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The effects of saltmarsh conservation grazing on breeding Common Redshank *Tringa totanus*

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The effects of saltmarsh conservation grazing on breeding Common
Redshank *Tringa totanus*

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Doctoral Thesis

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Summary

The Common Redshank *Tringa totanus* breeding population on British saltmarshes has reduced by > 50% since 1985, with declines linked to changes in grazing management. Conservation initiatives have encouraged low-intensity grazing of < 1 cattle ha⁻¹ but Redshank have continued to decline. This study investigated effects of grazing intensity on Redshank nest survival, and how conservation management can reduce nest mortality. This was studied by: (a) recording nest survival through monitoring of individual nests on saltmarshes subject to different livestock densities, (b) characterising vegetation at these nests, (c) quantifying spatial and temporal variation in nest trampling probability by GPS logging cattle, and (d) comparing nest survival on saltmarshes with alternative conservation management. Both nest trampling and predation risk increased with cattle density to almost 100% at 1 cattle ha⁻¹. Predation risk may be higher on grazed marshes because vegetation height at nests decreased to only 11cm at the highest grazing intensities. GPS logging showed that livestock concentrate their grazing, and therefore trampling, in the areas of most importance for breeding Redshank during the breeding season. A large scale comparison showed that grazing with adult cattle resulted in higher nest survival than mowing or no management, but that grazing with young cattle resulted in a 99% risk of trampling. These results therefore show that even < 1 cattle ha⁻¹ grazing can reduce Redshank nest survival to near zero, directly by trampling and indirectly by increasing predation through reducing vegetation height. The conservation benefits of grazing may be increased by reducing livestock densities, grazing with adult cattle, or by increasing the number of drinking troughs spread across the saltmarsh as this could move livestock away from Redshank breeding areas. Further research is needed into the potential role of rotational grazing systems that limit livestock access to Redshank habitat during the nesting season.

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1 General introduction

1.1 Introduction

Human needs have been, and continue to be satisfied at a cost to the natural world (Wackernagel & Rees 1998). This results in altered land use, climate, biogeochemical cycles and species distributions (Vitousek *et al.* 1997). Biodiversity is declining a thousand times faster now than at rates found in the fossil record (Mooney *et al.* 2005). Unless efforts to mitigate these losses are implemented, extinction rates are likely to intensify in the future (Barnosky *et al.* 2011). Humans have influenced all of the Earth's ecosystems, and much of the structure and functioning of these ecosystems cannot be understood without accounting for human modification (Vitousek *et al.* 1997). Continued threats posed by climate change (Walther *et al.* 2002; Walther, Berger & Sykes 2005) and changes in land use (Foley *et al.* 2005) put increasing pressure on natural habitats and the species that they support (Mooney *et al.* 2005). Habitat disturbance and destruction are major threats to the world's biodiversity, being one of the primary causes of species loss and a major driving force of an emerging extinction crisis (Soulé *et al.* 1986; Primack 1993; Spellerberg & Sawyer 1995). Population declines of once common birds are of major conservation concern in Europe (Newton 2004). Such declines have been widespread, with some species experiencing more than 80% reductions in former numbers and range in less than 20 years (Tucker & Heath 1994; Fuller *et al.* 1995).

Wading birds, also known as waders or shorebirds are members of the order *Charadriiformes*. There are around 210 species worldwide, the majority of which are associated with wetland or coastal environments (Hayman, Marchant & Prater 1991). Most species eat small invertebrates picked out of mud or exposed soil. Variation in bill length and shape enables a number of waders to feed in the same habitat, particularly on the coast, without direct competition for food (Lifjeld 1984). Many larger species for example Northern Lapwing *Vanellus vanellus* and Common Redshank *Tringa totanus* will take larger prey including terrestrial invertebrates and small crustaceans (Ausden *et al.* 2003). Breeding wader populations have declined severely in Europe in recent decades (Tucker & Heath 1994; Thorup 2004). Declines have been

linked to agricultural intensification and associated loss of breeding habitat (Donald, Green & Heath 2001; Vickery *et al.* 2001).

This chapter aims to introduce the reader to the needs for research into the effects of saltmarsh conservation grazing on breeding Redshank. To provide background information and context section 1.2 will cover the following topics: The importance of applied ecology, saltmarsh ecosystems, saltmarsh vegetation, saltmarsh waterbodies and tides, the Common Redshank and livestock grazing. The aims of the thesis are provided in section 1.3 and outline is provided in 1.4.

1.2 Background

1.2.1 The importance of applied ecology

Conservation management usually involves compiling management plans, which decide on appropriate actions to meet specific conservation objectives. Support for such decision-making can be poor if conservation managers do not have the opportunity to capture and evaluate the evidence for effectiveness of alternative management options (Pullin & Knight 2003; Pullin *et al.* 2004). This is likely to increase the probability that inappropriate management options will be adopted. Sutherland *et al.* (2004) provide a number of examples of how the failure to evaluate the effectiveness of conservation management can lead to the widespread implementation of ineffective treatments. For instance, winter flooding of lowland grasslands was widely considered to be beneficial for wading birds and was encouraged by Agri-environment grants. However, detailed analyses by Ausden, Sutherland and James (2001) showed that, although flooding previously un-flooded areas provided soft mud and bare soil that was suitable for foraging, it also killed the invertebrates upon which the birds fed. Further investigations suggested that a mosaic of flooded and un-flooded grasslands would be more beneficial (Ausden 2001). This demonstrates the importance of using an evidence-based approach founded on detailed data collection and analyses (Pullin *et al.* 2004; Sutherland *et al.* 2004). A thorough understanding of the ecology of a species, for example habitat use, breeding success and effectiveness of management practices is also necessary, to optimise

conservation management (Pullin & Knight 2003; Pullin *et al.* 2004; Sutherland *et al.* 2004).

1.2.2 Saltmarsh ecosystems

Saltmarshes are an important resource for wading birds and wildfowl (Charman & Macey 1978; Hughes 2004; Thyen *et al.* 2008). They act as high tide refuges for birds feeding on adjacent mudflats, as breeding sites for waders, gulls and terns and as a source of food for passerines (Greenhalgh 1971; Greenhalgh 1975; Cadbury, Green & Allport 1987; Burton *et al.* 2010). In winter, grazed saltmarshes are used as feeding grounds by large flocks of wildfowl (Charman & Macey 1978; White-Robinson 1982; Summers & Atkins 1991). Saltmarshes provide ecosystem services, for example from direct provisioning services such as harvesting of saltmarsh plants for food, animal fodder, thatch, grazing and regulating services such as nutrient filtering, carbon sequestration and coastal defence (Krutilla 1967; Costanza *et al.* 1998; Gedan, Silliman & Bertness 2009).

Saltmarshes occur in low-energy environments that allow for the accumulation of fine sediment (Adam 1990; Allen & Pye 1992). Therefore, they are usually restricted to relatively sheltered locations in five main physiographic situations: in estuaries, saline lagoons, at the heads of sea lakes, and on beach plains behind barrier islands (Allen & Pye 1992). The saltmarsh vegetation development is dependent on the presence of intertidal mudflats and other supplies of sediment (Boorman 2003). The extent of saltmarshes is restricted to between mid-tide level and high water spring tide level. Saltmarshes are subjected to dynamic long-term coastal processes such as waves, tides, sediment supply, currents, local geology and the movement of relative sea level, leading to natural cycles of erosion and accretion, which may span decades or hundreds of years (Carpenter & Pye 1996). Saltmarshes can be defined as areas, vegetated by herbs, grasses or low shrubs, bordering saline water bodies (Adam 1990). The lower limit of saltmarsh can be defined as the seaweed margin of vascular plant communities, excluding those composed of sea-grasses or other permanently submerged species. The presence of salt-tolerant plants or halophytes is considered a key defining feature of a saltmarsh (Jennings 1976).

There are some extensive saltmarshes in Europe, including the large estuaries of the UK and the Wadden Sea region of Germany, the Netherlands and Denmark (Adam 1990; Boorman 2003). The botanical features and geographic extent of the saltmarshes in Britain were described by Burd (1989), who concluded that England has the largest area of saltmarsh of the four countries that make up the UK, with an estimated total area of 32,500 hectares and 59 sites > 100ha. This is in comparison to 6748 ha in Scotland (14 sites > 100 ha), 6089 ha in Wales, and 239 ha in N. Ireland (Burd 1989).

1.2.3 Saltmarsh vegetation

Saltmarsh flora consists of a limited number of species adapted to regular immersion by the tides. In a natural saltmarsh system, these halophytic (salt tolerant) species form a clear zonation according to the frequency of inundation (Adam 1990; Boorman 2003). Typical pioneer saltmarsh plants are *Spartina* species, which colonise mudflats in circular patches; eventually these patches join to form a uniform sward (Ranwell 1964; Sanchez, SanLeon & Izco 2001). When established, rhizomes in the root structures of the *Spartina* ensure the stability of these vegetated patches, and trapping of sediment by the roots leads to the elevation of the marsh surface (Ranwell 1964; Stumpf 1983; Sanchez, SanLeon & Izco 2001). Shading by the pioneer species slows evaporation, therefore reducing soil salinity, allowing competitively superior species to colonise in the less-saline environment (Adam 1990; Shumway & Bertness 1994; Sanchez, SanLeon & Izco 2001).

Each of the zones represents a different stage of saltmarsh succession according to elevation and age, with the most mature marsh found at the highest elevations (Adam 1990; Boorman 2003). In the lowest level, the pioneer glassworts *Salicornia spp.* can withstand twice daily immersion (over 700 tides/year), while transitional species of the upper marsh can only withstand occasional inundation (Best, Massey & Prior 2007). The four recognisable zones from sea to land are known as the pioneer zone, low-mid marsh, mid-upper marsh and transition zone. Typically pioneer zones form a low, patchy cover of *Salicornia spp.*, *Suaeda maritima* and *Aster tripolium* with a bare mud

and sand surface. Low-mid marsh typically forms a continuous cover with *Puccinellia maritima* or *Atriplex portulacoides* often dominant. Mid-upper marsh typically forms a cover with *Festuca rubra*, *Limonium vulgare*, *Armeria maritima*, *Plantago maritima* and *Elymus repens* or *Elytrigia atherica*. A variety of communities may occur at the transition zone at the upper edge of the saltmarsh, where the upper edges are still present. These include mesotrophic grassland communities, tall fen community, brackish swamp communities or sand dunes (Weisbrod 1964; Adam 1990; Boorman 2003; Feagin *et al.* 2011).

Saltmarshes are naturally species poor. There are around 250 widespread species in the UK saltmarsh flora; of these only 45 species represent the halophytic environment and these 45 species represent only 12 families. The best represented are *Graminae* (10 spp.), *Chenopodiaceae* (7 spp.) and *Cyperaceae* (6 spp) (Adam 1990). Species are highly adapted to survive extreme conditions including: submersion by tide; high soil salinity; and smothering by deposition of sediment (Boorman, 2003). Plant species richness of the vegetation sward can be significantly influenced by grazing (Augustine & McNaughton 1998; Bouchard *et al.* 2003; Marty 2005). This is discussed in more detail in section 1.2.6.

1.2.4 Saltmarsh waterbodies and tides

A network of creeks (Fig. 1.1) is a key feature of most saltmarsh environments, often occupying a large part of the total marsh area (Adam 1990; Boorman 2003). These creeks tend to be cut between tidal stands by vegetation, or formed in the early development of the marsh when pioneer plant species develop around existing creeks on the mudflats (Chapman 1939; Adam 1990; Perillo & Iribarne 2003). Although considered stable, saltmarsh creeks are by no means static: creeks can continue to extend throughout the marsh, and merge with other creeks and salt pans. In extreme storm events, smaller creeks may change position completely (Perillo & Iribarne 2003). The creek network is the primary pathway for both tidal flooding and drainage. Well-drained soils can be found on the banks of creeks, whilst more waterlogged conditions are commonplace in areas furthest from the creeks (Pennings & Callaway 1992; Pennings & Bertness 2001; Temmerman *et al.* 2005). Creeks facilitate sediment and

nutrient transfer, both into and out of the marsh. Heavy and coarse sediment and materials are deposited on the banks of the creeks, whilst finer sediment is transported in suspension beyond the creeks (Stumpf 1983).

Small pools, known as salt pans, are another common feature of saltmarshes and there are numerous ways in which these pools are formed. Vegetation can form around a depression in the initial stages of saltmarsh formation, leaving a bare patch, which remains waterlogged and therefore makes it difficult for plants to colonise (Yapp, Johns & Jones 1917). Salt pans may develop on a mature marsh due to disturbance by ice or tidal litter, subsidence of the marsh surface after sub-surface drainage, or seasonal water logging of depressions in the surface of the marsh (Boston 1983). The blockage of creeks, either by vegetation growth, accumulation of sediment or tidal litter can form elongated salt pans (Yapp, Johns & Jones 1917).

Saltmarshes are regularly flooded with salt water; all vegetation is submerged at some point in time, but to varying extents. Topographical features lead to a greater than normal range of flooding and drainage regimes (Armstrong *et al.* 1985). In the high-mid marsh, tidal flooding with saltwater is rare and generally only occurs during spring tides (Adam 1990). However, climate change predictions suggest an increase in the frequency of extreme tides (Lowe *et al.* 2009). In the Wadden Sea, maximum high tide has increased twice as fast as mean high tide in recent decades, which has resulted in more frequent flooding of higher elevation sites (van de Pol *et al.* 2010). Projected increases in sea level (Hulme *et al.* 2002) could result in the eventual loss of intertidal habitats such as saltmarshes, which are vulnerable to coastal erosion (van der Wal & Pye 2004; Watkinson, Gill & Hulme 2004).

1.2.5 *The Common Redshank*

This thesis examines the effects of saltmarsh conservation grazing on breeding Redshank. The Redshank is a medium sized (27-29cm) wading bird in the Sandpiper *Scolopacidae* family. Table 1.1 summarises the range and conservation status of the Redshank. In the non-breeding season, two distinct races of Redshank occur in the UK - the nominate race *T .t. totanus*, which includes both UK breeding birds and

individuals from Scandinavia and the Icelandic race *T. t. robusta* (Hagemeijer & Blair 1997). In the field, the two races of Redshank are indistinguishable so attribution of counts to a race is problematic. Instead it is assumed that most non-breeding birds in the UK are Icelandic *T. t. robusta*, with *T. t. totanus* also occurring during passage periods (Cramp & Simmons 1983; Hagemeijer & Blair 1997). Some consider that those breeding in Britain and Ireland comprise the race *T. totanus britannica* (Engelmoer & Roselaar 1998). *T. t. totanus*, occurs throughout north-western Europe, with the UK population representing the north-western edge of this range (Cramp & Simmons 1983; Hagemeijer & Blair 1997). *T. t. robusta* breeds in Iceland and the Faeroes and moves in winter to the UK and the coasts of north-west Europe (Cramp & Simmons 1983). The remaining four sub-species occur in Asia, west of the Urals. (Hagemeijer & Blair 1997) Both *totanus* and *robusta* populations are classified as declining (Rose and Scott 1994).

Great Britain supports internationally important breeding numbers of Redshank, with over 18% of north-western Europe's estimated 100,000 – 172,000 breeding pairs (Piersma 1986; Batten *et al.* 2010). Redshank breed in various habitats, but saltmarshes hold approximately 50% of the British breeding population (Brindley *et al.* 1998), making British saltmarshes both nationally and internationally important for the species (Cadbury, Green & Allport 1987; Brindley *et al.* 1998). A lack of scientific literature on the importance of UK saltmarsh habitats for breeding birds existed until 1985 when the first national saltmarsh breeding Redshank survey was carried out (Cadbury, Green & Allport 1987). Allport, O'Brian and Cadbury (1986) found that Redshank sustained breeding populations across most of the UK, with the highest breeding densities found on the East Anglian and north-western coasts of England. On a national scale, UK saltmarsh breeding Redshank populations were surveyed in 1985 (Allport, O'Brian & Cadbury 1986; Cadbury, Green & Allport 1987), 1996 (Brindley *et al.* 1998; Norris *et al.* 1998) and 2011 (Malpas *et al.* 2013). Allport, O'Brian and Cadbury (1986) reported that Redshank were the most widespread saltmarsh breeding wader, present on 82% of the 77 sites surveyed. However, Malpas *et al.* (2013) highlighted that Redshank populations on saltmarshes declined by 52.8% between 1985 and 2011 (Table 1.2). Redshank have also declined in the wider countryside, as breeding populations declined by around 43% across all habitats between 1995 and 2013 (Balmer *et al.* 2013).

Thylen *et al.* (2008) discuss the conclusions of a workshop held in 2004, which highlighted that little is known about the factors that make saltmarshes important for wading birds. They also suggest that there are gaps in knowledge of the consequences (either at the level of the individual or population), for wading birds of selecting saltmarshes for breeding, roosting or feeding. A recommendation of the workshop was the use of integrated population monitoring schemes for the waders that use saltmarshes for breeding, wintering and passage, including systematic counts and the monitoring of breeding success (Thylen *et al.* 2008). It would appear that to date no such scheme has been developed for the UK. There was no nation-wide saltmarsh Redshank survey during the period 1996 - 2011, despite the known 23% population decline between 1985 and 1996 (Brindley *et al.* 1998; Norris *et al.* 1998). If the recommendations of (Thylen *et al.* 2008) had been put into practice, the Redshank declines that occurred between 1996 and 2010 could have been discovered sooner. This may have resulted in more targeted conservation action for Redshank through updated management guidelines for conservation management and agri-environment schemes.

Malpas *et al.* (2013) found that Redshank density declined across Britain despite the protected area status of most saltmarshes included in the study. They suggested that in their current form neither conservation management nor site designation represent an effective means of managing saltmarsh grazing levels and are therefore unlikely to provide a conservation solution for Redshank (Fig. 1.5). Malpas *et al.* (2013) recommended that conservation guidelines supplied to landowners need to be both detailed and enforced, and that future research should establish how improvements to conservation management can benefit breeding Redshank.

1.2.6 *Livestock grazing*

Livestock grazing is a common conservation management technique, used for the preservation of target species and communities or to conserve landscapes and ecosystems (WallisDeVries 1998). Historically, wild grazing animals (large herbivores) used saltmarsh habitat and evidence of this can be found dating back to the late

Pleistocene (Koch, Hoppe & Webb 1998; Levin *et al.* 2002). This suggests that saltmarshes and the wildlife they support evolved alongside grazing animals in a similar way to many terrestrial grasslands (Milchunas, Sala & Lauenroth 1988; Olf & Ritchie 1998). Farming of grazing animals on saltmarsh may date as far back as the Bronze Age (Britton, Mueldner & Bell 2008). In current times, agricultural grazing of saltmarsh habitats plays a crucial role due to losses of large wild grazing mammals (Adam 1990).

In the UK, the two most commonly used livestock types on saltmarshes are young cattle and sheep (Adam 1990; Adnitt *et al.* 2007) although horses, pigs and domestic geese are used in some areas. Wild and feral geese also graze saltmarshes (Smith & Odum 1981), however this is strongly influenced by existing grazing regimes and more likely to occur alongside more intensive livestock grazing (Mandema *et al.* 2014b). All saltmarsh grazing regimes can significantly affect the vegetation and soil characteristics of a saltmarsh and loss of vegetation height and density occurs in all cases (Jensen 1985; Andresen *et al.* 1990; Kiehl *et al.* 1996; Bouchard *et al.* 2003). However, the type of grazing animal used can affect the vegetation in differing ways (Jensen 1985). Wild and feral goose species are highly selective grazers, grazing mainly on plant roots, leaving the rest of the plant on the surface of the marsh. This can be destructive to individual plants, but also creates a patchy vegetation matrix (Smith & Odum 1981). Grazing by geese can often leave large open patches resembling salt pans, especially in stands of desirable plant species such as *Scirpus maritimus* in soft sediments (Jensen 1985). Sheep also graze selectively – choosing favourable grazing patches, plant species, individual plants or specific leaves of a plant. Sheep grazing of a saltmarsh therefore also creates a patchy vegetation matrix (Bakker, Deleeuw & Vanwieren 1984; Parsons *et al.* 1994). Preferential grazing by sheep can lead to stands of un-grazed undesirable plant species such as *Juncus* or *Atriplex*, which are taller and denser than the surrounding vegetation (Ranwell 1961; Bakker, Deleeuw & Vanwieren 1984). Cattle, in contrast graze as generalists, continually grazing as they move across the marsh regardless of vegetation type. Cattle grazing therefore results in a more uniform vegetation cover (Jensen 1985; Wallis De Vries, Laca & Demment 1999). Cattle grazing is more likely to produce longer and more heterogeneous swards suitable for some wading birds, compared with sheep grazing, which can produce close-cropped homogeneous swards of c. 3

cm (Green 1986; Beintema & Muskens 1987). Sheep graze shorter vegetation stands rather than taller vegetation, resulting in a higher grazing intensity in the shorter patches and an increased difference between patches of undesirable and desirable species. In comparison, cattle grazing would be less likely to create vegetation patchiness as vegetation height and density may be more uniform across the turf (Wallis De Vries, Laca & Demment 1999). Livestock density can have a dramatic effect on the vegetation community, as intensive grazing generally leads to a very short, uniform sward. In contrast, light grazing generally results in a more uneven, patchy sward (Jensen 1985; Kiehl *et al.* 1996).

Plant species composition of the vegetation sward can be significantly influenced by grazing. If a saltmarsh is left un-grazed, species richness can be low due to the dominance of the most competitive species. Low-intensity grazing can increase species richness by opening up the sward for less competitive species (Augustine & McNaughton 1998; Bouchard *et al.* 2003; Marty 2005). The initial plant community must be palatable to the grazing animals in use for this to occur (Schroder, Kiehl & Stock 2002). This opening up of the sward can also result in reverse succession as more resilient, earlier successional species invade open patches (Bakker 1985). Plant species richness will decrease under an intensive grazing regime, regardless of preferential grazing (Fleischner 1994; Olf & Ritchie 1998).

Grazing has the potential to alter habitat suitability by limiting or creating availability of vegetation patches on saltmarshes, which are used for Redshank nesting (Hale 1980; Mandema *et al.* 2014a). More intensive grazing leads to a very short, uniform sward and lighter grazing (Fig. 1.2) results in a more uneven patchy sward with diverse heights (Jensen 1985; Kiehl *et al.* 1996). Redshank population declines have been linked to changes in grazing management as breeding densities are higher in light and moderate grazing than on heavily grazed or un-grazed saltmarshes (Fig. 1.3) as demonstrated by Norris *et al.* (1998) and Malpas *et al.* (2013). However, Malpas *et al.* (2013) noted that populations declined by 28.8 pairs km² in northern England where grazing was more intensive, but also declined by 16.6 and 27.9 pairs km² respectively in eastern and southern England where light grazing prevailed (Table 1.2, Fig. 1.4). Nest survival is a critical component of reproductive success for many birds (Johnson, Nichols & Schwartz 1992; Hoekman *et al.* 2002). Livestock can reduce Redshank nest

survival by trampling nests (Beintema & Muskens 1987). Nest predation rates may be indirectly increased by grazing either through decreasing nest cover and vegetation heterogeneity, therefore making it easier for predators to find nests (Baines 1990; Hart *et al.* 2002), or by reducing the amount of patches of long vegetation that must be searched by a predator in order to find a nest (Chalfoun & Martin 2009). In a meta-analysis of European studies, Macdonald & Bolton (2008) found that > 50% of wader nests were predated in > 55% of the 544 examples that they reviewed. Potential Redshank nest predators on British saltmarshes include corvids *Corvus spp.*, gulls *Larus spp.*, Red Foxes *Vulpes vulpes*, Stoats *Mustela ermine* and non-native American Mink *Neovison vison* (Smart 2005).

Habitat quality is a key to an individual's fitness and organisms occupying habitats that maximise their lifetime reproductive success (e.g. by being exposed to a lower predation pressure) will contribute the most to future generations (Newton 1989; Block & Brennan 1993). Therefore, habitat quality can be defined in terms of the reproductive success attributable to a given habitat (Coulson *et al.* 2006; Johnson 2007). However, the effects of grazing on the quality of the habitat selected by nesting Redshank remain unclear. Elements of different grazing regimes employed on different saltmarshes may have a large effect on nest survival. Sheep are more likely to produce shorter vegetation swards than cattle (Green 1986; Beintema and Muskens 1987) and horses are more likely to trample nests (Mandema *et al.* 2013). In some habitats, for example agricultural meadows, young cattle have been found to trample more nests than adult cattle (Beintema & Muskens 1987) possibly due to their more lively nature (Ausden 2007)

To balance the potential conflict between the creation of a suitable vegetation sward whilst minimising grazing-induced nest mortality, densities of ~1 cattle ha⁻¹ have been recommended (Norris *et al.* 1997). This falls within the UK Environment Agency definition of light saltmarsh cattle grazing of 0.7-1 young cattle ha⁻¹, present from April to October (Adnitt *et al.* 2007). More recent evidence suggests that these light grazing levels may be too high for breeding waders. For example, Pakanen, Luukkonen & Koivula (2011) found that approximately 80% of experimental false wader nests were lost to trampling and predation in a Baltic coastal meadow with 0.83 cattle ha⁻¹. On saltmarshes, the effect may be even stronger, as Redshank typically nest in vegetation

communities associated with the high-mid marsh (Norris *et al.* 1997), which is typically closer to the landward side of the saltmarsh, and it has been suggested that grazing pressure can be higher in these areas (Patterson & Burrows 1998).

Guidelines for conservation management generally treat the distribution of grazing animals as homogenous across a landscape for example, Green (1986) and Adnitt *et al.* (2007). They tend to focus on simple calculations of numbers of livestock divided by the size of the site in hectares, taking into account the area of the saltmarsh that is accessible to livestock and the number of months that animals are present. This approach has a known constraint – that grazing across a saltmarsh can be localised closer to the sea wall (Patterson & Burrows 1998). However, it is recognised that livestock distribution can vary according to numerous biotic and abiotic factors (Bailey *et al.* 1996). These include the distance to drinking water and forage quality and quantity (Bailey 1995). Previous studies on spatial distribution of livestock have focused mainly on economically orientated pasture systems (Putfarken *et al.* 2008), therefore much of the current understanding is derived from simple model systems such as homogenous pastures of species-poor terrestrial grasslands (Rook & Tallwin 2003).

1.3 Thesis aims

In order to provide an evidence base for potential conservation interventions, this thesis aims to investigate effects of saltmarsh grazing on breeding Redshank.

- 1) To establish if light cattle grazing results in Redshank nest mortality: directly through livestock trampling and / or indirectly through grazer modification of habitat that accelerates predation risks.
- 2) To identify which nest vegetation conditions Redshank select at different spatial scales and if grazing limits the availability of higher quality nest sites.
- 3) To investigate spatial and temporal variation in livestock distribution and quantify effects on Redshank nest loss to trampling.

- 4) To quantify the effects of different conservation management practices on Redshank nest survival.

1.4 Thesis outline

This thesis consists of six chapters, of which four chapters present original data analyses (chapters 2-5), as well as this general introduction and an overall discussion (chapter 6).

The aim of chapter two is to establish if light cattle grazing results in Redshank nest mortality: directly through livestock trampling and / or indirectly through grazer modification of habitat that accelerates predation risks. Conservation initiatives have encouraged low-intensity grazing of < 1 cattle ha^{-1} but Redshank population numbers have continued to decline, even in regions where light grazing was predominant. The effects of grazing on Redshank nest survival are quantified over six lightly grazed saltmarshes with livestock densities between 0 and 0.82 cattle ha^{-1} . Cattle density was recorded both during the Redshank breeding season and for one year prior to the study, to account for both short-term trampling effects and the longer term effects on vegetation.

Chapter three aims to identify which nest vegetation conditions Redshank select at different spatial scales, and if grazing limits the availability of higher quality nest sites. This study identifies nesting habitat use as a measurement of nesting habitat selection, and how this varies with different levels of livestock density. This was studied by characterising vegetation height and composition at nests and control locations on six saltmarshes grazed between 0 and 0.55 cattle $\text{ha}^{-1} \text{y}^{-1}$ (approximately 0 - 1 cattle ha^{-1})¹ Fieldwork for chapters two and three was carried out on the Ribble estuary in north-western England between April and July 2012.

Chapter four investigates the drivers of spatial and temporal variation on livestock distribution and quantifies the effects of trampling on avian nest loss, to determine if Redshank nesting areas are exposed to higher trampling risk during the breeding season. This was carried out by quantifying spatial and temporal variation in livestock

distribution and nest trampling probability, using dummy nests and GPS loggers. Fieldwork for chapter four was carried out on five lightly grazed saltmarshes in the Wash estuary in eastern England between April and October 2013 and 2014.

As breeding Redshank populations are more stable on the saltmarshes of the Wadden sea regions of Germany, the Netherlands and Denmark than on British saltmarshes, chapter five aimed to investigate if conservation management techniques used in Wadden sea regions lead to lower rates of nest mortality than those used on British saltmarshes. This chapter combines data from six Redshank nest survival studies to estimate nest failure from a large sample size of 567 nests on nine saltmarshes subject to mowing, grazing or no active conservation management.

The thesis discussion (Chapter 6) presents an overview and overall conclusion for the work carried out in this thesis, and discusses the need for future research. Table 1.3 highlights the contributions of different authors to each of the chapters discussed above.

1.5 Tables

Table 1.1: The range and conservation status of the common Redshank *Tringa totanus*.

Global range	Wide but fragmented. Across both temperate and steppe areas of Eurasia, from the Russian Far east, as far as Iceland in the west (Cramp & Simmons 1983; Smit & Piersma 1989).
Breeding range	Breeds throughout Europe (Heath, Borggreve & Peet 2000) but most abundant in eastern Europe, Britain and Ireland, Scandinavia, Germany and the Netherlands (Smit & Piersma 1989). Smaller numbers breed in north Africa and the Middle East and a separate population breed from Kashmir, through northern India to Tibet and Tien Shan (Cramp & Simmons 1983).
Wintering range	Western: much of the European and African coastline. Eastern: around the coasts of India and south-east Asia as well as along major inland river systems. (Cramp & Simmons 1983).
Breeding habitats	The range of habitats throughout Europe includes coastal saltmarshes, wet grasslands including cultivated meadows, grassy marshes, swampy heathlands and swampy moors (Johnsgard 1981; Cramp & Simmons 1983; Smit & Piersma 1989; del Hoyo, Elliott & Sargatal 1996; Heath, Borggreve & Peet 2000). The nest site is often a shallow scrape or hollow on a hummock or at the base of a tuft of grass, often well hidden by overhanging vegetation (Hale 1980; Flint <i>et al.</i> 1984; Nethersole Thompson & Nethersole Thompson 1986; Hale 1988; del Hoyo, Elliott & Sargatal 1996; Snow <i>et al.</i> 1998). The species usually nests solitarily inland (less than 10 pairs km ⁻²) but can nest in loose colonial groups (up to 100-300 pairs/km ²) on the coast (del Hoyo, Elliott & Sargatal 1996).
Migration	Highly migratory; moving south to wintering grounds at the end of the breeding season (Cramp & Simmons 1983).
Wintering habitats	Mudflats, often where freshwater flows intersect (Burton <i>et al.</i> 2002; Burton, Rehfisch & Clark 2002).
Races/Sub-species	Six races identified (Cramp & Simmons 1983).
Conservation status	UK Amber list (Eaton <i>et al.</i> 2009). Species of European conservation concern (Heath, Borggreve & Peet 2000). Least Concern (IUCN Red List).
Known threats	Loss of habitat through agricultural intensification, wetland drainage, flood control, afforestation, land reclamation, industrial development (del Hoyo, Elliott & Sargatal 1996). Encroachment of <i>Spartina</i> spp. on mudflats (Evans 1986; del Hoyo, Elliott & Sargatal 1996). Improvement of marginal grasslands which can be, for example, by drainage, inorganic fertilising and re-seeding (Baines 1988; Baines & Dhle 1988; del Hoyo, Elliott & Sargatal 1996). Coastal barrage construction (Burton 2006), inappropriate grazing management (Norris <i>et al.</i> 1998; Malpas <i>et al.</i> 2013), disturbance on mudflats from construction work (Burton, Rehfisch & Clark 2002), foot-traffic on footpaths (Burton <i>et al.</i> 2002), and severe cold periods on western European wintering grounds (del Hoyo, Elliott & Sargatal 1996)
Diet	Breeding: consists of insects, spiders and annelid worms (del Hoyo, Elliott & Sargatal 1996). Non-breeding: polychaetes, molluscs, isopods (Sanchez, Green & Castellanos 2005).

Table 1.2: Regional and national population estimates for Redshank breeding on saltmarshes in Great Britain.

Observed mean densities and estimated numbers of breeding pairs are reported with bootstrapped 95% confidence intervals (in parentheses). Reproduced from Table 1 in Malpas *et al.* (2013) with permission from the author and the British Trust for Ornithology.

Population	Density (pairs km ⁻²) 1985	Density (pairs km ⁻²) 1996	Density (pairs km ⁻²) 2011	Saltmarsh area (ha)	Breeding pairs 1985	Breeding pairs 1996	Breeding pairs 2011
England	54.99 (42.95–66.54)	46.15 (35.50–57.71)	32.47 (23.92–43.25)	31066	17083 (13344– 20670)	14338 (11028– 17928)	10086 (7430– 13437)
<i>East Anglia</i>	66.75 (52.67–82.72)	57.61 (47.02–67.62)	50.59 (35.06–68.83)	12747	8508 (6714–10544)	7344 (5993–8619)	6449 (4468– 8773)
<i>North West</i>	55.81 (38.89–73.63)	40.30 (27.65–55.02)	26.99 (15.22–40.65)	10557	5892 (4105–7773)	4255 (2919–5808)	2850 (1607– 4291)
<i>South</i>	43.40 (20.75–69.48)	42.14 (17.32–70.27)	18.16 (8.10–30.43)	4712	2045 (978–3274)	1986 (816–3312)	856 (382– 1434)
Scotland	26.51 (16.67–36.84)	14.83 (9.07–20.74)	5.73 (1.56–10.60)	5969	1582 (995–2199)	885 (542–1238)	342 (93–633)
Wales	40.98 (21.07–69.11)	26.43 (8.26–48.43)	22.51 (9.83–34.89)	6748	2765 (1422–4664)	1784 (558–3268)	1519 (663– 2354)
Great Britain				43783	21431 (15761– 27532)	17007 (12127– 22434)	11946 (8186– 16424)

Table 1.3: The contributions of other authors to each chapter.

Chapter	Additional contributors	Notes
Chapter 1: General introduction	No additional contributors.	N/A
Chapter 2: Light grazing of saltmarshes is a direct and indirect cause of nest mortality in Common Redshank <i>Tringa totanus</i> .	Elwyn Sharps Jennifer Smart Martin Skov Angus Garbutt Jan Hiddink	This is a pre-publication version of a paper which has now been published. Elwyn Sharps is the primary author, but the other authors listed commented on drafts of this chapter.
Chapter 3: Does light grazing of saltmarshes reduce breeding habitat quality for Common Redshank <i>Tringa totanus</i> ?	Jennifer Smart Martin Skov Jan Hiddink	This is a draft of a manuscript currently being prepared to submit for publication. Elwyn Sharps is the primary author, but authors listed here have commented on early drafts.
Chapter 4: Nest trampling and ground nesting birds: can grazing density calculations assume homogenous livestock distribution on saltmarshes?	Jan Hiddink	This is an early draft of a manuscript currently being prepared to submit for publication. Only one other author (listed here) has commented on a previous draft.
Chapter 5: The impact of conservation management on nesting success of Common Redshank <i>Tringa totanus</i> in north-western Europe	Jan Hiddink	This is an early draft of a manuscript currently being prepared to submit for publication. Only one other author (listed here) has commented on a previous draft.
Chapter 6: Overall discussion	No additional contributors.	N/A

1.6 Figures

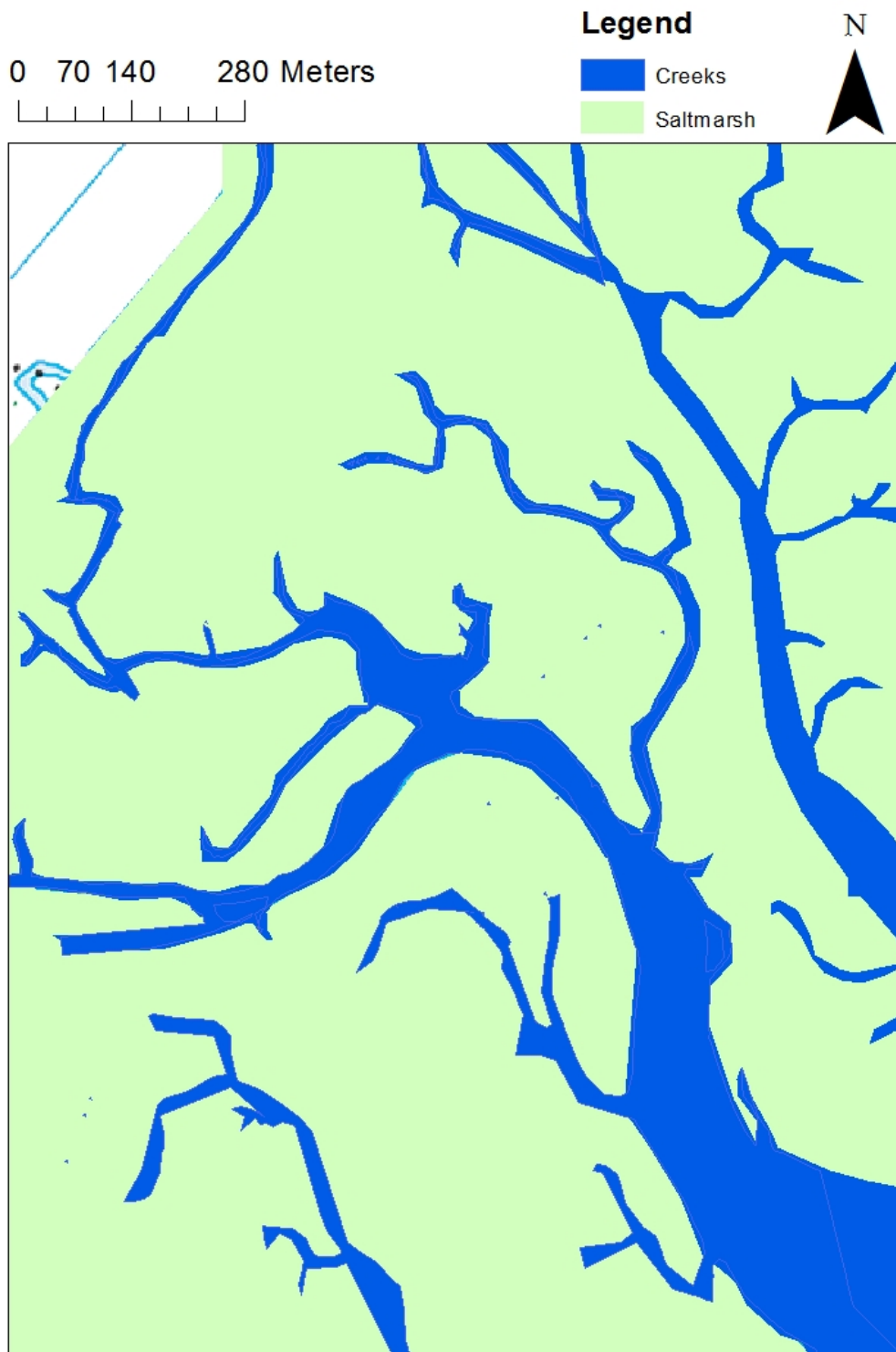


Figure 1.1: Frampton saltmarsh map illustrating creeks.

This map shows part of Frampton saltmarsh in eastern England and is provided to illustrate an example of the creek network on saltmarshes. A network of creeks is a key feature of most saltmarsh environments, often occupying a large part of the total marsh area.



Figure 1.2: The effects of light grazing on saltmarsh vegetation. The photograph on the top illustrates the difference between ungrazed (left hand side of fence) and grazed (right hand side of fence) saltmarsh vegetation. The photograph on the bottom illustrates the patchy vegetation structure produced by light grazing. Photographs are the copyright of Rachel Kingham and are reproduced with permission.

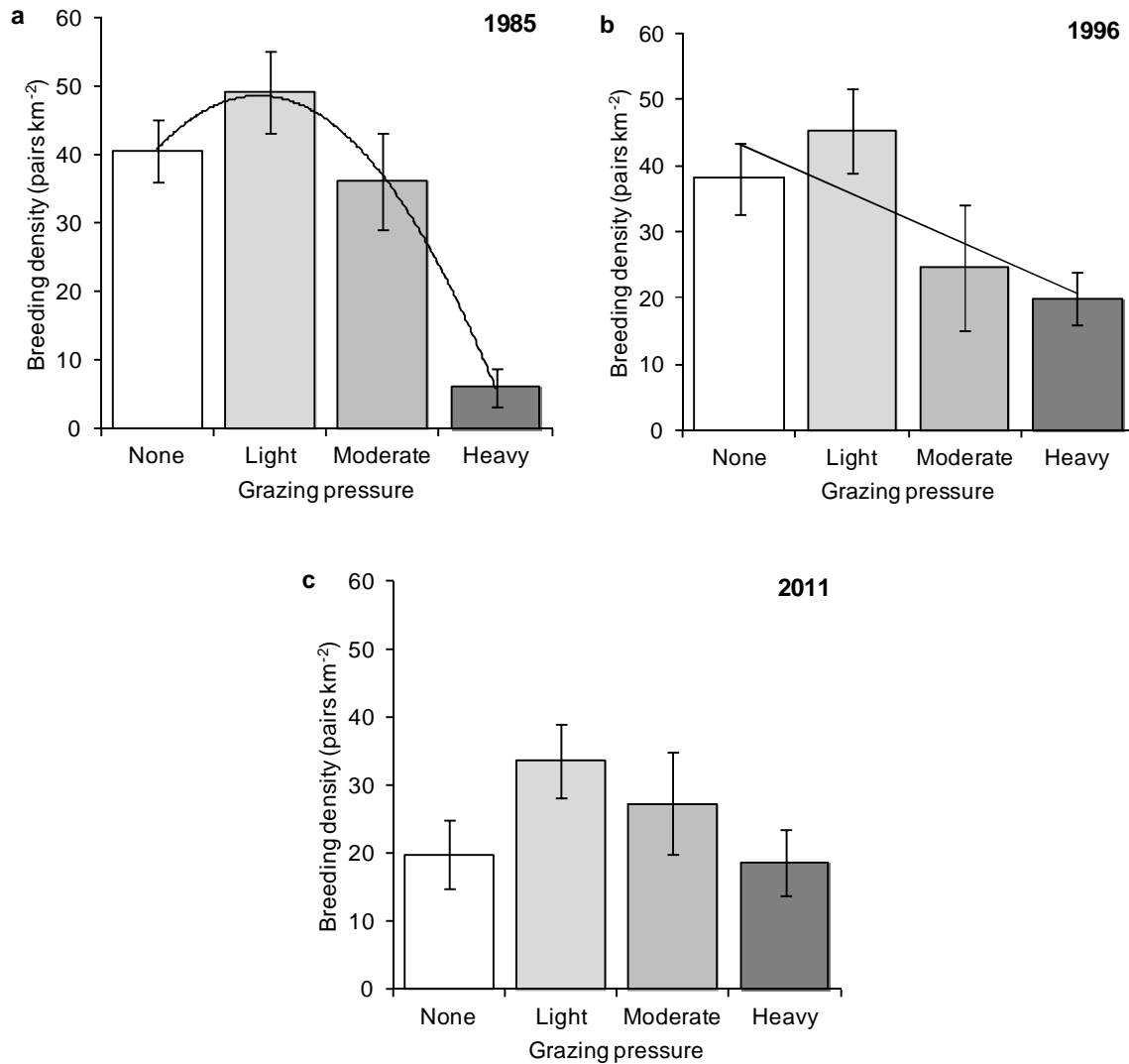


Figure 1.3: Redshank breeding density in relation to grazing. Showing mean (\pm se) Redshank breeding density (pairs km⁻²) in relation to grazing pressure across all saltmarshes surveyed in (a) 1985, (b) 1996 and (c) 2011, where both Redshank and grazing pressure data were available. Reproduced from Figure 2 in Malpas *et al.* (2013) with permission from the author and the British Trust for Ornithology. Grazing pressure was estimated subjectively in the field by Malpas *et al.* (2013) following the methodology described by Allport, O'Brian and Cadbury (1986). As the vegetation community is influenced by grazing, saltmarshes with *Halimione* sp. and *Elymus* sp. dominant or co-dominant were assumed to be lightly grazed or ungrazed. If *Puccinellia maritima*, *Festuca rubra* or *Juncus* sp. were predominant this was considered to be medium or high grazing pressure based on the vegetation height, number of livestock recorded or the presence of footprints.

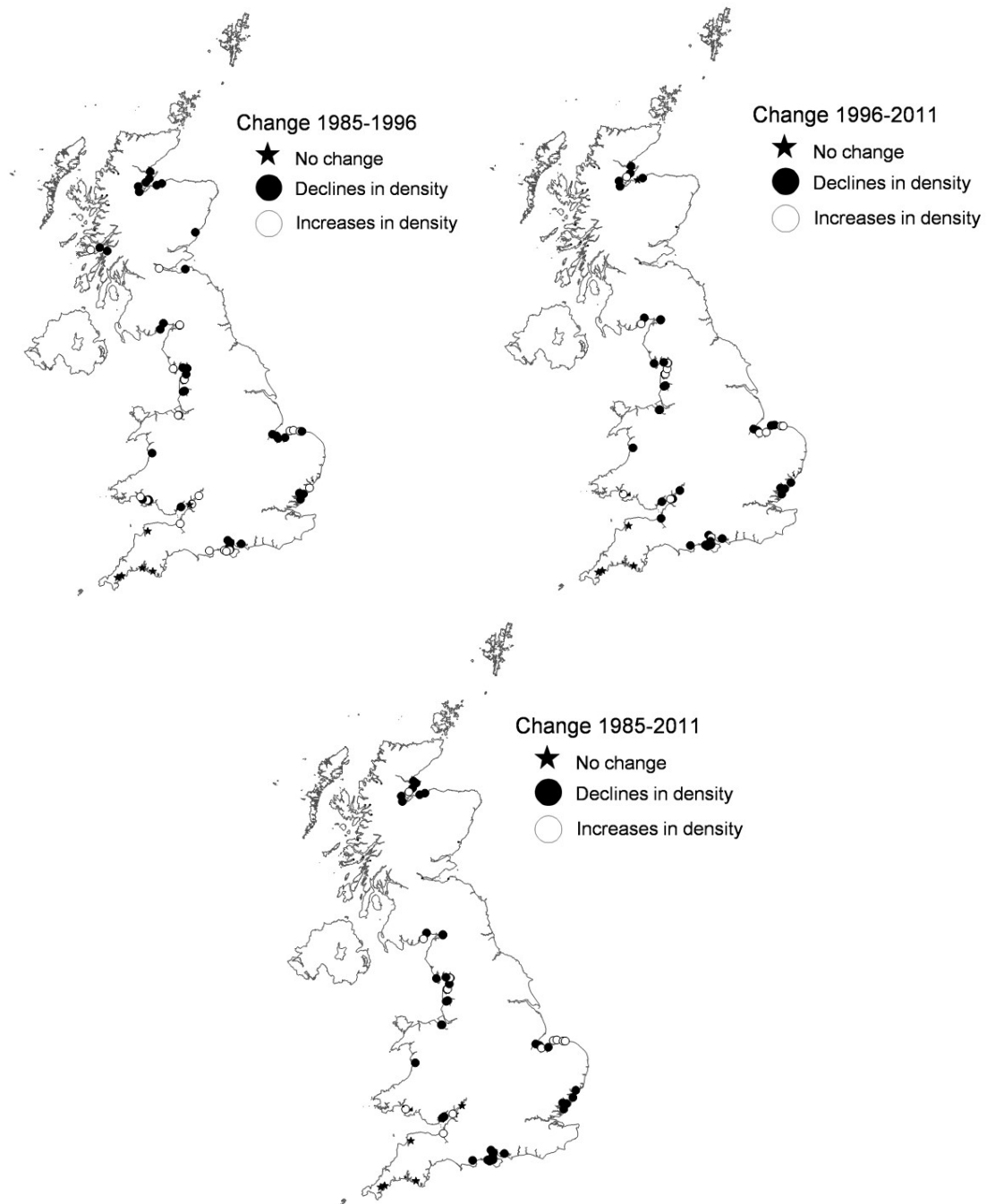


Figure 1.4: The change in Redshank breeding density at individual saltmarshes between 1985 and 2011.

Reproduced from Figure 2 in Malpas, Smart and Garbutt (2011) with permission from the author.

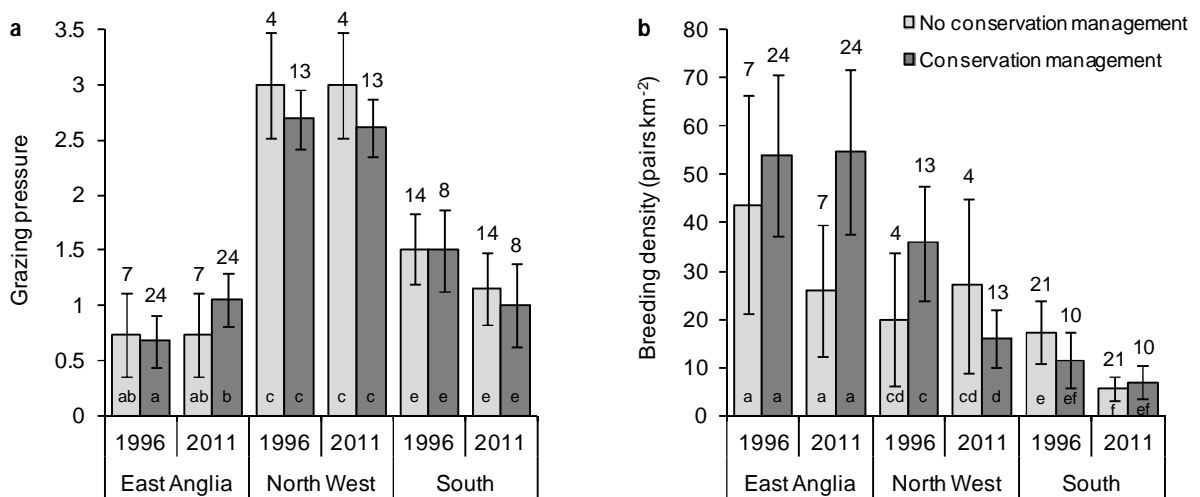


Figure 1.5: The effects of conservation management or no conservation management on grazing pressure and Redshank breeding density.

Mean±se (a) grazing pressure and (b) Redshank breeding density (pairs km⁻²) in 1996 and 2011 on plots with no conservation management and on plots which implemented conservation management after 1996 in East Anglia, the north west and the south of England. Numbers above bars indicate sample sizes (number of plots) for each factor level combination. Letters a–b, c–d and e–f show the results of pairwise comparisons between factor level combinations for East Anglia, the north west and south of England respectively, different letters indicating significant differences. Reproduced from Figure 3 in Malpas *et al.* (2013) with permission from the author and the British Trust for Ornithology. Grazing pressure was estimated in the field by Malpas *et al.* (2013) following the methodology described by Allport, O'Brian and Cadbury (1986). This assigned grazing pressure to an index where 0 = no grazing, 1 = lightly grazed, 2 = moderately grazed, and 3 = heavily grazed.

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2 Light grazing of saltmarshes is a direct and indirect cause of nest mortality in Common Redshank *Tringa totanus*.

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2.1 Abstract

The Common Redshank *Tringa totanus* breeding population on British saltmarshes has reduced by > 50% since 1985, with declines linked to changes in grazing management. Conservation initiatives have encouraged low-intensity grazing of < 1 cattle ha⁻¹ but Redshank have continued to decline, even in regions where light grazing was predominant. This study quantified effects of grazing intensity on Redshank nest survival over six lightly grazed saltmarshes with livestock densities between 0 and 0.82 cattle ha⁻¹, in the Ribble estuary, north-west England. We asked 'does grazing result in nest mortality: (a) directly through cattle trampling and/or (b) indirectly through grazer modification of habitat that accelerates predation risks'? Cattle density was recorded both during the Redshank breeding season and for one year prior to the study, to account for both short-term trampling effects and the longer term effects on vegetation. Results showed that risk of nest loss to trampling increased from 16% at 0.15 cattle ha⁻¹ to 98% at 0.82 cattle ha⁻¹ in the breeding season. The risk of a nest being predated increased from 28% with no grazing to 95% at 0.55 cattle ha⁻¹ based on all year grazing data. These results suggest that even light conservation grazing at less than one cattle ha⁻¹ can reduce Redshank nest survival rates to near zero. It may therefore be appropriate to reduce saltmarsh grazing intensities, or change the timing of saltmarsh grazing to reduce the number of livestock present during the Redshank breeding season.

2.2 Introduction

Breeding wader populations have declined severely in Europe in recent decades (Tucker & Heath 1994; Thorup 2004). Declines have been linked to agricultural intensification and associated loss of breeding habitat (Donald, Green & Heath 2001; Vickery *et al.* 2001). Great Britain supports internationally important breeding numbers of Common Redshank *Tringa totanus*, with over 18% of north-west Europe's estimated 100,000 – 172,000 breeding pairs (Piersma 1986; Batten *et al.* 2010). Redshank breed in various habitats, but saltmarshes hold approximately 50% of the British breeding population (Brindley *et al.* 1998), making British saltmarshes both nationally and internationally important for the species (Cadbury, Green & Allport 1987; Brindley *et al.* 1998). Recent surveys of British saltmarshes found a 52.8% reduction in nesting pairs between 1985 and 2011 and highlight the failure of conservation management to reverse historic Redshank declines (Malpas *et al.* 2013).

Grazing has the potential to alter habitat suitability by limiting or creating availability of vegetation patches, which are used for Redshank nesting (Hale 1980; Mandema *et al.* 2014). More intensive grazing leads to a very short, uniform sward and lighter grazing results in a more uneven patchy sward with diverse heights (Jensen 1985; Kiehl *et al.* 1996). Redshank population declines on British saltmarshes have been linked to changes in grazing management as breeding densities are higher in light and moderate grazing than on heavily grazed or un-grazed saltmarshes (Norris *et al.* 1998; Malpas *et al.* 2013). However, Malpas *et al.* (2013) found that the number of breeding pairs declined by 51.6% in northern England where grazing was more intensive, but also by 24.2% and 58.1% respectively in eastern and southern England where light grazing prevailed. Redshank breeding success can be higher in areas dominated by tall *Elymus spp.*, which occur predominantly in ungrazed saltmarshes (Thyen & Exo 2003). Nest survival is a critical component of reproductive success for many birds (Johnson, Nichols & Schwartz 1992; Hoekman *et al.* 2002). Livestock can reduce Redshank nest survival by trampling nests (Beintema & Muskens 1987). Nest predation rates may be indirectly increased by grazing either through decreasing nest cover and vegetation heterogeneity, therefore making it easier for predators to find nests (Baines 1990; Hart *et al.* 2002), or by reducing the amount of unoccupied patches of long vegetation that must be searched by a predator in order to find a nest

(Chalfoun & Martin 2009). In a meta-analysis of European studies, Macdonald & Bolton (2008) found that > 50% of wader nests were predated in > 55% of the 544 examples that they reviewed. Potential Redshank nest predators on British saltmarshes include corvids *Corvus spp.*, gulls *Larus spp.*, Red Foxes *Vulpes vulpes*, Stoats *Mustela ermine* and non-native American Mink *Neovison vison* (Smart 2005).

To balance this potential conflict between the creation of a suitable vegetation sward whilst minimising grazing-induced nest mortality, densities of ~1 cattle ha⁻¹ have been recommended (Norris *et al.* 1997). This falls within the UK Environment Agency definition of light saltmarsh cattle grazing of 0.7-1 young cattle ha⁻¹, present from April to October (Adnitt *et al.* 2007). More recent evidence suggests these light grazing levels may be too high for breeding waders. For example, Pakanen, Luukkonen & Koivula (2011) found approximately 80% of experimental false wader nests were lost to trampling and predation in a Baltic coastal meadow with 0.83 cattle ha⁻¹. On saltmarshes the effect may be even stronger, as Redshank typically nest in vegetation communities associated with the high-mid marsh (Norris *et al.* 1997), which is typically closer to the landward side of the saltmarsh, and it has been suggested that grazing pressure can be higher in these areas (Patterson & Burrows 1998).

In this study, we estimate Redshank nest survival over a range of livestock densities on six lightly grazed saltmarshes in the Ribble estuary, north-west England. We hypothesise that higher grazing pressure results in increased nest mortality: (1) directly through cattle trampling and (2) indirectly through grazer modification of habitat that accelerates predation risks.

2.3 Methods

2.3.1 *Field site*

This study was carried out on six saltmarshes of the Ribble estuary with grazing intensities that varied between 0 - 0.82 cattle ha⁻¹ present during the 2012 breeding season (Fig. 2.1). Much of the estuary is managed as a National Nature Reserve which supports one of the largest areas of grazed saltmarsh in the UK (Burd 1989; Skelcher

2010). The estuary is bounded by a river channel on the estuarine side and by a sea wall flood defence at the landward side. The vegetation is typical of saltmarshes on the west coast of the UK. The pioneer zone at the seaward boundary is composed of tussocks of *Spartina anglica* with scattered *Salicornia europaea* (Skelcher 2010). The saltmarshes are dominated by *Puccinella maritima* and *Festuca rubra* communities, which form a short turf with occasional grassy tussocks across most of their extent where grazed by livestock. In areas where livestock grazing is limited or absent, *Elymus repens* dominates, mainly through the central and upper parts of the saltmarsh extending on to the vegetated flood defences. Most of the site is grazed by young beef cattle; however ungrazed saltmarsh is present in parts. Variation in cattle density on the Ribble occurs largely due to the ability of local graziers to supply cattle, and to create different vegetation conditions on the saltmarshes (Skelcher 2010).

2.3.2 Nest finding

In May and June 2012, a total of 45 Redshank nests were found across the six saltmarshes (between 5 and 10 nests per marsh). These were found by systematic searches in which all parts of the study saltmarshes were walked to within 50m, observing adults flushed from their concealed nests to ensure consistency of nest discovery across all vegetation types. As we were concerned that we may miss nests in areas of longer vegetation, which could bias our estimate of the effect of grazing, we noted if we had flushed an adult bird but were not then able to find a nest. This occurred only twice, and on each occasion the nest was found upon completing a subsequent search. Nest searching occurred throughout May and June, on each weekly visit to each of the six saltmarshes. No new nests were found in July, but any active nests continued to be under observation. Unnecessary disturbance was avoided by following British Trust for Ornithology (BTO) guidelines (Ferguson-Lees, Castell & Leech 2011). To record nest survival, nests were revisited at least weekly, but occasionally more often where time allowed. This can improve the chances of successfully identifying the cause of nest failure, and we selected our analytical methods following Johnson (2007) to ensure this did not bias our results. Nests were revisited until the eggs had either hatched or the clutch had failed, and observed directly at the nest as it would not be possible to observe nests from a distance. Nests

were considered successful if one or more egg hatched. Causes of nest failure were classified following the methods of Smart (2005) and Bodey *et al.* (2010). If eggs were missing and/or remains of eaten eggs were present, the nest was classified as predated ($n = 14$). If the clutch was cold on two consecutive visits, the nest was classified as deserted ($n = 0$). If eggs had clearly been flattened/crushed in the nest they were classified as trampled by livestock ($n = 7$). If, immediately after high tides, all or part of the clutch was missing and nests were wet and/or covered in salt deposits, nests were classified as destroyed by tidal flooding ($n = 13$), and revisited again to be sure the nest was no longer active.

2.3.3 Grazing data and distance to the sea wall

As we hypothesised that nest loss to livestock trampling was a direct result of the presence of cattle in the breeding season, and nest loss to predation was an indirect result of the effects of grazing on vegetation structure, two measurements of stocking density were used. These were seasonal cattle density, indicative of trampling risk (Jensen, Rollins & Gillen 1990), and annual cattle density, a measurement of how grazing changes the vegetation landscape (Andresen *et al.* 1990). Seasonal cattle density measured the density of cattle within each saltmarsh during the breeding season; it was the mean direct count of cattle (ha^{-1}), based on weekly observations throughout the period when nests were active (May – July 2012). Annual cattle density took into account the number of cattle present within a saltmarsh over a 12 month period preceding the study to reflect the longer-term nature of grazing induced changes to the habitat (Andresen *et al.* 1990). Annual cattle density was calculated from stock numbers provided by land managers, using the formula: *Annual cattle density = number of cattle * number of months the animals were present / (size of site * 12)*. These calculations took into account only the grazed part of the saltmarshes. In the case of saltmarsh C (Fig. 2.1), a large network of creeks prevented grazer access to part of the saltmarsh so only the area accessible to livestock was included in this study. In all other grazed saltmarshes livestock used the entire extent of the area studied.

Distance to the sea wall was used as a measurement of the distance from the nest to the landward side of the marsh; it was included to account for effects of the position of

the nest within the landscape, which may influence both cattle and predator movements (Patterson & Burrows 1998; Eglington *et al.* 2009; Nolte *et al.* 2014). Fresh drinking water was available to cattle in drinking troughs situated near the landward extent of all grazed saltmarshes, but no known additional fodder or artificial shelter was provided.

2.3.4 Data analysis

Nest survival was computed in relation to both annual and seasonal cattle density and the distance from the nest to the sea wall using the nest survival model of Program MARK (Dinsmore, White & Knopf 2002). Program MARK uses numerical maximum-likelihood techniques and computes a quasi-likelihood AIC value (White & Burnham 1999). This enabled the selection of the variable(s) (cattle density, distance to the sea wall, or neither variable) that most strongly accounted for variation in nest survival. This method does not assume that nest visits are made at regular intervals (Johnson 2007). As we observed similar proportions of nests lost to predation throughout the study period, it was assumed that predation risk remained constant.

As both measurements of cattle density (annual and seasonal) were correlated ($r = 0.765$, $n = 6$) they could not be included in the same analysis, and to enable the evaluation of nest survival due to cause-specific components (Donovan *et al.* 1995), a total of four separate analyses were carried out (Tables 2 & 3). All 45 nests were used in each case, but depending on the analysis in question, following Fondell & Ball (2004), either nests trampled or predated were treated as unsuccessful and the remaining nests as successful (even though many of them were lost due to another cause). To estimate true probability of nest loss over the 24 day Redshank incubation period (Green 1984), the daily survival rate was raised to the power of 24 following Rotella, Cooch & White (2009) and converted from a survival to a mortality probability. Variance was calculated using the Delta method (Powell 2007) from which confidence intervals were calculated, following Armstrong *et al.* (2002).

Program MARK estimated all possible models. Model outputs were compared using the Akaike Information Criterion (corrected for small sample size, AICc) (Burnham &

Anderson 2002). The model with the lowest AICc value is the most parsimonious model. Models differing 2-7 AICc from the most parsimonious model have little empirical support; those with $\Delta AICc > 10$ have no support. Akaike weights, (the relative likelihood of the models, i.e. $\exp(-0.5 \cdot \Delta AICc)$ for each model, divided by the sum of these values across all other models) were used to measure the support for each model. To check the model fit, we obtained a \hat{C} value from Program MARK for the global model. Following Lebreton et al. (1992) if the model structure is adequate, \hat{C} is expected not to exceed 3.

At 0.82 cattle ha⁻¹, saltmarsh C had a higher seasonal cattle density than the other saltmarshes, which ranged between 0.00-0.34 cattle ha⁻¹. To be sure that the results from this saltmarsh were not biasing our conclusions, we re-ran the model using only data from the saltmarshes with 0.00-0.34 cattle ha⁻¹. These results were then plotted using program Mark's graphing facility, extrapolating to a seasonal cattle density value of 0.82. As this approach showed near identical results of 99% probability of nest loss to predation for this exercise compared with 98% in our original model, it led to the same conclusions. This approach was not necessary for annual cattle density as this was more evenly spaced between the saltmarshes.

2.4 Results

Forty-five Redshank nests were monitored for a total of 323 exposure days over a 65 day interval in 2012 (Table 2.2). Eleven nests hatched successfully, representing a 24.4% hatch rate. The cause of mortality was identified for all failed nests. Seven nests were trampled by livestock, 13 were destroyed by tidal flooding, and 14 were predated. The first nest with eggs was recorded on 06/05/12. The last nest with eggs was found on 26/06/12. Nest recording stopped on 09/07/12, when the last nest hatched. As the focus of this study was effects of grazing on Redshank nest survival, effects of tidal flooding were not considered.

Nest mortality due to trampling increased with seasonal cattle density. At a seasonal cattle density of 0.15 ha⁻¹, one of five nests were trampled (20%), compared to four of eight nests (50%) in the saltmarsh grazed at a seasonal cattle density of 0.82 ha⁻¹.

Distance to the sea wall for Redshank nests ranged from 56m to 995m, while the distance for nests lost to trampling ranged from 185 to 749m (mean = 409.78m) distance to the sea wall. Model fits showed that the risk of nest loss to trampling increased from 16% at 0.15 seasonal cattle density ha^{-1} to 98% at 0.82 seasonal cattle density ha^{-1} in the breeding season (Fig. 2.2; Table 2.1i, model set 1, AIC_c weight = 0.44, $\Delta\text{AIC}_c=10.62$ when this model was compared with the constant survival model without explanatory variables). The models “seasonal cattle density” and “seasonal cattle density + distance to the sea wall” were equally plausible ($\Delta\text{AIC}_c=0.12$ on addition of distance to the sea wall) while there was no support for an interaction between seasonal cattle density and distance to the sea wall (Fig. 2.3; $\Delta\text{AIC}_c=2.05$ on removal of the interaction). The probability of nest trampling was affected by the annual cattle density (Table 2.1ii, model set 3, $\Delta\text{AIC}_c=4.15$ when this model was compared with the constant survival model). However, there was no strong evidence of an effect of distance to the sea wall (the most parsimonious model did not include 'sea wall' and $\Delta\text{AIC}_c=1.95$ on addition of distance to the sea wall to this model).

Nest mortality due to predation increased with annual cattle density. At an annual cattle density of zero, none of five nests were predated compared to four of eight nests in the saltmarsh with 0.55 annual cattle density ha^{-1} . Nests lost to predation ranged from 56 to 995m (mean = 485m) distance to the sea wall. Model fits showed that the risk of nest predation during the incubation period increased from 28% with no grazing to 95% at a 0.55 annual cattle density ha^{-1} (Fig. 2.4; Table 2.1ii, model set 4, AIC_c weight = 0.55, $\Delta\text{AIC}_c=4.24$, when this model was compared with the constant survival model). The models “annual cattle density” and “annual cattle density + distance to the sea wall” were equally plausible ($\Delta\text{AIC}_c=1.52$ on removal of distance to the sea wall) while there was no strong evidence for an interaction between annual cattle density and distance to the sea wall (Fig. 2.5: $\Delta\text{AIC}_c=1.97$ on removal of the interaction). There was no strong evidence that the probability of nest predation was affected by seasonal cattle density (Table 2.1i, model set 2, $\Delta\text{AIC}_c=0.71$ when this model was compared with the constant survival model).

2.5 Discussion

The results of this study suggested that higher grazing pressure reduces Redshank nest survival rates, even across a range of saltmarshes grazed at < 1 cattle ha^{-1} . Results suggested support for hypothesis 1, that higher grazing pressure results in increased nest mortality directly through cattle trampling. However, results also suggested that the indirect effects of grazing on vegetation may affect the probability of nest loss to trampling. Results suggested a link between grazing and nest predation, supporting hypothesis 2 that higher grazing pressure results in increased nest mortality indirectly through grazer modification of habitat that increases predation risk. Mortality due to predation was positively related to the annual grazing management regime, but not to cattle presence in the breeding season.

Although Figure 2.3 suggested that the probability of nest loss to trampling is higher closer to the sea wall on saltmarshes with higher seasonal cattle density, the overlapping confidence intervals and the model results demonstrated no strong evidence to support an interaction. Cattle may spend more time closer to the sea wall due to the proximity of palatable vegetation or the provided fresh water source. Similarly, although Figure 2.5 suggested that the probability of nest loss to predation is lower closer to the sea wall on saltmarshes with higher annual cattle density, these overlapping confidence intervals and the model results demonstrated no strong evidence to support an interaction. Ground-based predators may cause nest mortality closer to the sea wall, however avian predators may cause nest mortality anywhere on the saltmarsh (Montevicchi 1977). It is therefore possible that Redshank nests used in this study were exposed to both aerial and ground predators. Identifying specific predator species was beyond the scope of this study.

When calculating livestock density our intention was to use a procedure similar to that used by land managers. These can vary but, management guidelines e.g. Adnitt *et al.* (2007) tend to focus on simple calculations of numbers of livestock divided by the size of the site in hectares, taking into account the part of the saltmarsh that is accessible to livestock and the number of months animals are present. This approach has a known constraint – that grazing across a saltmarsh can be localised closer to the sea wall (Patterson & Burrows 1998). However, calculating precise grazing pressure to

account for spatial variation was beyond the scope of this study. Of the cattle on our study saltmarshes, each individual could be translated to an LSU of 0.8 (DEFRA 2010), however our calculations of livestock density are based on numbers of individuals.

As this study was based on six saltmarshes of the Ribble estuary, it is not clear if these trends can be generalised and applied to other locations. The climax community of UK saltmarshes is dominated by Couch grass, *E. repens* in the west and *Elytrigia atherica* in the south and east (Rodwell *et al.* 2000). Both Couch grass species produce a similar vegetation sward, which if exposed to grazing is reduced in favour of *P. maritima* and *F. rubra*. Therefore, vegetation responses to grazing and indirect effects on breeding Redshank are likely to be similar around the UK. However, local predator communities may vary between different regions and years, with nest predation rates likely to be dependent on the habitat matrix of surrounding landscapes (Laidlaw *et al.* 2013). All saltmarshes included in this study were grazed by young cattle, which is commonplace on British saltmarshes (Adnitt *et al.* 2007), however young cattle may trample more nests (Beintema & Muskens 1987) possibly due to their more lively nature (Ausden 2007). Elements of different grazing regimes employed on different saltmarshes may have a large effect on nest survival. Sheep are more likely to produce shorter vegetation swards than cattle (Green 1986; Beintema and Muskens 1987) and horses are more likely to trample nests (Mandema *et al.* 2013). Different weather conditions, and tidal flooding rates in different regions and different years may also influence results and it is important to note this study was limited to only one breeding season.

Our study is based on a relatively low sample size of 45 nests, but it is within the range of other studies of bird species with low populations: e.g. 14 (Gregory *et al.* 2011), 31 (Tweed *et al.* 2006), 54 (Blanvillain *et al.* 2003) and 58 (Watson *et al.* 2006). Hensler and Nichols (1981) demonstrated that samples of ≥ 20 nests are sufficient to detect trends which are likely to be indicative of the wider population. A low sample size is unavoidable when studying species with low or declining populations, such as Redshank. These declines occurred even in regions where light grazing prevailed (Malpas *et al.* 2013).

Redshank nest survival on saltmarshes is also negatively affected by nest flooding (Thompson & Hale 1991; Norris 2000; Smart 2005). Losses of between 3 – 40% of all observed nests have been reported as lost to tidal flooding, a figure which can vary between different years and estuaries (Thompson & Hale 1991; Smart 2005). Although tidal flooding was not a focus of this study, we observed that all nest losses to tidal flooding took place during 'spring tides' $\geq 9\text{m}$. These are a regular occurrence in the Ribble estuary, which suggests results would be similar in other years. Climate change may increase tidal flooding. In the Wadden sea, for example the maximum high tide has increased twice as fast as mean high tide over the past four decades, resulting in more frequent and more catastrophic flooding of nests (van de Pol *et al.* 2010). However breeding Redshank populations in the Wadden sea are more stable, with only minor declines in some regions (Koffijberg *et al.* 2009). It is unclear if increased tidal flooding has played a part in Redshank population declines in the UK.

Model fits should be treated with some caution, as they show 0.01 probability of nest trampling at zero grazing, which when converted to a probability for the 24 day incubation period translates to a trampling probability of 0.08. This demonstrates that small differences in daily mortality probabilities translate to larger differences when converted to a mortality probability for the incubation period. Also, this probability of trampling > 0 at zero grazing seems improbable and may arise from fitting the nest survival model through data which included occasional nest loss to trampling at very low livestock densities. Although unexpected, results also suggested that the annual cattle density from the year prior to the breeding season was related to nest loss to livestock trampling. This could simply be because the two measurements of cattle density were correlated, or because the Redshank selected nests in vegetation that had been grazed previously. Previously grazed saltmarsh vegetation is more palatable to cattle and therefore more likely to be re-visited (Bakker 1985), therefore increasing nest trampling risk.

Considering the limitations discussed above, further research may be necessary before taking conservation action in areas other than the Ribble estuary. However, it should be noted that all saltmarshes included in this study were within a nature reserve, but those grazed at the highest annual cattle density of 0.55 ha^{-1} or with 0.82 ha^{-1} seasonal cattle density, would be unlikely to sustain stable populations of

Redshank due to the 98% risk of trampling and the 95% risk of predation. Translating the UK Environment Agency light grazing definition of 0.7-1 young cattle ha⁻¹ between April and October (Adnitt *et al.* 2007) to measurements used in this study, would mean seasonal cattle density of 0.7-1 ha⁻¹ and an annual cattle density of around 0.35 - 0.5 ha⁻¹. Model fits presented in this study suggested that Redshank nests on saltmarshes grazed at these intensities would have between a 90% - 98% risk of loss to livestock trampling and 75% - 92% risk of being predated. This suggests that the UK Environment Agency definition of light grazing is too intensive for saltmarsh breeding Redshank.

Our conclusions contrast with those of Norris *et al.* (1997) that cattle grazing of around 1 cattle ha⁻¹ is likely to maintain a high breeding density of Redshank, but the study did not account for nest survival. Birds preferentially selecting nesting sites that result in low nest survival (i.e. ecological traps) in landscapes modified by humans is not a new phenomenon (Best 1986; Schlaepfer, Runge & Sherman 2002) and if this is happening to Redshank, it could explain some of the differences between the findings of this study and the recommendations of Norris *et al.* (1997). Therefore although grazing can result in a higher breeding density of Redshank, further reducing grazing might improve nest survival. Research into this potential trade-off is necessary before suggesting ideal livestock densities for Redshank breeding. This work does not suggest that stopping livestock grazing on saltmarshes altogether will result in increased breeding populations of Redshank. Cessation of grazing in previously grazed saltmarshes can result in reductions in breeding Redshank as the vegetation becomes dominated by tall uniform vegetation (Norris *et al.* 1997). Furthermore, livestock grazing in saltmarshes can drive abundance and diversity of invertebrate prey (Ford *et al.* 2013). Further investigations into the ideal grazing management practices for Redshank may be necessary. If UK Environment Agency guidelines are followed, lightly grazed saltmarshes would have livestock present from April until October (Adnitt *et al.* 2007). Although our results suggest that Redshank breeding populations may benefit from reduced livestock grazing, we are by no means suggesting that this is the only solution to reverse declining population numbers. However, altering the timing of grazing, to introduce cattle outside of the breeding season would eliminate nest trampling. In other habitats, commencing grazing after the end of July has been shown to increase productivity in Redshank and other waders

(Green 1986). This may also increase vegetation height sufficiently to increase nest cover in the breeding season, whilst maintaining the more open sward structure created by grazing. Alternatively, a rotational grazing regime where saltmarshes are grazed and left fallow in alternate years may improve breeding success by eliminating nest trampling in the fallow year.

This work suggests that either previously recommended stocking levels may be too high, or that Redshank breeding areas are more intensively grazed than intended by land managers, due to the non-homogeneous distribution of cattle. This results in high nest mortality due to predation and trampling, therefore further investigations are needed into saltmarsh breeding habitat use, and movement of livestock on grazed saltmarshes. In conclusion, this work suggested that livestock grazing may negatively affect nest survival in saltmarsh breeding Redshank populations. Even in lightly grazed saltmarshes, grazing-induced nest mortality appeared to be high. This arose both indirectly through grazer modification of landscapes that accelerate predation risks, and directly through cattle trampling in the breeding season. Based on these results, it may be appropriate to review saltmarsh grazing management practices.

2.6 Tables

Table 2.1: Results of program MARK nest failure to trampling and predation analyses. ‘ ΔAIC_c ’: the difference between the model in question, and the top model. $\Delta AIC_c < 2$ suggests both models are plausible (Burnham & Anderson 2002). Akaike weights represent the relative likelihood of the models (i.e. $\exp(-0.5 * \Delta AIC_c)$ for each model, divided by the sum of these values across all other models). Model likelihood is the support of the model in question relative to other models in the candidate set. \hat{C} values between 1 and 3 suggest the model structure is adequate (Lebreton *et al.* 1992). SCD = Seasonal cattle density. ACD = Annual cattle density. DSW = Distance to the sea wall.

i) Seasonal cattle density as a measure of grazing intensity.

Response variable	Model	AIC _c	ΔAIC_c	Akaike Weight	Model Likelihood	\hat{C}
1. Daily nest failure from trampling.	SCD	52.40	0.00	0.44	1.00	1.12
	SCD + DSW	52.51	0.12	0.41	0.94	1.10
	SCD * DSW	54.56	2.16	0.15	0.34	1.13
	Constant survival	63.03	10.62	0.00	0.00	1.39
	DSW	65.02	12.62	0.00	0.00	1.42
2. Daily nest failure from predation.	Constant survival	76.05	0.00	0.35	1.00	1.68
	SCD	76.76	0.71	0.24	0.70	1.69
	DSW	77.31	1.26	0.18	0.53	1.70
	SCD * DSW	78.00	1.96	0.13	0.38	1.70
	SCD + DSW	78.61	2.56	0.10	0.28	1.73

ii) Annual cattle density as a measure of grazing intensity.

Response variable	Model	AIC _c	ΔAIC_c	Akaike Weight	Model Likelihood	\hat{C}
3. Daily nest failure from trampling.	ACD	58.87	0	0.59343	1	1.27
	ACD + DSW	60.82	1.95	0.22437	0.3781	1.30
	ACD * DSW	62.87	4.00	0.08	0.13	1.33
	Constant survival	63.03	4.15	0.07	0.12	1.39
	DSW	65.02	6.15	0.03	0.05	1.42
4. Daily nest failure from predation.	ACD	71.82	0.00	0.55	1.00	1.58
	ACD + DSW	73.33	1.52	0.26	0.47	1.60
	ACD * DSW	75.30	3.49	0.09	0.17	1.64
	Constant survival	76.05	4.24	0.06	0.12	1.68
	DSW	77.31	5.50	0.03	0.06	1.70

Table 2.2: Description of saltmarsh field sites and outcomes of individual nests. Exposure days were calculated using the Mayfield method (Mayfield 1961; Mayfield 1975).

Saltmarsh	Area (ha ⁻¹)	No. of nests	Exposure days (mean per nest)	Flooded	Predated	Trampled	Hatched
A	237	10	10.20	4	1	2	3
B	127	5	8.80	1	2	1	1
C	704	8	3.75	1	3	4	0
D	63	9	9.44	1	4	0	4
E	58	8	4.37	2	4	0	2
F	257	5	5.40	4	0	0	1

2.7 Figures

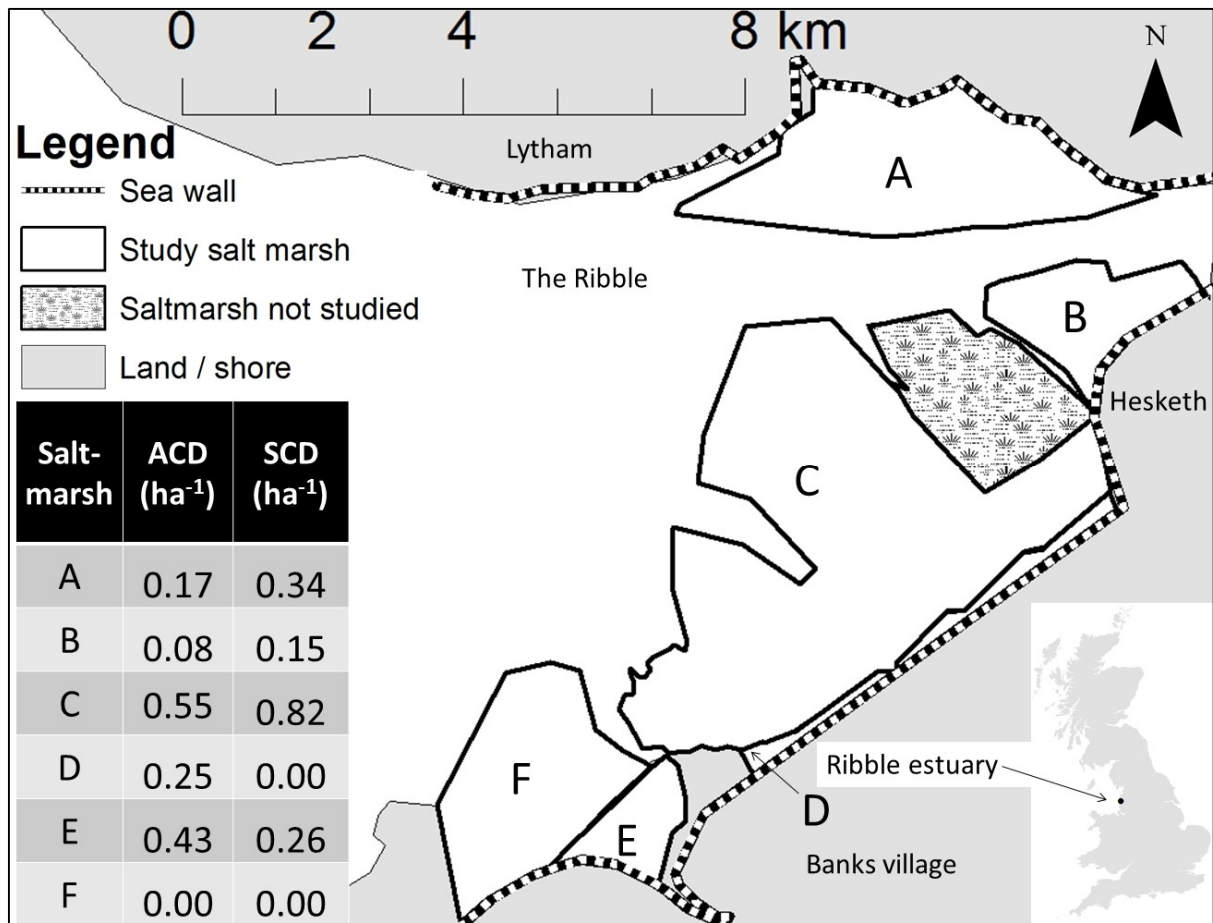


Figure 2.1: Ribble Estuary map.

This Shows the study saltmarshes and geographic location, with annual cattle density (ACD) and seasonal cattle density (SCD) for each of the saltmarshes included in this study.

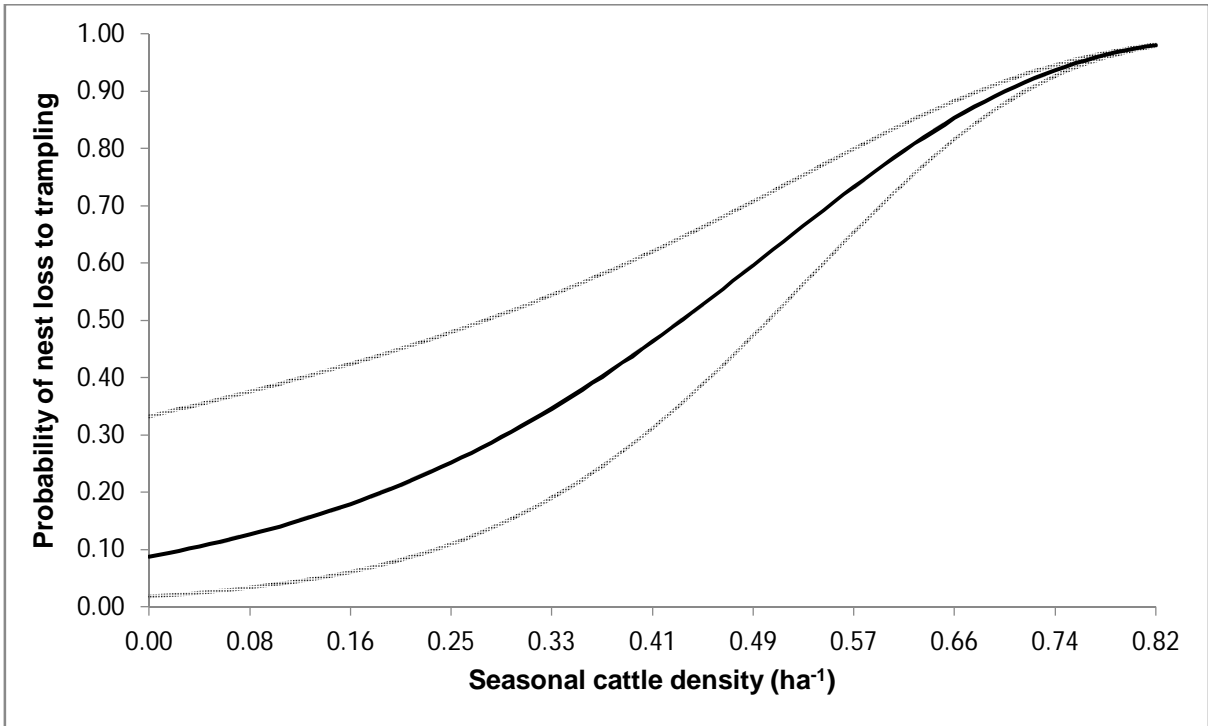


Figure 2.2: The probability of nest loss to trampling. This is the statistical fit of a nest survival model. The solid line shows predicted effects of seasonal cattle density on Redshank nest mortality to livestock trampling across a 24 day nest incubation period. Confidence intervals are represented by grey lines.

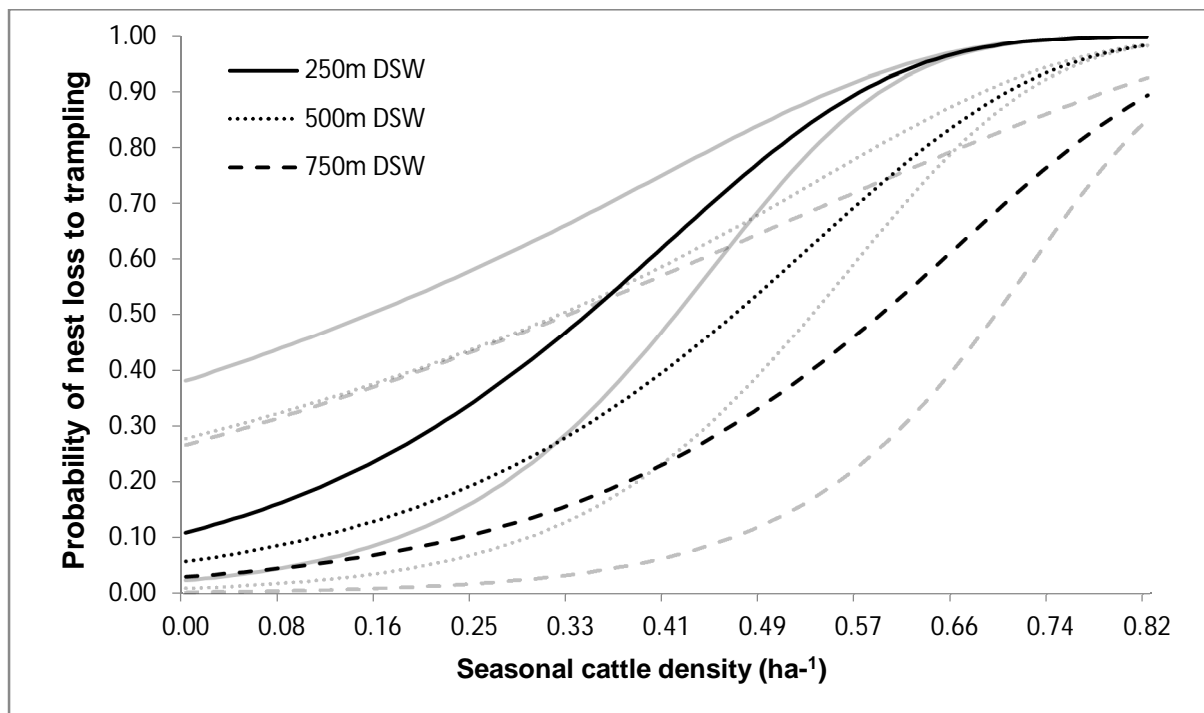


Figure 2.3: The probability of nest loss to trampling at different distances to the sea wall.

This is the statistical fit of a nest survival model showing predicted effects of seasonal cattle density on Redshank nest mortality to trampling across a 24 day nest incubation period, at 250m, 500m, and 750m distance to the sea wall (DSW). Light grey lines indicate confidence intervals.

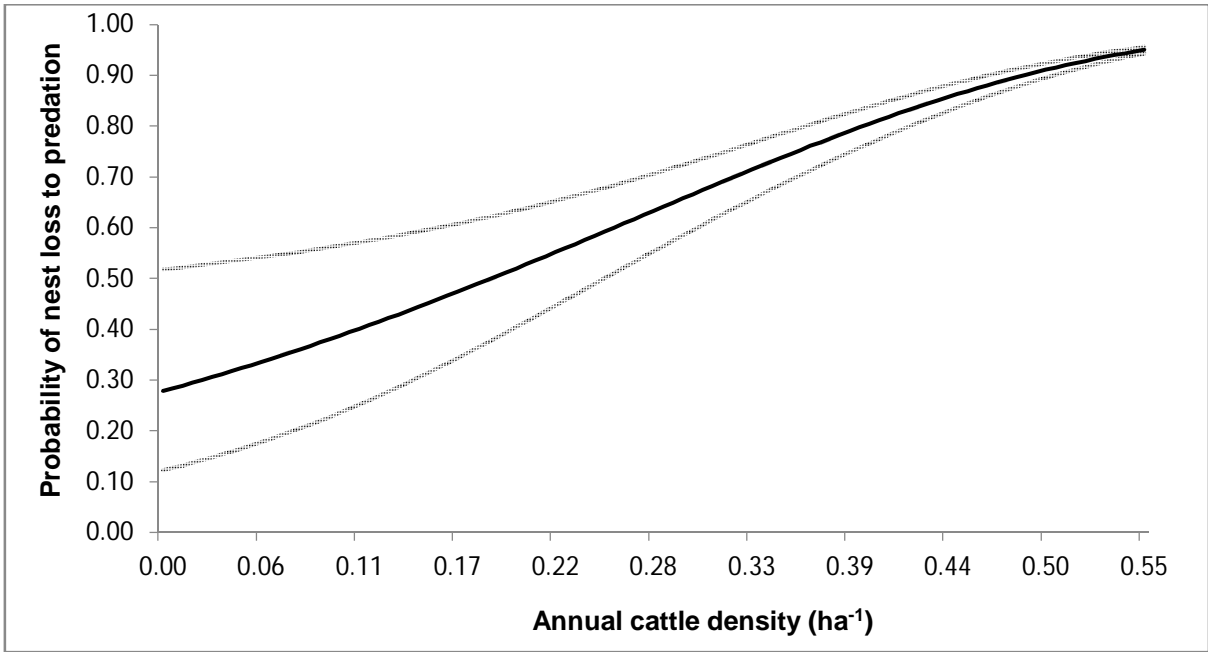


Figure 2.4: The probability of nest loss to predation.

This is the statistical fit of a nest survival model showing predicted effects of annual cattle density on Redshank nest mortality to predation across a 24 day nest incubation period.

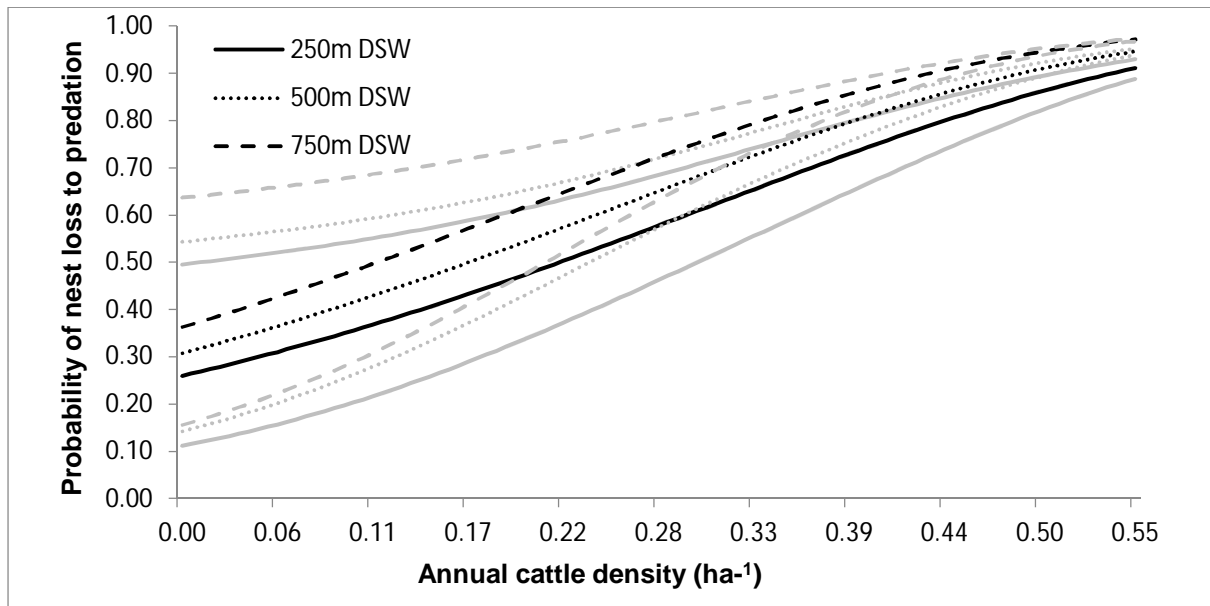


Figure 2.5: The probability of nest loss to predation at different distances to the sea wall.

This is the statistical fit of a nest survival model showing predicted effects of annual cattle density on Redshank nest mortality to predation across a 24 day nest incubation period, at 250m, 500m, and 750m distance to the sea wall (DSW).

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3 Does light grazing of saltmarshes reduce breeding habitat quality for Common Redshank *Tringa totanus*?

3.1 Abstract

The Common Redshank *Tringa totanus* breeding population on British saltmarshes has reduced by > 50% since 1985, with declines linked to changes in grazing management. Conservation initiatives have encouraged low-intensity grazing of < 0.5 cattle ha⁻¹ y⁻¹ but even light grazing can lead to high rates of nest mortality, and populations have continued to decline. To investigate Redshank nest site selection across saltmarshes with a gradient of low intensity livestock densities, this study aims to identify nesting habitat use as a measurement of nesting habitat selection, and how this varies with different levels of livestock density. We asked: (a) which nest vegetation conditions do Redshank select at different spatial scales? and (b) does grazing limit the availability of higher quality nest sites?'. This was studied by characterising vegetation height and composition at nests and control locations on six saltmarshes grazed between 0 and 0.55 cattle ha⁻¹ y⁻¹. Redshank optimally selected nest locations in the tallest vegetation available but grazing limited the availability of long vegetation. Mean vegetation height at Redshank nests decreased from 26±13cm with no grazing, to 11±7cm at 0.55 cattle ha⁻¹ y⁻¹, suggesting that grazing caused Redshank to nest in less optimal vegetation. However, Redshank also selected nest locations dominated by the grass *Festuca rubra*, which increased in saltmarshes with higher livestock densities. As nesting in shorter vegetation results in higher nest predation rates, results suggest that even light conservation grazing of < 0.55 cattle ha⁻¹ y⁻¹ can result in Redshank nesting in lower quality habitat. It may therefore be appropriate to reduce saltmarsh grazing intensities, or implement a rotational grazing system with saltmarshes grazed in alternate years.

3.2 Introduction

Habitat use affects the fitness of animals through variation in environmental conditions and resource availability, which generates selective pressure for habitat choice (Pulliam 2000). This in turn influences the survival and reproduction of individual birds (Brown 1969; Fuller 2012). However, the density of animals in a habitat can be a misleading indicator of habitat quality (Van Horne 1983) as species can preferentially use habitat which acts as an 'ecological trap' by lowering breeding success (Best 1986; Schlaepfer, Runge & Sherman 2002). For many ground nesting birds, predation is the largest cause of reproductive failure (Angelstam 1986; Macdonald & Bolton 2008). Consequently, variation in adaptations to reduce the risk of nest predation has evolved (Martin 1995). As predominantly ground-nesters, waders show various different kinds of anti-predator adaptations, including passive and active nest defence (Gochfeld 1984). Unlike Black-tailed Godwit *Limosa limosa*, and Oystercatcher *Haematopus ostralegus*, the Common Redshank *Tringa totanus* does not attack potential nest predators (Gochfeld 1984; Nethersole Thompson & Nethersole Thompson 1986). Instead, they build a cryptic nest obscured by tall vegetation and quietly leave the nest upon approach of a predator (Cervenci *et al.* 2011; Mandema *et al.* 2014). Habitat quality for breeding Redshank is higher with more vegetation cover, allowing the concealment of nests, which leads to higher breeding success (Thyen & Exo 2003).

Redshank breed in various grassland habitats, but British saltmarshes are nationally and internationally important for the species (Reed 1985; Cadbury, Green & Allport 1987; Brindley *et al.* 1998), holding around half of the British breeding population (Brindley *et al.* 1998). However saltmarsh populations of Redshank declined by > 52% between 1985 and 2011 (Malpas *et al.* 2013). Declines have been linked to grazing management as breeding densities are higher in light and moderate grazing than on either heavily grazed or un-grazed saltmarshes (Norris *et al.* 1998; Malpas *et al.* 2013). However, Malpas *et al.* (2013) found that populations declined by 28.8 pairs km² in northern England where grazing was more intensive, but also declined by 16.6 and 27.9 pairs km² respectively in eastern and southern England where light grazing prevailed. Grazing has the potential to alter habitat suitability of saltmarshes by limiting or creating availability of the vegetation patches that are used for Redshank nesting

(Hale 1980; Mandema *et al.* 2014). Intensive grazing leads to a very short, uniform sward, lighter grazing results in a more uneven patchy sward with diverse heights whilst no grazing can leave saltmarshes with dense communities of *Elymus spp.* or *Elytrigia spp.* (Couch grasses), (Jensen 1985; Kiehl *et al.* 1996). Grazing reduces the extent of Couch grasses in favour of *Festuca rubra* (Andresen *et al.* 1990). Redshank typically select grassy areas for nesting (Thyen & Exo 2003; Smart *et al.* 2006). On saltmarshes, these are usually within communities of *F. rubra*, *Puccinellia maritima*, or Couch grasses, (Hale 1988; Norris *et al.* 1997; Thyen & Exo 2003). Of these vegetation types Couch grasses can be regarded as the highest quality Redshank breeding habitat as they lead to the highest rates of nest survival (Thyen & Exo 2003).

Norris *et al.* (1997) demonstrated that breeding densities are higher in structurally diverse grazed vegetation, and to create a patchy vegetation sward, suitable for nesting Redshank, stocking densities of ~1 cattle ha⁻¹ were recommended (Norris *et al.* 1997). This density falls within the UK Environment Agency definition of light saltmarsh cattle grazing of 0.7 - 1 young cattle ha⁻¹, present from April to October (Adnitt *et al.* 2007) or 0.35 - 0.5 cattle ha⁻¹ y⁻¹. However, recent work has suggested that light grazing may not be optimal, as it can reduce nest survival both directly through nest trampling and indirectly through accelerating predation risks. Sharps *et al.* (2015) found that the risk of a Redshank nest being predated increased from 28% with no grazing to 95% at grazing of 0.55 cattle ha⁻¹ y⁻¹. Following the estimates of Macdonald and Bolton (2008), suggesting that waders typically need nest survival rates of > 49% for population stability, this implies that light grazing can negatively affect Redshank populations. Sharps *et al.* (2015) suggested that the increased nest mortality to predation is caused by livestock modification of vegetation. However, the effects of grazing on the quality of the habitat selected by nesting Redshank remain unclear.

Habitat quality is a key to an individual's fitness and organisms occupying habitats that maximise their lifetime reproductive success will contribute the most to future generations (Newton 1989; Block & Brennan 1993). Therefore, habitat quality can be defined in terms of the reproductive success attributable to a given habitat (Coulson *et al.* 2006; Johnson 2007). The terms '*nest-site selection*' or '*habitat selection*' refer to a process of behavioral responses that may result in the uneven use of habitats

(Hutto 1985; Jones 2001). The term '*nesting habitat use*' is a measurement of nesting habitat selection, as it refers to the way in which an individual or species uses habitats to meet its life history needs (Jones 2001; Fuller 2012). Habitat-use patterns are the end result of habitat-selection processes, and meaningful information about habitat use can be gained by comparing nesting sites used by a species with sites available to be used on a spatial scale appropriate to the species in question (Jones 2001). Redshank are highly site faithful, and have been found to typically return to within 50-1700m of their birthplace (Thompson & Hale 1989).

To investigate Redshank nest site selection across saltmarshes with a gradient of low intensity livestock densities, this study aims to identify nesting habitat use and how this varies with different levels of livestock density. We asked: (a) which nest vegetation conditions do Redshank select at different spatial scales? and (b) does grazing limit the availability of higher quality nest sites?'. This was studied on six saltmarshes grazed between 0 and 0.55 cattle ha⁻¹ y⁻¹, by characterising vegetation height and species composition at nests and control locations on a small spatial scale. We hypothesised that (1) Redshank optimally select nests in taller vegetation, but grazing limits the availability of preferred conditions; (2) Redshank select taller vegetation next to and in the wider area around the nest, but grazing limits the availability of these conditions; (3) The vegetation species composition varies between nests and control locations, with more of the grasses *F. rubra*, *P. maritima* and *Elymus repens* found near Redshank nests; (4) The proportion of *E. repens* found at Redshank nests is lower in saltmarshes with higher livestock densities and the proportion of *F. rubra* found at Redshank nests is higher in saltmarshes with higher livestock density.

3.3 Materials and methods

3.3.1 *Study location*

This study was carried out on six saltmarshes on the Ribble estuary with grazing intensities that varied between 0 – 0.55 cattle ha⁻¹ y⁻¹ (Fig. 3.1; Table 1). Much of the estuary is managed as a National Nature Reserve which supports one of the largest areas of grazed saltmarsh in the UK (Burd 1989; Skelcher 2010). The estuary is

bounded by a river channel on the estuarine side and by a sea wall flood defence at the landward side. The vegetation is typical of saltmarshes on the west coast of the UK. The saltmarshes are dominated by *F. rubra* and *P. maritima* communities, which form a short turf with occasional grassy tussocks across most of their extent where grazed by livestock. In areas where livestock grazing is limited or absent, *E. repens* dominates, mainly through the central and upper parts of the saltmarsh extending on to the vegetated flood defences. Most of the site is grazed by young beef cattle; however ungrazed saltmarsh is present in parts. The saltmarshes selected for this study were located in the same estuary to minimise the influence of environmental context on study findings, however they were different in terms of size and grazing management. Fieldwork was carried out between May and July 2012. Results of a simultaneous Redshank nest survival study are reported in Sharps *et al.* (2015).

3.3.2 Grazing data

To account for grazer modification of the habitat throughout a calendar year, an annual cattle density was calculated. This took into account the number of cattle present within a saltmarsh over a 12 month period preceding the study, to reflect the longer-term nature of grazing induced changes to the habitat which can occur over a number of years (Andresen *et al.* 1990). Annual cattle density is therefore a more appropriate measure of grazing on the vegetation than simply recording the number of livestock present at the time the nests were active. It was not possible to obtain precise livestock density data from before 2012, although it was roughly maintained at similar levels since at least 2007 (Mercer, pers. comm.). This was calculated from stock numbers provided by land managers, using the formula: *Annual cattle density = number of cattle * number of months the animals were present / (size of site * 12)*. These calculations took into account only the grazed part of the saltmarshes. In the case of saltmarsh C (Fig. 3.1), a large network of creeks prevented grazer access to part of the saltmarsh, so only the area accessible to livestock was included in this study and the annual cattle density calculations. In all other grazed saltmarshes livestock had access to the entire extent of the saltmarsh. All grazed saltmarshes were stocked with young cattle.

3.3.3 Nest finding and measurements

In May and June 2012, a total of 45 Redshank nests were found across the six saltmarshes (between 5 and 10 nests per marsh). These were found by systematic searches in which all parts of the study saltmarshes were walked to within 50m, observing adults flushed from their concealed nests to ensure consistency of nest discovery across all vegetation types. As we were concerned that nests in areas of longer vegetation may be missed, which could bias our estimate of the effect of grazing, we noted if we had flushed an adult bird but were not then able to find a nest. This occurred only twice, and on each occasion the nest was found upon completing a subsequent search. Nest searching occurred throughout May and June, on weekly visits to each of the six saltmarshes.

To investigate if Redshank select nests in taller vegetation (hypothesis 1), we recorded both the mean and maximum vegetation height at the nest. To investigate if Redshank select taller vegetation in the immediate vicinity of the nest and in the wider area around the nest (hypothesis 2), we recorded the mean vegetation height $\leq 1\text{m}$ from the nest and $\leq 10\text{m}$ from nest. All vegetation heights were recorded as direct measurements as studies have shown that this gives the more consistent and accurate results, compared to sward sticks or drop disks (Stewart, Bourn & Thomas 2001). The maximum vegetation height was the height of the tallest single strand of vegetation at the nest, while the mean vegetation height at the nest was calculated by taking the mean of 3 measurements at each nest. Vegetation height $\leq 1\text{m}$ from the nest was estimated by calculating a mean of 5 measurements taken within this area. Vegetation height $\leq 10\text{m}$ from the nest was estimated by taking 15 measurements over this larger area and calculating a mean. To look at selection on a small scale, for each of the nests studied, a control point was selected within 80-100m of the nest (mean distance between nests and control points = 91m). This was selected by generating a random number (between 80 and 100) and a random compass bearing (0-360°). All nest vegetation height measurements were duplicated at control points. As Redshank build their nests in vegetation, control points that fell within a creek were

reselected at random, to ensure only parts of a saltmarsh that had the potential to be vegetated were included.

To investigate if the vegetation species composition (referred to as the 'vegetation community' from now on) varies between nests, control locations and with grazing (hypotheses 3 and 4), percentage cover per species was estimated by eye (Kent 2011), at both $\leq 1\text{m}$ and $\leq 10\text{m}$ proximity to nests and control points. Sampling was carried out in a quadrat of 1m^2 around each nest and control point, and also sampled in five 1m^2 quadrats randomly selected within 10m of each nest and control point.

The observed nests were also recorded as part of a concurrent Redshank nest survival study, reported by Sharps *et al.* (2015). Unnecessary disturbance was avoided by following British Trust for Ornithology (BTO) guidelines (Ferguson-Lees, Castell & Leech 2011), therefore all nest measurements were sampled when the nest had either hatched or failed. To account for sampling vegetation at nests of unequal age, the age of the nest was estimated by weighing and measuring the eggs following the methods of Green (1984).

3.3.4 Data analysis

To investigate if Redshank optimally select nests in taller vegetation, but grazing limits the availability of preferred conditions (Hypothesis 1) and if Redshank select taller vegetation next to and in the wider area around the nest, but grazing limits the availability of these conditions (Hypothesis 2), a series of General Linear Mixed Models (GLMMs) were used in the lme4 package in the statistical programme, R (Bates *et al.* 2014) (Table 3.2, analyses i-iv). The different measurements of vegetation height (at different spatial scales) were used as response variables for each model set. In all cases the predictors were annual cattle density, point type (nest or control) and age of the nest when measured (days, to control for effects of measuring nests of unequal age on vegetation height). An interaction between annual cattle density and point type was added to test if annual cattle density affected the differences between nests and control points. In all cases the response variable was square root transformed to ensure normality of residuals.

To see if vegetation species composition varies between nests and control locations (Hypothesis 3), we followed the methods of (Ladwig *et al.* 2012). This involved using a Permutational Multivariate Analysis of Variance (PERMANOVA) to see if the vegetation community differed between the nests and control locations, and then applying similarity percentages (SIMPER) to assess the contribution of different species. PERMANOVA uses a permutation procedure to assess significance, and thus does not rely on the assumption of multivariate normality, which is frequently violated by ecological data (Anderson 2005). A modified Gower resemblance matrix was used with fourth root transformation to allow changes in composition to be specified explicitly (Anderson, Ellingsen & McArdle 2006). SIMPER identified which species had particular influence on the clustering of samples (Clarke & Gorley 2006).

To investigate effects of grazing on the proportion of *E. repens* and *F. rubra* found near Redshank nests (Hypothesis 4), we used the percentage cover of these species as the response variables in a series of Generalised Linear Mixed Models with binomial error (Table 3.2, analyses i & ii). Predictor variables included annual cattle density, proximity (levels $\leq 1\text{m}$ or $\leq 10\text{m}$ of nest, to account for effects of surrounding area vegetation type). To deal with the large number of zeros in the *E. repens* data, a zero-inflated binomial model was used. An interaction was added to test if the effect of annual cattle density on percentage cover of each species varied between proximity levels. All percentage cover models were run using the package glmmADMB in R (Fournier *et al.* 2011).

In all analyses, saltmarsh (A-F) was included as a random effect to account for spatial pseudo-replication caused by having multiple nests within each saltmarsh, while an additional random effect of Pair ID was included for analyses 1-4 (with each pair of nest and control points given a unique code). Model outputs were compared using the Akaike Information Criterion (corrected for small sample size, AICc) (Burnham & Anderson 2002). The model with the lowest AICc value is the most parsimonious model. Models differing 2-7 AICc units from the most parsimonious model have little empirical support; those with $\Delta\text{AICc} > 10$ have no support. Akaike weights, (the relative likelihood of the models, i.e. $\exp(-0.5 * \Delta\text{AIC})$ for each model, divided by the sum of

these values across all other models) were used to measure the support for each model.

3. Results

Mean and maximum vegetation height was taller at nests compared to control points indicating that Redshank select taller vegetation for their nests (mean vegetation height (cm) \pm sd: nests = 22 \pm 10, control points = 14 \pm 15; Fig. 3.2i; Table 3.2, analysis i, Δ AIC = 17.9 upon removal of type from the model; maximum vegetation height: nests = 38 \pm 15, control points = 13 \pm 15; Fig. 3.2ii; Table 3.2, analysis ii, Δ AIC = 60.4 upon removal of type). Mean and maximum vegetation height decreased in saltmarshes with higher annual cattle density (Mean: Δ AIC = 2 upon removal of the annual cattle density variable from the model; Maximum: Δ AIC = 3.2 upon removal of the annual cattle density variable from the model) and there was no clear support for an interaction between annual cattle density and point type demonstrating that the nature of the relationship was similar for vegetation height at both nests and control points (mean: Δ AIC = +1.7; maximum: Δ AIC = +2.3 upon addition of the interaction). Mean vegetation height at Redshank nests ranged from 26 \pm 13cm in the ungrazed saltmarsh to 11 \pm 7cm at an annual cattle density of 0.55 cattle ha⁻¹ (Fig. 3.2i) and maximum vegetation height ranged from 45 \pm 16cm on ungrazed saltmarsh to 23 \pm 16cm at an annual cattle density of 0.55 cattle ha⁻¹ (Fig. 3.2ii). Together these analyses suggest that Redshank preferentially select the tallest vegetation available but that grazing limits the availability of these conditions causing Redshank to nest in shorter than ideal patches. There was no clear need for the Age variable in either analysis 1 or 2 (mean: Δ AIC = +2.1; maximum: Δ AIC = +1.43 upon addition of age).

Vegetation height was taller next to and in the wider area around nests than at control points indicating that Redshank select nesting sites surrounded by taller vegetation. In the immediate vicinity of nests (\leq 1m), vegetation height ranged from 24 \pm 11cm for nests to 13 \pm 12cm for control points (Fig. 3.2iii; Table 3.2, analysis iii, Δ AIC = 37.02 upon removal of the type variable). Within 10m of sampling points, these trends were similar where vegetation height ranged from 20 \pm 14cm for nests to 15 \pm 14cm for control points (Fig. 3.2iv; Table 3.2, analysis iv, Δ AIC = 10.05 upon removal of the type

variable). Vegetation height next to and in the wider area around nests decreases with increasing annual cattle density (immediate vicinity of nests: $\Delta\text{AIC} = 2$ upon removal of the annual cattle density variable from the model; wider area around nests: $\Delta\text{AIC} = 5.34$ upon removal of the annual cattle density variable from the model) and there was no support for an interaction between annual cattle density and point type demonstrating that the nature of the relationship between changes in vegetation height next to and in the wider area around nests and control points were similar (immediate vicinity of nests: $\Delta\text{AIC} = +2.25$; wider area around nests: $\Delta\text{AIC} = +2.3$ upon addition of the interaction). Vegetation height in the immediate vicinity of Redshank nests ranged from $32\pm 15\text{cm}$ in the ungrazed saltmarsh to $13\pm 9\text{cm}$ at an annual cattle density of $0.55\text{ cattle ha}^{-1}$ (Fig. 3.2iii) and vegetation height in the wider area around nests ranged from $33\pm 17\text{cm}$ on ungrazed saltmarsh to $8\pm 6\text{cm}$ at an annual cattle density of $0.55\text{ cattle ha}^{-1}$ (Fig. 3.2iv). There was no need for the Age variable in either analysis 3 or 4 ($\leq 1\text{m}$: $\Delta\text{AIC} = +2$; $\leq 10\text{m}$: $\Delta\text{AIC} = +0.78$ upon addition of age). Together these analyses suggest that Redshank preferentially select nesting sites surrounded by tall vegetation, both next to and in the wider area around nests, but that grazing limits the availability of these conditions causing Redshank to nest amongst shorter than ideal vegetation swards.

The vegetation community was different next to and in the wider area around nests than at control points, indicating that Redshank select nest sites that are surrounded by particular species of vegetation. In the immediate vicinity ($\leq 1\text{m}$) and in the wider area ($\leq 10\text{m}$) around the nest or control point most of the dissimilarity between nests and control points was due to *F. rubra*, which was more abundant near nests than control points (immediate vicinity of nests: Table 3.3, analysis i, $p < 0.01$, dissimilarity = $22.05\pm 1.37\%$, abundance near nests = 71.67% , abundance near control points = 41.04% ; wider area: Table 3.3, analysis ii, $p < 0.01$, dissimilarity $20.04\pm 1.42\%$, abundance near nests = 59.18% nests, abundance near control points = 39.11%). *P. maritima* was more abundant near control points than nests (immediate vicinity of nests: dissimilarity = $4.45\pm 0.4\%$; abundance near nests = 1.78% , abundance near control points = 7.78% ; wider area: dissimilarity = $5.84\pm 0.47\%$; abundance near nests = 4.98% nests, abundance near control points = 8.36%). Trends for *E. repens* were not as clear, although this species contributed to the dissimilarity between the vegetation communities, the abundance was similar near nests and control points

(immediate vicinity of nests: dissimilarity = $9.18 \pm 0.62\%$, abundance near nests = 11.78%, abundance near control points = 9.98%; wider area: dissimilarity = $9.66 \pm 0.79\%$, abundance near nests = 9.97%, abundance near control points = 11.22%). Together these analyses suggest that Redshank preferentially select *F. rubra* next to and in the wider area around nests.

There was support for the inclusion of annual cattle density and proximity in the *F. rubra* models. The cover of *F. rubra* increased with increasing annual cattle density within 1m and 10m of nests (Fig. 3.3i; Table 3.4, analysis i, $\Delta AIC = 3.87$ on removal of annual cattle density from the model). The percentage cover of *F. rubra* ranged from $71 \pm 30\%$ within 1m of nests to $63 \pm 26\%$ within 10m of nests ($\Delta AIC = 176.63$ upon removal of proximity from the model). While there was an effect of proximity on the % cover of *E. repens* (Fig. 3.3ii, Table 3.4, analysis ii, $\Delta AIC = 10.71$ on removal of proximity from the model), there was no clear evidence of a decline in *E. repens* cover with increasing annual cattle density ($\Delta AIC = +2.15$ on addition of annual cattle density to the model).

3.4 Discussion

To investigate Redshank nest site selection across saltmarshes with a gradient of low intensity livestock densities, this study aimed to identify nesting habitat use as a measurement of nesting habitat selection, and how this varies with different levels of livestock density. This was studied on six saltmarshes grazed between 0 and 0.55 cattle $\text{ha}^{-1} \text{y}^{-1}$, by characterising vegetation height, and species composition at nests and control locations on a small spatial scale. Redshank optimally selected nest locations in the tallest vegetation available but grazing limited the availability of long vegetation, suggesting that grazing caused Redshank to nest in less optimal vegetation. Redshank were found to also selected nest locations dominated by the grass *Festuca rubra*, with the percentage cover of this species in the area surrounding the nest increasing in saltmarshes with higher livestock densities.

3.4.1 *Nest site selection and grazing*

Our results support hypotheses one and two, as Redshank nest vegetation was higher at nests than control locations and vegetation height at Redshank nests was lower in saltmarshes with higher livestock densities. This trend was consistent at the nest (maximum and mean) and in the area surrounding ($\leq 1\text{m}$ and $\leq 10\text{m}$) the nest. Results add support to the findings of Mandema *et al.* (2014), that Redshank built their nests in areas with relatively tall vegetation and high variation in canopy height, which was more abundant in low livestock densities of $0.25 \text{ cattle ha}^{-1} \text{ y}^{-1}$ than at $0.50 \text{ cattle ha}^{-1} \text{ y}^{-1}$.

Our results partially support hypothesis three as the vegetation species composition did vary between nests and control locations, with more *F. rubra*, found at Redshank nests, however slightly more *P. maritima* was found at control locations than Redshank nests and the cover of *E. repens* was similar between nests and controls. This suggests that Redshank select nests within communities of *F. rubra*, which is a species of high-mid saltmarsh (Adam 1990) *P. maritima* is associated with low and low-mid saltmarsh (Adam 1990; Boorman 2003) and was more abundant at control sites than nests, suggesting Redshank are selecting grassy areas at higher elevation saltmarsh. These findings are consistent with those of Thyen and Exo (2003), who found that although Redshank can nest amongst *P. maritima*, more nests were found amongst *F. rubra*. However 45% of the 83 nests recorded by Thyen and Exo (2003) were in *E. repens*. This difference could be due to grazing management, as 70% of the saltmarsh in their study was ungrazed, and *E. repens* is more abundant on ungrazed saltmarshes (Andresen *et al.* 1990). Our results partially support hypothesis four, as nests exposed to higher livestock densities had a higher percentage cover of *F. rubra* but annual cattle density did not affect the percentage cover of *E. repens*. Nesting amongst *E. repens* appears to lead to higher breeding success than *F. rubra* due to high vegetation cover and concealment from predators (Thyen & Exo 2003). As the proportion of *F. rubra* found at Redshank nests increases with grazing, this could indicate that grazing causes Redshank to nest in lower quality vegetation types.

3.4.2 Habitat quality

This study highlights a negative effect of livestock grazing for Redshank: birds optimally select longer vegetation but are selecting shorter vegetation for nest sites on grazed saltmarshes. *F. rubra*, is more abundant on grazed saltmarshes but can lead to lower rates of nest survival (Andresen *et al.* 1990; Thyen & Exo 2003). Throughout Europe, waders have been found to suffer high nest predation rates (Macdonald & Bolton 2008). Grazing decreases vegetation cover and vegetation heterogeneity, which makes it easier for predators to find nests (Baines 1990; Hart *et al.* 2002). Even in lightly grazed saltmarshes, grazing induced nest mortality is incurred and can arise indirectly through grazer modification of landscapes that accelerate predation risks (Sharps *et al.* 2015). Results of the current study may explain the relationship between livestock density and nest predation found by Sharps *et al.* (2015) as nests in shorter vegetation are more obvious to predators. Potential Redshank nest predators include aerial predators such as corvids (*Corvus spp.*), gulls (*Larus spp.*), and ground predators, such as Red Foxes (*Vulpes vulpes*), Stoats (*Mustela erminea*) and non-native American Mink (*Neovison vison*) (Smart 2005). It is likely that nests in short *F. rubra* are more vulnerable to both ground and aerial predators than those in longer vegetation (Thyen & Exo 2003; Maier 2014). Redshank do not actively defend nests against predators but can take advantage of the nest defence behaviour of other waders, such as Lapwing *Vanellus vanellus* and Black-tailed Godwit, by nesting in close association with larger colonies of these species (Dyrcz, Witkowski & Okulewicz 1981). However on most British saltmarshes Redshank are the most abundant breeding wader, with other species found in only very low numbers (Cadbury, Green & Allport 1987). This places emphasis on optimising grazing to benefit nesting Redshank.

3.4.3 Limitations

Age of nest when measured ranged between 3 and 24 days (Mean = 16, ± 7 sd). Therefore it is possible that nests could have been grazed after Redshank nest selection, but before they were measured. However, this was accounted for by taking a paired approach of recording vegetation at nests and control sites, and the inclusion of nest age as a predictor in all vegetation height models. Measuring nests of unequal

age did not affect our conclusions as results showed that the age variable was not necessary in the models.

This study investigated small scale selection of nesting habitat, therefore did not consider trends on a wider landscape scale. Breeding distribution changes between 1968 - 2011 (Balmer *et al.* 2013) suggest that Redshank populations are generally declining across the UK, but that there are a small number of increases in the uplands of northern England, the river systems of the English midlands and the highlands and islands of Scotland. However, as Redshank show both high breeding site fidelity and natal philopatry (Thompson & Hale 1989) it is unlikely that this represents wide scale habitat selection, and more likely to suggest that areas with increasing populations are due to higher breeding success. As this study was based on six saltmarshes of the Ribble estuary, it is not clear if these trends can be generalised and applied to other locations. However, it is likely that similar trends would be found on other north western European saltmarshes subject to grazing, as the climax community is usually dominated by Couch grass (Rodwell *et al.* 2000; Bakker, Bos & De Vries 2003). Therefore, our conclusions could potentially be applied to other European saltmarshes.

SIMPER results for *E. repens* should be treated with some caution as they showed some dissimilarity between nests and control points, which was not as clear when only considering average abundance data (Table 3.3). There were a high number of either zero values or high values (close to 100%) in the *E. repens* percentage cover data, which could account for these results. These percentage cover values are likely to be explained by the patchy nature of *E. repens* distribution on low intensity cattle grazed saltmarshes (Andresen *et al.* 1990). However, overall, the conclusion that Redshank select *F. rubra*, and that % cover of this species increases with grazing density is unaffected by uncertainty over *E. repens* results.

When calculating livestock density our intention was to use a procedure similar to that used by land managers. Methods can vary but management guidelines e.g. Adnitt *et al.* (2007) tend to focus on simple calculations of either numbers of livestock or livestock units (1 livestock unit = 1 adult cattle) divided by the size of the site in hectares, taking into account the part of the saltmarsh that is accessible to livestock

and the number of months animals are present. This approach has a known constraint – that grazing across a saltmarsh can be localised closer to the sea wall (Patterson & Burrows 1998; Sharps *et al.* 2015). Calculating precise grazing pressure to account for spatial variation was beyond the scope of this study. However, as Redshank nest in the high-mid marsh (Norris *et al.* 1997), this tends to be closer to the sea wall, and is therefore more accessible to livestock (Mandema *et al.* 2013; Nolte *et al.* 2014; Sharps *et al.* 2015). Therefore it is likely that Redshank nests may be exposed to higher grazing than intended by land managers and the conclusions of this study are unlikely to be affected by non-homogenous grazing.

3.4.4 Implications

All saltmarshes included in this study were low grazing intensity conservation managed areas. However, this study suggests that Redshank preferentially select nest sites in taller vegetation. This height is limited by grazing, which results in Redshank selecting nesting sites in shorter vegetation even across a range of lightly grazed saltmarshes with ≤ 0.55 cattle $\text{ha}^{-1} \text{y}^{-1}$. This is unlikely to represent the optimal grazing levels for Redshank breeding due to increased probability of nest loss to predation and cattle trampling (Sharps *et al.* 2015). Translating the UK Environment Agency light grazing guidelines of 0.7 - 1 young cattle ha^{-1} between April and October (Adnitt *et al.* 2007) to measurements used in this study would mean an annual cattle density of around 0.4 - 0.5 ha^{-1} . The saltmarsh with the highest annual cattle density in this study was 0.55 cattle ha^{-1} . This suggests that the UK Environment Agency definition of light grazing is too intensive for breeding Redshank. However it is unclear if the solution lies in simply reducing grazing levels, or employing more innovative approaches. Further investigations into the ideal grazing management practices for Redshank may be necessary. In lowland wet grasslands, commencing grazing after the end of July has been shown to increase productivity in Redshank and other waders, by preventing nest trampling (Green 1986). Alternatively, a rotational grazing regime where saltmarshes are grazed and left ungrazed in alternate years may improve breeding success by eliminating nest trampling in the ungrazed year, therefore increasing breeding success. Novel techniques may also be necessary to increase heterogeneity

in livestock distribution, but further investigations into possible interventions may be necessary.

3.5 Tables

Table 3.1: Description of saltmarsh field sites and numbers of nests found per saltmarsh.

In the case of saltmarsh C, a large network of creeks prevented grazer access to part of the saltmarsh, so only the area accessible to livestock was included in this study and the annual cattle density calculations.

Saltmarsh	Area (ha⁻¹)	Annual cattle density (ha⁻¹)	No. of nests
A	237	0.17	10
B	127	0.08	5
C	704	0.55	8
D	63	0.25	9
E	58	0.43	8
F	257	0.00	5

Table 3.2: Results of vegetation height analyses.

' ΔAIC_c ': the difference between the model in question, and the top model. $\Delta AIC_c < 2$ suggests both models are plausible (Burnham & Anderson 2002). Akaike weights measure the support for each model, and represent the ratio of ΔAIC_c values for each model relative to the candidate set. ACD = annual cattle density. Type = nest or control.

Analysis	Model	AIC _c	ΔAIC_c	Akaike weight
i) Mean vegetation height at nest	ACD + type	335.20	0.00	0.41
	ACD * type	337.00	1.73	0.17
	type	337.20	1.92	0.16
	ACD + type + age	337.40	2.13	0.14
	ACD * type + age	339.20	3.92	0.06
	type + age	339.30	4.06	0.05
	ACD	353.10	17.86	0.00
	intercept	355.10	19.81	0.00
	ACD + age	355.20	19.97	0.00
	age	357.20	21.92	0.00
ii) Maximum vegetation height at nest	ACD + type	353.10	0.00	0.44
	ACD + type + age	354.50	1.43	0.22
	ACD * type	355.50	2.33	0.14
	type	356.30	3.19	0.09
	ACD * type + age	356.90	3.82	0.07
	type + age	357.80	4.63	0.04
	ACD	413.50	60.39	0.00
	ACD + age	415.40	62.29	0.00
	intercept	416.60	63.47	0.00
	age	418.50	65.39	0.00
iii) Mean Surrounding area vegetation height (1m)	ACD + type	297.10	0.00	0.43
	type	299.10	1.97	0.16
	ACD + type + age	299.20	2.01	0.16
	ACD*type	299.40	2.25	0.14
	type + age	301.10	3.98	0.06
	ACD * type + age	301.50	4.32	0.05
	ACD	334.20	37.02	0.00
	ACD + age	336.20	39.02	0.00
	intercept	336.20	39.03	0.00
	age	338.20	41.02	0.00
iv) Mean Surrounding are vegetation height (10m)	ACD + type	299.60	0.00	0.43
	ACD + type + age	300.40	0.78	0.29
	ACD*type	301.90	2.30	0.14
	ACD * type + age	302.70	3.14	0.09
	type	304.90	5.34	0.03
	type + age	305.90	6.32	0.02
	ACD	310.10	10.50	0.00
	ACD + age	310.80	11.22	0.00
	intercept	315.50	15.89	0.00
	age	316.40	16.83	0.00

Table 3.3: Average abundance of dominant species at Redshank nests and control points based on SIMPER analyses.

The abundance is the average % cover found at Redshank nests or control locations. The average dissimilarity shows the difference between nests and control locations using Bray-Curtis dissimilarities. The percentage contribution shows the average contributions of each species to the overall Bray-Curtis dissimilarity. Prox. level = proximity level, meaning distance from the nest or control point (levels $\leq 1\text{m}$, $\leq 10\text{m}$).

Analysis	Species	Abundance (nests)	Abundance (control points)	Average dissimilarity \pm sd	% contribution
i) Prox. level $\leq 1\text{m}$	<i>Festuca rubra</i>	71.67%	41.04%	22.05 \pm 1.37%	35.28%
	<i>Elymus repens</i>	11.78%	9.98%	9.18 \pm 0.62%	14.70%
	Bare ground	2.22%	17.22%	8.51 \pm 0.68%	13.61%
	<i>Triglochin maritima</i>	1.27%	9.33%	4.81 \pm 0.44%	7.70%
	<i>Puccinellia maritima</i>	1.78%	7.78%	4.45 \pm 0.40%	7.13%
	<i>Aster tripolium</i>	4.07%	6.91%	3.90 \pm 0.78%	6.24%
	<i>Atriplex Prostrata</i>	3.53%	3.96%	2.79 \pm 0.66%	4.46%
	<i>Cochlearia officinalis</i>	2.82%	2.67%	2.49 \pm 0.32%	3.98%
ii) Prox. level $\leq 10\text{m}$	<i>Festuca rubra</i>	59.18%	39.11%	20.04 \pm 1.42%	30.95%
	Bare ground	7.15%	18.66%	9.66 \pm 0.79%	14.92%
	<i>Elymus repens</i>	9.97%	11.22%	8.49 \pm 0.65%	13.12%
	<i>Puccinellia maritima</i>	4.98%	8.36%	5.84 \pm 0.47%	9.02%
	<i>Triglochin maritima</i>	3.98%	8.71%	5.34 \pm 0.55%	8.24%
	<i>Aster tripolium</i>	5.58%	7.19%	4.49 \pm 0.72%	6.94%
	<i>Atriplex prostrata</i>	5.09%	5.60%	4.21 \pm 0.59%	6.50%
	<i>Glaux maritima</i>	2.62%	2.12%	2.07 \pm 0.40%	3.20%

Table 3.4: Results of percentage species cover analyses.

' ΔAIC_c ': the difference between the model in question, and the top model. $\Delta AIC_c < 2$ suggests both models are plausible (Burnham & Anderson 2002). Akaike weights measure the support for each model, and represent the ratio of ΔAIC_c values for each model relative to the candidate set. ACD = annual cattle density. Type = nest or control point.

Analysis	Model	AIC_c	ΔAIC_c	Akaike weight
i) Percentage cover of <i>Festuca rubra</i> .	ACD + proximity	2871.30	0.00	0.64
	ACD * proximity	2873.00	1.68	0.27
	proximity	2875.20	3.87	0.09
	ACD	3047.90	176.63	0.00
	intercept	3051.90	180.57	0.00
ii) Percentage cover of <i>Elymus repens</i> .	proximity	605.60	0.00	0.65
	ACD + proximity	607.70	2.15	0.22
	ACD * proximity	608.80	3.22	0.13
	intercept	616.30	10.71	0.00
	ACD	618.40	12.88	0.00

3.6 Figures

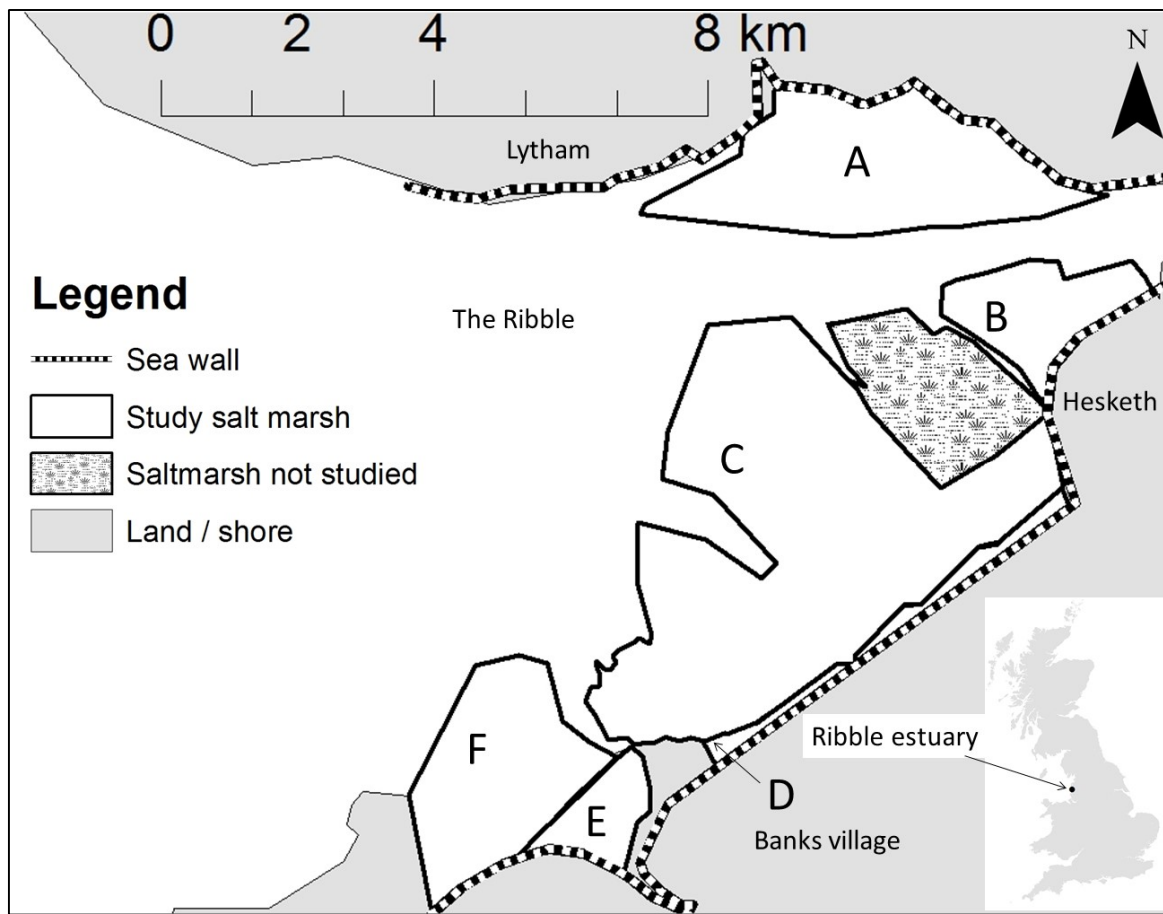


Figure 3.1: Ribble Estuary map showing the study saltmarshes (A-F) and geographic location.

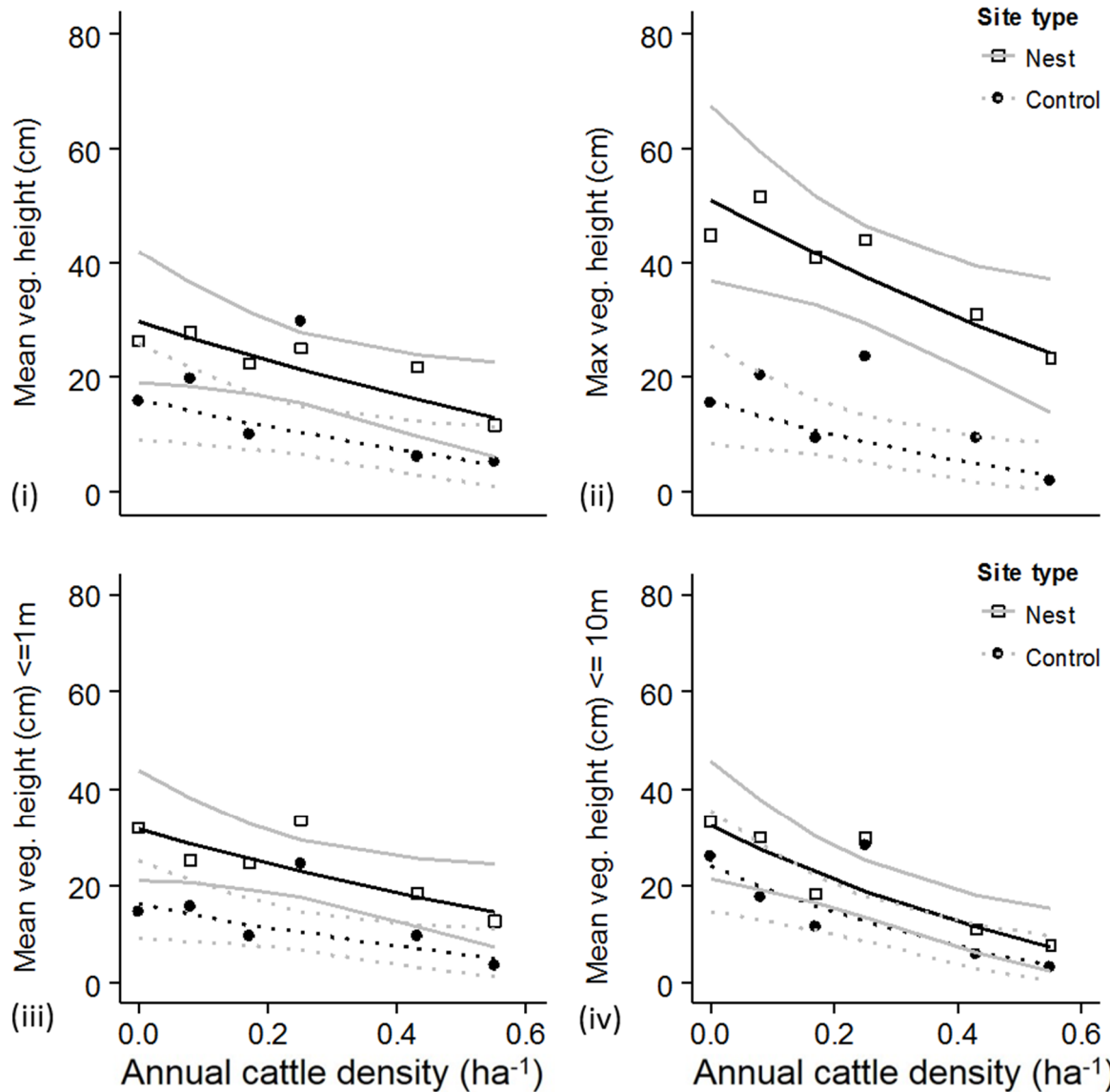


Figure 3.2: Effect of grazing on saltmarsh vegetation height and Redshank nest selection.

Top = vegetation height at nest and control points, mean (i) and maximum (ii). Bottom = vegetation height (iii) in the immediate vicinity ($\leq 1\text{m}$) of Redshank nests and control points (iv) in the wider area ($\leq 10\text{m}$) around Redshank nests and control points. Black lines are back-transformed model fitted lines, while grey lines are 95% confidence intervals.

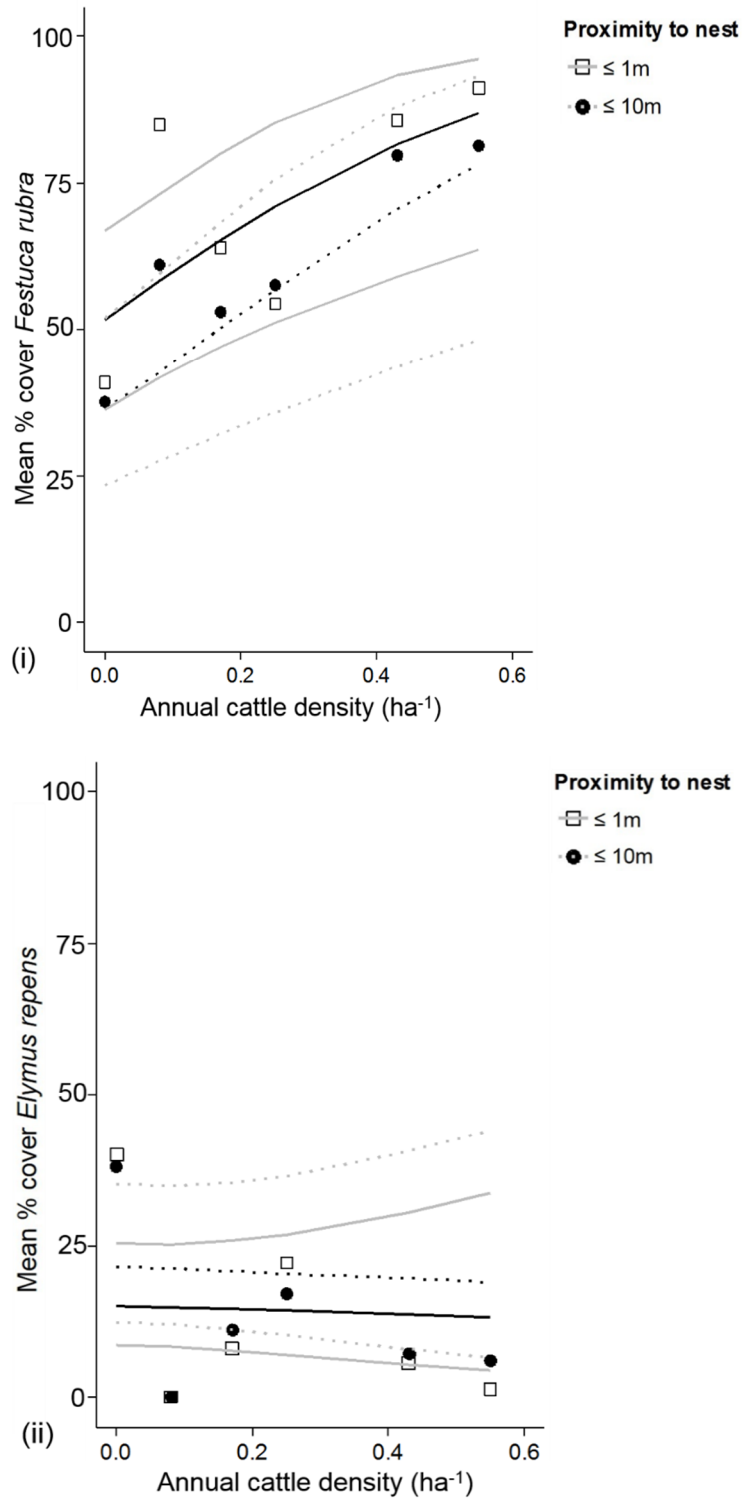


Figure 3.3: Effect of grazing and proximity to the nest (levels $\leq 1\text{m}$ and $\leq 10\text{m}$) on percentage cover of (i) *Festuca rubra* (ii) *Elymus repens*. Black lines are back-transformed model fitted lines, while grey lines are 95% confidence intervals.

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4 Nest trampling and ground nesting birds: can grazing density calculations assume homogenous livestock distribution on saltmarshes?

4.1 Abstract

The Common Redshank *Tringa totanus* breeding population on British saltmarshes has reduced by > 50% since 1985, with declines linked to changes in grazing management. Conservation initiatives have encouraged low-intensity grazing of < 1 cattle ha⁻¹ but even light grazing can lead to high rates of nest trampling, and livestock distribution may not be homogenous. This study investigated the possible drivers of spatial and temporal variation in livestock distribution and quantified the effects on avian nest loss due to trampling, to determine if Redshank nesting areas are exposed to higher trampling risk during the breeding season. We asked (a) is the probability of nest loss to trampling higher close to the landward extent of saltmarshes? and (b) does grazing homogeneity increase with time as livestock travel further away from the landward edge of the marsh? This was carried out by using dummy nests and GPS loggers on five lightly grazed saltmarshes, in the Wash estuary, eastern England. Probability of nest trampling was correlated to livestock distribution and was much higher closer to the landward extent of the saltmarsh, where most Redshank breed. The distribution of livestock was highly variable depending on both time in the season and the saltmarsh under study with grazing present on between 3% and 42% of the saltmarsh extent. Increasing grazing homogeneity and changing the timing of saltmarsh grazing to reduce the number of livestock present during the Redshank breeding season may increase Redshank breeding success,

4.2 Introduction

Livestock grazing is a common conservation management technique, used for the preservation of species and communities or to conserve landscapes and ecosystems (WallisDeVries 1998). Guidelines for conservation management tend to treat the distribution of grazing animals as homogenous across a landscape (e.g. Green 1986; Adnitt et al. 2007). However, it is recognised that livestock distribution can vary in space and depends on numerous biotic and abiotic factors (Bailey *et al.* 1996). These

include the distance to drinking water and forage quality and quantity (Bailey 1995). Previous studies on spatial distribution of livestock have focused mainly on economically orientated pasture systems that tend to have a homogenous and species-poor vegetation with universal accessibility (Rook & Tallowin 2003; Putfarken *et al.* 2008).

Many saltmarshes are grazed by livestock for conservation purposes (Boorman 2003; Adnitt *et al.* 2007). North-western European coastal saltmarshes can be defined as areas vegetated by herbs, grasses or low shrubs, bordering saline water bodies (Adam 1990). Saltmarshes typically consist of a limited number of species adapted to regular immersion by the tides, which ranges from a pioneer zone of extremely halophytic plants adapted to regular tidal immersion, through to a marsh largely composed of grassy species at higher elevations (Gray 1992; Boorman 2003). Many north-western European saltmarshes are bounded by man-made vegetation banks to the landward extent of the marshes, with the higher zone virtually absent and the transition zone appearing in a line along the sea wall (Boorman 2003). Due to this zonation, the vegetation is not homogeneous in space. The accessibility of many parts of the saltmarsh to livestock is also limited spatially by tidal channels, creeks and occasionally by tidal height.

Saltmarshes provide habitat for various breeding bird species including waders such as Common Redshank (*Tringa totanus*), Northern Lapwing (*Vanellus vanellus*) and Eurasian Oystercatcher (*Haematopus ostralegus*) and passerines such as Eurasian Skylark (*Alauda arvensis*) and Meadow Pipit (*Anthus pratensis*) which tend to nest in the higher elevation saltmarsh zones (Greenhalgh 1971; Allport, O'Brian & Cadbury 1986; Mandema *et al.* 2014; Van Klink *et al.* 2014). Of the breeding birds on British saltmarshes, it is the Redshank that is both nationally and internationally important (Cadbury, Green & Allport 1987; Brindley *et al.* 1998). Approximately 50% of the British breeding population occur on saltmarshes (Brindley *et al.* 1998) and Britain supports over 18% of north-west Europe's estimated 100,000 – 172,000 breeding Redshank pairs (Piersma 1986; Batten *et al.* 2010).

Recent surveys of British saltmarshes by Malpas *et al.* (2013) found a 52.8% reduction in nesting Redshank between 1985 and 2011. Livestock grazing can reduce Redshank nest survival both through nest trampling and indirectly through grazer modification of

habitat that may increase predation risks (Beintema & Muskens 1987; Sharps *et al.* 2015). Grazing also has the potential to alter saltmarsh suitability for Redshank as it reduces vegetation height and (depending on livestock density) can create or limit the availability of vegetation patches, which Redshank use for nesting (Mandema *et al.* 2014; Sharps 2015). Norris *et al.* (1997) recommended stocking densities of ~1 cattle ha⁻¹, in order to create the patchy vegetation sward used by Redshank (Norris *et al.* 1997). However, Sharps *et al.* (2015) found that risk of nest loss to livestock trampling increased from 16% at 0.15 cattle ha⁻¹ to 98% at 0.82 cattle ha⁻¹. The study also suggested that Redshank nests closer to the landward extent of the saltmarsh may be more vulnerable to nest trampling by livestock.

Redshank typically nest in vegetation communities associated with the high-mid marsh (Norris *et al.* 1997), which is usually closer to the landward side of the saltmarsh. Grazing pressure can be higher in these areas and lower in the pioneer zone which is closer to the seaward side of the saltmarsh (Patterson & Burrows 1998; Esselink, Fresco & Dijkema 2002). Little is known about how this non-homogenous distribution of livestock affects nest trampling of birds or how this is influenced by biotic and abiotic factors. Livestock distribution tends to be higher close to sources of fresh drinking water, due to their high intake requirements (Ali, Goonewardene & Basarab 1994; Ganskopp 2001). On saltmarshes these effects may be stronger, as there are typically no natural sources of freshwater and limited numbers of drinking troughs (typically 1-3 per saltmarsh placed at the landward extent due to practicalities of piping water). In temperate climates daily water intake can be in the region of 20 to 40 litres per animal per day depending on age, body size (weight), stage of production and the air temperature (Arias & Mader 2007). Cattle can spend up to 45% of their time grazing and 25% of their time walking (Hughes & Reid 1951). Diet choice of grazing animals is based on maximising energy intake and the quality and availability of forage intake (Vulink & Drost 1991). This may suggest that livestock distribution on saltmarshes can vary with time as forage quality may attract them to the different vegetation found in different zones, selecting preferential vegetation earlier in the season (Nolte 2014). Livestock are more likely to forage on previously grazed vegetation as it regrows (McNaughton 1984; Nolte *et al.* 2014). Therefore, livestock distribution is likely to vary with time, but relationships (e.g. with distance travelled) may not be linear due to depletion of preferential forage types or the need to return to drinking troughs more

often in warm weather. Management guidelines recommend starting grazing in April (Adnitt *et al.* 2007), this coincides with the April – July Redshank nesting season (Green 1984).

The aim of this study is to investigate the drivers of the spatial and temporal variation in livestock distribution and quantify effects on avian nest loss due to trampling, to see if Redshank nesting areas are exposed to higher trampling risk during the breeding season. Identification of the drivers of the distribution of livestock grazing may allow improvements to the management of grazing management that will maintain positive effects on the vegetation structure without the negative effects of trampling of nests. We hypothesise that (1) the probability of nest loss to trampling is higher in parts of saltmarshes subject to more grazing; (2) on saltmarshes exposed to higher grazing, the probability of nest loss to trampling increases closer to the landward extent of the saltmarsh; (3) grazing homogeneity and furthest distance travelled by livestock increases with time; (4) grazing pressure is higher closer to the landward extent of a saltmarsh; (5) livestock use of different saltmarsh zones varies with time.

4.3 Methods

This study was carried out on five saltmarshes of the Wash estuary with grazing intensities that varied between 0.11 – 0.50 cattle ha⁻¹ (Table 4.1, Figs. 4.1-4.5). To investigate the drivers of the spatial and temporal variation in livestock distribution we used GPS loggers placed on cattle on saltmarshes. To quantify effects on avian nest loss due to trampling we used dummy nests.

4.3.1 Field site

Much of the Wash estuary is managed as a National Nature Reserve and supports over 4000ha of saltmarsh (10% of the total UK saltmarshes), (Burd 1989; Murby 1997). The estuary is bounded by a river channel on the estuarine side and by a sea wall flood defence at the landward side. The vegetation is typical of saltmarshes on the east coast of the UK. The pioneer zone at the seaward boundary is composed of tussocks of *Spartina anglica* and *Salicornia europaea*. The saltmarshes are dominated

by *Puccinella maritima* and *Festuca rubra* communities, which form a short turf with occasional grassy tussocks across most of their extent where grazed by livestock. In areas where livestock grazing is limited or absent, *Elytrigia atherica* dominates, mainly through the central and upper parts of the saltmarsh extending on to the vegetated flood defences (Hill 1988; Murby 1997).

4.3.2 False nest study

Two nest trampling experiments were conducted. Firstly, to see if the probability of nest loss to trampling is higher in parts of saltmarshes subject to more grazing, and to check that the GPS positions of tagged cattle are correlated to the trampling pressure on the saltmarsh, we ran a dummy nest experiment. This involved using standard 110mm black clay pigeon shooting targets. These discs have a similar diameter to shorebird nests (e.g. 4 Redshank eggs approximately 45-48mm per egg), and like eggs they break if stepped on by livestock (Jensen, Rollins & Gillen 1990; Mandema *et al.* 2013). Thirty positions were selected at random across one of the saltmarshes to cover the full range of distances from the sea wall (minimum distance between points = 50m). At each of the 30 plots, 9 discs were placed in grids of 9m x 9m, with 3 metres between each disc. The precise location of each disc was recorded using a Leica Viva GS08 Global Navigation Satellite System (accuracy 60mm). Discs were placed on the saltmarsh before the cattle were introduced on 22/5/13 and checked after 14 and 28 days, when they were recorded as broken, intact or missing. When checking discs after 14 days missing/broken discs were replaced with a new disc and all debris was removed. When checking discs after 28 days all intact discs and debris were removed. These data were used to calculate a trampling probability for each of the 30 plots over the 28 days of the study and correlated to the GPS collar data relating to this period. Due to a suggested non-linear relationship between the probability of nest trampling and cattle density, a binomial Generalised Additive Model (GAM) was used, with the statistical program R (R Core Team 2013). Following Zuur *et al.* (2009), the residuals of the model were plotted against the spatial co-ordinates of the discs to test for spatial autocorrelation. Plots showed no clear clustering of (high or low) residuals. A variogram of the model residuals was also created by running the model as a Generalised Additive Mixed Model (GAMM) and using the 'Variogram' function in

the R package nlme (Bates *et al.* 2014) on the residuals. As the points in this plot were scattered along a horizontal line, independence could be assumed (Zuur *et al.* 2009). Therefore the final model was a GAM, with a smoothed term for cattle density, and no additional terms to allow for spatial autocorrelation.

The second dummy nest experiment aimed to identify if the probability of nest trampling increases closer to the landward extent (sea wall) of saltmarshes with more grazing. Discs were placed on five saltmarshes (A-E, Table 4.1, Fig. 4.1) between mid-April and May 2014, representing the time period in which Redshank lay their eggs. Transects were walked from the seaward edge of the saltmarsh to the landward edge and dummy nests ($n = 464$) were placed at points along the transect (mean distance between points = 52.3m, minimum = 8m). Between 2 and 8 transects were carried out per saltmarsh, depending on the size of the marsh. Dummy nests were relocated in late May and June using a handheld GPS and classified as broken or intact. Any nests not trampled upon this revisit were relocated a second time through July and their status recorded again, before being removed from the saltmarsh. A small number of nests ($< 1\%$) could not be successfully relocated. If this was in an area where the vegetation had been either churned up or completely flattened, and there were clear cattle footprints in the soil, nests were recorded as trampled. In all other cases where dummy nests could not be relocated, they were removed from the analysis. Nest trampling probability was computed in relation to the distance between the nest, sea wall, and livestock congregation point, using the nest survival model of Program MARK (Dinsmore, White & Knopf 2002).

The vegetation height at the nest was also included in the analysis to account for dummy nests placed in vegetation of different heights. Program MARK uses numerical maximum-likelihood techniques and computes a quasi-likelihood AIC value (White & Burnham 1999). This enabled the selection of the variable(s) (vegetation height, distance to: the nearest creek; the landward extent of the saltmarsh; livestock congregation point; cattle density, or no recorded variable) that most strongly accounted for variation in nest survival. This method does not assume that nest visits are made at regular intervals (Johnson 2007). To estimate true probability of nest loss over the 24 day Redshank incubation period (Green 1984), the daily survival rate was raised to the power of 24 following Rotella, Cooch & White (2009) and converted from

a survival to a mortality probability. Variance was calculated using the Delta method (Powell 2007) from which confidence intervals were calculated, following Armstrong *et al.* (2002).

Program MARK estimated all possible models. Model outputs were compared using the Akaike Information Criterion (corrected for small sample size, AICc) (Burnham & Anderson 2002). The model with the lowest AICc value is the most parsimonious model. Models differing 2-7 AICc from the most parsimonious model have little empirical support; those with $\Delta AICc > 10$ have no support. Akaike weights, (the relative likelihood of the models, i.e. $\exp(-0.5 * \Delta AIC)$ for each model, divided by the sum of these values across all other models) were used to measure the support for each model. To check the model fit, we obtained a \hat{C} value from Program MARK for the global model. Following Lebreton *et al.* (1992) if the model structure is adequate, \hat{C} is expected not to exceed 3.

4.3.3 GPS tracking

To investigate the drivers of the spatial and temporal variation in livestock distribution we used GPS loggers placed on cattle on the study saltmarshes. These were constructed from SiRFstarIV GSD4e-T GPS processor chipsets (recorded accuracy = 2.5m) and attached to cattle using a neck collar. Between May and October 2013, 12 young cattle across two saltmarshes (saltmarshes A and B, Table 4.1) were fitted with GPS loggers. This was repeated with eight young cattle on two different saltmarshes between April and August 2014 (saltmarshes D and E). GPS loggers were programmed to log a position every 20 minutes when satellite signals were available. They were retrieved at the end of the grazing season. Due to battery life some collars stopped earlier than planned, however approximately 50% of the collars per saltmarsh logged the entire period. The dates each of the saltmarshes were logged are noted in Table 4.1. Arc-GIS 10.1 was used to produce a grid over each saltmarsh, and to count the number of GPS records that fell into each 50 x 50m grid cell, on each day. To obtain an estimate of the true grazing per cell per day, this count of GPS records was increased to account for the size of the herd using the formula:

Grazing = (size of herd/number of GPS loggers active) * number of GPS records per cell.

This was converted to a percentage to account for unequal numbers of GPS records per day. The percentage of grid cells needed to account for all of the grazing was then calculated for each week, as a measure of grazing homogeneity.

In order to investigate if the homogeneity of livestock grazing increases with time, we used a general linear model (GLM) in R. As a measure of homogeneity, this included the percentage of cells containing all of the grazing (response variable), saltmarsh identity (A-D) and time (weeks) as explanatory variables. The response variable was \log_{10} transformed to deal with uneven spread in the residuals. A quadratic term for time (week^2) was included in the model to allow for the possibility of a curved relationship between time and the grazing distribution. An interaction between saltmarsh and time (and saltmarsh and week^2) was also included in the global model to determine if the relationship between the percentage of cells containing all of the grazing and time varied between saltmarshes. As measurements were recorded at regular temporal intervals (time series data), they may be temporally autocorrelated. This was tested by running the final optimal model using a generalised least squares (GLS) regression in R following Zuur *et al.* (2009). Model residuals were then tested using an autocorrelation function (ACF) plot. As this suggested some temporal autocorrelation, the final model run was a generalised least squares model, including a correlation structure term (corAR1). As we also aimed to investigate if the maximum distance travelled by livestock varies with time, for all grid cells visited by livestock, the 95th percentile of the distance from the sea wall was calculated, for each week. We used a GLS model which included distance travelled (95th percentile, metres) as the response variable and saltmarsh identity and time (weeks) as explanatory variables. A quadratic term for time (week^2), an interaction between saltmarsh and time, and saltmarsh and week^2 and a correlation structure term (corAR1) were also included in the global model.

To investigate if grazing pressure is higher closer to the landward extent of a saltmarsh, 'distance to the sea wall' was calculated from the centre of each grid cell. To obtain estimates of livestock density m^{-2} , the area of saltmarsh per grid cell was

calculated by subtracting the area of any creeks and any area which fell outside of the saltmarsh boundary. The percentage of true grazing per day was then divided by the area of saltmarsh per grid cell. Due to the accuracy of the GPS chipsets (recorded accuracy = 2.5m) only grid cells which contained saltmarsh > 6.25m² were included in the analysis. In order to reduce effects of spatial autocorrelation, and the high number of zeros in the data (grid cells which contained no grazing), we chose a set of integers for the distance to sea wall: (e.g. 0, 100, 200, 300, 400, 500, 600, 700, 800, 900, 1000 metres) which included the majority (89%) of all distance values. For each of these distances, we calculated the mean livestock density m⁻² for cells at a distance of >20 and < 20m of the whole number, (e.g. for 100m, the mean grazing was taken for cells between 80 and 120m from the sea wall). We fitted a GLM including mean livestock density m⁻² as the response variable, and as explanatory variables, saltmarsh identity (A-D), distance to the sea wall (metres), and an interaction between saltmarsh identity and distance to the sea wall. An additional quadratic term (distance to the sea wall²) was also included, as visual investigation of the data suggested a possible non-straight line relationship between grazing and distance to the sea wall. The response was square root transformed to ensure normality of residuals and to reduce heteroscedasticity.

4.3.4 Zonation assessment

In order to investigate seasonal trends in livestock use of different saltmarsh zones, we mapped the zonation of each saltmarsh (pioneer, low, mid, upper, drift line, transition) in the field based on vegetation communities recorded. This was then validated using aerial photographs to create a GIS layer of zonation for each saltmarsh. Cattle at saltmarsh D also had access to a non-saltmarsh paddock behind the sea wall so to account for this, we included a zone called 'non-saltmarsh'. We then identified the areas of each grid cell that fell within each of the zones. Where a grid cell fell within more than one zone, we used the zone that occupied the highest area of the grid cell. For each week, the grazing within each zone was summed and converted into a percentage of the total grazing on the saltmarsh in that week. For each zone we used a GLM with the percentage of the grazing as the response variable and saltmarsh identity and time (weeks) as explanatory variables. Where necessary,

the response variable was transformed (square root or $\log_{10}+1$) to ensure normality of residuals and reduce heteroscedasticity. A quadratic term for time (week²) and an interaction between saltmarsh and time (and saltmarsh and week²) were also included. Data were tested for temporal autocorrelation by running the final optimal model for each zone, using generalised least squares and tested using ACF plots. There was no evidence of temporal autocorrelation.

4.4 Results

The probability of nest trampling increased from zero where no cattle were recorded to 100% with >1 livestock GPS record m⁻² (Fig. 4.6, R²=0.72, edf=1.98, Ref df=2, Chi sq.=1756, p<0.001 for smoothed cattle density term). The fitted model shows that the probability of trampling increases with grazing, but levels out as trampling probability approaches 100%. This confirms that the GPS positions of tagged cattle are correlated to the nest trampling probability and that the probability of nest loss to trampling is higher in parts of saltmarshes subject to more grazing. Figures 4.2-4.5 show all GPS points recorded for each of the saltmarshes.

Nest trampling probability was highest closer to the sea wall at saltmarshes with higher cattle densities. In the saltmarsh grazed at 0.5 cattle ha⁻¹, 18 of 20 (90%) dummy nests placed between 0 and 250m of the sea wall were trampled, compared with 5 of 34 (15%) dummy nests placed between 500 – 750m from the sea wall in the saltmarsh grazed at 0.1 cattle ha⁻¹ (Fig. 4.7, Table 4.2, AICc weight = 0.34, Δ AICc = 239.62 when this model was compared with the constant survival model without explanatory variables). There was clear evidence for an interaction between seasonal cattle density and distance to the sea wall, as removal of the interaction from the top model resulted in a Δ AICc of 14.4. There was no strong evidence for an effect of distance to the nearest creek (Δ AICc of 0.56 on addition of 'creek' to the top model), distance to the nearest livestock congregation point (Δ AICc of 0.94 on addition of 'DCP' to the top model), or vegetation height (Δ AICc of 1.92 on addition of 'Height' to the top model).

Both the homogeneity of livestock distribution, and the maximum distance travelled by livestock increased with time, and then decreased again. In saltmarsh B the highest

level of homogeneity occurred in September at 42% of the available saltmarsh, and in saltmarsh D this occurred in week 7 at 30% of the available saltmarsh. In saltmarshes A and E homogeneity of livestock distribution was never more than 17% of the available saltmarsh area (Fig. 4.8, Table 4.3, analysis i, week $p=0.72$, week² $p<0.001$, saltmarsh identity $p=0.6$, interaction: saltmarsh identity and week $p=0.02$, saltmarsh identity and week² $p=0.16$). The maximum distance travelled by livestock for example, at saltmarsh B increased from 129m in week four to 1600m in week 20, but decreased to 1189m in week 24. Model results showed that maximum distance travelled by livestock varied between the saltmarshes (Fig. 4.9, Table 4.3, analysis ii, week $p<0.001$, week² $p=0.29$, saltmarsh identity $p = 0.02$, interaction: saltmarsh identity and week $p<0.001$, saltmarsh identity and week² $=0.012$). Results indicate that there was an interaction between saltmarsh and time, suggesting that the maximum distance travelled by livestock does vary with time, but the nature of this relationship varies between the saltmarshes.

Livestock density (m^{-2}) was highest closer to the sea wall at $0.58\pm 0.68m^{-2}$ (mean \pm sd) and lowest further from the sea wall (e.g. $0.02\pm 0.035m^{-2}$ at 700m). This response varied between the different marshes (Fig. 4.10, Table 4.3, analysis iii, DSW $p <0.001$, DSW² $p <0.001$, saltmarsh identity $p<0.001$, interaction: saltmarsh identity and DSW $p <0.001$, saltmarsh identity and DSW² $p=0.27$).

The percentage of the grazing in the grassland transition zone decreased with time, for example at saltmarsh B it decreased from 70% in May to 5% in October (Fig. 4.11, Table 4.4, analysis i, week, $p<0.001$ and week², $p=0.03$, saltmarsh identity $p<0.001$, interaction: saltmarsh identity and week $p=0.72$, saltmarsh identity and week² $p=0.34$). The percentage of the grazing in the drift line zone increased with time, for example at saltmarsh B it increased from 3% in May to 81% in October (Fig. 4.12, Table 4.4, analysis ii, week, $p < 0.001$, week², $p = 0.03$, saltmarsh identity $p<0.001$, interaction: saltmarsh identity and week $p=0.48$, saltmarsh identity and week² $p=0.15$). The percentage of the grazing in the mid-marsh zone showed a strong quadratic relationship with time, for example at saltmarsh B it increased from 27% in May, to 48% in July but then decreased to 9% in October (Fig. 4.13, Table 4.4, analysis iii, week, $p=0.97$ week², $p = 0.001$, saltmarsh identity $p<0.001$, interaction: saltmarsh identity and week $p=0.81$, saltmarsh identity and week² $p<0.001$). The percentage of

the grazing in the low-marsh zone showed no relationship with time but an interaction between time and saltmarsh identity (Fig. 4.14, Table 4.4, analysis iv, $p = 0.16$ for week, $p = 0.6$ for week², saltmarsh identity $p < 0.001$, interaction: saltmarsh identity and week, $p = 0.07$, saltmarsh identity and week² $p = 0.03$). This suggests that livestock use the low-marsh zone at some saltmarshes, but the nature of the relationship between time and the percentage of grazing in the low marsh zone varies between saltmarshes.

There was no grazing in the pioneer zone at saltmarshes A or B saltmarsh and no pioneer zone within the boundary of the study saltmarsh at saltmarsh D. Only data from saltmarsh E were included in the model where there was grazing in the pioneer zone, although even this was minimal. There was no relationship between time and the percentage of grazing in the pioneer zone (Fig. 4.15, Table 4.4, analysis v, $p = 0.77$ for week, 0.85 for week²). At saltmarsh D, cattle had access to a paddock of non-saltmarsh habitat behind the sea wall, which was treated as a separate zone for the purposes of this analysis. The percentage of grazing in this zone decreased from 73% in week 1 to 14% in week 11 demonstrating a strong linear relationship with time (Fig. 4.16, Table 4.4, analysis vi, week $p < 0.001$, week² $p = 0.052$). The quadratic term was not included in the final model for this zone, as there was no convincing evidence to suggest that it was required.

4.5 Discussion

The results of this study suggest that cattle distribution on coastal saltmarshes varies spatially and temporally. Early in the grazing season cattle generally concentrate on higher elevation saltmarsh zones close to the sea wall, and move out further onto the marsh as the season progresses. As breeding waders tend to be concentrated on the higher elevation zones and this breeding coincides with the early period of grazing, this pattern of grazing causes avian nest loss to trampling. Results support hypothesis 1, that the probability of nest loss to trampling is higher in parts of saltmarshes subject to more grazing. Results also suggest support for hypothesis 2, that on saltmarshes exposed to higher grazing, the probability of nest loss to trampling increases closer to the landward extent of the saltmarsh as more dummy nests were trampled closer to the sea wall on saltmarshes with more grazing. Results partially support hypothesis 3

as both the homogeneity of livestock grazing and the furthest distance travelled did increase with time during the wader breeding season, but decreased again after an initial peak. Results support hypothesis 4 as grazing pressure was highest closer to the sea wall on all saltmarshes studied. Results partially support hypothesis 5. Livestock use of different saltmarsh zones varies with time and is highly variable between the different zones with trends found for the higher marsh zones, but not the lower marsh zones, which received very little grazing.

As the probability of nest loss to trampling is higher in parts of saltmarshes subject to more grazing, this demonstrates that GPS records from livestock are a good indicator of nest trampling probability. Furthermore, these results show a concentration of grazing on the zones of the saltmarsh that are most important for breeding Redshank and several other bird species. Both the homogeneity of livestock distribution, and the maximum distance travelled by livestock increased with time, and then decreased again. Cattle are able to memorise spatial locations (Bailey *et al.* 1996; Howery *et al.* 1998). They can learn about habitat structure, and thus inform subsequent foraging decisions (Bailey *et al.* 1989). Therefore, the gradual expansion of the foraging area could reflect cattle becoming familiar with a saltmarsh over the grazing season – as their confidence with their environment increases, they are likely to venture further onto the saltmarsh, however with time it is likely that they begin to learn where the best grazing is and therefore stay in these locations more often. This could also be related to simple food depletion on the higher elevation saltmarsh zones, if cattle are forced to venture further afield during periods of slow vegetation growth. The effects of distance to the sea wall on livestock distribution could either be because the provided fresh water drinking sources were located at points on the sea wall, or because of the proximity to palatable grazing.

Although Redshank nest in the high and mid marsh zones (Norris *et al.* 1997; Sharps 2015). there was no high marsh zone in any of four of the saltmarshes with GPS logged cattle. However, the highest grazing pressure the mid marsh experienced was during the Redshank nesting season (April – July). Therefore, this suggests that Redshank breeding habitat is subject to higher rates of nest trampling during the breeding season. In the grassland transition zone livestock density was initially high and then decreased with time. This zone typically consists of mesotrophic grassland

(Burd 1989; Rodwell *et al.* 2000), which is likely to be highly palatable to livestock (Gardener 1980). This change in distribution with time may have been due to depletion of favourable grasses. On the other hand, in the drift line zone, which is largely composed of the low forage quality *Elytrigia atherica* (Burd 1989; Rodwell *et al.* 2000; Milotić *et al.* 2010) cattle grazing started low and increased with time. In the mid marsh zone, which is typically dominated by the highly palatable *P. maritima* and *F. rubra* (Pehrsson 1988; Burd 1989; Rodwell *et al.* 2000), cattle grazing increased with time and then decreased again. This mid marsh zone is typically further from the fresh water drinking source than the grassland transition zone, which may limit the time spent here by cattle. In the low marsh zone there were differences between the saltmarshes, but no noticeable effect of time on the cattle density in each zone. Vegetation in this zone is typically a transition between the mid marsh and pioneer zones (Burd 1989; Rodwell *et al.* 2000) and therefore may have mixed palatability to livestock. The low amounts of grazing in the pioneer zone may reflect the low forage quality these non-grass species (Hughes 2004; Doody 2008), the high distance from the freshwater drinking source or the high risk of tidal flooding (Adam 1990).

Although this study was based on British saltmarshes, these results suggest the need for similar research on other habitats. Livestock grazing is used to conserve a number of target species and communities in various landscapes and ecosystems (WallisDeVries 1998), this includes heathlands (Bakker *et al.* 1983), and grasslands (Smart *et al.* 2006; Eglington *et al.* 2009). However, it may be expected that livestock distribution would be more homogenous in habitats with a uniform coverage of palatable vegetation, and multiple sources of drinking water than habitats such as saltmarshes and heathlands, which have limited coverage of palatable vegetation. Livestock movements on saltmarshes are also likely to be influenced by the weather, tidal conditions and time of day. By definition, saltmarshes are affected by varying degrees of tidal flooding (Adam 1990). Total immersion of saltmarshes by sea water can occur on the highest tides of the spring neap tidal cycle (Armstrong *et al.* 1985), when livestock are forced to retreat to areas with high elevation such as the sea wall (Jensen 1985). Furthermore, as fresh drinking water is generally provided at the landward extent of the saltmarsh, in hot weather livestock may be forced to return more frequently. This has previously been demonstrated in north American pasture systems, where cattle stay close to their drinking water in the heat of the mid-day sun

(Bailey 1995). Therefore climate, tide and time of day may play a role in the non-homogeneity of livestock grazing, however investigating these effects was beyond the scope of this study. It should also be noted that all saltmarshes included in this study were grazed by young cattle, which is commonplace on British saltmarshes (Adnitt *et al.* 2007), however young cattle may trample more nests (Beintema & Muskens 1987) possibly due to their more lively nature (Ausden 2007). Elements of different grazing regimes employed on different saltmarshes may have a large effect on nest survival. Sheep are more likely to produce shorter vegetation swards than cattle (Green 1986; Beintema and Muskens 1987) and horses are more likely to trample nests (Mandema *et al.* 2013). Further studies may be needed into the distribution of sheep and horses on saltmarshes.

4.5.1 Limitations

The percentage of the herd wearing GPS loggers varied between 3.4 and 10.3%, depending on the saltmarsh. Whilst cattle are a herding animal, they often form small sub-groups, which move independently of the rest of the group (Howery *et al.* 1996; Howery *et al.* 1998). As it was beyond the scope of this study to observe individual cattle behaviour patterns, we assumed that these records were representative of the whole herd. As we logged at least 3 cattle per saltmarsh any effects of sub-herding are likely to be minimal and therefore unlikely to affect our conclusions.

4.5.2 Synthesis and applications

Grazing is generally at its highest on bird nesting areas during the breeding season. The actual livestock density is also much higher in these areas than expected if using standard calculations of cattle density. This study demonstrates that livestock grazing on saltmarshes is not homogenous. Therefore, it is likely high rates of Redshank nest loss to trampling are due in part, to this non-homogeneity of grazing during the breeding season.

However, the results of this work do not suggest that stopping livestock grazing on saltmarshes altogether will result in increased breeding populations of Redshank. It is

widely accepted that light grazing is beneficial to Redshank and is therefore an important part of saltmarsh management (Norris *et al.* 1997; Brindley *et al.* 1998; Norris *et al.* 1998). Cessation of grazing in previously grazed saltmarshes can result in reductions in numbers of breeding Redshank as the vegetation becomes dominated by tall uniform vegetation (Norris *et al.* 1997). Furthermore, livestock grazing in saltmarshes can drive abundance and diversity of invertebrate prey (Ford *et al.* 2013). If UK Environment Agency guidelines are followed, grazed saltmarshes would have livestock present from April until October (Adnitt *et al.* 2007). However, this study suggests that calculations of grazing density which assume homogenous distribution of livestock should not be applied to all saltmarshes. In order to counteract this problem there are several techniques which could help to make livestock distribution more homogenous and reduce nest trampling rates. Firstly commencing grazing earlier may move livestock beyond breeding areas during the nesting season. Secondly, the strategic placement of water troughs further away from breeding areas could naturally restrict livestock movements. Thirdly, installing many more drinking troughs spread around the marsh to even out livestock distribution may be beneficial. It may also be possible to control the homogeneity of grazing by fencing livestock into smaller, more targeted enclosures. A system of fences and gates could be used to more proactively manage livestock distribution, whilst balancing flooding risk to the livestock. However, as livestock would drown (due to tidal flooding) if restricted to the lower marsh zones, this would require considerably more management effort. Also, high amounts of fencing could be impractical on a saltmarsh particularly if trying to use temporary electric fences in parts of the marsh which could be subject to tidal flooding. Furthermore, allowing livestock access to the sea wall and/or additional non-saltmarsh grazing facilities may help to reduce effects of nest trampling.

The non-linear nature of the response of trampling probability to livestock grazing suggests that a rotational grazing regime where saltmarshes are grazed heavily then left fallow in alternate years may improve breeding success by eliminating nest trampling in the fallow year. This would be preferable to leaving saltmarshes ungrazed as breeding Redshank need some grazing to create suitable nesting habitat (Norris *et al.* 1997). However this approach may lead to total nest loss in the grazed year. Alternatively nest trampling could be eliminated by introducing livestock after the breeding season, but grazing more heavily afterwards. In other habitats, commencing

grazing after the end of July has been shown to increase productivity in Redshank and other waders (Green 1986). However, further investigations would be needed to find out the ideal livestock density for maintaining a patchy vegetation landscape needed by Redshank.

In conclusion, this work suggests that the areas of the saltmarsh where Redshank breed are much more intensively grazed than intended by land managers, due to the non-homogeneous distribution of livestock. This results in high nest mortality due to nest trampling, therefore changes in grazing management on saltmarshes may be necessary.

4.6 Tables

Table 4.1: Saltmarshes used in this study, showing seasonal cattle density per hectare (SCD) and number of GPS collars used.

Where seasonal cattle density per hectare in 2013 is recorded as N/A, this indicates that the saltmarsh was not included in the study during this year.

Salt-marsh	Salt-marsh size (ha).	Herd size 2013	Herd size 2014	SCD ha ⁻¹ 2013	SCD ha ⁻¹ 2014	No. cattle GPS tagged 2013	No. cattle GPS tagged 2014	% of herd tagged	Weeks GPS logged	Dummy nests used (year)
A	322	116	112	0.36	0.35	4	0	3.45%	19/5/13-26/10/13	2014
B	126	39	37	0.31	0.29	4	0	10.26%	19/5/13-10/8/13	2013 2014
C	347	N/A	38	N/A	0.11	0	0	Not GPS logged.	N/A	2014
D	201	N/A	100	N/A	0.50	0	5	5.00%	28/4/14-20/7/14	2014
E	477	N/A	60	N/A	0.13	0	3	5.00%	5/5/14-17/8/14	2014

Table 4.2: Results of program MARK nest trampling analysis.

' ΔAIC_c ': the difference between the model in question, and the top model. $\Delta AIC_c < 2$ suggests both models are plausible (Burnham & Anderson 2002). Akaike weights represent the relative likelihood of the models (i.e. $\exp(-0.5 * \Delta AIC_c)$ for each model, divided by the sum of these values across all other models). Model likelihood is the support of the model in question relative to other models in the candidate set. \hat{C} values between 1 and 3 suggest the model structure is adequate (Lebreton *et al.* 1992). SCD = Seasonal cattle density. DSW = Distance to the sea wall. Creek = Distance to creek. DCP = Distance to nearest cattle congregation point. Veg. height = height of vegetation where the dummy nest was placed.

Analysis	Model	AIC _c	ΔAIC_c	Akaike Weight	\hat{C}
On saltmarshes exposed to higher grazing, does the probability of nest trampling increase closer to the landward extent of the saltmarsh? (Hypothesis 2).	DSW * SCD	589.34	0.00	0.34	1.27
	DSW * SCD + creek	589.91	0.56	0.26	1.27
	DSW * SCD + DCP	590.29	0.94	0.21	1.28
	DSW * SCD + veg height	591.26	1.92	0.13	1.28
	DSW + SCD	603.79	14.40	0.00	1.31
	DSW	647.96	58.61	0.00	1.41
	Constant survival	828.97	239.62	0.00	1.80
	SCD	830.97	241.62	0.00	1.81

Table 4.3: Results of general linear models and generalised least squares models investigating spatial and temporal effects on livestock distribution and grazing pressure.

df= degrees of freedom. Res df = Residual degrees of freedom. F = F value.

Analysis	Predictor	df	Res df	F	p value
i) Does the homogeneity of livestock distribution increase with time? (Hypothesis 3)	Marsh (A-D)	3	50	0.62	0.6
	Time (weeks)	1	50	0.13	0.72
	Time (week ²)	1	50	9.08	<0.001
	Marsh*Week	3	50	3.66	0.02
	Marsh*Week ²	3	50	1.77	0.16
ii) Does furthest distance travelled by livestock increase with time? (Hypothesis 3)	Marsh (A-D)	3	50	3.34	0.02
	Time (weeks)	1	50	62.78	<0.001
	Time (week ²)	1	50	1.17	0.29
	Marsh*Week	3	50	7.53	<0.001
	Marsh*Week ²	3	50	3.98	0.012
iii) Is grazing pressure higher closer to the landward extent of a saltmarsh? (Hypothesis 4)	Marsh (A-D)	3	36	95.58	<0.001
	DSW	1	36	286.45	<0.001
	DSW ²	1	36	30.01	<0.001
	Marsh*DSW	3	36	16.06	<0.001
	Marsh*DSW ²	3	36	1.36	0.27

Table 4.4: Results of general linear models investigating variation in livestock distribution in different saltmarsh zones over time.
df= degrees of freedom. Res df = Residual degrees of freedom. F = F value.

Analysis	Predictor	df	Res df	F	p value
i) Transition zone	Marsh (A-D)	3	50	11.17	<0.001
	Time (weeks)	1	50	17.45	<0.001
	Time (week ²)	1	50	4.63	0.03
	Marsh*Week	3	50	0.43	0.72
	Marsh*Week ²	3	50	1.09	0.34
ii) Drift line zone	Marsh (A-D)	2	41	36.34	<0.001
	Time (weeks)	1	41	76.59	<0.001
	Time (week ²)	1	41	4.67	0.03
	Marsh*Week	2	41	0.75	0.48
	Marsh*Week ²	2	41	2.10	0.15
iii) Mid marsh zone	Marsh (A-D)	3	50	87	<0.001
	Time (weeks)	1	50	0.00	0.97
	Time (week ²)	1	50	11.70	0.001
	Marsh*Week	3	50	0.33	0.81
	Marsh*Week ²	3	50	12.55	<0.001
iv) Low marsh zone	Marsh (A-D)	3	50	23.43	<0.001
	Time (weeks)	1	50	2.10	0.16
	Time (week ²)	1	50	0.28	0.6
	Marsh*Week	3	50	2.50	0.07
	Marsh*Week ²	3	50	3.35	0.03
v) Pioneer zone	Time (weeks)	1	12	0.09	0.77
	Time (week ²)	1	12	0.04	0.85
vi) Non-saltmarsh zone	Time (weeks)	1	8	14.31	<0.01
	Time (week ²)	1	8	5.17	0.052

4.7 Figures

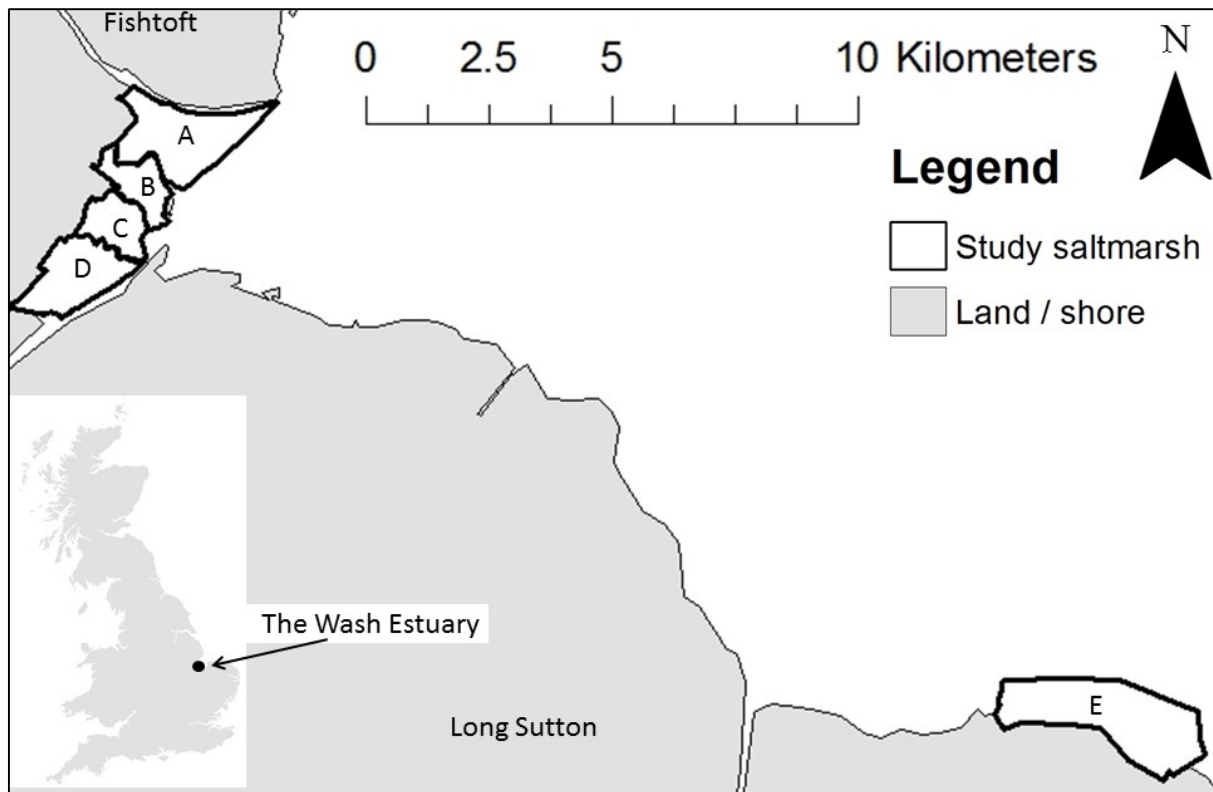


Figure 4.1: Map of the Wash estuary, showing the study saltmarshes.

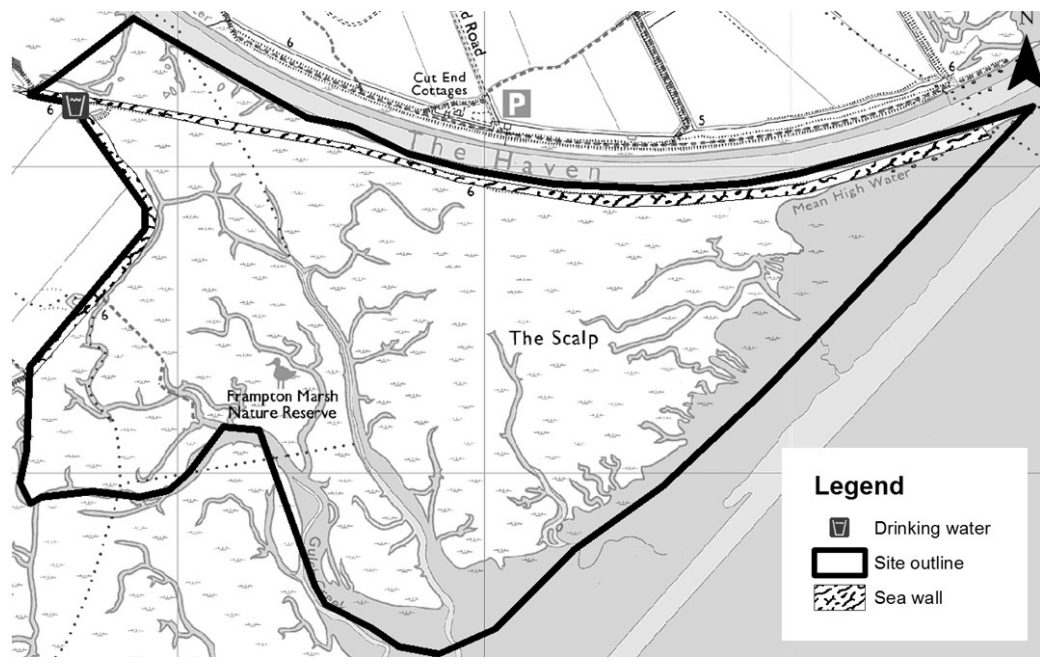
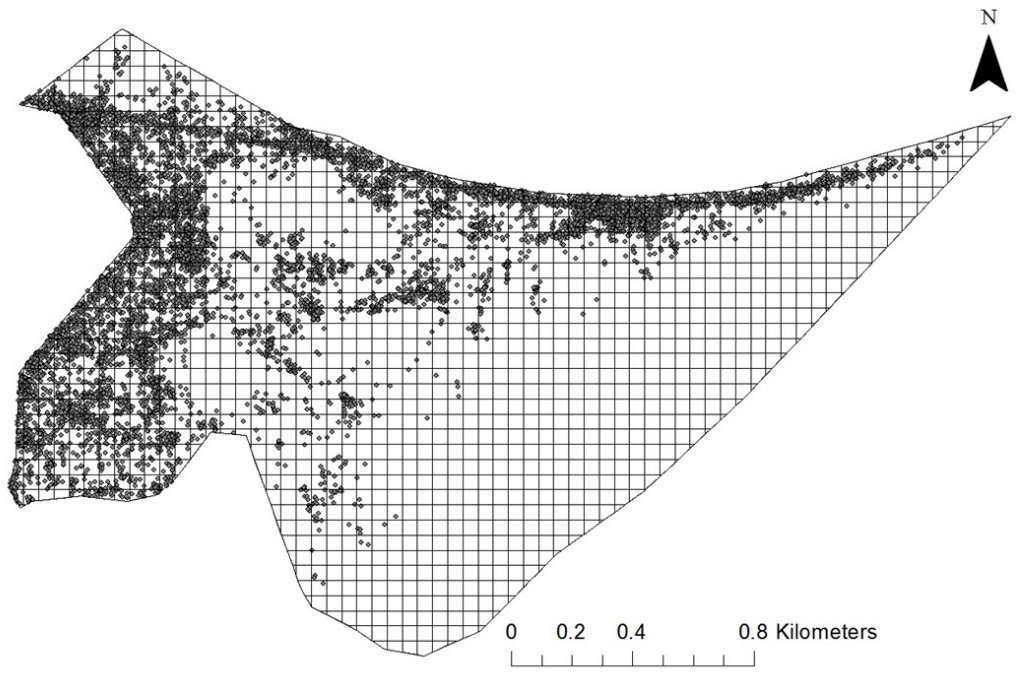


Figure 4.2: Maps of Saltmarsh A. This shows all GPS records (top) and a detailed site map (bottom).

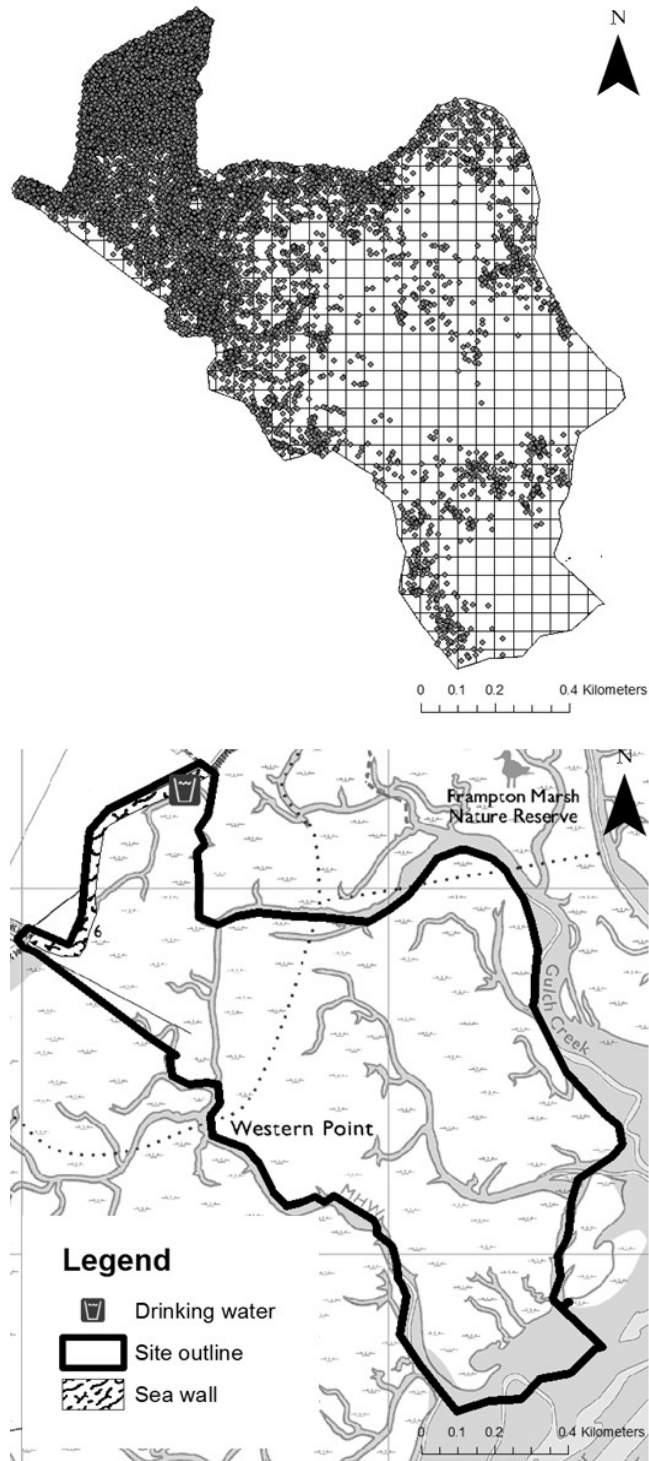


Figure 4.3: Maps of Saltmarsh B. This shows all GPS records (top) and a detailed site map (bottom).

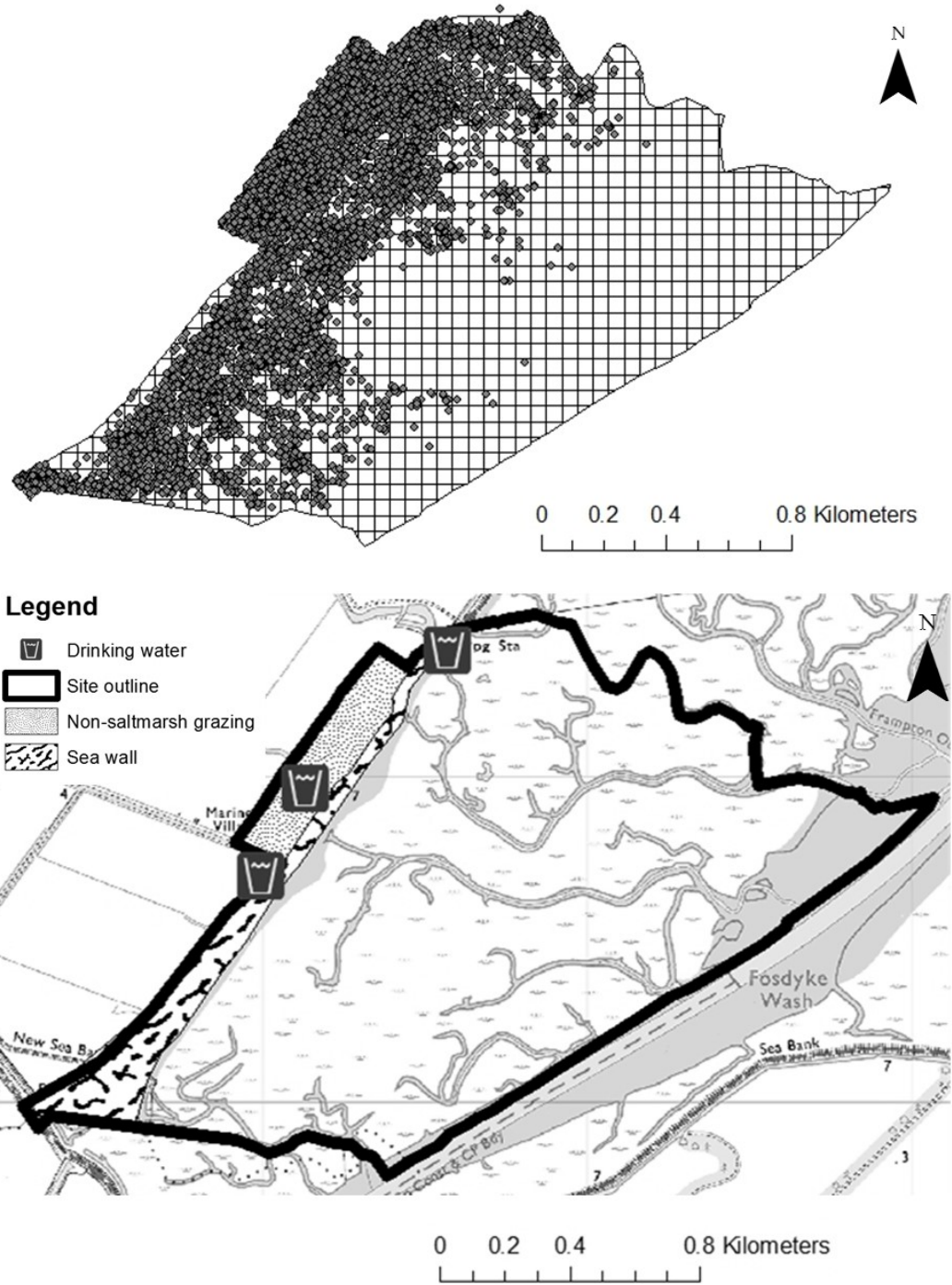


Figure 4.4: Maps of Saltmarsh D. This shows all GPS records (top) and a detailed site map (bottom).

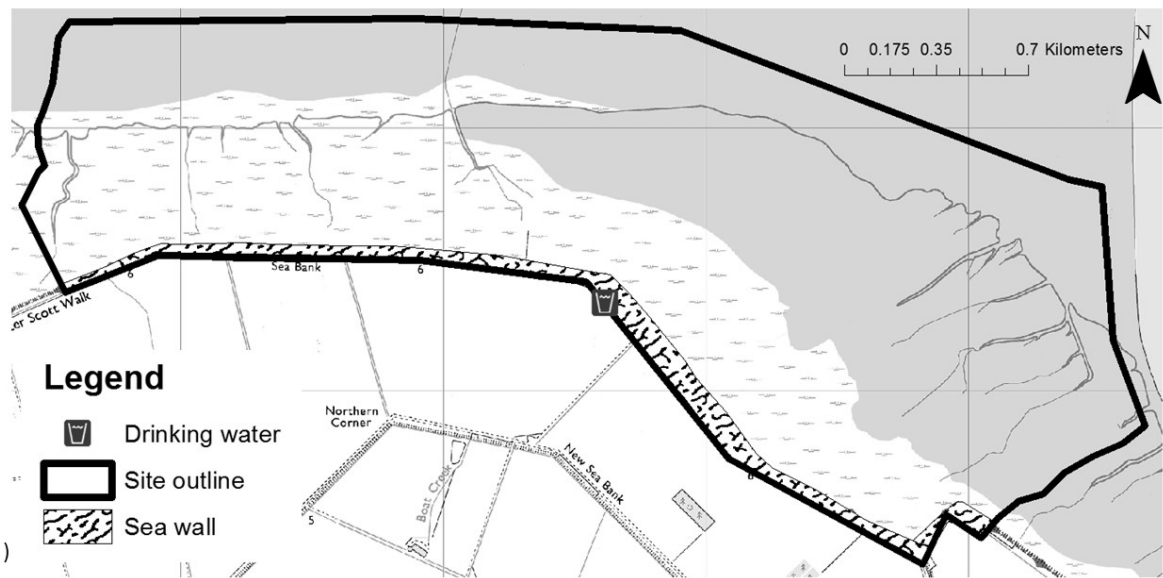
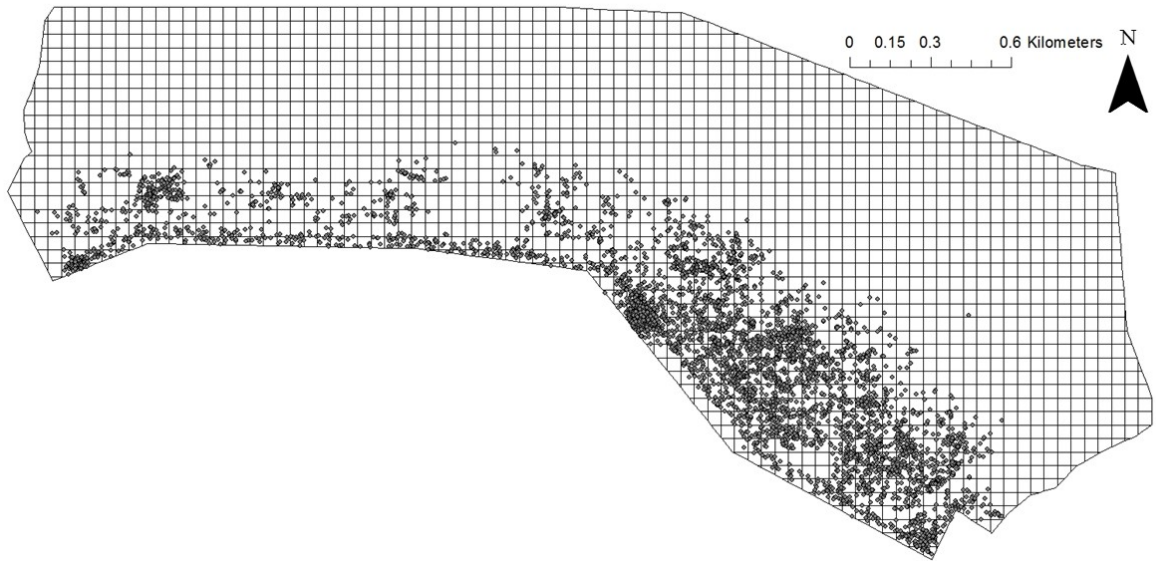


Figure 4.5: Maps of Saltmarsh E. This shows all GPS records (top) and a detailed site map (bottom).

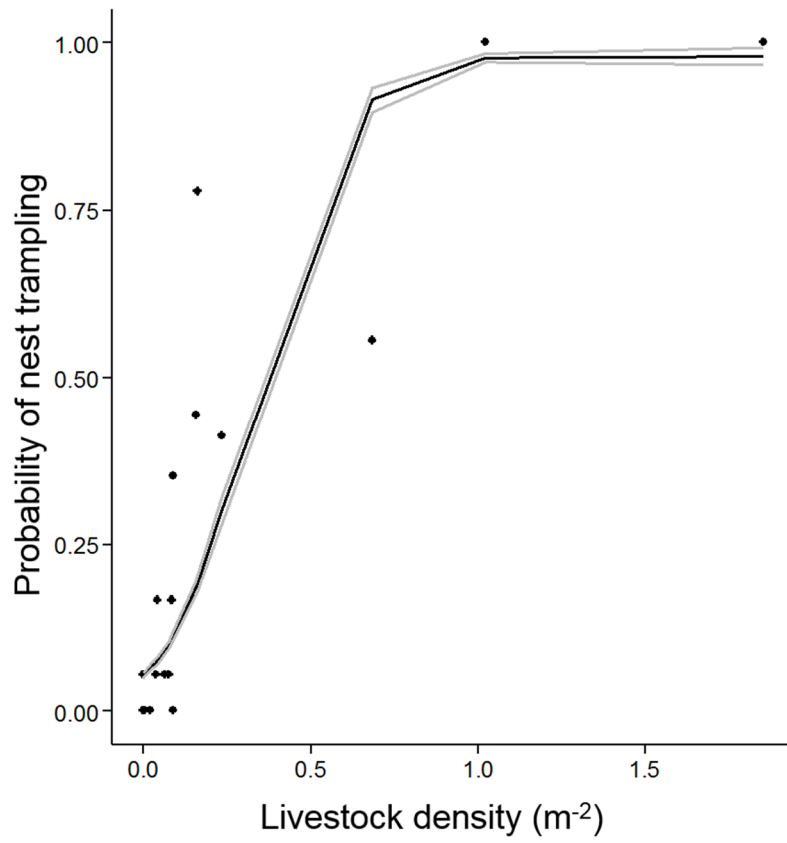


Figure 4.6: The probability of nest loss to trampling and livestock density (m⁻²).

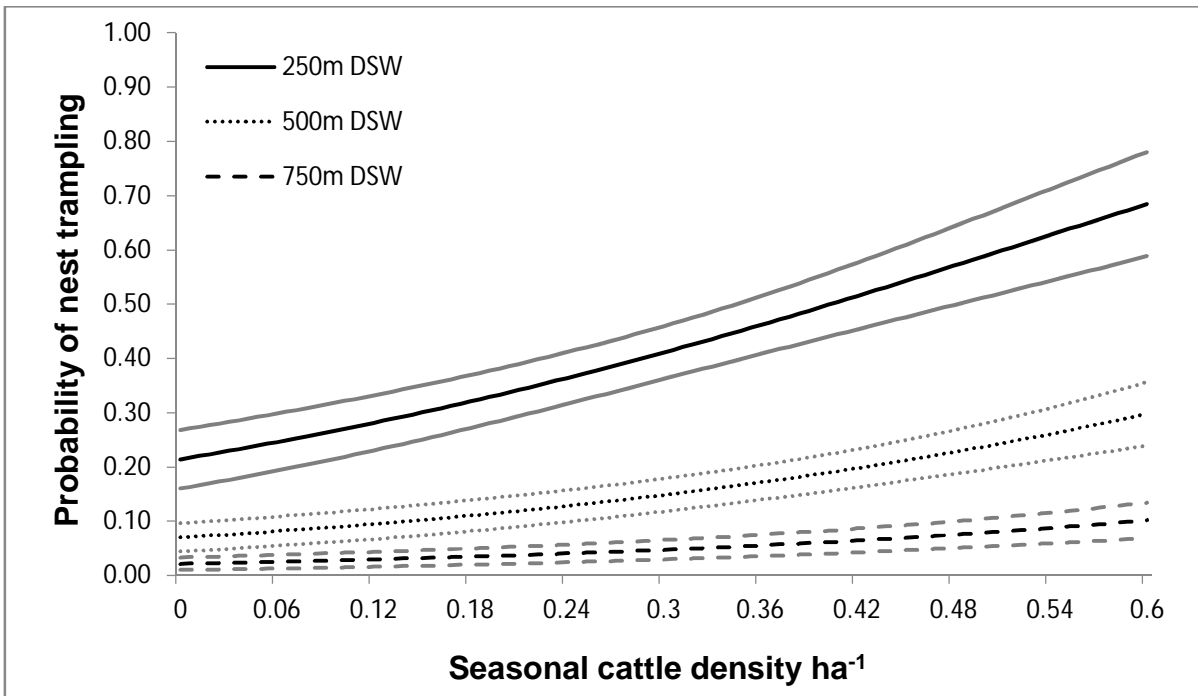


Figure 4.7: The probability of nest loss to trampling, showing the interaction between distance to the sea wall and seasonal cattle density. Confidence intervals (95%) are indicated by grey lines.

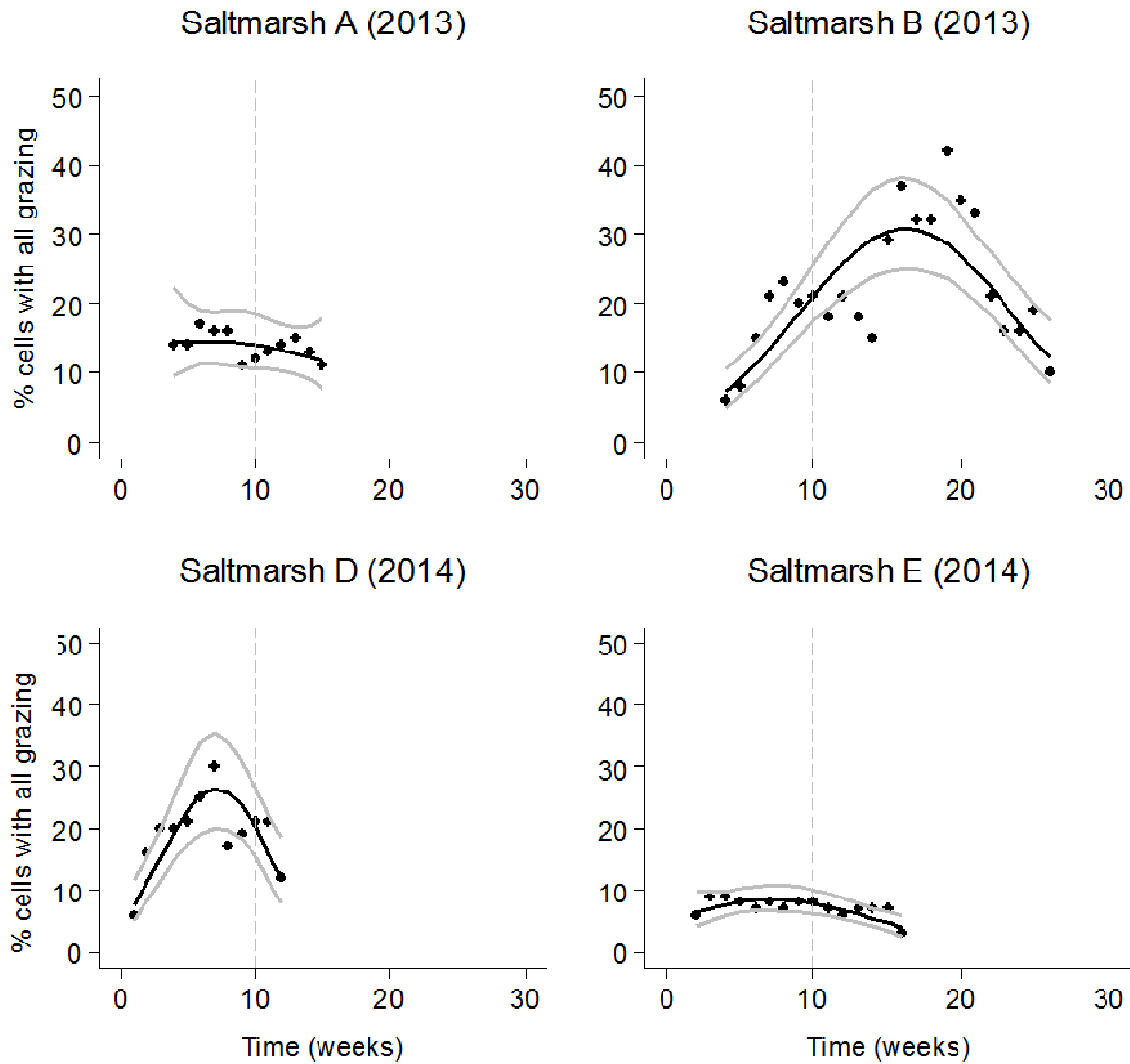


Figure 4.8: The percentage of cells containing all of the grazing and time. The percentage of cells containing all of the grazing is used as a measure of homogeneity of livestock distribution. Black lines are back-transformed model-fitted values. Confidence intervals (95%) are indicated by grey lines. The dashed grey vertical lines indicate the end of the Redshank nesting season (1st July). Week 1 was the week beginning 28th April. Week 26 (the last week) ended on the 26th October.

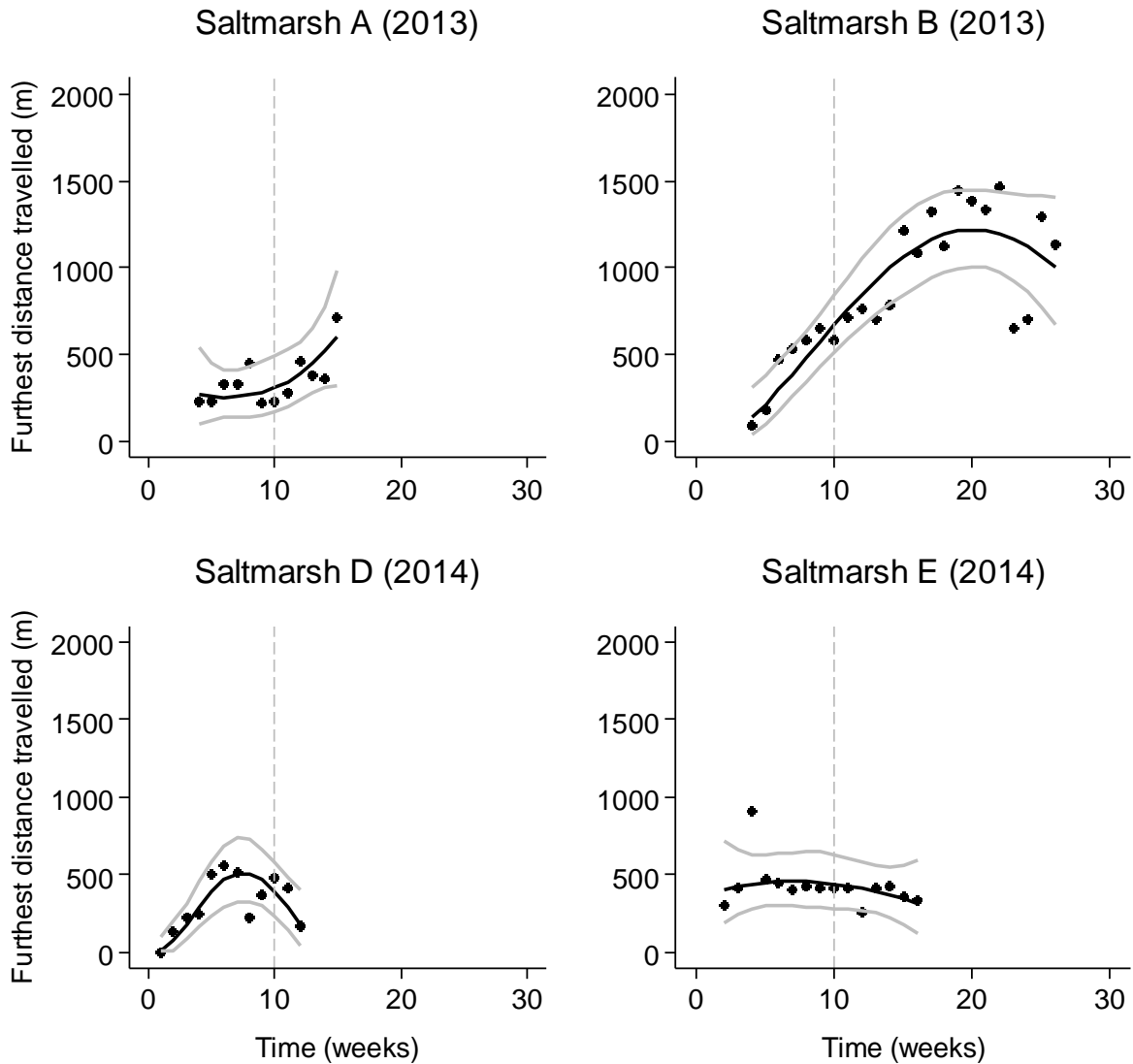


Figure 4.9: Maximum distance travelled by livestock and time. Black lines are back-transformed model-fitted values. Confidence intervals (95%) are indicated by grey lines. The dashed grey vertical lines indicate the end of the Redshank nesting season (1st July). Week 1 was the week beginning 28th April. Week 26 (the last week) ended on the 26th October.

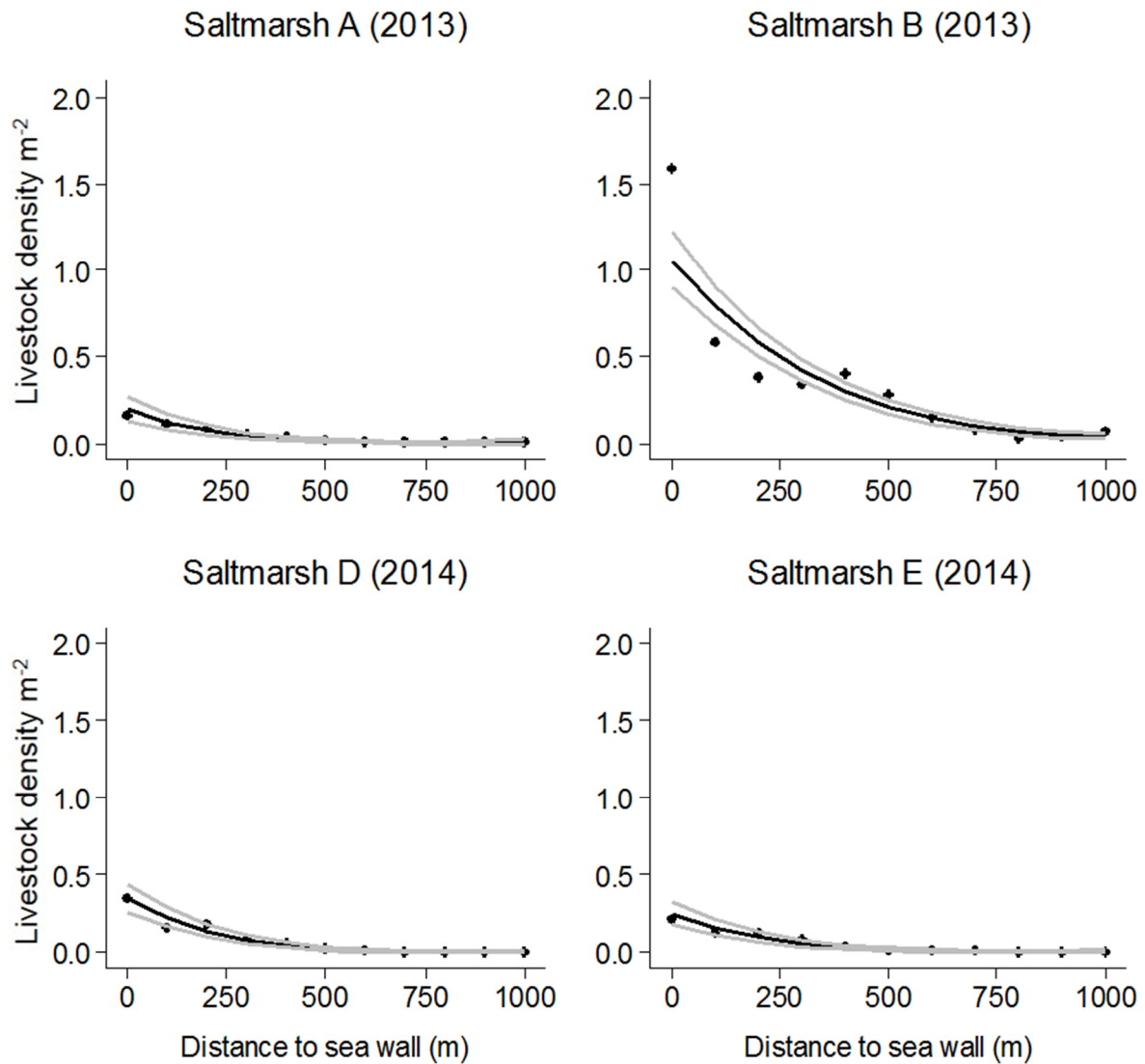


Figure 4.10: Livestock density (m⁻²) and distance to the sea wall. Black lines are back-transformed model-fitted values. Confidence intervals (95%) are indicated by grey lines.

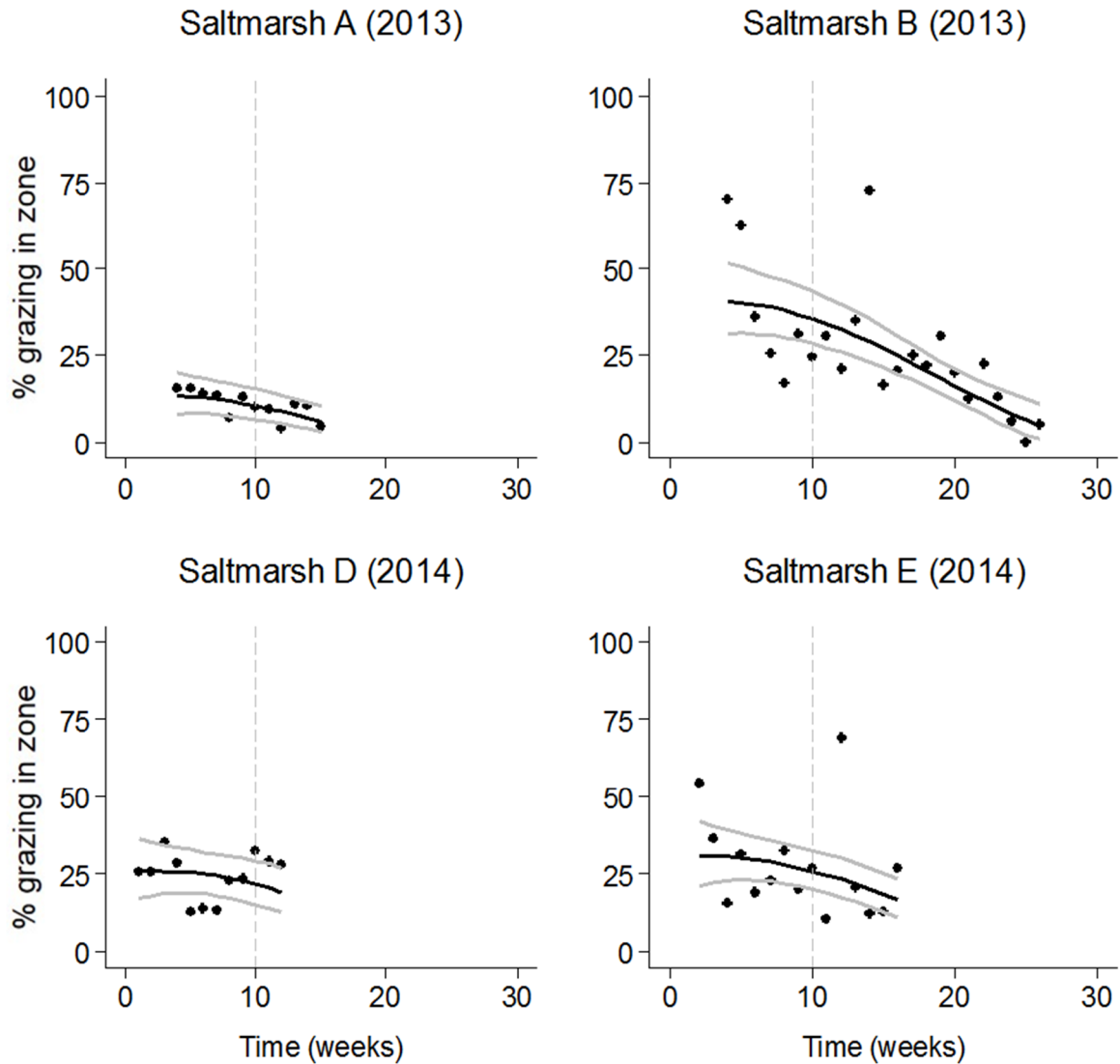


Figure 4.11: The percentage of the grazing in the grassland transition zone and time. Black lines are back-transformed model-fitted values. Confidence intervals (95%) are indicated by grey lines. The dashed grey vertical lines indicate the end of the Redshank nesting season (1st July). Week 1 was the week beginning 28th April. Week 26 (the last week) ended on the 26th October.

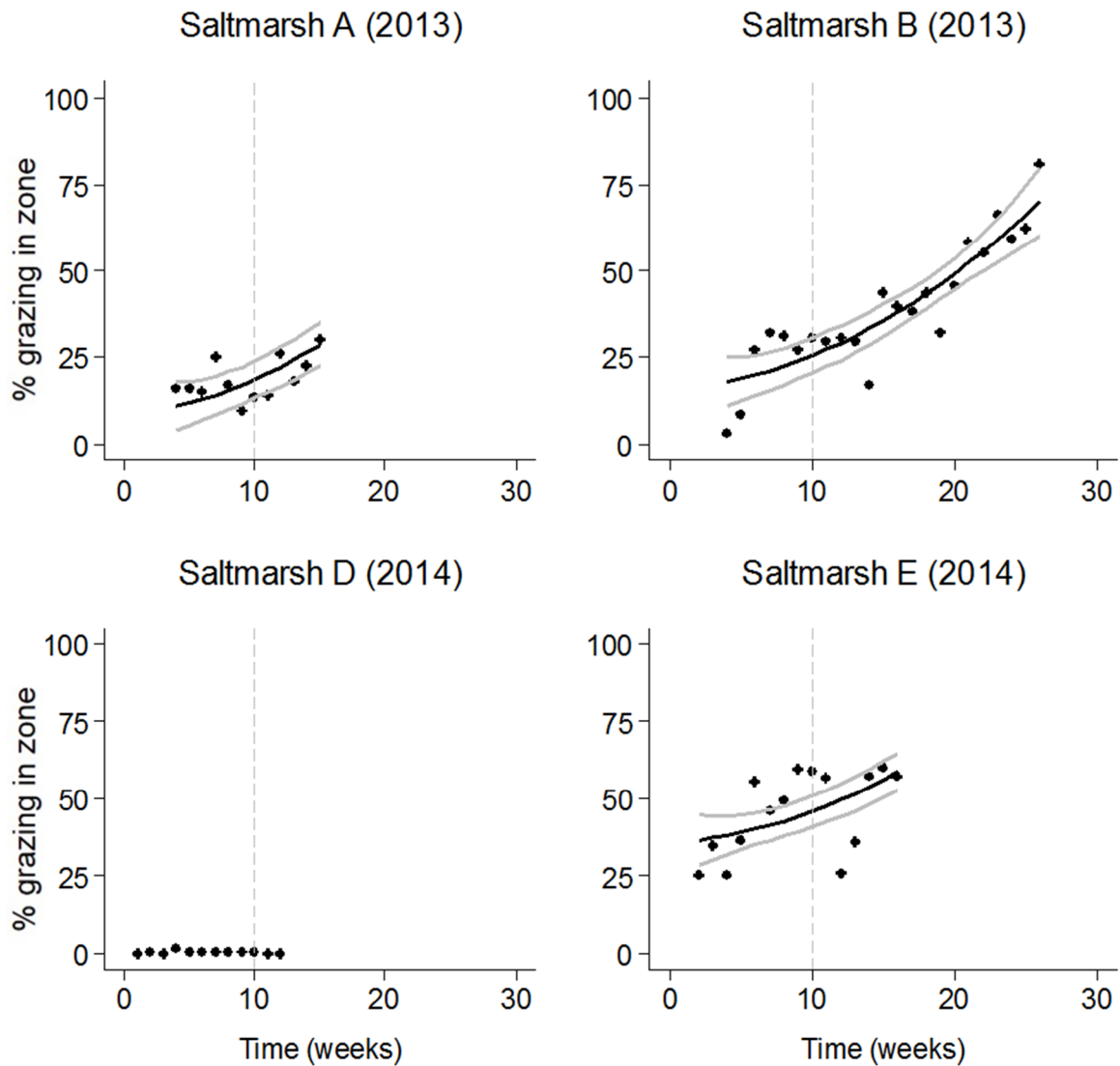


Figure 4.12: The percentage of the grazing in the drift line zone and time. Black lines are back-transformed model-fitted values. Confidence intervals (95%) are indicated by grey lines. The dashed grey vertical lines indicate the end of the Redshank nesting season (1st July). Week 1 was the week beginning 28th April. Week 26 (the last week) ended on the 26th October.

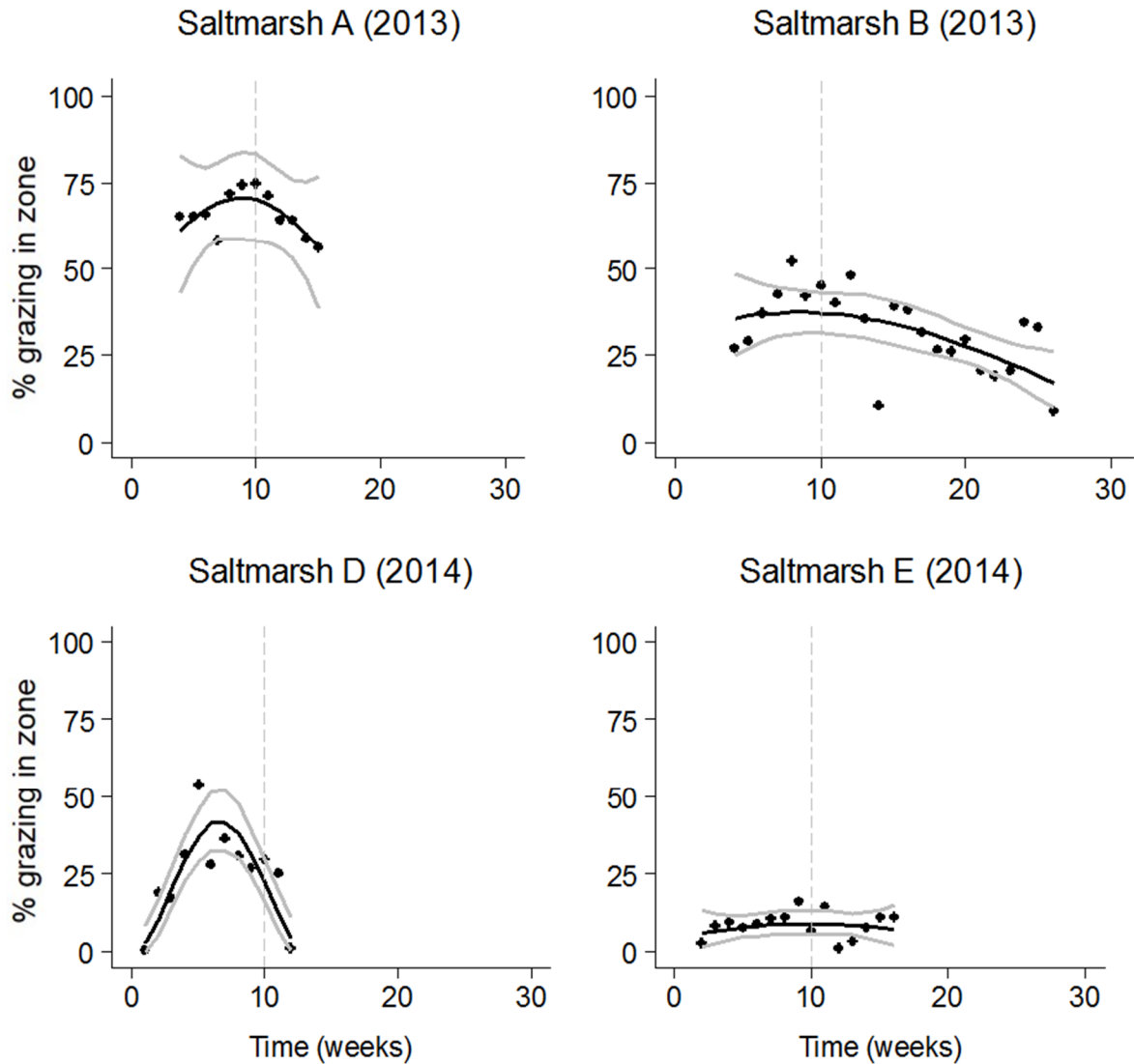


Figure 4.13: The percentage of the grazing in the mid marsh zone and time. Black lines are back-transformed model-fitted values. Confidence intervals (95%) are indicated by grey lines. The dashed grey vertical lines indicate the end of the Redshank nesting season (1st July). Week 1 was the week beginning 28th April. Week 26 (the last week) ended on the 26th October.

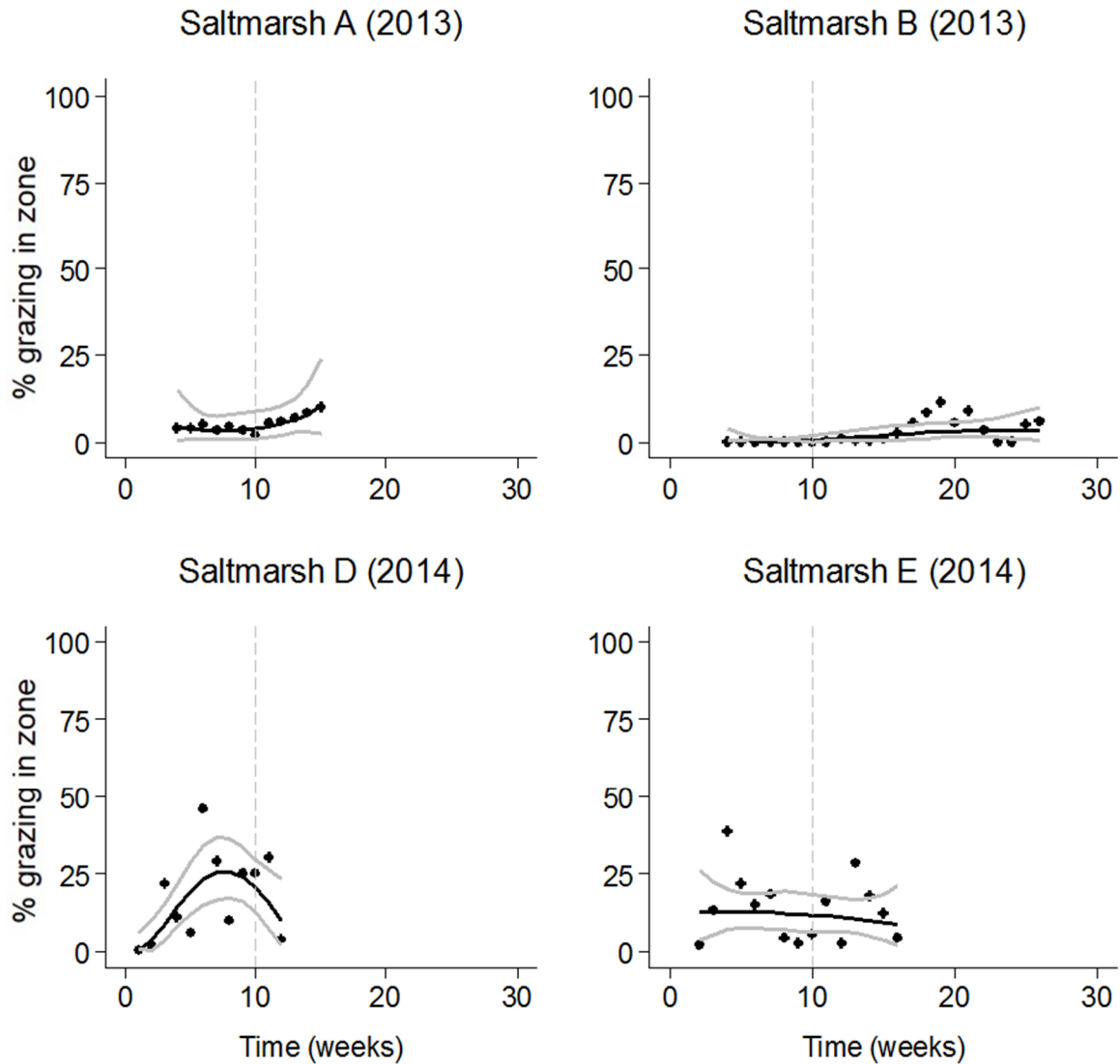


Figure 4.14: The percentage of the grazing in the low marsh zone and time. Black lines are back-transformed model-fitted values. Confidence intervals (95%) are indicated by grey lines. The dashed grey vertical lines indicate the end of the Redshank nesting season (1st July). Week 1 was the week beginning 28th April. Week 26 (the last week) ended on the 26th October.

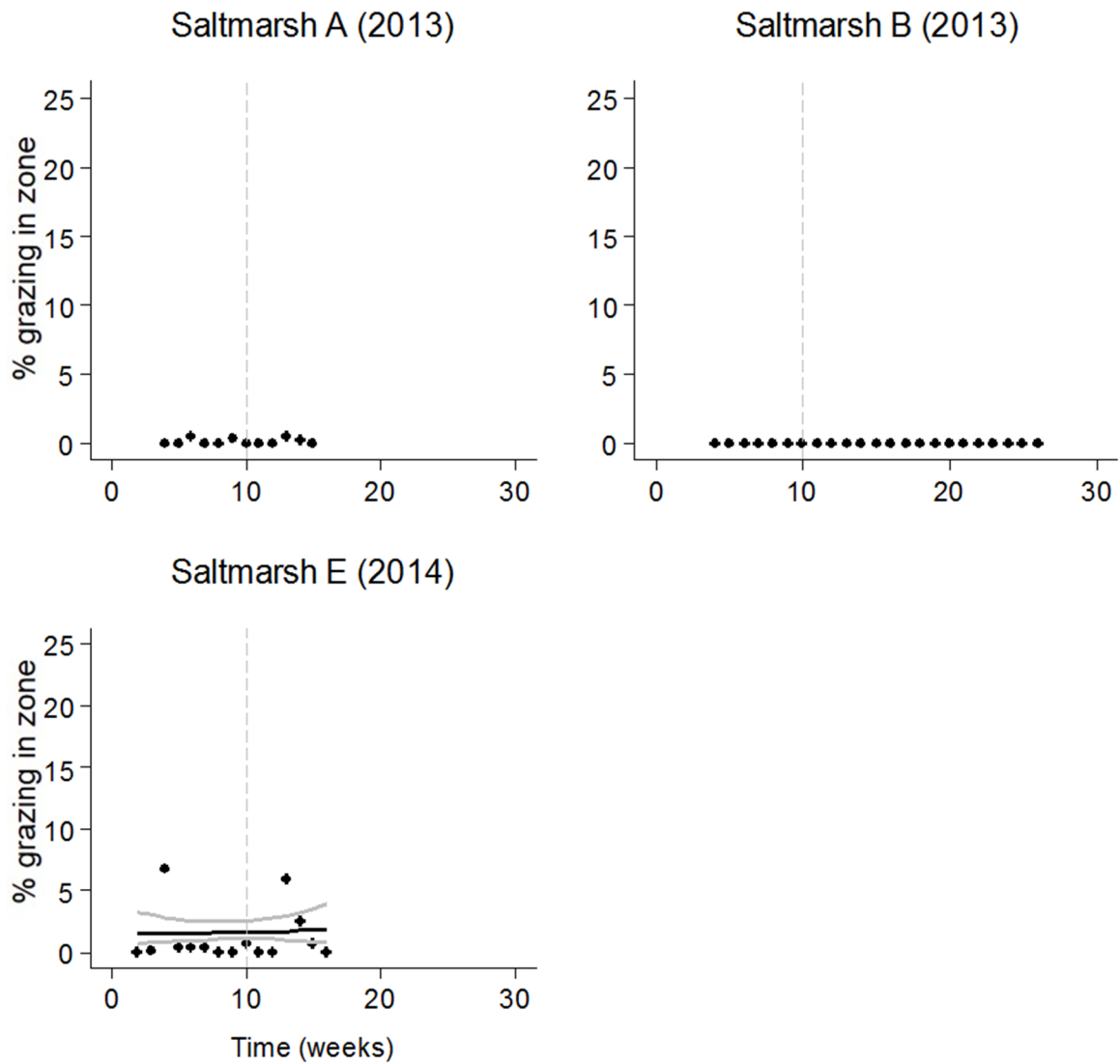


Figure 4.15: The percentage of the grazing in the pioneer zone and time. This zone was not present at saltmarsh D. Black lines are back-transformed model-fitted values. Confidence intervals (95%) are indicated by grey lines. The dashed grey vertical lines indicate the end of the Redshank nesting season (1st July). Week 1 was the week beginning 28th April. Week 26 (the last week) ended on the 26th October.

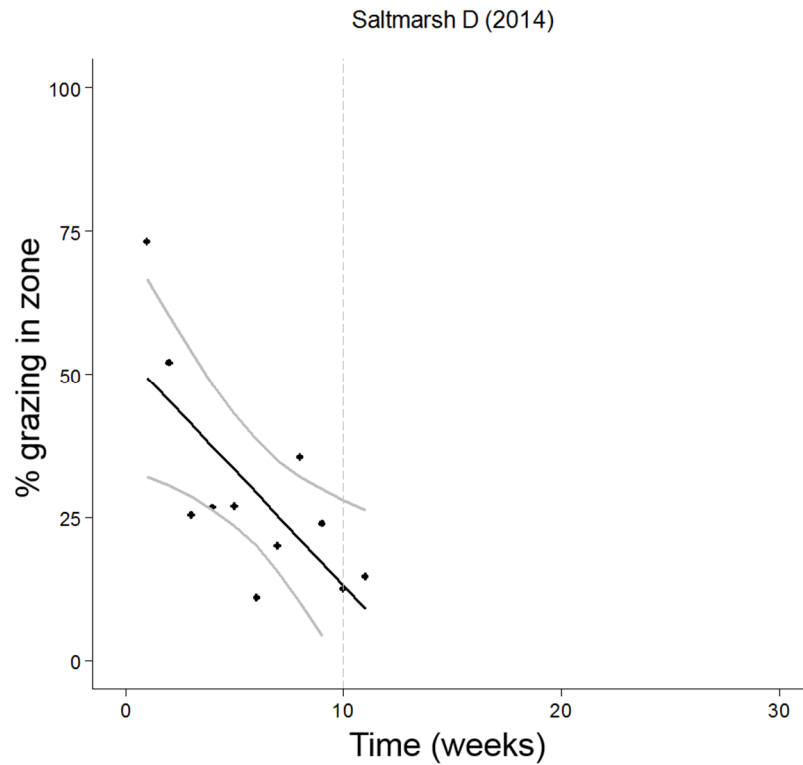


Figure 4.16: The percentage of the grazing in the non-saltmarsh zone and time. This zone was not present at saltmarshes A, B and E. The black line represents model-fitted values. Confidence intervals (95%) are indicated by grey lines. The dashed grey vertical lines indicate the end of the Redshank nesting season (1st July). Week 1 was the week beginning 28th April. Week 26 (the last week) ended on the 26th October.

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5 The impact of conservation management on nesting success of Common Redshank *Tringa totanus* in north-western Europe

5.1 Abstract

Breeding populations of Common Redshank *Tringa totanus* on British saltmarshes have reduced by > 50% since 1985. However, Redshank populations breeding on saltmarshes in the German and Dutch Wadden sea region are more stable. Low-intensity grazing of < 1 cattle ha⁻¹ leads to a high abundance of breeding Redshank, but low nest survival rates on British saltmarshes grazed by young cattle. This study aimed to investigate if conservation management techniques used in the Wadden sea regions lead to lower rates of nest mortality than those used on British saltmarshes. We asked: (a) does grazing result in more Redshank nest mortality than other conservation management techniques? and / or (b) does grazing with young cattle cause more Redshank nest mortality than grazing with adult cattle? We combined data from six Redshank nest survival studies to estimate nest failure from a large sample size of 567 nests on nine saltmarshes subject to mowing, grazing or no active conservation management. The probability of Redshank nest loss to predation or livestock trampling was 73% on grazed saltmarshes, 81% with no active management and 90% on mown saltmarshes and nest mortality rates were lower on saltmarshes grazed with adult cattle, than those with young cattle. We suggest that it may be best to manage saltmarshes with low densities of adult cattle, rather than with young cattle, mowing or no active management.

5.2 Introduction

Breeding wader populations have declined severely in Europe in recent decades (Tucker *et al.* 1994; Thorup 2004). Declines have been linked to agricultural intensification and associated loss of breeding habitat (Donald, Green & Heath 2001; Vickery *et al.* 2001). European saltmarshes provide a valuable breeding habitat for the Common Redshank, *Tringa totanus*, a species which has declined across its European range (Tucker *et al.* 1994; Papazoglou *et al.* 2004; Eaton *et al.* 2009; EBCC 2013). Approximately 18% of north-west Europe's estimated 100,000 – 172,000 nesting pairs, breed in Britain and 25% in the Wadden sea region of Germany, the Netherlands and Denmark (Piersma 1986; Rasmussen *et al.* 2000; Batten *et al.* 2010). Both regions are therefore internationally important for the species (Koffijberg *et al.* 2009; Batten *et al.* 2010). The British breeding population of Redshank decreased by around 43% between 1995 and 2013 (Balmer *et al.* 2013), with strong declines in coastal habitats (Malpas *et al.* 2013). In the Wadden sea, declines have been less severe; in the Netherlands and Lower Saxony, populations declined < 5% between 1991 and 2006. Further to the north-east, in Schleswig-Holstein populations remained stable and increased in Denmark by around 5% during the same period (Koffijberg *et al.* 2009).

In the UK, the most widespread forms of conservation management on saltmarshes are either grazing with young cattle (used for beef production), or grazing with sheep (Adnitt *et al.* 2007). In the Wadden sea regions, several other conservation management techniques are widely used, including grazing with adult cattle, horse grazing and mowing, which is commonplace in Germany (Maier 2014; Mandema *et al.* 2014a; Mandema *et al.* 2014b). Conservation management has the potential to alter habitat suitability by limiting or creating availability of vegetation patches, which are used for Redshank nesting (Mandema *et al.* 2014a; Sharps 2015a), but has failed to increase Redshank populations breeding on British saltmarshes (Malpas *et al.* 2013). Grazing is perhaps the most widespread form of saltmarsh management in north-western Europe (Adam 1990). Intensive grazing leads to a very short, uniform sward and lighter grazing results in a more uneven patchy sward with diverse heights (Jensen 1985; Kiehl *et al.* 1996). Mowing of saltmarshes, which is widely practiced in the Wadden sea region, can result in a more uniform sward, with heights dependent

on the mowing regime (Bakker 1978). If a saltmarsh is left un-managed, the most competitive plant species tend to dominate, leaving a dense cover of tall grasses (Bouchard *et al.* 2003). However, grazing can also cause high rates of nest mortality due to livestock trampling (Beintema & Muskens 1987; Sharps *et al.* 2015), although this could be eliminated by changing the timing of grazing so breeding areas are grazed after the breeding season has finished (Green 1986).

The large Redshank population declines in Britain have been linked to intensification or cessation of grazing as breeding densities are higher with light and moderate grazing than in heavily grazed or un-grazed saltmarshes (Norris *et al.* 1998; Malpas *et al.* 2013). Redshank breeding densities are higher in this structurally diverse vegetation on grazed saltmarshes (Norris *et al.* 1997). The patches of long vegetation provide cover for Redshank, which build a cryptic nest obscured by tall vegetation and leave the nest upon approach of a predator (Cervenci *et al.* 2011; Mandema *et al.* 2014a). To maintain suitable vegetation conditions, Norris *et al.* (1997) recommended livestock grazing densities of ~ 1 cattle ha^{-1} . This density falls within the UK Environment Agency definition of light saltmarsh cattle grazing of 0.7 - 1 young cattle ha^{-1} , present from April to October (Adnitt *et al.* 2007). However, light grazing by young cattle may not be optimal, as it can reduce nest survival both directly through nest trampling and indirectly through accelerating predation risks by reducing vegetation height, leading to near total nest mortality in some saltmarshes. This was demonstrated by Sharps *et al.* (2015), who found that risk of nest loss to livestock trampling increased from 16% at 0.15 young cattle ha^{-1} to 98% at 0.82 young cattle ha^{-1} in the breeding season. Young cattle are more lively than adults, therefore they have higher energy requirements (Basarab *et al.* 2003; Ausden 2007). Grazing with young cattle, rather than adult cattle, may lead to increased nest trampling and loss of vegetation cover resulting in increased rates of nest predation.

Potential Redshank nest predators include corvids (*Corvus* spp.), gulls (*Larus* spp.), Red Fox (*Vulpes vulpes*), mustelids (*Mustela* spp.), non-native American Mink (*Neovison vison*) and Brown Rat (*Rattus norvegicus*) (Smart 2005; Thyen & Exo 2005). Mammals may predate nests close to the sea wall as this is more easily accessible from their terrestrial habitats, however avian predators may cause nest mortality anywhere on the saltmarsh (Montevecchi 1977). Saltmarshes are often used by gulls, which tend to forage on adjacent coastal mudflats or nest in colonies in the

lower marsh zones (Greenhalgh 1971; Burger 1977). These areas tend to be closer to the seaward extent of the saltmarsh and therefore further from the sea wall. It may therefore be expected that more Redshank nest loss to predation occurs in nests further from the sea wall. Livestock distribution on saltmarshes is not homogenous (Sharps 2015b), and grazing can be higher closer to the landward extent of the saltmarsh (sea wall) during the Redshank breeding season. This can result in increased rates of nest trampling near the sea wall (Sharps *et al.* 2015; Sharps 2015b). Sharps *et al.* (2015) suggested that rates of predation were higher further from the sea wall, suggesting avian predators may have destroyed more of the nests under observation in the study. However, this result may have been partly obscured by the high rates of nest trampling closer to the sea wall.

For ground nesting birds, the presence of some large mammals (e.g. predators or domestic pets) causes disturbance which can lead to the permanent abandonment of clutches by the incubating adults (Madsen, Bregnballe & Hastrup 1992; Lord *et al.* 2001). This has been observed for Redshank, with the presence of ground predators leading to increased nest abandonment (Cervencl *et al.* 2011; Maier 2014), however it is unclear if this also occurs due to the direct presence of livestock on saltmarshes.

In this study, we combine data from six Redshank nest survival studies across nine saltmarshes in Britain and the Wadden sea regions. This approach provides the opportunity to compare the effect of saltmarsh management using a wider range of management measures than is possible in individual studies. The large amount of data involved and large scale of this analysis allows an unprecedented generalisation of the effects of saltmarsh management on breeding success of Redshank, and may elucidate why Redshank populations have declined so rapidly in the UK compared to the Wadden sea regions. We estimate Redshank nest failure on saltmarshes subject to mowing, grazing or no active conservation management. We hypothesise that (1) grazing results in higher Redshank nest mortality than other conservation management techniques, (2) grazing results in more Redshank nest abandonment than other conservation management techniques, (3) grazing with young cattle causes more Redshank nest mortality than grazing with adult cattle, (4) Redshank nest loss to predation is lower closer to the landward extent of the saltmarshes, regardless of the conservation management technique used.

5.3 Methods

Saltmarsh conservation land management and Redshank nest survival data (including data on the fate of the nest and number of days exposed to risk, $n = 567$ nests) were pooled from published and unpublished Redshank nest survival studies across nine saltmarsh regions in north-western Europe (Tables 1-2, Fig. 5.1). These included data from Germany (Thyen & Exo 2003; Thyen & Exo 2005; van de Pol *et al.* 2010; Cervenci *et al.* 2011; Maier 2014), the Netherlands (Mandema 2014; Mandema *et al.* 2014a) and the UK (Smart *et al.* 2003; Smart 2005; Sharps *et al.* 2015).

Typically, data collection involved nest observations between April and July. Nests were usually found by systematic walking in which all parts of the study saltmarshes were searched, observing adults flushed from their concealed nests to ensure consistency of nest discovery across all vegetation types. To record nest survival, nests were revisited during the nesting season. Nests were revisited until the eggs had either hatched or the clutch had failed. Nests were considered successful if one or more egg hatched. If eggs were missing and/or remains of eaten eggs were present, the nest was classified as predated. If the clutch was cold on two consecutive visits, the nest was classified as abandoned. If eggs had clearly been flattened/crushed in the nest they were classified as trampled by livestock. If, immediately after high tides, all or part of the clutch was missing and nests were wet and/or covered in salt deposits, nests were classified as destroyed by tidal flooding, and revisited again to be sure the nest was no longer active. Nests were classified as 'cause unknown' if there was no clear causal evidence of failure.

To test hypotheses 1 and 2, that grazing results in more Redshank nest mortality and nest abandonment than other conservation management techniques, we classified conservation management as grazed ($n=96$ nests), rotational (unoccupied) ($n=14$ nests) mown ($n=87$ nests) or no active management ($n=370$ nests). We defined rotational (unoccupied) as saltmarshes normally subject to some form of grazing management, but with no livestock present during the period the Redshank nests were active, whereas saltmarshes with 'no active management' had not been subject to any form of land management for at least 10 years prior to the start of nest recording. This

simply refers to the type of land use and does not refer to any protected area status, agri-environment scheme participation or predator control operation. Nest survival was computed in relation to conservation management (grazed, mown, rotational (unoccupied) or no active management).

To test hypothesis 3, that grazing with young cattle causes more Redshank nest mortality than grazing with adult cattle, we recorded the approximate age of cattle (adult or juvenile under 24 months) and the density of cattle within each saltmarsh during the breeding season from the nests on grazed saltmarshes with known cattle grazing data (n = 103 nests; 53 exposed to young cattle; 50 exposed to adult cattle). As nest loss to livestock trampling occurs as a direct result of the presence of livestock in the breeding season, and nest loss to predation occurs as an indirect result of the effects of grazing on vegetation structure (Sharps *et al.* 2015), two measurements of stocking density were used. These were seasonal cattle density, indicative of trampling pressure (Jensen *et al.* 1990), and annual cattle density, a measurement of how longer term grazing changes the vegetation landscape (Andresen *et al.* 1990). Seasonal cattle density measured the density of cattle within each saltmarsh during the breeding season. As grazing management was only available for one year prior to the start of most studies, annual cattle density took into account the number of cattle present within a saltmarsh over the 12 month period preceding the breeding season to reflect the longer-term nature of grazing-induced changes to the habitat (Andresen *et al.* 1990). Annual cattle density was calculated using the formula: Annual cattle density = number of cattle * number of months the animals were present / (size of site * 12). These calculations took into account only the grazed part of the saltmarshes. A further 7 nests from saltmarshes with horse grazing were excluded from the analysis, due to the small sample size. No saltmarshes exposed to other livestock types were present in the data. To see if grazing with young cattle causes more Redshank nest mortality than grazing with adult cattle, nest survival was computed in relation to age of livestock and livestock density.

To test hypothesis 4, that Redshank nest loss to predation is lower closer to the landward extent of the saltmarshes, regardless of the conservation management technique used, the distance to the sea wall was used as a measurement of the distance from the nest to the landward side of the marsh. This was measured using ArcGIS 10.1. This was only possible for Redshank nests for which we had GPS

coordinates (n = 387 nests). To investigate if Redshank nest loss to predation is lower closer to the landward extent of the saltmarshes (and if this varies between management types), nest survival was computed in relation to the distance from the nest to the sea wall and conservation management technique.

All data were tested to determine if there was a need to include the random effects of site (each saltmarsh) and year in the models. This involved using a generalised linear mixed model (GLMM) in the statistical program R (R Core Team 2013), with a log-link exposure function, following Shaffer and Burger (2004). In each case, the variance explained by the random effects was zero or close to zero suggesting that the analyses did not require the random effects. Therefore, these analyses were carried out using the nest survival model of Program MARK (Dinsmore, White & Knopf 2002). Program MARK accounts for the number of days a nest is active before mortality, based on the Mayfield approach (Mayfield 1961; Mayfield 1975). It uses numerical maximum-likelihood techniques and computes a quasi-likelihood AIC value (White & Burnham 1999). This enabled the selection of the variable(s) (Table 5.1) that most strongly accounted for variation in nest survival. As the frequency of observer revisits to nests may have been different between the studies, we selected this method because it does not assume that nest visits are made at regular intervals (Johnson 2007). As we observed similar proportions of nests lost to predation throughout the study period, it was assumed that predation risk remained constant.

To enable the evaluation of nest survival due to cause-specific components (Donovan *et al.* 1995), a total of six separate analyses were carried out (Table 5.1). Depending on the analysis in question, following Fondell & Ball (2004), either nests abandoned, trampled, predated, or predated and trampled combined were treated as unsuccessful and the remaining nests as successful (even though many of them were lost due to another cause). To estimate true probability of nest loss over the 24 day Redshank incubation period (Green 1984), the daily survival rate was raised to the power of 24, following Rotella, Cooch & White (2009) and converted from a survival to a mortality probability. Variance was calculated using the Delta method (Powell 2007) from which confidence intervals were calculated, following Armstrong *et al.* (2002).

Program MARK estimated all possible models. Model outputs were compared using the Akaike Information Criterion (corrected for small sample size, AICc) (Burnham &

Anderson 2002). The model with the lowest AICc value is the most parsimonious model. Models differing in 2-7 AICc from the most parsimonious model have little empirical support; those with $\Delta AICc > 10$ have no support. Akaike weights, (the relative likelihood of the models, i.e. $\exp(-0.5 * \Delta AIC)$ for each model, divided by the sum of these values across all other models) were used to measure the support for each model. To check the model fit, we obtained a \hat{C} value from Program MARK for the global model. Following Lebreton et al. (1992) if the model structure is adequate, \hat{C} is expected not to exceed 3. \hat{C} values between 3 and 4 suggest only a minor degree of over-dispersion (Lebreton *et al.* 1992; Anderson, Burnham & White 1994), whilst values between 0 and 1 indicate minor under-dispersion (Class, Klun & Brommer 2014), however following Rotella (2009), the models were not corrected for this underdispersion.

5.4 Results

Of the 567 nests included in this study, 9% were abandoned, 52% were predated and 30% successfully hatched (Table 5.1). Three percent of nests were trampled by livestock, representing 18% of all nests exposed to livestock grazing (Table 5.2). Six percent of nests were lost to tidal flooding, however as the hypotheses focus on conservation land management, it is beyond the scope of this study to investigate this cause of nest mortality further. Across all of the saltmarshes and years, the earliest date that nests were found was the 23rd April, and the latest date a nest was still active was the 21st July, when the last nest was predated.

Conservation management affected the probability of Redshank nest loss to predation. The highest rates of nest mortality were on mown saltmarshes with 59% of nests lost to predation. On saltmarshes with no active conservation management, 55% of nests were lost to predation. 35% of nests on rotational (unoccupied) and 32% of nests on grazed saltmarshes were lost to predation, but when considering nests lost to either livestock trampling or predation on grazed saltmarshes this increased to 51%. Accounting for the number of days nests were active before a nest mortality, model fits calculated that the probability of Redshank nest loss to predation was 90% on mown saltmarshes, 82% with no active management, 67% on rotational (unoccupied) saltmarshes and 59% on grazed saltmarshes (Fig. 5.2; Table 5.3 A_i , AIC_c weight = 1.00, $\Delta AIC_c=14.58$ when the conservation management model was compared with the

constant survival model without explanatory variables). When nest loss due to both predation and trampling was investigated the probability of nest loss on grazed saltmarshes was 73% (Table 5.3 Aii, AIC_c weight = 0.82, $\Delta AIC_c=2.97$ when the conservation management model was compared with the constant survival model without explanatory variables). These results therefore do not support hypothesis 1 that grazing results in higher Redshank nest mortality than other conservation management techniques, Conversely, results indicate that even when both trampling and predation are taken into account, nest mortality on grazed saltmarshes is still lower than on mown and unmanaged saltmarshes.

There was little evidence for an effect of conservation management on the probability of Redshank nest abandonment (Fig 5.3; Table 5.3 B, the constant survival model without explanatory variables was $AIC_c=3.94$ lower than the conservation management model). These results therefore do not support hypothesis 2 and suggest that grazing does not result in more Redshank nest abandonment than other conservation management techniques.

Seasonal cattle density and cattle age (adult or young) affected the probability of Redshank nest loss to trampling. Nest mortality was lower on saltmarshes grazed by adult cattle and at low livestock densities, as no nests were trampled on saltmarshes with 0.45 adult cattle ha^{-1} and 18% of nests were trampled on saltmarshes with 1 adult cattle ha^{-1} . On saltmarshes grazed by young cattle 33% of nests were trampled on saltmarshes with 0.13 young cattle ha^{-1} and 50% of nests were trampled on saltmarshes with 0.82 young cattle ha^{-1} . Models accounting for the number of days nests were active showed that both seasonal cattle density and cattle type could explain these results (Fig. 5.4; Table 5.3 Ci, AIC_c weight = 0.51, $\Delta AIC_c=22.58$ when the model containing cattle density and type was compared with the constant survival model without explanatory variables; removal of cattle density, $\Delta AIC_c=11.22$, removal of type, $\Delta AIC_c=23.37$), however there was no clear support for an interaction between seasonal cattle density and cattle type ($\Delta AIC_c = 0.06$ on removal of the interaction). The probability of Redshank nest loss to trampling increased from 25% at 0.04 cattle ha^{-1} to 99% at 1 cattle ha^{-1} when grazed by young cattle. When grazed by adult cattle the probability of Redshank nest loss to trampling increased from 2% at 0.04 cattle ha^{-1} to 28% at 1 cattle ha^{-1} . This suggests that grazing with young cattle causes more Redshank nest loss to trampling than grazing with adult cattle.

The probability of Redshank nest loss to predation was affected by both annual cattle density and cattle age. Less nests were predated on saltmarshes with lower livestock densities and grazed with adult cattle than those grazed by young cattle as 11% of nests were predated on saltmarshes with 0.19 annual cattle density ha⁻¹ adult cattle and 50% of nests were predated on saltmarshes with 0.42 annual cattle density ha⁻¹ adult cattle. Conversely, 25% of nests were predated on saltmarshes with 0.05 annual cattle density ha⁻¹ young cattle and 38% of nests were predated on saltmarshes with 0.55 annual cattle density ha⁻¹ young cattle. Model fits accounting for the number of days nests were active calculated that the probability of Redshank nest loss to predation increased from 34% at 0.02 annual cattle density ha⁻¹ to 94% at 0.55 annual cattle density ha⁻¹ when grazed by young cattle. When grazed by adult cattle the probability of Redshank nest loss to predation increased from 16% at 0.02 annual cattle density ha⁻¹ to 78% at 0.55 annual cattle density ha⁻¹ (Fig. 5.4; Table 5.3 Cii, AIC_c weight = 0.32, Δ AIC_c=12.36 when this model was compared with the constant survival model without explanatory variables; removal of annual cattle density, Δ AIC_c=10.3, removal of type, Δ AIC_c=2.21), however there was no clear support for an interaction between annual cattle density and cattle type (Δ AIC_c = 1.16 on removal of the interaction). This suggests that grazing with young cattle causes more Redshank nest loss to predation than grazing with adult cattle.

Distance from the nest to the landward extent of the saltmarsh also affected the probability of nest loss to predation (adding to the variance explained in the model by conservation management type). Nests further away from the sea wall were more likely to be predated, and whilst there were some differences between the nature of this relationship at saltmarshes subject to different conservation management, results suggest that there was little statistical support for these differences. Model fits calculated that the probability of Redshank nest loss to predation increased from 52% zero metres from the sea wall to 92% 995 metres from the sea wall on grazed saltmarshes. On mown saltmarshes, the probability of nest loss to predation increased from 90% zero metres from the sea wall to 100% 995 metres from the sea wall (Fig. 5.5; Table 5.3 D, AIC_c weight = 0.81, Δ AIC_c=15.48 when the model containing distance to the sea wall and conservation management was compared to the constant survival model without explanatory variables; removal of distance variable, Δ AIC_c=6.92, removal of management variable, Δ AIC_c=13.51). However there was no support for

an interaction between distance to the sea wall and conservation management (the top model contained distance + conservation management, $\Delta AIC_c = 3.2$ when the interaction was added to the model). This suggests that Redshank nest loss to predation is lower closer to the landward extent of the saltmarshes, regardless of conservation management.

5.5 Discussion

In this study, we combined data on 567 Redshank nests from six nest survival studies across nine saltmarshes in north-western Europe. This approach provided the opportunity to compare the effect of conservation management using a wider range of management measures than is possible in individual studies. The large sample size of nests and number of study sites allowed a robust comparison of differing saltmarsh management techniques on nest mortality and may elucidate why Redshank populations have declined so rapidly in the UK compared to the Wadden sea regions. The results of this study do not support hypothesis one, that grazing results in higher Redshank nest mortality than other conservation management techniques, as we found that Redshank nest survival rates were higher in grazed saltmarshes than those that were mown or had no active management. We found no support for hypothesis 2, that grazing results in more Redshank nest abandonment than other conservation management techniques. Rates of nest loss were higher on saltmarshes grazed with young cattle, than those grazed with adult cattle, supporting hypothesis 3. Rates of nest loss to predation were lower close to the sea wall, and were similar regardless of the conservation management technique used, suggesting support for hypothesis 4. Based on these results, the best saltmarsh management regime for Redshank would involve grazing by adult cattle at low livestock densities.

It should be noted that the large error bars for the probability of nest failure and abandonment on rotational (unoccupied) saltmarshes suggest that expected nest survival rates on saltmarshes normally subject to some form of grazing management, but with no livestock present during the period the Redshank nests were active remain uncertain. As this is not commonly practiced, there were only two saltmarshes subject to this conservation management technique. This represented a total of only 14 nests from an overall sample size of 567 nests. It is therefore likely that this result is due to the very low sample size of rotational (unoccupied) nests. It might be expected that

leaving a saltmarsh unoccupied by livestock during the Redshank breeding season would result in higher rates of nest survival due to reduced trampling rates, but improved vegetation structure due to previous grazing, but our results on this are inconclusive. Further research into the potential benefits of this management technique may be necessary.

Conservation management did not affect the probability of nest abandonment and grazed saltmarshes had the lowest nest abandonment rates. Although nest abandonment can be caused by the presence of mammal predators (Madsen, Bregnballe & Hastrup 1992; Cervenci *et al.* 2011), our results suggests that this is not the case for non-predatory mammals, such as grazing cattle. Nest abandonment in birds can be caused by numerous factors including adverse weather, observer effects, recreational disturbance or predation of the adults (Livezey 1980; Nichols *et al.* 1984; Lord *et al.* 2001). Therefore, nest abandonment in our data may have been influenced by factors that were beyond the scope of this study.

Although our results suggested that grazing with young cattle does cause more Redshank nest mortality than grazing with adult cattle, this should be treated with some caution. Grazing with young cattle was more common in the UK whilst grazing with adult cattle was more common in the Wadden sea region. British saltmarshes typically have an extensive network of tidal creeks occupying a large part of the total marsh area (Adam 1990; Boorman 2003). However, many of the Wadden sea saltmarshes do not have extensive creek networks (Hofstede 2003). It is therefore possible that the German and Dutch saltmarshes are more accessible to livestock, resulting in different patterns of grazing and nest trampling. However, the increased nest predation rates were expected as young cattle have higher energy requirements and therefore consume more vegetation (Basarab *et al.* 2003), which leads to increased nest predation (Sharps *et al.* 2015). Furthermore, Beintema and Muskens (1987) found that in agricultural pasture systems, grazing with adult cattle resulted in a 48% probability of surviving the incubation period, compared with a 13% probability with young cattle, which was attributed to increased rates of nest trampling. This suggests that young cattle do cause more Redshank nest mortality than adult cattle, but further research may be necessary to quantify the difference that changing from young to adult cattle would make on saltmarshes with older creek networks.

Our results showed that Redshank nest loss to predation is lower closer to the landward extent of the saltmarshes, regardless of conservation management. This suggests that more of the Redshank nest predation on saltmarshes is caused by avian predators than mammals. This is because ground predators may predate nests closer to the sea wall, however avian predators may cause nest mortality anywhere on the saltmarsh (Montevicchi 1977). In some habitats, lethal control of mammalian predators is used as a management tool to reduce shorebird nest predation rates (Bolton *et al.* 2007). Unfortunately, data on predator control was not available. However, the results of this study may suggest that increasing current rates of mammal control at the saltmarshes studied would be unlikely to benefit Redshank conservation. Sharps *et al.* (2015) showed that high rates of nest predation arise indirectly through conservation management, as grazing reduces the vegetation height (Sharps 2015a) and therefore makes it easier for predators to find nests. This suggests that nest predation rates could be reduced through conservation management change, by reducing livestock densities.

Local predator communities may vary between different regions and years, with nest predation rates likely to be dependent on the habitat matrix of surrounding landscapes (Laidlaw *et al.* 2013). However, understanding the full range of landscape features that lead to high rates of nest predation was beyond the scope of this study. Model fits of trampling levels with increasing grazing (Figure 5.3) should be treated with some caution, as they suggest livestock nest loss to trampling even with no cattle present. This is highly improbable and may arise from fitting the nest survival model through data which included occasional nest loss to trampling at very low livestock densities. When calculating livestock density our intention was to use a procedure similar to that used by land managers. These methods can vary but management guidelines e.g. Adnitt *et al.* (2007) tend to focus on simple calculations of numbers of livestock divided by the size of the site in hectares, taking into account the part of the saltmarsh that is accessible to livestock and the number of months animals are present. This approach has a known constraint – that grazing across a saltmarsh can be localised closer to the sea wall (Patterson & Burrows 1998; Sharps 2015b). However, calculating precise grazing pressure to account for spatial variation was beyond the scope of this study. Each young cattle on our study saltmarshes could be

translated to 0.80 LSU, and each adult cattle 0.65 LSU (DEFRA 2010), however our calculations of cattle density are based on numbers of individuals.

This study did not account for effects of nest loss to tidal flooding, however Redshank nest survival on saltmarshes is also negatively affected by nest flooding (Thompson & Hale 1991; Norris 2000; Smart 2005). Losses of between 3 – 40% of all observed nests have been reported as lost to tidal flooding, a figure which can vary between different years and estuaries (Thompson & Hale 1991; Smart 2005). Climate change may increase tidal flooding. In the Wadden sea, for example, changes in wind patterns have resulted in more storm surges during bird breeding seasons, resulting in more frequent and more catastrophic flooding of nests (van de Pol *et al.* 2010). It is unclear if increased tidal flooding has played a part in Redshank population declines.

This results of this work may appear to contrast with the findings of Sharps *et al.* (2015), who found that even light grazing of British saltmarshes would be unlikely to sustain stable populations of Redshank (98% risk of trampling at 0.82 ha⁻¹ seasonal cattle density and the 95% risk of predation at an annual cattle density of 0.55 ha⁻¹). However, the saltmarshes studied by Sharps *et al.* (2015) were on British saltmarshes grazed by young cattle. This is not only common practice on British saltmarshes, but is advocated in the UK Environment Agency guidelines of 0.7 - 1 young cattle ha⁻¹ between April and October (Adnitt *et al.* 2007). The current study therefore strengthens and builds upon the conclusions of Sharps *et al.* (2015), as we suggest that these high rates of trampling and predation might be reduced by grazing with adult cattle.

Unlike Black-tailed Godwit *Limosa limosa*, Oystercatcher *Haematopus ostralegus* and Lapwing *Vanellus vanellus*, Redshank do not attack potential nest predators (Gochfeld 1984; Nethersole Thompson & Nethersole Thompson 1986). Instead, Redshank rely on a patchy vegetation structure as their main form of nest defence, building a cryptic nest obscured by tall vegetation and quietly leaving the nest upon detection of a predator (Cervenci *et al.* 2011; Mandema *et al.* 2014a). Management by grazing (occupied) was found to lead to the lowest rates of nest mortality, accounting for effects of both predation and livestock trampling. Lighter grazing results in a more uneven patchy sward with diverse heights (Jensen 1985; Kiehl *et al.* 1996). Mowing of saltmarshes, which is widely practiced in the Wadden sea region, can result in a more

uniform sward, with heights dependent on the mowing regime (Bakker 1978). As the probability of nest failure was highest in mown saltmarshes than other land uses, it is likely that this practice leaves nests more vulnerable to predators by reducing vegetation height. Nest predation rates can be high when nests in long vegetation are exposed to severe weather, as this makes them more obvious to predators (Webb *et al.* 2012). As exposed coastal habitats, saltmarshes are exposed to high wind speeds (Allen & Pye 1992; Bockelmann *et al.* 2002). This could explain the high rates of nest failure in saltmarshes with no active conservation management, as this leaves an abundance of very long vegetation which Redshank use for nesting (Thyen & Exo 2003; Sharps 2015a).

Redshank breeding densities are higher in the structurally diverse vegetation on grazed saltmarshes (Norris *et al.* 1997), perhaps because species have evolved to recognise habitat that can lead to higher rates of survival (Battin 2004). Habitat use affects the fitness of animals through variation in environmental conditions and resource availability, which generates selective pressure for habitat choice (Pulliam 2000). This in turn influences the survival and reproduction of individual birds (Brown 1969; Fuller 2012). However, the density of animals in a habitat can be a misleading indicator of habitat quality (Van Horne 1983) when a species can preferentially use habitat which acts as an 'ecological trap' by lowering breeding success (Best 1986; Schlaepfer, Runge & Sherman 2002). We suggest that the practice of grazing with young cattle on British saltmarshes may be acting as an ecological trap for Redshank, if the vegetation structure resembles that of saltmarshes grazed by adult cattle. This is more likely to be the case at the start of the grazing season, which coincides with the peak period that Redshank are selecting nest locations.

Further investigations into the ideal grazing management practices for Redshank may be necessary. If UK Environment Agency guidelines are followed, lightly grazed saltmarshes would have livestock present from April until October (Adnitt *et al.* 2007). Although our results suggest that Redshank breeding populations may benefit from grazing with older cattle we are by no means suggesting that this is the only solution to reverse the decline in population numbers. Reducing livestock densities is also likely to reduce nest predation and trampling rates. In conclusion, this work suggests that managing saltmarshes with young grazing cattle, with mowing or with no active management can lead to high rates of Redshank nest loss due to predation and

trampling. Nest survival rates improve when land is managed with grazing adult cattle and at low livestock densities.

5.6 Tables

Table 5.1: List of saltmarshes included in this study.

Showing conservation management techniques ('Cons. Man. '), number of nests and percentage predated (%P), trampled (%T) and abandoned (%A). No active = No active management. Unoccupied = Rotational (unoccupied). All saltmarshes that are identified as being in Germany or the Netherlands are in the Wadden sea region.

Saltmarsh region & year(s)	Cons. Man.	Size (ha)	No. nests	%P	%T	%A
Essex (UK), 2004	No active	181 ha	38	38%	0	0
Jadebuurg East (Germany), 2007- 2008	Mown	8 ha	11	36%	0	0
Noard Fryslân Bûtendyks (Netherlands), 2010 - 2011	No active	8 ha	48	34%	0	15%
	Grazed	11 ha	28	29%	7%	4%
	Unoccupied	11 ha	5	20%	0	20%
Nordender Groden (Germany), 2007 - 2008	Mown	9 ha	7	71%	0	14%
	No active	9 ha	3	100%	0	0
Norderland (Germany), 2007 - 2008	No active	9 ha	3	67%	0	0
	Grazed	9 ha	9	22%	11%	0
Petersgroden (Germany), 2000 - 2008	No active	56 ha	195	77%	0	10%
	Grazed	8 ha	16	50%	19%	19%
	Mown	56 ha	69	72%	0	10%
Ribble Estuary (UK), 2012	No active	257 ha	5	0	0	0
	Grazed	58 – 704 ha	31	32%	23%	0
Wangerooge (Germany), 2003, 2005 – 2006.	Unoccupied	62 ha	9	44%	0	0
	No active	658 ha	37	16%	0	14%
The Wash (UK), 2003, 2005	No active	117ha	41	29%	0	12%
	Grazed	477 ha	12	25%	33%	0

Table 5.2: Livestock density and cause of mortality for grazed saltmarshes. Showing number of nests and percentage predated (%P), trampled (%T) and abandoned (%A). ACD = Annual cattle density, SCD = Seasonal cattle density. All saltmarshes that are identified as being in Germany or the Netherlands are in the Wadden sea region.

Saltmarsh region	Size (ha)	ACD	SCD	Livestock type	No. nests	% P	% T	% A
Noard Fryslân Bûtendyks (Netherlands)	11	0	0	Adult cattle	5	20%	0	20%
	11	0.19	0.45	Adult cattle	9	11%	0	11%
	11	0.38	0.91	Adult cattle	12	42%	16%	0
	11	N/A	N/A	Horses	7	18%	0	0
Norderland (Germany)	9	0.33	0.98	Adult cattle	3	33%	0	0
	9	0.34	1.02	Adult cattle	6	16%	16%	0
Petersgroden (Germany)	8	0.42	1.00	Adult cattle	16	50%	19%	19%
Ribble Estuary (UK).	127	0.08	0.15	Young cattle	5	40%	20%	0
	237	0.17	0.34	Young cattle	10	10%	20%	0
	62	0.25	0.00	Young cattle	9	44%	0	0
	58	0.43	0.26	Young cattle	8	50%	0	0
	704	0.55	0.82	Young cattle	8	38%	50%	0
The Wash (UK).	477	0.05	0.13	Young cattle	12	25%	33%	0

Table 5.3: Results of program MARK nest loss to trampling and predation analyses. ‘ ΔAIC_c ’: the difference between the model in question, and the top model. $\Delta AIC_c < 2$ suggests both models are plausible (Burnham & Anderson 2002). Akaike weights represent the relative likelihood of the models (i.e. $\exp(-0.5 * \Delta AIC_c)$ for each model, divided by the sum of these values across all other models). Model likelihood is the support of the model in question relative to other models in the candidate set. \hat{C} values between 1 and 3 suggest the model structure is adequate (Lebreton *et al.* 1992). \hat{C} values between 3 and 4 suggest only a minor degree of overdispersion (Anderson, Burnham & White 1994). \hat{C} values between 0 and 1 suggest only a minor degree of underdispersion (Class, Klueen & Brommer 2014). SCD = Seasonal cattle density. ACD = Annual cattle density. DSW = Distance to the sea wall. The “constant survival” model refers to the model with no explanatory variables. Table parts A and B are on this page and parts C and D are on the next page.

A) Hypothesis 1: Does grazing result in more Redshank nest mortality than other conservation management techniques?

Response variable (Cause of Daily nest failure)	Model	AICc	$\Delta AICc$	Akaike Weight	Model Likelihood	\hat{C}
i) Predation.	Cons. Man.	1659.04	0.00	1.00	1.00	2.93
	Constant survival	1673.62	14.58	0.00	0.00	2.95
ii) Predation or trampling.	Cons. Man.	1726.02	0.00	0.82	1.00	3.05
	Constant survival	1728.99	2.97	0.18	0.23	3.05

B) Hypothesis 2: Does grazing result in more Redshank nest abandonment than other conservation management techniques?

Response variable (Cause of Daily nest failure)	Model	AICc	$\Delta AICc$	Akaike Weight	Model Likelihood	\hat{C}
i) Nest abandonment.	Constant survival	448.55	0.00	0.88	1.00	0.79
	Cons. Man.	452.49	3.94	0.12	0.14	0.79

C) Hypothesis 3: Does grazing with young cattle cause more Redshank nest mortality than grazing with adult cattle?

Response variable (Cause of Daily nest failure)	Model	AICc	Δ AICc	Akaike Weight	Model Likelihood	\hat{c}
i) Trampling.	SCD + livestock age	121.19	0.00	0.51	1.00	1.15
	SCD * livestock age	121.25	0.06	0.49	0.97	1.14
	livestock age	132.41	11.22	0.00	0.00	1.27
	Constant survival	143.77	22.58	0.00	0.00	1.35
	SCD	144.56	23.37	0.00	0.00	1.34
ii) Predation.	ACD * livestock age	208.92	0.00	0.58	1.00	2.02
	ACD + livestock age	210.08	1.16	0.32	0.56	2.04
	ACD	212.49	3.57	0.10	0.17	2.06
	livestock age	220.38	11.45	0.00	0.00	2.14
	Constant survival	221.29	12.36	0.00	0.00	2.15

D) Hypothesis 4: Is Redshank nest loss to predation lower closer to the landward extent of the saltmarshes, regardless of conservation management technique used?

Response variable (Cause of Daily nest failure)	Model	AICc	Δ AICc	Akaike Weight	Model Likelihood	\hat{c}
i) Predation.	DSW + Cons. Man.	1223.39	0.00	0.81	1.00	3.20
	DSW * Cons. Man.	1226.59	3.2	0.16	0.20	3.22
	Cons. Man.	1230.31	6.92	0.02	0.03	3.22
	DSW	1236.90	13.51	0.00	0.00	3.23
	Constant survival	1238.88	15.48	0.00	0.00	3.23

5.7 Figures

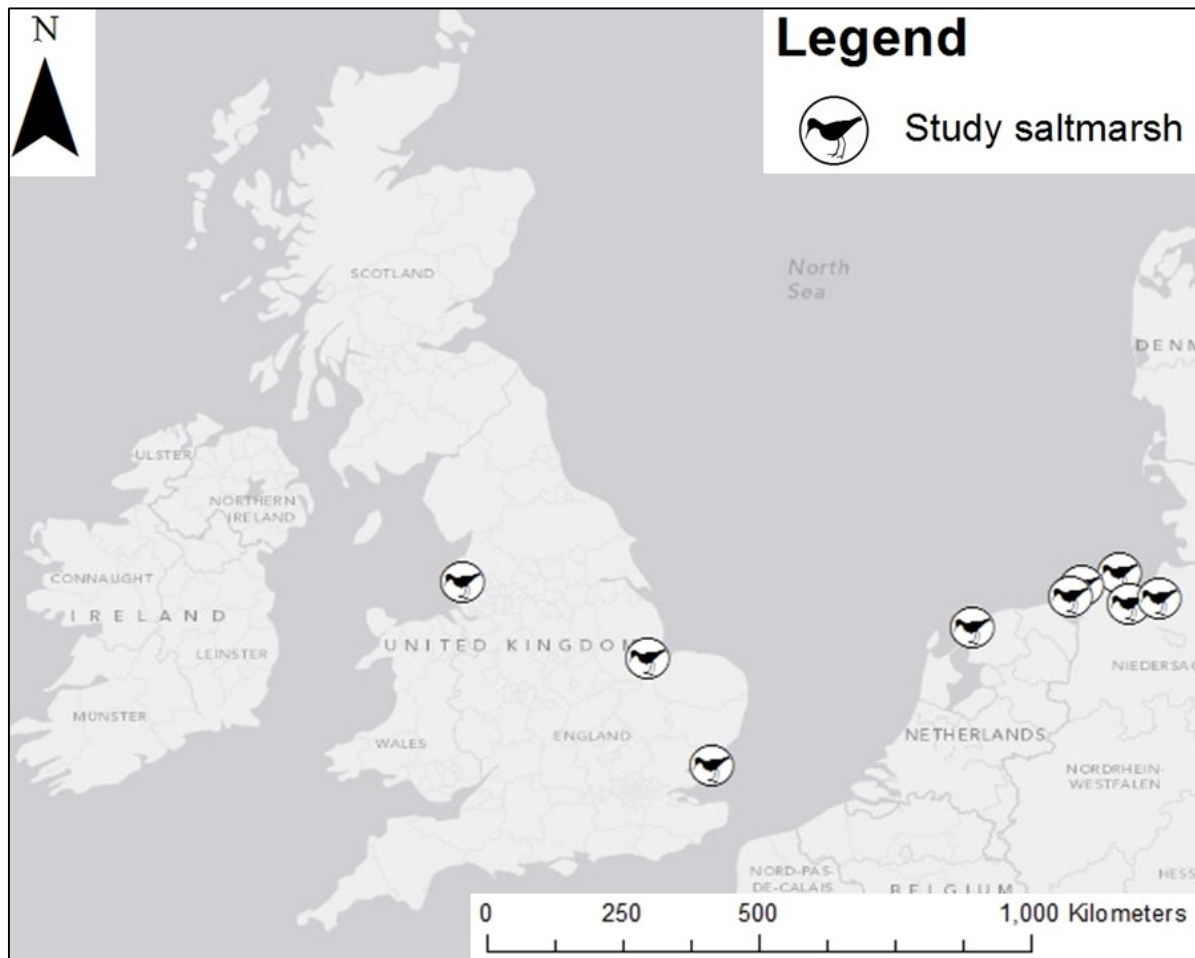


Figure 5.1: Map showing the location of the study saltmarshes.

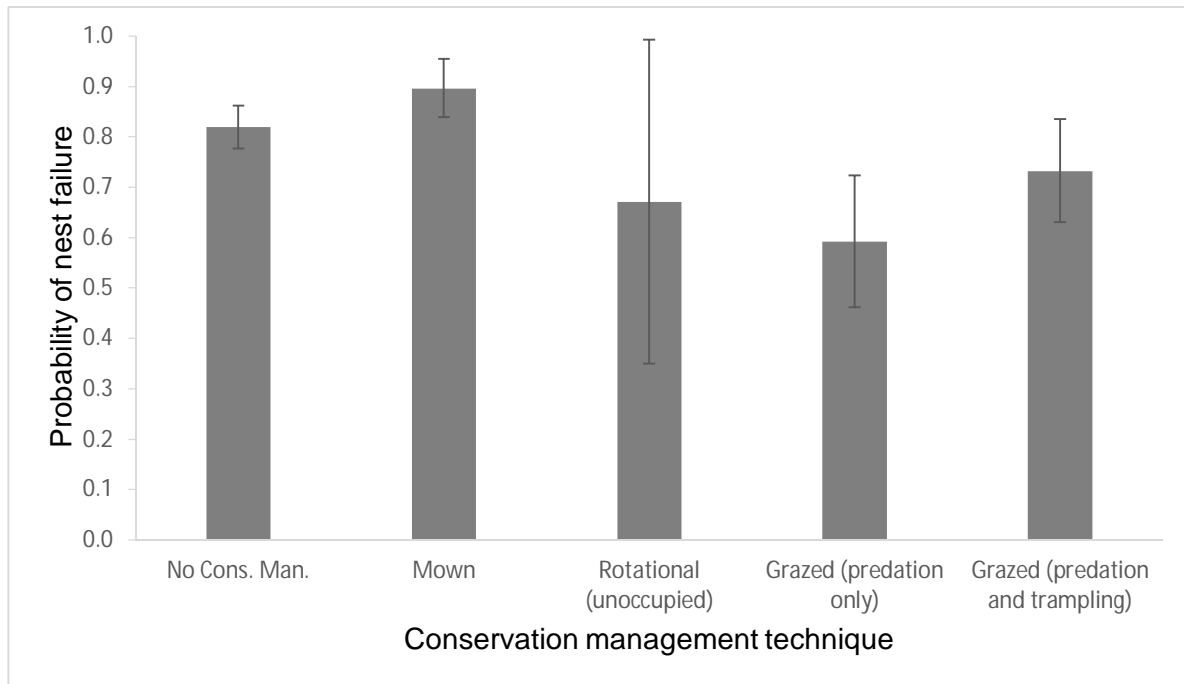


Figure 5.2: Predicted effects of conservation management technique on Redshank nest mortality.

Error bars represent 95% confidence intervals. All categories show probability of nest predation apart from the “Grazed (predation and trampling)” category which shows the probability of nest loss to either predation or trampling.

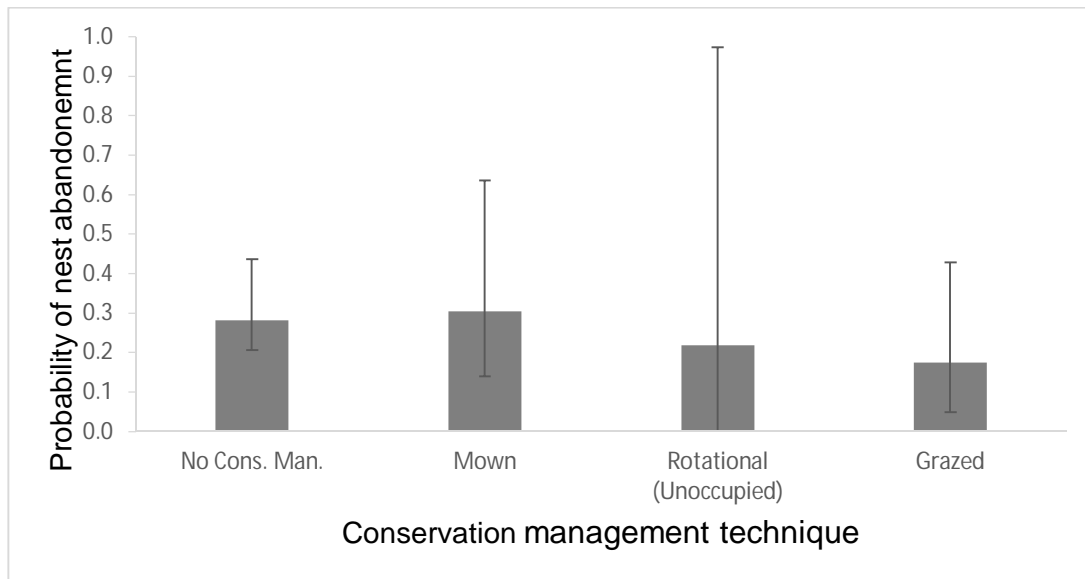


Figure 5.3: Predicted effects of conservation management technique on the probability of Redshank nest abandonment. Error bars represent 95% confidence intervals. No 'Cons. Man.' = no active conservation management technique used.

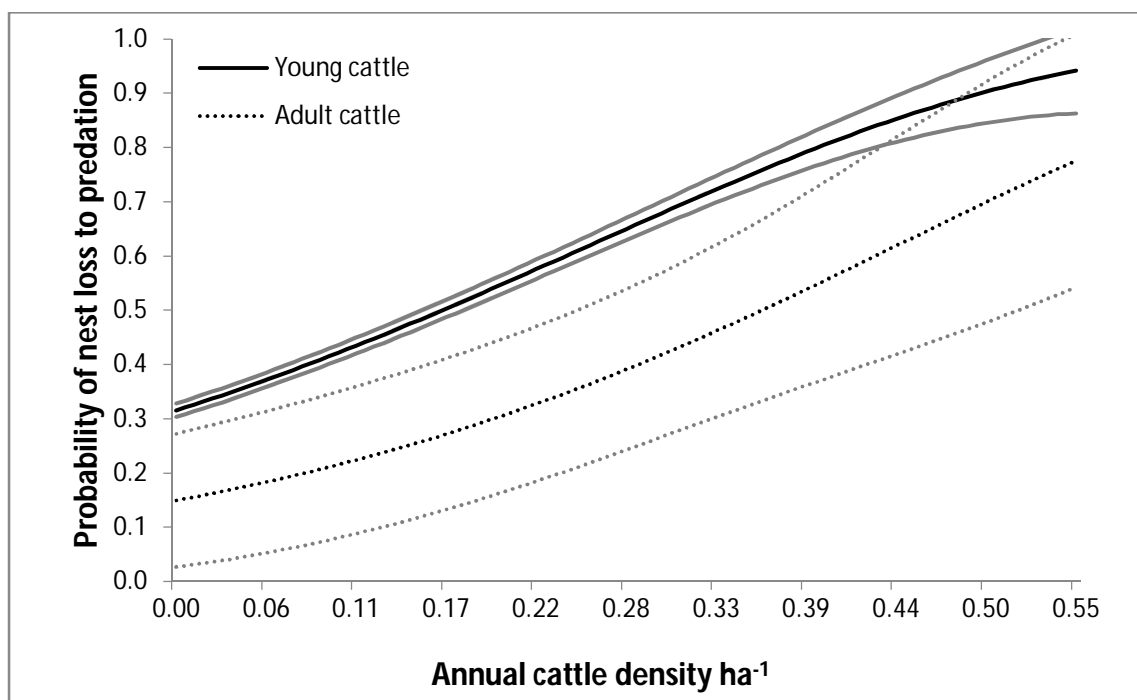
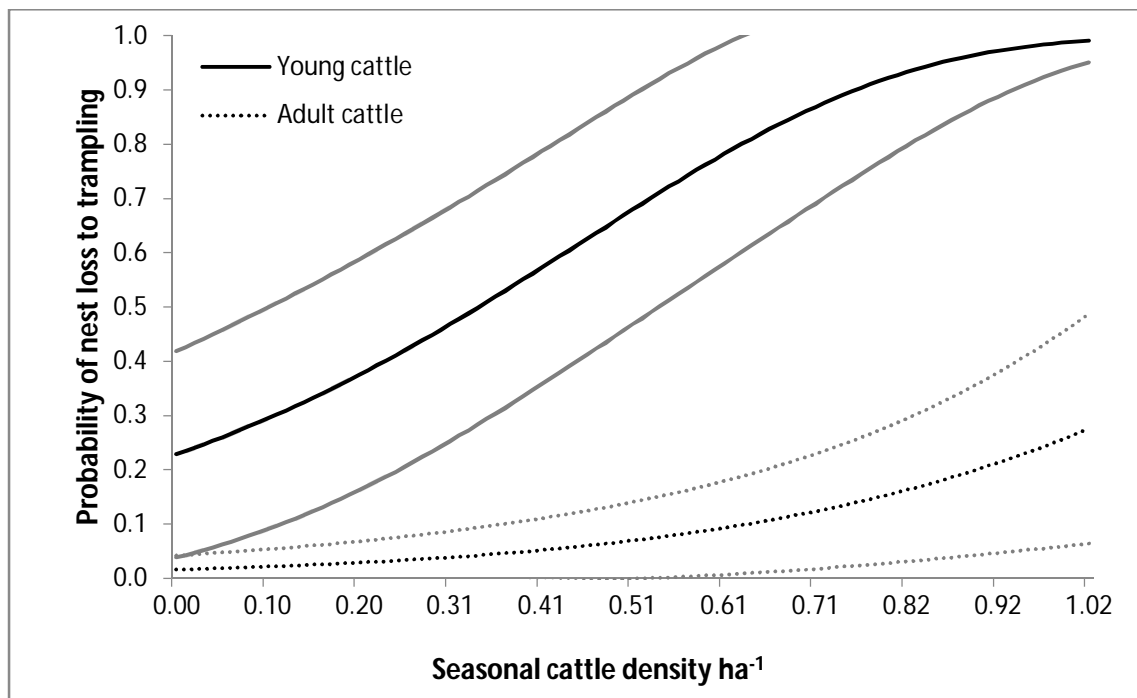


Figure 5.4: Statistical fit of nest survival models showing predicted effects of livestock density and livestock type on Redshank nest mortality. Showing trampling probability (top) and predation probability (bottom) across a 24 day nest incubation period. Light grey lines indicate confidence intervals.

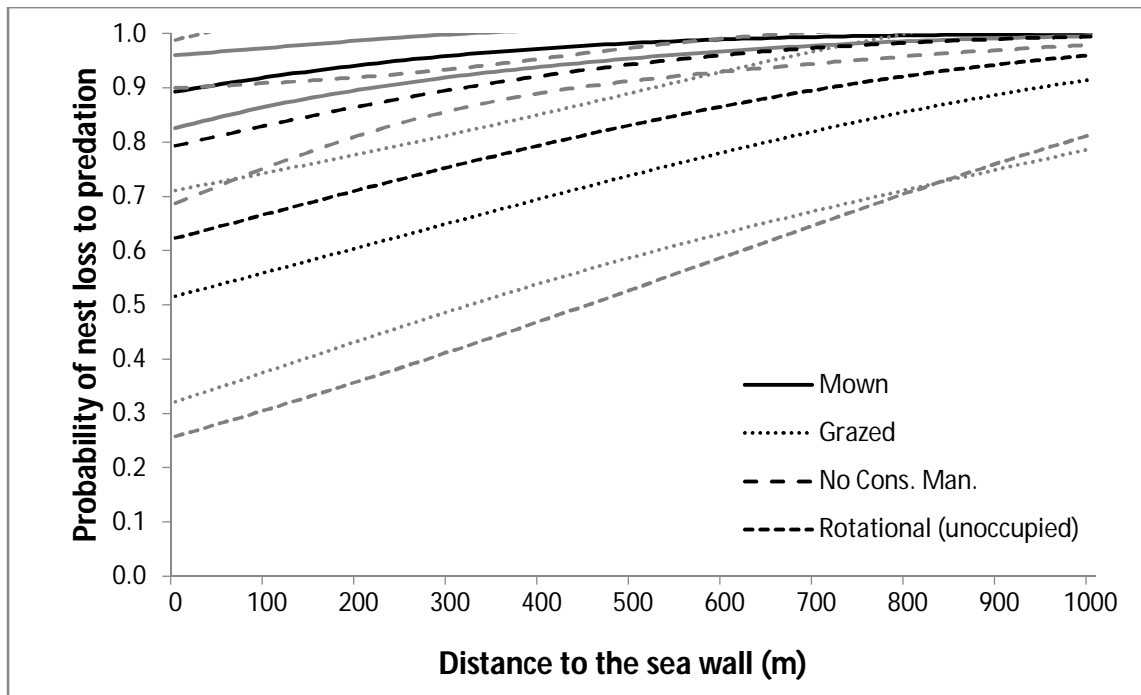


Figure 5.5: Statistical fit of nest survival models showing predicted effects of distance to the sea wall on Redshank nest mortality to predation. Light grey lines indicate confidence intervals. No 'Cons. Man.' = no active conservation management technique used.

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6 Overall discussion

6.1 Overview and conclusions

In order to investigate the effects of saltmarsh conservation grazing on breeding Common Redshank *Tringa totanus*, this thesis considered several factors. Chapter two aimed to establish if grazing results in nest mortality directly through livestock trampling and / or indirectly through grazer modification of habitat that accelerates predation risks. Higher grazing pressure reduced Redshank nest survival rates, even across a range of saltmarshes grazed at < 1 cattle ha^{-1} . Results demonstrated that higher grazing pressure can result in increased nest mortality directly through cattle trampling. Results also showed a link between livestock density and nest predation. Higher grazing pressure resulted in increased nest mortality indirectly, possibly through grazer modification of habitat leading to increased predation risk. Mortality due to predation was positively related to the annual grazing management regime, but not to cattle presence in the breeding season. These results suggest that either previously recommended stocking levels may be too high, or that Redshank breeding areas are more intensively grazed than intended by land managers, due to the non-homogeneous distribution of cattle.

The aim of chapter three was to investigate which nest vegetation conditions Redshank select at different spatial scales and if grazing limits the availability of higher quality nest sites. This was studied on the same six saltmarshes as chapter two, by characterising vegetation height and species composition at the Redshank nests and at control locations, selected at random. Vegetation was taller at Redshank nests than in control locations and had a higher cover of the grass *F. rubra*. Taller vegetation with more *F. rubra* was also found next to and in the wider area around nests. Redshank nests in saltmarshes exposed to lower grazing were found in taller vegetation. The proportion of *F. rubra* at Redshank nests increased in saltmarshes with higher livestock densities. As nesting in shorter vegetation is likely to result in higher nest predation rates, results suggest that even light conservation grazing at < 0.55 cattle $\text{ha}^{-1} \text{y}^{-1}$ (approximately 1 cattle ha^{-1}) can result in Redshank nesting in lower quality habitat.

Chapter four aimed to investigate the drivers of spatial and temporal variation in livestock distribution and quantify effects on avian nest loss due to trampling, to determine if Redshank nesting areas are exposed to higher trampling risk during the breeding season. Redshank usually nest in vegetation communities associated with the high-mid marsh, which is typically closer to the landward side of the saltmarsh (Norris *et al.* 1997). The results of chapter four show that grazing is generally at its highest on bird nesting areas during the breeding season. The actual livestock density is much higher in these areas than expected if using standard calculations of livestock density. This chapter demonstrates that livestock grazing on saltmarshes is not homogenous. Therefore, it is likely that high rates of Redshank nest loss to trampling are due, in part, to this non-homogeneity of grazing during the breeding season.

By analysing a large sample of Redshank nest data from both Britain and the Wadden sea region, chapter five aimed to quantify the effects of different conservation management techniques (grazing, mowing or no active management) on Redshank nest survival. The results of this chapter suggested that Redshank nest survival rates were higher in grazed saltmarshes than those that were mown or not actively managed. However, grazing with young cattle caused more Redshank nest mortality than grazing with adult cattle as predicted nest survival rates were shown to improve when land is grazed with adult cattle and at lower livestock densities.

The findings of chapter five, that grazing leads to a lower probability of Redshank nest mortality than other management techniques, may appear to contrast with the findings of chapter 2, which found that even light grazing of British saltmarshes would be unlikely to sustain stable populations of Redshank due to high rates of nest mortality. However, chapter two also highlighted that all of the study sites were grazed by young cattle. This is not only common practice on British saltmarshes, but is advocated in management guidelines (Adnitt *et al.* 2007). Chapter five suggests that rates of Redshank population declines on UK saltmarshes could be reduced by grazing with adult cattle.

While Norris *et al.* (1997) concluded that light cattle grazing of around 1 cattle ha⁻¹ is likely to maintain a high breeding density of Redshank, the study did not account for

nest survival. Birds preferentially selecting nesting sites that result in low nest survival in landscapes modified by humans is not a new phenomenon (Best 1986; Schlaepfer, Runge & Sherman 2002). If this is the case for Redshank on saltmarshes, it could explain some of the contradictions between the findings of this study and the recommendations of Norris *et al.* (1997). The density of animals in a habitat can be a misleading indicator of habitat quality (Van Horne 1983) as species can preferentially use habitat which acts as an 'ecological trap' by lowering breeding success (Best 1986; Schlaepfer, Runge & Sherman 2002). The results of this thesis suggest that saltmarshes grazed with young cattle may act as an ecological trap for Redshank because they show a preference for nesting sites that can result in almost total nesting failure.

All saltmarshes included in chapters 2 and 3 were within a nature reserve, but those grazed at the highest annual cattle density of 0.55 ha⁻¹ or with 0.82 ha⁻¹ seasonal cattle density, would be unlikely to sustain stable populations of Redshank due to the 98% risk of trampling and the 95% risk of predation. However, cessation of grazing in previously grazed saltmarshes can result in reductions in breeding Redshank as the habitat becomes dominated by tall uniform vegetation (Norris *et al.* 1997) and as suggested in chapter five, grazing can lead to higher rates of Redshank nest survival than other conservation management techniques. Therefore, this work does not suggest that stopping livestock grazing on saltmarshes altogether will result in increased breeding populations of Redshank. It is widely accepted that light grazing can be beneficial to Redshank and is therefore an important part of saltmarsh management. Furthermore, livestock grazing in saltmarshes can drive abundance and diversity of invertebrate prey (Ford *et al.* 2013).

Conservation management recommendations based on this work are highlighted in Table 6.1. Section 6.2 discusses the need for conservation management and research to balance the requirements of different bird species. Section 6.3 discusses the need to balance nature conservation with agricultural and economic productivity, and section 6.4 summarises the discussion and examines the need for future research.

6.2 Balancing the requirements of different bird species

Grazing of domestic livestock has been and continues to be one of the main drivers generating and maintaining heterogeneity of natural and semi-natural areas around the world (Senft *et al.* 1987; Cid & Brizuela 1998; Jacobo *et al.* 2006). As organisms respond differently to the intensity of livestock grazing, it can be used as a conservation management tool to create or limit suitable habitat for different species (WallisDeVries 1998; Bouchard *et al.* 2003; Van Klink 2014). Grazing modifies habitat complexity, affecting all aspects of avian habitat quality and selection from the microhabitat to regional scale (Wiens 2009; Mandema 2014), resulting in changes in bird composition and abundance associated with these habitats (Roxburgh, Shea & Wilson 2004; Mandema 2014). Habitat requirements of birds are highly variable depending on both the species in question and the role in which the habitat plays in its lifecycle (Law & Dickman 1998; Thyen *et al.* 2008). In open landscapes this can include species that use short grasslands (Myers & Myers 1979; Tucker, Davies & Fuller 1994; Lanctot *et al.* 2002) species adapted to live in tall grasslands (Isacch *et al.* 2004; Pavel 2004), and generalist species associated with various vegetation types (Fillooy & Bellocq 2007; Sweeney *et al.* 2010). Therefore, different species respond differently to different types and intensities of grazing (Zalba & Cozzani 2004; Mandema 2014).

Saltmarshes play various roles in the lifecycles of a number of bird species. They act as high tide refuges for birds feeding on adjacent mudflats, as breeding sites for waders, gulls and terns and as a source of food for passerines (Davidson 1991). In winter, grazed saltmarshes are used as feeding grounds by large flocks of wildfowl (Mandema *et al.* 2014b). To avoid creating unsuitable conditions for the non-target species, conservation management needs to be carried out in sympathy with the requirements of other species that use the target habitat or ecosystem (Simberloff 1998). Therefore the needs of bird species other than Redshank should also be considered (Greenhalgh 1971; Greenhalgh 1975; Thyen *et al.* 2008). A scarcity of scientific literature on the importance of UK saltmarsh habitats for breeding birds existed until 1985 when the first national saltmarsh breeding Redshank survey was carried out (Cadbury, Green & Allport 1987). This survey not only recorded Redshank populations, but also provided information on other bird species breeding on UK

saltmarshes (Allport, O'Brian & Cadbury 1986; Cadbury, Green & Allport 1987). Given the widespread declines in breeding wader populations and their associated breeding habitats, identifying the key habitat requirements of these species is crucial in order to improve management of the remaining areas. Allport, O'Brian and Cadbury (1986) identified three breeding wading bird species that were particularly dominant on saltmarshes: Redshank, Northern Lapwing *Vanellus vanellus* and Eurasian Oystercatcher *Haematopus ostralegus*. Furthermore, saltmarshes support breeding populations of passerines, such as Eurasian Skylark *Alauda arvensis* and Meadow Pipit *Anthus pratensis* (Mandema 2014; Mandema *et al.* 2014a; Van Klink *et al.* 2014), gulls (*Laridae*), terns (*Sternidae*) and wildfowl (*Anatidae*) (Greenhalgh 1971; Greenhalgh 1975). In order to consider management approaches which balance the requirements of different bird species, some examples of how recommendations provided for breeding Redshank may benefit or harm other species are discussed below.

Allport, O'Brian and Cadbury (1986) found Lapwing breeding on 33% of 77 saltmarshes surveyed around the UK. Although very little is known about the current status of the Lapwing population on British saltmarshes, declines have been reported in the general population in many parts of north-west Europe, including the UK and Ireland (Hudson, Tucker & Fuller 1994; Tucker, Davies & Fuller 1994). Most declines are likely to be due to agricultural intensification, however wetland drainage and egg collecting have also been contributing factors (Hudson, Tucker & Fuller 1994; Tucker, Davies & Fuller 1994; del Hoyo, Elliott & Sargatal 1996). In common with Redshank, Lapwing breed from April to July but form a shallow scrape in very short vegetation, where they generally nest in solitary pairs (Hayman, Marchant & Prater 1991; del Hoyo, Elliott & Sargatal 1996). However, when breeding in optimal habitat conditions, these pairs often nest close together or semi-colonally (Johnsgard 1981; Trollet 2003). Lapwing appear to show a preference for breeding on wet natural grasslands and hay meadows (del Hoyo, Elliott & Sargatal 1996; Trollet 2003), although they may also breed on grassy moors, swampy heaths, bogs and arable fields (Johnsgard 1981; del Hoyo, Elliott & Sargatal 1996). As Lapwing nest in very short grass (Devereux *et al.* 2004), it is likely that they would benefit from the short swards produced by intensive grazing on saltmarshes, however there is a need for further research to gain a more complete understanding of Lapwing requirements in this habitat. Lapwing are

also sensitive to nest trampling (Beintema & Muskens 1987; Baines 1990) therefore Lapwing populations on saltmarshes may also benefit from a form of rotational grazing, as suggested for Redshank. As Lapwing require a more open vegetation sward than Redshank, they may benefit from heavy grazing either before or after the breeding season, however overall reductions in livestock density may be detrimental to Lapwing populations as they would limit the availability of short vegetation swards.

Breeding Oystercatchers were found on 75% of the 77 saltmarshes surveyed by (Allport, O'Brian & Cadbury 1986). Although little is known about the status of the Oystercatcher population breeding on British saltmarshes, overall between 1972 and 2011 Oystercatchers experienced a 28% range expansion in the UK (Balmer *et al.* 2013). The main reason for this increase is thought to be behavioural, as Oystercatchers adapted their use of habitat to allow the exploitation of inland areas (Sutherland & Norris 2002; Balmer *et al.* 2013). Oystercatchers typically breed in coastal areas such as saltmarshes, sand and shingle beaches, dunes and cliff-tops with short grass and rocky shores. When breeding inland, Oystercatchers use habitats along the shores of lakes, reservoirs and rivers or on agricultural grass and cereal fields (Hayman, Marchant & Prater 1991; del Hoyo, Elliott & Sargatal 1996). The nest is a shallow scrape on the ground in either open or short vegetation (Hayman, Marchant & Prater 1991). Oystercatchers breed from April to July, usually in solitary pairs (Nethersole Thompson & Nethersole Thompson 1986; Ens 1992). Oystercatchers appear to show a preference for nesting sites in areas of short vegetation on more lightly grazed saltmarshes (Mandema *et al.* 2014a). As ground nesting birds, they are also vulnerable to nest trampling by livestock, therefore they would likely benefit from some of the measures suggested for Redshank, for example reduced livestock density and grazing with older cattle to reduce trampling risk.

Finding optimal grazing conditions to benefit Lapwing and Oystercatcher populations may also benefit Redshank. For many birds, predation is the largest cause of reproductive failure (Ricklefs 1969; Macdonald & Bolton 2008). Consequently, a multitude of adaptations to reduce the risk of nest predation has evolved (Martin 1995). As predominantly ground-nesters, waders show a variety of anti-predator adaptations, including passive and active nest defence (Gochfeld 1984). Unlike Redshank, Oystercatcher and Lapwing both practice active predator defence strategies, directly

attacking potential nest predators (Dyrzcz, Witkowski & Okulewicz 1981; Gochfeld 1984; Nethersole Thompson & Nethersole Thompson 1986; Ens 1991). Instead, Redshank simply build a cryptic nest obscured by tall vegetation and leave the nest upon approach of a predator (Cervencel *et al.* 2011; Mandema *et al.* 2014a). However, as Redshank can take advantage of the nest defence behaviour of other waders, nesting in close association with larger colonies of other species can result in lower nest mortality for Redshank. For example, Dyrzcz, Witkowski and Okulewicz (1981) found that for species which do not actively defend nests, the probability of nest survival was higher close ($\leq 39\text{m}$) to nests of wader species that actively defend nests, whereas further away from these species ($\leq 95\text{m}$), the probability of nest predation was high. They also suggested that these effects would be stronger in large colonies of breeding waders, which are more likely to be able to successfully defend against predators. Therefore, if populations of Lapwing and Oystercatcher increased on saltmarshes, the Redshank population may also increase.

Kalejta-Summers (1997) describes the diet of wintering passerines on saltmarshes, and recent work describes Meadow Pipit *Anthus pratensis* foraging ecology on saltmarshes during the breeding season (Van Klink *et al.* 2014). However literature on passerine use of saltmarshes for nesting is scarce. Eurasian Skylark *Alauda arvensis* and Meadow Pipit are primarily species of open habitats, either uncultivated or low-intensity agriculture, such as pasture, bogs, and moorland, however, they also occur in low numbers in arable croplands (Hagemeijer & Blair 1997; Chamberlain *et al.* 1999; Vanhinsberg & Chamberlain 2001; Donald 2004; Pavel 2004). Although Allport *et al.* (1986) make no mention of breeding Skylark and Meadow Pipit populations on saltmarshes, anecdotal reports suggest these species are abundant in these habitats (pers. obs.). Overall, Skylark populations in Britain declined by 58% between 1970 and 2010, while the British Meadow Pipit population declined by 46% between 1970 and 2010 (Eaton *et al.* 2012). A number of other farmland bird species also declined during this period and this coincided with a number of changes in agricultural practice (Campbell & Cooke 1997; Chamberlain *et al.* 1999). It is likely that a reduction in the number of nesting attempts made by each pair in a breeding season, linked to changes in crop type and structure, is a significant contributing factor (Wilson *et al.* 1997). Skylark and Meadow Pipit breed from April until July in a clump of grass or in a concealed depression in the ground (Vanhinsberg & Chamberlain 2001; Donald

2004). The choice of nesting site is influenced by the height and density of vegetation, and the ideal vegetation height is 20-50 cm (Wilson *et al.* 1997; Vanhinsberg & Chamberlain 2001; Donald 2004). As grazing reduces sward height, this suggests that zero or very light grazing on saltmarshes may be the most suitable management for Skylark and Meadow Pipit breeding. This suggests that the requirements of Skylark and Meadow Pipit are in direct conflict to that of Oystercatcher and Lapwing, which require more open areas. However, as Redshank require a patchy vegetation structure of diverse sward heights, management of saltmarshes for Redshank may indirectly benefit Skylark, Meadow Pipit, Oystercatcher and Lapwing.

While considering the competing breeding habitat requirements of the wader and passerine species found on saltmarshes, management should also take into account the requirements of other species. In addition to their importance for breeding birds, saltmarshes act as high tide retreats for birds otherwise feeding on adjacent mudflats, and as overwintering and stopover feeding grounds by large flocks of wildfowl, passerines and waders (Prater 2010). These different species again have conflicting requirements – for example breeding passerines need long grasses (Benton, Vickery & Wilson 2003) but overwintering geese (and foraging passerines) need shorter swards (Mandema *et al.* 2014b; Van Klink *et al.* 2014). Conservation management may aim to benefit one species, a selection of target species, or all bird communities (Carignan & Villard 2002; Groom, Meffe & Carroll 2006). Although the habitat management changes suggested by this thesis may help breeding populations of Redshank, further research is necessary to gain a complete understanding of the optimal management conditions for all target species and communities, to ensure that changing site management for Redshank does not adversely affect other important bird populations.

6.3 Nature conservation versus economic gain

As this thesis has shown that grazing with young cattle results in more Redshank nest mortality than grazing with adult cattle, and that Redshank populations may benefit from changed or reduced grazing regimes, this could result in loss of revenue for graziers. In recent years in the UK, government policy has focused on austerity

economics (Clarke & Newman 2012; Hutchison 2013). This places a stronger emphasis on conservation to be justified in terms of potential gains to the economy, in order to help non-ecologists and political decision makers appreciate the value of natural ecosystems and the species they support (Daily *et al.* 2009). In order to reduce Redshank population declines and facilitate conservation of saltmarshes, gaining an understanding of the ecosystem services provided by these habitats (and their associated species) may be necessary. Saltmarshes provide ecosystem services from direct provisioning services such as harvesting of saltmarsh plants for food, animal fodder, thatch, grazing and regulating services such as nutrient filtering, carbon sequestration and coastal defence (Krutilla 1967; Costanza *et al.* 1997; Gedan, Silliman & Bertness 2009). Birds play many roles as members of an ecosystem, including as predators, pollinators, scavengers, seed dispersers, seed predators, and ecosystem engineers (Sekercioglu 2006; Wallace 2007; Wallace 2008). However, very little is known about the ecosystem services that saltmarsh birds provide.

Mooney *et al.* (2005) and Kremen and Ostfeld (2005) distinguish four principal types of ecosystem service, and birds contribute to each of these. These are; (1) Provisioning services, such as production of fibre, clean water, and food. Provisioning services are provided by both domesticated (poultry) and non-domesticated species. Non-domesticated birds have been important components of human diets historically (Moss & Bowers 2007), and many still are today (Peres 2001). In developed countries, many species of birds are hunted for consumption and sport (Bennett & Whitten 2003). Bird feathers provide bedding, insulation, and ornamentation. (2) Regulating services, obtained through ecosystem processes that regulate climate, water, and human disease. Scavengers contribute regulating services, as efficient carcass consumption, which helps regulate human disease. (3) Cultural services, such as spiritual enrichment, cognitive development, reflection, recreation, and aesthetics. Via art, photography, religious custom, and bird watching, birds contribute cultural services. Bird watching, or birding, is a popular outdoor recreational activity around the world. In the United States, for example, 45 million bird watchers spent \$32 billion in retail stores in 2001, generating \$85 billion in overall economic impact, and supporting over 860,000 jobs (LaRouche 2001). (4) Supporting services, which include all other ecosystem processes, such as soil formation, nutrient cycling, provisioning of habitat, and production of biomass and atmospheric oxygen. Birds contribute supporting

services, as their foraging, seed dispersal, and pollination activities help maintain ecosystems throughout the world.

Many natural ecosystems and their associated ecosystem services have declined (Butler, Corvalan & Koren 2005). A rising human population and advances in technology have been the primary drivers behind these declines, advancing the conversion of previously uncultivable areas into functioning agricultural land (Vitousek 1994; Sala *et al.* 2000; Corvalan, Hales & McMichael 2005; Carpenter & Folke 2006). The loss of natural ecosystems and the associated loss of ecosystem services have increased our reliance on the ecosystem services of remaining natural areas (Cumming *et al.* 2005; Carpenter, Bennett & Peterson 2006).

The precise ecosystem services that British saltmarsh birds provide is currently unclear, as research has been limited or absent. However, in the Ribble estuary for example, the high number of birdwatchers that visit the saltmarshes and surrounding nature reserves are likely to contribute to the local economy through tourism and use of local facilities. Furthermore, saltmarshes can play a role in preventing wildfowl damage to surrounding crops. If the saltmarshes have short swards, they are more likely to host the large flocks of geese and prevent them from moving to surrounding arable land (Vickery *et al.* 1995; Mandema *et al.* 2014b). Aside from birds, saltmarshes themselves have important roles to play in ecosystem service provision through: maintaining good water quality, producing food, controlling erosion and flooding, providing habitat for other wildlife, and providing recreational and cultural use (Costanza *et al.* 1997). These services represent the direct and indirect benefits of saltmarshes to society (de Groot, Wilson & Boumans 2002). Further research into ecosystem services provided by saltmarsh birds may be necessary.

It should also be noted that a high proportion of grazing on saltmarshes is already subsidised by taxpayers through grazing agreements on National Nature Reserves (NNRs), (Skelcher 2010) and agri-environment schemes (Parrott & Burningham 2008). Malpas *et al.* (2013) found that 59% of English saltmarshes were implementing some form of conservation management, through either agri-environment schemes or NNRs. In the UK, saltmarshes which are not managed as part of a nature reserve are typically managed privately for agricultural production (Kingham 2013). Traditionally

the European agricultural landscape supported a wide variety of wildlife species, however due to large-scale changes in farm management, much of this diversity was lost or degraded in the latter half of the 20th century (O'Connor & Shrubbs 1990). One of the greatest challenges facing the human race is the provision of sufficient food and resources for an expanding global population, whilst maintaining farmland biodiversity and associated ecosystem services. Agricultural intensification increases the financial gain per unit area, but is negatively related to biodiversity (Donald *et al.* 2001; Donald, Green & Heath 2001; Green *et al.* 2005). Consequently, breeding wader populations have undergone severe declines in Europe during recent decades (Tucker & Heath 1994; Donald, Green & Heath 2001; Vickery *et al.* 2001; Thorup 2004). There is some disagreement amongst conservation biologists about the most effective methods of balancing the need to feed a growing population and the need to conserve biodiversity (Green *et al.* 2005). Some argue in favour of land sharing where less intensive agriculture is practiced alongside wildlife habitat conservation, whereas others argue in favour of land sparing, where farmland is managed as intensively as possible in order to save as much land as possible for conservation (Green *et al.* 2005; Pywell *et al.* 2012; Grau, Kuemmerle & Macchi 2013). The results of this thesis would suggest that land sharing is the most suitable option for Redshank conservation, as grazed saltmarshes show lower rates of nest mortality than those with no active management. Redshank may not sustain large populations in a land sparing scenario as neither intensive grazing or cessation of grazing creates a suitable vegetation structure for Redshank breeding (Brindley *et al.* 1998; Norris *et al.* 1998). Redshank population declines are linked to intensification or cessation of grazing (Norris *et al.* 1998). However the conservation management, through either protected area management or agri-environment schemes failed to prevent the continued declines (Malpas *et al.* 2013).

Agri-environment schemes are mechanisms by which farmers and other individuals and bodies responsible for farm management can be incentivised to manage their environment. In the European Union member nations, these schemes fall under Axis 2 of the Common Agricultural Policy (CAP), "Improving the environment and countryside". They award funding to farmers to manage their land for the benefit of biodiversity, landscape features and the improvement of water and soil quality and can be viewed as the state buying environmental goods and services from farmers who

would not supply them otherwise (Cooper, Hart & Baldock 2009). An objective of most agri-environment schemes is to enhance levels of biodiversity on farmed land (Whittingham 2011). However, Kleijn and Sutherland (2003) suggest that the evidence needed to assess the efficacy of agri-environment schemes for biodiversity conservation is often missing or when present suggests a limited effect. Since the work of Kleijn and Sutherland (2003) was published, the general consensus now indicates that agri-environment schemes provide only limited or moderate biodiversity gains (Kleijn *et al.* 2006; Birrer *et al.* 2007; Davey *et al.* 2010), although some studies have shown that larger agri-environment schemes do produce substantial conservation biodiversity benefits (Bengtsson, Ahnstrom & Weibull 2005; Jonason *et al.* 2011; Winqvist *et al.* 2011). Whittingham (2011) discuss the need for adaptive management of agri-environment schemes in the context of ecosystem service delivery. In this context, adaptive management is a process of decision-making that aims to reduce uncertainty and maximise system success via monitoring. Adaptive management can be used to learn about, and ultimately change a system (Holling 1978). Perkins *et al.* (2011) demonstrate how adaptive management can benefit a species of conservation interest. In this example, monitoring data facilitated identification of new management practices was used to refine agri-environment options to target populations of corn bunting *Emberiza calandra*. On sites where the farmer was given specialist advice on corn bunting conservation, populations increased significantly. Agri-environment schemes are used as a mechanism for biodiversity conservation, however there is a lack of knowledge of the requirements of bird species on saltmarshes. This highlights the need for research into both the habitat conditions and management techniques needed for different saltmarsh birds. This would then enable adaptive management to be used to tailor saltmarsh agri-environment schemes to benefit target species and communities. This could therefore help to reduce further Redshank declines.

The results of this study could be used to inform agri-environment schemes. A range of measures could be implemented to incentivise grazing with reduced livestock densities, grazing with older cattle, grazing only outside of the Redshank breeding season or to implement measures which may move livestock away from the parts of the saltmarshes used for Redshank nesting during the breeding season, such as a rotational grazing system. These are highlighted in Table 6.1.

6.4 Conservation management recommendations

Translating the UK Environment Agency light grazing definition of 0.7 - 1 young cattle ha⁻¹ between April and October (Adnitt *et al.* 2007) to measurements used in this thesis, would translate into a seasonal cattle density of 0.7 - 1 ha⁻¹ and an annual cattle density of around 0.35 - 0.5 ha⁻¹. This study suggests that these guidelines are not optimal for Redshank for several reasons. Chapter two suggested that the UK Environment Agency definition of light grazing is too intensive for saltmarsh breeding Redshank as it can lead to near total nest loss to predation and trampling, chapter three showed that grazing, even at these levels alters the vegetation height and composition, reducing the quality of nest sites for Redshank. Chapter four showed that this definition of light grazing could lead to Redshank nesting areas being exposed to higher trampling risk during the breeding season due to non-homogenous grazing. Chapter five showed that this definition of light grazing was not optimal for Redshank as grazing with young cattle leads to higher rates of nest trampling and predation than grazing with adult cattle. Table 6.1 highlights the key habitat management considerations raised by this thesis, and evaluates the strength of support for each intervention.

Chapter four demonstrates that calculations of livestock density by land managers cannot assume a homogenous distribution of livestock, and that densities are highest in the areas that are used for breeding by Redshank. It is likely that the high rates of Redshank nest loss to trampling shown in chapter two are due in part, to this concentration of grazing during the breeding season. This study suggests that calculations of grazing density which assume homogenous distribution of livestock should not be applied to all saltmarshes. In order to counteract this problem there are several techniques which could help to make livestock distribution more homogenous, and therefore reduce nest trampling rates. Firstly, commencing grazing earlier may move cattle beyond breeding areas during the nesting season. Secondly, the strategic placement of water troughs further away from breeding areas could naturally restrict livestock movements. Thirdly, installing many more drinking troughs spread around the marsh may even out livestock distribution.

It may also be possible to control the homogeneity of grazing through some form of rotational grazing management. The non-linear nature of the response of trampling probability to livestock grazing in chapter four suggests that a rotational grazing regime where saltmarshes are grazed heavily then left fallow in alternate years may improve breeding success by eliminating nest trampling in the fallow year. This would be preferable to leaving saltmarshes ungrazed as breeding Redshank need some grazing to create suitable nesting habitat (Norris *et al.* 1997), and grazing leads to lower nest loss to predation and trampling than other land uses (Sharps 2015). However, this approach may need to accept total nest loss in the grazed year. Alternatively, nest trampling could be eliminated by introducing livestock after the breeding season, but grazing more heavily afterwards. This would eliminate nest trampling, however any rotational grazing system could increase nest predation rates if a suitable vegetation structure is not maintained, due to the long term nature of grazing induced changes to the vegetation structure (Bakker 1985; Andresen *et al.* 1990). Another form of rotational grazing could involve fencing livestock into smaller, more targeted enclosures on saltmarshes and allowing livestock access to the sea wall and/or additional non-saltmarsh grazing facilities. This may help to reduce effects of nest trampling if timed to ensure that cattle graze these areas before spreading onto Redshank nesting areas. A system of fences and gates could be used to more proactively manage livestock distribution, whilst balancing flooding risk to the cattle. However, as cattle would drown due to tidal flooding if restricted to the lower marsh zones, this would require considerably more management effort. Increased fencing could also be impractical on a saltmarsh especially if trying to use temporary electric fences in remote parts of the marsh or areas with regular tidal flooding.

It should be noted that the large error bars for the probability of nest failure and on rotational (unoccupied) saltmarshes in chapter five suggest that expected nest survival rates on saltmarshes in an ungrazed period of a rotational grazing system remain uncertain. As this management technique is not commonly practiced on the saltmarshes studied, there were only two saltmarshes managed in this way. This represented a total of only 14 nests from an overall sample size of 567 nests. It is therefore likely that this inconclusive result is due to the very low sample size of nests from rotational (unoccupied) saltmarshes. It might be expected that leaving a saltmarsh unoccupied by livestock during the Redshank breeding season would result

in higher rates of nest survival due to reduced trampling rates, and improved vegetation structure due to previous grazing. Therefore further research into the potential benefits of this management technique may be necessary. In lowland wet grassland habitats, commencing grazing after the end of July has been shown to increase productivity in Redshank and other waders through reducing nest trampling mortality (Green 1986). However, further investigations would be needed to find out the ideal livestock density for maintaining a patchy vegetation landscape needed by Redshank.

Results of chapter five also suggested that Redshank nest loss to predation is lower closer to the landward extent of the saltmarshes, regardless of land use. This implies that Redshank nest predation on saltmarshes is primarily due to avian predators rather than mammals. This is because ground predators may predate nests closer to the sea wall, while avian predators may cause nest mortality anywhere on the saltmarsh (Montevecchi 1977). In some habitats, lethal control of mammalian predators is used as a management tool to reduce shorebird nest predation rates (Bolton *et al.* 2007). However, based on these results, it would seem that increasing any current rates of mammal control at the saltmarshes studied would be unlikely to benefit Redshank conservation. Chapters two and three suggested that high rates of nest predation arise indirectly through land use, as grazing reduces the nest vegetation height and makes it easier for predators to find nests. Chapter five showed that nest predation rates are lower when saltmarshes are grazed by adult cattle than when grazed by young cattle. It may therefore be possible to reduce nest predation rates by reducing livestock densities, and grazing with adult cattle.

As the results of chapter five showed that mowing or no active management may lead to lower nest survival rates, it is likely that improving grazing management would be more beneficial to Redshank than these alternative management techniques. Chapter five only considered the difference between adult and young cattle. Elements of different grazing regimes employed on different saltmarshes may have a large effect on nest survival. Sheep are more likely to produce shorter vegetation swards than cattle (Green 1986; Beintema and Muskens 1987) and horses are more likely to trample nests (Mandema *et al.* 2013). Further investigations may be needed to investigate the effects of using other livestock types.

6.5 Future research

Before implementing widespread changes to grazing management on British saltmarshes, further research is necessary. Reducing livestock density or changing the type of grazing animals may impact on species other than Redshank. Therefore, approaches are required that would balance the impacts of changes in conservation management practices on species which favour a different vegetation structure. This includes both shorter swards (e.g. overwintering geese, foraging passerines and some wader species) and longer swards (e.g. nesting passerines). The methodology may need to vary for species with known saltmarsh habitat requirements such as overwintering geese, than those which are speculative, such as Skylark and Meadow Pipit. For breeding passerines, research may need to focus on understanding their nesting habitat requirements, whereas for wildfowl, work should focus on understanding the impacts of reduced livestock density on populations using saltmarshes.

As this thesis demonstrated that non-homogenous livestock grazing may be detrimental to Redshank populations, research is necessary to understand the effects of implementing either an inter-annual rotational grazing system or changing the timing of grazing. In order to be successful for Redshank these approaches would need to establish the ideal balance between reducing the probability of nest trampling and maintaining a vegetation structure suitable for Redshank breeding and/or suitable for meeting the life history requirements of other birds on the saltmarshes. Methods would likely need to involve experimentally implementing these systems as part of a long term study across a sample of different saltmarshes. This investigation would need to consider the effects of changes in vegetation structure and bird populations over a number of years due to the long term nature of grazing-induced changes to the vegetation landscape.

In order to move livestock away from Redshank breeding areas on saltmarshes, research should also consider the effects of the location of drinking water troughs and the advantages of moving the drinking water locations or increasing the number of

drinking troughs. This could be investigated through the experimental manipulation of drinking water trough positions and the numbers of troughs placed on saltmarshes combined with recording the effects on livestock distribution with GPS tracking.

As this thesis considered the effects of cattle grazing on nesting Redshank, further research may be needed to understand if grazing with other domestic livestock such as sheep and horses would result in improved Redshank nesting success. This could be achieved firstly by recording nest survival rates in saltmarshes exposed to different livestock species and secondly by recording the distribution of other livestock species on saltmarshes with GPS tracking.

6.6 Conclusions

In conclusion, this work suggests that current livestock grazing on British saltmarshes is not optimal for breeding Redshank. This is because recommended light grazing levels lead to a high chance of nest loss to predation and trampling and because the areas of the saltmarsh where Redshank breed are much more intensively grazed than intended by land managers, due to the non-homogeneous distribution of livestock. This results in high nest mortality due to nest trampling, therefore changes in grazing management on saltmarshes may be necessary. Results also indicate that grazing could be optimised by reducing livestock density and by grazing with adult cattle, instead of young cattle. This thesis recommends that future research for Redshank focuses on designing an optimal system of rotational grazing, which directly reduces livestock trampling during the breeding season and maintains a suitable vegetation structure for Redshank to build cryptic nests, which are hidden from predators.

6.7 Tables

Table 6.1: Conservation management considerations raised by this thesis, and future research needed before implementing.

The column “Direct support” highlights if this thesis provides direct evidence in support of this management intervention or if the evidence is indirect and is therefore more speculative. The guideline livestock densities listed in interventions 1 and 2 are based on the nest survival models presented in chapter two of this thesis and following the estimates of Macdonald and Bolton (2008), that waders typically need nest survival rates of > 49% for population stability.

	Conservation management intervention	Direct support?	Evidence for intervention.	Considerations	Future research needs.
1	Reducing livestock density on British saltmarshes to levels beneath 0.4 cattle per hectare during the Redshank breeding season.	Direct	Near 100% probability of Redshank nest loss to livestock trampling (chapter 2).	Work based on young cattle.	Balancing the need to conserve Redshank populations with the habitat requirements of other bird species. The effects of grazing with other livestock species.
2	Reducing livestock density to beneath 0.2 cattle ha ⁻¹ y ⁻¹ the year before the breeding season.	Direct	Near 100% probability of Redshank nest loss to predation (chapter 2).	Work based on young cattle. Grazing induced changes to vegetation are medium-long term effects so changes may need several years to take effect.	As above.
3	Switching from grazing with young cattle to grazing with adult cattle.	Direct	Less probability of nest loss to trampling and predation by grazing with adult cattle than grazing with young cattle (chapter 3).		As above
4	Very light grazing, rather than mowing.	Direct	Less probability of nest mortality on grazed saltmarshes than	Success would be dependent on grazing regime.	The benefits of mowing on species other than Redshank.

			mown saltmarshes (chapter 3).		
5	Rotational grazing or commencing grazing later.	Indirect	Near 100% probability of Redshank nest loss to livestock trampling (chapter 2). Non-homogenous livestock distribution causing high rates of nest trampling during Redshank breeding season (chapter 4).	No evidence in support of this technique found in chapter 5, however the low sample size may not be representative.	Finding the right balance to maintain suitable vegetation structure for Redshank and other bird species.
6	Commence grazing earlier	Indirect	Non-homogenous livestock distribution causing high rates of nest trampling during Redshank breeding season (chapter 4).	May be issues with vegetation growth before April/May.	Continued GPS tracking to determine if this technique would result in livestock staying away from bird nesting areas during the breeding season.
7	Strategic placement of water troughs, or installing many more water troughs spread around the saltmarsh.	Indirect	Livestock distribution closer to the sea wall during Redshank breeding season may be due to the location of fresh drinking water (chapter 4).	Consider drowning risk for cattle in areas exposed to high rates of tidal flooding.	Continued GPS tracking to determine if this technique results in livestock staying away from bird nesting areas during the breeding season.

6.8 References

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