveys. However, field surveys using dGPS are time-consuming and expensive on a per point basis (Jones *et al*., 2007). Many studies that require detailed elevation data for large areas use LiDAR data due to its low costs (for pre-collected data) and high availability (Hodgson & Bresnahan, 2004; Hopkinson, *et al*., 2011; Moeslund *et al*., 2011; Mukoyama, 2011; Tarquini & Favalli, 2011).

LiDAR elevation data have been used in a variety of studies to accurately measure topography (Adams & Chandler, 2002; Bowen & Waltermire, 2002; Hodgson *et al*., 2003; Jones *et al*., 2007). Elevation accuracy is dependent on a variety of factors including flight altitude, LiDAR scanner, slope angle, vegetation cover and dGPS data recorded during the flight (Vain *et al*., 2009; Korpela *et al*., 2010; Vain *et al*., 2010). Some of these devices can record multiple reflected pulses and hence record the distance between the transmitter-receiver and each of the objects it strikes (figure 2.6). This has allowed LiDAR to be used in a variety of forestry applications such as measuring canopy heights (Dubayah *et al*., 2000; St-Onge, B. & Achaichia,

N. 2001; Hudak *et al*., 2002), stand volume (Nilsson, 1996; Drake *et al*., 2002), basal area (Means *et al*., 1999; Drake *et al*., 2002) and above ground biomass (Dubayah *et al*., 2000; Lim & Treitz, 2004).

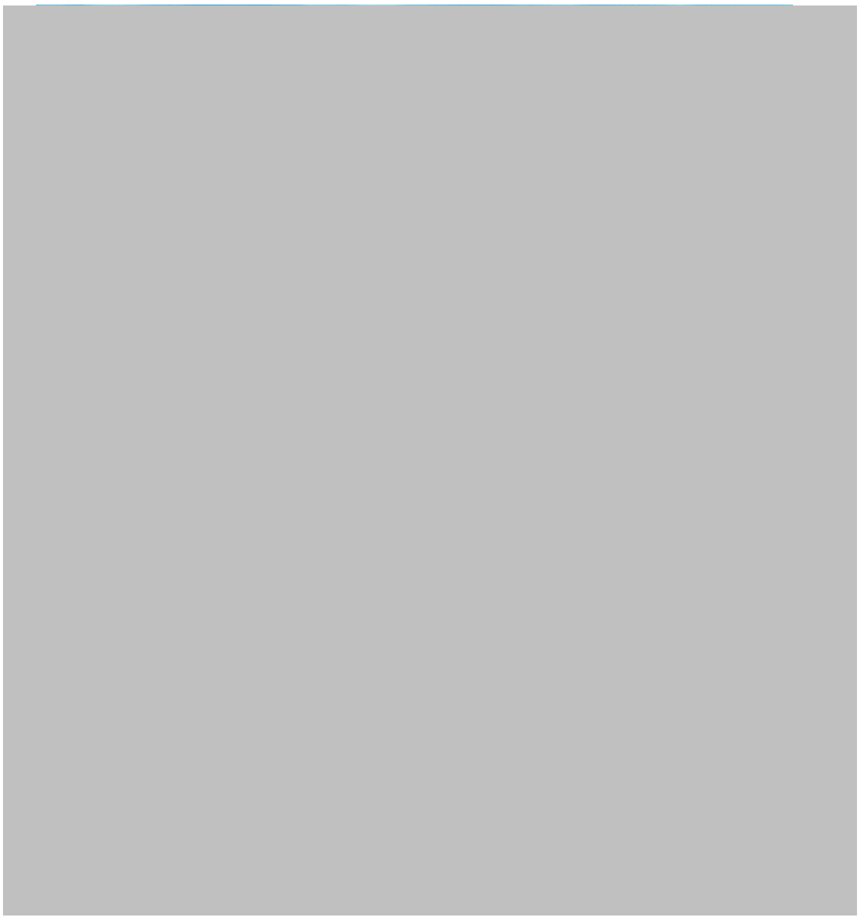

Figure 2.5: Fixed wing aircraft mounted LiDAR device. The aircraft mounted GPS and the Inertial Measurement Unit (IMU) provides detailed information on the exact latitude, longitude, altitude, heading, speed pitch and roll of the aircraft. This information is used together with LiDAR transmitter and receiver to provide accurate geographic (latitude, longitude and elevation) information of the ground topography (adapted from USGS, 2010).

More recently there have also been several studies using LiDAR in order to identify plant communities (Genç, 2004; Bork & Su, 2007; Verrelst *et al*., 2009). Identification of plant communities using LiDAR data radically reduces time and costs involved in plant community mapping (Vierling *et al*., 2008). Recently several authors have used an elevation-hydrology-plant community relationship to map plant communities in wetlands using LiDAR (Morris *et al*., 2005; Prisloe *et al*., 2006; Sadro *et al*., 2007; Korpela *et al*., 2009; Moeslund *et al*., 2011). The Prisloe *et al*. (2006) study differentiated between stands of the invasive species *Phragmites australis* and native lower growing salt marsh vegetation in tidal wetlands in the United States of America. The methodology used in this study calculated the difference between the first and last returns of the LiDAR pulse (figure 2.6). In stands of the tall (~ 2m) *P. australis* vegetation the elevation difference was much greater than that found in typical low growing salt marsh vegetation. However, this methodology was not able to differentiate between plant communities of similar heights. A study by Korpela *et al*. (2009) used the LiDAR intensity data to try to differentiate plant communities in mire ecosystems in Finland. Whilst this study had some success in distinguishing stands of different tree species, it was unable to differentiate ground flora and hence this limits the use of this methodology in coastal wetlands where many plant communities are comprised of low growing species. A study by Morris *et al*. (2005) was able to differentiate between the identified plant communities in tidal salt marshes using LiDAR derived elevation data, as was a study by Moeslund *et al*. (2011) in Atlantic salt meadows. A further study by Sadro *et al*. (2007) study used multi-spectral data to identify plant communities in coastal wetlands. However, this study also analysed LiDAR elevation accuracy in different vegetation and found that there were elevation inaccuracies (up to 0.17m) which resulted in poor model performance and led them to reject LiDAR for vegetation mapping. Therefore this may also have affected the accuracy of both the Morris *et al*. (2005) and the Moeslund *et al*. (2011) plant community models.

The location and extent of the plant communities within most Estonian coastal wetlands is not known. Due to the aforementioned elevation inaccuracies in LiDAR data, the methodologies developed by Morris *et al*. (2005), Sadro *et al*. (2007), and Moeslund *et al*. (2011) are unlikely to be able to distinguish proximal plant communities in such coastal wetlands due to the micro-topographical differences between plant communities (Burnside *et al*., 2008). However, more accurate elevation data can be obtained by using a dGPS. This thesis will develop a methodology to incorporate dGPS elevation data as a post processed correction into the LiDAR.

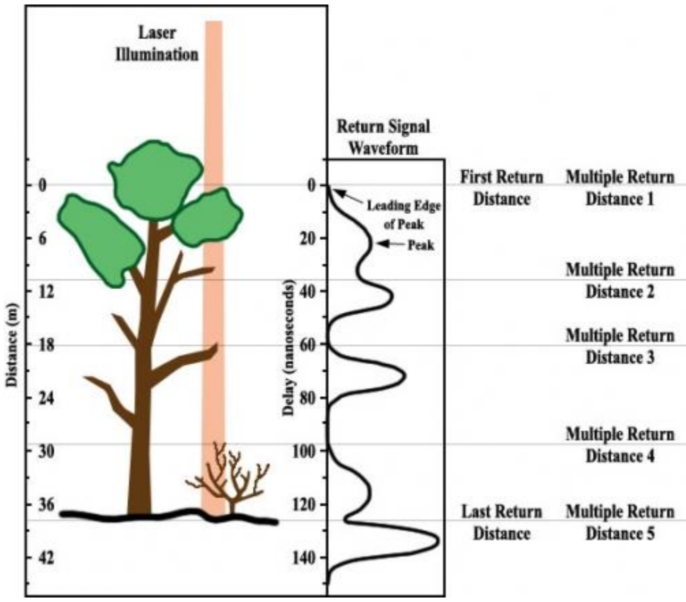

Figure 2.6: LiDAR recording multiple pulses and hence distance between transmitter-receiver and object. First return is the crown of the tree and last return is the ground.

2.4.4 Use of differential GPS (dGPS) for elevation mapping

GPS is a space-based, worldwide radio-navigation system. GPS uses a working constellation of 24 satellites in geosynchronous orbits, and a network of ground control stations to provide free, highly accurate positioning data. The position of a ground based user is calculated using trilateration based on the known location of the satellites and the known speed of radio frequency signals sent from the satellites to a user interface (Leica, 2012). The position of a GPS receiver is calculated relative to a predeterimned geoid model. The geoid model describes the equipotential surface of the Earths gravity field, which best fits global mean sea level. However, there are discrepencies in geoid predicted mean sea level and local mean sea level, due to the Earth’s rotation on its axis and the position of the continents preventing free movement of the oceans. Planimetric geoid errors can be as high as 40m although elevation errors can be greater dependent on location due (ESRI, 2008). This can be accounted for by using locally based ellipsoidal elevation difference error corrections (ESRI, 2008). The standard ellipsoidal error correction in Estonia is based on the BK77 Baltic mean sea level as measured at Kronstadt (Eesti Maa-amet, 2011).

However, there are systematic positional errors in GPS that are derived from a variety of sources (table 2.2). Atmospheric errors such as Ionospheric and Tropospheric are caused by the variable and slower travel time of radio signals through both the Ionosphere and Troposphere than through the vacuum of outer space (Leica, 2012). Positional calculations also require an accurate measure of time, however, whilst the satellites have built in atomic clocks which keep very accurate measures of time, GPS receivers do not have such accurate clocks. This problem is partially overcome by the transmission of code information to synchronise GPS receiver clocks in real time from the satellites, although due to calculation rounding, errors are still present and this translates into positional errors (Trimble, 2012). The multipath effect, another source of positional error, is caused by the reflection of satellite radio signals from objects such as buildings, which causes the reflected signal to reach the receiver after the unreflected signal. Whilst the GPS satellites are in a constant high orbit, subtle shifts can occur due to external gravitational forces such as those caused by the sun and moon. Satellite shifts obviously influence the positional accuracy of GPS devices, although this is limited by a data correction package sent to receivers called ephemeris data (Trimble, 2012).

In a further attempt to improve positional accuracy differential GPS (dGPS) systems can be employed. DGPS uses two GPS receiver devices. The moving GPS receiver (rover unit) position is corrected by comparing the carrier phase measurements of the rover unit with the corrected data from the second GPS unit (base station) which is static and located over a known position, usually a geodetic network point. Geodetic network points are precisely known locations and have been used since the 19th century in order to accurately map terrain (Eesti Maa-amet, 2011). Corrections can either be added in real time (real time kinematic or RTK) or post processed.

In RTK dGPS systems the positional error is continuously transmitted to the roving GPS receiver which can then remove the error in real time to calculate its position with a much greater accuracy, typically Horizontal Dilution of Precision (HDOP) 0.01m and Vertical Dilution of Precision (VDOP) 0.02m (Trimble, 2012).

Post processing GPS data uses a similar method. However instead of using a real time correction the base station GPS records the errors in the GPS satellites every second. These data can then later be used to post process the errors collected by the rover unit giving a similar accuracy to the RTK dGPS. There are many permanent GPS base stations currently operating (Dana, 2000). The main advantage to using the real time kinematic differential GPS methodology is that any problems can be detected immediately in the field.

Table 2.2: Main sources of positional errors when using a standard handheld GPS device (Trimble, 2012).

Error source	Value (m)
Ionosphere	5
Ephemeris	2.5
Clock	1.5
Multipath	0.6
Troposphere	0.5
Total	10.1

A study by Rayburg *et al*. (2009) compared digital elevation models (DEM) from different data sources, including both LiDAR and differential GPS, and found that in open areas dGPS provided a significantly more accurate DEM. However, in areas with forest coverage the inaccuracies of the dGPS derived DEM, due to limited satellite constellation and multipath errors, was much greater than the DEM derived from LiDAR. This suggests that, in appropriate open landscapes a dGPS derived DEM should be able to provide some form of error value for a LiDAR derived DEM located in the same area. A previous study investigated the effect of slope and sampling angle on LiDAR derived DEM accuracy and used dGPS data to assess LiDAR DEM accuracy (Su & Bork, 2006). However, dGPS derived elevation data has not previously been used to provide elevation corrections to LiDAR derived DEMs for plant community modelling. These data should be able to provide an elevation correction to improve plant community model accuracy based on LiDAR derived elevation data, which could then be used as a base correction for a larger dataset.

2.4.5 Predictive plant community modelling

Predictive plant community modelling has its basis in ecological niche theory and gradient analysis (Franklin, 1995). The development of predictive modelling in vegetation systems relies on the establishment of correlative relationships between plant communities and environmental and ecological variables. Further to this, data must be available to create maps based on a variable that is easier to map than the vegetation itself. As has been discussed throughout this literature review, coastal wetlands exhibit a range of environmental and ecological gradients including, elevation, hydrology, salinity, nutrients, and disturbance. It is these factors that have been shown by a variety of authors to influence the spatial arrangement of the plant communities (Tyler, 1971b; Wiegert & Freeman, 1990; Bertness, 1991; Puurmann & Ratas, 1998; Silvertown *et al*., 1999; Bertness *et al*., 2001; Silliman & Bertness, 2004; Araya *et al*., 2010b; Berg *et al*. 2011).

Predictive plant community modelling techniques have been described as static or dynamic models depending on whether they are time independent or time dependent dynamic responses to a changing environment (Franklin, 1995). Both static and dynamic predictive modelling use two different approaches correlative and mechanistic as described by Beerling *et al*. (1995). Correlative models are based upon strong, often indirect links between plant species distribution and environmental factors to construct a predictive model (Burnside & Waite, 2011). The mechanistic approach is based upon expert knowledge of the individual factors which influence the whole ecological system (Guisan & Zimmermann, 2000). Hoffman (2006) suggests that the mechanistic approach can often be more robust due to its basis on a complete understanding of the main factors influencing a plant community. Furthermore, Burnside & Waite (2011) highlight that correlative models can be affected by false absences. However, Robertson *et al*. (2003) state that mechanistic approaches have a major drawback as typically researchers have an incomplete understanding of cause and effect in complex ecological systems.

Robertson *et al*. (2003) tested both correlative and mechanistic models for predicting potential species distribution for a South African coastal dune plant. They demonstrated that the correlative approach could produce models that were very precise in modelling sites or ecosystems. The study went on to suggest that correlative models were able perform as well or better than simple mechanistic models. Hoffman (2006) confirmed that correlative models were equally or more precise than mechanistic models but went on to state that correlative models typically did not generalise or scale well. However, often models consist of a mixture of both correlative and mechanistic approaches (Hoffman, 2006).

Whichever modelling approach is used they are based on data concerning the relationship between environmental gradients and species presence and absence (Franklin, 1995). Austin & Smith (1989) identified three different categories of environmental gradients useful for plant community modelling: that of indirect gradients (gradients that indirectly influence plant species such as elevation, slope and aspect), direct gradients (gradients of environmental variables that directly influence plant species such as salinity, temperature, and pH), and resource gradients (gradients of environmental factors that are used by plants for growth and development such as light, water, nutrients). Data collected relating these environmental gradients to plant community type provide a basis to model the realised niche of the plant species within the plant community, which in itself is not problematic as long as the model is geographically or temporally specific (Franklin, 1995). Westman (1991) suggests that for any model assessing the effects of climate change a mechanistic approach based on knowledge of the fundamental niche of the resource gradients of each individual species is required. However, a variety of other authors have suggested that precise dynamic predictive plant community models can be developed based upon a correlative approach and including indirect or direct gradients, such as topography and edaphic factors (Moore *et al*., 1991; Kessell, 1995; Moffett *et al*., 2010; Coveney & Fotheringham, 2011; Fraser *et al*., 2012).

The rapid and simple correlative approach has been shown by previous authors to produce precise and accurate plant community models, both with regards to present day and future distributions (Beerling *et al*., 1995; Robertson *et al*., 2003). However, to improve the validity of a predictive model using the correlative approach it is useful to include autecological data of plant species (Beerling *et al*., 1995; Robertson *et al*., 2003). With regards to the measured environmental gradient used as a basis for the plant community model, resource gradient data have been suggested by some authors to produce the most accurate predictive maps. However, resource gradient data are much more difficult to collect over large areas than indirect gradient data and several authors have suggested that these data can accurately and precisely predict plant communities within environmentally similar ecosystems (Beerling *et al*., 1995; Franklin, 1995; Guisan & Zimmermann, 2000; Robertson *et al*., 2003; Hoffman, 2006).

With regard to modelling plant communities in Baltic coastal wetlands, limited research has been conducted. However, plant community models developed for other coastal wetland ecosystems have shown that precise plant community models can be produced using a correlative approach based on the indirect resource gradient of elevation above mean sea level (Morris *et al*., 2005; Moeslund *et al*., 2011). Accurate LiDAR derived elevation data are available for large areas and these data have been used to model plant community type in coastal wetlands (Morris *et al*., 2005; Moeslund *et al*., 2011). It should be noted that any predictive plant community model using a correlative approach based on an indirect resource gradient will be limited to a localised geographical extent without recalibration (Franklin, 1995).

2.5 Summary

This review has highlighted the main environmental factors that influence coastal wetlands and possible future threats. The chapter has also critically reviewed the use of GIS and remotely sensed data for use in plant community modelling. With regards to coastal wetlands, few studies have assessed the relationship between the main environmental variables and plant community type, and no studies have been conducted within Baltic coastal wetlands. Further to this the relationship between plant community type and the indirect resource gradient, elevation, has not been quantified.

Elevation has been used previously in coastal wetlands as a basis for predictive plant community modelling. However, previous studies have looked at modelling plant communities in tidal coastal wetlands where elevation differences between plant communities are greater and the plant communities less diverse than those found in Baltic coastal wetlands. The inaccuracies present in previous coastal plant community models would be likely to limit model validity in Baltic coastal wetland systems. Most inaccuracies would occur due to their micro-topographical relief and a new, more robust methodology needs to be developed. In order to improve the accuracy of any predictive plant community model to estimate the effects of sea level rise on plant communities in Baltic coastal wetlands, future sediment accretion must be estimated, but such data are currently missing for Baltic coastal wetlands. Finally, no assessment of the impact of sea level rise on the extent and distribution of the plant communities within Baltic coastal wetlands exists.

3. Study sites and methodology

3.1 Study sites

3.1.1 Location

Estonia is a small country located in north eastern Europe at the eastern end of the Baltic Sea (57° 30.56' and 59° 49.2' N and 21° 45.82' and 28° 12.73' E). The country covers 45 228 km^2 and is the most northerly of the Baltic states. It is bordered to the east by Russia (290km), to the south by Latvia (267km) and to the north and west is the Baltic Sea (figure 3.1). Estonia has a long shallow coastline of 3794km that is made up of shoals, bays, peninsulas and has 1520 low lying islands dotting the shoreline, including the two largest Saaremaa (2673 km^2) and Hiiumaa (989km^2).

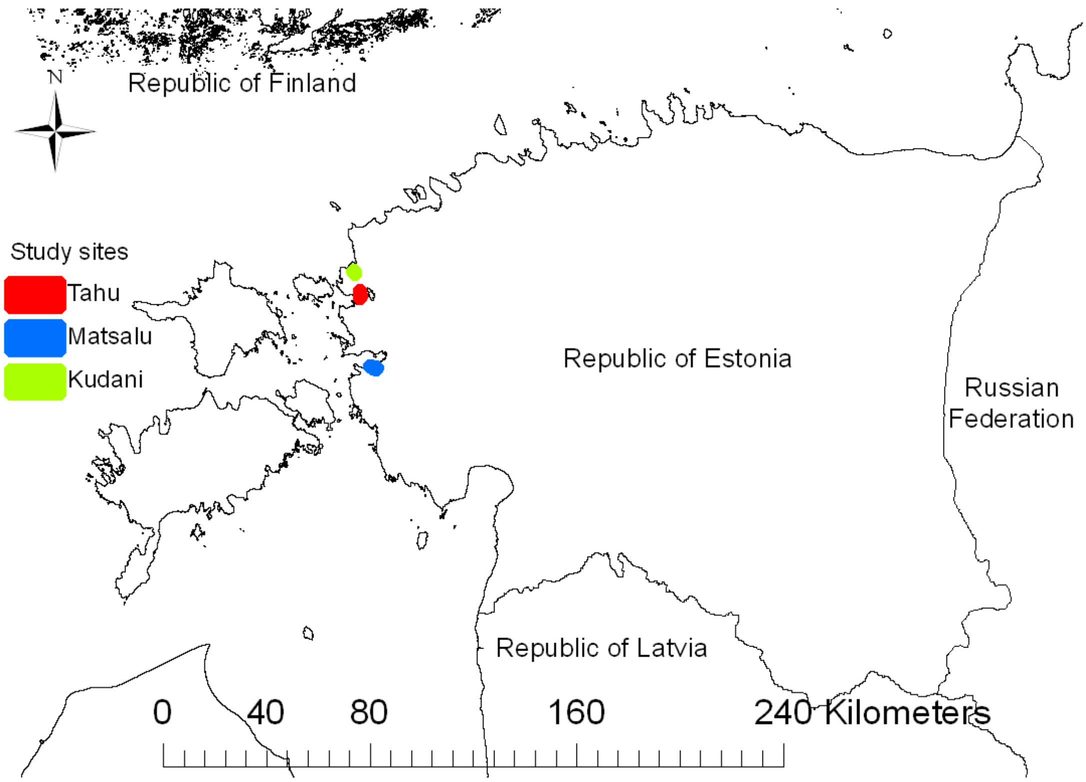

Figure 3.1: The location of the Tahu, Matsalu and Kudani study sites in a regional and national context.

3.1.2 Geology

Despite its small size Estonia contains quite significant variations in geology, morphology and climate. Estonia is situated on the southern slope of the Fennoscandian shield in the north western section of the East European platform. Since the end of the Proterozoic (2500 to 542 mya) period, the Fennoscandian shield, situated north of Estonia, has been mostly mainland. Crystalline stones in Finland are exposed on the ground, however in Estonia these stones are covered by sedimentary rocks. This bedrock lies at 100m depth in north Estonia but is much deeper in the south and has no relation to recent landforms (Ratas & Nilson, 1997). These igneous and metamorphic rocks are mostly made up of gneiss, quartzite and slates as well as Finnish red granites. The igneous and metamorphic rocks are overlain with sedimentary rocks from the Cambrian, Ordovician, Silurian and Devonian geological periods of the Palaeozoic era some 650-350 mya (Ratas & Nilson, 1997). The sedimentary rocks are mostly comprised of limestones and sandstones. The depth of these sedimentary rocks varies from 100m in North Estonia to 600m in the south (Raukas & Tavast, 1994). On top of the limestones and sandstones are quaternary deposits which are thin everywhere in Estonia: less than 5m in North, West and Central Estonia, on the limestone layer and in some areas completely absent. However, on the South-Estonian highlands the residual soil is between 100-200m deep. These quaternary soils are generally formed of moraine, sands, gravel, and peat.

There are considered to be three basic time periods of importance to the Estonian landmass: pre-glacial, glacial, and post glacial (Raukas, 1986).

In the pre-glacial period bedrock belonging to the Fennoscandian shield was subjected to erosion and the majority of the topography is flat, whereas the later Palaeozoic sedimentary rocks found in north and west Estonia are more uneven, as can be seen in the present day landforms of islands showing limestones and dolomites which have thin or absent quaternary covers (Kiipli *et al*., 1993).

During the last glaciation the territory of Estonia was covered by the Scandinavian ice sheet and this ended only during the gotiglacial time between about 13 500-11 000 years ago. Till from the last glaciation is quite extensively distributed in Estonia and is made up of clays with some crystalline material in gravel and pebble fraction (Kiipli *et al*., 1993).

Immediately following the retreat of the Scandinavian ice sheet vast tracts of Estonia were covered with ice dammed lakes in which varved clays accumulated, particularly in western Estonia. These glaciers also levelled off many topographical features due to glacial abrasion.

The post-glacial period has been shaped by the decrease in weight over the territory of Estonia, particularly in the north and west of the country, following the retreat of the ice sheets. This post glacial isostatic rebound has caused much of the landmass of northern and western Estonia to rise from the sea leading to the formation of the many islands and islets of Estonia as well as the joining of some former islands to the mainland (Vallner *et al*., 1988). The elevation of Estonia ranges between 0 and 318m, yet some parts of the territory are still rising, particularly in the north and west of the country. Average isostatic uplift rates of 2.5mm/year are found on the west coast with a maximum of 2.8mm/year in the far north west (Vallner *et al.*, 1988). Figure 3.2 shows the rate of isostatic uplift in Estonia, rising in the north and west but falling in the south east. Estonia is highest in the south east of the territory and gently sloping down to the north west.

Figure 3.2: Isostatic uplift rates in millimetres in Estonia (Vallner *et al.*, 1988).

3.1.3 Climate

Estonia is situated in the temperate zone in the transition area from maritime to continental climate. It is heavily influenced by the Baltic from the north and west and by the Eurasian continental landmass from the south and east (Ratas & Nilson, 1997). The maritime influence consists of the North-Atlantic cyclone belt (modified by the Baltic Sea) and produces a very high degree of climatic variability in Estonia. This causes higher precipitation, stronger winds and more rapid fluctuations in temperature than in other areas that exist at this latitude (Ratas & Nilson, 1997). Prevailing westerly winds push humid maritime air into the interior of Estonia and cause cooler winters and warmer summers than are typically found at this latitude. Typical seasonal start dates can be found in table 3.1. There is a distinct variation in season length with the West coast and the west Estonian Archipelago having shorter autumn and winter seasons than the north coast, north Estonian inland and south Estonian inland (table 3.1). The west coast and western Estonian archipelago have the longest growing season and the north Estonian and south Estonian inland have the shortest (table 3.1).

The vegetation growing period, with days having mean diurnal temperatures >5°C, typically lasts between 170-185 days/year and the period with mean diurnal temperatures >10°C is between 120-130 days/year dependent on the region (Estonian Metrological & Hydrological Institute, 2011). Mean annual precipitation is highest in southern Estonia and can be as high as 700mm/year whereas the large western islands of the Baltic (Saaremaa and Hiiumaa) have the lowest annual precipitation at ~550mm/year (Estonian Metrological & Hydrological Institute, 2011). Total precipitation for the year is ~ 675mm/year compared to evaporation rates of ~400mm/year, (Paal, 1997), creating a very damp climate. As can be seen from table 3.2 May is the month where the mean temperature rises above 5°C and this continues through until October, which is the typical duration for the growing season in west Estonia. Mean total precipitation for this period is about 402mm, and the majority of the precipitation that falls outside of the growing season falls as snow (particularly between November and March, see table 3.2). Typically snow cover varies throughout Estonia with the longest durations being in the south east (122 days) and the shortest being found on the western islands (65 days) (Estonian Meteorological and Hydrological Institute, 2011), however in mild winters much of Estonia has no lasting snow cover.

Table 3.1: Average start date and duration, in days, of each climatic season over five regional areas of Estonia (data obtained from the Estonian Meteorological and Hydrological Institute, 2011).

Region		Autumn	Winter	Spring	Summer
West Estonian	Start	30-Sep	28-Dec	27-Apr	13-Jun
Archipelago	Duration	27	96	47	109
West coast	Start	22-Sep	20-Dec	24-Apr	08-Jun
	Duration	29	105	45	106
North coast	Start	17-Sep	15-Dec	27-Apr	13-Jun
	Duration	33	109	47	96
North Estonian inland	Start	05-Sep	06-Dec	24-Apr	10-Jun
	Duration	39	119	47	87
South Estonian inland	Start	05-Sep	05-Dec	20-Apr	05-Jun
	Duration	41	117	46	92

Typically Baltic Sea ice cover around the Estonian coast varies greatly. The lowest mean number of days with ice cover between 1948-2004 was in Pakri on the western part of the north coast (47.7 days) and the highest mean number of days with ice cover was found in Pärnu located within the gulf of Riga on the western coast (137.8 days) (Jaagus, 2006). Sea ice affects wetland development due to the limited number of days over the winter period with inundation and the removal of vegetation and soil in some years by ice (Puurmann & Ratas, 1998).

The annual average wind speed in inland regions is less than 4 m/s whereas on the coast it is more than 6 m/s. The prevailing wind direction is from the south west (15-25%), and northerly, north-easterly and easterly winds are most frequent during spring and early summer (Estonian Metrological & Hydrological Institute, 2011).

Table 3.2: The mean daily, minimum and maximum temperature in degrees celsius, mean total precipitation in millimetres and mean number of precipitation days for each month for Estonia (data supplied by the Estonian Meteorological and Hydrological Institute, 2011).

Month	Mean t°C Daily min	Daily max	Mean total precipitation (mm)	Mean number of precipitation days
January	-7.5	-2.1	48.3	11
February	-8.2	-2.3	32.4	8
March	-4.9	1.6	32	8
April	0	8	36.5	7
May	4.9	14.6	37.9	7
June	9.8	19.2	56.9	8
July	12	21	78.4	11
August	11.5	19.8	82.2	11
September	7.4	14.7	74	12
October	3.5	9.3	72.7	11
November	-1.3	3.3	67.2	14
December	-5.2	-0.1	56.6	14

3.1.4 Hydrology

The Baltic Sea (including the Gulf of Finland and the Gulf of Bothnia) is the world's largest body of brackish water and has an area of 377 000km^2 (Ratas & Nilson, 1997). It is completely surrounded by land except for the straits between Denmark and Sweden. It drains an area of 1.6 million km^2 and has an annual freshwater input (including precipitation) of 660km^3 (Estonian Metrological & Hydrological Institute, 2010). Mean annual discharge of brackish water is 950km^3 and mean annual inflow of saline water is estimated to be ~475km^3. The Baltic is shallow throughout its area with the deepest point at 459m but mean average depth is 55m. However the coast of west Estonia is considerably shallower at an average of 4.9m with a maximum depth of 22m (Puurmann & Ratas, 1998).

Salinity is typically low and varies from 20ppt in the deep waters near the Danish-Swedish straits to 1-3ppt in the coastal waters off of west Estonia (Puurmann & Ratas, 1998). There are periodic influxes of saline water through the Danish-Swedish straits and salinity does vary throughout the year. In winter salinity levels are typically higher due to influxes from severe weather events through the Danish-Swedish straits and ice formation removing more fresh water from the sea (Ratas & Nilson, 1997). In spring this trend is reversed due to the melting of the sea ice and a large influx of fresh water from snowmelt throughout the catchment of the Baltic. Soil salinity of coastal areas is also at its lowest in spring as accumulated salts are leached out of the soils by snowmelt and rain (Puurmann & Ratas, 1998).

The Baltic Sea, along the Estonian coast, has almost no regular tide (Puurmann & Ratas, 1998). Previous work by Keruss & Sennikovs (1999) and Suursaar *et al*. (2001b) reports that the micro-tidal range is as low as 0.02m. The low micro-tidal range is due to the shallow bathymetry and limited access of the Baltic Sea to the ocean (Suursaar *et al*., 2001b). However major fluctuations in sea level do occur, caused by seasonally changing meteorological conditions and affected by storm surges and changes in barometric pressure. On the west coast of Estonia sea level is typically at its lowest in spring and early summer when easterly winds prevail (Suursaar *et al*. 2001a). In June there is usually a rise in sea level and periodic inundation of the more elevated parts of the coastal wetlands occurs. This combined with higher air temperatures and resulting increased evaporation rates raises the salinity of the soils (Puurmann & Ratas, 1998). Large rises in sea level do occur albeit rarely; the highest recorded sea level was during the 2005 storm Gudrun, which caused a 2.75m storm surge in Pärnu, Estonia. The previous highest storm surge was 38 years before that (2.53m) and sea level has been recorded as low as 1.2m below mean sea level, also in Pärnu (Orviku *et al*., 2003). More typically water levels do not vary more than between 0.3m below mean sea level and 0.4m above mean sea level (Puurmann & Ratas, 1998). The generally low relief of Baltic coastal wetlands (between -0.78m and +1.9m) means that these wetlands can be completely inundated during periods of high sea level. The frequent but variable inundation by brackish water is a controlling factor of plant community composition and abundance, location and species diversity (Tyler, 1971b).

3.1.5 Vegetation

Geobotanically Estonia belongs to the hemi-boreal vegetation zone (Ahti *et al.*, 1968) and the zonal climax vegetation is boreo-nemoral coniferous forest (Paal, 1998). The vegetation composition is diverse and consists of forests, mires and semi-natural grasslands interspersed with more intensively used agricultural land. 44-47% of the territory is made up of forest (including 7% coppices and scrub woodland), mires cover 31%, meadows/grasslands comprise 20%, and cropland covers 0.2% (Paal *et al*., 1997).

Estonian forests have a long history of management and have generally been influenced by fire, drainage or logging. Climax forests for this region are composed of Norway Spruce (*Picea abies*), Silver Birch (*Betula pendula*), Aspen (*Populus tremula*), and Scots Pine (*Pinus sylvestris*). Also found are stands of Small Leaved Lime (*Tilia cordata*), Ash (*Fraxinus excelsior*), Wych Elm (*Ulmus glabra*) and English Oak (*Quercus robur*). These forests make up the majority of the biomass of Estonia. The *Tilia*, *Fraxinus*, *Ulmus* and *Quercus* stands are typically found on thick fertile soils and consequently many of these stands have been cut and the land converted to agriculture. Swamp forests typically consist of Downy/White Birch (*Betula pubescens*) and Common Alder (*Alnus glutinosa*). Primary succesional forests develop in former meadow and grassland areas following abandonment. These forests are usually comprised of Aspen (*Populus tremula*), Grey Alder (*Alnus incana*) and Silver Birch (*Betula pendula*) (Paal, 1998).

Mires are another major land cover type in Estonia and can be found throughout the territory. Mires are divided into three categories in Estonia, minerotrophic fens, mixotrophic mires and ombrotrophic mires using the Masing (1984) classification. Mires are classified according to their hydrochemical conditions as well as their main water inputs. Minerotrophic fens are the most common and widespread mire type in Estonia. Minerotrophic fens occupy 515 000ha, which represents 57% of the total mire area and they acquire water from: precipitation, springs (soligenous), ground water (topogenous), floods, or through the terrestrialisation of lakes or other water bodies (limnogenous). Minerotrophic fens are an alkaline mire type. Fens can be species rich and the number of vascular plant species can exceed one hundred and thirty (Paal *et al*., 1997).

The next most common types of mire are ombrotrophic mires (bogs). They cover 278 000ha (31% of the total mire area). These mires gain their water, and consequently their nutrients, solely from precipitation and are an acidic type. The dominant plant species are *Sphagnum spp*. Mixotrophic mires are the least common of the Estonian mire types and cover an area of 114 000ha (12% of the total Estonian mire area). These mires are considered to be a transitional mire type between fens and bogs. They are usually found on lake and bog margins. Over 16 500 of these Estonian mires exceed 1ha and 143 exceed 1000ha (Paal *et al*., 1997).

Figure 3.3: The distribution of protected areas in green and semi-natural grassland areas as dots (EFN & RDSFNC, 2001)

Estonian grasslands are mostly the result of forest clearance and regular management. They can be divided into wooded meadows, alvars, floodplain grasslands and coastal wet grasslands (EFN & RDSFNC, 2001). Many of these grasslands have been traditionally managed by mowing for hay production, grazing or burning. This low intensity human intervention halts succession to coarser vegetation types such as scrub, woodland and reed beds (Burnside *et al*., 2007). Semi-natural grasslands can be found throughout the territory of Estonia (figure 3.3). Many lie in protected areas due to their high conservation value (figure 3.3) (EFN & RDSFNC, 2001).

Figure 3.4: The distribution of coastal wet grasslands in Estonia (EFFN&RDSFNC, 2001)

In the 1950s coastal wet grasslands occupied 28 750 ha (0.6% of the Estonian land area) (EFN & RDSFNC, 2001). According to the late 1970's land use inventory of Estonia, coastal wet grasslands had decreased to 9513 ha (Paal *et al.*, 1997). Despite its long coastline this comprised only 3% of the total Estonian grassland area. As can be seen in figure 3.4 this coverage is almost exclusively found on the coasts and islands of western Estonia with some exceptions in north Estonia.

On some small islands in Estonia coastal wetland vegetation can cover between 20-100% of the land area (Ratas & Nilson, 1997). Estonian coastal wetlands are characterised by irregular inundation by sea water which is the major hydrological influence. Low but variable salinity of the groundwater is another typical feature and in certain areas facilitates the growth of halophyte species which are otherwise absent from Estonian ecosystems. Estonian coastal wetlands occur in narrow strips along flat sheltered coastal areas and they rarely exceed 200m wide, although these can be much wider in protected bays. Coastal wetlands that occur on the northern coast of Estonia are usually much narrower as the coastal zone is limited due to greater slope angles, therefore they cover much smaller areas. Other major controlling factors influencing the development and species composition of Estonian coastal wetlands are:

(i) Elevation above mean sea level; typically these wetlands do not exceed 2m and the elevation determines site hydrology and consequently plant community type.
(ii) Soil type, which affects drainage and subsequently hydrology. Soils can vary but typically consist of clay, silt, sand, gravel and stones.
(iii) Slope, which will determine the area that is inundated and therefore the width of the wetland area.
(iv) Deposition of organic material (algae, foraminifera etc.).
(v) Exposure to wind and waves, and ice rafting/ erosion.
(vi) Anthropogenic influence, including grazing/ mowing/ burning or abandonment, and erosion due to trampling/ vehicles.
(vii) Neo tectonic uplift, as newly risen flat protected shorelines develop into coastal wetlands.

Estonian coastal wetlands sometimes have an underlying zonal character (Rannap *et al*., 2004) although the plant communities form a mosaic within this zonal arrangement (Burnside *et al*., 2007). The separation of the plant communities has been suggested to be dependent on the elevation above mean sea level and therefore the level of brackish water influence, although no research has yet quantified this relationship.

Paal (1998) has suggested that Estonian coastal wetlands can be simply divided into sublittoral, eulittoral and epilittoral zones. In a study by Rannap *et al.* (2004) it is suggested that: the sub-saline (sublittoral) zone is either permanently flooded or for long periods, and the lower parts of the plants are permanently submerged; the saline (eulittoral) zone is directly influenced by irregular inundation during periods of high water or during storm surges; the supra-saline (epilittoral) zone is more rarely inundated than the saline zone (figure 3.5). Rannap *et al*. (2004) suggested a further plant community, the dry coastal meadow zone, which is only inundated during extreme weather events and the vegetation is similar to that of an alvar meadow. The majority of Baltic coastal wetlands are in a primary succesional stage and have formed on bare unvegetated soil on land that has recently, in geological terms (1000 years, Rebassoo, 1975), risen out of the sea. A recent study by Burnside *et al*. (2007) has identified a greater number of plant communities (figure 3.5) than previously suggested by Paal (1998) and Rannap *et al*. (2004) and suggested that they are located in an irregular mosaic.

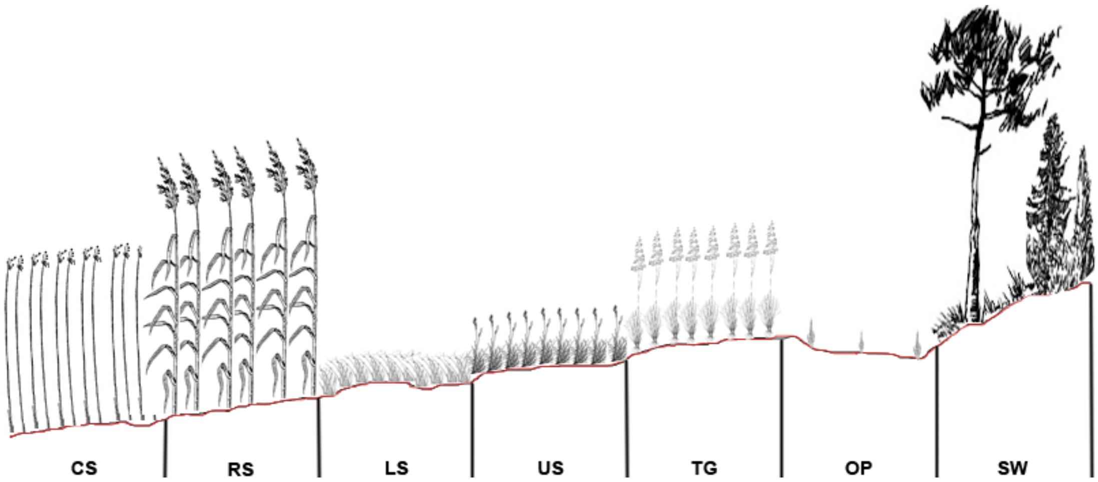

Figure 3.5: Simplified plant community elevation location of plant communities according to the Burnside et al. (2007) classification. CS (Clubrush Swamp) and RS (Reed Swamp) are related to Paal's (1998) sublittoral, LS (Lower Shore) and US (Upper Shore) are equivalent to eulittoral, and TG (Tall Grass) and SW (Scrub and developing Woodland) are equivalent to epilittoral. OP (Open Pioneer) is not included in Paal's (1998) classification.

3.2 Methods

3.2.1 Vegetation sampling

3.2.1.1 Vegetation sampling techniques

Frame quadrats are a commonly used method for visually assessing the percentage cover of species over a given area (Bullock, 1996). Quadrat size is dependent on the type of vegetation being studied (table 3.3). In this study the plant community type being investigated are wetlands with abundant grass species. Therefore the quadrat sizes recommended for this community type are 0.25-16m^2 (table 3.3). A quadrat size of 1m^2 was used to verify if the visually identified plant community contained vegetation corresponding to previous studies. Tilman (1999) states that species interaction in grasslands occur within an area of 1m^2 and several authors have used this quadrat size to assess species composition in wetlands and grasslands (Dix, 1961; Zobel & Liira, 1997; Sluis, 2002; Klimeš, 2003; Burnside *et al*., 2008)

Table 3.3: Recommended quadrat area size ranges for different vegetation types (Bullock, 1996).

Vegetation type	Quadrat size range m^2
Bryophytes, lichen and algal communities	0.01-0.25
Grassland, tall herb, short shrub, or aquatic macrophyte communities	0.25-16
Tall shrub	25-100
Trees in woods and forests	400-2500

A total of 105 quadrats were recorded across seven plant communities in this field research. Plant community was identified by eye using the indicator species provided in Burnside *et al.* (2007). Fifteen quadrats were randomly located within each of the plant communities. The random placing of the quadrats was done by initially identifying the location of all the patches of each plant community in each study site by a walk over survey. The patches of each separate community type were then allocated a number from 1 up to the maximum number of patches. A number was selected using a random number generator indicating in which patch the first quadrat would be placed. This was repeated fifteen times for each individual plant community, giving a total of one hundred and five quadrats per study site. Within each quadrat all the plant species were recorded and the percentage cover of each species estimated. Percentage cover of plant species was determined by estimating the total visible area of all the individuals of each species. The minimum percentage cover was 1% and although some attempt was made to achieve 100% for each quadrat, total percentage cover varied between 98 and 103%, although this was accounted for in the statistical analysis by recalculating the species percentage cover as a proportion of the actual total percentage cover.

There has been some criticism of frame quadrats and species cover estimates due to the subjectivity of this method. In a study by Vittoz & Guisan (2007) they showed that only 45-63% of all species in a grassland habitat were perceived by all observers. However, when using pairs of observers 10-20% less species were overlooked and the majority of species that were overlooked had a cover of <0.1%. Therefore in this study each quadrat was identified by pairs of recorders and full training was given at the beginning of the study in order to limit subjectivity and ensure the recorders were all recording consistently

3.2.1.2 Estonian coastal wetland plant community classification

Estonian coastal wetlands have been classified by a variety of authors and for clarity these classification systems are shown in detail in Appendix I.

The main classification systems have been Rebassoo (1975); the EC Habitat Directive 92/43/EEC (1992) on the conservation of natural habitats and of wild fauna and flora Boreal Baltic coastal meadows, Annex 1 (code: 1630); Paal (1998) or EUNIS European Nature Information System (Davies *et al*., 2004); and Burnside *et al.* (2007).

The Rebassoo (1975) study is the most wide-ranging and detailed of all the studies for Estonian coastal wetlands as it covers every coastal wetland site and plant community found throughout the country. However, the Rebassoo (1975) study failed to take into account issues of management in coastal wetlands, which is an important factor in determining the plant communities (Tyler, 1969; Wallentinus, 1973; Puurmann & Ratas, 1998).

Both Annex 1 of the EC habitat directive 92/43/EEC (1991) and the EUNIS European Nature Information System (Davies *et al*., 2004) cover the whole of Europe and are therefore not fine scale enough for this study. Moreover, the coastal wetlands are covered by a variety of overlapping community types. The descriptions of these communities are explained in Appendix I.

The Paal (1998) study is less detailed than that of Rebassoo (1975) as it only covers all rare and threatened plant communities in the territory of Estonia.

The classification of the coastal wetlands draws heavily on the earlier Rebassoo (1975) work with some corrections and amalgamations.

A more recent study by Burnside *et al.* (2007) used a TWINSPAN analysis to identify plant community types in managed Estonian coastal wetlands and further developed a phytosociological key for characteristic species. The advantage of the Burnside *et al*. (2007) classification is that it covers all the plant communities that occur in the majority of managed Estonian coastal wetlands. The classification also provides a list of indicator species to easily identify plant communities in the field and hence this classification was used for this study. The Burnside *et al*. (2007) study identified 11 different Estonian coastal wetland community types (figure 3.6). At the finest scale these consisted of: Scrub; Short Grass and Scrub; Tall Grass and Scrub; Tall Grass and Reeds; Uppershore and Tall Grass; Short Grass and Uppershore; Short Sedges; Lowershore; Reed Swamp; Clubrush Swamp and Open Pioneer. The phytosociological key was developed using only the plant communities that were more discernible in the field, as can be seen in table 3.4. These vegetation classification types included: Scattered Scrub (which consists of Scrub and Short Grass and Scrub and is elsewhere described as Scrub and developing Woodland and coded SW); Tall Grass (Tall Grass and Scrub and Tall Grass and Reeds and coded TG); Uppershore (Uppershore and Tall Grass; Short Grass and Uppershore and Short Sedges coded US); Lowershore (coded LS); Reed Swamp (coded RS); Clubrush Swamp (coded CS); and Open Pioneer (coded OP) (figure 3.6).

In the phytosociological table developed using TWINSPAN (table 3.4) species were assigned frequency values from I-V dependent on how often a species was present moving from one quadrat to another. The main indicator species for each plant community can be identified using this table. This is based on earlier work by Rodwell (1992) who developed a system for distinguishing plant communities in the field using indicator species presence and abundance derived from a phytosociological key and a species abundance measure (the Domin scale).

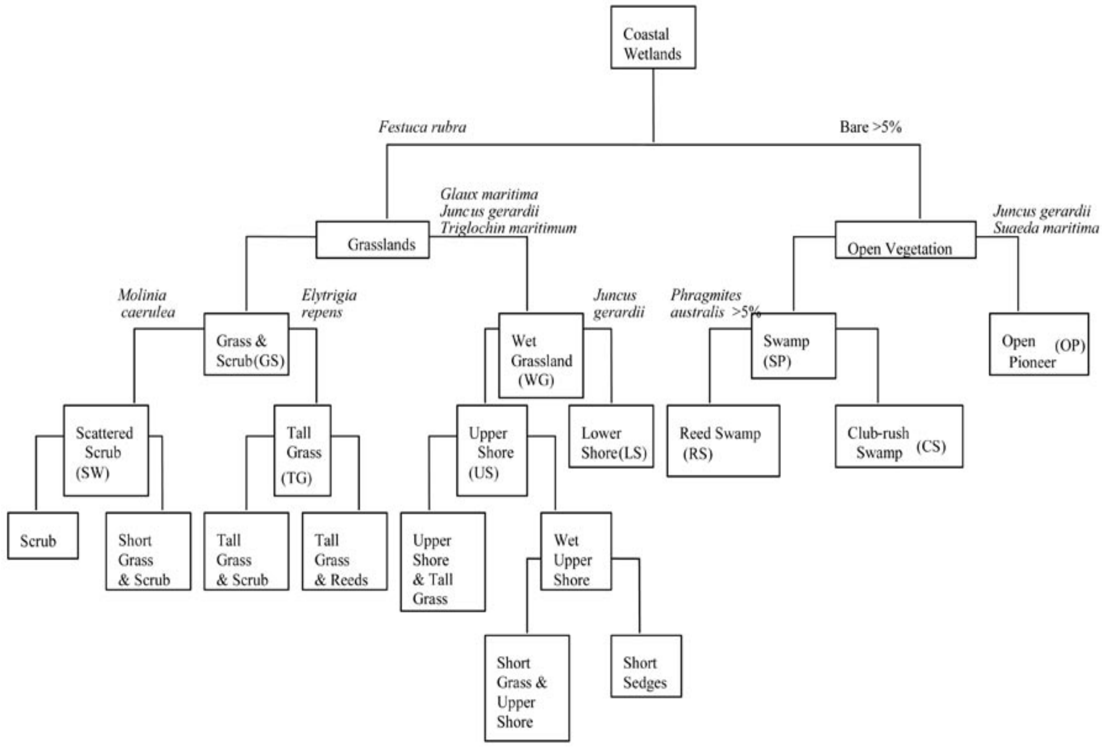

Figure 3.6: Hierarchical classification of coastal wetland plant communities in western Estonia. This was developed using TWINSPAN analysis based on species presence and percentage cover (Burnside *et al.*, 2007).

Table 3.4: Phytosociological classification of coastal wetland plant communities of west Estonia. OP = Open pioneer; CS = Clubrush Swamp; RS = Reed Swamp; LS = Lowershore Grassland; US = Uppershore Grassland; TG = Tall Grassland; SW = Scattered scrub and developing Woodland (Burnside *et al*. 2007)

Species	OP	CS	RS	LS	US	TG	SW
Bare (+5%)	V (7-9)	V (4-8)	V (1-8)	V (1-5)	III (1-4)	III (1-5)	V (1-7)
Suaeda maritima	V (1-6)						
Juncus gerardii	V (1-3)			V (4-9)	IV (1-5)	I (1-3)	
Litter	IV (1)	V (3-5)	V (3-8)	V (3-5)	V (1-7)	V (3-6)	V (3-5)
Agrostis stolonifera	III (1-7)	I (2-4)	II (2-4)	V (3-7)	IV (3-7)	III (3-5)	IV (3-4)
Triglochin maritimum	III (1-4)	IV (1-5)	II (1-4)	V (1-5)	V (1-4)	II (1-4)	I (3-4)
Bolboschoenus maritimus	II (1-5)	V (4-7)	II (1-3)	I (1-5)	I (1)		I (1)
Salicornia europaea	II (1-2)						
Glaux maritima	II (1)	I (1-4)	I (3-5)	V (1-7)	IV (1-4)	II (1-3)	I (1)
Plantago maritima	II (1)	I (1)	I (4-5)	V (1-5)	V (1-7)	II (1-5)	II (1-4)
Schoenoplectus lacustris		IV (4-8)	I (4)	I (1)	I (1)		
Eleocharis palustris		II (1-8)	I (1-3)	III (1-	II (1-7)		
Phragmites australis		I (1-3)	V (5-9)	III (2-	I (1-3)	IV (1-5)	III (1-4)
Elytrigia repens			II (2-4)		I (3-4)	V (3-8)	II (3-5)
Galium palustre			II (1-5)	III (1-	II (1-4)	III (1-5)	II (1-4)
Festuca rubra			I (1-3)	V (3-7)	V (4-8)	V (3-7)	III (3-5)
Peucedenum palustre			I (1-2)		II (1-5)	III (1-4)	III (1-3)
Vicia cracca			I (1)		I (1-4)	IV (1-4)	III (1-4)
Potentilla anserina			I (1)	II (1-4)	II (1-4)	V (1-5)	III (1-4)
Moss				II (1-4)	IV (1-3)	III (1-5)	III (1-7)
Leontodon autumnalis				II (1-3)	V (1-4)	II (1-4)	II (1-3)
Molinea caerulea				I (1-4)	II (1-7)	I (1-8)	III (4-7)
Carex nigra				I (1-2)	II (1-5)	I (1-3)	I (1-4)
Centaurium littorale				I (1)	II (1-3)		II (1-3)
Carex distans				I (1)	II (1-6)	II (1-6)	I (1-5)
Galium verum				I (1)	I (1)	II (1-4)	IV (3-5)
Trifolium pratense				I (1)	II (1-3)	III (1-4)	II (1-4)
Juniperus communis					I (1)		IV (1-7)
Carex glareosa					I (1-4)		
Valeriana officinalis						IV (1-5)	III (1-4)
Festuca arundinacea						II (3-7)	I (1-5)
Achillea millefolium						II (1-5)	III (1-5)
Angelica palustris						II (1-3)	I (1-3)
Centaurea jacea						I (1-4)	
Frangula alnus						I (1)	II (1-8)
Pinus sylvestris						I (1)	III (1-8)
Orchidaceae spp.							I (1)

3.2.2 Abiotic factors

A variety of environmental variables were recorded during this study including: soil nitrogen, phosphorus, potassium, organic matter content, moisture, salinity and pH. With the exception of soil moisture content all of these parameters were analysed in the laboratory of the Estonian University of Life Sciences, measured from soil samples collected in the field and air dried. A 40g soil sample was collected using a trowel from the centre of each quadrat, placed into a sample bag and transported to the laboratory.

3.2.2.1 Soil moisture and water table level

Soil moisture data were recorded using a Delta T WET Sensor over a four hour period starting at 10:00 and finishing at 14:00 in order to limit the effects of weather conditions. Three recordings were taken in each of the fifteen quadrats and a mean value calculated for each quadrat. The weather did not change throughout the duration of the data collection period. However, it should be noted that immediately prior to the study strong winds had raised the sea level in the coastal wetlands and inundated the CS, RS and LS plant communities.

In Estonia hydrological data are limited and there were none available for the study areas (Puurmann *pers. comm*. 2008). Therefore, in order to provide baseline hydrological data a series of dipwells were installed at the Tahu study area and site water levels measured (figure 3.7). Two dipwells were installed in each of the plant communities giving a total of 14. The advantages of this method are that soil permeability can be determined, the method is simple, quick and easy to set up and produces accurate results. The limitations are that dipwells give an average value of soil permeability and variations of the permeability in the soil profile are not able to be determined as the permeability of one soil layer may dominate.

Previous studies in Estonian coastal wetlands have had problems with infilling of the standpipe by fine sediments (Berg, 2009). Therefore in order to limit this problem two 1mm holes were drilled at 0.05m intervals opposite each other up the whole length of the pipe and these were covered with fine mesh gauze. The 1.5m standpipes were lowered into the boreholes and sealed at the top with a removable seal in order to prevent water penetration from surface water and to enable easy access when measuring the water table.

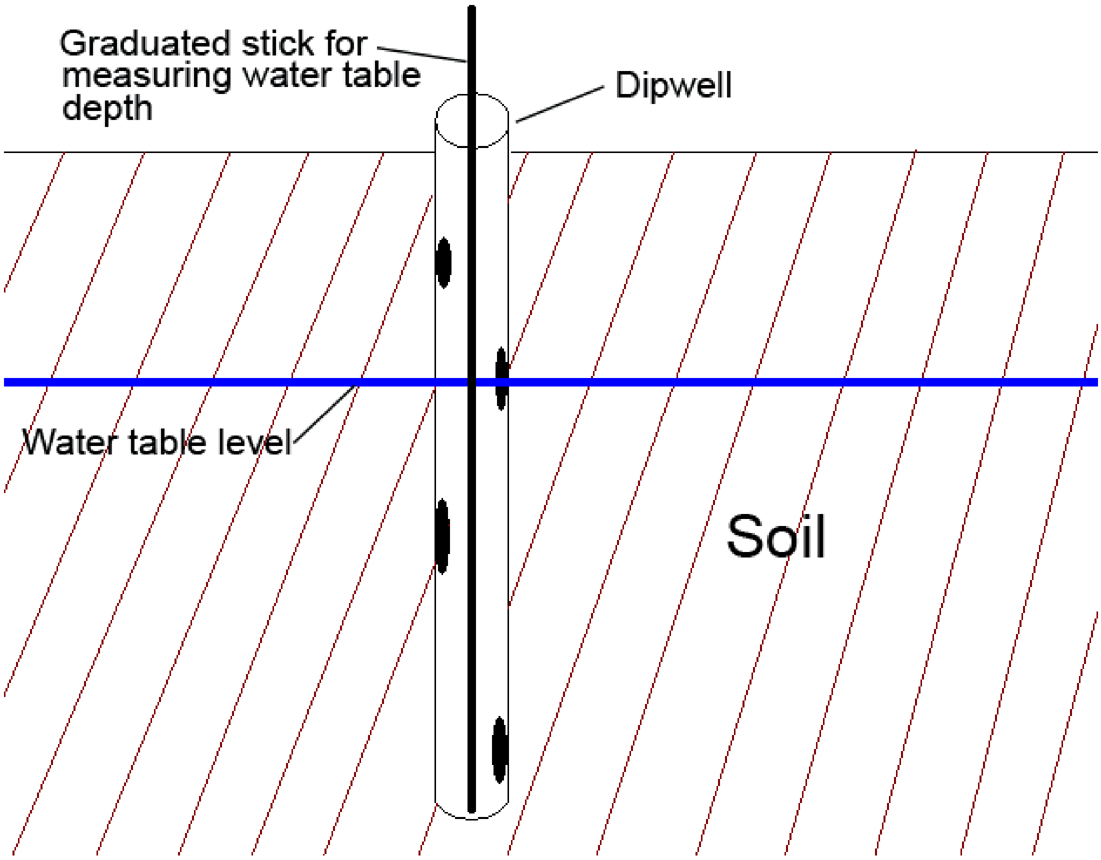

Figure 3.7: A dipwell with holes along its length. The dipwell is made from a perforated PVC tube placed into a hole in the ground. The dipwell was sealed at the top but the seal was removed to insert the graduated stick which was used to measure the water table depth.

3.2.2.2 Soil salinity and pH

Soil samples were taken in each quadrat and air dried in the laboratory, before being weighed out into 5g portions and mixed with 10ml of distilled water and stirred in a temperature controlled orbital shaker for 10 minutes (Jones & Reynolds, 1996). The resultant solutions were then tested with an electronic pH and salinity meter (WTW pH/conductivity 340i). The tests were repeated three times for each sample and a mean value calculated. In between each test the probes were washed with distilled water and dried

(Burnside *et al*., 2008). The salinity and pH probes were calibrated at the beginning of each day and again after every 2 hours of use.

3.2.2.3 Nitrogen, Potassium, Phosphorus and organic matter content

The methodology used to determine total nitrogen in the soil was the Kjeldahl method (Craft *et al.*, 1991) due to its universality, precision and reproducibility.

Available phosphorus was determined using an ammonium lactate extractable flow injection analysis (Egner-Riehm method) (Vucāns *et al.,* 2008). In a comparison by Vucāns *et al.* (2008) it was found that of the three methods tested (Olsen, Mehlich-3 and Egner-Riehm) the Mehlich-3 method and the Egner-Riehm methods were the best with the Mehlich-3 method providing better results in soils with pH>7.5. The soils of the study sites are pH <7 and so the Egner Riehm method was used.

Soil organic matter content was calculated using the loss on ignition method (Schulte, 1995). Ball (1963) in an examination of the loss on ignition method for determining organic carbon in soils, showed that it was a good indicator of soil organic matter content and the method was particularly good due to its simplicity. One of the main criticisms of this methodology is that if the ignition is at high temperatures then additional soil weight loss can occur due to the loss of CO_2 from carbonates in the soil and the loss of structural water from clay minerals. However, Keeling (1962) showed inaccuracies derived from the loss of structural water from the clay and the oxidation of carbonates in the soil are not significant at temperatures below 450°c for the former and 375°c for the latter and provided accurate results. The loss on ignition tests in this study took place over a 24 hour period at 360°c in order to limit the inaccuracies that occur at higher temperatures.

3.2.2.4 Sediment particle size analysis

Sediment particle size analysis was conducted as it can have a marked influence on plant species presence/absence (White, 2006). The sediment particle size analysis in this study used an accurate laser particle size analyser (Malvern Mastersizer 2000 Laser Particle Size Analyser) (Ashworth *et al*. 2011). This method sorts the sediment according to diameter of the individual grains and only requires very small sediment samples (Eshel *et al.*, 2004). The grain sizes were measured according to the Wentworth (1922) grading system, which is one of the most commonly used grading systems (Loveland & Whalley, 2000). The fraction size ranges are >2mm (Gravel), 2mm-1mm (Very coarse sand), 1mm-500μm (Coarse sand), 500μm-250μm (Medium sand), 250μm-125μm (Fine sand), 125μm-62.5μm (Very fine sand), 62.5μm-3.90625μm (Silt), 3.90625μm -1μm (Clay), <1μm (Colloid).

3.2.2.5 Assessment of sediment accretion rates

3.2.2.5.1 Field methods

Dating of discrete depths in soils in order to ascertain sediment accretion rates requires the collection of non-compacted sediment cores (Teasdale, 2011). Four sediment cores were collected by digging trenches and removing the core from the edge of the trench, placing the sediment core in a monolith tray and packing and sealing it for transport to the freezer. Suitable care was taken not to contaminate, degrade or otherwise change the state of the cores. The cores were stored at -26°C in sealed containers to preserve them until ready for laboratory preparation and analysis.

At the Matsalu study site two cores were taken, each 0.08m diameter and 0.4m deep. 0.4m was expected to cover the last 150 years of sediment accretion which is the maximum limit of ^{210}Pb dating. One of the cores was collected from LS community at an elevation of 0.04m above m.s.l. and the other was collected from the TG community at an elevation of 0.69m.

At Tahu two 0.4m deep, 0.08m diameter cores, were also collected. The core removed from the lower elevation plant community was located in the LS plant community at an elevation of 0.07m above m.s.l.. The core selected from the higher elevation plant community was located in the TG community at an elevation of 0.70m above m.s.l.. The elevation data were collected using a Leica dGPS system, mean vertical and horizontal error <0.02m.

3.2.2.5.2 Laboratory methods for determination of radionuclide activity

In order to determine detailed sediment accretion rates in the wetlands the cores were analysed using the methods described by Cundy *et al*. (2003) and Mizugaki *et al*. (2006). This method was selected so as to obtain high resolution results in order to ascertain detailed sediment accretion rates. The sediment cores were thawed in the laboratory overnight and then sliced into 0.01m depth increments. The samples were weighed prior to and after drying in an oven at 40°c until a constant weight was achieved (Guebuem *et al*., 2004; Mizugaki *et al*., 2006). These data were used to determine soil moisture content and hence dry bulk density down the profile for the CRS analysis (Appleby & Oldfield 1992). Following drying, each of the samples were lightly ground using a pestle and mortar to disaggregate the sediments.

Each disaggregated sample was accurately weighed (to 1/1000g) and then placed in a Canberra well type ultra-low background HPGe gamma-ray spectrometer to determine the activity of the ^{137}Cs, ^{210}Pb and ^{214}Pb. The spectra were accumulated using a 16k channel integrated multichannel analyser. Spectral analysis was conducted using the Genie 2000 system. Energy and efficiency calibrations were carried out using bentonite clay spiked with a mixed gamma-emitting radionuclide standard, QCYK8163, and checked against an IAEA certified sediment reference material (IAEA 135). Detection limits of radionuclides are dependent on time, radionuclide gamma energy, count time and sample mass. To achieve maximum quality of data within a minimum time period the samples were left counting until detection error was ≤5% for all the relevant radionuclides. Typically each sample count time was between 24 and 48 hours. Calculations for determining sediment accretion from these data are shown in chapter 6.

3.2.3 Geomatics

3.2.3.1 dGPS survey

This study used a differential GPS (dGPS) system in order to calculate the location and elevation within each individual quadrat. The dGPS system works in a similar way to a standard GPS system. As discussed in chapter 2, differential GPS uses differential correction techniques to enhance the quality of the location data to remove errors inherent in standard GPS receivers. Differential correction techniques can be applied in real time or by post processing the data. Both of these methods use the same underlying principles (Chivers, 2011). In this study an RTK dGPS system was used as it provides the opportunity to monitor GPS location accuracy in real time. A Positional Dilution of Precision (PDOP) value of 0.02m was used to provide accurate elevation data.

The system used in this study was a base station set over a geodetic network point. In Estonia the geodetic network points are accurate to 0.01m both horizontally and vertically as of August 2008 (Eesti Maa-amet, 2011). The base station used a radio transmitter to continuously send location corrections to the rover unit, allowing the rover unit to record data with a mean elevation accuracy of 0.02m (figure 3.8) (Leica, 2012).

Latitude, longitude and elevation were recorded in degrees, minutes, and decimal seconds using the WGS84 geoid. Elevation was recorded in metres above mean sea level. The elevation was later adjusted to take into account the geoid separation from the ellipsoid using the local orthometric height datum correction BK77 mean sea level as measured at Kronstadt. This datum is the standard for the eastern Baltic (Яковлев, 1989). At the Tahu study site the orthometric height datum correction was 20.943m, at Matsalu 12.230m and at Kudani 23.620m (data obtained from the Estonian Land Board, *pers. comm*.).

DGPS data were collected to assess the relationship between plant community type and elevation at Tahu (chapter 4) and to assess the accuracy of LiDAR elevation data in a variety of plant communities at Tahu (chapter 5).

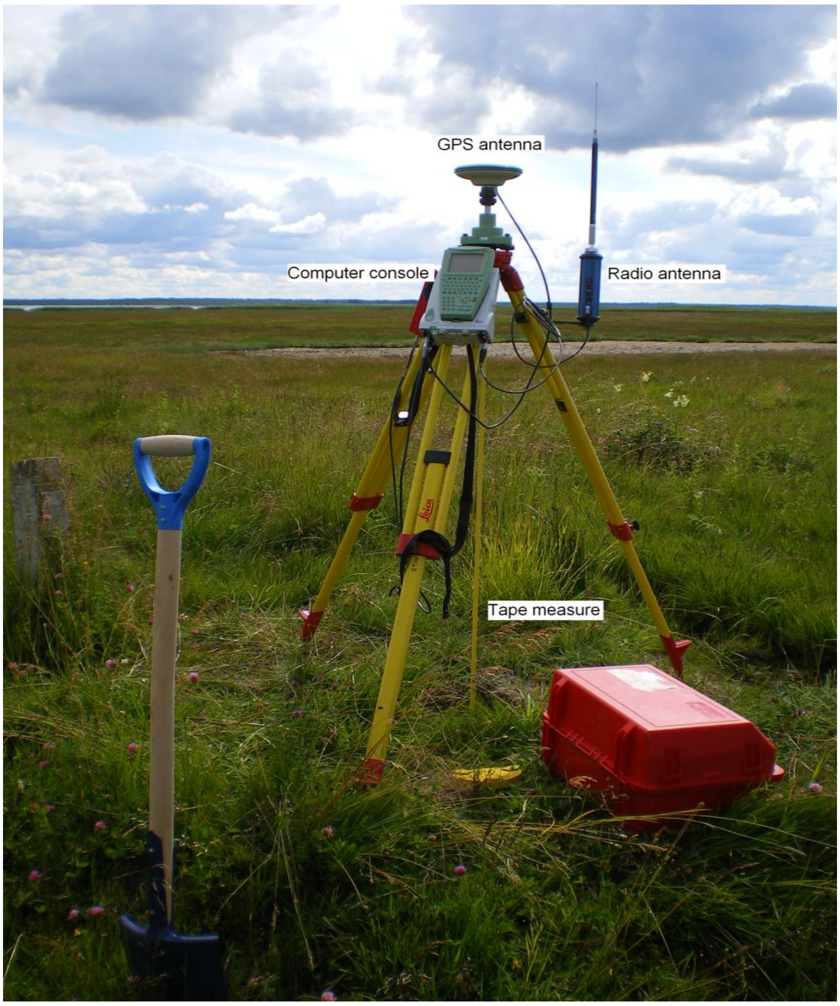

Figure 3.8: dGPS base station set up over geodetic network point. The computer located on the tripod has the co-ordinates of the geodetic network point. The height of the GPS antenna above the point is measured and entered into the GPS console. Corrections are then continuously transmitted to the rover unit by radio signal.

3.2.3.2 LiDAR data

The LiDAR data used for this study were supplied by the Estonian Land Board. Data were collected using an aeroplane mounted Leica ALS50-II scanner time-of-flight (TOF) based scanner with an operational altitude of 200m to 6000m. The mean resolution was 0.45pts/m^2, and data points were a maximum distance of 2.6m apart. The maximum scan rate of the Leica ALS50-II is 90 Hz (scan lines per second) and the maximum pulse rate is 150 kHz (pulses per second). The Leica ALS50-II uses an eye safe laser wavelength of 1064nm, which is standard for terrestrial LiDAR, although has limited penetrability through water, approximately 0.10m. It has an MPiA scanner (Multiple Pulse) which means the laser can fire one pulse before the return of the previous. The MPiA operates from 1200m altitude and provides more returns per m^2. The device can record four returns per pulse (first, second, third and last) and three intensities (first, second, third). Maximum scan angle is 75° from NADIR and it can record up to 300 GB of data. The laser footprint size on the ground is 0.54m and the Estonian Land Board estimated the elevation component accuracy to be between 0.07-0.12m averaged over all ground coverage in Estonia (Eesti Maa-amet, 2011).

3.2.3.3 Spatial interpolation

Spatial interpolation is a process involving the estimation of data values in unsampled areas within the region covered by existing observations (Longley *et al*., 2011). There are several methods for interpolating spatial data but these are all based on a central theme of spatial autocorrelation. The assumption of spatial autocorrelation is derived from Tobler's first law of geography "nearby things are more related than distant things" (de Smith *et al*., 2007). LiDAR is recorded as point elevation data and hence there are no available elevation data for the areas not covered by the points. Therefore, some form of spatial interpolation was necessary in order to produce a continuous DEM derived from the LiDAR data.

Three commonly used interpolation models are Triangulated Irregular Network (TIN), Inverse Distance Weighting (IDW) and Ordinary Kriging (OK) (Longley *et al*., 2011). All three of these techniques were used in this study. Further information discussing the underlying assumptions and processes involved in the TIN, IDW and OK techniques as well as an assessment of these three techniques is made in chapter 5. However, each of these approaches are introduced in this chapter.

There are a variety of methods for performing TIN interpolations. The standard form is linear interpolation using Delaunay triangles (Longley *et al*., 2011), which was used in this study. The Delaunay TIN method is a proximal method of interpolation which is particularly good at describing surfaces without important breaks of surface continuity. In TIN construction vertices are joined by connecting each other to form a series of irregularly shaped non-overlapping triangular elements. The Delaunay linear method of TIN construction ensures that no vertices lie within the circumscribing circle from each triangle (figure 3.9). This prevents the formation of any long thin triangles (de Smith *et al*., 2007).

Figure 3.9: Delaunay method of triangulation. No points lie within the circumscribing circle of any of the triangles (ESRI, 2008).

The IDW interpolation works on the premise that points further away should have their contributions to the immediate point diminished according to distance. This method is a local interpolator and statistically calculates from measured points within localised neighbourhoods (Davis, 2002). The basis of the interpolation is the assumption that data are spatially autocorrelated (Lo & Yeung, 2002). Therefore this method interpolates values in unmeasured locations from surrounding measured locations, giving a lesser weighting to data that are further away. Deterministic interpolations create surfaces from measured points (Negreiros *et al*., 2008). IDW is an exact interpolator in that its interpolated results pass through all the data values (Lo & Yeung, 2002; Longley *et al*., 2011; ESRI, 2008). This is because an interpolation that is located at a known data point produces infinities and must therefore be copied over (Burrough & McDonnel, 1998). IDW is a simple technique which often does not reproduce the local shape implied by the data. Because IDW is an averaging technique, no point in the interpolation can have a z value that exceeds the largest or smallest z value (Lo & Yeung, 2002), which can form bulls eyes in the interpolation (figure 3.10).

Figure 3.10: Profile of IDW interpolation of elevation. This IDW interpolation is exhibiting the bulls eye effect caused by averaging. (ESRI, 2008).

Ordinary Kriging (OK) is a widely used geostatistical technique. It uses a similar procedure to that used in IDW except weights are not based on the arbitrary function of distance but are derived from the model variogram. The OK interpolation model can take into account direction of variation as well as incorporate trends into the interpolation to create better predictions. However, this method is processor intensive and calculation speeds can be very low dependent on the number of points in the dataset.

The plant community model employed in the current study was initially developed for the Tahu site and all initial interpolation and model validation were performed there. Prior to interpolation the LiDAR data underwent data brushing (Burrough & McDonnel, 1998) in order to determine any statistical outliers and potentially erroneous elevation data points. In addition all but the last and only returns were removed to leave only the bare earth data points.

The data brushed LiDAR data were loaded into ArcGIS 9.3 and the Delaunay method of TIN interpolation was selected to produce a bare earth DEM (Morgan & Habib, 2002).

The IDW and OK methods required further analysis of the spatial dataset and hence the data were tested for spatial autocorrelation using a graduated Morans I test, a prerequisite for the IDW interpolation (Longley *et al*., 2011). The LiDAR data were also analysed to identify the average distance of the nearest neighbour which is used to assess the lag size, i.e. sample distance used to bin pairs of points, used in the IDW and OK methods (ESRI, 2010). If the lag size is incorrectly selected then the interpolation will not fit the known real-scale dependent spatial correlation (ESRI, 2010). If the lag size is too small then this can produce empty bins and the sample sizes within bins will be too small to obtain representative averages. If the lag size is too large then this will mask short range spatial autocorrelation (ESRI, 2010). In both the IDW and OK interpolation methods, changes to the minimum and maximum number of neighbours used in the interpolation affect the output (ESRI, 2010). In this study the minimum and maximum number of neighbours used in the IDW and OK interpolations were based on the LiDAR data points that were predicted to occur within plant community patches, based on site knowledge, and hence have similar elevation values. Directional influences in the dataset can affect the IDW and OK interpolations (de Smith *et al*., 2010). In ArcGIS these influences can be accounted for by selecting an anisotropic option and further to this allows the user to select the direction and distance of any influence to the interpolation, i.e slope or wind direction. The angle and distance of anisotropy were calculated using site knowledge and data acquired in the prior spatial data analysis.

The IDW method is based on interpolating data from a set of known points with a weight function based on inverse distance (de Smith *et al*., 2010). The IDW method allows the significance of known points to be changed based on their distance from the output point by using a power parameter (ESRI, 2010). ArcGIS has a power optimisation function which calculates the optimum significance level of known points to an interpolated point, i.e. that which will produce a surface with the lowest root mean square error value (RMSE) (de Smith *et al*., 2007). The power optimisation function was used in the IDW interpolation in this study.

The OK method relies on the assumption that there are no directional trends in the dataset (ESRI, 2010). Therefore any trends in the dataset will influence the validity of the model and hence must be identified and removed (de Smith *et al*., 2007). Trend levels in the LiDAR data were identified using a trend analysis within ArcGIS and a first order trend removal selected in the OK method based on the results of the trend analysis.

Following the formation of a TIN, IDW and OK elevation model of the LiDAR elevation data, cross validation was performed on all of the interpolations, a standard technique in assessing interpolation validity (Tomczak, 1998). Cross validation involved the removal of each data point sequentially and predicted the value that would occur at that location using the interpolation methodology. This produced a series of measured and predicted values for each data point and the similarity between the two was assessed using regression analysis.

3.2.3.4 Plant community modelling

The method that produced the best LiDAR derived elevation interpolation at Tahu in cross validation was selected as the basis for the plant community model. The study detailed in chapter 4 quantified the relationship between elevation above m.s.l. and plant community type at Tahu, although the OP community was not found to be located at a significantly different elevation to the TG plant community. The OP community is not present in many Estonian coastal wetlands and covers a small proportion of the total wetland area where it is present (Rannap *et al*., 2004). Therefore, the OP plant community was excluded from the plant community model.

Prior to producing a plant community model based on LiDAR elevation data an evaluation of the accuracy of the LiDAR data was performed. A comparison was made between elevation data collected using an RTK differential GPS and each point recorded was found to be accurate to within 0.02m both vertically and horizontally at Tahu. Although RTK dGPS elevation data were collected in all of the plant communities, it was not possible to collect elevation data for all of the plant communities within one patch due to their mosaic nature (Burnside *et al*., 2007). Therefore RTK dGPS elevation data were collected from a 20m x 100m patch that included five of the plant communities (CS, RS, LS, US and TG see table 3.4) and from a separate patch 20m x 20m covered only by SW. RTK dGPS elevation data were collected every 0.5m within both of the patches. The RTK dGPS and LiDAR elevation data for each plant community were compared in Matlab R2010a using a 2m x 2m moving window so as to include at least one LiDAR elevation data point in each step window (Wang *et al*., 2009). A correction was added to the whole LiDAR elevation dataset based on the RTK dGPS elevation data. The LiDAR elevation data were then categorised based on the Q1 and Q3 elevation values of each plant community (chapter 4). Where there was overlap between the quartile elevation values of two plant communities, a value half way between the two overlapping values was selected as the terminal value of one plant community and the beginning of the next. This categorisation was applied to the whole TIN interpolation.

In order to perform a validation of the plant community model at Tahu the plant community was identified at ninety points over the whole site using the same methodology as in chapter 4 (fifteen per plant community) during a stratified random walkover survey. The actual and predicted plant community were compared using a Fleiss's Kappa coefficient (Landis & Koch, 1977) for the IDW, TIN, and OK interpolation models.

The parameters used in the plant community model that were best able to predict plant community type at Tahu were used to develop a plant community model for two other Estonian coastal wetland sites, Matsalu and Kudani. Model validity was assessed by using a random number generated grid method (Greig-Smith, 1983) selecting fifteen points within each plant community as predicted by the model, totalling ninety per site. Each of these points were located in the field using a GPS and plant community type was evaluated using $1m^2$ quadrats to identify the plant species presence and abundance, using the same methodology as in chapter 4 (Rodwell, 1992). The predicted and actual plant community types were compared using Fleiss Kappa (κ) coefficients (Landis & Koch, 1977) to assess model validity.

The plant community models developed for Tahu, Kudani, and Matsalu provided a prediction of the present day location and extent of the CS, RS, LS, US, TG and SW plant communities based on elevation above m.s.l. (figure 3.11). The models for Tahu, Matsalu, and Kudani were used as a basis to predict the future location and extent of the plant communities, in

2099. The dynamic plant community models for Tahu, Matsalu, and Kudani predicted plant community location and extent by estimating the future elevation above m.s.l. (figure 3.11). For the Tahu, Matsalu, and Kudani sites, five models were developed taking into account isostatic uplift rates (Vallner *et al*., 1988), four IPCC sea level rise scenarios and one scenario assuming no change in eustatic sea level. In addition to this an estimation of sediment accretion was made taking into account both an increase in storminess, and hence sediment accretion, and no increase in sediment accretion rates (Schuerch *et al*., 2011). Sediment accretion was predicted by extrapolating from past sediment accretion rates, as calculated in chapter 6. Validation of the model was based on the assessment of the predicted and actual plant community as predicted by the present day model (figure 3.11).

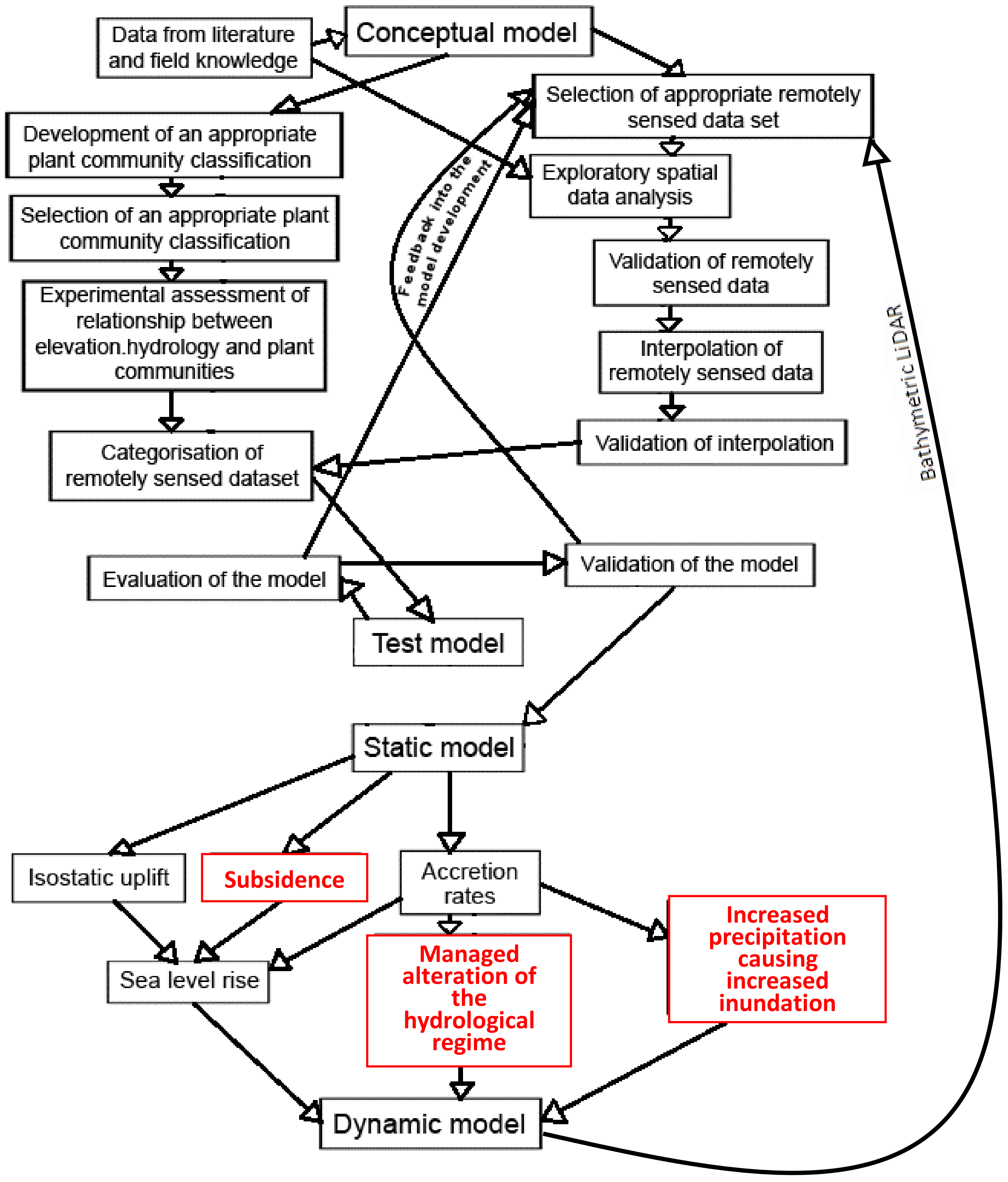

Figure 3.11: Overview of the conceptual system used in the development of the static and dynamic predictive plant community models. The dynamic model developed to predict changes due to sea level rise in Estonian coastal wetlands follows the final pathway taking into account isostatic uplift, accretion rates and sea level rise. Steps in red are potential factors required for using the model for wetland restoration, or for using the model in alternative sites such as floodplains and tidal coastal wetlands.

3.2.4 Statistical analysis

Initial statistical analysis of the edaphic and elevation data was undertaken using Kruskal-Wallis tests. The Kruskal-Wallis tests were conducted in order to establish if there were any differences in the environmental variables of different plant communities. This test is commonly used in ecology to test for differences between the medians of more than two samples, and is a non-parametric equivalent to a one way ANOVA used for data that are not normally distributed, as was the case for all the environmental variables. A post hoc test was required in order to assess where the variation in the data lies, the equivalent to a Tukey or similar comparison as used in ANOVA. In this study a series of Mann-Whitney tests were conducted. This test however requires multiple comparisons to be made on the same data, which can increase the likelihood of a Type I error occurring (Waite, 2000). This problem can be overcome by setting more stringent values to the test such as the Bonferroni or Šidák corrections. The Bonferroni method is more likely to cause Type II errors than the Šidák method (Šidák 1971). However, the Šidák correction method requires the additional condition that data are independent, which was not the case for these environmental data (Sokal & Rohlf, 1994). Therefore the Bonferroni correction was used in this study.

Canonical Correspondence Analysis (CCA) has become a very widely used tool in ecology (Makarenkov & Legendre, 2002; Wagner, 2004; Tornblom *et al*., 2011). It is derived from Correspondence Analysis but has been modified to incorporate environmental and species data (Legendre & Legendre, 2003). The CCA was used to indicate the main environmental variables influencing each plant community type. The outputs show direct relationships between species and their environmental preferences. The method is subject to possible curvilinear distortion, but this was not observed for the results in this study. The CCA can be influenced by the presence of rare species, therefore a down-weighting of species was undertaken in order to dampen this effect.

All of the plant species data collected from the quadrats were estimates of percentage cover, so these proportional data need to be transformed in order to meet the assumptions of the CCA (to fit a normal distribution). These data were transformed using an Arcsine transformation, which is the most appropriate for proportional data particularly if values occur outside of the 30-70% range (Sokal & Rohlf, 1995; Waite, 2000).

Validation of the TIN, IDW and OK interpolations were performed using a cross validation and regression analysis. Cross validation involves removing each data point sequentially and interpolating the value from the surrounding points. This provides a measured and predicted value and using these two sets of values a validation of the accuracy of the model can be made. This is a standard method of validating interpolations in geostatistics (Hofstra *et al*., 2008).

The assessment of the static predictive plant community model (chapter 5) used a Fleiss's Kappa coefficient to assess the reliability of agreement between the six predicted plant communities and the six actual plant communities. This is a commonly used statistical test for assessing the ability of an ecological model to predict habitats or communities (Monserud & Leemans, 1991; Karl *et al*., 2000; Moisen & Frescino, 2001 Robertson *et al*., 2003).

3.3 Summary

This chapter has introduced the biogeography of the study location in Estonia, namely the study sites Tahu, Matsalu and Kudani. The Tahu site was used to investigate and quantify the relationship between plant community and elevation and to develop a static predictive plant community model. The Matsalu and Kudani sites were used to validate the ability of the static model developed at Tahu to predict plant community location in other Estonian coastal wetland sites. The models developed to predict present day plant community location were used as a basis for the development of a dynamic predictive plant community model to estimate the future location and extent of the plant communities with respect to changes in local sea level. The factors used to assess changes in local sea level were: isostatic uplift, four IPCC global sea level rise scenarios and one assuming no change, and sediment accretion rates assuming an increase in sediment accretion due to greater storminess and no change in sediment accretion rates (derived from the results of the study in chapter 6). This chapter has also introduced the key sampling, modelling and statistical protocols for the research.

4 Relationships between micro-topography, edaphic factors and plant communities in Estonian coastal wetlands

4.1 Preamble

A review of the literature (chapter 2) has identified a gap in the knowledge regarding a quantification of the environmental characteristics that determine the location and extent of the plant communities within Estonian coastal wetlands. This chapter provides an analysis of the relationships between micro-topography, edaphic factors and plant community type. A series of Mann-Whitney tests were used to assess and quantify the relationship between each individual environmental factor, in turn, and plant community type. Following this a Canonical Correspondence Analysis was used to elucidate the relationship between all environmental variables and the plant communities. The analysis provided an insight into the main environmental variables that determine plant community and provided a basis for the development of the static correlative plant community model developed in chapter 5.

4.2 Introduction

Baltic coastal wetlands are of international ecological importance as they support characteristic biological diversity (Joyce & Wade, 1998a; Benstead *et al.*, 1999). This includes a wide variety of breeding and migratory bird species as well as rare plants (Rannap *et al.,* 2004). The limited extent and distribution as well as their recent decline in area mean these areas are of high conservation concern (Leito *et al*., 2008). The Baltic coastal meadow areas within Baltic coastal wetlands have been identified as a priority habitat in the EU Habitat Directive (1992). They are located in Estonia and to a lesser extent in Sweden, Latvia and Finland and are typified by a flat, broad landscape, situated between the micro-tidal, brackish Baltic Sea and an extensive forest interior (Burnside *et al*., 2007). Due to the micro-tidal nature of the Baltic Sea, inundation is primarily caused by changing meteorological conditions and is also effected by storm surges and changes in barometric pressure (Tyler, 1971b). Many of these wetland landscapes have been managed for thousands of years via traditional farming practices, usually grazing by cattle, with a comparatively stable non-tidal hydrological system (Puurmann & Ratas 1998) which forms wet grasslands. Without management in the form of grazing by livestock, mowing and/or periodic burning to maintain their biodiversity, these wet grasslands revert to less diverse communities such as reed beds and scrub/woodland (Rannap *et al*., 2004; Burnside *et al*., 2007; Berg, 2009). Increases in agricultural intensification in some areas and abandonment in others have affected the distribution of these grasslands in recent years (Berg, 2009). In the 1950's coastal wet grasslands covered 28 750ha in Estonia. However, following the late 1970's land inventory, these grasslands had decreased to 9513ha. A more recent study in 2001 put this figure at 6188ha with many coastal wet grasslands considered to be in poor condition (EFN & RDSNC, 2001). Recent efforts have been made in some areas to reinstate management to improve coastal wet grasslands in poor condition, reflecting the conservation value of these areas, and several of these projects have met with success (Lotman, *pers*. *comm*. 2009).

Baltic coastal wetlands are typically comprised of a range of plant community types, which combine to form a mosaic. Previous studies have suggested that the mosaic is determined by hydrology, soil type and management (Rebassoo, 1975, Ребассоо, 1987, Paal, 1998, Burnside *et al*., 2007, Burnside *et al.,* 2008) although these authors have investigated each of these factors in isolation.

Wetland hydrology has previously been shown to be strongly influenced by micro-topographic variation (Vivian-Smith, 1997). Micro-topography is described as topographic variation on the scale of individual plants, ranging from scales of one centimetre to one metre (Moser *et al*., 2007). Several studies have shown that micro-topography affects plant communities (Prach, 1992; Vivian-Smith, 1997; Cardinale *et al*., 2002; Werner, 2002; Peach & Zedler, 2006), due to its relationship with a variety of factors such as moisture gradient (Prach, 1992), spatial variation and temporal fluctuations of water levels affecting depth and duration of flooding, soil nutrient availability, resource specialisation (Vivian-Smith, 1997), substrate variability (Cardinale *et al*., 2002), increased surface area and the availability of multiple micro habitats (Peach & Zedler, 2006). It is therefore hypothesised that small-scale topographic differences and related edaphic factors influence vegetation composition and distribution across coastal wetland landscapes.

The aims of this study were to examine the influence of micro-topography and selected edaphic factors on plant community type and quantify any relationships in an Estonian coastal wetland.

4.3 Study site

This study took place on the Tahu coastal wetland (figure 4.1) located in the Silma Nature Reserve. The Reserve covers an area of 4780 ha and landowners include both the state and private persons. Silma Nature Reserve was first designated as a protected area in 1998 in order to protect bird species from hunting. The study site comprises 115.6 ha on the west side of Tahu Bay (figure 4.1). At Tahu the Baltic has negligible tidal influence (0.02m) due to the shallow bathymetry and limited access to the ocean (Suursaar *et al*., 2007).

However, major fluctuations in sea level do occur, caused by changing meteorological conditions on a seasonal basis (Estonian Meteorological and Hydrological Institute, 2011). In addition sea level is also affected by storm surges and changes in barometric pressure. During periods of high water much of the wetland is inundated by brackish water (~5ppt) from the Baltic and in areas where overland flow takes place sediment accretion occurs. At Tahu, sea level is typically at its lowest in spring and early summer when easterly winds prevail (Estonian Meteorological and Hydrological Institute, 2011). In June there is usually a rise in sea level and periodic inundation of more elevated coastal wetland environments (Estonian Meteorological and Hydrological Institute, 2011). During the winter period ice covers the sea in most years and coastal wetlands are covered with snow for a minimum of two months. The Tahu wetland is low lying with heights not exceeding 0.87 m above mean sea level (m.s.l.) and is currently rising at a rate of 2.8mm/year due to isostatic uplift (Vallner *et al.*, 1988).

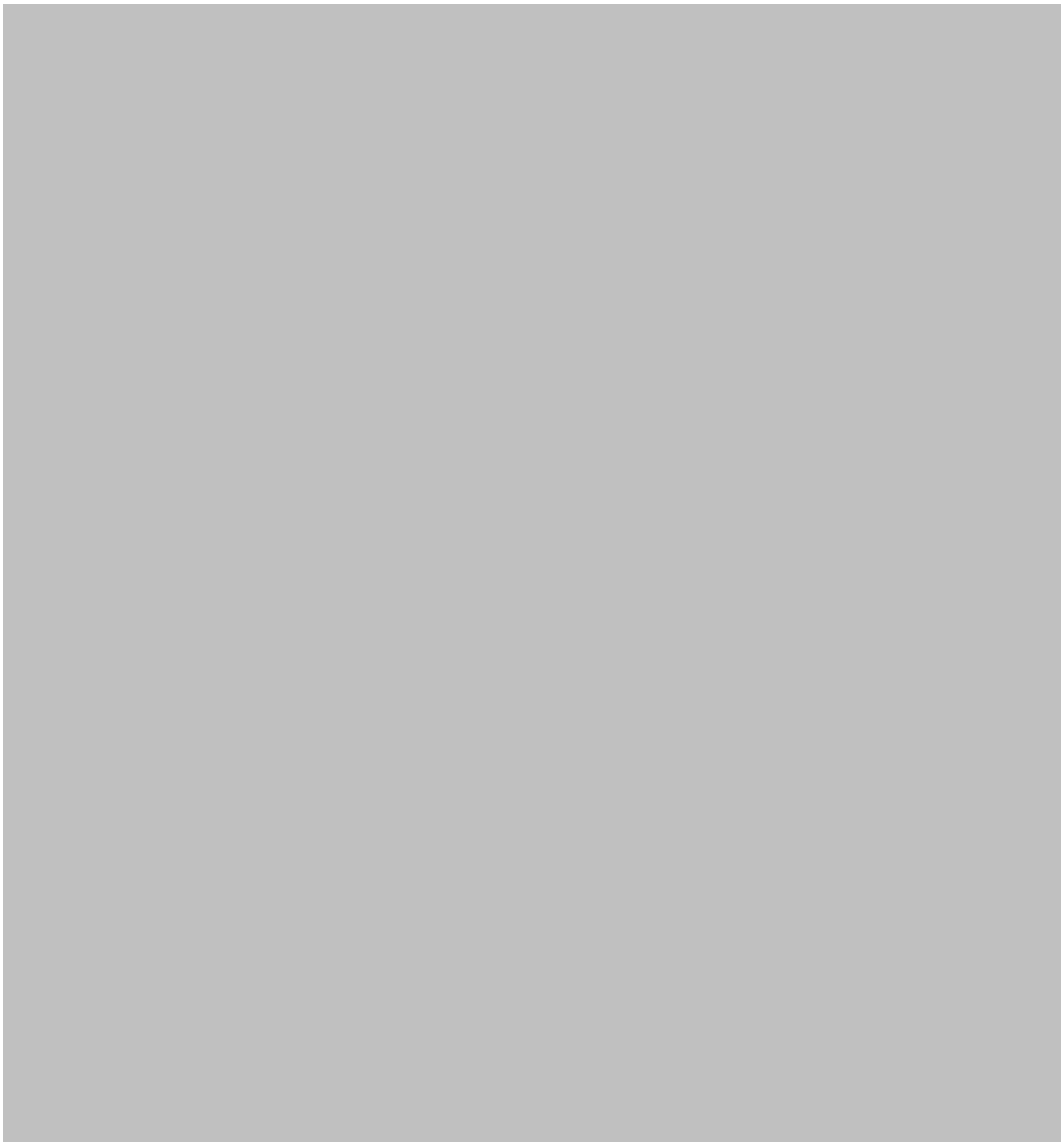

Figure 4.1: Tahu study site located in North West Estonia in the region of Läänemaa County (Eesti Maa-amet, 2010)

The southern half of the site had been abandoned for 25 years before the reinstatement of management in 2002 following the EU Life project "Restoration of Habitats of Endangered Species in Silma Nature Reserve", while the northern half has been under regular management for over 100 years. Both North and South Tahu have been managed through low intensity grazing by cattle and horses between May and September since 2002 (1 livestock unit per hectare). The reinstatement of management has reverted the site to a typical mosaic Estonian coastal wetland. Previously, following abandonment, south Tahu was covered by scrub and developing woodland in the higher elevation areas and reed swamp in the lower areas. During the study the whole site supported a mosaic of Estonian coastal wetland plant communities.

4.4 Methods

4.4.1 Field methods

The study took place in July 2009. Individual plant community patches were identified using the indicator species from the Burnside *et al*. (2007) phytosociological key. A range of community patches were identified by a walkover survey of the whole wetland. In total seven community types were identified: CS (Clubrush Swamp), RS (Reed Swamp), LS (Lower Shore), US (Upper Shore), OP (Open Pioneer), TG (Tall Grassland) and SW (Scrub and developing Woodland), across 15 patches per plant community, and the full species composition and indicator species for each plant community are shown in table 3.3. Fifteen quadrats per community type were then located using a stratified random approach. Quadrats were placed in randomly selected patches of each of the seven community types throughout the coastal wetland, giving a total of 105. 1m^2 quadrats were used (Bullock, 1996; Critchley *et al.,* 2002; Burnside *et al*., 2008) and within the quadrats plant species presence and abundance was recorded. Abundance was recorded using the Domin scale (Rodwell, 1992).

A real time kinematic differential Global Positioning System (dGPS) (Leica GPS1200 Surveying System) was used to record a series of ground positions and elevation values (estimated vertical accuracy <0.02cm) within sample quadrats. The dGPS elevation data were used to represent micro-topography. Four corner points were recorded along with a further 16 randomly selected points within each quadrat, giving a total of 2100 elevation data points. Random selection of the points within the quadrat was achieved by a closed eye method from a random standing point around the quadrat.

Two dipwells were placed within each plant community in the north of the Tahu site (figure 4.2) and the water table depth was measured manually each day between 10:00 and 14:00 for thirty seven days. The elevation of the soil surface at the location of each dipwell was recorded using a dGPS with an estimated vertical accuracy of <0.02m with relation to m.s.l. (BK77 as measured at Kronstadt) (Table 4.1).

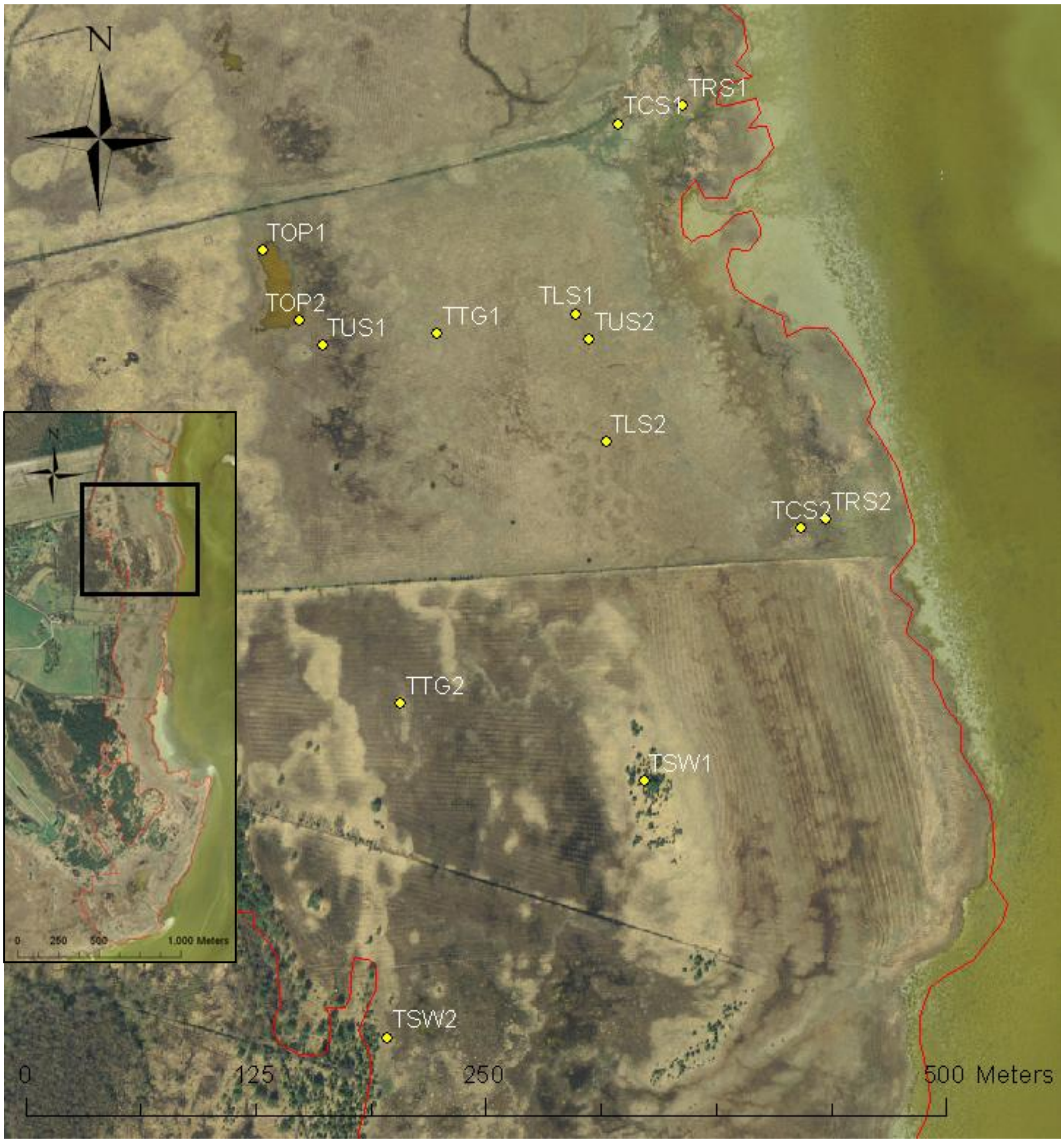

Figure 4.2: Location of the dipwells at Tahu. Dipwells are named by the plant community in which they are located and randomly allocated a number, either 1 or 2.

Eight edaphic parameters were recorded in each quadrat: moisture, organic matter content, nitrogen (N), phosphorus (P), potassium (K), salinity, pH and particle size. Soil moisture was recorded using a Delta T WET Sensor device and 3 readings per quadrat were taken over a single four hour period with very stable weather and no precipitation. The other soil parameters were recorded via soil samples. A total of one hundred and five soil samples (100g) were collected (one per quadrat) for analysis of soil organic matter

(Loss on Ignition), nitrogen (N) (Kjeldahl method), phosphorus (P) (Ammonium lactate extractable by flow injection analysis), potassium (K) (Flame photometric method), salinity (Super saturated solution), pH (Super saturated solution) and sediment particle size in the laboratory.

Table 4.1: The elevation of each dipwell at Tahu above mean sea level as measured at Kronstadt (BK77). Dipwells are named by the plant community in which they are located.

Dipwell	Elevation (m)
CS1	-0.15
CS2	-0.06
RS1	-0.02
RS2	0.10
LS1	0.13
LS2	0.16
US1	0.47
US2	0.24
OP1	0.31
OP2	0.31
TG1	0.41
TG2	0.54
SW1	0.68
SW2	0.60

4.4.2 Laboratory methods

Soil samples were initially dried for 24 hours to constant weight at 40°C in an oven. The samples were separated into two equal amounts. One half was tested for total N, P, K and soil organic matter, whilst the other was tested for sediment particle size, pH and salinity.

The methodology used for determining total kjeldahl soil Nitrogen was the block digestion with copper catalyst and steam distillation into boric acid method (Tecator ASN 3313 AOAC, 2001). Available phosphorus (P) was determined using the Determination of Available Phosphorus in Soil by Flow Injection Analysis method (ammonium lactate extractable, Egner-Riehmi method) (Tecator ASTN 9/84). Total available Potassium (K) was determined using the flame photometric method (956.01) (AOAC, 1990). Soil organic matter content was calculated using the loss on ignition method as per Schulte *et al.* (1987).

Soil pH and conductivity were measured using a WTW pH/conductivity 340i device in the laboratory. The device was calibrated prior to sampling each day using standard pH and conductivity solutions. Initially, each soil sample was weighed into 5g fractions and then put into a supersaturated state with distilled water, at a ratio of 1:2. Each sample was then stirred, in a temperature controlled agitator at 24°C, for 1 hour and left to settle for 15 minutes. In between each sample, the pH and conductivity devices were washed off with distilled water, dried, and each sample was triple tested and an average figure calculated.

Sediment particle size was calculated using a Malvern laser particle size analyser. Particle size fractions were sorted according to the Wentworth scale 2mm-1mm, 1mm-500µm, 500-250µm, 250-125µm, 125-62.5µm, 62.5-3.9µm and <3.9µm (Wentworth, 1922).

4.4.3 Statistical methods

Potential evapotranspiration rates were calculated using the Thornthwaite equation (Shenbin *et al*., 2006). The edaphic and elevation data were analysed using a Kruskal-Wallis test (Minitab 15, 2009) to identify if edaphic factors differed significantly between each community type. For those environmental factors where a significant difference was identified a series of Mann-Whitney tests were performed, in order to identify which factors differed between community types. To address the issue of Type I multiple comparison errors, a Bonferroni correction was used (Sokal & Rohlf, 1981; Waite, 2000). A Canonical Correspondence Analysis (CCA) (Legendre and Legendre 2003) was used to evaluate the relationship between species and community data and the associated environmental matrix. The CCA was conducted in MVSP 3.1 and for the subsequent Pearson correlation test all significance values were set at 95%. The CCA incorporated the species data (arcsine transformed) (Sokal & Rohlf, 1981) and all the edaphic and micro-topography data for each quadrat grouped by plant community. CCA scores of each axis were tested for correlations (Pearson's correlation coefficient) with edaphic variables and micro-topography.

4.5 Results

4.5.1 Plant community environmental characteristics

Analysis showed that the CS plant community occurred at the lowest elevation (median 0.01m) and was typified by high soil moisture (median 59.7%), salinity, (median 1981 µS/cm), intermediate soil organic matter (median 16.86%), N (median 0.9%), and K (median 227.2mg/kg), and the lowest soil pH (median 5.05) and P (median 19mg/kg).

The RS community also occurred at a low elevation (median 0.11m) and had the highest soil moisture and salinity values, with medians of 63.3% and 2145 µS/cm respectively, although these were not significantly different to CS or LS. The RS community was also found to have intermediate values for soil organic matter (median 25.98%), N median 0.96%), and K (median 389mg/kg), and an equally low pH as CS (median 5.05).

The LS community occurred at a slightly higher elevation than CS and RS (median 0.30m) and was found to have high soil moisture (median 56.6%) and intermediate soil salinity (median 1175 µS/cm), organic matter (median 15.01%), N (median 0.81%), K (median 323.8mg/kg), and pH (median 5.89).

The US plant community occurred at a higher elevation than LS (median 0.34m), and had higher soil salinity (median 1709 µS/cm), organic matter (median 23.05%), N (median 1.07%), K (median 417.2 mg/kg), and pH (median 6.01) than LS.

The OP plant community occurred at a median elevation of 0.38m and had the lowest soil moisture (median 17.1%), organic matter (median 0.96%), N (median 0.04%), K (median 123.9 mg/kg), and P (median 18.6 mg/kg). OP also had high soil salinity (median 1959 µS/cm) and the highest pH (median 6.89).

The TG and SW plant communities occurred at the highest elevation, with medians of 0.57m and 0.62m respectively, and were found to have the lowest soil salinity, with medians of 310.0 µS/cm and 296.5 µS/cm respectively. Both TG and SW contained the highest soil organic matter (medians 39.31% and 35.99% respectively), N (medians 1.62% and 1.42% respectively), and K (medians 477.2 and 380.9 respectively), and exhibited intermediate pH, with medians of 6.16 and 6.25 respectively. The TG plant community had high soil moisture (median 52.0%), whereas SW exhibited low soil moisture (median 21.0%). The data for all plant communities are summarised in Table 4.2.

4.5.2 Relationship between sea level and water level within plant communities

Water table depth was lowest in the TG (median -0.13m, minimum +0.07m, maximum -0.33m) and SW (median -0.20m, minimum +0.01m, maximum -0.46m) communities followed by the US (median -0.01m, minimum +0.17m, maximum -0.27m) and LS (median +0.08m, minimum +0.28m, maximum -0.12m) communities sequentially (figure 4.3). The highest water levels were found in the CS (median +0.10m, minimum +0.40m, maximum -0.20m) and RS (median +0.11m, minimum +0.37m, maximum -0.19m) plant communities, typically above the soil surface (figure 4.3). OP was the only plant community where water level did not greatly fluctuate and was consistently at or above the soil surface (figure 4.3) and above sea level (median +0.08m, minimum +0.13m, maximum 0.00m) (figure 4.3). The data in figure 4.3 show the water levels in relation to sea level. Both of the CS dipwells were located below m.s.l. and hence the majority of results reflect local sea level at the time recorded. Figure 4.4 shows that, over the short period of this study, the changes in sea level were reflected in the plant community water levels with the exception of OP. It is important to note however, that there is a considerable time lag in water level in the OP community, such that not all the variation that occurred in the other plant communities was reflected in OP (figure 4.4). The water levels were much lower in the higher elevation plant communities TG and SW than in the CS and RS plant communities (figure 4.4). However, the data in figure 4.4 also show that the water level in the higher elevation plant communities was considerably higher than sea level as measured at Rohuküla (figure 4.4).

Further to this the data presented in figure 4.4 show that over the time period of this study some of the variation in sea level can be explained by fluctuations in atmospheric pressure. During this same period there was only one precipitation event on the last day of the study (figure 4.4). Figure 4.5 shows monthly potential evapotranspiration (PET) for the study area as calculated using the Thornthwaite equation. During the period of study in June and July PET was 72mm and 109mm respectively with an annual PET of 447mm (figure 4.5).

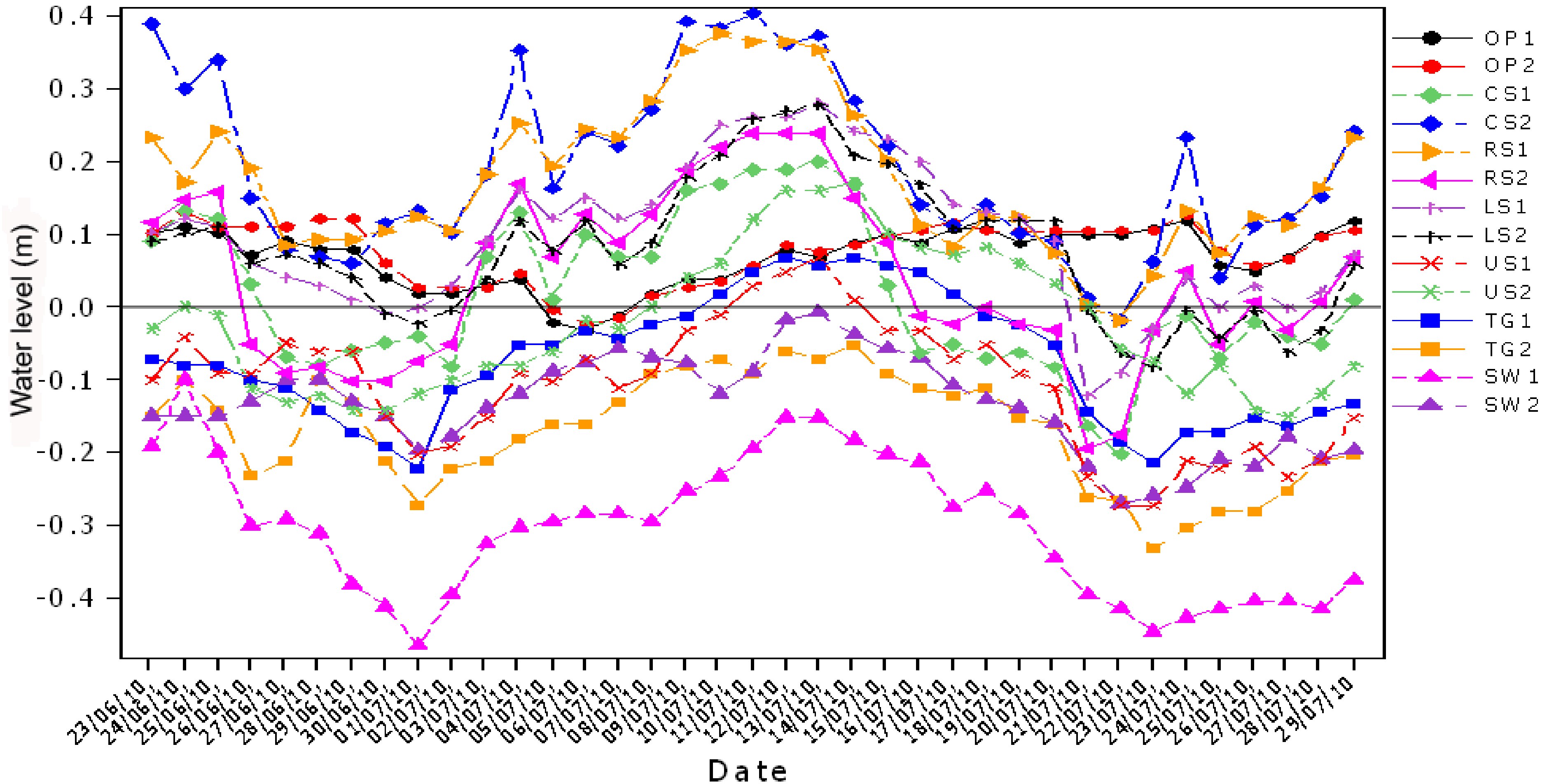

Figure 4.3: Water level (m) at the Tahu site for seven plant communities over a 37 day period in summer 2010. Plant communities are: OP = Open Pioneer, CS = Clubrush Swamp, RS = Reed Swamp, LS = Lower Shore, US = Upper Shore, TG = Tall Grass, and SW = Scrub and developing Woodland. Two dipwells were placed within each plant community and arbitrarily labelled 1 or 2. The black line denotes sea level.

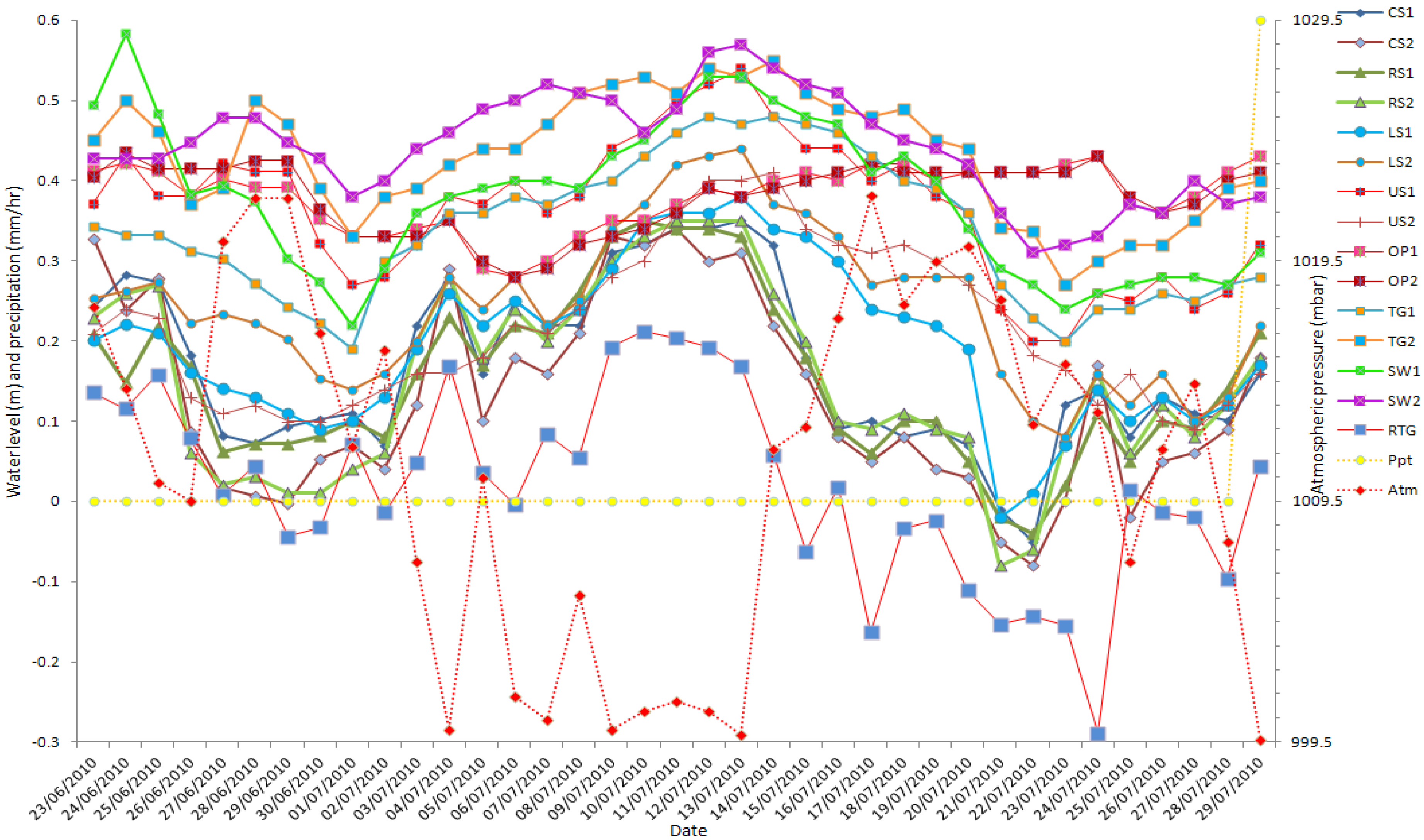

Figure 4.4: Level of the water table in comparison to sea level (m) at the Tahu site for seven plant communities over a 37 day period in summer 2010. RTG denotes sea level data, Atm denotes atmospheric pressure data and Ppt precipitation data from the nearest tide gauge and weather station at Rohuküla (14km from the site). Plant communities are: OP = Open Pioneer, CS = Clubrush Swamp, RS = Reed Swamp, LS = Lower Shore, US = Upper Shore, TG = Tall Grass, and SW = Scrub and developing Woodland. Two dipwells were placed within each plant community and arbitrarily labelled 1 or 2.

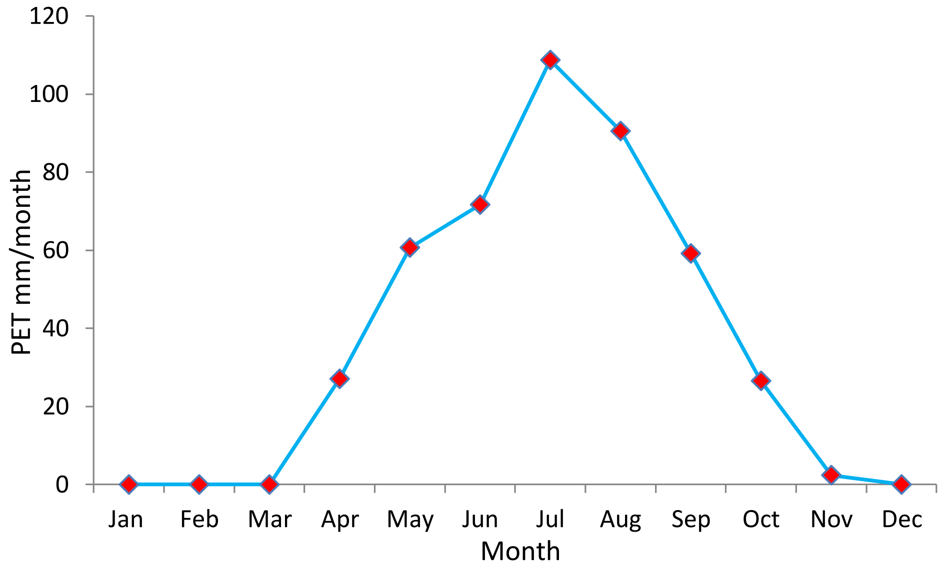

Figure 4.5: Potential evapotranspiration rates in the study area during 2010. Rates were calculated using the Thornthwaite equation.

4.5.3 Relationship between elevation and plant community

Analysis of plant community data and data obtained using the dGPS showed that the plant communities exhibited an elevation gradient (figure 4.6). The CS plant community was found at the lowest elevation (median 0.01m), followed by RS (median 0.11), LS (median 0.30m), US (median 0.34m), OP (median 0.38m), TG (median 0.57m), and SW (median 0.62m) (Table 4.2).

The Kruskal Wallis test identified that at least one of the plant communities was found to occur at a different elevation to at least one other plant community ($H = 1326$, $DF = 6$ $p < 0.01$). In a series of 21 Mann Whitney tests nearly all of the plant communities were found to occur at significantly different elevations from each other ($p < 0.0001$). The exceptions to this were OP and TG where there was some overlap in elevation ranges ($W = 85002$, $p = 0.321$) (figure 4.6). However, OP and TG were found to occur at significantly different elevations to the other plant communities.

There was some overlap in the interquartile ranges of elevation between the proximal LS - US and US – OP plant communities (Table 4.2). Mann Whitney analysis showed that the LS - US and US - OP plant communities occurred at significantly different elevations and yet the median elevation difference between the LS - US and US - OP was only 0.04m (figure 4.6). Additionally there was some overlap between TG and LS, US, OP, and SW due to the large range at which TG occurred (figure 4.6; Table 4.2). Some communities were observed to have very restricted elevation ranges, such as CS, and the data suggest that there is no overlap in the elevation interquartile ranges of the proximal plant communities CS – RS, and RS – LS (figure 4.6; Table 4.2).

Table 4.2: Summary of elevation and edaphic variable data from Tahu (excluding particle size). Data include the lower quartile (Q1), median and upper quartile (Q3) values for each of the environmental variables.

Variable	Statistic	CS	RS	LS	US	OP	TG	SW
Elevation	Q1	-0.029	0.067	0.241	0.266	0.349	0.317	0.56
(m)	Median	0.011	0.106	0.299	0.339	0.378	0.569	0.62
	Q3	0.037	0.147	0.351	0.467	0.488	0.698	0.682
Moisture	Q1	56.6	59.7	54.6	40.2	11.3	42.8	17.9
(%)	Median	59.7	63.3	59.5	46.4	17.1	49	20.5
	Q3	62.9	65.5	63.2	49	26.9	53.6	28.8
Salinity	Q1	1194	1887	966	1347	840	229	217
(µS/cm)	Median	1981	2145	1175	1709	1959	310	296.5
	Q3	2250	2260	1824	2270	3640	1252	438.5
Organic	Q1	3.21	16.31	6.41	20.29	0.63	32.28	29.29
matter	Median	16.86	20.37	15.01	23.05	0.96	39.31	35.99
(%)	Q3	20.89	25.98	18.85	29.78	1.38	48.5	41.04
N	Q1	0.17	0.76	0.31	0.89	0.02	1.37	1.27
(%)	Median	0.9	0.96	0.81	1.07	0.04	1.62	1.42
	Q3	1.05	1.13	0.94	1.28	0.07	2.19	1.83
K	Q1	156.5	319.4	203.4	329.2	87.5	357	341.2
(mg/kg)	Median	227.2	389	323.8	417.2	123.9	477.2	380.9
	Q3	327.6	483.8	405.6	468.6	140.9	594.6	510.4
P	Q1	12.5	23.9	22.4	19.5	6.9	8.2	22.9
(mg/kg)	Median	19	33.5	31.7	28.6	18.6	23.7	36.9
	Q3	25.7	42.8	40.6	36.6	55.8	36.8	63.5
pH	Q1	4.3	4.69	5.83	5.58	6.49	5.57	5.88
	Median	5.05	5.05	5.89	6.01	6.89	6.16	6.25
	Q3	5.69	5.3	6.36	6.23	7.41	6.33	6.38

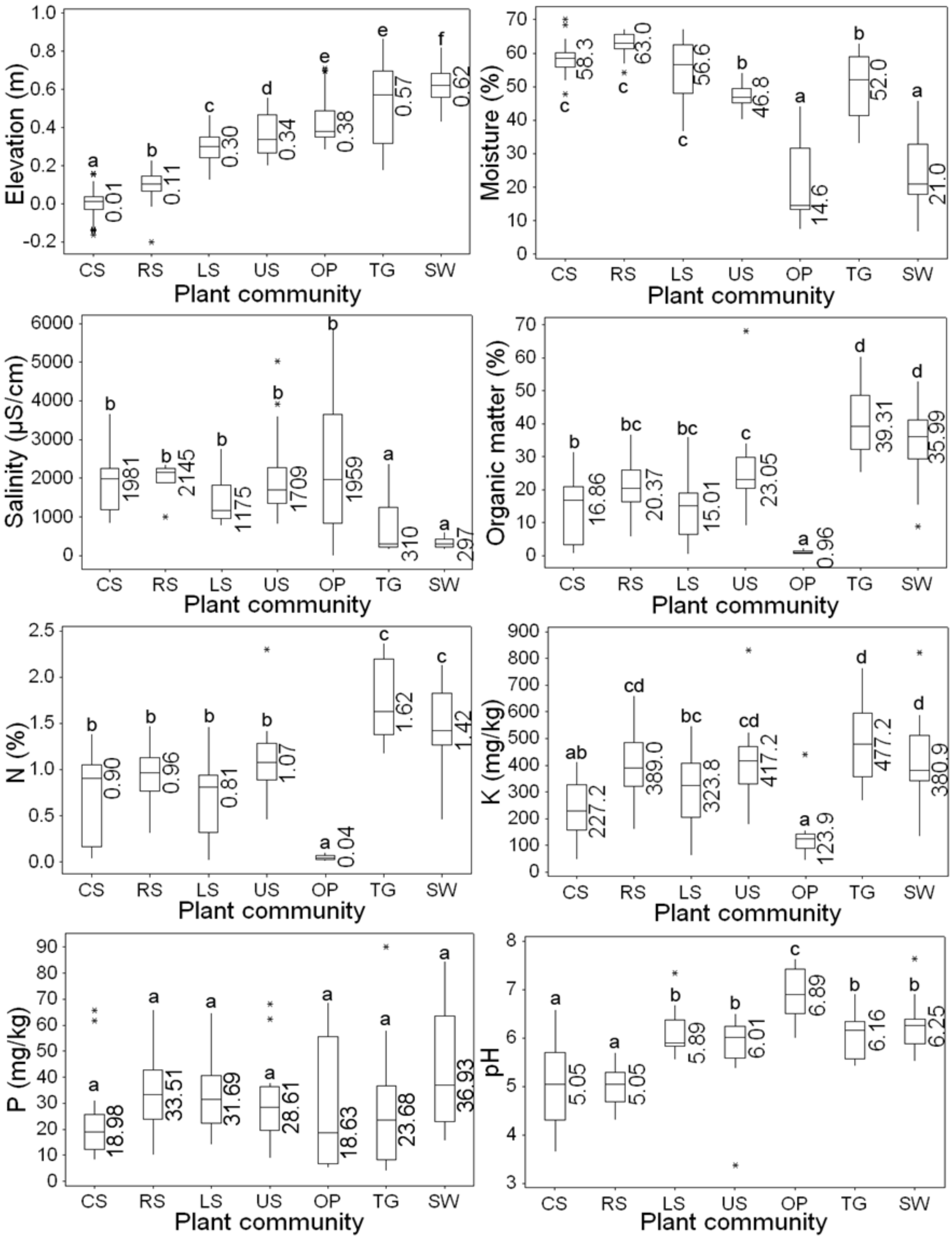

Fig. 4.6: Median values and interquartile ranges of measured edaphic factors (moisture, salinity, organic matter, N, K, P, and pH) and elevation above Baltic m.s.l., as measured at Kronstadt, 60º00'N; 29º44'E, of the seven plant community types. The boxes show the interquartile ranges and median values are situated next to the appropriate boxes. Asterisks represent outliers. Boxes with different letters show statistically different communities in Mann-Whitney tests ($p < 0.05$). For an explanation of plant community codes see table 3.3 in chapter 3.

4.5.4 Relationship between soil organic matter and plant community

The results of a Kruskal-Wallis test showed that soil organic matter was significantly different in at least one plant community ($H = 68.3$, $DF = 6$, $p < 0.01$). A series of Mann-Whitney tests showed that the OP community had significantly lower soil organic matter content to other plant communities with a median percentage of 0.96% and very little variation. The CS, RS and LS communities were not found to have significantly different percentages of soil organic matter from each other but formed a group (figure 4.6 group b) that significantly differed from the other communities, with the exception of US and LS which were not significantly different from RS (figure 4.6 group c). After OP, the CS, RS, and LS plant communities had the next lowest soil organic matter contents at 16.86%, 20.37% and 15.01% median values respectively. The US community, at 23.05% (median), had a significantly different percentage of soil organic matter from all the other communities except RS. The TG and SW communities had significantly higher soil organic matter from all the other communities but were not significantly different from each other with 39.31% and 35.99% median values respectively. The plant communities therefore formed two distinct groups by soil organic matter consisting of TG and SW (group d), with high soil organic matter and one other group split into two overlapping sub groups consisting of CS, RS and LS in the first sub group and RS and US in the second sub group (figure 4.6 groups b and c).

4.5.5 Relationship between soil Nitrogen and plant community

The results of the Kruskal Wallis test showed that at least one of the plant communities contained significantly different soil N than at least one other ($H = 67.06$, $DF = 6$, $p < 0.01$). Figure 4.6 identifies three significantly different community groups, as identified in a series of Mann Whitney tests, in relation to soil N. The lowest soil N was found in the OP community with a median value of 0.039%, the next lowest was in the group containing the CS, RS, LS and US communities with median values of 0.903%, 0.964%, 0.807% and 1.072% respectively, and the highest soil N value group consists of the TG and SW communities with median values of 1.623% and 1.419% respectively.

4.5.6 Relationship between soil Potassium and plant community

A Kruskal Wallis test identified that there was a significant difference in soil potassium between at least one plant community and one other ($H = 67.06$, $DF = 6$, $p < 0.0001$). In a series of Mann Whitney tests the soil K values of the CS (median 227.2mg/kg) community did not significantly differ from the RS, LS and OP communities (median 389.0mg/kg, 323.8mg/kg, and 123.9mg/kg respectively) and contained the lowest levels of soil K. The RS and LS communities only contained significantly different soil K values to the OP community. The US, TG and SW communities (median 417.2mg/kg, 477.2mg/kg, and 380.9mg/kg respectively) did not show significantly different soil K levels from each other or the RS and LS communities. The OP community had significantly less soil K than all the other communities except the CS community (figure 4.6).

4.5.7 Relationship between soil Phosphorus and plant community

The result of the Kruskal Wallis test ($H = 11.36$, $DF = 6$, $p = 0.078$) showed that there was no significant difference between the soil phosphorus values of the communities (figure 4.6).

4.5.8 Relationship between soil salinity and plant community

The Kruskal Wallis test showed that there was a significant difference between at least one of the communities ($H=49.52$, $DF = 6$, $p < 0.0001$). The Mann Whitney tests showed that the salinity values for the CS, RS, LS, US and OP communities were not significantly different from each other but were significantly different to the TG and SW communities, which themselves were not significantly different from each other. This resulted in two distinct groups: a higher salinity group consisting of CS, RS, LS, US and OP with median values between 1175 µS/cm and 1980.5 µS/cm; and a lower salinity group consisting of the TG and SW communities with between 296.5 and 310 µS/cm (figure 4.6; Table 4.2).

4.5.9 Relationship between soil moisture and plant community

A Kruskal Wallis test identified a significant difference between the soil moisture values of at least one plant community and one other ($H = 79.00$, $DF = 6$, $p < 0.0001$). The results of Mann Whitney tests showed that CS (median 59.7%), RS (median 63.3%) and LS (median 59.5%) showed significantly different soil moisture values to all the other communities but were not significantly different from each other (figure 4.6, group c). The TG (median 49.0%) and US (median 46.4%) communities did not have significantly different soil moisture values from each other (group b) but did have significantly different values from the other communities. Similarly the OP (median 17.1%) and SW (median 20.5%) communities did not have significantly different soil moisture values from each other (group a) but did have significantly different values from all other communities. The highest soil moisture occurred in the CS, RS and LS communities (figure 4.6), followed by the US and TG plant communities (figure 4.6). The lowest soil moisture values were found in the OP and SW communities (figure 4.6).

4.5.10 Relationship between soil pH and plant community

The result of the Kruskal Wallis test showed that there was a significant difference in soil pH between at least one of the communities and one other ($H = 57.87$, $DF = 6$, $p < 0.0001$). The results of the Mann Whitney tests showed that CS and RS had significantly different pH values from the other communities but were not significantly different from each other. CS and RS (group a) had the lowest pH, both having a median pH of 5.05 but with CS having a greater range of values (figure 4.6). The LS, US, TG and SW communities (medians 5.89, 6.01, 6.33, and 6.38 respectively, group b) were intermediate and did not have significantly different pH values from each other but had significantly different pH values from the CS, RS and OP communities. The OP community had significantly higher pH values (median pH of 6.89, c) at almost pH neutral.

4.5.11 Relationship between sediment particle size and plant community

The OP community type was associated with a much greater proportion of larger sediment particles than all of the other community types and had the highest proportion of the sediment fraction larger than 1mm. This community type was also underlain by scattered boulders and pebbles in the majority of the OP patches, which were not observed in the other community types. The TG and SW community types showed higher median fractions of medium sand (500-250µm) than the CS, RS, LS and US community types, and lower silt and clay fractions (62.5-3.9µm and <3.9µm) (table 4.3). CS, RS, LS and US had very similar percentages for each of the particle size fractions.

Table 4.3: Median percentages of each fraction of the sediments for each plant community type. Sediment fractions are according to the Wentworth scale (Wentworth, 1922).

Community type	>1mm	>500µm	>250µm	>125µm	>62.5µm	>3.9µm	<3.9µm
CS	0.8	1.6	13	12.5	11.6	26.8	33.3
RS	2.2	2.2	15.8	12.2	11.3	25.1	32.4
LS	0.7	1.9	11.5	14.5	15.3	24.7	31.7
US	0	1.9	15.5	16.5	24.8	23.9	17.3
OP	24.9	12.3	24	15.6	13.6	7.8	1.9
TG	9.8	11.7	23.2	18.9	16.1	12.1	7.8
SW	2.4	8.4	24.8	22.6	19.4	12.9	9.7

4.5.12 Relationships between plant community and environmental variables

Following the analysis of the environmental and plant community data individually a Canonical Correspondence Analysis (CCA) was used to describe the relationship between all of the environmental variables (elevation, soil moisture, organic matter, N, P, K, salinity, pH and particle size) and the plant communities to assess any trends. The results of the CCA showed that the plant communities were able to be separated using elevation and edaphic variables (figure 4.7).

Axis 1 of the CCA accounted for 52.8% of the variation in the data (eigenvalue = 0.566). Pearson correlation coefficients showed that axis 1 was significantly correlated with seven factors (table 4.4): soil organic matter, N, salinity, K, elevation above sea level, soil moisture and pH. Further associations were also evident with Axis 2, which accounted for 23.3% of the variation (eigenvalue = 0.250), and was significantly correlated with soil moisture, pH, particle size, N, K, organic matter and elevation above sea level (table 4.4). Edaphic variables and elevation above sea level have a strong influence on community composition, as can be seen from the significant species-environment correlations shown in table 4.4.

Table 4.4: Pearson correlation coefficients for all environmental variables, axes eigenvalues, percentage of axes explanation and species-environment correlations for the indicator species, community and environmental variables for the Canonical Correspondence Analysis shown in figure 4.7.

Variables	Axis 1	Axis 2
Soil Nitrogen %	0.726***	-0.489***
Soil Phosphorus mg/kg	0.099	-0.009
Soil Potassium mg/kg	0.474***	-0.447***
Soil organic matter %	0.747***	-0.417***
Soil pH	0.321***	0.739***
Soil salinity µs/cm	-0.673***	0.042
Elevation (m)	0.411***	0.195**
Soil moisture %	-0.339***	-0.811***
Sediment particle size (mean value)	-0.16	0.689***
Eigenvalues	0.566	0.250
Percentage variation	52.8	23.3
Species-environment correlations	0.898***	0.905***

** = $p < 0.01$, *** = $p < 0.001$

Each of the plant community types was separated from the others in the CCA (figure 4.7) with the exception of CS and RS where there was some similarity in the micro-topographic and edaphic factors that affect these two plant communities (figure 4.7). These two swamp community types have a strong relationship evident in the field where both sets of indicator species can coexist (Ward, *pers. obs.*). Both of these community types have high soil moisture and salinity and a low elevation above sea level, as well as low soil pH, organic matter content, N, K and P. RS has slightly higher soil moisture values than the CS community type, which accounts for its slightly higher values along axis 2.

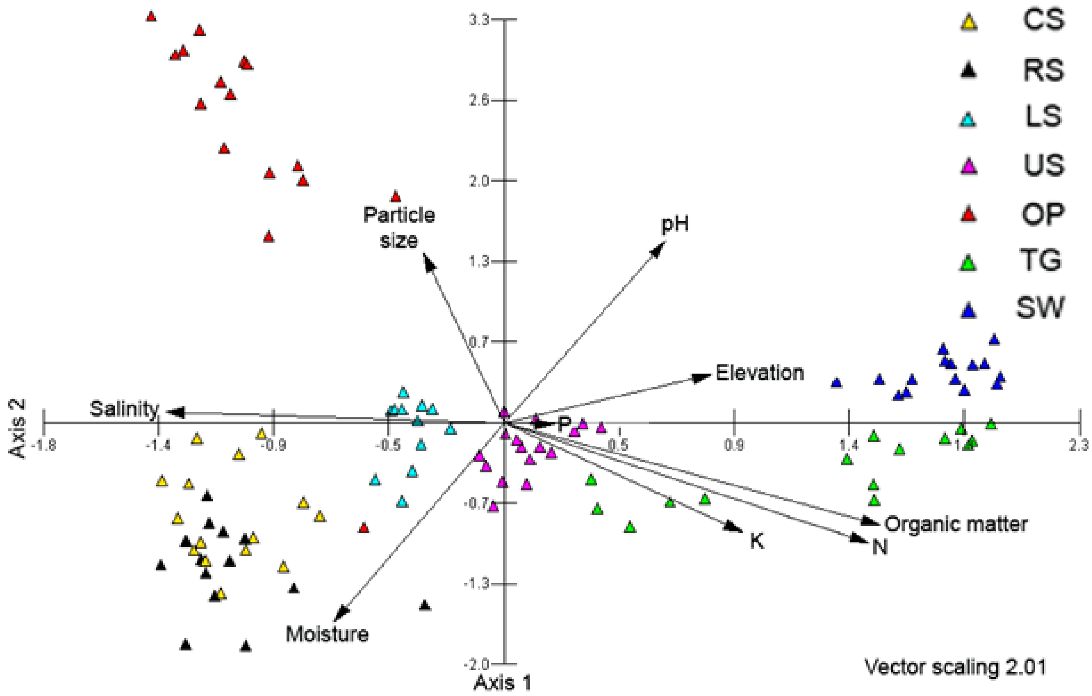

Fig. 4.7: Community ordination derived from Canonical Correspondence Analysis (CCA), showing plant communities against environmental vectors. Plant species data were Arcsine transformed and species with less than 5% cover were down weighted. The CCA was performed in MVSP 3.1 using all the plant species data grouped by plant community and all the environmental variables.

The LS community type is typified by high soil moisture (albeit lower than CS and RS) but low soil organic matter content, N and K. It also has lower soil salinity than the CS and RS community types, which explains, in conjunction with its greater elevation above sea level, its higher scores along axis 1 in figure 4.7.

The US community type sits centrally within the ordination on both axis 1 and 2, and exhibits intermediate values for most parameters (figure 4.7). The OP community type is very distinct from all the other community types along both axes (figure 4.7). The ordination shows that particle size is the main descriptor for the OP community (figure 4.7). It has very low soil organic matter content and N, and high soil salinity, which explain the high negative values along axis 1 (figure 4.7). The OP community also has very low soil moisture content and high pH.

The SW and TG plant communities are well separated from other communities along axis 1, and are strongly associated with high values for elevation above sea level, soil organic matter, N and K and low salinity values (figure 4.7; table 4.4).

4.5.13 Correlations between environmental variables

The environmental variables were tested using a Pearson correlation coefficient to examine any correlations between them. The results of this test are shown in table 4.4.

Soil Phosphorus was not significantly correlated with any of the other environmental values. Soil N, K, organic matter content and particle size were strongly positively correlated with each other ($p < 0.01$) with Pearson correlation coefficients of 0.980 for N and organic matter, 0.776 for K and organic matter and 0.767 for N and K (table 4.4). These are the most strongly correlated of all the environmental variables. Sediment particle size was slightly less strongly correlated with coefficients of 0.592, 0.575 and 0.449 for soil N, organic matter and K respectively. Elevation above sea level was positively correlated ($p < 0.01$) with K (Pearson coefficient 0.438), organic matter (Pearson coefficient 0.393) and N (Pearson coefficient 0.375) and negatively correlated ($p < 0.05$) with soil moisture (-0.345).
Salinity, pH and moisture were strongly correlated with each other ($p < 0.01$) and the correlation between pH and salinity (-0.467) and moisture (-0.363) was negative. Salinity and moisture were positively correlated ($p < 0.01$) with a coefficient of 0.418.

Table 4.5: Pearson correlation coefficients for each of the environmental variables. * = $p < 0.05$, ** = $p < 0.01$, *** = $p < 0.001$. All p values are shown in red and have been adjusted using a Bonferroni correction.

Environmental variable	N	P	K	Org. matter	pH	Salinity	Elevation	Moisture	Particle size
N		1.000	0.000	0.000	1.000	0.684	0.000	0.216	0.000
P	0.146		1.000	1.000	1.000	1.000	0.540	1.000	1.000
K	0.767***	0.218		0.000	1.000	1.000	0.000	1.000	0.000
Organic matter	0.980***	0.146	0.776***		1.000	0.216	0.000	0.108	0.000
pH	0.172	0.118	0.068	0.215		0.000	1.000	0.000	1.000
Salinity	-0.248	-0.121	-0.012	-0.287	-0.467***		1.000	0.000	0.324
Elevation	0.375***	0.256	0.438***	0.393***	0.201	-0.226		0.036	0.360
Moisture	-0.286	-0.162	-0.195	-0.308	-0.363***	0.418***	-0.345**		1.000
Particle size	0.592***	0.084	0.449***	0.575***	0.142	-0.275	0.270	-0.226	

Figure 4.8 is a diagrammatic representation of the correlations between all of the studied environmental variables recorded in the field based on the correlations shown in table 4.5.

Phosphorus was not correlated with any of the other environmental variables, and did not significantly vary between any of the communities so can be discounted from further study. The groups of correlated environmental variables in the red and purple boxes had a core set of N, K and organic matter, i.e. those that all correlated with each other. The environmental variables portrayed in a red box also contains Particle size (figure 4.8) and the environmental variables portrayed in a purple box also contains elevation above m.s.l. (figure 4.8). The group in the green box (figure 4.8) contains the variables Salinity, pH and moisture; this grouping is not correlated with the N, K, and organic matter set. The representation of environmental variables portrayed in figure 4.8 suggests that elevation along with soil moisture is the link between all correlated variables.

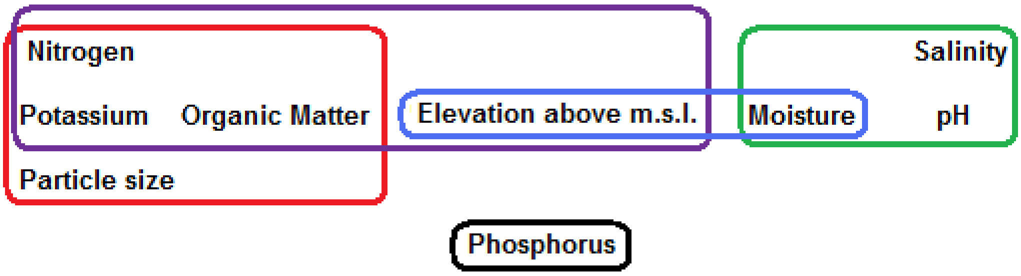

Figure 4.8: A representation of between variable correlation showing the environmental variables that are correlated with at least 95% confidence from the data in table 4.5.

4.6 Discussion

4.6.1 Environmental factors

The analysis in chapter 4 has assessed the relationship between environmental variables and plant community at an Estonian coastal wetland, Tahu. The methodology used in this study has produced data which provide a valuable insight into the factors that most strongly influence plant community location and extent. Traditional field approaches struggle to achieve this due to reduced accuracy and/or time constraints. A study by Vivian-Smith (1997) investigated the effects of micro-topography on plant community type using laboratory experiments to control substrate and water level heights. Whilst the method provided accurate data regarding these two controlled environmental factors in isolation it is unlikely to capture the more complicated relationships which exist between the plant communities and environmental variables in the field. Moser *et al*. (2007) made an attempt to assess the effects of micro-topography and hydrology on vegetation in the field. However, perhaps due to methodological restrictions, the study recorded vegetation, elevation and hydrology at one location in each of the four studied sites. This method would be unsuitable for use in any extrapolation of plant community location by elevation due to the limited within-site spatial coverage. The use of dGPS and appropriate ecological techniques in this study provided a rapid and accurate assessment of the main environmental variables which determine plant community type. More specifically, the results of this chapter have provided an accurate quantification of the relationships between the environmental variables and highlighted the importance of micro-topography in determining plant community. This study has shown that very small differences in elevation, as little as 0.04m, can affect plant community composition. The quantification of the elevation-plant community relationship provides a relatively easy to measure variable which can be used to predict plant community type (Moeslund *et al*., 2011).

Baseline hydrological information reinforces the assumption that sea level is an important factor controlling water levels in Baltic coastal plant communities, although the sampling period was short. The data presented in figure 4.4 show that water table level in each of the plant communities followed a similar trend to sea level, as recorded at the Rohuküla tide gauge. Further to this sea level was shown to have an inverse relationship with atmospheric pressure (figure 4.4) which has been reported by previous authors for the Baltic region (Tyler, 1971a; Puurmann & Ratas, 1998; Burnside *et al*., 2007). During the period of study there was no precipitation, except on the final day of the study (figure 4.4) and hence this factor had no influence on water table level. The study was conducted in June and July, which is the period of the year with the greatest PET in western Estonia, 72mm/month and 109mm/month respectively of a total yearly PET of 447mm (figure 4.5). However, actual losses through evapotranspiration are likely to be much lower (Lu *et al*., 2007), and due to continuous recharge of groundwater from inundation (figure 4.4) losses through evapotranspiration are unlikely to have contributed to a considerable decrease in water table level. In the OP plant community there appears to be much less variation in water table depth. This could be seen in the field as standing pools formed over the OP patches where the dipwells were located during the data collection period, and is supported by previous observations by Burnside *et al*. (2008). There was a much greater variation in water depth at the lower elevation plant communities, particularly in the CS and RS communities, which were near continuously inundated during the field sampling. There were less extreme fluctuations in water depth in the higher elevation plant communities. This was most likely due to slow percolation times of water through the silt rich soils, due to low hydraulic conductivity, the low elevation gradient over the wetland, and the distance to the main input source (i.e. the Baltic Sea) (Richardson & Vepraskas, 2001).

Three groups of plant community types could be identified for several of the environmental factors examined (figure. 4.4). These were firstly a group consisting of the two swamp communities (CS and RS), and invariably LS, and less frequently US grasslands. This group had low elevation above sea level, soil N, K, organic matter content, and pH, and high soil moisture and salinity. A previous study by Tyler (1971b) investigating hydrology and salinity within Baltic coastal wetland plant communities found similar results. Previous studies have also suggested that the RS community is the same as the CS community (Tyler, 1971b; Rebassoo, 1975; Paal, 1998). However, this study, which used the Burnside *et al*. (2007) classification where CS and RS were identified as separate plant communities, has shown that the CS and RS communities can be distinguished by the elevation at which they occur.

Secondly, the OP community type was distinct and not associated with any other communities other than swamp (CS and RS) and shore grasslands (LS and US) for salinity and with CS for soil K. The OP community, whilst it appears to be constrained to a relatively narrow elevation range, overlaps in elevation with the TG plant community. Burnside *et al*. (2007), in a study of 5 Estonian coastal wetlands, found that OP was absent from 3 of the 5 sites and covered between 3% and 5% of the total area at the other two sites. OP has also been suggested to be an ephemeral plant community formed by trampling of the vegetation and substrate which leads to localised pooling and subsequent precipitation of salts providing an open bare habitat suitable for halophytic vegetation (Tyler, 1971b).

Thirdly, a group consisted of TG, SW and occasionally US and rarely LS grassland. This group was situated at a high elevation, had high soil N, K, and organic matter content and neutral soil pH. In addition the TG and SW group had significantly lower soil salinity to the other plant communities, and SW had significantly lower moisture. The TG and SW plant community of Estonian coastal wetlands does not have an equivalent zone in other coastal wetland systems due to its much lower frequency of inundation. Typically in tidal coastal wetlands such as salt marshes, the frequency of inundation of the highest elevation plant community, or high marsh, is < 360 and ≥ 10 times a year (Chapman, 1960). However, the TG and SW plant communities can remain continuously exposed all year round in some years (Puurmann *pers. comm*. 2009). This, together with the low salinity of the Baltic Sea, would explain why, in contrast to the high marsh areas of salt marshes (Wiegert & Freeman, 1990), the TG and SW plant communities have low soil salinity.

In the Burnside *et al*. (2007) classification, the authors suggested that the community classification could be applied at different hierarchical levels. At higher levels the communities were grouped into broader classes. The next community level up from CS, RS, LS, US, OP, TG, and SW combined CS and RS together as Swamp vegetation (SP), LS and US together as Wet Grassland (WG), and TG and SW as Grass and Scrub (GS) (figure 3.6). The OP community is a stand-alone community not related to the others at this level from the hierarchical classification. It is interesting to note that the Burnside *et al*. (2007) study made these classifications on the basis of plant species frequency and abundance only. This corresponds to the groups found in this study with regards to a range of environmental variables and micro-topography. In particular, the results of this study have also shown that the finer scale CS, RS, LS, US, TG, and SW plant communities can all be distinguished by their elevation ranges.

All plant community types were contained within narrow elevation ranges, with the exception of the TG community. Moreover, the study indicated that very small differences in elevation above sea level, of the magnitude of 0.04m, can have a significant effect on the presence and composition of plant communities in wetlands. In temperate tidal wetlands such as salt marshes, with typically much larger tidal ranges (≥ 2m) than those of the Baltic, distinct vegetation zones (Mitsch & Gosselink, 2000) can be distinguished, and are clearly affected by the relationship between hydrology and topography (Allen & Pye, 1992; Woodroffe, 2003). In this study, some zonation determined by elevation-hydrological relationships was also described. Inland, where the SW and TG communities were often located, the land is elevated and the water table was shown to be further away from the rhizosphere. Conversely in the lowest sublittoral areas, where CS and RS were characteristic, the water table was often either at or above the soil surface (Tyler, 1971b). However, a more complicated community pattern forms within this zoned structure in these non tidal wetlands (Rebassoo, 1975; Ребассоо, 1987; Burnside *et al*., 2007; Berg, 2009). This is due to the relationship between a micro-topographic mosaic and an irregular, non-tidal, percolating hydrology which has a pronounced effect on community composition.

In coastal wetlands inundation of brackish or saline water will increase soil salinity, hence, areas more often submerged tend to be more saline, which affects plant species distribution (Chapman, 1939; Pennings *et al*., 2005). This study has shown a decrease in salinity associated with land elevation. The more frequently inundated areas were situated in low lying zones or patches with higher soil salinity, which in this study supported CS and RS, and LS and to a lesser extent US grassland community types. The OP community was an exception to this trend, because it showed high salinity despite being elevated. Moreover, Burnside *et al.* (2008) suggested that, due to low vegetation cover, pioneer patches experienced high levels of evaporation and subsequent salt accumulation following flooding.

4.6.1.1 Edaphic factors

Estonian coastal wetlands are situated in rising terrain due to post glacial isostatic uplift (Vallner *et al.,* 1988). This in turn means that the more elevated sections have risen above sea level much earlier than those at lower elevation. Much of the coastal wetland landscape rose above sea level less than 500 years ago, yet the upper reaches can be up to 2500 years old and the lowest areas have yet to rise above sea level and develop terrestrial organic soils (Truus, 1999). The higher elevation areas, populated by plant communities such as TG and SW, were found to have thicker, more organic and nutrient rich soils. Conversely, the lower elevation areas containing CS and RS possessed more mineral rich, less organic soils. Although the plant communities were shown to fall into three significantly different groups for pH, the indicator species for the CS and RS communities, *B. maritimus*, *S. lacustris* and *P. australis*, are all either not pH specific, as is the case for *P. australis* (Grime *et al.*, 1988) and *S. lacustris* (Fitter *et al.*, 1984), or are acidophiles as is the case for *B*. *maritimus* which is tolerant to pH from 3.7-5.3 (Dykyjova, 1986). In this study the LS, US, TG and SW communities were not found to have significantly different pH values, which ranged from 5.89-6.245, and values of ~6.0 to 7.5 are not considered to directly affect plant roots or soil microbial activity (Richardson & Vepraskas, 2001). Thus, these communities do not include any species which prefer very acid or alkaline soils.

The higher proportions of coarse, medium, and fine sand to silt and clay in TG and SW than in CS, RS, LS and US can be explained by the fact that deposition of marine sediments only occurs in these higher elevation communities during storms (Stumpf, 1983), at which time larger particles are likely to be deposited (Suursaar, 2001a). In the lower elevation communities, CS, RS, LS and US, deposition is likely to be dominated by silt/ clay sediments during the more frequent low intensity inundation that occurs in these communities.

4.6.2 Plant communities and distribution

4.6.2.1 Clubrush Swamp

The CS community in this study was found most often at the littoral edge, at a median elevation of 0.01m above m.s.l.. The CS community was also found to have high soil moisture content (58.3%), and low levels of soil N, K and organic matter. In wetlands the majority of soil N comes from atmospheric deposition and groundwater input (Mitsch & Gosselink, 2000). Soils that are developed under anaerobic conditions, as may occur in the CS and RS communities (and to a lesser extent in the LS and US communities) tend to have lower soil N due to the slower diffusion of N through water and the slow decomposition of N compounds (Richardson & Vepraskas, 2001). However, under anaerobic conditions five times the N is maintained as NH_4^+ than would occur under aerobic conditions mainly due to the prevention of nitrification (Patrick, 1982). This may in part explain the contrast in the lower levels of soil N that occur in the frequently inundated CS, LS, and US compared to the less frequently inundated TG and SW plant communities, which have significantly higher soil N.

The two indicator species for the CS community type, *Schoenoplectus lacustris* and *Bolboschoenus maritimus*, germinate below water level (Coops & van der Velde 1995, Clevering *et al.,* 1996) to a depth of 0.2m (Weisner *et al.,* 1993), which probably limited its seaward distribution in this study. CS species are capable of tolerating continued inundation of brackish water (Weisner *et al.,* 1993, Lillebø *et al.,* 2003). However, in conditions with higher salinity levels both of these species, in particular *S. lacustris*, suffer from restricted growth. A previous study has shown that under submerged conditions *B. maritimus* produced much taller shoots and greater seed production than individuals grown at or above water level (Lieffers & Shay, 1981). Similarly Coops & van der Velde (1995) found that from a group of helophyte species *P. australis* seeds were more likely to germinate in exposed soil. Conversely *S. lacustris* seeds were most likely to germinate in submerged conditions. Mauchamp *et al*. (2001) showed that during the heterotrophic stage *P*. *australis* seeds have high mortality rates in submerged conditions. This is evidenced by the fact that *P. australis* is a poor coloniser of areas that are regularly submerged during the growing season. This puts *B. maritimus* and *S. lacustris* at a competitive advantage with *P. australis* in less elevated, more frequently inundated areas. Shading also affects the relative growth rate of non submerged *B. maritimus* and *S. lacustris* (Clevering *et al.,* 1996), which is likely to limit their distribution at higher elevations in Estonian coastal wetlands, such as in RS which is dominated by the taller dense canopy of *P. australis* limiting light availability.

In addition to submergence, wave activity has an effect on littoral plant community type (Coops *et al*., 1991). Coops *et al*. (1994), in a comparative study of *P. australis* and *B. maritimus*, showed that the stiff hollow stems of *P. australis* are more resistant to wave energy than the flexible stems of *B. maritimus*. The study site used in this research programme (Tahu) is located in a sheltered bay and wave energy is limited and lower energy wavelets are less likely to cause breakages on the seaward edge of the wetlands, which are dominated by *B. maritimus* and *S. lacustris*. However, it has been noted that in higher energy coastlines in the Baltic both *B. maritimus* and *S. lacustris* are absent from the seaward edge, which is typically dominated by *P. australis* (Coops *et al.*, 1991).

4.6.2.2 Reed Swamp

The lower reaches of the RS community type coalesced with the upper reaches of CS. The predominant indicator species for RS was *P. australis* (Burnside *et al*., 2007) and no other species was present in more than 40% of the quadrats in this community. *P. australis* is a typical competitor strategist, which establishes itself and then grows rapidly and attains heights of 2-3m in vast stands (Grime *et al.*, 1988). *P. australis* forms one of the most productive ecosystems in the world (Brix, 1999). It has similar salinity tolerance to *S. lacustris* and is known to be found in both fresh and brackish water systems, often in between freshwater and salt marshes (Grime *et al.*, 1988) in Western Europe. *P. australis* is a poor coloniser of areas that are regularly submerged during the growing season (Mauchamp *et al.,* 2001). However, the RS community was found to be inundated less often than CS and thus *P. australis* dominates and prevents the expansion of the CS community to higher elevations. The upper reaches of RS were typically found at a higher elevation than CS, and patches could be found throughout the wetland landscape, although soil moisture levels were not significantly higher. A limiting factor for the spread of RS to higher, albeit wet, elevated land has been shown to be vegetation management, due to selective grazing by cattle of new shoots (Grime *et al*., 1988; Jutila, 1999; Berg, 2009). The three lowest elevation communities (CS, RS, and LS) did not exhibit significantly different soil moisture or salinity values from each other. In the day prior to the recording of the soil moisture values the barometric pressure had fallen which resulted in a rise in sea level, inundating these communities. Therefore the values recorded for the CS, RS and LS communities reflect super saturated soils. Such inundation events do not reflect the permanent situation throughout the growing season (Tyler, 1971b; Wallentinus, 1973). At times inundation events can be more severe, even on rare occasions to the elevation of the SW community, although this is typically associated with storm surges (Suursaar *et al*., 2001a) and was not recorded by the dipwells during this study. RS was also found to have relatively high soil K, which is likely to be related to litter produced by *P*. *australis* during winter dieback.

4.6.2.3 Lower Shore and Upper Shore

The LS and US community types were shown to be confined to specific elevation ranges (0.30m and 0.34m above m.s.l. respectively). These two wet grassland types are vitally important for nature conservation as they support nesting wading birds and migrating wildfowl. They contain similar plant indicator species, including *Juncus gerardii*, *Triglochin maritima*, *Glaux maritima*, *Plantago maritima, Agrostis stolonifera* and *Festuca rubra* (Wallentinus, 1973; Rebassoo, 1975; Jerling, 1983; Burnside *et al.,* 2007). However, the proportions of *Juncus gerardii* and *Festuca rubra* differ, with the former more abundant in the LS. All of the species found in these two communities are tolerant of inundation by brackish water although not prolonged submergence. This, and shading, limit their spread to the more frequently inundated RS community. Wallentinus (1973) found that there was a negative correlation between *J. gerardii* and *F. rubra* in Baltic coastal wetlands. In a study of grazed Baltic coastal wetlands in Sweden Jerling (1983) found that the geolittoral vegetation was dominated by *J. gerardii*. Inland from this was a belt of *F. rubra* dominated vegetation. In the Jerling (1983) study the seed belt was dominated by *J. gerardii* throughout the whole gradient, although *Juncus* seeds have been shown to have a low survivorship in established vegetation (Lazenby, 1955) and flourish better in bare ground (Jerling, 1983). *J. gerardii* is more tolerant of inundation than *F. rubra* (Cooper, 1982), which explains the lower elevation of the LS community in relation to US. Other species typical of the LS community, such as *Triglochin maritima* and *Glaux maritima,* were found in great numbers in the seed bank study by Jerling (1983). Both *T. maritima* and *G. maritima* can vegetatively propagate and have small seeds with innate or induced dormancy. Light and temperature are found to induce the germination of *G. maritima* seeds, as is found in the more open stands of the LS and US communities compared to dense CS, RS, TG and SW (Rozema & Riphagen, 1977). In a study by Jerling (1988a), it was found that *G. maritima* seeds were more likely to germinate in strongly fluctuating temperatures such as occur during inundation events. The seeds remain dormant in soils with limited changes in temperature during the growing season such as are found in the rarely inundated TG and SW communities and in the two communities with prolonged inundation, CS and RS. *Plantago maritima* and *T. maritima* require cold water and light as stimulants for germination, both of which are typical environmental conditions for the LS and US communities due to the frequency of inundation in these intermediate elevation plant communities with predominantly low growing plant species. *T. maritima* is a community engineer, in that it can influence the surrounding environment by developing a raised ring formation of soil around itself in response to increased waterlogging (Fogel *et al*., 2004). This increases its survivability at the adult stage in regularly inundated zones such as the LS and US communities.

F. rubra, the dominant species in the US community, is a coloniser of open habitats and not shade tolerant (Magee & Antos, 1992), which explains its absence from the CS and RS communities. *F*. *rubra* is tolerant to close grazing although over grazing leads to dieback (Howarth, 1920). Overgrazing however, does not occur in these lightly grazed meadows, which are more at risk from abandonment (Berg, 2009). In European populations *F. rubra* does not form a persistent seed bank (Champness & Morris, 1948; Roberts, 1981), which would explain the limited presence of *F. rubra* seeds in the Jerling (1983) study. It does however spread vegetatively by rhizomes (Eriksson, 1989), as do many species in waterlogged soils. *F. rubra* has been found to tolerate spring flooding and grows well in waterlogged soils but is also tolerant to drought (Chapman, 1939), which partly explains its greater dominance in the higher elevation US community compared to the wetter LS community. *F. rubra* is also tussock forming, which can alter elevation. Although *F. rubra* and *J. gerardii* have similar tolerance to flooding, *J. gerardii* is acknowledged to be slightly less negatively affected by inundation (Cooper, 1982) and salinity (Rozema & Blom, 1977). Therefore, it would appear that the lower elevation LS community, and consequent more frequent inundation, is one of the environmental factors controlling the abundance of *J. gerardii* in the LS plant community and *F. rubra* in US.

4.6.2.4 Open Pioneer

OP patches were found in localised depressions with little or no vegetation cover but at relatively high elevations in the landscape. After inundation events these OP areas appear to form standing pools, which then subsequently evaporate to leave surface salt deposits. This creates conditions for the establishment of halophytic plant species such as *Suaeda maritima* and *Salicornia europaea*, which are rare in the eastern Baltic area (EC Habitats Directive, 1992). Both *S. maritima* and *S. europaea* are commonly found in western European salt marshes as pioneer stress tolerant species in the low elevation sub-littoral zone (Chapman, 1939; Cooper, 1982; Hill *et al*., 1999). In Baltic coastal wetlands, these areas are characterised by the CS and RS swamp communities. The low salinity, 2.5-4ppt (Kotta *et al*., 2008), of the sea water in Estonia means that halophytic plants such as *S. europaea* and *S. maritima* are outcompeted by larger fast growing helophytes such as *B. maritimus*, *S lacustris* and *P. australis*. In this study the CS, RS, LS, US and OP communities did not have significantly different soil salinity levels from each other, although they were significantly higher than in the TG and SW communities. In a previous study, however, by Burnside *et al*. (2008) the OP community was found to have a two times higher soil salinity than the next most saline community type (CS). The study took place during a prolonged warm period (Burnside, *pers*. *comm*. 2009), whereas the data for this study were collected during a damp summer period and following an inundation event.

Other authors have shown that the OP community is often inundated for prolonged periods of the growing season (Rebassoo, 1975; Puurmann & Ratas, 1998), and fluctuations in salinity and inundation are conditions that suit the environmental preferences of both *S*. *europaea* and *S*. *maritima* (Ellison, 1987; Tessier *et al*., 2002). In a study of the effects of salinity and water logging on the growth of the salt marsh plants *F. rubra, J. gerardii, Armeria maritima, Plantago maritima, Aster tripolium, Triglochin maritima, Puccinellia maritima* and *S. europaea*, *S. europaea* was the only species to achieve maximum yield on the drained saline treatment (Cooper, 1982). Both *S. europaea* and *S. maritima* usually occur as scattered individuals, frequently in colonies (Chapman, 1947; Jefferies *et al.*, 1981). The same distribution was observed in this study with individuals in clustered groups surrounded by bare ground. It would appear that salinity is one of the main factors controlling the plant species in the OP community. This sparse vegetation is unlikely to develop organic, nutrient rich soils, and the OP community had the lowest soil N, K and organic matter in this study. In addition to the limited development of organic nutrient rich soils, the OP community had greater proportions of coarse, medium and fine sand than the other communities. Coarser soils have lower nutrient retention rates than soils with greater silt and clay fractions. *S. maritima* and *S. europaea*, commonly occur in areas with low soil N, and do not exhibit any increase in growth with the addition of N to soils, reflecting their adaptation to summer drought and saline conditions (Jefferies, 1977; Jefferies & Perkins, 1977). In general potassium in soils is derived from clay minerals and in organic matter such as plant litter (Spray & McGlothlin, 2004). The OP community type was the community with the lowest soil clay content, the lowest soil organic matter and plant litter was rare (Burnside *et al*., 2007).

4.6.2.5 Tall Grass and Scrub and developing Woodland

At higher elevations where TG and SW co-existed, grazing was limiting the encroachment of SW into the TG community (Puurmann & Randla, 1999). The indicator species for TG, which includes *Elytrigia repens*, *Potentilla anserina*, *Peucedenum palustre* and *Vicia cracca*, are not tolerant to regular or prolonged inundation, or salinity (Eriksson, 1988). These species prefer the water table to be within 0.20-0.50m of the surface dependent on species (Grime *et al.,* 1988; Meredith & Grubb, 1993; Newbold & Mountford, 1997). TG may be considered a mesic grassland within Baltic coastal wetlands and it may be that its extensive distribution is due in part to its somewhat broad classification (Burnside *et al*., 2007).

SW was located at the highest elevation, with a median value of 0.62m above m.s.l., and was found to have the lowest soil moisture (with the exception of OP) and lowest salinity. Interestingly the TG and SW communities had the highest organic matter and nutrients when compared to the other plant communities. The main indicator species of SW were *Molinea caerulea*, *Achillea millefolium*, *Juniperus communis* and *Pinus sylvestris*. *J. communis*, *M. caerulea* and *A. millefolium* are intolerant of waterlogging and absent from regularly inundated communities (Grime *et al.,* 1988, Newbold & Mountford, 1997). *J. communis* is typically associated with poor soils and harsh environments (Garcia *et al*., 2000). This species is also rarely grazed due to monoterpenes found in the needles, wood and cones, although it will be consumed by deer, cattle and sheep if food is scarce (Thomas *et al*., 2007). This means that in managed grasslands it does not dominate due to mowing but in unmanaged grasslands *J. communis* dominates the upper epilittoral zone with *P. sylvestris* and other woody species (Rannap *et al*., 2004) and once established is unlikely to be affected by grazing. *P. sylvestris* is tolerant of a wide range of environmental conditions including waterlogging but survival following germination is severely reduced in saline conditions (Werkhoven *et al.,* 1966). The TG and SW communities had the most fertile soils of all the communities in this study, although the thickness of the organic layer was no deeper than 0.25m (Puurmann & Ratas, 1998). For tree species the thickness of organic layer is a limiting factor to growth (Liski,

1995) and the *P. sylvestris* individuals found in Baltic coastal wetlands rarely exceed 5m in height, much smaller than individuals of the same species in other Estonian habitats (Carlisle & Brown, 1968). Hence, this community is excluded from lower elevations that are more frequently inundated.

4.7 Conclusions

This study has shown that micro-topography and edaphic factors play an important role in determining the location of the plant communities within Estonian coastal wetlands. Inundation occurs through both overland flow and throughflow and there is a clear relationship between elevation and inundation, although the hydrological data were recorded over a relatively short time period and other factors that can influence the hydrological regime, such as precipitation events did not occur during the study period. This interrelationship has led to the development of a mosaic within a broader zonation of plant community types across the wetland landscape. The main environmental variables shown to affect plant communities were elevation, soil moisture, organic matter, N, K and salinity. The OP community was also determined by sediment particle size.

The results of the CCA suggest that soil organic matter, N, salinity and moisture, and pH more strongly influence plant community type than elevation. However, elevation provides a single, readily measureable parameter that can be used to distinguish plant community types within Estonian coastal wetlands, with the exception of OP which overlaps with the TG elevation range. It therefore seems reasonable to assume that elevation could be used as a predictor of plant community type. However, the results of this study have shown that the elevation ranges at which the different plant communities occur are relatively small. Thus, accurate elevation data must be used in order to be able to accurately predict plant community type.

The results have implications for land managers in terms of nature conservation and ecological restoration, because it seems that relatively small alterations in surface topography or water levels (in the order of a few cm in this study) are likely to produce substantial vegetation changes, allowing targeting of valuable communities or rare species with specific ecohydrological requirements. The study also has implications for current concerns over sea level rise through climate change (Parry *et al.,* 2007).

The Baltic Sea, which has a low tidal range (typically <0.02m), may be particularly vulnerable to sea-level rise (Nicholls *et al.,* 1999). Although any sea level rise in the Baltic region would be partially offset by land uplift due to glacial isostatic rebound (Meier *et al*., 2004), the effects of sea level rise on coastal wetlands would be profound given the sensitivity of wetland plant communities shown in this study and their importance as a habitat for a variety of bird species. As such this is a topic which requires further study through the development of a plant community model derived from detailed elevation data, as will be conducted in chapter 5.

5. The use of medium point density LiDAR elevation data in determining the location of plant community types in Estonian coastal wetlands

5.1 Preamble

The study in this chapter examined the use of LiDAR elevation data in determining the location and extent of plant communities in Estonian coastal wetlands. Using the known elevation ranges of the plant communities, as derived from the study in chapter 4, a predictive plant community model was developed for Tahu using LiDAR elevation data calibrated and corrected by data derived from a dGPS survey. The plant community model developed at Tahu was applied to two non-contiguous Estonian coastal wetland sites, Matsalu and Kudani for model validation. The accuracy of the plant community models for Matsalu and Kudani was assessed with a ground truthing survey. The study demonstrated that LiDAR elevation data can be used, with dGPS elevation corrections, to accurately model the location of plant communities within Estonian coastal wetlands.

5.2 Introduction

5.2.1 Predictive modelling

Predictive modelling of habitats, plant communities and individual species is a commonly used tool in ecology (Franklin, 1995). There are a wide variety of remotely sensed data available which cover large areas and can, if properly used, reduce the amount of time required to collect data in the field (Jensen, 2007). These data can be used as a basis for the development of models to predict the location and extent of plant communities based on knowledge of ecological requirements. The two main techniques used in predictive modelling are often characterised as correlative and mechanistic (Burnside & Waite, 2011) as discussed in section 2.4.5 in chapter 2. Both the correlative and mechanistic approaches have been shown to produce accurate predictive models of plant community location (Robertson *et al*., 2003). Beerling *et al*. (1995) undertook a study to assess the validity of a correlative model in predicting the potential distribution of the invasive species *Fallopia japonica* in Europe. The study showed that while the correlative model worked very well in predicting distribution in the native Asian habitat, there were limitations in using the model in Europe due to factors that were not included in the model such as inter-species competition with native European plants. There are a number of drawbacks and limitations to correlative approaches: problems of scale (Hoffman, 2006); issues of available data (Franklin, 1995); and issues of false absences (Burnside & Waite, 2011). However, Robertson *et al*. (2003) state that, in tests, correlative models perform as well or better than mechanistic models in geographically similar areas. Mechanistic models have been shown to produce accurate predictive models (Hoffman, 2006). However, in plant community modelling, a mechanistic model requires in depth knowledge of the autecology of each species that is present in the plant community as well as knowledge of the inter-species competition within each plant community (Franklin, 1995) and often these data are not available. In summary, Hoffman (2006) suggests that often predictive models employ a mix of both correlative and mechanistic approaches to best represent the distribution of plant communities.

5.2.2 The use of Light Detection and Ranging (LiDAR) elevation data

Following the development of laser technology, laser instruments were used to measure distances by calculating the travel time of light from the transmitter to receiver. Light Detection And Ranging (LiDAR) became commercially available in the mid 1990's and provides high resolution terrain and feature data. Recent advances in aircraft mounted kinematic dGPS and inertial stabilising systems for consistently accurate georeferencing, in conjunction with improved GIS and CAD handling of LiDAR point data, has made this data source a common tool in ecological and vegetation studies (Lefsky *et al*., 2002; Raber *et al*., 2002; Rosso *et al*., 2006; Simard *et al*., 2006). LiDAR typically comes in the form of point data, therefore the data typically undergoes some form of interpolation in order to produce a digital elevation model (DEM). Common methods of interpolating point data are the deterministic methods of Triangulated Irregular Networks (TIN) and Inverse Distance Weighting (IDW), and geostatistical methods such as Kriging (de Smith *et al*., 2007). Several authors have made comparisons of different interpolators with LiDAR data (Gonçalves, 2006; Guo *et al*., 2010; Hernadez-Stefanoni *et al*., 2011), although there is no consensus as to the best interpolator.

Previous studies using LiDAR have demonstrated a range of applications. For example in forestry measuring canopy heights (Dubayah *et al*., 2000; St-Onge, B. & Achaichia, N. 2001; Hudak *et al*., 2002), stand volume (Nilsson, 1996; Drake *et al*., 2002), basal area (Means *et al*., 1999; Drake *et al*., 2002) and above ground biomass (Dubayah *et al*., 2000; Lim & Treitz, 2004), as well as bathymetric studies in areas too shallow for hydrographic vessels (Irish & White, 1998), elevation mapping of river valleys (Jones *et al*., 2007), and coastal and estuarine habitats (Chust *et al*., 2008). Studies have also been conducted using LiDAR data to model plant communities in coastal wetlands (Morris *et al*., 2005; Prisloe *et al*., 2006; Sadro *et al*., 2007; Moeslund *et al*., 2011).

The Prisloe *et al*. (2006) study distinguished stands of *Phragmites australis* from other plant communities in coastal salt marshes using the LiDAR data. They identified *P. australis* by the height of the stands, which was greater than other salt marsh species, using the first and last return of multi-pulse LiDAR. This methodology has been used in a variety of forestry studies (Dubayah *et al*., 2000; St-Onge, B. & Achaichia, N. 2001; Hudak *et al*., 2002). However, in Estonian coastal wetlands it would be difficult to differentiate between stands of *P. australis* (RS) and CS species or low shrubs in SW. A study by Sadro *et al*. (2007) used multi-spectral remotely sensed data to model plant communities in a coastal salt marsh, interestingly while this study did not use LiDAR to model the plant communities an assessment was made of the elevation accuracy of LiDAR, using a dGPS unit, in four plant communities. The study found that LiDAR data were inaccurate in dense plant communities. The Morris *et al*. (2005) and Moeslund *et al*. (2011) studies used LiDAR data to model tidal wetland plant communities. However, neither of the studies made any attempt to assess the accuracy of the LiDAR elevation data. A study by Rayburg *et al*. (2009) assessed the elevation accuracy of dGPS compared to LiDAR and found that in appropriate open habitats, i.e. not forest or urban areas, dGPS provided significantly greater elevation accuracy and thus is appropriate for use in assessing and calibrating the elevation accuracy of LiDAR in Estonian coastal wetlands.

The research study conducted in chapter 4 clearly established that plant communities within the coastal mosaic were significantly influenced by elevation and a range of edaphic factors. The study demonstrated that elevation, likely due to its relationship with hydrology, was a major factor in determining plant community type. It was therefore hypothesised that using accurate elevation data, plant community type could be predicted across the coastal wetland area.

The aim of this chapter was to develop a static correlative predictive plant community model for Estonian coastal wetlands using LiDAR elevation data. The objectives were: 1) to create a predictive plant community model at Tahu based on dGPS corrected LiDAR elevation data; 2) to assess and select the optimum modelling method at Tahu; 3) to apply the model to the Matsalu and Kudani coastal wetlands; and 4) to validate the models for the Matsalu and Kudani coastal wetlands with a ground truthing survey. The study in this chapter expands upon work previously published in Ward *et al*. (2012)

5.3 Study sites and methodology

5.3.1. Study area

This study took place in three areas of coastal wetland in north and western Estonia; Tahu, Kudani and Matsalu (figure 5.1). The Tahu and Kudani coastal wetlands are located in the Silma Nature Reserve. The Silma Nature Reserve is a regional reserve designated as an Important Bird Area (IBA) due to the large numbers of migratory and breeding birds that use the reserve. The total area of coastal wetlands within the reserve is >600ha. The Matsalu coastal wetland is located some 30km south and is in the Matsalu National Park (figure 5.1).

The Tahu site contains 104.4ha of coastal wetland and is located in the interior of Haapsalu Bay. The Kudani site extends to 78.7ha and is adjacent to Võõlameri, which is a former bay connected to the Gulf of Finland. Tahu and Kudani have been managed with low intensity grazing since the early 1990's and early 2000's respectively (Puurmann, *pers*. *comm*. 2009). The Matsalu coastal wetland is an area of 189.8 ha located on the south coast of Matsalu Bay, which is situated to the south of Väinameri. The Matsalu coastal wetland has been regularly managed since the 1940's (Truus, 1999).

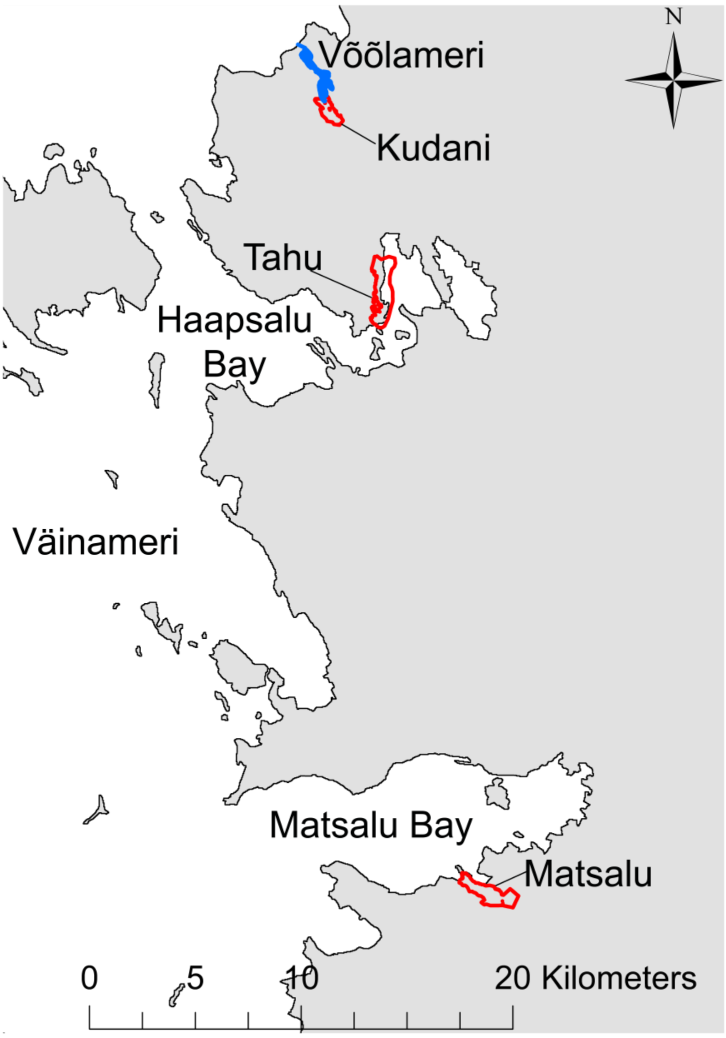

Figure 5.1: The location of the three wetland study sites in Estonia. The Tahu and Kudani sites are located in the Silma Nature Reserve. The Tahu site is located on the interior of Haapsalu Bay. The Kudani site is located on the coast of Võõlameri, a former bay which connects the wetland to the Baltic Sea. The Matsalu study site is situated on the south coast of Matsalu Bay in the Matsalu National Park.

5.3.2. Model development

5.3.2.1 LiDAR data collection

LiDAR data were collected by the Estonian Land Board in 2009 from an altitude of 2400m using an ALS50-II laser/detector. Average point density was 0.45 points/m^2 with a maximum point spacing of 2.6m. The illuminated footprint diameter was 0.54m (at 2400m altitude). Data elevation returns were categorised as Only (typically ground or grass), First (typically top of a tree), Intermediate (typically mid-section of a tree or tall grass), and Last (typically ground or grass) (Vain *et al*., 2010). In this study only the last and only returns were used to collect bare earth data.

5.3.2.2. Calibration and adjustment of LiDAR derived elevation using dGPS

LiDAR elevation data have been shown in a previous study to be inaccurate in dense vegetation (Sadro *et al*., 2007). Rayburg *et al*. (2009) showed that an RTK dGPS produced a more accurate DEM than LiDAR. Therefore, in order to improve the elevation accuracy of the LiDAR data a field survey was undertaken in July 2009, using an RTK dGPS (Leica GPS1200 Surveying System) to collect elevation data from the CS, RS, LS, US, TG and SW plant communities at Tahu. LiDAR data covering the same area were selected from the dataset for comparison. Due to the arrangement of the plant communities in a mosaic pattern it was not possible to include all of the plant communities in one contiguous patch. First a 20m x 100m area covering 5 of the plant community types (CS, RS, LS, US and TG) was surveyed with a dGPS. Elevation and associated accuracy through PDOP values were recorded. Following this a second neighbouring 20 x 20m patch, which contained an example of the SW community, was surveyed and elevation and PDOP values recorded.

Due to the difficulties in recording elevation at the same location as individual LiDAR data points, dGPS readings were recorded in a stratified manner at 0.5m intervals. This limited the distance between a dGPS recorded point to a LiDAR point to a maximum distance of 0.25m. The LiDAR data points that were located within the two dGPS surveyed patches were selected from the total data set and exported into Matlab R2010a for comparative analysis.

These comparative elevation data were used in a post-processing method to correct and calibrate the LiDAR elevation data. The difference between the dGPS and LiDAR derived elevation data was analysed by calculating a mean elevation difference value between the LiDAR and dGPS data using a 2m x

2m moving window in Matlab R2010a. This ensured that at least one LiDAR data point was captured and facilitated correction. All data were recorded using Estonian National Grid 1997 system and elevation at BK77 as measured at Kronstadt 59°59’43” N, 29°46’00”.

5.3.2.3. Exploratory Spatial Data Analysis (ESDA)

Prior to interpolation, it is necessary to examine the distribution and statistical properties of sample data points. This process is known as data brushing (ESRI, 2010). The Tahu LiDAR data were tested for spatial autocorrelation using a graduated Moran’s I test at distances of 5m, 10m, 20m, 50m, 100m, 200m, 500m, 1000m, 2000m and 4000m so as to assess at what distance the data were spatially autocorrelated, which would affect the method choice in the IDW interpolation. A trend analysis was used to determine whether trend removal was required prior to the Krige interpolation. The average nearest neighbour distance was calculated and used to determine an appropriate lag size for the interpolation. Data were pre-processed in order to identify and remove any statistical outliers that could adversely affect the interpolation models. The data were also tested for normality. A directional semivariogram was used to assess whether the data were anisotropic and, where relevant, assess the most appropriate bearing for anisotropy, and minor and major ranges.

5.3.2.4 LiDAR derived elevation interpolation models

Three interpolation methods were assessed and compared, comprising two deterministic interpolations, Triangulated Irregular Network (TIN) and Inverse Distance Weighting (IDW), and one geostatistical technique, Ordinary Kriging (OK). These three techniques are commonly used in interpolating LiDAR data (Drouin & Saint-Laurent, 2010).

In both the IDW and OK interpolations the LiDAR elevation data were tested for, and shown to be, spatially autocorrelated. Lag size was assessed by calculating the average distance of the nearest neighbour. An anisotropic model was chosen for the IDW and OK models and illustrated the directional differences in the LiDAR data. The number of neighbouring points was derived by estimating the minimum and maximum number of LiDAR points per plant community patch using field knowledge; these also produced the lowest Mean Prediction Error (MPE) for the IDW and OK interpolations.

Additionally for the IDW interpolation power optimisation was chosen in order to produce the lowest MPE. In the OK interpolation a first order trend was also removed from the data, derived from the results of a trend analysis.

Model accuracy was assessed via a cross validation technique. Cross validation of the TIN, IDW and OK interpolations was performed using a regression analysis. The residual error between the observed and predicted points was calculated and used to evaluate model success.

5.3.3 Model application

5.3.3.1 Categorisation of the elevation data for the three interpolation models by plant community elevation ranges

In order to create a predictive plant community model, the elevation ranges of the plant communities (with the exception of OP), derived from the study in chapter 4 were used. The interquartile ranges in elevation of the lower elevation plant communities CS, RS and LS did not overlap. In the field these plant communities exist proximally, therefore the elevation categories for the CS, RS and LS plant communities were extended to a mid-point between the two proximal interquartile values. In the LS, US, TG, and SW plant communities there was an elevation overlap in the interquartile elevations of proximal plant communities. In this case a mid-point value between the Q3 elevation of one plant community and the Q1 of the plant community at the next highest elevation was selected. Following interpolation the TIN, IDW, and OK LiDAR elevation models for Tahu were classified using the known interquartile elevation categories for each of the studied plant communities (chapter 4). The LiDAR elevation data were also post-processed to adjust for inaccuracies in the LiDAR as highlighted by the comparison between the dGPS – LiDAR elevation values.

5.3.3.2 Assessment and comparison of plant community model prediction at Tahu

The reliability of each surface interpolation to predict plant community type, at Tahu, was assessed via a stratified random field walkover survey in July 2009. Ninety $1m^2$ quadrats (fifteen per plant community) were used to evaluate model predictions and within the quadrats all plant species were recorded as well as their abundance using the Domin scale (Rodwell 1992).

Ground-truthed data were compared with each interpolation model to determine if the plant community found in the quadrats was in agreement with the predicted plant community type derived from each model. The reliability of agreement was assessed using Fleiss Kappa (κ) coefficients (Landis & Koch, 1977). The model that was best able to predict plant community type was used to assess the extent in hectares (ha) of each plant community at the Tahu site.

5.3.4. Model validation

5.3.4.1 Application of the best interpolation model at two further sites, Matsalu and Kudani

The most accurate interpolator of the LiDAR elevation data was used to model plant community distribution at two further coastal wetland sites, Matsalu and Kudani. The models were categorised using previously established plant community and elevation associations. In order to test the validity of the Matsalu and Kudani models to predict plant community, an independent ground truth survey was conducted in July 2010, using a stratified random approach. Fifteen points were selected within each plant community type in the model (yielding ninety quadrats per site) and all plant species presence and abundance were recorded (Rodwell, 1992). In order to assess the reliability of agreement between the plant community models and the ground-truthed data, a Fleiss κ coefficient was used.

5.4. Results

5.4.1 Model development

5.4.1.1 LiDAR elevation accuracy assessment and correction using a dGPS at Tahu

In Estonian coastal wetlands plant communities are rarely all proximal. Therefore, in order to obtain dGPS elevation data for all the plant communities two patches at Tahu were surveyed (figure 5.2). The difference in elevation between the dGPS and LiDAR for one patch, containing the CS, RS, LS, US and TG communities, was between +0.15m and +0.19m (figure 5.3). The second patch, containing SW, exhibited a greater elevation difference between the dGPS and LiDAR, of +0.18 and +1.70m.

At elevations between 0.01m and 0.69m the mean difference between the dGPS and LiDAR derived data was +0.177m (SD = 0.016) (figure 5.3, figure 5.4). These elevation ranges contained all of the studied plant communities except SW. For elevation values over 0.75m the mean difference was 0.799m (SD = 0.472m) (figure 5.3, figure 5.4). These values only occurred in the SW community (figure 5.3), where *Juniperus communis* shrubs and *Pinus sylvestris* trees were located (figure 5.4). The presence of an increased canopy height in the SW community is the probable cause for this greater discrepancy in elevation between the dGPS and LiDAR derived data (figure 5.4). Therefore, in order to improve the accuracy of the LiDAR data, 0.177m was added to all the LiDAR derived elevation points throughout the dataset; this will be considered in detail in the discussion. Although this was an underestimation for the SW community, it was not considered to alter the robustness of the prediction. SW has been shown (in chapter 4) to be located at the highest elevation, therefore it is still discernible and it would not unduly influence the output of the plant community prediction models.

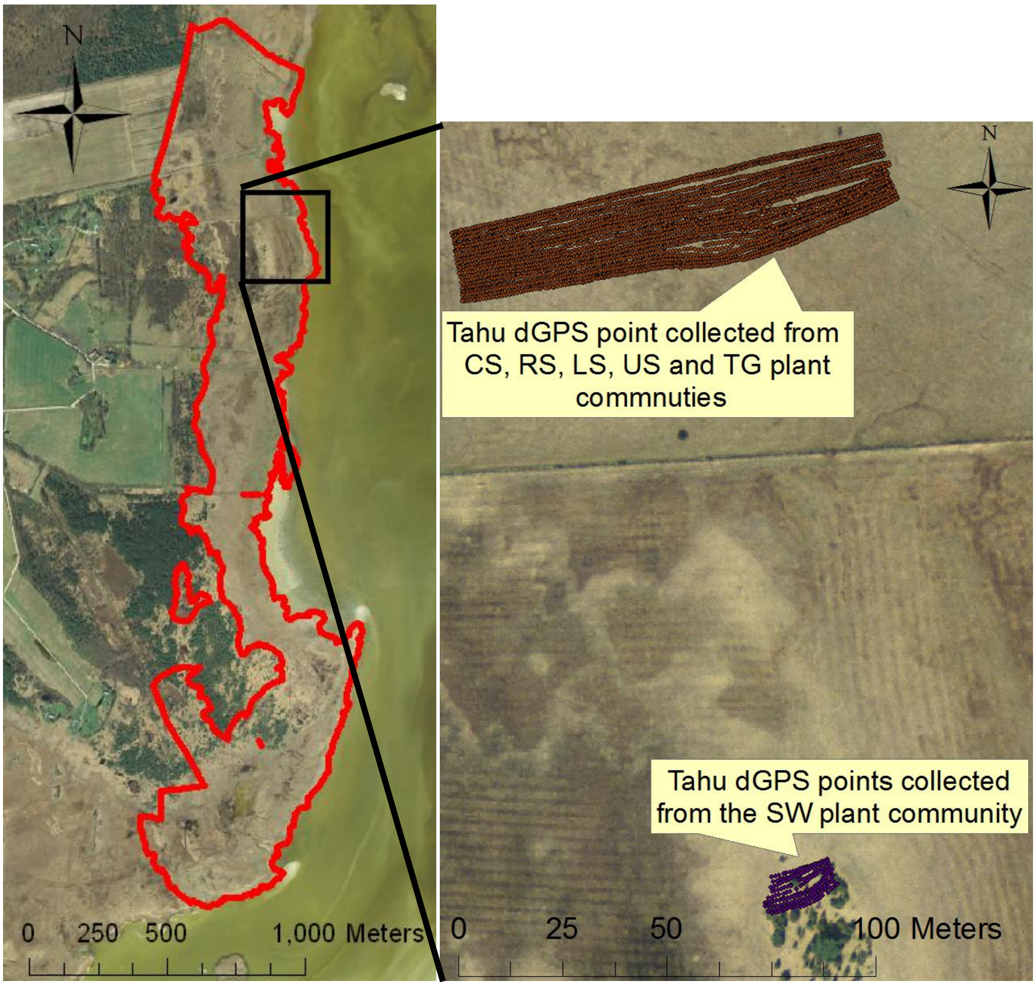

Figure 5.2: The distribution of the dGPS elevation measurements at the Tahu study site. Due to the patchy character of the plant communities they are rarely all located adjacent to each other. To obtain accurate elevation data for all the plant communities, two areas were therefore surveyed in order to assess the accuracy of the LiDAR elevation data.

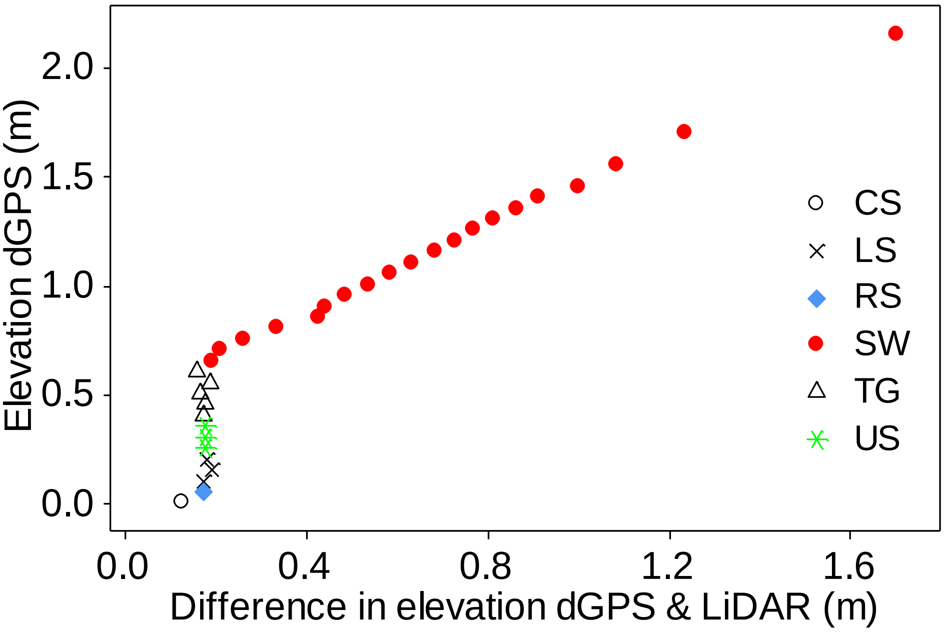

Figure 5.3: Elevation difference between the LiDAR and dGPS height values (m) for each 0.05m interval for the studied CS, RS, LS, US and TG and SW patches at Tahu

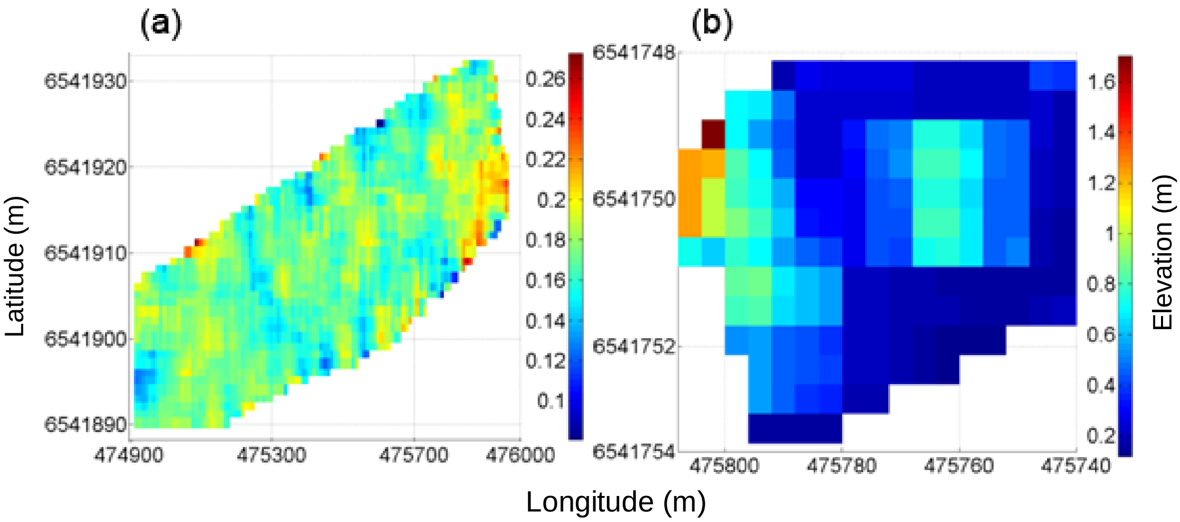

Figure 5.4: Relationship between dGPS and LiDAR elevation at Tahu. The co-ordinates of the points within each patch were recorded in Estonian National Grid (latitude y axis and longitude x axis). (a) Patch covering the CS, RS, LS, US and TG communities. The right side represents lower elevation adjacent to the sea. Elevation gradually increases moving from right to left. (b) Patch covering the SW community. Both a) and b) are coloured by difference in elevation (m) between dGPS and LiDAR derived data.

5.4.1.2 Exploratory spatial data analysis (ESDA)

The LiDAR elevation data for Tahu were spatially auto-correlated at all observed distances ($p < 0.01$). The Morans' index decreased rapidly with increased distance up to 500m at which point the rate of decay slowed (figure 5.5). Trend analysis showed that the Tahu site exhibited a first order trend of increased elevation from m.s.l. moving inland (figure 5.6; x axis). The average nearest neighbour distance between points was 1.001m and this value was then used as the lag size for the OK interpolation (table 5.1).

Anisotropy, minor range, major range and bearing were assessed using a directional semivariogram and the results showed that spatial autocorrelation was affected by directional influences. The strongest anisotropic effect was found at 315.2° and the minor and major ranges were 10.5m and 52.3m respectively (table 5.1) and this was accounted for in the IDW and OK interpolations. The minimum and maximum number of neighbours used in both the IDW and OK interpolations were 1 and 32 respectively (table 5.1).

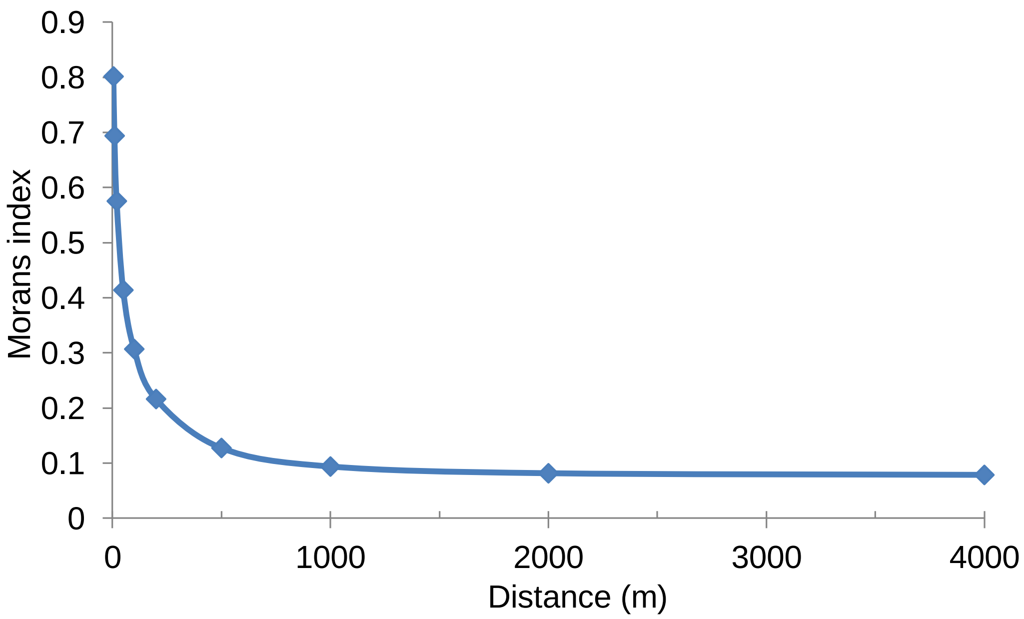

Figure 5.5: Morans index plotted against maximum distance tested (m). The data were significantly spatially autocorrelated at all distances, p < 0.000.

There was a first order trend in elevation increasing from east to west (figure 5.6), which can be related to the zonal character of Estonian coastal wetlands. However, figure 5.6 also shows that elevation does not consistently rise with distance from the sea which is evidence of the mosaic pattern of the plant communities accross the wetland. This was accounted for in the OK model (table 5.1).

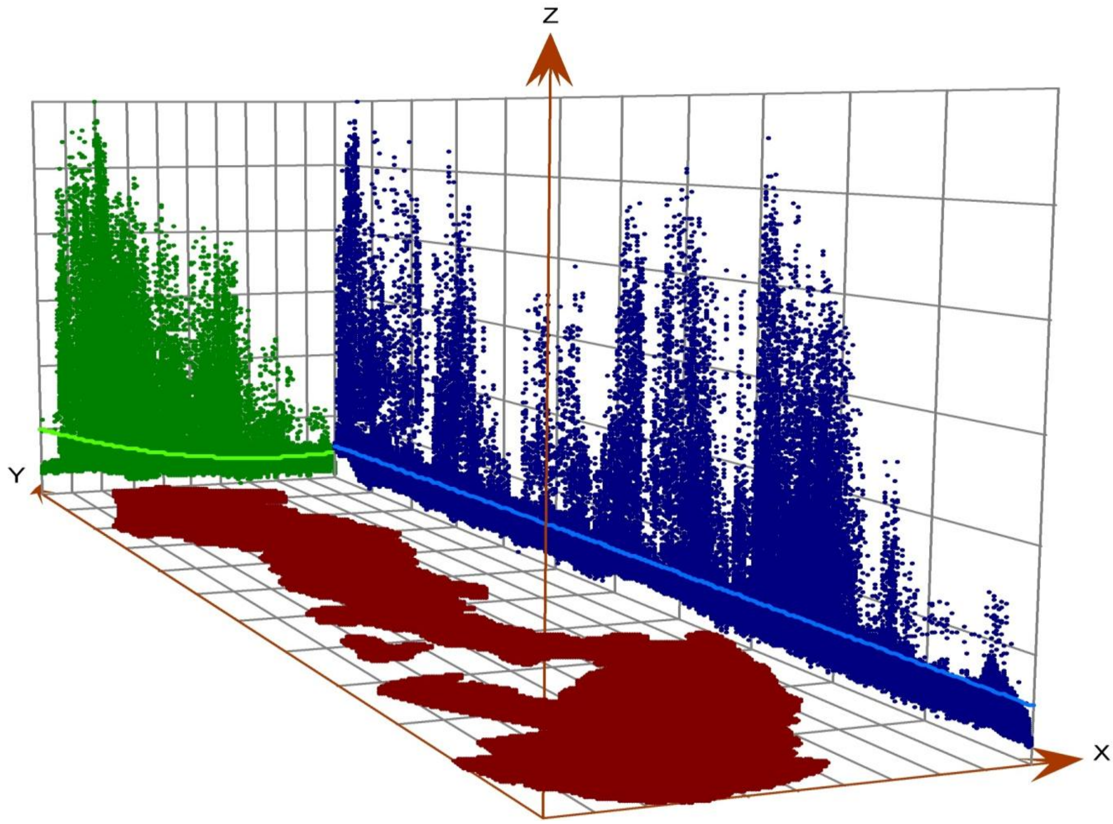

Figure 5.6: Trend analysis of the LiDAR data covering the Tahu site. The Y axis points to the north, the X axis point to the east and the Z axis shows the elevation of the Lidar points. In blue are the projections of all the individual Lidar points along the Y axis. The lighter blue line shows the trend line of the LiDAR elevation points from north to south. In green are the projections of all the individual LiDAR points along the X axis. The lighter green line is the trend line of the LiDAR points from east to west. The projection in red is the ground cover of the selected LiDAR data.

There are few method selections involved in TIN interpolation (table 5.1). In the IDW and OK interpolations the minimum and maximum number of neighbours selected were 1 and 32 respectively (table 5.1). Several different permutations were assessed but these figures produced the interpolation with the lowest MPE. A power optimisation function is provided as a method choice for the IDW interpolation and was selected for use in this study. Power optimisation calculates the optimum weighting, i.e. that which gives the lowest calculated error, by distance, for points used in the interpolation. A first order trend removal was selected for the OK interpolation, as a result of the trend analysis. An assumption of the OK method is that there are no directional trends, and hence trend identification and removal is necessary to enhance the validity of the interpolation. The ESDA identified the direction and distance of anisotropic effects. In order to improve the validity of the IDW and OK interpolations these factors were included. The selected lag size is equal to the mean distance between points and was calculated by assessing the average distance of the nearest neighbour of all points in the Tahu LiDAR dataset. The IDW and OK interpolations produce an assessment of the mean prediction error which, as can be seen in table 5.1, are 4.64×10^{-6}m and 1.3×10^{-4}m respectively.

Table 5.1: Summary of the method choices and validation results for the TIN, IDW and OK interpolations.

Variable	TIN	IDW	OK
Min neighbours	-	1	1
Max neighbours	-	32	32
Power value	-	Optimised	-
Trend removal	-	-	First
Anisotropy	-	Yes	Yes
Minor range (m)	-	10.5	10.5
Major range (m)	-	52.3	52.3
Lag size (m)	-	1.0001	1.0001
Number of lags	-	100	100
Mean prediction error (MPE) (m)	-	4.64×10^{-6}	1.3×10^{-4}
RMSE (m)	-	0.493	0.537
RMSE Standardised	-	1.001	0.989
R^2 predicted - observed (%)	92.4	64.7	62.9

The greater the number of points, the lower the MPE will be. The MPE is comparative if the same number of points is used, as in this case. The IDW has a lower MPE than the OK interpolation suggesting that it produces a more accurate surface. The root mean square error standardised (RMSE standardised) is a comparable prediction error statistic based on the relationship between the average standard error and the RMSE. A perfect interpolator would produce an RMSE standardised value of 1. In this case the IDW had RMSE standardised of 1.001 and the OK an RMSE standardised of 0.989 (table 5.1) further suggesting that in this case IDW is a better interpolator of these data than OK. The aforementioned prediction error values are not available for the TIN interpolation due to the different calculations involved in the interpolations. However, an R^2 comparison between the predicted and observed data is available for the IDW, OK, and TIN interpolations, which can be used to compare the accuracy of the interpolations. The results show that the TIN model produced the most accurate interpolator (R^2 92.4%), followed by the IDW (R^2 64.7%), and lastly OK (R^2 62.9%) (table 5.1).

5.4.1.3 Interpolation of the LiDAR data for Tahu

All three of the examined techniques were regarded as good interpolators. Analysis using cross validation enabled a comparison of the models to be conducted. Of the three interpolations the TIN was found to have the strongest relationship between observed and expected elevation values, R^2 = 92.4% as compared to 64.7% and 62.9% for IDW and OK respectively (figure 5.7).

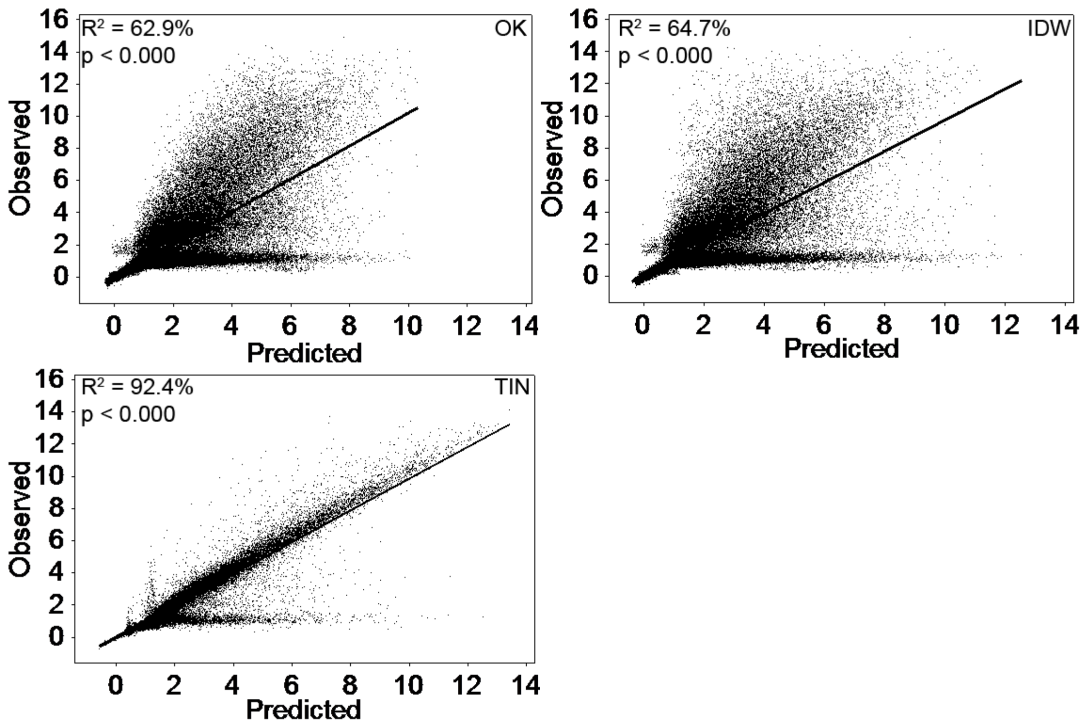

Figure 5.7: Cross validation regression analyses to compare the Triangulated Irregular Network (TIN), Inverse Distance Weighted (IDW) and the Ordinary Kriging (OK) LiDAR elevation (m) interpolations.

The results of the cross validation of the IDW showed an MPE of 4.64 x 10^{-6}m and the RMSE was 0.493. This interpolation underestimated variability in the dataset (Root Mean Square Error Standardised 1.001) (table 5.1). The chosen method for OK produced an interpolation with an MPE of 1.3 x 10^{-4}m and a RMSE of 0.537. The model was found to overestimate the variability in the dataset (Root Mean Square Error Standardised 0.989) (table 5.1).

5.4.2 Model application

5.4.2.1. Plant community elevation categories for the IDW, Kriging and TIN interpolations

Plant communities were modelled using the previously determined elevation ranges (chapter 4) to categorise the TIN, OK and IDW elevation interpolations for the Tahu site. The three interpolations were categorised into six elevation groups as related to the 1st and 3rd quartile heights of the six plant community types (CS, RS, LS, US, TG, and SW) (table 5.2). As noted previously, the upper range of elevation related to the SW category takes into account that last and only returns from the LiDAR elevation data could be reflected off the tops of trees. Tree heights typically do not exceed 5m in this community and therefore, 5m was added to the highest recorded elevation for the SW community.

Table 5.2: Elevation ranges of the six plant community types above mean sea level (BK77 as measured at Kronstadt, in m). The elevation ranges were calculated using a Leica RTK dGPS (adapted from Chapter 4).

Community	Elevation range (m)
CS	-0.03 to 0.07
RS	0.07 to 0.15
LS	0.15 to 0.27
US	0.27 to 0.47
TG	0.47 to 0.69
SW	0.69 to 5.85

5.4.2.2 Plant community models derived using the IDW, Kriging and TIN interpolations for Tahu

Plant communities were predicted at the Tahu site by using the IDW, OK and TIN elevation interpolations categorised by elevation ranges. All three models show the TG and SW plant communities predominantly on the west of Tahu (inland), the CS, and RS communities in the far south and east (on the coast) and the LS and US plant communities in between the CS – RS and TG – SW (figure 5.8). The three models showed similar distributions for the plant communities at Tahu although the TIN model offered more detail in its prediction of mosaic plant communities, with more small community patches (figure 5.8). At a coarse scale each of the models exhibited a similar extent of each individual community at Tahu (table 5.4). The IDW and OK models produced more similar predictions in both location and extent (figure 5.8, table 5.4) for the plant communities when compared to the TIN model. The greatest difference in predicted plant community extent between the different models was seen in the US plant community. Both the IDW and OK models predicted a greater extent of US than in the TIN model. This can be seen in the large uninterrupted patch of US in the southern section of Tahu predicted in both the IDW and OK models (figure 5.8). The TIN model also predicted a greater extent of TG than either the IDW or OK plant community model.

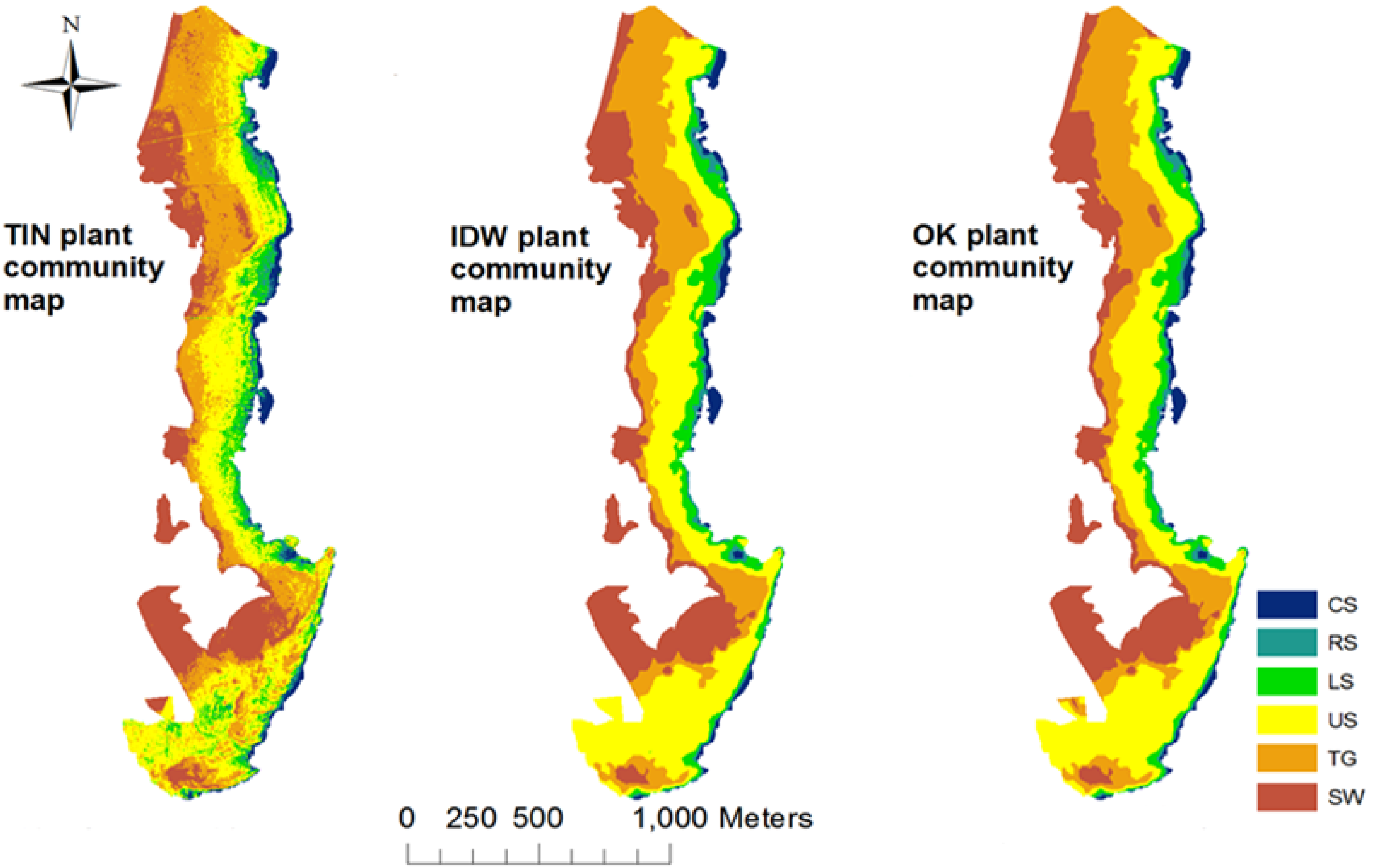

Figure 5.8: The outputs from the TIN, IDW and OK plant community models for the Tahu site.

In the TIN model there are patches of TG predicted in the south of Tahu, whereas these areas are predicted to be US in the IDW and OK models (figure 5.8).

The IDW model was able to correctly predict plant community location in 46.7%-80% of quadrats, with some of the plant communities more readily identified (table 5.3). The OK interpolation produced similar results to IDW for the lower elevation plant communities, CS, RS and LS (table 5.3), and correctly predicted plant community location in 43.3%-80% of all quadrats.

Table 5.3: Percentage of quadrats of each plant community type correctly identified using the IDW, OK and TIN models in the Tahu site. A Fleiss' Kappa coefficient (κ) was used to assess the reliability of agreement between the expected and observed plant community type for each interpolation.

Interpolation	Observed community	Expected Community						Correctly identified
		CS	RS	LS	US	TG	SW	
IDW	CS	66.7	20.0					66.7
	RS	33.3	70.0		40.0			70.0
	LS		10.0	80.0		46.7		80.0
	US			13.3	60.0		13.3	60.0
	TG			6.7		53.3	40.0	53.3
	SW						46.7	46.7
		κ coefficient 0.55						Mean 62.8
OK	CS	66.7	20.0					66.7
	RS	33.3	70.0		33.3			70.0
	LS		10.0	80.0	23.4	46.7		80.0
	US			13.3	43.3	0.0	6.7	43.3
	TG			6.7		46.7	40.0	46.7
	SW					6.7	53.3	53.3
		κ coefficient 0.52						Mean 60.0
TIN	CS	86.7	6.7					86.7
	RS	13.3	93.3.					93.3
	LS			80.0	53.3	26.7		80.0
	US			20.0	46.7	20.0		46.7
	TG					53.3	33.3	53.3
	SW						66.7	66.7
		κ coefficient 0.65						Mean 71.1

The TIN model was able to correctly predict plant community at Tahu in 46.7%-93.3% of the quadrats and had the highest mean percentage (71.1%)

of correctly identified quadrats compared to both the IDW (62.8%) and OK (60.0%) models (table 5.3). A Fleiss κ coefficient was calculated for all three interpolation models to compare observed and expected distribution of the plant communities. The OK and IDW models were both shown to be in moderate agreement with the field data, with κ coefficients of 0.52 and 0.55 respectively (Landis & Koch, 1977). The TIN model, however, was shown to be in substantial agreement with the observed data, with a κ coefficient of 0.65 (table 5.3) (Landis & Koch, 1977).

For completeness a further test of the IDW, OK and TIN derived plant community models was run without the dGPS – LiDAR calibration data. The results clearly demonstrated that without the addition of dGPS post-processing corrections, the TIN and OK models were only in slight agreement with the observed plant communities, with κ coefficients of 0.15 and 0.05 respectively (Landis & Koch, 1977). IDW was found to be in no agreement with a κ coefficient of 0.00 (Landis & Koch, 1977).

Table 5.4: The extent (ha) of the plant communities predicted by each model at Tahu.

Model	Plant community area (ha)						
for Tahu	CS	RS	LS	US	TG	SW	Total
TIN	4.1	8.5	22.8	35.5	16.8	16.7	104.4
IDW	3.8	8.3	21.6	40.5	12.1	18.1	104.4
OK	3.9	8.4	21.7	37.9	14.2	18.3	104.4

5.4.3 Model validation at Matsalu and Kudani

The TIN method produced the best interpolation of the LiDAR elevation data and also produced the most accurate predictive plant community model. It was thus used to predict plant communities in two geographically separate new sites: Matsalu and Kudani. An independent ground truthing of both sites was then undertaken to assess how robust the model was and validate the method.

5.4.3.1 Prediction at Matsalu

The prediction model developed for the Matsalu coastal wetland was able to correctly predict the RS, LS, US and TG communities with between 80%-100% accuracy (table 5.5) within the Matsalu site (figure 5.11). However, the model was unable to correctly predict the location of either the CS or SW communities at Matsalu. The patches predicted to support CS in the model were observed in the field to comprise species indicative of the RS community (table 5.5). The SW community, which was predicted in a number of cases, was found to be absent from the Matsalu site during the ground-truthing, as elevation range predicted to be SW was TG in all cases (table 5.5). Overall the observed and expected communities were still found to be in moderate agreement within the Matsalu site (K coefficient = 0.53).

Table 5.5: Percentage of quadrats of each community type that were located within each LiDAR elevation derived TIN plant community in the Matsalu and Kudani sites.

Site	Observed Community	Expected community CS	RS	LS	US	TG	SW	Correctly identified
Matsalu	CS	0.0						0.0
	RS	100.0	100.0					100.0
	LS			80.0	6.6			80.0
	US			20.0	86.7			86.7
	TG				6.6	100.0	100.0	100.0
	SW						0.0	0.0
		κ coefficient 0.53					Mean 61.1	
Kudani	CS	100.0						100.0
	RS		100.0					100.0
	LS			80.0	20.0	13.3		80.0
	US			20.0	60.0	26.7	6.7	60.0
	TG				20.0	60.0	13.3	60.0
	SW						80.0	80.0
		κ coefficient 0.81					Mean 80.0	

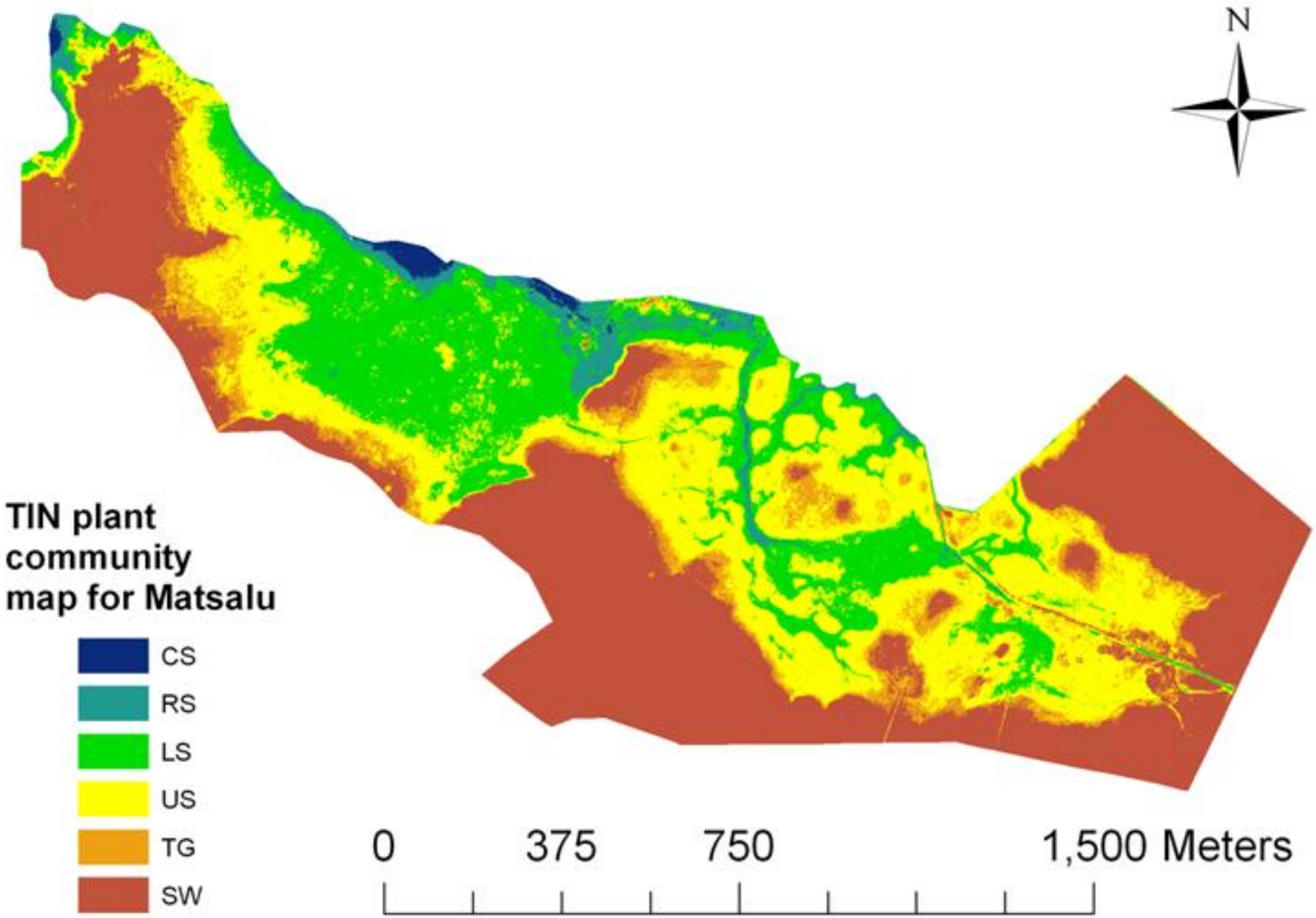

Figure 5.9: The output from the TIN derived plant community model for the Matsalu site.

The model output in figure 5.9 shows that the majority of the CS and RS plant community patches occur in the north west of the site adjacent to the sea. However, all of the predicted CS plant community patches adjacent to the sea were actually RS. Figure 5.9 also shows a patch of RS running from the north of the site through to the centre. This patch actually represented a shallow channel with RS associated vegetation. The large areas predicted to be covered with SW to the south and east of the site (figure 5.9) were all occupied by the TG plant community, as SW was absent from the site. The model output in figure 5.9 showed some zonation, in that the higher elevation plant communities are further from the sea than the majority of the lower elevation plant communities, but the mosaic nature of the plant communities is also apparent.

5.4.3.2 Prediction at Kudani

The predictions for the Kudani coastal wetland were accurate in 60% - 100% of cases, dependant on plant community type (table 5.5). As with the other models for all sites, the majority of the incorrectly determined plant communities were located within proximal elevation ranges (table 5.5). The expected and observed plant communities were found to be in almost perfect agreement (κ coefficient = 0.81) (Landis & Koch, 1977). Moreover the model had a mean accuracy of 80%. The lowest model accuracy was observed in the US and TG communities with values of 60%. In the CS and RS communities the model was able to correctly predict plant community type in 100% of the cases.

The output from the model for Kudani shows that the CS and RS plant community patches are all located in the north of the site. The indent section in the north of the site (figure 5.10) is the location of Võõlameri, a small brackish lake directly connected by a narrow channel to the Baltic Sea in the north (figure 5.1). The south west and south east of the site are covered by the TG and SW plant communities and there appears to be a depression, with LS and to a lesser extent US from the north west to the south east. This depression was previously a channel connecting Võõlameri in the north, and hence the Baltic Sea, to Tahu Bay in the south (Puurmann *pers. comm*. 2009)

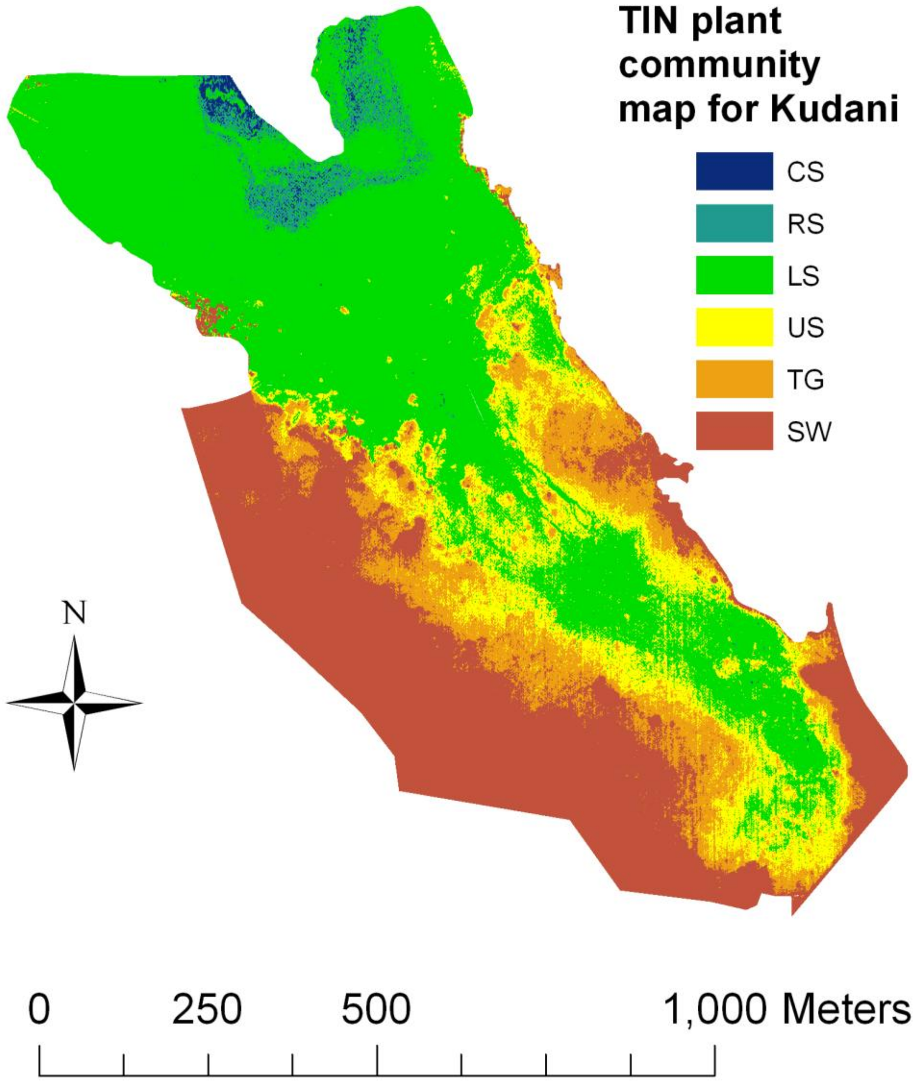

Figure 5.10: The output from the TIN derived plant community model for the Kudani site.

5.4.3.3 Model prediction of the extent of plant communities at Matsalu and Kudani

The plant community models for the Kudani and Matsalu sites (figures 5.9 and 5.10) were used to assess the extent of each of the six studied plant communities (table 5.6).

At the Matsalu site the SW community was predicted to cover 71.7 ha (table 5.6). However, in the field SW was absent and supported TG giving an actual area covered by TG as 85.4 ha, which was the largest extent of all the plant communities. The areas predicted to be CS, totalling 11.6 ha, were actually RS and hence RS covered a total 17.8 ha, the smallest extent of the plant communities at Matsalu.

Both the CS and RS plant communities covered a small area at Kudani, 2.3 and 2.7 ha respectively. At Kudani the plant communities with the greatest extent were LS and SW with 31.6 and 23 ha respectively.

Table 5.6: Predicted extent (ha) of the plant communities at Kudani and Matsalu according to the TIN derived plant community model. Ground truthing showed that the CS and SW communities were in fact absent from the Matsalu site. Field inspection showed that these areas were currently covered by RS and TG respectively.

	Plant community area (ha)						
Site	CS	RS	LS	US	TG	SW	Total
Kudani	2.3	2.7	31.6	9.8	9.2	23.0	78.7
Matsalu	11.3	6.2	36.9	50.6	13.7	71.7	189.8

5.5. Discussion

The analysis in chapter 5 has developed a useful and accurate predictive plant community model for Estonian coastal wetlands. Assuming a geomatic approach, this study has shown that LiDAR elevation data on their own are inaccurate for use in such models, but by post-processing LiDAR data with dGPS derived correction data, elevation accuracy can be significantly improved. The results of this chapter have shown that while all three commonly used interpolation techniques produce accurate DEM's, the TIN interpolation produces the most robust interpolation, and further to this, is the most accurate predictive plant community model. This study has shown that using a combination of ecological and geomatic techniques, a rapid and accurate assessment of the location and extent of plant communities can be undertaken.

5.5.1 LiDAR elevation accuracy

The results of the comparison between the LiDAR and dGPS elevations are consistent with other studies for both non-forested (Sadro *et al*., 2007) and forested environments (Hodgson & Bresnahan, 2004). Interestingly previous studies have shown that sudden breaks in terrain such as cliffs and slopes, or limited LiDAR ground penetration caused by tree and shrub canopies can be the cause of large scale fluctuations in the accuracy of LiDAR data (Huising & Perreira, 1998; Hodgson & Bresnahan, 2004; Bater & Coops, 2009; Guo *et al*., 2010). In Estonian coastal wetlands however, as in many wetlands, changes in topography are subtle and slope angle is negligible.

In this study, the lowest elevation points are no greater than 0.07m below m.s.l. and the highest points are rarely above 2m, while the planimetric distance between the highest and lowest points is typically between 100m and 1000m (Ward *pers*. *obs*.). The greatest source of error between LiDAR and dGPS derived elevations was found in habitats dominated by shrub species e.g. *Juniperus communis* and *Pinus sylvestris*. Hodgson *et al*. (2003), in a study of six land cover categories (Pavement, Low Grass, High Grass, Brush/ Low Trees, Evergreen Forest and Deciduous Forest) also found that the greatest errors from LiDAR elevation data were found in scrub vegetation (Brush/ Low Trees) due to limited LiDAR penetration of the canopy. The footprint of the LiDAR data used in this study is 0.54m, and a previous study by Hopkinson (2007) has shown that larger LiDAR footprints have reduced canopy penetration. Therefore, a smaller footprint could improve the LiDAR error values for plant communities dominated by shrubs as smaller footprint LiDAR data are more likely to pass through the canopy (Hodgson & Bresnahan, 2004).

In the CS, RS, LS, US and TG plant communities there was a mean elevation difference of +0.177m (0.016m SD) between the LiDAR and dGPS derived elevations. This was most likely to originate from ground and last returns of the LiDAR being derived from the top of the grass and forb plants that characterise the vegetation. Sadro *et al*., (2007) found similar results in a study on various open salt marsh vegetation types, with mean RMSEs of +0.18m for *Salicornia spp*, +0.13m for *Jaumea spp* and +0.17m for salt grass vegetation. However, in the Sadro *et al*. (2007) study there was no attempt made to model plant communities using the LiDAR data, instead it was used to assess inundation of the different wetland zones. This study has shown that without the elevation correction for the LiDAR as calculated from the dGPS data, the ability of the models to accurately predict the location of plant communities was considerably decreased.

5.5.2 LiDAR elevation interpolation

The results of the comparison between the TIN, IDW, and OK interpolation methods have demonstrated that the TIN method was the most accurate interpolator of medium point density LiDAR data for Estonian coastal wetlands. In the cross validation of the IDW, TIN and OK interpolations the regression analysis showed that all three interpolators had a significant relationship between the observed and expected values. The TIN interpolation however, exhibited the highest R^2 value, 92.4%, compared with 62.9% and 64.7% for the OK and IDW interpolations respectively, and was hence the most accurate elevation interpolator. Similarly the results showed that of the three plant community models, the TIN model was most readily able to predict plant community type by elevation, followed by the IDW and OK models respectively.

Several studies have assessed a variety of techniques for interpolating LiDAR data (Lloyd & Atkinson, 2002a; Lloyd & Atkinson, 2002b; Chaplot *et al*., 2006; Bater & Coops, 2009). All of these studies made an assessment of the IDW and OK methods of interpolation, although with no consensus as to which is the more accurate interpolator. Both of the Lloyd & Atkinson (2002a & 2002b) studies found that the OK technique produced the most accurate interpolation, whereas Chaplot *et al.* (2006) suggests that IDW is better, although all of these studies neglected to include TIN in their assessments. However, Delaunay linear TIN interpolation is commonly used for LiDAR elevation modelling due to its accuracy and efficiency of data storage in comparison with other techniques (Morgan & Habib, 2002; Hodgson *et al*., 2003; Hodgson & Bresnahan, 2004; Webster & Dias, 2006). Both TIN and IDW interpolation methods are very fast and simple interpolators in comparison to OK (Lo & Yeung, 2002; Longley *et al*., 2005; Guo *et al*., 2010). This study has shown that with close, evenly spaced data points, such as LiDAR, exact deterministic techniques, such as TIN and to a lesser extent

IDW, performed better than the geostatistical technique OK. These results have been found by previous authors comparing different techniques for LiDAR elevation interpolation (Bater & Coops, 2009; Drouin & Saint-Laurent, 2010). In the Drouin & Saint-Laurent (2010) study, assessing the accuracy of IDW, TIN, Universal and Ordinary Krige interpolation techniques using LiDAR derived elevation in areas with low relief, the TIN technique produced the smallest errors, whilst IDW and Universal Krige produced the highest. Bater & Coops (2009) found that using dense, regularly spaced point data the TIN method produced a better interpolation than IDW, and further to this the TIN method was better able to represent data in areas with sudden breaks in terrain. However, Bater & Coops (2009) neglected any form of Kriging from their assessment.

Where there is a high sampling density, as used in this study (e.g. $0.45m^2$, Eesti Maa-met, 2011), TIN (Drouin & Saint-Laurent, 2010) and IDW (Lloyd & Atkinson, 2002b; Anderson *et al*., 2006; Chaplot *et al*., 2006; Liu *et al*., 2007) methods are often found to be more efficient at producing interpolations than OK. This study confirms this, and infers that the use of Kriging should be limited to less regular and densely spaced point data interpolation. The strength of Kriging methods has previously been shown to be apparent in surface models derived using less dense point data (Lloyd & Atkinson, 2002a; Guo *et al*., 2010).

5.5.3 LiDAR elevation-derived plant community modelling

The static correlative plant community model developed at Tahu and tested at Matsalu and Kudani was shown in a model validation to be able to accurately predict plant community type at Kudani and to a lesser extent Matsalu. A difficulty occurred in the Matsalu site using elevation values to identify the CS and SW communities. In the Tahu and Kudani sites, the CS community was located at the seaward edge of the wetland in slightly deeper water than the RS community. In the Matsalu site however, this littoral zone was dominated by the RS community. This study site is located in Matsalu Bay and is more open to wave activity than both the original Tahu site and the modelled Kudani site. Coops & van der Velde (1995) have shown that

Phragmites australis, the main indicator for the RS community, is more resistant to wave energy than the flexible stems of *Bolboschoenus maritimus*, the main indicator for CS. Indeed, typically in higher energy coastal environments in the Baltic Sea, *P*. *australis* is the dominant plant species (Coops *et al*., 1991). Therefore it is suggested that the greater wave activity present at Matsalu could explain the absence of the CS community. The importance of this as a factor affecting littoral vegetation distribution would be an interesting area for future research.

The absence of the SW community from the Matsalu coastal wetland is probably due to a longer period of grazing management at this site when compared with the Tahu and Kudani sites. Matsalu has been almost continuously grazed for over 90 years at low grazing intensity (1 $LU.ha^{-1}$) (Lepik, *pers*. *comm*. 2010). However, the north of the Tahu site has been grazed only for the last 26 years at low intensity and the south of the site has been lightly grazed for only 8 years (Berg, 2009). The Kudani site has been grazed at a medium intensity for the past 10 years (Puurmann *pers*. *comm*. 2009). The periods during which the Tahu and Kudani wetlands were abandoned evidently allowed the higher elevations of the sites to become overgrown with SW associated vegetation. Following the reinstatement of management some of these patches were cleared, leading to a partial return of the TG community.

One of the problems associated with correlative modelling is that they do not generalise or scale well (Hoffman, 2006). In sites with different history of land use or different climatic effects such as Matsalu, problems arise in modelling, although the model still provided a reasonably accurate assessment of the plant communities (Kappa = 0.53). However, the limitations can be simply overcome. The SW community can be distinguished from the TG community, and CS from RS using detailed orthophotos. Using this refinement the model produced a Kappa value of 0.93 at Matsalu. If the model is to be used for wider applications and in other landscapes it may be prudent to examine the potential to integrate different management or atypical physical conditions within the model design. Other studies have combined LiDAR modelling with orthophoto visual verification in order to effectively assess land cover

(Schenk & Csatho, 2002; Koukoulas & Blackburn, 2005; Packalen & Maltamo, 2007). Therefore in order to verify the LiDAR derived elevation model, a visual assessment of vegetation should be considered in future. However, within the Estonian coastal wetland application the model delivers very good results, with an acceptable trade off between model complexity and applicability.

5.6 Conclusions

This study has shown that LiDAR elevation data can be used to accurately estimate the location and extent of plant communities that exist at different elevations in Estonian coastal wetlands. When accurate dGPS data are used to calibrate the model, the elevation accuracy of the LiDAR data is significantly improved and this in turn improves model validity. The study also found that the Delaunay linear TIN interpolation method is most suited to dense, evenly spaced LiDAR data points for plant community predictions in relatively flat coastal wetlands. Both the IDW and OK were found to be accurate interpolators, although less so when compared to the TIN. Both IDW and OK were less able to accurately predict plant community than the TIN derived model. In order to improve the accuracy of the models, either walk over surveys, orthophotos or a combination of both should be used, especially to verify plant community type. These methods provide a rapid and effective method to assess predictive quality of the plant community model. However, it should be noted that the model is unable to predict the presence of the OP plant community, which occurs at similar elevation range to TG. Thus the predicted extent of TG is likely to be an overestimate.

LiDAR elevation data are widely available both commercially and through government sources. Using the methodology developed in this study a rapid landscape assessment for open environments has been successfully developed and offers a framework to forecast environmental changes in a variety of different habitats. These environmental changes could include modelling the effects of sea level rise on coastal wetland communities or increased inundation in floodplain meadows.

6 The use of ^{210}Pb and ^{137}Cs dating for calculating sediment accretion rates in Estonian coastal wetlands

6.1 Preamble

This chapter examined three different radionuclide dating techniques for assessing sediment accretion rates in Estonian coastal wetlands in order to provide accurate information for use in the dynamic plant community model in chapter 7. Two of the methods involved different calculations using ^{210}Pb activity down the sediment profile to assess accretion rates, and the third used ^{137}Cs impulse dating to validate the results derived from the ^{210}Pb methods. Analysis was made of the sediment particle composition as well as the soil organic matter content as both of these factors can influence post depositional mobility of ^{137}Cs in the soil. Assessment of sediment accretion rates took place for two Estonian coastal wetland sites, Matsalu and Tahu, by taking cores in the lower elevation LS, and the higher elevation TG plant community patches. The sediment accretion rates for all three methods were found to be in agreement. A study of the literature showed that periods of increased sediment accretion, as identified in this study, were related to periods of increased storminess and prolonged elevated sea level. Prior to this study there were no available data for sediment accretion rates in Estonian coastal wetlands.

6.2 Introduction

This research has shown that there is a relationship between elevation and edaphic factors and that these are important determinants of plant communities in Estonian coastal wetlands. Moreover, elevation was shown to be a critical factor affecting the location of the plant communities within the wetland mosaic. The relationship between plant community type and elevation was used as a measureable resource gradient central to the development of a predictive plant community model. The model developed in chapter 5 was found to accurately predict the location and extent of plant communities in Estonian coastal wetlands, particularly when combined with orthophotos and ground truthed data. This static correlative model provides a basis for a dynamic correlative model, which can be used to predict the future location and extent of plant communities in Estonian coastal wetlands under a variety of scenarios. The threat of sea level rise to Estonian coastal wetlands has not previously been assessed in detail. However, a model predicting the future effects of sea level rise on coastal wetlands based on an elevation gradient, must take into account other factors that will affect the elevation above mean sea level. In Estonian coastal wetlands there are two main factors which will influence future elevation, isostatic uplift and sediment accretion. Isostatic uplift rates are known for the whole of Estonia (Vallner *et al*., 1988; Eronen *et al*., 2001), however, there are no sediment accretion data available for Estonian coastal wetlands.

6.2.1 Background

Anthropogenically induced climate change is one the greatest threats to biodiversity for the 21st century (Thomas *et al*., 2004). The threat of sea level rise will particularly affect low lying countries. Estonia with its predominantly low, flat and relatively long coastline is particularly at risk. Coastal wetlands cover many of the sheltered coasts of Estonia, supporting some of the highest concentrations of birds, protected mammals and rare plants in Europe (Joyce & Burnside, 2004). Many of these wetlands suffered from degradation due to overgrowth by *Phragmites australis* and woodland due to the cessation of management following the collapse of the Soviet Union (Truus & Tõnisson, 1998; Puurmann & Randla 1999; Rannap *et al*., 2004; Berg, 2009). However, with increased government support during the last ten years many of these wetlands are being managed again.

Estonian coastal wetlands rarely occur at elevations greater than 2m above mean sea level and their plant community types are located in narrow elevation ranges and influenced by hydrology (Ward *et al*., 2010; chapter 4). Sea level rise, therefore, poses a new and potentially damaging threat to these internationally important wetlands. Small increases in sea level could potentially affect the location and distribution of the plant communities ultimately leading to the decrease or loss of some important plant communities. Global sea level is expected to rise by between 0.18-0.59m, by 2099 (IPCC, 2007). Even the lowest predicted rise in sea level could radically alter the location of the plant communities within these wetland mosaics. This may also decrease the extent of the plant communities that occur in the upper reaches of these wetlands due to coastal squeeze.

Much of the land around the Baltic Sea is currently experiencing post-glacial isostatic uplift. In some areas, such as the north of the Bay of Bothnia, uplift is in the region of 9mm/year (Eronen *et al*., 2007), although in northwest Estonia it is between 2-3mm/year (Vallner *et al*., 1988). However, a sea level rise of 0.59m by 2099 equates to 6.4mm per year (IPCC, 2007), which far exceeds the offset provided by land uplift. Thus, present day sea level rise is likely to be partially compensated by isostatic uplift in Estonia (Kont *et al*., 2007).

Estonian coastal wetlands are predominantly depositional with sediment influx during periods of inundation. This deposits fine sediments such as silts and clays due to the typically low energy nature of the water influx (Puurmann & Ratas, 1990). However, during storm surges inundation occurs on a larger scale and is likely to wash coarser sediments such as fine, medium and coarse sands further inland (Mitsch & Gosselink, 2000). Sediment accretion in coastal wetlands can offset the effects of sea level rise, although in the coastal wetlands of Estonia rates of deposition are unknown.

A common method to assess accretion rates uses radionuclides to date horizons in the sediment profile. For example both ^{210}Pb and ^{137}Cs can be used to date sediments over timescales up to 150 years. Reliable data concerning past accretion rates can, with other important information such as sea level and meteorological data, be used to estimate future sediment accretion.

The aim of this study was to determine sediment accretion rates in Estonian coastal wetlands over the past 100 years in order to provide an indicator of future sediment accretion to improve the quality of the dynamic predictive plant community model. The main objectives were: 1) to determine sediment accretion rates using impulse radiometric dating via an artificial radionuclide, ^{137}Cs, and assess dating reliability; 2) to determine mean sediment accretion rates using the Constant Flux:Constant Sedimentation (CF:CS) method for ^{210}Pb; 3) to determine sediment accretion rates over discrete time periods using the Constant Rate of Sedimentation (CRS) method for ^{210}Pb; 4) to compare the sediment accretion rates for the different methods and describe any changes in sediment accretion with relation to past fluctuations in mean sea level and increased storminess.

6.3 Study sites and methodology

6.3.1 Study sites

Sediment accretion rates for the Tahu and Matsalu coastal wetlands were assessed for both the frequently inundated LS, and the rarely inundated TG plant communities. The altered hydrological regime of the Kudani coastal wetland severely limited sediment accretion from allochthonous sources, and hence was excluded from this study.

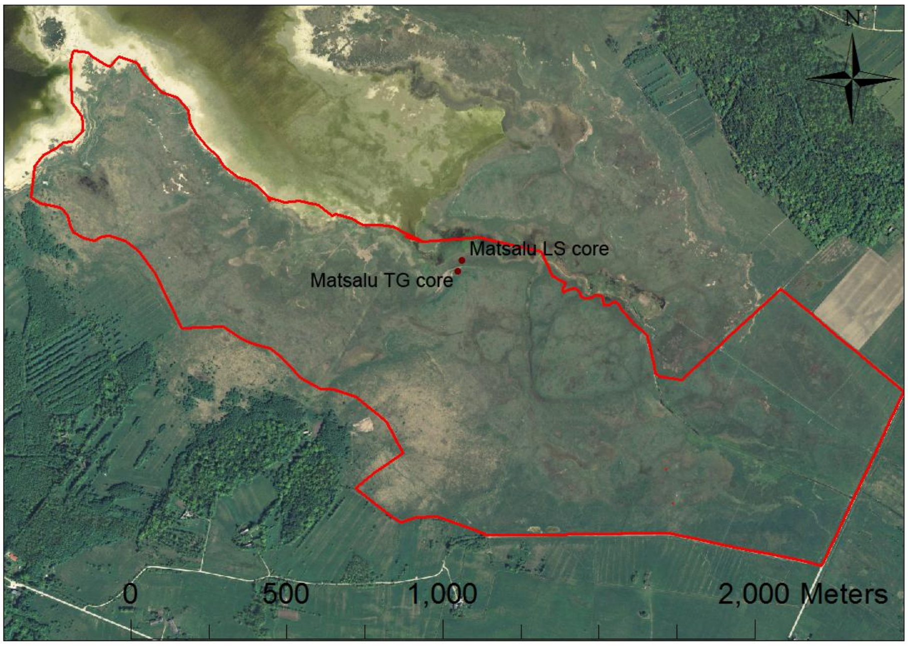

Figure 6.1: Location of the sediment cores at the Matsalu study site. The LS core was located in the north of the coastal wetland within 30m of the coastline. The TG core was located directly inland from the LS core.

The Matsalu site is located on the south of Matsalu Bay. This bay covers 67km^2 and has a large freshwater input from the Kasari River, with a catchment area of 3214km^2 (Kotta *et al.*, 2008). Annual freshwater input is seven times greater than the volume of the bay. This river contributes a large amount of the sediment input to the bay and the adjacent coastal wetlands. At Matsalu the core removed from the LS plant community was within 30m of the shoreline at an elevation of 0.04m above m.s.l. and the core from the TG plant community was located at the nearest TG patch, 20m directly inland of the LS core at an elevation of 0.69m above m.s.l. (figure 6.1).

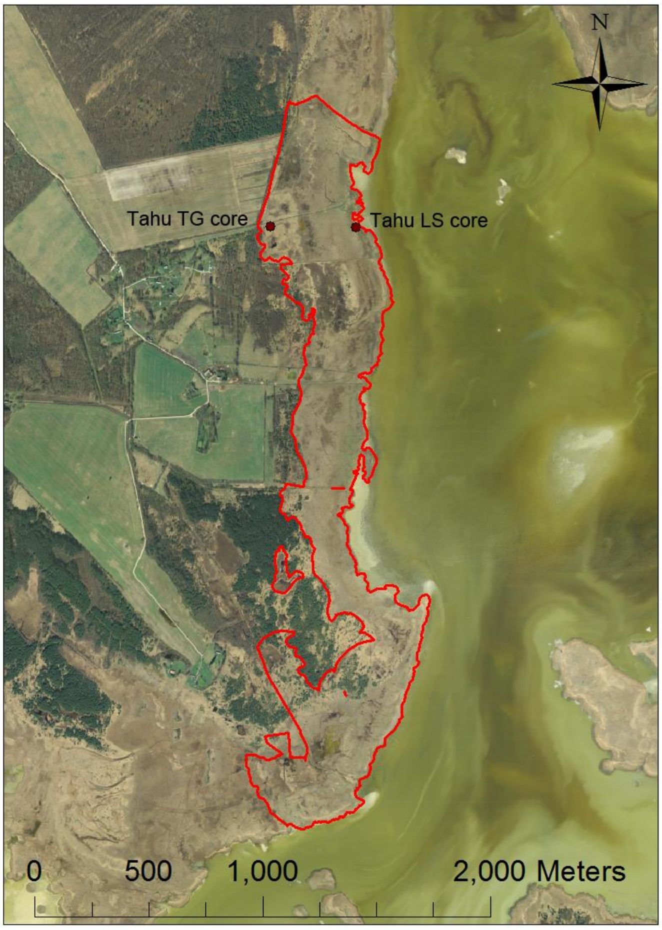

Figure 6.2: Location of the sediment cores at the Tahu study site. The LS core was located 30m from the seaward edge of the north of the coastal wetland. The TG core was located 250m directly inland of this.

The Tahu site is located on the north west of Haapsalu bay and covers 50km^2. This bay has moderate freshwater input from the Taebla River, which has a catchment of 107km^2 (Kotta *et al*., 2008). The hydrological conditions are mainly influenced by the exchange of water with Väinameri and hence sediment accretion is a mixture of riverine sediments and marine (Kotta *et al*., 2008). At Tahu the core removed from the LS community was within 30m of the shoreline at an elevation of 0.07m above m.s.l. and the TG community core was removed from the nearest TG patch located 250m directly inland from the LS core at an elevation of 0.70m above m.s.l. (figure 6.2).

Elevation for each core was recorded using a dGPS with a mean vertical accuracy of 0.02m. Elevation was recorded above mean sea level in metres as measured at Kronstadt using the BK77 elipsoid.

6.3.2 Field methods

Sediment cores were collected by digging trenches and removing the core from the edge of the trench, placing the core in a monolith tray and packing and sealing it for transport to the freezer. Suitable care was taken to not contaminate, degrade or otherwise change the state of the cores. The cores were stored at -26°C in sealed containers to preserve them until ready for laboratory preparation and analysis.

At Matsalu and Tahu two cores were taken at each site, each 0.08m diameter and 0.40m deep. This depth was expected to cover the last 150 years of sediment accretion, which is the maximum limit of ^{210}Pb dating.

6.3.3 Laboratory methods

The cores were thawed in the laboratory overnight and then sliced into 0.01m depth increments (Cundy *et al*., 2000; Mizugaki *et al*., 2006). This method was selected so as to obtain high resolution results, and to ascertain detailed sediment accretion rates. The samples were weighed prior to and after drying in order to assess soil moisture content through the profile. The soil moisture data were used to calculate dry bulk density for the CRS analysis. Drying was conducted in an oven at 40°C until constant weight was achieved (Guebuem *et al*., 2004; Mizugaki *et al*., 2006).

Following drying, each of the samples were lightly ground using a pestle and mortar to disaggregate the soils.

6.3.3.1 Gamma spectrometry

Each core sub sample was placed in a Canberra well type ultra-low background HPGe gamma ray spectrometer to determine the activity of the

^{137}Cs, ^{210}Pb and ^{214}Pb. The spectra were accumulated using a 16k channel integrated multichannel analyser. Spectral analysis was conducted using the Genie 2000 system. Energy and efficiency calibrations were carried out using bentonite clay spiked with a mixed gamma-emitting radionuclide standard, QCYK8163, and checked against an IAEA certified sediment reference material (IAEA 135). Detection limits of radionuclides are dependent on age, radionuclide gamma energy, count time and sample mass. To achieve maximum quality of data within a minimum time period the samples were left counting until detection error was ≤5% for all the relevant radionuclides. Typically each sample count time was between 24 and 48 hours.

6.3.3.2 Laser particle size analysis

The same method as chapter 4 was used to determine particle size distribution in each 0.01m core slice. A Malvern 2000 Laser Particle Size Analyser, graded according to the Wentworth scale, was used to identify the sand, silt and clay fractions.

6.3.3.3 Soil organic matter content analysis

In order to determine organic matter content for each 0.01m core slice the loss on ignition method was used (Schulte, 1995). Each sample was oven dried to constant weight at 40°C in order to remove moisture from the soil (Ball, 1963). The samples were then placed in a muffle furnace at 360°C for 24 hours and reweighed in order to calculate organic matter content using the calculation below.

$$\text{Organic matter content} = \left(\frac{\text{Mass of oven dried soil-Mass of soil following ignition}}{\text{Mass of oven dried soil}}\right) \times 100$$

6.3.4.1 Sediment accretion rate derivation for ^{137}Cs impulse dating

The calculation, shown below, for estimating sediment accretion rates uses the number of years since a known ^{137}Cs input event occurred and measures the amount of sediment that has accumulated following that event.

$$\text{Soil accretion rates} = \frac{\text{Depth of subsurface } ^{137}\text{Cs activity maximum (mm)}}{\text{Date sample taken - date of known } ^{137}\text{Cs production event}}$$

Two events were used for these cores: pre 1963 weapons testing and the Chernobyl nuclear reactor meltdown in 1986, which previous studies have shown are the second highest and highest ^{137}Cs input events in north east Europe respectively (Callaway *et al*., 1996; Rosen *et al*., 2009). Error values are calculated using the depths either side of the main event. The resultant sediment accretion rates can be used to independently verify the ^{210}Pb dating methods (Cundy & Croudace, 1996).

6.3.4.2 CF:CS (Constant flux: Constant sedimentation)

The CF:CS is the simplest of the methods used to assess sediment accretion using ^{210}Pb and is surprisingly robust. It is applicable when ^{210}Pb excess shows a uniform exponential decline with depth. The exponential decline in activity of ^{210}Pb with depth occurs when sediment accretion has been constant over time with no large scale variation. Average accretion rates can be calculated based on the gradient of the line of least squares regression of the natural logarithm of ^{210}Pb excess against depth. In order to calculate supported ^{210}Pb, ^{214}Pb in all samples down the core was calculated and then the mean was found. ^{214}Pb is the parent material of ^{210}Pb and in secular equilibrium, although it is typically only found in soils. Therefore ^{214}Pb can be used to estimate the level of ^{210}Pb already in the sediment and not from atmospheric deposition. Note, however, that the samples were not counted in sealed vials and hence the radionuclides may not be completely in secular equilibrium. The base level of ^{210}Pb was removed from the total ^{210}Pb to leave the unsupported ^{210}Pb or excess ^{210}Pb in the sediment which was used to date the sediment (figure 6.3).

The CF:CS method calculated average sediment accretion rates for the sediment cores by dividing the decay constant of ^{210}Pb by the gradient of the log normal line of the excess ^{210}Pb down the sediment core.

In order to calculate the gradient of the line a least squares regression analysis was used. The sediment accretion rates were derived from the calculation below.

$$\text{Soil accretion rates} = \frac{\text{decay constant } ^{210}\text{Pb}}{\text{gradient of Ln line of the excess } ^{210}\text{Pb in the soil}}$$

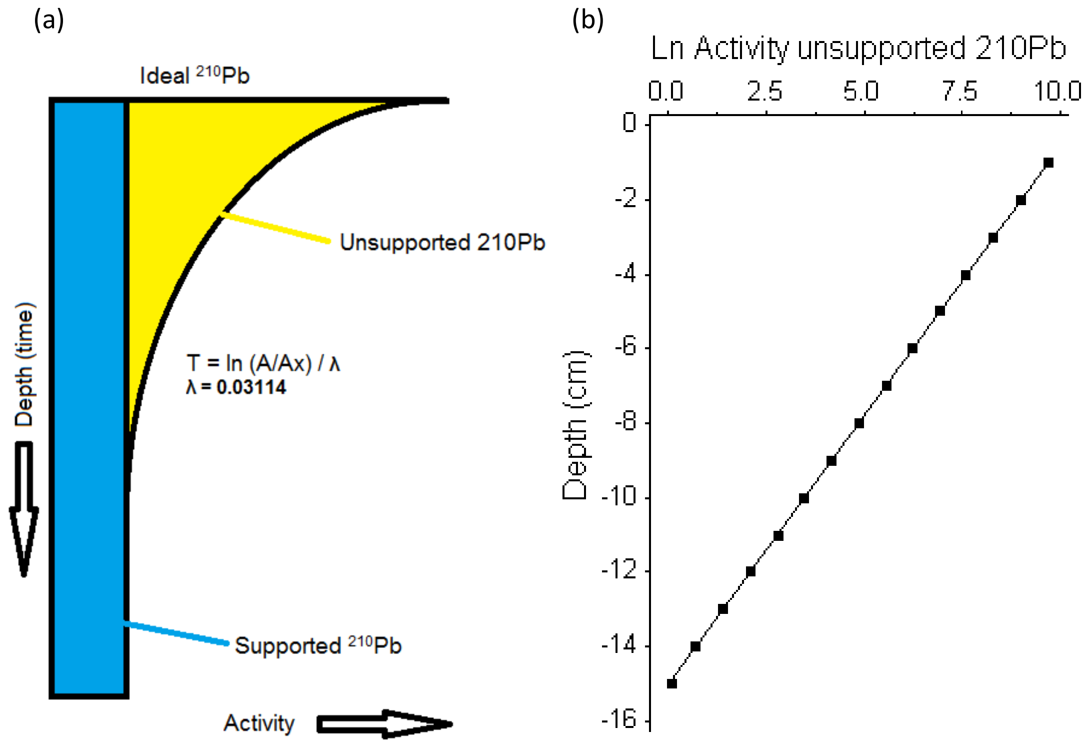

Figure 6.3: Ideal excess and the log normal plot of excess ^{210}Pb against depth. (a) shows the ideal supported (in blue) and the unsupported, or atmospherically derived (in yellow) ^{210}Pb activity, and (b) the log normal ideal excess ^{210}Pb activity with depth, which is plotted against a least squares regression line.

6.3.4.3 Constant rate of supply method (CRS)

The assumption of this method is that a constant rate of supply of ^{210}Pb from atmospheric sources is the major source of ^{210}Pb in the sediment profile. The method was used to calculate accretion rates at specific dates. These were linked to fluctuations in sediment supply by examination of historical meteorological and hydrological data. The age of sediment at depth x was calculated using the formula below. The inventory of each sediment slice was calculated using ^{210}Pb excess within the slice multiplied by the dry bulk density of the slice. Ages and hence sediment accretion rates were then calculated by using the calculation below.

$$\text{Age at depth x} = \left(\frac{1}{\text{decay constant of } ^{210}\text{Pb}}\right) \text{x} \ln\left(\frac{\text{unsupported inventory at depth x}}{\text{unsupported inventory of entire core}}\right)$$

6.4 Results

6.4.1 Soil organic matter analysis

Organic matter decreased down the sediment profile in all of the cores (figure 6.4). In both the LS and TG cores for the Tahu site and the LS core for the Matsalu site the upper layers of the core profile contained maximum organic matter contents of 53%, 57%, and 57% respectively. The TG core for the Matsalu site only contained a maximum of 28% organic matter.

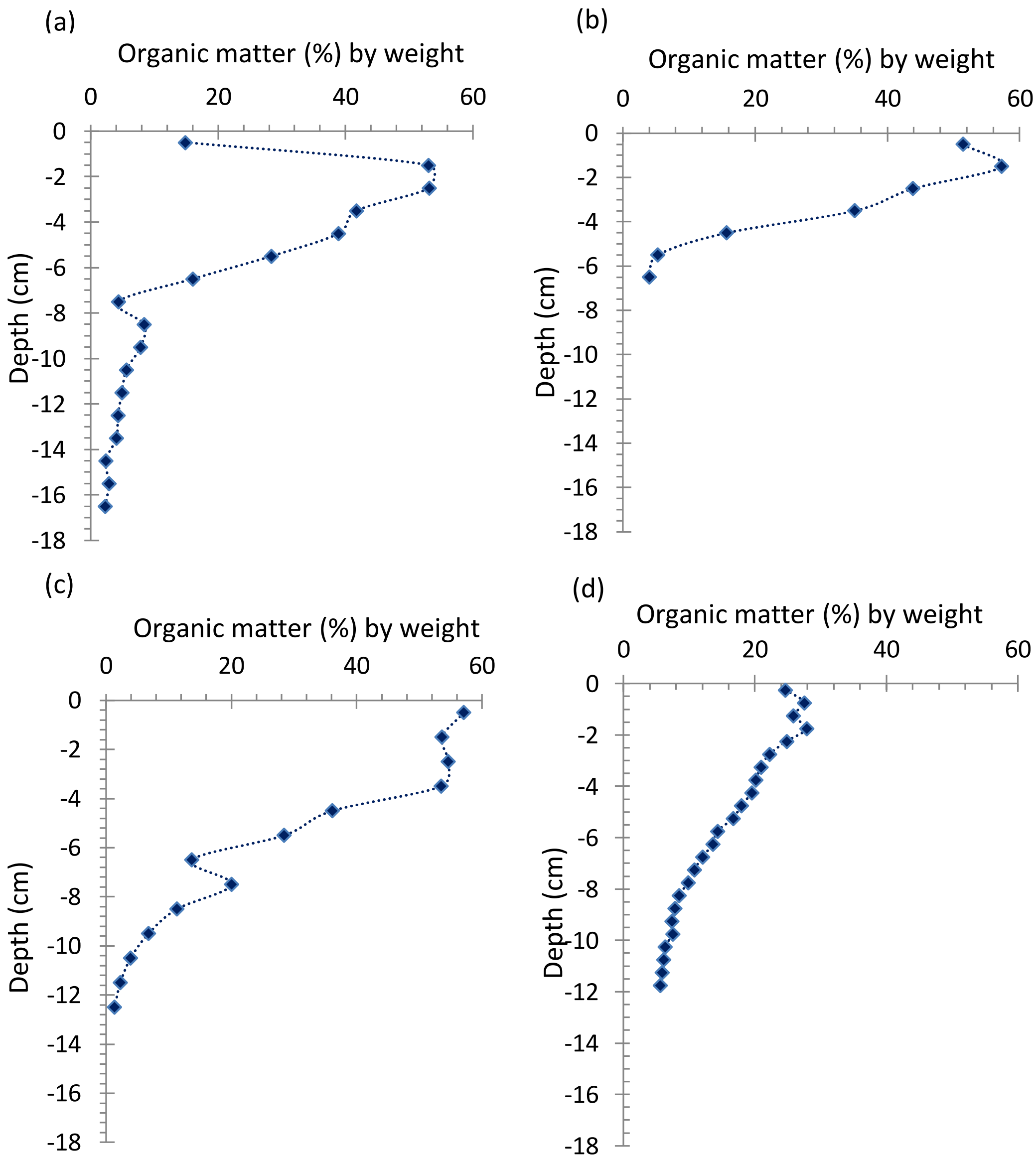

Figure 6.4: Soil organic matter content measured in percentage by weight for the Tahu site in the LS (a) community type and TG (b) and the Matsalu site in the LS (c) and TG (d) communities. The data was obtained from the sediment slices down the each core.

6.4.2 Sediment particle size analysis

All the cores contained clay, silt and sand throughout (figure 6.5), according to the Wentworth scale. The majority of the sediment for all depth increments of all cores was made up of sand (between 70% and 93%), with a lesser amount of silt (between 6% and 28%). Clay represented the smallest percentage in all of the cores, with a maximum of 3% of the total volume of all the cores from both the Tahu and Matsalu sites (figure 6.6). Clay is an important material in the sediment due to its ability to form cohesive bonds with a variety of compounds. This limits post depositional mobility of a variety of elements including ^{137}Cs. Data were only recorded until the maximum depth at which ^{210}Pb or ^{137}Cs activity was recorded.

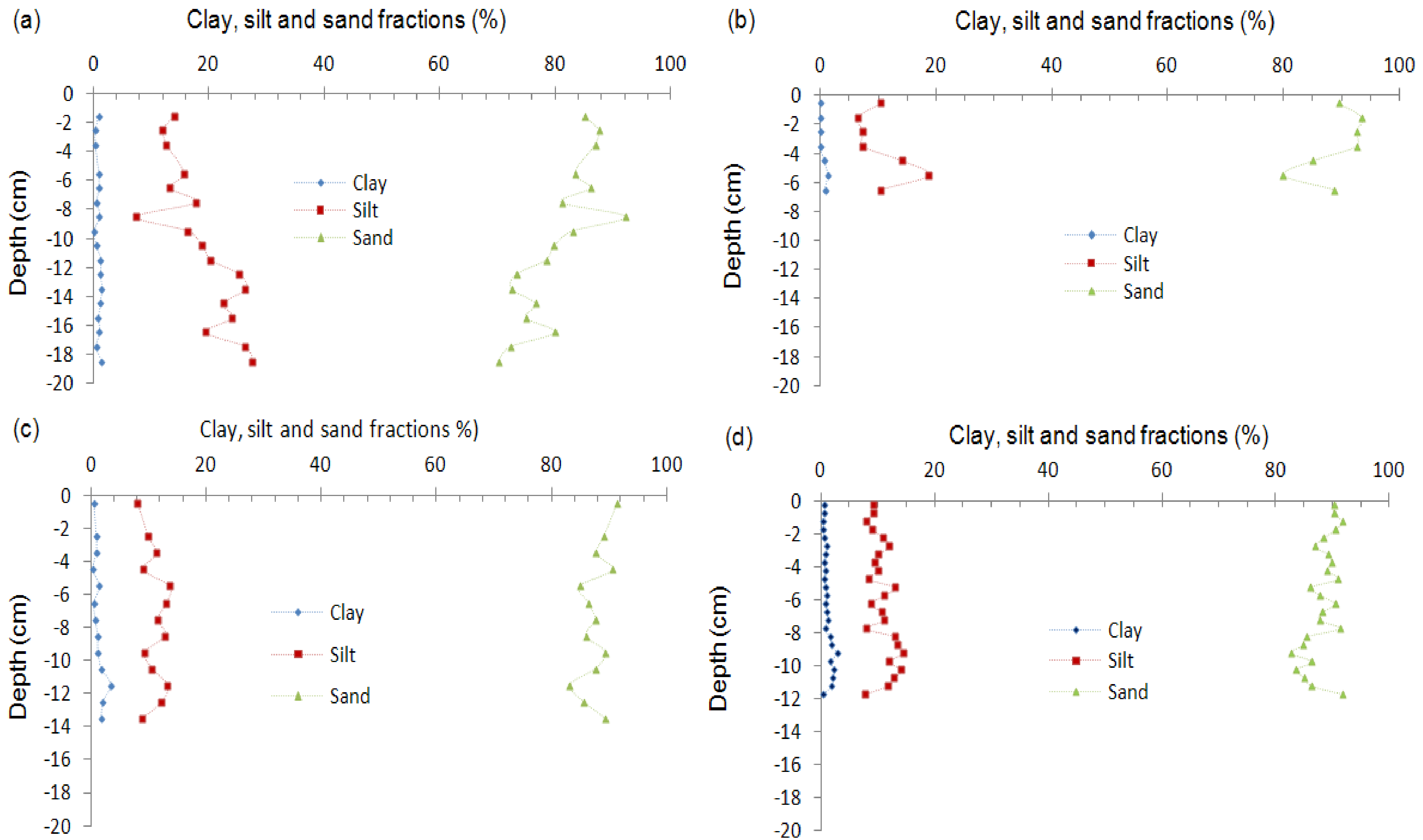

Figure 6.5 : The percentage by volume of clay, silt and sand in the cores taken from Tahu LS (a) and TG (b) and Matsalu LS (c) and TG (d) plant communities. These data show particle size percentages after organic matter has been removed

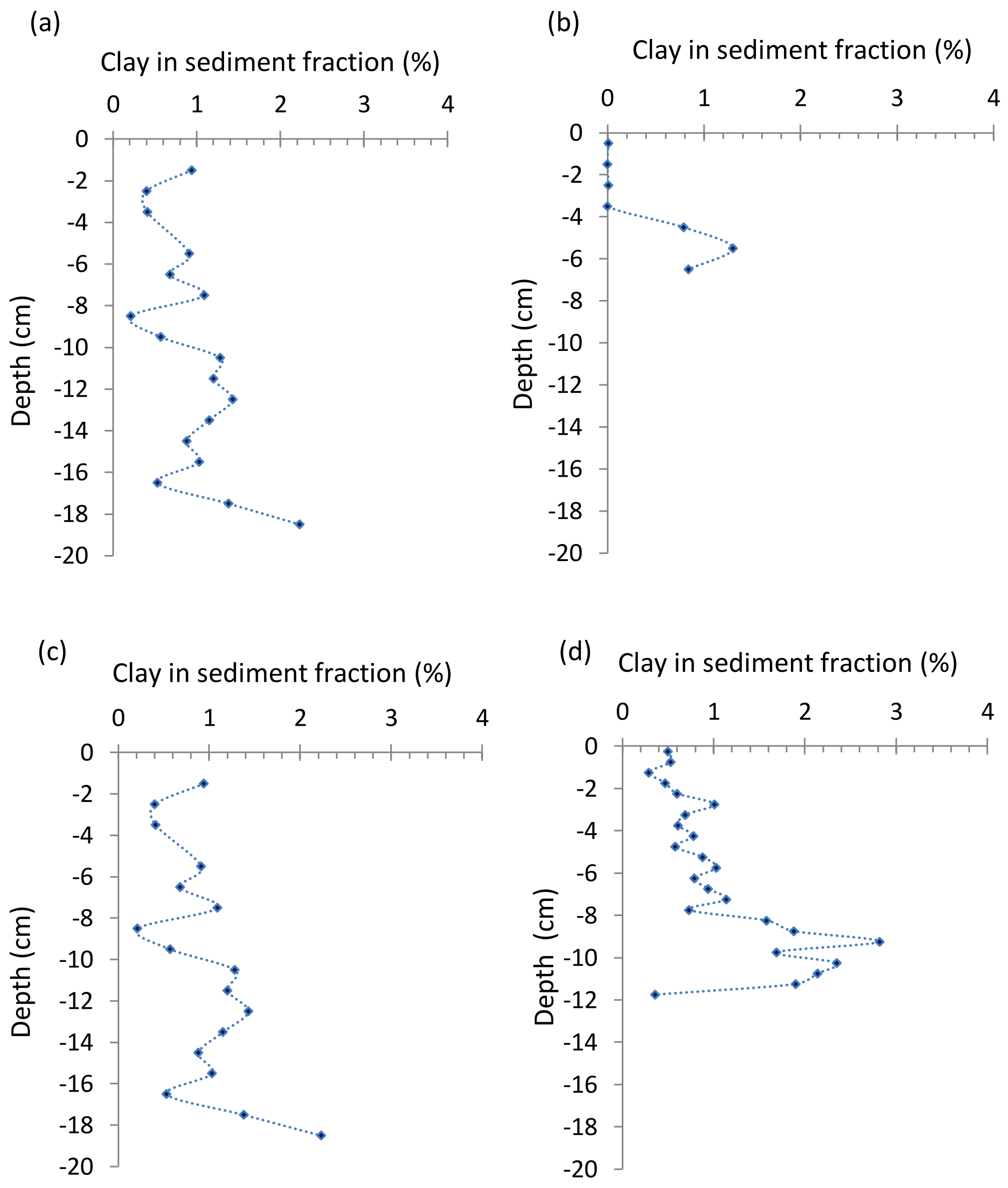

Figure 6.6: The percentage by volume of clay found in the Tahu LS (a) and TG (b) and Matsalu LS (c) and TG (d) cores. These data are shown against depth from surface.

6.4.3 ^{137}Cs accretion rates for Tahu

The larger peaks shown in figure 6.7, at depths of 6.5cm in the LS core and 2.5cm in the TG core, are consistent with activity levels from Chernobyl in Estonia (Realo *et al*., 1995). The smaller peak, at 9.5cm below the surface in the LS core and 4.5cm below the surface in the TG core, indicates a lower activity level consistent with the 1963 weapons testing peak (figure 6.7). This provided an age of 48 years for the sediment at a depth 9.5cm in the LS community and 4.5cm for the TG community. Using the formula for determining sediment accretion rates by using the ^{137}Cs radionuclide markers sediment accretion rates were calculated as shown in table 6.1.

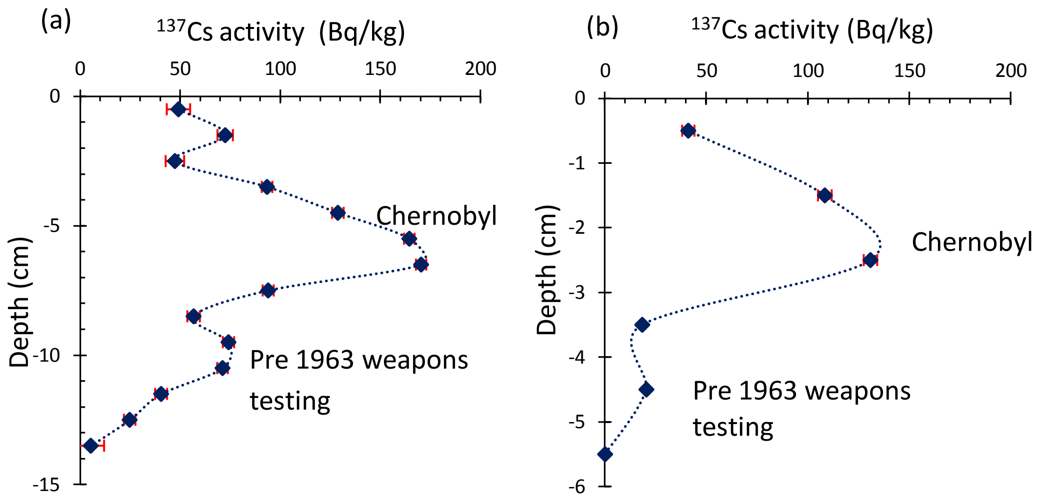

Figure 6.7: ^{137}Cs activity down the core profile in two cores from the Tahu site. (a) data recovered from the Tahu LS community, (b) data recovered from Tahu TG. The red bars show error margins in activity.

Table 6.1: Sediment accretion rates in mm/year at Tahu using the activity of ^{137}Cs as deposited during the 1963 weapons tests and 1986 Chernobyl nuclear power station release. Errors were calculated using the depths either side of each peak.

	Accretion rates since (mm/yr)					
		1963			1986	
Plant community	Lower	Average	Upper	Lower	Average	Upper
LS	1.2	1.4	1.5	2.3	2.7	2.9
TG	0.3	0.5	0.6	0.6	1	1.3

6.4.4 ^{137}Cs accretion rates for Matsalu

The larger peak, at 6.5cm in the Matsalu LS core, was likely to be related to the 1986 Chernobyl disaster (figure 6.8). This core was located at a similar elevation to the Tahu LS core and the same depth for the larger activity ^{137}Cs peak in activity (figure 6.7). The smaller peak at 9.5cm below the surface in the LS community has a lower activity level consistent with the 1963 weapons testing (figure 6.8). Interestingly this peak in activity also occured at the same depth in the Tahu LS core which was located at a similar elevation above m.s.l. suggesting similar sediment accretion rates (figure 6.7). This peak in activity provides an age of 48 years for the sediment at a depth 9.5cm in the LS community.

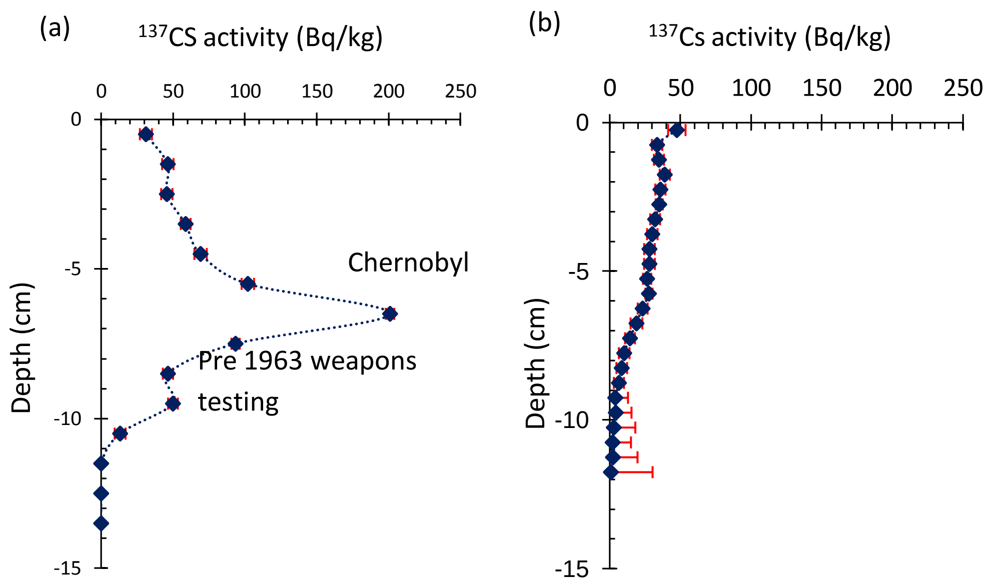

Figure 6.8: ^{137}Cs activity down the core profile in two cores from the Matsalu site. (a) data recovered from the Matsalu LS core, (b) data recovered from the Matsalu TG core. The red bars show error margins in activity.

The Matsalu TG core had much lower ^{137}Cs activity than either the Matsalu LS core or either of the cores taken from Tahu (figure 6.7; figure 6.8). The highest activity was found at the surface of the core, which was inconsistent with the other cores collected. This suggests that either there has been recent erosion leaving the 1986 Chernobyl marker horizon exposed, or there had been relocation of ^{137}Cs due to post depositional mobility, or more likely both. Therefore it was not possible to date this core using ^{137}Cs with any certainty.

Using the formula for determining sediment accretion rates by using the ^{137}Cs radionuclide markers, the results showed relatively rapid sediment accretion in the LS community with an increase in the last 24 years (table 6.2). These data showed rates of accretion in the lower (LS community) similar to regional isostatic uplift rates (2.0mm/yr, Vallner *et al*., 1988).

Table 6.2: Sediment accretion rates in mm/year at Matsalu are shown using the activity of ^{137}Cs as deposited during the 1963 weapons tests and 1986 Chernobyl nuclear power station release. Errors were calculated using the depths either side of each peak.

	Accretion rates since (mm/yr)					
	1963			1986		
Plant community	Lower	Average	Upper	Lower	Average	Upper
LS	1.8	2.0	2.5	2.3	2.7	3.1

6.4.5 Simple method accretion rates using ^{210}Pb for Tahu

The ^{210}Pb total activity/depth profile within the core sequences from the Tahu LS and TG exhibited a near exponential decline with depth down to 18cm and 5cm respectively (figure 6.9). This activity depth curve fits the theoretical profile described in this chapter (figure 6.3) although the LS core displayed a noticeable decrease in activity between 2-3 cm depth. The fluctuation in activity in the Tahu cores was likely to be due to an erosional event in the LS core. At the depths below 18cm for the LS core and 5cm for the TG core ^{210}Pb total activity was dominated by the supported (or background) component derived from the in-situ decay of ^{226}Ra. The total activity of ^{214}Pb, which is in secular equilibrium with ^{210}Pb, was used as a proxy to measure supported ^{210}Pb activity. Mean activity for ^{214}Pb in the Tahu LS core was 24Bq/kg, which was comparable to the average of the readings of ^{210}Pb total activity at the bottom of the core (22Bq/kg). Thus, supported activity was assumed to be 24Bq/kg for the Tahu LS core. Mean activity for ^{214}Pb in the Tahu TG core was 19Bq/kg, total activity for the bottom of the core 20Bq/kg, hence the supported activity was taken to be 19Bq/kg for the Tahu TG core.

The regression analyses for the LS and TG cores in the Tahu site both exhibited a significant ($p < 0.0001$ and $p < 0.0001$ respectively) and strong (R^2 0.853 and 0.865 respectively) linear relationship between log normal activity (ln) and depth (figure 6.10). For the LS core the coefficient of the regression line was 0.236 (lower 95% = 0.182, upper 95% = 0.289). Hence, according to the simple method equation sediment accretion rate was equal to 0.03114 (decay constant for ^{210}Pb)/0.236 (coefficient of the regression line). This was equal to 1.3mm/year. Error values were 1.7mm/year (upper value) and 1.1mm/year (lower value) (table 6.3).

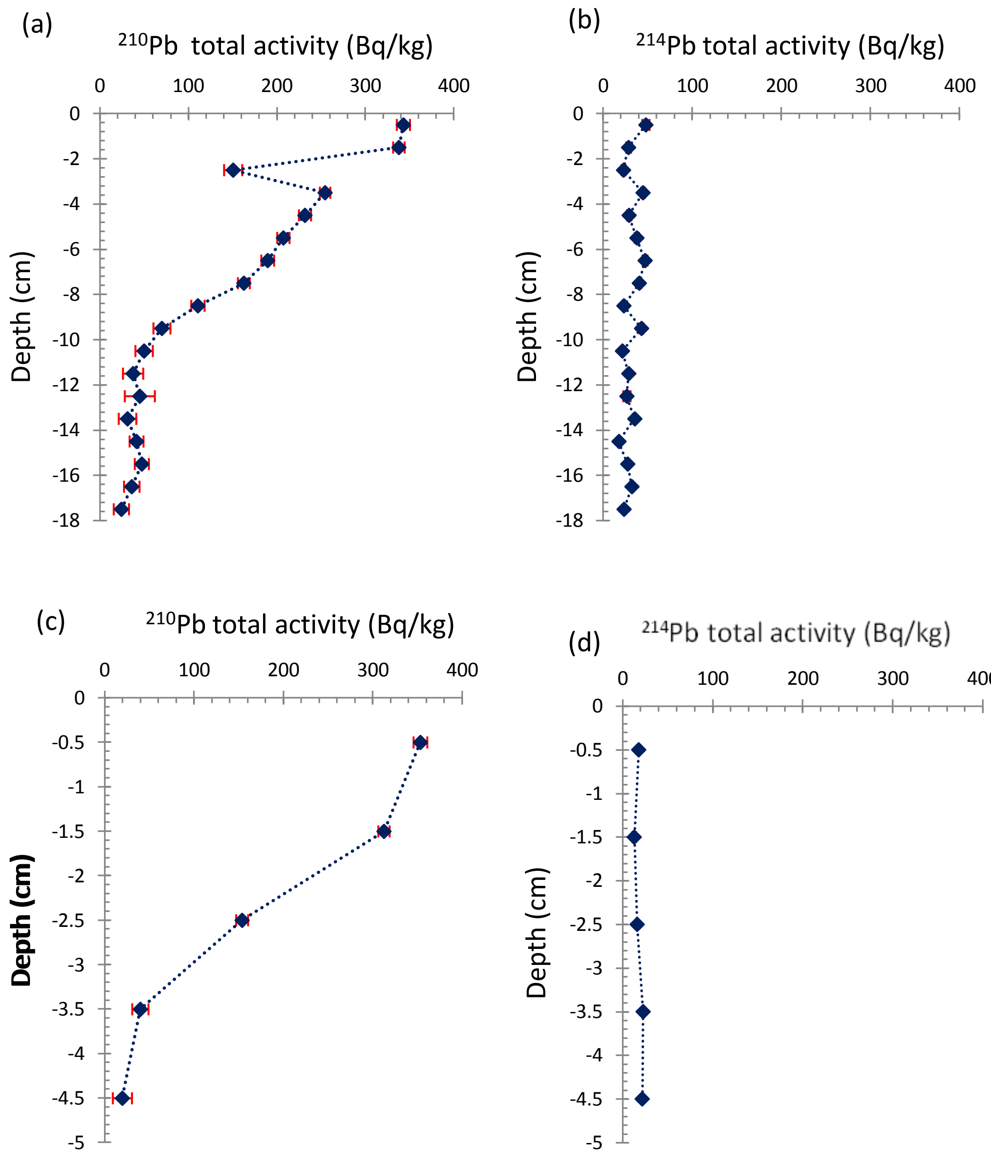

Figure 6.9: Total activity down the core for (a) ^{210}Pb LS, (b) ^{214}Pb LS, (c) ^{210}Pb TG, and (d) ^{214}Pb TG from Tahu. The red bars show error margins in activity.

For the TG core the coefficient of the regression line was 1.372 (lower 95% = 0.374, upper 95% 2.371). Thus, according to the simple method equation the TG community core has been accumulating soil, on average over the whole time period of the core, at a rate of 0.2mm/yr. Error values were 0.8mm/year (upper value) and 0.1mm/year (lower value) (table 6.3).

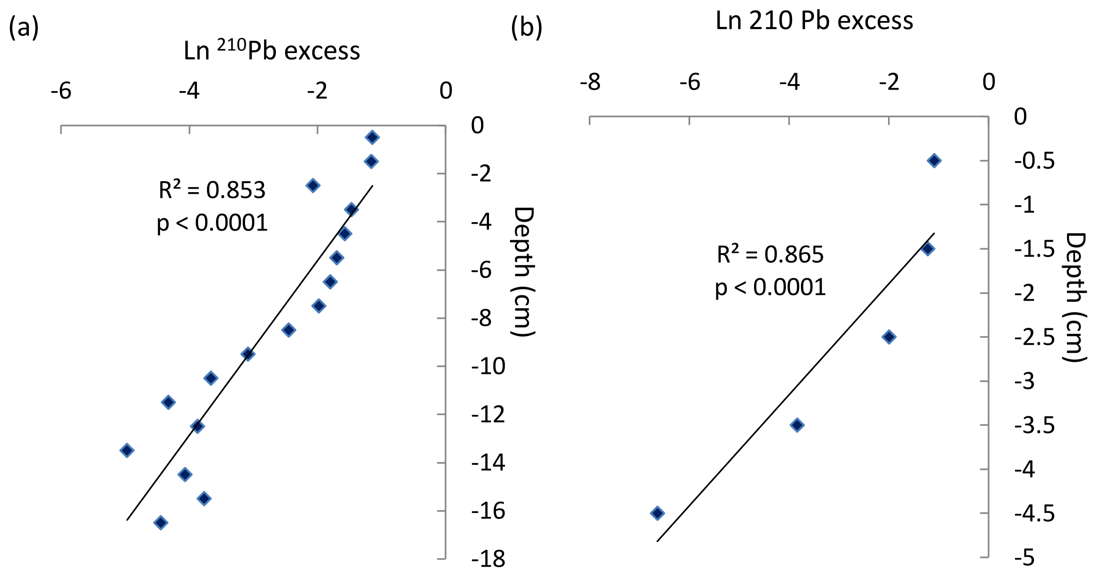

Figure 6.10: Log normal (Ln) ^{210}Pb activity with depth for the (a) LS core and (b) TG core at Tahu. Regression analysis was performed to determine if ln ^{210}Pb activity had a linear relationship with depth, a prerequisite of the CF:CS method.

Table 6.3: Sediment accretion rates for the LS and TG cores at Tahu calculated using the simple method from the regression line of the ^{210}Pb in the core profile over time. Lower, upper and average accretion rates were calculated using data for the whole core profile.

	Accretion rates (mm/yr)		
Plant community	Lower	Average	Upper
LS	1.1	1.3	1.7
TG	0.1	0.2	0.8

6.4.6 Simple method accretion rates using ^{210}Pb for Matsalu

The ^{210}Pb total activity/depth profile within the core sequences from the Matsalu LS and TG exhibited a near exponential decline with depth down to 14cm and 12cm respectively (figure 6.11). This curve was in agreement with the theoretical profile described in this chapter (figure 6.3) although the LS core does display noticeable decreases in activity between 2-3 cm and 5-6cm and the TG core displays a peak in activity between 0.5 and 2cm. The fluctuation in activity in the Matsalu LS core was likely due to an erosional event and occured at the same depth as in the Tahu LS core. There was a second decrease in activity which occured between 5 and 6cm and was also likely due to an erosional event.

The TG core displayed much lower activity than all the other cores at the surface. This could be, as suggested previously, due to the removal of sediment at the top of the core. There was a decrease in activity between 0.5-2cm in ^{210}Pb activity; this was also possibly due to an erosion event. At the depths below 14cm for the LS core and 12cm for the TG core ^{210}Pb total activity was dominated by the supported (background) component derived from the in-situ decay of ^{226}Ra. As with the Tahu cores total activity of ^{214}Pb, in secular equilibrium with ^{210}Pb, was used as a proxy to measure supported ^{210}Pb activity. Mean activity for ^{214}Pb in the Matsalu LS core was 26Bq/kg which was approximately equal to the average of the readings of ^{210}Pb total activity at the bottom of the core (25Bq/kg). Therefore supported activity was assumed to be 26Bq/kg for the Matsalu LS core. Mean activity for ^{214}Pb in the Matsalu TG core was 22Bq/kg, and total activity for the bottom of the core 22Bq/kg, therefore the supported activity was estimated to be 22Bq/kg for the Matsalu TG core.

The results of the regression analysis show that the observed values were significantly related to the expected values ($p < 0.0001$ for both cores) and therefore the simple method is valid. The R^2 values for the LS and TG cores in the Matsalu site both exhibited a good fit of the regression line to the Ln (log normal) data (0.853 and 0.913 respectively) (figure 6.12).

For the LS core the coefficient of the regression line was 0.399 (lower 95% = 0.264, upper 95% = 0.534). Therefore the mean estimated sediment accretion rate for the LS core derived from the least squares regression of the plot in figure 6.12 was calculated from the CF:CS (simple method) 0.03114 (decay constant for ^{210}Pb)/0.399 (coefficient of the regression line). This was equal to 0.7mm/year. For the TG core sediment accretion rates were equal to 1.0mm/year (table 6.4), errors were 0.9mm/yr (lower) and 1.3mm/yr (upper) table 6.4.

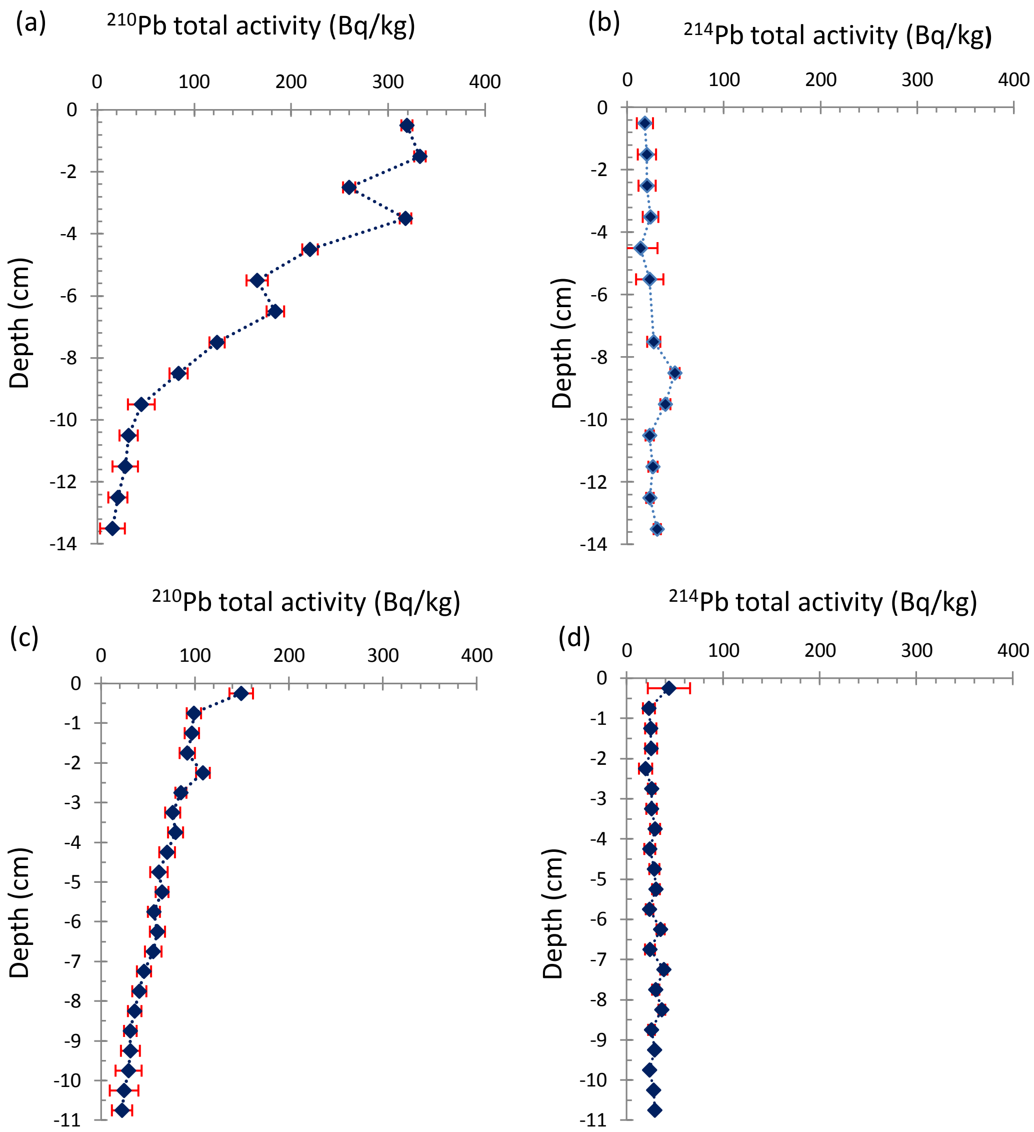

Figure 6.11: Total activity down the core for (a) ^{210}Pb LS, (b) ^{214}Pb LS, (c) ^{210}Pb TG, and (d) ^{214}Pb TG from Matsalu. The red bars show error margins in activity.

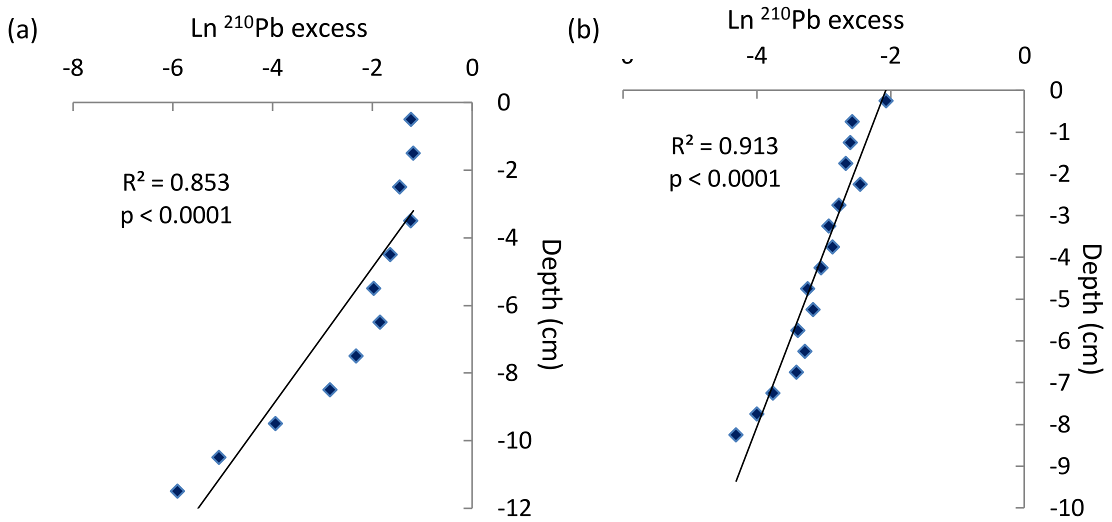

Figure 6.12: Log normal (Ln) ^{210}Pb activity with depth for the (a) LS core and (b) TG core at Matsalu. Regression analysis has been performed to determine if ln ^{210}Pb activity has a linear relationship with depth, a prerequisite of the CF:CS method.

Table 6.4: Sediment accretion rates for the LS and TG cores at Matsalu calculated using the simple method from the regression line of the ^{210}Pb in the core profile over time. Lower, upper and average accretion rates are calculated using data for the whole core profile.

	Accretion rates in (mm/yr)		
Plant community	Lower	Average	Upper
LS	0.5	0.7	1.1
TG	0.9	1.0	1.3

6.4.7 CRS method accretion rates using ^{210}Pb for Tahu

As shown in figure 6.13, the sediment from the LS core can be dated back to 1903 at 17-18cm depth although dating errors for these older sediments were much greater than for the newer soils. The oldest recorded sediments for the Tahu TG core were from 1810 at a depth of 4-5cm, although errors are likely to be high in these older fractions (figure 6.3). ^{210}Pb is most useful for dating sediments deposited during the last 100-150 years (Mizugaki *et al.*, 2006). The oldest Tahu TG core fraction had dating errors of 58 years older and 48 years newer than the given date of 1810, at these ages ^{210}Pb ceases to be a useful tool to assess sediment age.

Figure 6.14 shows that accretion rates have varied over the last 100 years at Tahu LS. During the majority of this time rates have fluctuated between 1.1

and 2mm/yr (figure 6.14). However, there are three periods where accretion rates have greatly increased, namely around 1995 and 1932 when there was an increase in yearly sediment accretion to 3.3mm/yr, and around 1924 when there was an increase in sediment accretion to 5mm/yr. The Tahu TG core did not show any large changes in sediment accretion over time. There did appear to be an overall increasing trend over time from 0.1mm/yr in 1810 to 0.5mm/yr in 1989.

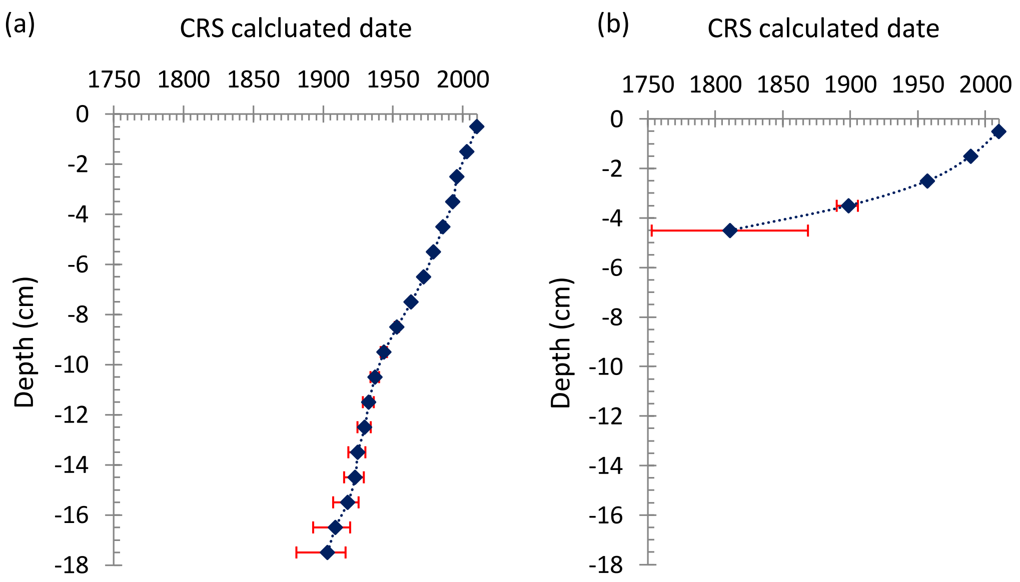

Figure 6.13: CRS calculated age of each depth for the (a) LS core, and (b) for the TG core at Tahu. Error calculations are shown in red using 5% errors in the supported ^{210}Pb activity.

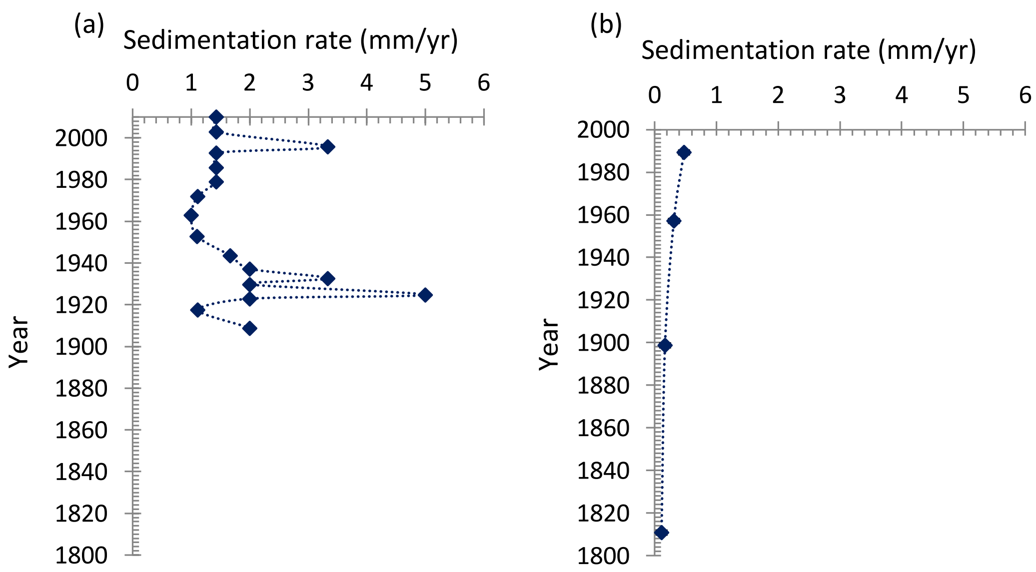

Figure 6.14: ^{210}Pb CRS method derived sediment accretion rates for the (a) LS core, and (b) TG core at Tahu.

6.4.8 CRS method accretion rates using ^{210}Pb for Matsalu

For the Matsalu LS community the oldest sediments assessed using ^{210}Pb were laid down around 1873 and were located at a depth of 11-12cm (figure 6.15). Sediments much older than this are not able to be dated using the ^{210}Pb isotope and even at this age errors were high (figure 6.15). The CRS method is not able to reliably date cores where structural damage has occurred. Due to the loss of the top of the core, sediment dating was not likely to be reliable for the TG core and hence was not conducted.

Sediment accretion rates were calculated over time using the CRS method (figure 6.16). The results for the Matsalu LS core showed accretion rates vary between 0.4-1.7mm/yr between 1873 and 2010. The data showed higher accretion rates around 1974 (1.4mm/yr), 1994 (1.3mm/yr) and 2010 (1.7mm/yr).

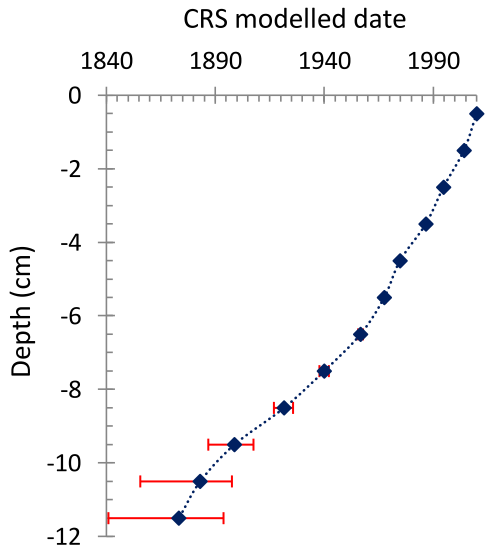

Figure 6.15: CRS calculated age of each depth for the Matsalu LS core. Error calculations are shown in red using 5% errors in the supported ^{210}Pb activity.

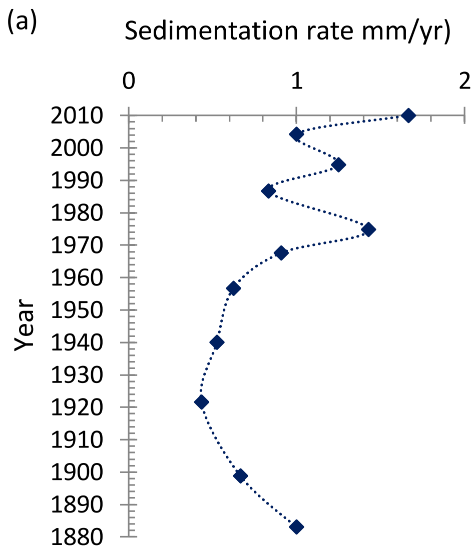

Figure 6.16: ^{210}Pb CRS method derived sediment accretion rates for the LS core at Matsalu.

6.4.9 Comparison of the ^{137}Cs and ^{210}Pb CRS derived accretion rates

The average accretion rate data from the ^{210}Pb CRS method from 1963 to present and 1986 to present were compared with the two ^{137}Cs accretion rates over the same time periods. As can be seen in table 6.5 the accretion rates derived from the ^{137}Cs method, with the exception of Tahu LS core from 1963-present, were all greater. This was likely to be due to limited post depositional movement of ^{137}Cs down the core.

Table 6.5: A comparison of the accretion rates for Tahu and Matsalu using the ^{137}Cs and ^{210}Pb methods. The CRS method does not provide upper or lower values. The Matsalu TG core was not compared as there were no reliable data obtained for the ^{137}Cs method.

		Accretion rates since (mm/yr)					
			1963			1986	
Method	Core	Lower	Average	Upper	Lower	Average	Upper
^{137}Cs	Tahu LS	1.2	1.4	1.5	2.3	2.7	2.9
^{210}Pb CRS	Tahu LS		1.6			1.8	
^{137}Cs	Tahu TG	0.3	0.5	0.6	0.6	1	1.3
^{210}Pb CRS	Tahu TG		0.4			0.5	
^{137}Cs	Matsalu LS	1.8	2	2.5	2.3	2.7	3.1
^{210}Pb CRS	Matsalu LS		1.2			1.2	

6.4.10 Comparison of the accretion rates derived from the ^{210}Pb CF:CS and CRS methods

Table 6.6 shows that all the CRS calculated accretion rates are higher than those calculated using the CF:CS method.

Table 6.6: A comparison of the accretion rates for Tahu and Matsalu from the ^{210}Pb CF:CS and CRS methods. The CRS method does not provide upper or lower values.

		Accretion rates (mm/yr)		
Method	Core	Lower	Average	Upper
CF:CS	Tahu LS	1.1	1.3	1.7
CRS	Tahu LS		1.9	
CF:CS	Tahu TG	0.1	0.2	0.8
CRS	Tahu TG		0.3	
CF:CS	Matsalu LS	0.5	0.7	1.1
CRS	Matsalu LS		0.9	

6.5 Discussion

6.5.1 Assessment of ^{137}Cs, ^{210}Pb CF:CS and CRS methods of measuring accretion rates

Soil organic matter and particle size have an effect on the location of radionuclides in the core profile particularly in areas with fluctuating water tables, which can move elements in solution through the core profile. Several studies have shown that ^{137}Cs is selectively fixed to clay minerals and is less mobile in clay rich soils (Robbins & Edgington, 1975; Coughtrey & Thorne, 1983; Walling & He, 1993; Cundy & Croudace, 1996; Rosen *et al*., 2009; Forsberg *et al*., 2000). Previous studies have shown that higher proportions of soil organic matter increase post depositional mobility of ^{137}Cs (Ritchie & McHenry, 1990; Rosen *et al*., 2009; Forsberg *et al*., 2000). The data from this study show that the soils for the both the LS and TG communities in both the Tahu and Matsalu sites are predominantly made up of sand with a lesser proportion of silt and a very small clay fraction (Puurmann & Ratas, 1998). However, this clay fraction increases with depth and is important with regards to limiting post depositional mobility of ^{137}Cs in the core profile. Soil organic matter has the opposite effect as it limits the retention of ^{137}Cs in the soil.

In the analyses of ^{137}Cs, both the Tahu cores and the Matsalu LS core exhibited similar activity levels and a similar trend down the profile, with ^{137}Cs activity showing two distinct peaks. The upper and larger peak related to the 1986 Chernobyl disaster, and the pre 1963 weapons testing showed as a smaller peak found at a greater depth. These cores also appeared to have little post depositional mobility as shown by the broad agreement in accretion rates derived from both the CF:CS and CRS methods for ^{210}Pb. However, the Matsalu TG core had much lower ^{137}Cs activity levels and did not show any obvious peaks in activity. Additionally, ^{137}Cs was present in the lowest reaches of the Matsalu TG core, which due to the rate of sediment accretion calculated using the CF:CS method was likely to be over 100 years old. ^{137}Cs is an artificial radionuclide whose only source is nuclear reactions and therefore is not more than 65 years old (Cundy & Croudace, 1996; Teasdale *et al*., 2011). This suggests post-depositional mobility of ^{137}Cs down the Matsalu TG core.

The activity of ^{137}Cs decreased down the profile of the Matsalu TG core but not in the other cores. The high proportion of organic matter, although lower than in other cores, and low proportion of clay in the upper sections of the Matsalu TG core was likely to have facilitated the migration of ^{137}Cs from the upper sections of the core downwards. The Matsalu TG core also appears to have undergone erosion, as shown by the very low activity of ^{210}Pb and the absence of an obvious 1986 ^{137}Cs Chernobyl signature. This reworking of the upper reaches of the core was likely to have supplied the ^{137}Cs that leached down the core profile. This would explain the greater spread of ^{137}Cs activity throughout the core. Therefore, the ^{137}Cs dating was likely to be unreliable in the Matsalu TG core.

^{210}Pb within the core profile is not as susceptible to post depositional mobility as ^{137}Cs (Leland & Shukla, 1973) although the sediments themselves can undergo mixing or bioturbation thereby reworking ^{210}Pb in the core profile

(Robbins & Edgington, 1975). Therefore this method is generally used to assess accretion rates, with ^{137}Cs used as an independent verification of the ^{210}Pb methods (Appleby, 2001).

The results for the Tahu cores and the Matsalu cores showed that there was an exponential decrease in ^{210}Pb activity with depth, suggesting a constant rate of supply of ^{210}Pb from atmospheric sources. This fits with the assumptions of ^{210}Pb supply for the CF:CS and CRS methods (Cundy *et al*., 2003). The Matsalu TG core however showed much lower activity in the surface of the core in comparison to the other cores. From this, it appears that there has been a removal of the surface layer of the soil. This also explains the lack of spikes in ^{137}Cs activity down the core profile that could be attributed to the Chernobyl and pre-1963 weapons testing signatures. Andersen *et al*. (2000) found similar results due to the reworking of sediments and erosion for both ^{137}Cs and ^{210}Pb in the Humber estuary, England. Erosion limits the use of ^{137}Cs to independently verify sediment accretion (Andersen *et al*., 2000), although it did not limit the use of ^{210}Pb to assess accretion rates for the Matsalu TG core using the CF:CS method. None of the Estonian coastal wetland cores exhibited great differences in sediment accretion rates between the ^{137}Cs and CF:CS and CRS ^{210}Pb methods. However, ^{137}Cs impulse dating can only estimate accretion rates since 1963 and is subject to post-depositional mobility, hence this method is more limited than ^{210}Pb dating for estimating sediment accretion in soils with small clay fractions.

The CF:CS and CRS ^{210}Pb methods for assessing accretion rates can be used to date much older sediments and therefore give a longer record of sediment accretion (Robbins & Edgington, 1975; Cundy & Croudace, 1996; Mizugaki *et al*., 2006). One of the assumptions of the CF:CS method is constant sediment accretion. Hence, this method can only provide an average of sediment accretion rates over the calculated age of the core (Mizugaki *et al*., 2006). The CRS method, as with the CF:CS method, assumes a constant supply of ^{210}Pb but does not require the assumption of constant sediment accretion and can be used to calculate rates of sediment accretion for each analysed section of a core (Appleby, 2008). Thus, in this study the CRS methodology was selected for calculating accretion rates to be used in the dynamic plant community model.

6.5.2 Sediment accretion rates

With the threat of sea level rise, it is important to know whether coastal areas are erosional or depositional. Previous studies have suggested that the main driving force behind the creation and development of Estonian coastal wetlands is isostatic uplift (Puurmann & Ratas, 1998). Estonian coastal wetlands are predominantly depositional with sediment derived from more exposed north and west facing coasts (Kotta *et al*., 2008), and so to predict their future extent it is necessary to obtain a more complete knowledge regarding their past development. As can be seen in the CRS output figures 6.14 and 6.16, with the exception of the Tahu TG core, sediment accretion rates at the study sites have fluctuated over the past 100-150 years. The Tahu TG community exhibited the lowest accretion rates (mean CRS 0.3mm/yr) and was located at a higher elevation above m.s.l. than LS. In tidal coastal wetlands most allochthonous accretion occurs in the lower more regularly inundated areas (Stumpf, 1983; Reed, 1989). Whilst Estonian coastal wetlands are non tidal, the lower elevation plant communities are more frequently inundated by the Baltic Sea than the higher elevation areas. This would explain the more rapid accretion rates in the Tahu LS core (mean CRS 1.9mm/yr) than in the Tahu TG core.

The Matsalu TG core (mean CF:CS 1.0mm/yr) exhibited slightly higher sediment accretion rates than the Matsalu LS core (mean CF:CS 0.9mm/yr). The Matsalu TG core was taken from a section of the TG community within 20m of the sea (figure 6.2). The Tahu TG core however, was removed from a section of the wetland with 250m of the LS and US plant community between it and the sea (figure 6.1). The study in chapter 4 showed that micro-topography affects the hydrology of these wetlands and inundation occurs by both overland flow and percolation. However, with respect to accretion, percolation does not supply sediment to the wetland. Therefore, whilst the plant community patches where the Tahu and Matsalu TG cores were located were likely to have similar water table levels, the frequencies and depths of inundation and the amount of sediment deposited on the Tahu TG patch was likely to have been appreciably lower. The data recorded from the dipwells located in the TG and SW plant communities in chapter 4 (figure 4.6) indicates that this was the case.

The results using the CRS method suggested that the Tahu TG core exhibited a slight increase in sediment accretion rates over time. The main supply of sediment in coastal wetlands is from allochthonous sources during inundation events (Mitsch & Gosselink, 2000). The low sediment accretion occurring in the Tahu TG core suggests that inundation events are rare although the increase in accretion rates may indicate that there has been more frequent inundation since the 1960's. This accords with other studies, which show that there has been a rise in relative sea level and an increase in storminess in Estonian coastal waters (Suursaar *et al.*, 2007; Keevallik & Soomere, 2008).

The results from the CRS method suggest that the Tahu LS community has undergone sediment accretion fluctuations. These varied from 1mm/yr to 5mm/yr, relatively large in comparison with mean accretion of 1.9 mm/yr (figure 6.16). These pulses of sediment deposition were most likely due to storm surges depositing relatively large amounts of sediments. The increase in sediment accretion around 1995 was likely to have been caused by a period of increased storminess in 1992-1993 (Kont *et al*., 2007; Orviku *et al*., 2009). The increases in sediment accretion around 1924 and again in 1930 were associated with two extended periods of high yearly average sea levels in west Estonia, according to the Pärnu tidal gauge (Suursaar *et al*., 2007).

The results from the CRS method for the Matsalu LS core showed an increase in sediment accretion during the mid 1970's. During the time period 1975-1976 there were a large number of storm days, which were recorded at both the Vilsandi and Pärnu tide gauges (Kont *et al*., 2007; Suursaar *et al*., 2007), and were likely to be the cause of the increase in sediment accretion rates.

There was a rise in sediment accretion rates that occurred around 1994 recorded in the Matsalu LS core. Suursaar & Sööäär (2007) showed that during this period there was an increase in maximal sea level recorded at both the Ristna and Pärnu tide gauges, located in north west and west Estonia respectively, which were likely to have been the main cause of increased sediment accretion. The Matsalu LS core records one further increase in sediment accretion in the sediment core after 2004. There was a winter storm in 2005 (Orviku *et al*., 2009) that deposited a large amount of material in Matsalu (Lotman *pers. comm*. 2010), which was likely to have been the cause of the 2004-2010 increase in accretion rates.

6.6 Conclusions

The methods using radionuclides to assess past sediment accretion rates in this study were robust. The ^{137}Cs and ^{210}Pb CF:CS and CRS methods were largely in agreement regarding sediment accretion rates over the time period of all the cores taken. However, the ^{137}Cs method was found to be more useful as a verification tool of the other methods than as a stand alone method due to its more limited time scale than either of the ^{210}Pb methods and the influence of post depositional mobility. Whilst the CF:CS method for assessing sediment accretion provides rates that are in agreement with the other methods, one of the main assumptions for this method, that of constant sediment accretion, was not met. Due to this, CRS was selected as the best methodology for use in assessing sediment accretion rates over a 100 year time period in the studied coastal wetlands and therefore these values were used in the dynamic plant community model. It should be noted, however, that whilst these data provide an insight into the sediment accretion rates at the Tahu and Matsalu site, sediment accretion rates are likely to vary over different areas within each site as well as between plant community types.

This study has provided the first evaluation of sediment accretion rates in Estonian coastal wetlands. Sediment accretion rates derived using the CRS method exhibit different rates in the LS and TG communities, both between the two sites and over time. Thus, any modelling of the effects of sea level rise on plant community types will need to take into account the causes of these different sediment accretion rates. The rates used in the model for predicting the impacts of future sea level scenarios should use both a mean value of the accretion rates as derived from the average over the whole core as well as an increased sediment accretion rate scenario to account for predicted increased storminess.

7 Applying the plant community model to assess the impact of sea level rise on three Estonian coastal wetland sites

7.1 Preamble

The study in this chapter developed a dynamic plant community model based upon the static correlative model developed in chapter 5. The dynamic plant community model was used to predict the future location and extent of six Estonian coastal wetland plant communities using four IPCC (2007) sea level rise scenarios and one scenario with no change in eustatic sea level. The model incorporated the two accretion rate estimates derived from the results of chapter 6. Assessment was made of the model performance using the results of the predicted and observed plant communities as shown in chapter 5. The dynamic correlative model provides a vision of the future location and extent of Estonian coastal wetlands based on changes in sea level, and taking into account sediment accretion and isostatic uplift. Constraints upon the model, which may be accounted for in other modelling techniques such as mechanistic, are: an assumption that temperature, precipitation, grazing management, soil pH, salinity, nitrogen, organic matter, potassium, and available sediment type will not vary to a degree that they will alter plant community composition.

7.2 Introduction

The research undertaken so far has assessed and quantified the relationship between plant community type, elevation and edaphic factors in Estonian coastal wetlands (chapter 4). Elevation has been shown to be closely related to plant community type, such that it can be used as a predictor. DGPS corrected LiDAR elevation data has been used to create a static correlative plant community model incorporating the elevation preferences of each wetland community type (chapter 5). The predictive plant community model was able to accurately predict the location and extent of the wetland plant communities in three non-contiguous Estonian coastal wetland sites. The study has also made an assessment of sediment accretion rates over the last 150 years for two Estonian coastal wetlands, Tahu and Matsalu (chapter 6). Chapter 7 further develops the plant community model by predicting the future location and extent of the plant communities at three sites, Tahu, Matsalu, and Kudani, up to and including 2099.

The model included the key factors affecting the location and extent of the plant communities. These were: (i) sea level rise including four IPCC (2007) sea level rise scenarios and one scenario assuming no change in eustatic sea level; (ii) sediment accretion assuming no increase in storminess and increased storminess; and (iii) isostatic uplift, using rates derived from Vallner *et al*. (1988). Model validation is problematic in scenario based predictive modelling (Franklin, 1995). Therefore model performance was assessed based on the ability of elevation to predict plant community type in the static model, an assessment of which was made in chapter 5 comparing observed and predicted plant community type.

7.2.1 Background

Sea level rise is one of the major climate change threats to low lying countries with maritime borders. Estonia with its relatively long, low lying coastline is potentially at risk from rising sea levels. Much of the coastline of Estonia is experiencing post glacial isostatic uplift. Uplift rates can be as high as 2.8mm/yr (Vallner *et al*., 1988; Ekman & Mäkinen, 1996; Eronen *et al*., 2001) and have been described, along with management, to be the driving force behind Estonian coastal wetland formation and maintenance (Rannap *et al*., 2004).

Previous studies have suggested that in spite of land uplift, sea level rise will cause degradation of the Estonian coastline due to more frequent inundation and increased storminess and hence erosion (Kont *et al*., 1997; Kont *et al*., 2003). The study by Kont *et al*. (2003) suggested that the majority of the wetlands on the west coast of Estonia would become permanently inundated causing a 100% loss of the swamp vegetation and an 80% loss of the wet grasslands, only partially alleviated by isostatic uplift. In the comparatively stable and well defined landscapes of Estonian coastal wetlands, any instability of climate and changes to inundation are likely to have a serious impact on the plant communities. Any increase in storminess could limit the distribution of the CS plant community in favour of RS (Coops *et al*., 1991).

Moreover in extreme examples the littoral vegetation could be completely removed, as is found in exposed areas Estonian coastal sites (Puurman & Ratas, 1998).

A report by the IPCC (2007) suggests that climate change will affect not only sea level and temperature but will lead to increased storminess, particularly in northern Europe and the Baltic (Orviku *et al*., 2003; Soomere *et al*., 2009; Rozynski & Pruszak, 2010). In both the Kont *et al*. (1997) and the Kont *et al*. (2003) studies looking at the effects of sea level rise on the Estonian coastal sediment accretion rates were ignored and both studies used an arbitrary sea level rise of 1m, and the basis for this rise was not explained. The study in chapter 6 suggested that in depositional areas, such as the Tahu and Matsalu coastal wetlands, increased storminess and rising sea levels have facilitated wetland development through sediment accretion. This was in agreement with previous studies in other coastal wetlands (Reed, 1990; Reed, 1995; Roman *et al*., 1997). Additionally sea level rise, predicted until 2099, is unlikely to be as high as 1m (IPCC, 2007). Actual sea level rise is estimated to be between 0.18m minimum under the “best” case scenario, and 0.59m, which is the maximum for the “worst” case scenario (IPCC, 2007). There are no predictions available for eustatic sea level rise in the eastern Baltic nor the effects of sea level rise on coastal plant communities located there.

There are numerous predicted rates of global eustatic sea level rise available. Rahmstorf (2010) has made an attempt to summarise the main sea level rise predictions (figure 7.1), which shows that the IPCC (2007) predictions are far lower those predicted by Rahmstorf (2007), Horton *et al*. (2008), Grinsted *et al*. (2009), Vermeer & Rahmstorf (2009), and Jevrejeva *et al*. (2010). The IPCC (2007) model has been criticised for not accurately assessing the rate of present day sea level rise due to an assumption of a zero contribution from the Greenland and Antarctic ice sheets (Rahmstorf, 2010). However, the semi-empirical models used by Rahmstorf (2007), Horton *et al*. (2008), Grinsted *et al*. (2009), Vermeer & Rahmstorf (2009), and Jevrejeva *et al*. (2010) take the assumption of a proportional relationship between Greenland and Antarctica ice sheet melting and temperature. This assumption ignores other factors such as an increase in Antarctic ice sheet size due to greater precipitation as suggested by the IPCC (2007). It must be noted that the semi-empirical models for sea level rise reproduce past sea level rise very well (Rahmstorf, 2007; Horton *et al*., 2008; Grinsted *et al*., 2009; Vermeer & Rahmstorf 2009, Jevrejeva *et al*., 2010) unlike the physical models used in the IPCC (2007) report. The main drawback to the semi-empirical models for sea level rise is that it is impossible to know whether historical relationships between temperature and sea level rise will continue to be valid in future scenarios. The IPCC (2007) sea level rise scenarios have been used as a standard in previous sea level rise modelling studies for coastal wetlands and hence they were chosen as the most valid and conservative basis for predicting the future location and extent of the plant communities in Estonian coastal wetlands.

Figure 7.1: A summary of the main sea level rise prediction ranges (Rahmstorf, 2010)

Predictive models have been used to predict potential distributions of plant species and communities under current climate conditions and using a variety of climate change scenarios (Araujo & Luoto, 2007; Lavergne *et al*., 2010; Bertrand *et al*., 2011; Moeslund *et al*., 2011; Stratonovitch *et al*., 2012). Typically ecological modelling is based on knowledge of environmental gradients, which are used as ecological predictors (Franklin, 1995). The static plant community model produced in chapter 5 will be used to develop a dynamic correlative plant community model based on predictions of elevation above sea level in 2099.

The aim of the study in this chapter was to use the plant community model, developed during the thesis, to predict the extent and distribution of wetland communities in the Tahu, Matsalu, and Kudani sites by 2099. The model integrated sea level scenarios, accretion rate estimates and isostatic uplift rates in Estonian coastal wetlands. The objectives were: 1) to calculate local sea level rise for each Estonian study site under different eustatic sea level scenarios and accretion rate estimates; 2) to use a GIS based approach to map the potential distribution of the plant communities in the three study sites, and 3) to calculate the extent of each plant community under each scenario in 2099 and discuss model performance.

7.3 Study sites and methodology

7.3.1 Study sites

Three study sites were used to assess the potential effects of sea level rise on Estonian coastal wetlands. The first two sites comprised the Tahu and Kudani coastal wetlands, located in the Silma Nature Reserve. The final site was the Matsalu coastal wetland located in Matsalu National Park. A detailed description of the sites is available in chapter 3.

7.3.2 Methodology

7.3.2.1 Baseline plant community map

In order to model the potential effects of sea level rise, the model required a baseline plant community map. The LiDAR elevation derived plant community maps produced in chapter 5 for the Tahu, Kudani and Matsalu sites were used as a basis for the exercise, incorporating the plant community types (CS, RS, LS, US, TG and SW) that can be differentiated using LiDAR elevation data. The OP community was excluded as it was not possible to determine its location using only elevation. This community type covered a small proportion of the total wetland area (average of 3.5% of the total area at Tahu and Kudani and absent from Matsalu).

7.3.2.2 Model development and sensitivity analysis

The baseline model was modified by integrating isostatic uplift rates, sea level rise scenarios and sediment accretion estimates (figure 7.2). Using these values ten scenarios were modelled for Tahu and Matsalu and five for Kudani. Within the GIS the raw values calculated for local sea level rise were fed into the TIN based model to re-categorise the elevations of the plant communities adjusting for future changes in elevation above mean sea level (figure 7.2). The resultant TIN models were converted to raster to assess the extent of each plant community within each scenario. Terrestrial LiDAR data has limited penetration through water, approximately 0.10m, and hence was unable to predict progradation of the wetland into the adjacent water body. The model was clipped to the extent of the present day wetland area with a 1m buffer to account for the estimated minimum distance from the shore at which this depth is reached based on site knowledge.

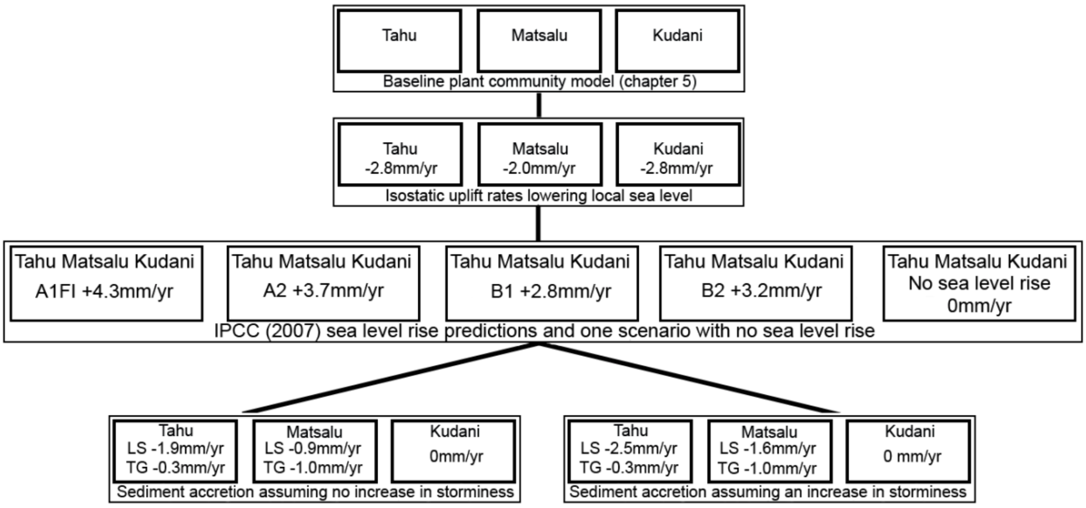

Figure 7.2: Steps involved in the assessment of local sea level rise at Tahu, Kudani and Matsalu.

7.3.2.2.1 Determination of the environmental modelling parameters

According to the IPCC (2007) the minimum global sea level rise between 1999 and 2099 is predicted to be between 0.18m and 0.38m, and the maximum to be between 0.26m and 0.59m dependant on scenario (Table 7.1) (IPCC, 2007). The A1 scenario assumes a world of very rapid economic growth, a global population that peaks mid-century and rapid introduction of new and more efficient technologies. A1 is divided into three groups that describe alternative directions of technological change: fossil intensive (A1FI), non-fossil energy resources (A1T) and a balance across all sources (A1B). The A1T and A1B sea level rise scenarios are very similar to the B1 and B2 scenarios respectively and therefore they were omitted from this study. The A1FI sea level rise scenario estimate quite different sea level rise from the other scenarios and was therefore selected for use in this study. The B1 scenario describes a world with the same global population as A1, but with more rapid changes in economic structures toward a service and information economy. The B2 scenario describes a world with intermediate population and economic growth, emphasising local solutions to economic, social, and environmental sustainability. The A2 scenario describes a very heterogeneous world with high population growth, slow economic development and slow technological change.

The time scale used in this study runs until 2099 consistent with the IPCC sea level scenarios. Four IPCC (2007) scenarios were used in order to take into account changes in hydrology, these were A1FI, A2, B1 and B2.

Table 7.1: Representative climate change scenarios and estimated related temperature and sea level rise. (Adapted from IPCC (2007).

	Sea level rise (m) 2099 relative to 1980-1999	
Scenario	Range	Mid value
A1FI	0.26-0.59	0.43
A1T	0.20-0.45	0.33
A1B	0.21-0.48	0.35
A2	0.23-0.51	0.37
B1	0.18-0.38	0.28
B2	0.20-0.43	0.32

No likelihood has been attributed by the IPCC to the scenarios or the sea level rise ranges. Since the model based sea level rise ranges are quite considerable, a mid range figure has been selected for each scenario. Thus, in this study the sea level rise predictions used are summarised in table 7.2. A no eustatic sea level rise (slr) scenario has been used for comparison.

In order to estimate future vegetation cover, isostatic uplift rates were taken from Vallner *et al*. (1988), which are similar to other studies (Ekman &

Mäkinen, 1996; Eronen *et al*., 2001). The Vallner *et al*. (1988) study used repeated levelling measurement data over the territory of Estonia collected over the period 1933 and 1985 and based on tide gauge data from west Estonia. In the Tahu and Kudani coastal wetlands, isostatic uplift rates were considered to be 2.8mm/year whereas in the Matsalu site rates were 2.0mm/year.

Table 7.2: IPCC sea level rise scenarios used in the modelling by 2099.

Climate change scenario	Mid range sea level rise (m)
A1FI	0.43
A2	0.37
B1	0.28
B2	0.32
No eustatic slr	0

Further variables that were taken into account in order to estimate the location and extent of the wetland plant communities were accretion rates. Past rates were calculated for the Tahu and Matsalu coastal wetlands (chapter 6) using the CRS method. Mean calculated rates over the last 100 years (Tahu LS), 180 years (Tahu TG), and 130 years (Matsalu LS) were used to predict future accretion rates by extrapolation (chapter 6). For Matsalu TG the results from the CF:CS method were used due to deficiencies in the other methods for this core. Mean accretion rates for Tahu were: LS = 1.9mm/yr, TG = 0.3mm/yr; and for Matsalu LS = 0.9mm/yr, TG = 1.0mm/yr (chapter 6). No accretion rates were used for Kudani as it is largely separated from the sea, and hence the main allochthonous sediment source, by a road. Sea water influx only occurs via a narrow 4m wide channel, which terminates in a 2m diameter pipe.

Increased storminess has been linked to increased sediment accretion in predominantly depositional wetlands by a variety of authors (Roman *et al.*, 1997; Allen, 2000; Kolker *et al*., 2009; Schuerch *et al*., 2012) and the results of chapter 6 suggest that this was the case for the Tahu and Matsalu sites. No assessment has been made in the IPCC (2007) report as to the increase in the frequency of storm events. However, a study by Bender *et al*. (2010) suggests that the frequency of storms will double by 2099. This, combined with a decrease of 50% in the number of days of winter sea ice in Estonia, which is when the majority of storms occur (Jaagus, 2006) means that this is likely to be reflected more frequent extreme inundations. In order to include the effects of an increase in the frequency of storm events to the model, the number of increased accretion events was doubled for the CRS derived data of accretion and a new mean calculated. This produced an accretion value of 2.5mm/yr for Tahu LS and no change for Tahu TG (due to lack of data), and a value of 1.6mm/yr for Matsalu LS and no change for Matsalu TG (due to a lack of data).

Both mean estimated accretion rates for the LS and TG communities and those estimated for periods of increased storminess were used at both Tahu and Matsalu. These data were extrapolated from the results of the CRS method for assessing sediment accretion in chapter 6. The estimated accretion rates for the LS community were used as a generalisation for the lower elevation CS, RS, and US plant communities for model simplicity. The estimated accretion rate data for the TG community were used as the rate for both the TG and SW communities as both of these communities are rarely inundated.

Accretion rates were not calculated for the Kudani coastal wetland. Due to the limited water exchange between Võõlameri Bay and the Gulf of Finland, the supply of allochthonous sediment is limited. Therefore accretion rates for the Kudani site were set at 0mm/yr.

7.3.2.2.2 Assessment of model performance

The assessment of the accuracy of a dynamic correlative model is problematic due to the difficulty in assessing future changes. Both Bakkenes *et al*. (2002) and Robertson *et al*. (2003) used the Kappa coefficient values of their static correlative model, upon which the dynamic model was based, to assess the performance of the dynamic model using benchmark data. Hence, in this study the Kappa values for the static correlative model developed in chapter 5 were used to assess the validity of the future predictions, although this was based upon the assumption that there will be no change in the management regime.

7.4 Results

7.4.1 Estimated sea level rise for the Tahu, Matsalu and Kudani sites

The results presented in table 7.3 show local predicted sea level at each site under all sea level scenarios. All data are presented in metres and were based on predictions until 2099 (i.e. not yearly rates). Each sea level scenario was presented using two accretion rate estimations, Accretion rate 1 assumes no increase in sediment accretion and accretion rate 2 assumes an increase in sediment accretion due to increased storminess and hence inundation frequency. As can be seen in table 7.3 local sea level falls in all sites and using both sediment accretion predictions in the scenario assuming no eustatic slr.

At Tahu in all scenarios there was a predicted decrease in local sea level due to the high isostatic uplift and sediment accretion rates. At Kudani, with no predicted sediment accretion, the high isostatic uplift rates were insufficient to keep pace with eustatic sea level rise except in the B1 scenario, which exhibited no change in local sea level (table 7.3). At Matsalu local sea level was predicted to fall in fifty percent of scenarios: B1 and no change in eustatic sea level (no eustatic slr) scenarios using both sediment accretion estimations, and the B2 scenario using accretion rate 2 (table 7.3). The greatest change in local sea level was predicted to occur in scenarios assuming no change in eustatic sea level for Tahu, Kudani and Matsalu (table 7.3). In the A1FI and A2 scenarios using both sediment accretion estimations and the B2 scenario using the accretion rate 2 estimation, local sea level was predicted to rise (table 7.3).

Table 7.3: Local sea level rise under different IPCC (2007) sea level rise scenarios and one scenario assuming no eustatic sea level rise for Tahu, Kudani, and Matsalu to 2099. Accretion rate 1 assumes no increase in storminess and accretion rate 2 increased storminess. Total 1 is equal to the total local sea level rise for each scenario using accretion rate 1 and total 2 uses accretion rate 2. All data are shown in metres.

Site	Sea level scenario	Isostatic uplift	Accretion 1	Accretion 2	Total 1	Total 2
	A1FI					
Tahu LS	+0.387	-0.252	-0.171	-0.225	-0.036	-0.090
Tahu TG	+0.387	-0.252	-0.027	-0.027	+0.108	+0.108
Matsalu LS	+0.387	-0.180	-0.081	-0.144	+0.126	+0.063
Matsalu TG	+0.387	-0.180	-0.108	-0.108	+0.135	+0.099
Kudani	+0.387	-0.252	0.000	0.000	+0.135	+0.135
	A2					
Tahu LS	+0.333	-0.252	-0.171	-0.225	-0.090	-0.144
Tahu TG	+0.333	-0.252	-0.027	-0.027	+0.054	+0.054
Matsalu LS	+0.333	-0.180	-0.081	-0.144	+0.072	+0.009
Matsalu TG	+0.333	-0.180	-0.108	-0.108	+0.081	+0.045
Kudani	+0.333	-0.252	0.000	0.000	+0.081	+0.081
	B1					
Tahu LS	+0.252	-0.252	-0.171	-0.225	-0.171	-0.225
Tahu TG	+0.252	-0.252	-0.027	-0.027	-0.027	-0.027
Matsalu LS	+0.252	-0.180	-0.081	-0.144	-0.009	-0.072
Matsalu TG	+0.252	-0.180	-0.108	-0.108	0.000	-0.036
Kudani	+0.252	-0.252	0.000	0.000	0.000	0.000
	B2					
Tahu LS	+0.288	-0.252	-0.171	-0.225	-0.135	-0.189
Tahu TG	+0.288	-0.252	-0.027	-0.027	+0.009	+0.009
Matsalu LS	+0.288	-0.180	-0.081	-0.144	+0.027	-0.036
Matsalu TG	+0.288	-0.180	-0.0108	-0.108	+0.036	0.000
Kudani	+0.288	-0.252	0.000	0.000	+0.036	+0.036
	No eustatic slr					
Tahu LS	0.000	-0.252	-0.171	-0.225	-0.423	-0.477
Tahu TG	0.000	-0.252	-0.027	-0.027	-0.279	-0.279
Matsalu LS	0.000	-0.180	-0.081	-0.144	-0.261	-0.324
Matsalu TG	0.000	-0.180	-0.108	-0.108	-0.252	-0.288
Kudani	0.000	-0.252	0.000	0.000	-0.252	-0.252

7.4.3 Estimated location and extent of the plant communities at Tahu in 2099

The model predictions for Tahu were divided into those assuming an increase in sediment accretion and those not. In all scenarios at Tahu there was a predicted decrease in local sea level. This decrease was greater in the scenarios assuming an increase in sediment accretion linked to an increase in storminess.

7.4.3.1 Scenarios assuming no increase in sediment accretion at Tahu in 2099

The greatest changes predicted for the Tahu wetland by 2099 compared to present day occurred in the no eustatic sea level rise scenario followed by B1, B2, A2 and A1FI sequentially (figure 7.3). The outputs from the model at Tahu for 2099 indicated that in the all of the eustatic sea level rise scenarios assuming no increase in sediment accretion the extent of the US, TG and SW communities would increase due to a local decrease of sea level, except the scenario assuming no change in eustatic sea level for TG and US (figure 7.3; table 7.4). This predicted increase in the extent of US, TG and SW was at the expense of the LS, RS and CS communities (figure 7.3; table 7.4). The greatest change in the extent of the SW plant community was predicted in the no eustatic sea level rise with an increase from 13.8% of the wetland area to 87.9% (figure 7.3; table 7.4). In the other scenarios the increase was less dramatic rising to 14.9% (A1FI), 17.1% (A2), 20.1% (B2), and 22.6% (B1) (figure 7.3; table 7.4). In the model validation (chapter 5), the model was able to correctly identify SW at Tahu in 66.7% of cases. In the remainder of cases the model identified TG as SW (table 5.3 chapter 5).

The greatest increase in the extent of TG was predicted in the B1 scenario rising from 13.9% to 29.9% of the total area (table 7.4). However, the ability of the model to correctly predict TG was limited, at only 53.3% of cases, due in part to its large elevation range. The validation performed in chapter 5 showed that in 20% of cases, areas predicted to have TG were actually US, and in 26.7% of cases with LS (table 5.3 chapter 5).

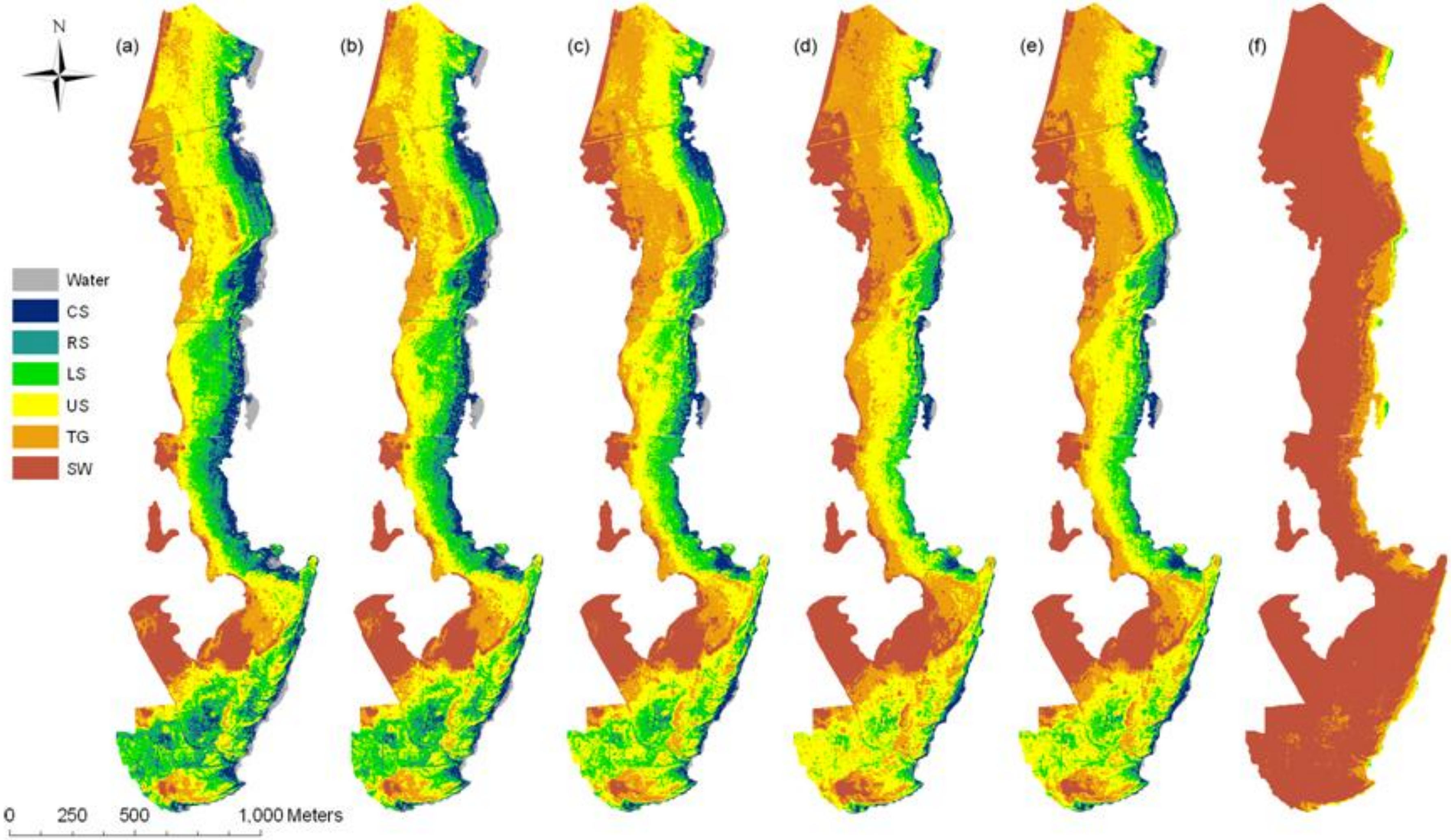

Figure 7.3: Outputs from the plant community model at Tahu assuming no increase in sediment accretion. (a) 2009 (b) A1FI, (c) A2, (d) B1, (e) B2 and (f) no eustatic sea level rise (outputs b-f are predictions for 2099). For sea level rise scenarios see table 7.3.

Whilst US was predicted to increase in extent in all modelled scenarios, with the exception of the no eustatic slr scenario, the increase was very limited. US was predicted to cover 29.3% at present but was predicted to increase to 29.7% (A1FI), 30.5% (A2), 30.7% (B2), and 30.5% (B1) (table 7.4). In the validation of the model in chapter 5 (table 5.3) US was the most difficult community to predict. More of the predicted US patches were observed to be LS than US, 53.3% and 46.7% respectively. The LS and US plant communities exist at similar and overlapping elevation ranges. Therefore this could affect the model outcome as US was predicted to increase in extent at the expense of LS in the majority of scenarios.

The LS plant community was predicted to decrease in extent in all scenarios although the decrease in extent was very limited in the A1FI scenario, from 18.8% of the total area in 2009 to 18.5% by 2099 (table 7.4). The decrease was predicted to be much greater in the B1 scenario, where extent was halved to 9% of the total area and greatest in the no eustatic slr scenario where it was reduced to 0.4% of the total area (table 7.4). In the validation the model was able to accurately predict the location of LS in 80% of cases.

RS was predicted to decrease in extent in the no eustatic slr, B1, B2 and A2 scenarios due to encroachment by the higher elevation plant communities US, TG, and SW (figure 7.3). The greatest reduction in extent was predicted in the no eustatic slr scenario from a present day extent of 7% of the total wetland area to 0.1% (table 7.4). CS was predicted to increase in extent in the A1FI, A2 and B2 scenarios. However in the A2 and B2 scenarios RS exhibited a decrease in extent (table 7.4). The predicted increase in extent of CS in the A1FI, A2 and B2 scenarios occured at the expense of open water. In the scenario assuming no eustatic slr the CS plant community was predicted to be absent. However, as mentioned previously the model was unable to predict an increase in the extent of the wetland into areas greater than 0.10m depth, due to limitations in the LiDAR. Therefore the predictions of a decrease in the extent of the lower elevation plant communities were limited to the present day wetland area. Both the CS and RS plant communities were well defined in the model. Validation showed that CS was correctly identified in 86.7% of cases and RS 90% of cases.

Table 7.4: Plant community extent in hectares and as a percentage of the total in five sea level scenarios predicted for 2099 at Tahu. The extent was calculated using estimated mean accretion. Total extent of the wetland is also shown. The figures with a + exhibit an increase from present day and the figures with a – exhibit a decrease. For sea level rise scenarios see table 7.3.

Plant community	2010	A1FI	A2	B2	B1	No eustatic slr
Water (ha)	16.6	4.2-	3.0-	2.1-	1.5-	0.0-
Water (%)	13.7	3.5-	2.5-	1.7-	1.2-	0.0-
CS (ha)	4.1	8.8+	6.7+	4.7+	3.8-	0.0-
CS (%)	3.4	7.3+	5.5+	3.9+	3.1-	0.0-
RS (ha)	8.5	9.6+	6.0-	5.0-	4.3-	0.1-
RS (%)	7.0	7.9+	5.0-	4.1-	3.6-	0.1-
LS (ha)	22.8	22.4-	19.1-	14.8-	10.9-	0.5-
LS (%)	18.8	18.5-	15.8-	12.2-	9.0-	0.4-
US (ha)	35.5	35.9+	36.9+	37.1+	36.9+	2.7-
US (%)	29.3	29.7+	30.5+	30.7+	30.5+	2.2-
TG (ha)	16.8	22.1+	28.6+	33.0+	36.2+	11.4-
TG (%)	13.9	18.3+	23.6+	27.3+	29.9+	9.4-
SW (ha)	16.7	18.0+	20.7+	24.3+	27.4+	106.3+
SW (%)	13.8	14.9+	17.1+	20.1+	22.6+	87.9+
Total (ha)	121.0	121.0	121.0	121.0	121.0	121.0
Total (%)	100.0	100.0	100.0	100.0	100.0	100.0

7.4.3.2 Scenarios assuming an increase in sediment accretion at Tahu in 2099

As in the scenarios assuming no increase in sediment accretion the greatest changes predicted for the Tahu wetland by 2099 occurred in the no eustatic slr scenario followed by B1, B2, A2 and A1FI sequentially (figure 7.4). The outputs from the model at Tahu for 2099 indicated that in all scenarios assuming an increase in sediment accretion the extent of the US, TG and SW communities would increase, except for the scenario assuming no eustatic slr for US and TG and B1 for US (figure 7.4; table 7.5). This increase in the extent of US, TG and SW was, as in the scenarios assuming no increase in sediment accretion, at the expense of LS, RS and CS. The increase in the extent of TG, US and SW was however, greater in the scenarios assuming an increase in sediment accretion (figure 7.3 figure 7.4; table 7.4; table 7.5). As can be seen in the model output (figure 7.4), the greatest change was predicted in the scenario assuming no eustatic slr where the SW community dominated at the expense of all other community types except TG. In this scenario SW covered 91.1% of the wetland area by 2099 compared to 13.8% in 2009 (figure 7.4; table 7.5). In the other scenarios the extent of SW was predicted to increase from 2009 and cover a greater extent than in the scenario assuming no increase in sediment accretion: 17.1% compared with 14.9% (A1FI), 20.9% compared with 17.1% (A2), 24.1% compared with 20.1% (B2), and 28.3% compared with 22.6% (B1) (figure 7.4; table 7.4; table 7.5). In the model validation (chapter 5), the model was able to correctly identify SW at Tahu in 66.7% of cases. In the remainder of cases the observed community was TG (table 5.3 chapter 5).

The greatest increase in the extent of TG was predicted in the B1 scenario rising from 13.9% to 32.9% of the total area (table 7.5). The accuracy of this prediction was likely to be limited due to the model validity for TG, which was able to correctly predict the location of this community in 53.3% of cases. The validation performed in chapter 5 showed that in 20.0% of cases, areas predicted to contain TG were actually US, and in 26.7% of cases LS (table 5.3 chapter 5). The scenarios presented in table 7.5 predicted a greater increase in the extent of TG than in the scenarios assuming no increase in sediment accretion rates except in the scenario assuming no eustatic slr. This loss was due to the predicted encroachment of SW into the TG plant community.

Whilst US was predicted to increase in extent in all modelled scenarios, with the exception of the B1 and no eustatic slr scenarios, the increase was limited. US was predicted to cover 29.3% of the total wetland area at present but was predicted to increase to 30.5% (A1FI), 30.9% (A2), 29.8% (B2) a maximum increase of 1.6% (table 7.5). In the B1 scenario US was predicted to decrease by 2.1% to 27.2% (B1) and in the no eustatic slr scenario a much greater decrease in extent by 27.6% to 1.7% was predicted (table 7.5). In the validation of the model in chapter 5 (table 5.3) US was the most difficult community to predict at Tahu because of similar elevation ranges to LS. More of the predicted US patches were observed to be LS than US, 53.3% and 46.7% respectively.

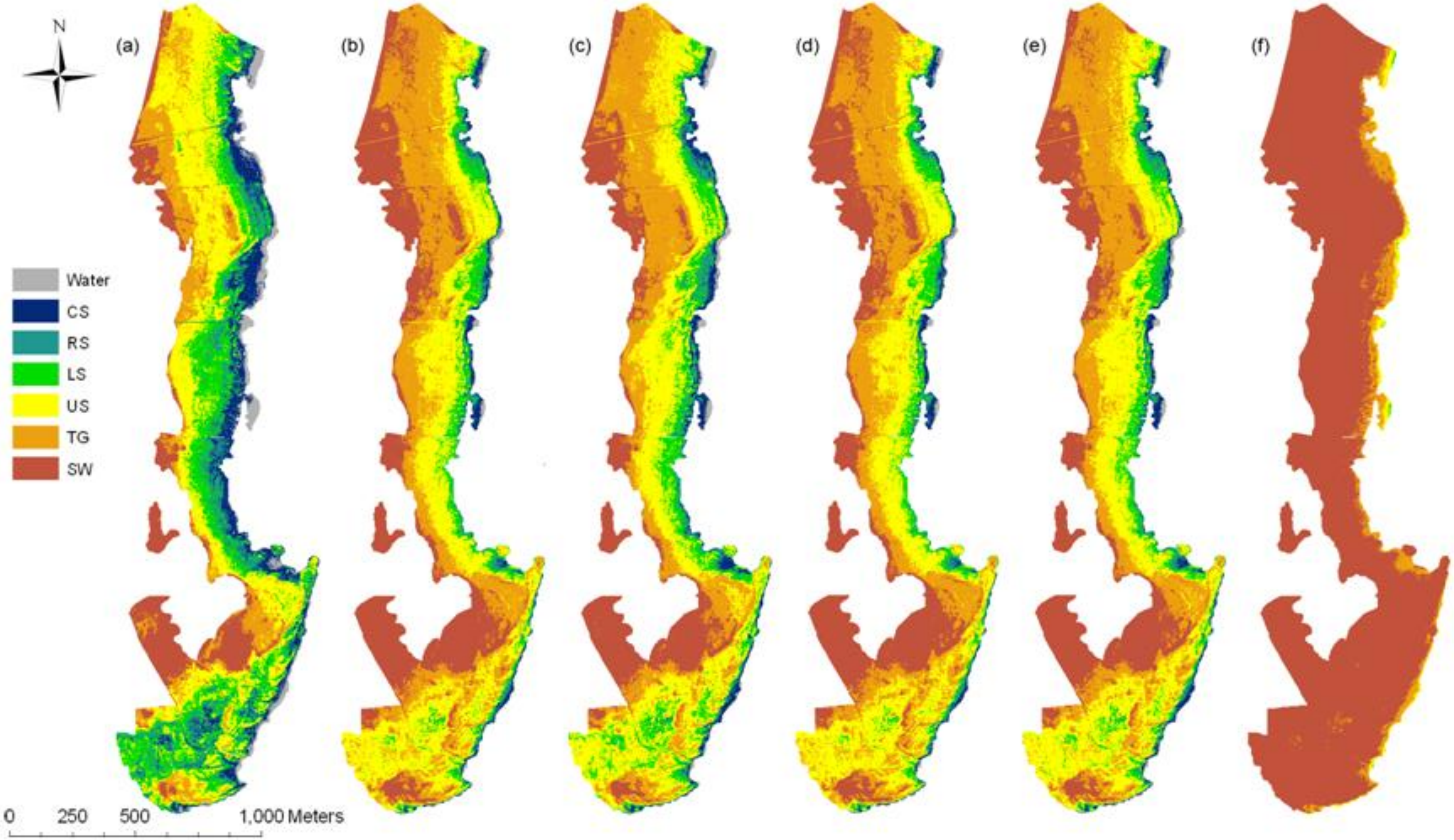

Figure 7.4: Outputs from the plant community model at Tahu assuming an increase in sediment accretion rates. (a) 2009 (b) A1FI, (c) A2, (d) B1, (e) B2 and (f) no eustatic sea level rise (outputs b-f are predictions for 2099). For sea level rise scenarios see table 7.3.

The LS plant community was predicted to decrease in extent in all scenarios although the decrease was limited in the A1FI scenario from 18.8% of the total extent in 2009 to 15.8% by 2099 (table 7.5). The predicted decrease was greater in the scenarios assuming an increase in sediment accretion than those assuming no increase (table 7.5). The greatest loss in extent for LS was predicted in the no eustatic slr scenario, where the extent was reduced to only 0.2% of the total wetland area (table 7.5). In validation the model was able to accurately predict the location of LS in 80% of cases.

Table 7.5: Plant community extent in hectares and as a percentage of the total in five sea level scenarios, by 2099 at Tahu. The extent was calculated using elevated estimated accretion rates in a scenario with increased storminess. Total extent of the wetland is also shown. The figures with a + exhibit an increase in extent from present day and the figures with a – exhibit a decrease from present day. For sea level rise scenarios see table 7.3.

Plant community	2010	A1FI	A2	B2	B1	No eustatic slr
Water (ha)	16.6	3.0-	1.9-	1.3-	1.0-	0.0-
Water (%)	13.7	2.5-	1.6-	1.1-	0.8-	0.0-
CS (ha)	4.1	6.7+	4.3+	3.4-	2.6-	0.0-
CS (%)	3.4	5.5+	3.6+	2.8-	2.1-	0.0-
RS (ha)	8.5	6.0-	4.9-	3.6-	2.6-	0.1-
RS (%)	7.0	5.0-	4.0-	3.0-	2.1-	0.1-
LS (ha)	22.8	19.1-	13.7-	9.5-	7.8-	0.3-
LS (%)	18.8	15.8-	11.3-	7.9-	6.4-	0.2-
US (ha)	35.5	36.9+	37.4+	36.1+	32.9-	2.0-
US (%)	29.3	30.5+	30.9+	29.8+	27.2-	1.7-
TG (ha)	16.8	28.6+	33.5+	37.9+	39.8+	8.4-
TG (%)	13.9	23.6+	27.7+	31.3+	32.9+	6.9-
SW (ha)	16.7	20.7+	25.3+	29.2+	34.3+	110.2+
SW (%)	13.8	17.1+	20.9+	24.1+	28.3+	91.1+
Total (ha)	121.0	121.0	121.0	121.0	121.0	121.0
Total (%)	100.0	100.0	100.0	100.0	100.0	100.0

RS was predicted to decrease in extent in all scenarios assuming an increase in sediment accretion due to encroachment of the higher elevation plant communities US, TG, and SW (figure 7.3). However, due to limitations in the remotely sensed data the model could not predict an increase in RS related to progradation of this community into Tahu Bay. The greatest reduction in extent was predicted in the no eustatic slr scenario from a present day extent of 7% of the total wetland area to 0.1% (table 7.5). CS was predicted to increase in extent in the A1FI and A2 scenarios, whilst in A1FI and A2 RS, which occured at a higher elevation, exhibited a decrease in extent (table 7.5). The predicted increase in the extent of CS in the A1FI and A2 scenarios occured at the expense of the area covered at present by water. In the no eustatic slr scenario the CS plant community was predicted to be completely absent. However, as mentioned previously the model was unable to predict any increase in extent of the wetland seawards due to limitations in the LiDAR. Therefore the predictions of a decrease in the extent of the lower elevation plant communities were limited to the present day wetland area. Both the CS and RS plant communities were well defined in the model. Validation showed that CS was correctly identified in 86.7% of cases and RS 90% of cases at Tahu.

7.4.4 Estimated location and extent of the plant communities at Matsalu in 2099

The model predictions for Matsalu in 2099 have been divided into those assuming an increase in sediment accretion and those not. In the scenario assuming no eustatic slr using both accretion rate estimates local sea level was predicted to decrease due to isostatic uplift and sediment accretion (table 7.3). In A1FI, A2 and B2 scenarios assuming no increase in sediment accretion rates, local sea level was predicted to increase, whereas in the B1 scenario local sea level was predicted to decrease (table 7.3). In the A1FI and A2 scenarios assuming an increase in sediment accretion, local sea level was predicted to increase, whereas in the B1 and B2 scenarios, local sea level was predicted to decrease (table 7.3).

7.4.4.1 Scenarios assuming no increase in sediment accretion at Matsalu in 2099

In the A1FI, A2 and B2 scenarios assuming no increase in sediment accretion at Matsalu, the model outputs suggest a decrease in the total extent of wetland plant communities by 2099 of 3.1%, 1.5% and 0.3% respectively compared to the present total wetland area (table 7.6). This was due to inundation of the lower elevation communities to the north of the wetland (figure 7.5). In the B1 and no eustatic slr scenarios the model indicated an increase in the total extent of the plant communities of 0.1% and 0.3% respectively. However, the increase in the total extent of the wetland covered by the plant communities predicted in the model was likely to be greater than predicted in the no eustatic slr scenario. This was due to the limited ability of the model to predict elevation in areas inundated to a depth greater than 0.10m. In the no eustatic slr scenario, the model predicted a decrease in local sea level of 0.261m (table 7.3) and hence the total wetland extent was likely to increase. However, as noted earlier this was not possible to predict using a terrestrial LiDAR based model.

In the A1FI, A2 and B2 scenarios the model suggested an increase in the extent of the lower elevation RS and LS communities at the expense of the higher elevation US and TG, which was most obvious in the north west and central areas of the wetland (table 7.6; figure 7.5). The LS community was predicted to have the greatest increase in extent in the A1FI, A2 and B2 scenarios from 19.3% of the total wetland area at present to 28.2%, 24.7%, and 21.2% respectively (table 7.6). The ability of the model to correctly identify the plant communities at Matsalu was greater than at Tahu using the orthophotos and ground truthing for model verification (Kappa 0.93) (chapter 5). The model was able to correctly identify the LS plant community in 80% of cases, in all incorrectly predicted cases the observed plant community was US. Whilst the model predicted an increase in the extent of LS, and simultaneously a decrease in US, in the A1FI, A2 and B2 scenarios this was only with an 80% prediction certainty (table 5.5 chapter 5).

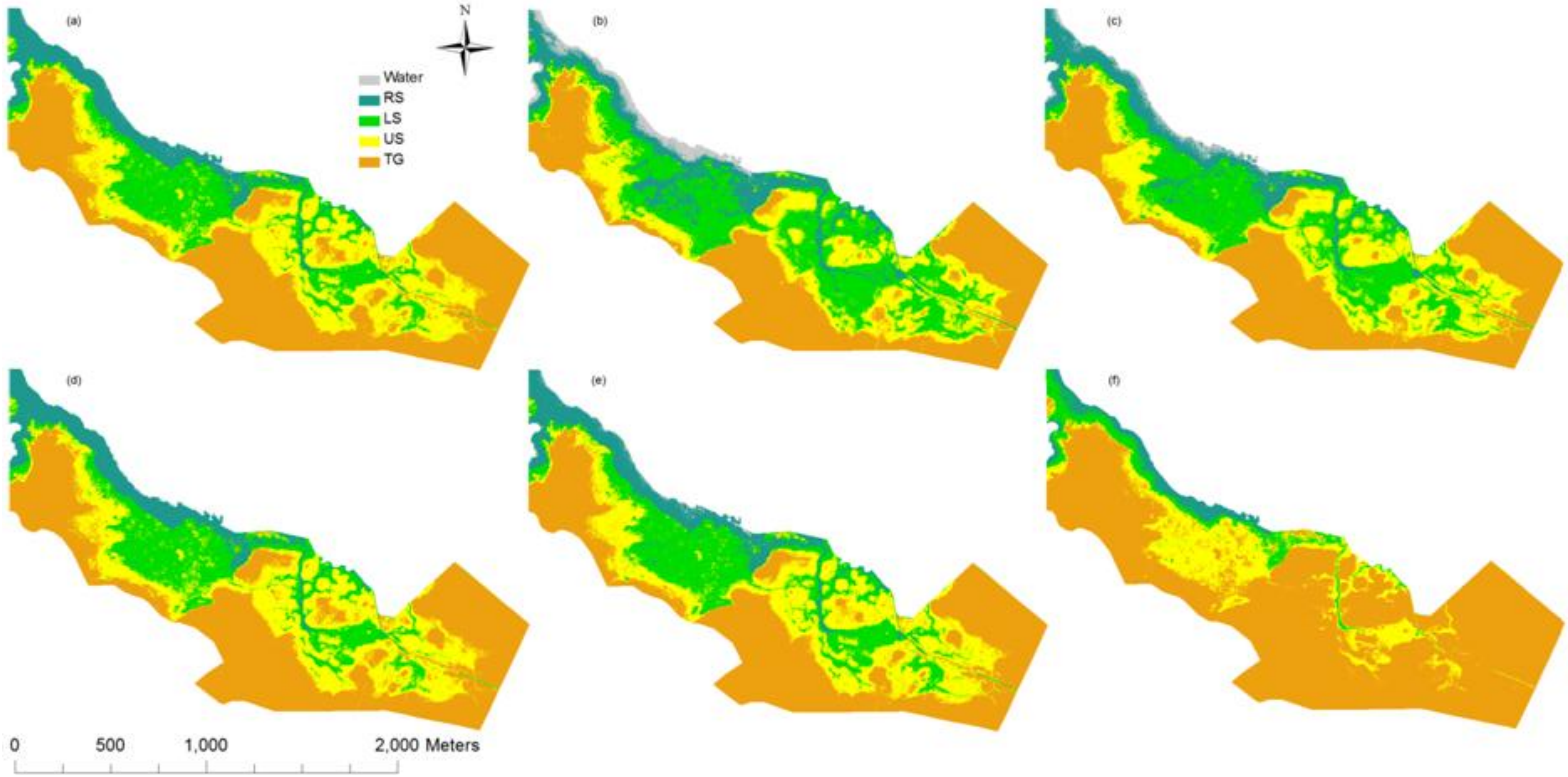

Figure 7.5: Outputs from the plant community model at Matsalu assuming no increase in sediment accretion rates. (a) 2009 (b) A1FI, (c) A2, (d) B1, (e) B2 and (f) no eustatic sea level rise (output a is for 2009, b-f are predictions for 2099). For sea level rise scenarios see table 7.3.

The RS community was predicted to increase from 9.2% of the total wetland area at present to 11.8% (A1FI), 10.3% (A2), and 9.7% (B2). However, in these scenarios the model predicts that some areas previously covered by RS are predicted to be inundated, suggesting a migration inland to higher elevation areas previously covered by LS (figure 7.5). The model developed for Matsalu was able to predict the location of the RS plant community in 100% of cases (table 5.5 chapter 5.

The model outputs indicated that in the A1FI, A2 and B2 scenarios both the TG and US plant communities were predicted to decrease with TG exhibiting a greater decrease than US (table 7.6; figure 7.5). TG was predicted to decrease from a present day extent of 44.7% of the total wetland area to 36.1% (A1FI), 39.1% (A2), and 42.2% (B2) (table 7.6). In the model validation in chapter 5, the plant community model developed for Matsalu was able to accurately predict the location of TG in 100% of cases (table 5.5; chapter 5), and hence the model prediction for TG was likely to be robust.

US was predicted to decrease from a present day extent of 26.5% of the total wetland area to 20.6% (A1FI), 24.2% (A2), and 26.3% (B2) (table 7.6). The decrease in the higher elevation plant communities in the A1FI, A2, and B2 scenarios was due to encroachment of the lower elevation plant communities and is an example of coastal squeeze linked to sea level rise (figure 7.5). The reliability of the model for predicting US was high, as 86.7% of cases were correctly identified (table 5.5 chapter 5). Of the cases where US was wrongly predicted to occur in the static model at Matsalu, 6.6% were observed to be LS and 6.6% were TG, both proximal plant communities with regards to elevation (table 5.5 chapter 5).

In the B1 scenario the model predicted limited change in the location of the plant communities from present. The model predicted a small increase in the extent of TG, from 44.7% of the total wetland area at present to 45.1%, at the expense of LS, which decreased from 19.3% to 18.6% (table 7.6). In the B1 scenario the model predicted no change in the extent of the RS or US plant communities (figure 7.5).

Table 7.6: Predicted plant community extent by 2099 in hectares and as a percentage of the total in five sea level scenarios using the estimated mean accretion rates at Matsalu. Total extent of the wetland is also shown. If there was a predicted increase in extent from present day this was designated with a +, where there was a predicted decrease in the area from present day this was designated with -. For sea level rise scenarios see table 7.3.

Plant community	2010	A1FI	A2	B2	B1	No eustatic slr
Water (ha)	0.7	3.2+	0.6-	0.0-	0.0-	0.0-
Water (%)	0.4	1.7+	0.3-	0.0-	0.0-	0.0-
RS (ha)	17.5	19.7+	18.4+	17.0-	15.4-	7.2-
RS (%)	9.2	10.3+	9.6+	8.9-	8.1-	3.8-
LS (ha)	36.9	47.2+	39.4+	33.0-	27.7-	11.4-
LS (%)	19.3	24.7+	20.6+	17.3-	14.5-	6.0-
US (ha)	50.6	46.3-	50.5-	50.3-	49.5-	30.5-
US (%)	26.5	24.2-	26.4-	26.3-	25.9-	16.0-
TG (ha)	85.4	74.7-	82.2-	90.8+	98.5+	142.0+
TG (%)	44.7	39.1-	43.0-	47.5+	51.5+	74.3+
Total (ha)	191.1	191.1	191.1	191.1	191.1	191.1
Total (%)	100	100	100	100	100	100

Of the scenarios assuming no change in sediment accretion the greatest difference in the location and extent of the plant communities was predicted in the no eustatic slr scenario (figure 7.5; table 7.6). In the no eustatic slr scenario the model predicted that the TG plant community would greatly increase in extent from 44.7% of the total wetland area to 80.5% by 2099 (table 7.6). This increase was predicted to occur at the expense of the US, LS and RS plant communities, which would decrease from 26.5% to 11.3%, 19.3% to 6.4% and 9.3% to 1.8% respectively. The model outputs show the greatest loss of RS and LS in the north west of the wetland and US in the west of the wetland (figure 7.5). Whilst this increase in the TG plant community was likely to be valid within the present day extent of the wetland, the inability of the model to predict a movement of plant communities into areas that are at present inundated to depths greater than 0.10m means that the predicted decrease in the extent of the US, LS and RS plant communities was unlikely to be wholly valid in this scenario.

7.4.4.2 Scenarios assuming an increase in sediment accretion at Matsalu in 2099

The model outputs for scenarios assuming an increase in sediment accretion at Matsalu by 2099 suggest a decrease in the total area of the wetland plant communities in the A1FI and A2 (2.9% and 0.3% respectively), and an increase in the B2, B1 and no eustatic slr scenarios (0.3% for all scenarios) (table 7.7). The decrease in the total extent was due to inundation of the lower elevation plant communities, which was predicted to occur in the north of the wetland (figure 7.6). The increase in the area of the wetland covered by the plant communities predicted in the B2, B1 and no eustatic slr scenarios was due to the expansion of RS into formerly inundated areas to the north of the site (figure 7.6). In scenarios where a decrease in sea level greater than 0.10m was predicted, such as in the no eustatic slr scenario, the model was limited in its ability to predict the extent and location of the lower elevation plant communities. Thus, the predicted decrease in the extent of RS and LS for the no eustatic slr scenario was not likely to be robust.

In the A1FI and A2 scenarios the model suggested an increase in the extent of the lower elevation RS and LS communities at the expense of the higher elevation US and TG. These differences were most obvious in the north west and centre of the wetland (table 7.7; figure 7.6). The LS community was predicted to have the greatest increase in extent in the A1FI and A2 scenarios from 19.3% of the total wetland area at present to 24.7% and 20.6% respectively (table 7.7). The ability of the model to correctly identify the plant communities at Matsalu was greater than at Tahu using the orthophotos and ground truthing (Kappa 0.93) (chapter 5). The model was able to correctly identify the LS plant community in 80% of cases, and in all incorrectly predicted cases the observed plant community was US. Whilst the model predicted an increase in the extent of LS, and simultaneously a decrease in US, in the A1FI and A2 scenarios, model validation suggested that this only had an 80% prediction certainty (table 5.5 chapter 5).

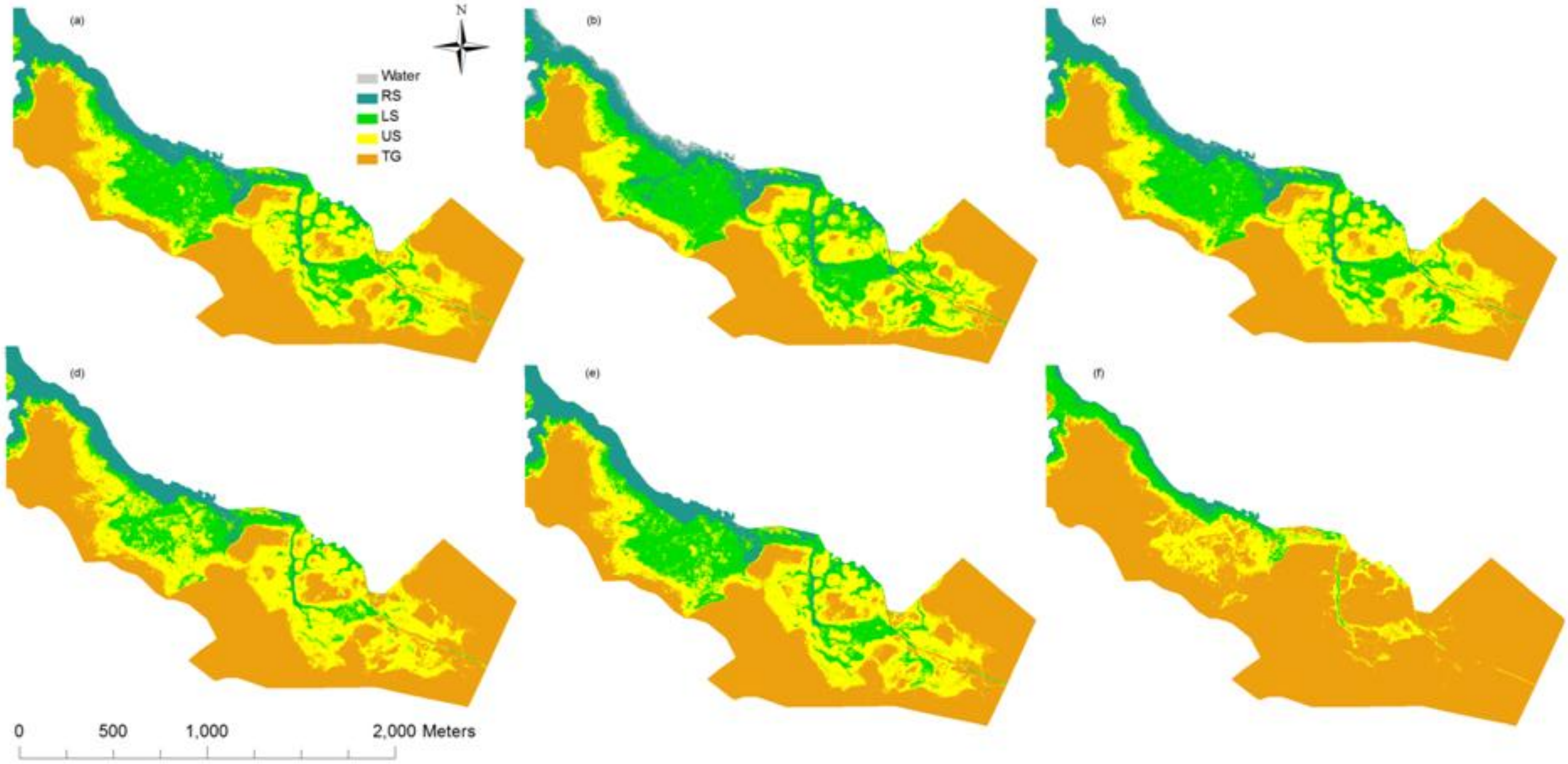

Figure 7.6: Outputs from the plant community model at Matsalu assuming an increase in sediment accretion rates. (a) 2009 (b) A1FI, (c) A2, (d) B1, (e) B2 and (f) no eustatic sea level rise (output a is for 2009 and b-f are predictions for 2099). For sea level rise scenarios see table 7.3.

The RS plant community was predicted to increase from 9.2% of the total wetland area to 10.3% (A1FI) and 9.6% (A2). However, in the A1FI and A2 scenarios some areas previously covered by RS were predicted to be inundated (figure 7.6) suggesting a movement landwards into areas formerly suitable for the LS plant community. The model developed for Matsalu was able to predict the location of the RS plant community in 100% of cases (table 5.5 chapter 5) and hence model validity was high.

The model outputs indicated that in the A1FI and A2 scenarios both the TG and US plant communities would decrease in extent. TG was predicted to exhibit a greater decrease than US (table 7.7; figure 7.6), from a present day extent of 44.7% of the total wetland area to 39.1% (A1FI) and 43.0% (A2) (table 7.7). In the model validation, the plant community model developed for Matsalu was able to accurately predict the location of TG in 100% of cases (table 5.5; chapter 5), and hence the model prediction for TG is robust.

US was predicted to decrease in extent in the A1FI and A2 scenarios from a present day extent of 26.5% of the total wetland area to 24.2% and 26.4% respectively (table 7.7). This decrease in the higher elevation plant communities at the expense of the lower elevation plant communities was a typical example of coastal squeeze linked to sea level rise, and can be most clearly seen in the centre and north west of the Matsalu wetland (figure 7.6). The reliability of the model for predicting US was quite high, 86.7% of cases were correctly identified (table 5.5 chapter 5). Of the cases where US was wrongly predicted at Matsalu 6.6% were observed to be LS and 6.6% were TG (table 5.5 chapter 5).

In the B1, B2 and no eustatic slr scenarios assuming no increase in sediment accretion the model predicted an increase in the extent of TG by 2099, from 44.7% of the total wetland area at present to 51.5%, 47.5% and 74.3% respectively, at the expense of US, LS and RS (table 7.7). In all these scenarios, LS was predicted to experience the greatest decrease in extent compared to the present from 19.3% to 17.3%, 14.5% and 6.0% respectively (table 7.7). The outputs from the model indicated that loss of LS would occur in the centre and north west of the Matsalu wetland. Both RS and US were predicted to experience limited loss in the B2 and B1 scenarios (RS 0.4% and 1.2%, and US 0.6% and 0.2% respectively) (table 7.7). However, in the no eustatic slr scenario assuming an increase in sediment accretion at Matsalu, the decrease in extent for the RS and US plant communities was predicted to be much greater than in the B2 and B1 scenarios (5.5% and 20.1% respectively) (table 7.7) and can be most clearly seen in the north west of the wetland (figure 7.6).

Table 7.7: Predicted plant community extent in hectares and as a percentage of the total in five sea level scenarios, by 2099 at Matsalu. The extent was calculated using elevated estimated accretion rates in a scenario with increased storminess. Total extent of the wetland is also shown. The figures with a + exhibit an increase in area from present day and the figures with a – exhibit a decrease in extent from present day. For sea level rise scenarios see table 7.3.

Plant community	2010	A1FI	A2	B2	B1	No eustatic slr
Water (ha)	0.7	6.4+	3.2+	1.0+	0.2-	0-
Water (%)	0.4	3.3+	1.7+	0.5+	0.1-	0-
RS (ha)	17.5	22.6+	19.7+	18.6+	17.8+	3.4-
RS (%)	9.2	11.8+	10.3+	9.7+	9.3+	1.8-
LS (ha)	36.9	53.8+	47.2+	40.6+	35.7-	12.3-
LS (%)	19.3	28.2+	24.7+	21.2+	18.6-	6.4-
US (ha)	50.6	39.3-	46.3-	50.2-	50.6	21.5-
US (%)	26.5	20.6-	24.2-	26.3-	26.4-	11.3-
TG (ha)	85.4	69.0-	74.7-	80.7-	87.2+	153.9+
TG (%)	44.7	36.1-	39.1-	42.2-	45.1+	80.5+
Total (ha)	191.1	191.1	191.1	191.1	191.1	191.1
Total (%)	100.0	100.0	100.0	100.0	100.0	100.0

7.4.5 Estimated location and extent of the plant communities at Kudani in 2099

The model predictions for Kudani in 2099 do not take into account sediment accretion due to limited access to an allochthonous sediment source. In the scenario assuming no eustatic slr, local sea level was predicted to decrease due to isostatic uplift (table 7.3). In the A1FI, A2 and B2 scenarios local sea level was predicted to increase, whereas in the B1 scenario local sea level was not predicted to change due to isostatic uplift compensating eustatic sea level rise (table 7.3).

7.4.5.1 Scenarios at Kudani in 2099

The model outputs indicated that in the A1FI, A2 and B2 scenarios there will be a decrease in the total extent of the Kudani wetland plant communities by 2099. In the B1 scenario the model suggested that there will be no change in the location or extent of any of the plant communities due to isostatic uplift compensating sea level rise. In the no eustatic slr scenario the model predicts that the CS and RS communities will be lost from the wetland due to a decrease in local sea level. However, the model was based on terrestrial LiDAR and hence was limited in its ability to predict the location and extent of lower elevation plant communities in situations with a decrease in local sea level greater than 0.10m. In the no eustatic slr scenario, local sea level was predicted to lower by 0.252m (table 7.3) and hence the predictions for this scenario were not likely to be valid for the CS, RS and LS plant communities. With the exception of the no eustatic slr scenario the model for Kudani has been shown to be a robust predictor of plant community, Kappa = 0.81 (table 5.5 chapter 5).

The model suggests that in the A1FI, A2 and B2 scenarios there will be an increase in the extent of both CS and RS. The increase in the extent of CS and RS was predicted to be greatest in the A1FI (by 8.5% and 4.0% respectively), followed by A2 (by 4.7% and 3.5% respectively) and B2 (by 2.2% and 1.2% respectively) (table 7.8). The majority of this increase in CS and RS was predicted to occur in the north of Kudani in the areas adjacent to Võõlameri (figure 7.7) The model was able to accurately predict the location of the CS and RS plant communities in 100% of cases at Kudani (table 5.5 chapter 5) and hence these predictions are likely to be highly valid for these three scenarios. In the A1FI scenario the increase in the extent of the CS and RS was predicted to be at the expense of LS, which decreased by 1.6% of the total wetland, US which decreased by 3.2%, TG which decreased by 4.2%, and SW which decreased by 8.5% (table 7.8). The model predictions for the LS plant community were shown to be correct in 80% of observations with the remaining incorrectly predicted observations occurring in the US (table 5.5 chapter 5). In the US and TG plant communities the model was only able to correctly predict plant community type in 60% of cases. For US 20% of the incorrect predictions were observed to be located in LS and 20% in TG. In 26.7% of the cases in which the model predicted TG the observed plant community was US and 13.3% LS. The model was able to accurately predict the location of SW in 80% of cases. In 13.3% of the cases in which the model predicted SW the observed plant community was TG and in 6.7% of cases US.

In the A2 and B2 scenarios the model predicted a limited increase in the extent of LS from 40.2% of the total wetland area in 2009 to 40.7% (A2) and 41.4% (B2) (table 7.8). The increase in the extent of CS, RS and LS predicted for the A2 and B2 scenarios was at the expense of US which decreased from 12.5% in 2009 to 10.3% (A2) and 11.4% (B2), TG from 11.7% to 8.9% (A2) and 10.5% (B2), and SW from 29.2% to 24.0% (A2) and 26.6% (B2) (table 7.8). In the A2 and B2 scenarios the US, TG and SW plant communities were therefore predicted to undergo coastal squeeze by a road to the south of the wetland. Most of the plant community loss was predicted to occur in the southwest and east of the Kudani wetland (figure 7.7).

The model outputs for the no eustatic slr scenario suggest an increase in the extent of the SW (from 29.3% of the total area predicted for 2009 to 57.7% in 2099), TG (11.7% to 12.7%) and US (12.5% to 13.8%). Increases were most obvious in the southern and central sections of the Kudani wetland (figure 7.7). The increase in the SW, TG and US plant communities was predicted to occur at the expense of LS which decreased from 40.2% of the total wetland area in 2009 to 15.7% in 2099 (table 7.8), and led to the loss of the CS and RS communities from the present day wetland area.

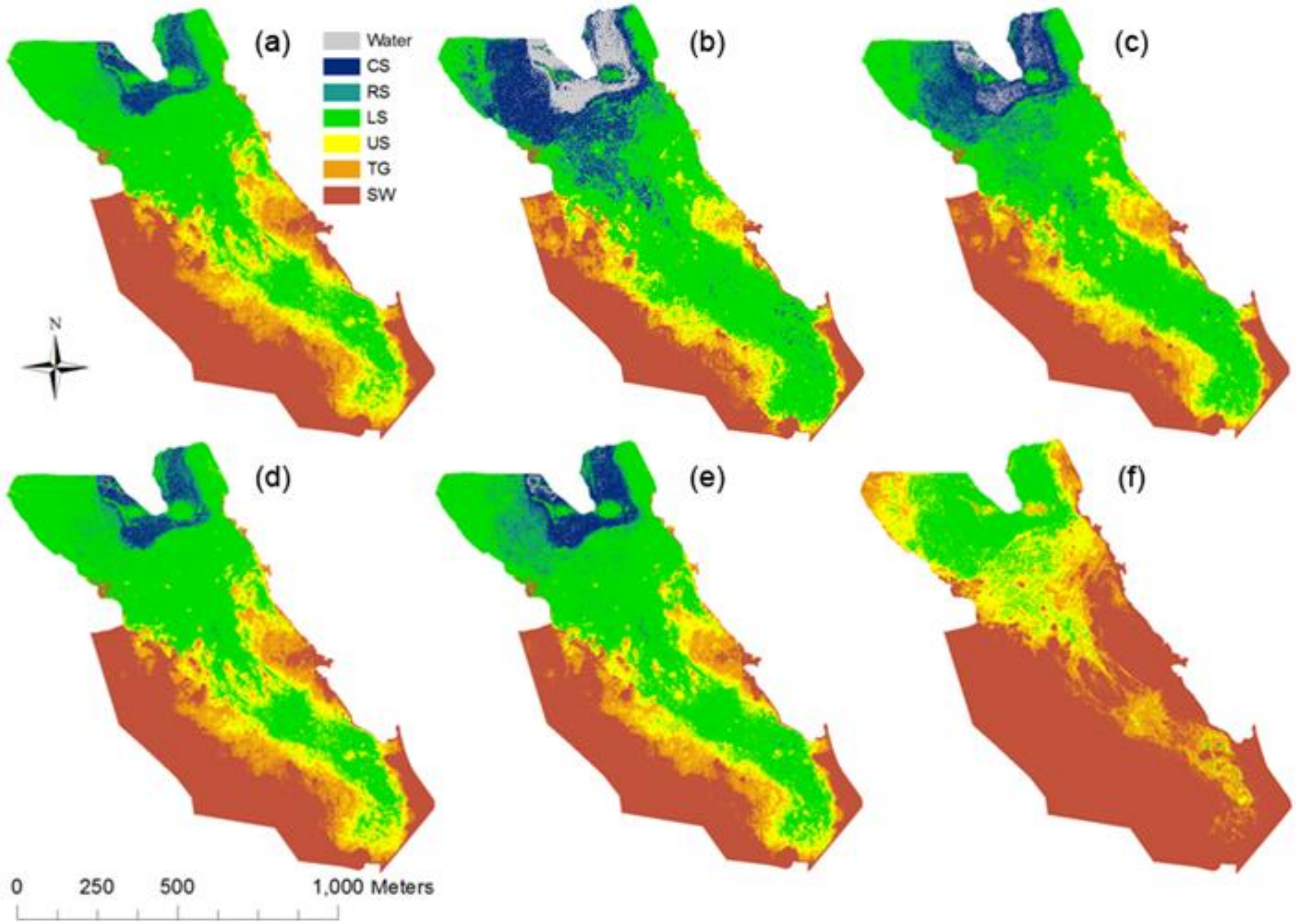

Figure 7.7: Outputs from the plant community model at Kudani. (a) 2009 (b) A1FI, (c) A2, (d) B1, (e) B2 and (f) no eustatic sea level rise (output a is for 2009 and b-f are predictions for 2099). For sea level rise scenarios see table 7.3.

Table 7.8: Predicted plant community extent in hectares and as a percentage of the total in five sea level scenarios, by 2099 at Kudani. Total extent of the wetland is also shown. The figures with a + exhibit an increase in area from present day and the figures with a – exhibit a decrease in extent from present day. For sea level rise scenarios see table 7.3.

Plant community	2010	A1FI	A2	B2	B1	No eustatic slr
Water (ha)	0.1	4.1+	1.3+	0.3+	0.1	0.0-
Water (%)	0.1	5.2+	1.7+	0.4+	0.1	0.0-
CS (ha)	2.3	9.0+	6.0+	4.0+	2.3	0.0-
CS (%)	2.9	11.4+	7.6+	5.1+	2.9	0.0-
RS (ha)	2.7	5.8+	5.4+	3.6+	2.7	0.0-
RS (%)	3.4	7.4+	6.9+	4.6+	3.4	0.0-
LS (ha)	31.6	30.4-	32.1+	32.6+	31.6	12.4-
LS (%)	40.2	38.6-	40.7+	41.4+	40.2	15.7-
US (ha)	9.8	7.3-	8.1-	9.0-	9.8	10.9+
US (%)	12.5	9.3-	10.3-	11.4-	12.5	13.8+
TG (ha)	9.2	5.9-	7.0-	8.3-	9.2	10.0+
TG (%)	11.7	7.5-	8.9-	10.5-	11.7	12.7+
SW (ha)	23	16.3-	18.9-	21.0-	23	45.5+
SW (%)	29.3	20.7-	24.0-	26.6-	29.3	57.7+
Total (ha)	78.8	78.8	78.8	78.8	78.8	78.8
Total (%)	100	100	100	100	100	100

7.5 Discussion

7.5.1 Estimates of sea level rise, storminess and sediment accretion

The results from the model used in this study suggests that sea level rise will be partially offset in Estonian coastal wetlands by isostatic uplift (Kont *et al*., 2003), although the extent of this was dependent upon site location. Kont *et al*. (1997) suggested that the Tahu and Kudani wetlands are less likely to be affected by sea level rise than the more southern coastal site at Matsalu due to greater rates of uplift. However, the study in chapter 6 has indicated that at times the accretion rates in Tahu and Matsalu exceed isostatic uplift, a factor that was not included in the Kont *et al*. (1997) study. Assuming increased storminess in the lower elevations of the Tahu site, accretion rates could have a stronger influence on wetland development than isostatic uplift. Previous studies have shown that sea level rise can drive coastal wetland progradation, particularly when linked with increased storminess (Reed, 1989; Reed, 1990; Ratas & Puurmann, 1995; Reed, 1995; Roman *et al*., 1997). The deposition of sediment on coastal wetland surfaces typically only occurs during inundation events (Reed, 1989) and Roman *et al*. (1997) have shown that, in tidal coastal wetlands, during storm free periods sediment accretion did not exceed 6.7mm/yr, yet during increased storm activity sediment accretion rates were found to increase to up to 24mm/yr. Boorman (1992) suggests that an increase in storminess will result in increased erosion of coastal wetlands due to the formation of larger and more powerful waves. Reed (1995) disputes this and suggests that erosion of the wetland surface sediment is unlikely to occur as long as the vegetation cover remains intact which is in agreement with other studies (Wang *et al*., 1993; Leonard *et al*., 1995). Furthermore large and powerful waves are unlikely to form over the short fetch of Võõlameri (maximum 1000m), Tahu Bay (maximum 2500m) and to a lesser extent Matsalu Bay (maximum 12 000m).

7.5.2 Predictive modelling at Tahu

The Tahu site was predicted to exhibit a decrease in sea level at the littoral edge of the wetland under all scenarios irrespective of accretion rate, due to both high sediment accretion and isostatic uplift. Isostatic uplift rates (-0.252m to 2099) compensate for sea level rise at Tahu in the B1 scenario (+0.252m to 2099) although not in the A1FI (+0.387m to 2099), A2 (+0.333m to 2099) or B2 (+0.288m to 2099) scenarios. Therefore, if sediment accretion was omitted from the modelling at Tahu the model would predict a loss of some of the wetland area, as was predicted at Kudani. This highlights the importance of sediment accretion in coastal wetland modelling, a factor which has been neglected in a variety of previous studies investigating the impacts of sea level rise on coastal wetlands (Kont *et al*., 1997; Kont *et al*., 2003; Poulter & Halpin, 2008; Moeslund *et al*., 2011).

7.5.3 Predictive modelling at Matsalu

The Matsalu site is located 30km further south than either Tahu or Kudani, and therefore has lower isostatic uplift rates to offset eustatic sea level rise (Vallner *et al*., 1988; Ekman, 1996; Eronen *et al*., 2001; Lidberg *et al*., 2007). The uplift rates that are estimated for Matsalu, 0.180m over the period 2009 to 2099, are insufficient on their own to compensate for sea level rise in any of the scenarios except no eustatic slr. Matsalu has an unmodified hydrological system and is predominantly depositional (Puurmann & Ratas, 1998). The model predicts that the Matsalu site would experience local sea level rise under the A1FI, A2 and B2 scenarios, assuming no increase in sediment accretion. However, under the B1 scenario accretion rates together with isostatic uplift exceed eustatic sea level rise such that local sea level was predicted to decrease. Assuming increased storminess, and therefore increased accretion rates, local sea level increases under the A1FI and A2 scenarios but falls under the B1 and B2 scenarios. As with both Tahu and Kudani, the Matsalu wetland was predicted to experience a considerable decrease in sea level between 2009 and 2099 in both the sediment accretion scenarios assuming no change in eustatic sea level (-0.261m assuming no increase and -0.324m assuming an increase in sediment accretion). This is likely to lead to progradation into the adjacent water. However, due to limitations in the terrestrial LiDAR data upon which the model was based the extent of the progradation was beyond the ability of the model to predict.

7.5.4 Predictive modelling at Kudani

At Kudani the model predicted a decrease in the total extent of the wetland covered in the A1FI, A2 and B2 scenarios, where sea level rise exceeded isostatic uplift. The isostatic uplift rates at Kudani are equal to those estimated for Tahu (Vallner *et al*., 1988) although with no predicted sediment accretion at Kudani, the wetland was predicted to undergo an increase in local sea level in the A1FI, A2 and B2 scenarios. In the B1 scenario at Kudani, as at Tahu, isostatic uplift was predicted to compensate sea level rise, and with no sediment accretion predicted to occur this would result in no change in the location or extent of any of the plant communities. In the no eustatic slr scenario a decrease in local sea level of 0.252m by 2099 was predicted at Kudani. Although there was likely to be an increase in inundation frequency in the A1FI, A2 and B2 scenarios, most particularly in the event of increased storminess, access between Võõlameri and the Gulf of Finland is through a 532m-long and 4m-wide channel which terminates in a 2m-wide pipe. This is likely to substantially limit the supply of allochthonous sediment which in the other sites was predicted to prevent considerable wetland loss due to local sea level rise. A study by Kont *et al*. (2003) suggested that there would be a likely loss in the extent of Estonian coastal wetlands following sea level rise, as was indicated by the model for Kudani by 2099 in the A1FI, A2 and B2 scenarios. However, the Kont *et al*. (2003) study used coarse elevation data and only made a prediction for the wetland area as a whole, not for the individual plant communities which contain significantly different vegetation (Burnside *et al*., 2007) and support a variety of different bird species (Mägi *et al*., 2004; Ward, 2007).

7.5.5 Model uncertainties

While the model was not able to predict any increase in extent beyond the present day wetland area, all three sites are likely to increase in total extent in at least one scenario by 2099 by expanding out into adjacent water. This is in character with the historical development of coastal landscapes in the north eastern Baltic as many coastal mainland areas were previously islands (Eronen *et al*., 2001) including the Tahu peninsula (Eesti Maa-amet, 2011). Historical maps show that some areas of southern Tahu were formerly islands that, due to wetland progradation, have coalesced with the mainland wetland (Eesti Maa-met, 2011). However, the LiDAR data, which formed the basis for the plant community model, were collected with a device unable to penetrate more than 0.10m below the surface of the sea, typical of eye-safe terrestrial LiDAR (Höfle *et al*., 2009; McNair, 2010; Mather & Koch, 2011; Pe'eri *et al*., 2011). The lowest elevation at which CS occurred was 0.03m below mean sea level (table 5.2 chapter 5). Thus, the robustness of the predictive models in scenarios with any decrease in sea level was limited for the lower elevation plant communities beyond the area of the present wetland.

Several authors have suggested that sediment accretion may be able to counterbalance the effects of eustatic sea level rise (Reed, 1995; Roman *et al*., 1997; Nielsen & Nielsen, 2002; Pedersen & Bartholdy, 2007; Kirwan & Murray, 2007). This study has shown that sediment accretion can have a profound effect on coastal wetland development, particularly in conjunction with isostatic uplift. However, the sediment accretion data in this study were collected from one LS and one TG core from the Tahu and Matsalu sites. Even under apparently similar conditions sediment accretion rates are known to experience high spatial and temporal variation (Cahoon & Reed, 1995; Roman *et al*., 1997). Thus, the accuracy of the sediment accretion data as representative of whole wetland system was likely to be limited although it has provided a good indication of sediment accretion rates for the first time in Estonia. At Kudani no sediment accretion rates were added into the model due to the limited access of the site to an allochthonous sediment supply. While it is unlikely that Kudani will accumulate sediment from external sources, there is likely to be some organic soil accumulation over the 90-year period, although this will be limited due to the relatively low productivity of the higher elevation plant communities in Baltic coastal wetlands in comparison to many other regional ecosystems (Tyler, 1971a).

The predictions of sea level rise used in this model were derived from the mid-range figures presented in the IPCC (2007) report. No likelihood was attributed to any of the IPCC (2007) climate change or sea level rise scenarios. The range of sea level rise for each climate change scenario by 2099 was relatively broad A1FI (0.26-0.59m), A2 (0.23-0.51m), B1 (0.18-0.38m), and B2 (0.20-0.43m) (table 7.1). Therefore, these ranges in predicted sea level rise introduced uncertainty into the model which should be acknowledged in the interpretation of the predictions.

The model does not take into account the ability of the indicator plant species to adapt to changes in soil hydrology nor the rapidity with which species can colonise new appropriate habitats. However, previous studies have shown that the main indicator species for the CS and RS plant communities are able to rapidly colonise suitable habitats via vegetative propagation (Weisner & Ekstam, 1993; Coops *et al*., 1994; Charpentier *et al*., 1998; Clevering & Hundscheid, 1998; Charpentier *et al*., 2000). Other authors have shown that whilst sexual colonisation is slower than vegetative, both *Bolboschoenus maritimus* and *Scirpus lacustris*, the main indicator species for CS, and *Phragmites australis*, can sexually colonise newly formed appropriate habitats fairly rapidly over periods of one or two growing seasons (Coops &

van der Velde, 1995; Clevering *et al*., 1996; Mauchamp *et al*., 2001; Alvarez *et al*., 2005) and hence much faster than the predicted sea level changes. The seed bank of Baltic coastal wetlands is considered to be highly developed and able to rapidly respond to changed water levels (Jerling, 1983; Jerling, 1988b; Jutila 1997; Jutila 1999). Jutila (2001) found that the majority of the species that occur in the CS, RS, LS, US and TG plant communities exist within the seed bank of the sublittoral, epilittoral and eulittoral zones, and that germination of the main indicator species following flooding or draw down disturbance occurred rapidly for the species in appropriate conditions. The indicator species that occur in the SW community, *Juniperus communis* and *Pinus sylvestris*, were not noted in the seed bank in previous studies. However, in studies concerning the cessation of management in Baltic coastal wetlands, it has been shown that SW species can rapidly colonise areas that are unmanaged, typically 20-25 years to become fully established (Ratas & Puurmann, 1995; Jutila, 2001; Berg, 2009; Antso *et al*., 2011) and it can be assumed that this would be the case following a decrease in local sea level. This would be particularly likely in the Tahu and Kudani wetlands which are bordered by scrub and woodland species and hence there is an appropriate seed source, but less likely at Matsalu where the SW plant community is absent and there is no immediately adjacent seed source. In scenarios where local sea level was likely to rise, such as was predicted to occur in the Kudani coastal wetland, the SW plant community is unlikely to rapidly die out and be replaced by more typical lower elevation vegetation even following a rise in the water table. This is due to the tolerance to high water levels of established individual plants of *P. sylvestris* and *J. communis* (Carlisle & Brown, 1968; Diotte & Bergeron, 1989).

The static model developed in chapter 5 makes an assumption that the wetlands undergo regular management in the form of grazing or mowing. Previous studies have shown that following the cessation of management Estonian coastal wetlands revert to later succesional plant communities (Tyler, 1969; Rebassoo, 1975; Berg, 2009). This typically involves the encroachment of scrub vegetation into areas formally covered by TG and eventual succession to woodland (Puurmann & Ratas, 1998; Burnside *et al*., 2007). In addition the mid-elevation plant communities LS and US rapidly become encroached by *Phragmites australis* and eventually the RS plant community dominates (Jutila, 1999; Burnside *et al*., 2007). The cessation of grazing and mowing management ultimately generally leads to a decrease in plant species richness and habitat variability (Jutila, 1997; Burnside *et al*., 2007; Berg, 2009).

7.5.5.1 Model uncertainties at Tahu

In the model for Tahu, uncertainties were predicted to affect only the CS plant community in the A1FI and A2 scenarios using both sediment accretion estimations and the B2 scenario assuming no increase in sediment accretion. This was due to the extent of the decrease in sea level being less than the range at which CS was predicted to occur, i.e. ≤-0.07m (table 5.2 chapter 5) plus the depth at which LiDAR was still valid (0.10m) (total -0.17m). The model at Tahu was unable to accurately predict the increase in the extent of both CS and RS in the B1 scenarios using both sediment accretion estimations and in the B2 scenario assuming an increase in sediment accretion due to the extent of the decrease in sea level being less than the range at which both CS and RS are predicted to occur, i.e. ≤0.15m (table 5.2 chapter 5) plus the depth at which LiDAR is still valid (0.10m) (total -0.25m). However, in the no eustatic slr scenario the predicted magnitude of the decrease in local sea level was such that the model was unable to accurately predict the increase in the extent of the CS, RS, LS and US plant communities.

Figure 7.8 shows the charted depth of the sea in Tahu Bay according to bathymetric studies conducted by the Estonian Maritime Administration in 2003. Whilst these data are coarse they suggest that in the no eustatic slr scenario the wetland could expand by at least 500m in the north of the site and 1000m in the centre of the site. In order to improve the models in scenarios with a decrease in local sea level greater than 0.17m, bathymetric LiDAR data are required which use a wavelength of <700nm.

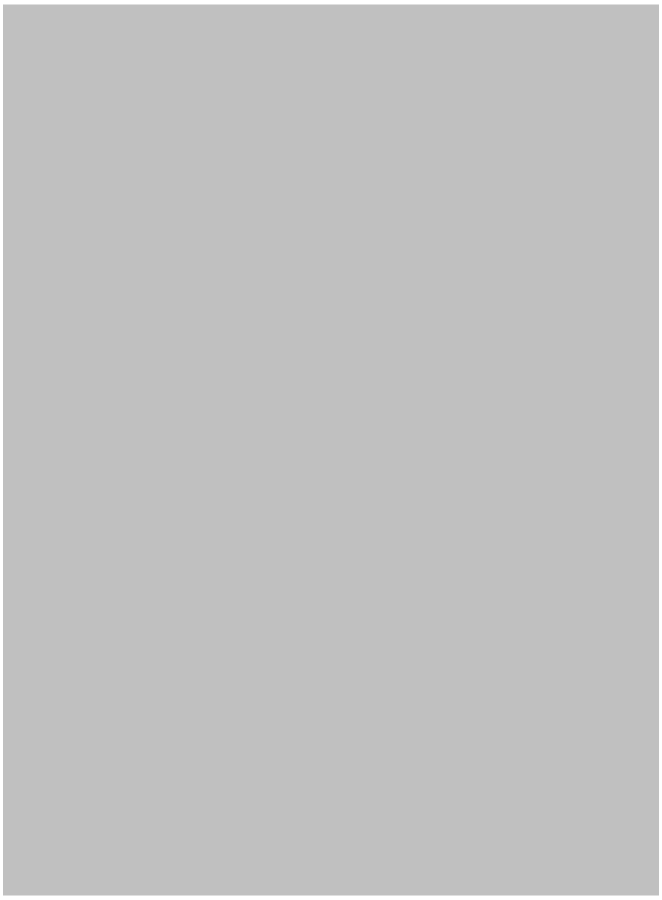

Figure 7.8: Nautical chart showing spot depths in metres in Tahu Bay. Data obtained from the Estonian Maritime Administration (2003).

7.5.5.2 Model uncertainties at Matsalu

At Matsalu, the decrease in local sea level was predicted to be much lower and occurred in fewer scenarios than at Tahu. The validity of the model at Matsalu for predicting the increase in extent of the Matsalu wetland was only limited in the no eustatic slr scenario using both accretion rate predictions.

As occurred at Tahu, the greatest decrease in local sea level at Matsalu was predicted to occur in the no eustatic slr scenario. In this scenario, using both the sediment accretion estimations, the decrease in local sea level was predicted to be -0.261m assuming no increase and -0.324m assuming an increase in sediment accretion. This was predicted to be of a magnitude which would increase the extent of the CS, RS and LS plant communities into areas at present submerged and hence not included in the plant community model. Further to this the limitation of the model to predict any growth in wetland plant community extent beyond a depth of 0.17m suggests that in the no eustatic slr scenario using both accretion rate estimations at Matsalu, model validity was limited for predicting the future extent of the CS, RS and LS communities beyond the boundaries of the present day wetland area. Therefore the future extent of the CS, LS and RS plant communities were likely to be under-represented in this scenario at Matsalu.

7.5.5.3 Model uncertainties at Kudani

The model outputs for Kudani predicted a decrease in the overall extent of the wetland due to an increase in local sea level in all scenarios except no eustatic slr. In the no eustatic slr scenario a decrease in local sea level of 0.252m was predicted. The model was limited in its ability to predict any increase in total wetland extent due to a lowering of local sea level and hence progradation of the lower elevation plant communities into the adjacent water body, Võõlameri. At Kudani, in the no eustatic slr scenarios, this resulted in the model being unable to accurately predict the total extent of the CS, RS and LS plant communities beyond the boundaries of the present day wetland area.

7.6 Conclusions

At Tahu in most scenarios there was a predicted increase in the extent of SW, TG and US by 2099 due to a decrease in local sea level. The lower elevation plant communities were predicted to increase in half the scenarios but decrease in the remaining scenarios. However, in all cases at Tahu the wetland was likely to prograde into Tahu Bay. Due to the limitations in the model in scenarios where large decreases in sea level occur, such as at Tahu, it was not possible to predict the change in extent of the lower elevation plant communities. However, historical trends suggest it is likely that there will be an increase in the extent of all the plant communities (Eesti Maa-amet, 2011). This predicted increase in the plant communities at Tahu was due to the high estimated sediment accretion and isostatic uplift rates. In the event of an increase in the extent of the plant communities at Tahu, this would provide a more substantial habitat for the wide variety of breeding and migrating birds which use the wetland.

At Matsalu there was little predicted change in the extent of the plant communities except in the no eustatic slr scenario where there was a considerable decrease in local sea level predicted by 2099. The predicted lowering of local sea level in the no eustatic slr scenario would, as at Tahu, likely lead to progradation of the wetland into the adjacent water body. There was also a considerable predicted increase in the extent of TG in this scenario at Matsalu by 2099.

At Kudani there was a predicted increase in local sea level in the A1FI, A2 and B2 scenarios, no change in the B1 scenario, and a decrease in local sea level in the no eustatic slr scenario. Therefore the model was able to predict the future location and extent of all the plant communities by 2099 in all scenarios except no eustatic slr. At Kudani, with the exception of the no eustatic slr scenario, there was a predicted increase in the extent of CS, RS and in two scenarios LS by 2099. The increase in CS, RS and LS occurred at the expense of US, TG and SW. In the no eustatic sea level rise scenario, as at Tahu and Matsalu, there was likely to be an increase in all plant communities due to progradation of the wetland, in this case into Võõlameri.

The study in this chapter has presented the dynamic correlative plant community model. It should be noted that any dynamic model, whether correlative, mechanistic or mathematical provides a vision of the future constricted by a series of assumptions. The dynamic model presented in this chapter provides a prediction of changes in plant community location and extent within three Estonian coastal wetland sites based on a series of sea level and sediment accretion scenarios. Thus the model assumes that over the modelling period, to 2099, other environmental variables that have been shown to influence plant community type in Estonian coastal wetlands, such as temperature, soil pH, N, K, organic matter, moisture, particle size, and salinity are unlikely to change.

Based upon these assumptions the model was able to accurately predict the future location and extent of the plant communities within the present wetland area. However, the model was limited in its ability to predict progradation of the wetland. In the Matsalu and Kudani sites, the limited ability of the model to predict progradation did not affect model performance in the majority of scenarios. However, in circumstances where there was a predicted decrease in local sea level of >0.17m the model was limited in its ability to predict the extent of the lower elevation plant communities as they extended into adjacent water. This was predicted to occur in the no eustatic slr scenario at all sites, and is consistent with the historical development of Estonian coastal wetlands. A decrease in local sea level of >0.17m was also predicted to occur in the majority of scenarios at Tahu due to high isostatic uplift and sediment accretion rates. These limitations in the model were derived from the low penetrability in water of the LiDAR data upon which the model was based. This could be overcome by using bathymetric LiDAR which has a much greater penetrability through water due to its shorter wavelength.

The model has shown that the Tahu coastal wetland was not likely to be at risk from sea level rise and was likely to increase in extent in all future scenarios by 2099. The Matsalu coastal wetland was unlikely to undergo considerable change in the majority of scenarios by 2099 and hence sea level rise was unlikely to impact the importance of this area as a habitat for birds. The Kudani coastal wetland was likely to undergo substantial loss of the higher elevation plant communities in the majority of scenarios due to coastal squeeze which could affect the bird species that utilise these plant communities for breeding and feeding.

8 Discussion

8.1 Preamble

Key results obtained from this research include the quantification of the relationship that exists between elevation (above m.s.l.) and plant communities in Estonian coastal wetlands. The study has developed a methodology to produce a robust large-scale static correlative plant community model in appropriate environments using LiDAR data. The addition of elevation corrections to the LiDAR data has been shown to significantly improve the accuracy of the plant community predictive model. The resultant plant community model has been used as a basis to produce a dynamic correlative model to inform debates on the implications of sea level rise on Estonian coastal wetlands.

8.2 Evaluation of methods

8.2.1 Study sites

Estonian coastal wetlands are often extensive and typified by a low range of relief due to their negligible tidal range, 0.02m (Keruss & Sennikovs, 1999; Suursaar *et al*., 2002) and the underlying geology (Raukas & Tavast, 1994). Previous authors have suggested that the plant communities within Estonian coastal wetlands were arranged according to elevation (Tyler, 1971a; Burnside *et al*., 2007; 2008) although with no empirical quantification and assessment. The environmental characteristics of Estonian coastal wetlands provided ideal conditions to develop and test a plant community model. The low relief required the model to provide a more detailed assessment of the vegetation – elevation relationship than has been accomplished in previous studies. The resultant model was more accurate than previous plant community models developed for coastal wetlands, such as those by Morris *et al*. (2005), Prisloe *et al*. (2006), Sadro *et al*. (2007), and Moeslund *et al*. (2011), because this study improved the elevation accuracy of the remotely sensed data. High levels of model accuracy were achieved through the integration of advanced geomatic techniques and differentially corrected GPS data.

Estonian coastal wetlands also have similarities to other wetlands such as floodplain meadows and are often classed as wet grasslands (Joyce & Wade, 1998b; Benstead *et al*., 1999) being largely non-tidal but irregular inundation periods. Estonian coastal wetlands are located next to a large body of brackish water, the Baltic Sea, and hence are also classed as related to Atlantic salt marshes (EU Habitats Directive, 1992; Davies *et al*., 2004). Such similarities mean that these systems are likely to be under threat from similar environmental changes, such as sea level rise and changes in the frequency and severity of extreme weather events, so the model potentially has wide applicability.

8.2.2 Environmental gradients

This study has made an assessment of the main environmental variables that influence the location of different plant communities within Estonian coastal wetlands. An assessment of environmental gradients is essential for the development of any predictive modelling of plant communities (Franklin, 1995). Further to this an assessment of a wide range of environmental gradients gives an holistic view of which factors can be best used as predictors of plant community type. Austin (1980) states that environmental gradients can be divided into indirect environmental gradients, resource gradients and direct gradients; a full description of these is presented in chapter 2. In this study all three of the aforementioned gradient types have been evaluated to assess their influence on plant communities: Indirect gradients (elevation, sediment particle size), resource gradients (soil moisture, organic matter, N, P, K); and direct gradients (pH and salinity). Austin & Smith (1989) suggest that, with regards to plant community modelling, resource and direct gradients are preferable as these predictors are spatially independent. However, Moore *et al*. (1991) have shown that indirect environmental gradients can be useful for plant community modelling if they explain sufficient variation in plant community location and are used for interpolations within similar environments. This study has also shown that in Estonian coastal wetlands the indirect variable, elevation, was a principal factor that could accurately distinguish the most common plant communities.

Within this study the relationship between elevation and plant communities was quantified. However, it is likely to be the case that elevation was a proxy for hydrology, which has been estimated to determine 50% of the properties of wetlands (Keddy, 2000). This study has shown that there was a strong correlation between elevation and soil moisture, and indicated a relationship between elevation and water level. This accords with studies on Swedish coastal wetlands (Tyler, 1971b), Estonian coastal wetlands (Berg *et al*., 2011), German coastal wetlands (Suchrow & Jensen, 2010) and various floodplain grasslands (Prach, 1992; Gowing *et al*., 2002; Wheeler *et al*., 2004; Araya *et al*., 2010a). A study in Estonian coastal wetlands by Berg (2009) showed that over a three year period the LS community had a higher mean water table than the higher elevation US community which in turn was found to have a higher mean water level than the higher elevation TG community (Table 8.1). Thus, the strong relationship between elevation and hydrology provided the foundation for a LiDAR elevation derived plant community model.

Table 8.1: Hydrological data for the LS, US and TG plant communities collected between June 2004 and August 2006 (adapted from Berg, 2011)

Plant community	Frequency of flood events	Mean summer inundation depth (m)	Water table depth during growing season (m)
LS	4	0.2	0-0.2
US	0	N/A	0-0.2
TG	0	N/A	>0.50

With the exception of the OP plant community each of the communities was located at significantly different elevations above sea level. The results of this study have shown that differences of only 0.04m can influence plant community distribution. Moreover, elevation appears to play an important role in determining the frequency and duration of inundation in Estonian coastal wetlands. The results presented in chapter 4 suggested that soil water level was related to sea level and that the higher elevation plant communities had a deeper water table than the lower elevation plant communities, with the exception of OP. Previous studies have shown similar results for tidal estuarine wetlands, coastal salt marshes and floodplains, relating the water level of the adjacent water body, sea or river, to soil water level (Montalto *et al*., 2007; Glamore & Indraratna, 2009).

8.2.3 Plant community classification

In order to develop a predictive plant community model, some form of vegetation classification must be used. There are several classification systems in place for Baltic coastal wetlands including those developed by Tyler (1969), Wallentinus (1973), Rebassoo (1975), the EC habitat directive (1992), Paal (1998), and Burnside *et al*. (2007). Each of these classifications has its merits, however, there are some regional differences in the plant communities of Baltic coastal wetlands (Rebassoo, 1975). The classification system that is most suited to managed coastal wetlands located in Estonia is that developed by Burnside *et al*. (2007). The Tyler (1969) and Wallentinus (1973) classifications, although somewhat similar to the Burnside *et al*. (2007) classification, are developed for Swedish coastal wetlands and have some subtle differences in species composition. The Rebassoo (1975) classification was developed for all seashore plant communities that exist on the Estonian Islands and hence is not easily applicable in mainland areas due to differences in plant species composition, including the abundance of rare species (Ratas & Nilson, 1997). The EC habitat directive classification has very broad definitions and many of the designated habitats that occur in Estonian coastal wetlands overlap and hence it is not applicable for use in the field. The Burnside *et al*. (2007) classification does not have any of these draw backs, represents the full suite of plant communities that exist in the study sites and hence was used in this research project.

8.2.4 Management of Estonian coastal wetlands

Vegetation management, typically in the form of mowing or low intensity grazing affects the plant communities in Estonian coastal wetlands and prevents succession to coarser community types (Jutila, 1997). Only managed Estonian coastal wetlands were used in this study due to their greater plant species diversity, importance as a habitat for breeding and migratory birds, and as a more varied plant community mosaic with which to develop the model. In a study of abandoned Estonian coastal wetlands, it was found that there was a loss of many plant communities in particular the wet grassland types (LS, US, and TG). Hence there was an overall loss of biodiversity, a phenomenon that has affected a variety of wet grasslands (Joyce & Wade, 1998b; Benstead *et al*., 1999; Busmanis *et al*., 2001; Joyce & Burnside, 2004; Rannap *et al*., 2004). In a study by Berg (2009) the biggest effect of abandonment came from the encroachment of *Phragmites australis* into the LS, US and TG plant communities. A study by Rannap *et al*. (2004) also showed that woody species such as *Juniperus communis*, *Pinus sylvestris* and *Alnus glutinosa* encroached into the TG plant community following the cessation of management. Therefore the elevation ranges at which these plant communities occur are most suitable for Estonian coastal wetlands managed by low intensity grazing or mowing.

8.2.5 Geomatic methodological considerations

The results of this study have shown that plant communities can be accurately identified using remotely sensed elevation data. The advantages of using remotely sensed data to model plant communities is that it does not require a large and costly field study to achieve accurate results over large areas. Remotely sensed data have been successfully used in many studies to develop plant community models over large field areas (Hardisky *et al*., 1986; Schmidt & Skidmore, 2003; Eva *et al*., 2004; Morris *et al*., 2005; Prisloe *et al*., 2006; Sadro *et al*., 2007; Chust *et al*., 2008; Sheffield *et al*., 2009; Muira & Jones, 2010; Moeslund *et al*., 2011). However, none of the aforementioned studies work at such a fine scale as that developed in this study.

There is a wide variety of remotely sensed data that can be used to identify plant communities including colour infrared photography, high resolution orthophotos, LiDAR and multispectral data such as Landsat, IKONOS and Quickbird (Jones & Vaughan, 2010). Wallentinus & Jonson (1972) showed that colour infrared photography is unsuitable for the identification of the different graminoid plant communities that exist in grassland communities in Baltic coastal wetlands. Furthermore similar studies have been undertaken in mosaic plant communities using Landsat imagery (Knick *et al*., 1997) and high resolution orthophotos (Bradter *et al*., 2011) with similar negative results.

The relationship between elevation above mean sea level and plant community type provided a basis for developing a predictive plant community model for Estonian coastal wetlands. However, the elevation differences between some of the plant communities were very small, with a minimum of 0.04m. Hence it was necessary to obtain highly accurate elevation data that covered a variety of Estonian coastal wetland sites. Due to the small size of some of the plant community patches, it was also necessary to obtain high resolution data for the study areas. The need for accurate elevation data, and a high spatial resolution, prohibited the use of Shuttle Radar Topography Mission (SRTM) and Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) data. The only large scale, high resolution and accurate elevation data available are LiDAR. LiDAR can be produced at a variety of spatial scales dependent on the pulse rate of the LiDAR device used, and the altitude and type of the LiDAR platform (Jensen, 2007). The data used in this study was medium point density, at 0.45 points per square meter, recorded from a fixed wing aircraft at 2400m using a Leica ALS50-II LiDAR device (Hurt, 2011). More points per square meter can be achieved using a lower and slower flight. However, this can only be achieved using either a helicopter and/or using a multipulse LiDAR device. The cost of obtaining these data is much higher, and the benefits are limited. A study by Anderson *et al*. (2006) showed that in coastal areas with low relief LiDAR data density could be reduced from 0.772 points per square meter to 0.0873 points per square meter without significantly affecting the accuracy of the resultant DEM. Therefore the horizontal resolution and data density of the LiDAR used in this study is unlikely to have adversely affected the elevation accuracy of the DEM's developed in chapter 5.

Therefore in environments with a strong relationship between plant community type and small changes in elevation LiDAR are a robust data choice due to the elevation accuracy and the density of the point cloud. However, as this study has shown, some form of calibration and adjustment must be performed on the LiDAR elevation data. The results of chapter 5 have shown that without dGPS calibration the resultant model was limited in its ability to predict plant community type in Estonian coastal wetlands due to their very low relief. In previous studies in tidal coastal wetlands, with a greater range in relief, calibration and adjustment was not used and was perhaps not necessary to produce a model that was able to distinguish plant communities based on their elevation range (Morris *et al*., 2005; Prisloe *et al*., 2006; Poulter & Halpin, 2008; Moeslund *et al*., 2011). However, the improvements in the accuracy by using dGPS calibration data of the model developed in this study could significantly improve the accuracy of plant community models developed for any tidal coastal wetlands with a greater relief.

LiDAR are available as point data which give, in the case of the Estonian Land Board, a 0.54m footprint with 0.45 points per square meter. Therefore this does not provide a continuous dataset and had to undergo some form of interpolation, as was described in chapter 5. This study has shown that the Delaunay linear TIN interpolation technique produced the most accurate DEM and was developed by elevation categorisation to produce the plant community model. Other studies have used a similar methodology to develop accurate plant community models based on vegetation-elevation relations in coastal wetlands (Morris *et al*., 2005; Chust *et al*., 2008; Millette *et al*., 2010).

In many coastal wetlands the driver of coastal wetland development is sediment accretion (Allen & Pye, 1992; Reed, 1995) and sediment accretion rates are therefore an important factor to be taken into account when modelling the effects of sea level rise on coastal wetlands (Reed, 1995). This study assessed past sediment accretion rates and extrapolated the results as used in other studies (Park *et al*., 1989; Park *et al*., 1991; Parkinson *et al*., 1994; Galibraith *et al*., 2002; Galibraith *et al*., 2003; Craft *et al*., 2009). The study presented in this thesis also integrated isostatic uplift rates (Kont *et al*., 1997; Kont *et al*., 2003) to improve model accuracy. Previous studies of the effects of sea level rise in Estonian coastal areas have only taken into account isostatic uplift and effectively ignored sediment accretion (Kont *et al*., 1997; Suursaar *et al*., 2006b; Kont *et al*., 2007). However the results of this study have shown that both sediment accretion and isostatic uplift rates have a considerable effect on local sea level rise modelling.

Wetland plant communities are dynamic and have been found in previous studies to respond rapidly to changes in soil hydrology (Keddy, 2000; Toogood, 2005). The model developed in this study did not take into account limitations on the speed with which species can adapt to changes in soil hydrology driven by sea level rise. Wetland soil hydrology influences plant community structure by limiting the plant species to suitably adapted types (Kozlowski, 1984) and modifies this through disturbance and stress (Barnes, 1978; Day *et al*., 1988). The results of higher local sea level, as has been predicted to occur in several sea level rise scenarios at Matsalu and Kudani, will include the loss of the dominant plant species in the more frequently inundated plant communities. The resultant gaps will be colonised by species more tolerant to inundation (Keddy & Resnicek, 1986). A study by Toogood (2005) in wet grasslands showed that response to flooding was greatest when maximum inundation depth was over 0.15m and plant species submergence persisted for 3-5 months. Gowing & Youngs (1997) showed that inundation most greatly affected plant species during periods when interspecific competition was most intense, typically at the start of the growing season, which in Estonian coastal wetlands starts in May/June (Puurmann & Ratas, 1998). An increase in the duration and depth of inundation of the magnitude of the Toogood (2005) study is only likely to happen over a prolonged period in Estonian coastal wetlands. Hence changes in the extent and location of the plant communities are likely to occur over a number of years in sites where local sea level rise occurs and is therefore unlikely to affect the validity of the predictive plant community model developed in this study.

The predictive plant community model developed in this study makes assumptions of increases in sediment accretion due to greater storminess as a result of climate change. The plant community model however, does not take into account the direct effects of sediment deposition on the plant communities. Sediment deposition has the effect of providing nutrients from allochthonous sources to the wetland thereby increasing vegetation fertility (deLaune *et al*., 1981). However, this simultaneously provides physical stress to individual plants to overcome the depth and physical structure of the depositional material and also limits light availability to low growing species (Ewing, 1996). These factors confer an ecological advantage to competitor species at the expense of stress tolerators (Grime *et al*., 1988). Therefore, assuming an increase in sediment accretion, competitor species such as *Bolboschoenus maritimus*, *Schoenoplectus lacustris* and *Phragmites australis* will be likely to rapidly colonise areas that have been previously submerged by water.

8.3 Development of the predictive model

This thesis develops and presents a rapid landscape tool for predicting the present location and extent of plant communities in Estonian coastal wetlands. The methodology combined field based ecological survey data with geomatic techniques to produce a static correlative plant community model which was used as a basis for predicting changes in plant community type with relation to sea level. In the model building process, a variety of successive steps were performed, which are illustrated in figure 3.11. The start of the model development process required the formulation of a conceptual model based on field knowledge. The development finished with the formation of a dynamic correlative predictive model for use in planning and/or land management (Guisan & Zimmerman, 2000). The basis for the conceptual model was the hypothesis that elevation above sea level is so strongly correlated to plant community type that it can be used as a predictor. Using this hypothesis as a basis it was necessary to select an appropriate plant community classification and assess the relationship between selected environmental variables and plant community type (chapter 4) as well as select an appropriate remotely sensed dataset.

Previous studies that have developed predictive plant community models in coastal wetlands have typically assessed only a few environmental variables such as salinity and elevation (Moeslund *et al*., 2011) or elevation alone (Morris *et al*., 2005; Poulter & Halpin, 2008). The results from this study showed that elevation was the environmental variable that differed significantly in the greatest number of plant community types and was also found to be correlated with the greatest number of other environmental variables (chapter 4). Assessing a variety of environmental variables provided an holistic evaluation of the main factors that influenced plant community type although, for modelling purposes, further studies could examine the elevation – vegetation relationship alone. Other studies have used multi-spectral data (Provoost *et al*., 2005; Sadro *et al*., 2006) or canopy height to determine plant community type in coastal wetlands (Prisloe *et al*., 2006). Whilst these have been shown to be robust for accurately mapping plant community types, these data are of little use for modelling plant community location and extent following changes in hydrology, such as sea level rise in coastal wetlands.

This study selected medium point density terrestrial LiDAR data due to its elevation accuracy. Previous studies have shown that these LiDAR data are useful for producing correlative plant community models for tidal coastal wetlands (Morris *et al*., 2005; Moeslund *et al*., 2011). However, the results from this study have shown that dGPS calibration and adjustment of the LiDAR data is essential in low relief Estonian coastal wetlands. This suggests that dGPS integration would also improve the robustness of correlative plant community models in wetlands with greater relief such as tidal salt marshes or floodplain meadows. Due to the nature of LiDAR data, some form of interpolation must be performed in order to produce a continuous elevation surface (de Smith *et al*., 2007). The results of this study have shown that for medium point density LiDAR data in appropriate open landscapes with few sudden breaks in terrain, such as Estonian coastal wetlands, a TIN interpolation was best able to represent the changes in relief and predict plant community location. Other studies have shown that, where data points are less dense or where there are sudden breaks in terrain, OK or IDW methods of interpolation are more robust than TIN (Lloyd & Atkinson, 2002a; Lloyd & Atkinson, 2002b). In order to determine which method of interpolation is appropriate a cross-validation of the interpolated surface must be performed (de Smith *et al*., 2007). The TIN derived elevation interpolation was selected for use in the static model (chapter 5) and following cross validation the resultant DEM was categorised according to the elevation preferences of the plant communities creating a test model. The test model was then evaluated and validated by ground truthing its predictive outputs at two further sites.

Independent verification is necessary in all forms of predictive modelling, whether correlative or mechanistic, if the model is going to be used and applied more widely than simply the test or development site (Franklin, 1995; Guisan & Zimmerman, 2000; Morris *et al*., 2005; Prisloe *et al*., 2006; Sadro *et al*., 2006; Moeslund *et al*., 2011). The validation of the static correlative model in this study showed that there were problems in the prediction accuracy at Matsalu. Burnside & Waite (2011) suggest that one of the main problems that can occur in both correlative and mechanistic plant community modelling is that of false absences due to unmodelled factors. At Matsalu the model incorrectly predicted the presence of CS and SW. This was most likely due to increased wave activity in the lower elevation areas for CS and a longer management history at Matsalu than at the test site for the SW. In order to overcome this, a visual assessment was made using a further remotely sensed dataset, in this case high resolution orthophotos, in order to identify any CS or SW plant community patches at Matsalu. Integrating further parameters into the model is important in correlative modelling due to the aforementioned problems of scaling. The resultant static correlative model at Matsalu, with additional parameters taken into account, was revalidated and shown to be considerably more accurate.

The static correlative model was then used as a basis for the development of the dynamic correlative model, to predict plant community changes by 2099. The factors that were predicted to influence the future location and extent of the plant communities in Estonian coastal wetlands were: isostatic uplift, eustatic sea level rise, and sediment accretion. Previous studies assessing the effects of sea level rise on Estonian coastal wetlands have been limited due to a variety of factors including: the exclusion of sediment accretion as a factor in limiting the effects of sea level rise, low resolution topography data (essential for these low relief coastal wetlands), and no verification of any modelling (Kont *et al*., 1997; Kont *et al*., 2003; Kont *et al*., 2008). Moeslund *et al*. (2011) justify the exclusion of accretion rates from their correlative model by suggesting that accretion rates vary greatly within sites and that the majority of sediment accretion data available is generalised from few samples. However, whilst extrapolation of sediment accretion data from few samples can be problematic due to the spatial variablilty of sediment deposition the exclusion of these data is likely to provide greater dynamic model prediction errors. This is highlighted in this thesis by the substantial differences in the predicted effects of sea level rise on the plant communities at Tahu (taking into account sediment accretion) compared with Kudani (assuming no sediment accretion) where both sites are experiencing the same rate of isostatic uplift.

The results of this study have shown that the relatively high isostatic uplift rates at Tahu in combination with sediment accretion rates are likely to offset sea level rise in all scenarios, and to such an extent that extensive progradation of the wetland into the shallow adjacent water body, Tahu Bay, is likely to occur. Due to the limited penetration of terrestrial LiDAR through water, any predicted progradation of the wetland into areas greater than 0.10m deep is beyond the ability of the model to predict. This requires a further feedback into the model and the use of an alternative remotely sensed dataset. A relatively recent development in LiDAR technology has been in bathymetric LiDAR which uses a shorter wavelength, typically 532nm instead of the longer 1064nm used in terrestrial LiDAR (Wang & Philpot, 2007). Bathymetric LiDAR has a much greater penetrability through water, to 40m, and the data have been used to model plant communities in tidal salt marshes (Collin *et al*., 2010). Bathymetric LiDAR data are not commonly used in terrestrial environments due to the greater potential risk of blindness to animals and people from the shorter wavelength laser used in this system, which requires the scanned area to be closed off for the duration of the scanning flight. However, the use of these shorter wavelength LiDAR data would overcome the limitations in model prediction that occur in scenarios with a >0.10m decrease in sea level.

8.4 Use of the predictive model in other appropriate environments

The model developed in this study has been shown to be able to accurately predict the location and extent of plant communities in non-tidal Estonia coastal wetlands. However, this model has potential applications in other appropriate open environments such as floodplain wetlands, coastal tidal marshes or for the restoration of wetlands. It does however, have limitations which must be taken into account when interpreting the results. As with any correlative model, there are issues with scaling and hence the model developed in this study is unlikely to be valid in locations further afield than Estonia without further ground truthing. The dynamic correlative model is also based on the assumption that sea level and sediment accretion will be the only environmental variables that will change by 2099. However, IPCC (2007) climate change scenarios suggest that temperature will also increase and several studies have suggested that this could cause a northward migration of some plant species (Huntley, 1991; Pitelka, 1997; Dullinger et al., 2004; Aitken *et al*., 2008; Hilyer & Silman, 2010), which would affect plant community composition in Estonian coastal wetlands. Further to this it must be noted that whilst the calculated sediment accretion rates have been shown to be useful in the dynamic correlative plant community model developed in this study, they are based on a limited number of samples and have been extrapoltaed for the studied sites.

In floodplain wetlands a similar methodology can be used as that developed in this study to predict plant community changes, due to for example altered flood dynamics. In floodplain wetlands the plant community prediction tool could be used to predict changes in the plant community location and extent due to changes in the duration or extent of inundation. Potential scenarios involving changes in the hydrological regime could be related to either increases or decreases in precipitation or managed alteration of the hydrological regime by the addition or removal of drainage ditches, all of which have been related to plant community type (Prach, 1992; Grevilliot *et al*., 1998; Joyce, 1998; Gowing *et al*., 2002; Wheeler *et al*., 2004; Toogood *et al*., 2008). Any predictive plant community model based on the methods developed in this study would require an appropriate plant community classification. There are a variety of plant community classifications available for a wide range of floodplain wetlands although the most commonly used in the UK are those developed by Rodwell (1992) and in continental Europe by Ellenberg (1988). In floodplain wetlands it would be necessary to assess the relationship between elevation and hydrology at each floodplain site due to changes in elevation above sea level of the water level at each site and within sites and differences in soil permeability. A variety of studies have looked at how differences in water levels (Prach, 1992; Newbold & Mountford, 1997; Gowing *et al*., 2002; Wheeler *et al*., 2004; Kalusova *et al*., 2009; Thompson *et al*., 2009; Araya *et al*., 2010a; Araya *et al*., 2010b) and soil permeability (Gowing *et al*., 2002; Wheeler *et al*., 2004) affect plant community type in floodplains. In addition to these assessments any plant community model for a floodplain would have to be subdivided into downstream sections for analysis and the width of the section will be dependent on the slope of the river. Other than subdividing sites based on slope angle of the river, the geomatic development of the static model in floodplains does not need to be altered from the methodology developed for Estonian coastal wetlands.

When considering the development of the dynamic model a range of different factors must be taken into account. The main factors that are likely to influence water table level in floodplains are changes in the duration and depth of inundation due to either artificial alteration of the hydrological regime, due to changes in the upstream hydrological input, or sediment accretion raising the level of the river or surrounding floodplain (Gell *et al*., 2009; Ashworth *et al*., 2011). Changes in the hydrological regime of floodplains are difficult to predict (Horritt & Bates, 2001) and hence a variety of scenarios should be modelled. An assessment should be made of the rapidity with which floodplain species can adapt. This is dependent on the availability of potential colonisers in the seed bank and precipitation (Bakker *et al*., 1996; Jansen *et al*., 2000; Klimkowska *et al*., 2007), as well as the autecology of potential colonisers (Strom *et al*., 2010), and the management regime i.e. grazing or mowing (Prach, 1992; Dyrcz, 1999; Kozulin, 1999).

Tidal salt marshes are another potential environment for the application of the plant community model. A variety of authors have discussed the zonation of plant communities in relation to elevation above mean sea level and related that to tidal range (Bertness, 1991; Gray, 1992; Nottage & Robertson, 2005; Cutini *et al*., 2010; Moffett *et al*., 2010; Moffett *et al*., 2012). However, tidal ranges vary greatly, hence the plant community model would likely be valid only for areas with similar tidal regimes. Correlative plant community models have been suggested to be location specific (Austin & Smith, 1989; Franklin, 1995) and hence in localities with different tidal regimes model parameters will need to be adjusted. Furthermore, salt marshes, whilst retaining a similar zonal character typically consisting of a low marsh, middle marsh and a high marsh (Nottage & Robertson, 2005), have a geographically varying species composition requiring the use of different plant community classifications dependent on location (Pennings *et al*., 2003). Both Rodwell (1992) and Ellenberg (1988) have produced plant community classifications for UK and continental European salt marshes which would be suitable for use in a plant community model. With regards to the geomatic stages involved in applying the plant community tool to salt marshes, the same procedure would apply as used in the development of the plant community tool for Estonian coastal wetlands. This involved the selection of an appropriate remotely sensed dataset, which in scenarios that are likely to include progradation of the marsh should include bathymetric LiDAR, or cases with no predicted decrease in local sea level, terrestrial LiDAR.

The test site plant community model should be applied to independent sites with similar tidal regimes and vegetation to validate the static model. Dependent on location either isostatic uplift rates or subsidence rates must be calculated if applicable to the geographical location of the modelled marsh, which are typically well known for the Baltic region (Ekman, 1996) and for the UK (Shennan, 1989). Finally sediment accretion rates or erosion rates for the studied marshes must be calculated although in many cases these are also available (Callaway *et al*., 1996; Cundy & Croudace, 1996; Kolker *et al*., 2009; Teasdale *et al*., 2011; Schuerch *et al*., 2012). However, for areas where no sediment accretion rates are available, ^{210}Pb with ^{137}Cs

validation have been shown to provide accurate estimations (Appleby & Oldfield, 1992; Cundy & Croudace, 1995a; Yeager *et al*., 2004; Dyer *et al*., 2002; Teasdale *et al*., 2011).

A further potential application of the plant community modelling tool is for restoration, such as of salt marshes or floodplains. Often restoration wetlands are located in inappropriate areas or inappropriate species are selected due to a lack of knowledge of species suitable for specific sites (Zedler, 2000). Roman *et al*. (1997) have suggested that hydrological modelling can be used as a basis for ecological restoration of wetlands as hydrology is the main factor influencing the location of plant communities. This study has shown that in Estonian coastal wetlands, elevation, may be a proxy for hydrology and has such a strong relationship with plant community type that it can be used as a predictor.

Often the restoration of wetlands fails due to site differences and the fact that the selection of plant species is based on autecological factors, which typically do not account for interspecific competition and hence do not relate to the realised niche of the selected species (Manchester *et al*., 1999). Correlative models use site or region specific correlations between one or more environmental variables and plant community type (Burnside & Waite, 2011). Robertson *et al*. (2003) suggest that this incomplete knowledge of the autecology of many plant species is one of the reasons that correlative models can be superior to mechanistic models, although this is also the reason why they do not scale well. The plant community tool developed in this study could be used to improve the success of restoration projects in appropriate wetland environments. This would require the development of a static correlative model for a suitable established site in an area with similar geology, soil type, management regime, and with similar plant communities. The static model could then be used to assess which vegetation would be appropriate for seeding/ planting at specific elevations (and at different distances dependent on soil permeability) from the adjacent water body.

9 Conclusions

Baltic coastal wetlands are included as a priority habitat in Annex I of the EU Habitat Directive (1992) and have been designated as such due to their international importance, supporting high biodiversity (Rannap *et al*., 2004). The wetlands are shaped by isostatic uplift, inundation by the brackish Baltic Sea, sediment accretion and regular low intensity management. The resultant plant communities are made up of a variety of rare and specialised species many of which are at the extremes of their ranges (Rebbassoo, 1975; Paal, 1998; Kull *et al*., 2002; Wotavová *et al*., 2004). The vegetation also provides habitats that support a wide diversity of migratory and breeding birds, including waders, ducks, geese, passerines and raptors (Puurmann & Randla, 1999; Truus, 1999; Mägi *et al*., 2004a; Ward, 2007).

A rise in eustatic sea level is likely to pose a threat to coastal wetlands worldwide (IPCC, 2007). These problems have been suggested to be exacerbated in large micro-tidal water bodies such as the Baltic Sea and the Mediterranean (Nicholls *et al*., 1999; Parry *et al*., 2007). However, many Estonian coastal wetlands are experiencing isostatic uplift which, it has been suggested (Suursaar *et al*., 2004), will offset eustatic sea level rise. Other factors such as sediment accretion have not been taken into account in previous studies on Estonian coastal wetlands and these studies have used arbitrary eustatic sea level rise values (Kont *et al*., 1997; Kont *et al*., 2003; Kont *et al*., 2008).

The aim of this research was to develop a predictive model of plant community location and extent in Estonian coastal wetlands. The model was developed through the application of field based ecological and geomatic techniques, and used LiDAR elevation data to assess the effects of sea level rise on the distribution of the plant communities. In order to address this aim four objectives were stated:

(i) to determine and characterise the relationship between a range of coastal wetland plant community types, elevation and edaphic conditions.

(ii) to assess the use of LiDAR data to develop a model to determine plant community distribution based on fine scale elevation data.

(iii) to conduct a sensitivity analysis of the plant community model using ground recorded dGPS elevation data and sediment accretion rates.

(iv) to assess the effects of sea level rise on the plant communities of Estonian coastal wetlands under five sea level scenarios and two accretion rate scenarios and factoring in isostatic uplift rates.

9.1 Key findings

(i) This study has shown that Estonian coastal wetland plant communities were determined by a variety of edaphic factors and elevation above m.s.l.. The main environmental variables shown to effect plant community type were elevation, soil moisture, organic matter, N, K, and salinity, although elevation was the variable which was able to significantly distinguish the greatest number of plant community types, i.e. all except OP and TG. Micro-topography appears to play an important role in determining the frequency and duration of inundation in such wetlands. The study has shown that inundation frequency and water table depth is related sea level by the processes of overland flow and throughflow, and sea level is itself related to changes in atmospheric pressure. The interrelationship between hydrology and elevation has led to the development of a mosaic within a broader zonation of plant community types across Estonian coastal wetland landscapes and as such plant community type can be readily and effectively modelled in the field using a proxy measure of elevation above m.s.l..

(ii) Medium point density LiDAR elevation data can be used to produce an accurate predictive plant community model based on the relationship between elevation and plant community type. The methodology requires ground truthing of plant community type using vegetation quadrats, ground-truthed dGPS calibration, adjustment of the vertical accuracy of the LiDAR data, and exploratory spatial data analysis in order to improve model validity. The results of the study have shown that, due to the limited ability of LiDAR to penetrate vegetation canopy, a mean elevation correction value of 0.177m was required to improve model accuracy within micro-topographical Estonian coastal wetlands. The addition of the elevation adjustment procedure to the model has been shown, in this study, to be an improvement to previous LiDAR models (e.g. Morris *et al*., 2005; Chust *et al*., 2008; Poulter & Halpin, 2008; Huang *et al*., 2011; Moeslund *et al*., 2011). Kappa values, using similar modelling procedures to the Morris *et al*. (2005), Chust *et al*. (2008), and Moeslund *et al*. (2011) studies, were between κ 0.15 in Estonian coastal wetlands, compared to κ 0.63 using the elevation correction and TIN interpolation at Tahu. The study has also shown that using densely spaced regular point data the most accurate interpolation method is Triangulated Irregular Network. Several previous studies have used Ordinary Kriging or Inverse Distance Weighting methods (Lloyd & Atkinson, 2002a; Lloyd & Atkinson, 2002b; Bater & Coops, 2009), which this study has shown are less accurate, more time consuming and processor intensive.

(iii) Accretion has been largely ignored as a factor in the development of Estonian coastal wetlands and no sediment accretion rates were available. The ^{137}Cs impulse dating and the ^{210}Pb CRS and CF:CS methods have been shown to be relevant for use in coastal wetland environments. The results determined from these techniques have shown that sediment accretion in some Estonian coastal wetland sites can contribute to greater vertical growth at the littoral edge of the wetland than isostatic uplift (2.8mm/yr at Tahu, 2.0mm/yr at Matsalu), particularly during periods of increased storminess (5.0mm/yr at Tahu, 1.7mm/yr at Matsalu). The addition of accretion data to the model substantially altered the predicted extent of the plant communities. This is particularly evident at Tahu, where the wetland was predicted to prograde into Tahu Bay. However, at Kudani, which has the same rates of isostatic uplift yet no predicted sediment accretion, the wetland was predicted to decrease in extent in all sea level rise scenarios.

(iv) The results of the dynamic model have shown that sea level is unlikely to pose a serious threat to Estonian coastal wetlands, such as Tahu (all scenarios predict a decrease in local sea level) and Matsalu (half of the scenarios predict a decrease in local sea level). However, wetlands that lack sediment accretion may lose the higher elevation plant communities due to coastal squeeze (e.g. at Kudani in all scenarios except B1 where eustatic sea level rise is predicted to keep pace with isostatic uplift, and the no eustatic sea level rise scenario). This could be overcome at Kudani by reinstating the natural hydrological regime to the wetland, which would allow an influx of allochthonous sediment.

9.2 Recommendations for further study

9.2.1 Further data collection

The hydrological study in chapter 4 gave an interesting indication that sea level was the main factor influencing water table level in the plant communities in Estonian coastal wetlands. Previous authors have also suggested that water table level is governed by sea level in these wetlands (Tyler, 1971a; Burnside *et al*., 2007; Berg *et al*., 2011). However, the hydrological study presented in chapter 4 was conducted over a limited time period compared to other studies (Kaplan *et al*., 2010; Rodhe & Seibert, 2011) and hence was likely to underestimate the impact of other factors such as precipitation (which did not occur during the period of the hydrological study). Further to this, the results of the hydrological study suggested that there was a time lag in water movement to the higher elevation plant communites, which are often located further form the sea than the lower elevation plant communities. However, no work was conducted to assess the rate or direction of throughflow in these wetlands nor any assessment made of whether the predominant process of throughflow is matrix flow, macropore flow or pipe flow. Therefore, in order to improve the understanding of the soil hydrological regime it is suggested that a longer period study could be conducted to assess water table level over the wetland. In order to clarify the relationship between sea level and water table level it would be useful to assess the direction of groundwater flow during periods of low and high sea level throughout the wetland sites (Winter, 1988; Rosenberry & Winter, 1997; Poole *et al*., 2006; Bradley *et al*., 2010). Previous studies assessing groundwater flow direction have used piezometers located in a transect from the main water source, in the case of Estonian coastal wetlands this is likely to be the sea, moving away from the water source (Winter, 1988; Rosenberry

& Winter, 1997; Poole *et al*., 2006; Bradley *et al*., 2010; Kaplan *et al*., 2010; Rodhe & Seibert, 2011). Groundwater flow direction can be assumed to be equal to the slope direction of the groundwater table (hydraulic gradient) derived from the piezometer transect data (Rodhe & Seibert, 2011). Further study could also be undertaken to assess rates of groundwater flow using Darcy's law, groundwater flow (Q) = K(dH/dL). K = hydraulic conductivity and is related to soil texture and structure and dH/dL = hydraulic gradient which can be calculated using data collected from the piezometer transect (Richardson & Vepraskas, 2001). Assessment of the hydraulic conductivity would also provide information as to which are the main mechanisms of water movement through the soil.

The results of chapter 4 also showed that the OP plant community was the only plant community not able to be identified by elevation alone, which prevented its inclusion in the modelling process. A suggestion has been made that the formation of the OP plant community could be related to trampling by cattle altering the soil texture and structure (Puurmann *pers. comm*. 2010). However, whilst a variety of authors have described the environmental variables that occur within this plant community (Tyler, 1969; Walentinus, 1973; Burnside *et al*., 2007; 2008; Ward *et al*., 2010) there have been no conclusive suggestions as to the mechanism of formation, which provides an opportunity for further study.

The results of chapter 6 provided an interesting indication of sediment accretion rates which were then incorporated into the dynamic correlative plant community model. However, sediment accretion rates have been shown to vary greatly over different areas of coastal wetlands (Cahoon & Reed, 1995; Roman *et al*., 1997). Further to this the ^{137}Cs in the Matsalu TG core appeared to have undergone post depositional mobility down the core and there appeared to have been an erosional event at the top of the core, thereby limiting the use of the CRS method for assessing sediment accretion. Therefore further study should be undertaken to assess sediment accretion variation across each wetland site including collecting further cores in the LS and TG plant communities to improve model accuracy.

In all the IPCC (2007) sea level rise scenarios using both accretion rate estimates, the Tahu coastal wetland site was found to considerably prograde into the adjacent bay. However, the accuracy of the terrestrial LiDAR data used in this study is limited at depths greater than 0.10m below the water surface. In order to overcome this problem a bathymetric LiDAR system using a maximum wavelength 700nm, typically 532nm, would be suitable. Bathymetric LiDAR are able to provide accurate elevation data for both submarine and terrestrial surfaces. However, previous studies have found problems linking terrestrial LiDAR data with bathymetric LiDAR data (Quadros *et al*., 2008; Gallant & Austin, 2009) and there are at present no commercially available data for the study areas. Therefore a bespoke flight using the bathymetric LiDAR device to record data for the whole study area would be necessary.

9.2.2 Development of a LiDAR plant community model for use in other environments

The results of this study have shown that LiDAR data can used to model plant communities in Baltic wetland environments with low relief and where the arrangement of the plant communities is related to elevation. The addition of dGPS calibration and adjustment and the use of TIN interpolations improved the accuracy of the model to such an extent that it is likely to be able to be used to improve previously developed static plant community models in other environments such as salt marshes and floodplains. Further to this a static plant community model could be used as a baseline to more accurately assess the impacts of changes in hydrology related to, for example, sea level rise and managed retreat of sea defences on salt marshes; soil accretion or erosion, the addition or removal of drainage ditches, in salt marshes or floodplains; and changes in precipitation and hence hydrological input to rivers on floodplains. For the UK there are already a range of data available regarding the relationship between the hydrological regime of both salt marshes (Ranwell *et al*., 1964; Stubbings & Houghton, 1964; Gray, 1992) and floodplain meadows (Gowing *et al*., 2002; Toogood, 2005; Grapes *et al*., 2006; Duranel *et al*., 2007) and plant community type. These data could be readily used in the plant community model tested in this study to develop accurate models for such wetlands, which would be an important resource for researchers, land managers and nature conservationists.

APPENDIX I

A.I.I. Summary of plant community types

Estonian coastal wetlands correspond to the EC habitat directive 92/43/EEC of May 1992 habitat Boreal Baltic coastal meadows, Annex 1 (code: 1630) and the EUNIS (European Nature Information System) habitat classification A2.5 Coastal saltmarshes and saline reedbeds (for full classification information see Davies *et al.*, 2004). The EUNIS habitat types are either included within EC habitat directive 92/43/EEC classification or these can overlap several EUNIS habitat types. Estonian coastal wetlands are a priority natural habitat type, which means they are in danger of disappearance and hence the European Union has a particular responsibility to protect. The vegetation of this habitat type is typified by low growing graminoids including grasses such as *Festuca rubra*, and *Agrostis stolonifera*, rushes such as *Juncus gerardii*, and *Eleocharis uniglumis* and sedges including *Blysmus rufus*, *Blysmus compressus*, and *Carex nigra*. These graminoids dominate, but many forbs exist within the sward. Many halophytic species occur on these wetlands including *Plantago maritima*, *Triglochin maritimum*, *Aster tripolium*, *Suaeda maritima* and *Salicornia europea* particularly in the lower and more frequently inundated areas of the wetlands. Estonian coastal wetlands are interspersed with other boreal habitat types dependent on micro-topography, salinity, sediment type and management. In the lowest areas surrounding these wetlands are found Large shallow inlets and bays (EU habitat code 1160 and EUNIS classification A1.1, A1.2, A1.3, A1.4, A2.1, A2.3, A2.5, A2.6, A5.5 and A7.9), which cover 20 000 ha in Estonia. Typical species found in these bays are *Phragmites australis*, *Potamogeton spp*, *Zostera marina*, and *Fucus vesticulosus* (EEA, 2008). Within and on the edges of the grasslands are Coastal lagoons (EU habitat code 1150 and EUNIS classification A1.3, A2.2, A2.3, A2.5, A3.34, A5.31, C1.5 and C3.4) covering 6000 ha throughout Estonia. These habitats commonly support species such as *Schoenoplectus tabernaemontanii* and *Bolboschoenus maritimus* and are considered to be a priority habitat type (EU Habitats Directive, 1992). Coastal lagoons are separated from the sea either temporarily or permanently and some dry out during droughts. Salinity varies greatly dependent on how often water exchange occurs between the Baltic Sea (EU Habitats Directive, 1992). Vegetation varies due to salinity, grazing and sediment type *Phragmites australis* tends to dominate in ungrazed areas. Mudflats and sandflats not covered by seawater at low tide (EU habitat code 1140 and EUNIS classification A2.1, A2.2, A2.24, A2.3 and A2.6) occur in Estonian coastal wetlands (although there is practically no tide in this area, Estonia accepted this EU habitat type). Mudflats and sandflats cover 24 500 ha throughout the territory of Estonia and typically have no vegetation cover although in unmanaged areas they become overgrown with *P. australis*. Mudflats and sandflats are important habitats for both migrating and breeding waterfowl and waders who feed there during low water. Throughout the coastline of all European coastal countries exists the habitat Annual vegetation on drift lines (EU habitat code 1210 and EUNIS classification B2.1); in Estonia this habitat type covers 500 ha and is a very dynamic and is considered to be an essential part of a larger coastal habitat complex. These are important feeding sites of migrating waders and other birds. Along the edges of these coastal wetlands are perennial vegetation of stony banks (EU habitat code 1220 and EUNIS classification B2.3). Perennial vegetation of stony banks covers an area of 1500ha in Estonia and is usually found directly inland of the habitat type 1210 although is less dynamic, which allows the development of perennial vegetation. Perennial vegetation of stony banks is an important nesting site for some waders such as *Charadrius hiaticula* and *Haematopus ostralegus*. *H. ostralegus* also use the mudflats as feeding grounds. Common vegetation species that occur in the Perennial vegetation of stony banks habitat type are *Elytrigia repens*, *Atriplex littoralis*, *Lepidium latifolium*, *Crambe maritima* and *Salsola kali*. In a mosaic throughout Estonian coastal wetlands are bare areas that are lower lying than the surrounding area. These bare patches are created by disturbance, either by trampling, vegetation removal following turf removal by machinery or due to local changes in sediment type. These bare patches are named *Salicornia* and other annuals colonising mud and sand (EU habitat code 1310 and EUNIS classification A2.5). Salinity is the main controlling factor of vegetation species and typical species occurring here are *Salicornia europea, Suaeda maritima, Halimione pedunculata, Salsola* and other halophyte plants. They cover an area of 500 ha in Estonia, which is very small in comparison with other listed types. Within the Estonian coastal wetlands, in damper nitrogen rich areas, patches of hydrophilous tall herb fringe communities of plains and of the montane to alpine levels develop (EU habitat code 6430 and EUNIS classification E5.5). Hydrophilous tall herb fringe communities areas cover 2000 ha within Estonia and typical species that occur there are *Epilobium hirsutum*, *Alliaria petiolata*, *Filipendula ulmaria* and *Cirsium oleraceum*. Hydrophilous tall herb fringe communities typically occur at the edges of the meadows just outside of the studied area.

In the upper area of the Estonian coastal wetlands on the edge of the frequently inundated zones are found *Juniperus communis* formations on heaths or calcareous grasslands (EU habitat code 5130 equivalent to EUNIS classification F3.1). The *Juniperus communis* formations habitat type covers 5300 ha in Estonia and is typically a later succesional stage on the edges of Estonian coastal wetlands and alvar grasslands, and with insufficient grazing, can develop into a forest habitat. Typical species include *Juniperus communis*, *Molinia caerulea*, *Sesleria caerulea* and *Carex nigra*.

Other habitats that occur on the periphery of Estonian coastal wetlands are Seminatural dry grassland and scrubland facies on calcareous substrates (EU habitat code 6210, EUNIS classification E1.2). This habitat type covers 5000 ha over the territory of Estonia and can be found in conjunction with *Juniperus communis* formations where more calcareous substrates occur.

European dry heaths (EU habitat code 4030, EUNIS classification F4.2) are also found on the boundaries of the Estonian coastal wetlands and are the earlier succesional stage of the previous two habitat types, Seminatural dry grassland and scrubland facies and *Juniperus communis* formations. Without management European dry heaths will develop into the more scrub-like vegetation types of Seminatural dry grassland and scrubland facies on calcareous substrates and *Juniperus communis* formations on heaths or calcareous grasslands. The European dry heaths habitat type covers an area of 450 ha in Estonia and common species that occur are *Thymus serpyllum*, *Galium verum* and *Hieracium pilosella*.

Table A1: Habitat types found occurring on and adjacent to coastal wetlands in Estonia.

Primary habitat type		Associated habitat types	
Code	Within Estonian coastal wetlands	Code	Name
*1630	Boreal Baltic coastal meadows	1140	Mudflats and sandflats not covered by seawater at low tide
		1210	Annual vegetation of drift lines
		1220	Perennial vegetation of stony banks
		1310	*Salicornia* and other annuals colonizing mud and sand
		1620	Boreal Baltic small islands and islets
		6410	*Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (*Molinia caeruleae*)
		5130	*Juniperus communis* formations on heaths or calcareous grasslands

Location communities found in relation to the Boreal Baltic coastal meadows	Code	Name
Found on the coastal edges	1150	Coastal lagoons
	1160	Large shallow inlets and bays
Found on the landward edges	6430	Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels
	7230	Alkaline fens
	4030	European dry heaths
	6210	Seminatural dry grasslands and scrubland facies on calcareous substrates
	6280	Nordic alvar and precambriam calcareous flatrocks

Nordic alvar and precambriam calcareous flatrocks (EU habitat code 6280, EUNIS classification E1.2) can occur at the edges of Estonian coastal wetlands in areas where the soil is extremely thin or absent over limestone/limestone gravels. Nordic alvar and precambriam calcareous flatrocks can also occur in complexes with *Juniperus communis* formations on heaths or calcareous grasslands. This habitat covers an area of 10 000 ha in Estonia and typical plant species that occur there are *Geranium sanguineum*, *Filipendula vulgaris* and *Helianthemum nummularium*.

Amongst the landward edges of Estonian coastal wetlands are found Alkaline fens (EU habitat code 7230, EUNIS classification D4.1). This habitat type is a mire occupied by peat producing small sedge and brown moss communities with minimal water level fluctuation. They can be open due to hydrological conditions or due to historical grazing/ mowing. Following succesional lines they can develop into woodlands following the cessation of management.

Many Estonian coastal wetlands exist on and within the Boreal Baltic islets and small islands (EU habitat code 1620 and EUNIS classification B3.2) which cover an area of 1060 ha. These islands are very important as breeding grounds for many bird species including *Vanellus vanellus*, *Tringa totanus*, *Alauda arvensis* and *Anthus pratensis*.

The location, both in and around Estonian coastal wetlands, of the associated EU habitat types is summarised in table A1.

A detailed classification of Estonian coastal plant communities has also been produced by Rebassoo (1975) in which 15 communities were distinguished. These consisted of *Eleocharetum parvulae*, *Tripolio-Triglochinetum maritimi*, *Halo-Bolboschoenetum maritimi*, *Eleocharetum uniglumis*, *Salicornietum europaeae*, *Puccinelietum maritimae*, *Spergularietum salinae*, *Glauco maritimae-Juncetum gerardii*, *Festucetum rubrae*, *Festucetum arundinaceae*, *Seslerietum caeruleae*, *Arrhenatheretum elatioris*, *Caricetum distichae*, *Hierochloetum odoratae* and *Elytrigietum repentis*.

The *Eleocharetum parvulae* community was found in very few Estonian coastal wetland sites on the larger islands of Estonia and was not found on any of the mainland or smaller island sites of the Burnside *et al* (2007) study. This habitat type was also absent from the sites used in this study. Similarly the *Tripolio-Triglochinetum maritimi* plant community was only noted on the islands of the western Estonian archipelago by Rebassoo (1975).

The *Halo-Bolboschoenetum maritimi* plant community was a species poor habitat type, typically submerged for most of the growing season. Rebassoo (1975) found that this plant community was made up of three subassociations. These subassociations were: *Phragmitetosum australis*, *Bolboshoenetosum maritimi and Schoenoplectosum lacustris*. The *Phragmitetosum australis* was dominated with *Phragmites australis* and related to the RS plant community described by Burnside *et al*. (2007). The *Bolboshoenetosum maritimi* was dominated by *Bolboschoenus maritimus* and the *Schoenoplectosum lacustris* by *Schoenoplectus lacustris* and these two communities related to the CS plant community described by Burnside *et al*. (2007). Both Rebassoo (1975) and Burnside *et al*. (2007) suggested that *Phragmites australis, Schoenoplectosum lacustris* and *Bolboshoenus maritimus* existed in low abundance in each of the sub-associations. The *Schoenoplectosum lacustris* sub-association was found to be very rare. Rebassoo (1975) suggested that this was due to the species sensitivity to grazing.

The *Eleocharetum uniglumis* plant community was not found in many Estonian coastal wetlands. Due to its low area coverage in the areas where it did occur, it was not considered by previous or later authors (Lippmaa 1931; Laasimer, 1965; Burnside *et al*., 2007) to be an independent community.

Salicornietum europapaeae was an open halophilous community type characterised by *Salicornia europapaea* and *Suaeda maritima*. This habitat type was described as ephemeral by Rebassoo (1975), although it was found in many Estonian coastal wetlands throughout the north west, west coast of the mainland as well as on some western islands of Estonia. This plant community type was considered to be synonymous with the OP plant community described by Burnside *et al*. (2007).

The *Puccinelietum maritimae* was only found on the western islands of Estonia and was rarely found in Estonian coastal wetlands close to or on the mainland. Rebassoo stated that due to the mix of species from other plant community types this might not be considered to be a distinctly marked unit. The *Puccinelietum maritimae* plant community type was absent from other classification systems.

Similarly the *Spergularietum salinae* plant community type was also only found on a few west Estonian islands and was not found further east. This was suggested by Rebassoo (1975) to be due to the species being at the north eastern limit of its distribution.

The *Glauco maritimae-Juncetum gerardii* plant community type was considered to be the most widespread of all the plant community types. The dominant species in grazed areas is *Juncus gerardii* with *Glaux maritima*, *Triglochin maritima* and other halophytic species as secondary species in the canopy. This plant community was synonymous with the LS plant community in the Burnside *et al*. (2007) study. Rebassoo suggested that at a slightly higher altitude the *Festucetum rubrae* community type occurs which related to the US plant community described by Burnside *et al*. (2007). This plant community type was considered by Rebassoo (1975) and Paal (1998) to be a transition to more terrestrial vegetation and was found to have a very wide distribution throughout Estonian coastal wetlands.

Festucetum arundinaceae as a community was found to be widespread, although this community varied greatly from site to site and in some cases existed in place of the *Festucetum rubrae* community (Rebassoo, 1975). This plant community related to the TG plant community described by Burnside *et al*. (2007).

Seslerietum coeruleae, Caricetum distichae and *Arrhenatheretum elatioris* were communities more commonly found in fresh water inundated areas such as floodplains and swampy meadows in Estonia. These communities rarely occured in Estonian coastal wetlands and where they did occur were found in areas rarely influenced by sea water but with some influence of fresh water flowing from inland (Rebassoo, 1975).

Hierochloetum odoratae was found only locally in Estonian coastal wetlands on western islands and was not found to cover large areas.

The final plant community type described by Rebassoo (1975) was *Elytrigietum repentis and* was occasionally recorded displacing the *Festucetum rubrae* in areas with high soil nitrogen contents.

Appendix II

See CD of environmental data.

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