## *Determinants of nesting and nest success in saltmarsh breeding Common Redshank, Tringa totanus, in North West England*

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# Determinants of nesting and nest success in saltmarsh breeding Common Redshank, *Tringa totanus,* in North West England

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# Declaration

The material contained within this thesis has not previously been submitted for a degree at Durham University or any other university. The research reported within this thesis has been conducted by the author unless indicated otherwise.

Dominic R. Harmer

March 2020

# Acknowledgements

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# Abstract

Breeding populations of Common Redshank, *Tringa totanus*, Redshank hereafter, on British saltmarshes halved between 1985 and 2011, with the North West of England experiencing some of the largest losses. Understanding the drivers of these declines and exploring conservation management options to redress losses is the primary focus of this thesis.

I first evaluated the reliability of a standardised survey method (SSM) for estimating Redshank nesting density on saltmarshes. This involved multiple walked censuses on four saltmarshes in North West England. Estimates of peak nesting density derived from the SSM were compared to detailed nest monitoring information gathered at the same sites. The SSM was found to overestimate nesting density by 42% across the study sites. Reasons for this discrepancy were considered to be, (i) the presence of non-breeding birds, (ii) differing causes of nest failure across different habitats and areas, and (iii) geographical variation in, and temporal changes to, nesting phenology, the latter likely related to ongoing climate change.

I then examined the temporal and spatial distribution of wildfowl and livestock, and their effect on saltmarsh vegetation height in relation to Redshank nesting attempts. To do this, I used observational and experimental exclusion approaches on Banks Marsh, part of the Ribble Estuary National Nature Reserve, over a 30-month period. Cattle usage was variable spatially across the site with higher usage of the inner (landward) marsh compared to the outer marsh, but with a consistency of areas with greatest and least use between years. The number of nests trampled by cattle was relatively low (15%), occurring in areas of shorter vegetation and higher cattle use. Winter wildfowl herbivory played a crucial role in reducing saltmarsh vegetation height, with wildfowl grazed vegetation typically one third of the height of ungrazed vegetation (wildfowl excluded) during the peak Redshank nesting period. Redshank selected nest sites in taller vegetation and successful nests were in significantly taller vegetation than nests that failed. Using the data collected from Banks Marsh, I developed a logistic regression model based on key biotic (cattle, duck, and goose herbivory) and abiotic variables, including elevation above sea level, to predict where Redshank nest on saltmarshes. Winter grazing by Eurasian Wigeon, *Mareca penelope*, had a strong negative impact on Redshank nesting, whereas light, late summer grazing by cattle had a positive impact. A modelled reduction in Wigeon use whilst maintaining light cattle grazing optimised the availability of suitable nesting habitat. Under such a regime, Redshank numbers were projected to remain relatively unchanged in future under a scenario of a sea level rise of 0.25m. Under other scenarios of sea-level rise and management, Redshank populations were much reduced. The model developed provided a framework for simulating potential 'trade-offs' between wildfowl and breeding wader populations, where a conservation conflict could occur, and for long-term conservation management planning for future climate change impacts.

Finally, I investigated the effects of livestock and wildfowl herbivory, flooding, and predation on Redshank nest survival. I analysed self-collected and long-term nest-record data from Banks Marsh between 1969 and 2018, using Program MARK. Redshank nest survival was most strongly negatively affected by dramatic increases in winter duck herbivory. Increasing cattle grazing intensity during the Redshank breeding season also negatively impacted nest survival but to a lesser extent. Identifying the key environmental variables that influence Redshank nest survival should assist conservation managers to reflect on strategies to maintain this vulnerable species whilst also conserving other target species.

Overall, this project highlights, (i) the benefits of retesting established survey methods and developing improved population estimates, (ii) the need to address the impact of both wildfowl and livestock grazing in future research and conservation management for Redshank, and (iii) the value of long-term data that permit new insights into population dynamics of species.

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# 1 General introduction

## 1.1 Motivation

Biodiversity loss is occurring globally at an alarming rate with an average 68% decrease in population sizes of mammals, birds, amphibians, reptiles and fish between 1970 and 2016 (Almond *et al.*, 2020). The reasons identified for the biodiversity loss include, (i) human-driven land and coastal use change, (ii) over-exploitation, (iii) climate change, (iv) pollution and (v) invasive alien species (Almond *et al.*, 2020; Sala *et al.*, 2000).

Human existence relies fundamentally on the many goods and services that are contributed to and regulated by biodiversity such as food, clean water, climate mitigation and cultural connections (IPBES, 2019). Declines in the abundance and diversity of wild species compromise the functioning of ecosystems making them more vulnerable to further change and less able to supply humans with needed services (Hooper *et al.*, 2005). The global threats to both biodiversity and human society are so profound that scientists worldwide have conveyed repeated warnings to humanity, calling for an urgent improvement in conservation measures (Ripple *et al.*, 2017).

Current species range contractions and population declines combined with predicted future changes in climatic distributions, make managing the remaining suitable habitats even more important. Wildlife conservation is increasingly important in stemming declines in plant and animal populations globally. Establishing protected areas, restoring habitats, and the management of target populations have been the most consistently successful mechanisms for tackling biodiversity loss (Bolam *et al.*, 2021). An estimated 15% of terrestrial and 7% of marine habitats currently have some level of protection but this falls short of The Aichi Biodiversity Target goals (Almond *et al.*, 2020). Wilson (2016) advocates that half the earth's land surface must be set aside for nature to stave off a mass extinction. A primary reason why there has not been more success in biodiversity conservation is that resourcing is far too low (Waldron *et al.*, 2013). Conserving biodiversity and dealing with global threats requires intervention at different spatial and temporal scales. Site protection and applied conservation management are the foundations for conserving most threatened species, and when complemented with landscape scale interventions can prevent mass extinctions (Boyd *et al.*, 2008).

## 1.2 The importance of conservation science

Waders or 'shorebirds' are members of the order *Charadriiformes* and are commonly associated with wetland or coastal environments, where they wade in mud and sand for food. Worldwide, there are around 210 species of waders, (Hayman *et al.*, 1991), many of which are declining in abundance (Zockler *et al.*, 2003), though available census methods can lead to high uncertainty in population estimates across this sometimes cryptic breeding group. Waders were once among the most common of all breeding birds in European lowland habitats, but substantial recent population declines, and range contractions mean that many now require conservation management to prevent local extinctions (Birdlife International, 2021; EBCC/BirdLife/RSPB/CSO, 2019).

Taking effective actions through conservation management to halt the decline of breeding waders requires detailed knowledge of their ecology and the effectiveness of conservation practices. The key steps central to effective conservation management are, (i) good population monitoring, to determine the size of the current population and for measuring success at restoring populations, (ii) understanding species habitat requirements, (iii) establishing whether habitat management results in favourable populations, or (iv) if other changes are needed to promote population stability and growth (Mason, 2019).

In the absence of scientific evidence, a common approach to conservation management has been to mimic traditional land management practices, such as the livestock grazing regimes, that facilitated the communities before the declines (Pullin and Knight, 2003). However, often such conservation management actions are not based upon well-designed experiments and a range of scientific studies, instead, they are a best-guess approach (Sutherland *et al.*, 2004). Failure to evaluate the effectiveness of conservation management can lead to the widespread implementation of ineffective treatments (Sutherland *et al.*, 2004). One such example is the winter flooding of grasslands, which was formerly widely considered to be beneficial for wading birds, and hence was encouraged by governmental grants. However, research showed that whilst flooding previously unflooded grasslands did provide soft mud and bare soil suitable for foraging waders, it also killed the invertebrates upon which they fed (Ausden *et al.*, 2001). By adopting evidence-based approaches, as used elsewhere in the fields of medicine and public health, conservation management is more likely to produce effective policy and conservation outcomes (Pullin *et al.*, 2004).

Below I describe key features of coastal saltmarshes relevant to their conservation value and for ecological research, with emphasis on different forms of saltmarsh grazing which are important for their conservation management. I then provide background information on Redshank, the primary focal species of this research. I introduce approaches to Redshank monitoring and review population trends identifying reasons for the population decline. Finally, I briefly outline the following chapters of this thesis which evaluate methods used for censusing breeding Redshank on saltmarshes, and the drivers of habitat suitability and nest survival for Redshank breeding on saltmarshes in North West England.

## 1.3 Saltmarshes

### 1.3.1 Description

Coastal saltmarshes are the vegetated areas of intertidal mudflats lying approximately between mean low-water spring tides and mean high-water spring tides (Adam, 1990; Boorman, 2003). They occur worldwide, but most extensively outside of the tropics and they are a common habitat of estuaries. A conservative estimate of global saltmarsh extent is 5.5 million ha with approximately 8% of this occurring in North West Europe (Mcowen *et al.*, 2017). The UK supports 45 thousand ha of saltmarsh habitat with 12 thousand ha in North West England (Phelan *et al.*, 2011).

Saltmarsh plants are the structural and ecological foundation of saltmarsh ecosystems and consist of a halophytic, salt tolerant species adapted to regular immersion by the tide. Zones from the sea to the landward edge are characterised by distinct communities following successional stages (Figure 1.1), (Adam, 1981; Burd, 1989; Rodwell, 2000).

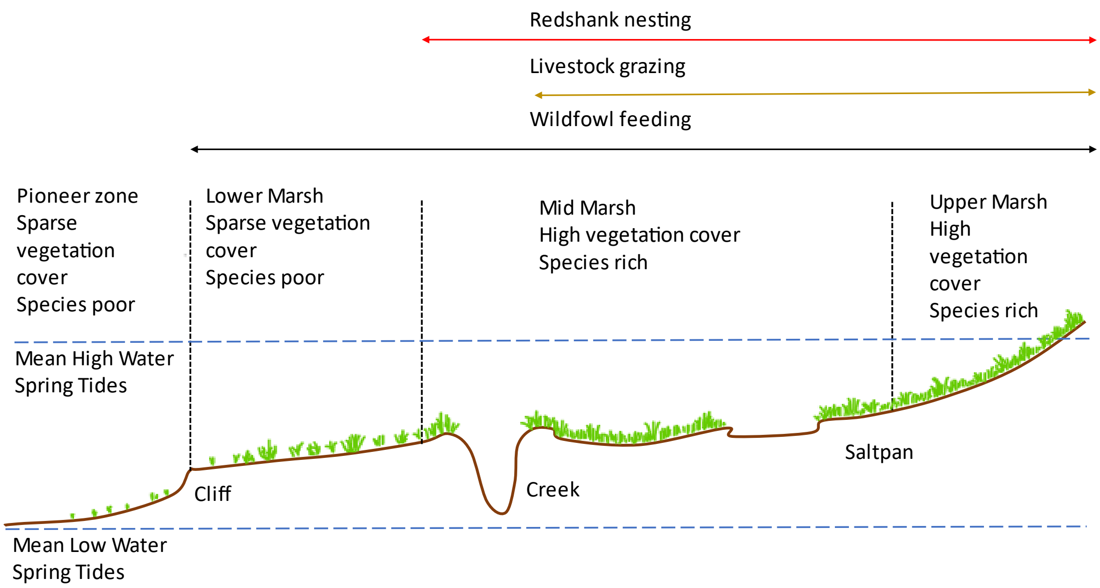



Figure 1.1 Profile of a typical saltmarsh from the seaward (left) to the landward edge (right). The main vegetation zones and typical features are illustrated along with an indication of where livestock and wildfowl graze and Common Redshank nest.

Pioneer zone plants, such as *Spartina* and *Salicornia* species can colonise mudflats, trapping sediment and elevating the saltmarsh surface. By providing shade, evaporation is slowed and soil salinity is reduced allowing competitively superior species to colonise (Adam, 1990; Sánchez *et al.*, 2001). Mid and upper saltmarsh zones typically comprise a continuous vegetation cover and experience progressively less frequent tidal inundation, with upper marshes usually only flooding during spring high tides (Adam, 1990; Best *et al.*, 2007). The UK National Vegetation Classification system (NVC) recognises twelve lower, eight middle and five upper saltmarsh vegetation communities (Rodwell, 1991).

Saltmarshes are an important resource for agricultural production and for wildfowl and wading birds, but agricultural management may not always be suitable for their conservation management (Mason *et al.*, 2019). *Puccinella maritima* and *Festuca rubra* grassland communities can develop a short turf across saltmarsh zones when grazed by livestock and or migratory wildfowl (Bos *et al.*, 2005; Cadwalladr *et al.*, 1972). Livestock and wildfowl grazing has the potential to alter habitats for breeding birds by limiting or creating the availability of suitable vegetation patches for nesting and foraging (Hale, 1980; Madsen *et al.*, 2019; Mandema *et al.*, 2014). Common Redshank, Redshank hereafter, typically nests in the middle and upper saltmarsh zones, where livestock may also spend most of their time (Sharps *et al.*, 2017). Creek networks, dissecting stands of vegetation, and saltpans are common features of many saltmarshes and often occupy a large part of the total marsh area (Adam, 1990; Boorman, 2003). Different saltmarsh types arise through the interaction of sediment type with climatic, biotic, and historic factors. Saltmarshes dominated by fine grassland swards are characteristic in North West England and occur around much of the UK, though coarse grassland, dwarf shrub and herb saltmarshes also commonly occur e.g. in North Norfolk (Chatters, 2004). Livestock grazing is regarded as one of the most important causes of the different vegetation characteristics of marshes on the West coast and in the South East of the United Kingdom by some (e.g. Jefferies, 1972; Pigott, 1969). An alternative opinion is that the importance of *Puccinellia maritima* on UK West coast marshes is a result of their sandy substrate, making them particularly suitable for grazing, hence suggesting that the vegetation dictates the grazing rather than vice versa (Chapman, 1941). The northern limits of some saltmarsh species also influence plant community variation between the north and south of Britain (Adam, 1978).

### 1.3.2 Conservation value and past and future threats

Saltmarshes are highly productive ecosystems and can be rich in biodiversity for some taxa (Boorman, 2003; Brindley *et al.*, 1998; Rickert *et al.*, 2012). For example, saltmarshes are important breeding sites for waders, gulls, and terns (Burton *et al.*, 2010; Cadbury *et al.*, 1987; Greenhalgh, 1971) and in winter are used as feeding grounds by large flocks of wildfowl (Cadwalladr *et al.*, 1972; Frost *et al.*, 2019; Mandema *et al.*, 2014; Owen and Williams, 1976). Saltmarsh habitats are protected under national and international legislation, reflecting their importance for birds and other taxa, and for the critical ecosystem services they provide, including tidal defence and carbon storage (Davidson *et al.*, 2017; Doody, 2008). Despite their importance, saltmarsh ecosystems are often overlooked, being perceived as difficult and dangerous places to visit. Their protected status has not prevented declines in their biodiversity, making their conservation management an area of active ecological research (Garbutt *et al.*, 2017).

Historically, saltmarsh losses resulted primarily from land reclamation and drainage associated with agricultural conversion and development. For example, on the Wash Estuary, UK, the area of saltmarsh has declined from 47,000 ha to 4,000 ha since the 17th Century due to reclamation (Chatters, 2017). Van Der Wal *et al.* (2002) detail the extensive land reclamation on the Ribble Estuary since 1810, which culminated in the establishment of Banks Marsh National Nature Reserve (NNR) following the embankment and conversion to arable land of the neighbouring Hesketh Marsh saltmarsh in 1979. Loss of saltmarsh habitat to the process of "coastal squeeze" results from the combined effect of reclamation of former saltmarsh habitat and sea-level rise and is most evident on the saltmarshes of Essex and North Kent, UK (Adnitt *et al.*, 2007). Such coastal squeeze is often cited as a main cause for the loss of intertidal habitats (Doody, 2013). Further widespread loss of saltmarsh habitat is likely to accelerate as a result of climate change and sea level rise (Alexander, 2020; Hughes, 2004). Managed realignment schemes are an increasingly popular approach for recreating intertidal habitats whereby planned breaches of coastal defences allow tidal waters to flow through creating new intertidal areas (French, 2006; Shirres, 2015). The landward realignment of coastal defences can rapidly produce intertidal mudflats which are colonised by saltmarsh plants, but nature conservation targets, such as increases in Redshank populations, may be more difficult and costly to achieve (Garbutt *et al.*, 2006). It is unclear whether managed realignment and conservation management can provide a sustainable long-term coastal management approach whilst stemming saltmarsh biodiversity loss (Esteves, 2013; Greenwood *et al.*, 2016).

Ecological modelling can provide predictions of how future climate change scenarios may impact intertidal habitats in coastal areas (McFadden *et al.*, 2007; Simas *et al.*, 2001). Such modelling can also contribute both to the assessment of future change in species breeding distributions and inform decision-making by forecasting likely ecosystem responses to management actions (Huntley *et al.*, 2007; Mason *et al.*, 2018).

### 1.3.3 Saltmarsh research

Saltmarshes provide valuable study systems for applied ecological research and conservation management practices, owing to their steep environmental gradients, relatively low species richness and protected designations that require them to be monitored and assessed (Garbutt *et al.*, 2017). Key research themes that have emerged chronologically over the last century include, (i) saltmarsh vegetation, including biotic and abiotic interactions, (ii) species-specific research on birds and other taxa that breed or use saltmarshes, (iii) the effect of livestock grazing on plants and animals, in particular on breeding birds, small mammals and invertebrates, (iv) saltmarsh restoration and managed realignment schemes, and (v) the effect of climate change on carbon sequestration and greenhouse-gas emissions from salt marshes (Garbutt *et al.*, 2017).

### 1.3.4 The importance of grazing

Natural and semi-natural grasslands occur extensively around the globe, but their successful management for agricultural and/or biodiversity depends critically upon the level of grazing, its timing and the animal species involved (Watkinson and Ormerod, 2001). Grazing can positively influence the spatial heterogeneity of grassland vegetation, affecting ecosystem processes and biodiversity (Adler *et al.*, 2001). Too much grazing can result in land degradation and biodiversity loss, while too little grazing may lead to succession changes and the loss of the grassland habitat (Watkinson and Ormerod, 2001).

Livestock grazing is unquestionably a major factor in determining the suitability of sward structure and habitat selection for breeding waders (Durant *et al.*, 2008), and intensive grazing regimes have been highlighted as failing to provide suitable habitat for breeding Redshank on saltmarshes (Mason *et al.*, 2019). Livestock grazing is important both for agricultural purposes and for conservation management. As part of the latter, livestock grazing is used across a wide range of grassland habitats, where natural grazers have been lost (Ausden, 2007). Livestock grazing is still the most common form of saltmarsh management in the UK (Adnitt *et al.*, 2007; Boorman, 2003), though throughout the 20$^{th}$ century the practice declined locally in many places to the point of abandonment, particularly in Southern England. However, grazing remains common on the saltmarshes of North West England (Chatters, 2004). The current consensus is that light to moderate levels of cattle grazing between April–October is beneficial for maintaining, enhancing, or restoring overall saltmarsh biodiversity (e.g., Davidson *et al.*, 2017; Doody, 2008; Ford *et al.*, 2013; van Klink *et al.*, 2016; Lagendijk *et al.*, 2017) in particular for breeding Redshank, (e.g., Mandema *et al.*, 2015; Mason, 2019; Norris *et al.*, 1998). However, Chatters (2004) suggests no such consensus exists as to the value of livestock grazing on saltmarshes amongst conservation managers.

Malpas *et al.* (2013) present evidence that continued declines in breeding Redshank populations on British saltmarshes are associated with unsuitable livestock grazing, suggesting that the most severe Redshank declines, found in North West England, are likely to be driven by a lack of suitable nesting habitat resulting from persistent, heavy livestock grazing. Inappropriate livestock grazing may not deliver the breeding habitat conditions required for Redshank, which are a mixed structure of tussocky longer vegetation for nesting interspersed with shorter vegetation to assist predator detection and to provide suitable areas for chicks to feed (Green, 1986; Hale, 1988; Smart *et al.*, 2006). Conservation grazing needs to balance the positive effects on vegetation structure against the potential negative effects of nest trampling which may occur even from light grazing (Mandema *et al.*, 2013; Sharps, 2015; Sharps *et al.*, 2017).

### 1.3.5 Measurement of livestock grazing intensity

The measurement of livestock grazing in relation to wader conservation has been approached in different ways including simple presence or absence (Hart *et al.*, 2002)**,** but more commonly using estimates of stocking densities (Triplet *et al.*, 1997), or qualitative assessments of sward appearance during the breeding bird season (Malpas *et al.*, 2011; Norris *et al.*, 1998; Ottvall *et al.*, 2005). Interestingly, the same saltmarsh can be simultaneously recorded as lightly grazed using a livestock/ha assessment (e.g., Sharps *et al.*, 2015) whilst being categorised as heavily grazed using a qualitative assessment (Malpas *et al.*, 2011; Norris *et al.*, 1998). The UK Environment Agency defines light grazing as 0.7–1 cows/ha between April and October, moderate grazing as 1-1.5 cows/ha and heavy grazing as levels up to 2 cows /ha (Adnitt *et al.*, 2007). If the distribution of livestock on saltmarshes is not spatially or temporally homogenous, calculating a more precise measure of livestock grazing to account for this variation is required and can effectively be achieved using methods such as dropping counts or GPS collars (Rankin, 1979; Sharps *et al.*, 2017).

### 1.3.6 Other grazers

Livestock grazing is the possible driver of saltmarsh species declines that has received the most attention to date, as it is easy to measure and influence through direct conservation management. It is also thought that manipulating grazing could mitigate the effects of larger-scale drivers of change such as climate change and sea level rise (Clausen *et al.*, 2013; Mason, 2019). In addition to livestock, wild herbivores can also graze saltmarshes, affecting their vegetation and impacting the landscape (Chatters, 2004). For example, in North America, White-tailed deer, *Odocoileus virginianus* and Wild horse, *Equus ferus,* play an important role in shaping saltmarsh ecosystems (Gaskins *et al.*, 2020). Similarly, in the UK, non-native Sika deer, *Cervus nippon*, have been shown to influence saltmarsh vegetation (Hannaford *et al.*, 2006).

Herbivorous wildfowl (swans, geese and some ducks), are the most numerous avian herbivores on saltmarshes, and many populations have increased rapidly and/or markedly altered their range during the last 50 years, as a result of recovery from hunting pressures, the establishment of protected areas, and the impacts of recent climate change (Fox *et al.*, 2017; Hirons and Thomas, 1993). Saltmarshes are used as feeding grounds by large flocks of wildfowl throughout the winter and into spring, creating progressively shorter sward heights for ground nesting birds (Cadwalladr *et al.*, 1972; Mandema *et al.*, 2014; Owen and Williams, 1976; Watts Mayhew, 1985). Two important species of migratory herbivorous wildfowl, which feed on UK saltmarshes during winter and spring, are Eurasian Wigeon, *Mareca Penelope,* and Pink-footed Goose, *Anser brachyrhynchus*. These two species have shown increases in abundance of 147% and 548% respectively since the mid-1970s (DEFRA, 2020).

Wildfowl can often have a positive effect on biodiversity by, for example, recycling nutrients and dispersing seeds, but some populations have seen such spectacular increases in recent years, with consequent adverse impacts, that research is required to establish both the positive and negative impacts (Green and Elmberg, 2014). Chatters (2004) argued that the grazing pressure of herbivorous wildfowl was minor, being insufficient to modify saltmarshes ungrazed by livestock into fine-grass saltmarshes. However, more recently, the feeding of expanding, and now abundant, populations of Lesser Snow Goose, *Chen caerulescens,* in Arctic and sub-Arctic saltmarsh habitats has triggered a devastating trophic cascade, resulting in vegetation loss, changes to soil properties, adverse effects on invertebrate, and changes to passerine and breeding wader populations (Cargill and Jefferies, 1984; Jefferies *et al.*, 2006; Jefferies and Rockwell, 2002; Rockwell *et al.*, 2009). Indeed, it has been suggested that conservation management actions to control the abundance of herbivorous wildfowl may be needed to prevent further declines in other species of conservation concern (Koons *et al.*, 2014).

While wildfowl herbivory may be an obvious and significant feature of the ecology of saltmarshes, its potential importance in relation to the decline of breeding waders appears largely overlooked, occurring primarily before the wader breeding season, and considered beyond the scope of many studies, though with a few notable exceptions. An assessment of the impact of wildfowl grazing on saltmarsh vegetation communities was included in the first survey of Redshank breeding on British saltmarshes along with livestock grazing, to produce a measurement of total grazing (Allport *et al.*, 1986). However, the follow up surveys (Brindley *et al.*, 1998; Malpas *et al.*, 2011) dropped the wildfowl grazing measure due to difficulties in quantitatively assessing the impact of winter wildfowl herbivory during the Redshank breeding period (G. Allport 2020, pers. comm*.,* 9 March). Vickery *et al.* (1997) found that winter grazing geese can modify coastal habitats, negatively impacting breeding waders, but conclude that potential conflicts between intensive wildfowl grazing and breeding wader requirements may be relatively easily resolved by managing areas close to wildfowl roosts for wildfowl, and other areas specifically for breeding waders. By contrast, on farmed freshwater wet grasslands, (Madsen *et al.*, 2019) found no negative effect of intensive Barnacle Geese, *Branta leucopsis*, and Brent Geese, *Branta bernicla,* herbivory on Redshank nest occupancy.

### 1.3.7 Conservation conflicts

Balancing the need to increase suitable habitat for nesting Redshank populations with the requirements of other bird species of conservation concern, such as migratory wildfowl, can be viewed as a conservation vs conservation conflict. Numerous examples exist where species of conservation concern occur in the same habitat and may biologically impact or potentially conflict with each other. Protected predators may consume protected prey e.g., Pine Marten, *Martes*, and Capercaillie, *Tetrao urogallus,* (Young *et al.*, 2010). A potential conflict between the requirements of feeding wildfowl and breeding Redshank was identified by Lambert (2000), specifically in relation to Banks Marsh and the

Ribble Estuary, which now routinely support the highest density of feeding Wigeon in Britain but have seen a spatial shift and decline in its breeding Redshank population (Ashcroft, 1978; Booth and Haywood, 2017; Frost *et al.*, 2021). Such direct conflicts between species of conservation concern require detailed investigation and present challenges for conservation management, with potential trade-offs.

While the potential conservation conflict between herbivorous wildfowl and breeding waders is little studied, conflicts between increasing wildfowl populations and agricultural production are well documented (Fox *et al.*, 2017; Mason *et al.*, 2018; Vickery and Gill, 1999). It has been suggested that saltmarshes can play a role in preventing wildfowl damage to surrounding crops if they are managed through livestock grazing to have short swards more likely to attract large flocks of ducks and geese (Mandema *et al.*, 2014). However, from a nature conservation viewpoint, concentrating herbivorous wildfowl into small areas results in their reduced value for other forms of wildlife (Vickery and Gill, 1999). Also, if wildfowl become very dependent on just a few refuge sites they become more vulnerable to adverse conditions, such as disease outbreaks or pollution incidents, as seen recently with bird flu outbreaks in migratory geese along the Solway Firth. As a result, dispersing populations may be beneficial for such species (Meirei and Kuijken, 1991).

## 1.4 Redshank

### 1.4.1 Description

The Redshank is a medium-sized (27-29cm) ground-nesting wading bird of the Sandpiper family, *Scolopacidae.* Adults have grey-brown upperparts with whitish underparts, heavily streaked and spotted dark brown on breast flanks and belly and most notably long orange-red legs, and a medium length bill (Figure 1.2a). Redshanks are socially monogamous and non-territorial and can nest semi-colonially during the breeding season. They breed in a wide range of habitats throughout Europe, including coastal saltmarshes, lowland wet grasslands and upland swampy moors (del Hoyo *et al.*, 1996). The nest site is a shallow depression on the ground, typically located at the base of a grass tussock, which is formed into a covering ‘roof’ to conceal 3-5 eggs (Figure 1.2b). The species usually nest solitarily inland at densities of less than 10 pairs km$^2$ but can nest in loose colonial groups (of up to 100-300 pairs/km$^2$) on the coast (Hale, 1980; del Hoyo *et al.*, 1996). Incubation is undertaken by both adults and lasts about 24 days, with the precocial young feeding themselves soon after hatching before fledging 23-25 days later (del Hoyo *et al.*, 1996), (Figure 1.2a).

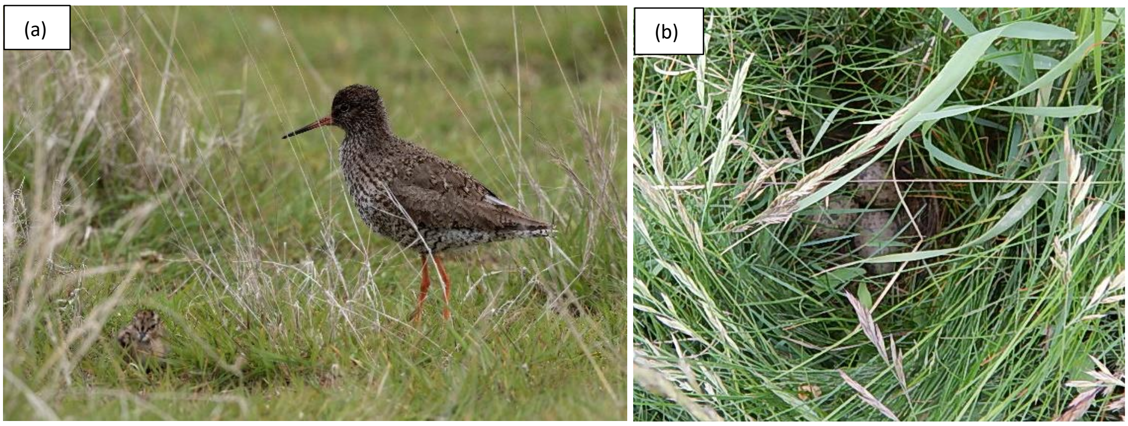



Figure 1.2 (a) Adult Redshank and precocial Redshank chick (lower left), photograph used with permission from David Hosking (b) Redshank nest.

### 1.4.2 Population and trends

Redshanks have a large but fragmented global range, with six recognised races breeding across temperate and steppe areas of Eurasia, from Iceland in the west to the Far East of Russia (Cramp and Simmons, 1983). The global population is broadly estimated at between 1.3-3.1 million, with an uncertain overall population trend, with some populations decreasing, while others are stable, increasing, or unknown (Wetlands International, 2022). Breeding occurs throughout Europe but most abundantly in eastern Europe, Britain and Ireland, Scandinavia, Germany and the Netherlands (Smit and Piersma, 1989). There are estimated to be 100-172 thousand breeding pairs in North West Europe, and British saltmarshes support around 8% of the European population (Batten *et al.*, 1990; Piersma, 1986). The nominate race *Tringa totanus totanus* breeds in Britain and are the focus of this study (Hagemeijer and Blair, 1997).

The UK breeding population is estimated at 25,000 pairs, while 130,000 birds winter around the coast, with many birds from mainland Europe joining the UK breeding population in winter (RSPB, 2021a). In England, of the estimated 14,000 breeding pairs, 66% nest on saltmarshes, 21% on lowland wet grassland and 12% in upland habitats (Mason, 2019). A recent national survey of British saltmarshes found a 53% reduction in nesting Redshank between 1985 and 2011 (Malpas *et al.*, 2013) which equates to a 24% loss of the total British breeding population. Consequently, in the UK, Redshank is on the Amber list of Birds of Conservation Concern (with Red being the highest conservation priority and Amber the next most critical group) due to declines in the breeding population (Stanbury *et al.*, 2021).

### 1.4.3 Monitoring

Species monitoring, the regular, systematic collection of data to detect long-term changes in the populations of wild species, is a critical element of biodiversity conservation practice and policy (Moussy *et al.*, 2022). However, monitoring is often given low priority because it can be difficult and expensive to implement (Danielsen *et al.*, 2009). Since it is not practical to monitor all species, monitoring some well-known taxa as indicator species is frequently adopted to indicate broader environmental changes, assess the efficacy of management, and provide warning signals of impending ecological shifts (Siddig *et al.*, 2016). Wild birds can act as useful indicators of ecosystem health, reacting quickly to changing environmental conditions, being sensitive to anthropogenic impacts and occupying high trophic levels (Browder *et al.*, 2002; Gregory and Strien, 2010). Wading birds, and Redshank, in particular, are useful indicators of wetland ecosystem health, occurring in abundance across a range of coastal and inland habitats and proving a good model species to study issues relating to conservation management (Furness *et al.*, 1993).

The dramatic decline of Redshank populations on saltmarshes, and their usefulness as an indicator of both saltmarsh ecosystem health and the effectiveness of current conservation management (Exo *et al.*, 2017; Malpas *et al.*, 2013), was a major motivation for the work undertaken in this thesis.

The first published assessment of Redshank breeding in Great Britain (Thomas, 1939) indicates that, before 1900, the population was concentrated on the eastern side of Great Britain, with only a small number of breeding pairs at a few localities known in North West England. Subsequently, Redshank experienced a steady increase in breeding range and numbers across England, Wales, and southern Scotland (Thomas, 1939). Much of our current understanding of the breeding Redshank population on British saltmarshes is derived from three surveys conducted between 1985 and 2011 (Allport *et al.*, 1986; Brindley *et al.*, 1998; Malpas *et al.*, 2011). The first survey found the highest breeding densities in East Anglia and North West England. Follow up surveys indicated that North West England, and the Ribble Estuary, in particular, had suffered the largest Redshank population declines (Malpas *et al.*, 2011). Local scale but longer-term Redshank nest monitoring at Tipperne, Denmark between 1928 and 2016 show dramatically fluctuating breeding populations, with a relatively stable population of 25-75 pairs up to the late 1970s, a very significant increase to a peak of 400-575 pairs in 1990 followed by a return to lower levels of 100-140 pairs from the year 2000 onwards (Meltofte *et al.*, 2018).

Redshanks were once among the most common breeding birds in lowland habitats, but populations declined by an estimated 61% between 1980 and 2019, which resulted in their conservation status being classified as vulnerable to extinction in Europe (Birdlife International, 2021; PECBMS, 2019) (Figure 1.3).

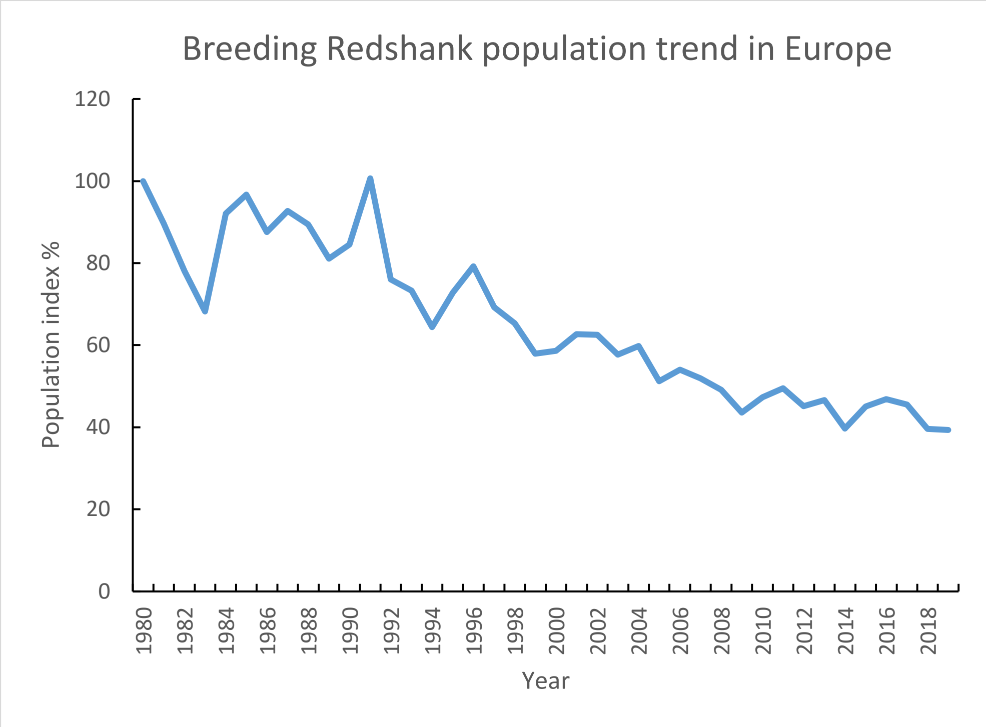



Figure 1.3 The collated annual population trend for breeding Redshank, *Tringa totanus,* across Europe between 1980-2019 (PECBMS, 2019; with index set at 100 in 1980).

Breeding waders can be problematic to census due to their non-territoriality and their often-cryptic nests and behaviour around the nest, and there is no single unified approach for assessing breeding populations between different habitats or countries. Two widely used methods for surveying breeding attempts for a variety of waders are those presented by O'Brien and Smith (1992), for lowland breeding wader populations, and Brown and Shepherd (1993), for upland populations (Gilbert *et al.*, 1998). The method developed by Green (1986) specifically for censusing breeding Redshank has been used in the British survey of saltmarshes (Allport *et al.*, 1986; Brindley *et al.*, 1998; Malpas *et al.*, 2011). One research aim of the current project was to test the efficacy of the Green (1986) survey method and determine if opportunities exist to improve estimates of the breeding Redshank population on saltmarshes. Research that explores systematic patterns of bias in censusing approaches, and if such errors contribute to flawed conservation management, serves to enhance the future application of ecological knowledge (Elphick, 2008). There have been calls to increase the robustness of monitoring schemes and to develop methods that can harmonise data from across different monitoring schemes (Normander *et al.*, 2012). The establishment of a centralised set of survey methods for bird monitoring schemes, such as occurs in the Pan-European Common Bird Monitoring Scheme (PECBMS) has promoted the collection of comparable data across national monitoring schemes and the exploration of forces driving changes in populations (PECBMS, 2019).

### 1.4.4 Identifying reasons for population declines

There are many potential reasons for the decline in breeding wader populations, which necessitates the collection of detailed ecological data to understand such complex issues. Direct loss of breeding and wintering habitats because of land reclamation, wetland drainage, flood control and coastal barrage construction are considered major causes for the population decline of Redshank (Burton *et al.*, 2006; del Hoyo *et al.*, 1996). Redshank declines have been linked to deterioration in habitat quality because of agricultural intensification and increases in livestock grazing intensity, but as discussed above, other herbivores including wildfowl may play a role which has not been fully considered. Redshanks are also vulnerable to severe cold periods on wintering grounds and are also susceptible to avian influenza (del Hoyo *et al.*, 1996; Melville and Shortridge, 2006).

Low breeding success, resulting from increased nest predation and declines in chick survival has been identified as the mechanism for recent population declines of several wader populations across Europe, including Redshank, Eurasian Oystercatcher, *Haematopus ostralegus*, Northern Lapwing, *Vanellus vanellus*, Black-tailed Godwit, *Limosa limosa*, and Eurasian Curlew, *Numenius arquata* (Roodbergen *et al.*, 2012). Wader nests are particularly susceptible to failure due to predation (Macdonald and Bolton, 2008), livestock trampling (Beintema and Muskens, 1987) and flooding (Green, 1986). The precocial chicks are also susceptible to predation or starvation (Teunissen *et al.*, 2008). Predation of wader nests in western Europe has increased by more than 40% across a suite of species, including Redshank, during the last four decades (Roodbergen *et al.*, 2012). In many studies, predation is the most important cause of Redshank nest and chick failure (Jackson *et al.*, 2004; Ottvall, 2004; Smart, 2005). Livestock can reduce Redshank nest survival by trampling nests (Beintema and Muskens, 1987; Sharps, 2015), but Mandema *et al.* (2013) suggest that Redshank may select nest sites with a relatively low trampling risk. Tidal flooding can cool or wash away the eggs of saltmarsh ground-nesting birds preventing incubation or drown chicks causing brief and localized population declines (Michener *et al.*, 1997). Redshank nest survival on saltmarshes may also be negatively affected by nest flooding (Norris, 2000; Smart, 2005; Thompson and Hale, 1991) but nesting in longer vegetation, which forms a cover over the eggs, can prevent them from being washed away, and incubation may successfully continue after tidal inundation (Hale, 1988). Climate change predictions suggest future increases in the frequency of extreme tides will occur, and in the Wadden Sea, maximum high tides have already increased twice as fast as mean high tide in recent decades, resulting in more frequent flooding of higher elevation saltmarsh sites (Howard *et al.*, 2019; van de Pol *et al.*, 2010).

### 1.4.5 Long-term studies and nest survival

Caution is required in drawing conclusions about the factors contributing to low breeding success because many studies occur in single years and there may be high annual variation in the factors influencing breeding success. Long-term studies based on repeated measurements of the same entities can be the most informative (Lindenmayer *et al.*, 2012). Without long-term research, potentially serious misjudgements can occur with conservation management recommendations (Magnuson, 2008). Long-term studies provide core ecological data as well as vital information on variability and trends and permit a greater understanding of complex systems where many factors might be operating concurrently (Lindenmayer *et al.*, 2012). One such example of pioneering data collection and research, which has led to a long-term dataset accumulating has been that of Redshank nesting on saltmarshes in North West England, conducted over decades and led by Professor W G Hale. This dataset offers a rare opportunity to explore the factors affecting Redshank nest survival over an extended period. Measuring nest survival is key to understanding population dynamics and is crucial to many studies of breeding birds. Nest survival can be modelled using the program MARK

(Dinsmore *et al.*, 2002; Dinsmore and Dinsmore, 2007; Rotella *et al.*, 2004). This approach can incorporate covariates such as livestock and wildfowl grazing, predation and flooding to allow the investigation of complex questions around the processes that affect nest survival. This is an advance from the Mayfield nest survival analysis approach which cannot include nest covariates in an efficient manner (Dinsmore and Dinsmore, 2007). Such modelling has been widely applied across a range of species and habitats and incorporated into conservation management actions (Colwell *et al.*, 2011; Polak, 2016; Stephens, 2003).

A key component of undertaking conservation-orientated research is that the outcomes are applicable and can be translated into actions on the ground. Conservation site managers are often repositories of detailed knowledge and understanding of the systems they manage and of their past changes, management, history and ecology (Duffield *et al.*, 2021); information that can be invaluable to researchers. In turn, they need to know which actions they can undertake are likely to provide suitable conditions for target species. In the case of ground-nesting waders, relevant site level management can be aimed at improving nest success and a better understanding of factors that explain nest survival (Shew *et al.*, 2019). One of the aims of the research presented in this thesis is to improve understanding of the critical factors impacting nest success in breeding Redshank, so that this can be conveyed to site-based conservation managers, linking applied ecology research to the delivery of solutions, as advocated by Palmer *et al.* (2005).

## 1.5 Thesis aims and outline

Below, I briefly describe the different chapters in my thesis, which use both contemporary, self-collected data, and long-term data to explore the usefulness of methods used for censusing breeding populations and the drivers of habitat suitability and nest survival in Redshank breeding on saltmarshes in North West England.

**Chapter 2 - Testing the efficacy of the standardised survey method (SSM) for censusing breeding Common Redshank, *Tringa totanus*, on saltmarshes.**

In this chapter, I will test the efficacy of the current best practice censusing approach for breeding Redshank on saltmarshes by comparing the survey results against detailed nest-searching across four saltmarshes in North West England. The aim is to determine if the survey results are significantly biased and if so to recommend ways in which they might be improved to make population estimates more reliable.

**Chapter 3 - Quantifying the temporal and spatial impact of wildfowl and livestock herbivory on saltmarsh vegetation in relation to breeding Common Redshank, *Tringa totanus*.**

Here, I examine the temporal and spatial impact of both wildfowl and livestock herbivory on saltmarsh vegetation height and relate this to Redshank nesting attempts and outcomes, as well as explore the relationship between nest trampling and livestock grazing pressure. I use exclusion experiments to manipulate cattle and wildfowl herbivory and assess the impacts on sward height. In parallel, I explore the characteristics of sward height around successful and unsuccessful Redshank nests.

**Chapter 4 - Determinants of Common Redshank, *Tringa totanus*, nest-sites on saltmarsh and potential responses to changes in management and sea level rise.**

In this chapter, I develop a predictive model to identify areas of saltmarsh habitat suitable for nesting Redshank. I use the model to explore possible conservation management options for maintaining and increasing breeding Redshank habitat by manipulating wildfowl and livestock use and assess the impact of plausible future changes in sea level.

**Chapter 5 - Nest survival of Common Redshank, *Tringa totanus,* on saltmarsh: the effects of wildfowl herbivory, livestock grazing, flooding, and predation**

Here, I examine the key factors affecting Redshank nest survival on saltmarshes. By analysing nest records spanning five decades and environmental variables that influence Redshank nest survival I attempt to identify the causes of population decline to assist conservation managers in taking positive action to protect this vulnerable species.

**Chapter 6 - General discussion**
Finally, I provide an overview of my key findings, linking them to previous research and discussing how they can contribute to our understanding and future conservation management of Redshank, as well as discussing the wider implications of the research.

# 2 Testing the efficacy of the standardised survey method (SSM) for censusing breeding Common Redshank, *Tringa totanus*, on saltmarshes

## 2.1 Abstract

The effective conservation management of species requires knowledge of their population size, with repeat surveys being vital to understanding changes over time and the potential drivers of such change. Breeding waders can be problematic to census due to their non-territoriality and often cryptic nests. Common Redshank, *Tringa totanus*, Redshank hereafter is one such species, which breeds in a range of wetland habitats from wet grassland and moorland to coastal saltmarshes. In the latter case, the much-dissected nature of inter-tidal saltmarshes, and the occasional flooding of areas by high tides present additional challenges for standard census methods. Current best-practice for surveying saltmarsh breeding Redshank populations in Britain relies upon a standardised survey method (SSM) that involves counting adult birds, which are then related to peak nesting, using a calibration curve that was developed primarily on lowland grasslands in southern and eastern England.

Here, I assess the efficacy of the SSM on four saltmarsh habitats in North West England over four years and contrast population estimates from such censuses with detailed nest-searching in the same areas. I found differences in breeding density estimates from nest recording and those from the SSM, which suggest that an alternate calibration curve better fitted my saltmarsh datasets. The SSM overestimated nesting density by an average of 42% indicating a relationship of 1.42:1 between birds counted and nests across the saltmarsh sites monitored in North West England. These differences likely arose due to the more dynamic nature of inter-tidal areas (*cf.* lowland grassland), the increased likelihood of the presence of non or failed breeders, and geographical variation in nesting phenology. To account for regional variation in breeding phenology, and climate-mediated temporal shifts in breeding, I recommend that the timing of surveys should reflect geographical and temporal variation in nesting phenology and should account for the impact of tidal flooding. Finally, I recommend the use of a double-sampling strategy to locate nests for future validation and highlight how technology can assist nest finding whilst also reducing disturbance.

The cautious interpretation of previous saltmarsh survey results for Redshank and modifications to the current SSM may be required to improve our understanding of populations and population changes.

## 2.2 Introduction

Reliable estimates of breeding bird populations, based on consistent and accurate methods, are an essential requirement for identifying population changes, establishing conservation priorities, and assessing the success or failure of conservation management (Brouwer *et al.*, 2003). Estimates of breeding bird populations come from a wide variety of sources. The Common Bird Census (CBC), a territorial mapping approach, and the Breeding Bird Survey (BBS), a line transect method, have been the main schemes for monitoring the population changes of common and widespread breeding birds in the UK (Freeman *et al.*, 2007). However, such methods are not appropriate for some types of birds, e.g., ducks and waders, which have secretive nesting habitats and non-territorial behaviour. For such species, alternative monitoring methods have been developed (Bibby *et al.*, 2000).

The two most widely used methods for surveying breeding attempts for a variety of waders, though not necessarily the most appropriate for individual species, are those presented by O'Brien and Smith (1992), for lowland breeding wader populations, and Brown and Shepherd (1993), for upland populations (Gilbert *et al.*, 1998). Within the waders, Redshank are somewhat problematic to survey using these standard methods because (i) they are non-territorial; (ii) their nests are typically well-hidden; (iii) adults display over wide and overlapping areas; (iv) they are inconspicuous during incubation, and (v) individuals may feed at considerable distances from nests. Smith (1983) was among the first to use a mapping-based approach to assess the distribution and abundance of breeding wader populations in lowland habitats. The method involved mapping the number, location and behaviour of all wader species recorded and using these to estimate the number of pairs present. However, for Redshank, Smith found a lack of correlation between the number of birds recorded and the estimated numbers of pairs. Mapped surveys are thought to underestimate breeding Redshank numbers during the incubation period and possibly overestimate numbers when the birds have young. Furthermore, such techniques, uncalibrated for Redshank breeding on saltmarshes, have been shown to underestimate substantially the breeding population when compared to detailed nest finding (Green and Johnson, 1984). Similarly, intensive nest finding on the Dutch Wadden Sea showed that numbers of Redshank individuals recorded during single-visit territory surveys represented only around 42% of the nest sites located (Dallinga, 1993).

The current best-practice standardised survey method (SSM) adopted in Britain specifically for censusing breeding Redshank on saltmarshes was calibrated from intensive nest-finding studies and bird counts undertaken simultaneously at several lowland wet grassland (LWG) sites in Cambridgeshire, Norfolk, Kent and Somerset between 1982 and 1984 and a single saltmarsh, Kirton Marsh, in Lincolnshire, UK (Green, 1986; Green and Johnson, 1984). Lowland wet grasslands are seasonally flooded freshwater grasslands characterised by networks of drainage ditches, typically occurring in lowland river valleys and behind sea defences, and traditionally managed by grazing or for hay production (Ausden and Treweek, 1995). Coastal saltmarshes comprise the upper, vegetated portions of intertidal mudflats, lying approximately between mean high water neap tides and mean high water spring tides (Natural England and RSPB, 2014). Saltmarshes have been under-represented in large scale surveys of lowland breeding waders compared to lowland wet grasslands (O'Brien and Smith, 1992).

Green and Johnson (1984) and Green (1986) concluded that using the SSM, the peak number of Redshank nests can be estimated with reasonable accuracy from the mean counts of Redshanks if standard rules on the timing of counts and the exclusion of flocks and adult birds with young are applied. Furthermore, Green and Johnson (1984, p.3) proposed that the widespread adoption of the SSM 'opens the way for extensive surveys of breeding Redshank on saltmarshes in Britain without the need to locate nests'.

National, regional, and local studies of breeding Redshank conducted in Britain over the past four decades have adopted the SSM approach. Consequently, much of our current understanding of the breeding Redshank population and its changes over time in Britain is founded upon the reliability of the SSM. The limited resource requirements and simplicity of the method are attractive and lend themselves to large scale implementation. However, little, or no additional systematic testing of the efficacy of the SSM on saltmarshes has been undertaken, despite the limited sample size and geographical range of the original calibration work. This is perhaps understandable given the resources required to conduct detailed and systematic nest finding research and the desire to focus on national surveys rather than rechecking a method which has become established over time through repetition. However, ensuring quality assurance in ecological data is critical for ensuring the reliability of subsequent analysis and associated evidence-based decision-making (Ferretti 2011).

Later research from the same region of the UK as the original calibration studies, conducted on non-saltmarsh sites, generally supports the initial findings regarding the efficacy of the SSM, although some anomalous relationships are reported where counts of Redshank severely underestimated the number of nests found (Smart *et al.*, 2006). On saltmarshes, concern has been expressed regarding the potential for the SSM to overestimate populations, especially in relation to the effects of tidal flooding during the count period (Smart 2005). On several important saltmarshes for breeding Redshank in North West England, where regular monitoring is a statutory requirement, the SSM has not been adopted due to concerns regarding its utility (Booth and Haywood, 2017, C Wells 2016 (RSPB Dee Estuary Reserve manager), pers. comm., 7 August)

UK breeding wader populations are in decline due to a combination of habitat loss, unfavourable habitat management and predation (BTO, 2021). At the current rate of decline, assessed using the SSM, breeding Redshank are likely to disappear from the majority of British saltmarshes within the next 25 years,(from 2011) (Malpas *et al.*, 2013) and the highest reported declines have occurred in North West England (Malpas *et al.*, 2011). However, the results of the periodic British surveys using the SSM do not necessarily concur with annual monitoring at the individual site level, using nest finding methods, which some site-based conservation staff regard as more reliable (e.g., K. Scott 2015, (Reserves Officer, Cumbria Wildlife Trust), pers. comm., 19 November).

Opportunities for improvement to the SSM may exist and this is a key aim in order to achieve progress in evidence-based nature conservation. Research that looks at systematic patterns of bias in censusing approaches, and if such errors contribute to flawed conservation management, enhances the future application of ecological knowledge (Elphick, 2008). Although there can be a temptation to retain established methods for continuity, if shortcomings are recognized, changing a methodology is recommended (Buckland *et al.*, 2005).

It is beyond the scope of this study to test all past methodological approaches employed at a local level. Instead, I focus on the SSM, my aim is to determine whether it produces a reliable and accurate population estimate when compared to intensive nest finding approaches.

I contrast breeding pair estimates of Redshank on four saltmarshes in the North West of England derived from the SSM method and from concurrent systematic nest finding over four breeding seasons. This re-examination of the efficacy of the long-established SSM may encourage its more common adoption outside of the British national survey or identify discrepancies that merit further consideration.

Specifically, I address the following questions:

1. Is there a significant difference between the slopes of linear regression models fitted to the calibration data for the SSM, (Green, 1986) and those from my study sites on saltmarshes in North West England?
2. Does the mean count of Redshank, using the SSM, provide an accurate predictor of the peak nesting density, at saltmarsh sites in North West England?

## 2.3 Methods

### 2.3.1 Study Area

More than one third of the total area of 33,571 ha of coastal saltmarsh habitat in England is located in the North West region (Phelan *et al.*, 2011). The sites for the current study are shown in Figure 2.1 and comprise Rockcliffe Marsh, Aldcliffe Marsh, Hesketh Out Marsh and Banks Marsh.

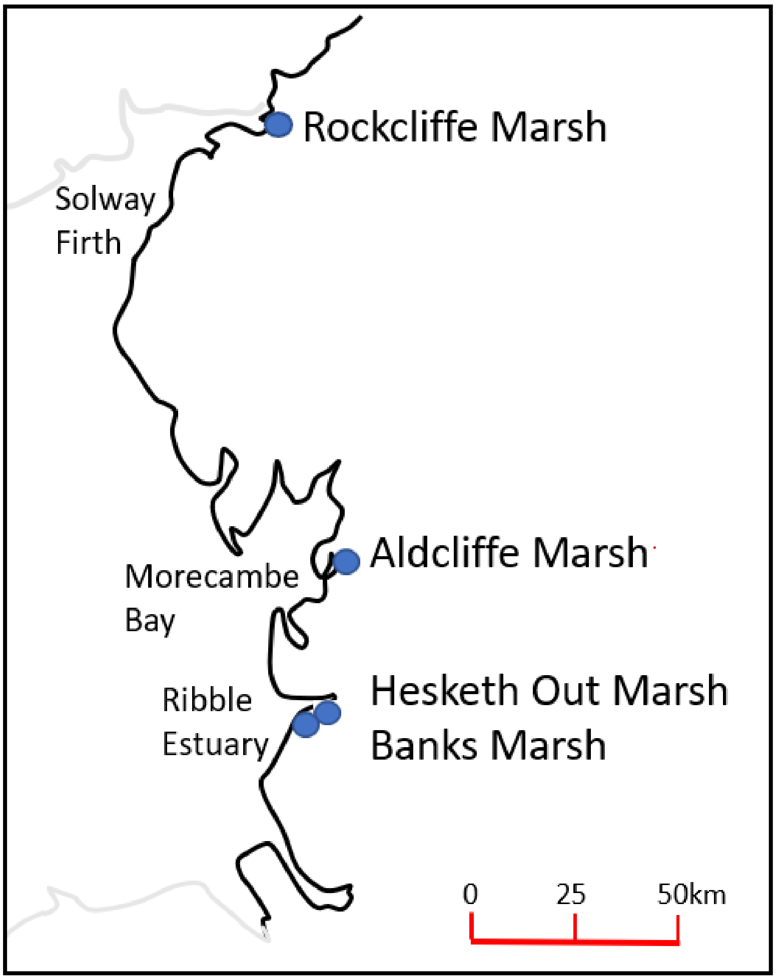


Figure 2.1 Saltmarsh study sites on the coast of North West England where simultaneous surveys using the SSM and intensive nest finding were undertaken.

In this study, 17 plots, ranging in size from 10.8 to 53.9 hectares(ha), mean of 26.2 ha, were established across the four saltmarshes. Six plots corresponded to those originally selected for the national surveys of Redshanks on British saltmarshes (Allport *et al.*, 1986). These plots were selected to cover a range of communities and grazing regimes representative of those prevalent in the region (Allport *et al.*, 1986). These plots were supplemented by 11 new, randomly selected plots on Banks Marsh and Hesketh Out Marsh. Some saltmarsh areas were excluded from the additional random plot selection process due to concerns regarding disturbance to other breeding species or because they could not be accessed without crossing dangerous tidal channels. All study plots are subject to tidal flooding during the breeding season. The additional plots were relatively homogeneous in terms of vegetation community and livestock grazing levels.

All sites except for Aldcliffe Marsh are managed as nature reserves and have long time series of breeding Redshank data, though these data were gathered using alternative census methods to the SSM. Additionally, Banks Marsh was the site of continuous research on breeding Redshank between 1969 and 1989. The availability of intensive Redshank nest recording for Rockcliffe Marsh and Banks Marsh in 1985 allowed comparison with concurrent results using the SSM for these sites (Cumbria Trust for Nature Conservation, 1985; W G Hale 2016, pers. comm., 30 December). SSM count and nest data were kindly provided for Hesketh Out Marsh in 2018 ( R McCloud 2020, pers. comm., 9

November) All other plots were surveyed by the author using both the SSM and intensive nest finding between 2016 and 2018.

### 2.3.2 Study Species

The Redshank is a medium-sized (27-29cm) ground nesting wading bird of the sandpiper family, *Scolopacidae*. Redshanks are non-territorial and can be semi-colonial during the breeding season (Cramp and Simmons 1983; Hale 1988). Redshank breed in a range of habitats, including lowland wet grassland (LWG) and upland grassland, but in Britain saltmarshes hold the highest proportion of breeding pairs. In these areas, nesting occurs between April and early July. There is a strong association with nesting in *Festuca rubra, Elytrigia spp*. and, less frequently, *Puccinellia maritima* grassland communities in the mid to upper zone saltmarsh (Hale, 1980; Norris *et al.*, 1997; Sharps *et al.*, 2016; Thyen and Exo, 2005). During laying and incubation, Redshanks form a grass canopy above a nest cup by pulling together vegetation. Dallinga (1993) reported a median nest vegetation height of 29cm, and 20cm, in short, grazed areas. Replacement clutches are often laid after a failed breeding attempt, with Green & Johnson (1984) reporting a mean of 1.96 nesting attempts per breeding pair.

The UK breeding population of Redshank is estimated at 25,000 pairs, with approximately 45% nesting on saltmarshes and the highest densities in North West England and East Anglia (Brindley *et al.*, 1998; RSPB, 2021a). British saltmarshes are both nationally and internationally important for breeding Redshank, supporting more than 18% of the northwest European breeding population (BirdLife International, 2004). Due to declines in the breeding population, Redshank is on the Amber list of UK Birds of Conservation Concern, with Red being the highest conservation priority and Amber the next most critical group, (DEFRA, 2020; Stanbury *et al.*, 2021). On British saltmarshes, the decline of breeding Redshank is estimated at 9,500 pairs or 53%, between 1985 and 2011, with 3,042 pairs estimated to have been lost in North West England over that period (Malpas *et al.*, 2013).

### 2.3.3 Standard Survey Method (SSM) bird counts

Green (1986) demonstrated a very strong correlation between the mean density of Redshank seen during three counts in April and May and the estimated peak density of incubated nests. The strength of this relationship provides the justification for using bird counts as an SSM to estimate the breeding population without having to find nests.

I followed the published guidelines for conducting the SSM fully, as detailed below (see Allport *et al.*, (1986)). The SSM involves walking a pre-planned route passing within 100m of all areas of a survey plot. I followed routes using a Garmin GPSMAP 64S handheld navigator, walking at a slow methodical pace and maintaining consistency between surveys. All Redshank observed were counted and their locations were recorded on a study plot map. Movements of birds within a plot were noted to avoid double counting. Behaviours including displays, song flighting, obvious pairs, birds flushing from creeks or vegetation and adults with young were noted. Immediately after completing the surveys, the data were checked to resolve any potential ambiguities. Each plot was required to be surveyed on at least three occasions between April 15$^{th}$ and May 30$^{th}$ with a minimum 10-day interval between visits under the SSM. I undertook additional surveys where possible to assess variation in Redshank counts in relation to different visit dates within the prescribed survey period. All surveys were completed between 08.00 and 17.00 British Summer Time (BST), as no diurnal fluctuations in the activity patterns of breeding Redshank have been documented. Surveys were undertaken during different tidal states but not during periods of spring high tides, rain, or strong wind. The mean number of adult Redshanks counted in each plot, during three survey visits, was calculated. Two important exclusions from the mean count calculation are necessary when following the SSM: (i) flocks of more than six Redshank are excluded, as the method assumes that these are likely to be non-breeding birds and, (ii) Redshank behaving as if they had young are omitted from the calculation. For each study plot, the mean birds counted per $km^2$ was calculated to allow comparison between plots. Published counts for plots on Banks Marsh and Rockcliffe Marsh in 1985 were also included in my analysis from Allport *et al.* (1986). Surveyors had prior experience using the SSM as part of a past study (Norris *et al.*, 1997).

Some divergence from the recommended SSM timing and the number of counts have been found in the literature. For example, Cook *et al.* (1994) commenced counts as early as the 2nd of April on saltmarshes in Essex, and Smart *et al.* (2003) in a follow-up survey, used two survey counts not three. Similarly, Norris *et al.* (1997) adopted a 6th April start date on the Wash estuary, East Anglia. The justification for and potential impact of these variations is not reported.

### 2.3.4 Nest finding and monitoring

Nest finding and monitoring were conducted in a manner consistent with the original calibration of the SSM (Green and Johnson, 1984; Green 1986). Details from published and grey literature, accounts from field workers involved in the original studies and advice from recognised experts were used to ensure that the methods adopted in this study were appropriate and comparable. Nest finding can be considered the 'gold standard' benchmark for estimating breeding populations, providing all nests are found without detrimentally influencing their outcomes through disturbance. Nest finding provides an estimate of breeding Redshank density independent of counts of adults. My testing of the SSM required an accurate assessment of the peak number of incubated nests in each study plot based on finding and monitoring the nests present.

The timing and frequency of nest searching and monitoring are critical if all nests are to be found and the peak of incubated nests accurately calculated. I conducted a nest-finding pilot study on Banks Marsh in 2015 to gain a general understanding of the local timing of the Redshank breeding season. My main nest-finding fieldwork, from 2016 to 2018, started on the third week of March each year, and continued until late June or early July, the end date is determined as being one week after the final nest had failed or hatched and no more nesting activity had been detected. Each study plot was searched every 3-4 days during this period. Nest searching was limited to a maximum 30-minute session in a 1 ha area before moving to another distant area to avoid excessive disturbance in the same location. Field experiments have demonstrated that a 30-minute nest disturbance time did not reduce the daily survival of clutches up to the point of hatching or increase the daily clutch predation rate for breeding waders (Fletcher *et al.*, 2005).

The frequency of systematic searches for nests for the original calibration of the SSM is not reported (Green & Johnson 1984; Green 1986). However, the fieldworker conducting the Kirton Marsh saltmarsh SSM calibration study (Green and Johnson, 1984) recalled conducting surveys throughout weekdays from mid-April to early July (D Collins 2019, pers. comm., 18 December).

The techniques I adopted for nest finding, evaluated during the pilot study, were selected based on maximising nest detection, whilst minimising disturbance, and being achievable given the human resources available. Scrape finding and active searching for nests were the primary approaches, with flushing and *ad hoc* passive searching employed as secondary approaches. I did not attempt rope dragging for nest finding as, (i) it was not a practical option with one fieldworker, (ii) Green (1986) found it to be the least efficient method, and (iii) some consider it ineffective on saltmarshes (D Collins 2019, pers. comm., 18 December). Previous long-term nest finding studies on Banks Marsh have demonstrated that an experienced fieldworker using the above approaches (detailed further below) can complete a thorough nest finding search on an area of approximately 250ha, working 5 days per week over the breeding season (W G Hale 2015, pers. comm., 10 April). I applied this level of nest finding effort, with field workers experienced in locating Redshank nests on saltmarshes. Below I describe the four methods of nest detection in more detail.

1. Scrape finding

I commenced searching for scrapes three weeks in advance of the expected first egg laying, based on the pilot and ongoing surveys. Male Redshank may form 15-20 readily recognisable scrapes, comprising 14-15cm circles of bare ground, often in tufts of vegetation, two to three weeks before laying takes place (Hale 1988). Scrape locations were marked with a Garmin GPSMAP64S, and clusters were identified and used to target nest searching effort, as one may be selected as the basis for a nest. Preferred scrapes for nesting may be identifiable before the first egg is laid by the absence of vegetation or the presence of 2cm x 1cm pellets containing crustacean and mollusc fragments. Green (1986) does not describe using scrape finding in the SSM calibration studies, but it is a commonly employed technique that improves nest finding efficiency, particularly in the early stages of the breeding season (W G Hale 2015, pers. comm., 10 April).

2. Active searching

During laying and incubation, vegetation is pulled over the nest to conceal it. I used active searching for field signs that might indicate the presence of a hidden nest, such as suitable grassy tussocks, runs or entrances to a bower concealing a nest or a partially hidden bird on a nest. Partially covered nests can be seen from above and fully concealed nests can be located by cautiously opening grass tussocks. Incubating birds frequently left nests when approached within 5m, aiding nest location. My active searching technique is taken to be similar to the 'cold' searching approach described by Green (1986), which involved searching all tussocks.

3. Flushing

Systematic flushing of adults from their nests was used as a secondary method of nest detection. The flushing method described by Green (1986) comprised walking a systematic route, which approaches all points of the search area to within 50m whilst scanning the ground 50-100m ahead. If a bird appears to fly up from a nest, the area was actively searched as described above. Green demonstrated flushing to be less efficient than active searching.

My pilot study indicated that Redshank frequently left nests only when approached at a shorter distance than 50m. A simple field experiment was undertaken on Banks Marsh during the 2017 and

2018 breeding seasons to test the flushing method. Two randomly selected study plots, where all nests had been located, were searched by an independent observer experienced in the use of the flushing method, and the success rate for finding the known nests was recorded.

4. Passive searching

Passive searching, or watching birds return to nests from a distant vantage point, can be a useful technique but was only suitable on my saltmarsh study sites when close to the sea wall, where it was possible to get an elevated observation position.

For each nest located, I recorded clutch size, incubation stage, GPS coordinates, and vegetation characteristics of nests. Nests found before clutch completion were dated based on the assumption that a typically clutch of four eggs, takes an average of 6 days to complete (W G Hale 2015, pers. comm., 10 April). The incubation stage of completed clutches was gauged by observing the degree of flotation of eggs in water (van Paassen, *et al.* 1984). Nest disturbance and predator attraction were limited by not physically marking nest locations but instead using GPS co-ordinates to allow efficient relocation within a 3-5m radius. Georeferenced photographic records of nests and surroundings also aided efficient, low-impact monitoring. The use of a Pulsar Helion XP28 Thermal Imaging Scope to detect the presence of incubating birds or warm eggs from 8-10m also proved useful in minimising nest disturbance. The estimated incubation start and end dates were used to calculate the peak of nesting i.e., the maximum daily number of incubated nests for each study plot. This is reported as peak nesting density per $km^2$ for comparison between plots. Maxim DS1921G-F5 data loggers were placed in the Redshank nests in the 2018 breeding season to record the temperature. This enabled improved measurement of the timing of nest failure or hatching as the ending of incubation could be established by associated temperature change, and hence peak nesting density was more accurately assessed.

I adopted the approach to estimating nest finding efficiency developed by Green (1985, 1986), to ensure comparability. By measuring the incubation stage when a nest is found, it can be determined if the nest was previously missed during regular repeated systematic searches, and thus nest finding efficiency can be determined.

### 2.3.5 Statistical Analysis

I fitted a linear regression model to the published calibration data of the original SMM (Green 1986, Box 1.4), a plot of mean birds counted $km^2$ (y-axis) versus peak nests $km^2$ (x-axis). Additional data from the saltmarsh study site used to develop the SSM were not available for inclusion but a similar relationship (approximately 1:1) between birds counted and peak nests was reported (Green and Johnson, 1984). A linear regression model was similarly fitted to the data collected in the current study. As I conducted additional bird counts on twelve of my study plots, multiple qualifying values of mean birds counted per $km^2$ were available for analysis. To make my census structure identical to the SSM, I used a series of ten random draws of qualifying bird counts and associated peak nesting density calculations for each plot and year, i.e., I included, on each occasion, only three SSM visits per season.

Wald tests were used to examine whether the slopes of my regression models for each random draw differed significantly from that of Green (1986).

In my data, a single outlying data point was identified for a plot where very high numbers of birds were counted but there was no correspondence with nests in the area. The high number of birds counted on this occasion was a result of flooding and displacement of Redshank from an adjacent area of saltmarsh. Hence, I undertook analyses with and without this outlier. I used the LinearHypothesis() function in the 'car' Companion to Applied Regression package, version 3.0 -10, (Fox and Weisberg, 2019) for analyses in R.

## 2.4 Results

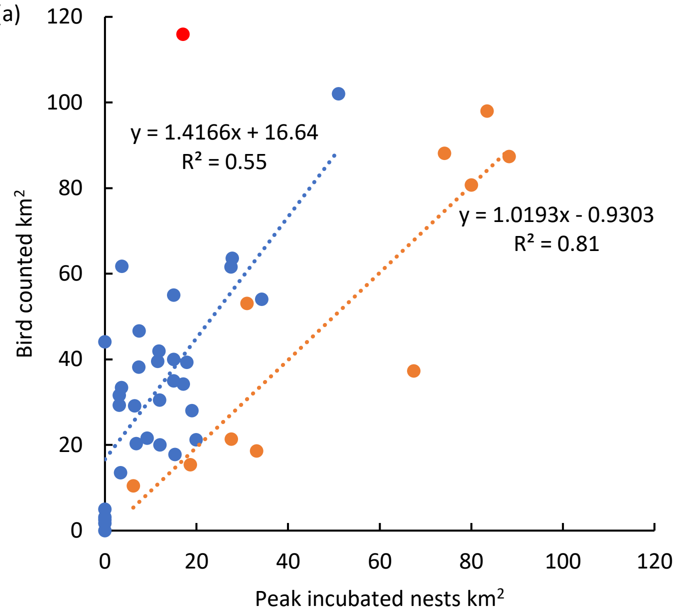
Across all study plots the mean peak nesting density = 11.91 nests /km$^2$ (SD= 11.29). There was a difference in the relationship between the mean number of Redshank counted and the peak number of nests identified by intensive searching between my data and that of Green (1986). The mean slope from my random draw datasets indicates an average ratio of 1.42:1 for birds counted relative to peak incubated nests compared to the approximately 1:1 ratio measured by Green (1986) (Table 2.1). The mean intercept from my data was 16.13 SD 1.197 compared to -0.93 in the Green (1986) model (Table 2.1 and Figure 2.2). There was no justification for forcing the regression line through the origin.

After excluding the outlier point, I did not find a statistically significant difference between the mean slope from my saltmarsh monitoring and that of the SSM at the level of $P \leq 0.05$ (Wald test, $W_T$ = 0.084, (Figure 2.2a). However, the results of the Wald test for 2 out of 10 individual draws reached a significant difference between the slopes for the saltmarsh and SSM models($P \leq 0.05$), with individual levels of significance varying between $W_T$ = 0.048 and $W_T$ = 0.136.

Table 2.1: Wald Test results examining the difference in the slope of linear regression models of peak incubated nests and birds counted, from saltmarsh plots in NW England and lowland wet grassland plots in SE and S England (Green 1986). Results for ten random draws of the saltmarsh plots in NW England are shown. Bold text indicates when slopes differed significantly (P≤0.05). Section (a) excludes one outlier point and (b) includes the outlier in the analysis.

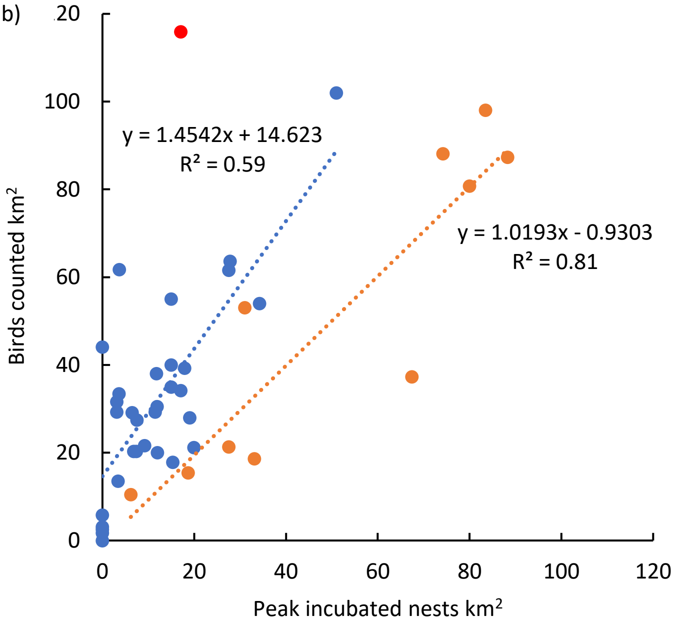
| (a) All Plots excluding outlier n=32 | | | | Wald Test Statistic $W_T$ | (b) All Plots including outlier n=33 | | | | Wald Test Statistic $W_T$ |
|---|---|---|---|---|---|---|---|---|---|
| Random Draw | Slope | Intercept | $R^2$ | Green slope =1 | Random Draw | Slope | Intercept | $R^2$ | Green slope =1 |
| **Draw 8** | **1.454** | **14.623** | **0.589** | **0.048** | Draw 8 | 1.551 | 15.783 | 0.458 | 0.078 |
| **Draw 5** | **1.456** | **14.677** | **0.586** | **0.050** | Draw 5 | 1.553 | 15.833 | 0.457 | 0.078 |
| Draw 10 | 1.450 | 14.700 | 0.585 | 0.053 | Draw 10 | 1.547 | 15.859 | 0.456 | 0.081 |
| Draw 7 | 1.453 | 16.147 | 0.559 | 0.064 | Draw 7 | 1.548 | 17.284 | 0.446 | 0.087 |
| Draw 2 | 1.417 | 16.640 | 0.551 | 0.084 | Draw 2 | 1.512 | 17.778 | 0.437 | 0.107 |
| Draw 1 | 1.419 | 15.724 | 0.543 | 0.088 | Draw 1 | 1.516 | 16.875 | 0.430 | 0.110 |
| Draw 3 | 1.423 | 17.132 | 0.534 | 0.091 | Draw 3 | 1.518 | 18.261 | 0.430 | 0.110 |
| Draw 9 | 1.397 | 16.483 | 0.525 | 0.112 | Draw 9 | 1.492 | 17.629 | 0.418 | 0.129 |
| Draw 4 | 1.381 | 17.088 | 0.533 | 0.116 | Draw 4 | 1.477 | 18.229 | 0.422 | 0.135 |
| Draw 6 | 1.384 | 18.130 | 0.505 | 0.136 | Draw 6 | 1.478 | 19.254 | 0.409 | 0.145 |
| Mean | 1.423 | 16.134 | 0.551 | 0.084 | Mean | 1.519 | 17.279 | 0.436 | 0.106 |
| SD | 0.029 | 1.197 | 0.028 | 0.031 | SD | 0.030 | 1.185 | 0.018 | 0.024 |

(a)



120
100
80
60
40
20
0
Bird counted km²
y = 1.4166x + 16.64
R² = 0.55
y = 1.0193x - 0.9303
R² = 0.81
0
20
40
60
80
100
120
Peak incubated nests km²


(b)



120
100
80
60
40
20
0
Birds counted km²
y = 1.4542x + 14.623
R² = 0.59
y = 1.0193x - 0.9303
R² = 0.81
0
20
40
60
80
100
120
Peak incubated nests km²

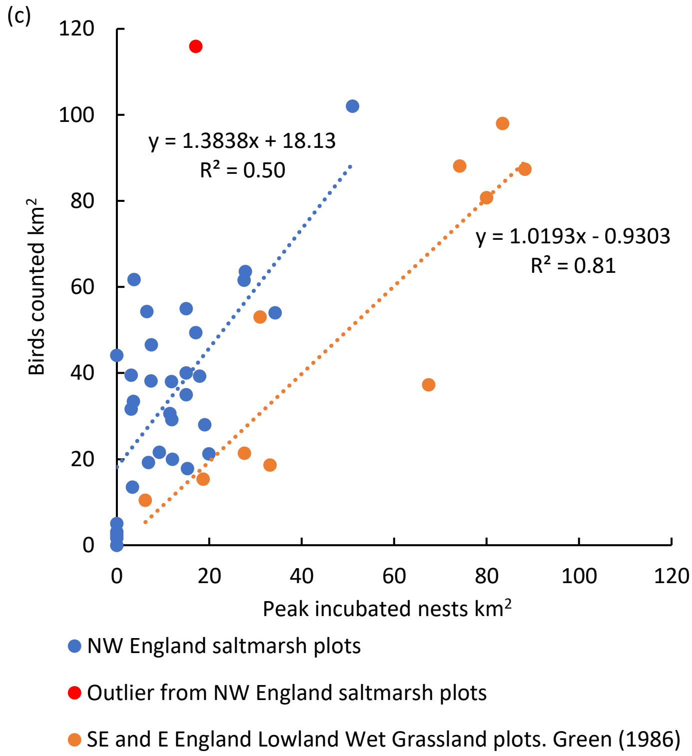



Figure 2.2 Comparison of the relationship between peak incubated nests and birds counted for Saltmarsh plots in NW England and the lowland wet grassland habitats for (a) the mean $W_T$ result, (b) the lowest $W_T$ result, and (c) the highest $W_T$ result. A single outlier point for the saltmarsh plots is excluded from the analysis presented in the graphs but is shown in red.

In assessing the accuracy of the SSM on my saltmarsh study sites, I found a consistent positive bias towards the overestimation of peak nesting density using counts of birds. My results suggest that using a 1.42:1 ratio of birds counted to peak nests on my study sites would provide a better estimate of peak nesting density rather than the 1:1 ratio currently used in the SSM. However, equally importantly, I detected higher variability in the bird count to peak nest relationship on my saltmarsh sites compared to the LWG calibration sites of the SSM ($R^2$ =0.551 ± 0.028 for Saltmarsh plots compared to $R^2$ = 0.808 for SSM plots).

## 2.5 Discussion

I found that (i) applying the SSM for breeding Redshank on my study saltmarsh sites in North West England consistently overestimated peak numbers of nests, compared to intensive nest finding, and (ii) there was higher variability in the bird count to peak nest relationship on these saltmarshes than the grassland sites where the SSM was primarily calibrated. This could have important implications for the conservation status designation and management of Redshank across the UK, and for other regions and species where similar methods are applied. Overestimation and inconsistency in population estimates derived using the SSM reduce their scientific validity and usefulness for conservation management, e.g an overestimation of a breeding Redshank population could lead to a false impression that conservation measures had been successful. No census method of breeding birds will return estimates that exactly match those arising from intensive nest-finding. Instead, an ideal method should provide a very close approximation of the real nesting population but require a much-reduced survey effort. The SSM is a species-specific approach explicitly designed to overcome known difficulties with estimating breeding Redshank populations and should ideally provide a reasonable approximation of nesting numbers, such that changes in populations over space or time can be detected. The ultimate goal of a population census is to obtain population estimates with low (or no) bias and high precision in a cost-effective and logistically feasible manner (Thompson, 2002). The pragmatic aim of producing populations estimates 'within 25% of the true value' has been suggested, (Bibby *et al.*, 2000, p. 28). The results of my testing of the SSM on saltmarshes in North West England indicate a strong positive bias leading to an average overestimation of peak nesting density by 42% and a high degree of variance across repeated surveys. Potential reasons for the overestimation of the SSM across out study sites are explored in the following sections, and suggestions for improvements to the method proposed.

### 2.5.1 Methodological effects

Nest finding provides a direct measure of the breeding population and 'for some bird species with weak signs of territorial behaviour, nest finding is the only really good way of counting them' (Bibby *et al.*, 2000). Nest finding underpins the calibration work for the SSM, which was developed specifically to address recognised difficulties in censusing Redshank through territorial mapping approaches (Green and Johnson, 1984). An obvious issue in using nest finding as a benchmark for the SSM is that it requires a measurement of confidence that all or most nests are found.

An estimate of nest finding efficiency, based on whether a nest is missed during regularly repeated searches, and assessed through the incubation stage, was employed by Green (1985, 1986) and repeated in my study. In my study, 93% of nests were discovered during the first search, and frequently before clutch completion. This is a substantially higher nest-finding success rate than the 59% reported during the original calibration of the SSM (Green, 1986). Possible reasons for this discrepancy include the additional use of scrape finding and the application of new technology but could also reflect differences in the grassland sward of LWG versus saltmarsh grassland. Buckland *et al.* (2005) suggest that the issue of detectability can be safely ignored if detection 'is certain, or nearly so' without defining certainty. I propose that the methods and nest finding efficiency demonstrated in this study are comparable to the original calibration for the SSM, and possibly more efficient.

Flushing birds from nests by walking within 50m, rather than the active searching primarily used in my study, has been adopted as the method for estimating Redshank nests in multiple studies (Feather *et al.*, 2016; Sharps *et al.*, 2015). Green (1986) found a lower probability of finding nests using flushing compared to active searching, but flushing does have advantages in terms of speed and ease of use. In support of the use of the 50m flushing distance, it has been suggested that most Redshank rise from nests when approached at 50-200m (Ferguson-Lees *et al.*, 2011). However, this contrasts with others who recorded a wider range of Redshank behaviours, ranging from remaining on nests when an observer is present to leaving a nest when approached within 600-700m (Hale, 1988). Mandema *et al.* (2014) suggested that, upon the approach of a predator, Redshanks only leave the nest at the very last minute. My limited testing of the flushing method demonstrated a relatively low nest finding success, with only 23% of known active nests found by an independent experienced observer. Further, I recorded Redshank most frequently leaving nests when approached within 0-15m at the Banks Marsh study site. In comparable studies on Banks Marsh, Sharps *et al.* (2015) found 17 nests in 767ha using the 50m flushing method, compared to my later study on the same site, using active searching, in which I located, on average, 62 nests in 263 ha in 2017 and 2018. Although this difference might represent inter-annual variability, an order-of-magnitude difference in nesting density (2.22 nests per $km^2$ versus 23.57 nests per $km^2$) does highlight a potential methodological limitation of the flushing approach. Redshank population estimates between the 2011 and 2012 breeding seasons on the wider Ribble Estuary indicated very different densities based on different methods. In 2011, using the SSM, Malpas *et al.* (2013) estimated 17.67 pairs per $km^2$ whilst in 2012, using the flushing method to find nests, Sharps *et al.*(2015) located only 3.11 nests per $km^2$. While it is not possible to directly compare these measurements in different years, they may highlight the risk of over or under-estimating Redshank populations using different methods. Further testing of the flushing method for nest finding should be considered a sensible precaution.

Observer bias in counting and interpreting counts was minimised in my study by strictly following the guidelines of the SSM. Observer competence in conducting the SSM was based on previous experience, including data collection using the SSM for Norris *et al.* (1997). Variations to the SSM guidelines, such as using two counts instead three (Smart *et al.*, 2003), and varying start dates outside the prescribed range (Cook *et al.*, 1994) were avoided as they may have unintentionally introduced bias.

Given no systematic differences in the gathering of nest and bird count data between my study and the original calibration studies of the SSM (Green and Johnson, 1984; Green, 1986), I suggest that differences found in the relationship between birds counted and peak nesting density arise due to differences in Redshank behaviour and habitat between the study areas.

### 2.5.2 Bias related to nesting density

It is possible that the number of nests is overestimated when sampling areas of lower nesting density using the SSM. The mean peak nest densities in the LWG calibration data of the SSM was 50.9 nests per $km^2$, n = 10, compared to 11.9 nests per $km^2$, n = 33 in my test plots, i.e., a four-fold difference. Allport *et al.* (1986, p.64) report that 'at very low densities some surveyors felt that the SSM overestimated the population' of Redshank on saltmarshes across Britain. Further testing of the SSM across a greater range of measured nesting densities is recommended.

### 2.5.3 Bias related to different habitats

The very strong correlation between birds counted and nests found on LWG, (r = 0.90, slope = 1.02, n=10 P <0.001; calculated from raw data in Green (1986), is broadly supported by the later findings of Smart *et al.* (2006) working in similar habitats and the same geographical region. On coastal grazing marsh sites, Smart, *et al.* (2006) found that the estimated numbers of pairs were strongly correlated with the peak number of nests found (r = 0.91, slope = 1.1, n = 24, P <0.001) and on some inland LWG sites there was a significant correlation (r = 0.81, slope = 1.03, n = 13, P < 0.001). However, a severe underestimation of nesting density, using the SSM, was reported at other inland sites in the same study (r = 0.55, slope = 0.22, n = 12, P = 0.07) (Smart *et al.*, 2006). No further published research measuring the relationship between counts of Redshank and nests found, on LWG or saltmarsh habitats have been found.

The results of Green and Johnson (1984) calibration of the SSM for saltmarsh, from Kirton Marsh Lincolnshire, are described as similar to those for LWG reported in Green (1986). The results of my more extensive testing of the SSM on saltmarshes (r= 0.77, slope=1.42, n= 32) demonstrate the possibility of overestimating peak nesting density by an average of 42% using the SSM in this habitat. Below I discuss the potential causes of this overestimation on saltmarshes in North West England.

### 2.5.4 Overestimation of breeding numbers by counting non-local breeders

The high bird counts associated with zero or a very low number of incubated nests (Figure 2.2) are likely to be partly explained by the presence of non-local breeding birds. Spring passage birds may be present with local breeding Redshank early in the SSM counting period. Few or no non-breeding flocks occur on LWG, whereas non-breeding flocks on saltmarshes are much more likely (M O'Brien 2019, pers. comm., 20 December). Archer and Branston (2014) acknowledged that coastal sites may support large non-breeding populations of Redshank during the breeding season but found little evidence of this at Poole Harbour and other south coast estuaries in Britain. Comparing Wetland Bird Survey (WeBS) monthly counts of Redshank on Banks Marsh indicates a consistently higher number of Redshank in April and an average reduction of 86% by May, (Figure 2.3) suggesting the likely presence of a significant number of non- breeding birds in the April count period for the SSM.

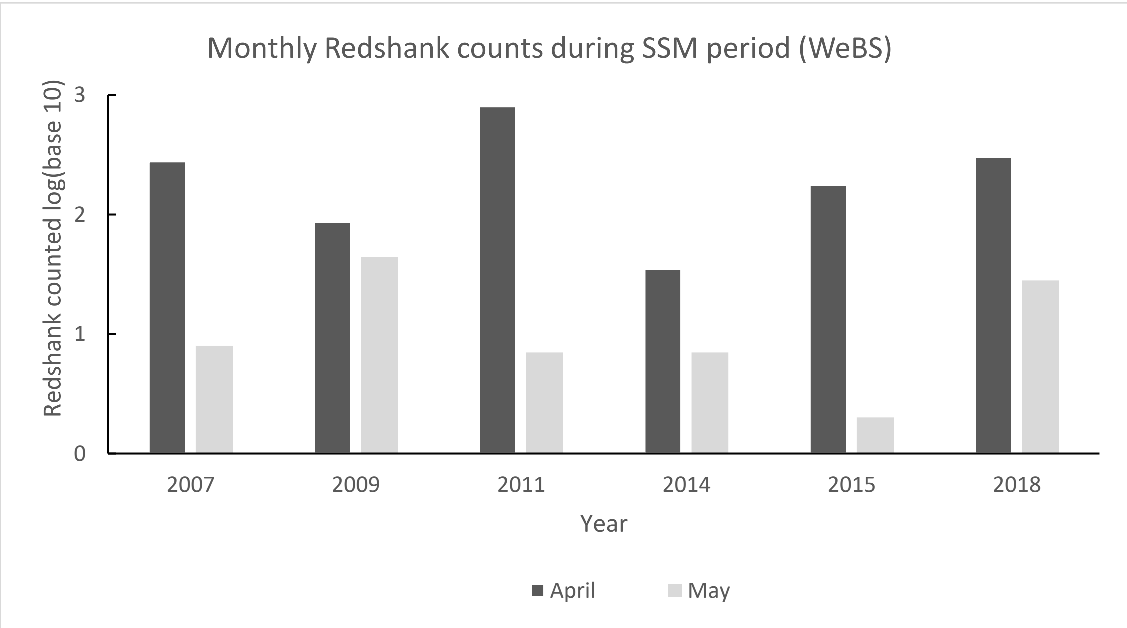



Figure 2.3. Redshank numbers (log scale) recorded during April and May by Wetland Bird Survey (WeBS) counts on Banks Marsh Frost *et al.* (2021). Data shown are years when both WeBS count dates fell within the April 15th to May 31st timing guidelines for the SSM.

The SSM deals with potential non-breeding birds by excluding any group of six or more co-occurring birds from the counts used to estimate peak nesting density. I have been unable to find the rationale behind this figure, although the concept of ignoring larger groups from the potential breeding population counts seems intuitively sensible. A study contemporary with the development of the SSM observed that Redshanks tend to occur in discrete pairs, but also in groups of three or more in areas of the highest breeding density (Reed and Fuller, 1983). Due to the difficulty in distinguishing passage birds from breeders, Greenhalgh (1971) only included pairs or birds leaving nests in his estimation of breeding Redshank populations for saltmarshes in Lancashire. Ausden *et al.* (2003) identified the importance of estuarine feeding areas for breeding Redshank nesting on nearby coastal grazing marshes. Such travelling by Redshank to feed on saltmarsh habitats could explain discrepancies between my calibration of the SSM for saltmarshes and those derived from LWG sites. During my surveys, groups of up to 13 Redshanks were counted together but subsequently excluded based on the SSM protocols. Further interpretation of the SSM, six bird count rule, based on a surveyor's experience of identifying migrant bird behaviour has been suggested but this is problematic to assess objectively, and concerns remain regarding a potential bias resulting from the inclusion or exclusion of groups of non-breeding birds in the estimation of breeding populations.

### 2.5.5 Overestimation of breeding numbers due to saltmarsh flooding.

Tidal inundation during spring high tides and storm events presents an increased risk of nest failure for ground nesting bird species on saltmarshes, something not typically experienced in LWG habitats. The SSM assumes that only one individual from a pair is counted because not all incubating birds are flushed from nests (Smart, 2005). However, in saltmarsh habitats, there is clear potential for both birds in a pair to be recorded during counts following nest failure due to tidal flooding, and hence an overestimation of peak nesting density could occur.

The likelihood of nest failure during a flood event depends on interrelated factors including tidal height, weather, nest location and the stage in the breeding season. The extent of nest failure from tidal flooding can vary significantly between years (Thompson and Hale, 1991). Variability among years could not be considered in the original SSM calibration study as data were collected during one breeding season when a 14% loss of nests to flooding was recorded (Green & Johnson, 1984). The mean percentage of Redshank nests lost due to flooding in a breeding season at my study sites on Banks Marsh between 1969-89 and 2016-18, was similar at 18.4% (SD = 13.3%). Clutch replacement following loss due to flooding took an estimated 10-12 days. This may result in some overestimate of nests (based on birds counted) as both birds from a pair may be recorded on consecutive SSM counts, which are required to be a minimum of 10 days apart.

Smart (2005) and McCloud (2019) recognised that tidal flooding can result in a considerable overestimate of nests using the SSM, particularly on lower elevation saltmarshes. Smart (2005) astutely suggested the first SSM count be undertaken 10 days after the first spring tide in the peak laying period to ensure that most breeding pairs should be incubating a first nesting attempt. However, the application of this approach does require prior knowledge of when the peak laying period will occur. Records of the peak nesting period from long term studies at Banks Marsh indicate considerable annual variation, from 8$^{th}$ May to 11$^{th}$ June (Figure 2.4). Adopting the timing recommendation of Smart (2005) could have been a practical solution for two of the three breeding seasons in my study to avoid significant flooding impacts. However, in years where the peak of nesting occurs later, adjustment of the SSM timing rule requiring three survey counts to be completed before the end of May would be required. Long-term nest monitoring data for Banks Marsh from 1969 to 2018 shows that nest losses due to tidal flooding occur throughout the breeding season from April to July, so a workable correction for the tidal flooding impact on the SSM may require supplementary nest monitoring.


Figure 2.4 Dates of the start of laying, peak nesting and completion of last Redshank nesting on Banks Marsh in relation to the period for undertaking SSM surveys. Data for 1969-1989 were obtained from unpublished nest records courtesy of W.G. Hale.

In addition to causing nest failure, tides may also affect the SSM in other ways. Firstly, high spring tides occurring just before the start of laying can delay the breeding season. On Banks Marsh in 2016, laying commenced 10 days later than anticipated on April 23rd following extreme spring high tides in the second week of April. Similar reports of tidal flooding in early April at Rockcliffe Marsh, Cumbria in 1985 delayed nesting by at least two weeks (Cumbria Trust for Nature Conservation, 1985). This could result in more individuals being counted in early counts, as birds delay nesting, and consequently result in an over-estimate of breeding populations.

### 2.5.6 Other potential causes of nest failure impacting SSM estimates

Nest failure rates due to predation and trampling by livestock may also influence the SSM by increasing the likelihood of counting more than one bird from a breeding pair if they remain in the area, potentially to re-lay. Green & Johnson (1984) report a 49% probability of newly laid clutches being taken by a predator and a 35% probability of being trampled by cattle whilst maintaining a 1:1 relationship of mean birds counted to the peak incubated nests. In two of my study plots, where 100% nest failure was recorded because of predation and nest trampling, ratios of mean birds counted to the peak incubated nests were 10:1 and 44:0. Many instances of nest predation occurred before the start of incubation, increasing the ratio of birds counted to the peak of incubated nests used in the SSM. A high degree of temporal and spatial variability in the rate of nest loss due to predation and trampling has been observed at the Banks Marsh study site over the 1969-89 and 2016-18 breeding seasons. The mean percentage of nests lost to predation was 13.4% (SD = 12.3%), and the mean losses to nest trampling by livestock was 11% (SD = 12%). The highly variable predation and trampling rates among my saltmarsh sites, combined with flooding effects, may explain the weaker relationships between mean counts of birds and peak nest counts observed during the testing of the SSM.

### 2.5.7 Geographical variation in nest timing and changes in nesting phenology

The fixed timings of the SSM, between April 15th and May 31st, may impact population estimates because of geographical variation in the timing of peak nesting. The SSM calibration of Green & Johnson (1984) and Green (1986) was focused on southern and eastern England, but it is used in saltmarsh surveys across Britain with a common timing guideline (Allport *et al.*, 1986; Brindley *et al.*, 1998; Malpas *et al.*, 2011). The suitability of the SSM timings for the first British saltmarsh survey in 1985 was based on expert opinion of when peak nesting occurred (M O'Brien 2019, pers. comm., 20 December). Smart (2005) recorded the peak laying period for Redshank on saltmarshes in East Anglia between 2003 and 2005 being 15th April to 13th May, fitting within the SSM guidelines. By contrast, on Banks Marsh in North West England, peak nesting occurred later than the prescribed SSM count period in 8 out of 24 years for which data were available (Figure 2.4). Additional Redshank nest record data was requested from the British Trust for Ornithology, to evaluate phenological variation in breeding across Britain, but were not available. The O'Brien & Smith (1992) method for censusing lowland breeding waders' populations addresses geographical variation in nesting timing, with an earlier start in southern England and Wales compared to northern England and Scotland. Timing modifications to the SSM were adopted by Cook *et al.* (1994) with an earlier start date of the 2nd of April on Essex Saltmarshes rather than the 15th of April.

Changes in nesting phenology over time may also affect peak nesting estimates from the SSM given its fixed timings since 1985. Climate change is having an impact on the timing of life cycles in birds, with a trend towards earlier laying-dates (Winkler *et al.*, 2002). At a long-term study site in Denmark, Redshank advanced breeding initiation by about one week between 1926 and 2016 (Meltofte *et al.*, 2018). Available records for Banks Marsh indicate a tendency for first laying to advance but the completion date of the last nest to remain unchanged between 1969 and 2018, but with considerable inter-annual variation (Figure 2.4). This may lead to a mismatch between established timings for the SSM and changing Redshank breeding phenology.

### 2.5.8 Future research directions and alternative approaches

Given the evidence that I have found that applying the SSM on my study system overestimates breeding Redshank populations, the use of a revised relationship between Redshank counted and nests estimated (using the ratio of 1.42:1 rather than assuming a 1:1 relationship) for saltmarshes in

North West England could be cautiously suggested for the region. SSM estimates based on the 1.42:1 ratio could also be considered for UK saltmarshes. However, this is likely to be unsatisfactory as further testing of these relationships would be required. There is clearly scope for further study to better understand how breeding season Redshank counts, in relation to peak nesting, can be impacted by habitat, flooding and phenology. Further use of the SSM across a wider geographical range of saltmarshes, in parallel with nest finding, over several seasons may help refine the SSM methodology for saltmarshes and could consider the impact of the timing of counts based on tides, geographical variation in nest timing, and changes in nesting phenology. The additional costs of further testing of the SSM need to be balanced against the risk of compromised inferences from its results. The current study has demonstrated how effective testing can be achieved with small resources and person-time when aided by appropriate technology such as thermal imaging. Drone surveying may present further opportunities for improved Redshank nest finding whilst potentially reducing disturbance during censusing in challenging saltmarsh environments (Valle and Scarton, 2020).

Bart and Earnst (2002) suggest using a double-sampling approach for censusing waders which involves surveying a large sample of plots using a rapid count-based assessment, similar to the SSM, combined with intensive nest finding in a subsample of the same plots to determine actual nesting density. The ratio of the mean counts to the mean actual density on the double sampled plots can be used to correct the results of all the rapidly surveyed plots. Such an approach works well when results from the rapid count method are highly correlated with actual nest density and would allow for the recording of additional information such as impacts of flooding, predation and nest trampling on the intensively searched plots with little additional effort. I suggest that the feasibility of double sampling be evaluated to test the assumptions of the SSM wherever it is applied.

Distance sampling has become one of the most widely used methods for estimating the density and abundance of birds with several new developments making the methods more widely applicable, reducing bias and increasing precision (Marques *et al.*, 2007). Distance sampling approaches have been used with Breeding Bird Survey (BBS) data to produce revised population estimates for breeding Redshank in the UK (Newson *et al.*, 2008). A key assumption of distance sampling is that birds are certain to be detected at zero distance, so birds that remain concealed and undetected will produce a negative bias in abundance estimates (Buckland *et al.*, 2008). My study indicates that incubating Redshanks frequently do not leave when an observer is at the nest site. This was confirmed through interviews with recognised experts (W G Hale 2015, pers. comm., 10 April; J Smart 2016, pers. comm., 31 March). Newson *et al.* (2008) also acknowledge that distance-sampled estimates tended to be higher for species with a large proportion of non-breeders, which is a potential source of overestimation bias for Redshank breeding on saltmarshes. I suggest distance sampling methods for Redshank could be assessed in further field trials with sample nest finding.

The bias inherent in any survey method can be revealed by well-designed field trials based on nest finding, supported by the application of new technology, to provide the most reliable baseline for comparison (McCafferty, 2013; Valle and Scarton, 2020). Greater confidence in tested survey methods can then be demonstrated to researchers and conservation practitioners. This is more likely to result in the standardisation of methods and comparability of results in the UK and other countries, a fundamental requirement for species population monitoring schemes that may currently operate in isolation (Moussy *et al.*, 2022).

## 2.6 Conclusion

This study aimed to test the efficacy of the SSM and assess whether opportunities exist to improve estimates of the breeding Redshank population on saltmarshes. My results indicate that the SSM consistently overestimated nesting densities on saltmarshes in North West England compared to intensive nest finding. Sources of bias likely included, (i) the presence of non-breeding birds, (ii) the variable impact of tidal flooding and other causes of nest failure and (iii) geographical variation and changes in nesting phenology over time. These can be systematically adjusted for with additional nest monitoring or a double sampling approach and I recommend that future use of the SSM for saltmarshes in North West England should adopt a relationship of 1.42:1 between birds counted and peak nesting density as it represents the current best knowledge.

My results potentially indicate the need to reassess the conservation status of breeding Redshank on saltmarshes in Britain as the population may have declined more than the SSM suggests, due to its potential to overestimate populations. Redshank populations may face a greater threat than currently envisaged, and conservation actions aimed at reversing declines may be getting a false signal of success from the SSM.

# 3 Quantifying the temporal and spatial impact of wildfowl and livestock herbivory on saltmarsh vegetation in relation to breeding Common Redshank, *Tringa totanus*.

## 3.1 Abstract

The population of breeding Common Redshank, *Tringa totanus,* Redshank hereafter, on British saltmarshes is estimated to have halved in the last 30 years. Overgrazing by livestock, which can result in vegetation too short for concealing nests, and a high incidence of eggs being trampled in nests have been suggested as key drivers of population declines, particularly in North West England. However, light to moderate livestock grazing has been shown to benefit both breeding Redshank and wintering herbivorous wildfowl by creating structural diversity in the sward. Consequently, light cattle grazing is the management practice adopted where these features are conservation priorities. However, a largely unstudied potential impact on Redshank breeding is that of overwinter grazing by wildfowl. High intensity wildfowl grazing can adversely impact habitat suitability for breeding Redshank.

Here, I examine the temporal and spatial impact of both wildfowl and livestock herbivory on saltmarsh vegetation height and relate this to Redshank nesting attempts and outcomes, as well as explore the relationship between nest trampling and grazing pressure. I use exclusion experiments to manipulate cattle and wildfowl herbivory and assess the impacts on sward height. In parallel, I explore the characteristics of sward height around successful and unsuccessful Redshank nests. This work, conducted over three years on a saltmarsh of national and international importance for both breeding Redshank and winter wildfowl, demonstrates the critical role wildfowl herbivory can play in limiting sward height during the Redshank nesting period, and the potential conservation conflict this creates. Trampling of nests occurred in the areas of highest livestock use but these areas tended not to correlate with high Redshank nesting densities. Finally, I found that nest predation was not significantly affected by sward height.

My findings on how habitat suitability for breeding Redshank is affected by both wildfowl and livestock herbivory illustrate the need to consider the impact of wildfowl in conserving Redshank.

## 3.2 Introduction

Coastal saltmarshes are highly productive ecosystems occurring worldwide but most extensively in temperate and high latitudes. Comprising the upper, vegetated portions of intertidal mudflats between mean low-water spring tides and mean high-water spring tides, saltmarshes support a range of habitats and species of nature conservation significance. Saltmarsh pastures provide important breeding habitats for waders and other species and a food source for herbivorous migratory wildfowl (Adnitt *et al.*, 2007). These nature conservation attributes are often the primary reason that saltmarshes are afforded legal protection and are the primary targets of conservation management (Doody, 2008)

Migratory wildfowl, which feed on saltmarsh vegetation, notably Eurasian Wigeon, *Mareca penelope*, and Pink-footed Goose, *Anser brachyrhynchus*, have shown long term increases in abundance of 147% and 548% respectively, in the UK, since the mid-1970s (DEFRA, 2020). In contrast, populations of breeding Redshank the most abundant wader species breeding on British saltmarshes, have undergone significant declines, estimated at 53% between 1985 and 2011 in Britain, and the species is now classified as amber listed in terms of conservation priority (Eaton et al., 2015). This decline has been linked to a deterioration in habitat quality as a result of increases in livestock grazing intensity (Norris et al., 1998; Eglington & Noble, 2010; Malpas et al., 2013). Changes in sward height associated with livestock grazing intensification can negatively affect the availability of nest sites and foraging efficiency for breeding Redshank (Vickery et al., 2001). However, wildfowl herbivory can also markedly affect saltmarsh vegetation structure, reducing the height and the balance of vegetation (Cadwalladr *et al.*, 1972; Smith & Odum, 1981; Cargill & Jefferies, 1984). The impact of wildfowl herbivory in this context remains relatively unstudied, yet requires consideration alongside livestock herbivory amid a growing body of evidence of wildfowl effects on ecosystems (Green and Elmberg, 2014). Understanding the temporal and spatial impact of wildfowl and livestock herbivory on saltmarsh vegetation in relation to breeding Redshank is the primary aim of the work presented in this chapter.

Saltmarsh grazing by livestock is a long-established tradition in North West England, where some of the largest conservation-managed sites in Europe occur (Doody, 2008). Livestock grazing is widely used to deliver beneficial nature conservation outcomes on saltmarshes and is promoted by agri-environment schemes (Chatters, 2004; Mason et al., 2019). Light summer cattle grazing (<1 Livestock Unit [LU] per ha) is considered beneficial for both breeding Redshank and winter wildfowl (Beeftink, 1977; Norris et al., 1997; Lambert, 2000; Adnitt et al., 2007). It can create a structurally diverse sward of longer vegetation suitable for concealing Redshank nests as well as areas of shorter vegetation for feeding and which aid predator detection (Milsom *et al.*, 2000). The availability of such well-structured nest-sites of advanced succession stages on saltmarshes significantly improves the reproductive success of Redshank and other ground nesting birds (Thyen and Exo, 2005). However, the impact of light summer grazing on Redshank nesting may be detrimental if it is concentrated in areas where, and at times when, Redshank nest, due to the possibility of nest trampling (Sharps *et al.*, 2017).

Livestock grazing on saltmarshes can also facilitate feeding opportunities for migratory wildfowl that preferentially feed on grazed saltmarshes (van der Graaf et al., 2002; Mandema et al., 2014). However, following intense winter or spring usage by wildfowl, saltmarshes may be overgrazed, limiting suitable habitat for nesting Redshank at the start of the season (Cadwalladr et al., 1972). Changes in nesting phenology with earlier start dates for Redshank may also be exacerbating this problem (Meltofte *et al.*, 2018).

The impacts of migratory wildfowl species on saltmarsh swards are less predictable and controllable than livestock grazing and are subject to change unrelated to local livestock grazing such as population growth as a result of off-site factors such as hunting pressure, breeding ground conditions or alternative over-winter forage (Norris, 2000). The conflict between increasing wildfowl populations and agricultural production is well documented but the potential conservation conflict between wildfowl and breeding waders is less studied (Fox et al., 2017; Vickery et al., 1997). As wildfowl feeding occurs mostly outside of the Redshank nesting period it has received less attention than livestock grazing, being notably absent from assessments of the causes of the decline in Redshank breeding on British saltmarshes (e.g., Malpas et al., 2013).

The requirement to understand the spatial and temporal distribution of both wildfowl and livestock herbivory is critical to understanding their influence on saltmarsh vegetation characteristics important for nesting Redshank. This study conducted, over three years, on a saltmarsh of international conservation importance adopts a holistic and detailed approach to understanding the drivers of Redshank nesting success. Thereby my intention is to inform evidence-based practical solutions to the problem of declining Redshank populations which can be adopted both at the focal study site and more widely.

Here, I investigate:

1. The patterns of herbivorous duck, geese, and cattle use of a saltmarsh National Nature Reserve over 30 months and relate these factors to Redshank nesting attempts.
2. The impact of experimentally excluding key herbivore populations on saltmarsh sward structure to assess the key drivers of sward condition and hence suitability for breeding Redshank.
3. If differences in vegetation height at and around Redshank nests result in differences in nest outcomes.

## 3.3 Methods

### 3.3.1 Study Area

The study area is Banks Marsh, Lancashire, United Kingdom, the core of the Ribble Estuary National Nature Reserve (NNR) and the Ribble and Alt Estuaries Special Protection Area (Figure 3.1). A mosaic of *Puccinellia maritima* and *Festuca rubra* saltmarsh communities covers most of the 10km$^2$ site. Conservation grazing by cattle has been practised on site between May and October at recommended levels of approximately 0.7 cattle per ha since the establishment of the NNR in 1979 (Lambert, 2000; Adnitt et al., 2007). Internationally important populations of herbivorous wildfowl feed on the saltmarsh vegetation between late September and early May. Eurasian Wigeon and Pink-footed Goose are the most numerous species, with peak winter counts for the Ribble Estuary ranging between 48-75 thousand, and 11.6-34 thousand individuals during the study period 2016/17-2018/19 (Frost et al., 2021). Five-year mean peak counts for 2017/18, Wigeon 51.9 thousand and Pink-footed Goose 22.5

thousand, for the Ribble Estuary (Figure 3.2) represent 11.5% and 4.4% of their British winter populations respectively (Frost *et al.*, 2019). An estimated 80-120 nesting attempts of Redshank occur on Banks Marsh between April and July (Booth and Haywood, 2017). This represents a marked decline of approximately 50% from 1979 when 164 nests were recorded in the same area, as part of a long-term study (W G Hale 2015, pers. comm., 10 April).

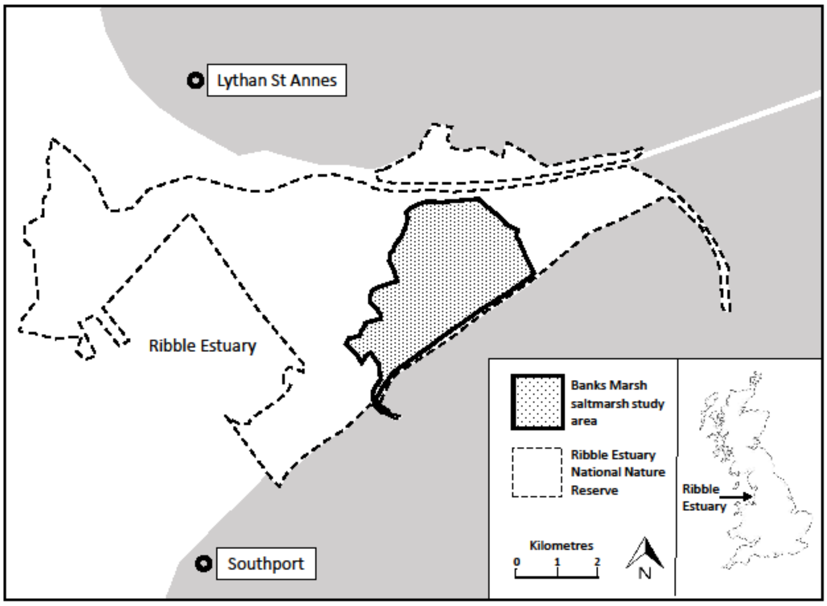



Figure 3.1 Location of the Banks Marsh saltmarsh study site within the Ribble Estuary National Nature Reserve in North West England.

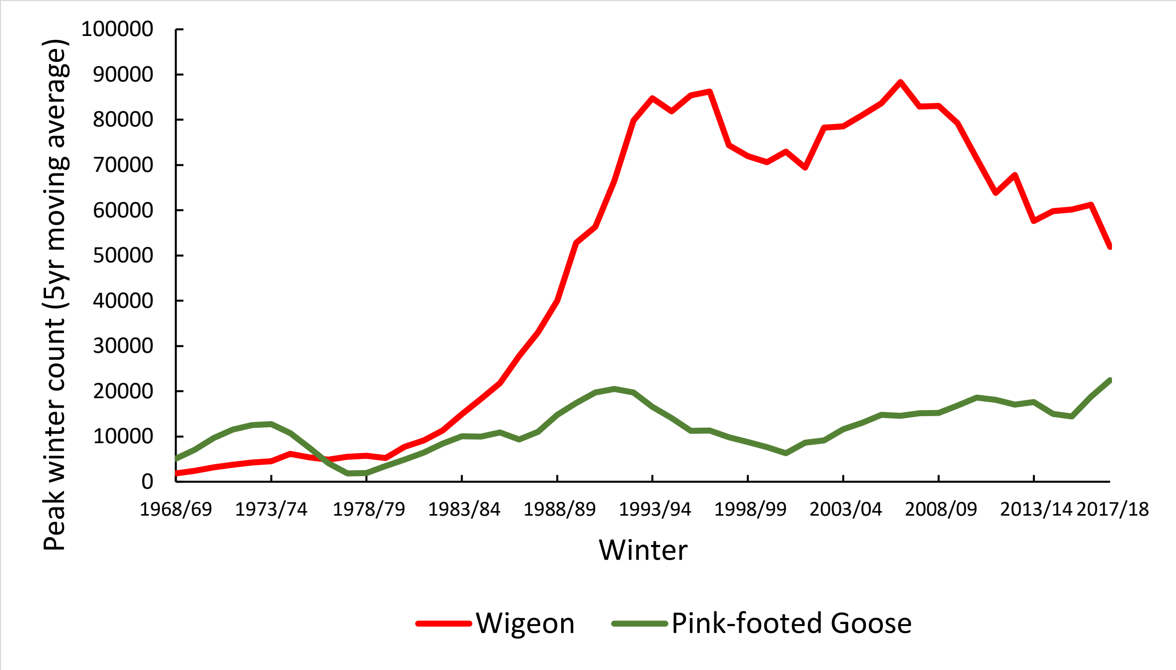



Figure 3.2 Wigeon and Pink-footed Goose population trends for the Ribble Estuary 1968/69 to 2017/18. Peak winter wildfowl counts are Wetland Bird Survey (WeBS) data (Frost et al., 2021)

### 3.3.2 Grazing intensity indices

Dropping counts are a widely used and reliable method of assessing wildfowl and cattle grazing intensity on saltmarshes (Owen 1971; Rankin 1979; Bos et al., 2005), each goose dropping being equivalent to 3.99 duck droppings, based on published mean dry weights for Wigeon and Pink-footed Goose droppings (Kear, 1963; Mayhew, 1988). I used dropping density counts to provide a standardised estimate of duck, goose, and cattle use intensity across the site. Monthly counts at fixed sampling points arranged in a 250m X 250m grid covering all accessible areas of the site were conducted between August 2016 and May 2019 (Figure 3.3). Cowpats were sampled within an 8m radius (approximate area of 200m$^2$), and duck and goose droppings within an annulus between 3-5m (approximate area 50m$^2$) around the fixed sampling points.

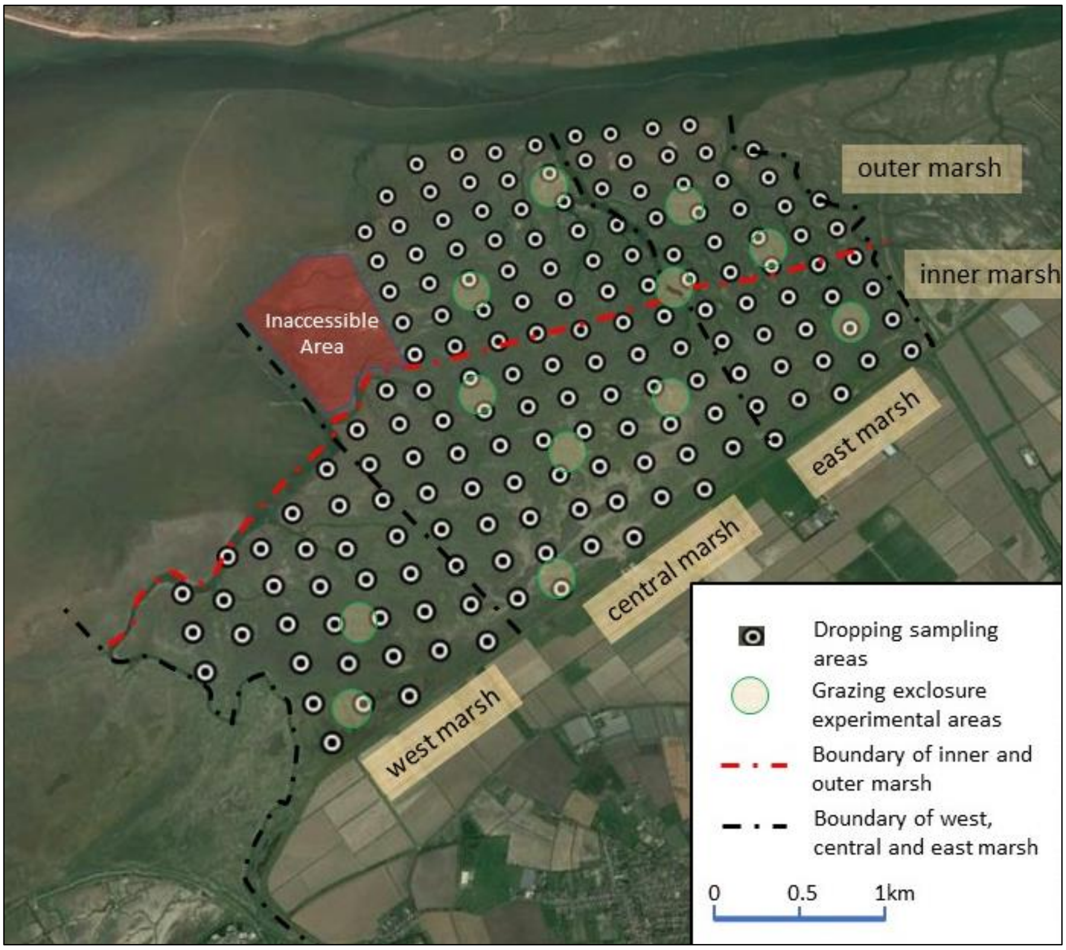



Figure 3.3 Map of Banks Marsh study area showing; (i) the locations of dropping sampling points (n=171), (ii) the boundaries of the marsh sub-regions, and (iii) the grazing exclusion experimental areas (n=12).

At each sampling point, duck and goose droppings were identified and counted separately. Duck and goose droppings could be differentiated by size (Figure 3.4). Single droppings produced while grazing were recorded, whereas clumped piles of three or more sets of droppings were ignored, as they were assumed to indicate roosting sites rather than local feeding activity (Owen, 1971). All duck droppings were assumed to be from Wigeon, which is the dominant duck species on site. Similarly, all goose droppings were attributed to Pink-footed Geese. A composite wildfowl dropping count was also calculated for each sampling point, weighted as 1 goose dropping  being equivalent to 3.99 duck droppings, based on published mean dry weights for Wigeon and Pink-footed Goose droppings (Kear, 1963; Mayhew, 1988)

Wildfowl droppings were counted shortly before the highest monthly tides, the high tides then removing/dissolving droppings, allowing a new accumulation before the next pre-high-tide count. This process was repeated monthly throughout the entire study period. A storm driven high tide in January 2017 led to the removal of wildfowl droppings from 25 of the 171 sampling points before they could be counted. All other datasets were fully collected. A similar process was used to survey cowpats, though as the latter were less likely to be fully removed between tidal cycles, individual pats were marked to prevent double counting and to distinguish old from fresh dung in subsequent counts.

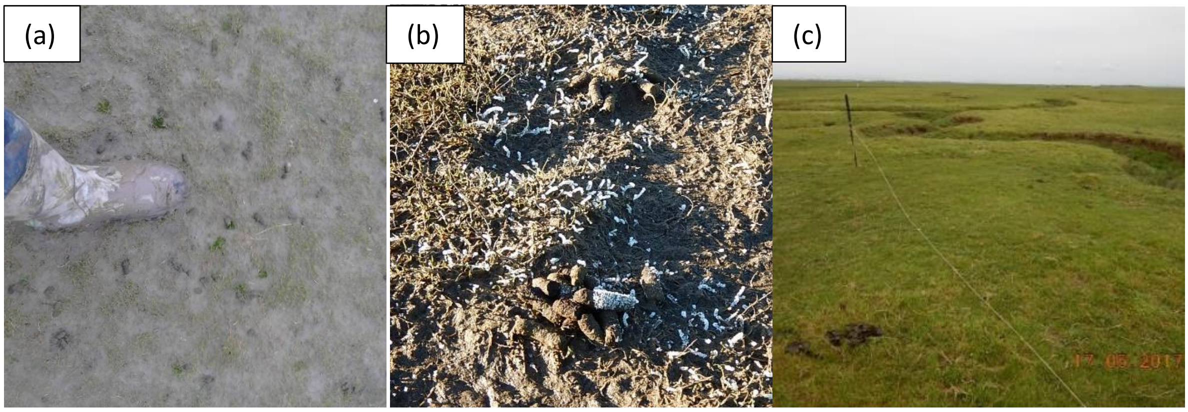



Figure 3.4 (a) Wigeon droppings (b) Pink-footed Goose droppings and (c) cowpats, recorded at fixed sampling points.

Spearman's rank correlation ($r_s$) was used to assess the correlation in site use among species, and among years within species using the sampled data. Possible facilitation effects of summer cattle grazing promoting winter duck and goose are examined.

Ordinary Kriging was used to produce interpolated maps of dropping densities for each of the three groups, which I took to reflect the intensity of use by cattle, ducks and geese across the study area, at a 200m$^2$ (14.142m x 14.142m ) spatial resolution (Wackernagel, 1995; Nunes et al., 2019). Kriging was performed using the gstat package version 2.0-7e in R (Gräler et al., 2016).

Interpolated cattle use intensity at Redshank nest locations was examined in relation to nest outcomes. Wilcoxon rank sum tests were used to determine if cattle use intensity differed significantly between nests grouped by outcome, with the potential nest outcomes being trampled by cattle, hatched, predated, or flooded.

### 3.3.3 Exclosure experiments

The impact of cattle and wildfowl on sward height was assessed by excluding these herbivores from experimental plots. To do this, the study site was first divided into 1km$^2$ areas. In each area, the nearest homogenous vegetation stand to a randomly selected point was identified. Four 1m$^2$ experimental plots were randomly selected within the homogenous stands at distances between 50-100m apart. Four different treatments were randomly applied to one of the four localities as follows: (i) cattle exclusion only, May to October, (ii) wildfowl exclusion only, October to May, (iii) year-round cattle and wildfowl exclusion, and (iv) a control, fully open to both cattle and wildfowl grazing.

Exclosures were constructed of wooden posts, reinforcing bars, and wire mesh. The design was effective in preventing cattle and wildfowl herbivory inside the plots throughout the experimental period from February 2017 through to May 2019 (Figure 3.5)

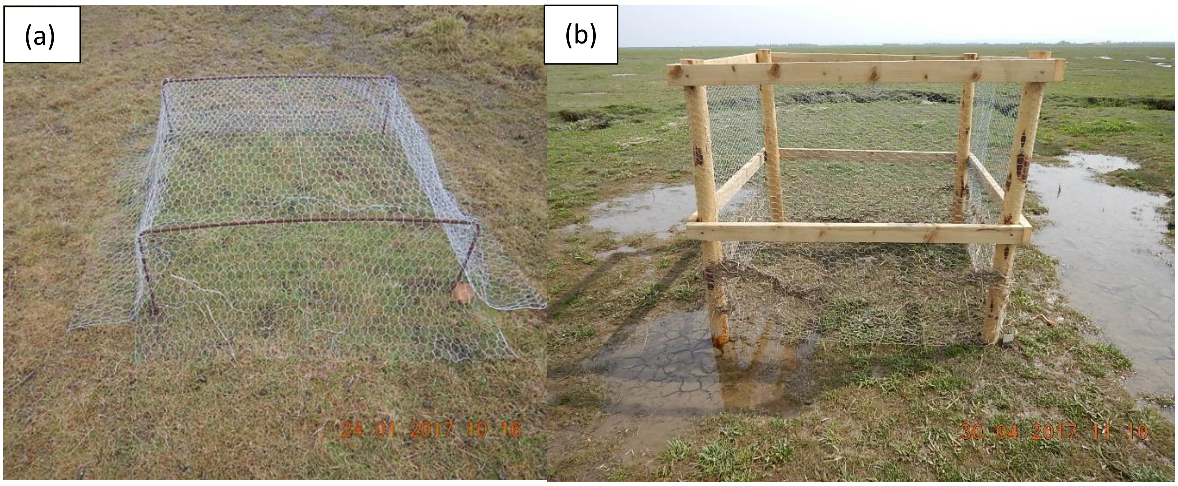


Figure 3.5 Herbivore exclosure designs (a) Wildfowl exclosure - wire mesh covers the top and sides and the height is adjustable to accommodate vegetation growth and (b) Cattle exclosure – wire mesh and rails prevent cattle feeding from the sides or over. The combined cattle and wildfowl exclosure design, not shown, is the same as a cattle exclosure with the addition of wire mesh covering the top to prevent wildfowl from entering the excluded area.

Sward height, vegetation species, cover and abundance were recorded in each plot at the start, middle and end of the exclusion experiment, following National Vegetation Classification (NVC) guidelines (Rodwell, 2006). However, as there was negligible change in vegetation species, cover and abundance over this time period, this information was not used any further.

At monthly intervals, the vegetation height in each experimental plot was measured at 10 random sample points to the nearest 0.5cm, and the mean value was calculated. This provided consistent and accurate results and is more suitable for measuring variation in short turf than sward sticks or drop disks (Stewart *et al.*, 2001). Wilcoxon signed rank tests were used to determine if the vegetation heights in the grazed control plots and the exclosure treatment plots differed significantly from each other across all 12 grazing exclosure sites.

### 3.3.4 Vegetation and nest outcomes

A systematic nest finding approach, described in Chapter 2, was used to locate Redshank nests as part of a concurrent investigation of the efficacy of censusing methods. Vegetation heights were measured (to the nearest 0.5cm) at and around 127 nests, located over two breeding seasons. Vegetation height was randomly sampled at i) the nest cup ii) ≤ 1m from the nest cup and iii) at a random distance and direction between 1-10m from the nest. Vegetation height measurements were replicated at a randomly determined control point, between 80m and 100m from each nest. Measurements were made either on the day nests were first located or, if adults were considered to be present, within 3 days afterwards, to avoid unnecessary disturbance to incubation. This method contrasts with previous studies (e.g. Sharps, 2015) where vegetation height measurements were taken after nesting had finished. I considered the latter approach to introduce unnecessary variance, as vegetation could have continued growing after hatching or conversely subsequent grazing may have occurred, such that vegetation height might not reflect conditions at the time of laying/brooding. Wilcoxon signed rank tests were used to determine if vegetation height at nests were significantly different to their surroundings and to more distant paired control points. Wilcoxon rank sum tests were used to compare vegetation heights between nests that hatched and failed, and between nests that failed due to predation, flooding, or trampling by cattle.

## 3.4 Results

### 3.4.1 Temporal and spatial patterns of cattle use

The distribution of cattle dung, considered as a proxy for cattle usage of the site, was highly variable spatially though with high consistency of areas with the greatest and least use between years (Figure 3.6 and 3.7). Most notable was the much higher usage of the inner (landward) marsh compared to the outer marsh, the latter area also being the site of a large gull colony (Chapter 4, Figure 4.2). The central and east inner marsh had a higher density of cattle use than the inner west marsh, with noticeable hotspots in the central inner marsh consistent between 2017 and 2018. Less intense cattle use within these high-density areas occurred in frequently flooded and unvegetated areas (Figures 3.6 and 3.8). Patterns of early season cattle usage from May to July in 2017 and 2018, which overlap the Redshank nesting season, varied more than usage over the whole cattle season from May to October. The inner western marsh area had low cattle use in early 2017 compared to 2018, corresponding to the preceding very intensive duck use during the winter of 2016, something which did not occur in the winter of 2017 (Figure 3.10).

Cattle usage over the whole grazing season, May to October, was strongly positively correlated between 2017 and 2018 when sample point data were compared ($r_s$=0.8) (Table 3.1). Early and late season cattle grazing in 2017 were strongly positively correlated ($r_s$= 0.76) and moderately correlated in 2018 ($r_s$=0.60). Similarly, early season cattle use, from May to July, was strongly correlated in 2017 and 2018 ($r_s$= 0.73). All Spearman's rank correlations were statistically significant at the $p< 0.001$ level. The positive correlations between all the time periods indicate that the spatial and temporal patterns of cattle use were relatively consistent and predictable over the study years.

Cattle use intensity around Redshank nests active during the cattle grazing seasons 2016- 2018, was examined in relation to nest outcome (Figure 3.9). The median cattle use intensity at cattle trampled nests was more than 2.2 times greater than at nests that successfully hatched young and 2.5 times greater than at nests that were predated. These differences are highly statistically significant ($p <0.001$) and indicate that where nesting attempts occurred in areas of high cattle use, they were more likely to be trampled. However, the number of nests trampled by cattle (n=16) was relatively low compared to the number of nests that hatched (n=49) or were predated (n=36). The difference in median cattle use intensity between hatched and predated nests was not statistically significant suggesting that nest predation was not the result of differing cattle use. In addition to the above, two nests were flooded, and one nest was crushed by a vehicle.

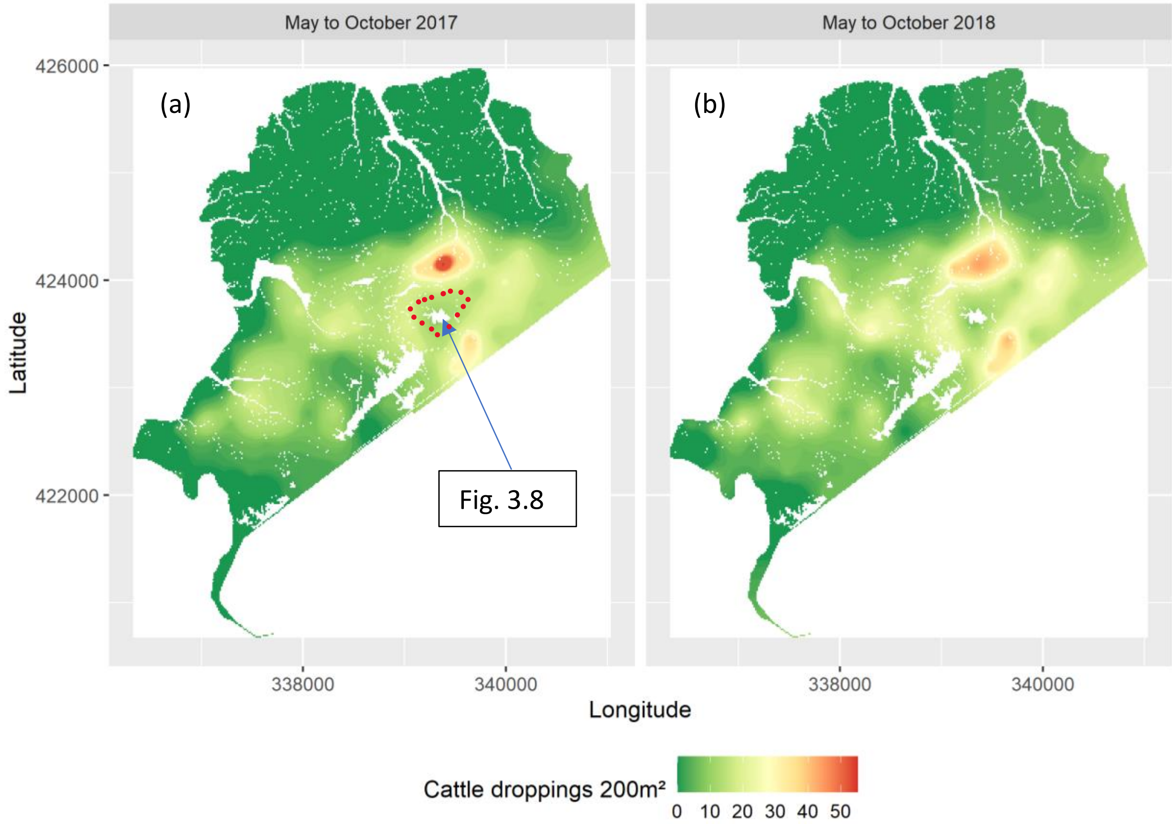



Figure 3.6 Cattle use intensity interpolated from monthly cattle dung counts on Banks Marsh during the whole cattle grazing seasons, (May to October) for (a) 2017 and (b) 2018. The location of the area less used by cattle shown in Figure 3.8 is marked.

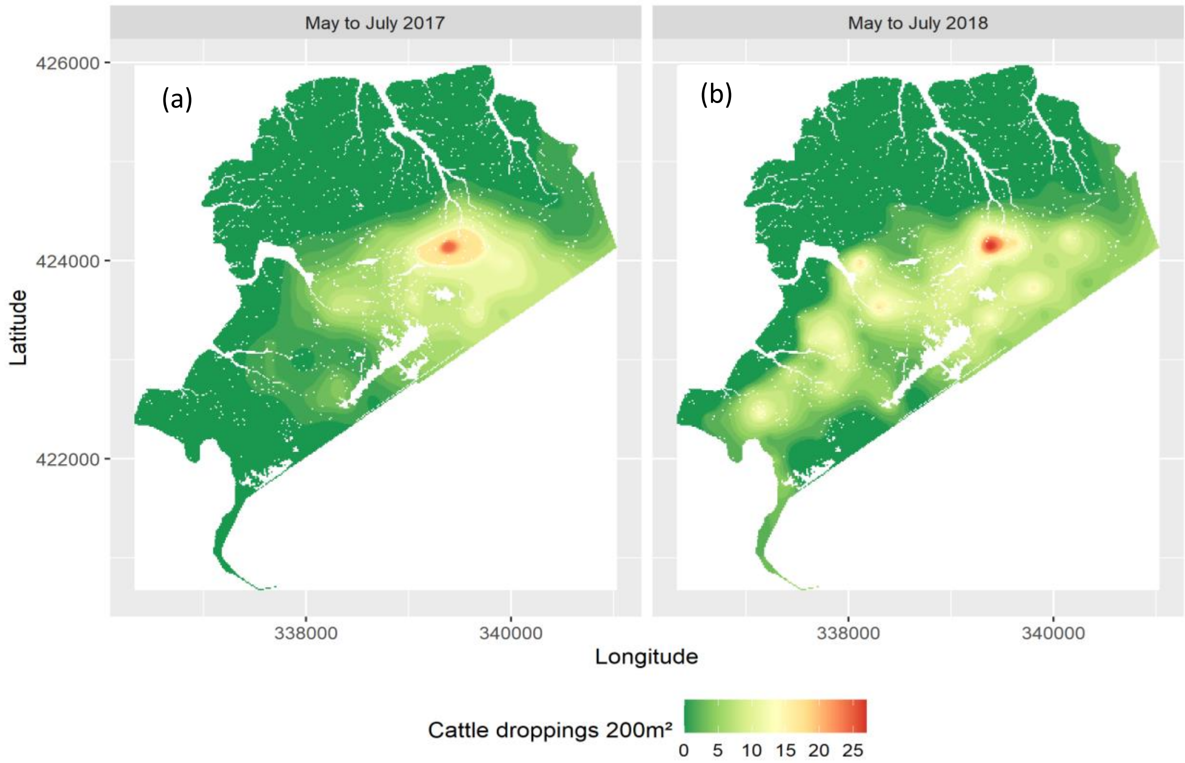



Figure 3.7 Cattle use intensity interpolated from monthly cattle dung counts on Banks Marsh during the Redshank nesting seasons, (May to July) for (a) 2017 and (b) 2018.

Figure 3.8 An example of a less vegetated and partially flooded area of Banks Marsh where lower cattle use was recorded (outlined in red). The location is shown in Figure 3.6 (a) surrounded by more intensive cattle use in areas of better forage.

Table 3.1 Spearman's rank correlation coefficients ($r_s$) for cattle use intensity on Banks Marsh during the cattle grazing periods in 2017 and 2018: Whole season = May to October, Early season = May to July (during the Redshank nesting period), Late season = August to October. Correlation coefficients are calculated from sampled data at fixed points (n=171). All correlations were found to be highly significant at the p< 0.001 level. Mean and standard deviation are calculated from interpolated points (n=51,506).

| | **Mean cowpats (200m²)** | **SD** | **1** | **2** | **3** | **4** | **5** |
|---|---|---|---|---|---|---|---|
| **1.Cattle 17(Whole season)** | 6.33 | 7.92 | | | | | |
| **2.Cattle 18 (Whole season)** | 7.93 | 8.50 | 0.80 | | | | |
| **3.Cattle 17(Early season)** | 2.45 | 3.90 | 0.89 | 0.71 | | | |
| **4.Cattle 18 (Early season)** | 3.34 | 4.03 | 0.81 | 0.85 | 0.73 | | |
| **5.Cattle 17(Late season)** | 3.46 | 4.34 | 0.96 | 0.78 | 0.76 | 0.78 | |
| **6.Cattle 18(Late season)** | 3.93 | 4.08 | 0.70 | 0.92 | 0.60 | 0.63 | 0.69 |

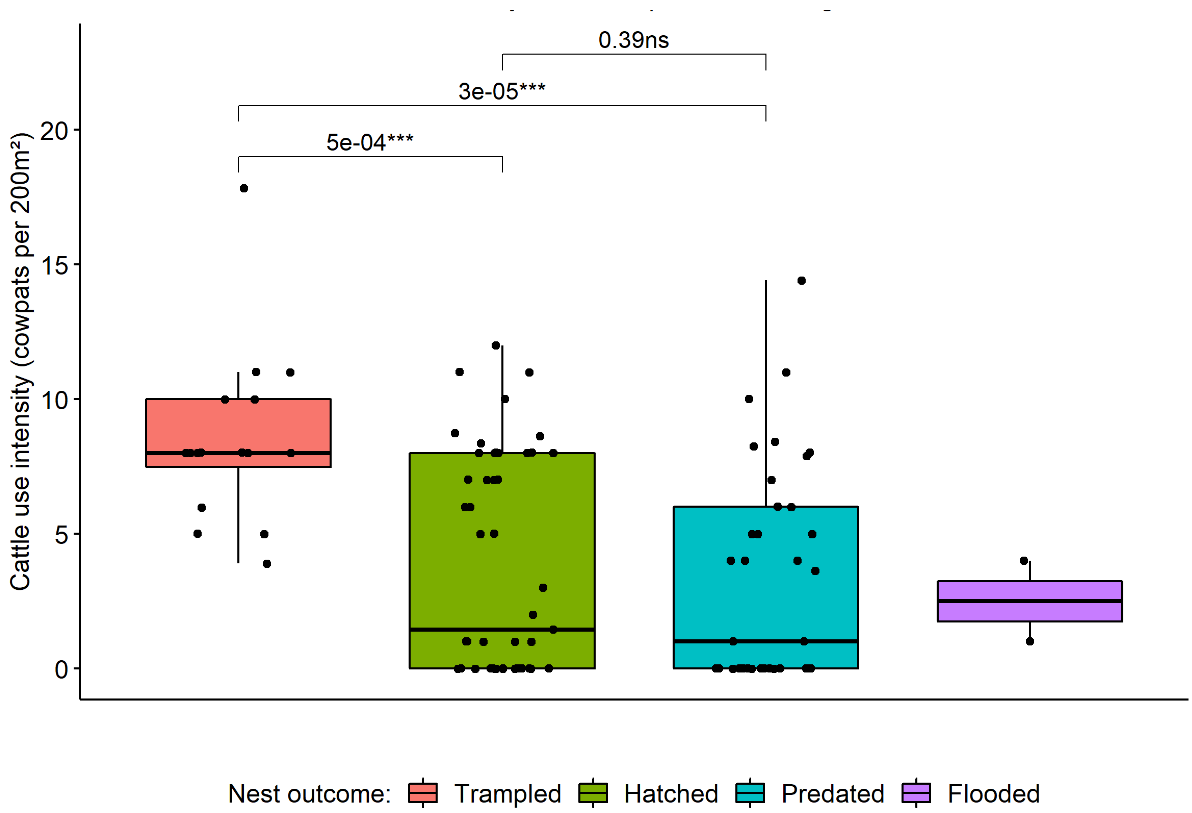



Figure 3.9 Median breeding season cattle use intensity (cowpats per 200m²) around Redshank nest locations, (n=103), in relation to nest outcomes on Banks Marsh 2016-2018. Wilcoxon rank sum test p values and significance levels are shown above outcome group comparisons.

### 3.4.2 Temporal and spatial patterns of wildfowl use

The distribution of wildfowl droppings, considered a proxy for wildfowl herbivory, was highly variable both spatially and between the years.

Duck herbivory

Duck dropping density across the site in the winter of 2016-17 was much higher than in 2017-18 and 2018-19 (Figure 3.10). An area of very high dropping density on the west and central marsh in 2016-17 followed a large influx of Wigeons, which grazed the vegetation in the area very short (Figure 3.11a). Lower duck herbivory occurred consistently in the east marsh area compared to the west and central areas. The spatial distribution of duck droppings was reduced in the winter of 2018-19 compared to 2016-17 and 2017-18. Duck use in the winters of 2016-17 and 2017-18 was positively correlated ($r_s$= 0.76, p<0.001) and moderately positively correlated between 2017-18 and 2018-19 ($r_s$= 0.64, p<0.001) (Table 3.3).

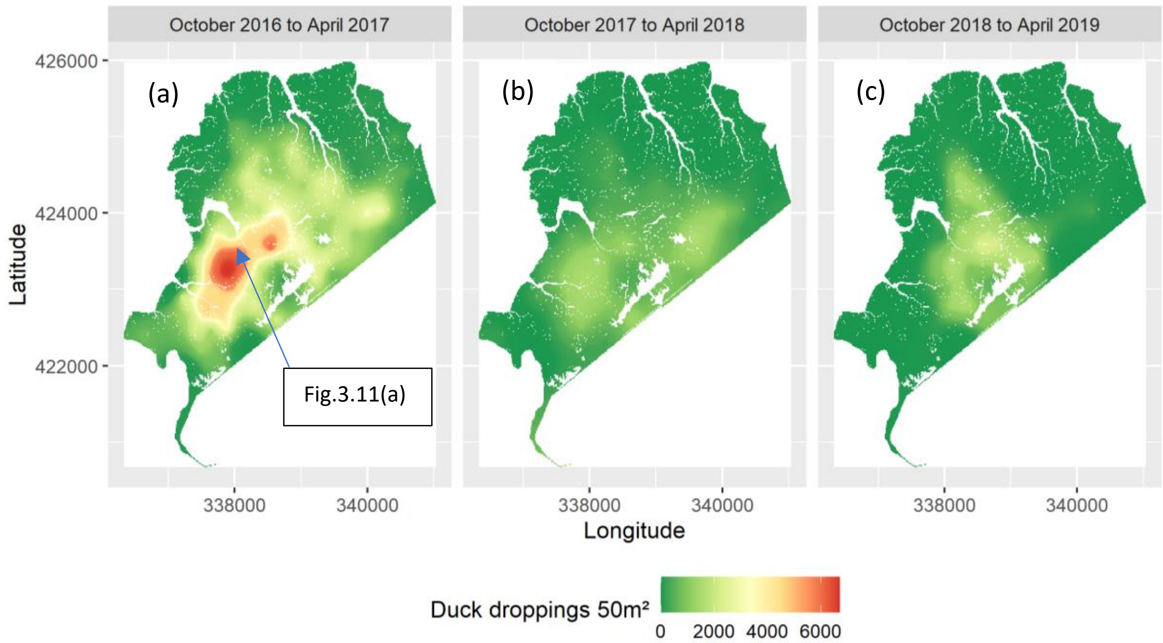



Figure 3.10 Duck use intensity interpolated from monthly dropping counts on Banks Marsh during the winter feeding seasons (October to April) for (a) 2016-17, (b) 2017-2018 and (c) 2018-2019.

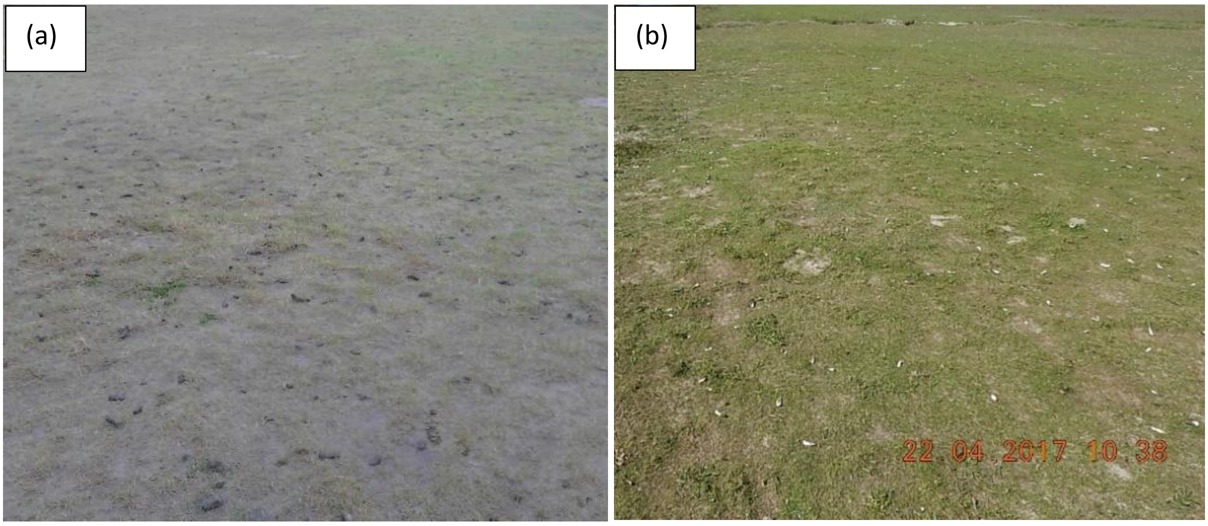



Figure 3.11 Examples of (a) high monthly density of duck droppings, and (b) high monthly density of goose droppings from the winter of 2016-17. The locations where the photographs were taken are shown in Figure 3.10 and Figure 3.12, respectively. Note the very short vegetation associated with intensive wildfowl herbivory.

Goose herbivory

Goose dropping densities occurred at an order of magnitude lower than duck droppings (Figure 3.12). The intensity of goose use in the central inner central marsh was low compared to other areas of the site. The most westerly area of the marsh in 2016-17 and 2017-18 and the outer east marsh in 2018-19 had notably high concentrations of goose droppings. Goose use in the winters of 2016-17 and 2017-18, was found to be moderately positively correlated ($r_s$=0.64 p<0.001) and strongly positively correlated in 2017-18 and 2018-19 ($r_s$=0.83, p<0.001) (Table 3.3)

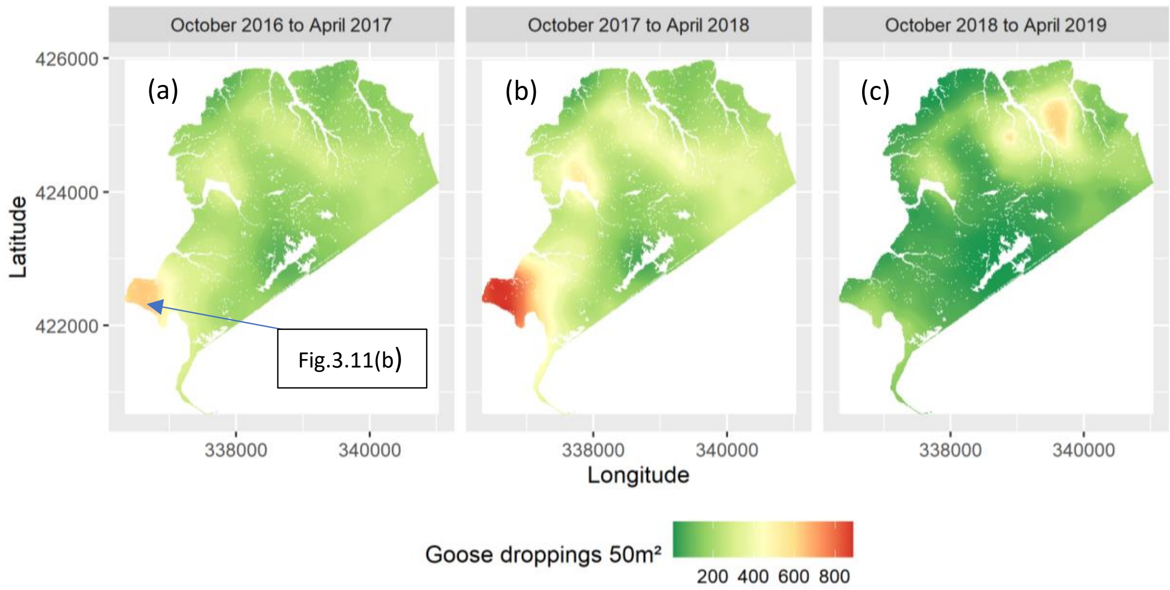



Figure 3.12 Goose use intensity interpolated from monthly dropping counts on Banks Marsh during the winter feeding seasons, (October to April) for (a) 2016-17, (b) 2017-2018 and (c) 2018-2019.

Winter duck and goose use intensity in the same year was found to be either weakly positively or weakly negatively correlated. Duck 2016-17 and Goose 2016-17($r_s$=0.18, p=0.02).
Duck 2018-19 and Goose 2018-19 ($r_s$ = -0.29, p = 0.001) (Table 3.3).

Combined wildfowl herbivory

The combined weighted measure of wildfowl dropping density, duck and goose, revealed widespread use across the majority of the site (Figure. 3.13). The level of wildfowl use varied between years, being lower in the winter of 2018-19 compared to the two previous winters. Combined wildfowl use in the winters of 2016-17 and 2017-18, was moderately positively correlated ($r_s$=0.67 p<0.001) and strongly positively correlated in 2017-18 and 2018-19 ($r_s$=0.77, p<0.001) (Table 3.3).

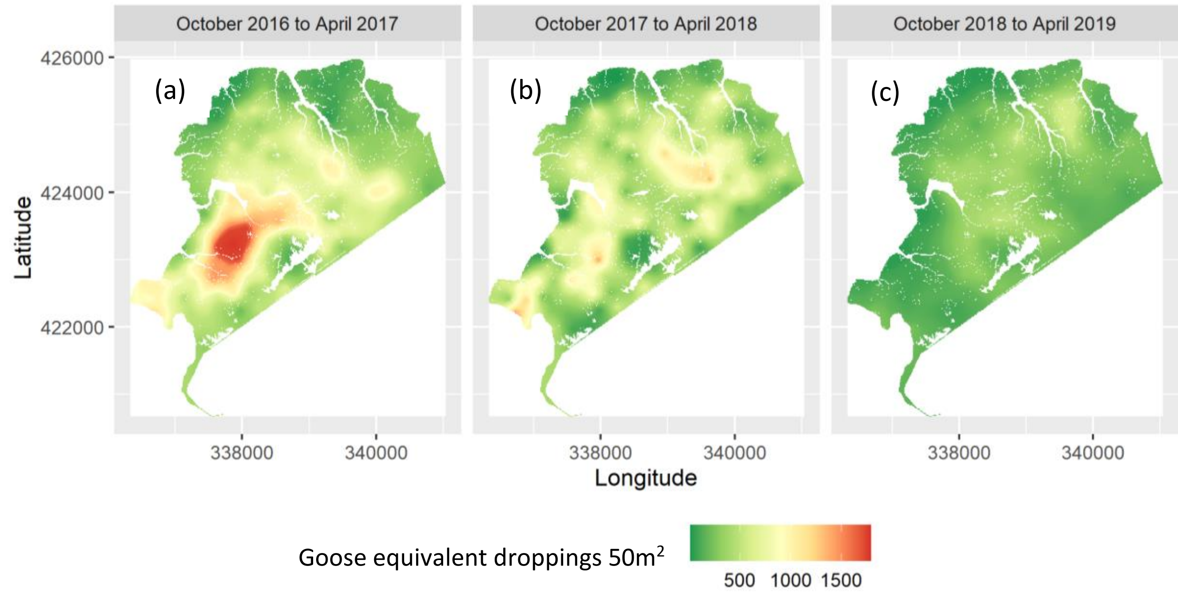



Figure 3.13 Wildfowl use intensity, combined duck and goose weighted as goose equivalents, interpolated from monthly dropping counts on Banks Marsh during the winter feeding seasons, (October to April) for (a) 2016-17, (b) 2017-2018 and (c) 2018-2019.

### 3.4.3 Facilitation of use between cattle, ducks, and geese

Spearman's rank correlation was used to assess if cattle grazing, the main conservation management option for the site, facilitated duck and goose use, a primary objective of the site management. Cattle use intensity in 2017 preceding duck use in 2017-18 was found to be moderately positively correlated $r_s = 0.6$, $p<0.001$ and weakly positively related in the following year $r_s=0.38$, $p<0.001$ (Table 3.4). This result suggests that cattle grazing may be facilitating duck use, but the strength of the relationship is not consistent over two years when cattle livestock numbers were stable. Cattle use intensity preceding goose use was found to be weakly correlated $r_s=0.21$, $p = 0.007$ in 2018-19, and very weakly and non-significantly correlated $r_s=0.09$, $p=0.23$ in 2017-18 (Table 3.4) suggesting no facilitation effect by cattle grazing on goose use.

Table 3.3 (a) Spearman's rank correlation coefficient matrix for duck, goose, and wildfowl (combined duck and goose) use intensity on Banks Marsh during the period October to April 2016-17, 2017-18 and 2018-19. Correlations are based on monthly sampled field data at fixed points (n=171), mean and standard deviation from interpolated points (n=51506), (b) p values for the Spearman's rank correlations.

| (a) | Mean droppings ($50m^2$) | SD | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 |
|---|---|---|---|---|---|---|---|---|---|---|
| 1.Duck 16-17 | 1285 | 1431 | | | | | | | | |
| 2.Duck 17-18 | 369 | 434 | 0.76 | | | | | | | |
| 3.Duck 18-19 | 285 | 508 | 0.64 | 0.70 | | | | | | |
| 4.Goose 16-17 | 224 | 88 | 0.17 | -0.05 | -0.23 | | | | | |
| 5.Goose 17-18 | 288 | 142 | 0.13 | -0.19 | -0.27 | 0.64 | | | | |
| 6.Goose 18-19 | 129 | 116 | 0.08 | -0.18 | -0.29 | 0.71 | 0.83 | | | |
| 7.Wildfowl 16-17 | 494 | 242 | 0.83 | 0.56 | 0.40 | 0.60 | 0.41 | 0.39 | | |
| 8.Wildfowl 17-18 | 494 | 242 | 0.48 | 0.29 | 0.08 | 0.57 | 0.82 | 0.70 | 0.67 | |
| 9.Wildfowl 18-19 | 226 | 117 | 0.38 | 0.09 | 0.12 | 0.59 | 0.73 | 0.83 | 0.61 | 0.77 |

| (b) | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 |
|---|---|---|---|---|---|---|---|---|
| 1.Duck 16-17 | | | | | | | | |
| 2.Duck 17-18 | < 0.001 | | | | | | | |
| 3.Duck 18-19 | < 0.001 | < 0.001 | | | | | | |
| 4.Goose 16-17 | 0.03 | 0.54 | 0.003 | | | | | |
| 5.Goose 17-18 | 0.09 | 0.01 | 0.005 | < 0.001 | | | | |
| 6.Goose 18-19 | 0.28 | 0.02 | 0.001 | < 0.001 | < 0.001 | | | |
| 7.Wildfowl 16-17 | < 0.001 | < 0.001 | < 0.001 | < 0.001 | < 0.001 | < 0.001 | | |
| 8.Wildfowl 17-18 | < 0.001 | 0.001 | 0.311 | < 0.001 | < 0.001 | < 0.001 | < 0.001 | |
| 9.Wildfowl 18-19 | < 0.001 | 0.23 | 0.13 | < 0.001 | < 0.001 | < 0.001 | < 0.001 | < 0.001 |

Table 3.4 Spearman's rank correlation coefficients for cattle use intensity (May to October) preceding duck and goose use (October to April). Correlations are based on monthly sampled field data at fixed points (n=171).

| | **Duck 17-18** | **Duck 18-19** | **Goose 17-18** | **Goose 18-19** |
|---|---|---|---|---|
| **Cattle 17 (Whole season)** | $r_s$= 0.6 $p$< 0.001 | | $r_s$= 0.09 $p$= 0.23 | |
| **Cattle 18 (Whole season)** | | $r_s$= 0.38 $p$ <0.001 | | $r_s$= 0.21 $p$< 0.007 |

### 3.4.4 The impact of herbivory exclusion on saltmarsh vegetation height

Winter wildfowl exclusion

Excluding wildfowl for part of the winter or the full winter season resulted in highly significantly taller saltmarsh vegetation compared to areas subject to wildfowl herbivory. Wildfowl exclusion from February 2017 to May 2017 resulted in a vegetation height more than twice the height of the control treatments (Median = 10.9cm, SD = 4.25, Median = 4.73cm, SD = 2.05) (Figure 3.14). Following exclusion over the full winter wildfowl season, October to May 2017-18 and 2018-19 (Figure 3.15 and 3.16) median vegetation heights in wildfowl excluded plots were approximately 3 times greater than the wildfowl grazed control plots (Median = 15.45cm, SD = 2.54 compared to Median = 5.28cm, SD = 2.25, May 2018, Median = 21.05cm, SD = 1.81 compared to Median = 6.65cm, SD = 3.14, May 2019). These results demonstrate conclusively that winter wildfowl herbivory results in highly significantly shorter saltmarsh vegetation across the site in all years of this study.

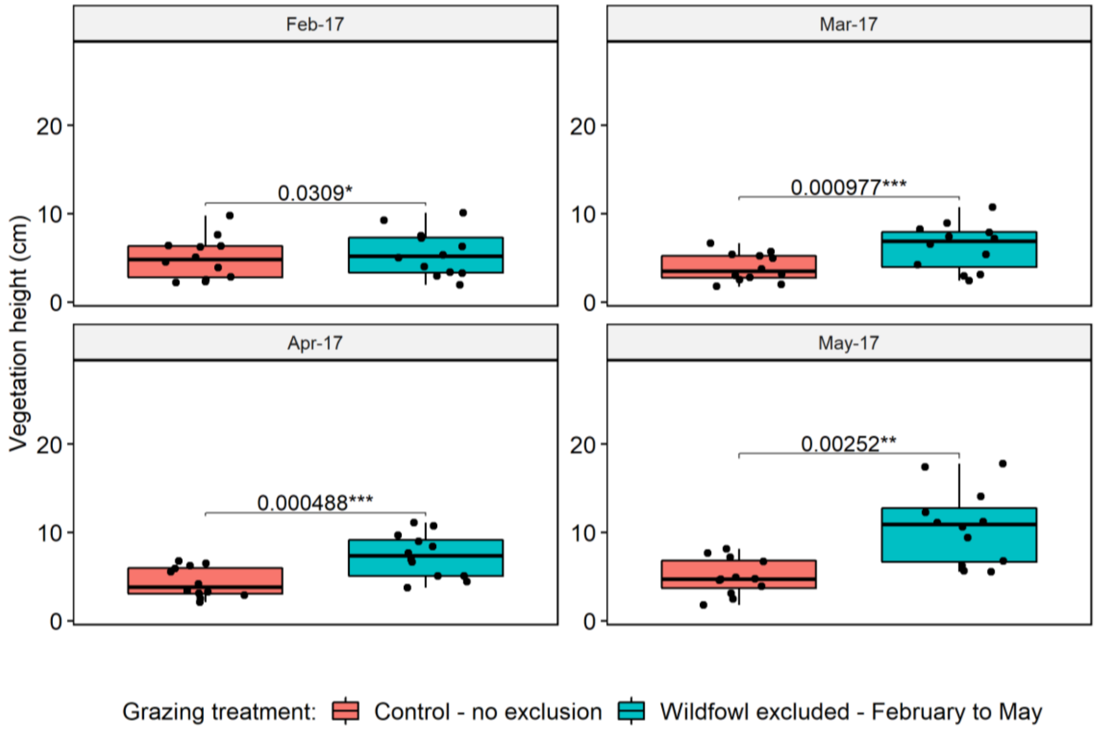



Figure 3.14 Monthly vegetation heights for control treatments (no exclusion), paired with wildfowl excluded treatments (n=12) during the late winter wildfowl feeding period, February 2017 to May 2017. Wilcoxon signed rank test p values and significance levels are shown above pairwise comparisons.

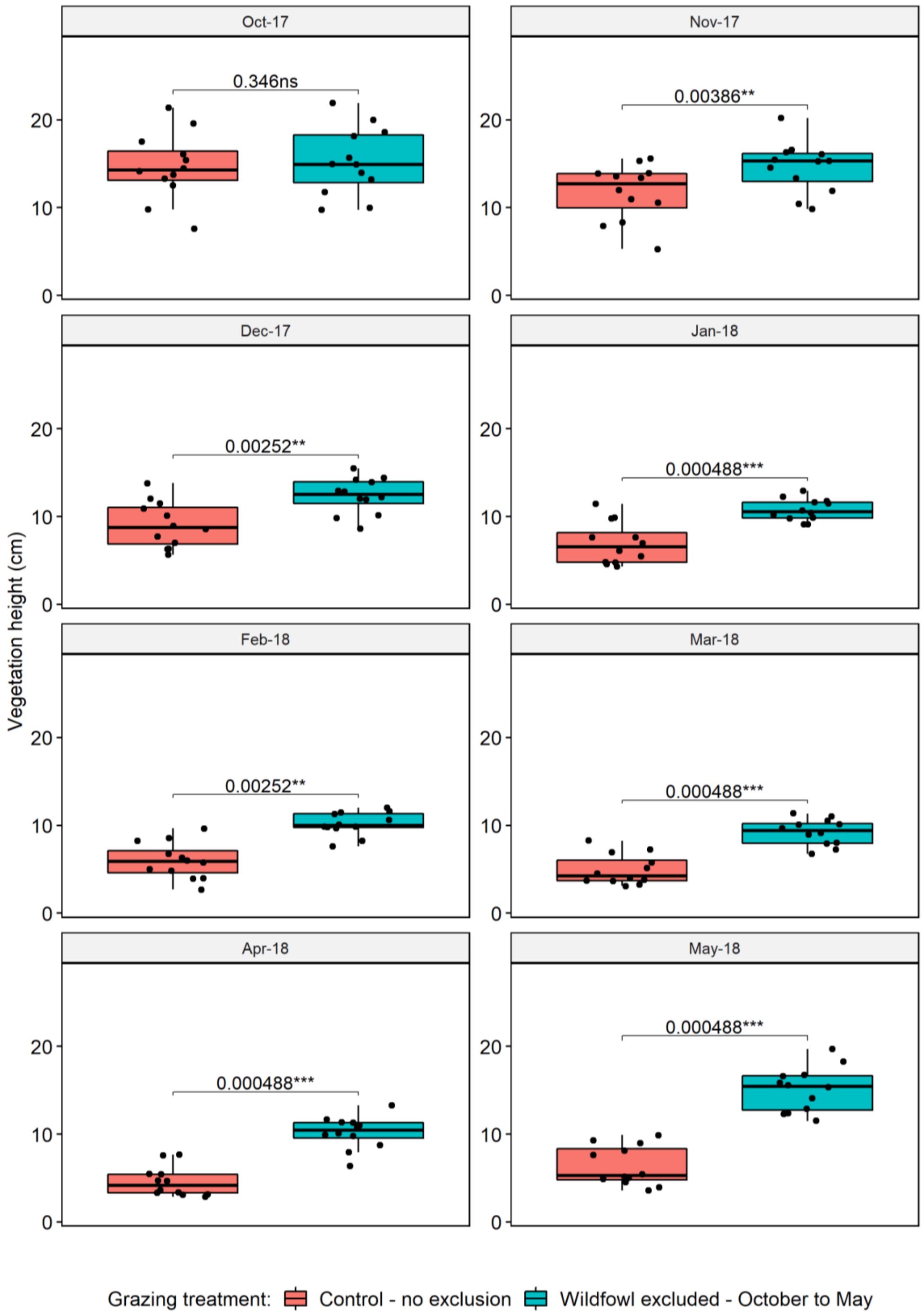



Figure 3.15 Monthly vegetation heights for control treatments (no exclusion), paired with wildfowl excluded treatments (n=12) during the wildfowl feeding period, October 2017 to May 2018. Wilcoxon signed rank test p values and significance levels are shown above pairwise comparisons.

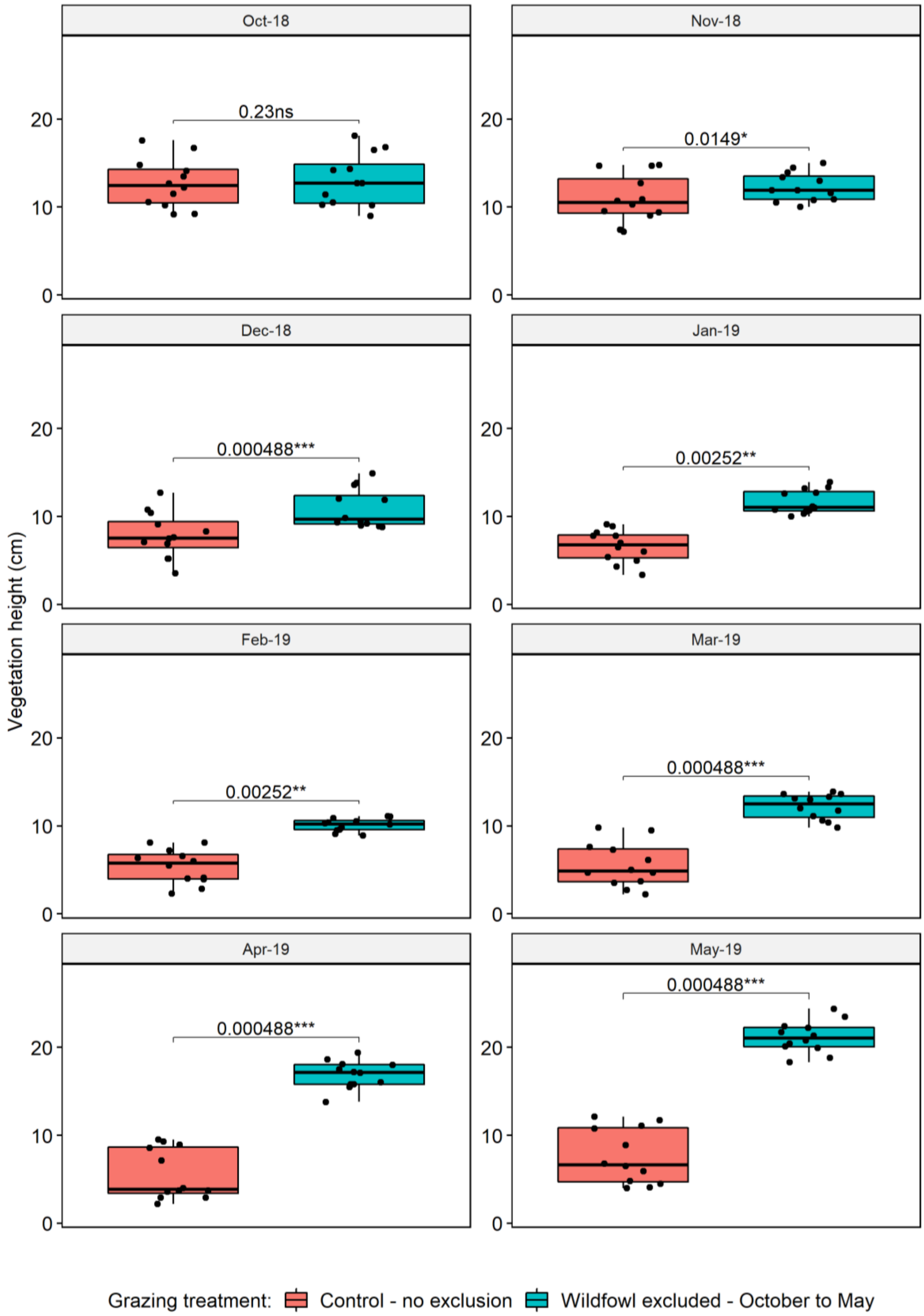



Figure 3.16 Monthly vegetation heights for control treatments (no exclusion), paired with wildfowl excluded treatments (n=12) during the wildfowl feeding period, October 2018 to May 2019. Wilcoxon signed rank test p values and significance levels are shown above pairwise comparisons.

Summer cattle exclusion

During the summer cattle grazing season May to October, the vegetation height in the cattle excluded treatment plots is significantly taller than the control treatment plots after one month in both 2017 and 2018 (Figures 3.17 and 3.18). By October the median vegetation height in the cattle excluded plots was 25% and 30% taller than in control plots in 2017 and 2018 respectively (median = 19cm, SD = 2.34cm compared to median = 14.3cm, SD = 3.84cm October 2017, and median = 17.9cm SD = 1.46cm compared to median = 12.45cm, SD = 2.76cm, October 2018).

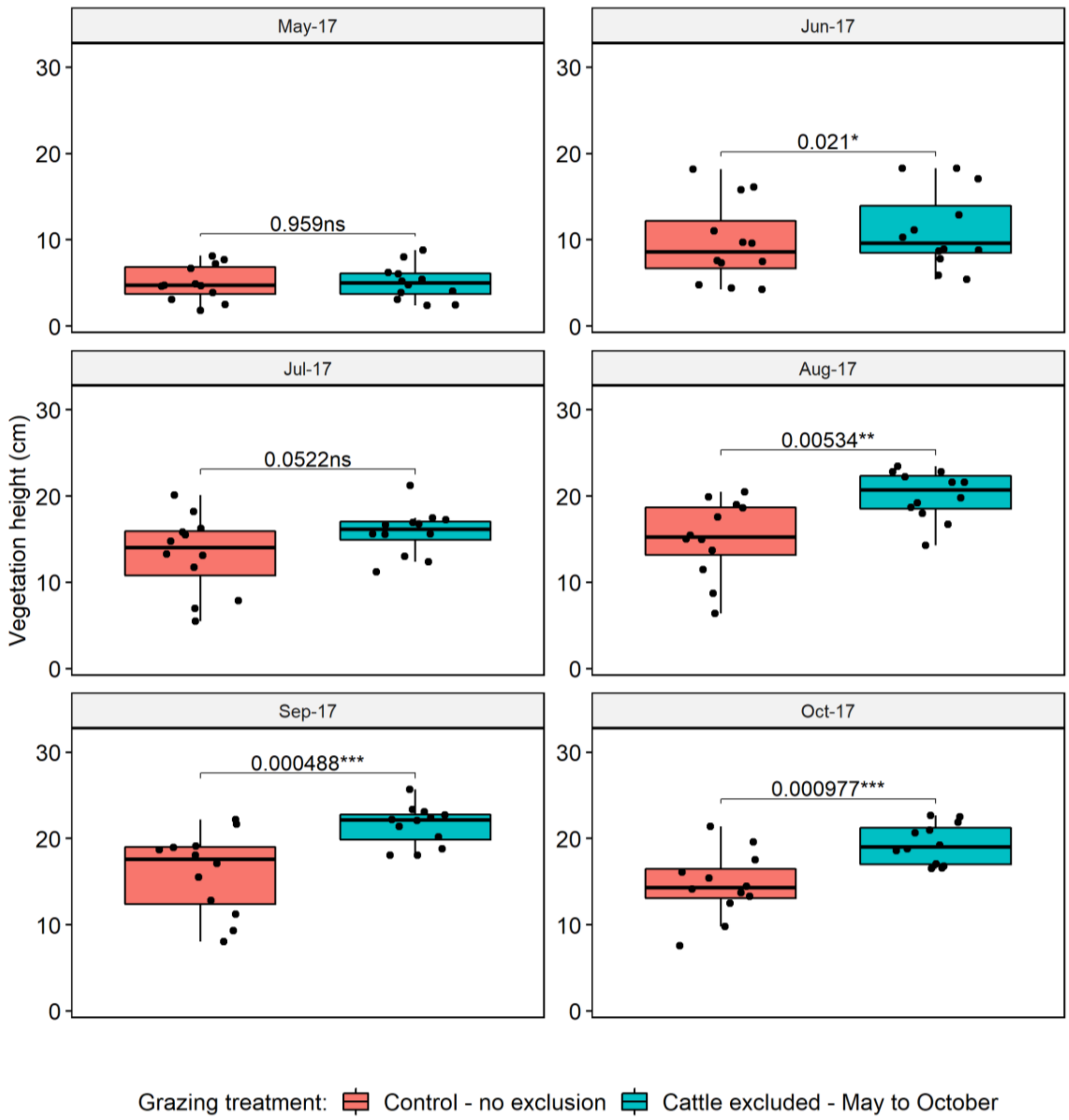



Figure 3.17 Monthly vegetation heights for control treatment plots, no exclusion, paired with cattle excluded treatment plots, (n=12) during the summer cattle grazing season, May to October 2017 on Banks Marsh. Wilcoxon signed rank test p values and significance levels are shown above pairwise comparisons.

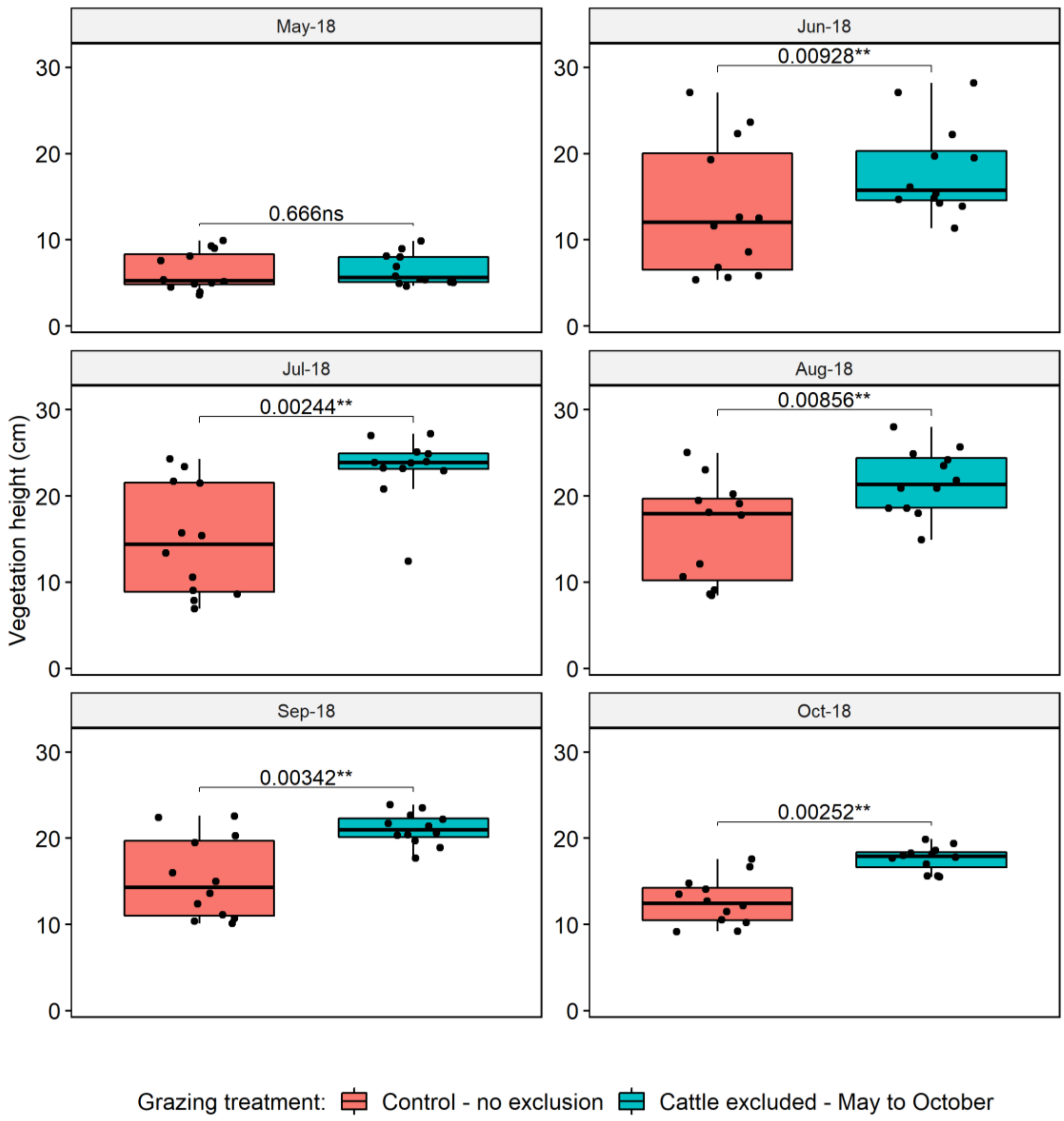



Figure 3.18 Monthly vegetation heights for control treatment plots, no exclusion, paired with cattle excluded treatment plots, (n=12) during the summer cattle grazing season, May to October 2018 on Banks Marsh. Wilcoxon signed rank test p values and significance levels are shown above pairwise comparisons.

### 3.4.5 Herbivory exclusion during the Redshank nesting season

The effects of all exclusion treatments, including combined wildfowl and cattle exclusion, on saltmarsh vegetation height during the Redshank breeding period, March to July 2017 and 2018 are shown in Figures 3.19 and 3.20, and for March to May 2019 in Figure 3.21. Notably, there is no significant difference in median vegetation height between the control treatments and the cattle excluded plots between March and May in all years, demonstrating that the effect of preceding summer cattle exclusion is not maintained through the winter wildfowl feeding period and into the following Redshank breeding season. Vegetation height in the wildfowl excluded plots is significantly higher than in the controls up to May, which is the peak of the Redshank nesting, and this height difference is maintained after the exclosure is removed and cattle can graze the plots. No consistent additive effect of excluding both wildfowl and cattle herbivory compared to only seasonally excluding wildfowl was recorded, with non-significant differences between these treatments in 8 of 10 months. These results provide compelling evidence that wildfowl herbivory plays a crucial role in determining saltmarsh vegetation heights relevant for Redshank nesting.

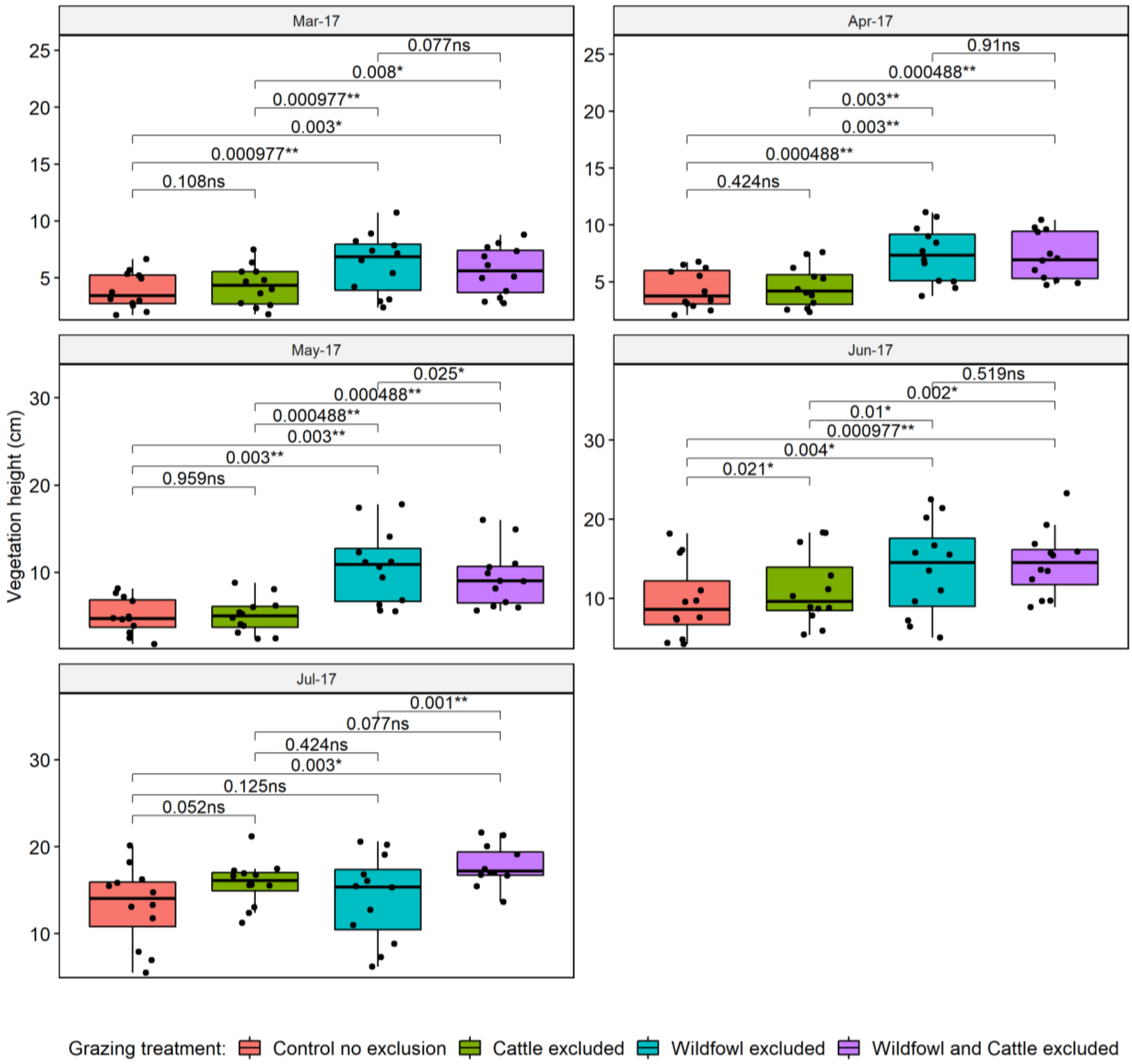



Figure 3.19 Boxplots of monthly vegetation heights (n=12) for all grazing treatments (i) control - no exclusion (ii) cattle exclusion May to October (iii) wildfowl exclusion February to May (iv) wildfowl and cattle exclusion February 2017 onwards, during the Redshank breeding period March to July 2017. Wilcoxon signed ranked test p values and significance levels are shown for pairwise comparisons.

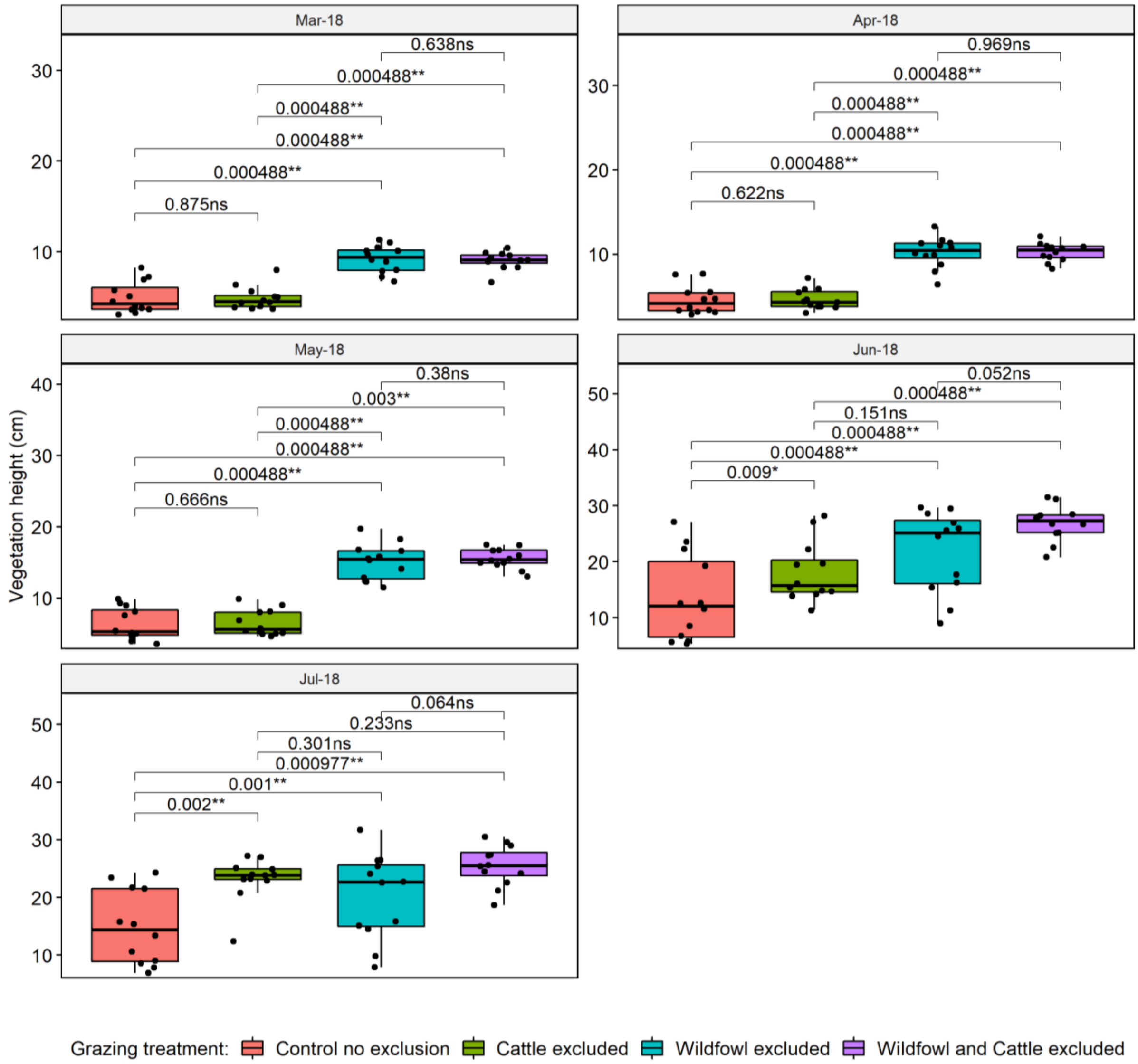



Figure 3.20 Boxplots of monthly vegetation heights (n=12) for all grazing treatments (i) control - no exclusion (ii) cattle exclusion May to October (iii) wildfowl exclusion February to May (iv) wildfowl and cattle exclusion February 2017 onwards, during the Redshank breeding period March to July 2018. Wilcoxon signed ranked test p values and significance levels are shown for pairwise comparisons.

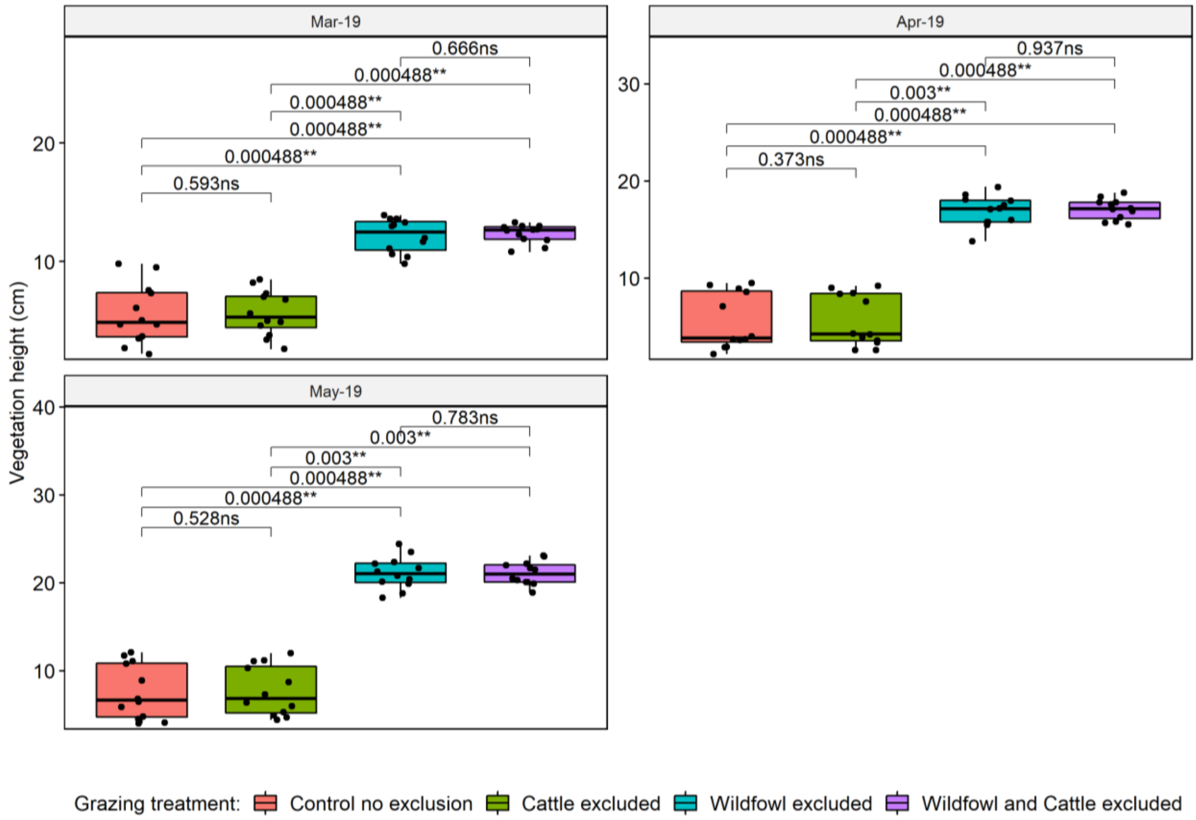



Figure 3.21 Boxplots of monthly vegetation heights (n=12) for all grazing treatments (i) control - no exclusion (ii) cattle exclusion May to October (iii) wildfowl exclusion October to May (iv) wildfowl and cattle exclusion February 2017 onwards, during the Redshank breeding period March to May 2019 on Banks Marsh. Wilcoxon signed ranked test p values and significance levels are shown for pairwise comparisons.

### 3.4.6 Saltmarsh vegetation heights at and around Redshank nests

Redshank nests were in vegetation significantly taller than their surroundings (Figure 3.22). The median vegetation height at Redshank nests (median = 18.3cm, SD = 4.89) was significantly taller than vegetation sampled at <1m from the nest (median = 12cm, SD = 3.33). At the greater distances of <10m (median = 9.7cm, SD = 3.28), and 80-100m (median = 8.7cm, SD = 3.89) the median vegetation height was approximately 50% of the nest vegetation height. These vegetation height differences were highly significant at the $p< 0.001$ level, strongly suggesting that nest site selection occurs in the tallest vegetation available in an area. Examples of contrasting short and long vegetation heights at nests are shown in Figure 3.23.

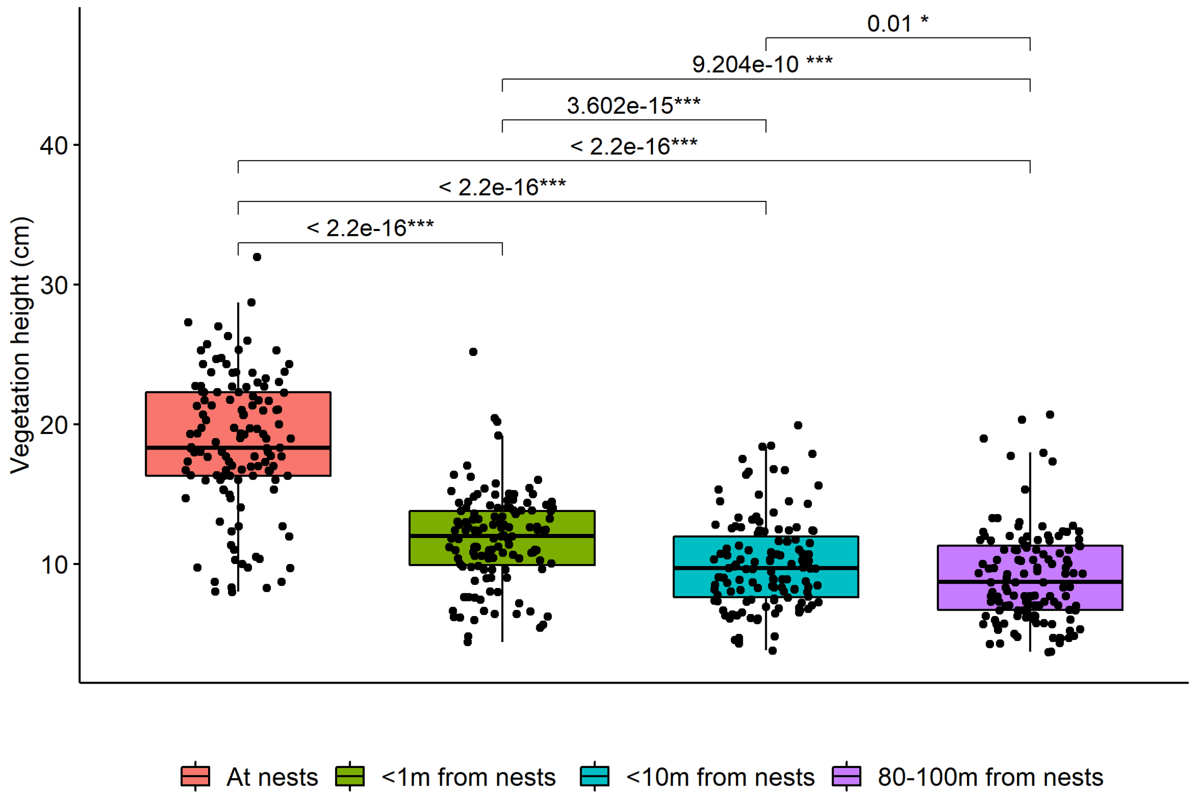
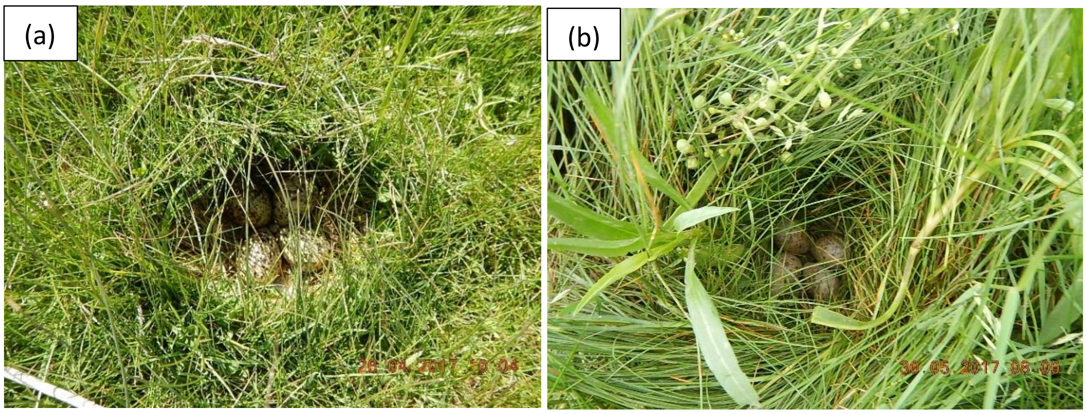



Figure 3.22 Vegetation height at Redshank nests (n=127), within 1m and 10m radius and at random control points 80-100m, on Banks Marsh 2016-18. P values and significance levels (Wilcoxon signed rank test) are shown above pairwise comparisons.




Figures 3.23 Contrasting examples of vegetation heights at Redshank nests on Banks Marsh in 2017. (a) Nest vegetation height is 8.3cm, and 6.7cm <10m from the nest. (b) Nest vegetation height is 22.7cm, and 9.7cm <10m from the nest.

### 3.4.7 Saltmarsh vegetation heights at and around Redshank nests that succeed or fail

The median vegetation height at Redshank nests where eggs successfully hatched (median=21.7cm, SD =5.43, n=39) was found to be significantly taller than at all failed nests (median = 17.7cm, SD = 4.51, n = 88), Figure 3.24(a). The difference in vegetation height at hatched nests and predated nests (n = 52, median = 18.85cm, SD = 3.97) was found not to be statistically significant. In contrast, the vegetation height at nests that were trampled by cattle (n = 16, median = 11.4cm, SD = 3.24) was significantly shorter than at both hatched and predated nests.

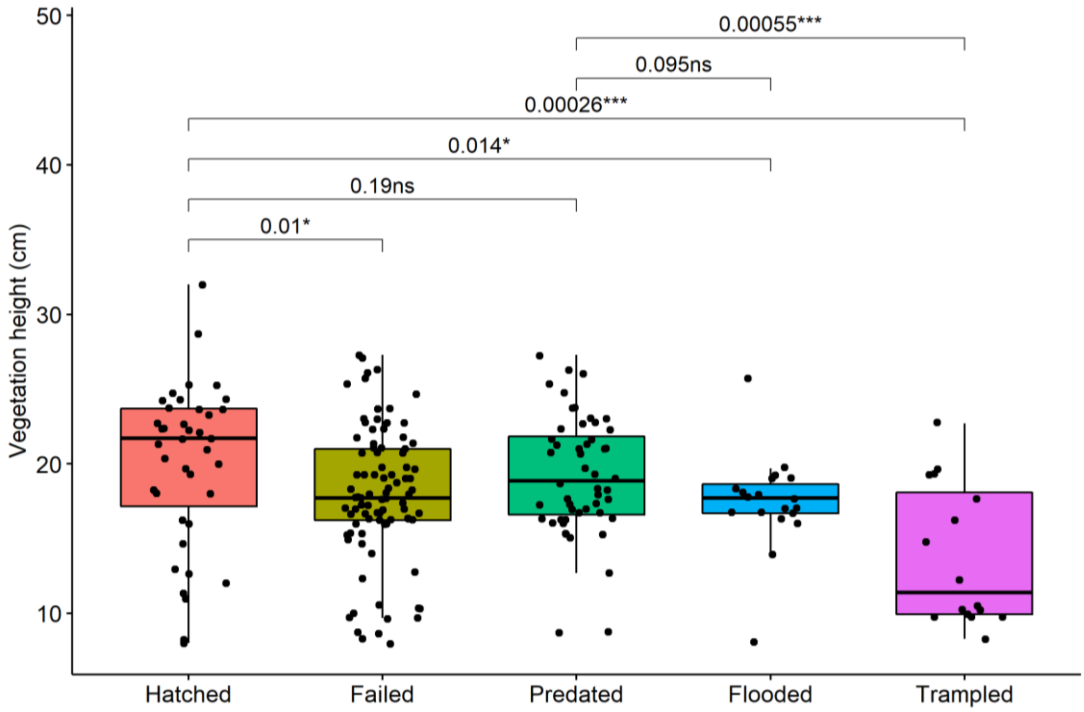



Figure 3.24(a) Vegetation heights at Redshank nests (n=127), grouped as hatched or failed and separately by causes of failure, during the 2016 to 2018 breeding seasons on Banks Marsh. Wilcoxon rank sum test p values and significance levels are shown.

A similar pattern occurs at <1m from Redshank nests with the vegetation height at hatched nests (median = 12.8cm, SD = 4.28) being significantly taller than at all failed nests (median = 11.3cm, SD = 2.66, n=88), Figure 3.24(b). No significant differences in vegetation height at <1m distance from hatched and predated nests were recorded. However, the vegetation height at <1m from cattle trampled nests (median = 8.3cm, SD = 2.74) was a significantly shorter $p < 0.01$ than both hatched and predated nests.

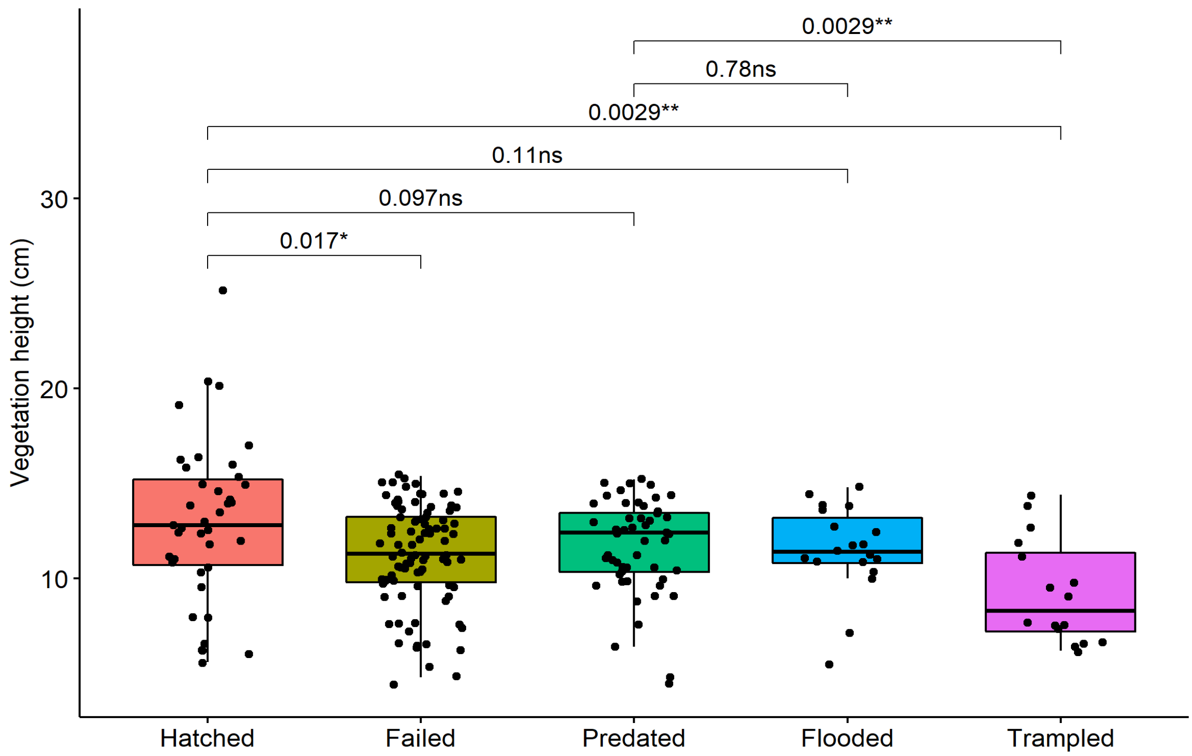



Figure 3.24(b) Median vegetation heights <1m from Redshank nests (n=127), grouped as hatched or failed and separately by causes of failure, during the 2016 to 2018 breeding seasons on Banks Marsh. Wilcoxon rank sum test p values and significance levels are shown.

At <10m from nests the difference in vegetation height between hatched nests (median = 10.3cm, SD = 4.04) and failed nests (median = 9.92cm, SD = 2.76) was approaching significance at the p = 0.05 level, Figure 3.24(c). The vegetation height <10m from nests trampled by cattle (median = 7cm, SD = 2.13) was significantly shorter than for <10m from hatched and predated nests (both p<0.01).

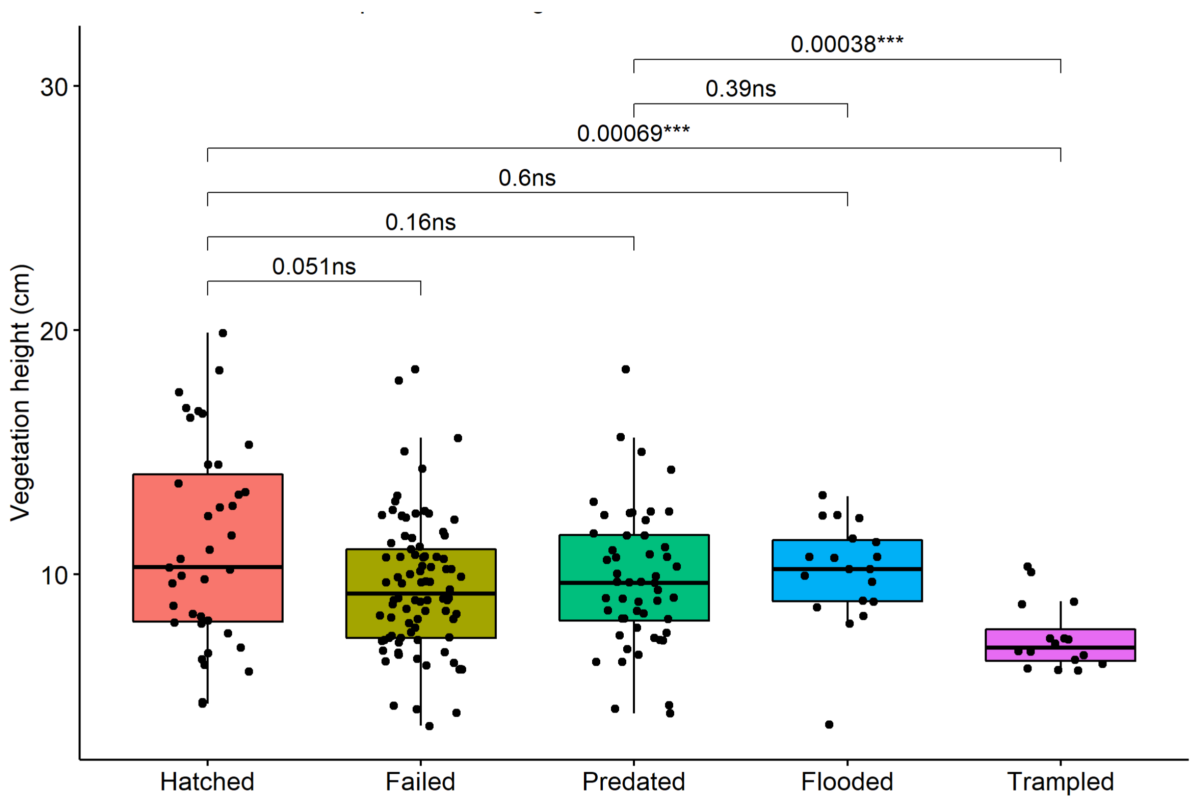



Figure 3.24(c) Median vegetation heights <10m from Redshank nests (n=127), grouped as hatched or failed and separately by causes of failure, during the 2016 to 2018 breeding seasons on Banks Marsh. Wilcoxon rank sum test p values and significance levels are shown.

## 3.5 Discussion

An understanding of the temporal and spatial use of wildfowl and livestock on saltmarshes is critical to identifying how these grazers modify the saltmarsh habitat and potentially impact breeding Redshank. This information is important for managing populations of breeding Redshank and other wader species of conservation concern with their differing nesting sward height preferences e.g. 15-20cm for Redshank and 2-3 cm for Northern Lapwing, *Vanellus vanellus,* (Durant *et al.*, 2008; Milsom *et al.*, 2000).

Previous studies have suggested that marked declines in saltmarsh breeding Redshank are driven by a lack of suitable nesting habitat as a result of high livestock grazing pressure, this being particularly the case in North West England (Malpas et al., 2011; Malpas et al., 2013). Further, it has been shown that even low intensity conservation livestock grazing results in low nest survival (Sharps et al., 2015). However, measures of livestock grazing based on infrequent direct observations, assumptions of grazing homogeneity, or a rapid visual assessment of sward condition may not adequately reflect impacts on saltmarsh vegetation. Fundamentally, such measures ignore the additive impacts of high and increasing levels of wildfowl herbivory on saltmarsh habitats (Frost *et al.*, 2019). The research presented in this chapter identifies how a detailed understanding of both wildfowl and livestock herbivory can be achieved, with limited resources, to address this significant gap in our understanding of grazing impacts on saltmarshes.

### 3.5.1 Cattle herbivory patterns and impact on sward height

The temporal and spatial distribution of cattle is important for breeding Redshank as overgrazing may create unsuitable sward conditions and result in nest trampling. The distribution of livestock can vary markedly depending on multiple factors including forage quality and quantity, the positions of drinking water and the availability of shelter (Putfarken *et al.*, 2008). The distribution of cattle on Banks Marsh was highly consistent between study years, with livestock utilising approximately two-thirds of the marsh area and grazing up to 1.5km from the landward edge of the marsh within the first month of access. Grazing in the remaining one third of the saltmarsh is apparently restricted by the presence of an extensive nesting gull colony, which mob and defecate on approaching cattle. Sharps *et al.*,(2017), reporting on livestock movements around the Wash estuary, on the East coast of England, observed a contrasting scenario where livestock stay close to the landward edge of saltmarshes, until the later stages of the livestock grazing period. A plausible explanation for the greater distance travelled on my study site is the limited forage available in the early part of the livestock grazing period created by intensive winter wildfowl feeding. Cattle require a minimum vegetation height of 5-6 cm for feeding (Tolhurst and Oates, 2001) and the average vegetation height across the study site in May between 2016-19 was at or below this threshold following the wildfowl feeding period, thus requiring livestock to travel greater distances to feed.

Over the cattle grazing season from May to October on Banks Marsh the average reduction in saltmarsh vegetation sward height between cattle grazed plots and cattle excluded plots was only 25% in 2017 and 30% in 2018. These results indicate a light cattle grazing intensity with variable sward heights and the retention of vegetation standing crop compared to a heavy livestock grazing regime where all the standing crop is removed and sward height reduced to <10cm (JNCC, 2004). Heavy livestock grazing pressure was recorded for Banks Marsh in both the 1996 and 2011 British surveys of breeding redshank on saltmarshes (Brindley *et al.*, 1998; Malpas *et al.*, 2011) which didn't consider the impact of wildfowl herbivory on sward height.

### 3.5.2 Wildfowl herbivory and impact on sward height

Wildfowl provide a key functional role in ecosystems including as herbivores (Green and Elmberg, 2014). The dominant effect of wildfowl herbivory on saltmarsh vegetation height at Banks Marsh was demonstrated through their widespread occurrence and the measured impact of wildfowl exclusion. Murdock *et al.* (2017) show similar patterns of wildfowl use at a coarser scale using peak count data from the Wetland Bird Survey (WeBS) for Banks Marsh. The variation in the intensity of wildfowl use between winters reflects the population level present, which has increased dramatically over the past 50 years, Figure 3.2. This contrasts with stable or declining numbers of livestock grazing on the site.

The dropping counts approach chosen to measure cattle and wildfowl use provides a simple, tested, and practical approach that can be widely adopted elsewhere. However, it does require a sufficient understanding of local tidal regimes, dropping decomposition rates, and careful assessment of factors such as the position of water sources which can potentially introduce bias into estimates of use and herbivory. Alternative approaches such as GPS tracking of cattle and wildfowl can provide an alternative method for measuring the movement and saltmarsh use by individuals, but there are considerable practical and resource challenges in scaling up the measurement to the population levels found on extensive saltmarshes such as Banks Marsh.

My experimental exclusion experiment revealed that the average vegetation height at the peak of the Redshank breeding season is one third as long as it would be without winter wildfowl feeding. However, this intensive wildfowl grazing may not necessarily result in declines in breeding Redshank. Madsen *et al*. (2019) working on freshwater polder grasslands on the island of Mandø, Denmark detected no negative effect of very intensive goose grazing on nesting wader occupancy. Vickery et *al.* (1997) in a study on coastal grazing marshes, in North Norfolk, United Kingdom, concluded that potential conflicts between intensive wildfowl grazing and breeding wader requirements may be relatively easily resolved by managing areas close to wildfowl roosts for wildfowl and other fields specifically for breeding waders. A clear conflict between the requirements of wildfowl and breeding Redshank was identified by Lambert (2000) specifically in relation to the Banks Marsh study site, which now routinely supports the highest density of feeding Wigeon in Britain and increasing numbers of staging geese, but has seen a spatial shift and decline in its breeding Redshank population. Wildfowl grazing on Banks Marsh may represent an example of severe overgrazing recognised by Cadwalladr *et al.* (1972) and Jefferies & Rockwell (2002) with the resulting severe loss of the standing crop and changes in soil conditions. The decision to manage a particular area for breeding Redshank or wintering wildfowl may have to be made based on current conservation priorities. With wintering wildfowl populations substantially increasing and breeding Redshank declining, the latter could be considered a higher conservation priority, and current site management may require reassessment. Reducing the intensity of wildfowl herbivory where it markedly limits the height of vegetation in the Redshank breeding season is likely to have the beneficial effects, increasing suitable Redshank nesting habitat. This scenario is explored through modelling in Chapter 4.

The experimental exclusion methods used in my study were effective in preventing wildfowl and cattle grazing of representative experimental plots, but these are still limited in scale compared to the size of the site. Some caution in interpreting the vegetation height in treatment plots is therefore required. In addition, Mandema *et al.* (2014) report that winter flooding reduces saltmarsh canopy height through flattening but to a lesser extent than grazing. With monthly spring tides covering Banks Marsh, such an effect was considered constant between plots. The winter decrease in vegetation height in wildfowl excluded plots are similar to those recorded by Kleijn & Bos (2009) at 0.6cm per month, suggesting consistency between the studies. Grazing of unfenced plots by Brown Hares, *Lepus europaeus* and Rabbits, *Oryctolagus cuniculus*, could not be practically prevented but their impact on sward height was assessed as minor compared to wildfowl and cattle based on preliminary observations of their low abundance and limited distribution on the intertidal saltmarsh. The impact of invertebrate herbivory on all treatments could not be practically controlled or differentiated in my experiment.

### 3.5.3 Herbivory facilitation

Livestock grazing of saltmarshes is a widely accepted management practice shown to facilitate feeding by wild ducks and geese (Bos et al., 2005), and to maintain a favourable vegetation structure for Redshank breeding (Norris *et al.*, 1997). The moderate to weak positive correlation between areas used by cattle and subsequently by ducks indicates that cattle grazing may be facilitating duck use by creating shorter swards of nutritious younger shoots. Wigeons, with their short bills, feed repeatedly in these areas throughout the winter (Mayhew & Houston, 1999). A relatively low sward height has been suggested as an important determinant of the distribution of geese on saltmarshes (van der Graaf *et al.*, 2002). Livestock grazing facilitating goose feeding is less evident from my results on Banks Marsh, with only a weak positive correlation, and Mandema *et al*. (2014) found that the distribution of spring staging geese was not affected by livestock grazing treatments on saltmarshes of the Netherlands Wadden Sea. Weak positive correlations between goose and cattle use and weak negative correlations between goose and duck use may indicate the preference of geese for longer vegetation. Goose feeding patterns in spring may be driven by resource depletion in intensively duck grazed areas. Interestingly, localised grubbing out of rhizomes by geese was observed following overgrazing by ducks on Banks Marsh in the winter of 2016-17. Intensity of wildfowl use is also influenced by a range of biotic and abiotic factors other than livestock grazing management including substantial natural population growth and the creation of wildfowl refuges, the latter providing uninterrupted feeding opportunities (Hirons and Thomas, 1993).

### 3.5.4 Vegetation characteristics where Redshank nest

I found that Redshank selected nest sites in taller vegetation, presumably to conceal the nest. By contrast, the surrounding area typically comprised shorter vegetation, possibly to assist predator detection. These findings are similar to those of Smart (2006) in a study of inland and coastal nesting Redshank in East Anglia. The average height of nest vegetation on Banks Marsh (18.3 ± 4.89cm) was comparable to preferred grass heights in the East Anglian study (17·9 ± 6·6cm inland, and 14·1 ± 3·8cm, coastal). They are, however, markedly different from an earlier study on Banks Marsh where the average nest vegetation height was 11 ± 7cm (Sharps, 2015), though the latter was based on a low sample size relative to the previous two studies. Globally, a nesting sward height of ca. 15–20cm seems suitable for Redshank (Durant *et al.*, 2008). Interestingly, Smart *et al*. (2006) reported no difference in the daily survival rates of nests between those found in shorter and longer vegetation and those in inland and coastal habitats. By contrast, I found nests that hatched were in significantly taller vegetation than nests that failed.

### 3.5.5 Trampling

Cattle trampling has an obvious and direct impact on Redshank nest survival, being highlighted as a driver of population decline and a focus of previous investigations at the study site. Even the light cattle grazing practised on Banks Marsh results in some nest losses due to trampling. However, my results reveal that, (i) the number of nests trampled by cattle was relatively low at 15% of nests active during the grazing period, (ii) nests trampled by cattle occurred in shorter vegetation, the latter primarily determined by wildfowl herbivory and (iii) cattle use intensity at trampled nests was more than 2.2 times greater than at nests that successfully hatched young.

Sharps (2015) simulated the risk of Redshank nests being lost to trampling on Banks Marsh at an extraordinarily high 98% at the estimated light grazing intensity of 0.82 cattle per ha. In a follow up study on Frampton marsh, on the Wash estuary, the probability of 'dummy nest' trampling was approximately 30 % at the same cattle grazing level, and only reaching a 98% probability of loss at cattle grazing levels at >3 cattle per ha (Sharps et al., 2017). My use of actual Redshank nests and locally measured cattle use intensity is more robust than the methods adopted by Sharps et al, (2017 and 2015) which used dummy nests randomly placed on the marsh surface or generalised stocking levels (livestock units /ha) for whole sites to estimate trampling risk. Potential sources of bias were avoided using my approach as, (i) Redshank nests on saltmarshes are generally hidden by covering vegetation (Hale, 1988), and are less likely to attract the innate curiosity of cattle compared to unusual objects on the marsh surface, (ii) Redshank do not select nest locations at random (Sharps et al., 2016) and (iii) generalised stocking levels cannot account for heterogeneous cattle grazing behaviour.

Between 1980-86 cattle were excluded from the main Redshank nesting area on Banks Marsh, preventing nest trampling. This initially appeared to be an effective management approach which reduced the incidence of nest trampling, but it was found that the breeding Redshank population within the cattle excluded area continued to decline suggesting an alternative driver of population change might be at least partly responsible (Lambert, 2000). This is explored further in my analysis of Redshank nest survival in Chapter 5.

My findings indicate that maintaining light cattle grazing as the primary conservation management tool for saltmarshes, with its consistent and well understood impact in creating structurally diverse sward (Adnitt *et al.*, 2007; Doody, 2008), which benefits breeding Redshank by providing taller vegetation for nesting and shorter vegetation for identifying approaching predators (Norris *et al.*, 1998) should remain a priority. The limited impact of nest trampling observed on Banks Marsh could be avoided entirely by delaying livestock grazing until mid-July after the Redshank breeding season. However, such a change is unlikely to increase the availability of preferred Redshank nesting habitat, with shorter vegetation around nests and the resulting taller uniform sward might increase adult predation levels.

### 3.5.6 Predation

Predation of Redshank nests accounted for more than half of annual nest losses in this study. This is similar to observations from other sites and predation losses for other wader species but is more than double the average predation rate for Banks Marsh during the period 1973-86 (McCloud, 2019; Macdonald & Bolton, 2008; W G Hale 2018, pers. comm., 12 February). Sharps (2015) suggests that livestock grazing on the Ribble Estuary indirectly impacts Redshank nest survival by causing them to nest in shorter vegetation where they experience increased nest predation, estimated as a 95% risk of predation with light grazing at a level of 0.5 cattle per ha. I found that the difference in both vegetation height and livestock grazing intensity between hatched nests and predated nests was not statistically significant and that wildfowl grazing was the dominant factor in driving the shorter vegetation during the breeding season. However, the question of whether the short vegetation increases predation remains. It is plausible that nests in shorter vegetation are more vulnerable to ground and aerial predators due to a lack of concealment by vegetation cover (Maier, 2014; Thyen & Exo, 2003). However, Laidlaw et al.,(2020) found that poorly hidden examples of Redshank and other nest concealing species were only 10% more likely to be predated than better hidden nests and Ottvall *et al*. (2005) reports that vegetation concealment did not have any significant effect on Redshank nest survival rates.

The low rates of vegetation growth during the early nesting period for subarctic breeding waders reported by Laidlaw et al.,(2020) and the reduced vegetation height following intensive wildfowl herbivory on Banks Marsh, both reduce the capacity to effectively hide nests in vegetation early in the breeding season. However, the majority of nests on Banks Marsh were not located in short vegetation intensively grazed by wildfowl (see Chapter 4 Figure 4.2(a) and (b)). Conversely, tall vegetation may obscure approaching predators, delay the departure of incubating adults and risk their capture (Laidlaw *et al.*, 2015). On Banks Marsh incubating adult Redshank were killed at 20% of all the nests lost to Red Foxes, *Vulpes vulpes,* predation between 2016-18. The vegetation height at these nests was greater than 20cm, with no adult predation at a nest recorded in shorter vegetation. The predation of both adults and eggs is more likely to limit populations than the predation of eggs alone as Redshanks typically produce replacement clutches in the event of failure of the first nest and are relatively long lived with an average lifespan of 4 years (Roos *et al.*, 2018; BTO, 2020).

A Europe-wide review suggested that levels of predation on wader nests are unsustainably high, with predation rates over 50% in situations where the breeding habitat is otherwise considered in favourable condition (Macdonald and Bolton, 2008). If increased nest predation rates are not linked to reduced vegetation height and nest concealment, they may indicate changes in predator abundance and community composition. There has been a marked increase in predation over recent years compared to earlier decades when Carrion Crow, *Corvus corone*, were the most frequently reported nest predators on the Banks Marsh study site (W G Hale 2018, pers.comm., 12th February). It is also likely that Red Fox numbers have increased locally in recent decades (T Baker 2021, (RSPB Ribble Estuary Reserve manager), pers. comm., 6 July). These changes are investigated further in my analysis of Redshank nest survival in Chapter 5.

## 3.6 Conclusion

Grazing is unquestionably a major factor in determining sward height, and habitat selection for breeding waders on saltmarshes and wet grassland habitats (Durant *et al.*, 2008). Understanding the timing, impact and interactions between livestock grazing and wildfowl grazing components on sward height is important for managing populations of breeding Redshank and other wader species of conservation concern with their differing nesting sward height preferences.

I have shown that the application of a simple and practical dropping count method can provide an effective approach to quantifying the temporal and spatial use by both wildfowl and livestock on a large saltmarsh. With more comprehensive wildfowl and livestock grazing information than typically gathered in other studies, I can examine the drivers of sward vegetation height and nest success and failure characteristics with more confidence. Crucially my experimental exclusion approach has demonstrated the role of different grazers in determining sward height conditions relevant for breeding Redshank, conclusively showing the key role of wildfowl herbivory at this study site.

My findings suggest that, (i) both wildfowl and livestock grazing impacts should be assessed in future studies of breeding Redshank and (ii) published research that attributes all impacts of herbivory to livestock grazing, might need to be revisited considering my findings of the importance of wildfowl grazing.

Across a range of saltmarshes, there is likely to be a different mix of effects of livestock and wildfowl herbivory, but on sites which attract nationally and internationally important numbers of ducks and geese, there is a clear potential for their feeding to reduce vegetation height critical for nesting Redshank. Whilst this study focuses on Redshank breeding on saltmarshes my findings have broader relevance for other habitats e.g. wet grasslands and for other species of conservation concern with similar breeding ecology and sward structure preferences such as Black-tailed Godwits, *Limosa limosa* and Curlews, *Numenius arquata* (Durant *et al.*, 2008).

Understanding how different grazers modify habitats and how this relates to nest outcomes is vital for informing evidence-based practices to conserve breeding wader populations. My approach to assessing the impact of wildfowl and livestock grazing on habitat condition and nesting success has been communicated through applied workshops and adopted by conservation managers engaged with providing suitable habitat conditions for breeding waders.

# 4 Determinants of Common Redshank, *Tringa totanus*, nest-sites on saltmarsh and potential responses to changes in management and sea level rise.

## 4.1 Abstract

An estimated 53% decline in the breeding population of Common Redshank, *Tringa totanus,* Redshank hereafter, on British saltmarshes between 1985 and 2011 has led some to predict their disappearance from most saltmarshes by 2036. Population declines have been attributed principally to reductions in suitable nesting habitat driven by unsuitable livestock grazing. I address the issue of saltmarsh habitat suitability for breeding Redshank by analysing the effect of wildfowl and livestock grazing, and abiotic factors, including elevation above sea level, found to be significant determinants of Redshank nest sites, on an internationally important site for nature conservation.

By developing a predictive model to identify areas of saltmarsh habitat suitable for nesting Redshank I explore future management options for maintaining and increasing breeding Redshank by manipulating wildfowl and livestock use, and I assess the impact of plausible future changes in sea level.

I found that winter duck herbivory is of primary importance in limiting the availability of suitable nesting habitat at this site, while light cattle conservation grazing has a positive but less significant effect. The negative impact of a moderate sea level rise prediction up to 2050 might be offset by optimised site management but this would involve a conservation trade-off between migratory wildfowl and breeding gull populations, also of conservation concern.

I argue for the consideration of both wildfowl and livestock herbivory holistically wherever they co-occur and significantly alter habitats and species of conservation concern rather than a focus solely on livestock management. The more widespread use of ecological models to predict the outcomes of different management alternatives should be made to inform multi-species conservation planning and practical conservation delivery.

## 4.2 Introduction

Breeding wader species (order Charadriiformes) are an important component of biodiversity, particularly at higher latitudes. Recent reviews have shown that many species of wader are declining across Europe, and hence are amongst the most threatened bird guilds. The Redshank, is still a common and widespread breeding species but populations have declined moderately across Europe since 1980 (Leyrer, Brown, *et al.*, 2018). Declines in population and range in the UK are attributed to a combination of habitat loss, unfavourable habitat management and predation (Harris *et al.*, 2020, Balmer *et al*., 2013). An estimated halving of saltmarsh-nesting Redshank in Great Britain, to 12,000 pairs, occurred between 1985 and 2011. Breeding Redshank have been projected to disappear from the majority of British saltmarshes by 2036, at the current rate of decline, due to a lack of suitable nesting habitat as a result of unsuitable livestock grazing (Malpas *et al.*, 2013).

Given the widespread decreases in wader populations and their breeding habitats, identifying their key habitat requirements is crucial for improving their conservation. Vegetation structure is an important factor for determining suitable Redshank nesting localities (Thyen and Exo, 2005). Smart *et al*. (2006) concluded that nest site selection is principally driven by vegetation characteristics, in particular by the presence of taller vegetation. Redshanks require a heterogenous sward, with grass tussocks in which to hide their nests and shorter vegetation for feeding and to ease predator detection. Structurally diverse saltmarsh vegetation supports higher breeding densities of Redshank (Adnitt *et al.*, 2007).

Consumption of saltmarsh plants changes the vegetation structure and, importantly for Redshank, reduces vegetation height. Intensive herbivory leads to a short, uniform sward, whilst lighter feeding produces a more uneven, patchy sward with diverse heights. By contrast, the removal of grazing can leave saltmarshes with dense communities of coarse grasses. In many terrestrial ecosystems, large mammals are the most important grazers (McNaughton, 1976), but wildfowl grazing can predominate in coastal and aquatic ecosystems, reducing plant standing crop by up to 100% (Wood *et al.*, 2012). Livestock and wildfowl herbivory are the main drivers of changes in saltmarsh vegetation structure, in the absence of other large non domesticated herbivores, or other human management such as mowing. Light intermittent grazing by cattle, at densities of < 1.0 young animal per hectare, and grazing by wildfowl have long been considered to provide good structural diversity for saltmarsh vegetation (Beeftink, 1977). However, others have suggested that even light cattle grazing can be problematic, in terms of the loss of Redshank nests to trampling and predation (Sharps *et al.*, 2015)

Many wildfowl populations have increased dramatically in recent decades in different regions of the world and in some areas, this has directly resulted in damage to important wetland habitats (Koons *et al.*, 2014). For example, Lesser Snow Goose, *Chen caerulescens,* populations increased from the late 1960s by 5-7% annually. Their overabundant foraging severely degraded coastal breeding habitat across large areas of the Arctic and sub-Arctic (Cargill and Jefferies, 1984; Jefferies *et al.*, 2006). High intensity grazing by Eurasian Wigeon, *Mareca Penelope,* on saltmarshes can result in very short vegetation at the end of the overwintering period in the UK (Cadwalladr *et al.*, 1972). Since the mid 1970's the UK wintering population of Wigeon has experienced a long term increase of 146% (DEFRA, 2020) and the peak winter counts on my study site have increased approximately 10 fold over the same period (Frost *et al* ., 2021). Allport *et al.* (1986) acknowledged the potential importance of wildfowl herbivory for breeding birds, estimating both a wildfowl grazing and livestock grazing index in the first survey of Redshank breeding on British saltmarshes. However, the follow up surveys

(Brindley *et al.*, 1998; Malpas *et a*l., 2011) dropped the wildfowl grazing measure due to difficulties in quantitatively assessing the impact of winter wildfowl herbivory during the Redshank breeding period, focusing solely on attempting to assess livestock grazing (G Allport 2020, pers. comm*.,* 9 March).

Having gathered monthly usage data for cattle, ducks and geese over several years on a large conservation managed saltmarsh system in North West England, of importance for both wintering Wigeon, Pink-footed Goose, *Anser brachyrhynchus*, and breeding Redshank (Murdock *et al.*, 2017), I examine the contributions of cattle and herbivorous wildfowl in determining habitat suitability for nesting Redshank and model the likely impacts of managed changes of these populations.

Current climatic change predictions project mean increases in sea level in the UK ranging from 44-78cm by 2100 based on different emission scenarios (Howard *et al.*, 2019). Such changes are likely to result in a substantial loss of breeding bird habitats on saltmarshes as a result of increased flooding risk (Alexander, 2020). Optimising conservation management might counterbalance some of the loss expected in Redshank breeding habitat in the short and medium term with managed realignment and near-coastal habitat management also required in the longer term (Clausen *et al.*, 2013).

Ecological modelling can contribute both to the assessment of the drivers of change in populations (Mason *et al.*, 2018), and inform decision-making by predicting likely ecosystem responses to management action and future scenarios (Williams *et al.*, 2011). I use modelling to identify where suitable Redshank nesting habitat currently exists and what can be done through habitat management, and under rising sea level scenarios to increase, maintain and mitigate future habitat loss.

Balancing the need to increase suitable habitat for nesting Redshank populations with the habitat requirements of other bird species of conservation concern can be viewed as a conservation vs conservation conflict. Numerous examples exist where species of conservation concern occur in the same habitat and may biologically impact or potentially conflict with each other. Protected predators may consume protected prey e.g. Pine marten, *Martes martes,* and Capercaillie, *Tetrao urogallus,* (Young *et al.*, 2010) or wintering grazing geese may modify coastal habitats negatively impacting breeding waders (Vickery *et al.*, 1997). Such direct conflicts between species of conservation concern present unique challenges for conservation planning and require appropriate management and potential trade-offs. I explore example scenarios, based on the quantitative data presented in Chapter 3, of the compromise between breeding Redshank habitat, and wildfowl use to objectively propose proactive management to meet the needs of multiple species of conservation concern in the face of impending sea level rise.

Mason *et al*. (2019) argue that there is good evidence about how saltmarshes should be managed for breeding Redshank, through a limited focus on livestock grazing. By investigating the impacts of both wildfowl and livestock herbivory and other important abiotic factors, including sea level rise, I can contribute to evidence based multi-species conservation management.

The aims are:

1. To understand the relationship between and relative importance of key biotic variables, (cattle, duck, and goose herbivory) and abiotic variables, including elevation above sea level, for predicting where Redshank nest on saltmarshes.
2. To simulate the potential impacts of manipulating these variables to optimise breeding Redshank habitat in balance with other conservation priorities.
3. To investigate whether losses of saltmarsh habitat for breeding Redshank associated with climate change induced sea level rises can be compensated for by changes in conservation management.

## 4.3 Methods

### 4.3.1 Study area

Data were collected from Banks Marsh saltmarsh, Lancashire, United Kingdom, part of the Ribble Estuary National Nature Reserve (NNR) (Figure 4.1). The site is 1030ha in extent and most of the site is covered by a mosaic of common saltmarsh grass, *Puccinellia maritima,* and red fescue, *Festuca rubra,* saltmarsh communities. Conservation grazing by cattle is undertaken between May and October each year. Internationally important populations of herbivorous wildfowl feed on the saltmarsh vegetation between late September and early May. Eurasian Wigeon and Pink-footed Goose are the most numerous grazing wildfowl, with peak winter counts of circa 50,000 and 11,300 individuals respectively during the study period (2016-18) representing 11.1% and 2.2% of the British winter populations, respectively (Frost et al., 2019). In a typical year, an estimated 80-120 nesting attempts of common Redshank occur on Banks Marsh (Booth and Haywood, 2017), representing up to 1% of the UK saltmarsh breeding population.

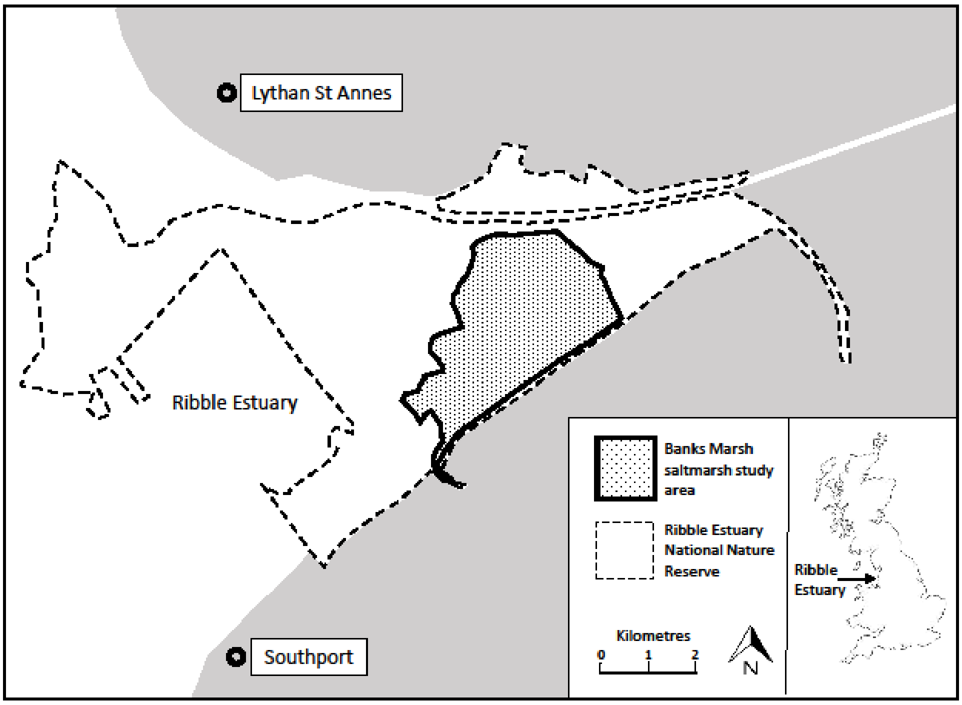



Figure 4.1. Ribble Estuary map showing the location of the Banks Marsh saltmarsh study site, Ribble Estuary National Nature Reserve boundary, location in North West England.

### 4.3.2 Data collection and preparation

To locate Redshank nests, I used standard monitoring approaches detailed in Chapter 2. Nest-finding commenced in the third week of March each year, and continued until late June or early July, dependent upon the phenology of the breeding season in any one year, covering the period from nest site selection until the completion of all nesting activity. Systematic nest searches were conducted every 3-4 days in 10 plots covering an area of 2.6 km$^2$, approximately 1/4 of the total site. Nest finding efficiency was estimated using the method of Green (1985) to determine if a nest was missed during a search and to infer a level of confidence that all nests present were found. The area of the study site occupied by the gull colony, shown in Fig 4.2 was largely omitted from Redshank nest searches due to the unacceptable risk of disturbance to breeding gulls. An assumption that Redshanks avoid nesting in the area is made based on preliminary observations and published research on the competition for breeding habitat and Redshank nest predation by gulls (Oro and Martínez-Abraín, 2007).

As a proxy of duck, goose, and cattle use across the marsh, I collected count data of fresh droppings and dung. Droppings and dung were considered likely to reflect grazing use as both wildfowl and cattle graze extensively and do not use, for example, discrete areas of the site for roosting/resting. Monthly counts of droppings and dung were conducted across all accessible areas of the site at fixed points from August 2016 to April 2018 (see Figure 3.3 Chapter 3). Count data were grouped into biologically relevant time periods to maximise the information content and produce more compact and easily interpreted models. Cattle use was split into early season (May to July) and late season (August to

October) periods, with cattle being removed from the site outside of these two periods. The early season cattle use period partly coincides with the Redshank nesting season. Over-winter duck and goose use was divided into early winter (October to January), the peak winter-feeding period and late winter (February to April), the latter covered the period of early vegetation growth and the start of the Redshank breeding season. Ordinary Kriging was used to produce interpolated raster layers of use for cattle, ducks and geese for the periods discussed (Wackernagel, 1995).

Elevation across the site was estimated using the 2017 Environment Agency (EA) LiDAR (Light Detection and Ranging) composite DTM (Digital Terrain Model) to create an elevation raster layer for the study site with a 1m horizontal resolution and ±5cm relative height error. A proximity map raster layer was generated to allow the distance measurement from any location on the study site to the nearest saltmarsh creek or water body.

Cattle use, duck use, goose use, elevation and distance to water body data are extracted from the raster layers at Redshank nest locations and an equal number of randomly selected nest absence locations using the raster package in R (Hijmans and Etten, 2012). The number of nest absence locations randomly selected was in proportion to the area of the Redshank nest search plots, with the largest plot having 10 nest absence points and the smallest 3. A circular buffer of 10m radius around Redshank nest locations was excluded from the area where nest absence sites could be selected to provide what was considered a reasonable level of spatial differentiation between a nest and a nest absence location. Resampling of nest absence locations was used to generate five replicate sets of data to provide more representative estimates of the nest absence parameters.

### 4.3.3 Model development

To understand the potential drivers of Redshank nesting localities, I used logistic regression, with the presence or absence of a Redshank nest (with absences selected as described above) as the dependent variable and the environmental variables described above (cattle use, duck use, goose use, elevation, distance to water bodies) as the independent variables. Explanatory variables were scaled to make effect sizes comparable.

For the logistic regression, I used a generalized linear model (GLM) procedure using a logistic link function and a binomial error. Bivariate correlations between predictor variables were checked to ensure correlation coefficients $\leq$0.7, indicating no problem with multicollinearity (Dormann *et al.*, 2013).

An interaction effect between duck use, distance to a water body and elevation was considered biologically reasonable as field observations indicated that ducks favoured landing on water at lower elevation saltmarsh areas before feeding. The inclusion of interaction terms did not significantly improve the models so were not considered any further. The potential for nonlinear effects for duck, goose and cattle use were assessed by including quadratic polynomial terms in the model and found to not significantly improve the model fit so I only present the output of the linear models.

I used the dredge function in the MuMIn package in R (Barton, 2009) to identify all candidate models (n=256) for each of the 5 data sets. A model selection framework, using Akaike's Information Criterion $AIC_c$, corrected for small sample sizes to assess model performance, was used to identify a top model set, thereby limiting the potential bias associated with stepwise selection. All models having a ∆AIC value ≤ 6 were selected and more complex models with a higher ∆AIC than simpler nested models were removed (Richards, 2008). The remaining candidate models for each data set were combined and a single averaged model was used to interpret effect sizes and make predictions.

The predictive ability of the averaged model was quantified using Area Under the Curve (AUC). The averaged model was applied across the whole study area, using the predict function in R, to infer suitable nesting areas for Redshank. A probability threshold of 0.5 was used to produce a binary nesting suitability map from the continuous suitability predictions. Finally, the area of suitable nesting habitat was calculated for the two focal years.

### 4.3.4 Scenario modelling

The significant predictor variables for Redshank nesting suitability in the averaged model, namely late season cattle use, early winter duck use and elevation (Table 4.1) were increased and decreased to represent plausible changes in the management of cattle and duck herbivory on the site and future sea level rise predictions.

Measured late season cattle use and early winter duck use were incrementally reduced to zero and increased by up to three times to reflect previously recorded levels and realistic potential management options. Cattle use on the study site has remained relatively stable over the last 40 years during its management as a nature reserve (1979-2018 mean = 730 cattle/grazing season, SD = 137) except for years when reduced availability limited numbers to 240 cattle/grazing season, or when disease outbreaks prevented any cattle use. Prior to this period of conservation management numbers peaked at 1500 cattle/ grazing season. Winter duck use of the site has undergone a dramatic increase over the same period 1979-2018. Wigeon, the duck species which accounts for the overwhelming majority (93%+) of duck use on the site has increased by approximately 10 times to a mean peak winter level of approximately 50,000 individuals in my focal years. A modelled 2/3 reduction in early winter duck use levels is estimated to maintain the on-site population above the threshold of international conservation importance for this species of 14,000 individuals (Frost *et al.*, 2021).

I manipulated the late season cattle use and early winter duck use variables using a simple proportional increase or decrease to the raster layers. A simplifying assumption was made that the relative distribution of cattle and duck use remains the same, based on field observation over several years that spatial cattle and duck use patterns stayed broadly consistent irrespective of the numbers of individuals. Further, I assume that cattle and duck use are not significantly altered by future sea level changes with expected impacts in terms of increased tidal inundation frequency rather than major changes in saltmarsh geomorphological or vegetation communities. Additional investigations would be needed to better understand what might happen in terms of actual cattle and duck carrying capacity under sea level change.

I apply projected sea level increases based on the latest UKCP18 (UK Climate Projections 2018) (Howard *et al.*, 2019), using greenhouse gas concentration trajectories adopted by the IPCC (Intergovernmental Panel on Climate Change) ranging from the low emission RCP2.6 (Representative Concentration Pathway), the low to moderate emission RCP 4.5, and the high emission RCP 8.25. The impact of sea level rises of 0.25m by 2050 and 0.44m, 0.54m and 0.78m by 2100 are assessed using a basic inundation model where sea level rise height is uniformly subtracted from the site elevation raster. Due to the complexity of intertidal areas, I did not consider the potential impacts of future saltmarsh accretion or erosion rates, or factors such as post glacial isostatic rebound, though the latter has negligible effects on sea-level changes in the study region (Shennan *et al.*, 2009). The modelled sea level changes are consistent with other published local predictions for the Ribble estuary of 84 cm by the year 2100 (Environment Agency, 2009) and similar to the nearby Mersey Estuary at 0.5m and 0.7m by 2095, based on UKCPO9 low emission and high emission scenarios (Alexander, 2020).

## 4.4 Results

From the averaged model (Table 4.1), it is evident that Redshank nesting is most influenced by the extent of duck grazing early in the winter period, with higher duck usage of an area resulting in a much-reduced likelihood of Redshank nesting. By contrast, increased use of an area by cattle in late summer has a positive effect on nesting likelihood, though the effect size is much smaller than for early winter duck grazing. There was also a positive relationship between saltmarsh elevation and nesting likelihood. All other variables including cattle use in the Redshank breeding season had non-significant impacts on the likelihood of Redshank nesting.

Table 4.1: Model coefficients for the GLM used to predict areas of suitable Redshank nesting habitat on Banks Marsh 2017-18.

| | Estimate | Std. Error | z value | Pr(>\|z\|) | Signif. code |
|---|---|---|---|---|---|
| (Intercept) | -0.1160 | 0.1505 | 0.7670 | 0.4430 | |
| Cattle use (Late season Aug-Oct) | 0.5109 | 0.2321 | 2.1950 | 0.0282 | * |
| Distance to creek or water body | 0.1896 | 0.1798 | 1.0520 | 0.2926 | |
| Duck use (Early winter Oct to Jan) | -1.2136 | 0.2927 | 4.1300 | 3.64E-05 | *** |
| Duck use (Late winter Feb to April) | 0.1421 | 0.2158 | 0.6570 | 0.5110 | |
| Elevation | 0.6880 | 0.2708 | 2.5290 | 0.0115 | * |
| Goose use (Early winter Oct to Jan) | 0.0932 | 0.1690 | 0.5510 | 0.5820 | |
| Cattle use (Early season May to July) | 0.0087 | 0.0595 | 0.1460 | 0.8837 | |
| Goose use (Late winter Feb to April) | 0.0039 | 0.0350 | 0.1110 | 0.9117 | |
| Signif. codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1 | | | | | |

The mean AUC 0.774, SD= 0.0026 for the averaged model, across the five data sets, demonstrates the acceptable level of discrimination and useful application of the model in predicting where Redshanks nest across the study site (Swets, 1988, Hosmer and Lemeshow, 2013, Duan et al., 2014).

The percentage of the study site modelled as suitable for Redshank nesting was 47.08% in 2017 increasing to 54.81% in 2018 (Table 4.2). Unsuitable areas for Redshank nesting map well with observed high duck use areas in the western and central marsh in the winter before the nesting season (see Chapter 3. Figure 3.10). There is a reasonably good agreement between the area predicted as being suitable nesting habitat and the location of nests found during systematic searches in 2017 and 2018 (Fig.4.2(a) and (b)). The model predicts that the majority of the 300ha of the study site occupied by the colony of breeding Herring Gull, *Larus argentatus, and* Lesser black-backed Gull, *Larus fuscus,* would be suitable Redshank nesting habitat.

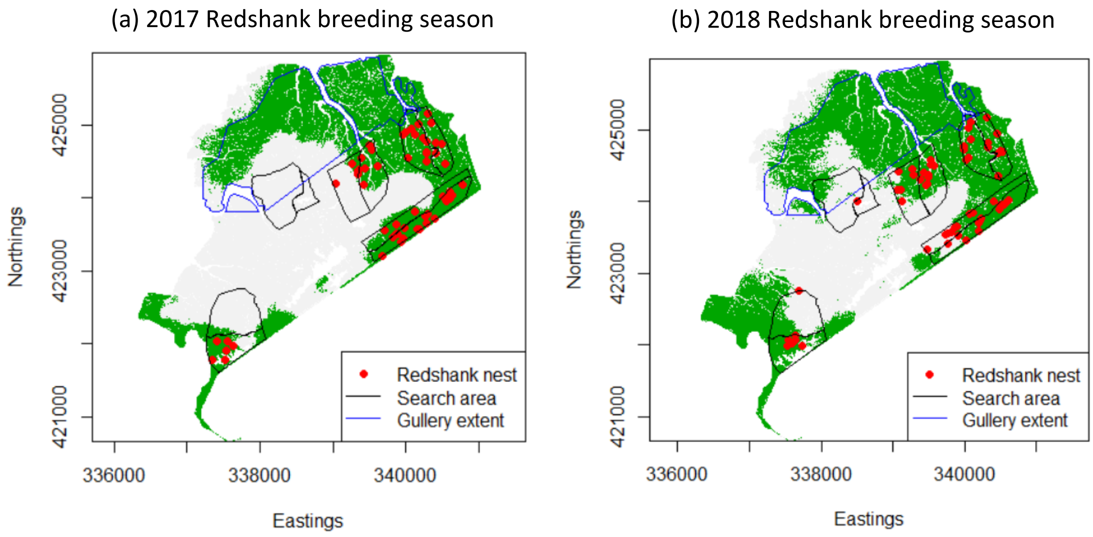



Figure 4.2(a) and (b) show the areas of saltmarsh habitat on Banks Marsh predicted to be suitable for Redshank nesting (green)and unsuitable for nesting (grey) in the 2017 and 2018 breeding seasons, based on the averaged model and < 0.5 threshold.

The impact of decreasing and increasing the values of the significant covariates in the averaged model, (i) Late season cattle use, (ii) Early winter duck use, and (iii) Elevation (reduced to simulate sea level rise), on the extent of suitable Redshank nesting habitat, are shown in Table 4.2 (a-b). These manipulations indicate how cattle and wildfowl use might be managed in the future to maintain or increase the availability of breeding Redshank habitat and the likely reductions resulting from sea level rise based on the most recent predictions of climate change.

Reducing or removing late season cattle use reduces the area suitable for nesting whereas a threefold increase in late season cattle use increases the area suitable for nesting in 2018 by 19%. This amount of increase in cattle use is equivalent to the level of cattle grazing prescribed for the site which was not achieved in 2017. Reducing early winter duck usage by 2/3 dramatically increases the area suitable for Redshank nesting to between 74-82%. This level of reduction is likely to be equivalent to maintaining the 1% threshold of international importance for wintering Wigeon, the species which accounts for the overwhelming majority of duck use on Banks Marsh (Frost *et al.*, 2021, Murdock, Cox and Thomas, 2017). A sea level rise of 0.25m, equivalent to the mid-century RCP4.5 prediction, see methods, reduces the area suitable for nesting Redshank to 26-31%, with a further reduction to less than 7% by 2100.

Table 4.2(a): The effect of modelled decrease and increase in late season cattle use and early winter duck use on the area suitable for Redshank nesting on Banks Marsh 2017-2018.

| Model variable | Nesting Year | % Area suitable for Redshank nesting | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Baseline | ↓1/3 | ↓2/3 | ↓3/3 | ↑1/3 | ↑2/3 | ↑X2 | ↑X3 |
| Late season cattle use | 2017 | 47.08 | 42.82 | 38.24 | 32.92 | 50.76 | 54.09 | 57.76 | 67.16* |
| | 2018 | 54.81 | 51.73 | 49.18 | 47.21 | 58.3 | 62.01 | 65.8 | 74.05 |
| Early winter duck use | 2017 | 47.08 | 55.03 | 73.76 | 92.91 | 42.35 | 38.85 | 36.15 | 31.83* |
| | 2018 | 54.81 | 65.67 | 81.93 | 89.66 | 49.9 | 47.15 | 45.28 | 41.46* |

*Modelled increases in cattle and duck use significantly above levels previously experienced on site.

Table 4.2(b): The effect of modelled increases in sea level (Elevation↓) on the area suitable for Redshank nesting on Banks Marsh 2017-2018. Mid-century and 2100 sea level rise predictions based on; (i) RCP 2.6, a low emissions scenario, (ii) RCP 4.5, a medium–low emissions scenario and (iii) RCP8.5, a high emissions scenario (Howard *et al.*, 2019).

| Model variable | Nesting Year | % Area suitable for Redshank nesting | | | | |
|---|---|---|---|---|---|---|
| | | Baseline | ↑0.25m (2050, RCP4.5) | ↑0.44m (2100, RCP2.6) | ↑0.54m (2100, RCP4.5) | ↑0.78m (2100, RCP8.5) |
| Sea level rise | 2017 | 47.08 | 25.61 | 11.36 | 6.93 | 2.57 |
| | 2018 | 54.81 | 31.28 | 12.99 | 6.06 | 1.05 |

Using the mean Redshank nesting suitability area from the 2017 and 2018 predictions, 49.3% of the study area, as a baseline I simulated the impact of a mid-century 0.25m sea level rise prediction (RCP 4.5, a medium–low emissions scenario) which reduced the area suitable for Redshank nesting to only 28.7%. A modelled increase in late season cattle use by 1/3 and decreased duck use by 2/3, referred to as optimised management, more than compensates for the loss of nesting habitat from the 0.25m sea level rise, with 52.7% of the site becoming suitable for Redshank nesting (Figure 4.3(a-c)). It is notable that areas of the western and central marsh which were unsuitable for Redshank nesting, primarily due to overuse by ducks, become suitable with optimised management. Modelling a range of low to high emission, end of century sea level rise predictions with the improved cattle and duck use management predicts that between 25% and 7% of the study site would remain suitable Redshank nesting habitat (Figure 4.3 (d-f)).

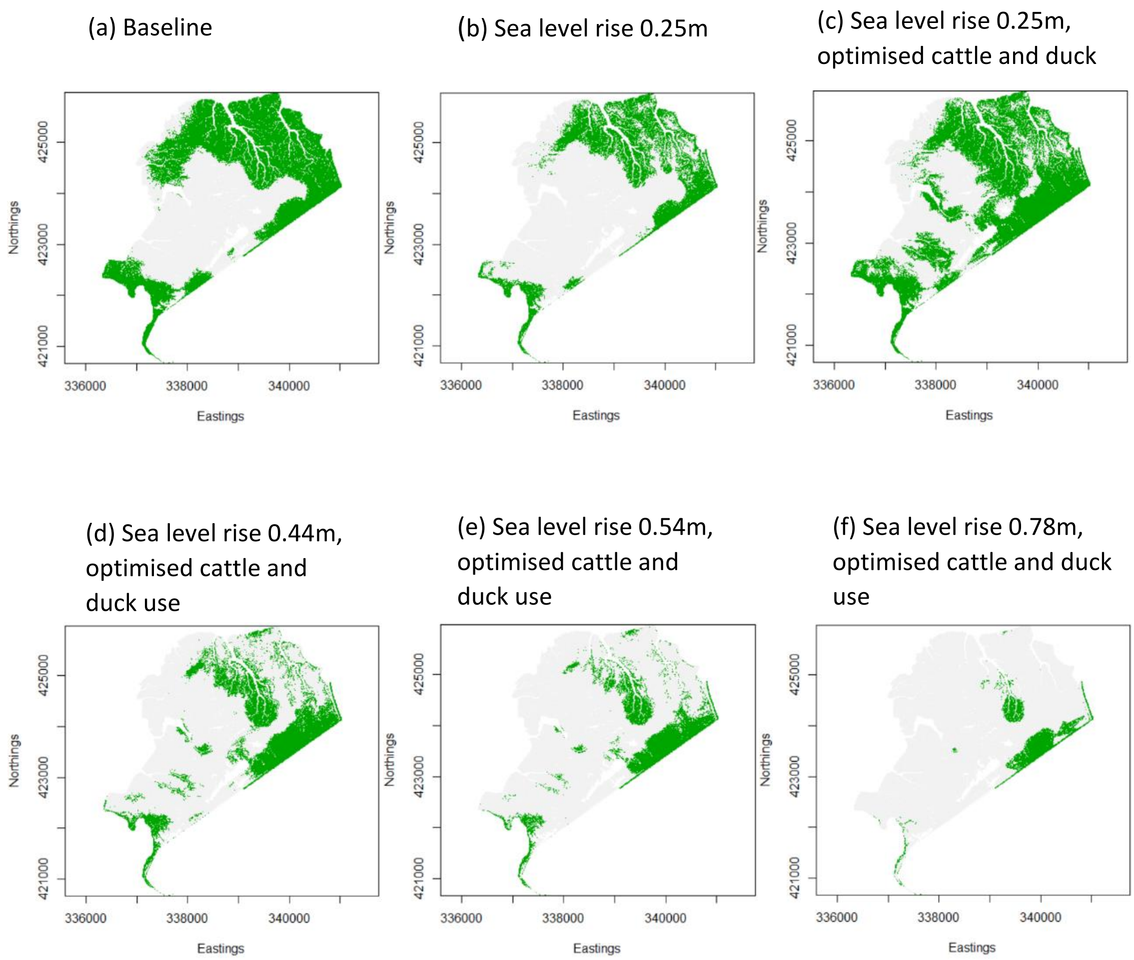



Figure 4.3(a-f) Predictions of the area of Banks Marsh suitable or nesting Redshank (green) under sea level rise and optimised late season cattle use (↑1/3) and early winter duck use (↓2/3) scenarios (a) Baseline (mean 2017/2018) = 49.3% of the area suitable, (b) Sea level rise 0.25m (2050 RCP4.5) = 28.7% of the area suitable, (c) Sea level rise 0.25m (2050 RCP4.5, a medium–low emissions scenario), with optimised cattle and duck use = 52.7% of the area suitable, (d) Sea level rise 0.44m (2100 RCP2.6, a low emissions scenario) with optimised cattle and duck use = 25.2% of the area suitable (e) Sea level rise 0.54m (2100 RCP4.5 a medium–low emissions scenario), with optimised cattle and duck use = 16.4% of the area suitable, (f) Sea level rise 0.78m (RCP8.5, a high emissions scenario), with optimised cattle and duck use = 7% of the area suitable (Howard *et al.*, 2019).

## 4.5 Discussion

A lack of suitable nesting habitat for saltmarsh breeding Redshank is widely suggested as being a contributory factor to recent population declines (Norris *et al.*, 1998, Malpas *et al.*, 2013). Here, I explored the factors influencing Redshank nesting localities across an extensive UK saltmarsh and national nature reserve. I found that, contrary to previous research, a critical factor explaining nesting localities for this species was not cattle grazing during the breeding season but instead was principally over-winter grazing by ducks, which had a strong negative impact and to a lesser extent, late summer grazing by cattle in the previous year, which had a positive impact. Using these relationships, I was able to simulate the potential change to suitable nesting habitat for Redshank under changing management regimes and incorporate the potential impacts of sea-level rise on the extent of suitable habitat under various future change scenarios.

### 4.5.1 Wildfowl herbivory and Redshank nesting

Herbivorous wildfowl are ecological engineers and can significantly alter the habitats where they feed (Green *et al.,* 2013). They are widespread and growing in numbers to an estimated 2.1 million ducks, and 1.1 million geese in the UK ( Frost *et al.*, 2019). The creation of wildfowl sanctuaries on conservation managed saltmarshes similar to the Banks Marsh study site elsewhere in North West England has facilitated dramatic increases in local wintering wildfowl populations with average winter peak wildfowl numbers up by nearly 3 times and 2.5 times on the Dee Estuary and Morecambe Bay (Hirons and Thomas, 1993). Malpas *et al*. (2013) report significant declines in breeding Redshank populations on saltmarshes managed for nature conservation in the North West of England but not in areas outside of conservation management. These declines were attributed to heavy livestock grazing pressure reducing suitable nesting habitat but the possible contributory effect of increased wildfowl herbivory was not investigated.

The Ribble Estuary and Banks Marsh are of outstanding international importance for wintering Wigeon, with peak winter counts averaging 50,000 individuals between 2016-18. I have shown that a modelled reduction in duck use by 2/3 results in a marked increase in suitable nesting habitat for Redshank, which could be achieved whilst maintaining the Wigeon population above the threshold of international importance of 14,000 individuals, 1% of the biogeographic population (Frost *et al.*, 2021). My finding that goose use was not a significant covariate in the averaged model concurs with the study by Madsen *et al.* (2019) which found no negative effect of very intensive goose herbivory on Redshank nest occupancy. Vickery *et al.* (1997) identified a potential conflict between overwintering geese and breeding waders on coastal grazing marshes. Future increases in Pink-footed Geese feeding on many saltmarshes may negatively impact the habitat condition for nesting Redshank in coming years and should be closely monitored.

The potential importance of increased wildfowl herbivory in relation to declining breeding Redshank populations may have been overlooked as a result of being considered beyond the scope of studies focused on the Redshank breeding season. My research reveals that where a significant linear relationship between wildfowl use and Redshank nesting suitability exists the likely impact of both natural and managed changes to wildfowl populations on breeding Redshank habitat can be estimated.

The combination of high intensity winter wildfowl herbivory and the earlier start of the Redshank nesting period (Meltofte *et al.*, 2018) may present a growing conservation conflict that requires management across the many saltmarsh sites where they both occur. Several studies have demonstrated that wildfowl herbivory can reduce populations of birds, small mammals, and invertebrates (Sammler *et al.,* 2008, Samelius and Alisauskas, 2009), but whether these decreases are typical or exceptional requires further investigation.

Increased wildfowl populations may eventually regulate themselves, but not before substantial changes to salt marsh vegetation and impacts on other species have occurred. Conservation management approaches for reducing the negative impact of intense duck use could include removing constraints on disturbance during parts of the winter period to allow recovery of the saltmarsh vegetation before the Redshank breeding season. The impact of human disturbance on wildfowl, and Wigeon, in particular, results in strong local decreases when the disturbance is increased but disturbance need not reflect population level consequences as the fitness costs are low (Gill et al., 2001).

### 4.5.2 Breeding gulls and Redshank conservation conflict

An additional potential conservation conflict exists between nesting Redshank and the breeding colony of Herring Gull, *Larus argentatus, and* Lesser black-backed Gull, *Larus fuscus,* which are also conservation priority species in the UK, being red and amber listed respectively (Stanbury *et al.*, 2021). The 18-fold expansion in the area occupied by gulls to its current 300ha extent has occurred across an area that my model predicts would be largely suitable for nesting Redshank based on the covariates assessed. Breeding species including Redshank that previously nested in the area before the gullery extension do not appear to overlap with large gulls (Murdock *et al.*, 2017). The occupation of nesting habitat suitable for other species and the predatory habits of large gulls have been identified as significant causes of change in many ecological communities (Oro and Martínez-Abraín, 2007). Future research into the displacement of breeding Redshank and the predation of their nests and young by large gulls on Bank Marsh and other sites including Rockcliffe Marsh, which saw similar increases in gull numbers in the 1990s, may identify if gulls drive local population declines of Redshank. A cull of large gulls was initiated on the Banks Marsh study site to address aircraft safety issues but was discontinued (Ecological Solutions, 2013). However, the result of my modelling suggests that if maximising the availability of suitable habitat for breeding Redshank is the priority, reducing the extent of the gullery could be an option.

### 4.5.3 Cattle herbivory and Redshank nesting

I identified the positive impact of late season cattle grazing before the breeding season in increasing the availability of suitable Redshank nesting habitat and quantified how maintaining the prescribed light grazing level for the site would contribute to improving the breeding Redshank habitat. My results support the widely held view that light cattle grazing is beneficial for saltmarsh breeding Redshank as it produces a heterogenous sward of longer vegetation suitable for concealing the nest and incubating adult and shorter vegetation for feeding and identifying approaching predators (Norris *et al.*, 1998). The modelled reductions of late season cattle use reduced the area of saltmarsh suitable for nesting. This supports the findings of Vickery and Gill (1999) that the cessation of livestock grazing on saltmarshes will result in the loss of sites of high conservation value for breeding Redshank. Practical approaches to maintaining appropriate light cattle grazing levels may require reducing fees for commercial graziers, establishing conservation grazing herds for key sites, or improving the delivery of agri-environment schemes (Mason *et al.*, 2019).

My finding that cattle use during the Redshank breeding season is a relatively uninformative variable in the averaged model was not unexpected. Redshanks on my study site begin selecting potential nest sites in late March, five weeks prior to cattle grazing commencing in early May. The light cattle grazing that does occur during the breeding season is likely to have only a limited impact on nesting habitat quality. A simple predictive model using only cattle use during the nesting season data was no better than random at predicting Redshank nesting occurrence with an AUC= 0.52. Crucially, my early season cattle grazing variable is comparable in timing to the livestock grazing measure used extensively in studies linking the decline of breeding Redshank on British saltmarshes to unsuitable livestock grazing. The 'grazing index' adopted by (Brindley *et al.*, 1998; Malpas *et al.*, 2013; Mason *et al.*, 2019) provides only a crude reflection of livestock grazing pressure, but may not recognise the impact of earlier cattle and wildfowl use before the breeding season. Allport *et al.* (1986) acknowledged the importance of both wildfowl and cattle herbivory in the first survey of Redshank breeding on British saltmarshes attempting to assess these separately, but the follow up surveys in 1995 and 2011 discontinued this approach. My findings provide an important caveat to the conclusions of Malpas *et al.* (2013) that inappropriate livestock grazing is a driver of breeding Redshank decline and suggest the need to adopt an alternative holistic approach for assessing the impact of cattle and wildfowl herbivory in future surveys.

### 4.5.4 Elevation and sea level rise

The future decline in available saltmarsh habitat for breeding Redshank seems to be an inevitable consequence of the rising sea levels associated with climate change. My model demonstrates the positive impact of higher elevation for Redshank nesting suitability, with a reduced risk of tidal flooding. Maier (2014) found that a 10cm higher nest site had a 0.07 reduced risk of flooding. Under a moderate to low emissions scenario (RCP4.5) predicted sea level rises resulted in unprecedented decreases in suitable Redshank nesting to 28% and 6.5% of the study area respectively by 2050 and 2100.

Differences in the estimates of habitat suitability between different sea level rise scenarios highlight the importance of accurate forecasting and a more sophisticated elevation modelling approach but my predictions are generally in good agreement with other studies where nesting sites for all avian species were predicted to be restricted by tidal flooding under lower sea level rise scenarios, with no successful breeding under the top-end estimate of sea level increase (Alexander, 2020).

Optimising in situ management of saltmarsh habitat offers the opportunity for buffering the negative impact of seal level rise. My future scenario modelling demonstrates how improving conservation management of cattle and duck use could more than compensate for the losses of suitable Redshank nesting habitat to sea level rise in the medium term up to 2050. However, there is a limit to how much existing habitats can be manipulated to buffer species against the adverse effects of climate change. My results show that even with optimised in situ management only 16% of the study site is likely to be suitable for nesting Redshank by 2100 in a medium–low emissions scenario (RCP4.5).

To secure the availability of suitable saltmarsh habitats for breeding Redshank by the end of the century improved management of existing sites must be accompanied by managed realignment to create new coastal habitats adjacent to existing ones (Wolters *et al.*, 2005). One of the most important factors in the success of managed realignment schemes is the surface elevation as this will determine their ecological community composition. The managed realignment project at Hesketh Out Marsh, completed in 2017 and immediately adjacent to the study site, has multiple objectives including nature conservation, targeting breeding waders and winter wildfowl (RSPB, 2021b). However, the newly created saltmarsh habitat is of significantly lower elevation than my study site. Consequently, there is a greater risk of nest flooding, making the site a potential 'ecological trap' for breeding Redshank (McCloud, 2019). An indirect benefit from the realigned site for Redshank breeding at the estuary level may arise from dispersing the negative impact of intensive duck use, currently heavily focused on my study site, across a wider protected area. Further managed realignment at higher elevation sites would be required to provide additional suitable saltmarsh habitat for breeding Redshank. Alternatively, focusing on optimising conservation management on non-tidal, lowland wet grassland and upland grassland sites where currently <50% of the UK Redshank population breed may be the most effective long-term approach.

#### 4.5.5 Limitations of the model

It is important to consider plausible explanations why my predictive model performance is not better than good, (Duan *et al.*, 2014), in discriminating between suitable and unsuitable nesting habitats for Redshank. This may relate to the response of breeding Redshank to changes in habitat conditions. Observed wildfowl and cattle use intensity changed between the study years (see Chapter 3, Figures 3.6, 3.10 and 3.12). Thompson and Hale (1989) demonstrated breeding site fidelity, and natal philopatry in Redshank selection of nest locations so it is plausible that there may be a lag in the response of Redshank to changing environmental conditions in the short term. Maier, (2014) also reports that saltmarsh breeding birds including Redshank demonstrate a lack of an adaptive response to flooding risks with some individuals repeatedly nesting at elevations that are nearly always flooded. Additional model variables such as saltmarsh vegetation community were not included as a preference for nesting in a particular vegetation type was not detected in this study or that of Thyen and Exo (2003). Also, nest absence points were randomly selected after the fieldwork period and covid 19 restrictions meant that the vegetation community at these points could not be surveyed.

Malpas *et al*. (2013) pose the question 'is there a solution to the conservation problem of Redshank breeding on saltmarshes?' calling for an urgent and in depth understanding of livestock grazing practices and improvement to conservation management guidelines. Unfortunately, the repeated national surveys which provide the data to support this argument did not also collect data on wildfowl herbivory on saltmarshes. As suitable Redshank nest sites are determined by livestock grazing and also often by winter wildfowl herbivory (Vickery *et al.*, 1997) it is possible that an emphasis solely on livestock grazing is misdirected and based on assumption than data. Crucially my modelling has examined holistically wildfowl and livestock herbivory and predicted sea level rise, considering objectively alternative conservation management mechanisms to enhance and maintain Redshank nesting habitat. I have 'separated the objective prediction of consequences from subjective valuations about the importance of management objectives' as suggested by Schuwirth *et al.* (2019), to inform evidence based conservation planning.

## 4.6 Conclusion

By demonstrating the critical importance of duck herbivory alongside cattle grazing and elevation in determining breeding Redshank habitat suitability on saltmarshes I can better understand the potential drivers of Redshank population declines and how habitat management can be improved. I have taken a step forward in identifying in situ conservation management strategies for increasing the availability of Redshank nesting habitat whilst identifying the potential 'trade offs' between other priority conservation interests. Establishing predictive maps of suitable nesting areas and objectively assessing conservation priorities have made it possible to improve applied conservation management outcomes under future sea level rise scenarios. The relative impact of wildfowl herbivory will vary between geographical locations and change over time, but existing wildfowl monitoring programs such as Wetland Bird Survey (WeBS) in the UK can indicate where herbivorous wildfowl are likely to significantly alter saltmarsh habitat conditions. The results of this modelling are likely to be applicable beyond my study site and across Northwest Europe, where the avian populations and saltmarsh communities considered here co-occur.

# 5 Nest survival of Common Redshank, *Tringa totanus*, on saltmarsh: the effects of wildfowl herbivory, livestock grazing, flooding, and predation.

## 5.1 Abstract

Determining the key factors affecting the reproductive success of nesting birds is crucial for understanding their population dynamics and developing effective conservation programmes. Breeding Common Redshank**,** *Tringa totanus,* Redshank hereafter, across Europe have experienced a 61% population decline over the past 40 years and are now vulnerable to extinction. By analysing nest records spanning six decades, I assessed the effect of key variables on Redshank nest survival for a conservation managed saltmarsh part of a National Nature Reserve (NNR) in North West England. Program MARK software was used to estimate daily nest survival in relation to wildfowl herbivory, livestock grazing, flooding, and predation.

Winter duck herbivory across the site was found to be the primary driver of nest success, with the highest numbers of ducks over the period reducing Redshank daily nest survival rates by 4.7%. Cattle grazing in the breeding season was also found to have a negative, but smaller effect, driven in large part by nest trampling.

To date, the focus on successful management of breeding Redshank has been on managing cattle numbers, to produce a suitable vegetation sward for nesting and feeding, whilst minimising nest loss. Here, I identify an overlooked but critical factor impacting nest success, namely winter wildfowl grazing. My results highlight the value of long-term datasets for understanding drivers of population change. They also flag a potential conservation dilemma, as increases in wildfowl numbers over recent decades have been a conservation success story. Yet, this success is detrimentally impacting the status of another species of conservation concern. I suggest feasible options to conserve both breeding Redshank populations and wildfowl populations of international importance on coastal grasslands.

## 5.2 Introduction

Wading birds were once among the most common of all breeding birds in European inland and coastal grazing marshes and saltmarshes, but over recent decades have experienced substantial recent population declines and range contractions. Breeding population reductions of more than 60% have occurred between 1980 and 2019 for species including Northern Lapwing, *Vanellus vanellus,* and Redshank, resulting in them now being classified as vulnerable to extinction in Europe (EBCC/BirdLife/RSPB/CSO, 2019; Birdlife International, 2021).

Of the three key demographic parameters that typically contribute to population declines namely (i) breeding success, (ii) post-fledging juvenile mortality, and (iii) adult mortality, low breeding success has been suggested as the primary cause of recent population declines (and the prevention of possible recovery) in five wader species across Europe, namely Eurasian Oystercatcher, *Haematopus ostralegus*, Northern Lapwing, Black-tailed Godwit, *Limosa limosa*, Eurasian Curlew, *Numenius arquata*, and Redshank (Roodbergen *et al.*, 2012). Low breeding success may result from low nest survival, high chick mortality, or a combination of both. Nest survival is defined here as the probability that a nest survives from initiation to having at least one offspring leave the nest. Measuring nest survival is often key to understanding population dynamics in birds (Rotella *et al.*, 2004). Advanced methods exist for modelling daily survival rates (DSR), which estimate the probability of a nest surviving each day as a function of specific covariates (Dinsmore *et al.*, 2002; Dinsmore and Dinsmore, 2007; Rotella *et al.*, 2004). Such modelling has been widely applied across a range of species and habitats and subsequently driven conservation management actions (Colwell *et al.*, 2011; Polak, 2016; Stephens, 2003).

Lowland wader nest survival has been linked to local habitat conditions (Morrison *et al.*, 2016) and nests may be susceptible to failure as a result of livestock trampling where grazing occurs at high density (Beintema and Muskens, 1987). Nest predation is also a common cause of nest failure (Macdonald and Bolton, 2008) and flooding can be important too, particularly in intertidal habitats such as saltmarshes (Plaschke *et al.*, 2019).

The breeding Redshank population on British saltmarshes is estimated to have halved since 1985 (Malpas *et al.*, 2011). Heavy grazing by livestock, which can result in vegetation being too short for concealing nests, and a high incidence of eggs being trampled in nests have been suggested as key drivers of Redshank population declines (Malpas *et al.*, 2013). Nest survival studies of Redshank on saltmarshes and coastal meadows have focussed to date on livestock grazing as a direct and indirect cause of nest (Sharps *et al*., 2015), the impact of nest flooding (Smart, 2005), habitat characteristics and predators (Ottvall *et al.*, 2005), and the impact of different conservation management strategies (Sharps, 2015). Madsen *et al.* (2019) examined the effect of wildfowl herbivory on field occupancy by nesting and chick-rearing waders including Redshank, but no published research examining the effects of wildfowl and livestock herbivory on lowland wader nest survival could be found. However, in Chapters 3 and 4, I have demonstrated the crucial role of intense wildfowl herbivory in altering nesting habitat suitability for Redshank on saltmarshes. In North America, it has been suggested that reduced nest survival for Semipalmated Plover, *Calidris pusilla,* and other nest-concealing wader species, may be attributed to overabundant feeding by Lesser Snow Goose, *Chen caerulescens*, which can result in the loss of nest vegetation cover on which they depend (Rockwell *et al.*, 2009)

Research focused on the factors affecting nest survival is arguably more valuable for conservation management than those relating avian densities to habitats, and helps to determine if management actions, as opposed to other factors affecting nest survival, result in improved nest success (Shew *et al.*, 2019). Here, I examine Redshank nest records over the period from 1969 to 2018, using a rare long-term and high-quality nest monitoring dataset from Banks Marsh, part of the Ribble Estuary National Nature Reserve, a site of international conservation importance in the North West of England. This comprehensive, long time-series data offers a unique opportunity to assess possible drivers of variation in DSR. The principal objective of this research is to identify those factors significantly affecting the daily nest survival of Redshank at this site. By modelling DSR as a function of cattle and wildfowl numbers, flooding, predation, and the observer effect of intensive fieldwork I aim to:

1. Identify the key drivers of varying nest survival rates among years, and hence help to understand possible causes of population decline of breeding Redshank at the site, whose population trends are indicative of breeding Redshank nationwide.

2. Based on the findings from (1), to suggest conservation management approaches for improving Redshank nest survival, to prevent further population declines and promote recovery.

## 5.3 Methods

### 5.3.1 Study area

All Redshank nests used in this chapter were located on Banks Marsh (Figure 5.1) between 1969 and 2018. Before 1979, Banks Marsh was a privately owned, commercially grazed saltmarsh. Pioneering research into the ecology of saltmarsh breeding Redshank, led by Professor W G Hale, was undertaken on the site between 1968 and 1989, which resulted in the importance of the site for breeding Redshank being recognised and in its declaration as a Site of Special Scientific Interest (SSSI) and National Nature Reserve (NNR) in 1979. Since this time, it has been actively managed primarily for the conservation of breeding birds and winter wildfowl populations.

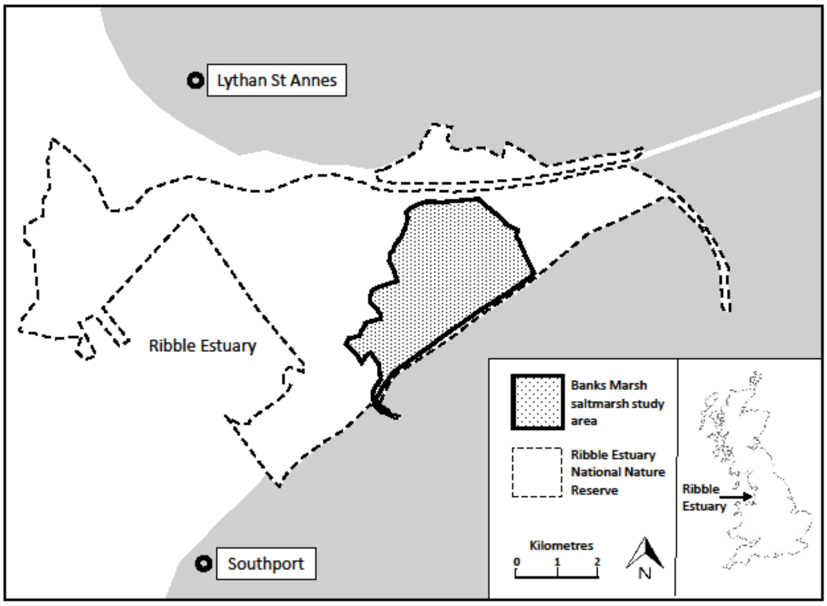



Figure 5.1 Location of the Banks Marsh saltmarsh study site within the Ribble Estuary National Nature Reserve in North West England.

### 5.3.2 Data collection

Nest finding and monitoring

Throughout the entire study period, Redshank nests were located by systematic searching throughout the nesting season (April to July), with active searching the primary method adopted, and flushing and passive searching employed as secondary approaches (Bibby *et al.*, 2000; Green, 1986). Further details of these methods are provided in Chapter 2. After locating a nest, nest monitoring occurred at a frequency of 4 days or fewer, to acquire the necessary information for the DSR analysis. This continued until a nest outcome occurred (nest loss or hatching). The following data were collected, (i) the date the nest was found, (ii) the estimated start of laying/incubation, (iii) the last day the nest was known to be active, (iv) the last day a nest was checked, and (v) the fate of the nest. Nest fate was recorded as failed when eggs were cold, damaged, or missing and no adults were present. Nest success was recorded by the presence of newly hatched, precocial chicks or an empty nest with tiny fragments of eggshell in the lining demonstrating hatching (Green, 2004). The start of incubation and likely date of hatching was gauged by observing the degree of floatation of eggs in water (van Paassen *et al.*, 1984), assuming an incubation period of 24 days (Cramp and Simmons, 1983). In nests found before clutch completion, the first egg laying date was estimated based on 1.5 day intervals between egg-laying events (Ottvall, 2004). The presence of adults at a nest or warm intact eggs indicated continuing incubation. For failed nests, the probable cause of nest failure was also recorded. Flooding was considered the probable cause of failure when mud and tidal debris were observed at the nest and on eggs of abandoned nests. Predation was indicated by either partly eaten eggs, large fragments of eggshell around the nest, or an empty nest with no shell fragments in the nest cup (Green, 1986). Eggs crushed in the nest cup with hoof prints, or the presence of other indicative trampling signs were taken to indicate livestock trampling caused nest failure. Nest were recorded as abandoned if there was no clear evidence of the cause of failure.

Environmental covariates

A range of environmental covariates considered to affect nest survival in redshank and other species were obtained from published and grey literature sources. A lack of fine-scale data related to nest occurrence in the early years of the study period precluded an analysis of nest fates at individual nests. Instead, we take a site-level approach to understand the impacts of livestock grazing, wildfowl numbers and flooding and incorporate national trend data for predator populations, which I assume to be representative of the broad scale predation pressure at the site level. Hence, individual Redshank nests are considered replicate 'trials' within a particular breeding season. A summary description of variables used to model DSR is provided in Table 5.1.

Table 5.1 Variables used to model Daily Survival Rates (DSR) of Redshank nests on Banks Marsh between 1968 and 2018.

| Variable Code | Description |
|---|---|
| Wigeon | Peak winter Wigeon counts (October to March) preceding the Redshank nesting period (April to July) |
| Cattle1 | Cattle numbers in the summer (May to October) preceding the Redshank breeding season |
| Cattle2 | Cattle numbers during the Redshank breeding season (May to July) |
| Flooding | High tides events that resulted in multiple Redshank nest failures |
| Capture | Nest active during a period of intensive nest study work, including tagging adults |
| Crow | Crow population from the National Gamebag Census (NGC) national index |
| Nest Age | Day since the first egg laying |
| Year | Year of nesting event (1969 = year 1) |

Livestock grazing records and herbivorous wildfowl counts act principally as a proxy of habitat suitability for nesting, as vegetation height and nest concealment have been shown to be positively correlated with nest survival (Bentzen *et al.*, 2017; Walpole *et al.*, 2008), though clearly, cattle in the Redshank breeding season can also directly impact nest fate through trampling. Complete records of the number of cattle grazed each year of nest monitoring were compiled (W G Hale 2019, pers. comm., 18 July, L Barber 2014 (Natural England), pers. comm., 19 September). Two cattle variables are included, (i) Cattle 1 – the number of cattle present in the grazing period May – October in the year prior to the breeding season, which reflects livestock grazing induced changes to saltmarsh habitat prior to the Redshank nesting, and (ii) Cattle 2 – the number of cattle present in the Redshank breeding season (May-July). The latter has a limited influence on habitat condition during the breeding season (See Chapter 3) and impacts levels of Redshank nest trampling, which is assumed to be proportional to the number of cattle grazed. Cattle were experimentally excluded from Redshank nesting areas in the study years 1980-86.

Dramatic changes in the number of herbivorous wildfowl feeding on the study site occurred between 1969 and 2018 (Chapter 3, Figure 3.2). Systematic, standardised counts of Eurasian Wigeon, *Mareca penelope*, from the Wetland Bird Survey (WeBS) are used in my analyses as an indicator of wildfowl numbers as this species accounts for the majority of wildfowl feeding on site (Frost *et al.*, 2021). Pink-footed Goose, *Anser brachyrhynchus*, numbers have also fluctuated over the study period. However, in the latter case recorded numbers do not represent herbivory on Banks Marsh saltmarsh, as the geese fed primarily on neighbouring agricultural fields until the later years of this study, 2016-19. Peak winter counts are used as a standard to produce indices and trends in the abundance and distribution of wintering wildfowl across the UK. As WeBS counts are conducted at high tide when wildfowl are highly mobile, the Ribble estuary scale count is the most appropriate for my analysis, with Banks Marsh supporting the majority of all wildfowl recorded on the Ribble Estuary (K. Abram 2018, (Ribble Estuary WeBS coordinator), pers. comm., 9 February).

Redshank nests in intertidal wetlands are subject to flooding caused by high tides, which may affect nest survival (Norris, 2000; Smart, 2005; Thompson and Hale, 1991). As I was unable to determine all nest elevations relative to published tidal heights, I could not establish flood likelihood for individual nests in any season. Instead, I produced a simpler metric to indicate site level flooding events based on high tides recorded at the nearby Liverpool Tidal gauge (UK Tidal Gauge Network) adjusted for changes in marsh elevation over time. Hence, I derived a binary covariate for potential flooding events.

As nest predation is a main reason for the lack of reproductive success in many avian species (Martin, 1992), I attempted to incorporate a measure of predation pressure into my nest survival analyses. Red Fox and Carrion Crow, *Corvus corone*, (Crow hereafter) were the most recorded predators of Redshank nests on Banks Marsh during the study period. However, estimates of local predator populations were not available for all the nest monitoring years. Instead, I planned to include predator population indices from the National Gamebag Census (NGC) (Aebischer, 2019), as a proxy for predation pressure. Sample sizes for the NGC in North West England before 1980, were too low to produce reliable population estimates, so indices for the whole of the UK were considered (N. J. Aebischer 2021 (Game and Wildfowl Conservancy Trust), pers. comm., 11 August). These indices were compared with incomplete local predator data to establish if there was a correlation between local and national trends, before inclusion in the models. Three reliable sources with lengthy experience of the study site indicated that the UK index for Red Fox did not reflect the local population between 1969 and 1989, probably due to the very strong impact of control of this species locally on-site use. Consequently, I did not include this measure in my analysis. By contrast, there was no compelling evidence for a discrepancy between the NGC UK Crow index and local records, so this index was retained as a predator index in my models (Figure 5.2).

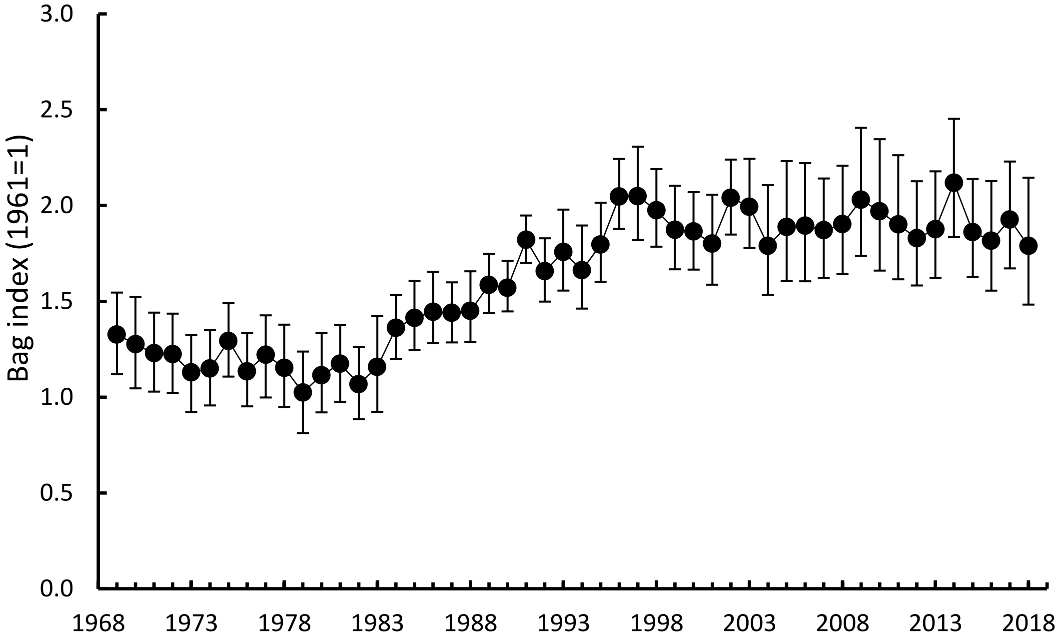



Figure 5.2 UK Crow population index based on the National Gamebag Census (NGC) 1969-2018. Upper and lower confidence intervals are shown. Data provided by N Aebischer.

Research activities can influence the DSR of nests being monitored, resulting in biased estimates of nest survival (Rotella *et al.*, 2000). A nuisance variable 'Capture' is included in this analysis to cover a period when intensive fieldwork, including the capture of adult Redshanks on the nest, was undertaken to establish an individually marked population. A binary 'Capture' variable is allocated to all nests located during such intensive fieldwork period (1), and all outside (0).

Nest age, measured as days starting with 0 on the first day of the study season and ending with 112 on the last day of the season, was included in my models, although I did not have preconceived ideas on how this might influence daily nest survival. Finally, I included a 'Year' variable in my models, to represent annual variation including e.g., weather effects, not accounted for by other covariates. Its inclusion allows prediction of how much DSR varies over years after controlling for other factors and is important in long-term studies.

### 5.3.3 Data analysis

Explanatory variables were scaled to make effect sizes comparable for modelling purposes and back transformed for the presentation of the results of key covariates to make them more readily interpretable. Bivariate correlations between predictor variables were checked to ensure correlation coefficients $\leq 0.7$, indicating no issues with multicollinearity (Dormann *et al.*, 2013).

Daily survival of Redshank nests was modelled using the R package RMark, version 2.2.7 (Laake, 2019) which provides an interface to Program MARK, version 9.0 (White, 2021). The list of models was based on variable subsets that were considered *a priori* as candidates to affect Redshank nest survival. I modelled the variables alone and in combinations, and evaluated model performance using Akaike's information criterion corrected for small sample sizes (AICc) (Burnham and Anderson, 2002). The model with the lowest AICc, the top model, was considered to represent the best compromise between goodness-of-fit and model complexity (Whittingham *et al.*, 2006). Interaction effects considered biologically reasonable were also assessed in the model selection process. The possibility of model overfitting was reduced due to the available large sample size of 2,990 individual nests. Overdispersion was assessed and corrected by using an overdispersion parameter to adjust standard errors and likelihood statistics, following Lebreton *et al.* (1992). Quasi AIC*c* values, (QAICc) a modification of AICc for overdispersed count data are presented for the candidate models. To evaluate which variables had the greatest influence on nest DSR in the top model, the 'sensitivity' for each predictor was estimated for a range of values and presented graphically.

## 5.4 Results

This nest survival analysis examines 2990 Redshank nests, of which approximately 41% hatched, 11% were trampled by cattle, 10% were predated, 17% were lost to flooding and 18% were abandoned. The fates of the remaining <2% of nests were either unknown or failed as a result of egg collection, eggs being addled or trodden on by researchers.

Model selection

The top model to explain the DSR of Redshank nests on Banks marsh included all the available variables, and also included an interaction between wildfowl and cattle numbers in the period prior to the breeding season. The next best model was 2.44 QAICc units higher and differed only in excluding the capture variable (Table 5.2). The candidate models in the top model set to predict DSR (Delta QAICc <6) all included the effect of cattle (in both the season preceding and the breeding season), wildfowl (in the preceding winter), flooding events, predation level and year, with nest age being in four of the top five models.

Table 5.2 Candidate top models predicting Daily Survival Rates (DSR) of Redshanks nests on Bank Marsh 1968-89 and 2016-18. The Quasi Akaike's information criterion (QAICc) is a modification of Akaike's Information Criterion (AICc) for overdispersed count data.

| Candidate model (n= 2990 nests) | QAICc | DeltaQAICc | Weight |
|---|---|---|---|
| S(~Wigeon + Cattle1 + Wigeon:Cattle1 + Cattle 2 + Flooding + Capture + Crow + NestAge + Year) | 3000.32 | 0.00 | 0.55 |
| S(~Wigeon + Cattle1 + Wigeon:Cattle1 + Cattle 2 + Flooding + Crow + NestAge + Year) | 3002.75 | 2.44 | 0.16 |
| S(~Wigeon + Cattle1 + Cattle2 + Flooding + Crow + NestAge + Year) | 3003.45 | 3.14 | 0.11 |
| S(~Wigeon + Cattle1 + Cattle2 + Flooding + Capture + Crow + NestAge + Year) | 3005.20 | 4.88 | 0.05 |
| S(~Wigeon + Cattle1 + Cattle2 + Flooding + Crow + Year) | 3005.46 | 5.14 | 0.04 |

Estimates from the top model indicate that, of the three habitat-related herbivory variables, Wigeon numbers had the greater negative affect on DSR compared to both livestock variables, and that cattle grazing in the current breeding season (Cattle2) was, in turn, more influential on DSR than cattle in the preceding season (Cattle1). There was also a positive interaction between Cattle1 and Wigeon. The occurrence of Flooding had a small positive impact on the daily nest survival rate and an increase in the Crow variable also had a positive effect. The Capture variable had a negative impact in the top model. NestAge had a small negative impact on daily survival, i.e., older nests had lower DSR while the Year variable had a small positive effect i.e., in later years of the study DSR was higher (Table 5.3).

Table 5.3 Top model estimates for individual variables that explained the Daily Survival of Redshank nests on Banks Marsh between 1968-89 and 2016-18

| Independent variable | estimate | se | lcl | ucl |
|---|---|---|---|---|
| (Intercept) | -12.87 | 0.00 | -12.87 | -12.87 |
| Wigeon | -0.38 | 0.00 | -0.38 | -0.38 |
| Cattle1 | -0.07 | 0.03 | -0.13 | -0.01 |
| Cattle2 | -0.30 | 0.02 | -0.33 | -0.27 |
| Flooding | 0.22 | 0.06 | 0.11 | 0.33 |
| Capture | -0.34 | 0.11 | -0.55 | -0.12 |
| Crow | 0.42 | 0.06 | 0.30 | 0.54 |
| NestAge | -0.01 | 0.00 | -0.02 | -0.01 |
| Year | 0.01 | 0.00 | 0.01 | 0.01 |
| Wigeon x Cattle1 interaction | 0.23 | 0.00 | 0.23 | 0.23 |

The influence of changes in key individual variables on DSR in the top model are shown in Figures 5.3, 5.4 and 5.5. Of the three variables related to habitat condition the change recorded in peak Wigeon counts in the winter before the Redshank breeding season has the greatest effect on DSR. For the lowest recorded Wigeon count (3,932) the modelled mean DSR = 0.991; at the highest Wigeon level (71,533) the DSR= 0.944, a 4.7% reduction in DSR. In the final year of my study, peak Wigeon counts were 48,243, resulting in a DSR of 0.972 (Figure 5.3a).

Removing cattle in the breeding season, which affects both habitat conditions and direct nest loss through cattle trampling, was estimated to generate a DSR of 0.991, compared to a DSR = 0.979 when the site was grazed at a density of 1.5 cattle/ha. Hence, removing cattle would increase DSR by 1.2%. At the currently prescribed stocking rate for conservation grazing on Banks Marsh (800 cattle = 0.8 cattle/ha) the DSR = 0.985 a 0.6% decrease or increase from the lowest and highest recorded cattle grazing levels respectively (Figure 5.3b).

Changes in the cattle numbers in the grazing season preceding the Redshank nesting period resulted in a DSR = 0.99 (242 cattle= 0.24 cattle/ha) and DSR = 0.986 (1500 cattle =1.5 cattle/ha) a decrease in DSR of 0.4% between the lowest and highest recorded stocking level (Figure 5.3c).

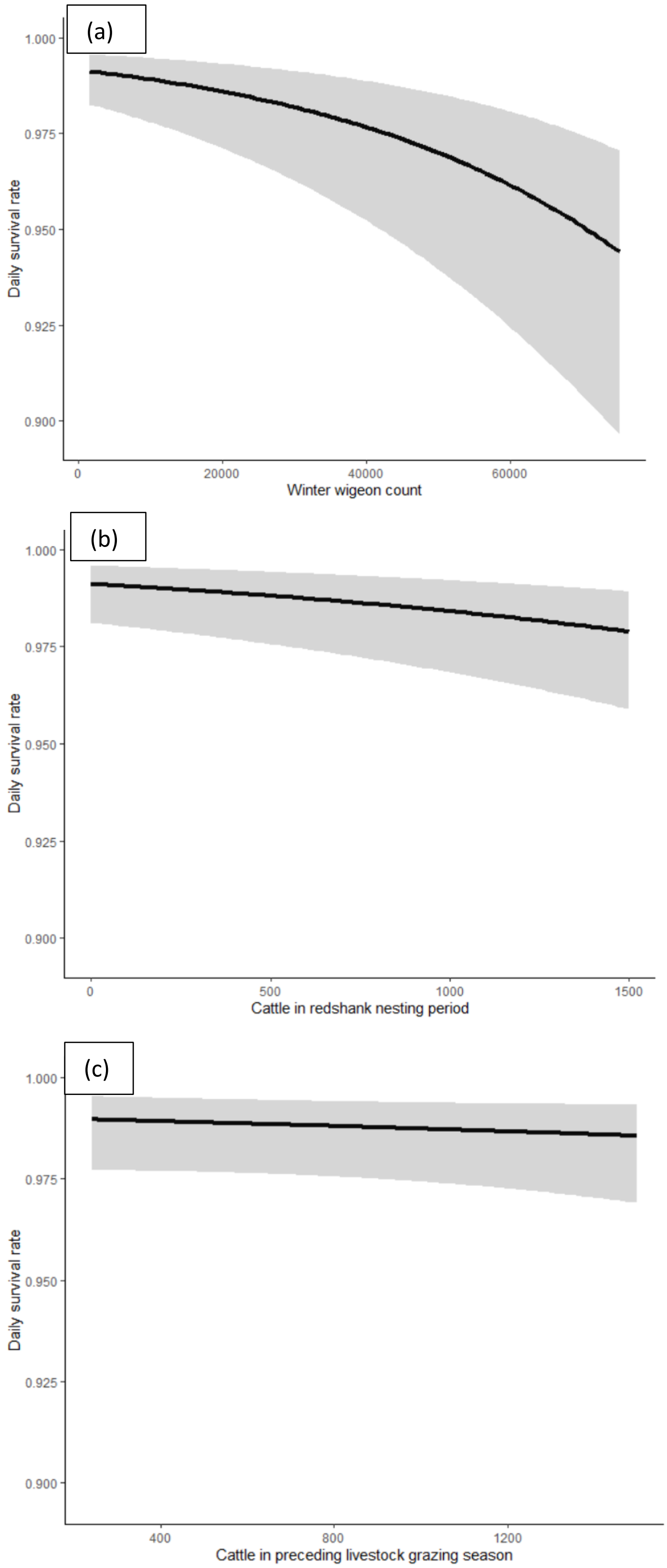



Figure 5.3 Redshank nest Daily Survival Rate on Banks Marsh in relation to (a) peak Wigeon counts in the winter preceding the Redshank breeding season, (b) cattle in the Redshank nesting period, and (c) cattle in the preceding livestock grazing season. The grey shading indicates upper and lower 95% confidence intervals.

The interaction of Wigeon and cattle in the livestock grazing season preceding Redshank nesting shows that estimated Daily Survival Rates are lowest at high Wigeon and low cattle levels (Figure 5.4). Indicating the benefit of cattle grazing for Redshank nesting habitat but the dominant detrimental effect of high Wigeon numbers which have occurred at the lower levels of cattle grazing.

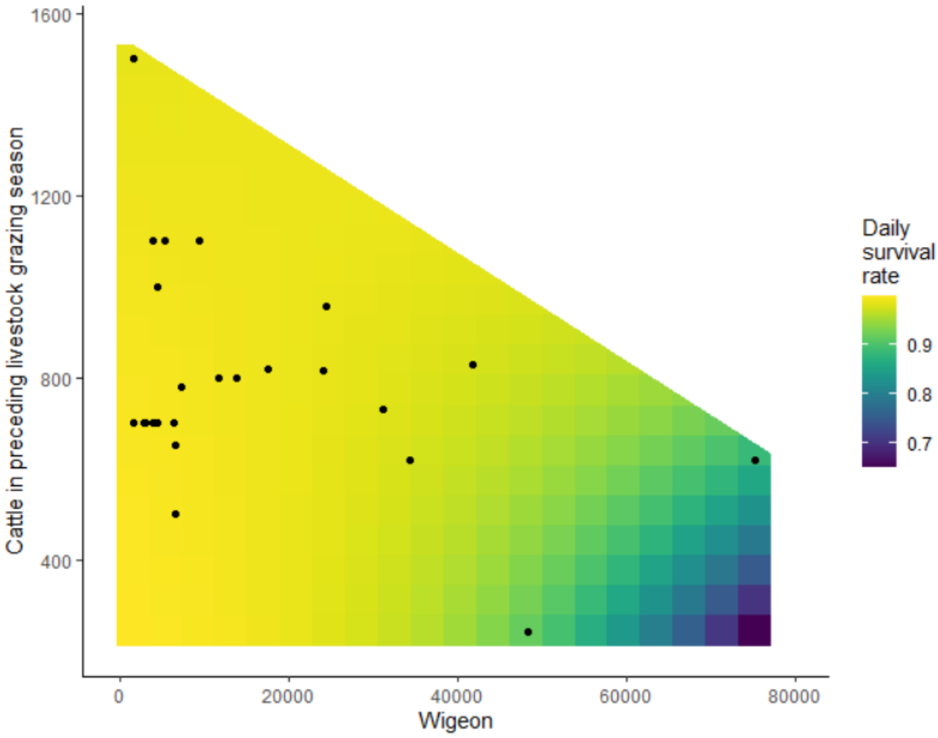



Figure 5.4 Interaction effect of Wigeon and cattle grazing in the preceding livestock grazing season on Redshank nest Daily Survival Rate on Banks Marsh. Observed data points are shown (black points).

The occurrence of tidal flooding appears to result in an 0.3% increase in DSR across all nests in my model (Figure 5.5a). Intensive research involving the capture of adult birds at nests results in an estimated 0.3% decrease in DSR (Figure 5.5b)). At the lowest level, the Crow index results in a DSR= 0.979 which increases to DSR= 0.997 at its highest level, a positive change of 1.8% (Figure 5.5c). By contrast, Nest Age and Year have minimal impacts on nest survival.

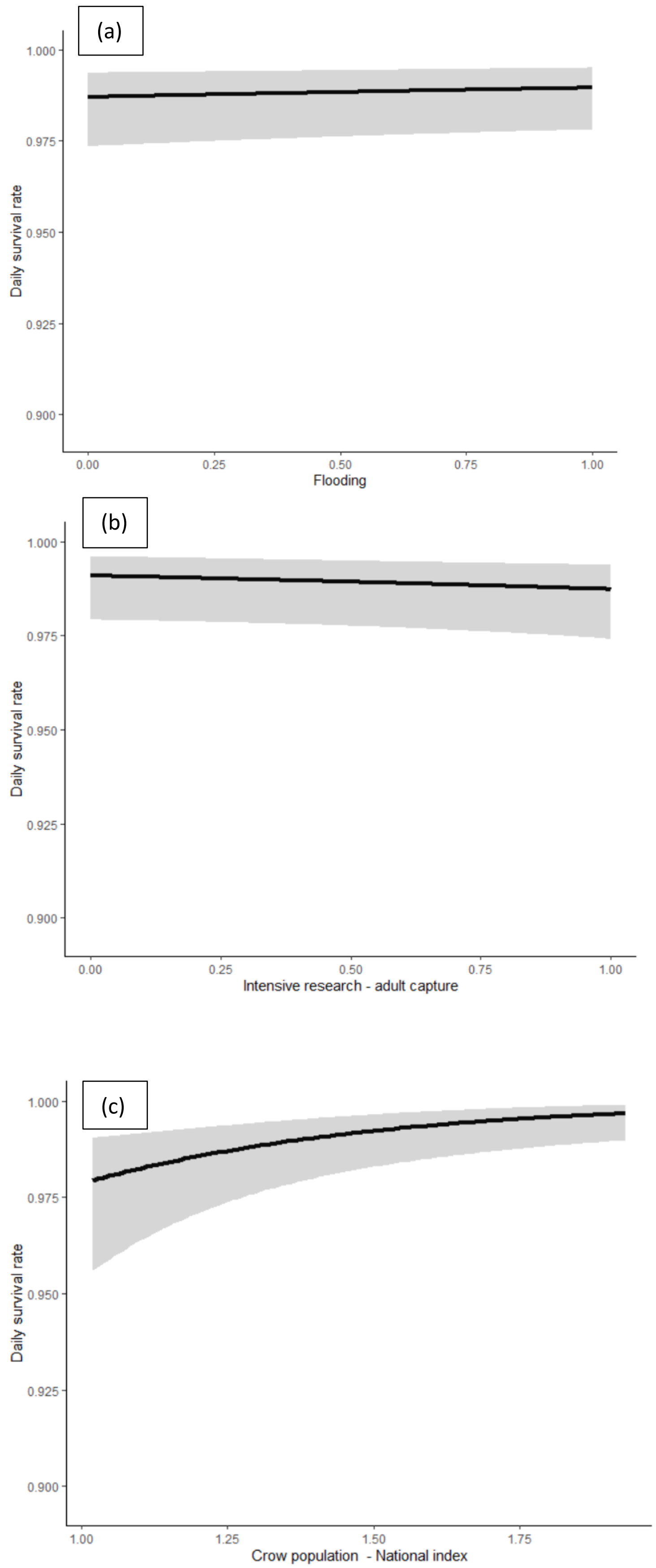



Figure 5.5 Redshank nest Daily Survival Rate on Banks Marsh in relation to (a) spring tide flooding, (b) intensive research period when Redshanks were captured at nests, and (c) Crow population, a UK index. The grey shading indicates upper and lower 95% confidence intervals.

## 5.5 Discussion

Several key points emerge from my analyses. Firstly, that Redshank nest DSR on Banks Marsh over the last six decades has been most strongly negatively affected by a dramatic 18-fold increase in wintering Wigeon (Frost *et al.*, 2019). Variation in the intensity of cattle grazing during the Redshank breeding season also negatively impacted DSR but to a lesser extent over the range of plausible densities. At the levels of light conservation grazing adopted, the impact of cattle is small relative to the effect of typical recent winter Wigeon feeding. The impact of increases in the UK Crow population and of flooding have an unexpectedly weak positive effect on DSR, though the confidence intervals are broad suggesting their impact may be negligible.

A preliminary analysis of adult Redshank survival and population change on Bank Marsh between 1973-1986 indicates a constant adult survival but a declining population, suggesting either low nest survival or high chick mortality (C Howard, per. comm., 18th October 2018). This supports the review of Roodbergen *et al.* (2012) for wader species across Europe that indicated most population declines are caused by a decline in reproduction, not in adult survival. The work I present here goes some way to understanding these drivers of nest mortality, and hence putative drivers of population declines, for coastal breeding Redshank.

No other nest survival analysis for Redshank and waders more generally, which assessed the combined impacts of wildfowl and livestock grazing, could be found. Previous chapters have demonstrated the role of intensive wildfowl herbivory in reducing Redshank breeding density. Here, my modelling provides convincing evidence that increasing numbers of feeding wildfowl also reduced Redshank nest DSR over a long time series. Increases in Wigeon numbers wintering on the Ribble Estuary have been particularly dramatic, with the area supporting a higher concentration (11.5%) of the UK wintering population than any other site (Frost *et al.*, 2019). From a conservation management perspective, balancing the maintenance of both breeding Redshank and wintering wildfowl at saltmarsh sites might necessitate adapting management regimes in future. Management approaches for limiting the negative impact of abundant wildfowl herbivory on Redshank nest survival could include bird scaring and redistributing wildfowl from sites that coincide with prime Redshank breeding habitats. Alternatively, based on the impacts of exclosures on sward condition presented in Chapter 3, the use of small, temporary winter wildfowl exclosures on Redshank nest success could be evaluated at a site level. Disturbing wildfowl can result in large local declines but need not reflect population level consequences as the fitness costs are low (Gill *et al*., 2001). At Banks Marsh, carefully controlled disturbance could be an effective management tool for maintaining Wigeon populations above the threshold of international importance across the entire NNR but below a level at which Redshank nest survival is highly impacted within the Banks Marsh section of the NNR. Coincidentally, the restriction of wildfowl shooting over parts of the NNR may serve as an attractant to Wigeon driven from elsewhere on the estuary where shooting is permitted, to the potential detriment of Redshank on the NNR. Two management options to permit the two species to persist could be summarised as either a habitat sharing (local exclosures to retain tussocky nesting habitat whilst maintaining wildfowl throughout) or habitat sparing (whereby wildfowl and Redshank are encouraged to use different areas of a site) approach.

When no cattle were grazed during the breeding season the DSR was estimated at 0.991, whilst at the highest level of commercial livestock grazing, DSR was reduced by 1.2%. At the prescribed light conservation grazing level (Skelcher, 2010) the reduction in DSR was 0.6%. Higher livestock grazing pressure in the nesting period may result in reduced nest survival because of increased nest trampling, although my results indicate that this risk is small at recommended conservation management grazing levels of < 1.0 cattle per hectare. Under such management, a balance between the creation of a suitable vegetation sward for Redshank nesting and the risk of nest trampling is maintained (Norris *et al.*, 1997)(Norris *et al.*, 1997). Despite the recognised limitations of using crude livestock numbers, rather than the more precise measure of grazing pressure indices adopted in Chapters 3 and 4, the results in the current chapter demonstrate broad support for established guidelines for livestock grazing beneficial for nesting Redshank on saltmarshes. Sharps *et al.* (2015) proposed that the UK Environment Agency definition of light saltmarsh cattle grazing of 0.7-1 young cattle per hectare (Adnitt et al., 2007) is too intensive for Redshank nest survival, reporting a 98% probability of nest trampling at a seasonal cattle density of 0.82 ha from research on Banks Marsh in 2012. My results, which are based on a larger sample size(n = 2990) and long-time series do not support the findings of the Sharps study.

Altering the timing of livestock grazing by introducing cattle after the Redshank breeding season, would eliminate nest trampling but would have practical implications for the livestock farming community who rely on spring grazing on saltmarshes. Further investigations into the benefits for Redshank nest survival of alternative grazing management practices such as rotational grazing are currently underway (L R Mason 2022, pers. comm., 12 January).

Livestock grazing may negatively affect nest survival in saltmarsh breeding Redshank population, but the impact of increased abundance and overabundance of wildfowl had the more significant impact in my models, particularly when livestock grazing levels are within recommended guidelines. Conservation management approaches that encourage lower winter wildfowl numbers in optimal Redshank breeding habitats whilst permitting livestock grazing at low densities are likely to result in the greatest improvement in DSR. Interestingly, an analysis of Redshank nest survival rates from data pooled from multiple studies across Europe revealed nest survival rates on livestock grazed saltmarshes are higher than those where mowing or no active management is undertaken (Sharps, 2015). So, where livestock grazing is practised at recommended conservation levels, we need to consider more fully the potential that it is wildfowl feeding in addition to cattle impacts that lead to a reduction in nest survival.

Nest predation is the cause of the lack of reproductive success in most bird species (Martin, 1992). High levels of predation on wader nests have resulted in nest failure rates of 50% across Europe, and in many cases, this unsustainably high level of predation is likely to be associated with declining populations even where breeding habitat is otherwise favourable (Macdonald and Bolton, 2008). There are many reasons why predators are increasing and, as a consequence, wader populations are facing increasing pressure from both aerial and ground predators (Leyrer, Frikke, *et al.*, 2019).

Increases in the Crow variable in my model have a positive effect on DSR that cannot be fully explained. The UK NGC Crow index may simply not reflect changes in the local Crow population around Banks Marsh over the study period. However, a plausible explanation as to why Redshank nest DSR increased with the Crow population index is that the attention of Crows may have been more drawn to Redshank nests in the early years of the study (when the Crow population were lower) as a result of physically marking nests (Picozzi, 1975). In recent years, with a higher Crow population, physical nest marking was replaced by GPS tagging. In addition, the use of thermal imaging reduced the time spent at nests, minimising the likelihood of revealing nest localities to Crows.

Redshank nests are also vulnerable to ground predators and a substantial proportion of recent nests on Banks Marsh were predated by Red Fox. However, it was not possible to use the UK Red Fox index in my model (see methods 5.3.2 for reasoning). Identifying nest predators and quantifying the contribution of different predators to wader nest losses has improved in recent years through advances in camera technology, which can provide valuable insight to determine whether targeted predator control could be successfully employed to increase Redshank nest survival (Macdonald and Bolton, 2008; Sabine *et al.*, 2005). Obtaining a fuller understanding of predator impact on nest survival is the planned next step on Banks Marsh before considering undertaking predator control measures. The latter is more likely to result in increased nest survival where predator densities are high (Bolton *et al.*, 2007).

Nesting on intertidal saltmarshes presents a potential problem due to periodic tidal flooding. Each breeding season a proportion of the Redshank nests on Banks Marsh is flooded by tides. The inclusion of a flooding variable in my model results in a minor increase in DSR of 0.3%, which may appear counterintuitive. However, Redshank nests are typically enclosed by vegetation which helps to contain eggs within a nest during a flood event. All nests that hatch will have survived at least one of the highest monthly spring tides as the combined laying and incubation period is 28-30 days. Many nests in my study were inundated by spring tides but continued to be incubated afterwards and successfully hatched young; similar survival to hatching following inundation was reported by Smart (2005). Individual Redshank may select nest sites less likely to flood or build nests in vegetation most suitable for retaining eggs within the nest cup during flood events (Smart, 2005). Other coastal breeding species such as Snowy Plovers, *Charadrius nivosus*, in North America appear well-adapted to the risk of nest flooding by spring tides (Plaschke *et al.*, 2019). Further analysis of nests that flooded and survived may be required. However, future sea level rise is predicted to increase the nest flooding vulnerability of coastal bird populations (van de Pol *et al.*, 2010) and reduce the available habitat for intertidal nesting (see Chapter 4). There is certainly a need for future work to assess the likely impacts of ongoing sea level rise around the UK on the populations of Redshanks and other key bird species that rely on saltmarshes to nest. Though such work is necessarily complicated by a tendency for saltmarshes habitats to naturally accrete and erode over time due to changing tidal conditions and sediment loads.

The influence of disturbance by researchers undertaking studies of breeding birds is an often unrecorded factor that may have adverse effects on nest survival (Rotella *et al.*, 2000). Estimates of nest survival may be biased if nest visits provide predators cues for finding nests (Nichols *et al.*, 1982), or if visits deter predators from approaching nests (MaCivor *et al.*, 1990). Fletcher *et al.* (2005) found no evidence that nest visits every four days during the incubation period reduced the daily survival of Lapwing nests. The inclusion of the nuisance variable 'Capture' in my model, to account for an increased intensity of fieldwork to establish an individually marked population, resulted in a decrease in daily nest survival of 0.3%. Capture attempts early in the incubation period increased the number of nest abandoned, but when birds were allowed to incubate for 2-3 days before trapping started the number of nest abandoned declined (Ashcroft, 1978). Thorup (1995) reports no nest desertion following c.400 capture attempts of Dunlin, *Calidris alpina,* in a study at Tipperne, Denmark, but Kania (1992) in a general review of the safety of catching birds in the nest found that capture did reduce daily survival in some wader species. Ethical review of the catching of birds at the nest where it is found to reduce daily survival rates is essential and the adjustment of capture techniques may be required to minimise the possible risk of nest abandonment.

The large sample of Redshank nests available in this study permitted the flexibility to generate reliable estimates of daily nest survival using multiple meaningful environmental covariates. This has added significantly to our understanding of the underlying causes of Redshank population declines, both at my study site and more generally. However, nest survival analysis is only part of the work required to fully understand the decline of bird species. Further research effort is required to measure the number of chicks that fledge, and how many of these are recruited into the breeding population. Developments in tracking technologies are facilitating an increase in the number of studies capable of generating such data, for example by the use of miniaturized radio trackers and drone based signal triangulation (Lahoz-Monfort and Magrath, 2021).

## 5.6 Conclusion

This study has shown that conservation management for breeding Redshank should focus on the impact of wildfowl herbivory as well as livestock grazing, as this will most benefit Redshank nest survival. Management strategies for conserving wildfowl populations that utilise saltmarshes should consider the requirements of breeding Redshank populations and should aim to achieve a balance of population targets where a potential conflict is identified. Limiting the impact of livestock trampling on nests by delaying the start of grazing may also benefit nest survival although at currently prescribed conservation grazing levels the risk of nest trampling is less significant for nest survival than wildfowl overgrazing. It is currently unclear to what extent these findings can be generalised and applied to other saltmarshes that are conservation grazed and support high numbers of winter wildfowl, but they are being incorporated into ongoing field experiments across major UK saltmarsh estuaries commencing in 2022.

# 6 General Discussion

In this thesis, I first tested the efficacy of the standard census method used to estimate Redshank populations on saltmarshes. I then explored the temporal and spatial impacts of wildfowl and livestock herbivory on saltmarsh vegetation, in relation to breeding Redshank. As part of this work, I developed a predictive model to identify areas of saltmarsh habitat suitable for nesting. Finally, by analysing long-term nest records I identified key environmental variables that affect nest survival. In the following sections, I briefly discuss the key findings of these three research areas, relating my results to previous research, and discussing how they can contribute to our understanding and future conservation management of Redshank, as well as discussing the wider implications of this work. I consider the potential causal pathways affecting breeding Redshank on saltmarshes based on the evidence of my results and from other research. Finally, I summarise recommendations for monitoring and management based on my findings.

## 6.1 Summary of key findings

### 6.1.1 Standardised Survey Method

Monitoring is a crucial part of effective conservation management, which fundamentally requires consistent and reliable methods to produce high-quality data. My testing of the Standardised Survey Method (SSM) for breeding Redshank on saltmarshes (as first proposed by Green, 1986; Green and Johnson, 1984) demonstrated an overestimation of nesting density by an average of 42% across the saltmarsh sites monitored in this study. The potential reasons for this deviation from the original validation data are likely to relate to, (i) the presence of non-breeding birds, (ii) differing causes of nest failure across different habitats and areas, and (iii) geographical variation in, and temporal changes to, nesting phenology. In chapter 2, I propose some potential solutions to these issues. Extended validation of monitoring methods is fundamental to applied ecology, and even minor refinements to established approaches can substantially improve population estimates (Brook *et al.*, 2008). Research that identifies patterns of bias in censusing approaches, and if such errors contribute to flawed conservation management, is crucial for conservation prioritisation and management (Elphick, 2008). My work highlights the value of questioning and revisiting established survey methods to see if they are fit for purpose. The potential for phenological shifts in Redshank nesting on saltmarshes, across both space and time, to impact population estimations may be an example of a more widespread phenomenon in population monitoring, which could impact monitoring protocols to drive apparent population trends. This is worthy of further research, as climate-driven phenological shifts in phenomena across many taxa are now widespread (e.g., Hällfors *et al.*, 2020; Lawrence *et al.*, 2022). By contrast, long-term monitoring protocols have typically remained unaltered over time (e.g., UK Breeding Bird Survey [BBS] requires two site visits between April and June, whilst the US BBS suggests you maintain the same survey date across years). Similar temporal biases could equally apply to surveys of wild mammal populations, such as Elk, *Cervus elaphus*, during migration (e.g., Middleton *et al.*, 2013).

Greater confidence in an improved SSM may encourage its more widespread use amongst future researchers and conservation managers, with impacts likely to occur at multiple scales from improved measures of site-level changes to influencing national and international conservation priorities. The testing undertaken here is timely, as the suggested improvements in timing to reflect geographical and temporal variation in nesting phenology and the impact of tidal flooding and incorporating a double-sampling approach to validate results based on counts of adult Redshank, can be introduced into the next decadal survey of Redshank breeding on British saltmarshes. Improved estimates of breeding Redshank populations may result in an improved assessment of the conservation status of Redshank, thereby prioritising scarce conservation resources to areas and habitats of greatest need.

### 6.1.2 The importance of wildfowl and livestock grazing for Redshank nesting habitat

By analysing the temporal and spatial impact of both wildfowl and livestock herbivory on saltmarsh vegetation height, and relating this to Redshank nesting attempts and outcomes, I explored a fundamental question of whether livestock grazing is likely to be the primary cause of declining habitat suitability, and hence Redshank populations, on saltmarshes or if the issue is more complex than previously considered. Contrary to previous research, (Malpas *et al.*, 2013; Sharps *et al.*, 2015) the principal factor explaining the nesting density of Redshank at my study site was not cattle grazing during the breeding season but instead over-winter grazing by wildfowl, which had a strong negative impact. To a lesser extent, late summer grazing by cattle in the previous year, at recognised levels for conservation management had a positive impact on habitat suitability (Adnitt *et al.*, 2007). This negative impact of increasing wildfowl feeding on breeding waders appears to have been largely overlooked to date (though see e.g., Madsen *et al.*, 2019; Vickery *et al.*, 1997). This issue may be becoming more acute due to the growing wildfowl numbers to an estimated 2.1 million ducks, 1.1 million geese in the UK (Frost *et al.*, 2019) which are often concentrated in wildfowl sanctuaries on conservation managed saltmarshes such as Banks Marsh, the Dee Estuary and Morecambe Bay in North West England (Hirons and Thomas, 1993). More generally evidence is emerging across temperate European and in Arctic and sub Arctic habitats that increasing wildfowl populations are negatively impacting breeding waders (Swift *et al.*, 2017; Tamis and Heemskerk, 2020)

My findings strongly suggest that both wildfowl and livestock grazing impacts should be assessed in future studies of breeding Redshank, where they co-occur. Such a reorientation of future research focus from just livestock grazing to a holistic approach to wildfowl and livestock grazing is something I have encouraged through a grazing management workshop for researchers and site managers in North West England. Horizon scanning for future changes in herbivorous wildfowl populations resulting from climate change or altered agricultural practices would also be relevant in assessing their potential future impact on saltmarsh habitat (Drever *et al.*, 2012; Wisz *et al.*, 2008). The predictive model I developed to identify areas of saltmarsh habitat suitable for nesting Redshank can be used to inform management prescriptions for protected areas and more widely on all saltmarshes through agri-environment scheme options, to demonstrate how the manipulation of wildfowl and livestock use can enhance Redshank breeding. Moreover, the models also have the potential to consider the impacts of future changes in UK sea levels on Redshank habitat. This consideration of the impacts of sea level rise on coastal breeding species (e.g., Von Holle *et al.*, 2019) is another overlooked area of research that should be a high research priority, given the ongoing and relatively rapid sea-level rise around the UK and across the globe.

The most effective future conservation management scenario for increasing breeding Redshank on my focal study site at Banks Marsh might involve reducing Wigeon use of the saltmarsh, at least locally, whilst still maintaining their populations above international importance thresholds, and maintaining cattle grazing to recommended levels for light conservation management. These changes were simulated on my model to more than compensate for the loss of saltmarsh nesting habitat resulting from a 0.25m rise in sea level. This represents a significant step forward in identifying in situ conservation management strategies for increasing the availability of Redshank nesting habitat whilst simultaneously future proofing the potential breeding populations and explicitly addressing the potential 'trade offs' between populations of conservation interest.

A similar approach to managing conservation trade-offs involves the non-lethal removal and relocation of Pine marten, *Martes martes*, where they heavily predate Capercaillie, *Tetrao urogallus*, which could improve Capercaillie breeding success and restore populations (e.g., Summers *et al.*, 2009). In a similar vein, Sutherland and Allport (1994) demonstrated, using a depletion model, that the suitability of grasslands for wintering Bean geese, *Anser fabalis*, a species of considerable conservation interest within Britain, could be maintained by reducing the winter Wigeon population, whose numbers had increased consistently since the 1970's in their study area of the Yare Valley, Norfolk.

Revisiting saltmarsh conservation management prescriptions may be required to address the impact of increasing wildfowl populations on Redshank nesting. Where breeding Redshanks are declining and wintering wildfowl are less threatened, it may be appropriate to set maximum target levels for wildfowl use of a site, in the same way that upper limits for livestock grazing are set. Wildfowl dispersion measures could then be deployed when these limits are reached. This would provide an effective means for managing the potential conservation conflict between breeding Redshank and wildfowl populations. There is a precedent for such targets to be set in European law as the EU Birds Directive (Article 2) commits Member States to maintain the population of bird species to 'a level which corresponds in particular to ecological, scientific and cultural requirements, while taking account of economic and recreational requirements' or to 'adapt the population of these species to that level' (Crabtree *et al.*, 2010, p.33). Such situations normally arise when there is a conflict between people, typically landowners whose grazing is denuded, and wildfowl (Bainbridge, 2017). The latter conflicts are often resolved by shooting, to reduce or redistribute locally problematic populations.

An alternative approach to allowing the coexistence of Redshank and wildfowl, and one that would benefit from future research, is the potential to create small artificial wildfowl grazing exclosures over the winter to maintain areas of long vegetation that Redshank use for nesting. Of course, such areas might be highly attractive to livestock in the spring if the remainder of a site has been heavily overgrazed, which could result in increased nest mortality through trampling.

### 6.1.3 The effects of wildfowl herbivory, livestock grazing, flooding, and predation on Redshank nest survival

By analysing the effects of wildfowl herbivory, livestock grazing, flooding, and predation on Redshank nest survival on Banks Marsh over decades, I found that nest survival has been most strongly negatively affected by the dramatic increase of wintering Wigeon (and their impact on Redshank

breeding habitat condition). Increasing cattle grazing intensity during the Redshank breeding season also negatively impacted nest survival, but to a lesser extent, likely due to increased nest trampling. Determining the key factors affecting the reproductive success of nesting birds is crucial for understanding their population dynamics and for developing effective conservation programmes. The opportunity to examine long term data spanning decades provided advantages over previous studies conducted over a small number of seasons. My results reached different conclusions to a recent single season study (Sharps, 2015), albeit the latter study could not consider wildfowl numbers due to its duration.
Identifying key environmental variables that influence nest survival can assist conservation managers to take positive action for vulnerable species. Crucially this study has shown that management for breeding Redshank should focus on the impact of wildfowl herbivory as well as livestock grazing, as this will most benefit Redshank nest survival. Site-level management strategies for UK saltmarsh habitats need to consider the requirements of breeding Redshank populations and those of important wildfowl populations concurrently, to achieve a balance of population targets.

Importantly, high quality data were not available for all the major nest predators at my study site, so their impact on breeding Redshank could not be assessed. Future research at the study site could try to monitor nest predators more closely, to consider this important additional factor in nest success, which would provide further clarity on the relative importance of the major potential drivers of nest success. I did attempt, as part of the PhD to investigate predation events, by adding ibutton temperature probes to nests, to monitor times of predation events (which can help identify bird versus mammal predation). However, unfortunately, this work did not generate sufficient data for further consideration. Given that predation is a common cause of wader nest failure, and that remote camera technology is now readily available to facilitate improved nest predator recognition (Ellis *et al.*, 2018; Macdonald and Bolton, 2008), such monitoring would be tractable, albeit more difficult in an environment that is regularly inundated by seawater.

Delaying the start of livestock grazing or introducing a rotational grazing system may also benefit nest survival (Sharps *et al.*, 2017), although at currently prescribed conservation grazing levels at my study site, the risk of nest trampling is low. It is unclear to what extent the findings on nest survival from Banks Marsh can be applied to other saltmarshes, but they are being incorporated into planned grazing experiments by RSPB and Natural England on the Ribble Estuary and other UK saltmarshes estuaries to address future management options for breeding Redshank.

This work has clearly shown the benefits of the continued data collection on a single species over an extended period, permitting new insights into their population dynamics. Such long-term insights cannot be gained within the typical three-year research grant funding cycle (unless a long-term dataset is already available). To continue such long-term studies, researchers must continually apply for repeat funding, risking gaps in valuable long-term datasets, whose multiple benefits are often not realised until the future. The Ribble Estuary saltmarshes system has great potential for a long-term study site, given its existing history of Redshank research and the ongoing managed realignment and rewilding projects. The latter could provide a particularly interesting long-term comparison study and investigation into the effect of climate change on intertidal habitats. The benefits of such long-term monitoring approach have been demonstrated elsewhere by the highly influential and informative Isle of Rum Deer, St Kilda Soay Sheep and Wytham Tit projects (Coulson *et al.*, 2001; Pemberton and Kruuk, 2015; Wytham Tit Project, 2022).

## 6.2 Causal pathways for Redshank decline

### 6.2.1 Wildfowl impacts

My research has demonstrated that wildfowl herbivory has limited the distribution and impacted the survival of Redshank nests. The combined observational data, experimental exclusion data and modelling undertaken indicate cause-effect mechanisms by which winter wildfowl overgrazing is diminishing both the amount and quality of Redshank breeding habitat. Increased wintering Wigeon populations, with their feeding strategy of returning repeatedly to favoured areas through the season (Mayhew & Houston 1989), have created large areas of uniformly very short vegetation, 'Wigeon lawns', which lack sufficient cover to conceal Redshank nests. These lawns have replaced the mosaic of sward heights which formerly attracted 100s of nesting Redshank before the establishment of a wildfowl sanctuary area and subsequent increases in wildfowl numbers. Now only very infrequent Redshank nesting attempts are recorded in these areas. In addition to the direct abandonment of unsuitable nesting habitat, intensive wildfowl herbivory can also result in reduced nest survival in several ways related to changes in habitat quality. Insufficient nest vegetation cover may allow eggs to float out of nests during regular tidal flooding (Hale, 1988) and reduced nest concealment may potentially make it easier for predators to locate nests (Sharps *et al.*, 2015). Conversely, if Redshanks abandon the most heavily overgrazed areas and relocate to ungrazed areas with a uniformly long sward incubating adults may be prevented from visually detecting approaching predators resulting in the loss of both adults and eggs, as observed during this research.

Similar mechanisms whereby increasing herbivorous wildfowl impact breeding waders have been proposed for other species and habitats. Geese herbivory driving change in habitat condition in European meadows has been suggested as a factor in the decline of Eurasian Oystercatcher, *Haematopus ostralegus*, Northern Lapwing, *Vanellus vanellus*, Black-tailed Godwit, *Limosa limosa* and Common Redshank, *Tringa totanus*, although the functional mechanisms remain unclear (Kleijn *et al.*, 2011). On coastal grazing marshes, Vickery (1997) found that fields grazed most intensively by geese in the winter supported lower densities of breeding waders in the summer than fields that were rarely grazed by geese. Heavy overgrazing as a result of massive increases in the numbers of migrating Lesser Snow Geese, *Chen caerulescens*, and populations and breeding Canada Geese, *Branta canadensis*, in Eastern Canadian Arctic habitats have destroyed wader nesting habitats. Hudsonian Godwits, *Limosa haemastica*, select specific nesting habitats with high grass and moderate shrubby cover and avoid the barren areas caused by goose overgrazing (Swift et al., 2017). Goose overgrazing is limiting Godwit populations through increases in competition for nest sites, or increased nest predation as habitat changes improve the search efficiency by predators (Martin, 1993).

The sustained increases in abundance of many herbivorous winter wildfowl species across North West Europe which has occurred since the 1950's, because of hunting restrictions and the establishment of protected areas is a conservation success, but this research highlights the potential for a conservation conflict with breeding waders where extreme overgrazing by wildfowl occurs. Looking forward, there are signs of change in this situation regarding Wigeon with stabilisation and decline in the North West Europe flyway (Fox *et al.*, 2016). Fox *et al.* (2016) present evidence of a 'short stopping' where Wigeons are now overwintering in areas closer to their breeding grounds, because of milder winters related to climate change in the north and east of their former wintering range. With declines already detected in the western end of the flyway in Ireland a continuation of this trend may result in a reduced influence on saltmarsh vegetation in North West England, but a redistribution of the impact elsewhere. Monitoring such changes and informing future conservation management at sites impacted by Wigeon herbivory should remain a priority.

### 6.2.2 Livestock impacts

A counterargument is that increased livestock grazing intensity on saltmarshes in North West England is driving declines in breeding Redshank with livestock overgrazing creating an unsuitable short sward structure for nesting and also the direct trampling of nests by livestock (Malpas *et al.*, 2013; Sharps *et al.*, 2015). It is necessary to question the inference of causality from the correlation between data gathered on Redshank breeding densities and apparent increases in livestock grazing intensity from the surveys of British saltmarshes (Allport *et al.*, 1986; Brindley *et al.*, 1998; Malpas *et al.*, 2011), as data from the methods used i.e., the SSM (Green and Johnson, 1984) and the grazing index score do not necessarily concur with detailed records from the same sites. Intensive grazing by sheep can undoubtedly result in very short swards of similar appearance to Wigeon grazed areas (Cadwalladr *et al.*, 1972) and livestock can trample nests (Sharps *et al.*, 2017) but livestock grazing intensity was assessed based on sward appearance and no nest finding or monitoring was undertaken in the (Allport *et al.*, 1986; Brindley *et al.*, 1998; Malpas *et al.*, 2011). On conservation managed sites including Banks Marsh, Hesketh Marsh and Rockcliffe Marsh, where light conservation livestock grazing has been consistently implemented over the last 40 years, Malpas *et al.* (2011) report that livestock grazing has increased to the maximum, heavy intensity. Detailed site data for livestock grazing do not correlate with changes in habitat suitability for Redshank, but changes in wildfowl populations do. Validation of results from the British saltmarsh survey data with other available data sources could improve the usefulness of the national breeding Redshank population index.

The detailed examination and experimental exclusion of wildfowl and cattle herbivory undertaken, and the predictive modelling of habitat suitability and nest survival analysis demonstrates clearly that wildfowl herbivory should be considered as a contributory cause driving the habitat change and the decline of breeding Redshank on saltmarshes. My findings convincingly weaken the case for livestock herbivory alone driving the decline of breeding Redshank in my study area. Whether wildfowl have enough of an effect to be a major factor in the wader declines elsewhere is a crucial issue for further investigation but ignoring the possibility may result in poor decision making and hamper attempts to conserve important breeding Redshank populations.

### 6.2.3 Additional causal pathways

The direct displacement of Redshanks from areas where they had formerly nested as a result of the increased areal extent of the breeding gull colony is a plausible mechanism for the reduced Redshank population at Banks Marsh. During this research, no overlap in the distribution of Redshank nests and those of gulls were observed and evidence of gulls predating Redshank nests and young was recorded but this causal pathway requires further investigation.

The impact of climate change with more extreme tides increasing the vulnerability of Redshank nests to flooding and the devastating loss of saltmarsh nesting habitats due to sea level rise inundation is likely to accelerate in the coming years. This highlights the need for further research which is complicated by a tendency for saltmarshes habitats to naturally accrete and erode over time due to changing tidal conditions and sediment loads.

My results should not be viewed uncritically due to the remaining limitations in the data and the simplicity of the models used. I have looked at the obvious causes of change operating immediately before the declines in Redshank numbers, but additional causal pathways may be affecting change before this time, although data is not available. Difficulties remain in teasing apart the causal role of wildfowl herbivory from the confounding proximal variables of predation and tidal flooding however these are being explored in ongoing research.

## 6.3 Summary of recommendations

### 6.3.1 Monitoring

- Improve the SSM by using the revised ratio of birds counted to peak nesting density demonstrated for saltmarshes in North West England. Combine the SSM rapid assessment with sample nest finding to validate the results in a double sampling approach. Review survey timings to account for potential bias associated with geographical variation, changes in nesting phenology and the impact of flooding.
- Future research incorporating nest finding should examine the efficacy of the flushing method commonly used to locate Redshank nests if employed. An approach to estimating nest finding efficiency should be adopted. The use of thermal imaging and drone surveillance to aid nest finding should be further explored.
- Assess both wildfowl and livestock grazing impacts where they co-occur, in future studies of breeding Redshank on saltmarshes. Simple, proven low-cost methods for measuring grazing impacts can be employed with relative ease and integrated with detailed site records where they are available to validate assumptions about the relative contribution of different grazers. Horizon scanning for future wildfowl population changes is advised given the potential impact demonstrated in this study.
- Further monitoring of wildfowl and livestock grazing impacts on saltmarsh sward conditions, nest concealment and nest predation should be undertaken to provide greater clarity on how this causal pathway may influence Redshank nest success and population change.

### 6.3.2 Management

- Use ecological modelling to assess the drivers of change in breeding Redshank populations and inform future conservation management by predicting ecosystem responses to different management scenarios.
- The optimised conservation management scenario for breeding Redshank at Banks Marsh shown by my modelling, involving reducing the intensity of duck herbivory and maintaining light cattle grazing should be trialled. These changes are predicted to compensate for the future loss of saltmarsh nesting habitat resulting from a 0.25m rise in sea level. The use of the same modelling approach, at the wider flyway scale, to promote extensive areas of well-managed saltmarshes is an opportunity to counterbalance declines observed in many wading bird populations and the expected future losses from rising sea levels at least in the medium term.
- By adopting my modelling approach to conservation conflict situations, it is possible to consider objectively how conservation management can achieve a balance between the requirements of different species of conservation concern.
- A final recommendation for future research is to ensure structures exist for researchers and conservation managers to collaborate closely. Protected areas are central to efforts to protect biodiversity and utilizing the practical expertise and knowledge of their site managers to drive research questions and facilitate relevant research activities is highly desirable, but not always done sufficiently well (Duffield *et al.*, 2021; Watson *et al.*, 2014). Realising the full potential of National Nature Reserves in the UK, as examples of natural laboratories for experimentation and long-term monitoring, to increase our understanding and ability to manage the natural environment for nature conservation, should be a priority.

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