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The effects of fieldwork on Hen Harrier *Circus cyaneus* breeding success

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While information on the ecology of endangered species is essential to inform conservation measures, the acquisition of the necessary scientific data during the breeding season involves field methods that could potentially have a negative impact on the study species. Studies of the impact of fieldwork on the breeding success of endangered species make an important contribution to the development of field methodologies that have a minimal impact on species and ecosystems. Hen Harriers *Circus cyaneus* are one of the most vulnerable bird species in Ireland at present and also act as a flagship species of upland habitats, which are under considerable pressure from human activities. In this study, which used data from five breeding seasons, we investigated the effects of nest visits for the purpose of data collection on the breeding success of Hen Harriers. Success rates were compared between groups of nests at which different types of fieldwork were carried out, including: remote observations only (no visits); nest visits; and nest visits with camera deployment. No negative effect of nest visits on breeding success was observed. At visited nests, the additional deployment of nest cameras had no apparent effect on nesting success. These findings show that fieldwork during this study did not have a negative impact on overall Hen Harrier breeding success. The absence of a negative effect of fieldwork should be considered in the context of the study, which involved highly trained, experienced staff adhering to detailed fieldwork protocols that ensured that the welfare of birds and their nests was the main priority.

Introduction

Data collected on different aspects of the ecology of endangered species are essential to improve our knowledge of these species' ecological requirements and to inform effective conservation actions (Bird & Bildstein 2007, Hardey *et al.* 2013). However, data collection in these studies may sometimes involve field methods that could potentially have a negative impact on the species under study, particularly as they are conducted during the breeding season (Fletcher *et al.* 2005, Rosenfield *et al.* 2007). The conservation of these vulnerable species and the ethical issues of environmental research and data collection must be given due consideration

Plate 234. Hen Harrier (Richard T. Mills).

in the design of ecological fieldwork. In this context, research on the impact of these studies themselves on the study species are invaluable so that scientists can ensure that their research does not have a negative impact on species or ecosystems (Costello *et al.* 2016).

Hen Harriers *Circus cyaneus* are birds of particular conservation concern in Ireland (Colhoun & Cummins 2013) and throughout much of their range (Burfield & von Bommel 2004). Under the EU Birds Directive all member states are required to take measures to ensure the survival of Hen Harriers at a favourable conservation status. The advancement of scientific knowledge and the development of effective conservation management plans rely on relevant, up-to-date, evidence-based scientific research (Bird & Bildstein 2007, Lerner 2009), which in turn rely on data collected in the Hen Harrier's natural habitat (Irwin *et al.* 2008, 2011). In this way, research will yield results that are directly relevant to the welfare and conservation of this species (Bird & Bildstein 2007, Ibáñez-Álamo *et al.* 2012). It is important that the fieldwork carried out to collect these data impacts minimally on the fitness of individual birds and on the viability and conservation status of Hen Harrier populations. To this end, negative impacts on breeding success must be avoided (Rosenfield *et al.* 2007, Hardey *et al.* 2013).

The collection of data on Hen Harrier breeding biology involves locating nest sites to gather data on breeding success. One or more visits may be required over the course of a breeding season to determine the basic information on breeding biology and productivity, and to determine the outcome of the breeding attempt (Bird & Bildstein 2007, Lerner 2009, Hardey *et al.* 2013). Much attention has been paid by ornithological researchers to the potential for their activities to negatively impact on their study species (Willis 1973, Major 1990, Götmark 1992, Fair *et al.* 2010, Uher-Koch *et al.* 2015, Smallwood 2016). Despite some variation both between and within species in how individuals respond to nest visits (Fair *et al.* 2010), raptors as a group are less susceptible to disturbance associated with research activity than many other bird groups (Götmark 1992). Evidence suggests that nest visits for research purposes do not negatively impact on the nest survival of a number of species investigated (Ibáñez-Álamo *et al.* 2012), particularly where nests visits are infrequent (MacIvor *et al.* 1990, Major 1990, Fletcher *et al.* 2005, Fair *et al.* 2010). However, no studies have yet been undertaken specifically on Hen Harriers.

However, there are a number of ways in which the activities of researchers involved in data collection during the breeding season could have adverse impacts. Possible impacts include nest desertion, egg or chick mortality, premature fledging, increased predation risk, decreased nest attendance and injury to handled birds, all of which can negatively impact on breeding success (Blackmer *et al.* 2004, Fletcher *et al.* 2005, Rosenfield *et al.* 2007, Ibáñez-Álamo *et al.* 2012). Increased predation risk is a particularly important consideration when working with ground nesting birds such as the Hen Harrier (Willis 1973, Major 1990, Hannon *et al.* 1993). Given the vulnerable status of Hen Harriers across their range, and the continued need for fieldwork on this species, it is important to gather information specific to this species in order that the need for further research can be balanced with ensuring that disturbance pressures on its small and declining population are minimized.

In order to ensure that sufficient data are collected from nests while minimizing the number of nest visits, several researchers have opted to use nest cameras to record breeding biology and predator data (Bolton *et al.* 2007). Nest cameras have been associated with both negative (Pietz & Granfors 2000) and positive (Herranz *et al.* 2002) impacts on nest success through their effect on nest predation rates (Richardson *et al.* 2009, Humphreys *et al.* 2012). Therefore, to minimize negative impacts of fieldwork on the survival or reproductive success of avian study populations, we must evaluate the effects of fieldwork on nest survival. This study set out to examine the effect of fieldwork on the breeding ecology of Hen Harriers. These data were collected as part of a larger, long-term study investigating optimum scenarios for Hen Harrier conservation in Ireland (Irwin *et al.* 2012). This project involved visits to nests by fieldworkers under license over a five-year period, during which information was collected on breeding biology, foraging behaviour and nestling development. In order to facilitate data collection, cameras were deployed at a subset of nests. This study investigates the effects of nest visits on the breeding success of Hen Harriers by comparing the success of visited and unvisited nests.

Methods

As part of a five-year study on Hen Harrier conservation in Ireland (Irwin *et al.* 2008, 2011, Wilson *et al.* 2009, 2010, 2012), data on breeding Hen Harriers were collected at four study sites (Slieve Aughty Mountains, West Clare, Kerry and Ballyhoura Mountains) in the south of Ireland between 2007 and 2011. During this time 173 nests were monitored in order to collect information on breeding ecology, including first egg-laying and fledging dates, clutch size, brood size, nest success and productivity. Nest visits were undertaken at a subset of these nests (under license from National Parks and Wildlife Service) to gather breeding biology data where appropriate. The success of all visited nests was recorded, and unvisited nests were recorded as successful where recently-fledged juveniles were observed flying in the nesting area (Watson 1977). Wherever possible, stage of failure at unvisited nests was determined during a post-failure nest visit.

Nests that were least vulnerable to disturbance associated with visits and that were the most accessible in terms of fieldwork logistics were selected for nest visits (n = 103) and all other nests were monitored from a distance (n = 70). Before deciding whether or not to visit each nest, we assessed the risks to that nest deriving from inherent vulnerability of the nest to discovery by humans or predators and the potential effects of visits on nest success. Risk in the former category was assessed in relation to distance from the nearest path, track or road, level of human activity along this path, visibility from this route, and ease of access. Fieldwork-related risk was assessed according to distance from the nearest route, human activity along this route, conspicuousness of observation points, reaction of adult birds to fieldworkers, and the degree to which ease of access to the nest was altered by fieldworker activity. Where any of these factors were deemed to pose a high risk to nest success, the nests were not visited. Where nests were deemed suitable for visits based on these criteria, only those that were suitable in terms of fieldwork logistics were selected. The factors considered in this assessment were costs in terms of fieldwork time and ease of access to the nests, with difficult terrain and long cable runs being avoided.

Nests selected for visits were visited between one and four times while they were active. Nests were not visited until one week after incubation had begun in order to be certain that egg-laying was complete. This was determined by the behaviour of the female following food-passes (Hardey *et al.* 2013). The first visit to a nest was made only when we were confident that the nest location had been identified to within an accuracy of about 10 m. Nest visits were typically made by two or three people, one of these acting as a distant observer using a telescope and hand-held radio to guide the others to the location of the nest. However, in situations where the location of the nest was readily identifiable 'on the ground', nest visits were sometimes made by a single, unaccompanied fieldworker.

The first priority during fieldwork was the welfare of the birds, and every effort was made to avoid distress or loss of the nest as a result of visits. Fieldworkers stayed at nests no longer than was necessary to carry out the required actions. On first visiting a nest, fieldworkers recorded its position using a GPS, and took a photograph of the nest contents. At 25 nests, discreet 'bullet' cameras were deployed (see Irwin *et al.* (2012) for methodological details) to record activity at the nest. Real-time footage of the nest could be viewed at a base station 50-300 m away. Camera deployment precluded the need for further visits to assess progress of the clutch or brood. Visits to nests containing eggs or young chicks were not made in wet or cold weather or during the early morning, to ensure that any temporary avoidance of the nest by the female during and after the visit did not expose the nest contents to inclement conditions. After visiting a nest, observers attempted to replace any vegetation disturbed on approach to ensure that access to the nest was no easier than prior to the visit.

Statistical analyses were carried out in R version 2.13.1 (R Core Project Development). Binomial generalized linear models (GLMs) with logistic-exposure link function (Schaffer 2004) were carried out using the GLM function in the MASS package. This technique allows for testing of the effects of fieldwork activities on nest success (the probability of nests successfully fledging at least one chick) while controlling for the number of observation days at each nest (Schaffer 2004). As well as the two research-related variables 'visited' and 'camera', we also tested for effects of 'year' and 'region' on nest success using GLM. Final models were selected by backwards selection performed from fully specified models including all explanatory variables, until no terms could be removed from the model without incurring an increase in AIC (Akaike's Information Criterion). Statistically significant differences between levels of factors with more than two levels were identified using Tukey post-hoc comparison tests, carried out using the GLHT function in the Multcomp package.

Results

Of a total of 173 nests monitored across the four study areas between 2007 and 2011, 103 were visited and 70 were monitored from a distance. Of the 103 nests that were visited, 66 fledged young successfully and 37 failed to fledge any young. Fifteen of the 25 nests where cameras were deployed fledged young successfully while ten failed to fledge any young. Of the 70 nests that were not visited, 27 fledged young successfully and 43 failed to fledge any young. Among all failed nests (80), 23 failed during egg-laying or incubation, 41 failed at the chick stage, and 16 nests (all unvisited) failed at an indeterminate stage.

In a model describing the fledging success of all 173 nests (103 visited, 70 unvisited) monitored during this project, visit status and study area were both retained in the final model with year being excluded (Table 1). Post-hoc tests showed that success of nests was higher in West Clare than in the Slieve Aughty Mountains ($z = 3.15$, $P = 0.001$) and higher at visited than at unvisited nests ($z = 3.85$, $P = 0.0001$). Nest success was significantly higher at visited nests than at nests where no visits were carried out. In order to control for biases related to nests failing before they could be visited, we conducted the analyses again using only the 136 nests (91 visited and 45 unvisited) that had survived beyond hatching. This revealed that the effect of nest visits on nest survival was marginally non-significant in this case. In common with the model of all 173 nests, the final model retained study area and nest visits

(Table 2), with inter-study area differences lying principally between Area 1 (Slieve Aughty Mountains) and Area 4 (West Clare) ($z = 2.35_{1,56}$, $P = 0.019$). However, the apparent positive influence of nest visits was greatly diminished and marginally non-significant ($z = 1.85_{1,134}$, $P = 0.064$).

In order to investigate the effect of camera deployment, a model describing the fledging success of 103 visited nests (25 with cameras, 78 without) revealed no difference in fledging success between nests with and without cameras. The only term retained in the final model was study area (Table 3). Post-hoc tests showed that success of nests was lower in the Slieve Aughty Mountains than in the Ballyhoura Mountains ($z = 2.70_{1,62}$, $P = 0.007$), Kerry ($z = 2.90_{1,54}$, $P = 0.004$) and West Clare ($z = 2.74_{1,43}$, $P = 0.006$).

Table 1. Summary of a binomial generalized linear model (GLM) describing the apparent effects of study area and nest visits on nest success of 173 Hen Harrier nests*.

Variable	Estimate	SE	*z* value	*P*
Intercept	2.89	0.29	9.91	< 0.0001
Area 2 (Ballyhoura)	0.71	0.34	2.08	0.04
Area 3 (Kerry)	0.53	0.33	1.63	0.10
Area 4 (West Clare)	1.37	0.43	3.15	0.001
Visited	1.00	0.26	3.85	0.0001

*The fully specified model included year, as well as the two factors retained in the final model, study area and nest visits. The effects of one level of each factor (Area1, being the Slieve Aughty Mountains; and Unvisited) are included in the intercept of the model. Null deviance = 238.9_{172}, residual deviance = 255.7_{163}, AIC = 268.6.

Table 2. Summary of a binomial generalized linear model (GLM) describing the apparent effects of study area and nest visits on nest success of 136 Hen Harrier nests that survived beyond hatching*.

Variable	Estimate	SE	*z* value	*P*
Intercept	3.62	0.37	9.84	< 0.0001
Area 2 (Ballyhoura)	0.65	0.42	1.55	0.12
Area 3 (Kerry)	0.65	0.42	1.53	0.13
Area 4 (West Clare)	1.29	0.55	2.35	0.019
Visited	0.63	0.34	1.85	0.064

*The fully specified model included year, as well as the two factors retained in the final model, study area and nest visits. The effects of one level of each factor (Area1, being the Slieve Aughty Mountains; and Unvisited) are included in the intercept of the model. Null deviance = 167.4_{134}, residual deviance = 171.0_{130}, AIC = 181.0.

Table 3. Summary of a binomial generalized linear model (GLM) describing the apparent effects of study area and nest visits on nest success of 103 visited Hen Harrier nests*.

Variable	Estimate	SE	*z* value	*P*
Intercept	3.50	0.27	12.93	< 0.0001
Area 2 (Ballyhoura)	1.10	0.41	2.70	0.007
Area 3 (Kerry)	1.43	0.50	2.90	0.004
Area 4 (West Clare)	2.84	1.04	2.74	0.006

*The fully specified model included year and camera deployment, as well as study area, the only variable included in the final model. The effects of Area1 (Slieve Aughty Mountains) are included in the intercept of the model. Null deviance = 135.5_{102}, residual deviance = 144.8_{99}, AIC = 152.8.

Discussion

The output from our nest success model, including failures at all stages, suggests that breeding success at Hen Harrier nests was not negatively affected by nest visits. Although breeding success was slightly higher at nests that were visited than at nests that were not visited, the observed difference in breeding success was greatly diminished when only nests that survived beyond hatching were considered. This indicates that the apparently higher success of visited nests was due in large part to the fact that many of the nests that failed during laying and incubation did so before it was possible for us to visit them. After nests had been located, fieldworkers did not visit them until they were satisfied that females at these nests had started incubating, in order to minimize the risk of nest abandonment (Dickinson *et al.* 1987, Craik & Titman 2009, Hardey *et al.* 2013). If nests failed during this time, they were unavailable for visits, resulting in the observed bias in the comparison of success rates between visited and unvisited nests.

By conducting the analysis using only nests that had survived beyond hatching we considered only nests that were old enough to be visited, thereby greatly diminishing the potential for failure before nests were visited, and biasing our assessments. However, visited nests still had a slightly higher rate of survival than unvisited nests, though this difference was not statistically significant. It is possible that at some nests with chicks visits were delayed due to uncertainty of nest stage. However, this is unlikely to have been the case at many nests, as the majority (50 out of 61) of nests that were visited when they had chicks were originally discovered before hatching. Furthermore, changes in female behaviour make it possible to determine when Hen Harrier clutches have hatched, making it unnecessary for fieldworkers to delay the first visit to a nest.

It is also likely that our strict criteria for selecting nests for fieldwork resulted in our not visiting the nests that were most vulnerable, and which may have been either positively or negatively affected by disturbance, and our findings are presented in this context. While an ideal experimental design would have been to randomly assign nests to each of the categories, this is not possible with scarce and vulnerable bird species. We also cannot rule out the possibility that nest visits may, in some instances, have a positive effect on survival. Positive effects of nest visits on rates of nest success have been previously described in several studies (Leighton *et al.* 2010, Ibáñez-Álamo *et al.* 2012). Such effects appear to be most commonly realized through avoidance of, or reduced activity in, nest areas by predators in response to fieldworker presence and activity.

Among visited nests in this study, we found no difference in breeding success between nests with and without nest-cameras. Previous reviews of daily nest survival rates at nests with and without cameras have shown that, on balance, cameras may have a positive effect on survival by acting as a deterrent to potential nest predators (Richardson *et al.* 2009). This may occur where predators associate signs of human activity with danger and therefore avoid the nest (Picman & Schriml 1994). However, our findings indicate that the deployment of discreet 'bullet' cameras at Hen Harriers nests did not significantly affect predation rates either positively or negatively. This suggests that data on nest survival rates collected from nests using cameras may be directly comparable to data collected using alternative methods, which is a concern in the interpretation of camera-derived data (Bolton *et al.* 2007).

In this five-year study of possible effects of nest visits and nest camera deployment we found no measurable disturbance effect of research activity across the island of Ireland on Hen Harrier breeding outcomes. The absence of a negative effect of fieldwork should be considered in the context of the study, which involved highly trained, experienced staff adhering to carefully devised fieldwork protocols that ensured that the welfare of birds and their nests was the main priority, and every effort was made to minimize disturbance associated with research activity. The implications of our findings are that, if nests are selected appropriately and fieldwork methods are strictly standardized to minimize negative effects, Hen Harriers in Ireland are sufficiently robust to disturbance to allow visits to nests without noticeable negative impacts on their breeding success. Previous work on other ground-nesting species has arrived at similar conclusions (O'Grady *et al.* 1995, Lloyd *et al.* 2000, Verboven *et al.* 2001, Ibáñez-Álamo *et al.* 2012). Regulators of fieldwork on protected species such as Hen Harriers should be careful to minimize risks to these species. This will require weighing the likelihood of negative impacts on individuals resulting from fieldwork against the potential for positive effects on the wider population due to information that can be used to improve conservation management strategies.

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Migration phenology of Jack Snipe *Lymnocryptes minimus* at an Irish coastal wetland

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Plate 235. Jack Snipe (Tony Hisgett, Wiki Commons).

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Migration times of Jack Snipe *Lymnocryptes minimus* were monitored at North Bull Island in Dublin Bay during 2011/2012 to 2016/2017. Average arrival times in autumn centred on 2 October and average departure times in spring on 23 April. Although these results were site and habitat specific, they were similar to recent migration data for Ireland. While the time series examined for Ireland and Britain were of different lengths, migration times were extraordinarily similar. The average autumn arrival date for Ireland as a whole was 16 September while that for Britain was 23 September, and departure times in spring for Ireland centred on 30 April, one day later than in Britain. The close agreement suggests that migration times across both islands possibly occur synchronously. Other recently generated data for Ireland provides tantalising evidence that passage migration may take place and that Jack Snipe could be more frequent in upland areas than previously suspected. In both instances greater clarity will only be possible through increased observer effort and higher detection rates of this enigmatic species.

Introduction

Jack Snipe *Lymnocryptes minimus* are difficult to detect in winter largely due to their solitary behaviour, nocturnal or crepuscular habits, cryptic colouration and reluctance to take to the wing when disturbed. Therefore, it is not surprising that there are no reliable estimates of the winter population in Ireland (Crowe *et al.* 2008). The source of information on their migration phenology has changed little for over a century, as most data were, and still are, generated by hunters or through casual observations by birdwatchers (Thompson 1850, Ussher & Warren 1900, *Irish Birding* 2017). In autumn, earliest migrants are thought to arrive in Ireland during October and November with a return passage in April (Lack 1986, Hutchinson 1989). There is evidence from ringing recoveries that autumn passage migration takes place in Britain, but not in Ireland (Smiddy 2002). In winter, Jack Snipe appear to be widely but thinly distributed across much of Ireland with highest densities in counties along the west coast (Balmer *et al.* 2013). They are mainly reported from lowland habitats with low sparse vegetation, muddy substrates and fresh or brackish water (Olivier 2008).

Most wader migration times and population trends are relatively easy to monitor as the species are gregarious and are easy to locate and record. Monitoring data have demonstrated that many species are altering the timing of their migrations, and their breeding and wintering distributional ranges, in response to changing climatic conditions (Rehfisch *et al.* 2004, MacLean *et al.* 2008, Godet *et al.* 2011).

Unfortunately, due to their secretive nature, no such data exist for Jack Snipe and therefore their true status in Ireland is unknown. This lack of data is a conservation concern as it is not known if unrecorded population declines are occurring (Smiddy 2002) or if this species is advancing or delaying migration times in response to climate change. The objective of this study was to generate baseline data on arrival and departure times of Jack Snipe at a wetland on the east coast of Ireland. The results are compared to new data for Ireland and published data from Britain. Other aspects of their migrations and occurrences in Ireland are also discussed.

Study area

The study site was a small section of Mediterranean salt meadow *(Juncetalia maritimi)* on North Bull Island (53.3705° N, 6.1440° W) on the northern shore of Dublin Bay, Ireland. This habitat was situated between Atlantic salt meadow *(Glauco-Puccinellietalia maritimae)* and low fixed dunes dominated by Marram Grass *Ammophila arenaria.* The study area was broadly rectangular in shape, approximately 300 m x 40 m, and covered an area of 1.3 ha. Vegetation cover was dominated by Sea Rush *Juncus maritimus* which was thinnest on the margins of the area where small shallow muddy patches and pools were frequent. This particular habitat was selected for this study because it has been known to the author as a regular site for Jack Snipe during migration times for over four decades.

Methods

There are no recommended methods for field recording of migratory Jack Snipe. In this study, earliest arrivals and latest departures were recorded by walking a single line through the site two to four times weekly from late August to mid-October and again from late March to mid-May during 2011/2012 to 2016/2017. This line transect route was walked in a zig-zag manner to maximise coverage. This method was used because it was quick and easy for one observer to complete and is considered suitable for open, uniform or species poor habitats and is efficient in terms of data gathered (Bibby *et al.* 1992). Although it was likely that some birds may have occasionally evaded detection, it is worth noting that Pedersen (1988) demonstrated that most Jack Snipe will flush at a distance of less than 6 m. It was planned that by systematically surveying the site at North Bull Island a pattern would eventually emerge. Surveys were not undertaken during severe weather or extreme high tides.

For comparative purposes, earliest arrival and departure times for Ireland over the same time period were extracted from the *Irish Birding* website (www.irishbirding.com). This popular website reports bird sightings for the entire island of Ireland. A record for a single bird in County Mayo on 2 June 2015 was omitted from the Irish calculations as the mid-summer date suggests it may have been summering in that area. Comparisons were also made with data from Britain spanning a longer period of a total of 34 years (Sparks & Mason 2004). Calendar dates were converted to Julian days (DOY = Day of Year) with the first week of the year starting with Julian day = 1. Data were adjusted for the 2012 and 2016 leap years.

Results

Average arrival times at North Bull Island were centred on 2 October with departure times centred on 23 April (Table 1). The earliest recorded arrival was on 18 September 2012 and latest departure on 8 May 2015. Duration of stay in winter was, on average, 202 days. For the island of Ireland as a whole over the same time period, the average arrival date was earlier, on 16 September, while the average departure date was only seven days later than that on North Bull Island, on 30 April

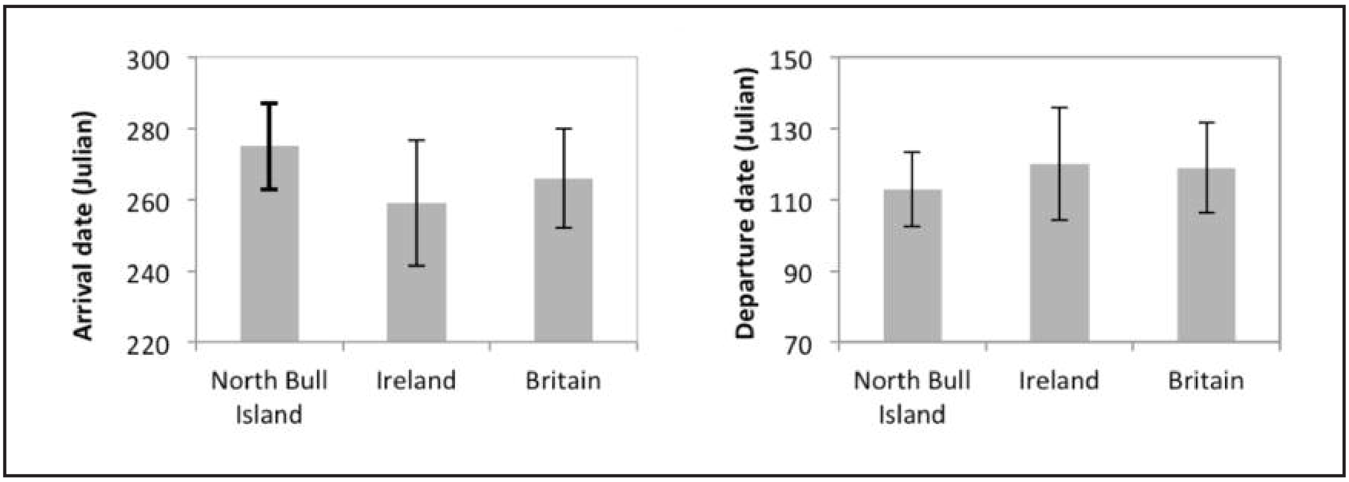

Figure 1. Jack Snipe mean arrival and departure times (+/- sd).

Table 1. Mean dates, standard deviations (sd) of arrivals, departures and duration of stay of Jack Snipe at North Bull Island, 2011/2012 to 2016/2017 (DOY = Day of Year; see Methods).

	Arrival date		Departure date		Duration of stay
Season	**Date**	**DOY**	**Date**	**DOY**	**Days**
2011/2012	30-Sep	273	23-Apr	114	206
2012/2013	18-Sep	262	27-Apr	117	220
2013/2014	18-Oct	291	24-Apr	114	188
2014/2015	15-Oct	288	08-May	128	205
2015/2016	02-Oct	275	15-Apr	106	196
2016/2017	19-Sep	263	07-Apr	97	199
Average	02-Oct	275	23-Apr	113	202
sd		12.2		10.5	10.9

Table 2. Mean dates, standard deviations (sd) of arrivals, departures and duration of stay of Jack Snipe in Ireland 2011/2012 to 2016/2017 (DOY = Day of Year; see Methods).

	Arrival date		Departure date		Duration of stay
Season	**Date**	**DOY**	**Date**	**DOY**	**Days**
2011/2012	08-Oct	281	26-Apr	117	201
2012/2013	19-Sep	263	15-Apr	105	207
2013/2014	08-Sep	251	17-May	137	251
2014/2015	25-Sep	268	08-May	128	225
2015/2016	19-Sep	262	15-Apr	106	209
2016/2017	16-Aug	229	07-May	127	263
Average	16-Sep	259	30-Apr	120	226
sd		17.6		12.9	25.6

Table 3. Mean DOY, standard deviations (sd) of arrivals, departures and duration of stay of Jack Snipe at North Bull Island, Ireland and Britain (DOY = Day of Year; see Methods).

	Arrival date		Departure date		Duration of stay	
Site	**DOY**	**sd**	**DOY**	**sd**	**Mean days**	**sd**
North Bull Island[1]	275	12.2	113	10.5	202	10.9
Ireland[2]	259	17.6	120	12.9	226	25.6
Britain[3]	266	14.0	119	12.8	219	23.5

[1] this study; [2] Irish Birding (2017); [3] Sparks & Mason (2004)

(Table 2). The earliest arrival date for Ireland was 16 August 2016 and the latest departure date was 17 May 2014. Average duration of stay was 226 days, 24 days longer than for North Bull Island. The Irish and British data were also very similar with average arrival times for Britain being seven days later than for Ireland and average departure times one day earlier (Table 3, Figure 1). The average duration of stay for Britain was 219 days, seven days less than for Ireland. The datasets for North Bull Island and Ireland were considered too short for trend analyses.

Discussion

This study has identified average arrival and departure times of migratory Jack Snipe at North Bull Island and in Ireland for the first time. The study at North Bull Island has also demonstrated that it is possible to generate migration data by systematic field recording at a specific location as the results were generally comparable to data generated in Ireland and Britain as a whole. However, it is acknowledged that the process of generating these data was very time consuming as multiple visits over long periods in autumn and spring were required before the first and last migrant birds were recorded. The results for North Bull Island were also clearly site and habitat specific and this may account for slight discrepancies between them and the Irish and British data. It is likely that an improved methodology, or the use of a method for two observers (Jackson 2004), and a longer time series of data from a variety of habitats would produce results even closer to those for the whole of Ireland and Britain.

The metrics generated from the Irish and British data were remarkably similar with only seven days separating arrival times and duration of stay, and only one day separating departure times. This suggests that the timing of migration occurs synchronously across both islands. This is not too surprising as ringing recoveries indicate that birds from the same part of the breeding range in northern Europe move southwest in autumn to winter in Ireland and Britain (Smiddy 2002, Robinson *et al.* 2017). It is worth noting that although trend analysis was not carried out on the Irish data, a significant trend (0.62 ± 0.27, $P < 0.05$) was previously reported for Britain with Jack Snipe arriving six days later per decade in autumn (Sparks & Mason 2004). If timing of migrations on both islands coincide then it is not unreasonable to suspect Irish wintering populations are also altering their migration phenology.

Adult and juvenile Jack Snipe usually remain close to their breeding areas to moult during August and September (Van Gils *et al.* 2017) and reports for the month of August in Ireland and Britain are rare (BTO/RSPB/BirdWatch Ireland/SOC/WOS 2017). Although the record on 16 August 2016 was exceptionally early, it is apparently not unique in Ireland. There are reports dating to the nineteenth century that specimens were obtained in August, although the details of these records have not been published (Ussher & Warren 1900, Kennedy *et al.* 1954). Olivier (2008) considered August occurrences 'exceptional' with only five records for France and four for Britain. With the increased popularity of birdwatching and online websites now available to record observations, it is likely that more August records will be reported in the future than previously.

Currently there is no evidence from ringing recoveries that passage migration takes place in Ireland. However, it has previously been reported that a distinct passage takes place on the southwest coast of Ireland in October, late March and April (Sharrock 1973). In recent years two more records add support to the view that passage migration may take place in

Plate 236. Jack Snipe habitat, North Bull Island, Co. Dublin (Tom Cooney).

Ireland, but records are either rare or very difficult to detect. On 30 March 2014, a bird was recorded on Great Saltee Island off the County Wexford coast (*Irish Birding* 2017). Perhaps more convincingly, a bird was observed circling the Irish research vessel R.V. *Celtic Explorer* 170 nautical miles west-south-west off the County Cork coast on 20 March 2016 (Niall T. Keogh personal communication). The direction that the bird flew was not noted but based on the date and location it is reasonable to conclude it was on passage. Considering that so little is known about the migration patterns of Jack Snipe in Ireland, both records are considered noteworthy. They could potentially represent the first evidence of a previously undetected migration pattern of Jack Snipe that may have wintered further south in France, North or West Africa. If this proves to be the case, it would not be unique as a Water Rail *Rallus aquaticus* was recorded on the same date in 2016. Other wader species recorded 'at sea' on migration in spring off the coast of Ireland include Northern Lapwing *Vanellus vanellus,* Common Snipe *Gallinago gallinago,* Whimbrel *Numenius phaeopus,* European Golden Plover *Pluvialis apricaria,* Icelandic Black-tailed Godwit *Limosa limosa* and Eurasian Oystercatcher *Haematopus ostralegus* (Niall T. Keogh personal communication). Species recorded at other times include Ruddy Turnstone *Arenaria interpres,* Purple Sandpiper *Calidris maritima,* Dunlin *Calidris alpina,* Ruff *Calidris pugnax* and the pelagic Grey Phalarope *Phalaropus fulicarius.* So little is known about wader migration at sea off the coast of Ireland that it was only within the last decade, using satellite transmitters and geolocators, that the much larger, more common and obvious species, Whimbrel, was proven to migrate 'at sea' to and from Iceland and West Africa (Alves *et al.* 2016).

In addition to the lack of clarity on the timing of migration there are also some questions over the habitats used by Jack Snipe in Ireland. Jack Snipe are usually associated with damp lowland wetlands and it has been suggested that they are probably absent from mountains and moorlands (Smiddy 2002). A number of recent observations raise the possibility that this might not be the case. From 2007 to 2017 there were at least six records from upland areas in Ireland (*Irish Birding* 2017). The highest locations reported were the summit of Mount Leinster 796 m (2,612 feet) on 9 October 2016 and 487 m (1,597 feet) in the Slieve Blooms on 16 March 2008. All records were in October, February, March and April. These records have to be considered in the context of the difficulties associated with finding Jack Snipe even at well-known wintering sites and observer effort in these very large and remote locations. The records, though few in number, raise the possibility that Jack Snipe occur more frequently in uplands areas than previously reported. The fact that most of these birds were recorded during known migration periods may not be entirely coincidental.

At a time when bird species, including many familiar waders, are reported to be in decline or under threat globally for a variety of reasons including climate change (BirdLife International 2017, Pearce-Higgins *et al.* 2013, 2017), it is imperative that trends in less commonly encountered species are not overlooked simply because they are difficult to observe or monitor. This is particularly relevant to Jack Snipe as they are also a quarry species in Ireland (NPWS 2017).

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A transcript of records of Jay *Garrulus glandarius* from the books of a Dublin taxidermist, 1881-1912

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Keywords: *Garrulus glandarius,* Jay, taxidermist records

The only known surviving records of the business of E. Williams and Son, taxidermist, of Dublin is a transcript by Richard John Ussher of details of some 388 Jays *Garrulus glandarius* handled by the firm during 1881 to 1912. This paper concentrates on locating the addresses of the clients involved. Apart from five specimens of the British subspecies, *Garrulus glandarius rufitergum,* all birds appear to have been sourced in Ireland. Most of the clients lived within the accepted area of distribution of the Irish Jay *Garrulus glandarius hibernicus* at that time, while the tendency for very small numbers of birds to travel well outside their normal range is also reflected in the addresses of clients. It seems that most of the clients were rural dwellers living close to where the specimen was obtained together with a number of professional and business people in nearby towns and a few in Dublin.

Introduction

The reliable identification of rare birds in the field is now taken for granted, but this has depended upon the development of accurate field guides, efficient optical aids and digital photography, and not least, ease of transport to allow observers find and watch suitable sites. One hundred years ago most studies on ornithology involved evidence from dead specimens. Even early migration studies depended on recovery of birds killed by flying into the marine navigation lights at lighthouses and lightships around the coast of Ireland (Barrington 1900). As a result of this study, which began in 1881, the regular migration of common birds such as Starling *Sturnus vulgaris,* Blackbird *Turdus merula* and Skylark *Alauda arvensis* was discovered for the first time and despite limitations of the investigation, Barrington deemed for lack of evidence to the contrary, that apart from Rook *Corvus frugilegus,* all the other corvids including Jay *Garrulus glandarius* could be confidently classed as resident and non-migratory.

Plate 237. Two Jays set up by E. Williams and Son of Dublin in February 1897 for R. St George Robinson of Sligo (Gordon Ledbetter).

During the period before World War One began in 1914 there was great interest in having birds preserved and mounted, hence the local taxidermist's premises was where unusual birds could most likely be seen, and leading ornithologists of the day often kept in close touch with them. In Dublin, Williams and Son of Dame Street, from 1872 to about 1940, were not only three generations of taxidermists but also very careful and accurate observers of the birds which they handled and the authors of notes and a number of papers on the subject (Frost 1987, Ledbetter 2010, Nairn 2014). Edward Williams was the first person credited with noticing the distinctive plumage of the Irish Jay *Garrulus glandarius hibernicus* (More 1885, 1890, Fox & Caffrey 2013). As late as 1949 the botanist Robert Lloyd Praeger noted that Williams' records "contain a large amount of information relative to the identity, place and date of capture of interesting specimens and were very often consulted by ornithologists" (Praeger 1949).

So when the Irish Jay was recognised as a separate subspecies (Witherby & Hartert 1911) it was normal practice for a taxidermist's records to be examined to see if any useful information could be obtained. It seems that Richard John Ussher, the leading ornithologist of the day, went through the books of Williams and Son sometime in early summer 1912 (as that is when the records end) and transcribed any references to Jay that he found. Unfortunately, Ussher died after a short illness on 12 October 1913, hence the transcript has lain with his papers ever since. The only significant study of the Irish Jay in the last 100 years has been by George Rayner Humphreys, and the ensuing report (Humphreys 1928) established his reputation as a leading ornithologist with a special interest in breeding birds. The result was that when the seminal *Handbook of British Birds* (Witherby *et al.* 1938) was being prepared, such was Humphreys' standing among British ornithologists that the sections on breeding and nesting habits were sent to him for comment. It is interesting that Charles Bethune Moffat's observations on young Irish Jays as they leave the nest, which is quoted in this handbook, also first appeared in Humphreys' paper. It is worthy of note that Moffat's work is undergoing somewhat of a reappraisal at present, as his explanation for some aspects of bird population dynamics has been found to accord with recent theory (Hunt 2015).

No records of Williams' taxidermy business have survived apart from this transcript by Ussher, which stands in contrast to the situation with James Sheals, the former Belfast taxidermist, details of whose work can be found in invoice books and other documentation which have been carefully preserved (McKee 1983). The purpose of this paper is to record the details in the Ussher transcript as accurately as the constraints of handwriting, abbreviation, identification of well-known land owners, place names and addresses, will allow and see what information can be found. Hence data are presented and allowed to stand on their own merits, with no attempt at detailed analyses which are unlikely to show significant results.

The transcript entries

The transcript is in Ussher's own handwriting, is entitled "Williams & Sons Books" and runs to twenty columns of dates and addresses, sometimes with names identified, giving a total of 356 records of Jays received for mounting, of which 21 refer to two birds, two to three birds, one to four birds and one reference to five "jay skins". The remaining 331 records consist of one bird each, with the exception of one being skin only and another wings only, giving 388 birds in total. The wings are listed to Dundrum and these are likely to be a ladies fashion item, as one year later the invoice ledger of Sheals, the Belfast taxidermist, records the receipt of 3 shillings and 6 pence for "dressing of four wings for a lady's hat," on behalf of a businessman from the Newtownards Road just prior to Christmas 1911.

The five jay skins are credited to Lord Ashtown at Woodlawn which had a station on the main Dublin to Galway railway line about half way between Athenry and Ballinasloe, well outside the species normal range, at that time, in September 1893. Dead birds were often skinned in the event of any delay in getting them to a taxidermist, as skinning was considered to be the most urgent and lowest skilled task in mounting specimens. These birds certainly came from a stronghold like Portarlington only 60 miles away by rail and are extremely unlikely to have been obtained locally. However, in 1893 Lord Ashtown, Frederick Trench, who was only 25 years old inherited the title on coming of age, as both his father and grandfather were deceased. He married Violet Cosby of Stradbally Hall, County Laois in January 1894 and was certainly in a position to obtain five Jay skins wherever they might have been available, but his purpose is unclear. Also, the birds would be unlikely to have all their new feathers fully grown, whether juvenile or following the post breeding moult. On the side of the transcript there is a marginal annotation in Ussher's handwriting which reads "I have missed many." So, it may be assumed that his search for Jay records did not get them all.

Allocation of 10 km square grid reference to each entry

The writing in the transcript is generally fairly clear and legible and in most cases the details are self-explanatory, with the address of only one entry proving to be completely unidentifiable. With this exception, a 10 km grid reference has been allocated to each entry (Figure 1). It seems that Ussher's main priority was to give the name of the town in the address of

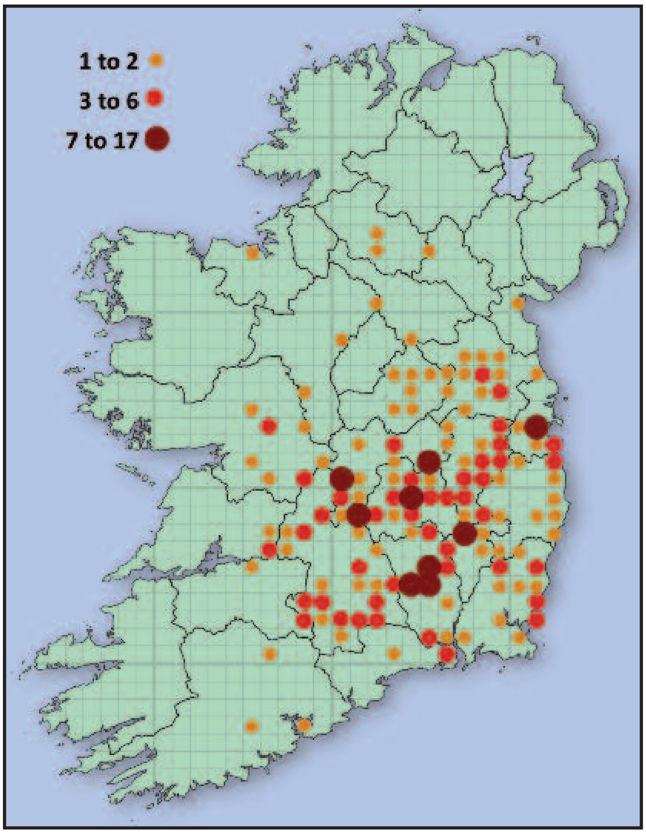

Figure 1. Number of Jays coming from clients with addresses in each 10 km square (five English birds and two birds with no location details are omitted).

each entry and further details of names, professions, townlands and so on were only given when the person was well known or someone whom Ussher knew personally. One of the more difficult classes of entry to track down were his personal acquaintances with no address – for example, "sent by Fitzherbert" is taken to be from Blackcastle, Navan and "W.W. Despard" was traced to Mountrath. There are 14 or 15 Ballingarry's in Ireland, but when the subsequent word was deduced to be "collieries" the location was certain, as only one had a coal mine. When an address gives a town which could be from two counties, usually the county is given, but Kells is an exception – County Meath is specified once and a well-known County Meath estate in another, while the other five mention no county and these are all taken to be from the County Kilkenny town. An address at Ballyheigue is taken as County Wexford rather than County Kerry because Jays are much more likely to have been available in Wexford rather than Kerry; the name Mr Kinnear gives no clue, as it is absent from both 1901 and 1911 census returns, old telephone directories, and other documents, and local enquiries also drew a blank.

Sometimes the main postal address is a provincial town at the corner of a 10 km square, e.g. Roscrea, and the result is that most of the records for perhaps up to four squares are shown to be from one square. On other occasions only County Kilkenny or County Tipperary are given and then it is necessary to allocate a square to the record in question, and generally the one with the most records is selected as being most likely to be correct. The result is that the maps reflect (Figure 1) a broad pattern of the records, and little significance should be attributed to one square with many records surrounded by others with few or none. It is certain that a small number of errors must have occurred in the allocation of grid squares, but not enough to have any effect on the general picture.

The relationship between a client's address and where the bird originated

At first glance the relationship between a client's address and the source of the bird may appear to be tenuous. For example, the five birds of the British race were known to have come from England and the three clients involved were residents of rural parts of counties Longford, Londonderry, and Dublin. However, if the relationship between the density of the human population and that of the Jay is examined, a different conclusion might be considered. Summarising the available data on Jay distribution in Ireland during 1875 to 1900, Holloway (1996) gives the bird as common in Counties Laois and Kilkenny, uncommon in Counties Offaly, Kildare, Carlow, Wexford, Waterford and Tipperary, probable in County Louth and not breeding in any other county. By 1908, Ussher stated that the bird was "Resident in those counties watered by the Suir, Barrow and Nore and has of late extended beyond the basins of those rivers in several counties and into Kildare & Meath" (Ussher 1908). In the same year Barrington (1908), writing about the Dublin area, said "the bird was occasionally wandering into Wicklow and possibly Dublin, but were not known with certainty to have bred in either county." Since Dublin city and suburbs, west and south County Dublin and northeast County Wicklow are therefore unlikely areas for Jays at that time and were heavily populated districts and close to Williams' city centre business, it is interesting to see how many clients have an address in this area. This might give some idea of the number of clients who lived some distance from where birds were obtained. In the seven relevant 10 km squares, e.g. O02, O03, O12, O13, O21, O22 and O23 (which includes all of County Dublin, south of a line from Howth to Clonee and northeast County Wicklow, e.g. Bray, Greystones and Enniskerry), counting all English birds, Jay wings and skins gives 24 birds from 21 clients. Therefore, out of a total of 388 birds and 356 clients, only about 6% of the birds can be shown to have definitely come some distance from where they were obtained. This would tend to indicate that many birds came from clients in rural parts of Ireland close to where they were caught or shot. As well as those living in the countryside, other clients who feature frequently are those resident in south midland provincial towns and the addresses are given as a street, a quay, the Mall, a bank, Court House, or an Army or Royal Irish Constabulary Police Barracks and, once, a Convent of Mercy, a Parish Priest or a Rectory. According to Martyn Anglesea (1997), Keeper of Fine Art at the Ulster Museum, displays of stuffed birds and other animals usually shot on the Estate formed an important feature in the Victorian Country House both in Ireland and Britain.

Since the Jay is a very shy and wary bird it can be very difficult to shoot and often when two or three birds came from a single client they were received by Williams one at a time, over the course of a week or two. Whenever this happened it is likely that the birds were obtained near to where the client was living. The two birds illustrated (Plate 237) came from R. St George Robinson of Sligo, agent for a number of local estates, on 2 and 5 February 1898; the exact locality where the birds occurred cannot be ascertained, but Drummond Nelson always believed them to have been obtained nearby.

Another method of identifying and verifying the occurrence of birds listed in the transcript is by cross-referencing them with other letters and data contained in Ussher's papers. For example, William Hamilton of Castlehamilton near Killeshandra reported to Ussher that the first Jay seen in County Cavan was shot on 10 November 1909 and sent to Williams. This bird appears in the transcript nine days later. Interestingly, Lord Rossmore, living less than 30 miles away near Monaghan town, is credited with a Jay on almost the same day, 17/18 November 1909. In both cases birds were certainly sourced locally.

Plate 238. Jay (Terry Flanagan).

Equally, the letters can provide details of Jays that do not feature in the transcript; for example, Lord Rosse at Birr informed Ussher that his keeper had "caught" no less than 26 Jays when trapping hen Pheasants *Phasianus colchicus* for breeding purposes in spring and early summer 1910, none of which appears to have found their way to Williams. The transcript does not record any Jays between 11 April and 23 July that year, with the only ones with an address in the Birr area being one dated 21 February and one dated 9 September. Another factor which might have influenced the address spread of clients is the existence of alternative taxidermists who might have been in competition with Williams for business. However, it appears that there was no other well-known taxidermist in Dublin (Ledbetter 2010) at that time, and Sheals in Belfast is known to have handled seven Jays between 1892 and 1914. Fredrick Raynor Rohu, the long-established Cork taxidermist, was a keen observer of birds and had items published in *The Irish Naturalist* (Rohu 1904, 1909, 1913) and was also listed as a contributor among those providing records for Ussher and Warren's *Birds of Ireland* (1900). However, he certainly did not get all the business in the area; for example when Mr Longfield's keeper "unluckily sent" a Peregrine Falcon *Falco peregrinus* which he had shot "to a local man in Mallow" the skin was quickly retrieved and sent to Williams to be mounted. Also, three Jays from County Cork appear in the transcript; one in 1888 from Doneraile, one in 1898 from Rostellan and one in 1900 from Upton.

Since Ussher went to the trouble of going through Williams' records, it shows that he considered the addresses to be relevant to the status and distribution of the species in Ireland at that time. The fact that birds were generally received in small numbers at any one time and at all times of the year would indicate that no major commercial exploitation of the species was taking place which might distort the validity of the addresses as an indication of the species distribution. Sensational stories that Williams handled 739 Barn Owls *Tyto alba* during one ten-year period (Shawyer 1987) or that many of the 383 Kingfishers *Alcedo atthis* prepared by Walter Lowne of Great Yarmouth were "aviary reared" (Morris 2012), or the apocryphal tale that Jays of the lower River Blackwater valley in east Cork and west Waterford were exterminated about 1840 to provide feathers to make salmon fishing flies (Ussher & Warren 1900) are all irrelevant.

The novelty effect of a spectacular looking bird with very secretive habits and which is rarely seen must also be considered. It is probable that once someone had obtained a bird and had it mounted, others would have wanted to do the same. This does not appear to have been a major influence

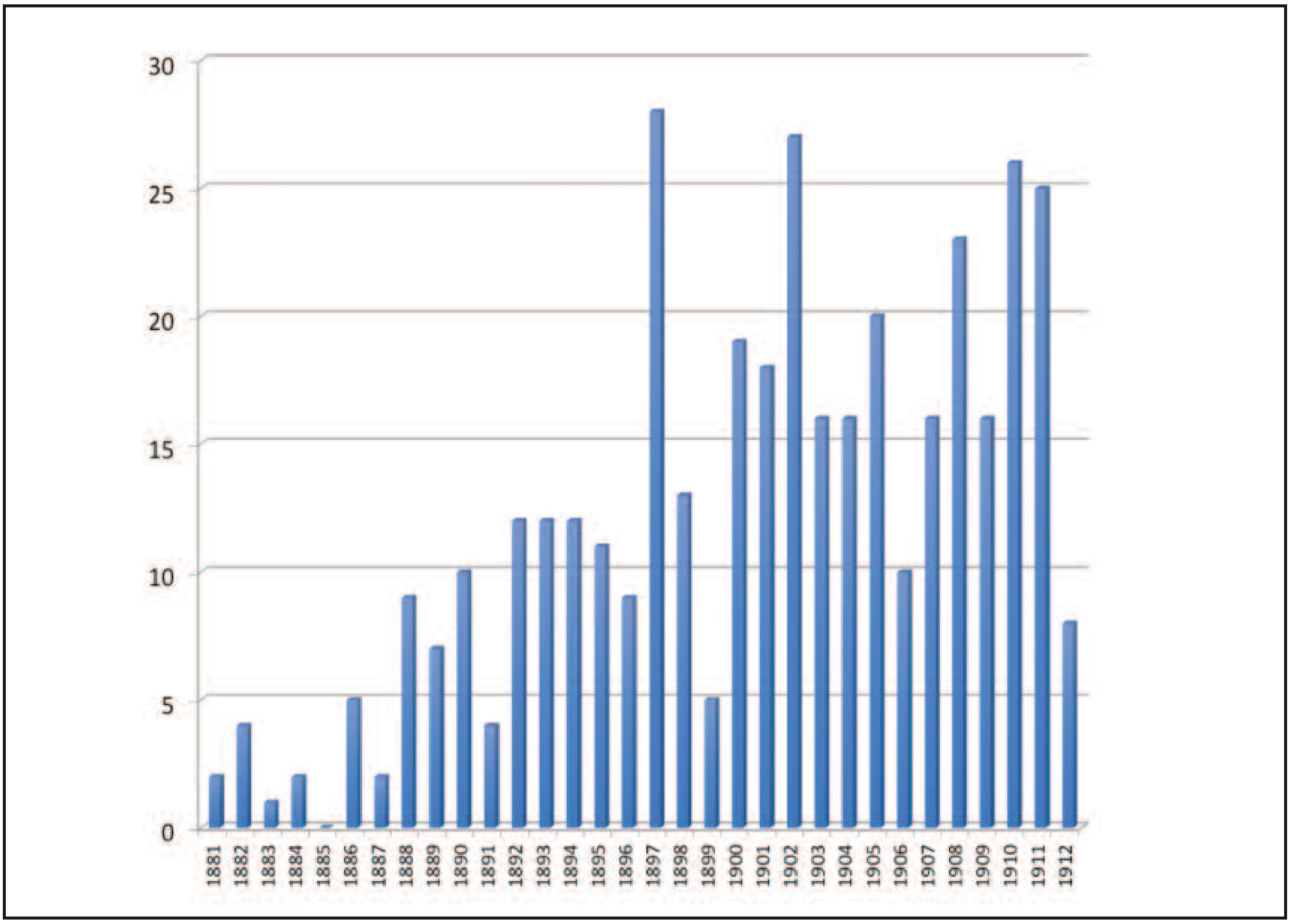

Figure 2. Number of Jays received for taxidermy by E. Williams and Son of Dublin, by year.

on the overall scatter of records, particularly balanced out over the entire 32 years, but certainly would have had a distorting effect over a shorter period. An attempt to track population expansion by sequential mapping of addresses for each successive eight-year period failed to show any consistent trends. So it appears that a high proportion of the Jays were obtained close to where the relevant clients lived and that, with some obvious exceptions, the distribution of the bird in Ireland at that time tended to mirror the spread of addresses given in the transcript.

The records year by year and month by month

The yearly totals are shown in Figure 2. Despite continuous variation in numbers from one year to the next a definite increase occurred over the 32 years. Monthly totals for the whole period are given in Figure 3. Since Jays were considered vermin, they were killed all year round as the opportunity arose. The numbers were reasonably constant from October to April, during the shooting season. However, during the breeding season of this single-brooded bird its secretive nesting habits would explain low numbers in May, June, July

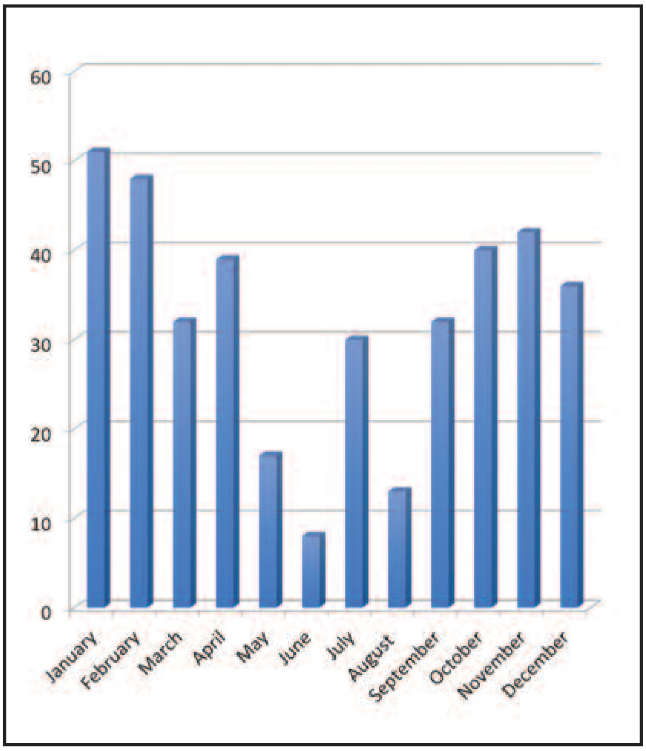

Figure 3. Number of Jays received for taxidermy by E. Williams and Son of Dublin, by month.

and August. The increase for July came about by Williams often receiving two or three birds together in this month, with one client from Carlow sending four birds on 13 July 1897. This suggests that many were vulnerable young birds just out of the nest. Although these young birds would not yet have attained full adult plumage they would still have made a very striking exhibit if mounted together. Late summer and early autumn is the moulting period for adults, who would not have been in their best condition from the end of June until the completion of their moult.

The illustration

The transcript records two Jays sent to Williams in February 1897 by R. St George Robinson of Sligo, the agent for a number of local estates. Drummond Nelson, a relative of Robinson, drew the author's attention to these birds (which he always believed to have been obtained locally) at Woodville House, Sligo, the home of his daughter and her husband, Linda and Richard Wood Martin, to whom we are grateful for facilitating the photograph by Gordon Ledbetter (Plate 237). These birds are in a glass case bearing the Williams' trade label.

Acknowledgements

Paddy Sleeman first sparked my interest in Jays due to the implication of salmon fly fishermen in their supposed disappearance from the Munster Blackwater valley. Staff of the Royal Irish Academy Library helped me extract relevant data from Ussher's papers. Sophie Wilcox of the Alexander Library, Oxford is thanked for a copy of George Humphreys' elusive paper on Irish Jays, and Paddy Mackie who searched over 20-years of Sheals Invoice Books for records of Jays. Linda Wood Martin and her father, Drummond Nelson, are thanked for tracking down the Sligo birds, and Gordon Ledbetter for the photograph. The Wood Martin family must be commended for the excellent condition of the birds after 120 years. Stephen Newton looked at the data and encouraged me to tabulate it and both he and Niall Hatch of BirdWatch Ireland kindly commented on the manuscript. Kevin Kenny carried out a provisional sequential mapping of addresses for each of four successive eight-year periods, which failed to show any consistent trends. Michael O'Clery produced the map for Figure 1. Kenny McGarvey and my sister, Lois Jarvis, helped get everything into an electronic and email-friendly format.

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Occurrences of non-breeding waders in summer on the east coast of Ireland

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This study investigated the occurrence of fifteen non-breeding wader species at North Bull Island in Dublin Bay between spring and autumn. This is a period when species diversity and populations at coastal wetlands in Ireland are believed to be at their lowest. Weekly observations from May to mid-July 2012 to 2015 and in 2017 confirmed that Oystercatcher *Haematopus ostralegus,* Curlew *Numenius arquata,* Bar-tailed Godwit *Limosa lapponica,* Black-tailed Godwit *Limosa limosa* and Dunlin *Calidris alpina* occur continuously throughout summer. Another seven species were absent intermittently, mainly in mid-June, and three species rarely occurred. Diversity and total numbers were highest in May and from late-June to mid-July. However, a minimum of nine species and 450 birds were recorded in the first three weeks in June. Although populations in summer 2017 varied according to the species, Curlew and Oystercatcher were on average the most numerous. In June both species dominated summering populations representing 79% to 90% of all birds present. Although weekly counts were only carried out in 2017, it is noteworthy that seven species occurred in concentrations in excess of their respective thresholds for national importance. The results show that North Bull Island is an important site in summer for a range of wader species some of which have an unfavourable conservation status not just in Ireland, but globally. It is likely that a coordinated systematic monitoring scheme at a range of coastal wetland sites would establish the importance of Irish coastal wetlands for non-breeding and migratory wader populations in summer.

Introduction

There is an increasing concern for the conservation status of many European waders including familiar species such as Curlew *Numenius arquata,* Bar-tailed Godwit *Limosa lapponica* and Black-tailed Godwit *Limosa limosa* (BirdLife International 2015, Pearce-Higgins *et al.* 2017). Understanding the migration patterns and the phenology of these species is critical for the future management of both the species and their habitats. Ireland is a key European location for wading bird species due to its situation on the East Atlantic Flyway and to the diversity of Irish wetland habitats, both coastal and inland. Approximately ninety-five species of wading bird have been recorded in Ireland as breeders, winter visitors, and passage migrants or as vagrants (Hobbs 2017). Non-breeding and passage migrant waders occur in the summer months in Ireland either regularly or infrequently (Ussher & Warren 1900, Kennedy *et al.* 1954, Hutchinson 1989, *Irish Birding* 2017). However, this aspect of wader distribution has to date received very little attention. Large scale studies have generally concentrated on the status and distributions of breeding and

Plate 239. Black-tailed Godwit (Liam Kane).

wintering species (Sharrock 1976, Gibbons *et al.* 1993, Balmer *et al.* 2013).

The absence of data on waders at coastal wetlands in summer months could lead to the misconception that such sites are of little conservation value as bird habitats outside of the main breeding, wintering and migration times. Non-breeding waders of several species have been known to occur at North Bull Island in the May to July period since the early nineteenth century. Kennedy (1953) mentions that 12 wader species occur commonly or rarely in summer, including Oystercatcher *Haematopus ostralegus,* Curlew, Redshank *Tringa totanus,* Grey Plover *Pluvialis squatarola,* Turnstone *Arenaria interpres* and Sanderling *Calidris alba.* In more recent times two studies have been undertaken on waders at North Bull Island in Dublin Bay. In the early 1970s, monthly counts of all waterbird species indicated that several non-breeding wader species occurred regularly in summer, occasionally in large numbers (Hutchinson & Rochford 1974). A more detailed study based on systematic weekly observations reported that five species were always present from May to August, including moulting Oystercatcher, Bar-tailed Godwit and Curlew (Holohan 2008). That study also concluded, based mainly on count data and plumage details for all wader species, that there was a gap of approximately ten days in June between the end of northward spring migration and the appearance of birds on post-nuptial migration. Unfortunately the full results of this study were not published so the distribution of waders over the summer at this site remained unclear.

Evaluating the role of such coastal wetlands for non-breeding and migratory waders in summer is particularly important for species that are reported to be in decline nationally and globally (Colhoun & Cummins 2013, BirdLife International 2017). The objective of the current study was to establish the frequency of occurrence of fifteen common wader species in the ten-week period from May to mid-July and to evaluate the size of the summering populations at North Bull Island. The application of the methodology used to other sites could provide useful information for assessing the status of such species at national and regional level.

Study area

North Bull Island (53.3705° N, 6.1440° W) on the north shore of Dublin Bay is a National Nature Reserve (NNR), Special Protection Area (SPA) for birds and Special Area of Conservation (SAC) for habitats (NPWS 2017). The island is approximately 5.5 km in length and is separated from the mainland by inter-tidal mud and sandflats. From autumn to spring wetland bird populations of national and international importance occur (Crowe 2005), some of which are of conservation concern in Ireland (Colhoun & Cummins 2013). At high tide the majority of waders roost on the island's saltmarshes which are located on the landward side of the island.

Methods

The fifteen species investigated in this study, including their codes in parentheses, were Oystercatcher (OC), Lapwing *Vanellus vanellus* (L), Golden Plover *Pluvialis apricaria* (GP), Grey Plover (GV), Ringed Plover *Charadrius hiaticula* (RP), Whimbrel *Numenius phaeopus* (WM), Curlew (CU), Bar-tailed Godwit (BA), Black-tailed Godwit (BW), Turnstone (TN), Knot *Calidris canutus* (KN), Sanderling (SS), Dunlin *Calidris alpina* (DN), Redshank (RK) and Greenshank *Tringa nebularia* (GK). Bird species were recorded weekly on the islands inter-tidal habitats, saltmarshes, sandy foreshore and rocky habitats over a ten-week period from the first full week in May (week 19) to mid-July (week 28). These weekly observations were carried out in each year from 2012 to 2015, and again in 2017. The data generated were used to score the occurrence of each species (with a score of 1 when it was present, and zero (0) when it was absent) in each of the ten weeks over the five years of the study. The maximum score, therefore, for a given week over the five years was 5 and the minimum was zero (0). The frequency of occurrence of each species over the ten-week period May to mid-July and the four weeks in June are expressed as a percentage. In addition to recording presence or absence of species, weekly counts were also made in 2017 for the same ten-week period to quantify the populations present in summer. Over the course of this survey a number of supplementary records were provided by several observers known to the author (see Acknowledgements).

Results

All fifteen species were recorded at least once during the survey period in each year (Table 1). Three species (Oystercatcher, Curlew and Bar-tailed Godwit) were recorded in all weeks of the survey period. Although Dunlin and Black-tailed Godwit were not recorded on five and four occassions respectively, it is possible that birds were present but overlooked and that both species also occur thoughout summer annually. Seven species occured less frequently (on 86% to 64% of occassions) and were generally least common in the first three weeks of June (weeks 23 to 25). These were Whimbrel, Ringed Plover, Redshank, Lapwing, Sanderling, Turnstone and Grey Plover. Spring passage of Whimbrels was recorded regularly up to the third week of May (week 21) but they were scarce thereafter. Redshank and Ringed Plover occurred least frequently in the first three weeks of June (weeks 23 to 25). Lapwing were least commonly recorded from mid-May to the first week of June, after which they were

Table 1. Frequency of occurrence of fifteen wader species from May to mid-July over the five-year period 2012-2015 and 2017 at North Bull Island, Dublin Bay. Species listed by frequency of occurrence.

	May				June				July		% occurrence, all weeks	% occurrence, June
Week number	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**		
Oystercatcher[1]	5	5	5	5	5	5	5	5	5	5	100	100
Curlew[1,2]	5	5	5	5	5	5	5	5	5	5	100	100
Bar-tailed Godwit[3]	5	5	5	5	5	5	5	5	5	5	100	100
Black-tailed Godwit[1,3]	5	5	5	4	4	4	4	5	5	5	94	85
Dunlin[2]	5	5	5	5	5	4	3	3	5	5	90	75
Whimbrel	5	5	5	4	4	5	4	4	3	4	86	85
Ringed Plover	5	5	5	5	2	3	2	5	5	5	84	60
Redshank[2]	5	5	5	4	2	2	3	5	5	5	82	60
Lapwing[2]	3	3	2	2	2	5	5	5	5	5	74	85
Sanderling	5	5	5	3	3	4	3	3	3	2	72	65
Turnstone	5	5	4	3	3	3	2	3	5	2	70	55
Grey Plover[3]	5	4	4	3	2	2	2	3	3	4	64	45
Knot[1,3]	4	2	3	3	3	0	0	1	2	2	40	20
Greenshank	3	2	1	1	0	1	2	2	3	4	38	25
Golden Plover	1	2	2	1	0	0	2	1	0	1	20	15

[1] Globally Near Threatened (BirdLife International 2017); [2] Red List and [3] Amber List species of conservation concern in Ireland (Colhoun & Cummins 2013).

recorded in all weekly surveys. Sanderling occurred regularly until about mid-May but irregularly for the rest of the summer period. Turnstone were also regular up to mid-May and either occurred infrequently or were absent occasionally in most summers. Grey Plover were most frequently recorded in early May but were erratic in occurrence in most summers. It is of note, however, that although this species was absent periodically in four of the five summers of this survey, at least five birds were present continuously in 2015.

Weekly counts in 2017 showed that in excess of 700 birds in total (across all 15 species) were recorded in May (weeks 19 to 22) and from mid-June into early July (weeks 25 to 28), with a peak count of 2,498 birds in week 28 (Table 2). Lowest weekly counts were in the first two weeks of June with 483 (week 23) and 459 (week 24). Average populations were 1,177 in May, 699 in June and 2,068 in the first two weeks in July.

On average, the Oystercatcher and Curlew were the most numerous species throughout the survey in 2017. They dominated the summering populations in June and repesented 79% to 90% of all birds recorded (Figure 1). The average number of Oystercatchers for the entire survey period was 355 birds, with 321 present in June. Weekly counts ranged from a low of 231 (week 22) to a high of 490 birds (week 28). Curlew were the second most numerous species with averages of 312 birds for the entire survey period and 278 for June. The number of Curlew present varied with lowest values of 35 in mid-May (weeks 20 and 21), followed by a steady increase thoughout summer to a peak of 1,098 in July (week 28). The highest numbers of Bar-tailed Godwit occurred in May (peak 203) with lowest values of two to four birds in mid-June (weeks 24 and 25), followed by a small increase in July. Dunlin were common in May with a peak count of 689 (week 21) but most birds had departed by the end of the month leaving only five to eleven birds present throughout June. A small increase was observed in July with the return of post-nuptial migrants. Up to 100 Black-tailed Godwit were present in May but only eleven to sixteen birds were recorded in the first three weeks of June (weeks 23 to 25). An increase was observed from late June with a peak count of 293 in mid-July (week 28).

Of the species that were absent occasionally or for short periods during summer 2017, fewer than thirteen Redshank were recorded from May up to the third week of June with none in the first week of June (week 23). A noticable increase took place from the last week of June (week 26) from 31 birds to 325 birds by mid-July (week 28). Small numbers of Whimbrel were present in all weekly counts in 2017 with a peak of 26 at the end of June (week 26). Sanderling were absent in two weekly counts and were generally scarce in 2017 with a peak of only 49 birds in May (week 21). Unlike other wader species in this survey, Lapwing were absent at the beginning of summer 2017 but became frequent from the last week of May to mid-July. Lapwing were the first post-nuptial migrants to arrive each year. Small numbers of Grey Plover

Table 2. Weekly totals of fifteen wader species from May to mid-July 2017 at North Bull Island, Dublin Bay. Species listed by numerical abundance.

	May				June				July		Total average	June average
Week number	**19**	**20**	**21**	**22**	**23**	**24**	**25**	**26**	**27**	**28**		
Oystercatcher	480	390	266	231	290	240	298	454	410	490	355	321
Curlew	52	37	35	63	92	160	354	506	724	1098	312	278
Dunlin	376	248	689	535	11	5	5	7	29	109	201	7
Bar-tailed Godwit	187	147	203	189	29	4	2	13	48	72	89	12
Black-tailed Godwit	104	81	86	79	11	16	13	47	139	293	87	22
Redshank	12	7	5	3	0	2	13	31	170	325	57	12
Whimbrel	13	5	8	5	18	18	16	26	21	31	16	20
Sanderling	1	4	49	34	22	5	0	3	9	0	13	8
Lapwing	0	0	0	3	3	5	18	26	20	27	10	13
Grey Plover	20	13	1	4	0	0	0	7	26	16	9	2
Knot	3	8	5	2	2	0	0	5	13	4	4	2
Greenshank	1	0	0	1	0	0	5	4	14	24	5	2
Turnstone	1	2	1	5	3	4	0	0	5	7	3	2
Ringed Plover	5	2	2	3	2	0	0	1	10	2	3	1
Golden Plover	0	0	0	0	0	0	0	0	0	0	0	0
Totals	1255	944	1350	1157	483	459	724	1130	1638	2498		
No. of species	13	12	12	14	11	10	9	13	14	13		
Average monthly	1177				699				2068			
Average 10 weeks	1164											

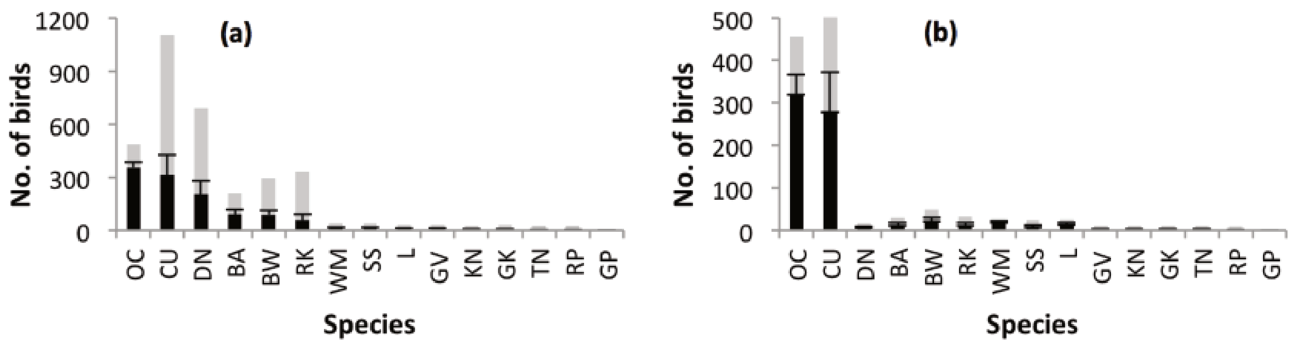

Figure 1. Populations of fifteen wader species in summer 2017 at North Bull Island, Dublin Bay (a) May to mid-July (weeks 19 to 28); (b) June (weeks 23 to 26). Black bars are mean number of birds (+/-S.E.); grey bars represent a maximum weekly count. See Methods for species codes.

were recorded in May (peak of 20 in week 19) and from the last week of June to mid-July (26 in week 27) but were absent for most of June (weeks 23 to 25). Knot were scarce in 2017 with less than eight birds from May to the first week of June. They were not recorded again until five birds, including juveniles, were recorded in the last week of June (week 26) but numbers remained low to mid-July. Single Greenshank were recorded in only two out of six weekly counts up to mid-June but autumn passage commenced in week 25 with the arrival of adults and juveniles. Numbers increased slightly to 24 birds by mid-July (week 28). Turnstone were very scarce throughout 2017 occurring in single digits in most weeks but were absent in the latter part of June (weeks 25 and 26). Less than ten Ringed Plover were present throughout summer of 2017 with none recorded in mid-June (weeks 24 and 25). Golden Plover were absent in 2017.

Plate 240. Oystercatcher (M.O'Clery).

Discussion

This survey has established that twelve wader species occur at North Bull Island either continuously, frequently or occasionally over the summer period annually. Although the majority of birds were probably migrants moving to and from breeding grounds, particularly during known migration times in May and from late June onwards, it is clear that several hundred birds, mostly immature Oystercatcher, Curlew and Bar-tailed Godwit also use the island in summer to moult. The diversity of species and the total numbers present highlight the importance of this site and perhaps other coastal wetlands in Ireland for migratory and non-breeding waders in mid-summer. Based on a combination of lowest numbers, lowest species diversity and the reappearance of birds in juvenile plumage in early June, there appeared to be a gap of about two weeks when most northward spring passage ended and return passage in autumn commenced. Holohan (2008) reported a similar pattern of occurrence with a gap of about ten days between the end of northward migrations and the arrival of the first Lapwing.

Of the fifteen species investigated, Oystercatcher, Curlew and Bar-tailed Godwit are present at all times, while Dunlin and Black-tailed Godwit were very rarely absent. A further seven species, Whimbrel, Ringed Plover, Redshank, Lapwing, Sanderling, Turnstone and Grey Plover occur in small numbers, frequently but not continuously in most summers. The rarest species to occur were Knot, Greenshank and Golden Plover. The frequency of occurrence and numbers for each species in this study is generally consistent with their known trends in Ireland and Britain (BTO/RSPB/BirdWatch Ireland/SOC/WOS 2017, Frost *et al.* 2017).

It was not surprising that Oystercatcher was found to be the most common species, as they have been known to be 'common at all times' in Dublin Bay since the nineteenth century (Patten 1898). The mid-summer population of about 300 birds appears to have remained fairly constant since the 1970s (Hutchinson & Rochford 1974, Hutchinson 1975, Cooney *et al.* 1986). Although smaller numbers of Curlew occured in May, again mostly moulting birds, the population increased steadily throughout June and July with the arrival of post-nuptial migrants. Similar numbers and trends were reported in the early 1970s (Hutchinson & Rochford 1974). Although few Bar-tailed Godwit were present in mid-summer 2017, summering populations are known to vary considerably from year to year at North Bull Island e.g. 75 in 1939 (Kennedy 1953), 300 in 1959 (O'Mahony 1959), three in 1969 (Cummins *et al.* 1970), 167 in 1974 (Hutchinson & Rochford 1974) and

300 in 2012 (Cooney 2013). A similar pattern has been apparent for Dunlin e.g. 32 birds were reported in June 1972 (Hutchinson & Rochford 1974), 420 on 7 June 2012 and 2,000 on 20 May 2013 (Cooney 2013, 2014, 2017). It is likely that *C.a. schinzii, C.a. alpina* and *C.a. arctica* occur simultaneously at the site as all three sub-species are known to occur during migration times in Ireland (Crowe 2005). The number of Black-tailed Godwits wintering at North Bull Island has increased substantially in recent decades so it is not surprising that they now occur regularly, though usually in low numbers, during the summer months.

Although Whimbrel are primarly spring and autumn migrants in Ireland, a recent study has demonstrated that small numbers occur regularly in summer (Cooney 2016). Numbers recorded in summer 2017 were slightly above average. Ringed Plover are the only wader to have nested regularly at North Bull Island (Patten 1898, Hutchinson 1975), however the last known successful nesting attempt was in 2002 (Coombes & Murphy 2009). Their local extirpation as a breeding species probably explains why they were quite scarce or absent in mid-summer in the current study. Lapwing are generally scarce in early summer but are present regularly from about mid-June with the arrival of the first post-nutial migrants, adults and juveniles. Redshank, Sanderling, Turnstone and Grey Plover all occur sporadically in summer but are probably least frequent and in lowest numbers in June. However, it is possible that in some years they may occur throughout summer. For example, in 2017 Grey Plover were recorded in all weekly surveys. Knot, Greenshank and Golden Plover are rare in summer.

In recent decades phenological studies have reported that many migratory bird species, including waders, are altering the distribution range and the timing of their migrations in response to changing climatic conditions (La Sorte & Thompson 2007, Pavón-Jordán *et al.* 2015, Miles *et al.* 2016). Phenological studies that report on arrivals and departures are appropriate for those migratory species that are clearly absent for long periods, either in summer or in winter. In such circumstances, there is little or no ambiguiety regarding what constitutes, for example, a last departure date or a first arrival date. The current study, however, has highlighted that a group of twelve wader species were present continuously or frequently throughout the summer at the study site. For these species, therefore, it would not be possible to indentify birds which had over-wintered at North Bull Island and were still present in spring and early summer from those present in spring for other reasons (e.g. as passage migrants). This overlap between wintering and migrant populations and a lack of any period of absence highlights that for these species departure times of over-wintering birds cannot be established. Without clear departure times it would evidently not be possible to calculate duration of stay in winter. These conclusions contrast markedly with recently reported changes in the migration phenology of migrant Irish waterbirds (Donnelly *et al.* 2015, 2016). For example, those studies reported 'departure' and 'arrival' dates for Bar-tailed Godwit. This species has long been known to occur in summer in Ireland, occasionally in 'considerable numbers' (O'Mahony 1959, Ruttledge 1966) and have been reported to be present continuously through the summer in the current study and by Holohan (2008). It is of note that the studies of Donnelly *et al.* (2015, 2016) were based on edited bird report data (Cooney *et al.* 1981-1995, Madden & Cooney 1996, 1997, 2001, Coombes & Murphy, 2003, 2004, 2006, 2009, Madden 2005), with some of the published dates for sightings being misinterpreted by those authors as examples of departure times in spring. It is also highly likely that the inclusion of such data, which would be inappropriate for phenological studies, into the statistical analyses contributed to results that were inconsistent with observed migration patterns of waterbird species in Ireland.

The current study focused on a wetland on the east coast of Ireland, however it is unlikely that the trends observed are confined to this one location. Summering non-breeding and migratory populations have been documented occassionally from various locations in Ireland for over a century (e.g. Ussher & Warren 1900, Merne 1971, O'Sullivan 1983, NIBA 1993, Lysaght *et al.* 1994, *Irish Birding* 2017). The importance of monitoring and protecting these species at wetlands in summer should not be underestimated. Of the fifteen species recorded at North Bull Island in this survey, four are 'Globally Near Threatened' (BirdLife International 2017), while five are 'Red list' and four are 'Amber list' species of conservation concern in Ireland (Colhoun & Cummins 2013). These include species like Curlew that are undergoing severe range retractions and population declines in Ireland and Britain (Balmer *et al.* 2013, Franks *et al.* 2017). As many summering waders, including Curlews, are moulting immature birds (Prater 1981, Holohan 2008) they will eventually become the breeding populations of the future. Along with other conservation measures, it is therefore vital that they are fully protected from disturbance at coastal sites in summer, as part of conservation strategies devised to reverse declines in breeding populations. In addition, although weekly counts were only carried out during the ten week survey period in 2017, it is noteworthy that seven species occurred in concentrations in excess of their respective thresholds for national importance. Although it is likely that populations fluctuate annually, longer term systematic monitoring at this site in summer would provide a more accurate assessement of bird numbers.

While conservation efforts have correctly focussed on monitoring and protecting breeding and wintering populations in Ireland, this study has demonstrated the importance of coastal wetlands for non-breeding wader species in summer, some of which have an unfavourable conservation status nationally and globally.

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Nocturnal communal roosting behaviour in Great Crested Grebes *Podiceps cristatus*

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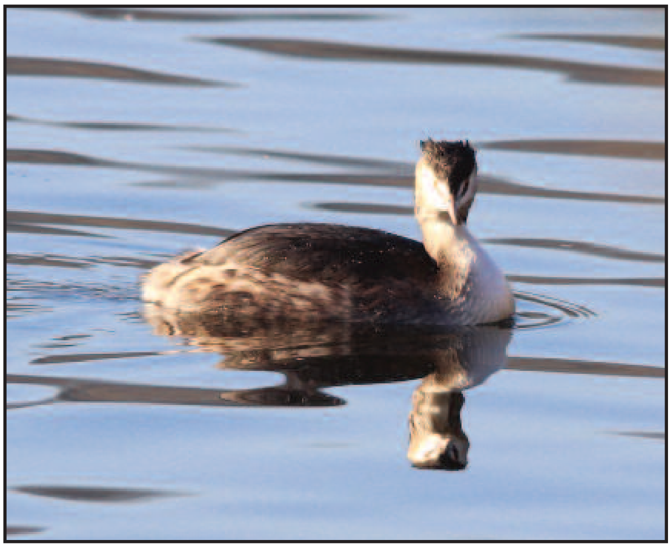

This paper documents nocturnal communal roosting as a distinct behavioural phenomenon in Great Crested Grebe *Podiceps cristatus* populations at three coastal sites in southern Ireland with the following distinctive features: the grebes roost in the same general areas each night; the size of the roosting flocks are larger than at daytime roosts; typically all the birds in an area assemble into a single flock; and birds actively travel long distances to join the roosts. Regular counts in Cork Harbour showed that dusk roost counts provide a better method of monitoring Great Crested Grebe numbers and distribution than routine daytime waterbird counts. Recognition of the importance of nocturnal roosts is required for managing Special Protection Areas (SPAs) designated for this species, but the Cork Harbour SPA does not include most of this grebe's roosting areas. The roosting grebes are sensitive to disturbance from vessel activity and environmental impact studies for plans or projects that may include crepuscular and/ or nocturnal vessel activity should consider the potential impact on roosting grebes.

Introduction

There is very little published information on the nocturnal behaviour of Great Crested Grebes *Podiceps cristatus*. Campbell *et al.* (1978) made reference to Great Crested Grebes forming nocturnal roosts in the Firth of Forth (Scotland), but did not describe this behaviour in detail. Piersma *et al.* (1988) described diel patterns in the activity of moulting Great Crested Grebes in the Netherlands. These birds were most active around dawn and dusk, with roosting flocks forming during the middle of the night and the middle of the day. However, the general prevalence of nocturnal communal roosting in Great Crested Grebes appears to be unknown. Fjeldså (2004) referred to social nocturnal roosting in grebes generally but did not provide any specific information for Great Crested Grebes, while the species account in Cramp and Simmons (2004) refers only to birds roosting "in loose parties" at night.

During waterbird surveying in Cork Harbour in recent winters I became aware that Great Crested Grebes habitually assemble into flocks in specific locations to roost nocturnally. From discussions with other observers, there appears to be a lack of awareness of this behaviour, and, as discussed above, it is not adequately described in the scientific literature. This paper describes my observations on nocturnal roosting behaviour by Great Crested Grebes in Cork Harbour, and at two other coastal Irish sites, and discusses the conservation implications of this behaviour, and its utility for waterbird surveying.

Plate 241. Great Crested Grebe (Terry Hughes - WikiCommons).

Methodology

I carried out observations in Cork Harbour (County Cork), Dungarvan Harbour (County Waterford) and Wexford Harbour (County Wexford) during the winters of 2014/15, 2015/16 and 2016/17. Initially, I searched these sites to find the locations of the nocturnal roosts of these grebes. At each roost I attempted to map the position of the main roosting flocks on multiple dates by taking bearings to the flock from two separate locations and using these to triangulate the position (taking account of the grid declination). I tested this method, by mapping the position of navigation buoys, and the results indicated that it was accurate to ± 100-200 m. However, as this method only works if the flock is stationary, I was only able to use it on a minority of occasions. In Dungarvan Harbour, I mapped the position of one of the roosts (D2) by using the known positions of aquaculture marker buoys and posts. I used 150 m buffers around the mapped positions to generate maps of the approximate extent of the roosting areas.

For comparison between the positions of the nocturnal roost sites and the daytime foraging distribution, I mapped the extent of grebe habitat at each site. This mapping was based on a review of Irish Wetland Bird Survey (I-WeBS) and National Parks and Wildlife Service (NPWS) Waterbird Survey Programme data for each site, combined with my personal knowledge of grebe distribution in Cork Harbour, and consultation with regular counters for particular subsites within Cork Harbour. As grebes typically feed in waters less than 4 m deep (Cramp & Simmons 2004), I used the 5 m depth contour from the Admiralty charts to help map the boundaries of suitable habitat at each site.

In Cork Harbour, I carried out opportunistic counts of the nocturnal roosts of the grebes on 86 days in January to March 2015, September 2015 to March 2016 and September 2016 to February 2017. On most of these days, I made multiple counts, starting at times ranging from 180-60 minutes before dusk, to record the pattern of the build-up of the roosts. I also made observations on the roosting behaviour at dawn, and during the day. On 7 February 2015 and 20-21 February 2016, I organised co-ordinated counts in which the Cork Harbour roost sites were counted simultaneously (2015), or across two consecutive evenings (2016), at dusk by teams of four or five counters. I also carried out dusk roost counts at Dungarvan Harbour on seven days in February and March 2015, November 2016 and January 2017, and at Wexford Harbour on eight days in February and March 2015, September and October 2015 and January and March 2016. On each count I categorised the behaviour of the grebes as either feeding, roosting, or swimming. The swimming category referred to birds that were purposefully swimming in one direction, usually while travelling to the night roost site. The roosting category included all non-feeding birds that were not actively swimming. I also recorded the size of each roosting group. I used the time of the end of civil twilight from www.timeanddate.com/sun as the time of dusk.

For reporting and analysis of numbers using the roosts, I used counts only from days when I obtained accurate counts within at least 60 minutes of dusk. For analysis of seasonal variation in population size in Cork Harbour, I have only used data from October, November, January and February in each winter. I excluded data from September and March as the grebes are arriving or departing in these months and the counts are, consequently, highly variable. I did not use data from December as the number of counts for this month was considered too small. For analysis of patterns of build-up of the roosts, I assigned counts to five time periods relative to dusk: 0-30 minutes, 31-60 minutes, 61-120 minutes, 121-180 minutes, and > 180 minutes, before dusk. I used counts only where a final count was taken in the 0-30 minutes time period and another count was taken in at least one other time period.

Results

Roosting sites and numbers

Great Crested Grebes are widespread in Cork Harbour but appear to be limited to waters less than 5 m deep. They also do not usually occur in the upper estuarine areas. Therefore, Cork Harbour can be divided into four discrete sectors of grebe habitat and there are grebe roosts associated with each of these sectors (Figure 1). Based on the distribution of the roost sites in relation to their feeding habitat, grebes in Cork Harbour may travel over 3 km from the roost sites to feed during the day. The primary roosts (E1/E2 and W1) in the East and West sectors occur in open water around 0.5-1.5 km out from the shoreline. However, in the Inner and North sectors, the primary roosts (IN1 and N1) occur close to the shoreline and, on high tides, the roosts can occur over the intertidal zone. In three of the sectors, there was one primary roost site (IN1, N1 and W1) that held all the birds on nearly all counts, with secondary roosts in two of these sectors (IN2 and N2) that were only occasionally used. No secondary roosts were identified in the West sector. In the East sector there were three roost sites (E1, E2 and E3). All three of these roost sites were occupied on the majority of the counts, although E3 usually held small numbers of grebes (Figure 1). There appeared to be a seasonal shift between the E1 and E2 roosts: E2 was the main roost used at the start of the season, with birds gradually switching to E1 as the winter progressed (Figure 2). The East sector held the largest numbers of roosting grebes with mean counts ranging from 79-109, the Inner sector held the next largest numbers with mean counts ranging from 33-54, while the North and West sectors were

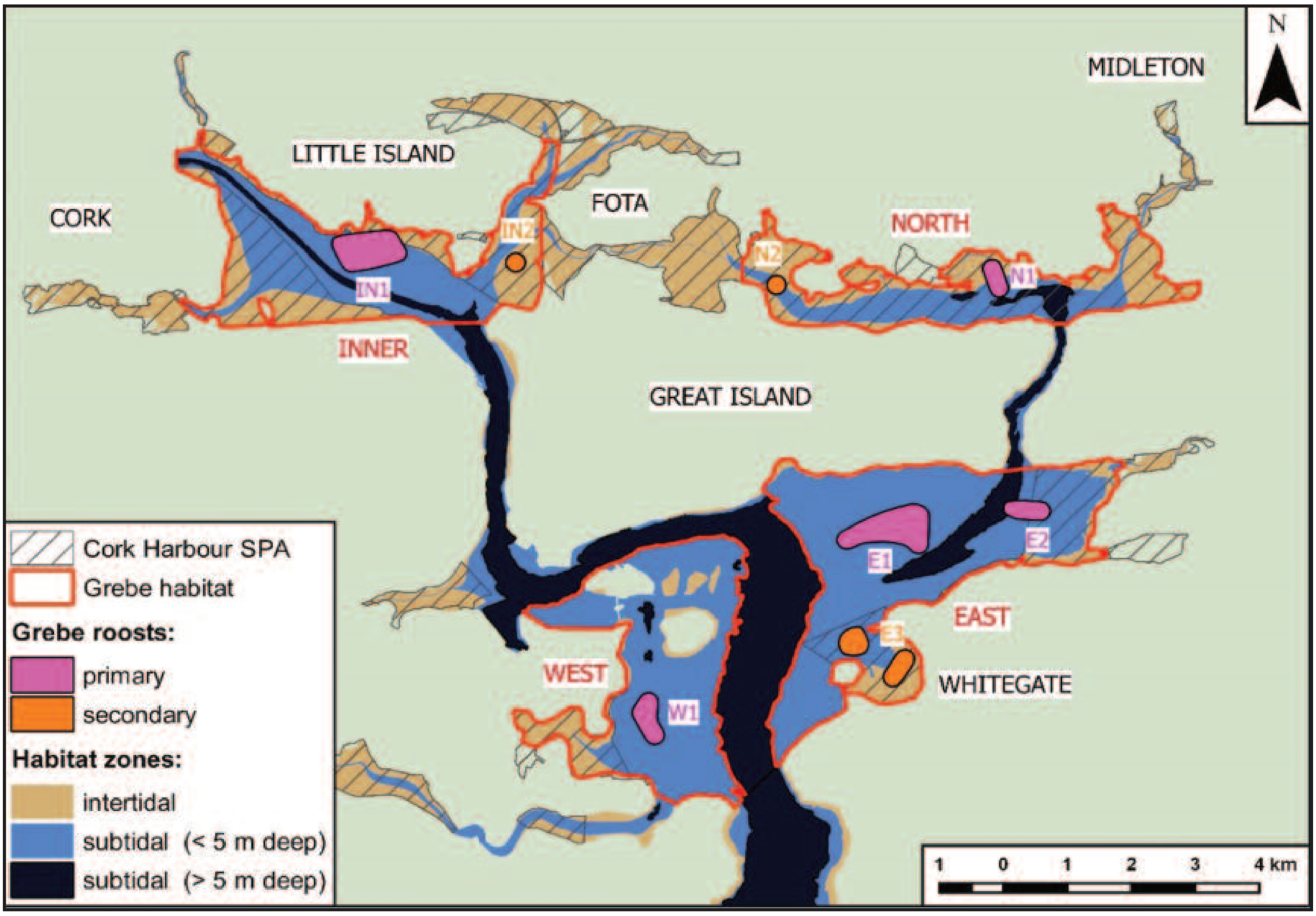

Figure 1. Nocturnal Great Crested Grebe roosting sites in Cork Harbour.

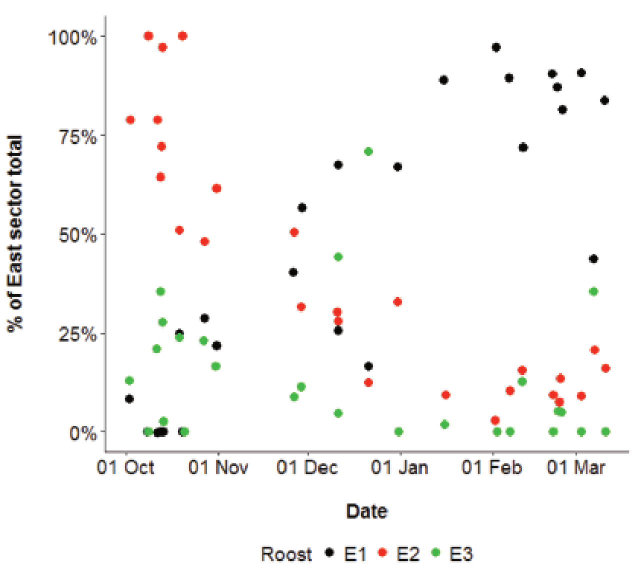

Figure 2. Occupancy of the three Great Crested Grebe roosting sites in the East sector of Cork Harbour. Combined data from 2014/15 to 2016/17; only includes dates with total counts of 35 or more grebes.

less favoured by grebes with mean counts of 5-16 for each sector in October and November (Table 1). However, these sectors held larger numbers in late winter, with mean totals of 26-49 in January and February, apart from the North sector in January and February 2017 (Table 1). The distribution of roosting grebes between the sectors was broadly similar to the distribution of foraging habitat, although the East sector held relatively higher densities compared to the North and West sectors (Table 2).

In Dungarvan Harbour Great Crested Grebes mainly occur in the Outer part of the harbour, and the whole area out to the eastern boundary of the Special Protection Area (SPA) is within their preferred depth zone, while small numbers may occur in the Inner part of the harbour (Figure 3). There is also potentially suitable grebe foraging habitat at Clonea Strand, but its usage by grebes is not known, as this area is not regularly covered by waterbird counts. In February and March 2015, the grebe roosts were mapped on evening low tides and occurred in the southern part of the Outer harbour below the tideline (D1; Figure 3). In November 2016 and January 2017, the grebe roosts were mapped on evening high tides and occurred a few hundred metres off the Cunnigar over the intertidal zone (D2; Figure 3). On most of

Table 1. Means (and ranges) of dusk roost counts of Great Crested Grebes in Cork Harbour.

Sector	Parameter	2014/15 Jan-Feb	2015/16 Oct-Nov	Jan-Feb	2016/17 Oct-Nov	Jan-Feb
East	mean	105	93	79	86	109
	range	(95-114)	(76-120)	(54-103)	(49-119)	(98-120)
	n	3	8	4	4	2
Inner	mean	33	35	50	54	35
	range	(20-42)	(24-50)	(41-64)	(42-64)	(28-47)
	n	4	4	3	3	3
North	mean	39	8	26	8	9
	range	(35-44)	(4-12)	(26-27)	-	-
	n	3	2	3	1	1
West	mean	35	5	39	16	46
	range	(30-39)	(3-9)	(35-45)	(11-21)	(44-47)
	n	3	3	3	2	2
Total	-	212	142	194	164	199

Table 2. Comparison of distribution of Great Crested Grebes in Cork Harbour with the availability of grebe foraging habitat. Two values are shown for the percentage of grebe habitat: the percentage of suitable subtidal habitat; and the percentage of suitable subtidal and intertidal habitat, with the intertidal habitat being weighted by a factor of 0.5 to reflect its availability to foraging grebes. The values for the percentages of the grebe population are calculated from the means of the mean values in Table 1 with the ranges in parentheses.

Sector	% of grebe habitat subtidal	intertidal and subtidal	% of grebe population
East	42%	37%	53% (41-66%)
Inner	21%	23%	23% (16-33%)
North	10%	15%	9% (5-18%)
West	27%	25%	15% (10-23%)

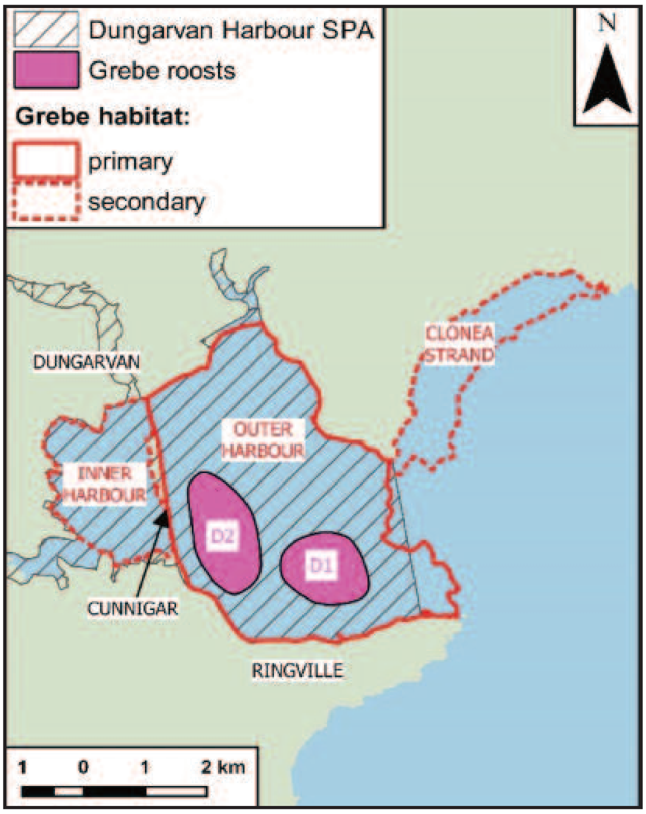

Figure 3. Nocturnal Great Crested Grebe roosting sites in Dungarvan Harbour.

the days surveyed the grebes still appeared to be assembling to roost as it became dark, so the mapped positions may exaggerate the size of the areas used for the roosts. Birds from these roosts may travel 5-6 km to feed in the Inner part of the harbour during the day. The numbers using these roosts ranged from 34-39 on three dates in February and March 2015, and 49-176 on four dates in November 2016 and January 2017.

Great Crested Grebes occur throughout Wexford Harbour and the whole of the Main harbour and Ferrycarrig Reach is within their preferred depth zone (Figure 4). Suitable grebe habitat also extends into the Outer bay outside the harbour and small numbers of grebes are regularly recorded off Raven Point. The only roost that I identified occurs in the northern part of the Main harbour around 1 km off Ardcavan. The numbers using this roost ranged from 70-115, apart from

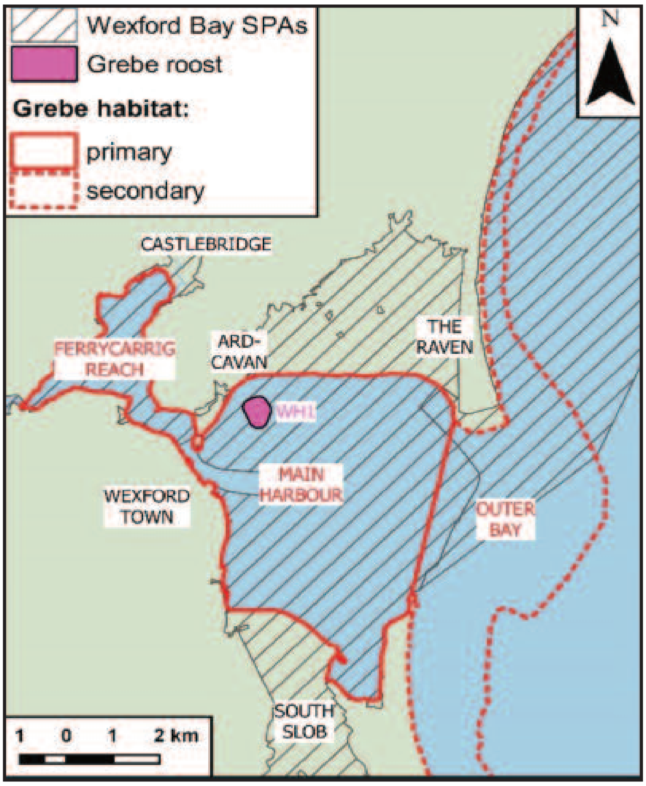

Figure 4. Nocturnal Great Crested Grebe roosting sites in Wexford Harbour.

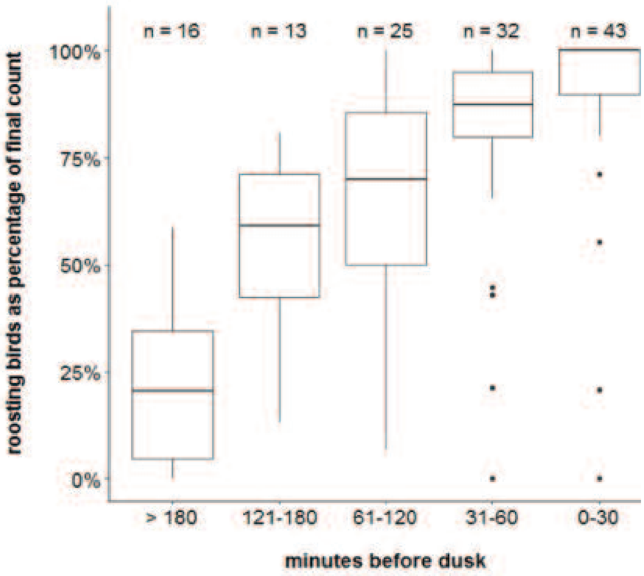

Figure 5. Boxplot showing build-up of dusk roosts of Great Crested Grebes, as percentages of roosting birds in the dusk roost locations in five time periods relative to dusk. Data is only included from counts where a final count was taken in the 0-30 minutes before dusk time period and another count was taken in at least one other time period.

one count of 15 birds at the start of the season. No roosts appear to occur in the Ferrycarrig Reach section of the harbour, and birds from the Ardcavan roost may travel 5-6 km to feed in Ferrycarrig Reach during the day. I did not search the southern part of the Main harbour for roosts.

Roosting behaviour

Roosting flocks of Great Crested Grebes can occur at any time, but are of erratic occurrence during the day. The regularity of the roosting behaviour, and the size of the roosting flocks, increases towards dusk (Figure 5). All grebes in an area will usually be assembled into one or more roosting flocks by dusk, apart from occasional lone feeding birds. However, in late December 2016 and early January 2017, the grebes in the Inner sector of Cork Harbour displayed atypical behaviour, with roosts breaking up and birds dispersing to feed as dusk approached (represented by the two most extreme outliers in the 0-30 and 31-60 minute categories in Figure 5).

Roosts typically form gradually and birds that are already in the general area of the roost often just stop feeding and gradually drift towards the nearest group. Birds that have been feeding away from the general area of the roost site actively swim towards the roosting area and I observed some birds travelling several kilometres. In Cork Harbour, I observed birds swimming from Whitegate Bay to the E1 roost (about 2-3 km), from the upper part of the Fota Channel to the IN1 roost (about 3-4 km), and from the western end of the North Channel to the N1 roost (about 3-4 km). In Dungarvan Harbour, I observed birds swimming from the Inner harbour to the D2 roost (about 4 km), and in Wexford Harbour I observed birds swimming from Ferrycarrig Reach to the WH1 roost (about 5 km). More generally, the distribution of grebe foraging areas in relation to roost sites indicates that commuting distances of 3-5 km are not uncommon (see above). Flight activity was very rare with only four observations of a total of 12 birds flying to the roosts, compared to 131 records of 987 birds swimming to the roosts.

As dusk approached groups gradually coalesced, usually by drifting together rather than actively swimming towards each other. Often they would have formed a single group by the time darkness fell. On some days, the birds were still in more than one group at this point, but probably continued to drift together after it was too dark to see them. Even when a single group had formed, the birds could continue to move up until, and after, dusk. For example, in October 2015 and 2016, the grebes in the East sector of Cork Harbour showed a consistent pattern of assembling into a single flock at roost E2, but, as darkness fell, beginning to actively swim, or drift, west.

The roosting grebes typically occurred in a compact flock, each bird a few bird lengths from its nearest neighbour (Plate 242). When the flock was stationary, the birds often drifted apart but then came back together. The flocks could also move in a co-ordinated fashion, either by drifting, or by actively swimming. During the day, the roosting grebes were often sleeping. As the night roosts assembled before dusk, the incidence of sleeping was often reduced due to movement of the flock and disturbance from new arrivals.

Only a few dawn watches were made, but the grebe roosting flocks appear to begin to break up before it gets light enough to see. By the time that I was able to make observations, the birds would usually be widely dispersed around the general roost location, with some birds actively swimming away from the roost and others drifting around.

Disturbance

Foraging Great Crested Grebes appear to be generally very tolerant of vessel activity. In Wexford Harbour, I observed numerous instances of boats travelling past foraging grebes within a few hundred metres without any discernible response from the grebes, although I did observe one instance of feeding grebes flushed by a boat when the boat drove through an area with grebes directly in its path. However, Great Crested Grebe roosting flocks appear to be much more sensitive to disturbance. I observed eleven instances of flocks being apparently disturbed by vessel activity (Table 3). These

Table 3. Observations of disturbance to Great Crested Grebe roosting flocks by vessel activity.

Date	Time	Site	Vessel type	Flock size	Disturbance response
03/10/2015	19:41	Cork Harbour (Inner)	Large ship	28	Flock close to navigation channel, broke up and swam into shore as ship approached.
12/10/2015	18:56	Cork Harbour (East)	RIB	54	Flock reacted to noise of boat when boat was over 1 km from flock and not visible to flock. Flock scattered with birds swimming rapidly away.
29/10/2015	17:55	Wexford Harbour	Cot	44	Flock broke up with birds swimming away from shoreline as cot approached along route parallel to shoreline.
31/10/2015	15:50	Cork Harbour (East)	RIB	24	Flock broke up and swam into shore.
31/10/2015	17:27	Cork Harbour (East)	Cabin cruiser	68	Flock broke up and swam into shore. The flock then gradually re-coalesced over a period of 10-20 minutes.
13/02/2016	16:37	Cork Harbour (West)	Inshore potting vessel	13	Flock broke up and swam into shore when boat was over 1.5 km from flock. The flock then re-coalesced over a period of 20 minutes.
28/02/2016	15:10	Cork Harbour (Inner)	Currach	9	Flock close to navigation channel disturbed by rowers and flew in towards the Little Island shore.
19/11/2016	14:45	Cork Harbour (Inner)	Small ship	24	Flock close to navigation channel broke up and swam in towards another flock which was further away from the navigation channel.
19/11/2016	15:45	Cork Harbour (Inner)	Trawler	35	Flock some distance from the navigation channel reacted to passage of vessel by becoming alert and swimming around, and a few diving, but flock not breaking up.
19/11/2016	16:40	Cork Harbour (Inner)	Small ship and trawler	40	Flock close to navigation channel broke up and became widely dispersed with the flock not reforming in the next 35 minutes before dusk.
07/01/2017	16:40	Cork Harbour (West)	Small boat	23	Flock began spreading out and some diving when hit by wake of boat going into Crosshaven.

mainly occurred when boats were heading on routes that would pass within a few hundred metres of the roosting flock and the birds could respond at distances of more than 1.5 km from the boat. The disturbance caused the roosting flocks to break up, with birds swimming rapidly away looking alert and looking around behind them, and with some birds diving and/or flapping their wings. Usually the grebes swam into the shore. On one occasion, the birds instead swam out away from the shore, but this was when the boat was approaching along the shoreline. I also observed a few instances of flocks breaking up and birds dispersing when there was no obvious disturbance source, which may be analogous to the way that roosting flocks of waders can "spook" for no apparent reason. The E1, E2 and IN1 roosts are in areas of Cork Harbour with regular vessel activity and I observed four incidences of boats passing within around 500 m of roosting flocks without any apparent disturbance response.

Comparison with I-WeBS counts

The mean grebe roost count totals for Cork Harbour in January and February 2015 and 2016 compare well with the total recorded in the co-ordinated counts in February 2015 and February 2016, respectively, indicating that the mean counts provide an accurate index of grebe numbers in Cork Harbour (Table 4). However, the mean grebe roost count totals for Cork Harbour were 1.6-4.3 times greater than the numbers recorded during the I-WeBS counts for the same periods, apart from November 2016, and the January and February totals are greater than the maximum number recorded during any I-WeBS count since the winter of 2002/03 (Table 4). In contrast to the mean grebe roost count totals (Table 1), the I-WeBS counts do not show consistent seasonal patterns across winters (Figure 6).

Table 4. Comparison between Irish Wetland Bird Survey (I-WeBS) count totals and dusk roost count totals of Great Crested Grebes in Cork Harbour. Data source for the I-WeBS count totals: annual summaries of Cork Harbour I-WeBS counts on www.gittings.ie.

	2014				2015				2016	
Count type	**Jan**	**Feb**	**Oct**	**Nov**	**Jan**	**Feb**	**Oct**	**Nov**	**Jan**	**Feb**
I-WeBS counts	94	114	87	-	59	46	87	159	126	87
Sum of grebe roost count means	212		143		199		164		199	
Co-ordinated roost counts	-	220	-	-	-	190	-	-	-	-

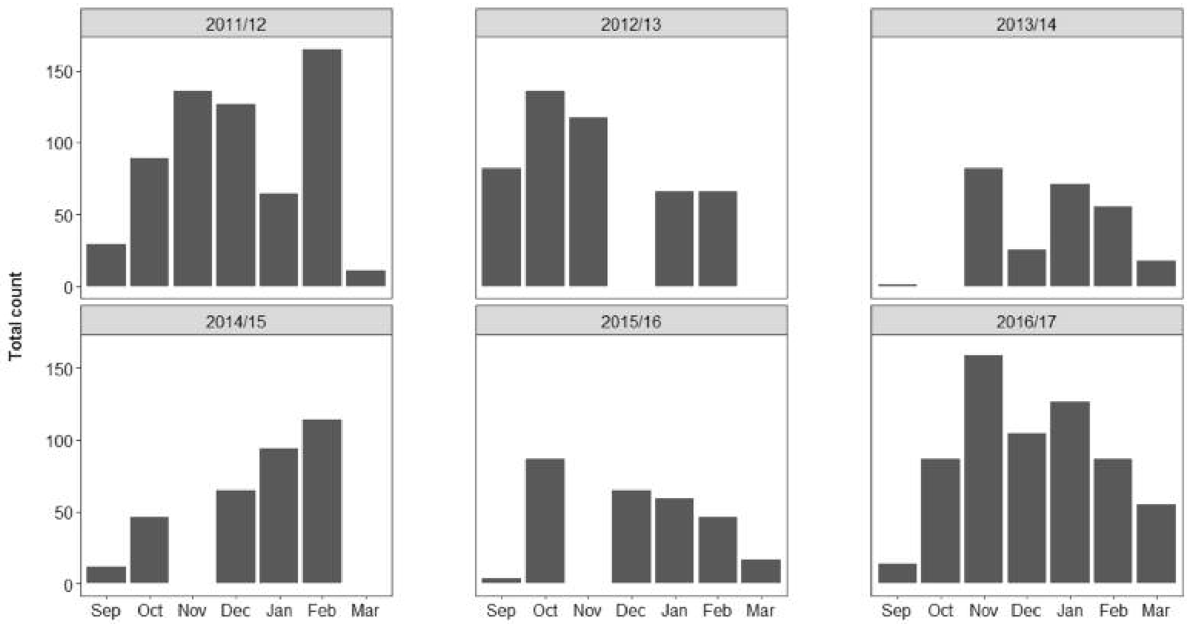

Figure 6. Total numbers of Great Crested Grebes recorded in Irish Wetland Bird Survey (I-WeBS) counts of Cork Harbour. Counts are only included for months when there was complete coverage of the subsites that hold significant numbers of Great Crested Grebes. Data source: annual summaries of Cork Harbour I-WeBS counts on www.gittings.ie.

Discussion

Daytime aggregations of Great Crested Grebes are not uncommon. For example, Konter (2008) states that "outside the breeding season, loose flocks of Great Crested Grebes roosting together are a common picture on many bodies of water", while Simmons (1968) describes how loose flocks of up to 250 non-feeding grebes form on Chew Valley Lake in winter. In Cork Harbour, as documented in this study, daytime flocks are of regular occurrence. However, the assembly of nocturnal roosts appears to be a distinct behavioural phenomenon. It differs from daytime roosting in the following ways: flock sizes are larger; typically all birds in the area will have assembled into a single flock; and birds actively travel long distances to join the flocks. In contrast to the above literature references to daytime flocking, which refer to "loose flocks", the birds in the night roosts typically form compact flocks. However, the daytime roosts that I observed in Cork Harbour often also involved compact flocks (as illustrated in Plate 242).

Within each site, night roosts appear to consistently occur in specific areas, although there is some variation in precise locations. The distribution of the mapped locations in the two roosts that were mapped most regularly (E1 and IN1) have a maximum separation of around 1.0-1.5 km. However, this may overestimate the degree of variation due to inaccuracies involved in the triangulation method if flocks move position between the times when the bearings are taken.

The distribution of foraging habitat in relation to the roost sites indicate that Great Crested Grebes frequently swim several kilometres to the roost sites, and this was confirmed by direct observation of swimming birds. Given a theoretical maximum speed for a swimming bird of 0.7 m/sec (Fjeldså 2004), these grebes must spend an hour or more swimming to their roost sites. Therefore, some benefit from attending the roost would be expected. Potential benefits of communal night roosting in birds include defence against predators, reduced thermoregulation demands, information sharing (particularly on foraging areas), and maintaining family relationships (Beauchamp 1999). It seems unlikely that Great Crested Grebes have any significant predation risk at night, and the birds are not close enough together to benefit from reduced thermoregulation demands. Head-shaking displays occur frequently amongst grebes in the roosting flocks, so some social component to roosting behaviour is likely. The grebes do rely on a prey resource (fish) that is probably unpredictably distributed. Therefore, information sharing might seem to be an explanation for the formation of nocturnal communal roosts. However, as pointed out by Beauchamp (1999), while unsuccessful foragers will benefit from communal roosting, there is no reason why successful foragers need to travel to communal roosts. However, the following scenario might explain some aspects of the roosting behaviour: if it is assumed that the main grebe roosts occur in usually productive foraging areas, then the birds that travel far from the roosts are likely to have been the most

Plate 242. Part of a raft of 120 Great Crested Grebes off the southern shore of Great Island, Cork Harbour, at 09:00 hours on 26 January 2010. While this photo was taken during the day, the arrangement of the flock is typical of dusk roosts (Seán Cronin).

Plate 243. Great Crested Grebe (Dick Coombes).

unsuccessful foragers and have the most incentive to return to the roost to benefit from information sharing. This could also explain the erratic occurrence of roosts in the secondary roosting sites with such roosts only forming when birds that have travelled a long distance from the main roosting location have been relatively successful and decide to remain near their foraging location rather than return to the main roost.

However, the above scenario still does not explain why birds form flocks rather than remaining as dispersed individuals while another factor requiring explanation is why the birds consistently choose the same general location for the nocturnal roosts. If there was some particular advantage to roosting in these locations, this might answer both these questions. A possible explanation could be that the roosting locations are in areas with weak tidal currents, minimising the energy expenditure required to maintain their position over the night when they are not otherwise active. The roosting location in Wexford Harbour is an area with relatively slack tidal currents on the ebb and flood tides (tidal current model produced for the UISCE project; Bord Iascaigh Mhara, unpublished data), but I was not able to find similar data for Cork Harbour or Dungarvan Harbour.

Great Crested Grebes are difficult to count in coastal waters as birds are often widely dispersed and spend much of their time diving. The routine waterbird monitoring of Cork Harbour through I-WeBS counts has produced highly variable counts within winters and does not show consistent seasonal patterns between winters (Figure 6). The highest I-WeBS counts occurred when the counters happened to pick up large daytime roosting flocks, or when the counts took place close to dusk. While my roost count data only covers three winters, and only includes early winter counts for two of those winters, there is a high degree of consistency between winters in the total numbers recorded, the distribution between sectors, and the seasonal patterns. The comparison of the I-WeBS counts and the roost counts indicate that the I-WeBS counts usually significantly underestimate the Cork Harbour population and do not provide reliable information on seasonal changes in numbers. These comparisons show that, at coastal sites, dusk roost counts provide a better method of monitoring Great Crested Grebe numbers than routine waterbird counts. The dusk roost counts also appear to provide useful information on the distribution of grebes within Cork Harbour with the distribution of grebes between the roosts reflecting the amounts of suitable foraging habitat around the roosts. Seasonal changes in the usage of the roosts probably reflect seasonal changes in the distribution of foraging grebes: for example, the shift over the winter from roost E2 to E1 in the East sector of Cork Harbour was reflected in the numbers of foraging grebes recorded in daytime counts around these two roost sites. However, dusk roost counts, particularly for the more distant roosts such as E1 in Cork Harbour, require relatively good weather conditions with a sea state of 3 or less, and there is often a very narrow time window to get a count between the birds assembling at the roost and the light fading.

There are conservation implications to the recognition of night roosting as a component of grebe behaviour. As the grebes appear to deliberately select specific areas for their nocturnal roosts, it is clearly important that these areas are identified and included in SPAs designated for their grebe populations. It is notable that four of the five primary grebe roosts in Cork Harbour are outside, or partly outside, the SPA, while the majority of the grebe foraging habitat is also outside the SPA (Figure 1). My observations also indicate that roosting grebes are sensitive to disturbance from vessel activity. As roosts can re-assemble within a short period of time, occasional vessel passage is unlikely to have a significant impact. However, sustained activity (e.g. night fishing) could have a more serious impact. Therefore, environmental impact studies for plans or projects that may include crepuscular and/ or nocturnal vessel activity should consider the potential impact on roosting grebes.

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Using within-site level trends of non-breeding waterbirds as a monitoring tool: a case study using data from Sruwaddacon Bay, County Mayo

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Waterbirds wintering at Sruwaddacon Bay, County Mayo, have been subject to intensive monitoring since 2002 in connection with the Corrib Gas Project. Sruwaddacon Bay is part of the larger Blacksod Bay and Broad Haven site complex which is monitored each winter by the Irish Wetland Bird Survey (I-WeBS). The combination of detailed site and site complex data enabled trends in waterbird numbers at these two scales to be compared and contrasted, while site trends were examined in the light of national, all-Ireland and international level trends. The site trends calculated for Sruwaddacon Bay were positive (increasing/ stable numbers) for six of the nine species, with declines noted for three (Ringed Plover *Charadrius hiaticula,* Dunlin *Calidris alpina* and Redshank *Tringa totanus*). The decline for Dunlin is consistent with the trend observed at site complex and national level and it is likely that broad-scale declines are driving the decline at site level. However, the trends for Ringed Plover were driven by a low index in the final season, having been largely stable previously. Therefore, future monitoring will be important to investigate this further. We discuss the suitability and use of trend analyses for waterbirds at site level, as well as their ability to provide a means of assessing population change over time and to determine conservation status. In addition, they may provide an important early warning system to identify the beginning of potential long-term declines, whilst overall being a useful tool in practical ecological monitoring and management programmes.

Introduction

Blacksod Bay and Broad Haven Special Protection Area (SPA) (Birds Directive 2009/147/EC) is a large wetland complex in north-west County Mayo. Ten waterbird species are listed for this SPA, which include nine wintering (non-breeding) species (Light-bellied Brent Goose *Branta bernicla hrota,* Common Scoter *Melanitta nigra,* Red-breasted Merganser *Mergus serrator,* Great Northern Diver *Gavia immer,* Ringed Plover *Charadrius hiaticula,* Sanderling *Calidris alba,* Dunlin *Calidris alpina,* Bar-tailed Godwit *Limosa lapponica* and

Plate 244. Red-breasted Merganser (Clive Timmons).

Plate 245. Curlew (Colum Clarke).

Curlew *Numenius arquata*) and one breeding species (Sandwich Tern *Sterna sandvicensis*). Light-bellied Brent Goose and Great Northern Diver occur in numbers of international importance and all other species occur in numbers of national importance.

One constituent of the wetland complex is Sruwaddacon Bay, located on the north-eastern side of Broad Haven Bay. The waterbirds of this bay have been subject to monitoring since 2002 in connection with the Corrib Gas Project in accordance with the extensive Ecological Monitoring Programme of the Environmental Management Plan for the project. Monitoring has been undertaken through the Project Ecologist (Ecological Advisory and Consultancy Services – EACS) on behalf of their client (Shell E. & P. Ireland Ltd) (SEPIL) and will continue in the post-construction (operational) phase for a further three years. BirdWatch Ireland was commissioned by SEPIL through their Project Ecologist to undertake an analysis of the large body of data on waterbirds.

Ecological monitoring of Sruwaddacon Bay has been essential to site management through what has been a period of near continuous adjacent development since 2002 and to ensure the effectiveness of mitigation measures that were put in place to prevent and/ or minimise disturbance to waterbirds. Given that the site is part of the much larger Blacksod Bay and Broad Haven SPA implies that its conservation status should remain in favourable condition, which in part is defined by the population status of selected species. This necessitated a monitoring scheme that could track changes in the status of waterbirds at the site, and an analysis method that would allow the status of waterbird populations to be determined.

In recent years, waterbird population trends have been produced for individual coastal wetland sites that are SPAs in order to monitor populations at site level and to assist at assigning conservation status (e.g. NPWS 2013). These analyses were conducted using data collected by the Irish Wetland Bird Survey (I-WeBS). However, where data exist for smaller areas or 'subsites' and where these data have been collected in a consistent and standardised manner, there is no reason why waterbird population trends cannot be calculated for small-scale areas (e.g. Wright *et al.* 2008, Austin & Calbrade 2010). Therefore, by following the principles used to calculate SPA trends, we assessed the trends of wintering waterbirds of Sruwaddacon Bay, and examined these against their status at the wider site complex level, as well as at national and flyway scales.

Study area and methods

Sruwaddacon Bay (Grid reference F830375) is a small bay, approximately 320 ha in size, located on the north-eastern side of Broad Haven Bay. The bay is characterised by a high-energy (strong currents and wave action) environment which results in the sediments being dominated by sand (SEPIL 2014). Seaward sections of the bay comprise rippled mobile sand sediments with a sparse benthic infauna while the remainder of the site comprises more poorly-drained sandflats, with gravelly mixed sediments and varying amounts of mud confined to inner estuarine areas and small inlets away from the main channel (RSK Group PLC 2007). Saltmarsh habitat occurs around the majority of the upper intertidal area and below this zone there is often a wrack zone comprising a mixture of brown and green algae.

Sruwaddacon Bay forms a constituent part of the Blacksod Bay and Broad Haven SPA which is a site complex comprised of large, sheltered and mostly shallow inlets which stretch south and north respectively, either side of a causeway linking the mainland to the Mullet Peninsula (Figure 1). Blacksod Bay, to the south, is the larger of the two bays and is more complex with many small bays and inlets dominated by sandy sediments. The inner section of Broad Haven Bay, to the north, is dominated by coarse sediments to sandy muds and opens out into a subtidal bay with a rocky shoreline.

Waterbird monitoring in the Sruwaddacon Bay area has been ongoing since 2002 in connection with the Corrib Gas Project. This is in line with the Corrib Development Biodiversity Action Plan (BAP)[1] which, under Action 10[2], commits SEPIL to setting up partnerships, including for the

[1] http://www.shell.ie/content/dam/shell-new/local/country/irl/downloads/pdf/corrib-development-biodiversity-action-plan-2014-2019.pdf.

[2] Corrib BAP Action 10: "Set up partnerships with relevant and appropriate bodies such as academic institutions, NGOs, peer experts and others".

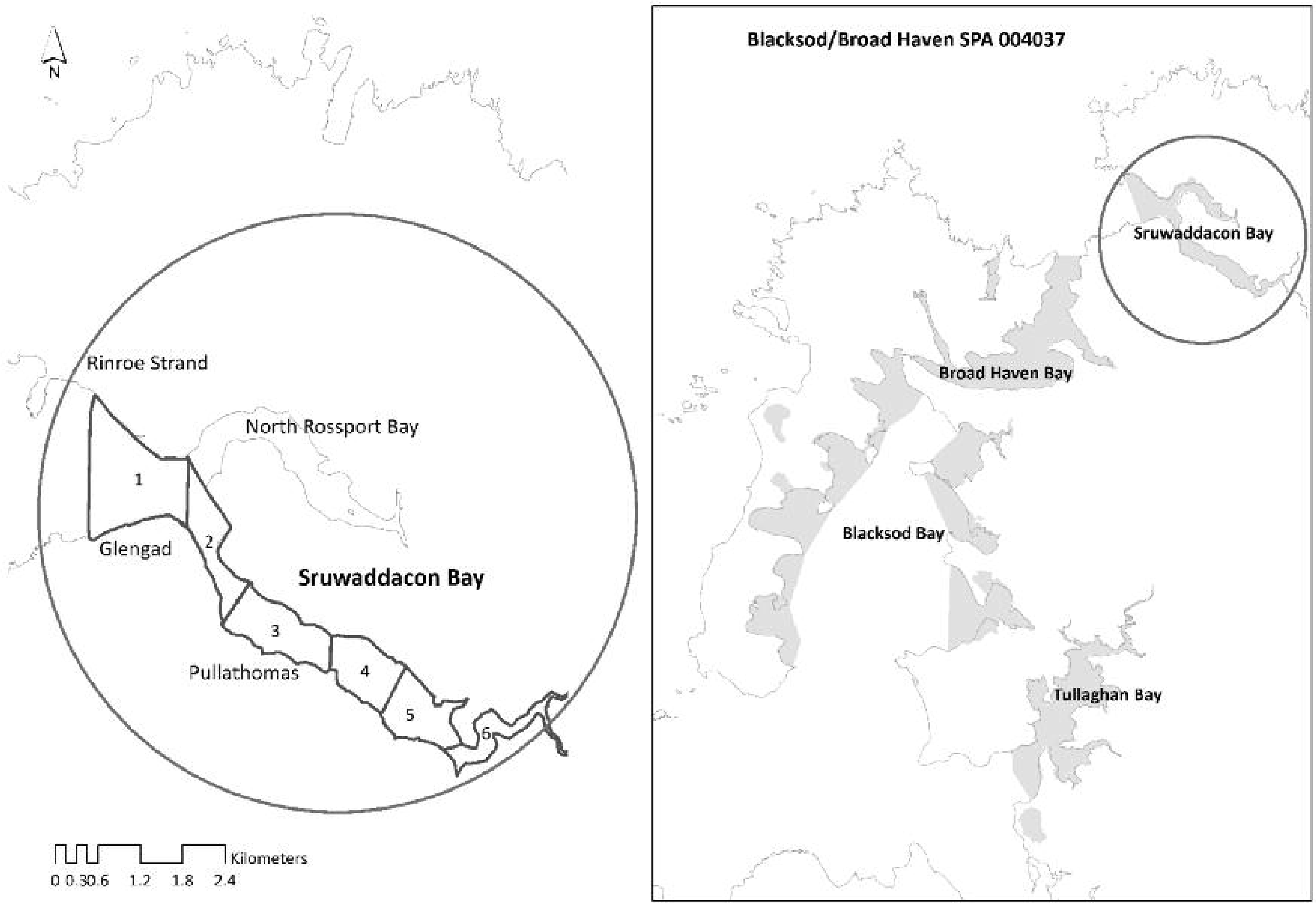

Figure 1. Sruwaddacon Bay and the wider Blacksod Bay and Broad Haven site complex, County Mayo.

independent analysis of the monitoring data. This paper uses waterbird data collected during the period 2002/03 to 2012/13. Surveys were undertaken across the period October to March in the 2002/03 survey season and thereafter extended into April of each season. Surveys were initially conducted at fortnightly intervals and increased to weekly intervals during the pre-construction and construction phases. Standard waterbird counting techniques were used throughout, using 'look-see' methodology (Bibby *et al.* 2000) to count waterbirds within a series of count sections (Figure 1). A series of set vantage points were used and the bay was counted in a systematic manner by two field surveyors, typically within a two-hour period, taking care to avoid double-counting due to bird movements. Counts were conducted during both low and high tide periods (i.e. two hours either side of low tide or high tide respectively) typically on two consecutive days per survey period. Field surveys were undertaken when suitable weather conditions prevailed, and using standard optical equipment (i.e. telescope and binoculars).

Wintering waterbirds have been monitored across the wider Blacksod Bay and Broad Haven SPA site complex by I-WeBS since the winter of 1994/95. The site complex comprises two separate I-WeBS count areas as follows (1) Blacksod and Tullaghan Bays; (2) Broad Haven and Sruwaddacon Bays. With an approximate survey area of 10,808 ha (Crowe 2005), the site is difficult to count in terms of size, access, visibility and often rapidly changing weather conditions. Consequently, count coverage has been variable across the years and the coverage of Blacksod and Tullaghan Bays has exceeded that of Broad Haven and Sruwaddacon Bays.

Waterbird low tide and high tide peak counts were compiled for each season and the five-year mean peak of counts was calculated for the latter five-year period 2008/09-2012/13. The relative importance of Sruwaddacon Bay, in the context of the overall site complex, was then assessed by comparison with I-WeBS data. The calculation of waterbird trends for Sruwaddacon Bay ('site trends') largely followed the methods used to produce trends for coastal SPA sites and those at national level (e.g. as reported by Crowe *et al.* 2012). Both high tide and low tide trends were calculated for the most abundant waterbird species that occurred in the bay. Peak monthly waterbird counts were at first summed across a standard subset of months (in most cases November to February) and annual indices calculated. These annual indices were then modelled using Generalised Additive Models (GAM) which fitted a smoothed curve to the indices and served to dampen annual fluctuations (Fewster *et al.* 2000, Atkinson *et al.* 2006) enabling better interpretation of trends and the calculation of proportional change in population size between one season and another. The calculation of long-term site trends was not possible as waterbird counts were not carried out at Sruwaddacon Bay during the 2006/07 season and raw data were not available for the 2007/08 season. Site trends were therefore calculated for the latter short-term (five-year) period 2008/09-2012/13. The short-term change was calculated from the GAM-fitted indices using: change = $((I_y - I_x) / I_x)$ x 100 where Iy is the index value for the final year and Ix is the index value for the start year. The trends produced therefore represent the percentage change in index (population) values across the specified time period; positive values equating to increases in population size while negative values reflect a decrease in population size. For waterbirds, threshold levels of >25.0% and >50.0% are generally used to highlight 'medium' and 'high' declines respectively (e.g. Lynas *et al.* 2007, Leech *et al.* 2002), while the 'intermediate' range (1.0-24.9% decline) allows for natural fluctuations and represents the range within which relatively small population declines have the potential to be reversible (Leech *et al.* 2002). These thresholds have also been adopted for assigning conservation condition (status) to waterbird populations at Irish coastal SPA sites, and these thresholds were therefore used to assess the conservation status of the selected species at Sruwaddacon Bay.

Waterbird trends for Sruwaddacon Bay were also examined in the context of trends for the wider site complex, plus national, all-Ireland and international trends for the selected waterbird species. Due to the variable count coverage of the 'Broad Haven and Sruwaddacon Bays' I-WeBS subsite (see above), data for the 'Blacksod and Tullaghan Bays' I-WeBS subsite were used to serve as a proxy for the wider Blacksod Bay and Broad Haven site complex (hereafter called 'site complex trends'). The analysis methods used for these wider scales are described in Boland and Crowe (2012) and include an additional step to that described above, whereby raw counts are first modelled using a multiplicative log-linear model (Underhill & Prŷs-Jones 1994) and the fitted values are used to impute values where counts are missing or are of poor quality (e.g. underestimated) (this step being unnecessary for Sruwaddacon Bay given the frequency and robustness of the dataset).

Results

Species composition and numbers

Across the long-term data period 2002/03-2012/13, 42 waterbird species were recorded at Sruwaddacon Bay (range 21-35). This compares with a total of 84 species recorded for the I-WeBS count subsite 'Blacksod and Tullaghan Bays' during the data period 1994/95-2012/13. Light-bellied Brent Goose was by far the most numerous species recorded within Sruwaddacon Bay and regularly occurred in numbers of all-Ireland importance and on occasion in numbers of international importance. Of note is that numbers could represent a relatively large proportion of those occurring across the wider site complex (Blacksod and Tullaghan Bays serving as a proxy for the wider Blacksod Bay and Broad Haven site complex) (Table 1).

Oystercatcher *Haematopus ostralegus* and Curlew were two of the most regularly occurring species, being recorded in all counts and in good numbers and their high tide peak counts represented 35% and 28%, respectively, of the numbers occurring across the site complex in the same season. At low tide, Ringed Plover occurred on occasion in numbers of all-Ireland importance, while the peak high tide count represented up to 6% of the numbers occurring across the wider site complex. Of further note were Mallard *Anas platyrhynchos*, Black-tailed Godwit *Limosa limosa,* Redshank *Tringa totanus* and Greenshank *Tringa nebularia,* all of which occurred within the bay in high proportions relative to the peak count across the wider site complex in the same season (Table 1). Of the nine wintering species listed for Blacksod Bay and Broad Haven SPA, only the Common Scoter did not occur on a regular basis within Sruwaddacon Bay, while the Bar-tailed Godwit occurred in very low numbers (maximum 13; 2011/12).

Species trends

Of the nine waterbird species for which trends were calculated for Sruwaddacon Bay (Table 2), five (Light-bellied Brent Goose, Mallard, Red-breasted Merganser, Oystercatcher and Curlew) showed a positive (increasing) trend over the short-term based on high tide data. With the exception of Oystercatcher, these species also showed a positive trend based on low tide data. Although Sanderling occurred too infrequently at high tide to enable trend analysis, this species showed a positive trend based on low tide data (Table 2). These six species can therefore be described as having a favourable conservation status at site level. The trends for Light-bellied Brent Goose (Figure 2a), Red-breasted Merganser and Sanderling were similarly positive at the site complex, national and all-Ireland levels. Site numbers of Oystercatcher at high tide showed a progressive increase since

Table 1. Five-year mean peak data for (a) Blacksod and Tullaghan Bays, (b and c) Sruwaddacon Bay high tide and low tide data for non-breeding waterbird species listed for Blacksod Bay and Broad Haven SPA (shaded blue) and other regularly occurring waterbird species, (d) the overall high tide peak count recorded at Sruwaddacon Bay during the period 2008/09-2012/13 and (e) importance of the peak count in column 'd' relative to the peak count recorded in the same season by I-WeBS.

Waterbird species	BoCCI Category[1]	(a) Blacksod & Tullaghan Bays[2]	(b) Sruwaddacon Bay[3] HT	(c) Sruwaddacon Bay[3] LT	(d) Peak HT count recorded at Sruwaddacon Bay (season)	(e) % relative to peak count-Blacksod & Tullaghan Bays
Light-bellied Brent Goose *Branta bernicla hrota*	Amber	658 (i)	252	233	344 (2008/09)	51
Mallard *Anas platyrhynchos*	Green	105	32	19	95 (2010/11)	95
Common Scoter *Melanitta nigra*	Red	494 (n)	0	0	-	-
Red-breasted Merganser *Mergus serrator*	Green	70 (n)	8	8	12 (2011/12)	11
Great Northern Diver *Gavia immer*[4]	Amber	79 (n)	3	3	4 (2011/12)	4
Oystercatcher *Haematopus ostralegus*	Amber	297	80	57	117 (2011/12)	35
Ringed Plover *Charadrius hiaticula*	Green	595 (n)	13	117 (n)	46 (2010/11)	6
Sanderling *Calidris alba*	Green	285 (n)	14	22	37 (2008/09)	14
Dunlin *Calidris alpina*	Red	687 (n)	24	55	52 (2010/11)	14
Black-tailed Godwit *Limosa limosa*	Amber	2	13	10	24 (2012/13)	100
Bar-tailed Godwit *Limosa lapponica*[4]	Amber	627 (n)	5	8	13 (2011/12)	1
Curlew *Numenius arquata*	Red	471 (n)	83	79	103 (2010/11)	28
Greenshank *Tringa nebularia*	Green	31 (n)	9	9	11 (2011/12)	35
Redshank *Tringa totanus*	Red	173	44	55	76 (2010/11)	27
Turnstone *Arenaria interpres*	Green	126 (n)	19	13	28 (2012/13)	38

[1] BoCCI category (after Colhoun & Cummins 2013)(i) denotes numbers of international importance (after Wetlands International 2012)
[2] 5-year mean peak for the period 2008/09-2012/13 (data from I-WeBS)(n) denotes numbers of all-Ireland importance (after Crowe & Holt 2013)
[3] 5-year mean peak for the period 2008/09-2012/13 (site monitoring data)
[4] Annex I species

the base year while also increasing at site complex level (long-term) against a background of declining and stable trends at national and all-Ireland levels respectively.

Mallard exhibited an increasing site trend against declining trends at site complex, national and all-Ireland level. The high tide index for Mallard was particularly high in 2010/11 and was attributed to a relatively large count of roosting birds recorded in February 2011, while the annual site index doubled in most seasons since the base year (Figure 2b). The same pattern was evident for Curlew, with a consistent increase in the annual high tide index over time (Figure 2c), against a background of sustained decline at site complex, national and all-Ireland level. To examine this trend further we plotted the annual low tide peak count for Curlew for the site across the longer data period (2002/03-2012/13) (Figure 3). This revealed that the increase in numbers was only applicable in the short-term. Site numbers have varied considerably between seasons, sometimes doubling or halving, but the trend line does suggest a decline over the long-term with peak numbers recorded in the two most recent seasons being lower than those recorded in the 2002/03 and 2005/06 seasons, albeit when survey effort was lower.

Declining site trends and unfavourable conservation status were exhibited by three species, Redshank (high tide

Table 2. Population trends for non-breeding waterbird species listed for Blacksod Bay and Broad Haven SPA (shaded blue) and other regularly occurring waterbird species at Sruwaddacon Bay, in the context of the wider site complex, national, all-Ireland and international trends (scientific names as in Table 1).

Waterbird species	Sruwaddacon Bay (short-term)[1]		Site complex trend (long-term)[2]	Site complex trend (short-term)[3]	National trend[4]	All-Ireland trend[5]	International trend[6]
	HT data	LT data					
Light-bellied Brent Goose	+ 61	+ 22	+ 152	+ 91	+ 89	Increasing	Increasing
Mallard	+ 79	+ 32	- 5	- 5	- 16	Declining	Unknown
Red-breasted Merganser	+ 144	+ 85	+ 24	+ 57	+ 6	Stable	Unknown
Oystercatcher	+ 62	- 16	+ 34	- 7	- 13	Stable	Declining
Ringed Plover	-	- 31	+ 31	+ 29	- 6	Stable	Fluctuating
Sanderling	-	+ 320	+ 235	+ 79	+ 65	Stable	Increasing?
Dunlin	-	- 33	- 65	- 34	- 39	Declining	Stable
Curlew	+ 88	+ 43	- 19	- 3	- 22	Declining	Declining
Redshank	- 38	- 1	+ 38	+ 25	- 10	Stable	Stable/Increase

1 Sruwaddacon Bay - population trend analysis for the period 2008/09-2012/13 (based on high tide (HT) and low tide (LT) data
2 Site complex - long-term population trend analysis for the period 1995/96-2010/11
3 Site complex - short-term population trend analysis for the period 2004/05-2010/11
4 National trend - % change 2002/03-2012/13 (data from I-WeBS)
5 All-Ireland trend - where a species is deemed increasing or declining if annual rate of change is equal to or greater than (+/-) 1.2% (after Crowe & Holt 2013)
6 Current international trend (after Wetlands International 2012)

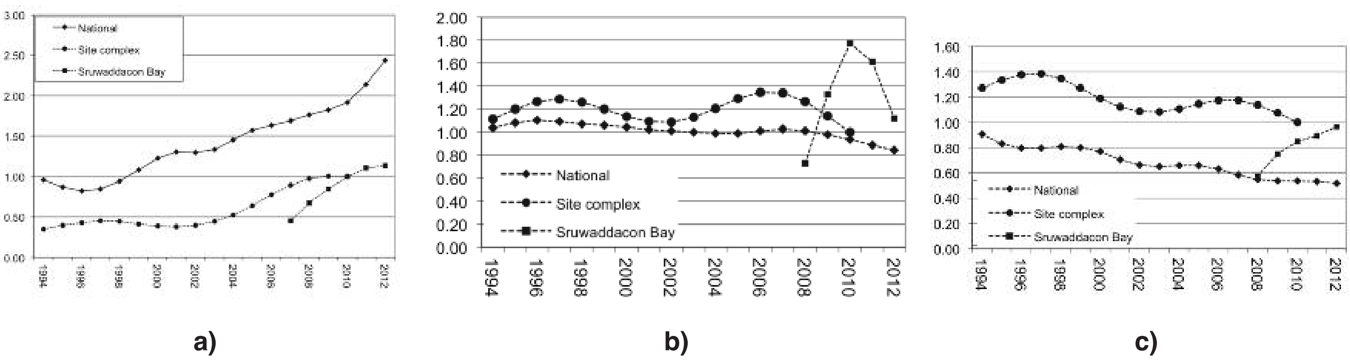

Figure 2. Comparative trends (smoothed indices) for waterbirds at national level (1994/95-2012/13), site complex level (1994/95-2010/11) and for Sruwaddacon Bay (2008/09-2012/13) (a) Light-bellied Brent Goose, (b) Mallard and (c) Curlew.

data), Ringed Plover and Dunlin (low tide data). Numbers of Redshank at low tide were stable. The declining site trend for Ringed Plover contrasts to the observed increase at site complex level and can be attributed wholly to the site index for 2012/13, which was the lowest in the dataset. The decline observed for Dunlin is consistent with trends observed at site complex, national and all-Ireland levels. However, as for Ringed Plover, this trend is highly influenced by the site index for 2012/13, which was the lowest in the dataset.

Discussion

This paper demonstrates that the methods used to produce smoothed site and site complex, national and all-Ireland trends can be extended to produce trends for small areas where, as in the case of Sruwaddacon Bay, data have been collected in a consistent and standardised manner (e.g. Austin & Calbrade 2010). The methodology can therefore serve as a useful tool and has wide applications where there is a need to

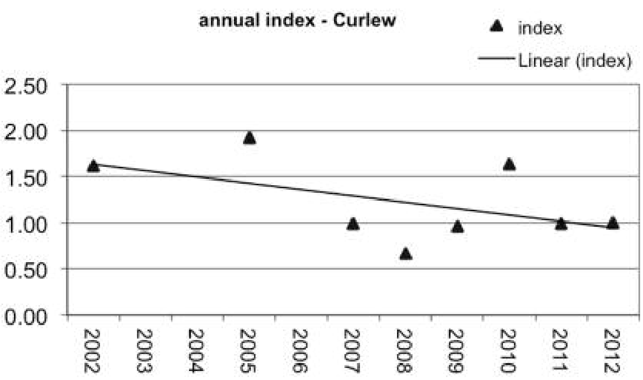

Figure 3. Annual low tide peak counts of Curlew at Sruwaddacon Bay.

monitor populations and assess conservation status in a consistent manner (e.g. Lewis *et al.* 2017). In addition, comparing the results with national and all-Ireland trends can give some insight into the possible causes of recorded declines. For example, if a species had a declining trend at site level (Sruwaddacon Bay) and was also exhibiting a declining trend at site complex and at national level, then it would be reasonable to expect that the underlying causative factors were external to Sruwaddacon Bay. Conversely, if a species showed a declining site trend not observed across the wider site complex then this may point to within-site causative factors. It would be most useful to identify where several species were undergoing declines at site level, but where these declines were not observed across the wider site complex. This has particular relevance for Sruwaddacon Bay, because the site has been the focus of continuous adjacent development, in the form of the Corrib Gas Project, for the past decade. Various site-specific measures have been employed during this period that seek to prevent or reduce disturbance and other impacts upon waterbirds, such as screening to reduce noise and light emanating from the construction compound into the bay, and the use of green artificial 'bird-friendly' lighting (Poot *et al.* 2008). Sadly, long-term trends were not available for Sruwaddacon Bay as a complete raw dataset dating back to 2002/03 was not available, but short-term trends can be useful in identifying the start of population declines, particularly if related to within-site features that have changed (e.g. increased levels of human disturbance).

On the whole, the trends calculated for the selected study species at Sruwaddacon Bay were positive (increasing/ stable numbers). For those species that exhibit a declining trend (Ringed Plover, Dunlin and Redshank), one of these, Dunlin, has exhibited a marked downwards trend across wintering sites monitored by I-WeBS (Boland & Crowe 2012), while similar declines have been observed in Britain and Northern Ireland (Holt *et al.* 2012). It is likely therefore that these broad-scale declines in Dunlin are driving the observed decline at site level. However, the indices for Dunlin, Ringed Plover and Redshank were at their lowest in the final season (2012/13), a factor which drove the trend for decline in all three species. Annual indices for these species for the four previous seasons remained largely stable. Such nuances in the trend data reveal how appropriate analysis can be useful in highlighting the start of a possible long-term downward trend. While such a trend could point to a within-site causative factor during 2012/13, such as lower macroinvertebrate abundances, in the case of Ringed Plover and Redshank, the declining site trend against an increasing site complex trend could simply indicate a re-distribution of the species within the site complex; particularly as the national and all-Ireland trends are relatively stable. Only future monitoring will determine whether this downward trend continues or if numbers return to their former levels.

The increase in numbers of Mallard at the site has occurred against a background of declining numbers at site complex, national and all-Ireland level. The particularly high count recorded in February 2011 may have been due to the re-distribution of birds following an extremely cold weather spell in November and December 2010 (Met Éireann 2010) but the overall increases observed suggest that within-site conditions are attractive for this species, despite the adjacent on-going development, which suggests that mitigation measures employed to reduce or avoid disturbance to waterbirds have been effective. Similarly, the positive site trend for Curlew occurred against a background of sustained decline at site complex, national and all-Ireland level, and on further examination, a long-term trend for decline at site level. However the short-term positive trend does nonetheless suggest that conditions at the site are attractive for this wader relative to the wider site complex.

As for larger scales, a standardised method of data collection is required to enable trend analysis at smaller scales, including as at Sruwaddacon Bay, a minimum of three months of data per season is required to provide reliable estimates of population change (Atkinson *et al.* 2006). The interpretation of site trends is also an important consideration. Firstly, it is important to recognise that the numbers of birds underlying the site trends are lower than those underlying the site complex trends. For example, a hypothetical 50% decline in numbers at Sruwaddacon Bay (e.g. 100 birds to 50 birds) would be much less significant than a 50% decline at site complex level (e.g. 1,000 birds to 500 birds) (Leech *et al.* 2002). Therefore, absolute population size should be taken into consideration when examining the size and direction of a trend. Similarly, the seemingly large population increases for Red-breasted Merganser, although reflecting a more than doubling of birds across time, relates to a very small number of birds (maximum ten) that frequent the site.

Many individual site-specific monitoring projects are put in place following Environmental Impact Assessments. However, assessing waterbird numbers or attempting to determine conservation status in isolation, or in the context of a larger site, is inherently difficult while comparisons of absolute numbers are meaningless unless factors such as seasonality and area (density) are taken into account. The methods described here and elsewhere (e.g. Leech *et al.* 2002, Austin & Calbrade 2010) provide a useful way to examine waterbird numbers across time on a smaller scale and to assess the conservation status of the species. Importantly, this method can provide an early-warning system to identify the beginning of potential long-term downward trends and as such can be extremely useful as a tool in the practical management of a waterbird site.

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Fluctuations in breeding activity of Sand Martins *Riparia riparia* at a coastal site in the west of Ireland

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Keywords: Breeding, ecological monitoring, *Riparia riparia,* Sand Martin, site trends

This study charts the changes that occurred in a coastal population of Sand Martins *Riparia riparia* over a 13-year period from 2002 to 2014. The colony, at Sruwaddacon Bay, County Mayo, was studied intensively during the years 2008 to 2014 as part of ongoing long-term monitoring in connection with the Corrib Gas Development's activities at Glengad. During the monitoring period, which included construction of the offshore pipeline landfall and the Landfall Valve Installation adjacent to the core Sand Martin colony at Glengad, this colony doubled in size reflecting positive national trends at that time, with additional nesting cliffs then colonised within a 3 km radius. Overall, construction activities were not considered to have had any long-term impact on the local breeding population. In 2013, colony sizes were smaller possibly due to the effects of weather locally (colder springs).

Introduction

Patterns of arrival in spring have shown that amongst aerial insectivores, Sand Martins *Riparia riparia* are the first to arrive from their wintering grounds, in the Sahel region of sub-Saharan Africa, followed by Barn Swallows *Hirundo rustica,* House Martins *Delichon urbicum* and Common Swifts *Apus apus* (Baillie *et al.* 2006). In Ireland, the breeding range of Sand Martins declined from the 1970s to the 1980s followed by a recovery (Balmer *et al.* 2013) with the species currently amber-listed (Colhoun & Cummins 2013). Little has been published on this colonial species in Ireland although it is known that the earliest arrivals are usually in March with birds commencing autumn passage from the end of July, with first brood juveniles on the move by then (Hutchinson 1989, Cullen & Smiddy 2008).

This study charts the changes that occurred in a coastal population of Sand Martins over a 13-year period from 2002 to 2014, with intensive monitoring carried out in the years 2008 to 2014. The colonies at Sruwaddacon Bay, County Mayo were located along predominantly soft sandy coastal cliffs continually reshaped by the sea. Sruwaddacon Bay forms part of a wider coastal wetland known as Blacksod Bay and Broad Haven Special Protection Area (SPA). Monitoring of Sand Martins commenced initially in 2002 in connection with the Corrib Gas Project as part of the Environmental Impact Assessment process, and latterly (since 2008) as part of the Project Environmental Management Plan's extensive ecological monitoring programme. Breeding activity of the

Plate 246. Sand Martin chicks (John Fox).

Sand Martin population was assessed for the period 2008 to 2014 which enabled the variability between colonies, and the likely factors affecting breeding productivity and annual survival to be explored.

Study area

A Sand Martin colony at Glengad (Colony A) located along exposed soft coastal cliffs at the southern mouth of Sruwaddacon Bay (Figure 1), was identified during baseline surveys conducted as part of the Environmental Impact Assessment for the Corrib Gas pipeline in 2002 prior to initial landfall construction activities at that site, with subsequent colony counts carried out in 2004 and 2005. No monitoring was conducted in 2003, 2006 and 2007. From 2008 onwards, regular monitoring at Glengad was implemented with the cliff faces checked on at least a fortnightly basis throughout the breeding season where possible (Table 1). An additional colony was identified at Glengad in 2008 (Colony B) (Figure 1) with Colony A situated 30 m north of the pipeline landfall, and Colony B approximately 200 m southwest of the landfall. In addition to these 'core' colonies, a further colony at Rinroe Strand (Colony C) was discovered in 2008 at the northern mouth of the bay. A newly active colony (Colony D) was observed at Pullathomas in 2010 at a location approximately 2 km southeast of the Glengad colonies, while in 2011 Colony E was established near the mouth of North Rossport Bay (Figure 1). Most of these colonies lie wholly or partially within the existing SPA, except for Colony C. During landfall construction in 2008 and 2009 exclusion zones were set up in front of the colonies at Glengad to minimise disturbance.

Monitoring methods

Detailed monitoring occurred during the years 2008 to 2014. Monitoring of Colonies A and B was the most intensive, with at least fortnightly surveys carried out in all years throughout the breeding season (Table 1). Each survey was carried out by two surveyors using optical equipment (e.g. binoculars, digital cameras). Periods of heavy rain or poor visibility were avoided. Surveyors were positioned at a safe setback distance from the colony (i.e. 50 m) so as not to interfere with the natural behaviour of the Sand Martins. The first task on each survey visit was to photograph the extent of the colony. This provided a catalogue of digital images which was subsequently used as a survey tool to record any changes in the number and physical status of burrows (such as collapse or new burrow construction) between field visits. Burrows deemed suitable for Sand Martins were classed as 'viable' burrows. Thereafter, each colony was monitored for a minimum of 45 minutes (or until such time as no new active burrows were identified) during which time the number of Sand Martin visits to each burrow and observations of chicks at the burrow entrances were recorded in order to confirm activity at each burrow. 'Active' burrows were defined by having one or more of the following criteria:

- Where adults are seen entering/ exiting the burrow and/ or;
- Where chicks are either seen or heard at a burrow entrance.

Care was taken to distinguish active occupied burrows from old holes or trial borings (Kuhnen 1978). Additional signs of activity such as recent claw marks or fresh faeces below the burrow entrance were also recorded to indicate burrow

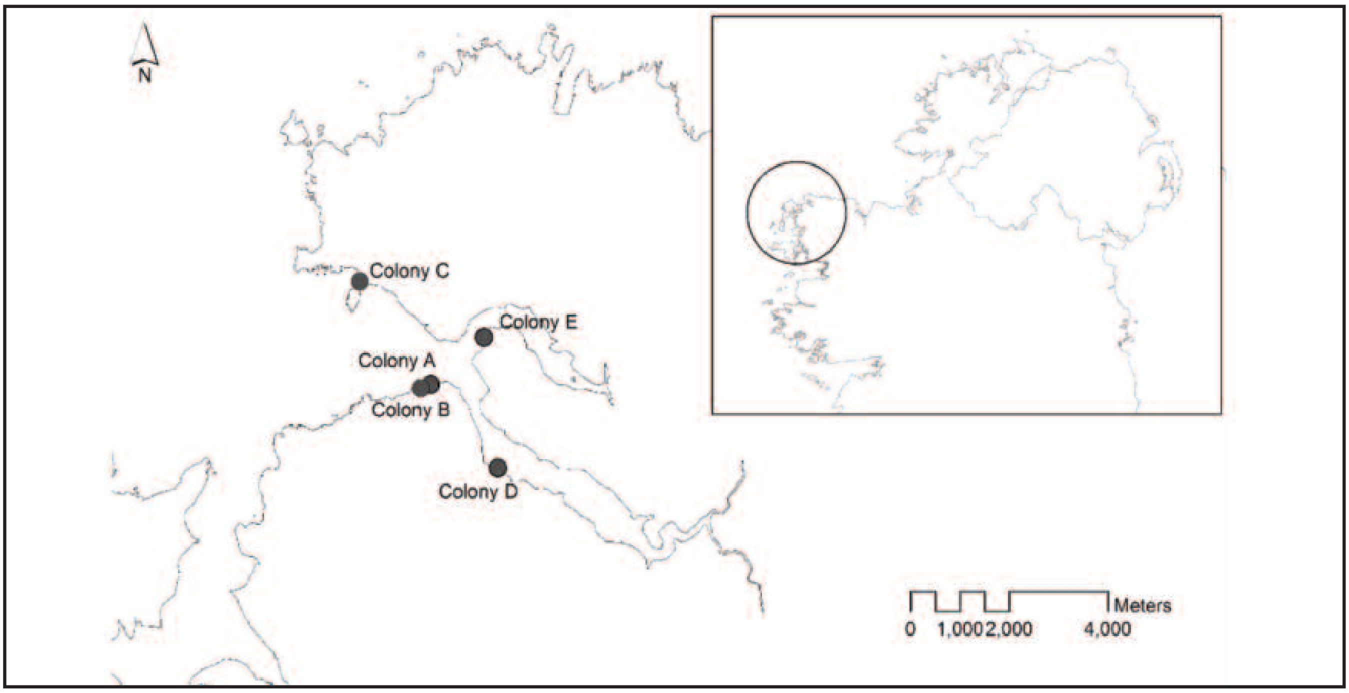

Figure 1. Map of study area indicating locations of Sand Martin colonies at Sruwaddacon Bay, County Mayo.

Table 1. Number of completed counts of Sand Martin Colonies A and B per season (2008-2014) at Sruwaddacon Bay, County Mayo.

Year	Range of survey dates	Maximum no. of visits	No. of incomplete/ abandoned counts	Completed counts
2008	13/05/08 – 18/08/08	7	-	7
2009	01/04/09 – 26/08/09	16	5	11
2010	31/03/10 – 25/08/10	25	-	25
2011	22/03/11 – 14/09/11	23	-	23
2012	27/03/12 – 21/08/12	13	2	11
2013	17/04/13 – 21/08/13	20	3	17
2014	01/04/14 – 21/08/14	17	1	16

occupancy and burrows at Colonies A-D were not confirmed to be active unless one or both of the above criteria were satisfied. Inactive burrows were defined as burrows that show none of the criteria of active burrows and that had no obvious signs of recent usage. Detection of Sand Martin activity at Colony E was difficult due to the constraints of distance from the observers, and/ or infrequent access by boats so estimates of the numbers of viable burrows were made either by direct observations using a telescope at a distance, or by examining high resolution digital images of the colony.

Burrows that were (or became) inaccessible to Sand Martins either due to collapse or vegetation growth were also recorded as inactive. Furthermore, where a viable burrow was seen to be visited on only one occasion during the course of the monitoring season, it was categorised as inactive. In this case it is likely that the bird was merely prospecting the available nest sites before choosing a nesting burrow. Some burrows deemed not viable at the outset of a breeding season (e.g. those that had a protrusion of vegetation) became active during the summer following fresh excavation by Sand Martins. Similarly, several burrows classed as viable at the beginning of the breeding season were later described as not viable, typically due to partial collapse or the growth of vegetation.

The monitoring results from Colonies A, B and C were used to determine approximate timing of breeding, relative population sizes of colonies from burrow counts, changes in colony sizes across the bay and across years, and known occupancy of burrows from counts undertaken. In addition, these results helped to ascertain the occurrence of second broods and provided information on any obvious nest failures. Counts of the 'core' Glengad Colonies A and B pre-construction of the pipeline landfall (2002-2005) together with the intensive monitoring data (2008-2014) were used to assess whether changes in breeding activity were the result of adverse impacts of construction or whether other natural processes were at play.

Data from the nearest weather station at Belmullet, County Mayo were examined to explore if Sand Martin arrival could be influenced by local weather patterns. Weather data were obtained from Met Éireann (2015) (mean monthly minimum temperatures (degrees Celsius) and total monthly rainfall (mm)) for the period March to July and was used to identify any obvious relationships between local weather patterns and timing of breeding (measured as date of earliest recorded arrival at the Glengad Colonies A and B) and as a measure of breeding success (number of viable burrows).

Results

Arrival dates

Sand Martin spring arrival dates showed some variation over the years 2008 to 2014 from 30 March to 23 April with arrival somewhat later in 2012, 2013 and 2014 when compared with 2011 (Table 2). This compares with the average earliest dates of spring sightings in Ireland of 15-16 March (BirdTrack data), while April and May are the most important months for sightings of Sand Martins.

Colony count totals

Across all colonies at Sruwaddacon Bay, the total number of potential viable burrows more than doubled from 80 to 167

Table 2. Spring arrival dates of Sand Martins at Sruwaddacon Bay, County Mayo.

Year	First arrival
2008	Not given
2009	Late March/early April
2010	14 April
2011	30 March
2012	19 April
2013	23 April
2014	23 April

between 2008 and 2010 and peaked in 2011 with 247 (Table 3). Numbers remained constant in 2012 with 230 viable burrows recorded. The total in 2013 declined, with less than 100 viable burrows confirmed. However, this is probably an underestimate because Colony E was not surveyed. The estimated number of viable burrows for 2013 was thus >150. Numbers declined further in 2014 with just 71 viable burrows recorded in the study area, with Colony D inactive and Colony E (unsurveyed) estimated to hold only about five viable burrows (Table 3).

Plate 247. Sand Martin (Michael O'Clery).

The overall occupancy of burrows varied across the survey period from 19% to 85% (Table 4) but it is likely that this is due to difficulties in assessing breeding activity at Colony E. Total numbers of viable burrows at Glengad increased in 2008 with the establishment of Colony B. Totals for the Glengad Colonies (A & B) combined peaked in 2010 with 88 viable burrows, with 41 burrows recorded in 2014 (Table 4). Breeding activity through the season fluctuated with highest numbers of active nests recorded in June in most years (Appendix 1). Between 2008 and 2014, the average colony size in the bay was 36.1 ± 4.6 viable burrows, with colonies ranging in size from 16 to 81 viable burrows in any given year.

Local weather patterns

The exceptional storms in the winter of 2013/14 (Met Éireann 2014) contributed to significant losses of viable burrows between the end of the 2013 breeding season and the beginning of the 2014 season. The two areas most obviously affected were Colony A at Glengad and the sandy cliff portion

Table 3. Maximum counts of viable burrows of Sand Martins at all colonies monitored in 2002 and 2004-2005 and during subsequent more intensive monitoring during 2008-2014 at Sruwaddacon Bay, County Mayo. No monitoring was conducted in 2003, 2006 and 2007.

Year	Colony A Glengad	Colony B Glengad	Total 'core' Glengad Colonies	Colony C Rinroe	Colony D Pullathomas	Colony E Sandy Point	Total
2002	40	0	40	0	0	0	40
2004	46	0	46	0	0	0	46
2005	50	0	50	0	0	0	50
2008	52	16	68	12	0	0	80
2009	51	16	67	56	0	0	123
2010	71	17	88	49	25-30	0	162-167
2011	57	22[a]	79	72	20[b]	76	247
2012	39	17	56	88	18	68	230
2013	31	17	48	49-51	?[c]	?[c]	97-99[d]
2014	24	17	41	30	?[e]	?[e]	71[e]

[a] No Sand Martins bred at this location after early prospecting in spring.

[b] Burrows lost due to erosion early in the season.

[c] These colonies are accessible by boat only and no dedicated boat survey was undertaken in 2013 but observations were made from a distance using a telescope and/or digital images were taken; the Pullathomas colony was believed to be inactive, while the Sandy Point colony was believed to be active.

[d] Underestimate with total number of viable burrows likely to be >150 as Colony E appeared to be of a similar size as in 2012 but detailed evaluation not possible from remote Vantage Point.

[e] The Pullathomas colony was believed to be inactive, while the Sandy Point colony was believed to have some active burrows; neither colony was approached closely during the 2014 breeding season. In addition, there were a small number (about 5) of isolated burrows observed along the northern shore of Sruwaddacon Bay, but these were not approached closely to assess in detail, consequently, the total is believed to be an underestimate.

Table 4a and b. Summary information on activity of Sand Martins at Sruwaddacon Bay, County Mayo for each survey season. Intensive monitoring of the Sand Martin population commenced in 2008. Prior to that, only summary totals are available. Some colonies (D and E) could not be adequately assessed in terms of activity of burrows due to distance of the colony from land-based vantage points and accessibility (i.e. requires boat survey). (A = not assessed; B = active burrows, but number not assessed; C = loss due to slippage).

(a)

Year	Colony A			Colony B			Colony C		
	Active	Viable	% occ	Active	Viable	% occ	Active	Viable	% occ
2002	-	40	-	-	-	-	-	-	-
2004	34	46	-	-	-	-	-	-	-
2005	37	50	-	-	-	-	-	-	-
2008	33	52	63	16	16	0	-	12	-
2009	31	51	61	-	16	0	-	56	-
2010	64	71	90	15	17	88	-	49	-
2011	28	57	49	1	22	5	B	72	-
2012	21	39	54	4	17	24	-	88	-
2013	9	31	29	0	17	0	3	51	6
2014	12	24	50	0	17	0	0	30	0

(b)

Year	Colony D			Colony E			Overall		
	Active	Viable	% occ	Active	Viable	% occ	Active	Viable	% occ
2002	-	-	-	-	-	-	-	40	-
2004	-	-	-	-	-	-	34	46	74
2005	-	-	-	-	-	-	37	50	74
2008	-	-	-	-	-	-	49	80	85
2009	-	-	-	-	-	-	31	123	41
2010	-	25-30	A	-	-	-	79	162-167	51-53
2011	0	2	C	-	76	-	29	227	26
2012	0	18	-	-	68	-	25	230	19
2013	0	18	-	-	-	A	12	117	29
2014	0	0	-	-	-	A	12	71	34

of Colony C at Rinroe. The more elevated Colony B at Glengad was largely unaffected by the winter storms. In some years, Sand Martins were recorded actively scoping burrows in the early part of the season at some colonies, but latterly no confirmed breeding activity was recorded. These outcomes were sometimes due to physical losses of burrows, for example in 2014 Colony C had a substantially reduced number of viable nest burrows at the outset having lost all of the burrows located in the sandy cliff due to winter storms (Fennessy 2015). This colony showed some recovery throughout that breeding season but the level of activity was lower than in previous years and reflected the overall pattern of reduced breeding numbers observed at other local colonies.

There was a negative relationship between spring temperature and arrival date (day of year) (Figure 2) indicating that as spring temperatures increased, the arrival date was earlier, although the relationship was weak and not statistically significant. There was no relationship between mean minimum air temperatures and the number of active burrows (Figure 3) or between monthly rainfall and the number of active burrows (Figure 4).

Discussion

The Sand Martin population at Sruwaddacon Bay underwent a period of rapid growth during the survey period as indicated by the increase in numbers of viable burrows since 2002. While it is possible such changes could be partly attributed to increased survey intensity in recent years (2008-2014) other indications suggest that the patterns of changes in colony sizes recorded at Sruwaddacon Bay appear to be consistent with national trends from the Countryside Bird Survey (unpublished Countryside Bird Survey (CBS) data, BirdWatch Ireland). The patterns also appear consistent with the United Kingdom, where numbers of Sand Martins peaked in the late

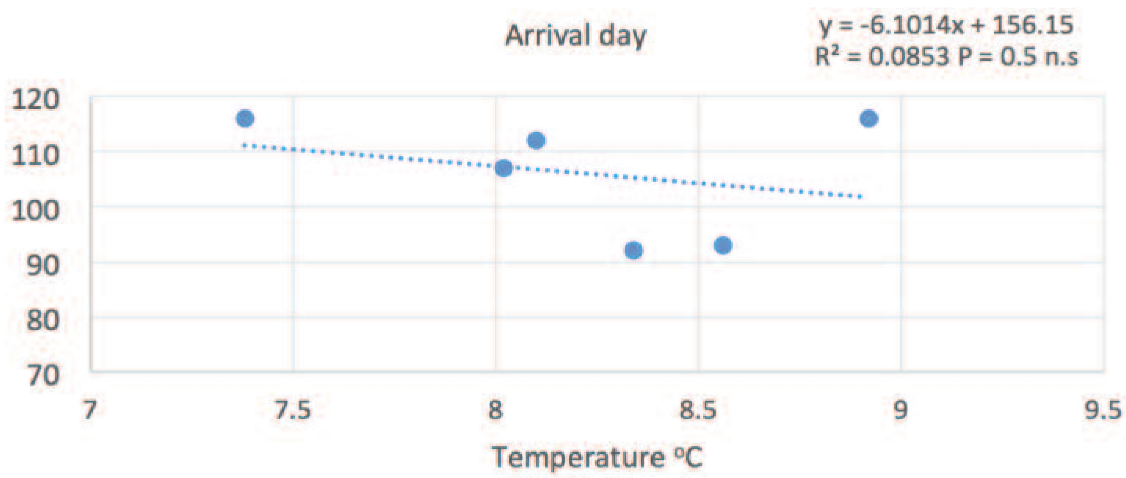

Figure 2. Relationship between mean minimum temperature and arrival date (day of year) of Sand Martins at Sruwaddacon Bay, County Mayo.

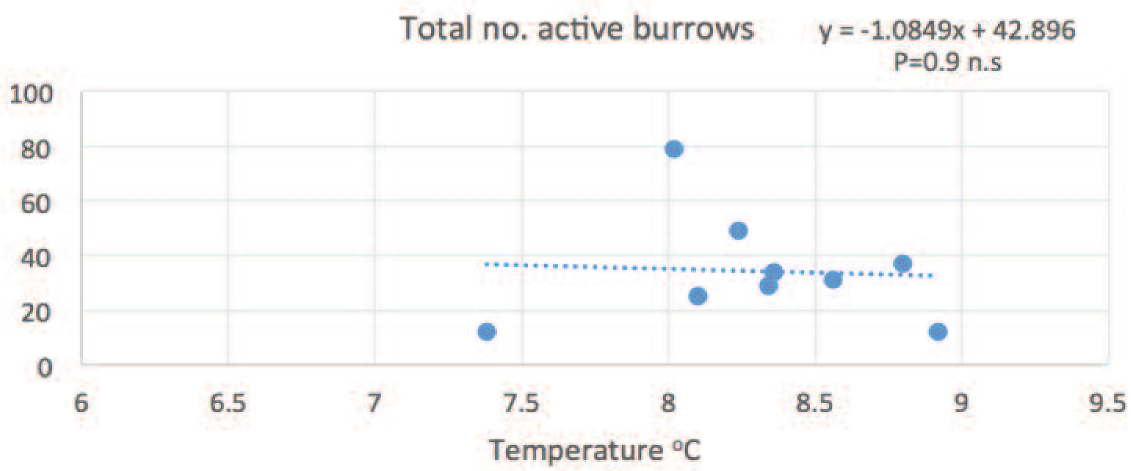

Figure 3. Relationship between mean minimum temperature and total number of active burrows of Sand Martins at Sruwaddacon Bay, County Mayo.

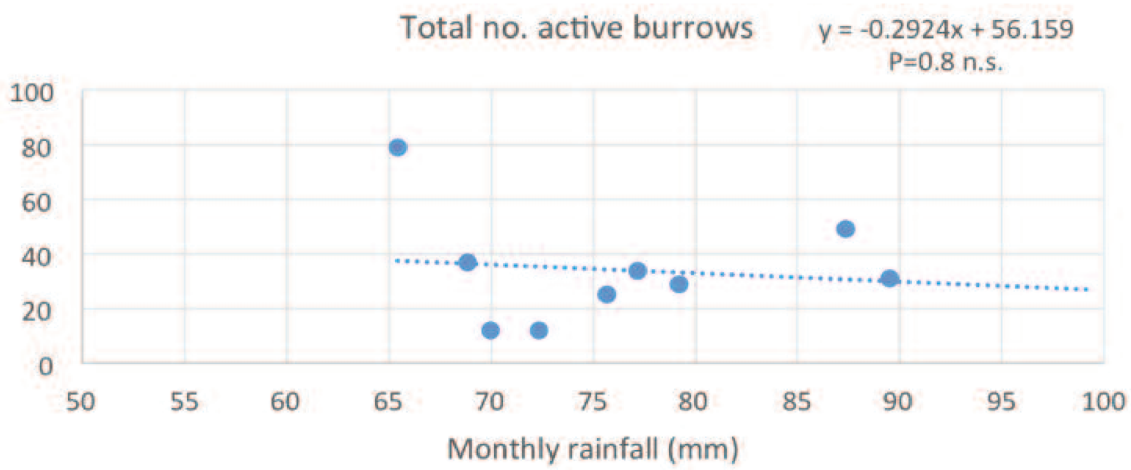

Figure 4. Relationship between mean monthly rainfall (mm) and number of active burrows of Sand Martins at Sruwaddacon Bay, County Mayo.

1990s, before declining sharply in the early 2000s with a recovery by 2010, before numbers fell again (Harris *et al.* 2015).

The data for this study were collected for the purpose of assessing changes in Sand Martin colonies at Glengad (Colonies A and B). During the years of construction activity at the pipeline landfall (including boat activity associated with laying the pipeline) there was no indication of any negative impacts arising from such activities (FTC 2010, 2011). The establishment of new colonies from 2008, coupled with the increased occupancy of viable burrows (particularly during the period 2010-2012) would suggest that breeding success and recruitment at colonies in the bay was good during this period. In particular, the number of viable burrows increased from 2008 to 2011, with peak values coinciding with periods of offshore pipeline construction activities. This trend was maintained in 2012 and 2013 during onshore construction activity, including foreshore activities associated with the construction of the Landfall Valve Installation. However the colony size dramatically fell in 2014, a year in which there were no foreshore-based activities. Sand Martins are site faithful with an estimated 93% of adults and 87% of juveniles returning the following year to within a distance of 10 km of the same colony (Mead 1979, Szép & Moller 1999). Any likely impacts of construction activities on the local breeding population are therefore likely to be detected within a short time-frame and it is probable that other factors were driving the observed local population trends.

Colony sizes varied across Sruwaddacon Bay and across the main survey period (2008-2014), with a maximum of 88 burrows recorded at Rinroe (Colony C) in 2012, up from a minimum of 12 burrows two years earlier. Published records for 24 Irish colonies surveyed during the 1960s indicate a mean colony size of 66 pairs (Ruttledge 1966), higher than the mean colony size at Sruwaddacon Bay but lying within the range of colony sizes recorded there. In Britain, the average colony size is 38 pairs (n = 57) with colonies perhaps larger in the north than in the south; while throughout its range, most Sand Martin colonies hold about 50 pairs, although 100 or more pairs is not unusual (Morgan 1979).

Changes in numbers of viable burrows at the colonies across the survey period highlight the annual variation in the local breeding Sand Martin population. In southwest Sweden, the ratio of pairs to nest holes is in the order of 60-65% based on birds ringed at colonies (Persson 1987). In Hungary, endoscopic inspections of nest burrows revealed 64% of them contained nests (Szép *et al.* 2003). Therefore, counts of nest burrows can provide a reasonably accurate measure of the size of local breeding populations.

The Sand Martins at Sruwaddacon Bay are utilising soft coastal breeding cliffs shaped by the sea that are susceptible to collapse and erosion. The resilience and flexibility of the local population to adapt to environmental changes has been demonstrated repeatedly over the survey period. In 2011, Pullathomas (Colony D) was almost entirely lost to erosion in the early part of the season. However, a new colony (Colony E) was quickly established and more than offset the loss of viable burrows at Colony D and the non-use of Colony B at Glengad. In addition, it may have drawn nesting birds from the other monitored colonies, as Colony A had only a moderately successful season. At Rinroe (Colony C) there was relocation within the colony itself from birds using burrows along the sandy cliffs to using others along the muddy soft cliffs closer to Rinroe Pier.

Sand Martins have been arriving on breeding grounds earlier in more recent years (Sparks & Tryanowski 2007, BTO News 2011, 2012) and this pattern is consistent with the trends in other species such as Blackcap *Sylvia atricapilla* and Reed Warbler *Acrocephalus scirpaceus* (Donnelly *et al.* 2009). At Sruwaddacon Bay, arrival dates for Sand Martins each spring varied across the survey period (2010-2014), with birds arriving at the bay consistently later each year than along east coast counties (BTO/RSPB/BirdWatch Ireland/SOC/WOS 2015). Observers noted in the later survey years (2013-2014) that pairs largely reared single broods, whereas in the earlier survey years (2008-2010) second broods were suspected. Sand Martin mating systems can be variable, including extra-pair paternity, con-specific brood parasitism and extra-pair maternity (Alves & Bryant 1998, Augustin *et al.* 2007) and therefore teasing out patterns with regard to timing and number of breeding attempts is not straightforward. Furthermore, first-year birds arrive later than older birds (up to three weeks), which can contribute to a succession of nesting attempts at larger colonies through the season (Mead & Harrison 1979). Later arrival of first-year birds will often result in them occupying more peripheral burrows at established colonies (Szabó & Szép 2010) or they may have to excavate new burrows (Cowley & Siriwardena 2005). Ringing records have shown that the earliest free-flying juvenile Sand Martins are usually captured at the end of May or early June, but young can continue to fledge from successful colonies for a further ten weeks (Mead 1979). Only young birds seen in June and early July can be assured of having come from the colony.

Local weather patterns have been shown to affect Sand Martin survival (Cowley & Siriwardena 2005). Furthermore, spring arrival may be negatively correlated with March temperatures (Donnelly *et al.* 2009). While the data for Sruwaddacon Bay followed this pattern, i.e. as spring temperature increased, arrival time was earlier, the relationship was weak and based on relatively few years of data; longer-term monitoring would possibly strengthen this observed pattern over time. It follows however, that in the latter years of this study, colder springs may have been responsible for the later arrival dates of Sand

Martins to Sruwaddacon Bay. Elsewhere, colder temperatures have been shown to reduce breeding success (Cowley 1979), with a knock-on effect on the number of broods reared and on annual breeding success.

A review of monthly rainfall data for County Mayo showed a weak negative relationship with breeding success (number of viable burrows). Certainly, heavy rainfall can suppress the flight of adult insects making them less available to aerial feeding birds and may cause collapse of nest burrows (Cowley & Siriwardena 2005). The obvious colder springs of 2013 and 2014 in County Mayo may have limited breeding opportunities with fewer second broods observed and overall lower occupancy of burrows at the colonies. Although monitoring is unlikely to extend beyond three years in the post-construction phase of the project, keeping track of changes in the breeding phenology of Sand Martins in the study area, particularly the time of first arrival of birds and first breeding, could help detect whether climate is negatively impacting the local breeding success and overall conservation status of this summer migrant.

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Appendix 1. Chronology of activity across months and years at Sand Martin colonies located at Sruwaddacon Bay, County Mayo.

Year	Annual peak/Monthly peak	Colony	No. of active burrows	No. of viable burrows
2002	**Peak**	**A**	**-**	**40**
2004	**Peak**	**A**	**-**	**46**
2005	**Peak**	**A**	**-**	**50**
2008	May	A	13	-
2008	June	A	26	-
2008	July	A	20	-
2008	August	A	0	-
2008	**Peak**	**A**	**33**	**52**
2008	**Peak**	**B**	**16?**	**16**
2008	**Peak**	**C**	**-**	**12**
2008	**Peak**	**D**	**-**	**Not present**
2008	**Peak**	**E**	**-**	**Not present**
2009	**Peak**	**A**	**48 total A & B**	**51**
2009	**Peak**	**B**	**48 total A & B**	**16**
2009	**Peak**	**C**	**-**	**56**
2009	**Peak**	**D**	**-**	**Not present**
2009	Peak	E	-	Not present
2010	March	A	-	41
2010	May	A	-	63
2010	June	A	53	70
2010	March	B	-	-
2010	May	B	-	13
2010	June	B	-	17
2010	July	B	10	-
2010	August	B	1	-
2010	April	C	-	42
2010	May	C	-	39
2010	June	C	-	49
2010	July	C	-	47
2010	August	C	-	43
2010	August	D	-	25-30
2010	**Peak**	**A**	**64**	**71**
2010	**Peak**	**B**	**15**	**17**
2010	**Peak**	**C**	**-**	**49**
2010	**Peak**	**D**	**-**	**25-30**
2010	**Peak**	**E**	**-**	**Not present**
2011	March	A	-	43
2011	May	A	28	57
2011	June	A	29	50
2011	July	A	22	-
2011	August	A	-	51
2011	**Peak**	**A**	**-**	**57**
2011	**Peak**	**B**	**1**	**22**
2011	**Peak**	**C**	**-**	**72**
2011	**Peak**	**D**	**-**	**20**
2011	**Peak**	**E**	**-**	**76**
2012	March	A	0	38
2012	April	A	0	39
2012	May	A	2	38
2012	June	A	15	30
2012	July	A	8	36
2012	August	A	5	36

2012	April	B	4	17
2012	May	B	0	17
2012	June	B	0	17
2012	July	B	4	17 or 18
2012	March	C	-	46
2012	July	C	-	86
2012	August	C	-	88
2012	September	C	-	84
2012	March	D	0	13
2012	**Peak**	**A**	**15**	**39**
2012	**Peak**	**B**	**4**	**18**
2012	**Peak**	**C**	**-**	**88**
2012	**Peak**	**D**	**0**	**13**
2012	**Peak**	**E**	**0**	**68**
2013	April	A	0	22
2013	May	A	4	28
2013	June	A	9	28
2013	July	A	10	31
2013	August	A	5	31
2013	April	B	0	17
2013	May	B	0	17
2013	June	B	0	17
2013	July	B	0	17
2013	August	B	0	17
2013	**Peak**	**A**	**10**	**31**
2013	**Peak**	**B**	**0**	**17**
2013	**Peak**	**C**	**0**	**51**
2013	**Peak**	**D**	**0**	**0**
2013	**Peak**	**E**	**0**	**0**
2014	April	A	0	14
2014	May	A	6	20+
2014	June	A	6	24
2014	July	A	10	24
2014	August	A	7	24
2014	**Peak**	**A**	**12**	**24**
2014	**Peak**	**B**	**0**	**17**
2014	**Peak**	**C**	**0**	**30**
2014	**Peak**	**D**	**0**	**0**
2014	**Peak**	**E**	**-**	**-**

Waterbird populations on non-estuarine coasts of Ireland: results of the 2015/16 Non-Estuarine Coastal Waterbird Survey (NEWS-III)

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Keywords: Coastal waterbirds, monitoring, non-estuarine waterbirds

The third all-Ireland survey of waterbirds wintering on non-estuarine coasts (NEWS-III) was carried out between December 2015 and February 2016. A total of 2,095 km of non-estuarine coastline across Northern Ireland and the Republic of Ireland was surveyed representing 63% of the total all-Ireland non-estuarine coastline. Linear densities (totals per kilometer) were used to compare distributions between counties and coastal regions. A bootstrap method was used to generate estimates of 25 species within each county and in Ireland overall. Totals of 110,061 birds, belonging to 72 species, were recorded and waders comprised nearly half of the total. Herring Gull *Larus argentatus* was the most numerous species, with a total count of 19,681 birds recorded across 85% of sectors. Black-headed Gull *Chroicocephalus ridibundus* was the second most numerous of the gulls. Oystercatcher *Haematopus ostralegus* was the most numerous wader (12,990) and the most widespread waterbird (86% of sectors). Shag *Phalacrocorax aristotelis,* Cormorant *Phalacrocorax carbo* and Common Scoter *Melanitta nigra* were the most numerous wildfowl species (including allies). There were considerable differences in the estimates generated for species on non-estuarine coasts between NEWS-III and NEWS-II. Largest increases in population size were for Teal *Anas crecca,* Eider *Somateria mollissima,* Shag, Purple Sandpiper *Calidris maritima,* Snipe *Gallinago gallinago* and Greenshank *Tringa nebularia*. Five species showed declines in excess of 50% (Shelduck *Tadorna tadorna*, Red-throated Diver *Gavia stellata,* Golden Plover *Pluvialis apricaria,* Knot *Calidris canutus* and Bar-tailed Godwit *Limosa lapponica*). Overall, NEWS-III showed that non-estuarine coastal habitats continue to support a broad diversity of waterbirds, and is especially important for divers, seaducks, Cormorant, Grey Heron *Ardea cinerea* and several waders, notably Ringed Plover *Charadrius hiaticula,* Sanderling *Calidris alba,* Purple Sandpiper, Greenshank and Turnstone *Arenaria interpres.*

Introduction

The Wetland Bird Survey (WeBS) in Northern Ireland and Britain, together with the equivalent scheme in the Republic of Ireland, the Irish Wetland Bird Survey (I-WeBS), are Ireland's key monitoring surveys of wintering waterbirds, and are ongoing since the early 1990s. These surveys focus on key wetlands that support high densities of birds, predominantly

Plate 248. Herring Gulls (Michael O'Clery).

coastal estuaries and inland lake complexes. However, these surveys are not adequate for monitoring some wintering species, particularly those that occur on non-wetland sites such as swans and geese, and also species that are prevalent along open coastline. Periodic special surveys of the populations are thus undertaken to improve our baseline information on population sizes and distribution (e.g. Crowe *et al.* 2014).

In December 1997 and January 1998 the first thorough national survey of waterbirds occurring on non-estuarine coasts was undertaken (NEWS-I) (Colhoun & Newton 2000, Rehfisch *et al.* 2003). The second survey of this kind, undertaken nine years later, NEWS-II, was carried out in December 2006 and January 2007 (Austin *et al.* 2008, Crowe *et al.* 2012). In these surveys, a random selection of coastal stretches was visited from which estimates of total numbers of waterbirds occurring in these habitats were generated. The survey results revealed that substantial proportions of several waterbird species could occur along non-estuarine coasts highlighting that I-WeBS core counts at predominantly estuarine sites may not adequately reflect the national trends of these species (Crowe *et al.* 2012).

During December 2015 to February 2016, the third all-Ireland survey of waterbirds on non-estuarine coasts was carried out (NEWS-III). This paper reports on the results and provides an update on the total numbers of waterbirds occurring on open coasts, a broad-scale assessment of habitat usage, and an indication of changing proportions using Ireland's non-estuarine coast. These estimates will feed into the forthcoming revision of the non-breeding waterbird population estimates.

Methods

Field methods

Non-estuarine habitat is defined here as open rocky and sandy coastline not already covered by coastal WeBS and I-WeBS counts (mostly estuarine), but excluding areas of tall cliffs with little or no exposed shoreline. The methodology used closely followed that of NEWS-I (Colhoun & Newton 2000) and NEWS-II (Crowe *et al.* 2012) and was broadly similar to the first survey of this kind, the Winter Shorebird Count (WSC), undertaken in Great Britain and Northern Ireland in the mid-1980s (Moser & Summers 1987, Moser & Prŷs-Jones 1988), entailing coverage of pre-defined short sections of non-estuarine coasts within a period of 3.5 hours either side of low tide, preferably in favourable weather conditions. It was recommended that counts be carried out between 1 December 2015 and 31 January 2016, but an extension was made to the end of February 2016 due to poor weather (strong winds and heavy rain) during the main count period. The non-estuarine sections defined during NEWS-I and NEWS-II were digitized using ArcGIS; their boundaries based on obvious changes in substrate type. Most sections were between 2 km and 4 km in length. As in NEWS II in 2006/07, the aim was to achieve overall coverage equating to at least 50% of the non-estuarine coast and for organisers to assign priority stretches first, which were the same randomly selected count stretches designated in NEWS-II, to avoid geographic gaps and bias from counter preferential selection of more productive areas of coast. Selected count stretches were omitted only on the grounds of practical complications such as difficulty of access or remoteness from available counters, but explicitly not on prior expectations of waterbird numbers. Counters were asked to cover adjacent sectors synchronously on a given count day. A small number of professional counters were also deployed to enhance coverage in poorly covered areas, particularly those in the west and northwest.

The NEWS-III count methodology was based largely on that used during NEWS-I (Moser & Summers 1987, Rehfisch *et al.* 2003) and NEWS-II (Crowe *et al.* 2012). Observers were asked to record on standard recording sheets all waterbirds (waders, wildfowl, divers, grebes, Cormorant *Phalacrocorax carbo,* herons and gulls) observed in three main habitat types: (1) intertidal habitat, (2) sea - viewable from the high tide mark with binoculars and (3) land - inland habitat viewable from the high tide mark. Observers were also asked to record details of tidal state, weather, disturbance and coverage which may account for the quality of counts. Data recording was expanded for NEWS-III to collect additional information on habitat, specifically strandline deposits of beach-cast seaweed, and their use as a foraging resource. NEWS III was run as an online survey, using additional functionality within the existing WeBS Online data submission system run by the BTO. This allowed participants to select count sectors and after the survey, to enter data online.

Analytical methods

Species totals were generated and compared between habitat types. To examine the geographical distribution of birds, linear densities, defined as total number of birds per kilometer, were calculated for each species of wildfowl (including allies) and waders, and compared across coastal regions defined as Northwest (Donegal, Leitrim, Sligo), West (Mayo, Galway, Clare), Southwest (Kerry, Cork), Southeast (Waterford, Wexford), East (Wicklow, Dublin, Meath, Louth), Northeast (Down, Antrim, Derry).

The analytical approach used to estimate the total numbers of waterbirds on the open coast during the winter of 2015/16 was similar to that used previously in NEWS-I (Colhoun & Newton 2000, Rehfisch *et al.* 2003) and NEWS-II (Austin *et al.* 2008, Crowe *et al.* 2012). A bootstrap approach was used to derive estimates at the regional level, summing across the relevant bootstrapped samples to derive country estimates. A key difference to the method used previously was that the bootstrap approach was used to estimate populations for the whole of the region, whereas in the previous analysis the counted stretches were treated effectively as a census and only populations on the uncounted stretches were estimated using the bootstrap approach to add to the 'census' value. Thus, for each species, for each region and for each habitat (intertidal, sea and land) 119 bootstrap realisations of the total number of birds counted within that region were obtained from a sample with replacement of stretches drawn from the region in question until that sample equated to the length of non-estuarine coast within that region. In order to produce the best possible estimates of numbers, 119 all-habitat region estimates were obtained by summing the unordered estimates from each of the three habitats. Equivalent repetitions upon which to derive overall estimates for the Republic of Ireland, Northern Ireland and all-Ireland, were derived by summing across unsorted repetitions for the relevant constituent counties. The latter approach was used so that, for example, estimates of numbers on uncounted stretches in southwest Ireland would not be influenced by counts in northeast Ireland and *vice versa* when estimating the numbers across these larger regions, as would be the case had the sample itself been drawn from the whole of Ireland. For each geographical extent, the point (median), lower and upper 95% confidence limit estimates for the number of birds were obtained by taking the 60th, 3rd and 116th ascendant ordered bootstrap values respectively (Austin *et al.* 2017).

Regions used for reporting NEWS-III in the United Kingdom (including Scotland, Wales and Northern Ireland) were reappraised to take into account current county boundaries and geographic coverage. Furthermore, precise details of the analysis were adjusted because the counted part of the coast within each region was no longer treated as a census to which an extrapolated estimate for the remainder is added, but rather counts were treated as sample repetitions to be extrapolated to the entire coast within that region. As a consequence, previous NEWS survey data were reanalyzed, including the Irish data, to reflect these improvements (Austin *et al.* 2017).

Comparison between NEWS-III and NEWS-II

Population estimates generated for NEWS-III were compared with those generated for NEWS-II for the Republic of Ireland only. However, given the considerations described above, care must be taken when comparing data between surveys presented in the results below. In the future, estimates of population change will be made following a paired count stretch approach as used by Rehfisch *et al.* (2003). This is a more powerful approach than that of direct comparison of estimates of total numbers between surveys, the latter being vulnerable to differences in geographical coverage between surveys or unknown biases in count-stretch selection.

The percentage change in linear densities (birds per kilometer) of non-breeding waterbirds between NEWS-III and NEWS-II were calculated for six coastal regions in Ireland. Densities were generated from counts of birds within intertidal habitats only for waders. However, wildfowl (and their allies) are generally more widely distributed in all three habitats, so densities for this group were based on counts from all habitats combined.

Results

Coverage

In total, 2,095 km and 675 sectors of coastline were covered during NEWS-III (Table 1), which represents 63% of the total all-Ireland non-estuarine coastline available (Figure 1). Coverage ranged from 52% (Clare) to 100% in Leitrim and Meath (Table 1). All coverage was completed between 2 December 2015 and 29 February 2016, with less than 10% of the total number of sectors (63) covered during February 2016 due to poor weather during the main count period.

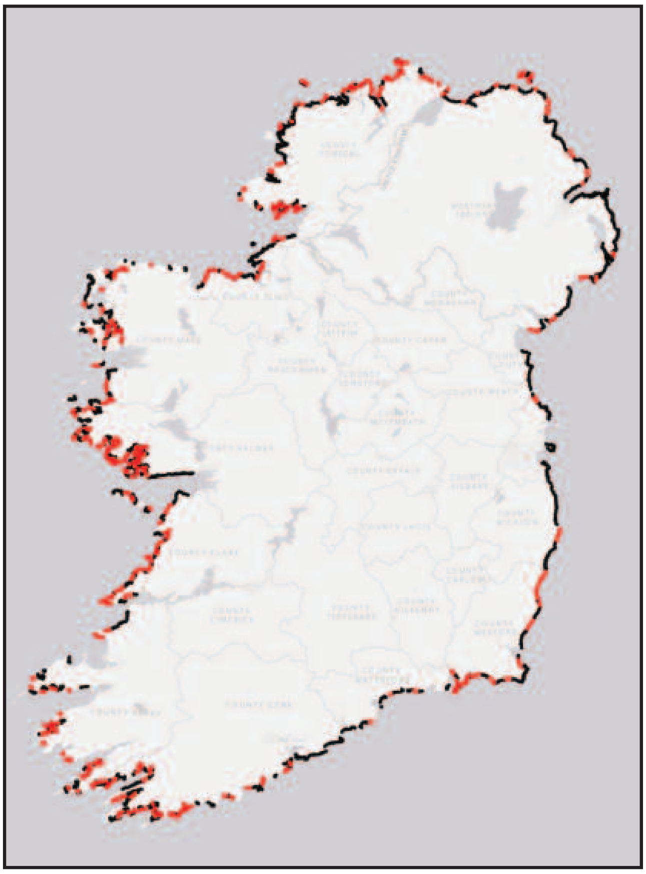

Figure 1. Coverage achieved during NEWS-III, 2015/16, illustrating stretches of coast that were covered (black) and not covered (red).

Table 1. Total coverage of non-estuarine coasts by county and overall during NEWS-III, 2015/16. Lengths of coast are expressed in km.

	Total length	Length covered	Proportion coast (%) surveyed	No. of sectors covered
Northern Ireland				
Ards and North Down	102	99	97	62
Causeway Coast and Glens	137	106	77	52
Mid and East Antrim	68	61	89	37
Newry Mourne and Down	72	61	85	62
Republic of Ireland				
Clare	223	117	52	33
Cork	605	357	59	85
Donegal	560	329	59	79
Dublin	63	59	93	11
Galway	576	306	53	66
Kerry	207	128	62	28
Leitrim	5	5	100	2
Louth	42	35	84	8
Mayo	266	161	60	42
Meath	7	7	100	3
Sligo	132	84	64	45
Waterford	51	47	92	20
Wexford	153	83	54	27
Wicklow	59	50	85	13
NI Total	**378**	**326**	**86**	**213**
RoI Total	**2950**	**1769**	**60**	**462**
All-Ireland	**3329**	**2095**	**63**	**675**

Overall numbers and distribution

Overall, 110,061 birds of 72 species were recorded. This includes 36 wildfowl species (and their allies), 22 waders, 12 gull species, one tern species (Sandwich Tern *Sterna sandvicensis*) and Kingfisher *Alcedo atthis*. Waders were the most numerous group (Table 2), comprising 43% of the total waterbirds counted, with the majority of waders (86%) recorded in the intertidal zone. Wildfowl and their allies accounted for 18% of the total waterbirds recorded, with more than 50% recorded on the sea, and 32% recorded in intertidal habitat and 17% on land (Table 2). Gulls and terns comprised 39% of the total waterbirds recorded with the greatest majority (70%) recorded in the intertidal zone.

The Herring Gull *Larus argentatus* was the most numerous species, with a total count of 19,681 birds recorded across 85% of sectors (Table 2). Black-headed Gull *Chroicocephalus ridibundus* was the second most numerous of the gulls with 6,851 recorded in the intertidal zone, and over 10,000 overall. Oystercatcher *Haematopus ostralegus* was the most numerous wader, with a total count of 12,990 birds recorded across 86% of sectors making it the most widespread species overall. Curlew *Numenius arquata,* Dunlin *Calidris alpina,* Lapwing *Vanellus vanellus* and Ringed Plover *Charadrius hiaticula* were also especially numerous among the waders (all >4,000 birds), with largest proportions using intertidal habitat with the exception of Curlew, which was recorded in terrestrial zones (land) with almost the same abundance as intertidal habitat. Golden Plover *Pluvialis apricaria* and Turnstone *Arenaria interpres* were also relatively numerous with their total counts exceeding 3,000 birds. Shag *Phalacrocorax aristotelis,* Cormorant and Common Scoter *Melanitta nigra* were the most numerous wildfowl species (including allies) recorded. The latter was recorded exclusively at sea, but while the former two species were more numerous at sea, they were also recorded in both intertidal and terrestrial zones.

Oystercatcher, Curlew and Redshank *Tringa totanus* were the most widely distributed wader species, recorded in over 50% of sectors (Table 2), while Turnstone was recorded in over 40% of sectors. Cormorant, Shag and Great Northern Diver *Gavia immer* were the most widely recorded of the wildfowl and allies group, while after Herring Gull, Great Black-backed Gull *Larus marinus,* Common Gull *Larus canus* and Black-headed Gull were the most widespread of the gulls.

Table 2. Waterbird totals recorded within each of the main habitat types and overall, together with distribution (number and proportion of sections in which recorded) on non-estuarine coasts in the Republic of Ireland and Northern Ireland combined during NEWS-III, 2015/06.

Species	Intertidal	Sea	Land	Total	Number of sectors present in (with %)
Mute Swan *Cygnus olor*	53	42	30	125	26 (4)
Whooper Swan *Cygnus cygnus*	1	-	35	36	6 (<1)
Pink-footed Goose *Anser brachyrhyncus*	-	-	5	5	1 (<1)
Greenland White-fronted Goose *Anser albifrons flavirostris*	-	1	-	1	1 (<1)
Greylag Goose *Anser anser*	13	-	45	58	8 (1.0)
Barnacle Goose *Branta leucopsis*	-	28	1,440	1,468	15 (2.2)
Light-bellied Brent Goose *Branta bernicla hrota*	1,565	199	575	2,339	101 (15.0)
Feral/hybrid goose	3	-	-	3	1 (<1)
Shelduck *Tadorna tadorna*	50	17	6	73	25 (3.7)
Wigeon *Anas penelope*	268	333	484	1,085	48 (7.2)
Teal *Anas crecca*	1,157	466	184	1,807	61 (9.1)
Mallard *Anas platyrhynchos*	532	456	255	1,243	157 (23.4)
Feral/hybrid Mallard type	3	-	-	3	1 (<1)
Shoveler *Anas clypeata*	-	-	2	2	1 (<1)
Tufted Duck *Aythya fuligula*	-	-	12	12	1 (<1)
Scaup *Aythya marila*	-	1	-	1	1 (<1)
Long-tailed Duck *Clangula hyemalis*	-	117	-	117	10 (1.5)
Eider *Somateria mollissima*	40	972	-	1,012	78 (11.6)
King Eider *Somateria spectabilis*	-	1	-	1	1 (<1)
Common Scoter *Melanitta nigra*	-	2,221	-	2,221	25 (3.7)
Goldeneye *Bucephala clangula*	-	13	-	13	3 (<1)
Red-breasted Merganser *Mergus serrator*	44	431	2	477	86 (12.8)
Red-throated Diver *Gavia stellata*	-	310	-	310	82 (12.2)
Black-throated Diver *Gavia arctica*	-	16	-	16	10 (1.5)
Great Northern Diver *Gavia immer*	8	894	1	903	264 (39.4)
Little Grebe *Tachybaptus ruficollis*	1	64	-	65	30 (4.5)
Great Crested Grebe *Podiceps cristatus*	-	85	2	87	19 (2.8)
Slavonian Grebe *Podiceps auritus*	-	1	-	1	1 (<1)
Black-necked Grebe *Podiceps nigricollis*	-	2	-	2	1 (<1)
Cormorant *Phalacrocorax carbo*	805	1,592	62	2,459	395 (59.0)
Shag *Phalacrocorax aristotelis*	1,173	1,839	51	3,063	338 (50.4)
Little Egret *Egretta garzetta*	91	-	10	101	81 (12.1)
Grey Heron *Ardea cinerea*	490	-	33	523	237 (35.4)
Glossy Ibis *Plegadis falcinellus*	-	-	8	8	1 (<1)
Water Rail *Rallus aquaticus*	-	-	3	3	2 (<1)
Moorhen *Gallinula chloropus*	1	-	6	7	2 (<1)
Oystercatcher *Haematopus ostralegus*	11,808	5	1,177	12,990	578 (86.3)
Ringed Plover *Charadrius hiaticula*	3,961	60	19	4,040	197 (29.4)
Golden Plover *Pluvialis apricaria*	3,003	-	479	3,482	38 (5.7)
Grey Plover *Pluvialis squatarola*	157	-	-	157	43 (6.4)
Lapwing *Vanellus vanellus*	3,315	-	928	4,243	79 (11.8)
Knot *Calidris canutus*	277	-	163	440	7 (1.0)
Sanderling *Calidris alba*	2,514	-	23	2,537	86 (12.8)
Little Stint *Calidris minuta*	60	-	-	60	1 (<1)
Purple Sandpiper *Calidris maritima*	395	-	27	422	70 (10.4)
Dunlin *Calidris alpina*	4,252	-	-	4,252	113 (16.9)
Ruff *Calidris pugnax*	1	-	-	1	1 (<1)
Jack Snipe *Lymnocryptes minimus*	1	-	3	4	4 (<1)

Table 2 (Continued).

Species	**Intertidal**	**Sea**	**Land**	**Total**	**Number of sectors present in (with %)**
Snipe *Gallinago gallinago*	99	-	321	420	73 (10.9)
Woodcock *Scolopax rusticola*	-	-	2	2	2 (<1)
Black-tailed Godwit *Limosa limosa*	125	-	-	125	6 (<1)
Bar-tailed Godwit *Limosa lapponica*	516	-	-	516	42 (6.3)
Whimbrel *Numenius phaeopus*	17	-	-	17	7 (1.0)
Curlew *Numenius arquata*	3,838	1	3,090	6,929	443 (66.1)
Common Sandpiper *Actitis hypoleucos*	3	-	-	3	2 (<1)
Greenshank *Tringa nebularia*	251	-	8	259	114 (17)
Redshank *Tringa totanus*	2,867	-	123	2,990	347 (51.8)
Turnstone *Arenaria interpres*	3,308	-	121	3,429	299 (44.6)
Mediterranean Gull *Larus melanocephalus*	38	24	26	88	12 (1.8)
Little Gull *Hydrocoloeus minutus*	1	2	-	3	2 (<1)
Sabine's Gull *Xema sabini*	2	-	-	2	2 (<1)
Black-headed Gull *Chroicocephalus ridibundus*	6,851	2,384	865	10,100	306 (45.7)
Ring-billed Gull *Larus delawarensis*	2	-	2	4	4 (<1)
Common Gull *Larus canus*	4,676	2,191	1,827	8,694	321 (47.9)
Lesser Black-backed Gull *Larus fuscus*	824	65	55	944	77 (11.5)
Herring Gull *Larus argentatus*	15,038	2,960	1,683	19,681	566 (84.5)
Yellow-legged Gull *Larus arg. cachinnans michahellis*	1	1	1	3	3 (<1)
Iceland Gull *Larus glaucoides*	7	1	3	11	10 (1.5)
Glaucous Gull *Larus hyperboreus*	7	2	2	11	11 (1.6)
Great Black-backed Gull *Larus marinus*	2,558	706	264	3,528	427 (63.7)
Sandwich Tern *Sterna sandvicensis*	3	8	-	11	5 (<1)
Kingfisher *Alcedo atthis*	5	3	5	13	11 (1.6)
Total Wildfowl and allies	**6,298**	**10,101**	**3,251**	**19,650**	**-**
Total waders	**40,768**	**66**	**6,484**	**47,318**	**-**
Total gulls	**30,005**	**8,336**	**4,728**	**43,069**	**-**

Population estimates were generated for 25 species (Table 3). A comparison of these estimates with all-Ireland estimates based on the 2006/07 to 2010/11 period (Crowe & Holt 2013) shows that substantial proportions of Eider *Somateria mollissima,* Red-throated Diver *Gavia stellata,* Ringed Plover, Sanderling *Calidris alba,* Greenshank *Tringa nebularia* and Turnstone occur along non-estuarine coasts. Proportions of Purple Sandpiper *Calidris maritima* on non-estuarine coasts are notably high (78%). There was considerable change between NEWS-III and NEWS-II in the population estimates of many species (Table 3), with the most notable increases occurring for Teal *Anas crecca,* Eider, Shag, Purple Sandpiper, Snipe *Gallinago gallinago* and Greenshank, while five species showed declines in excess of 50% (Shelduck *Tadorna tadorna,* Red-throated Diver, Golden Plover, Knot *Calidris canutus* and Bar-tailed Godwit *Limosa lapponica*). The population estimates of Oystercatcher, Sanderling, Dunlin and Redshank were highly consistent between the two surveys, indicating that these proportions of their overall populations are relatively stable.

Geographical patterns of distribution

The highest concentrations of wildfowl were recorded in the eastern region while wader densities were highest in the northeast (Table 4). Densities ranged between 5.6 and 20.0 birds per kilometer for wildfowl and between 12.6 and 45.4 birds per kilometer for waders. Highest densities of most species occurred in the east and northeast with smaller numbers of species having highest densities in other regions (Table 5). Of the 15 wader species shown in Table 5, ten had higher densities in the east and northeast, with one notable exception being Ringed Plover with highest linear densities in the west. The regional distribution of densities of a selection of wildfowl and wader species is presented in Figure 2. The overall densities of 21 species were lower when compared

Table 3. Population estimates of a selection of waterbirds on non-estuarine coasts in the Republic of Ireland, Northern Ireland and in Ireland overall. For each species, the all-Ireland population estimate is given where known, together with the proportion on non-estuarine coasts based on the estimate generated during NEWS-III, 2015/16.

Species	Republic of Ireland	Northern Ireland	All-Ireland	Percentage change NEWS-II - NEWS-III (RoI)	All-Ireland population estimate[a]	Proportion using non-estuarine coasts
Light-bellied Brent Goose	2,038 (1,409-2,909)	1,066 (471-2,317)	3,167 (2,157-4,472)	28[b] -	36,380 -	8.7 -
Shelduck	62 (21-115)	56 (23-98)	120 (60-183)	-61 -	11,760 -	1 -
Wigeon	1,839 (1,007-3,234)	14 (0-59)	1,859 (1,034-3,262)	18 -	62,980 -	3 -
Teal	1,235 (724-1,855)	1,121 (462-2,223)	2,318 (1,414-3,474)	147[b] -	34,370 -	6.7 -
Eider	1,002 (546-1,753)	446 (261-704)	1,449 (963-2,223)	195[b] -	3,550 -	40.8 -
Red-breasted Merganser	717 (514-1,021)	54 (17-90)	772 (551-1,058)	22 -	2,130 -	36.2 -
Red-throated Diver	344 (180-922)	63 (35-123)	411 (246-992)	-59 -	920 -	44.7 -
Little Grebe	104 (58-160)	4 (0-16)	107 (59-164)	- -	2,230 -	4.8 -
Great Crested Grebe	104 (47-185)	1 (0-4)	105 (49-187)	32 -	4,060 -	2.6 -
Cormorant	2,756 (2,336-3,422)	889 (462-1,452)	3,717 (2,994-4,726)	-34 -	11,920 -	31.2 -
Shag	4,147 (3,212-5,592)	531 (403-696)	4,664 (3,660-6,086)	90 -	- -	- -
Grey Heron	655 (530-844)	139 (101-180)	793 (674-999)	22[b] -	2,500 -	31.7 -
Oystercatcher	14,530 (12,347-16,844)	3,603 (2,807-4,501)	18,025 (15,686-20,729)	-8 -	68,930 -	26.1 -
Ringed Plover	5,811 (4,121-7,582)	539 (290-768)	6,350 (4,576-8,154)	-18 -	10,290 -	61.7 -
Golden Plover	1,724 (727-3,385)	2,446 (339-6,352)	4,486 (1,582-8,118)	-69 -	118,480 -	3.8 -
Grey Plover	177 (99-271)	28 (11-50)	203 (129-310)	-17 -	3,050 -	6.7 -
Knot	496 (85-1,466)	3 (0-9)	502 (94-1,475)	-68 -	28,030 -	1.8 -
Sanderling	3,270 (2,188-4,556)	460 (30-1,635)	3,785 (2,422-5,336)	-4 -	5,830 -	64.9 -
Purple Sandpiper	340 (147-619)	156 (58-279)	499 (281-776)	54[b] -	640 -	78 -
Dunlin	4,703 (2,546-7,391)	1,395 (537-2,673)	6,199 (3,304-9,460)	6 -	56,700 -	11 -
Bar-tailed Godwit	635 (282-1,463)	26 (6-89)	672 (307-1,486)	-59 -	15,100 -	4.5 -
Curlew	8,821 (7,028-10,549)	1,839 (1,317-2,412)	10,517 (8,754-12,401)	-15 -	35,250 -	29.8 -
Greenshank	416 (322-541)	5 (1-11)	420 (328-548)	116 -	1,040 -	40.4 -
Redshank	2,502 (1,895-3,194)	1,392 (1,101-1,772)	3,911 (3,221-4,606)	-1 -	29,520 -	13.2 -
Turnstone	3,126 (2,394-3,951)	1,518 (1,202-1,766)	4,665 (3,912-5,446)	-26 -	9,630 -	48.4 -

[a] after Crowe & Holt (2013).
[b] Calculated as simple percentage change between raw count data (NEWS-III versus NEWS-II) due to data missing from main analysis.

Table 4. Densities of wildfowl (including their allies) and waders within six coastal regions in Ireland during NEWS-III, 2015/06.

Region	Non-estuarine length covered (km)	Total wildfowl	Wildfowl density (birds/km)	Total waders	Wader density (birds/km)
Northwest	414.6	5,209	12.6	6,911	16.7
West	583.7	4,095	7.0	11,006	18.9
Southwest	475.8	2,651	5.6	6,000	12.6
Southeast	123.8	811	6.6	2,513	20.3
East	151.6	3,031	20.0	5,857	38.6
Northeast	330.8	3,853	11.6	15,031	45.4

with NEWS-II, while increases were found for 13 species. Relatively large increases in density were found for Teal, Long-tailed Duck *Clangula hyemalis,* Eider, Little Egret *Egretta garzetta,* Snipe and Greenshank. Both Wigeon *Anas penelope* and Teal increased substantially in the northwest. Of the waders, 11 out of 15 species increased in density in the northwest, while all except one species (Bar-tailed Godwit) decreased in density in the east (Table 5).

Discussion

More than one-third of the coastline of Ireland comprises open, non-estuarine coastline that represents suitable habitat for non-breeding wintering waterbirds. NEWS-III, the third all-Ireland survey of non-estuarine coastal waterbirds, coincided with a prolonged period of poor weather. Despite this, nearly two-thirds of the all-Ireland non-estuarine coast was surveyed, which represents an increase on that surveyed previously. NEWS-III recorded ten additional waterbird species when compared to NEWS-II (e.g. Pink-footed Goose *Anser brachyrhyncus,* Greenland White-fronted Goose *Anser albifrons flavirostris* and Woodcock *Scolopax rusticola*) and the recorded total of 110,000 waterbirds is also an increase on the total recorded by NEWS-II and is likely to be a reflection of the increased coverage. It is clear that non-estuarine coastal habitats continue to support a broad diversity of waterbirds, and supports substantial proportions of the national estimates of divers, seaducks, Cormorant, Grey Heron *Ardea cinerea* and several waders; and perhaps most notably Oystercatcher, Ringed Plover, Sanderling, Purple Sandpiper, Greenshank and Turnstone.

Of note was the abundance of the Herring Gull, the most numerous species overall and in numbers that were more than double than recorded during NEWS-II. Shag, Cormorant and Common Scoter were the most numerous wildfowl species (including allies) recorded, with numbers that represented a substantial proportion of the estimated all-Ireland population. The population estimate for Eider corresponds to a very high proportion (>40%) of the all-Ireland population estimate; these birds being recorded in Antrim, Derry and Down but overall more abundant in the northwest (Donegal and Sligo). Oystercatcher, Curlew, Dunlin, Lapwing and Ringed Plover were especially numerous among the waders (>4,000 birds), with numbers of Oystercatcher using non-estuarine coasts equivalent to over a quarter of the all-Ireland population. Similarly, a relatively high proportion of the all-Ireland population of Sanderling utilised non-estuarine coasts, consistent with a species that favours wide open sandy beaches (e.g. Summers *et al.* 2002). However, it should be borne in mind that the current all-Ireland population estimates (Crowe & Holt 2013) are becoming out-of-date. Importantly, the non-estuarine coastal estimates reported here will input into the next analysis of the population estimates of Irish wintering waterbird populations. The differences observed in estimates generated for species in

Plate 249. Purple Sandpiper (Michael O'Clery).

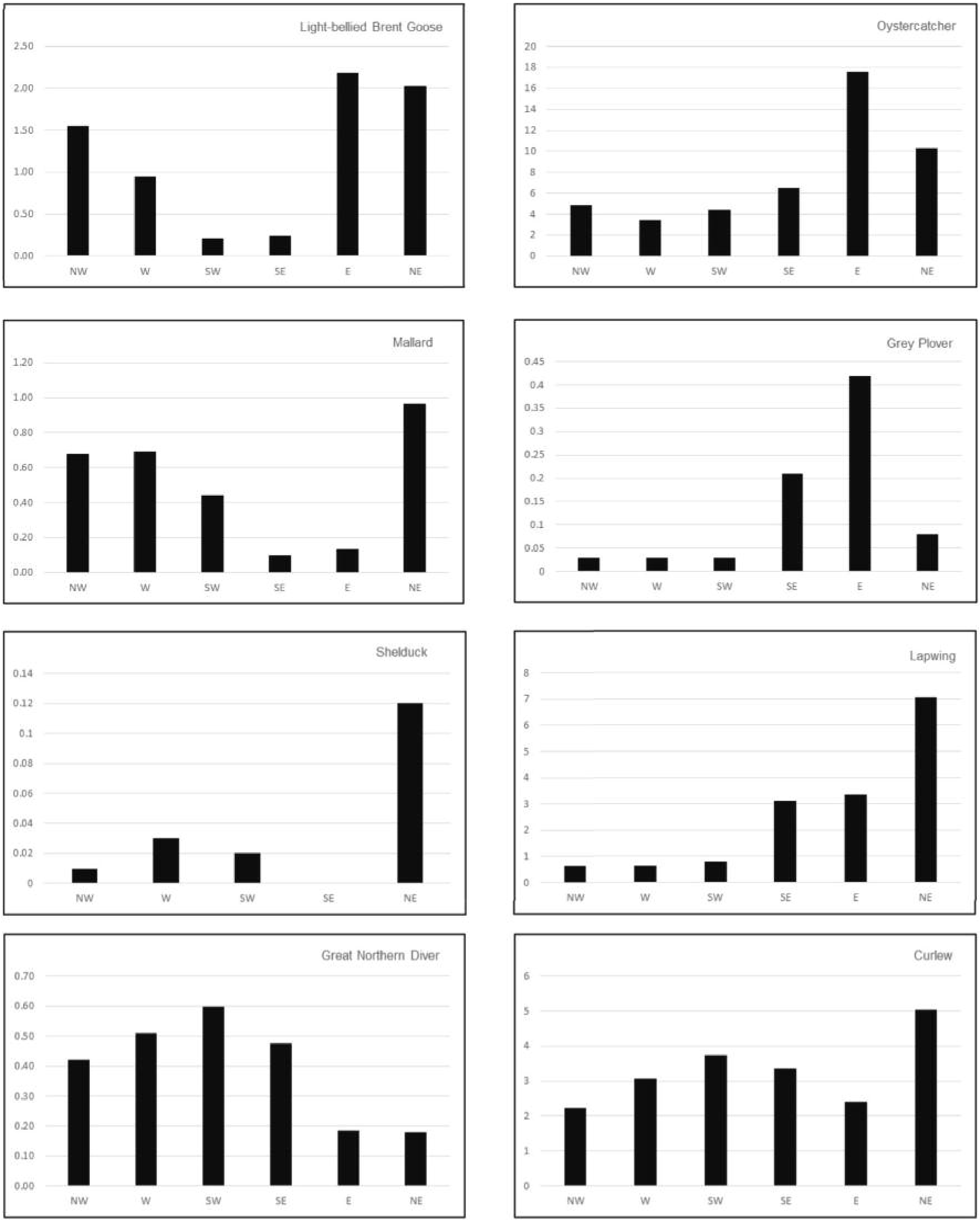

Figure 2. Linear densities of a selection of wildfowl (left) and waders (right) between regions as recorded during NEWS-III, 2015/16.

Table 5. Linear densities (birds per km) of non-breeding waterbirds on non-estuarine coasts during NEWS-III, together with the percentage change between NEWS-II and NEWS-III within six coastal regions in Ireland. No change estimate was possible where a species was absent during NEWS-II, while an asterisk indicates where present during NEWS-II and absent during NEWS-III, 2015-16.

Species	NW	W	SW	SE	E	NE	Overall
Mute Swan	0.07 (-17.1)	0.03 (144.73)	0.08 (123.35)	0.02	0.18 (-72.01)	0.02	0.06 (-34.47)
Barnacle Goose	2.18 (14.59)	0.97 (328.09)	-	-	-	-	0.71 (0.69)
Light-bellied Brent Goose	1.55 (23.4)	0.94 (162.5)	0.21 (37)	0.23 (-94.2)	2.18 (-54.13)	2.02 (-29.5)	1.12 (-27.50)
Shelduck	0.01 (-91.7)	0.03 (267.1)	0.02 (-70.4)	*	0.01 (-56)	0.12 (187.5)	0.04 (-36.08)
Wigeon	1.25 (119.1)	0.42 (6.7)	0.63 (54.5)	-	0.03 (-99.2)	0.05	0.52 (-28.70)
Teal	0.7 (408.4)	0.55 (116.3)	0.35 (203.1)	-	0.19 (-93.9)	3.01 (812.4)	0.87 (79.71)
Mallard	0.68 (25.5)	0.69 (34.7)	0.44 (41.8)	0.1	0.13 (-91)	0.96 (465.3)	0.60 (23.01)
Long-tailed Duck	0.18 (355.5)	0.07 (42.09)	-	*	-	-	0.06 (200.00)
Eider	1.53 (39)	-	-	-	*	1.14 (418.2)	0.49 (104.76)
Common Scoter	0.03 (-99)	0.03 (-41.1)	0 (-98.7)	3.1 (339.3)	11.91 (-30.2)	-	0.01 (-99.76)
Red-breasted Merganser	0.24 (24)	0.3 (32.9)	0.24 (132.3)	0.02 (-90.2)	0.3 (-46.8)	0.12 (303.3)	0.23 (17.53)
Red-throated Diver	0.02 (-77.8)	0.03 (-63.5)	0.33 (-34.9)	0.14 (-76.7)	0.36 (-51.2)	0.16 (35.8)	0.15 (-45.61)
Great Northern Diver	0.42 (42.42)	0.51 (7.34)	0.6 (66.84)	0.48 (173.73)	0.18 (-45.03)	0.18 (493.33)	0.43 (36.88)
Little Grebe	0.01 (-40.1)	0.03 (-35.22)	0.09 (644.05)	-	*	0.01	0.03 (11.71)
Great Crested Grebe	0.03 (948.23)	0.01 (51.16)	-	0.12 (1152.68)	0.32 (-27.2)	0 (-97.7)	0.04 (-38.27)
Cormorant	0.84 (10.09)	0.74 (-22)	0.94 (-42.3)	1.83 (-11)	2.07 (-61.43)	2.09 (254.07)	1.18 (-21.31)
Shag	2.45 (253.1)	1.24 (107.6)	1.29 (55)	0.31 (-3.8)	1.79 (15.9)	1.22 (-24.6)	1.47 (63.06)
Little Egret	-	0.03 (447)	0.13 (77.1)	0.05 (0.2)	0.03 (-26.7)	0.03	0.05 (113.15)
Grey Heron	0.25 (19.1)	0.27 (-36.2)	0.16 (-37.2)	0.15 (-0.8)	0.27 (-4.1)	0.38 (31.4)	0.25 (-16.15)
Oystercatcher	4.89 (2)	3.41 (-9.7)	4.4 (-4.3)	6.5 (-9)	17.56 (-12)	10.31 (-22.6)	6.24 (-18.34)
Ringed Plover	1.55 (-5)	3.45 (-10.6)	0.73 (-23.4)	1.06 (-50.7)	2.45 (-30)	1.626 (-7.1)	1.94 (-18.91)
Golden Plover	0.87 (19.5)	0.66 (-62.2)	0 (-91.9)	0.9 (-77.9)	2.72 (-80.4)	6.68 (-57.4)	1.67 (-65.00)
Grey Plover	0.03 (105.9)	0.03 (-48.2)	0.03 (-51.6)	0.21 (3.4)	0.42 (-3.6)	0.08 (105)	0.08 (-17.54)
Lapwing	0.64 (94.5)	0.64 (-73.3)	0.79 (-11.7)	3.11 (-66.2)	3.34 (-78.1)	7.059 (-5.1)	2.04 (-52.59)
Knot	0.17 (-50.01)	0.31 (1102.6)	-	*	1.23 (-84.5)	0.01 (-70)	0.21 (-75.23)

Table 5 (Continued).

Species	NW	W	SW	SE	E	NE	Overall
Sanderling							
	1.23 (103.8)	1.58 (-22.1)	0.29 (14.3)	1.87 (-16.1)	1.59 (-23.9)	1.508 (357)	1.22 (5.37)
Purple Sandpiper							
	0.3 (20.6)	0.13 (-59.3)	0.06	0.06 (33.6)	0.18 (-16.8)	0.49 (69)	0.20 (-2.32)
Dunlin							
	2.13 (64.7)	2.7 (7.4)	0.25 (-7.1)	0.36 (-32.9)	2.09 (-27.2)	3.96 (-52.5)	2.04 (-23.92)
Snipe							
	0.05 (103.2)	0.18 (381.9)	0.45 (146.6)	0.1 (0.2)	0.01 (-87.8)	0.21 (87.3)	0.20 (111.06)
Bar-tailed Godwit							
	0.36 (-76.6)	0.39 (8.9)	0.1 (1605.9)	0.16 (-58.2)	0.3 (49.9)	0.08 (-51.8)	0.25 (-46.28)
Curlew							
	2.23 (-20.8)	3.06 (15.3)	3.73 (-12.6)	3.35 (-9)	2.39 (-74.8)	5.03 (17.5)	3.33 (-21.53)
Greenshank							
	0.14 (189.5)	0.14 (115.9)	0.2 (168)	0.08 (4.4)	0.06 (-55)	0.02 (-10)	0.12 (94.46)
Redshank							
	1.03 (49.75)	0.75 (-12.83)	0.74 (-12.06)	0.86 (78.72)	2.28 (-17)	4.04 (-40.82)	1.44 (-25.72)
Turnstone							
	1.07 (6.76)	1.32 (-23.73)	0.61 (-37.78)	1.66 (-15.67)	2 (-39.87)	4.3 (-16.58)	1.65 (-24.21)

the Republic of Ireland between NEWS-III and NEWS-II was quite variable. The largest increases in population size were for Teal, Eider, Shag, Purple Sandpiper, Snipe and Greenshank. Numbers of Teal increased more than five times in the northwest, although overall wildfowl density in the northwest was relatively similar between the two surveys. Purple Sandpiper increased in both numbers and distribution between the two surveys; recorded in eight counties in the Republic of Ireland during NEWS-II with an increase to 14 counties during NEWS-III, including Antrim and Down in Northern Ireland. However, the well documented flock of Purple Sandpipers at Quilty (Clare) was not encountered during NEWS-III. During recent seasons of I-WeBS counts, Purple Sandpipers have been recorded in numbers in excess of 130, with a peak count of 80 during the 2015/16 season (I-WeBS unpublished data), so the lack of detection of this key flock during NEWS-III has undoubtedly affected the population estimate.

While the population estimate for Red-throated Diver has seemingly declined since NEWS-II this could be as a result of the poor weather conditions during the counting period resulting in poor conditions in general for counting sea ducks and divers. For example, January 2016 saw above average rainfall and several bouts of gale force winds (http://www.met.ie/climate/monthly-weather-reports.asp). For this reason population estimates for Common Scoter and Great Northern Diver are not presented. In contrast, detection of Eiders may have been better as they tend to occur closer to shore than species such as Common Scoter (Milne & Campbell 1973). Weather may also have affected the results for wader species such as Knot and Bar-tailed Godwit. Both species are associated with open sandy coasts, therefore poor weather may have caused them to move to estuaries for shelter resulting in lower detections rates during NEWS-III.

Relatively high proportions of Dunlin and Ringed Plover were recorded along non-estuarine coastlines, despite these two waders being considered highly reliant on intertidal sand and mudflat habitat for foraging, compared to, for example, Oystercatcher, Curlew and Lapwing which show flexibility and can forage amongst a variety of habitats including mudflats, rocky shores and on land. Dunlin and Ringed Plover are thought to be faithful to roost sites (e.g. Rehfisch *et al.* 2003), with a key criteria of suitable roosting sites being proximity to feeding areas (e.g. Dias *et al.* 2006). At the wetland sites of Dublin Bay and Malahide Estuary, significant proportions of both species' site populations have been shown to roost along open coastline and often outside of the areas designated as Special Protection Areas (SPA) (e.g. Lewis *et al.* 2016, Lewis & Butler 2017). This has various management implications. Firstly, it may indicate that adequate roost sites are unavailable within the estuarine sites or that within-site roost sites are unsuitable due to factors such as human disturbance. Secondly, it highlights the importance of taking into account cumulative *ex-situ* pressures acting upon waterbirds when undertaking ecological impact assessments of plans and projects within the SPA network.

Estimates of the size of waterbird populations should be regularly updated as they provide a basis for population and site conservation (Crowe *et al.* 2008). NEWS-III has again shown that non-estuarine coastal habitats can support substantial proportions of the national estimates of several waterbird species and these habitats are undoubtedly important in the life cycles of many of these species. While population estimates determined during NEWS-III will feed into the forthcoming updated national and all-Ireland waterbird population estimates, some caution is required when interpreting the observed trends or making comparisons between NEWS-III and NEWS-II because these surveys were undertaken a relatively long time apart. With weather being such an influential factor in the results, such factors support the need for more regular monitoring of the non-estuarine coast, for example, as part of I-WeBS.

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Winter diet of the Hen Harrier *Circus cyaneus* in coastal east County Cork

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The diet of Hen Harriers *Circus cyaneus* was studied through an examination of 163 pellets collected at night-roost sites during winter in east County Cork. These Hen Harriers hunted over farmland and wetland habitats, roosted in marshland and rough ground vegetation, and fed upon a range of birds and mammals. Mammal prey such as Brown Rat *Rattus norvegicus,* Bank Vole *Myodes glareolus,* House Mouse *Mus musculus,* Field Mouse *Apodemus sylvaticus,* Rabbit *Oryctolagus cuniculus* and Pygmy Shrew *Sorex minutus* formed 39.9%, while various bird species formed 77.2% of their diet (by percentage frequency). The birds were predominantly passerines, although few could be identified to species level, and Common Snipe *Gallinago gallinago* and pigeons (Columbidae) were also taken.

Introduction

The Hen Harrier *Circus cyaneus* is currently Amber-listed on Birds of Conservation Concern in Ireland (Colhoun & Cummins 2013). It can be considered a rare species with fewer than 200 breeding pairs recorded during recent surveys (Ruddock *et al.* 2012, 2016), and it has declined in some areas (e.g. O'Donoghue 2012). The Hen Harrier has not been known to breed within the present study area, at least not within the last 40 years, although it may have done so during an earlier period before the intensification of grasslands. During approximately the last 40 years, the closest that Hen Harriers have bred to the present study area is in the Drum Hills (Waterford), along the Cork and Waterford border between Tallow and Youghal and in the Nagles Mountains (Cork), approximately 10 km to 20 km away to the north.

Plate 250. Hen Harrier (Shay Connolly).

The Hen Harrier has been the subject of much research in Ireland over the last 20 years (e.g. Wilson *et al.* 2009, 2012, Irwin *et al.* 2011, Fernández-Bellon *et al.* 2015), mainly because of its legal status as an Annex 1 species under the EU Birds Directive. Chief among the concerns for its survival as a breeding species in Ireland are its small and fragmented population, and the fact its natural open habitat in upland areas has been targeted for conversion to grassland and conifer plantation, and more recently as sites for windfarms in an effort to meet Ireland's need to reduce the use of fossil fuels.

Study area and methods

The present study area is a coastal band consisting of a mixture of agricultural land (pasture and tillage) situated between Cork Harbour and Youghal. Along the coast there is steep-sloping heath with European Gorse *Ulex europaeus,* Western Gorse *Ulex gallii,* Bell Heather *Erica cinerea* and Bracken *Pteridium aquilinum.* There are also wetland habitats near the coast, such as at Ballycotton, Ballymacoda and Ballyvergan; these include saltmarsh, rough grazing marsh and reedbed *Phragmites australis.* Hen Harriers have regularly been seen hunting over all of the above habitats (personal observations), but especially in the wetland areas, usually between autumn and spring. Three roost sites are known within the study area, where usually fewer than five birds settle on the ground for the night within dense vegetation. Two of the roosts are on level ground among wetland vegetation, while the third is in the steep side of a small valley.

At each site searches of the area where it was known that the harriers roosted were undertaken in January, February, March and December 2012. All searches took place during the mid-day period when birds were absent from the roost area, in order to minimise disturbance. Pellets (n = 163) were teased apart and dissected in the manner described by Yalden (2009). Results are expressed as percentage frequency of occurrence of each prey species within each individual pellet. In analysis of pellets of diurnal birds of prey it is usually not possible to count individual prey items, therefore the more accurate calculation of biomass is more difficult to carry out (e.g. Village 1990, Smiddy 2018), as can be done with analysis of owl pellets (Cullen & Smiddy 2012, Smiddy 2013).

Results

Pellets had a mean length and width of 31.1 x 17.2 mm, respectively (n = 123). Among the mammal prey, rodents were the most frequently recorded (Brown Rat *Rattus norvegicus*, Bank Vole *Myodes glareolus,* House Mouse *Mus musculus* and Field Mouse *Apodemus sylvaticus*), although Rabbit *Oryctolagus cuniculus* was also recorded, as was a single case of Pygmy Shrew *Sorex minutus* (Table 1). Most of the bird prey species were passerines, although few were identified to species level (Meadow Pipit *Anthus pratensis,* Wren *Troglodytes troglodytes,* Common Starling *Sturnus vulgaris,* Chaffinch *Fringilla coelebs,* Linnet *Carduelis cannabina* and Bullfinch *Pyrrhula pyrrhula*). Non-passerines (Common Snipe *Gallinago gallinago* and pigeons (Columbidae)) also occurred in low frequency, compared to the passerines.

A careful examination of the pellets which contained passerine bird remains showed that there were plant seeds in 53 of them, while a further eight pellets had remains of small beetles. It is considered most likely that the plant seeds and beetles were taken accidently, they having been initially swallowed by the passerines concerned, which then became prey of the Hen Harrier. Plant seeds of five or six species were recognised (but not identified), and samples of each were sown in potting compost free of contamination from weed seeds. However, only two species grew to enable identifi-

Table 1. Winter diet of Hen Harriers in coastal east County Cork, 2012, expressed as percentage frequency of occurrence of each prey species within each individual pellet.

Prey species	% frequency
Rabbit	4.3
Brown Rat	11.7
Field Mouse	3.1
House Mouse	4.3
Bank Vole	5.5
Pygmy Shrew	0.6
Small rodent[1]	10.4
All mammal prey	**39.9**
Common Snipe	5.5
Pigeon species[2]	3.7
Meadow Pipit	1.2
Wren	1.8
Common Starling	1.2
Chaffinch	0.6
Linnet	0.6
Bullfinch	0.6
Small passerines[3]	62.0
All bird prey	**77.2**

[1] Small rodent; most probably refer to Field Mouse and/or House Mouse, rather than Bank Vole.

[2] Pigeon species; none identified to species, most probably Woodpigeon *Columba palumbus* and/or Rock Dove/Feral Pigeon *Columba livia,* rather than Stock Dove *Columba oenas*.

[3] Small passerines; none larger than Song Thrush *Turdus philomelos* and Common Starling, and most were probably seed-eating species.

Plate 251. Hen Harrier (Shay Connolly).

cation; these were Redshank *Persicaria maculosa* and Knotgrass *Polygonum aviculare,* both of which are common ruderal weeds of cultivated and disturbed ground (O'Mahony 2009). A few pellets also contained small amounts of grass, straw and grit, all indicative of material taken accidently while feeding on bird and mammal prey on the ground in an agricultural landscape.

Discussion

Despite the conservation importance of the Hen Harrier in Ireland, very little research has been undertaken on its feeding ecology, and much of what has been published here is in the form of casual observations rather than dedicated research (e.g. Watson 1977). The most substantial collection of prey items in the form of an analysis of pellets has been published for Northern Ireland by Scott (2005). Most of the pellets from Northern Ireland appear to have come from nests, with some also from roosts apparently outside the breeding season, but the data from pellets from different seasons are presented together (see Tables 1 & 2 in Scott (2005)). Therefore, this makes a direct comparison between the east Cork and Northern Ireland studies difficult. The Hen Harriers in Northern Ireland were feeding predominantly on birds, with the Meadow Pipit, Common Starling and Skylark *Alauda arvensis* being the most important prey items over the period 1991 to 2005. They also preyed upon a number of mammal species, with lagamorphs being the most important. Most prey items in the east Cork study were also birds, but most could not be identified beyond the level of 'small passerine' (Table 1).

A preliminary analysis of Hen Harrier pellets and prey remains collected at Duhallow, south-west Ireland has shown marked differences between breeding season and winter season diet (McCarthy *et al.* 2017). Breeding season diet was dominated by small passerines (71% by number of prey items) with small mammals, primarily Bank Vole, comprising 27% of the diet. Twelve species of small passerine were identified. Although small passerines remained the most important prey group during the winter season (58%), small mammals comprised a significantly higher proportion of winter diet (42%) compared to breeding season diet. The remains of beetles were also found within pellets during both seasons at Duhallow (McCarthy *et al.* 2017), and this was also the case in Northern Ireland (Scott 2005) and in the east Cork winter study reported here. It is considered unlikely that beetles were deliberately targeted by Hen Harriers, and were more likely to have been secondary prey from the stomachs of small bird prey species.

This study in east Cork, although admittedly restricted in nature, adds one small piece to the complicated picture of Hen Harrier feeding ecology in Ireland. Further dietary studies throughout the year are urgently required from as representative a range of habitats as possible, and given the size difference between males and females ways in which the diet of the different sexes could be identified should be investigated.

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Factors affecting the choice of roost sites by wintering waders in South Dublin Bay, Ireland

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Keywords: Disturbance, roost sites, roosting, waders

Plate 252. Wader roost, Dublin Bay (John Coveney).

A study of the use of high tide roost sites by waders in South Dublin Bay was carried out from January to March 1991. Primary roost sites were used by all twelve species on the majority of dates but, when these sites were covered on high spring tides, secondary roost sites were used by some of the birds. Bar-tailed Godwit *Limosa lapponica* and Knot *Calidris canutus* left the South Bay entirely in these conditions. This suggests that these species have different roosting requirements to other waders. On dates when disturbance was recorded at the preferred roost sites the overall numbers of waders roosting in the study area was reduced. Oystercatcher *Haematopus ostralegus* was the only species present in all roosts and its numbers in the area at low tide and high tide were positively correlated. The importance of Cockles *Cerastoderma edule* in the diet of this species may define the principal foraging areas which are close to the primary high tide roost sites. Changes in the total wader populations of this area over a 25-year period are reviewed.

Introduction

Dublin Bay is among the most important wetlands in Ireland for non-breeding waders (Crowe 2005) and they use traditional high tide roosts from year to year. The close proximity of Dublin city, the largest urban area in Ireland, often puts these roost sites under pressure either from human disturbance or, more permanently, due to land-claim from the sea. Thus, in planning and assessing the possible impacts of recreational uses or new developments close to the intertidal area, it is important to know the exact location and relative importance of wader roosts. This is also helpful for designing and constructing new roost sites to replace any that become unavailable or too heavily disturbed (e.g. Burton *et al.* 1996).

It is generally understood that communal roosting in waterbirds reduces the individual's risk of predation (Cresswell 1994) and conserves energy (Wiersma & Piersma 1994). The choice by non-breeding waders of high tide roost sites in estuaries depends primarily on tide height, proximity to foraging areas, all-round visibility for detecting potential predators and the level of human disturbance (Furness 1973, Kirby *et al.* 1993, Colwell 2010). Given the importance of the existing wader roosts in South Dublin Bay this study aimed to examine the factors affecting their use.

Study area

Dublin Bay borders the eastern side of Dublin city which has a population of over one million people. The bay has high cultural value and is extensively used for recreation. Dublin Port effectively subdivides the Bay into northern and southern halves. There are also a considerable number of industries, including a major power station, waste water treatment plant, waste incineration facility, oil storage, scrap metal recycling and cement manufacturing. Most of the bay is surrounded by hard engineered shorelines (Nairn *et al*. 2017).

The intertidal area of Dublin Bay covers at least 2000 ha with the area subdivided into northern and southern bays by the River Liffey channel (Jeffrey *et al*. 1992). The main foraging areas for waders in South Dublin Bay are within a triangular area of sandflats covering approximately 840 ha and stretching from the South Wall of Dublin Port to the West Pier of Dun Laoghaire Harbour (Figure 1). At the northern end of this area the distance from spring high water mark to spring low water mark is over 2.5 km. The sediments are predominantly fine sands but there are small areas of muddy sand in the upper shore area. The macroinvertebrate community is dominated by molluscs, with polychaete worms and all other groups contributing less than 20% of total invertebrate biomass (Wilson 1982).

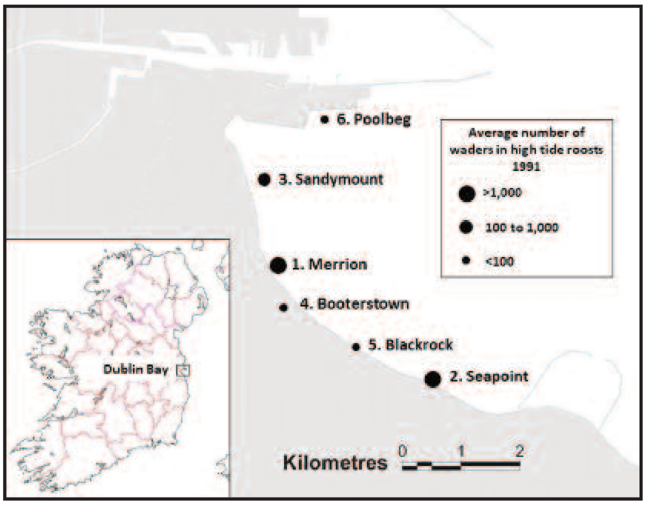

Figure 1. Location and size of wader roost assemblages in South Dublin Bay in 1991.

Since the late 1970s, Dublin Bay has consistently held one of the largest concentrations of waders in the Republic of Ireland (Hutchinson 1979, Sheppard 1993, Crowe 2005). A general account of the waterbird populations and their habitats in Dublin Bay was given by Ó Briain (1987), while more complete population estimates were given by Crowe (2006). More recent studies confirm the importance of this coastal area for waterbirds throughout the year (Tierney *et al.* 2017) and some population data from the latter study are given here for comparison. The largest wader roosts are on the north side of the bay within saltmarsh at Bull Island (Hutchinson & Rochford 1974, Ó Briain 1987, Tierney *et al.* 2017). However, South Dublin Bay also has a number of large wader roosts that have been present since at least 1991.

Field observations

This paper is based on a dedicated survey undertaken in 1991. On 11 dates between early January and late March 1991, full counts of all waders in each high tide roost site were carried out. Each survey involved a series of counts of all species within two hours of high tide and the figures used here were the maximum counts for each species. Survey dates were chosen to represent all stages in the spring-neap cycle and hence a range of tide heights. Counts were carried out on both weekday and weekend dates to include different levels of human disturbance activity. All observations were carried out by the same two observers using telescopes with 20x magnification and all registrations were marked on maps at a scale of 1: 5,000. All flocks of waders foraging within two hours of low tide were also counted and mapped on the same dates plus an additional three dates. A Spearman Rank Correlation Coefficient was calculated to compare low tide and high tide counts of Oystercatcher *Haematopus ostralegus* carried out on the same dates. This was designed to compare the populations of one widespread species foraging at low tide with those roosting at high tide in the same area. Population counts undertaken during rising tides in 2014 to 2017, as part of the Dublin Bay Birds Project (see Tierney *et al.* 2017), are presented for comparison with the earlier estimates.

Results

Usage of roosts within South Dublin Bay

South Dublin Bay held nationally important winter populations of seven wader species and smaller numbers of several other species in 1991. Since then there have been some significant changes in the wader populations in South Dublin Bay and peak counts for the equivalent winter months in 2014-2016 are given in Table 1. There is a striking similarity between the peak counts of Oystercatcher, Curlew *Numenius arquata*, Knot *Calidris canutus* and Dunlin *Calidris alpina* in 1991 and the mean peaks in 2014-2017. There have been increases in numbers of Bar-tailed Godwit *Limosa lapponica*, Redshank *Tringa totanus*, Black-tailed Godwit *Limosa limosa* and Greenshank *Tringa nebularia*. The latter two species were not recorded in South Dublin Bay in 1991. The main roosts identified in South Dublin Bay in 1991 are shown in Figure 1 and described in Appendix 1.

Usage of high tide roost sites by each wader species in 1991 is shown in Table 2. The major roost sites, each used by

Table 1. Peak counts of waders in South Dublin Bay in 1991 and during 2014 to 2017.

	Jan-Mar 1991[1]	Jan-Mar 2014[2]	Jan-Mar 2015[2]	Jan-Mar 2016[2]	Jan-Mar 2017[3]	Mean peak 2014-17
Number of counts	14	3	3	3	3	-
Oystercatcher *Haematopus ostralegus*	1522	1055	930	1616	2089	1423
Ringed Plover *Charadrius hiaticula*	200	103	75	167	188	133
Grey Plover *Pluvialis squatarola*	30	76	0	2	18	24
Knot *Calidris canutus*	2000	1150	450	1813	5027	2110
Sanderling *Calidris alba*	400	60	57	317	111	136
Dunlin *Calidris alpina*	2530	1570	550	2568	5402	2523
Black-tailed Godwit *Limosa limosa*	0	80	176	163	11	108
Bar-tailed Godwit *Limosa lapponica*	620	1867	524	1033	484	977
Curlew *Numenius arquata*	60	133	32	12	38	54
Greenshank *Tringa nebularia*	0	8	12	18	10	12
Redshank *Tringa totanus*	170	215	292	511	427	361
Turnstone *Arenaria interpres*	60	59	27	22	40	37

Sources: 1. This study; 2. Dublin Bay Birds Project (Tierney *et al.* 2017); 3. Irish Wetland Birds Survey (I-WeBS).

more than two species, were at Merrion, Seapoint and Blackrock. All other sites were used by one or two species only. Oystercatcher used many different roosting habitats including sandbanks and railway embankments. By comparison, Knot and Bar-tailed Godwit were only recorded at the Merrion roost site. Comparison of Oystercatcher counts in South Dublin Bay at high and low tide periods on the same dates show that low tide estimates were mostly higher than high tide counts and there was a positive correlation (r^2 = 0.853) between high tide and low tide counts (Figure 2).

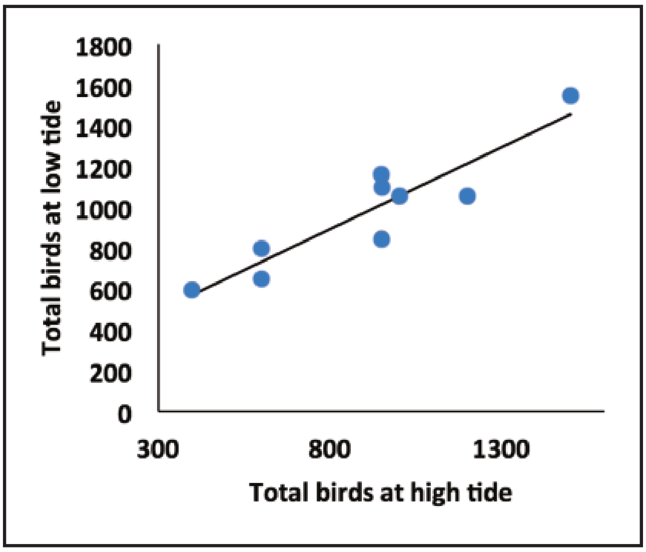

Figure 2. Oystercatcher numbers at high and low tide on nine separate dates in South Dublin Bay in 1991.

Patterns of roost usage during different tide heights

The largest single high tide roost site in South Dublin Bay throughout the entire period was at Merrion. On neap tides waders also roosted on sandbars at Sandymount and Poolbeg.

Table 2. Roost site usage by wader species in South Dublin Bay in 1991.

Species	Roost site: Merrion	Seapoint	Sandymount	Booterstown	Blackrock	Poolbeg
Oystercatcher	•	•	•	•	•	•
Ringed Plover	•	•			•	
Grey Plover	•	•				
Knot	•					
Sanderling	•	•				
Dunlin	•	•	•		•	
Bar-tailed Godwit	•					
Curlew	•	•				
Redshank	•	•		•	•	
Turnstone	•	•			•	

However, in 1991, these three roost sites were covered by spring high tides (≥4.1 m OD). On such dates the majority of the waders flew south to the other roost sites at Seapoint, Booterstown and Blackrock (Figure 3). The Sandymount and Poolbeg roosts were prone to disturbance at higher tides as people and dogs had direct access to the reduced shoreline from the adjacent car parks.

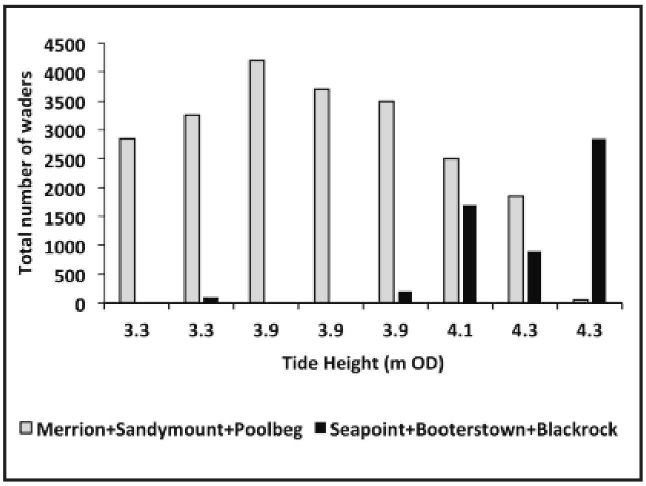

Figure 3. Oystercatcher numbers at high and low tide on nine separate dates in South Dublin Bay in 1991.

The Merrion roost was used by the greatest numbers of waders on all occasions except when it was covered by spring tides. The Seapoint, Booterstown or Blackrock roost sites did not hold the full range and density of species that roosted at Merrion (Table 2). Both Knot and Bar-tailed Godwit left South Dublin Bay on high tides over 4.1 m, suggesting that the habitat at the more southerly roost sites was unsuitable for them.

Effects of disturbance

Most of the human activity on the intertidal area of South Dublin Bay involved people walking, jogging or exercising dogs. In the present study the disturbance levels were highest when low tide occurred in the middle of the day and when this coincided with fine weather at weekends. Of the 11 dates when high tide roost counts were conducted, disturbance events causing birds to take flight were recorded on eight days, seven days and six days respectively at the Poolbeg, Sandymount and Merrion roost sites. Waders disturbed at the Poolbeg or Sandymount roost sites generally flew first to the Merrion site. If further disturbance occurred here, some of the birds flew on to the Seapoint roost while others left South Dublin Bay entirely. The total counts of waders in all roost sites were lower on dates when the Merrion site was disturbed, irrespective of the height of the tides.

Discussion

Several authors have discussed the factors affecting choice of high tide roost site by coastal waders. The main factors are considered to be predation risk, disturbance rates and the energetic costs of remaining thermoneutral at the roost site and flying to the roost from the feeding grounds (Rogers 2003, Rogers *et al.* 2007). Luís *et al.* (2001) proposed a quantitative method for assessing the quality of high tide roost sites used by Dunlins in Portugal based on the same three attribute categories. Rosa *et al.* (2006) studied whether avoidance of predators or maximising feeding opportunities were the dominant factors and concluded that the preference by most species of waders for roosting in upper intertidal flats was influenced by lower predation pressure here. Colwell *et al.* (2003) proposed that the substantial variation in abundance of waders at roosts was due to height of the tide at roosts differing in elevation, availability of alternative foraging habitats, disturbance by predators or humans and migratory movements.

A positive correlation between frequency of occurrence and the abundance of waders at roost sites has been found elsewhere (Colwell *et al.* 2003). With their observation that multiple species shared the top-ranked roosts this supported the concept of a few primary roost sites. The occurrence of multiple species in the Merrion and Seapoint roosts suggests that these were the primary wader roost sites in South Dublin Bay in 1991 (Table 2).

Plate 253. Oystercatcher (John Fox).

Links between roost sites and foraging areas

Oystercatcher was the only species present at all high tide roosts and was the subject of more detailed investigation in this study. It was shown by Quinn and Kirby (1993) that the Cockle *Cerastoderma edule* was the most important prey item for the Oystercatcher in South Dublin Bay. This species contributed 74% of the biomass of bivalves present in the sandflats. They found that, over all the plots sampled, the mean number of Oystercatchers counted was highly correlated with both Cockle density and biomass. Using focal observations on foraging Oystercatchers, Quinn (1988) showed that Cockles were by far their most important prey species and that other prey species were rarely taken. The macroinvertebrate fauna of Dublin Bay was sampled systematically in 1977 by Wilson (1982) using 0.25 m^2 quadrats laid out in a grid at 250 m intervals. Using these results, a plot of the distribution of Cockles in South Dublin Bay shows that the highest densities occurred in a north-south band between Sandymount and Merrion in the mid-tide region of the intertidal sandflats. This coincides with the primary foraging area used by Oystercatchers that were colour-ringed and radio-tracked at the main high tide roost at Merrion in February and March 2014 (Tierney *et al.* 2017).

Higher estimates of wader numbers at low tide compared with high tide have also been noted in other studies (Burton *et al*. 2004, Lewis *et al.* 2016) and may reflect the difficulty of separating individuals in tightly-packed roosting flocks leading to underestimation of total numbers. A positive correlation between the counts of Oystercatchers in South Dublin Bay at high and low tide periods on the same dates in this study suggests that the birds may roost closest to the foraging areas to conserve energy. This is to some extent supported by an assessment of the movements of individual Oystercatchers through colour-ringing and radio-tracking undertaken as part of the Dublin Bay Birds Project (Tierney *et al.* 2017). A consistency in the use of subsites in Dublin Bay by all waterbirds at low and high tide periods was reported by Lewis *et al.* (2016), implying a degree of subsite fidelity and that some of the birds may forage preferentially close to their high tide roost sites. A positive correlation between the numbers of Oystercatchers in high tide roosts and the size of available tidal flats within a 1.5 km radius was described by Swennen (1984). Dias *et al.* (2010), studying use of space by Dunlin in the Tagus estuary (Portugal), reported that the density of these waders on foraging areas declined significantly with distance to the nearest roost, with fewer than 20% of individuals foraging more than 5 km from two roosts. They concluded that lack of suitably located high tide roosts can limit the total number of waders present regardless of the quality of foraging areas. These studies confirm that some wader species, at least, prefer to roost as close as possible to their foraging areas.

Plate 254. Roosting Knot at Sandymount Strand (John Coveney).

Effects of tide height and disturbance

In the present study, most species were found to move to other roosts within South Dublin Bay, and possibly elsewhere, on high spring tides or during sustained disturbance at the preferred roost sites, while Bar-tailed Godwit and Knot often left the bay entirely in these conditions. This is consistent with the findings of Rehfisch *et al.* (2003) that over 97% of within-year inter-roost movements of adult Oystercatcher, Ringed Plover *Charadrius hiaticula*, Purple Sandpiper *Calidris maritima,* Dunlin, Curlew, Redshank and Turnstone *Arenaria interpres* were within a single section of the Moray Firth, while 63% of adult Knot and 13% of Bar-tailed Godwit inter-roost movements involved changes of section. In the Tagus Estuary, tidal flat roosts became unavailable around spring tides and waders were forced to roost in saltpans where predation pressure was higher (Rosa *et al.* 2006).

This study shows that, on high spring tides, most of the birds were forced to leave some of the most frequently used roost sites and to move greater distances to other sites which were available during all tide phases. However, it is difficult to separate this factor from the effects of human disturbance. Roost site preferences of waders studied on the Forth estuary by Furness (1973) noted that spring tides can influence the use of roosts because the birds were forced to roost closer to human disturbance, the effects of which were thus intensified. By subjecting the flocks to experimental disturbance he found that some species responded differently and eventually moved to different, secondary roost sites. He showed that species such as Bar-tailed Godwit and Dunlin, which were the most sensitive to human disturbance, tended to remain in the preferred roost for longer than Oystercatcher and to fly further to their secondary roost sites when disturbed (Furness 1973). Kirby *et al.* (1993) found that Grey Plover *Pluvialis squatarola* Knot, Dunlin and Bar-tailed Godwit commonly left the Dee Estuary altogether when disturbed at high tide. The absence of Knot and Bar-tailed Godwit from South Dublin Bay on high spring tides in the present study suggests that the other roost sites here were unsuitable for them, or that there were other more preferable roost sites available elsewhere within Dublin Bay or further afield. Some of these birds may have remained in flight throughout the high tide period when disturbance levels were high. Such flights, often referred to as aerial roosts, commonly occur over water and may last for hours (Hale 1980, Dekker 1998, Hötker 2000, Conklin & Colwell 2007).

Plate 255. Knot and Bar-tailed Godwit roosting in South Dublin Bay (John Coveney).

In this study disturbance (at least one incident causing birds to take flight) was recorded in 1991 on the majority of survey days at the Poolbeg, Sandymount and Merrion roost sites. As these were the preferred roost sites in South Dublin Bay and the closest to the largest intertidal foraging area it is possible that these disturbance events may have had a negative impact on survival of the birds as the energy required in flying is high. The longer-term and indirect impacts of disturbance on shorebird survival are most likely influenced by the extent and timing of disturbance events, but this is very difficult to confirm. It has been shown that Knots incur significant energy costs associated with travel to and from their roost sites which tends to limit their use to just a few locations (van Gils *et al.* 2006, Rogers *et al.* 2007). A study of the effects of human disturbance on birds in part of South Dublin Bay during 2001 found that an average of 112 people and 37 dogs per hour used one area near the upper shore (about 38 ha in area) (Phalan & Nairn 2007). In another study in South Dublin Bay, where direct disturbance occurred to individual Oystercatchers, their prey capture rates, time spent searching for prey and the numbers of prey-finding activities all declined. Disturbance to individuals resulted in increased time spent running, flying, preening and remaining stationary or being vigilant (Fox 2014).

Sustained disturbance has a significant impact on foraging success, energetic costs, use of feeding and roosting sites and may ultimately result in population declines (Pfister *et al.* 1992, Townshend & O'Connor 1993, Rehfisch *et al.* 1996, Coleman *et al.* 2003, Béchet *et al.* 2004), especially where other factors are present (e.g. habitat loss from development and climate change). Several authors have found that availability of roosting habitat may limit local population size of waders (van Gils *et al.* 2006, Rogers *et al.* 2007 Conklin *et al.* 2008). This is supported by changes in population size when roosts are lost or created (Furness 1973, Burton *et al.* 1996).

As the entire area of South Dublin Bay is designated a Special Protection Area (SPA), some of the most important roost sites should be actively secured from disturbance, especially during the winter months when largest numbers of birds are present and when energetic requirements may be greater due to adverse weather conditions. There is a need for education and outreach to promote the importance of the bay for this group of birds among the wider public. The Natura 2000 Management Plan for the site should be made public, and monitoring of human activities in the SPA should be among the key priorities.

Acknowledgements

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Appendix 1.

Description of high tide roosts used by waders in South Dublin Bay in 1991.

1. Merrion: This is the main wader roost site in South Dublin Bay. It is located on a sandbank beside the railway embankment north of Booterstown Station. The outfall of the Trimblestown Stream separates the roost site from the main public access at Merrion Gates but people and dogs can approach quite closely. This site is approximately equidistant from the major foraging areas used by roosting waders. In 1991, the Merrion roost site was an unvegetated sandbank which was covered at high spring tides. By 2016 this had developed into a significant spit with extensive sand dune vegetation, but continues to be covered on occasions each winter during high spring tides.

2. Seapoint: This roost site is on the railway embankment north of Seapoint Station near the southern end of the study area. At high tide this roost site is virtually inaccessible to people except by way of private gateways onto the beach.

3. Sandymount: The higher sandbanks on Sandymount Strand are used by roosting waders on neap tides. On intermediate and neap tides these banks are connected to the shoreline and are frequently disturbed by people and dogs.

4. Booterstown: Booterstown Marsh and the railway embankment near Booterstown Station. This area is inaccessible at high tide.

5. Blackrock: Railway embankment between Booterstown and Blackrock. This area is inaccessible to people at high tide.

6. Poolbeg: Supratidal sandbanks and rock armoury at Poolbeg Peninsula at the northern end of the study area. The sandbanks are accessible at high tide.

Breeding of the Common Tern *Sterna hirundo* in Cork Harbour, 1983-2017

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Common Terns *Sterna hirundo* recolonised Cork Harbour in 1983 and they continue to breed there in 2017, thirty-five years later. Seven different sites have been used and all, apart from one, have been artificially constructed, usually within industrial complexes. The number of Apparently Occupied Nests has varied each year from 18 (2008) to 157 (2015), and although birds were present in 2001, none bred that year. Numbers increased from 1983 to 1991, remained stable to 1997 and then declined to 2008, following which there has been a steep increase each year to 2017. Productivity, although imperfectly monitored, has obviously varied across the years due to factors associated with the nature of the sites, such as human disturbance, tidal flooding and loss to avian and mammalian predators.

Introduction

Common Terns *Sterna hirundo* bred regularly in Cork Harbour during the period 1983 to 2017. Details of their breeding during 1983 to 2000 have already been published (Wilson *et al.* 2000). Despite setbacks over the years due to various factors, this colony has survived, and it is important as it is the only colony along the south Irish coast between Wexford and west Cork of a species that has an essentially westerly and northerly distribution in Ireland with a significant inland component to the population (Hannon *et al.* 1997, Ratcliffe 2004, Balmer *et al.* 2013). The Irish population has been surveyed a number of times since the late 1960s (Ratcliffe 2004), therefore, its population trend is reasonably well known. The Common Tern is Amber-listed on Birds of Conservation Concern in Ireland (Colhoun & Cummins 2013).

Methods

Surveys for breeding Common Terns have been carried out in Cork Harbour every year since 1983, since when they have been breeding regularly. At least one, usually two, and sometimes three visits are made to each breeding site, and potential new breeding sites are also periodically checked. Basic data are gathered on each visit, especially a direct count of the number of Apparently Occupied Nests (AONs) often supplemented with counts of the number of birds in the air on approaching the site (flush counts). As many chicks as possible are ringed each year, although it is not possible to make visits as frequently as would be necessary to ensure all are ringed. A small number of adult birds have been trapped

Plate 256. Common Tern (Brian Burke).

at the breeding sites, and some mist-netting of adults and fledged juveniles in pre-migration roost flocks has also been carried out in Cork Harbour away from the breeding sites.

Results and discussion

Data relating to breeding by Common Terns in Cork Harbour for the period 1983 to 2000 has already been published (Wilson *et al.* 2000) (Table 1). However, an amendment needs to be made to these data; in 1999 two pairs were observed on nests (outcome unknown, but probably failed) on a flat-topped mooring buoy at the Marino Point barges site, raising the total for that year to 70 AONs (Table 1).

Although Common Terns were present in Cork Harbour throughout the 2001 breeding season (e.g. 25 seen feeding in Lough Mahon on 4 July), there was no evidence that any were breeding. However, there remains the possibility that a few pairs did actually breed at a site (or sites) remote from the main tidal channels of the harbour, but this is considered highly unlikely. Considerable attention was focused on Cork Harbour during the summer of 2001 by one of us (PS) with regard to the welfare of a pod of Killer Whales *Orcinus orca* which were present within the harbour for about six weeks from 10 June (Ryan & Wilson 2003), and in the circumstances it is unlikely that breeding Common Terns would have been overlooked. The most likely scenario is that they failed, for

Table 1. Number of Apparently Occupied Nests (AONs) of Common Terns at seven sites in Cork Harbour, 1983-2017. Sites: MPB = Marino Point barges; MPM = Marino Point Martello tower; RCP = Raffeen Creek pool; BPC = Ballybricken Point causeway; BPJ = Ballybricken Point ADM jetty; POC = Ringaskiddy Port of Cork docks; LBE = Lough Beg.

Year	MPB	MPM	RCP	BPC	BPJ	POC	LBE	Total
1983	23	-	-	-	-	-	-	23
1984	27	-	-	-	-	-	-	27
1985	26	-	-	-	-	-	-	26
1986	50	-	-	-	-	-	-	50
1987	60	-	-	-	-	-	-	60
1988	55	-	-	-	-	-	-	55
1989	90	-	-	-	-	-	-	90
1990	84	-	-	-	-	-	-	84
1991	91	-	-	-	-	-	-	91
1992	53	-	-	-	-	-	-	53
1993	88	-	-	-	-	-	-	88
1994	36	-	-	-	-	-	-	36
1995	102	-	-	-	-	-	-	102
1996	63	-	-	-	-	-	-	63
1997	89	-	-	-	-	-	-	89
1998	81	-	-	-	-	-	-	81
1999	2	68	-	-	-	-	-	70
2000	-	58	-	-	-	-	-	58
2001	-	-	-	-	-	-	-	0
2002	-	4	-	19	-	-	-	23
2003	-	5	19	1	-	-	28	53
2004	-	2	7	-	-	-	44	53
2005	-	3	14	-	-	-	45	62
2006	-	3	22	-	-	-	21	46
2007	-	3	34	-	-	-	-	37
2008	-	8	10	-	-	-	-	18
2009	-	13	26	-	-	-	-	39
2010	-	14	12	-	-	8	-	34
2011	-	44	18	-	-	13	-	75
2012	-	42	-	-	-	47	-	89
2013	-	51	-	-	-	69	-	120
2014	-	42	1	-	-	74	-	117
2015	-	62	19	-	-	76	-	157
2016	-	25	40	-	-	40	4	109
2017	-	28	4	-	8	61	31	132

unknown reasons, at the Marino Point Martello tower site very early in the season before we began our annual monitoring, although they had bred successfully there for the two previous years (Table 1), and they did not re-nest elsewhere within the harbour.

During the period 1983 to 2017 the number of AONs varied from 18 in 2008 to 157 in 2015. Numbers increased from 1983 to 1991, remained stable to 1997 and then declined to 2008, following which there has been a steep increase each year to 2017 (Figure 1, Table 1). Seven different sites have been used and all, apart from one, have been artificially constructed, usually within industrial complexes (Appendix 1). The Cork Harbour population is an important one in conservation terms and in 1995 formed 40.3% of the Cork and 3.3% of the Irish population totals (Hannon *et al.* 1997).

Although it was not possible to monitor productivity accurately, it was evident that there were years when many nests were lost and when many young did not survive to fledging (Wilson *et al.* 2000, Appendix 1). Several causal factors could be identified, some of which are associated with the nature of the sites concerned, especially human disturbance and tidal flooding, while predation by avian and mammalian predators has been identified as significant at some sites. Failure due to tidal flooding was most severe at the Raffeen Creek pool site and we suspect that, following failure, some may then have re-nested at other sites, especially Ringaskiddy Port of Cork docks and Lough Beg.

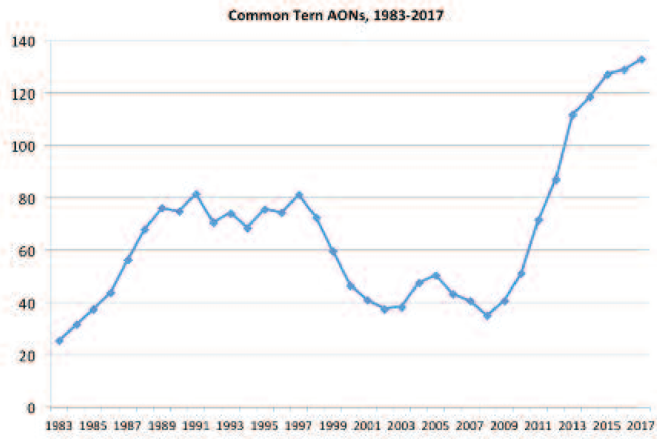

Figure 1. Five-year running mean of Apparently Occupied Nests (AONs) of Common Terns in Cork Harbour, 1983-2017.

Plate 257. Common Tern at Cork Harbour (Barry O'Mahony).

Plate 258. Common Tern (Brian Burke).

The following real or potential predatory birds have been seen at or near almost all of the breeding sites: Buzzard *Buteo buteo* (only in recent years), Sparrowhawk *Accipiter nisus,* Kestrel *Falco tinnunculus* and Peregrine Falcon *Falco peregrinus.* Among these, only the Sparrowhawk and Kestrel have been observed in predation events where tern chicks have been taken, in both cases at the Ringaskiddy Port of Cork docks site. Large gulls (Laridae) have also been present in small numbers at and near the breeding sites, but we have no direct evidence of predation on tern eggs or chicks. However, in 2017 two pairs of Great Black-backed Gulls *Larus marinus* bred at the Ballybricken Point ADM jetty site and one of the gull chicks was seen to accidently bump against a tern chick and push it off the edge of the jetty and into the sea. Another pair of Great Black-backed Gulls built a nest at the Ringaskiddy Port of Cork docks site in 2017; but the nest was removed with the permission of National Parks and Wildlife Service. Grey Herons *Ardea cinerea* have been diurnal and nocturnal predators at the Ringaskiddy Port of Cork docks site since 2014 and have been observed to take unfledged tern chicks as well as taking fledged chicks in the air. Hooded Crows *Corvus cornix,* on the other hand, while often present in the vicinity of tern nesting sites, have not been recorded as direct predators on either tern eggs or chicks although at the Lough Beg site they have been observed taking abandoned tern eggs.

Evidence of mammalian predators at and near tern nesting sites has frequently been observed. A Fox *Vulpes vulpes* seen within the Ringaskiddy Port of Cork docks site, having got through the security fence, is believed to have taken tern eggs. At the Lough Beg site many dead and eaten tern chicks were observed in 2005 and 2017. Brown Rat *Rattus norvegicus* and Mink *Neovison vison* were considered possible predators, but predation by the Otter *Lutra lutra* must be considered equally likely as in 2017 considerable evidence of Otter activity (spraint and rolling sites) was observed at the site. The Otter has been recorded as a predator at some seabird colonies (Leonard & Preston 2013). Seabirds are vulnerable to many aspects of the fishing industry as well as to leisure fishing and general debris floating in the sea (Acampora *et al.* 2016). A single adult entangled in fishing line has been found dead at one of the breeding sites.

Totals of 1,835 nestlings and 23 adults have been ringed at the breeding sites in Cork Harbour, while another 45 fully grown birds (adults and juveniles) have been ringed at pre-migration roosts within the harbour. Four nestlings ringed in Cork Harbour have been recovered in winter on the west coast of Africa in Mauritania, Senegal and Togo. This is consistent with the results of ringing of nestlings elsewhere in Ireland and Britain (Norman 2002). Seventeen nestlings ringed in Cork Harbour have been recovered back at Cork Harbour in a subsequent year, and are known (6) or presumed (11) to have been breeding there. Three nestlings ringed in Cork Harbour have been recovered in Dublin colonies in subsequent years, while four nestlings ringed in Dublin colonies have been recovered in Cork Harbour in subsequent years. There have also been movements of single Cork Harbour nestlings to France and Merseyside (where they were sighted in five and three subsequent breeding seasons, respectively), and a nestling from Flintshire found dead at a Cork Harbour site was also likely to have been a breeding bird. A nestling ringed in Sweden and recovered in Cork Harbour on 19 August was probably on its southward migration.

Concluding remarks

Across its range Common Tern distribution and populations have been greatly influenced by predation, mainly by mammals and especially by the Mink and Fox. Such predators, unless controlled, can lead to severe declines or total abandonment of sites. Gulls can also cause declines or abandonment of sites, but if gull numbers are controlled in the vicinity of colonies then terns can increase again (Ratcliffe 2004). Common Terns take a broader range of fish prey than other tern species and this has resulted in their being less affected by food shortages than other tern species (Ratcliffe 2004). While many nestlings come back to the natal site to breed, many also settle down at colonies far distant from the natal one, although adults are much more site faithful (Norman 2002). Their ability to recruit into distant sites probably partly explains why the Cork Harbour colony has persisted and increased, despite apparent low productivity in some years. Populations can also be severely affected within their winter range in Africa where there is a tradition of killing birds for sport, for food and in order to collect rings. First-year birds are mostly affected by this practice and this has the potential to reduce recruitment in subsequent years (Ratcliffe 2004). Breeding Common Terns can be badly affected by human disturbance, but when protected they can nest successfully at artificially constructed sites in close association with humans, as has been shown here and at Dublin Port (Merne 2004) and elsewhere. Efforts are underway in Cork Harbour to provide a series of more permanent and protected sites for breeding Common Terns (Richard Nairn, personal communication) and works to protect the Raffeen Creek pool site from tidal flooding are also under consideration.

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Appendix 1. Sites occupied for breeding by Common Terns in Cork Harbour, 1983-2017.

Marino Point barges (MPB) (W772701): Steel barges moored off Marino Point in shallow estuarine habitat. These barges are no longer in existence (Wilson *et al.* 2000).
Conservation: Human disturbance caused by movement, and eventual complete removal, of barges.

Marino Point Martello tower (MPM) (W780705): One of a series of 19th century defensive towers built around Cork Harbour (Wilson *et al.* 2000).
Conservation: Occasional human disturbance with chicks prone to falling or being blown off the high sloping wall.

Raffeen Creek pool (RCP) (W758647): Small impounded brackish inlet of Cork Harbour with vegetated artificial island surrounded by golf course.
Conservation: Flooding by spring tides.

Ballybricken Point causeway (BPC) (W772652): Narrow man-made causeway in shallow estuarine habitat connected to mainland.
Conservation: Flooding by spring tides and disturbance by Cormorants *Phalacrocorax carbo,* Grey Herons and large gulls roosting at the site.

Ballybricken Point ADM jetty (BPJ) (W775652): Industrial jetty (currently disused) in shallow estuarine habitat.
Conservation: Disturbance by large gulls.

Ringaskiddy Port of Cork docks (POC) (W776643): Series of mooring dolphins within working industrial docks and passenger ferry port.
Conservation: Human disturbance and predation by Grey Herons.

Lough Beg (LBE) (W781630): Vegetated low limestone reef in shallow intertidal estuarine habitat; the only truly natural habitat of the seven sites used by Common Terns in Cork Harbour.
Conservation: Predation by mammalian predators, possibly involving Brown Rat and Mink, but also probably involving Otter.

An overview of Rare Breeding Birds in Ireland in 2017

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Keywords: Breeding birds, rare birds, estimates, numbers, trends

Introduction

The Irish Rare Breeding Birds Panel received a significant boost late in 2017 when funds were made available by the National Parks and Wildlife Service to cover costs of a part-time development position over a 12-month period. This has enabled the creation of a bespoke website (www.irbbp.org) and we urge readers to look at this and keep up to date with Panel news, including which species to report on.

We are working through options for creating a fit-for-purpose database for the archive of records collected since 2002 and an on-line portal for the submission of rare breeding bird observations and information into the future. The latter may include a mapping option, reducing the need for grid references and providing a better tool for site-definition but at the same time ensuring the security of sensitive site information. Additionally, we are seeking greater involvement by the birdwatching community in the working of the Panel. To this end, we intend running training and information workshops at various locations in early spring 2018. Initially, we will run one in the Limerick area for potential recorders in the south and west and a second in the Dundalk area for those in the North and east. If these are successful, it may be possible to run others later in the year.

While this work is in progress, we have decided against compiling a full 2017 Report for this issue of *Irish Birds* and in

Plate 259. Dotterel flock on a Kerry mountain summit in April 2017 (M.Connaughton).

this account, I summarise some of the key headlines and findings and will report fully on 2017 with a review of the previous five years of data in the 2018 issue of *Irish Birds*.

Freshwater wildfowl

Much of the data for this group comes from a couple of 'supersites': Loughs Neagh and Beg in the north and Tacumshin and Lady's Island Lakes in the southeast. Other large sites that perhaps could do with more regular monitoring include Lough Ree and some smaller, important sites such as Lough Ourna (Tipperary), Lough Funshinagh (Roscommon), Cahore (Wexford) and the Murrough and Broad Lough (Wicklow).

Lough Neagh remains the premier site for scarce breeding ducks and two species in particular stand out: Gadwall *Anas strepera* and Pochard *Aythya ferina*. A minimum of 12 female Gadwall were seen with broods totalling 38 young (mean brood size 3.2) and for Pochard, a minimum of 12 successful females with 31 young (mean brood size 2.6). There is also an intriguing report of two female Scaup *Aythya marila* each with broods of four on 10 June. They were reported together with Tufted Duck *Aythya fuligula* and Mallard *Anas platyrhynchos* broods, but unfortunately nobody else could relocate and confirm the identification of the ducks. We need some regular monitoring of this area in 2018. A pair of Whooper Swan *Cygnus cygnus* (with no young) were present at Lough Neagh and one adult over-summered on Inishbofin (Galway).

Gadwall also bred at Lady's Island Lake with 24 pairs reported and at Tacumshin where two females and an excess of male suitors were seen. A female Shoveler *Anas clypeata* was also flushed from a probable nest at Lady's Island Lake and a male was noted 'in suitable habitat' at nearby Tacumshin. It was a poor year for Garganey *Anas querquedula* with only single males and females reported from south Wexford on different dates in April and with no evidence of breeding. Four male Common Scoters *Melanitta nigra* were present on Lough Corrib, but no females or young were reported.

Upland habitats

Breeding Dotterel *Charadrius morinellus* numbers have declined significantly in Scotland and southbound autumn migrants are often enjoyed by Irish birdwatchers on mountain tops in September. The breeding range in Scotland is shrinking eastwards in the Highlands so observations at western Irish mountains in spring 2017 were intriguing. Initially, five were seen near the summit of a Kerry mountain on 23 April and then almost a month later one was recorded calling for two hours on 20 May almost at the same location. Further north, seven were reported from the summit of a Sligo mountain on 7 May and another two in the Wicklow Mountains on 8 May. These birds are presumed migrants, but the Kerry records indicate follow up surveys should be made in 2018.

Our other mountain top speciality is the Ring Ouzel *Turdus torquatus*; only one report has been received so far, a male was seen and heard calling at 500 m in the MacGillycuddy Reeks (Kerry) on 27 May in what we think is one of the remaining key mountain ranges for this species.

Waders and gulls

There was a single record of potential breeding in Ireland for Black-tailed Godwit *Limosa limosa*, with a 'recently' fledged juvenile accompanied by a pair of adults seen at the Quoile Pondage (Down) on 7 July. Dunlin *Calidris alpina* bred successfully at two coastal and machair sites (Mayo) with up to six pairs present. About 25 Red-necked Phalarope *Phalaropus lobatus* were noted at recently used western locations, including two incubating males and four copulating 'pairs', and a pair was seen at a new site. However young were not recorded at any site, with several wetlands drying out during July. The Mediterannean Gull *Larus melanocephalus* colony at Lady's Island Lake (Wexford) held 56 pairs in 2017 and a pair bred successfully at a new site on one of the 'great' western lakes, raising two young.

Great Skua

Great Skuas *Stercorarius skua* are well covered in County Mayo and the population continues to increase: overall 11 pairs reported so far with many raising at least one chick and a pair at a new site where two young were raised. Systematic coverage is not achieved in Donegal and Sligo, but to date one pair has been reported.

Woodland habitats

The Nightjar *Caprimulgus europaeus* is perhaps the least well searched for rare woodland breeder, and single observations from two sites, Oranmore (Galway) on 30 April and Gooig Bog (Limerick) on 29 July were of note, but neither is particularly close to the forested southern upland areas (e.g. Tipperary and Cork), where breeding is suspected. A single pair of Wood Warblers *Phylloscopus sibilatrix* was reported from Wickow and they successfully raised young. Neither Common Redstart *Phoenicurus phoenicurus* nor Pied Flycatcher *Ficedula hypoleuca* was seen in Wicklow woodlands in 2017. Numbers of occupied territories of nesting Great Spotted Woodpeckers *Dendrocopos major* were slightly down in County Wicklow, but more worrying is the frequency of nest predation events by Pine Martens *Martes martes*. However, the population has now spread into much of County Wexford with pairs reported in Kilmuckridge and Johnstown Castle. Finally, although not a strict woodland bird, a Garden Warbler *Sylvia borin* was heard signing near Cahir (Tipperary) on 15 May.

Farmland and lowland marsh and reedbed habitats

One species rarely reported nowadays is the Quail *Coturnix coturnix.* A single calling male heard at a site in Tipperary at 03:30 in the morning of 12 June, while the observers were looking for Barn Owls *Tyto alba,* was the best proof of breeding though a single bird was seen on Great Saltee (Wexford) on 28 May and a late calling bird was recorded near Tacumshin on 5 August. Bearded Reedlings *Panurus biarmicus* have become regular breeders in southeastern counties in recent years. Maxima of 15 were present at Cahore (breeding confirmed), 12 at Tacumshin, five in the Lady's Island Lake area (all Wexford) while in Wicklow five were reported near Newcastle. The Reed Warbler *Acrocephalus scirpaceus* continues to increase as a breeding species in the southeast: at least eight territory-holding males were in south Wexford, and certainly one more was at Cahore in north Wexford, with three at the East Coast Nature Reserve in Wicklow (one seen feeding young). The range of the species could have extended as far as County Leitrim as a bird was in song on Upper Lough MacNean on 28 August. A singing male Marsh Warbler *Acrocephalus palustris* in north Wexford on 1 July was a good find, although it is unknown if this bird managed to find a mate.

Non-native species

We are interested in receiving more reports of non-native and introduced species apart from Common Pheasant *Phasianus colchicus* and Red-legged Partridge *Alectoris rufa* which are widely released for shooting. Black Swan *Cygnus atratus* is a regular presumed escapee from collections and a single bird resided at Nimmo's Pier (Galway) through the 2017 breeding season. A Tawny Owl *Strix aluco* has resided in County Louth for a couple of years and another, presumed a falconer's escapee, was reported in Killarney National Park (Kerry) in spring 2015.

Plate 260. Bearded Tit, Wexford (Eric Dempsey).

Plate 261. Nightjar (Andy Hay, RSPB-images).

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