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Light grazing of saltmarshes increases the availability of nest sites for Common Redshank *Tringa totanus*, but reduces their quality

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ABSTRACT

The breeding population of Common Redshank *Tringa totanus* on British saltmarshes has reduced by >50% since 1985, with declines linked to changes in grazing management. Conservation initiatives have encouraged low-intensity grazing of 0.5 cattle ha^{-1} y^{-1} but even light grazing can lead to high rates of nest mortality. To avoid predators, Redshank nest in patches of tall vegetation, but the effects of grazing on the availability and quality of habitat selected by Redshank remain unclear. We investigated Redshank nest site selection in relation to cattle grazing and asked (a) which nest vegetation conditions do Redshank select and (b) does grazing limit the availability of higher quality nest sites? We characterised vegetation height and composition at nests and control locations on six saltmarshes grazed between 0 and 0.55 cattle ha^{-1} y^{-1}, which falls within or near the UK Environment Agency definition of light grazing. Redshank selected nest locations in the tallest vegetation available (26 ± 13 cm with no grazing), but grazing limited the availability of such tall vegetation (11 ± 7 cm at 0.55 cattle ha^{-1} y^{-1}). However, Redshank also selected nest locations dominated by the grass *Festuca rubra*, which increased with higher livestock densities. By causing Redshank to nest in shorter vegetation, but with more of their preferred grass species, grazing presented a trade-off for Redshank. As previous work has shown that nesting in shorter vegetation results in higher nest predation rates, results of this study suggest that even light conservation grazing can result in Redshank nesting in lower quality habitat. Reducing saltmarsh grazing levels below 0.55 cattle ha^{-1} y^{-1} may therefore increase Redshank populations by maintaining a vegetation structure with patches of *F. rubra* but with longer sward heights for nesting.

1. Introduction

Habitat quality affects animal fitness, survival and reproductive success through variation in environmental conditions and resource availability, which generates selective pressure for habitat choice (Brown, 1969; Pulliam, 2000; Fuller, 2012). The density of animals in a habitat can be a misleading indicator of habitat quality (Van Horne, 1983), as species can preferentially use habitat which acts as an 'ecological trap' by lowering breeding success (Best, 1986; Schlaepfer et al., 2002). For many ground-nesting birds, predation is the largest cause of reproductive failure (Angelstam, 1986; Macdonald and Bolton, 2008). Consequently, ground-nesting wading birds show a variety of anti-predator adaptations, including passive and active nest defence (Gochfeld, 1984). Unlike Black-tailed Godwit *Limosa limosa*, and Oystercatcher *Haematopus ostralegus*, the Common Redshank *Tringa totanus* does not attack potential nest predators (Gochfeld, 1984; Nethersole Thompson and Nethersole Thompson, 1986). Instead, they build a cryptic nest obscured by tall vegetation and quietly leave the nest upon approach of a predator (Cervencl et al., 2011; Mandema et al., 2014a). Tall vegetation is therefore a key habitat requirement for successful Redshank breeding as it allows better concealment of nests, which leads to higher breeding success (Thyen and Exo, 2003).

Although Redshank breed in various grassland habitats, saltmarshes are internationally important for the species (Reed, 1985; Cadbury et al., 1987; Brindley et al., 1998). In Britain, saltmarshes once held around half of the national breeding population (Brindley et al., 1998), but the number of Redshank breeding on British saltmarshes declined by >52% between 1985 and 2011 (Malpas et al., 2013b). Declines have been linked to grazing management as breeding densities are higher with light and moderate grazing than on heavily grazed or un-grazed saltmarshes

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(Norris et al., 1998; Malpas et al., 2013b). However, Malpas et al. (2013b) found that the number of breeding pairs declined by 51.6% in Northern England, where grazing was more intensive, but also by 24.2% and 58.1% respectively in Eastern and Southern England where light grazing prevailed. Grazing has the potential to affect habitat suitability of saltmarshes by altering availability of the tall vegetation patches that are used for Redshank nesting (Hale, 1980; Mandema et al., 2014a). Intensive grazing leads to a very short, uniform sward, lighter grazing results in a more uneven patchy sward with diverse heights whilst no grazing can leave saltmarshes with dense communities of the coarse grasses *Elymus* spp. (Jensen, 1985; Kiehl et al., 1996). In general, the aim of saltmarsh grazing management is to reduce the extent of *Elymus* spp. and open the vegetation sward (Adam, 1990). This results in an increased presence of other species, including the fine grass *Festuca rubra* (Andresen et al., 1990). On saltmarshes, Redshank have been recorded nesting within high-mid marsh communities of the grasses, *F. rubra*,*Puccinellia maritima*, or *Elymus* spp. (Hale, 1988; Norris et al., 1997; Thyen and Exo, 2003).

Norris et al. (1997) found that breeding densities are higher in structurally diverse grazed vegetation, and recommend low stocking densities of ~1 cattle ha^{-1}, in order to create a patchy vegetation sward, suitable for nesting Redshank. This falls within the UK Environment Agency definition of light saltmarsh cattle grazing of 0.7–1 young cattle ha^{-1}, present from April to October (Adnitt et al., 2007) which translates to an annual cattle density of 0.35–0.5 cattle $ha^{-1}\,y^{-1}$. However, recent work has suggested that light grazing can reduce nest survival both directly through nest trampling and indirectly through accelerating predation risks. Sharps et al. (2015) found that the risk of Redshank nest predation increased from 28% with no grazing to 95% with light grazing of 0.55 cattle $ha^{-1}\,y^{-1}$, and suggested that the cause was livestock modification of vegetation. Therefore, following the estimates of Macdonald and Bolton (2008), that waders typically need nest survival rates of >49% for population stability, implies that light grazing can negatively affect Redshank populations. Potential Redshank nest predators in Britain include ground predators, such as foxes (*Vulpes vulpes*), stoats (*Mustela erminea*) and non-native American mink (*Neovison vison*) and aerial predators such as corvids (*Corvus* spp.), gulls (*Larus* spp.) (Smart, 2005). There is no evidence to suggest that raptors predate Redshank nests on UK saltmarshes and in areas where raptors are more abundant, such as lowland wet grasslands, they are not a key predator (Malpas et al., 2013a; Laidlaw et al., 2015). Although grazing may decrease habitat quality, the effects of grazing on the sward height and vegetation species composition selected by nesting Redshank remain unclear.

Nesting habitat-use is the end result of habitat-selection processes, which can occur at a range of scales ranging from landscape to nest level (Jones, 2001; Hebert et al., 2006). In lowland wet grasslands, Smart et al. (2006) demonstrated that Redshank select tall nest vegetation, with some additional cover in a wider 10 m area around the nest. Redshank are highly site faithful, and have been found to typically return to nest within 50–1700 m of their birthplace (Thompson and Hale, 1989). Habitat quality and site fidelity provide strong site selection pressures and may reduce the ability of a species to adapt to changing habitat conditions, and therefore limit lifetime reproductive success (Newton, 1989; Gonzalez-Zevallos and Yorio, 2006).

We investigated nest site selection by Redshank breeding in lightly grazed saltmarshes. Habitat use was taken as indicative of nesting habitat selection, and how this varies with different levels of livestock density. We asked: (a) which nest vegetation conditions do Redshank select on saltmarshes? and (b) does grazing limit the availability of preferred nest vegetation conditions?. The study sampled six saltmarshes grazed between 0 and 0.55 cattle $ha^{-1}\,y^{-1}$, and characterised vegetation height and species composition at nests and control locations. We hypothesised that (1) Redshank select taller vegetation at, next to and in the wider area around the nest, but grazing limits the availability of these conditions; (2) The vegetation species composition varies between nests and control locations, with more of the grasses *F. rubra, P. maritima and Elymus repens* found near Redshank nests; (3) The proportion of *E. repens* found at Redshank nests is lower in saltmarshes with higher livestock densities and the proportion of *F. rubra* found at Redshank nests is higher in saltmarshes with higher livestock densities.

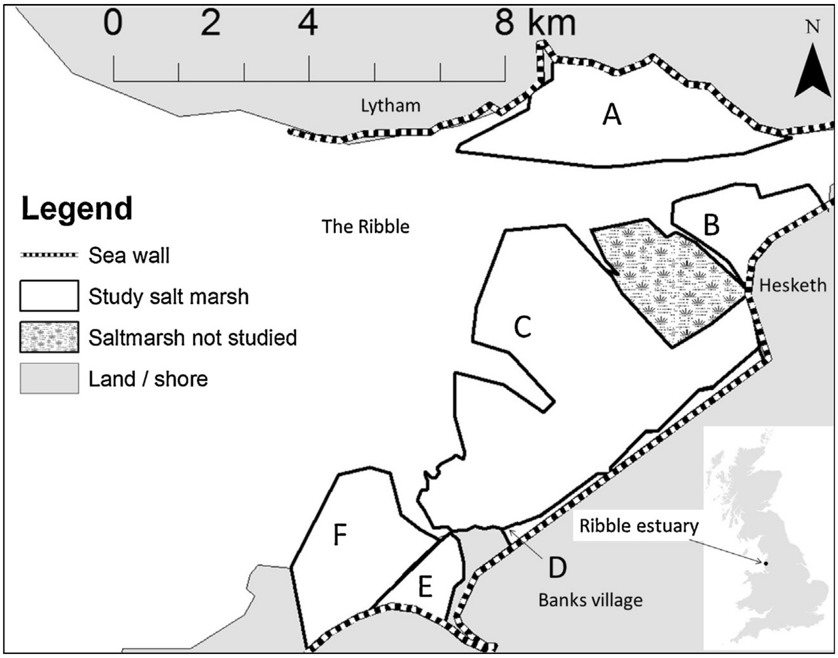

Fig. 1. Ribble Estuary map showing the study saltmarshes (A–F) and geographic location.

2. Materials and methods

2.1. Study location

The study concentrated all observations on one estuary, the Ribble (Fig. 1) to minimise the influence of variation in environmental conditions. Six saltmarshes were selected to cover grazing intensities from 0 to 0.55 cattle $ha^{-1}y^{-1}$ (Fig. 1; Table 1). Much of the Ribble estuary is managed as a National Nature Reserve, which supports one of the largest areas of grazed saltmarsh in the UK (Burd, 1989; Skelcher, 2010). The estuary is bounded by a river channel on the estuarine side and by a sea wall flood defence at the landward side. The vegetation is typical of saltmarshes on the west coast of the UK (Rodwell et al., 2000). The saltmarshes are dominated by *F. rubra* and *P. maritima* communities, which form a short turf with occasional grassy tussocks across most of their extent in grazed areas. In areas where livestock grazing is limited or absent, *E. repens* dominates, mainly through the central and upper parts of the saltmarsh extending on to the vegetated flood defences. Fieldwork was carried out between May and July 2012.

2.2. Grazing data

Grazing intensity was quantified as the annual cattle density over the 12 month period preceding the study: annual cattle density = number of cattle × number of months the animals were present/(size of site × 12). Livestock numbers were provided by land managers and had been similar since at least 2007. As grazing induced changes to vegetation develop over a number of years, annual cattle density is a better reflection of grazing induced changes to the breeding habitat than the number of livestock present at the time the nests were active (Andresen et al., 1990). For saltmarsh C (Fig. 1), a large network of creeks prevented grazer access to part of the saltmarsh, therefore only the area accessible to livestock was sampled and included in the annual cattle density calculations. As all of the grazed saltmarshes were stocked with young beef cattle, each individual was equivalent to 0.8 livestock units (DEFRA, 2010).

2.3. Nest finding

In May and June 2012, a total of 45 Redshank nests were found across the six saltmarshes (between 5 and 10 nests per marsh). These were found by systematic searches in which all parts of the study saltmarshes were walked to within 50 m, observing adults flushed from their concealed nests to ensure consistency of nest discovery across all vegetation types. As we were concerned that nests in areas of longer vegetation may be missed, which could bias our estimate of the effect of grazing, we noted if we had flushed an adult bird but were not then able to find a nest. This occurred only twice, and on each occasion the nest was found upon completing a subsequent search. Nest searching occurred throughout May and June, on weekly visits to each of the six saltmarshes. To avoid unnecessary disturbance, all nest vegetation measurements were made as soon as the nest had been recorded as either hatched or failed, which ranged from 8th May to 9th July, when the last nest hatched. To account for sampling vegetation at nests of unequal age, the age of the nest was estimated by weighing and measuring the eggs following the methods of Green (1984).

Table 1
Description of saltmarsh field sites and numbers of nests found per saltmarsh. In the case of saltmarsh C, a large network of creeks prevented grazer access to part of the saltmarsh, so only the area accessible to livestock was included in this study and the annual cattle density calculations.

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

2.4. Vegetation height

To investigate if Redshank select taller vegetation at, next to and in the wider area around the nest, but grazing limits the availability of these conditions (hypothesis 1), we recorded the mean and maximum vegetation height at the nest and the mean vegetation height at both ≤1 m and ≤10 m from nest. The mean vegetation height at the nest was calculated from 3 measurements taken at random directly at each nest location. The maximum vegetation height was the height of the tallest single strand of vegetation at the nest. Vegetation height ≤1 m from the nest was a mean of 5 measurements taken at random within this area. Vegetation height ≤10 m from the nest was a mean of 15 measurements taken at random within this larger area. To compare vegetation at the nest with non-nest areas, for each of the nests studied, a control point was selected and sampled on the same day. To investigate selection on a small spatial scale, control points were within 80–100 m of the nest (mean distance between nests and control points = 91 m). Control points were selected by generating a random number (between 80 and 100) and a random compass bearing (0–360°). All nest vegetation height measurements were duplicated at control points. As Redshank build their nests in vegetation, control points that fell within a creek were reselected at random, to ensure only parts of a saltmarsh that had the potential to be vegetated were included. Vegetation heights were recorded as direct measurements as studies have shown that this gives more consistency and accuracy than sward sticks or drop disks (Stewart et al., 2001).

A series of General Linear Mixed Models (GLMMs) were run using the 'lme4' package in the statistical program, R (Kubetzki and Garthe, 2007) (Table 2, analyses i–iv). Measurements of vegetation height (at different spatial scales) were used as the response variable for each model set. In all cases, the predictors were annual cattle density, nest presence (nest or control), age of the nest when measured (days, to control for effects of measuring nests of unequal age on vegetation height) and day measured (day of study, to control for measuring nests at different times in the season). An interaction between annual cattle density and nest presence was added to test if annual cattle density affected the differences in vegetation height between nests and control points. In all cases, the response variable was square root transformed to ensure normality of residuals. Saltmarsh (A–F) was included as a random effect to account for spatial pseudo-replication caused by having multiple nests within each saltmarsh. An additional random effect of Pair ID was included (with each pair of nest and control points given a unique code). Model outputs were compared using the Akaike Information Criterion (Tables 2 & 4).

2.5. Vegetation species composition

To determine if vegetation species composition varies between nests and control locations (Hypothesis 2), percentage cover per plant species was estimated by eye (Kent, 2011), at both ≤ 1 m and ≤ 10 m proximity to nests and control points. Sampling was carried out in a quadrat of 1 m^2 around each nest and control point, and also sampled in five 1 m^2 quadrats randomly selected within 10 m of each nest. To maintain the same spatial autocorrelation structure for nests and control points, we also sampled in five

Table 2

Results of vegetation height GLMM analyses. Models are ranked by their AIC_c. 'ΔAIC_c': the difference between the model in question, and the top model. $\Delta AIC_c < 2$ suggests both models are plausible (Burnham and Anderson, 2002). Akaike weights measure the support for each model, and represent the ratio of ΔAIC_c values for each model relative to the candidate set. ACD = annual cattle density. Nest presence = nest or control point. Age = age of nest when vegetation height was recorded. This Table shows only those with $\Delta AIC_c < 2$ and the intercept. Full model results are reported in SOM Table 2.

Analysis	Model	AIC_c	ΔAIC_c	Akaike weight
(i) Mean vegetation height at nest	ACD + nest presence	335.20	0.00	0.22
	ACD + nest presence + day measured	335.60	0.32	0.19
	ACD × nest presence	337.00	1.73	0.09
	nest presence	337.20	1.92	0.09
	intercept	355.10	19.81	0.00
(ii) Maximum vegetation height at nest	ACD + nest presence	346.30	0.00	0.28
	ACD + nest presence + day measured	346.50	0.15	0.26
	intercept	387.90	41.63	0.00
(iii) Mean immediate vicinity vegetation height (1m)	ACD + nest presence	297.10	0.00	0.31
	ACD + nest presence + day measured	299.00	1.83	0.12
	nest presence	299.10	1.97	0.11
	intercept	336.20	39.03	0.00
(iv) Mean wider area vegetation height (10 m)	ACD + nest presence	299.60	0.00	0.33
	ACD + nest presence + age	300.40	0.78	0.22
	intercept	315.50	15.89	0.00

1 m^2 quadrats randomly selected within 10 m of each control point. There was no significant difference between the two sets of control points ($p = 0.84$). Permutational Multivariate Analysis of Variance (PERMANOVA) was used to test if vegetation composition differed between the nests and control locations; posthoc, similarity percentages (SIMPER) assessed the contribution of different species to the clustering of samples (Clarke and Gorley, 2006). PERMANOVA is akin to ANOVA, but uses a permutation procedure to assess significance, and thus does not rely on the assumption of multivariate normality, which is frequently violated by ecological data (Anderson, 2005). A modified Gower resemblance matrix was used with fourth root transformation following Anderson et al. (2006) as this is suitable for species abundance data, such as percentage cover.

To investigate the effects of grazing on the proportion of *E. repens* and *F. rubra* found near Redshank nests (Hypothesis 3), we used the percentage cover of these species as the response variables in a series of Generalised Linear Mixed Models with binomial error (Table 4, analyses i & ii). Predictor variables included annual cattle density and proximity (levels ≤ 1 m or ≤ 10 m of nest) to account for effects of surrounding area vegetation type. To deal with the large number of zeros in the *E. repens* data, a zero-inflated binomial model was used. An interaction was added to test if the effect of annual cattle density on percentage cover of each species varied between proximity levels. All percentage cover models were run using the package 'glmmADMB' in R (Fournier et al., 2011). In all analyses, saltmarsh (A–F) was included as a random effect to account for spatial pseudo-replication caused by having multiple nests within each saltmarsh.

3. Results

3.1. Vegetation height

Vegetation height was taller at, next to, and in the wider area around nests compared to control points for all spatial scales studied (Table 2, SOM Tables 1 & 2; mean nest vegetation height: Fig. 2i, $\Delta AIC = 17.9$ upon removal of nest presence from the model; maximum nest vegetation height: Fig. 2ii, $\Delta AIC = 37.8$ upon removal of nest presence; vegetation height in the immediate vicinity of nests: Fig. 2iii, $\Delta AIC = 37.0$ upon removal of nest presence; vegetation height in the wider area around nests: Fig. 2iv, $\Delta AIC = 10.5$ upon removal of nest presence). Vegetation height at, next to, and in the wider area around nests and control points decreased in saltmarshes with higher annual cattle density for all spatial scales studied (SOM Table 3; Fig. 2; mean: $\Delta AIC = 2.0$ upon removal of the annual cattle density variable from the model; maximum: $\Delta AIC = 3.8$ upon removal of the annual cattle density variable from the model; immediate vicinity:

Table 4

Results of percentage species cover analyses. 'ΔAIC_c': the difference between the model in question, and the top model. $\Delta AIC_c < 2$ suggests both models are plausible (Burnham and Anderson, 2002). Akaike weights measure the support for each model, and represent the ratio of ΔAIC_c values for each model relative to the candidate set. ACD = annual cattle density. Proximity = proximity to the nest (levels ≤ 1 m and ≤ 10m).

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

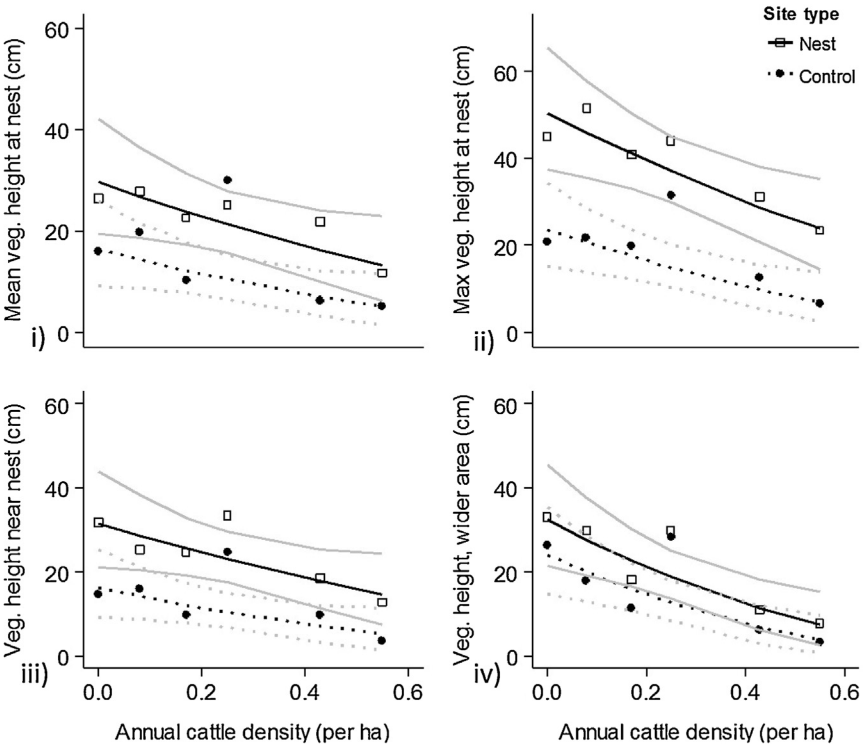

Fig. 2. Effect of grazing on saltmarsh vegetation height and Redshank nest selection, showing difference between nests and control points. Top = vegetation height at nest and control points, mean (i) and maximum (ii). Bottom = vegetation height (iii) in the immediate vicinity (≤1 m) of Redshank nests and control points (iv) in the wider area (≤10 m) around Redshank nests and control points. Black lines are back-transformed model fitted lines (best fitting model), while grey lines are 95% confidence intervals.

ΔAIC = 2.0 upon removal of the annual cattle density variable from the model; wider area: ΔAIC = 5.3 upon removal of the annual cattle density variable from the model). There was no clear support for an interaction between annual cattle density and nest presence demonstrating that the nature of the relationship was similar for vegetation height at both nests and control points (mean: ΔAIC = +1.7; maximum: ΔAIC = +2.3 upon addition of the interaction; immediate vicinity: ΔAIC = +2.2; wider area: ΔAIC = +2.3 upon addition of the interaction). There was no clear need for the Age of nests when recorded (Table 2; mean: ΔAIC = +2.1; maximum: ΔAIC = +2.3; immediate vicinity: ΔAIC = +2.0; wider area: ΔAIC = +0.8 upon addition of age). There was no clear need for the day of nest measurement (Table 2; mean: ΔAIC = +0.3; maximum: ΔAIC = +0.1; immediate vicinity: ΔAIC = +1.8; wider area: ΔAIC = +2.3 upon addition of age).

3.2. Vegetation species composition

The vegetation composition was different in the immediate vicinity and in the wider area around nests than at control points, indicating that Redshank select nest sites surrounded by particular species of vegetation. Most of the dissimilarity between nests and control points was due to *F. rubra*, which was more abundant near nests than control points (immediate vicinity of nests: Table 3,

Table 3
Average abundance of dominant species at Redshank nests and control points based on SIMPER analysis. The abundance is the average% cover found at Redshank nests or control locations. The average dissimilarity is the contribution of each taxa towards dissimilarity between nests and control points. Dissimilarity is the average Bray Curtis contribution/standard deviation of the contribution over groups (i.e., consistency). The percentage contribution shows the average contributions of each species to the overall dissimilarity. Prox. level = proximity level, meaning distance from the nest or control point (levels ≤ 1m, ≤ 10m).

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

analysis i, $p < 0.01$; wider area: Table 3, analysis ii, $p < 0.01$). *P. maritima* was more abundant near control points than nests. Trends for *E. repens* were not as clear, although this species contributed to the dissimilarity between the vegetation communities, the abundance was similar near nests and control points.

Both annual cattle density and proximity were found to affect the percentage cover of *F. rubra*. The cover of *F. rubra* increased with increasing annual cattle density within 1 m and 10 m of nests (Fig. 3i; Table 4i, ΔAIC = 3.9 on removal of annual cattle density from the model). The percentage cover of *F. rubra* ranged from 71 ± 30% within 1 m of nests to 63 ± 26% within 10 m of nests (ΔAIC = 176.6 upon removal of proximity from the model). While there was an effect of proximity on the% cover of *E. repens* (Fig. 3ii, Table 4ii, ΔAIC = 10.7 on removal of proximity from the model), there was no clear evidence of a decline in *E. repens* cover with increasing annual cattle density (ΔAIC = +2.1 on addition of annual cattle density to the model).

4. Discussion

4.1. Nest site selection and grazing

Our results support hypotheses one, as Redshank nest vegetation was higher at, next to and in the wider area around nests than control locations but grazing limited the availability of these conditions. Vegetation height at, next to and in the wider area around Redshank nests was lower in saltmarshes with higher livestock densities. Although trends were similar next to, and in the wider area around nests, the difference between nests and control points was weaker for the wider area around nests. This could be due to the patchy nature of cattle grazing on saltmarshes (Jensen, 1985) as Redshank appear to choose the longest vegetation for their nests, with some cover in the wider area around the nests (Smart et al., 2006). This therefore suggests that for future Redshank nest vegetation studies, recording nest vegetation at the nest, and in the wider area ≤ 10 m of the nest provides useful information. Results add support to the findings of Mandema et al. (2014a), that Redshank build their nests in areas with relatively tall vegetation and high variation in canopy height which, in their study was more abundant in low livestock densities of 0.25 cattle ha^{-1} y^{-1} than at 0.50 cattle ha^{-1} y^{-1}.

Our results partially support hypothesis two as more *F. rubra* was found at Redshank nests, however slightly more *P. maritima* was found at control locations than Redshank nests and the cover of *E. repens* was similar between nests and controls. This suggests that Redshank select nests within communities of *F. rubra*, which is a species of high-mid saltmarsh (Adam, 1990). *P. maritima* is associated with low and low-mid saltmarsh (Adam, 1990; Boorman, 2003) and was more abundant at control sites than nests, suggesting Redshank are selecting grassy areas at higher elevation saltmarsh. These findings are consistent with those of Thyen and Exo (2003), who found that although Redshank can nest amongst *P. maritima*, more nests were found amongst *F. rubra*. Our results suggest that Redshank on the Ribble estuary were not selecting for *E. repens*. However 45% of the 83 nests recorded by Thyen and Exo (2003) were in *E. repens*. This difference could be due to grazing management, as 70% of the saltmarsh in their study was ungrazed, and *E. repens* is more abundant on ungrazed saltmarshes (Andresen et al., 1990).

Our results partially support hypothesis three, as nests exposed to higher livestock densities had a higher percentage cover of *F. rubra*. However, annual cattle density did not affect the percentage cover of *E. repens* found at Redshank nests. This suggests that Redshank are able to select more nest sites in *F. rubra* in higher livestock densities. Our results did suggest that more *E. repens* cover was found in the wider area around the nest than closer to the nest. Although overlapping confidence intervals suggest that differences between the cover of both *E. repens* and *F. rubra* at different proximities from the nest are only minor. It is interesting that there was no apparent effect of grazing intensity on the percentage cover of *E. repens*. This suggests that the increase in *F. rubra* with grazing intensity must have been at the expense of another vegetation type. However, as we found that there was no difference in the% cover of *E. repens* between nests and control points on these saltmarshes this suggests a trade-off for Redshank between selecting nests with ideal vegetation height in the saltmarshes with the lowest annual cattle density and their preferred vegetation type being more prevalent in saltmarshes with a higher annual cattle density.

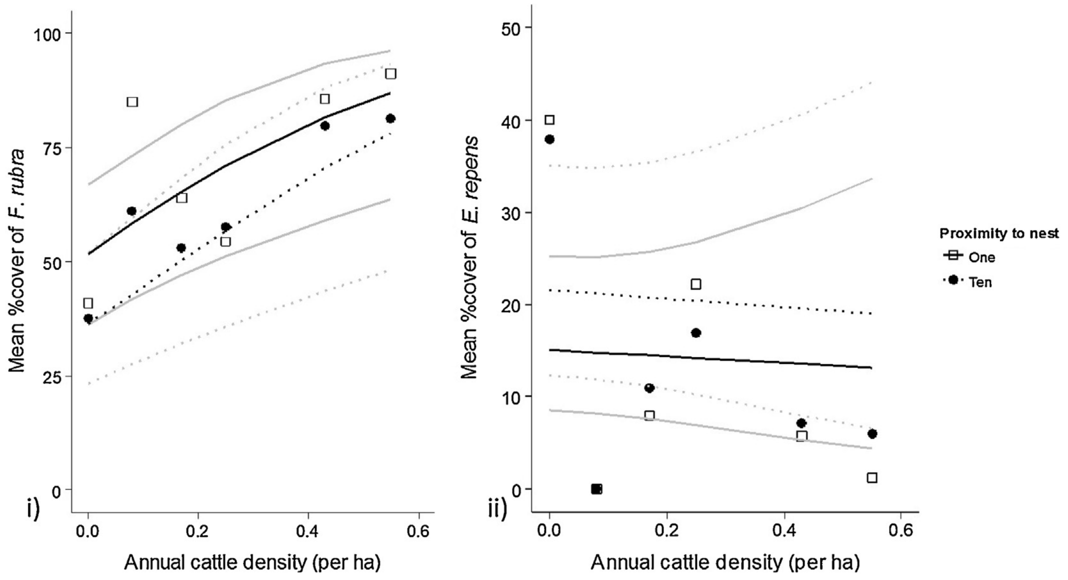

Fig. 3. Effect of grazing and proximity to the nest (levels ≤ 1 m and ≤ 10m) on percentage cover of (i) *Festuca rubra* (ii) *Elymus repens*. Black lines are back-transformed model fitted lines (best fitting model), while grey lines are 95% confidence intervals.

4.2. Habitat quality

This study suggests that livestock grazing can have a negative effect on Redshank: birds prefer longer vegetation but are forced to breed in shorter vegetation for nest sites on grazed saltmarshes. *F. rubra* is more abundant on grazed saltmarshes but can lead to lower rates of nest survival (Andresen et al., 1990; Thyen and Exo, 2003). Even in lightly grazed saltmarshes, grazing induced nest mortality is incurred and can arise indirectly through grazer modification of landscapes that accelerate predation risks (Sharps et al., 2015). Results of the current study may explain why more nest predation occurs at higher livestock densities as found by Sharps et al. (2015), as nests in shorter vegetation are more obvious to predators. It is likely that nests in short *F. rubra* are more vulnerable to both ground and aerial predators than those in longer vegetation (Thyen and Exo, 2003; Maier, 2014). This places emphasis on optimising grazing to benefit nesting Redshank. However, dense swards of *E. repens* caused by grazing abandonment can reduce the number of breeding Redshank (Norris et al., 1997; Norris et al., 1998) and there may be differences in vegetation structure between saltmarshes which have never been grazed and those formerly grazed saltmarshes which have been abandoned (Adnitt et al., 2007).

4.3. Limitations

Age of nest when measured ranged between 3 and 24 days (mean = 16, ±7 sd) and the dates measured ranged from day 7 to day 69 of the study (mean = 38, ±17 sd). Therefore it is possible that nests could have either grown or been grazed after Redshank nest selection, but before they were measured. This could have resulted in recording of vegetation heights that were different from when the bird started nesting. However, this was accounted for by taking a paired approach of recording vegetation at nests and control sites, and the inclusion of both nest age and date measured as predictors in all vegetation height models. Measuring nests of unequal age or at different times in the season did not affect our conclusions as results showed that neither the age variable, or the date measured variable were not included in the best models.

This study investigated small scale selection of nesting habitat, therefore did not consider trends on a wider landscape scale. Breeding distribution changes between 1968 and 2011 (Balmer et al., 2007) suggest that Redshank populations are generally declining across the UK, but that there are a small number of increases in the uplands of northern England, the river systems of the English midlands and the highlands and islands of Scotland. However, as Redshank show both high breeding site fidelity and natal philopatry (Thompson and Hale, 1989) it is unlikely that this represents wide scale habitat selection, and more likely to suggest that areas with increasing populations are due to higher breeding success. As this study was based on six saltmarshes of the Ribble estuary, it is not clear if these trends can be generalised and applied to other locations. However, it is likely that similar trends would be found on other north western European saltmarshes subject to grazing, as the climax community is usually dominated by *Elymus* spp. such as *E. repens* (Rodwell et al., 2000; Bakker et al., 2003). Therefore, our conclusions could potentially be applied to other European saltmarshes.

Our analysis of the vegetation composition showed that *E. repens* was a contributor to the dissimilarity between nest and control sites, but this difference is not obvious from the actual abundance data. This discrepancy may be the result of a large number of either zero values or high values (close to 100%) in the *E. repens* percentage cover data. This could be explained by the patchy nature of *E. repens* distribution on low intensity cattle grazed saltmarshes (Andresen et al., 1990). However, overall, the conclusion that Redshank select *F. rubra*, and that% cover of this species increases with grazing density is unaffected by uncertainty over *E. repens* results.

4.4. Implications

The results of this study suggest that livestock grazing plays an important role in creating the *F. rubra* nesting habitat preferred by Redshank, but that even low intensity conservation grazing can create a shorter than ideal sward height which could leave Redshank nests more vulnerable to predators. Complete abandonment of previously grazed saltmarshes is therefore unlikely to help Redshank conservation. Translating the UK Environment Agency light grazing guidelines of 0.7–1 young cattle ha^{-1} between April and October (Adnitt et al., 2007) to measurements used in this study would mean an annual cattle density of around 0.4–0.5 $ha^{-1} y^{-1}$. The saltmarsh with the highest annual cattle density in this study was 0.55 cattle ha^{-1}. This suggests that the UK Environment Agency definition of light grazing is too intensive for breeding Redshank. Stopping grazing altogether would reduce the availability of *F. rubra*, so solutions should focus on designing a grazing regime which increases sward heights for Redshank nesting. This could include delaying the start of grazing until most Redshank stop nesting in mid-July, but then grazing more intensively afterwards. This would allow vegetation to remain tall during the nesting season, but would still maintain a cover of *F. rubra* due to the longer term effects of grazing on vegetation. Alternatively, implementing a rotational grazing system whereby saltmarshes are grazed and left ungrazed in alternate years may improve habitat quality by allowing the vegetation grow taller in the ungrazed year. However, on a small scale Redshank may then select the part of a saltmarsh that was ungrazed in the previous year, and therefore the area that will be grazed whilst breeding occurs. Therefore, this could be carried out on a large scale, for example no grazing across a whole estuary every second year. Further investigations into the ideal saltmarsh management conditions for Redshank may be necessary. Whilst saltmarshes in the UK do not hold significant populations of other nesting wader species (Allport et al., 1986), in other parts of Europe they support important populations of Black tailed godwit *Limosa Limosa* (Møller, 1975) and Oystercatcher *Haematopus ostralegus* (Mandema et al., 2014a) which given the similar ecology and timing of nesting could be comparably affected by saltmarsh grazing. It may also be appropriate to consider the role that saltmarshes play in the lifecycle of various other bird species with contrasting needs to Redshank. These include passerines, which generally require both short and long vegetation, and over-wintering wildfowl, which require shorter sward heights (Mandema et al., 2014b; van Klink et al., 2014).

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2016.01.030.

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