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Saltmarsh Function and Human Impacts in Relation to Ecological Status (SAMFHIRES)

Authors: Philip M. Perrin, Stephen Waldren, Marcin R. Penk and Fionnuala H. O'Neill




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Rialtas na hÉireann
Government of Ireland

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Cover images: top left: recording ground elevation with a differential global navigation satellite system, Castletown River, County Louth; top right: grazing exclosure on saltmarsh, Ballyteigue Burrow, County Wexford; bottom left: *Salicornia* bed, Tramore, County Waterford; bottom right: saltmarsh created by managed realignment, Kilmacleague West Wetlands, County Waterford.

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Executive Summary

Coastal saltmarshes provide a range of important ecosystem services but face increasing challenges because of human activities. Ecological assessments of saltmarshes are required to inform reporting in compliance with the European Union (EU) Habitats Directive (HD) (92/43/EEC) and Water Framework Directive (WFD) (2000/60/EC). A WFD saltmarsh assessment tool was recently developed through the Environmental Protection Agency (EPA)-funded SMAATIE (Saltmarsh Angiosperm Assessment Tool for Ireland) project. This report addresses knowledge gaps identified by SMAATIE that limited the development and application of the tool, and provides an additional, synergistic understanding of saltmarsh processes.

Anthropogenic Pressures on Saltmarshes

Nutrient enrichment can severely impact saltmarshes, but there have been few European investigations of such impacts. Data on soils, vegetation and elevation in 15 saltmarshes were recorded and compared with existing water quality data for corresponding water bodies. Plant-available soil P and N were only weakly related to water nutrients. Plant community composition was highly significantly related to soil N and, to a lesser degree, soil P. This was most clearly evident in the positive association of the cover of *Atriplex portulacoides* and the weaker, negative association of *Plantago maritima* with nitrate, indicating competition between these key and functionally contrasting species. Above-ground biomass increased and below-ground biomass decreased along the oxidised N gradient in soil, with a 6.6-fold increase in the above-to-below-ground-biomass ratio. Nutrient-driven decreases in diversity, shifts in community composition, and strong shifts in above- and below-ground biomass allocations would have important consequences for ecosystem functions, although the causality is yet to be ascertained. Saltmarsh soils and plant communities are unlikely to serve as general sentinels of nutrient conditions in corresponding water bodies and may need separate assessment criteria.

Livestock grazing is a major pressure on European saltmarshes, yet very little is known about its effects on Irish vegetation. These effects were assessed through (1) an observational survey utilising existing spatial differences in grazing regimes and (2) 2-year experimental manipulation of grazing levels through fenced exclusion of livestock. Grazing resulted in shorter, more species-rich and more open vegetation, with positive effects on annual species and, at least in the longer term, on *Puccinellia maritima,* but negative effects on *Aster tripolium* and *A. portulacoides*. Responses of vegetation to grazing are to some extent site specific, and management plans should reflect both this and the vegetation structure requirements of fauna of conservation value.

Functions and Processes of Saltmarshes

Saltmarshes provide unique habitats for flora. This function was characterised by comparing the biodiversity value of saltmarshes with that of other Irish habitats of high conservation value. Three related rarity indices were developed and applied at ecosystem, habitat and community levels. Saltmarshes were subsequently categorised as “specialist vascular plant habitats”, whose biodiversity value lies in the occurrence of specialist salt-tolerant species rather than high species richness. The integration of rarity indices into site conservation assessments is recommended.

The resilience of saltmarshes, through the processes of accretion and landward migration, to the potential threat of forecast sea level rise was studied by developing a statistical model. MARGOT (Marshes Governed by Tides) simulated the effect of different sea level rise scenarios, accretion rates and managed realignment strategies on saltmarsh extent, and was applied to areas within two Irish estuaries. Without managed realignment, which allows saltmarshes to migrate inland as a response to sea level rise, saltmarshes were predicted to decline at both sites in a majority of simulations. Ireland is lagging behind other European countries in the adoption of managed realignment as part of a sustainable coastal management strategy.

Monitoring and Assessing Saltmarshes

Comprehensive definitions of saltmarsh zones are vital for assessments under EU directives. To address potential gaps in the saltmarsh division of the Irish Vegetation Classification (IVC), a field survey of rare and under-recorded vegetation communities of Irish saltmarshes was conducted. Plot data were classified according to the current IVC scheme and the results were used to recommend amendments to it.

A single protocol for saltmarsh field surveys is required so that data necessary for both HD and WFD reporting can be collected efficiently. Existing mapping and assessment procedures were reviewed and amended. A major change was switching to IVC categories for mapping communities. Five sites across Ireland were field surveyed to test the amended procedures. Resultant maps were more detailed and informative than those previously produced. Further recommended amendments to these procedures have been outlined, e.g. mapping *Elytrigia* swards.

1 Introduction

1.1 Background

Saltmarshes occur within the upper intertidal zone of sheltered coastlines, such as in protected bays, lagoons and estuaries. They have a global distribution, although they are gradually replaced by mangroves towards the tropics. Ireland has approximately 40 km^2 of saltmarshes. These habitats support a specialised assemblage of salt-tolerant plants (halophytes). The vegetation varies significantly with the degree of tidal inundation and thus saltmarshes can display distinct zonation up the shore. The lower fringes adjoin tidal flats and may support pioneer communities of glassworts (*Salicornia* spp.) and non-native cord-grasses (*Spartina* spp.). Above this occurs the lower saltmarsh, usually dominated by common saltmarsh-grass (*Puccinellia maritima*). This is followed by the middle marsh characterised by thrift (*Armeria maritima*) and sea plantain (*Plantago maritima*). Behind this occurs the upper saltmarsh, typically dominated by red fescue (*Festuca rubra*) and rushes (*Juncus maritimus* and *J. gerardii*), which gradually transitions to terrestrial vegetation (Sheehy Skeffington and Wymer, 1991). The breadth of this zonation depends on geomorphological settings and anthropogenic pressures. A characteristic network of pans and creeks develops throughout these zones, further conferring a unique character to saltmarsh habitats.

Saltmarshes are typically characterised by high rates of biological processes, including productivity (Mcleod *et al.*, 2011), that promote strong linkages across ecological guilds, habitats and ecosystems. Such linkages underpin a broad range of high-value ecosystem services (Barbier *et al.*, 2011; de Groot *et al.*, 2012). These services are conveyed mainly through vascular plants, which form the structural and functional foundation of the saltmarsh ecosystem. Plants absorb nutrients and C as they produce biomass (Rozema *et al.*, 2000; Mcleod *et al.*, 2011), which then fuels terrestrial and aquatic food webs through grazing and detrital exports (Silliman and Zieman, 2001; Svensson *et al.*, 2007; Schrama *et al.*, 2013). Plant shoots also attenuate wave energy and intercept suspended solids, whereas roots and rhizomes bind and thus stabilise the sediment (Gedan *et al.*, 2011; Shepard *et al.*, 2011). Saltmarshes have a relatively small global extent but a disproportionately large capacity to sequester C into their substrates, thereby mitigating climate change (Chmura *et al.*, 2003; Gedan *et al.*, 2011).

Similar to other coastal habitats, saltmarshes are under pressure from a multitude of anthropogenic impacts (Weis *et al.*, 2016) and are declining globally. Sea level rise may gradually shift shore zonation and associated habitats landwards, but the migration of these habitats can be impinged by coastal defences, leading to “coastal squeeze” (Pontee, 2013). In addition, Irish saltmarshes are commonly intensively grazed by cattle and sheep (particularly in the west of Ireland), which is well known to impair saltmarsh functions (Davidson *et al.*, 2017). Furthermore, eutrophication has been shown elsewhere to impact saltmarshes, but there is no evidence to evaluate such impacts in Ireland. An estimated 50% of saltmarshes have been lost or degraded worldwide (Barbier *et al.*, 2011). In Ireland, approximately 90–150 km^2 of intertidal areas, including saltmarshes, had been converted to land by 1900, mostly for agricultural use (Devoy, 2008). It is therefore vital and urgent to assess the condition, functions and resilience of Irish transitional habitats, and their importance to the wider environment, so that the various conflicting interests in these habitats (e.g. wave energy attenuation vs livestock grazing; Davidson *et al.*, 2017) can be prioritised based on evidence.

The unique character of saltmarshes is recognised under the European Union (EU) Habitats Directive (HD) (EU, 1992), which lists them as habitats whose conservation requires the designation of Special Areas of Conservation. Furthermore, the EU Water Framework Directive (WFD) (EU, 2000) aims to protect and enhance the quality of water bodies, including estuaries and coastal waters. One of the biological quality elements to be assessed is “angiosperms”, which includes saltmarsh communities. Thus, Ireland has a legal requirement to assess and protect saltmarsh habitats. A WFD status assessment tool for saltmarshes has been recently developed and tested on existing data through the desk-based Saltmarsh Angiosperm Assessment Tool for Ireland (SMAATIE) project (Devaney and Perrin, 2015a,b) funded by the Environmental Protection Agency (EPA). The SMAATIE project has identified a number of knowledge gaps that have limited the development of the tool and its application. In particular, it has highlighted the absence of specific data linking livestock grazing and the eutrophication of saltmarshes to ecological measurements. Furthermore, deficiencies have been noted in quantifying the ecological functioning of saltmarshes, in describing certain saltmarsh communities and finally in testing the applicability of SMAATIE in the field.

1.2 Project Scope

The Saltmarsh Function and Human Impacts in Relation to Ecological Status (SAMFHIRES) project was a 36-month multidisciplinary collaboration between Botanical, Environmental & Conservation Consultants Ltd and the Department of Botany, Trinity College Dublin. The project consisted of three work packages (WPs): WP1–WP3.

WP1 (reported in Chapters 2 and 3) aimed to address anthropogenic pressures on Irish saltmarshes with a research focus on two impacts perceived to be important. First, through field surveys of soil properties, vegetation characteristics and ground elevation, combined with existing EPA water quality data for the corresponding water bodies, this WP investigated the extent to which nutrient pools in saltmarsh soils are related to those in tidal waters across Irish saltmarshes. It then tested whether or not saltmarsh plant diversity, community composition and biomass are related to nutrients. Second, this WP sought to provide empirical data on the effects of livestock grazing on Irish saltmarsh vegetation structure and composition. A dual approach was used, which comprised both conducting an observational survey at multiple sites by utilising existing spatial differences in grazing regimes and experimentally manipulating grazing levels at two contrasting sites through fenced exclusion of livestock.

WP2 (reported in Chapters 4–6) aimed to investigate some of the functions and processes of saltmarshes in Ireland. First, the provision of a suitable habitat for a diverse flora was examined by comparing the biodiversity value of saltmarshes at ecosystem, habitat and community levels with that of other habitats of high conservation value. To achieve this, a group of related rarity indices was developed. Second, the resilience of saltmarshes to forecast sea level rise (SLR) through the processes of accretion and landward migration was studied by developing a spatially explicit statistical model. In addition, changes as a result of SLR in the provision of ecological functions, such as wave attenuation and “blue carbon” storage, were predicted. Finally, the potential for the application of managed realignment in Ireland, a coastal defence strategy that mitigates some impacts of SLR, was investigated through both modelling scenarios and a series of real-world case studies.

WP3 (Chapters 7 and 8) aimed to address the practical aspects of monitoring and assessing saltmarshes in Ireland. First, because the characterisation of zones underpins key components of saltmarsh assessments, a field survey recorded data from a number of rare or under-recorded saltmarsh communities that are not currently recognised by the Irish Vegetation Classification (IVC). Second, the assessment tool developed by the SMAATIE project for the purposes of WFD reporting and the saltmarsh assessment criteria for the purposes of HD reporting were both reviewed and refined. These were then tested using mapping and assessment data from a dedicated field survey. This WP also aimed to consider outputs from WP1 and WP2 in recommending further amendments to these assessment procedures.

This project conducted an extensive literature review, which is compiled in a bibliographic file (EndNote XML format). A portfolio of vegetation maps (PDF format) from five saltmarsh sites assessed and mapped as part of WP3 has also been compiled. Both resources can be accessed through the SAFER-Data web-based interface to the EPA’s Environmental Research Data Archive at http://erc.epa.ie/safer/.

2 The Relationships between Saltmarsh Plant Community Composition, Biomass and Nutrients

2.1 Introduction

Nutrient enrichment can severely impact saltmarshes, but the few investigations of such impacts in Europe have been strongly localised and have not addressed community-level properties. Elucidating these impacts at broad spatial scales demands a better understanding of the underlying nutrient linkages between saltmarsh soils and tidal waters. This study first investigated the extent to which nutrient pools in saltmarsh soils are related to those in tidal waters across Irish saltmarshes. It then tested whether or not saltmarsh plant diversity, community composition and biomass are related to nutrients.

2.2 Methods

2.2.1 Data collection

Fifteen Irish saltmarshes were selected from water bodies on the east and south coasts of Ireland, representing a broad range of labile P and inorganic N concentrations. Between 12 and 20 plots per saltmarsh (246 in total), distributed within the section of the saltmarsh with the maximum variation of vegetation types, were surveyed. Within each plot, the percentage cover of each plant taxon was estimated between July and August 2016. Species nomenclature follows Stace (2010). At each plot at low tide in August 2016, the project team (1) collected above-ground (AG) plant biomass from a 46-cm-diameter area, (2) combined three soil cores of 35-mm-diameter to a depth of 30 cm within the cleared patch for determination of below-ground (BG) plant biomass and (3) combined eight soil cores of 22-mm diameter to a depth of 10 cm from throughout the plot for soil analyses. Ground elevation was measured at the centre of each plot as an index of tidal inundation regime. Biomass was determined as dry weight. Salinity data were expressed per weight of soil moisture; all other soil data were expressed per weight of field-moist soil. Elevation was expressed as a proportion of the highest astronomical tidal amplitude within each saltmarsh to standardise for differences in tidal amplitude among saltmarshes. Water nutrient concentrations from 2010–2016 EPA water quality data were modelled as linear functions of water salinity over all data relating to a particular saltmarsh (2–6 sites and 15–246 data points per saltmarsh) and then predicted for 75% of maximum water salinity at that saltmarsh.

2.2.2 Statistical analyses

All statistical analyses were conducted in R (R Core Team, 2018). The relationships between, on the one hand, soil nutrients and salinity and, on the other hand, water properties were investigated using linear regressions. Similarly, the relationships between, on the one hand, species richness and the Shannon diversity index and, on the other hand, soil and water nutrients were also tested using linear regressions. For plant multivariate community composition, relationships with soil and water nutrients were analysed using multivariate redundancy analysis (RDA). The relationships between plant biomass (AG, BG, total and AG:BG ratio) and soil nutrients were analysed using generalised additive models (GAMs). Ground elevation (as the first- and second-degree polynomial terms) and soil pH, moisture and sand content were additional explanatory variables in each model. Soil salinity was included in each vegetation model.

2.3 Results

2.3.1 The relationships between soil and water nutrients

The concentration of labile P in soil was associated positively with water PO_4^{3-}, which accounted for ≥7.5% of variance (Figure 2.1a). Non-labile P in soil was associated positively with water PO_4^{3-}, which accounted for ≥10.0% of variance (Figure 2.1b). Soil NO_x^- concentration was associated positively, although weakly, with water dissolved inorganic N (DIN), which accounted for ≥5.2% of variance (Figure 2.1c). Soil NH_4^+ and organic N concentrations were not significantly related to water DIN.

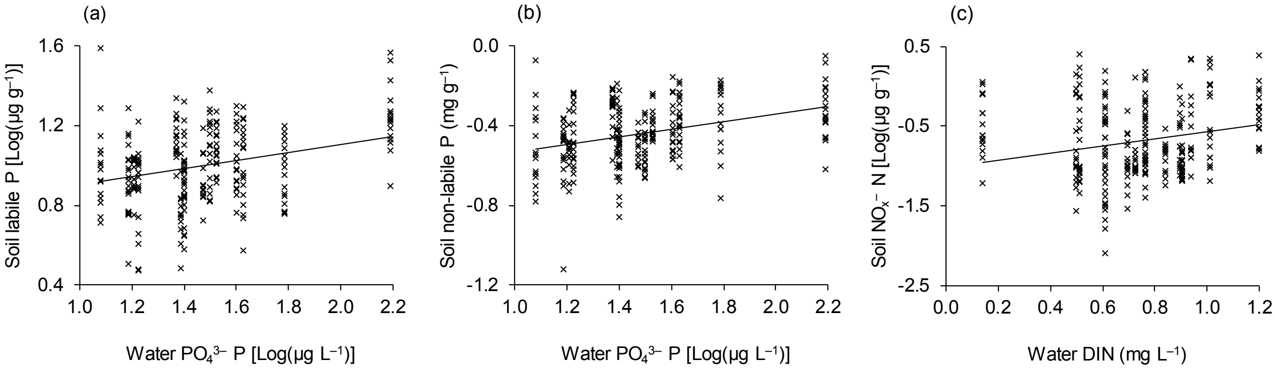

Figure 2.1. Significant relationships between water and soil nutrients: (a) soil labile P vs water PO_4^{3-}, (b) soil non-labile P vs water PO_4^{3-} and (c) soil NO_x^- vs water DIN. Reproduced from Penk *et al.* (2019a).

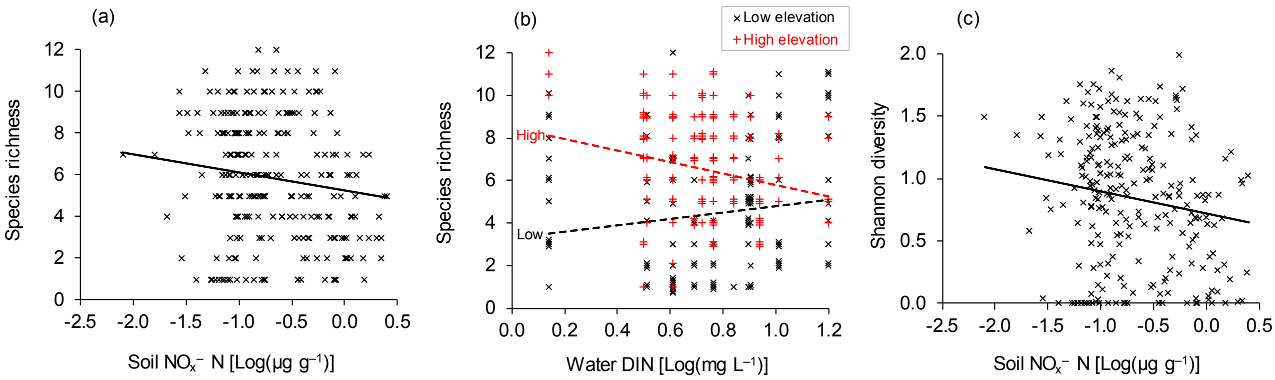

Figure 2.2. Significant relationships between nutrients and species diversity indices: (a) species richness vs soil NO_x^-, (b) species richness vs water DIN and (c) Shannon diversity vs soil NO_x^-. Solid lines indicate significant main term ($p \leq 0.05$). Broken lines indicate significant interaction with elevation and are plotted for 25% (low) and 75% (high) of the elevation range. Repetitive richness–DIN combinations are offset for better visual representation.

2.3.2 The relationships between plant diversity and community composition and nutrients

Species richness ranged from 1 to 12 species per plot, and Shannon diversity ranged from 0 to 1.99 among plots. Both were related negatively to soil NO_x^- (Figure 2.2a and c). There was also a negative interaction, such that the association of richness with water DIN appeared to change from positive to negative with increasing elevation (Figure 2.2b). Water DIN and PO_4^{3-} and their interactions with elevation and soil NH_4^+ and labile P were not significant.

The RDA model showed that multivariate community composition was related to soil NO_x^-, which was among the most significant explanatory variables in the model. Community composition was also significantly related to soil labile P, water DIN, water PO_4^{3-} and its interaction with elevation. Soil NH_4^+ and the interaction of water PO_4^{3-} with elevation were not significant. Partial RDA showed that *Atriplex portulacoides* in particular, but also other species such as *Pl. maritima,* were associated with nutrients. Ground cover of *A. portulacoides* was strongly positively associated with soil NO_x^-, whereas *Pl. maritima* was negatively associated with the same nutrient (≥17.3% and ≥5.2% of variance explained, respectively; Figure 2.3).

2.3.3 The relationships between plant community biomass and nutrients

Above-ground community biomass was positively related to NO_x^- concentration, increasing with increasing NO_x^- concentration at an accelerating rate (Figure 2.4a). Modelled AG biomass increased from 0.946 to 2.130 kg m^{-2} over the recorded NO_x^- range. BG community biomass was negatively related to NO_x^- concentration (Figure 2.4b). Modelled BG

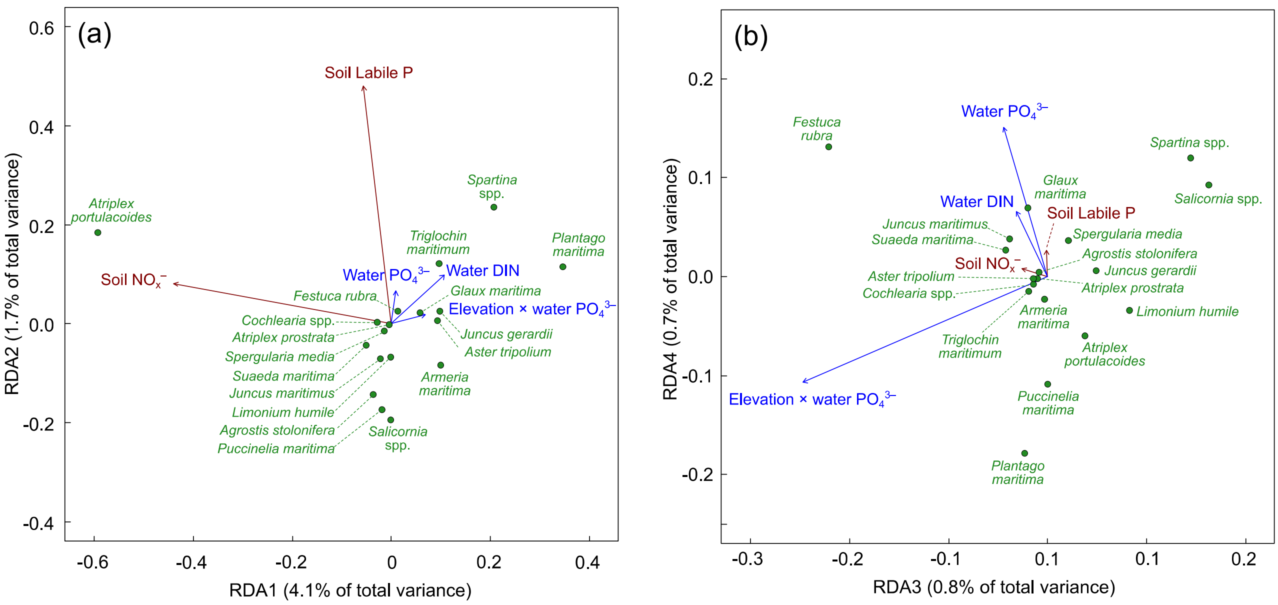

Figure 2.3. (a) The first and the second axes (best representing soil nutrients) and (b) the third and the fourth axes (best representing water nutrients) of the partial RDA of plant community composition in relation to nutrients, with all other explanatory variables as conditions.

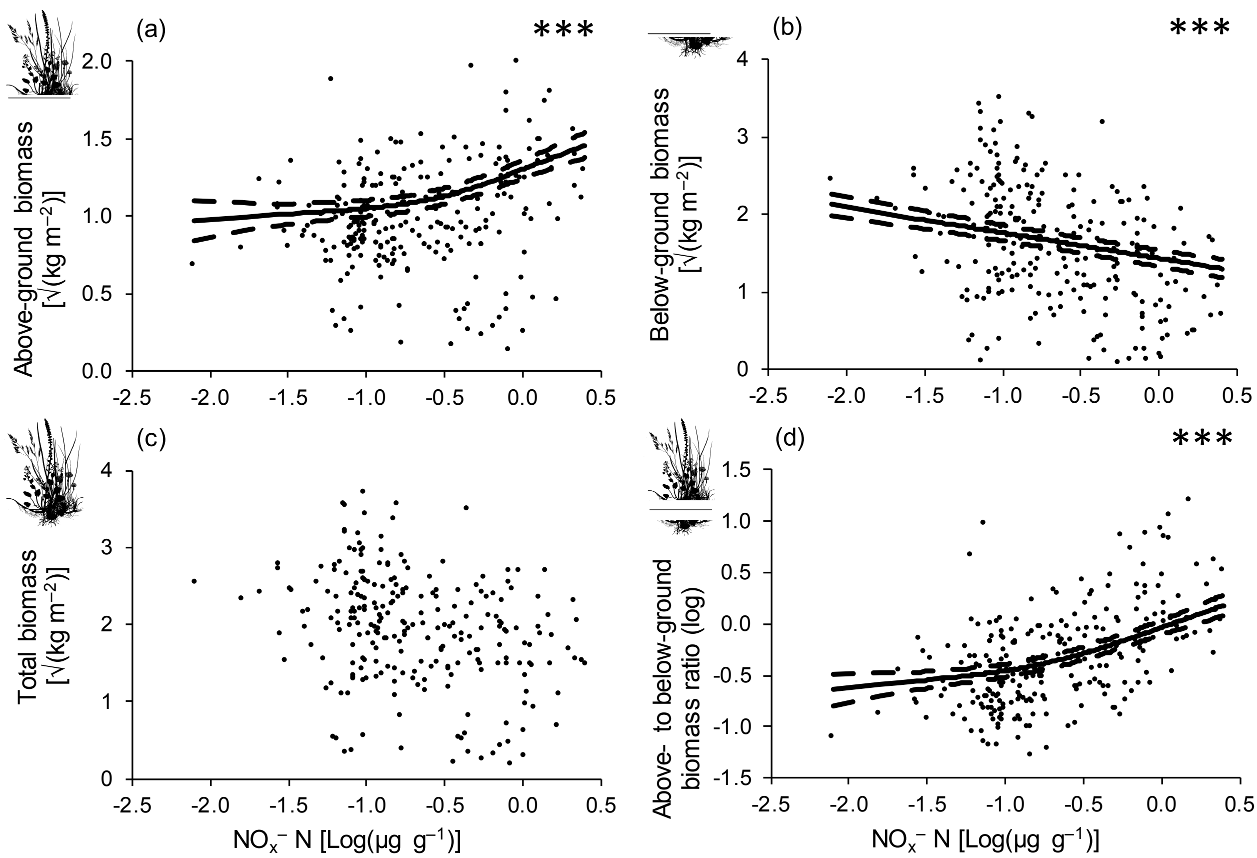

Figure 2.4. Significant relationships between soil NO$_x^-$ and different measures of plant community biomass: (a) AG biomass, (b) BG biomass, (c) total biomass and (d) AG:BG biomass ratio. Parts (a), (b) and (d) reproduced from Penk *et al.* (2019b).

biomass decreased from 4.501 to 1.691 kg m^{-2} over the recorded NO_x^- range. Total community biomass was not significantly related to NO_x^- (Figure 2.4c). The AG:BG community biomass ratio was positively related to NO_x^- concentration, increasing with increasing NO_x^- concentration at an accelerating rate (Figure 2.4d). The modelled AG:BG biomass ratio increased from 0.229 to 1.518 over the recorded NO_x^- range. AG biomass, BG biomass and AG:BG biomass ratio, but not total biomass, were also marginally significantly related to labile P concentration. None of the biomass components was significantly related to NH_4^+.

The strength of the relationship between AG biomass and NO_x^- decreased markedly after adding the ground cover of *A. portulacoides* as an explanatory variable to the model, implying its high mechanistic importance in conveying this relationship. The strength of the relationship between BG biomass and NO_x^- decreased the most after adding the ground cover of *Pl. maritima* as an explanatory variable to the model, followed by *Spartina* spp., implying their mechanistic importance in conveying this relationship. The strength of the relationship between AG:BG biomass ratio and NO_x^- decreased the most after adding the ground cover of *A. portulacoides* as an explanatory variable to the model, followed by *Pl. maritima*, implying their mechanistic importance in conveying this relationship. The low significance of the other relationships between biomass and nutrients hampers the interpretation of the influential taxa.

2.4 Discussion

2.4.1 The relationships between soil and water nutrients

This study found that both N and P pools in soil across saltmarshes were only weakly related to those in tidal waters, suggesting that other internal and external influences on nutrient budget are more important. In spite of a very strong soil salinity gradient with elevation, there was no support for the decreasing importance of tidal water influences up the elevation gradient, previously shown in a Dutch saltmarsh (Schrama *et al.*, 2013). Although such an interaction seems intuitive, it may be obscured by other factors affecting the nutrient budget, such as bird droppings, the redistribution of macroalgae, atmospheric deposition and internal cycling acting at various spatial scales.

The statistically significant relationship between soil NO_x^- and water DIN, which was mostly in the form of NO_x^-, is in line with the paradigm that saltmarshes tend to be net sinks of NO_3^- from tidal waters (Rozema *et al.*, 2000). A considerable amount of variability in soil P was associated with PO_4^{3-} in tidal waters. The correlation between labile and non-labile P fractions, and the similarity in their relationships with water PO_4^{3-}, could suggest that a large proportion of water PO_4^{3-} is sequestered in non-labile soil fractions. Alternatively, these relationships could indicate a potential tidal import of non-reactive dissolved and particulate P fractions correlated with PO_4^{3-}, but water data for the former two variables were not available.

2.4.2 The relationships between plant diversity and community composition and nutrients

This study found that both species richness and Shannon diversity were inversely related to N. It also found that community composition was highly significantly related to nutrients, in particular soil NO_x^-. This was most clearly evident in the very strong positive association of the ground cover of *A. portulacoides* and a weaker negative association of *Pl. maritima* with soil NO_x^- and appeared to be linked at least in part to the negative association of diversity indices with the same nutrient. The dominance of *A. portulacoides* was associated with a strong dip in species richness and Shannon diversity, in contrast to *Pl. maritima*. *A. portulacoides* was frequently found in near-monospecific stands, whereas plots dominated by *Pl. maritima* were among the most species rich in this survey. Furthermore, these two species co-occur in the field, have strongly overlapping elevation ranges and are both positively associated with salinity. These similarities are conducive to competition. Stands of *A. portulacoides* have half the root biomass and an order of magnitude higher shoot-to-root ratio than *Pl. maritima* (M. Penk, Trinity College Dublin, unpublished data, 2018), which is conducive to poorer competitiveness for nutrients. However, under relaxation of nutrient limitations, competition for light is considered the main mechanism structuring plant community composition (Hautier *et al.*, 2009). *A. portulacoides* is an evergreen dwarf shrub forming a thicket of obliquely arranged branches, whereas *Pl. maritima* is a short rosette-forming forb. The former physiognomy appears more advantageous for competition for light. Therefore, *A. portulacoides* may be benefiting from increased NO_x^- concentrations to outcompete *Pl. maritima*. Where present, *A. portulacoides* would lend itself well as an indicator of nutrient enrichment. It is easy to identify and map, even from aerial photographs, owing to its distinct physiognomy, and therefore could be easily incorporated into assessment programmes.

A. portulacoides was the only shrub in the dataset, and its nutrient-driven proliferation could promote stronger wave attenuation. However, in contrast to the forbs occupying similar shore elevations, it was associated with low plant diversity. Higher diversity can boost important ecosystem functions such as productivity and resilience to perturbations (Hector *et al.*, 2010). Thus, a nutrient-driven competitive shift from a herb-rich *Pl. maritima* community towards monospecific *A. portulacoides* shrub stands could have significant implications for ecosystem functioning.

The negative relationship between diversity and N is in line with findings from the USA (Theodose and Roths, 1999; Wigand *et al.*, 2003). However, the functional implications of such relationships are likely to depend on the functional traits of individual species and their associations. A potential nutrient-driven transition from forb-rich communities towards monospecific shrub stands would contrast with North America, where nutrient enrichment changes dominance among grasses. These findings highlight the potential pitfalls of uncritical global extrapolation from biogeographically restricted studies. Across the entire saltmarsh elevation gradient in our study, the relationship between diversity and N was weak, but it is likely to be spatially heterogeneous. For example, a stronger relationship could be anticipated at mid-marsh elevations, because this is where communities dominated by *A. portulacoides* and *Pl. maritima* can typically be found. Further studies to ascertain causality could determine whether or not nutrient enrichment results in the competitive decline of species-rich saltmarsh communities.

2.4.3 The relationships between plant community biomass and nutrients

This study found strong evidence for shifts in AG and BG biomass allocations, which are key functional traits of saltmarsh plant communities, along nutrient gradients. The positive relationship between AG biomass and NO_x^- was mainly mediated through *A. portulacoides*; however, the strong negative relationship between BG biomass and NO_x^- was most strongly but not exclusively driven by *Pl. maritima*, and both of these species were key contributors to the strong positive relationship between AG:BG biomass ratio and the same nutrient. These findings corroborate the observed relationships between these species' cover abundances and NO_x^- reported above and demonstrate a broad, community-wide link between biomass and nutrients. The positive association of AG biomass with soil inorganic N appears credibly causal and, together with the negative association of BG biomass with NO_x^-, corroborates the more general paradigm of a nutrient-driven shift in growth allocation from roots to shoots (Hermans *et al*., 2006; Hautier *et al*., 2009). It is also largely in line with extensive experimental findings from North American saltmarshes (reviewed in Wong *et al*., 2015).

The estimated 2.3-fold and 2.7-fold differences in AG biomass and BG biomass, respectively, along the recorded NO_x^- gradient would have significant implications for multiple saltmarsh functions. More AG biomass implies higher organic matter availability for terrestrial and aquatic consumers. It also suggests a higher potential to attenuate wave energy and intercept suspended solids. However, lower BG biomass could weaken sediment binding and thus increase susceptibility to erosion, and the higher AG:BG ratio is also conducive to higher drag in water currents relative to anchorage (Schutten *et al*., 2005), increasing plant vulnerability to storm damage. Furthermore, BG biomass was on average much higher than AG biomass across our plots. As roots die, they can be readily incorporated into the substrate, whereas dead leaf material is prone to be more widely circulated by tidal action, and so BG biomass sequesters C in the substrate better than AG biomass (Kell, 2012). The contrasting relationships between AG and BG biomass and NO_x^- demonstrated in this study drove a 6.6-fold increase in AG:BG ratio with increasing NO_x^-. Therefore, high NO_x^- probably drives poorer C sequestration in saltmarsh soils.

The strong gradients of saltmarsh plant biomass associated with nutrient concentrations, including a sevenfold difference in AG:BG biomass ratio across the recorded NO_x^- range, highlight the sensitivity of saltmarsh ecosystems, and thus the valuable ecosystem functions that they provide, to further nutrient enrichment. Ireland has recently lost considerable saltmarsh sections in Bannow Bay (at Grange) and in Wexford Harbour (at Rosslare) to coastal erosion (M. Penk, Trinity College Dublin, personal observation, 4 and 12 July 2017). Both of these water bodies are “potentially eutrophic” according to EPA (2005), the second worse category on a four-point scale, but there are no data to assess causality of these occurrences.

2.4.4 Limiting nutrients of Irish saltmarshes

The relationships between plant communities and N found in this study were stronger than those between plant communities and P, indicating that soils were more saturated with P. The positive, albeit weak, relationships between the low-marsh species *Spartina* spp. and P found in this study are consistent with an experimental study in the Wadden Sea, the Netherlands, which concluded that P limitation may prevail in marshes with low soil organic matter (Van Wijnen and Bakker, 1999), such as the colonising front of the saltmarsh, which generally has little organic matter.

Furthermore, NO_x^- was generally a less abundant form of inorganic N than NH_4^+ across the saltmarshes, and yet NO_x^- had stronger relationships with plant diversity, community composition and biomass. Even in an abundance of NH_4^+, some plants need at least some NO_3^- for optimal growth (Falkengren-Grerup, 1995). This may be particularly relevant in environments where nitrification is inhibited (Falkengren-Grerup and Lakkenborg-Kristensen, 1994).

2.4.5 Nutrient sources of Irish saltmarshes

The relationship between plant community characteristics and water nutrients was far weaker than that between plant community characteristics and soil nutrients. This supports previous assertions that root uptake of N in saltmarsh plants dominates strongly over foliar uptake (Rozema *et al*., 2000). It also implies that sources other than tidal waters probably dominate the nutrient budgets of saltmarsh plant communities in Ireland, despite regular tidal connectivity to these water bodies. A better understanding and, if necessary, better management of the contribution of tidal, airborne and biogenic nutrient sources acting at spatial scales relevant to individual saltmarshes would help in promoting saltmarsh resilience, which is particularly important and urgent in the face of progressing SLR and intensifying storm surges.

2.4.6 The alignment of saltmarsh trophic assessment with national schemes

The weak relationship between plant community characteristics and water nutrients compared with soil and the weak correspondence between soil and water nutrients found in this study indicate that saltmarsh soils are unlikely to serve as general sentinels of nutrient conditions in their corresponding water bodies. Saltmarshes share many hydromorphological characteristics with other intertidal habitats and are therefore likely to respond to similar hydromorphological pressures. European guidance proposed the use of biological elements, including saltmarshes, as one of the principal features for assessing changes due to hydromorphological pressures across the EU (CEN, 2014). Therefore, saltmarshes should be an essential part of integrated management initiatives, such as the WFD (Best *et al*., 2007; Devaney and Perrin, 2015a). However, the findings of this study indicate that their trophic conditions and nutrient pressures need separate assessment criteria and management tools. Disentangling localised influences on both soil nutrient budgets and biological communities from whole-saltmarsh influences would improve our understanding of their structure and functioning. This, in turn, would permit an assessment of change over time and link these changes to relevant anthropogenic pressures, ideally in a predictive framework, so that these changes can be used to indicate pressures.

2.5 Conclusions

The results of this study are that:

- concentrations of plant-available nutrients in saltmarsh soils correspond poorly with those in adjacent tidal waters;
- plant community composition changes, diversity decreases and the AG:BG biomass ratio increases with increasing soil nitrate concentration.

Physiognomic and functional differences between the species underlying these relationships are likely to have implications for ecosystem services. Further studies should seek to ascertain the causality of these relationships.

3 The Impacts of Livestock Grazing on Irish Saltmarsh Vegetation

3.1 Introduction

A national saltmarsh survey in Ireland (McCorry and Ryle, 2009) highlighted that more than half of the area of each of two EU HD saltmarsh habitats – 1330 Atlantic salt meadows and 1410 Mediterranean salt meadows – is grazed by livestock (mainly cattle or sheep), with 15.0% and 9.3% of these habitats, respectively, assessed as overgrazed. However, research on the effects of livestock grazing on Irish saltmarshes is scarce.

This present study examined the effects of livestock grazing on Irish saltmarsh vegetation and involved (1) an observational survey at multiple sites, utilising existing differences in grazing regimes, and (2) an exclusion experiment using fencing to manipulate grazing levels at two contrasting sites. Data are presented from the first 2 years of the ongoing exclosure experiment, and these complement the broader view provided by the observational survey.

3.2 Methods

3.2.1 Site selection

Sites for the observational survey were selected by reviewing McCorry and Ryle's (2009) saltmarsh site reports (by searching for references to grazing), and by selection in the field, where fences divided areas of ungrazed and grazed saltmarsh habitat.

Two sites were selected for the exclusion experiment: Ballyteigue Burrow Nature Reserve in County Wexford and Sheskinmore Nature Reserve in County Donegal. Both sites are winter grazed by cattle from around September to April; grazing intensity is low to medium. Both sites may also be grazed all year round by rabbits; this was factored into the experimental design.

3.2.2 Exclosure establishment

Fencing was manually erected by the project team at the two exclusion experiment sites in June 2016. At each site, four cattle-proof exclosures were built in saltmarsh corresponding to habitat 1330 Atlantic salt meadows. Three permanent plots were associated with each exclosure. The first two were marked out within the exclosure, one inside a rabbit-proof fence and the other outside it. A third plot, the control, was located 5 m outside the exclosure at the same elevation. The experiment therefore examined the effects on vegetation of three grazing treatments: (A) no cattle grazing but potential rabbit grazing; (B) no cattle or rabbit grazing; and (C) cattle grazing and potential rabbit grazing (control).

3.2.3 Field survey

Observational survey sites were surveyed between June and August in 2017 and 2018. Paired plots were positioned, one on either side of a fence dividing two different grazing regimes, on comparable elevations approximately 2 m from the fence. The percentage cover of each vascular plant species and disturbed ground in each plot was estimated visually in vertical projection. Within each quarter of each plot, the maximum height of leaves was recorded.

Baseline recording for the exclusion experiment occurred in late July 2016, with follow-up monitoring occurring at the same time of year in 2017 and 2018. Within each quarter of each plot, the maximum heights of flowers and leaves were recorded. The percentage covers of vascular plant species, individually and in total, were estimated visually in vertical projection.

3.2.4 Statistical analysis

All analyses were conducted in R (R Core Team, 2018) using the default arguments for each function, unless otherwise stated. For each plot, the median of the four maximum height measurements was used. Analysis proceeded with only the leaf height data.

For the observational survey, linear mixed-effects models (R package *nlme*; Pinheiro *et al.*, 2017) were used to examine the relationships between height and treatment (grazed or ungrazed) and between species richness and treatment. Generalised additive models with zero-inflated beta regression were used to examine the relationship between treatment and the percentage cover of selected species/disturbed ground (R package *gamlss*; Rigby and Stasinopoulos, 2005).

For the exclusion experiment, linear mixed-effects models were used to examine the change in height, species richness and total vegetation percentage cover between 2016 and 2017 and between 2016 and 2018. The interaction between treatment and site was also included. Changes in percentage cover of selected frequent species between 2016 and 2017 and between 2016 and 2018 were analysed separately for each site, again using linear mixed-effects models. Absolute changes rather than relative changes were used for percentage cover data. For example, a decline in cover from 90% to 80% would be reported as a 10.0% change, not an 11.1% change.

To examine the effects of grazing treatments on overall vegetation composition, non-metric multi-dimensional scaling (NMDS) ordinations were used (R package *vegan*; Oksanen *et al.*, 2017). The Bray–Curtis dissimilarity was used to compare plots. Separate ordinations were conducted for the experimental and observational datasets.

3.3 Results

3.3.1 Observational survey

Fieldworkers recorded 26 pairs of plots across 11 sites. A single pair of plots was recorded from habitat 1410, with the rest being from habitat 1330. Cattle were the main grazing livestock, with three sites grazed by sheep. The main findings were as follows:

- There was as a tendency for annual species, such as *Salicornia* spp., *Suaeda maritima* and *Spergularia marina*, to be more frequent in grazed plots.
- Species richness was significantly higher in grazed plots.
- The cover of disturbed ground and the cover of *Pu. maritima* were significantly higher in grazed plots.
- Vegetation was significantly taller in ungrazed plots.
- The cover of *Pl. maritima*, *J. gerardii* and *A. portulacoides* was significantly higher in ungrazed plots.
- No significant differences between treatment levels were found for the cover scores of *Glaux maritima*, *F. rubra*, *Triglochin maritima* and *Aster tripolium*.

In the NMDS ordination, the woody species *A. portulacoides* was associated with ungrazed plots and taller vegetation; this was also the case for *Elytrigia repens,* but this species occurred only in a single plot. The annual plants *S. maritima* and *Salicornia* spp. were associated with grazed and more disturbed plots. Within the plot pairs, the grazed treatment tended to promote vegetation characteristic of the lower marsh.

3.3.2 Exclusion experiment

Changes in height, species richness and total vegetation cover are summarised in Table 3.1. At Ballyteigue in 2017, the mean vegetation height of all plots had increased, but the magnitude of the increase in control plots was significantly less than in treatments A and B. Significant differences following a similar pattern were observed in 2018. At Sheskinmore in 2017, there were no significant differences between treatments; however, in 2018 there was a small but significant difference in mean

Table 3.1. Changes in 2017 and 2018 from the 2016 baseline in height (cm), species richness and total vegetation cover (%) within the three exclosure treatments

Variable	Year	A	B	C
Ballyteigue				
Height	2017	+18.7	+20.8	+2.0
	2018	+3.2	+4.6	–8.9
Species richness	2017	–1.75	–2.00	–0.25
	2018	+0.75	+0.25	+0.25
Total vegetation cover	2017	+3.8	+3.0	+1.3
	2018	+3.5	+3.0	–5.6
Sheskinmore				
Height	2017	–0.6	–1.1	–1.8
	2018	+1.3	+1.5	–1.1
Species richness	2017	–1.00	–0.75	0.00
	2018	+0.25	0.00	+0.25
Total vegetation cover	2017	+3.0	+3.4	+2.0
	2018	+3.5	+4.2	+0.8

A, no cattle grazing but potential rabbit grazing; B, no cattle or rabbit grazing; C, cattle grazing and potential rabbit grazing (control).

change in height between control plots and treatment B, but no significant difference between either of these treatments and treatment A.

At Ballyteigue in 2017, the mean change in species richness in control plots was significantly different from that in treatments A and B. The main reason for this difference was the loss of annual species from the fenced plots. There were no significant differences in species richness changes between treatments at Ballyteigue in 2018. At Sheskinmore, there were no significant differences in species richness changes between treatments in either year.

There were no significant differences in the change in total vegetation cover at either site in 2017 or at Sheskinmore in 2018. At Ballyteigue in 2018, the mean change in total vegetation cover in control plots was significantly different from that in treatment A, but there was no significant difference between either of these treatments and treatment B.

There were no significant differences between treatments in the change in cover in either 2017 or 2018 for four of the six species analysed at Ballyteigue (Table 3.2). Change in the mean cover of *A. tripolium* was, however, significantly different between the treatments in both years. *A. tripolium* cover increased for all three treatments in 2017 and 2018. The magnitude of the increase was significantly less for treatment C than for treatment A or B; A and B did not differ significantly from each other. Similar results were found for *Pu. maritima* in 2018.

At Sheskinmore, there were no significant differences between treatments in the changes in cover in 2017 for any of the six species analysed (Table 3.3). In 2018, however, the mean changes in cover of *J. gerardii* in the two fenced treatments (slight increases) were significantly different from the control plots (slight decrease).

In the NMDS ordination, axis 1 clearly differentiated between the two sites, with all plots from Ballyteigue being lower and all plots from Sheskinmore being higher on the axis. Axis 2 was related to differences in total vegetation cover and height. At Ballyteigue, there was no clear pattern of change over time in the vegetation of the control treatment (C) plots. However, most of the treatment A and treatment B plots in 2017 and 2018 were clustered separately from the corresponding plots in 2016 and were associated with taller vegetation. At the same site, the annual plants *S. marina*, *S. maritima* and *Salicornia* spp. were associated with those plots that were higher on axis 2, which had shorter vegetation and lower total percentage cover. At Sheskinmore, there was no consistent change in vegetation for any of the treatments after 2 years.

Table 3.2. Changes in 2017 and 2018 from the 2016 baseline in percentage cover of selected species within the three exclosure treatments at Ballyteigue

Species	Year	A	B	C
Aster tripolium	2017	+33.8	+35.0	+10.0
	2018	+15.0	+11.3	+0.2
Puccinellia maritima	2017	+4.3	+2.3	+8.5
	2018	+41.8	+40.5	+11.5
Glaux maritima	2017	+0.5	–0.1	–0.8
	2018	+0.8	+0.3	+0.9
Limonium humile	2017	+5.0	+3.0	+4.5
	2018	+3.8	+2.5	+4.5
Plantago maritima	2017	–17.5	–21.3	+1.3
	2018	–17.5	–20.0	–2.5
Triglochin maritima	2017	–0.6	+0.1	+0.8
	2018	+0.4	–0.8	–1.5

A, no cattle grazing but potential rabbit grazing; B, no cattle or rabbit grazing; C, cattle grazing and potential rabbit grazing (control).

Table 3.3. Changes in 2017 and 2018 from the 2016 baseline in percentage cover of selected species within the three exclosure treatments at Sheskinmore

Species	Year	A	B	C
Festuca rubra	2017	+4.5	+4.1	+1.0
	2018	+2.2	+3.8	–0.8
Armeria maritima	2017	+7.8	+6.8	+4.3
	2018	+10.3	+9.5	+11.0
Glaux maritima	2017	–5.0	–4.4	0.0
	2018	–3.8	–3.8	+3.8
Aster tripolium	2017	+1.5	+0.7	+2.9
	2018	+1.0	+0.7	+1.7
Juncus gerardii	2017	+2.5	+2.8	–0.8
	2018	+2.0	+3.0	–1.1
Plantago maritima	2017	–1.3	0.0	–2.5
	2018	+2.5	+2.5	–2.5

A, no cattle grazing but potential rabbit grazing; B. no cattle or rabbit grazing; C, cattle grazing and potential rabbit grazing (control).

3.4 Discussion

3.4.1 General observations

Changes in vegetation following the exclusion of cattle occurred rapidly at Ballyteigue (Figure 3.1) but not at Sheskinmore. The vegetation at the two sites differs, but it was *Pl. maritima*-dominated at both sites prior to the experiment. The sandier, less fertile soil and the more exposed locality of the saltmarsh at Sheskinmore may be limiting the response to treatment (Greatorex, 2010).

After 2 years of the exclusion experiment, little evidence was found for a significant impact of rabbit (or hare) grazing at either site. Rabbit grazing on saltmarshes is likely to be found at low intensity, as rabbits tend to graze more intensely close to their burrows (Bakker *et al.*, 2005) and these are unlikely to be within the intertidal zone.

The observational survey was limited to situations in which gross livestock grazing impacts were apparent, and as a result the grazed treatment in this survey probably tended to be of heavy intensity. However, livestock grazing intensity was not quantified. Other factors could also have had an influence, e.g. frequency of grazing or the presence of geese.

3.4.2 Vegetation impacts

Increases in the cover of *A. tripolium* and an associated increase in vegetation height were obvious changes within the fences at Ballyteigue, after just 1 year of exclusion. *A. tripolium* is an important plant for pollinators, thus heavy grazing could have major impacts on local populations of flower-visiting species.

A. portulacoides is also known to be a grazing-sensitive species. The observational survey found a significantly higher abundance of this species in ungrazed areas. Grazing sensitivity is likely to be a significant factor in the scarcity of *A. portulacoides* along the Irish west coast, where grazing is so prevalent. Reductions in grazing pressure on west coast saltmarshes could result in significant changes in saltmarsh communities.

Grazing livestock can have positive indirect effects on annual saltmarsh species by reducing the growth of perennial competitors and maintaining open spaces for seedlings (Jensen, 1985; Kiehl *et al.*, 1996). A positive association was found between the presence of livestock and the annuals *S. maritima* and *Salicornia* spp. in both the observational survey and the exclusion experiment. The promotion of these lower marsh species within middle and upper marsh areas is an example of retrogressive succession (Bakker, 1985).

Figure 3.1. Change in vegetation at Ballyteigue after 1 year of cattle exclusion: (a) plot under treatment A (no cattle grazing but potential rabbit grazing) in July 2016; (b) the same plot in July 2017.

Evidence was found in the observational survey that grazing favoured *Pu. maritima*. However, the exclusion experiment suggested that, at least in the short term, release from grazing pressure can result in increases in *Pu. maritima*, which may recover faster than its competitors.

In Europe, reduced levels of grazing can result in the dominance of *Elytrigia* spp. in the upper marsh, with resultant declines in species diversity (Andresen *et al.*, 1990; Bockelmann and Neuhaus, 1999; Bos *et al.*, 2002; Veeneklaas *et al.*, 2011; Ford *et al.*, 2013; Lagendijk *et al.*, 2017). In Irish marshes, the dominance of *Elytrigia* spp. may be limited by embankments and upper marsh reclamation rather than grazing. Whereas *E. repens* is widespread across Ireland, *E. atherica* is largely restricted to the south and east coasts and can be locally dominant. The scarcity of this palatable species on the west coast may be influenced by the prevalence of livestock grazing in that region. The impact of undergrazing on Irish saltmarshes is likely to have been underestimated, because *Elytrigia* swards were not regarded as corresponding to an Annex I habitat by McCorry and Ryle (2009).

In the current study, reduced species richness in ungrazed vegetation is associated with the loss of annual species, which favour disturbed niches. In rank, abandoned vegetation, the dominance of competitive species and the build-up of litter are also causes (Bakker, 1985).

3.4.3 Management implications

Site-specific grazing plans are likely to be required for conserving saltmarsh sites. Providing a range of grazing intensities may benefit overall diversity, although this is likely to be practical only at larger sites. At sites such as Ballyteigue and Sheskinmore, where livestock can roam across multiple habitats, considerations are more complicated.

3.5 Conclusions

The results of this study are that livestock grazing:

- results in shorter, more diverse and more disturbed vegetation;
- has positive effects on annual species;
- has positive effects (at least in the longer term) on *Pu. maritima*;
- has negative effects on *A. tripolium* and *A. portulacoides*.

Chronic heavy grazing on the Irish west coast is therefore likely to have resulted in retrogressive succession. The responses of vegetation to grazing are, in part, site specific, and management plans should reflect this and the requirements of faunal taxa of conservation value.

Data collected by this study have predominantly been from habitat 1330. Studies in the future that examine grazing in other saltmarsh zones, such as habitat 1440, will elucidate how widely these conclusions can be applied.

4 Vegetation Richness and Rarity in Saltmarsh and Other Habitats of European Conservation Value in Ireland

4.1 Introduction

Species richness is widely used as a measure of biodiversity in quantifying the conservation value of sites or habitats and in researching ecological functioning (Mace *et al.*, 2012). However, species richness does not take into account the varying contribution that different species make to biodiversity when assessed at particular geographical scales. These contributions can be evaluated at a community level by indices that incorporate some measure of rarity.

This study applied a group of related rarity indices to vegetation samples from a range of habitats, including saltmarshes, in Ireland. The primary aims were to (1) determine if the conservation valuation of these habitats differed between measures reflecting species richness only, measures reflecting species rarity only or both; (2) determine if these conservation valuations differed when these measures were applied to data representing different major taxonomic groups; (3) determine if categorisations of habitats could be made based on patterns in these conservation values; and (4) examine the application of these indices at ecosystem, habitat and community levels.

4.2 Methods

4.2.1 Data preparation

Quantitative vegetation plot data for 23 Annex I habitats from seven ecosystems (saltmarsh, coastal shingle, coastal dunes, grassland, bog, heath and scree) were sourced from national surveys for the National Parks & Wildlife Service (NPWS) and supplemented with data from the SAMFHIRES project. Plot data from rush pasture, a non-Annex I habitat of perceived low conservation value, were also included. Only 2 m × 2 m plots were used to prevent species-area effects. To reduce the potential for pseudoreplication caused by plots in the dataset often being clustered at a site level, the dataset was stratified such that, for each 1 km × 1 km grid square (Irish National Grid), only one random plot per habitat was retained. Domin cover scores were converted to mid-range percentages. Plant records made only to the genus level were excluded, and if the sum of their cover abundance within a plot was ≥ 5% the whole plot was excluded.

4.2.2 Data analysis

Each native plant species in the dataset was first assigned a Rarity Co-efficient (R) from 1 to 10, with higher values denoting higher rarity, based on the number of 10 km × 10 km grid squares (Irish National Grid) from which it has been recorded across the island of Ireland (Preston *et al.*, 2002; Blockeel *et al.*, 2014; O. Pescott, British Bryological Society, unpublished data, 2015). Non-native species were assigned an R of zero.

Using these data, four different indices were calculated in addition to species richness (S, which, for the sake of simplicity, is also hereafter referred to as an “index”). Indices were calculated for each plot separately for vascular plants and for bryophytes.

As a measure of both richness and rarity, the sum of R values for each plot (Sum R) was calculated, where R_i is the R of species i:

$$\text{Sum } R = \sum_{i=1}^{S} R_i \tag{4.1}$$

For a closer measure of just rarity, the mean R ($\overline{R}$) was used:

$$\overline{R} = \frac{\sum_{i=1}^{S} R_i}{S} \tag{4.2}$$

To describe rarity weighted by species abundance, an abundance-weighted mean R ($\overline{R}_w$) was calculated, where A_i is the percentage cover of species i, as:

$$\overline{R}_w = \frac{\sum_{i=1}^{S} R_i A_i}{\sum_{i=1}^{S} A_i} \tag{4.3}$$

The reciprocal form (1/D) of Simpson’s index of diversity (D) was also calculated, which takes into account species richness and the relative abundance of each species, where p_i is the proportion of species i:

$$D = \sum_{i=1}^{S} p_i^2 \tag{4.4}$$

If a plot lacked bryophytes or vascular plants, it was assigned a 1/D value of zero for that component.

All statistical tests were conducted in R (R Core Team, 2018). Differences in indices among ecosystems were analysed using Kruskal–Wallis tests with post hoc multiple comparisons (package *pgirmess*; Giraudoux, 2017). The mean values for each of the five indices by habitat, calculated separately for vascular plants and bryophytes, were then ranked. Then, *k*-means clustering was used to create several partitions of the habitats based on these 10 rankings (k = 1, 2, 3 … 10) and the optimal number of clusters was selected with the Calinski–Harabasz criterion (package *vegan*; Oksanen *et al.*, 2017). To examine how these rankings at the habitat level differed between indices, Kendall's coefficient of concordance was used (package *synchrony*; Gouhier and Guichard, 2014). To examine community-level patterns, the saltmarsh plots were classified to IVC community, using the ERICA application (Perrin *et al.*, 2018).

4.3 Results

4.3.1 General

Stratification yielded a dataset of 2613 plots containing 453 vascular plant species and 266 bryophyte species. Of the native vascular plant species within the dataset, 62.6% were very common ($1 < R \leq 2$) at a national level, with a median R value of 1.27. In contrast, only 27.5% of native bryophyte species were categorised as very common, with a median R value of 3.86.

4.3.2 Ecosystem-level analysis

There were significant differences between ecosystems in the index values calculated on the vascular plant data (Figure 4.1). Grassland had a higher species richness (median S = 26 species/plot) than all other ecosystems, with saltmarsh, scree and shingle being the most species poor (medians of S = 6, 5 and 4, respectively). Notably, non-Annex I rush pasture was the second most species-rich category (median S = 14). Simpson's index of diversity (1/D) yielded a similar pattern of results (Figure 4.1). In contrast, the median value of $\overline{R}$ of 1.14 for grasslands was the lowest of all habitat groups, except rush pasture (median $\overline{R}$ = 1.05). Shingle (median $\overline{R}$ = 3.60) and saltmarsh (median $\overline{R}$ = 2.66) had the highest values, whereas dunes displayed a notably greater variation in values than other ecosystems. When the elements of richness and rarity were combined as Sum R, grassland again scored the highest (median Sum R = 30.43), but rush pasture (median Sum R = 14.67) was now statistically comparable with scree, shingle and dunes (median Sum R = 11.21, 14.34 and 16.66, respectively). Bog and heath had the lowest scores for this index (median Sum R = 10.89 and 11.27, respectively). $\overline{R}_w$ produced a pattern similar to that of its unweighted counterpart.

There were also significant differences between ecosystems in the values of indices calculated on bryophyte data. Bryophyte species richness was highest in heath, bog and scree (median S = 10 for all three ecosystems), with intermediate levels of richness in grassland (median S = 5) and low levels in rush pasture (median S = 2). Bryophytes were largely absent from saltmarsh and shingle and from many of the dune plots. Unlike the vascular plant data, the same pattern of results was obtained for the other four indices, as there are proportionately more rare bryophytes than rare vascular plants.

4.3.3 Habitat-level analysis

The *k*-means procedure partitioned habitats into three types:

- Type A – *specialist vascular plant habitats.* This consisted of all shingle and saltmarsh habitats, together with early successional dune habitats, all of which lacked bryophytes. Rankings here were low for S and 1/D, intermediate for Sum R and high for both $\overline{R}$ and $\overline{R}_w$.
- Type B – *species-rich vascular plant habitats*. This consisted of the late successional dune habitats, grassland habitats and rush pasture. Rankings based on vascular plant data were typically high for S, 1/D and Sum R, and low to intermediate for $\overline{R}$ and $\overline{R}_w$, whereas rankings based on bryophyte data were intermediate.
- Type C – *rich and rare bryophyte habitats*. This contained the bog, heath and scree habitats, which were ranked highly based on bryophyte data and lowly/intermediately based on vascular plant data.

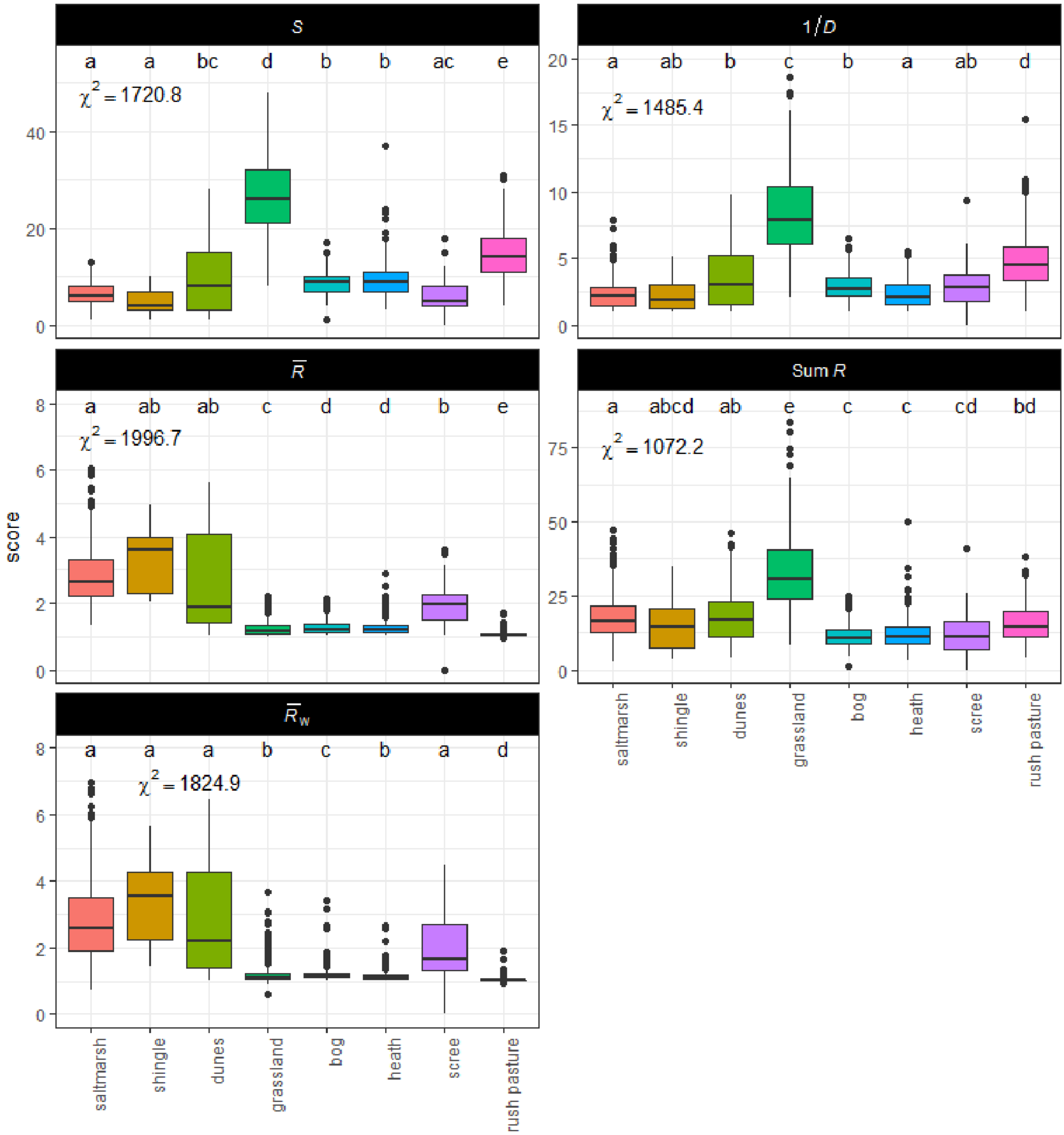

Figure 4.1. Kruskal–Wallis rank sum test analysis of five indices applied to vascular plant data from eight ecosystems. For each index, ecosystems with different lower-case letters are significantly different ($p < 0.05$), according to post hoc pairwise comparisons adjusted for multiple comparisons.

There was significant concordance between the rankings of S, $1/D$ and Sum R and between $\overline{R}$ and $\overline{R}_w$ for vascular plant data; a similar pattern was noted for bryophyte data. There was, however, almost no significant concordance between the rankings for vascular plants and for bryophytes.

4.3.4 Community-level analysis

Saltmarsh plots were classified according to 17 communities of the IVC, which belong to six groups – SM1–6; these groups represent a general progression from lower to upper marsh vegetation. There were trends that showed S increasing and $\overline{R}$ decreasing up the saltmarsh elevation gradient, resulting in the highest levels of Sum R at intermediate elevations in the SM2 group (highest median Sum $R = 29.43$ in SM2C). Abundance weighting (in indices $\overline{R}$ and $\overline{R}_w$) reduced the index scores of communities typically dominated by non-native or nationally common species but increased the scores of communities characteristically dominated by saltmarsh specialist species. Dominance of *Salicornia* spp. in SM1A, *Pu. maritima* in SM2D and *F. rubra* in SM4D drove down Simpson's diversity.

4.4 Discussion

This study demonstrates that the conservation valuation of habitats may differ depending on whether measures reflecting species richness only, species rarity only or both are used. For example, habitat 1420 Halophilous scrub, defined in Ireland by the very rare *Sarcocornia perennis*, had an intermediate rank based on vascular plant S, but it was the top-ranking habitat based on vascular plant $\overline{R}$ and ranked second based on vascular plant Sum R. Conversely, habitat 6510 Lowland hay meadows was the third most species-rich habitat but came second to last in terms of $\overline{R}$, as in Ireland these meadows are chiefly composed of generalist vascular plants.

The application of these indices at ecosystem, habitat and community levels revealed informative patterns of results, particularly across successional gradients. The dune ecosystem displayed a larger degree of variation in $\overline{R}$ and $\overline{R}_w$ than other ecosystems, as it contains both early successional habitats (a few specialist species) and late successional habitats (more generalist species). Community-level results for saltmarsh showed how abundance weighting can have a significant effect on indices. Significant variation in richness and rarity can also occur within a single Annex I habitat; this has been demonstrated by the indices for IVC groups SM2, SM3, SM4 and SM6 and community SM1B, which all occur in habitat 1330 Atlantic salt meadows.

A general lack of concordance was found between rankings based on vascular plant data and those based on bryophyte data; therefore, these taxonomic groups are not suitable as surrogates for each other, and this should be reflected in monitoring procedures. Based on the categorisation of habitat conservation value, it is proposed that assessment criteria should include an appraisal of the presence of vascular plant specialists (e.g. halophytes) for type A habitats, vascular plant species richness for type B habitats and the bryophyte flora for type C habitats. A fourth category, type D, could be surmised, comprising low conservation value habitats, with both low species richness and low species rarity.

This categorisation does not account for ecosystem services other than the provision of suitable habitats. Neither does it account for value represented by other taxa (e.g. invertebrates, birds). Such other taxa could be assessed in a similar fashion if distribution records are sufficient to provide a reasonable estimate of the true R for each species. Our rarity indices are based on the national distribution of plant species, but the approach could be applied at a variety of spatial scales (e.g. regional, national and continental) if adequate data are available.

4.5 Conclusion

This approach provides for the integration of species rarity into biodiversity assessments and could be readily applied in other countries or regions, based on localised datasets. Once R values have been derived for the relevant species, the related indices are straightforward to calculate and allow the conservation value of vegetation samples to be objectively assessed in the context of national rarity. These indices also have the advantage of being conceptually simple to interpret. Site conservation value investigations at the plot or habitat scale should incorporate an assessment of species rarity, such as $\overline{R}$ and $\overline{R}_w$, as, based on the results of this study, these may highlight aspects of conservation value other than species richness.

5 A Review of Managed and Unmanaged Realignment in Ireland with Respect to Saltmarshes

5.1 Introduction

Sea level rise poses a significant threat to the Irish coastline and lands along estuaries. In light of predicted SLR and increased storm frequency resulting from climate change (Devoy, 2008; Wang *et al.*, 2008), the saltmarsh functions of wave attenuation, shoreline stabilisation and floodwater storage (Shepard *et al.*, 2011) are becoming increasingly relevant. The loss of saltmarsh habitat and its associated protective functions can have significant financial ramifications, as narrower fronting saltmarshes need higher sea defences behind them, which are costlier to install and maintain (Doody, 2008). Managed realignment (MR), also known as managed retreat or de-embankment, is a landscape management strategy whereby previously reclaimed land along coasts, estuaries or rivers is surrendered back to natural tidal processes. MR projects may seek to offset habitat losses caused by “coastal squeeze” (Pontee, 2013).

Several Irish saltmarshes have formed through the inundation of reclaimed land, owing to the failure of levees or other defences (McCorry and Ryle, 2009). Such “unmanaged realignment” (UR) can provide an insight into how MR sites may develop. This review examines the four existing Irish MR projects (including one from Northern Ireland) and three examples of UR, to stimulate debate on the role of MR as a national coastal strategy in light of predicted SLR and storm surges.

5.2 Case Studies

5.2.1 Kilmacleague West Wetlands, County Waterford (managed realignment)

In 2005, the Court of Justice of the European Communities successfully brought a case against Ireland for general and persistent breaches of the Waste Directive (74/442/EEC, as amended by 91/156/EEC) in County Waterford (Case C 494/01). One of the complaints was the unauthorised operation of the municipal landfill on Tramore Back Strand since 1939, which adjoined and encroached on now-protected areas within the Tramore Dunes and Backstrand Special Area of Conservation. Part of Ireland’s response to this judgement was a commitment to create a compensatory wetland through MR in an area of agricultural land at Kilmacleague West, adjacent to existing areas of saltmarsh. The aim was to create 5.0 ha of mudflats, 1.0 ha of transitional saltmarsh, 0.5 ha of upper saltmarsh and 1.0 ha of pioneer marsh (B. Guest, Waterford City & County Council, personal communication, August 2018). Works to create the new levee began in May 2012, with breaching of the old levee occurring in April 2013 and the works concluding in May 2013 (B. Guest, Waterford City & County Council, personal communication, August 2018). Rock armour was used to prevent tidal erosion from broadening the breach point.

A 2015 survey by Wetland Surveys Ireland noted that an intertidal zone with sand and mud substrates had formed, with some narrow shingle zones along the shoreline. Some saltmarsh species were present in a 2- to 4-m zone of maritime vegetation on mixed sediment above the high-water mark. However, none of the habitats present was deemed to correspond to any EU HD Annex I habitats. As of August 2018, saltmarsh vegetation with distinct zonation has established itself along the shoreline of the created wetland. The lowest zone corresponds to Annex I habitat 1310 *Salicornia* mud. Isolated clumps of the invasive non-native *Spartina anglica* have established themselves here. A narrower zone of habitat 1330 Atlantic salt meadows occurs above this area. On the northern and eastern shores, a third zone occurs on the lower slope of the levee, dominated by *Elytrigia* spp., which should also be considered as habitat 1330.

5.2.2 Harper’s Island, County Cork (unmanaged realignment)

This island was formerly managed as farmland, with the northern part comprising improved grassland behind a levee. By around 2006, a small unintentional breach in the northern levee appears to have occurred, allowing saline waters to enter and saltmarsh vegetation to rapidly establish itself (T. Gittings, independent ecological consultant, personal communication, 20 January 2017). By 2014, a lagoon (~1.9 ha) had developed where particularly low ground occurred inside the levee on the northern and eastern sides of the island (O'Neill *et al.*, 2014). Adjacent to the lagoon was a large muddy area of pioneer *Salicornia* saltmarsh (~3.2 ha).

The island has been owned by Cork County Council since the 1980s but is now managed as a bird reserve in partnership with Birdwatch Ireland and the Glounthaune Community Association. As of August 2018, extensive and dense beds of *Salicornia* agg., corresponding to Annex I habitat 1310, continue to dominate the saltmarsh, with patches of *J. gerardii* and *Bolboschoenus maritimus* having also established themselves. To the rear of these beds, there is only a narrow band of other saltmarsh vegetation with abundant *A. tripolium* before the land steps up to grassland; hence, there is little natural zonation. A single clump of *S. anglica* was observed.

5.2.3 Youghal, County Cork (managed realignment)

The Youghal bypass was completed in 2003 and now forms part of the N25 national road. An environmental assessment of the project, conducted by RPS Cairns (1999), noted that crossing the Tourig Estuary would result in the loss of 0.4 ha of saltmarsh and 1.0 ha of mudflats. It was planned that 1.7 ha of compensatory intertidal habitat would be created, with a new levee to be formed in front of a tidal drain on farmland south of the existing levee, which would then be removed. It was noted that the relative proportions of mudflats and saltmarsh that would develop in this new intertidal area would be dependent on topography. The aim was for the majority of the intertidal habitat created to be saltmarsh.

These planned compensatory measures were not fully implemented, however, resulting in this example of MR failing to achieve its intended goal (T. Gittings, independent ecological consultant, personal communication, 20 January 2017). The new levee was constructed, but the older levee was left *in situ* and intact, so the intervening area of about 2.5 ha gradually flooded, forming a small brackish lake. An intentional breach in the older levee was subsequently made to rectify the situation, and a connecting channel was dug through a remnant area of saltmarsh. This allowed the lake area to drain and become intertidal. Re-profiling works to adjust the topography of the newly created intertidal zone were not implemented. As a result, the area of managed retreat was too low for any saltmarsh to develop. When visited in August 2018, the compensatory habitat was still a tidal mudflat. Any accretion of sediment within this area has not been sufficient for any saltmarsh to develop in the time since construction.

5.2.4 Ballymacoda, County Cork (unmanaged realignment)

The Womanagh River has been embanked on either side as it flows south from Crompaun Bridge into Ballymacoda Estuary. Around 2000, the levees on either side of the river were unintentionally breached during storms. According to a report by McCorry and Ryle (2009), the levee on the western side was soon repaired by the Office of Public Works (OPW) because of the possibility of flooding in Ballymacoda village. Attempts by the landowner on the eastern side of the river to repair the levee there were not successful, and an area of agricultural grassland (16.7 ha) was left open to the tide through two main breaches. The extent of inundation was limited by rising ground to the east.

In 2007, several zones of vegetation were already evident, including *Spartina* swards and Annex I habitats 1310 and 1330 (McCorry and Ryle, 2009). However, the majority of the inundated area was intertidal mudflats. When visited in August 2018, it was seen that the breaches have remained open and adjacent areas of the levee appear to have continued to erode. The inundated land can be divided into two parts: a northern section in which saltmarsh has developed and a larger southern section that is predominantly unvegetated mudflats. In the northern section, a variety of saltmarsh communities was noted, but the abundance of *S. anglica* swards was particularly noticeable.

5.2.5 Turvey, Rogerstown Estuary, County Dublin (managed realignment)

The Rogerstown Estuary to the west of a viaduct on the Dublin–Belfast railway line has been embanked on both the northern and the southern shores. Saltmarsh has developed both in front of and behind these levees because of a number of breaches. On the southern shore, most of the land is owned and managed by Fingal County Council as part of the Turvey Nature Reserve. In 2015, a 1.4-km section of a 1.5-m-high levee was intentionally removed for the purposes of habitat creation and restoration of the natural hydrology (Woodworth, 2015). No new levee was constructed to the rear of the site; instead, the rising topography of the nature reserve has been allowed to restrict flood waters. The area behind the levee already supported some saltmarsh vegetation prior to this MR project (McCorry and Ryle, 2009), as tidal waters had access through a gap in the levee to the west and probably also through a culvert in the central section of the levee through which a drain flows; at some point in the past this culvert would have had a tidal flap to prevent the ingress of saline water. The removal of the levee will, however, now allow unimpeded flooding of this area of approximately 24 ha.

5.2.6 Tramore Back Strand, County Waterford (unmanaged realignment)

Tramore Back Strand is an intertidal area divided from Tramore Bay by a large sand spit. In the 19th century, a levee around 2.5 km in length was constructed across the western end of the Back Strand, enabling the reclamation of approximately 130 ha of the intertidal area. According to J.P. Quigley (quoted from 1946 in Gault *et al.*, 2006), work started in 1853 and was completed in 1857, with a racecourse and golf links soon established on the polder. During storms in 1912, an unintentional breach occurred in the levee at the site of one of the sluices, inundating the site. Subsequent proposals for the repair of the levee were deemed uneconomical (Houses of the Oireachtas, 1952) and the breach has remained open for over a century. Tramore Intake now consists largely of mudflats and saltmarsh. A municipal landfill site was established in the intake in 1939. In 2006, there was approximately 46 ha of saltmarsh here, but of this about 24 ha was dominated or invaded by the non-native *S. anglica* (McCorry and Ryle, 2009).

5.2.7 Castle Espie, County Down (managed realignment)

Castle Espie is a Wildfowl & Wetlands Trust (WWT) reserve on the western shore of Strangford Lough that has been developed on a 19th-century industrial site. Ordnance Survey maps from 1834 show three limestone quarry pits at the site, close to the shoreline. In 1864, the land was purchased by the Murland family with the intention of reopening the quarries and a levee was constructed to reclaim land from the lough. This permitted three further quarry pits to be opened, from which clay and limestone were extracted (Hanna, n.d.). As these pits fell into disuse, they became flooded.

The site received funding for a wetland restoration project from the UK Heritage Lottery Fund in 2007, primarily for the purposes of creating an improved habitat for birds. Today, the older pits are freshwater lakes, whereas the newer pits have been developed as a saline lagoon, a freshwater lagoon and an area of saltmarsh and mudflats. Saline water enters the saltmarsh and saline lagoon through tidal flaps built into the levee, making this an example of controlled tidal regulation. To the east of the pits, a further area of saltmarsh has been created on former grassland by breaching the levee. The topography of the land behind the breach has been adjusted by creating a series of scrapes (M. Turley, WWT, personal communication, 25 January 2017).

5.3 Discussion

5.3.1 Ecological considerations

These case studies highlight some important ecological issues worthy of consideration, should MR become more popular in Ireland:

- MR/UR can result in the creation of not only saltmarsh but also mudflats and lagoons, which themselves may constitute Annex I habitats 1140 and 1150, respectively. Predicting the proportion of habitats created as a result of MR requires high-accuracy elevation surveys. Re-profiling may be needed to ensure that saltmarsh develops, and to ensure that this saltmarsh has appropriate zonation.
- Although MR projects create new areas of intertidal habitat, there may be conflicts with other ecological interests. Conflicts may arise when MR projects are within or adjacent to designated conservation areas and might necessitate a loss of habitats within these areas (Esteves and Williams, 2017).

- Saltmarsh plants are able to rapidly establish themselves within MR/UR sites with suitable elevation, provided that there are local source populations of halophytes (Wolters *et al.*, 2005). However, the non-native invasive *S. anglica* is also quick to establish itself. Swift action is therefore recommended at MR/UR sites if *S. anglica* plants are found, as this will make removal easier and less expensive. Ongoing monitoring will, however, be required.
- Long-term monitoring is needed to assess the success of MR projects relative to project-specific aims.

5.3.2 The national strategy

The three MR case studies presented from Ireland were the only three projects for which evidence could be found (and one of these failed to achieve its aims), with a total area of approximately 48.5 ha. This places Ireland behind most other countries in north-west Europe in terms of the number of MR projects carried out and significantly behind all of these countries in terms of the area of these projects (Figure 5.1). This indicates that Ireland has yet to adopt MR as a mainstream coastal engineering option.

Several European countries have published strategies for managing coastlines and estuaries that explicitly involve MR (Esteves and Williams, 2017). Ireland, however, lacks a national coastal management strategy, meaning that its approach is reactive rather than proactive. The Irish Coastal Protection Strategy Study (ICPSS) conducted by the OPW does not itself present a national strategy, as decision-making is left to local authorities, resulting in a lack of a common approach (Murphy, 2015).

Adopting MR as an approach would not be without its challenges, however, as the public can have negative perceptions of surrendering land to the sea. Engaging with the local population, educating them about the benefits of MR and increasing trust in public authorities

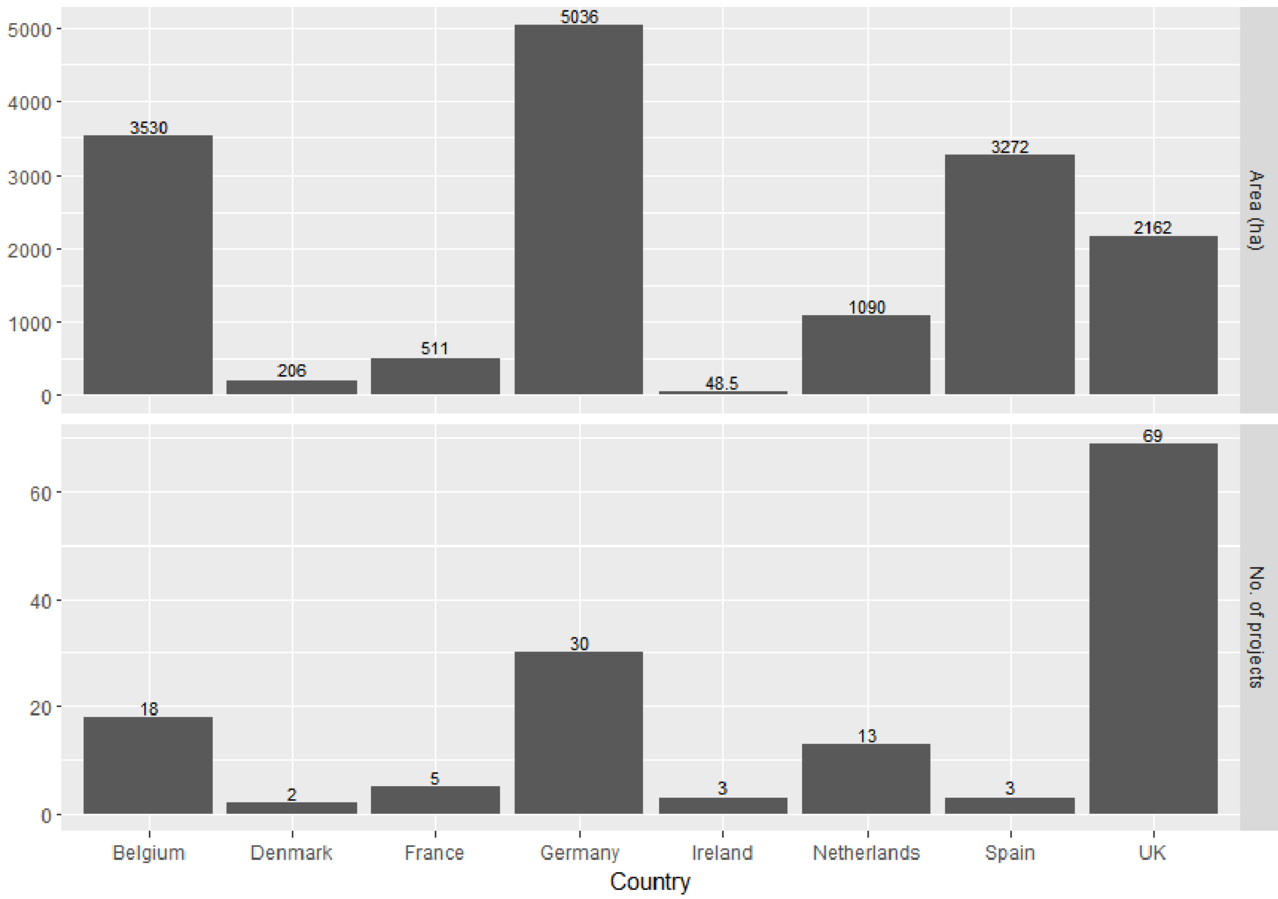

Figure 5.1. Number (bottom) and total size (top) of MR projects implemented or under construction in north-west Europe at the end of 2015. Data for all countries other than Ireland have been taken from Table 2 in Esteves and Williams (2017).

have been identified as important stages in an MR project (Myatt-Bell *et al.*, 2002; Myatt *et al.*, 2003a,b,c; Roca and Villares, 2012) and would need to form part of any national strategy. There is a wealth of relevant guidance available should Ireland develop policy in this area (Nottage and Robertson, 2005). Despite Ireland's current lack of a coastal management strategy, in light of projected climate change and SLR, MR is likely to become an increasingly attractive option for necessary adaptation in coastal areas.

6 Modelling the Effect of Sea Level Rise on Saltmarsh Extent and Functions in Irish Estuaries

6.1 Introduction

Sea level rise driven by climate change has the potential to be an over-riding threat to saltmarshes; global losses of coastal wetlands have been predicted as a consequence of SLR (Nicholls *et al.*, 1999). SLR threatens saltmarshes through increased erosion at the seaward edge and increased submergence. Ecogeomorphic feedbacks reduce the vulnerability of saltmarshes to SLR, as rates of accretion through accumulation of mineral sediments and organic matter, and thus vertical marsh growth, can increase with greater inundation (Kirwan *et al.*, 2016), although the degree of resilience that these processes provide has been debated (Kirwan *et al.*, 2017; Parkinson *et al.*, 2017). Saltmarshes may also survive SLR by migrating landward at rates faster than or comparable to those at which seaward erosion occurs (Kirwan *et al.*, 2016; Borchert *et al.*, 2018), but this progression may be impeded by steep natural topography or coastal defences, such as seawalls and levees, leading to the gradual narrowing of the saltmarsh; this is an example of the process termed “coastal squeeze” (Doody, 2004, 2013).

In areas susceptible to coastal squeeze, losses of saltmarsh habitats and the associated biodiversity and ecological functioning may be mitigated through MR (see Chapter 5). This is a coastal defence strategy whereby seawalls or levees are intentionally breached or removed, allowing previously reclaimed land to be given back to the intertidal area (e.g. Atkinson *et al.*, 2004; Midgley and McGlashan, 2004; Symonds and Collins, 2007). New defences may be constructed on the landward side of the new intertidal area, or the site may be backed by naturally rising ground.

Several countries in north-west Europe have been actively engaged in MR and saltmarsh creation for many years (Wolters *et al.*, 2005), but in Ireland there has been relatively little work conducted to date on saltmarsh creation, and there is a lack of a national strategy on coastal erosion (Murphy, 2014). It is estimated that there is about 37.7 km^2 of saltmarsh habitat in Ireland. This has been assessed as being relatively stable in extent in the recent past, resulting in no targets having been set for saltmarsh habitat creation (NPWS, 2013). The reclamation of land from the intertidal area has, however, been widespread and extensive in the past; Devoy (2008) estimated that approximately 90–150 km^2 had been reclaimed in Ireland by 1900. As a consequence, SLR, together with increased storm frequency resulting from climate change (Devoy, 2008; Wang *et al.*, 2008), poses a significant threat to the low-lying sections of the Irish coastline and land along estuaries. This threat is strongest in the south of the country where relative SLR is greater owing to glacial isostatic adjustment (GIA) (Edwards and O’Sullivan, 2007).

In this study, a reduced-complexity, spatially explicit model was used to forecast the long-term effects of different (1) SLR scenarios, (2) accretion rates and (3) MR strategies on the extent of saltmarsh within two Irish estuaries. The study also sought to examine the effects that predicted changes in habitat extent would have on ecological functioning.

6.2 Methods

6.2.1 Site selection

Two sites for which the required data were available and that had contrasting levels of GIA were selected. Great Island (GIA = –0.15 mm $year^{-1}$), which covers parts of the counties of Kilkenny, Wexford and Waterford, is located just to the east of the city of Waterford, at the confluence of the Suir and the Barrow, two of Ireland’s largest rivers. Most of the surrounding land is agricultural, but a major piece of infrastructure, the Great Island electricity power station, is also located on high ground overlooking the estuary. Rogerstown Estuary (GIA = +0.25 mm $year^{-1}$) in the north of County Dublin is located in the upper part of an estuary fed primarily by the small watercourses of the Ballyboghil River and the Ballough Stream. Areas on either side of the estuary are managed as part of nature reserves, but the site also contains agricultural land, a golf course and a former landfill site.

6.2.2 Parameters

The Marshes Governed by Tides (MARGOT) model was built in the R statistical environment (R Core Team, 2018), using functions from the *raster* package (Hijmans, 2016). Simulations were defined by a series of parameters:

- *SLR scenario* – three SLR scenarios were modelled that represent increases of 500 mm (strong mitigation scenario), 1000 mm (unmitigated scenario) and 2000 mm (worst-case scenario) between 1990 and 2100. Changes over time for these scenarios were scaled from the Intergovernmental Panel on Climate Change A1B maximum scenario, which predicted a global mean SLR of 694 mm during this period (IPCC, 2001).
- *Accretion rate scenarios* – in the absence of site-specific data on accretion rates, the approach of Tabak *et al*. (2016) was followed and three generic curves were generated that describe the feedback between saltmarsh elevation and accretion rate. These curves represent a range of plausible accretion scenarios with high (8 mm year^{-1}), medium (4 mm year^{-1}) and low (2 mm year^{-1}) maximum accretion rates. The approach of Tabak *et al*. (2016) was also followed in setting a constant tidal mudflat accretion rate of half the maximum saltmarsh accretion rate. In addition to these three scenarios, a "bathtub model" with zero accretion (Rogers *et al.*, 2012) was used to examine the importance of the accretion process in the localised survival of saltmarsh habitats.
- *MR strategies* – four managed realignment strategies were simulated. These were applied based on the categorisation of current land cover and not as a consequence of a practical assessment of the placement of current and potential defences. Strategy A represented a "hold the line" approach, whereby all areas of dryland at the start of the simulation (i.e. in 2006, see data sources below) were protected from inundation. Strategy B represented a "moderate realignment" approach, whereby semi-natural habitats and rough grazing, which were deemed to be of low economic value, were allowed to be inundated from 2006 onwards in the simulations. Strategy C represented a "significant realignment" approach, whereby losses of productive agricultural land (arable land and improved pasture) and forestry were permitted, but infrastructure, housing and amenities were protected. Finally, strategy D represented an "abandon defences" scenario with unrestrained inundation. To facilitate the application of these strategies, a land cover map for each of the study sites was digitised from satellite imagery.
- *Connectivity check* – to support intertidal habitats, areas of land not only need to be at an elevation that is within the intertidal range but must also have hydrological connectivity to saltwater (Clough *et al*., 2016). To facilitate a check for this connectivity, for each site an arbitrary mid-channel, subtidal point in the estuary was selected as a saltwater source to which intertidal habitats needed to have hydrological connectivity, defined as a continuous surface water interaction (Colón-Rivera *et al*., 2012).

6.2.3 Data sources

A digital elevation model (DEM) derived from classified ground point light detection and ranging (LiDAR) data recorded in autumn 2006 was provided by the OPW for each of the selected sites and converted to raster format. Maps of saltmarsh habitats were provided by the NPWS. These were based on field surveys in 2006 (Rogerstown Estuary) and 2007 (Great Island) (McCorry and Ryle, 2009), with some additional desktop mapping from 2009. Hence, the habitat maps were essentially contemporaneous with the LiDAR data. Contour maps were produced from the DEMs, with 0.1-m contour intervals. A visual comparison of the contour maps with the saltmarsh habitat maps and satellite imagery was used to derive elevation ranges for six habitat categories for each site: subtidal estuary, tidal mudflat, lower marsh, middle marsh, upper marsh and dryland. These habitat elevation ranges and the DEMs were then used to classify a raster format habitat map representing year zero of the model simulation (i.e. 2006), this being the same extent and resolution as the DEM. This year-zero map was then validated through a statistical comparison with the original habitat map.

6.2.4 Model functions

For each year of a simulation, inundation was modelled by calculating the relative SLR as a combination of the predicted rise in sea level and the GIA. In addition, for each year of a simulation, the accretion rate was calculated for each raster cell, based on its classification in the current habitat map. For saltmarsh habitats, this was derived from the relevant accretion rate curve and the current elevation of that cell, whereas for tidal mudflats the relevant constant was used. Accretion rates were zero for all other habitats. The DEM was then updated to reflect inundation and accretion. The habitat map was then reclassified using the updated DEM according to the habitat elevation ranges, with the exception that areas indicated as either (1) protected in the land cover map under the selected MR scenario or (2) unprotected but also without hydrological connectivity always remained dry land.

6.2.5 Model simulations

Simulations covering the period 2006–2100 were run using MARGOT for all combinations of the three SLR scenarios, the four accretion rate scenarios and the four MR strategies, both with and without the connectivity check, hence 96 simulations for each site. Each simulation started with the year-zero map after the application of strategy A and the connectivity check if applicable.

6.2.6 Ecological functioning

The consequences of forecast habitat extent changes for saltmarsh vegetation biomass and soil concentrations of organic C, total N and total P were examined, specifically the change over time as a consequence of coastal squeeze. Data on these variables from biomass and soil samples collected from 15 saltmarshes along the south and east coasts of Ireland were used (see Chapter 2). Each sampling point was classified as lower ($n = 32$), middle ($n = 160$) or upper ($n = 32$) marsh on the basis of vegetation composition. The mean value of each of the four determinands for the lower, middle and upper marsh was then extrapolated to the spatial coverage of these marsh habitats in (1) 2006 and (2) 2100 for each of the simulations with strategy A.

6.3 Results

6.3.1 Forecast changes in saltmarsh extent

Implementing the connectivity check had a proportionately small but significant effect, typically decreasing the predicted total area of saltmarsh in 2100. Therefore, results are presented only from simulations run using the connectivity check.

Overall, the forecast area of saltmarsh in 2100 was positively associated with the accretion rate and negatively associated with the degree of SLR. The bathtub simulations (with 0 mm year^{-1} maximum accretion) forecast markedly less saltmarsh than those that modelled accretion curves; for Great Island, the median difference in percentage change between the 0 mm year^{-1} simulations and the 2 mm year^{-1} simulations which represent low accretion scenarios was 18.2%, whereas for Rogerstown Estuary the median difference was 20.6%.

Under strategy A, the “hold the line” approach, the total area of saltmarsh at Great Island was forecast to decline markedly (loss of > 5%) because of coastal squeeze by six of the nine non-bathtub simulations, and marginal declines (loss of < 5%) were forecast by two of these nine simulations. Only with low SLR (500 mm) and a high accretion rate (maximum 8 mm year^{-1}) was the original extent of saltmarsh retained (change = +12.1%). Losses under the 2000-mm SLR scenario were catastrophic, ranging from –84.8% to –91.3%.

At Rogerstown Estuary, five of the nine strategy A non-bathtub simulations predicted marked losses of saltmarsh and one of the nine simulations predicted a marginal decline. Only the simulations of 500-mm SLR and 8 mm year^{-1} accretion, 500-mm SLR and 4 mm year^{-1} accretion, and 1000-mm SLR and 8 mm year^{-1} accretion yielded predicted saltmarsh increases (6.4%, 10.6% and 4.2%, respectively). Again, there were major losses under the 2000-mm SLR scenario.

The application of MR strategies B, C and D was forecast to result in progressively greater areas of total saltmarsh area. At Great Island, net gains in area resulted from five of the nine non-bathtub simulations under strategy B and all nine simulations under strategies C and D. Strategy B did not greatly increase saltmarsh area in comparison with strategy A because of the limited area zoned as semi-natural habitat or rough grazing; however, moving to strategy C greatly increased the forecast area because of the relatively large area of reclaimed land zoned as productive agriculture that would be inundated under this plan. Moving to strategy D yielded relatively minor further increases in area. At Rogerstown Estuary, net gains in saltmarsh area resulted from six of the nine non-bathtub simulations under strategy B, seven under strategy C and all nine under strategy D. At this site, moving from strategy A to strategy B yielded considerable increases in forecast area under most of the scenario combinations because of the larger proportion of land zoned as semi-natural habitat or rough grazing, with further increases on moving to strategies C and D. See Figure 6.1 for visualisations of some subjectively chosen simulations.

6.3.2 Forecast changes in the extent of individual habitats

The forecast proportions of upper, middle and lower marsh within the total saltmarsh area in 2100 under strategy A indicated a loss in the proportion of upper marsh because of coastal squeeze in all simulations at both sites. Lower marsh tended to become the dominant habitat under the 1000-mm and 2000-mm SLR scenarios. Middle marsh tended to have higher proportions under the 500-mm SLR scenario.

Tidal mudflats were forecast as the dominant habitat within the intertidal zone at Great Island under all combinations of scenarios and strategies, with the proportion of the intertidal zone comprising saltmarsh varying from 4.3% to 24.6%. At Rogerstown Estuary, the relative extent of saltmarsh within the intertidal zone varied more widely, ranging from 44.8% to 75.0% under the 500-mm SLR scenario, from 21.1% to 82.5% under the 1000-mm SLR scenario and from 4.3% to 42.4% under the 2000-mm SLR scenario.

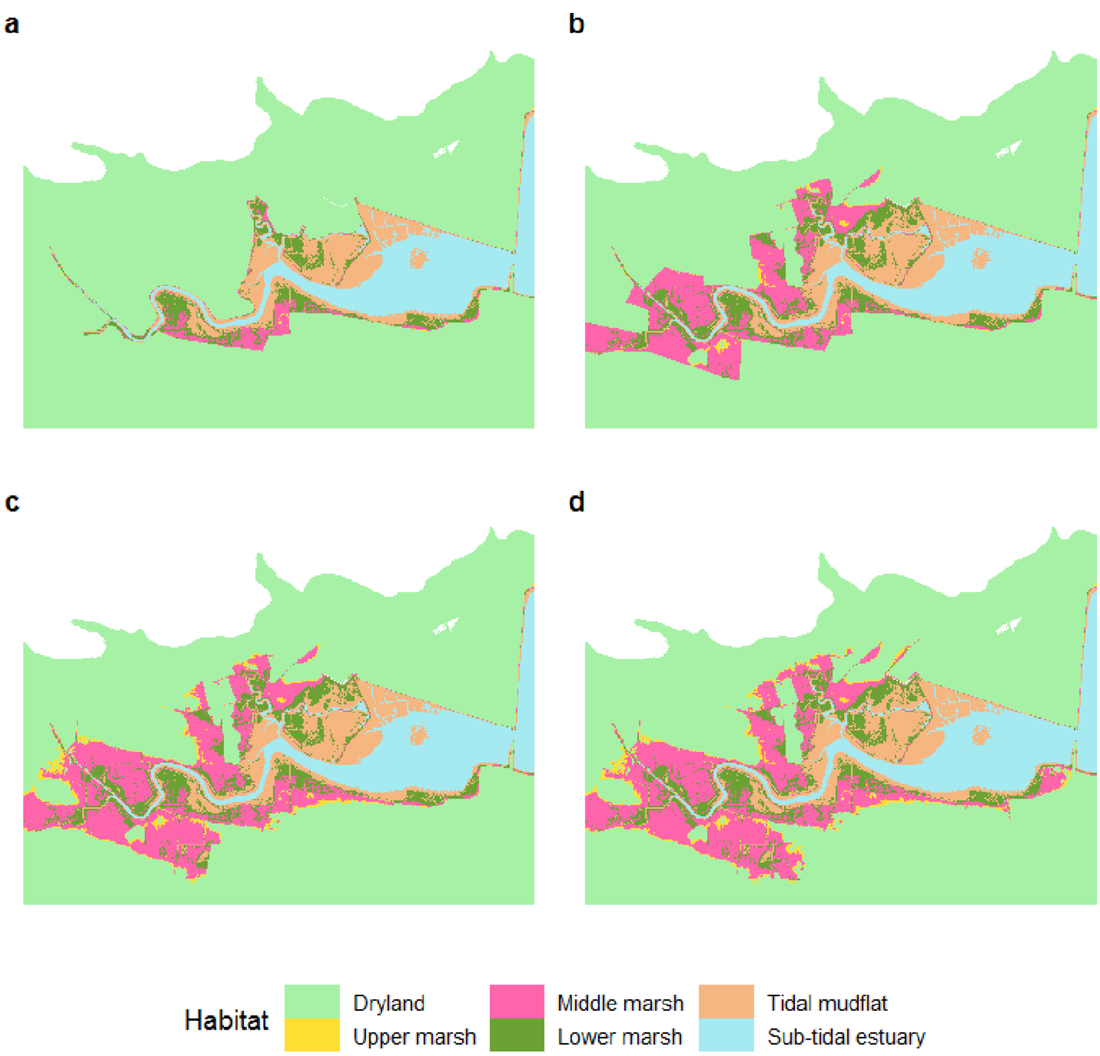

Figure 6.1. An example of the effects on forecast habitat distribution of the different MR strategies. The habitat maps are for Rogerstown Estuary in 2100 with 1000 mm SLR and 4 mm year^{-1} maximum accretion, implementing the connectivity check. Panels (a)–(d) represent strategies A–D.

6.3.3 Impacts on ecological functioning

For the simulations under strategy A where loss of saltmarsh area was forecast in 2100, forecasts for saltmarsh plant biomass tended to be more negative than those for saltmarsh area. For example, the simulation for 1000-mm SLR and 2 mm year^{-1} maximum accretion at Great Island predicted a 39.9% loss in saltmarsh area but a 56.3% loss in plant biomass. This was because of the different amounts of plant biomass being supported by the different saltmarsh habitats; upper marsh supports the highest amount of plant biomass, but the proportion of this habitat within the marsh declined as a result of coastal squeeze. Consequently, although 9 of the 12 simulations at Rogerstown Estuary predicted loss of saltmarsh area, 11 of the 12 simulations predicted loss of saltmarsh plant biomass and thus associated ecological functionality. A similar but rather more variable pattern was observed for forecasts for soil organic C and soil total N.

In contrast, for the simulations under strategy A where loss of saltmarsh area was forecast in 2100, forecasts for soil total P tended to be less negative than those for saltmarsh area. For example, the simulation for 1000-mm SLR and 2 mm year^{-1} maximum accretion at Rogerstown Estuary predicted a 64.4% loss in saltmarsh area but only a 59.0% loss in soil total P. This resource is at its greatest in the lower marsh, and the proportion of this habitat within the marsh tended to increase as a result of coastal squeeze. Therefore, although 11 of the 12 simulations at Great Island predicted loss of saltmarsh area, only 9 of the 12 simulations predicted loss of soil total P. Under the 2000-mm SLR scenario, the differences between the change in the biomass/soil variables and the change in area were minor compared with the size of the forecast losses.

6.4 Discussion

6.4.1 Influencing factors

The SLR scenario was the parameter that had the greatest influence on the simulation results, but considerable uncertainty exists in terms of the estimates of the magnitude and rate of SLR. In the face of this uncertainty concerning climate change impacts, an approach of active adaptive management has been proposed in which ecosystems are closely monitored and management strategies are altered to address changes (Lawler *et al.*, 2010). MR projects would benefit from incorporating this approach, which emphasises the selection of flexible coastal defence options that allow adjustments over time (Linquiti and Vonortas, 2012).

The model predictions also varied considerably with accretion rate. The results of the bathtub simulations (0 mm year^{-1} maximum accretion) indicate that saltmarsh models not incorporating an accretion function will consistently underestimate the resilience of saltmarshes to SLR. Site-specific data on past and current rates would assist in selecting the most appropriate scenarios. However, suspended sediment concentration is a key factor that drives accretion rates, and this is not constant over time (Temmerman *et al.*, 2004). Therefore, to reduce model uncertainty, localised predictions on change in suspended sediment concentrations are also required.

A zoning approach based on land use and perceived economic value at the start of the modelling period was used as the basis of the MR strategies. In practice, the land use and net value of land along low-lying coastland and estuaries may change in the event of significant SLR (Bernstein *et al.*, 2018). Net value equates to the value of the land itself, less the cost of installing and maintaining adequate sea defences. Higher rates of SLR amplify the financial ramifications of these defences, because narrower fronting saltmarshes provide less wave attenuation, resulting in a requirement for higher and more costly sea defences behind them (Doody, 2008). These linkages highlight the need for site-specific cost–benefit analyses of management options (Jonkman *et al.*, 2013).

6.4.2 Changes in habitat extent

At the subsiding Great Island study site, simulations following the strategy of maintaining current sea defences indicated that saltmarsh was only resilient to SLR under the single simulation of low SLR and high accretion rates. Under all other scenario combinations, it was predicted that some degree of MR will be required to prevent (potentially precipitous) declines in saltmarsh extent as a consequence of coastal squeeze. These predictions could have serious consequences for nature conservation and indicate that long-term planning for the migration of habitats must be integrated with short-term management prescriptions addressing other pressures (e.g. Adnitt *et al.*, 2013). The loss of habitat conservation value at these estuarine sites is exacerbated under scenarios in which the proportion of lower marsh increases, since this marsh zone is predominantly composed of non-native and invasive *S. anglica* swards of low conservation value (McCorry and Otte, 2001; McCorry *et al.*, 2003). Such planning should be cognisant that MR at sites that have large areas of low-elevation reclaimed land, such as Great Island, may result, at least initially, in areas dominated by tidal mudflats (Blott and Pye, 2004).

The Rogerstown Estuary site, which is uplifting, was rather more resilient to SLR, with gains in saltmarsh area occurring in three simulations with the hold-the-line approach; nevertheless, coastal squeeze still occurred under the majority of simulations under strategy A. This is one of only a few Irish sites at which MR has been conducted (see Chapter 5). Based on these simulations, it is predicted that these works will offset some of the potential habitat losses due to coastal squeeze.

6.4.3 Ecological functioning

As a result of the differences in plant biomass and soil chemistry between saltmarsh habitats, these simulations indicate that forecast proportional changes in saltmarsh area will not be directly reflected by changes in these attributes. The rates of loss of plant biomass may exceed rates of loss of saltmarsh area, and, as biomass is a key driver of the ecological functions of wave attenuation and shoreline stabilisation (Shepherd *et al.*, 2007), this may magnify the effects of habitat loss on coastal defences. Biomass is also frequently regarded as a part of "blue carbon" stocks, although it is more volatile than the component sequestered in soil organic matter (Mcleod *et al.*, 2011; Howard *et al.*, 2014). Therefore, the results for change in plant biomass in combination with those for soil organic C suggest an amplified effect of habitat loss on "blue carbon" storage. Conversely, soil total P storage capacity may be less affected by the loss of habitat area, which is positive for the role of saltmarsh as a P sink (Jiménez-Cárceles *et al.*, 2010; Freitas *et al.*, 2014).

6.5 Conclusions

This study is an example of downscaling global climate change scenarios to the local level, which is critical for understanding and mitigating the impacts of climate change at relevant spatial scales. It is the first time that a spatially explicit model, MARGOT, has been used to forecast the long-term effects of different SLR scenarios, accretion rates and MR strategies on the extent of saltmarsh in Ireland. The results of the simulations on two study sites exhibited strong site-specific variations based on topography, extant intertidal habitats and GIA. Net losses of saltmarsh habitats were predicted as a result of sequential coastal squeeze under a majority of plausible simulations that did not incorporate MR. However, the necessarily reduced complexity of the model means that these forecasts must be regarded with appropriate caution. Furthermore, there is potential for amplified effects of habitat loss on some associated ecological functions. The implementation of MR at both of the study sites could result in substantial increases in saltmarsh habitat area. Although the efficacy of MR is not without criticism (Mazik *et al.*, 2010; Esteves, 2013), these results strongly support the inclusion of this option within national coastal management strategies and site-specific saltmarsh management plans. Further research on accretion rates, suspended sediment concentrations and the processing of DEM data would improve confidence in forecasts.

7 Rare and Under-recorded Vegetation Communities of Irish Saltmarshes

7.1 Introduction

The IVC (Perrin *et al.*, 2018) currently describes 18 plant communities within Irish saltmarshes. There are some clear similarities with the British National Vegetation Classification (NVC; Rodwell, 2000), but there are also some differences. For example, a number of NVC communities with diagnostic species that occur in Ireland have no obvious counterparts in the saltmarsh division of the IVC. These differences are worthy of further investigation.

This study focused on six of these diagnostic species: *Ruppia maritima, S. maritima*, *S. perennis*, *Blysmus rufus*, *E. atherica* and *E. repens*. *J. acutus* was also included, as Stace (2010) notes that it occurs on sandy seashores and in drier parts of saltmarshes. The objectives of this study were to collect data on the vegetation assemblages within which these species occur, compare them with the current version of the IVC, assess existing IVC data and make recommendations for the amendment of the IVC if required.

7.2 Methods

7.2.1 Data collection

Sites supporting the target species were selected using the reports of the Saltmarsh Monitoring Project (SMP) (McCorry and Ryle, 2009), other literature (Cott *et al.*, 2013) or while conducting other saltmarsh fieldwork. Vegetation data from these sites were recorded during summers 2017 and 2018 in a series of 2 m × 2 m plots. Within each plot, the percentage cover in vertical projection of all vascular plant and bryophyte species was recorded. A total of 85 plots were recorded across 11 counties.

7.2.2 Data analysis

Vegetation plot data were analysed using the ERICA v4.0 application, which was developed as part of the IVC (Perrin *et al.*, 2018). Based on the analysis, a degree of membership (DOM) for each new vegetation sample to each IVC community was produced; the higher the DOM, the greater the affinity of the sample with the community.

For each vegetation plot, the community with the highest DOM ("first match") was identified. The communities with the second and third highest DOMs ("second match" and "third match", respectively) were also identified when DOMs were ≥ 5.

7.3 Results and Discussion

7.3.1 *Ruppia maritima*

Five plots with *R. maritima* were recorded, all from intertidal mudflats on Bull Island, County Dublin. The first match for all five of these plots was with the SW1A *Ruppia maritima/cirrhosa–Potamogeton pectinatus* lagoon community of the IVC; each had a DOM of 100%, indicating a very high correspondence. SW1A is typically a submerged community but one within which *R. maritima* or *R. cirrhosa* is dominant and often the only species, similar to the Bull Island mudflat vegetation. Rodwell (2000) noted in the NVC that, in addition to occurring within a submerged community of pans, creeks and brackish dykes, *R. maritima* occurs as a plant of estuarine flats where it overlaps with the NVC SM8 Annual *Salicornia* salt-marsh community. This is a similar finding to that in the current study, in which this *Ruppia* vegetation on Bull Island frequently transitioned to IVC community SM1A. *R. maritima* has also been observed in the lower intertidal zone at Inch, County Kerry, where it grades into *Zostera* beds, and in pans at multiple sites (M. Penk, Trinity College Dublin, personal communication, 7 February 2019). This analysis indicates that *R. maritima* mudflat vegetation should be included within the IVC as a component of community SW1A.

7.3.2 *Suaeda maritima*

Two plots where *S. maritima* was dominant were recorded at Ballyteigue Burrow, County Wexford, and one was recorded at Jamesbrook Hall, County Cork. In all three plots, there was a low cover of *Salicornia* agg., and in two plots there was a sparse cover of *Pu. maritima*. The first match for all plots was with the SM1A community. At Ballyteigue, this vegetation occurred near the strandline on a mixed substrate of sand and shingle. At Jamesbrook Hall, it occurred at the front of the marsh, but on a coarse substrate (10% gravel, 22% sand, 68% mud). In the NVC, the SM9 *Suaeda maritima* salt-marsh community is noted as occurring on gravelly mud in the lower marsh, at the base of shell banks, on piles of dumped sediment and with accumulations of drift litter at the foot of sea walls, with *Salicornia* agg. also being a constant taxon (Rodwell, 2000). This tallies with observations at Baltray, County Louth, where a belt of large *Suaeda* plants occurs on drift material near the sea wall (M. Penk, Trinity College Dublin, personal communication, 6 February 2019). The IVC reference dataset currently contains 16 plots in which *S. maritima* is the most abundant species. In these plots, *Salicornia* agg. and *Pu. maritima* are the most frequent associates. Based on this analysis, it is recommended that *Suaeda*-dominated vegetation is either (1) defined as a new community, SM1C, within the SM1 group or (2) defined as a sub-community within the SM1A community.

7.3.3 *Sarcocornia perennis*

A single plot with *S. perennis* – a species with a very limited distribution in Ireland – was recorded at Ballyteigue Burrow, County Wexford, bordering an area of SM1A. *S. perennis* dominated this plot, growing alongside a plentiful amount of *Limonium humile* and a sparse amount of *S. maritima*, *Pu. maritima* and *Spergularia media*. The first match was with the SM2C community. The IVC reference dataset currently contains 16 plots containing *Sarcocornia*, but its cover exceeds 50% in only one of these, and it is dominant or co-dominant in only three plots. Typically, in these plots it is subordinate to *Pu. maritima* or *Spartina*. Based on this analysis, it is recommended that vegetation in which *S. perennis* is an important constituent is either (1) treated as part of the overall SM2C community or (2) defined as a sub-community within the SM2C community.

7.3.4 *Juncus acutus*

Eight plots with *J. acutus* were recorded at three sites: Harbour View, County Cork; Buckroney, County Wicklow; and Dungarvan, County Waterford. There was considerable variation in the vegetation between these sites. At Harbour View, the first match for both of the plots was with community SM4A. At Dungarvan, results varied: for one plot the first match was with SM5A; a second plot was matched with SM2B; for a third plot the first match was with SM2A but with a low DOM (< 50%), indicating that it was somewhat transitional to the second match community of SM3B; and for the last plot of this site the first match was with GL3F *Festuca rubra–Lotus corniculatus* grassland but again with a low DOM, indicating relatively poor correspondence. At Buckroney, plots had first matches with GL2A *Agrostis stolonifera–Ranunculus repens* marsh/grassland and GL3C *Festuca rubra–Plantago lanceolata* grassland. In the NVC, *J. acutus* is noted as occurring occasionally in three dune-slack communities but not in any saltmarsh communities (Rodwell, 2000). In contrast, in Ireland it appears to be associated with a broader range of communities. Based on our analysis, we do not recommend any amendments to the saltmarsh section of the IVC to accommodate *J. acutus*-dominated vegetation, but this does not preclude the future definition of a dune-slack community characterised by this species.

7.3.5 *Blysmus rufus*

Twenty-nine plots with wide-ranging covers of *B. rufus* were recorded at several sites in the counties of Donegal, Sligo and Mayo. The best matches were with SM4B (11 plots), SM6D (10 plots), SM3A (5 plots), SM6B (2 plots) and SM4C (1 plot). SM4B and SM6D are both communities characterised by the presence of *J. gerardii*. First match DOM values were typically high, indicating that this vegetation has been accommodated reasonably well within the current IVC framework. In the NVC, the SM19 *Blysmus rufus* salt-marsh community is noted as occurring in small depressions in the upper marsh, often surrounded by vegetation of the *Juncetum gerardii*. The IVC reference dataset currently contains 49 plots containing *B. rufus*, but only in eight of these plots is it abundant (> 30% cover). Based on this analysis, it is recommended that vegetation within which *B. rufus* is an important constituent is either (1) treated as a regional variation within the several existing IVC communities or (2) defined as a sub-community within both the SM4B and SM6D communities.

7.3.6 *Elytrigia atherica and Elytrigia repens*

Plots containing *E. atherica* were recorded from several ungrazed sites, and this species tended to strongly dominate the vegetation, sometimes in combination with *E. repens*. Very low first match DOM values (<7%) were calculated for 12 of the 18 plots, which had trivial matches with communities of the freshwater habitats (FW) division. The remaining plots' first matches were with SM6B (two plots), SM5A (two plots), SM6D (one plot) and SM4C (one plot), but only one first match DOM value was >50%, indicating that these plots are poorly accommodated within the current IVC framework. In the NVC, such vegetation is represented by the SM24 *Elymus pycnanthus* salt-marsh community (*Elymus pycnanthus* = *E. atherica*), which is noted as being dominated by the titular species, often terminating the zonation at the upper limit of saltmarshes in the south and east of England.

Plots containing *E. repens* (but not *E. atherica*) were recorded at several ungrazed sites. This species again tended to strongly dominate the vegetation community. Very low first match DOM values (<8%) were calculated for 5 of the 21 plots, which had trivial matches with communities of the FW division or grasslands (GL) division. The remaining plots matched best with SM6B (five plots), SM6C (three plots), SM6D (two plots), SM6A (two plots), SM4D (two plots), SM4C (one plot) and SM4A (one plot), with nine of these plots having first match DOM values that were >50%. In the NVC, *E. repens* swards are represented by the SM28 *Elymus repens* salt-marsh community (*Elymus repens* = *E. repens*), which is described as the north-western equivalent of SM24, similarly occurring at the upper limit of saltmarsh (Rodwell, 2000). The Joint Nature Conservation Committee (JNCC) (2011) acknowledges that *E. repens* grassland occurs inland on some floodplain systems.

Based on this analysis, it is recommended that *E. atherica* swards and *E. repens* swards should form separate IVC communities within a new SM7 saltmarsh group. Inland swards of *E. repens* could be accommodated within the *E. repens* community, potentially as a separate sub-community.

7.4 A Summary of Recommended Irish Vegetation Classification Amendments

- *R. maritima* vegetation of mudflats (Figure 7.1) should be included within the SW1A *Ruppia maritima/cirrhosa–Potamogeton pectinatus* lagoon community; this should possibly be renamed as a "lagoon/mudflat community".

Figure 7.1. Clockwise from top left: vegetation dominated by *Ruppia maritima*, *Sarcocornia perennis*, *Blysmus rufus* and *Juncus acutus*.

- *S. maritima*-dominated vegetation should be either (1) defined as a new community, SM1C, within the SM1 group or (2) defined as a sub-community within the SM1A community.
- Vegetation in which *S. perennis* is an important constituent (Figure 7.1) should be (1) treated as part of the overall SM2C community or (2) defined as a sub-community within the SM2C community.
- No amendments should be made regarding vegetation with *J. acutus* (Figure 7.1).
- Vegetation in which *B. rufus* is an important constituent (Figure 7.1) should be (1) treated as a regional variation within the several existing IVC communities or (2) defined as a sub-community within the both the SM4B community and the SM6D community.
- *E. atherica* swards and *E. repens* swards should form separate IVC communities within a new SM7 group.

8 Field Survey and Ecological Assessment of Saltmarshes

8.1 Introduction

Ireland has obligations under Article 17 of the HD (92/43/EEC) to report to the EU every 6 years on the status of habitats listed in Annex I of that Directive. These include four saltmarsh habitats: 1310 *Salicornia* mud; 1330 Atlantic salt meadows; 1410 Mediterranean salt meadows and 1420 Halophilous scrub. To facilitate this reporting, field survey and assessment protocols for saltmarsh were developed and applied by McCorry and Ryle (2009) in the SMP.

Ireland also has an obligation under the WFD (2000/60/EC) to periodically assess and classify the quality of transitional and coastal waters. One of the biological quality elements to be assessed in these water bodies is "angiosperms", which includes saltmarshes. To facilitate this assessment, SMAATIE – a tool composed of several metrics – was devised by Devaney and Perrin (2015a,b). No dedicated field surveys were conducted to test the tool and the data used were not optimal for this purpose. One of the recommendations made by the SMAATIE project was to devise a single field survey to collect data for both the HD and the WFD.

This chapter reports on work conducted to satisfy this recommendation, much of which was conducted in tandem with a second national saltmarsh monitoring project (Brophy *et al.* 2019; referred to hereafter as SMP-II), co-funded by the EPA and the NPWS.

8.2 Initial Amendments

8.2.1 Mapping protocol

The SMP mapped saltmarsh predominantly on the basis of Annex I habitats, although other categories were also utilised. The main problem with this approach was that it did not adequately record zonation of communities; furthermore, estimates of habitat area were sometimes less accurate. To record zonation adequately, and following a recommendation of the SMAATIE project, it was decided that mapping would be carried out using groups and communities from the IVC (Perrin, 2016; Perrin *et al*., 2018), with some additional non-IVC categories, as indicated in Table 8.1. To allow more accurate estimates of habitat area, the proportion of each of the categories was to be recorded for every mapped polygon. No specific IVC category had an affinity with habitat 1420, so this was to be individually noted.

8.2.2 HD assessment criteria

The SMP derived eight criteria from the JNCC (2004) to assess the "Structure and Functions" (S&F) parameter for Article 17 reporting. The criteria addressed aspects of the physical and vegetation structures of the habitat as well as negative indicators (such as disturbance) and indicators of local distinctiveness. These criteria were reviewed for the current project and for SMP-II, and amended versions were developed as part of a new S&F assessment procedure, with new criteria being added to assess vegetation composition.

8.2.3 WFD tool metrics

The SMAATIE tool consists of five metrics applied at a water body level (Devaney and Perrin, 2015a). Metrics that referenced saltmarsh zones were updated to reflect the use of the new list of categories. The categories in Table 8.1 were each considered to constitute a zone, with the exception of bare ground, *Ruppia* mudflats, pans and non-saltmarsh.

Table 8.1. List of categories used to map the test sites

IVC	Non-IVC
SM1A	Swamps (including SM6A)
SM1B	*Ruppia* mudflats
SM2	Pans (part of 1330)
SM3	Bare ground (including large creeks)
SM4	*Elytrigia repens* or *E. atherica*/Driftline
SM5	Non-saltmarsh (e.g. rocks, dunes)
SM6BCD	–

8.3 Field Survey

8.3.1 Site selection

Five sites were selected to test the survey protocol: Sheskinmore, County Donegal; Dundalk Bay, County Louth; Bull Island, County Dublin; Ballyteigue Burrow, County Wexford; and Barleycove Dunes, County Cork. Sites were chiefly surveyed between May and August in 2017, with some follow-up work conducted between July and August 2018. These sites fell within eight water bodies: Ballyteige Channels and Bridgetown Estuary (associated with Ballyteigue Burrow); South Western Atlantic Seaboard and Lissagriffin Lake (associated with Barleycove Dunes); North Bull Island and Tolka Estuary (associated with Bull Island); Inner Dundalk Bay (associated with Dundalk Bay); and Owenea Estuary (associated with Sheskinmore).

For the WFD assessment, Barleycove Dunes and Ballyteigue Burrow were classified as lagoon types, Bull Island as an estuary type, Dundalk Bay as a bay type and Sheskinmore as a sandflat type.

8.3.2 Mapping

Each of the polygons mapped by the SMP was revisited and recorded using the new category system, including recording the proportion of each category for every mapped polygon. Where necessary, polygon boundaries were amended.

8.3.3 Monitoring stops

Each of the monitoring stops recorded by the SMP was relocated using a global positioning system (GPS). A 2 m × 2 m assessment plot was recorded in the Annex I habitat assessed at that stop by the SMP. At some stops, the plot had to be moved, for example to locate it within the correct Annex I habitat. Further additional plots were also recorded at new, subjectively selected monitoring stops. In all cases, the co-ordinates of where the plot was actually placed were recorded.

The following data were recorded for each plot: the Annex I habitat, the IVC community, a list of vascular plants, percentage cover of disturbed ground, percentage cover of *S. perennis* (1420 habitat only), density of annual plants per square metre (1310 habitat only), percentage cover of *Spartina* spp. within a 5-m radius of stop centre, and maximum leaf height within each quadrant of the plot (1330 habitat only). These data were used to assess the HD criteria. The list of plants was also needed for the WFD metrics.

8.4 Maps and Assessment Results

8.4.1 Mapping

A sequence of maps was generated for each site (e.g. Figure 8.1). Polygons were coded according to the dominant habitat category if this was greater than 75% or the top two categories, with the secondary category indicated in brackets, e.g. SM1A (SM1B). This meant that there was a consistent approach to displaying mosaics. The use of IVC and associated categories with this approach allowed more detailed maps to be produced. More accurate category areas were estimated using the percentage of each recorded in each polygon; this allowed sites to be compared in terms of their composition. For example:

- *Salicornia* flats (SM1A) were more abundant at Bull Island than at any other site.
- *J. maritimus* swards (SM5) were abundant only at Sheskinmore and Barleycove Dunes.
- Extensive areas of swamps were only found in the inner part of Barleycove Dunes.
- Pans were an important feature of the marsh at Bull Island.
- *E. repens* swards and driftline vegetation were most abundant at Dundalk Bay.

8.4.2 HD assessment

The S&F of habitat 1310 was assessed as “Favourable” at three of the sites but “Unfavourable – Inadequate” at two sites because of an estimated expansion of *Spartina* swards. Habitat 1330 was assessed as “Unfavourable – Inadequate” at all of the sites owing to one or more of the following: the negative effects of cattle grazing, the expansion of *Spartina* swards, inadequate zonation and the presence of a coastal embankment. Habitat 1410, in contrast, was deemed to be of “Favourable” status at all sites. Only a single stop was recorded in habitat 1420 (at Ballyteigue Burrow), so this habitat was not assessed.

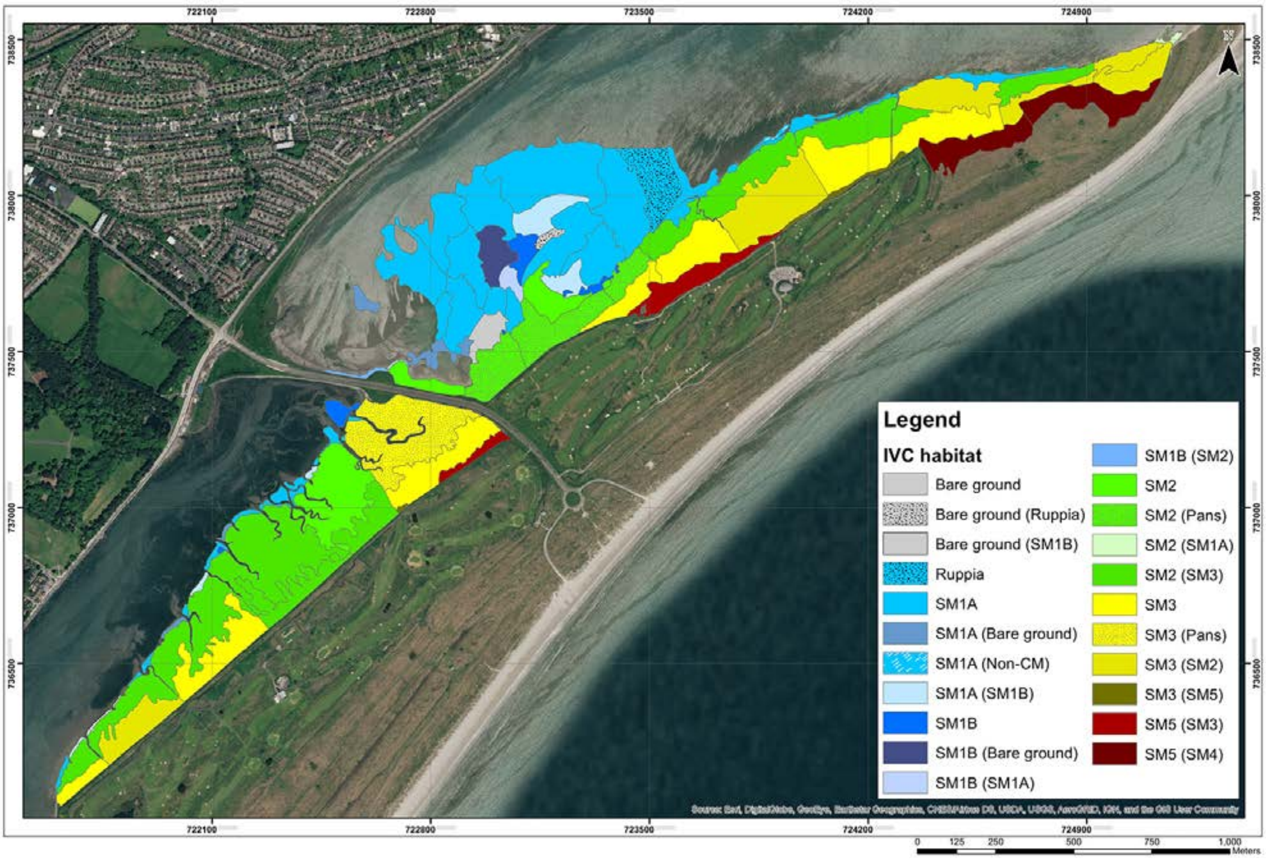

Figure 8.1. Bull Island test site mapped using the IVC and additional categories.

8.4.3 WFD assessment

The results of the WFD assessment using the amended version of the SMAATIE tool are shown in Table 8.2. Following the procedure of Devaney and Perrin (2015a), an ecological quality ratio (EQR) was produced for each of the five metrics, and a weighted average was then taken from these EQRs to calculate an overall EQR, which corresponded to an ecological status. Metrics II–V were calculated using data gathered from the test sites only, even though, within some of the water bodies assessed, other saltmarsh habitats in addition to the test sites were present but not surveyed. Metric I, which relates to the area of extant saltmarsh, was calculated at the water body level using a combination of mapping data from the test sites and previously collected mapping data for the rest of the water body.

A key objective of the WFD is that surface water bodies achieve good or high status. Ballyteige Channels and Bridgetown Estuary both failed to pass this threshold, chiefly because they scored very badly on metric I, which assesses the extent to which saltmarsh habitat has been lost by reclamation. Inner Dundalk Bay scored poorly on metric III, which relates to the balance of zonation.

According to guidance in Devaney and Perrin (2015a), water bodies with less than 400 ha of saltmarsh and in which current and potential saltmarsh area comprise less than 10% of the combined area of current saltmarsh area, potential saltmarsh area and remaining water body area should not be assessed for the purposes of the WFD in respect of saltmarsh. Therefore, according to these guidelines, the results for both Owenea Estuary and the South Western Atlantic Seaboard should not be used to inform the overall ecological status of those water bodies.

8.5 Discussion

The changes in mapping protocol greatly increased the resolution and accuracy of the recorded spatial data. The amended HD assessment criteria and WFD tool metrics detected several recognised management issues at the test sites. Some further amendments to these procedures are discussed below.

Table 8.2. Assessment of the angiosperm biological quality element of WFD water bodies corresponding to five test sites using the amended SMAATIE tool

Test site	Water body	EQR_I (3)	EQR_{II} (1)	EQR_{III} (0.5)	EQR_{IV} (0.5)	EQR_V (1)	Overall EQR	Ecological status
Ballyteigue Burrow	Ballyteige Channels	0.01	1.00	NA	1.00	0.87	0.44	Moderate
	Bridgetown Estuary	0.16	1.00	NA	0.56	0.80	0.47	Moderate
Barleycove Dunes	South Western Atlantic Seaboard	0.88	1.00	NA	1.00	0.87	0.91	High
	Lissagriffin Lake	0.52	0.67	NA	1.00	NC	0.61	Good
Bull Island	North Bull Island	0.94	0.67	0.88	0.77	0.93	0.87	High
	Tolka Estuary	0.81	0.50	0.85	0.86	0.87	0.78	Good
Dundalk Bay	Inner Dundalk Bay	0.76	0.60	0.33	0.51	0.73	0.67	Good
Sheskinmore	Owenea Estuary	1.00	0.60	0.55	1.00	0.80	0.86	High

Colours indicate status: red = bad; orange = poor; yellow = moderate; green = good; blue = high. Numbers in brackets indicate weighting applied to that metric.

EQR_I, saltmarsh area; EQR_{II}, the number of zones; EQR_{III}, zone dominance; EQR_{IV}, occurrence of *Spartina*; EQR_V, halophyte diversity; NA, not applicable for lagoons; NC, not calculated.

- *Elytrigia* swards and driftline vegetation clearly belong within the definition of habitat 1330 and form part of the saltmarsh zonation. It is recommended that the mapping category *E. repens* or *E. atherica*/Driftline be recoded once this vegetation has been incorporated into the IVC, for example as group SM7 (Chapter 7).
- It may be unreasonable to have the same vegetation height targets for small and large sites. It is therefore recommended that site-specific targets for the plant height criterion be researched and established.
- It is recommended that the presence of extensive algal mats deposited on saltmarsh be listed as a possible impact under the HD "Other negative indicators" criterion.
- The study on species richness and rarity indices (Chapter 4) demonstrated that the conservation value in these habitats is the occurrence of halophile species rather than high plant diversity. This justifies the focus of the HD vegetation composition criteria and metric V of the WFD SMAATIE tool on the presence of halophiles.
- The study on eutrophication (Chapter 2) revealed that community composition was highly significantly related to N and, to a lesser degree, P. However, the cause of this relationship is unknown. Pot experiments growing *A. portulacoides* and *Pl. maritima* in combination with different levels of fertiliser could address this issue. There are also other variables that may correlate (e.g. grazing level, biogeography, substrate type). Plant cover would be a useful tool for the biomonitoring of eutrophication, but further research is needed before such criteria can be developed.
- Currently, the reference value for metric I of the SMAATIE tool is calculated as follows:

$$Area_{ref} = Area_{current} \times (Area_{PSA} \times 0.75) \tag{8.1}$$

  Observations made in the field (Chapter 5) and on modelling MR strategies (Chapter 6) suggest that a greater downweighting of $Area_{PSA}$ may be appropriate, as significant proportions of reclaimed land may develop into mudflats rather than saltmarsh following realignment operations. For now – pending further research – it is recommended that a subjective weighting of 0.5 (rather than 0.75) be considered for future assessments.
- As it stands, the development of SM1B within a previously uninvaded water body will not result in a decline in the status of metric IV. This approach conflicts with that taken for the HD assessment. An amendment is therefore proposed such that both the upper and the lower boundaries for high status for metric IV are set at 0%, with the lower boundary of good status remaining unchanged at 15%. This will mean that the spread of *Spartina* into any water body will be flagged by a decline in metric status.

Two other points are also worthy of emphasis. First, although the HD assessments are conducted at a site level, the WFD tool is applied at the level of water bodies. Ideally, the SMAATIE metrics would use field data from all areas of relevant habitat within a water body. Therefore, it is recommended that sites selected for these dual-purpose field surveys are clustered on a water body basis. Second, it is important that swamps are adequately surveyed. Swamps are included within the broad concept of “saltmarsh” for the purposes of the WFD assessment; however, areas of swamp not associated with significant areas of saltmarsh *sensu stricto* have not been selected as field survey sites for HD assessments by either the SMP or the SMP-II. Such sites need to be included in future field surveys, and the network of monitoring stops needs to be expanded so that representative plots are recorded from areas of swamp.

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Abbreviations

AG	Above-ground
BG	Below-ground
D	Simpson's index of diversity
DEM	Digital elevation model
DIN	Dissolved inorganic nitrogen
DOM	Degree of membership
EPA	Environmental Protection Agency
EQR	Ecological quality ratio
EU	European Union
FW	Freshwater habitats
GAM	Generalised additive model
GIA	Glacial isostatic adjustment
HD	Habitats Directive
ICPSS	Irish Coastal Protection Strategy Study
IVC	Irish Vegetation Classification
JNCC	Joint Nature Conservation Committee
LiDAR	Light detection and ranging
MARGOT	Marshes Governed By Tides
MR	Managed realignment
NMDS	Non-metric multi-dimensional scaling
NPWS	National Parks & Wildlife Service
NVC	National Vegetation Classification
OPW	Office of Public Works
R	Rarity Co-efficient
RDA	Redundancy analysis
S	Species richness
S&F	Structure and Functions
SAMFHIRES	Saltmarsh Function and Human Impacts in Relation to Ecological Status
SLR	Sea level rise
SMAATIE	Saltmarsh Angiosperm Assessment Tool for Ireland
SMP	Saltmarsh Monitoring Project
UR	Unmanaged realignment
WFD	Water Framework Directive
WP	Work package
WWT	Wildfowl & Wetlands Trust

AN GHNÍOMHAIREACHT UM CHAOMHNÚ COMHSHAOIL
Tá an Ghníomhaireacht um Chaomhnú Comhshaoil (GCC) freagrach as an gcomhshaol a chaomhnú agus a fheabhsú mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaol a chosaint ó éifeachtaí díobhálacha na radaíochta agus an truaillithe.

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

Rialú: *Déanaimid córais éifeachtacha rialaithe agus comhlíonta comhshaoil a chur i bhfeidhm chun torthaí maithe comhshaoil a sholáthar agus chun díriú orthu siúd nach gcloíonn leis na córais sin.*

Eolas: *Soláthraímid sonraí, faisnéis agus measúnú comhshaoil atá ar ardchaighdeán, spriocdhírithe agus tráthúil chun bonn eolais a chur faoin gcinnteoireacht ar gach leibhéal.*

Tacaíocht: *Bímid ag saothrú i gcomhar le grúpaí eile chun tacú le comhshaol atá glan, táirgiúil agus cosanta go maith, agus le hiompar a chuirfidh le comhshaol inbhuanaithe.*

Ár bhFreagrachtaí

Ceadúnú
Déanaimid na gníomhaíochtaí seo a leanas a rialú ionas nach ndéanann siad dochar do shláinte an phobail ná don chomhshaol:
- saoráidí dramhaíola *(m.sh. láithreáin líonta talún, loisceoirí, stáisiúin aistrithe dramhaíola)*;
- gníomhaíochtaí tionsclaíocha ar scála mór *(m.sh. déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta)*;
- an diantalmhaíocht *(m.sh. muca, éanlaith)*;
- úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe *(OGM)*;
- foinsí radaíochta ianúcháin *(m.sh. trealamh x-gha agus radaiteiripe, foinsí tionsclaíocha)*;
- áiseanna móra stórála peitril;
- scardadh dramhuisce;
- gníomhaíochtaí dumpála ar farraige.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil
- Clár náisiúnta iniúchtaí agus cigireachtaí a dhéanamh gach bliain ar shaoráidí a bhfuil ceadúnas ón nGníomhaireacht acu.
- Maoirseacht a dhéanamh ar fhreagrachtaí cosanta comhshaoil na n-údarás áitiúil.
- Caighdeán an uisce óil, arna sholáthar ag soláthraithe uisce phoiblí, a mhaoirsiú.
- Obair le húdaráis áitiúla agus le gníomhaireachtaí eile chun dul i ngleic le coireanna comhshaoil trí chomhordú a dhéanamh ar líonra forfheidhmiúcháin náisiúnta, trí dhíriú ar chiontóirí, agus trí mhaoirsiú a dhéanamh ar leasúchán.
- Cur i bhfeidhm rialachán ar nós na Rialachán um Dhramhthrealamh Leictreach agus Leictreonach (DTLL), um Shrian ar Shubstaintí Guaiseacha agus na Rialachán um rialú ar shubstaintí a ídíonn an ciseal ózóin.
- An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaol.

Bainistíocht Uisce
- Monatóireacht agus tuairisciú a dhéanamh ar cháilíocht aibhneacha, lochanna, uiscí idirchriosacha agus cósta na hÉireann, agus screamhuiscí; leibhéil uisce agus sruthanna aibhneacha a thomhas.
- Comhordú náisiúnta agus maoirsiú a dhéanamh ar an gCreat-Treoir Uisce.
- Monatóireacht agus tuairisciú a dhéanamh ar Cháilíocht an Uisce Snámha.

Monatóireacht, Anailís agus Tuairisciú ar an gComhshaol
- Monatóireacht a dhéanamh ar cháilíocht an aeir agus Treoir an AE maidir le hAer Glan don Eoraip (CAFÉ) a chur chun feidhme.
- Tuairisciú neamhspleách le cabhrú le cinnteoireacht an rialtais náisiúnta agus na n-údarás áitiúil *(m.sh. tuairisciú tréimhsiúil ar staid Chomhshaol na hÉireann agus Tuarascálacha ar Tháscairí)*.

Rialú Astaíochtaí na nGás Ceaptha Teasa in Éirinn
- Fardail agus réamh-mheastacháin na hÉireann maidir le gáis cheaptha teasa a ullmhú.
- An Treoir maidir le Trádáil Astaíochtaí a chur chun feidhme i gcomhair breis agus 100 de na táirgeoirí dé-ocsaíde carbóin is mó in Éirinn.

Taighde agus Forbairt Comhshaoil
- Taighde comhshaoil a chistiú chun brúnna a shainaithint, bonn eolais a chur faoi bheartais, agus réitigh a sholáthar i réimsí na haeráide, an uisce agus na hinbhuanaitheachta.

Measúnacht Straitéiseach Timpeallachta
- Measúnacht a dhéanamh ar thionchar pleananna agus clár beartaithe ar an gcomhshaol in Éirinn *(m.sh. mórphleananna forbartha)*.

Cosaint Raideolaíoch
- Monatóireacht a dhéanamh ar leibhéil radaíochta, measúnacht a dhéanamh ar nochtadh mhuintir na hÉireann don radaíocht ianúcháin.
- Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as taismí núicléacha.
- Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta.
- Sainseirbhísí cosanta ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

Treoir, Faisnéis Inrochtana agus Oideachas
- Comhairle agus treoir a chur ar fáil d'earnáil na tionsclaíochta agus don phobal maidir le hábhair a bhaineann le caomhnú an chomhshaoil agus leis an gcosaint raideolaíoch.
- Faisnéis thráthúil ar an gcomhshaol ar a bhfuil fáil éasca a chur ar fáil chun rannpháirtíocht an phobail a spreagadh sa chinnteoireacht i ndáil leis an gcomhshaol *(m.sh. Timpeall an Tí, léarscáileanna radóin)*.
- Comhairle a chur ar fáil don Rialtas maidir le hábhair a bhaineann leis an tsábháilteacht raideolaíoch agus le cúrsaí práinnfhreagartha.
- Plean Náisiúnta Bainistíochta Dramhaíola Guaisí a fhorbairt chun dramhaíl ghuaiseach a chosc agus a bhainistiú.

Múscailt Feasachta agus Athrú Iompraíochta
- Feasacht chomhshaoil níos fearr a ghiniúint agus dul i bhfeidhm ar athrú iompraíochta dearfach trí thacú le gnóthais, le pobail agus le teaghlaigh a bheith níos éifeachtúla ar acmhainní.
- Tástáil le haghaidh radóin a chur chun cinn i dtithe agus in ionaid oibre, agus gníomhartha leasúcháin a spreagadh nuair is gá.

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil
Tá an ghníomhaíocht á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóirí. Déantar an obair ar fud cúig cinn d'Oifigí:
- An Oifig um Inmharthanacht Comhshaoil
- An Oifig Forfheidhmithe i leith cúrsaí Comhshaoil
- An Oifig um Fianaise is Measúnú
- Oifig um Chosaint Radaíochta agus Monatóireachta Comhshaoil
- An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag comhaltaí air agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair imní agus le comhairle a chur ar an mBord.

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Saltmarsh Function and Human Impacts in Relation to Ecological Status (SAMFHIRES)

Authors: Philip M. Perrin, Stephen Waldren, Marcin R. Penk and Fionnuala H. O'Neill

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Identifying pressures • Informing policy • Developing solutions

Identifying Pressures

Saltmarshes are intertidal habitats that provide a broad range of high-value ecosystem services, such as carbon storage, dissipation of wave energy and nutrient cycling, and are also recognised as important for biodiversity. However, Irish saltmarshes are subject to a number of pressures that need to be addressed if these habitats and their services are to be maintained or improved. The SAMFHIRES project (1) investigated the effects of nutrient enrichment on saltmarsh soils, plant communities and plant biomass allocation; (2) studied the effects of livestock grazing on the structure, diversity and composition of saltmarsh plant communities; (3) modelled the impacts of different sea level rise scenarios on saltmarsh extent, zonation and functions; and (4) reviewed how the effects of sea level rise could be mitigated through the managed realignment of coastal defences. In the face of these pressures, regular and informative monitoring and assessments of saltmarshes are required. This project sought to assist these processes by reviewing and amending field survey protocols, developing and applying species rarity indices, and examining the classification of less common saltmarsh communities.

Informing Policy

Weak relationships were found between nutrient conditions within saltmarsh soils and those within associated water bodies, indicating that saltmarsh soils may be poor sentinels of eutrophication in those systems. This has repercussions for assessment procedures under the Water Framework Directive and the integrated management of these coastal and transitional water bodies. Changes in above- and below-ground biomass allocation and the abundance of key saltmarsh species with increasing nutrient enrichment are likely to have implications for ecosystem services. Overall, heavy grazing on saltmarshes results in retrogressive succession. Tall, dense vegetation resulting from a lack of livestock grazing benefits some taxa but has negative effects on others; therefore, a policy to provide a range of grazing intensities on larger sites is recommended. Losses of saltmarsh habitat and consequently some functions that provide ecosystem services occurred under a majority of plausible sea level rise simulations that did not incorporate some measure of managed realignment. However, Ireland, which lacks a national coastal defence strategy, is lagging far behind many other countries in Europe in the adoption of managed realignment as a coastal management tool.

Developing Solutions

Developed by this project, MARGOT (Marshes Governed by Tides) is a reduced-complexity, spatially explicit model in R that forecasts the impacts of different sea level rise scenarios, accretion rates and managed realignment strategies on saltmarsh habitats. Based on national plant distribution data, a "Rarity Co-efficient" and related indices were developed to incorporate taxonomic rarity into conservation evaluations that are often based solely on species richness. Community zonation is a key characteristic of Irish saltmarshes but has not been accurately mapped by past surveys because of the coarseness of the available classification systems. The mapping procedures used by field surveys that inform the European Union Habitats Directive Article 17 reporting on Annex I saltmarsh habitats have therefore been amended to incorporate more detailed categories from the Irish Vegetation Classification (IVC). Recommendations have also been made on how to incorporate less common saltmarsh communities into the IVC. Assessment criteria for Annex I habitats and metrics for assessing saltmarshes for the purposes of the Water Framework Directive have both been updated to reflect our improved understanding of saltmarsh ecology.

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