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Reintroduction of White-tailed Eagles *Haliaeetus albicilla* to Ireland

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White-tailed Eagles *Haliaeetus albicilla* were extirpated as a breeding species in Ireland in the early 20th century following decades of population decline due to human persecution. Preparatory studies, including population modelling, site selection and identification of a donor population, resulted in the initiation of a reintroduction programme for the species in the Republic of Ireland. Between 2007 and 2011 one hundred young White-tailed Eagles (51 males and 49 females) were collected from nests in Norway under licence and transported to Ireland for release in Killarney National Park, Co. Kerry. Birds were held for 6-10 weeks before release. Wing-tags and radio and/or GPS satellite transmitters were attached to birds for individual identification and tracking post-release. Birds tended to remain in the Killarney area for the first few months after release, moving away in late winter but remaining in south Kerry. Most birds dispersed in spring, tending to return towards the 'natal' area in autumn. First pairing occurred in 2010 when birds were still sub-adult. First nesting took place in 2012 with chicks fledged successfully in 2013. The number of territorial pairs increased rapidly, but declined after 2014 with the loss of some adult birds. However, the number of breeding pairs and the number of young fledged continues to increase, with 14 chicks fledged to date. Comparisons with the first phase of the Scottish west coast reintroduction suggest that the outlook for the Irish population is reasonably optimistic. Illegal poisoning (64% of known mortalities) has had a serious impact on population growth and continues to threaten the viability of the reintroduction programme.

Plate 147. First Irish-bred White-tailed Eagle chick in over 100 years in nest, Mountshannon, Co. Clare, June 2013 (Allan Mee).

Introduction

The White-tailed Eagle *Haliaeetus albicilla* (along with the Golden Eagle *Aquila chrysaetos*) was once widespread throughout the island of Ireland, with an estimated 800-1,400 pairs in Britain and Ireland some 1,500 years ago (Evans *et al.* 2012). The evidence suggests the White-tailed Eagle may have been more widespread and abundant than the Golden Eagle in Ireland, perhaps related to its more diverse diet (Watson *et al.* 1992), especially its ability to catch fish, and to its largely lowland aquatic habitat requirements for breeding (Radović & Mikuska 2009, Evans *et al.* 2010). It is likely that White-tailed Eagle populations began to contract and recede towards the coastal fringe as human populations impacted on the landscape, especially with large-scale forest clearance (Everett 2014, Tierney 1998) and the consequent destruction of former tree nest sites.

However, human persecution and especially the advent of the breech loading shotgun, and more latterly the use of poisons to eliminate predators would have had a more drastic and ultimately catastrophic impact (Thompson 1849, D'Arcy 1999, Lysaght 2004). During this period of intense predator elimination Ireland lost several of its large apex predators including Golden Eagle, Red Kite *Milvus milvus,* Osprey *Pandion haliaetus* and Goshawk *Accipiter gentilis* (D'Arcy 1999). Marsh Harrier *Circus aeruginosus* also declined to extinction during this period although habitat loss, especially drainage of the large bogs, may have had at least as great an effect. Common Buzzard *Buteo buteo* was also eliminated as a breeding species in the late 19th century, but began to slowly recolonise the north-east of the island from the 1930s onwards (Hutchinson 1989).

The first hard evidence of the population size, range and trends of White-tailed Eagles in Ireland came with accounts written by various Victorian naturalists, most notably William Thompson (Thompson 1849). Weld (1807) goes into some detail on eagles in the Killarney area including the 'Eagle's nest', an historical nest site, and refers to two 'types' of eagle, "of a very dark brown and of an ash colour", the latter almost certainly the White-tailed Eagle. This species was said to be "commonly seen on the small islands of the lower lake" where they "exhibit all the appearance of tameness and familiarity". He concludes by stating that "notwithstanding the eager endeavours of the people of the country to destroy them... the number of these birds is supposed to be increasing in Kerry". One interesting account from 1824, apparently written by the administrator of a will for a priest in south Kerry, documented a "...very large eagle chained with iron... of the dark grey species" in the yard of a Killarney inn (Seán Mac a tSighigh personal communication).

White-tailed Eagles were clearly still resident in the remote parts of their range in the early-mid 19th century. Thompson (1849) states that of "thirteen or fourteen eagles killed at the Horn (Horn Head, Co. Donegal) within four years, all but one were the *Haliaeetus albicilla*" and "in winter the sea eagle is comparatively numerous" in the Dunfanaghy area, suggesting that several pairs were still resident. John Vandeleur Stewart (in Thompson 1849), a naturalist and bird collector living in Donegal, states that "the golden eagle is resident and rare (in Donegal) the sea eagle is resident and common". Thompson also observed two active White-tailed Eagle nests at Horn Head, one on eggs followed by a second with fully feathered chicks, "less than a furlong (200 m) distant". Such close proximity of eagle nests is highly unusual but not unknown today in high density eagle populations in Norway (T. Nygård personal communication) suggesting that food was at least seasonally superabundant as eagle pairs are normally highly territorial and aggressive towards other adults in the proximity of nests.

However, the species was clearly being heavily impacted upon, even in its last strongholds, through direct human persecution such as shooting, destruction of nests and eggs, collection of eggs and live chicks, and through the use of poison baits, in particular strychnine (Thompson 1849, Ussher & Warren 1900, D'Arcy 1999, Lysaght 2004). By the end of the 19th century White-tailed Eagles were all but extinct as a resident breeder in Ireland with the last pairs clinging on in remote parts of north-west Mayo and Kerry (Table 1). The last confirmed breeding appears to be on the north coast of Mayo in 1909 where this eagle had nested for several years near Portacloy, but Ussher was unable to prove breeding during a visit to the area in 1910 (Lysaght 2004). Occasional birds recorded during the 20th century were presumably continental vagrants (Hutchinson 1989).

Systematic reintroductions began in Scotland on the island of Rum (1975-1985), followed by a second release in Wester Ross (1993-1998), with the first successful breeding in 1985 (Love 1983, Bainbridge *et al.* 2003). A third release took place in East Scotland (2007-2012) concurrently with the Irish reintroduction. As of 2015 the Scottish population stood at 106 territorial pairs (Royal Society for the Protection of Birds (RSPB) unpublished data). Range expansion away from the core Scottish west coast populations on Skye, Mull and the adjacent mainland has been relatively slow, but has recently extended south as far as the Isle of Islay, 37 km north of the Irish coast. Despite this there has been very little movement of Scottish released or wild bred birds (one satellite tagged bird in Antrim over a period of days) into Ireland in the past ten years (D. Sexton RSPB, personal communication). Given this scenario, the likelihood of natural recolonisation of Ireland by the White-tailed Eagles within the next 25-50 years was considered low. Moreover, there are as yet few signs of the west coast population expanding into east Scotland or dispersing further south in the United Kingdom mainland into England and Wales.

Table 1. Chronology of selected White-tailed Eagle extinctions in Ireland. Year refers to last reported breeding or other significant event.

Year	Area	County	Notes
<1831	Mourne Mts.	Down	3-4 pairs reported from Mournes[1]
1832	Lough Bray	Wicklow	Eggs robbed, site subsequently deserted[2]
>1838	Lough Derg	Galway	3 birds (family party?) feeding on sick/dying fish 1835-38 in late June/July[1]
>1841	Connemara	Galway	'Common' throughout the district
1854	Comeragh Mts.	Waterford	Female 'collected' at nest[2]
<1866	Rathlin Island	Antrim	Pair killed on mainland
<1880	Blasket Islands	Kerry	Pair resident to 1870s and probably later[2]
1880	Horn Head	Donegal	3-4 pairs had bred along cliffs[1, 2]
1909	Portacloy	Mayo	Bred successfully for at least 7 years up to 1909[3]

[1]Thompson 1849, [2]Ussher & Warren 1900, [3]Lysaght 2004

Thus, a project steering group, consisting of the National Parks and Wildlife Service (NPWS) and the Golden Eagle Trust, decided to investigate the feasibility of a reintroduction of the species in Ireland. As part of this process an analysis based on the International Union for the Conservation of Nature (IUCN) criteria for reintroductions (IUCN/SSC 2013) was undertaken (O'Toole 2006), an Appropriate Assessment evaluated potential effects of the reintroduction on other Annex 1 species (Mee 2007), a suitable release area was identified and evaluated as part of a site selection process (Halley *et al.* 2006), and population modelling to investigate survival probabilities was undertaken (Fielding & Haworth 2007). Project collaborators, the Norsk Institutt for Naturforskning (NINA) and Norwegian Ornithological Society (NOS), identified a suitable source population in Norway. Killarney National Park, Co. Kerry was chosen as the release site due to its size (10,235 ha), location, historical significance as a former breeding site with potential to meet the ecological requirements of newly released eagles.

Methods

Young White-tailed Eagles were collected under licence (Directorate of Nature Management, Norway) from nests in west central Norway in the provinces of Sor-Trøndelag and Nord-Trøndelag west and north of the city of Trondheim. As part of the licencing requirements, eaglets were collected only from nests holding at least two chicks, thus leaving the breeding pair with at least one chick to rear. White-tailed Eagles lay between 1-3 eggs, although the latter is rare and clutches are almost invariably of two eggs (Willgohs 1961, Helander 1985). Likewise, most successful pairs rear only single chicks as mortality can be high in the first few weeks after hatching. On average, only one in five nests monitored in the project areas in Norway held two or more chicks. Thus, over 100 territories were monitored each breeding season to yield enough suitable sites for collection (Nygård *et al.* 2010).

Eaglets were 6-10 weeks old when they were collected from nests in mid-late June each year. Nests were primarily in trees, but also on cliffs or on the ground on islands free of ground predators. Young were sexed at the nest site using biometrics (Helander *et al.* 2007) to try and obtain as balanced a sex-ratio as possible within each release cohort. Eaglets were held for 2-6 days at a holding site near Trondheim airport and checked by a veterinarian prior to transport to Ireland. Birds were flown directly from Trondheim to Kerry airport by charter and immediately transferred to the release site in a remote part of Killarney National Park (henceforth Killarney) on arrival. Birds were housed in large outdoor aviaries with two to three birds per pen for 6-10 weeks before release using previously described methods (see Bainbridge *et al.* 2003).

Every effort was made to minimise human contact during the holding, transport and pre-release phases. Birds were fed a diet of fresh fish, venison (deer meat), Rabbits *Oryctolagus cuniculus* and crows (Corvidae) through a hatch at the rear of the cage but had no human contact apart from tagging, several days before release. Prior to release birds were ringed with a British Trust for Ornithology (BTO) metal ring and fitted with vinyl patagial tags to identify the release site, year of release and individuals (number/letter/symbol). Birds were also fitted with VHF radio (Biotrack Ltd. UK) or solar powered GPS satellite transmitters (Microwave Telemetry Inc. USA) to aid in post-release monitoring and the recovery of dead birds. White-tailed Eagles usually fledge at 10-12 weeks old in the wild (Hardey *et al.* 2013). Birds were released in Killarney at 12-17 weeks old depending on their development. Releases were staggered over a three-week period (4-29 August). This was partly due to variation in age and development but also to allow individuals to be tracked more easily in the first week post-release. In the first year of releases (2007) supplementary food was provided post-release and throughout the winter at 2-4 locations within the National Park. In subsequent years, supplementary food was provided until the end of September only.

Eagles were monitored and tracked post-release by visual observation, radio telemetry or remotely via the Argos satellite system (Collecte Localisation Satellites, Toulouse, France). Dead birds were, where possible, recovered and submitted to Regional Veterinary Laboratories (Department of Agriculture, Fisheries and the Marine) for post-mortem. Tissue samples were sent to the State Laboratory, Celbridge, Co. Kildare and, in earlier years, also to Science and Advice for Scottish Agriculture (SASA), Edinburgh, for toxicological analyses. As well as routine monitoring, much invaluable data on dispersing eagles came from public sightings of wing-tagged birds or through digital photographic images. These were reported directly to the project manager, entered in an online portal (www.goldeneagle.ie), or via other national or regional bird reporting platforms. All data were entered into a database which allowed individual birds to be tracked over their lifetime. Data were analysed using the SPSS statistical package (SPSS Inc.) and mapped using free software (Google Earth, GPS Visualizer).

Results

Release and dispersal

One hundred (100) White-tailed Eagles were released over a five-year period from 2007-2011 in Killarney: 15 in 2007, 20 in 2008 and 2009, 22 in 2010 and 23 in 2011. The sex-ratio of the release cohort varied somewhat from year to year but over the five-year period was close to 1: 1 (51 males, 49 females).

Most birds remained in the Killarney area in the first few months post-release. However, by mid-late winter many birds had moved away from Killarney but remained in south Kerry, primarily within the Iveragh peninsula (5-50 km from the release site). Some birds (n = 4) left the release area within the first few weeks of release. Two satellite tagged juveniles departed independently of each other ten days after release in 2009 having never visited a supplementary feeding site and were tracked to the Sligo and north Antrim coasts respectively (Figure 1).

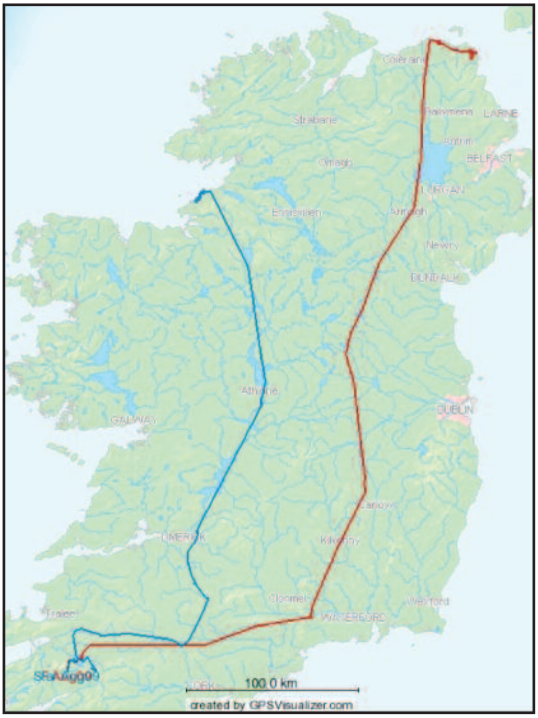

Figure 1. Dispersal of two satellite tagged juvenile White-tailed Eagles from Killarney in August 2009; male Star (blue track) to Sligo and female Fiadhna (red track) to north Antrim coast.

Most large-scale dispersal of released birds away from Killarney took place in spring with radio-tracking and satellite data showing that birds tend to return towards the 'natal' (i.e. release) area in their second and third calendar year (Mee *et al.* unpublished). Although most movement was within the island of Ireland, at least six released birds (four males and two females) dispersed to Scotland, including one male which undertook an eight-month exploratory trip in 2009 (Figure 2). The GPS satellite data showed this male left Killarney on 20 April, reaching Lough Gill, Co. Sligo, later that day (260 km). He crossed the Mull of Kintyre on 21 May, reaching Westray, Orkney Isles (910 km) on 28 May. He summered on the Kyle of Durness in north-west Sutherland before returning south in late September, reaching the Irish coast in early-mid November and finally Killarney on 23 December having travelled a minimum of 4,381 km (cumulative distance between all data points).

Territoriality and pairing

Birds began to settle on territories in their third or fourth year although some pairs began to form as early as their second (third calendar) year post-release. The first pair apparently formed in early 2010 on the west Kerry coast although the female moved out of the area for much of the summer. The pair had reformed by late summer and was highly mobile, covering a large 'territory' encompassing a 50 km stretch of coast, before settling at a site in late 2010. A second pair was located in spring 2011 at another coastal site in south-west Kerry. Over the next few years the number of territorial pairs increased but then declined after 2014 (Figure 3).

Figure 2. Dispersal of 2008-released satellite tagged male White-tailed Eagle from Killarney April to December 2009 to Orkney Isles, Scotland.

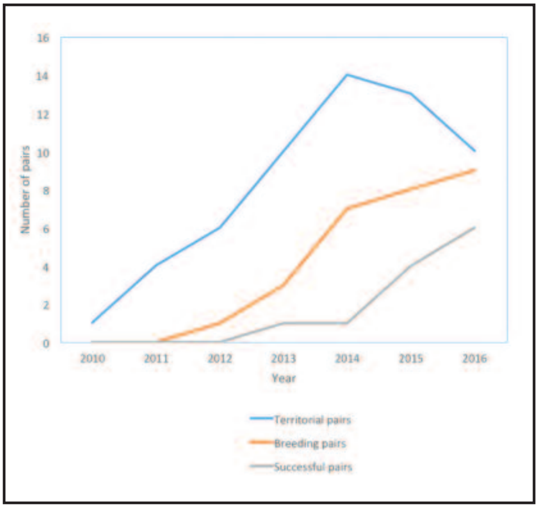

Figure 3. pairs of White-tailed Eagles in Ireland, 2010-2016.

Breeding and nest success

The first nesting attempt occurred on Lough Derg, Co. Clare, in 2012 but failed at the hatching stage. The same pair again nested at a new site in 2013 after the old nest tree fell during Storm Darwin, fledging two chicks, the first young White-tailed Eagles fledged in Ireland in over 100 years (Plate 147). The same pair re-nested in 2014-2016 at a new nest site nearby, being successful in 2014 and 2015 (Plate 148).

Plate 148. Recently fledged White-tailed Eagle near its nest at Lough Derg, Co. Clare, July 2014 (Arthur Ellis).

Fourteen chicks have fledged to date from 12 successful nesting attempts in counties Kerry, Cork, Clare and Galway (Table 2). All but two broods at fledging were of single chicks (mean = 1.2). A pair nested in Connemara, Co. Galway in 2014 but failed to hatch chicks. A nesting attempt by the same pair in 2015 failed when the female was found dead on the nest on the point of laying, having been poisoned (Plate 149). Nesting attempts in Kerry in 2015 and 2016 by 'pairs' included two trios: two males and a female and two females and a male. Both trios laid eggs in both years but failed to hatch chicks. Although not part of the Irish breeding population, an Irish released female nested in Argyll, Scotland in 2015 and 2016, successfully fledging a single chick in 2016 (RSPB unpublished data).

Only 11 out of 28 nesting attempts to date have been successful (Table 2), most failures occurring during incubation (n = 14), most of these being at the hatching stage (n = 8), all of which were to first time breeders and likely resulted from chick death at the point of hatching. Other causes of nest

Plate 149. Adult female White-tailed Eagle dead on the nest, Connemara, Co. Galway, April 2015; subsequent toxicology analysis revealed the bird had been poisoned (Dermot Breen, NPWS).

failure were chick death after hatching (n = 3), including two chicks that died at or near fledging, and trios laying, but abandoning, nests during incubation (n = 4). One nest failed due to human disturbance during incubation in 2014 and another may have also failed due to disturbance in 2016, both pairs incubating 2-3 weeks after the due hatch date.

Age of first breeding

Age of first breeding is often seen as a good indicator of the quality of the habitat available to nesting pairs, although this may also be related to the size of the release population, survivorship and the likelihood of finding a mate having

Table 2. Productivity and nest success of White-tailed Eagles in Ireland, 2012-2016.

Year	No. pairs breeding	No. pairs successful	No. young fledged	No. fledged/ breeding pair	No. fledged/ successful pair
2012	1	0	0	-	-
2013	3	1	2	0.66	2.0
2014	7	1	1	0.14	1.0
2015	8	4	4	0.50	1.0
2016	9	6	7	0.78	1.2
Total	**28**	**12**	**14**	**0.50**	**1.2**

dispersed away from the natal or release area. Age of first breeding in the current Irish population ranged from 4-7 years for males and 3-7 years for females and compares well with the Scottish released and wild-bred birds (Table 3).

Mortality factors

The first eagle mortality of the project was recorded in November 2007, only three months after release, a female recovered within Killarney National Park having died due to poisoning. Overall, poisoning accounted for 42% of mortalities of birds recovered dead (Table 4). However, poisoning was also suspected in as many as 10 of 11 birds where the cause of death was undetermined. One bird died due to haemorrhaging of the liver, an indicator of poisoning possibly due to high levels of rodenticides. Of birds where the cause of death was determined following post-mortem and/or toxicology analysis (n = 22), poisoning accounted for 64% of mortalities. Most confirmed poisoning incidents took place in the first years of the reintroduction but have continued to persist in most years to date (Figure 4). All but two confirmed poisonings occurred in late winter and spring (December to May) with suspected poisonings following a similar pattern (Figure 5). Poisons identified as the cause of White-tailed Eagle deaths in poisoning incidents included Carbofuran (n = 4), Nitroxynil (n = 7), Alphachloralose (n = 2) and Bromadiolone (n = 1).

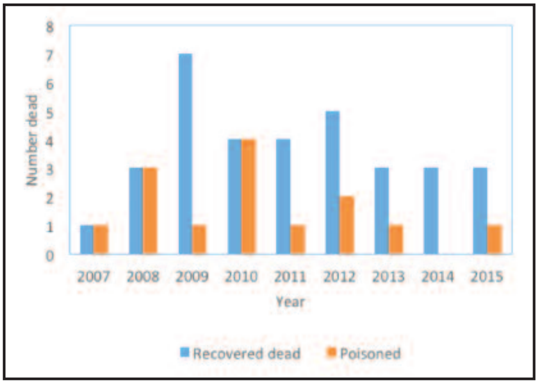

Figure 4. Number of White-tailed Eagles recovered dead and confirmed poisoned in Ireland (2007-2016). Note that no birds were confirmed poisoned in 2014 and no birds were recovered dead in 2016.

Three birds were killed when struck by turbine blades at windfarms, two in 2011 and one in 2012. Birds killed in 2011 were an adult and sub-adult female, while the 2012 mortality was a second calendar year female. All mortalities took place

Table 3. Age of first breeding in the Irish and Scottish White-tailed Eagle populations (Scottish data from Evans *et al.* 2009).

Population	**Mean age of first breeding ± SE (n)**		
	Males	**Females**	**All**
Ireland	4.9 ± 0.9 (12)	5.2 ± 0.9 (12)	5.0 ± 0.9 (24)
Scotland, Phase 1	5.2 ± 0.7 (5)	6.4 ± 1.2 (7)	5.9 ± 0.7 (12)
Scotland, Phase 2	4.7 ± 0.5 (6)	5.1 ± 0.3 (10)	4.9 ± 0.3 (16)
Scotland, wild bred	4.8 ± 0.3 (37)	5.4 ± 0.3 (39)	5.1 ± 0.2 (76)

Table 4. Mortality factors for White-tailed Eagles in Ireland. Population includes 100 released birds (2007-2011) and 13 Irish-bred birds (2013-2016).

Mortality factor	**No. recovered**	**% birds recovered**	**% known mortalities**	**% population (n = 113)**
Poisoning	14	42.4	63.6	12
Shooting	2	6.1	9.1	2
Turbine strike	3	9.1	13.6	3
Power lines	1	3.0	4.5	1
Starvation	1	3.0	4.5	1
Other	1	3.0	4.5	1
Unknown	11	33.3		10
Total	**33**	**-**	**-**	**30**

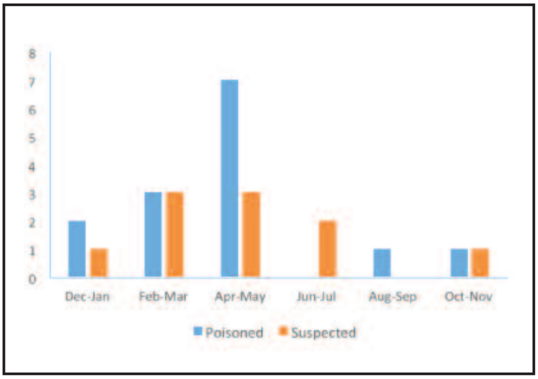

Figure 5. Timing of recovery of White-tailed Eagles confirmed (n = 14) and suspected (n = 10) poisoned in Ireland (2007-2016).

in spring and early summer. One adult male was killed in 2014, apparently in collision with a powerline, in west Kerry.

An additional mortality factor has been deliberate persecution through shooting (Table 4). A first-year female was reported dead by a party of kayakers on Lough Neagh, Northern Ireland, in October 2009. Although the carcass was not relocated following a search of the shore, the radio transmitter removed by the finders revealed the impact of shot gun pellets. In February 2014 one of the first two juveniles to fledge from a nest on Lough Derg, Co. Clare, in 2013 was discovered dead 18 km north-east of its natal site having been shot (Plate 150). In addition, a third year male found poisoned in Co. Mayo in 2012 had survived a previous shooting, its radiograph showing a number of shotgun pellets embedded in the body.

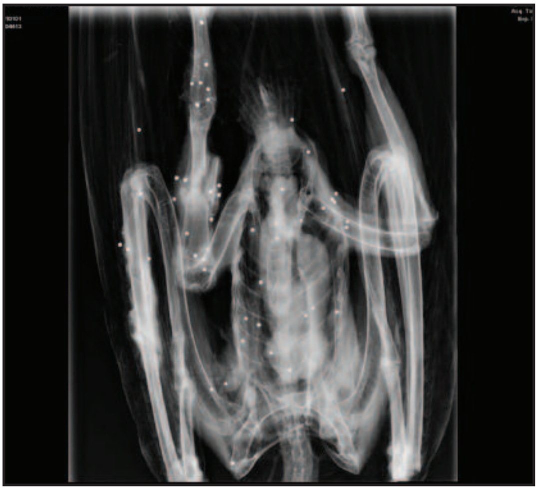

Plate 150. Radiograph of White-tailed Eagle shot near Ballinderry, Co. Tipperary, recovered in February 2014. Some 49 shotgun pellets are identifiable as white spots in the X-ray image. (Alan Johnson, Regional Veterinary Laboratory, Meelick).

Losses of breeding adults or members of potentially breeding pairs can have profound negative effects on the growth of a breeding population. Losses of paired adults in this population led to the break up or loss of at least six potentially breeding pairs. For example, loss of the male of the first territorial pair in early 2011 resulted in the female deserting the territory and dispersing from Kerry to Donegal some weeks later. The female returned to Kerry but did not pair again until 2014, lost her second mate to a powerline collision, re-paired a third time and finally nested in 2015, rearing her first chick successfully in 2016, some five years after losing her first mate.

Discussion

Establishing a viable, self-sustaining breeding population is the ultimate conservation goal of reintroduction projects (Sarrazin & Barbault 1996, Yu *et al.* 2015). Reintroduction has proven to be a valuable tool in conservation biology in restoring populations that have become regionally extinct, to supplement already existing but rare or threatened populations or species, to restore populations or species that have become globally extinct in the wild, and as a conservation measure to introduce a threatened species outside its historical range (Mee & Snyder 2007, Seddon *et al.* 2014). As human pressure on species and habitats increases through habitat loss, habitat degradation, direct or indirect persecution, contaminants, and perhaps most insidiously global climate change, reintroduction has increasingly become important in averting local, regional or global extinction (e.g. Armstrong & Ewen 2002, Jones & Merton 2012).

The demise of White-tailed and Golden Eagles, along with a major proportion of our large raptor avifauna, left Ireland with a hugely reduced guild of avian predators and scavengers, even more extreme than the similarly striking species extinctions in Britain, also largely as a result of human persecution (e.g. Brown 1979, Love 1983). Effects of large carnivore population extinctions include trophic cascade effects such as mesopredator release (see review in Ripple *et al.* 2014). Likewise, the loss of the largest component of avian scavengers (eagles, kites and buzzards), in Ireland in particular, is likely to have had profound effects on species diversity such as an increase in the abundance of other scavengers such as corvids. Thus, recovery of large avian scavenger populations through natural processes such as population expansion and dispersal (e.g. Common Buzzard) following relaxation of human persecution and through conservation actions, such as reintroduction, are likely to result in changes in other predator and scavenger populations over time. Further, long-term studies in Sweden have shown that apex predators, such as White-tailed Eagles, act as excellent biomonitors of harmful environmental pollutants, such as Dichlorodiphenyltrichloroethane (DDT) and Polychlorinated biphenyls (PCBs), in the marine ecosystem (Helander *et al.* 2008). Both pollutants were found to greatly reduce eagle productivity and nest success in the mid-20th century. Following a ban on DDT and PCBs in the early 1970s eagle productivity and nest success began to recover to former levels.

White-tailed Eagles bred in the wild in Ireland for the first time since their extinction in 2012 and successfully fledged the first young in 2013 six years after the first cohort were released. This compares favourably with the first phase of the Scottish reintroduction programme (1975-1985) from which the first young fledged ten years after the first releases (Figure 6). Likewise, the Irish population has shown promising signs of increasing the number of young fledged annually. However, to date mean brood size at fledging (1.2) and mean number of young fledged per breeding pair (0.46) is lower than that in the Scottish population (mean = 1.52 and 0.66 respectively; 1981-2000; Bainbridge *et al.* (2003)), or most other European populations (mean brood size = 1.2-1.8; Evans *et al.* (2009)). The high rate of nest failure during incubation in the Irish population to date (14 of 17 nest failures) is at least partly explained by parental inexperience with most of these failures occurring in first-time breeding pairs (Newton 1979). Nest success is positively related to parental age and experience in many raptor species (e.g. Ferrer & Bisson 2003, Penteriani *et al.* 2003). Brood reduction, often following poor weather in the first few weeks after hatching, may also be partly a function of parental inexperience.

The high number of nest failures at the point of hatching (n = 8), all to first-time breeders is particularly striking. Behavioural observations at a number of nests in Ireland suggest this is a critical point in the breeding effort with a notable increase in visits and time spent at the nest by both adults, occasionally resulting in apparent competition between the sexes for 'control' of the hatching egg (Golden Eagle Trust unpublished data). This could explain such nest failures if parental inexperience resulted in egg breakage or accidentally standing on a hatching chick. The fact that such failures did not occur in subsequent breeding attempts by the same pairs strongly suggest that parental experience and age is important, especially in long-lived species with long-term pair bonds (Ferrer & Bisson 2003, Penteriani *et al.* 2003). Thus, we would expect to see productivity and breeding success increase in the Irish population over time as more of the breeding population consists of older, experienced birds (Evans *et al.* 2009). However, the first Irish-bred White-tailed Eagles should also be recruited into the breeding population in the next few years as birds begin to reach maturity. Although some of these birds may also experience nest failure during initial breeding attempts, nest success in Scotland has been higher for pairs where one or both partners were wild-bred as opposed to released birds (Evans *et al.* 2009).

However, the attrition rate of losses to various mortality factors, especially illegal poisoning, has been highly significant (Table 4 & Figure 5). Moreover, it is highly likely that several undetermined deaths were also due to poisoning based on the location and the time of year birds were recovered, prior to and during the main lambing period in the Irish uplands.

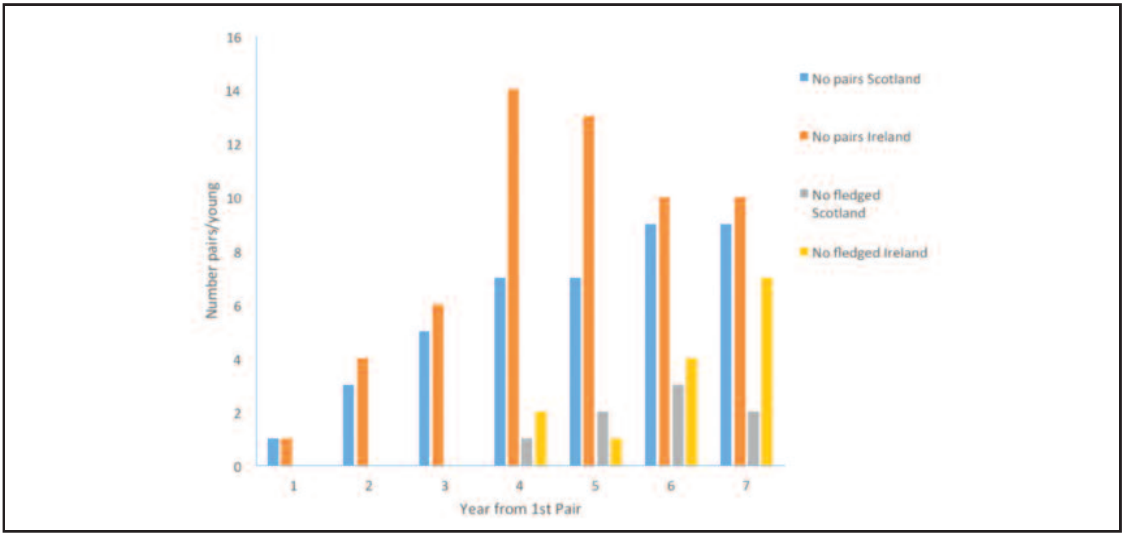

Figure 6. Growth of the Irish breeding population of White-tailed Eagles in comparison to west coast of Scotland First Phase release (1975-1985). Year 1 denotes the year the first territorial pair was formed (2010 in Ireland, 1982 in Scotland) Scottish data from Bainbridge *et al.* (2003). Number of pairs refers to territorial (breeding and non-breeding) pairs.

Poisoning has been a major cause of mortality in raptors and vultures worldwide resulting from the widespread use of anti-inflammatories in cattle (Oaks *et al.* 2004), lead poisoning from hunter shot game (Finkelstein *et al.* 2012, Mee & Snyder 2007), and the illegal use of pesticides to kill predators (Hernández & Margalida 2008, 2009). Poisoning accounted for almost half of all mortalities of White-tailed Eagles recovered dead in the reintroduction programme to date, but 64% of mortalities with known causes. This is one of the highest rates of loss to poisoning in any wild raptor population in Europe, compared to 31% in Spanish Imperial Eagles *Aquila adalberti* in Spain (González *et al.* 2007) and 38% in Bearded Vultures *Gypaetus barbatus* in the European Union (Margalida *et al.* 2008). The known losses of White-tailed Eagles to poisoning in Ireland were highest in the early years of the reintroduction (9 in 2007-2010), and birds dispersing away from the release area into the uplands in late winter-early spring were especially vulnerable. Immature White-tailed Eagles have a high dependency on carrion, and sheep mortality can be high during the late winter. Thus, the application of poisons, largely to control Foxes *Vulpes vulpes* and corvids, in contravention of regulations introduced in 2010 (Statutory Instrument No. 481 of 2010, Restrictions on the Use of Poison Bait Regulations) continues to pose a threat to scavenging birds and mammals. In one case, single poisoned carcasses may have resulted in the deaths of two and three eagles in 2008 and 2010 respectively. Indiscriminate and deliberate use of toxins, such as the carbamate pesticide Carbofuran, has resulted in mass poisonings of vultures in Africa (Ogada *et al.* 2011), and four eagle mortalities in this project to date. Other toxins which have had a significant impact on eagle mortality in Ireland include Nitroxynil (brand name Trodax), an antihelmintic drug used to control liver fluke in sheep and cattle. The misuse of veterinary medicines and their use on illegal poison baits in the countryside has serious implications, not just for wildlife but also for domestic animals and pets.

Following the poisoning of eagles in 2008 a working group was set up in Kerry involving farming organisations and other local stakeholders. Leaflets were issued to all sheep farmers in south Kerry and west Cork in 2010 and 2011 and distributed to marts and Agricultural Co-operatives in an effort to raise awareness regarding the effects of poisons on eagles, other wildlife and farm animals. Arising out of this a local initiative, the Kerry Sustainable Rural Environment Group, including Teagasc, the Irish Farmers Association, the Irish Creamery Milk Suppliers Association, and An Garda Síochána was set up to better promote awareness and examine opportunities to demonstrate legal predator deterrence and control at the farm level. While it is hard to determine if such measures have had a major impact on the use of poisons at a population level, both these measures and the high profile of several poisoning incidents in the national and local media undoubtedly heightened awareness of the risk the illegal use of poisons continue to pose (Burke *et al.* 2014). Further, a protocol for processing and reporting raptor mortalities nationally, including poisoning cases, has been established by NPWS since 2011 (see www.npws.ie/publications). EU Life project funding to tackle illegal poisoning of raptors such as Eastern Imperial Eagles *Aquila heliaca* in Hungary (LIFE15 NAT/HU/000902) and Bearded Vultures in Spain (LIFE04 NAT/ES/000056) has been instrumental in the recovery of these species. A similar dedicated research based initiative would be highly beneficial for a suite of raptor species in Ireland in countering the effects of poisons and other contaminants.

No evidence of mortality due to lead poisoning was detected in the programme, but the lethal and sub-lethal effects of lead poisoning has been a significant source of mortality for White-tailed Eagles elsewhere (Kurosawa 2000, Krone *et al.* 2006, Helander *et al.* 2009), as well as for other large scavenging eagles and vultures (Hall *et al.* 2007). However, most White-tailed Eagles in the Irish population are likely to carry at least background levels of lead resulting from consuming small amounts of lead pellets in hunter-shot carcass remains, especially in areas where a large number of deer are culled. The long-term implications of lead ingestion by White-tailed Eagles in Ireland, even at low levels, are unknown but it is the single most critical factor slowing the recovery of species such as the critically endangered California Condor *Gymnogyps caliafornianus* (Mee & Hall 2007, Finkelstein *et al.* 2012).

Electrocution is a major mortality factor for large eagles (e.g. Real *et al.* 2001), and has been an important mortality factor for recently released White-tailed Eagles in east Scotland, but has not been a known mortality factor to date in the Irish population. Electrocution was an important mortality factor for reintroduced California Condors in the 1990s mainly due to birds landing on power poles. Aversion training in pre-release pens, using mock power poles designed to give birds a mild shock on landing, was successful in largely eliminating this behaviour (Mee & Snyder 2007). Releases of White-tailed Eagles in Ireland took place in a heavily wooded area remote from power structures (utility poles, pylons). Although birds were never documented landing on power poles post-release one bird was reported landing on a utility pole at a windfarm site in Co. Antrim three months after release (B. Dunlop personal communication) and a wild-fledged juvenile, released following successful rehabilitation for a wing injury, consistently landed on dangerous power poles over several days risking electrocution. Thus, electrocution may become an important mortality factor for White-tailed Eagles and other large birds of prey (e.g. Common Buzzard) in Ireland as their populations recover and expand. Moreover, much of the

Plate 151. Adult female White-tailed Eagle, Co. Kerry, June 2014 (Jimmie Bannon).

existing power infrastructure in Ireland at present is inherently dangerous to large perching raptors. Careful planning in siting, utilising eagle friendly designs in new utility structures and retrofitting existing power lines has been shown to greatly reduce this risk (López-López *et al.* 2011, Chevallier *et al.* 2015).

Collisions, including turbine strikes and a powerline collision have been more significant. All three wind turbine strikes reported here were at two windfarms 19-25 km from the Killarney release site. Importantly, two of the mortalities included an adult and sub-adult female. White-tailed Eagles appear to be particularly vulnerable to collision risk showing little if any avoidance behaviour (A. Mee personal observation, Dahl *et al.* 2013). Turbine strikes have been the major mortality factor for White-tailed Eagles nesting on the island of Smøla, Norway (75 to date), where the largest windfarm in the country is sited in the densest population of breeding eagles in Norway (Dahl *et al.* 2012). Moreover, breeding success within the environs of the Smøla windfarm collapsed following windfarm construction and operation due to a combination of losses and desertion by adults, displacement and disturbance (Dahl *et al.* 2012). In contrast, the first reported White-tailed Eagle death at a windfarm in the United Kingdom was not until 2014. However, as yet few windfarms are sited near White-tailed Eagle breeding areas in Scotland (Bright *et al.* 2008). Moreover, post-construction monitoring, such as carcass searches for turbine fatalities, is carried out at relatively few existing windfarms in the United Kingdom and in Ireland. Disturbance displacement in and around windfarms during construction (Pearce-Higgins *et al.* 2012), and operational effects on some breeding bird species, including raptors, have resulted in significant negative population declines (Pearce-Higgins *et al.* 2009). As well as collision risk, White-tailed Eagles are vulnerable to disturbance displacement at or near nest sites. Future siting of windfarms >5 km from nest sites, communal roost sites and avoiding important flyways should be important considerations in future planning (Bright *et al.* 2008).

Bird species which have long reproductive lifespans but low reproductive rates are most affected by changes in the rate of adult mortality (Newton 1979). Thus, as has been seen, the loss of a pair member can have long-term effects with

Plate 152. Adult male White-tailed Eagle returning to nest, Co. Galway, May 2016 (Robert Foyle).

potentially breeding adults remaining unpaired, or remaining unpaired for some years before finding an alternative mate. Illegal poisoning has had the most profound effects on adult mortality and pair break-up in the putative Irish White-tailed Eagle breeding population.

The success of any reintroduction programme is contingent upon several factors such as the size of the release cohort, the quality of the release area, release strategies and post-release monitoring, and perhaps most critically, eradication, or a significant reduction, of the mortality factors that caused the species extinction (IUCN/SSC 2013). White-tailed Eagles are now breeding across four counties in Ireland and are showing signs of expanding their breeding range into the north-west and midlands. While the increase in breeding pairs and the number of chicks fledged to date is encouraging, the long-term viability of the reintroduction programme is contingent upon minimising losses of adult and sub-adult eagles in particular. Elimination of illegal poisoning as a threat to large scavengers, both avian and mammalian, remains a major goal for the species conservation in Ireland.

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The legacy of Christopher W. Bailey: his Belfast contribution to the Common Bird Census and Waterways Bird Survey (1964-1990)

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Keywords: Abundance patterns, Common Bird Census, population change, Waterways Bird Survey

Plate 153. Christopher W. Bailey (Courtesy of Ruth McNabb).

Between 1964 and 1980 Chris Bailey ran a Common Bird Census plot in Belfast and between 1974 and 1990 he ran two Waterways Bird Survey plots along a Belfast stretch of the River Lagan. The aim here is to show the changes in patterns of abundance of birds during the 17 years of each of these surveys. Whilst a few species increased in abundance, like Collared Dove *Streptopelia decaocto* and Mallard *Anas platyrhynchos,* some remained stable, like Great Tit *Parus major,* but far more species showed a decline, like Whitethroat *Sylvia communis* and Greenfinch *Chloris chloris.* The reasons for changing patterns are discussed, especially in the light of long-term monitoring throughout the United Kingdom by the British Trust for Ornithology's BirdTrends annual report. Long-term monitoring of birds, as achieved by Chris Bailey, is of significant importance in detecting changes in bird abundance throughout Ireland.

Introduction

In the 1960s, 1970s and 1980s Chris Bailey (Appendix 1) made an important contribution to knowledge of birds in the greater Belfast area through his participation in the British Trust for Ornithology's (BTO) Common Bird Census (CBC) and Waterways Bird Survey (WBS) programmes. Indeed, his contribution extended beyond Belfast: his survey of the River Lagan was the longest-running WBS plot in the United Kingdom (UK) which he ran from 1974 to 1990 (see Appendix 2 in Marchant *et al.* 1990). Most WBS sites were in England with smaller numbers in Wales, Scotland and Ireland (see Figure 2.3 in Marchant *et al.* 1990); hence the importance of his work on the River Lagan. He published a paper on his two River Lagan WBS sites for the first eight years (1974-1981) of his work there (Bailey 1982). Nationally, the WBS ran from 1974 to 2007 (http://www.bto.org/survey/complete/wbs.htm (accessed 5 March 2016)). The role of the WBS is now undertaken by the Waterways Breeding Bird Survey (WBBS) (http://www.bto.org/volunteer-surveys/wbbs (accessed 5 March 2016)). Combining both his WBS sites, Chris Bailey recorded 29 species, but only 18 of these were regarded as breeding birds of the River Lagan (Bailey 1982). He found that there were considerable variations in the breeding populations of Little Grebe *Tachybaptus ruficollis,* Moorhen *Gallinula chloropus,* Sedge Warbler *Acrocephalus* *schoenobaenus,* Grey Wagtail *Motacilla cinerea* and Reed Bunting *Emberiza schoeniclus* and that the main factors responsible for the fluctuations were climatic conditions, pollution and the arrival of American Mink *Neovison vison*, whilst habitat changes were not regarded as substantial, so were unlikely to have caused changes in bird numbers (Bailey 1982).

Although he never published anything on his contribution to the CBC, Chris Bailey ran a site on a college campus in Belfast from 1964 to 1980. This was an important national contribution to the CBC as most sites were concentrated in the south of England with a mere handful in Ireland (see Figure 2.1 in Marchant *et al.* 1990). Nationally, the CBC ran from 1962 to 2000 (http://www.bto.org/survey/complete/cbc.htm (accessed 5 March 2016)). Although most CBC plots were on either farmland or woodland, Chris Bailey's was one of the rarer 'special' plots being situated in a suburban setting (see Figure 1.2 in Marchant *et al.* 1990). The role of the CBC is now undertaken by the Breeding Bird Survey (BBS) (http://www.bto.org/volunteer-surveys/bbs (accessed 5 March 2016)).

The purpose of this paper is two-fold: firstly, to examine patterns of bird abundance as revealed by the CBC on the campus of Stranmillis University College in Belfast using Chris Bailey's data from 1964 to 1980; and secondly, to re-examine patterns of bird abundance as revealed by the WBS along the River Lagan by using Chris Bailey's extended data set from 1974 to 1990.

Part 1: Common Bird Census at Stranmillis University College, Belfast

Study site and methods

The campus of Stranmillis University College occupies about 20 ha and is situated in south Belfast (J3371) and lies about 2 km from the city centre. Today, the campus is surrounded mostly by housing and is dominated by buildings but also includes sports pitches, roads and car-parks, paths, lawns, parkland, mixed deciduous woodland, marshland and a little open water. However, when Chris Bailey began his CBC in 1964 there were slightly fewer buildings, but during the late 1960s and early 1970s there was a building programme on the campus (Beale & Phoenix 1998).

Chris Bailey used the BTO instructions for the CBC (http://www.bto.org/survey/complete/cbc.htm) to obtain his yearly lists for 1964-1980. Between eight and ten visits were made annually between March and July and sightings of birds drawn on visit maps. The BTO determined the number of territories of each species each year from these maps.

Today, more than 50 years after the CBC began the BTO publishes BirdTrends on-line which allows users to easily access the latest information on trends in population size, breeding performance, as well as survival rates as measured by monitoring schemes like BBS (Robinson *et al.* 2015). This information was unimaginable in Chris Bailey's time, but today provides a comparator for Chris Bailey's CBC Stranmillis data. Using the relative abundance of birds, together with their diversity, is useful in detecting long-term change in avian community structure. However, using, choosing and interpreting diversity measures can be fraught with difficulties (Magurran 1988). The fact that the Stranmillis CBC had small sample sizes (just the low hundreds each year) and that some species were abundant, whilst others were not, meant that the usual suite of measures was not available (Magurran 2004). For this reason the only measure of species richness that could be used for the Stranmillis CBC was the number of species each year, which has been used widely for similar studies (e.g. Crowe 2011).

It is now well established that both weather and climate affect many aspects of the lives of birds, including their abundance. For instance, over 50 years ago Perrins (1965) reported the influence of spring-time temperature on laying dates of Great Tits *Parus major* in Oxfordshire, whilst more recently Crick *et al.* (1997) have demonstrated that earlier laying-dates in many birds in the UK was caused by warmer springs as a result of global warming. Crick and Sparks (1998), examining UK breeding birds data, showed convincing relationships between laying dates and temperature and rainfall over the period 1939-1995. Similar phenological responses are being reported throughout Britain and Ireland (Sparks & Collinson 2006) and beyond (Sanz 2002). As climate changes, many birds are changing their ranges (Huntley *et al.* 2007); hence abundance of some populations might change. Weather can also affect the population levels of birds, as shown by Newton (1998). Because of weather effects, historical Meteorological Office data were analysed (www.metoffice.gov.co.uk/climate/uk/stationdata/) for Armagh, the nearest meteorological station available. Relationships between species abundance and species richness with the meteorological data (monthly mean maximum temperature; monthly mean minimum temperature; days of frost; monthly mean rainfall) were determined using least squares linear regression (Zar 2010). Regressions were calculated using Excel.

Results

Thirty-two species were recorded in the 17 years of the Stranmillis CBC. Eighteen of these species occurred with sufficient abundance that trends could be discerned and these are shown in a series of graphs. The abundance of Collared Dove *Streptopelia decaocto*, which bred for the first time in 1969, is shown in Figure 1. The abundance of Magpie *Pica*

Plate 154. Closed woodland at Stranmillis (Julian Greenwood).

pica and Hooded Crow *Corvus cornix* is shown in Figure 2. Both species have increased in abundance, although at the time of the Stranmillis CBC Hooded Crow was not separated as a species from Carrion Crow *Corvus corone* (Parkin & Knox 2010). There were four species, Blackbird *Turdus merula,* Blue Tit *Cyanistes caeruleus*, Robin *Erithacus rubecula* and Chaffinch *Fringilla coelebs,* whose numbers increased very slightly following an initial decline (Figure 3). These four species were the most abundant during the Stranmillis CBC. Great Tit, Willow Warbler *Phylloscopus trochilus* and Mistle Thrush *Turdus viscivorus* abundances, though fluctuating, showed stable numbers overall (Figure 4). Goldcrest *Regulus regulus*, Coal Tit *Periparus ater* and Wren *Troglodytes troglodytes* showed identical trends with stable numbers, but which rose during the early to mid-1970s (Figure 5). The abundances of five species declined; Chiffchaff *Phylloscopus collybita*, Song Thrush *Turdus philomelos,* Spotted Flycatcher *Muscicapa striata,* Dunnock *Prunella modularis* and Greenfinch *Chloris chloris* (Figure 6).

The remaining 14 species occurred either in low numbers or so infrequently that no trends could be discerned. Mallard *Anas platyrhynchos* (maximum one breeding female) bred in only four years, Tufted Duck *Aythya fuligula* (maximum one) bred twice, Sparrowhawk *Accipiter nisus* (maximum one) in eight years, Little Grebe (maximum one) bred twice, one pair of Moorhens bred every year except three, Jackdaw *Corvus monedula* (maximum two) bred in five years, Long-tailed Tit *Aegithalos caudatus* (maximum one) bred in one year, Blackcap *Sylvia atricapilla* (maximum two) bred in six years, Treecreeper *Certhia familiaris* (maximum two) bred in 12 years, Pied Wagtail *Motacilla alba* (maximum one) bred just once, Bullfinch *Pyrrhula pyrrhula* (maximum three) bred in

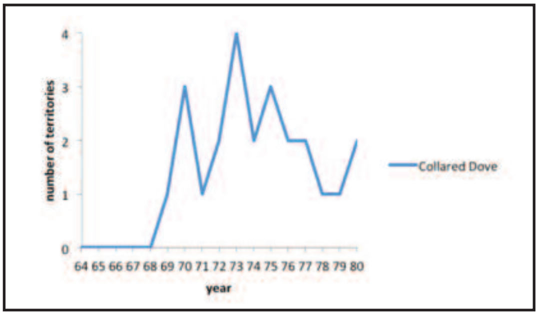

Figure 1. The abundance of Collared Dove at Stranmillis CBC; it became a breeding species only in 1969.

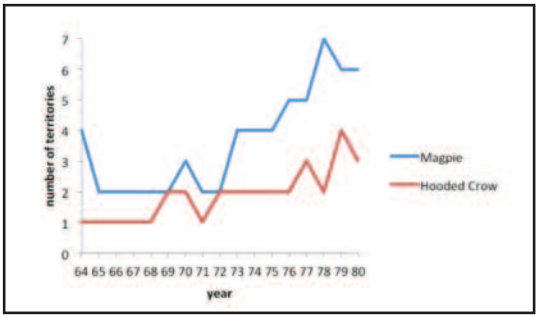

Figure 2. The abundance of Magpie and Hooded Crow at Stranmillis CBC; both species have increased in abundance.

Plate 155. Open woodland at Stranmillis (Julian Greenwood).

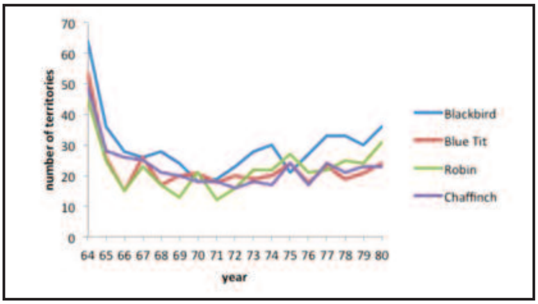

Figure 3. The abundance of Blackbird, Blue Tit, Robin and Chaffinch at Stranmillis CBC; the abundance of these species increased very slightly following an initial decline.

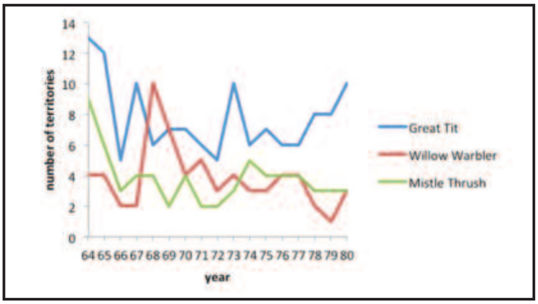

Figure 4. The abundance of Great Tit, Willow Warbler and Mistle Thrush at Stranmillis CBC; these species showed fluctuating abundances, but remained stable overall.

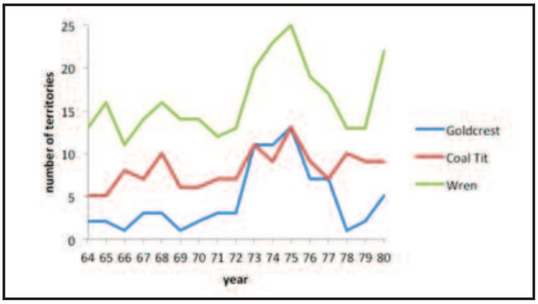

Figure 5. The abundance of Goldcrest, Coal Tit and Wren at Stranmillis CBC; these species showed stable abundances, except during the 1970s when numbers increased.

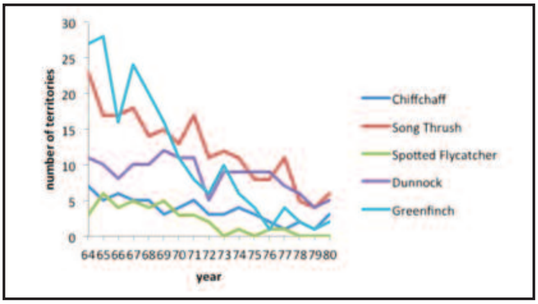

Figure 6. The abundance of Chiffchaff, Song Thrush, Spotted Flycatcher, Dunnock and Greenfinch at Stranmillis CBC; these species declined throughout the survey period.

Plate 156. Pond at Stranmillis (Julian Greenwood).

all except two years, Redpoll *Carduelis cabaret* (maximum three) bred in six years, Goldfinch *Carduelis carduelis* (maximum two) bred twice and Siskin *Carduelis spinus* (maximum one) bred just once. In addition, neither Woodpigeon *Columba palumbus* nor Starling *Sturnus vulgaris* are included as these were regarded as difficult species in which to assess territories.

Species richness (the number of species) recorded over the 17 years at the Stranmillis CBC is shown in Figure 7. It can be seen that richness increased slightly during the middle years of the study period. The change in species richness can be explained in terms of whether individual species were present or not; for instance Collared Dove was not present for the first five years, Spotted Flycatcher was present in only two of the last eight years, whilst Blackcap was only present in the middle years of the CBC. There were no significant relationships between species richness and any of the four meteorological factors. The only significant relationships were between some species and meteorological factors: Dunnock and March rainfall ($_{adj}R^2 = 0.400$, $F_{1,15} = 13.4$, $P = 0.002$); Magpie and April rainfall ($_{adj}R^2 = 0.334$, $F_{1,15} = 9.0$, $P = 0.009$); Wren and April rainfall ($_{adj}R^2 = 0.348$, $F_{1,15} = 9.5$, $P = 0.007$). In each of these three relationships, the regressions are negative, indicating that drier conditions lead to greater abundance.

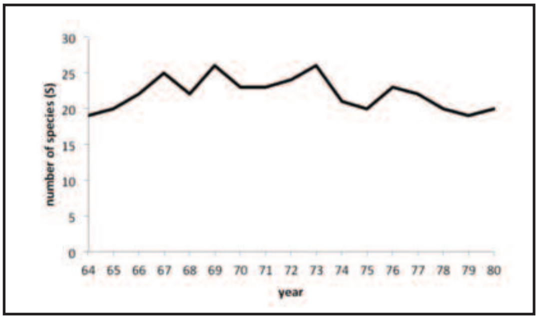

Figure 7. Species richness (S) during seventeen years at Stranmillis CBC.

Discussion

The Collared Dove was unique in the Stranmillis CBC in being a coloniser during the study period. It arrived in Britain in 1955, first bred in Ireland in 1959 and bred in Belfast for the first time in 1963 (Hutchinson 1989). The first breeding at Stranmillis in 1969 fits this pattern of colonisation. It has been suggested that the spread was because the species filled an empty niche and it had utilised garden food supplies (Robinson *et al.* 2015).

The Stranmillis CBC showed an increase in Magpies; they have undergone an expansion in range and abundance throughout Ireland in the 20th century (Hutchinson 1989) and throughout the UK from 1970 to the late 1980s (Robinson *et al.* 2015) and have adapted well to urban landscapes like Stranmillis (Gooch *et al.* 1991). Also of interest is the negative relationship between Magpie abundance and April rainfall which may be related to accessibility of invertebrate prey in the soil; Møller (1983) found that an increase in water level in

Plate 157. Kingfisher (Tom Ormond).

soils drove many invertebrates to the surface. The increase in Hooded Crows seen at Stranmillis mirrors that which has occurred throughout Ireland, birds having moved into urban areas to take advantage of food available by scavenging (Hutchinson 1989). Prior to the separation as a species of Hooded Crow from Carrion Crow, the latter showed a constant rise in abundance from the late 1960s to the present day (Robinson *et al.* 2015).

Strong declines were found during the late 1960s at the Stranmillis CBC for Blackbird, Blue Tit, Robin and Chaffinch, which for all four species bottomed out in the early 1970s before showing a very gradual increase. The UK pattern showed a Blackbird decline throughout the period of the Stranmillis CBC (Robinson *et al.* 2015). Siriwardena *et al.* (1998) suggested that the decline was due to reduced survival, and whilst population changes might best be correlated with adult survival, there seem to be marked differences between western and eastern Britain (Amar *et al.* 2006, Robinson *et al.* 2012). Adult survival, along with greater recruitment in woodland compared to farmland (Hatchwell *et al.* 1996), paint a complex picture for population changes in Blackbird which may account for the difference in abundance between the UK and Stranmillis from the early 1970s onwards. The difference between Blue Tit abundance at Stranmillis and across the UK (Robinson *et al.* 2015) is profound, with numbers increasing nationally but declining at Stranmillis in the mid-1960s after which numbers remained rather stable. The UK increases were greatest in Northern Ireland (Parkin & Knox 2010), although fledgling numbers at urban sites, like Stranmillis, was lower than elsewhere (Cowie & Hinsley 1987). The reason for the difference between Stranmillis and the UK might therefore be complex. Prior to the mid-1980s Robin numbers were low caused by a series of cold winters (Robinson *et al.* 2015); the relative stability during the Stranmillis CBC being the same as that throughout the UK. Parkin and Knox (2010) have stated that the smallest increase from the mid-1980s onwards was in Northern Ireland. The decline in Chaffinch at Stranmillis from the mid-1960s is similar to the UK pattern (Robinson *et al.* 2015) and was caused by the cold winters of the early 1960s (Parkin & Knox 2010). From the early 1970s numbers of Chaffinches in the UK increased (Robinson *et al.* 2015) whilst the Stranmillis numbers increased only marginally.

Although Great Tit, Willow Warbler and Mistle Thrush numbers fluctuated, there was, none the less, some stability. However, the Great Tit increased throughout the UK (Robinson *et al.* 2015) and whilst the presence of broadleaved woodland at Stranmillis would be considered ideal for them (Sweeney *et al.* 2011) the reason for their general stability there remains unknown. In Ireland, Willow Warblers have a preference for open woodland and scrub (Sweeney *et al*. 2011); this habitat is somewhat limited at Stranmillis, hence the small number of birds. The abundance of Willow Warblers throughout the UK declined at this time (Robinson *et al*. 2015). The Willow Warbler was one of many trans-Saharan migrants (in this case to the Ivory Coast and Ghana (Flegg 2004)) that experienced population declines (Ockendon *et al.* 2012, 2014). The small and fluctuating numbers at Stranmillis make comparison with the UK trends difficult. During the period of the Stranmillis CBC, Mistle Thrushes remained stable; it was subsequent to the late 1970s that UK numbers declined (Robinson *et al.* 2015) which Siriwardena

Plate 158. Chaffinch (Kevin Murphy).

et al. (1998) attributed to reduced annual survival. The small numbers of Mistle Thrushes at Stranmillis make comparison with UK trends difficult.

The increases of Goldcrest, Coal Tit and Wren at Stranmillis in the early to mid-1970s followed by declines, shows a remarkable parallel to the trends of the three species throughout the UK (Robinson *et al*. 2015). The increases throughout the late 1960s were attributable to recovery after the 1962/63 severe winter (Parkin & Knox 2010). Baillie *et al.* (2006) suggested for Goldcrest at least, that the subsequent decline might have stemmed from oscillations as the species recovered to previous levels of abundance. The nature of the habitat at Stranmillis is far from ideal for Goldcrest and Coal Tit; a survey of Irish forests showing the importance of mature closed stands of conifers for both species (Sweeney *et al.* 2011). However, the nature of habitats at Stranmillis, with the availability of open woodland containing lots of understory, bushes, as well as formal and informal planting, particularly suits Wrens (Sweeney *et al.* 2011). The relationship between lower April rainfall and increased Wren abundance is also of interest as Parkin and Knox (2010) mentioned that the cold wet spring of 1995 reduced Wren numbers in Britain and Ireland.

The five species that declined during the Stranmillis CBC also declined throughout the UK at that time. The decline of Chiffchaff mirrored that of other trans-Saharan migrants (Chiffchaffs winter between Mauritania and Guinea-Bissau (Flegg 2004)) at that time (Siriwardena *et al.* 1998). The decline of Song Thrush at Stranmillis and throughout the UK was substantial; Parkin and Knox (2010) saying that adverse weather alone was insufficient to explain all the changes and that other unidentified effects must have been involved. The decline in Spotted Flycatcher was catastrophic both at Stranmillis (from six pairs in 1965 to one or no pairs from 1973 onwards) and throughout the UK with an 88% decrease in numbers between 1970 and 2010, including a 21% range contraction in Ireland (Balmer *et al.* 2013). The Spotted Flycatcher is one of several long-distance migrants currently in steep population decline (Hewson & Noble 2009) that winter in the humid zone of West Africa (in particular between Camaroon and Angola (Flegg 2004)) which suggest that these declines have a common cause (Ockendon *et al.* 2012). Parkin and Knox (2010) suggested that the Irish decline had started by the 1980s, while the evidence from Stranmillis suggests that the decline had started even earlier. There was a substantial decline in Dunnock abundance across the UK from 1975 onwards (Robinson *et al.* 2015), which was also seen at Stranmillis at that time. Explanations concerning the decline of farmland and woodland Dunnock populations (Parkin & Knox 2010) do not adequately account for declines in urban areas like Stranmillis. However, of interest is the negative relationship between Dunnock abundance and March rainfall. Could it be that a drier March increases their invertebrate prey, reducing the 'hungry-gap' reported by Robinson *et al.* (2015), thus allowing more males to establish territories? The decline of the Greenfinch at Stranmillis was even more severe than throughout the UK at that time (Robinson *et al.* 2015); why that should have been so is unknown.

The rise in species richness in the late 1960s and early 1970s coincided with the building programme on the Stranmillis campus (Beale & Phoenix 1998). Common sense might draw the conclusion that building disturbance should have had a deleterious effect upon species richness. This was not the case at Stranmillis and many of the birds encountered might be, as Goode (2014) describes them, 'urban colonisers' able to withstand close contact with people.

Conclusion

One species, Collared Dove, colonised during the years of the Stranmillis CBC and two others, Magpie and Hooded Crow, increased in abundance. Five species, Chiffchaff, Song Thrush, Spotted Flycatcher, Dunnock and Greenfinch showed great declines, whilst the remaining ten species showed fluctuating fortunes. Perhaps it was the reductions in abundance of these five species, particularly the Spotted Flycatcher, which caused a reduction in species richness in the latter half of the Stranmillis CBC. Dry springs enhanced numbers of Wren, Dunnock and Magpie whilst the other three meteorological factors (frost, minimum and maximum temperatures) had no influence on abundance of other birds.

Plate 159. River Lagan above Shaw's Bridge (Julian Greenwood).

Part 2: Waterways Bird Survey on the River Lagan

Study site and methods

Chris Bailey's two WBS sites on the River Lagan ran from Drum Bridge (J305670) to Shaw's Bridge (J325691), a distance of about 5.6 km, and from Shaw's Bridge to Stranmillis Weir (J342704), a distance of about 4.3 km. For the present analysis both sections have been combined. A full description of the river was provided by Bailey (1982); suffice to say here that in the 10 km of the river surveyed it flowed by woodlands, parklands, golf-courses, a disused canal, sports fields, water meadows, one sewage works and very little housing, a situation that is remarkably similar today. Chris Bailey used the BTO instructions for the WBS (http://www.bto.org/survey/complete/wbs.htm) to obtain his yearly lists for 1974-1990. About nine visits were made annually between March and July and sightings of birds drawn on visit maps. The BTO determined the number of territories of each species each year from these maps. Like the Stranmillis CBC, a comparator has been BirdTrends (Robinson *et al*. 2015), along with species richness and meteorological data.

Results

Thirty-five species were recorded in the 17 years (1974-1990) of the River Lagan WBS of which 18 occurred with sufficient abundance that trends could be discerned which are shown in a series of graphs. The remaining 17 species occurred occasionally in small numbers and have not been considered further in the analysis of data. Of the 18 breeders only one, the Mallard, increased in abundance (Figure 8). Little Grebe and Moorhen showed very similar patterns of change; they increased in the late 1970s, declined through the 1980s before increasing slightly at the end of the 1980s (Figure 9). Many other species declined during the Lagan WBS; the declines of the two non-passerines, Tufted Duck and Kingfisher *Alcedo atthis,* are shown in Figure 10 whilst the declines of the passerines (Whitethroat *Sylvia communis,* Sedge Warbler, Grey Wagtail and Reed Bunting) are shown in Figure 11. The declines of Tufted Duck and Whitethroat led to their eventual loss as breeding birds in the early 1980s. The remaining nine species occurred infrequently or in low numbers so that trends could not be discerned. Mute Swan *Cygnus olor* (maximum three) bred every year, Coot *Fulica atra* (maximum two) bred in ten years, Lapwing *Vanellus vanellus* (maximum three) bred in six years, Snipe *Gallinago gallinago* (maximum one) bred in three years, Sand Martin *Riparia riparia* (maximum three) bred in seven years, Blackcap (maximum seven) bred only in 1977, Grasshopper Warbler *Locustella naevia* (maximum one) bred only in 1975, Dipper *Cinclus cinclus* (maximum two) bred in 15 years and Pied Wagtail (maximum three) bred every year.

The other 17 species that were seen without any evidence of breeding were all non-passerines: Mandarin *Aix galericulata,* Teal *Anas crecca,* Shoveller *Anas clypeata,*

Plate 160. River Lagan above Stranmillis weir (Julian Greenwood).

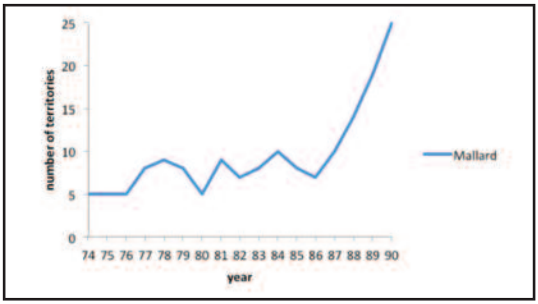

Figure 8. The abundance of Mallard at the River Lagan WBS; Mallard was the only species to increase during this survey.

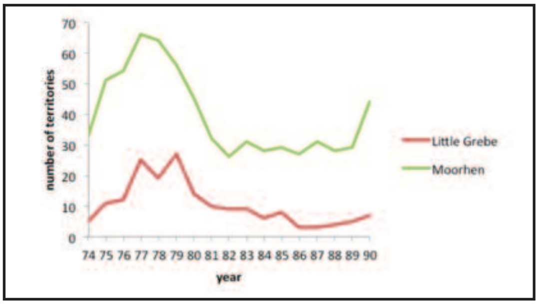

Figure 9. The abundance of Little Grebe and Moorhen at the River Lagan WBS; both species increased in the late 1970s before declining and increasing again in the late 1980s.

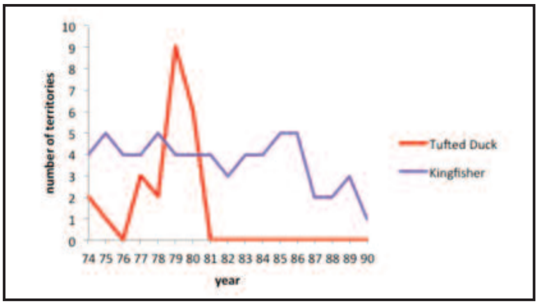

Figure 10. The abundance of Tufted Duck and Kingfisher at the River Lagan WBS; both species showed a strong decline.

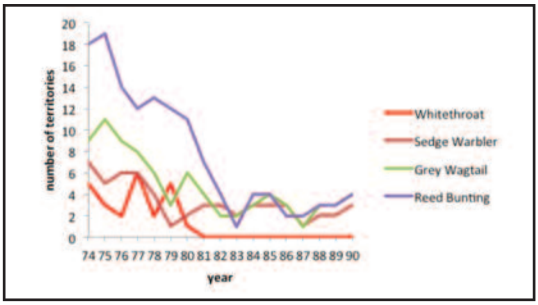

Figure 11. The abundance of Whitethroat, Sedge Warbler, Grey Wagtail and Reed Bunting at the River Lagan WBS; all species showed strong declines.

Plate 161. River Lagan at Drum Bridge (Julian Greenwood).

Goldeneye *Bucephala clangula,* Cormorant *Phalacrocorax carbo,* Grey Heron *Ardea cinerea*, Water Rail *Rallus aquaticus,* Oystercatcher *Haematopus ostralegus*, Ringed Plover *Charadrius hiaticula,* Curlew *Numenius arquata,* Common Sandpiper *Actitis hypoleucos,* Green Sandpiper *Tringa ochropus,* Redshank *Tringa totanus,* Black-headed Gull *Chroicocephalus ridibundus*, Lesser Black-backed Gull *Larus fuscus,* Herring Gull *Larus argentatus* and Common Tern *Sterna hirundo*.

With declines shown by most of the birds and the disappearance of Tufted Duck and Whitethroat along the River Lagan it is not surprising that species richness also declined (Figure 12). There were no significant relationships between species richness and any of the four meteorological factors. The only significant relationships were between some species and meteorological factors. There were three significant relationships for Mallard: March maximum temperature ($_{adj}R^2 = 0.323$, $F_{1,15} = 8.6$, $P = 0.01$); March minimum temperature ($_{adj}R^2 = 0.311$, $F_{1,15} = 8.2$, $P = 0.012$); February rainfall ($_{adj}R^2 = 0.275$, $F_{1,15} = 7.1$, $P = 0.018$). Each of these regressions was positive indicating that Mallard numbers were higher when March maximum and minimum temperatures as well as February rainfall were higher. There was also a relationship between March rainfall: Sedge Warbler ($_{adj}R^2 = 0.285$, $F_{1,15} = 7.4$, $P = 0.016$); Grey Wagtail ($_{adj}R^2 = 0.239$, $F_{1,15} = 6.0$, $P = 0.027$). These latter regressions were negative indicating that their numbers were higher when March rainfall was lower.

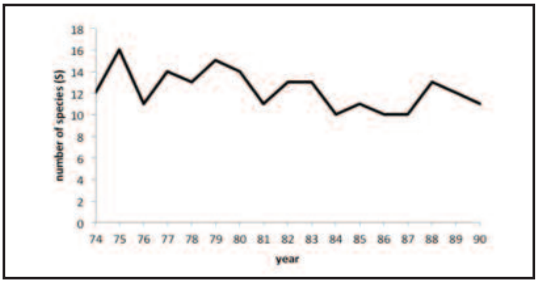

Figure 12. Species richness (S) during seventeen years at River Lagan WBS.

Discussion

Of the nine species that were sufficiently numerous for meaningful analysis only one, Mallard, increased on the Lagan WBS, as it did across the UK (Robinson *et al.* 2015). However, Robinson *et al.* (2015) were unable to identify the drivers for the increase. Bailey (1982) concluded that Mallard had increased slightly during his Lagan WBS from 1974 to 1981; however, the inclusion of data from 1982 to 1990 in the present analysis showed the increase to be substantial, with abundance more than doubling. Cabot (2009) has suggested that the increase in Mallard across the UK may have been due to the release of hand-reared birds although he did acknowledge that the story was complicated. Certainly nowadays many Mallards on the River Lagan show traits of being descended from captive stock, exhibiting aberrant

Plate 162. Spotted Flycatcher (Shay Connolly).

plumage for instance (personal observation). However, warmer and wetter conditions might have allowed greater adult survival in the late winter and early spring prior to the onset of breeding (for breeding seasons see Ferguson-Lees *et al.* (2011)).

The pattern of change in abundance of Little Grebe during the Lagan WBS coincides exactly with the UK trends (Robinson *et al.* 2015) which also peaked in the late 1970s, but was then followed by a decline and a slight recovery in the late 1980s. Parkin and Knox (2010) suggested that national declines might have been due to a combination of American Mink and loss of waterside vegetation and Bailey (1982) regarded American Mink as a major factor in the decline of Little Grebe along the River Lagan. Likewise, the pattern of change of Moorhen, which was identical to Little Grebe, matched exactly the pattern of change throughout the UK (Robinson *et al.* 2015). Bailey (1982) suggested that the decline through the 1980s was due to American Mink predation along the River Lagan, a view consistent with Parkin and Knox (2010).

Of the two non-passerines that have declined during the Lagan WBS, the disappearance of Tufted Duck is exactly the opposite of UK trends where abundance increased during the same period (Robinson *et al.* 2015). However, at that time Tufted Duck suffered a decline in breeding numbers on Loughs Neagh and Beg which Cabot (2009) attributed to increased competition for invertebrate food from an expanding Roach *Rutilus rutilus* population. Roach are a relative newcomer to Irish waters (Maitland & Campbell 1992) and at the time of the Lagan WBS were expanding both in terms of numbers and distribution (David Cragg-Hine personal communication). Judging by anglers' information boards along the River Lagan nowadays, Roach are a common coarse quarry. It seems likely then that competition from Roach may have been an important cause of Tufted Duck disappearance on the Lagan at that time. There was no detectable decline in Bailey's (1982) analysis as the disappearance occurred from 1981 onwards. The decline of Kingfishers on the River Lagan occurred in the late 1980s so was not detected by Bailey (1982); prior to that time Kingfishers averaged about four pairs each year. At that time there was a UK decline in Kingfishers followed by a recovery from the mid-1980s onwards (Robinson *et al.* 2015). The reasons for the UK decline between 1975 and 1985 are unclear (Parkin & Knox 2010), and although Kingfishers are susceptible to freezing conditions, no such association was detected in the present analysis. Indeed Ruttledge (1968) suggested that the very cold 1962/63 winter did not impact upon Irish Kingfishers and more recently Crowe *et al.* (2010) reported that the severe 2009/10 winter had no impact upon Kingfishers in the Munster Blackwater and Boyne river systems.

Four passerine species suffered substantial declines during the Lagan WBS: Whitethroat, Sedge Warbler, Grey Wagtail and Reed Bunting. Bailey (1982) found no Whitethroats in the last year of his analysis (1981). The present analysis shows that Bailey had witnessed the beginning of the end for Lagan Whitethroats, as although some birds were found between 1981 and 1990, none of them bred. The infamous UK crash was the topic of the seminal paper by Winstanley *et al.* (1974). It is now understood that the loss of Whitethroats was due to problems in their wintering area in the Sahel region of sub-Saharan Africa (Parkin & Knox 2010). In particular, reduced rainfall and agricultural change both impacted on open habitats which led to widespread desertification and habitat degradation (Sanderson *et al.* 2006). Such changes have been clearly linked to declines in a number of European migrants that wintered in the Sahel in the 1980s (Baillie & Peach 1992). Another sub-Saharan migrant (to the coastal countries between Senegal and Ghana (Flegg 2004)), the Sedge Warbler, has had declines across the UK similar to Whitethroat (Robinson *et al*. 2015), and along the River Lagan, but patterns of recovery in Sedge Warblers in the UK have been a little more substantial than for Whitethroat. Throughout the UK there have been four troughs in abundance of Sedge Warblers, one of which in 1985 coincided with the period of the Lagan WBS. The small numbers of Sedge Warblers on the River Lagan WBS meant that there was no good correlation between the Lagan and UK trends;

Plate 163. Willow Warbler (Shay Connolly).

however it is clear that Sedge Warbler numbers were high in the late 1970s after which they declined. Peach *et al.* (1991) concluded that changes in abundance were attributable to adult survival and that numbers were especially low in years of poor rainfall in the wintering grounds of West Africa. Also of interest is the relationship between abundance and lower March rainfall during the Lagan WBS.

The reasons for the decline of Grey Wagtail and Reed Bunting are different from Whitethroat and Sedge Warbler as only the warblers migrate to Africa. A few British and Irish Grey Wagtails may migrate to the near continent, but most Reed Buntings are resident (Wernham *et al.* 2002). Throughout the UK, numbers of Grey Wagtails showed a marked decline from the mid-1970s to the early 1980s, after which there was a recovery, though stalled throughout all of the 1990s. In the last ten years numbers have again declined (Robinson *et al.* 2015). Whilst the abundance of the River Lagan Grey Wagtails was broadly similar to BirdTrends, Parkin and Knox (2010) could provide no conclusive explanation regarding patterns of change. However, Robinson *et al.* (2015) state that the patterns of change are similar to those in Pied Wagtail and similar factors may be affecting both species. They further suggest that reduced survival of young may be the cause of population decline. There is also a correlation between increased abundance and lower March rainfall. The gradual decline in Reed Buntings that Bailey (1982) observed in his analysis continued through the 17-year period of the Lagan WBS following the pattern observed in BirdTrends (Robinson *et al.* 2015). Bailey (1982) could not account for the losses. Parkin and Knox (2010) said that loss of winter stubble probably played a part in the decline across the UK, whilst Peach *et al.* (1999) have said that agricultural intensification, perhaps with the loss of damp patches, might have influenced the decline. Bearing in mind that Bailey (1982) mentioned that there had been no significant drainage in the area concerned, changes in agricultural practice might seem unimportant. Whilst there are meadows adjacent to the river there is no arable farming in this part of the catchment. Therefore, the loss of Reed Buntings during the Lagan WBS cannot be easily explained.

Bailey (1982) concluded that the arrival of American Mink and climate conditions might have been responsible for changes in the bird species monitored by the Lagan WBS. Both of these factors can be seen as influential in the declines of some of the species along the River Lagan. Bailey also concluded that pollution may have led to population declines. Unfortunately there are no data to substantiate this claim as the Biology and General Chemical Assessment schemes did not begin in Northern Ireland until 1990 (Information Management – DOENI, personal communication).

Conclusion

The declines in many birds that Bailey (1982) began to detect worsened in the remaining years of the River Lagan WBS. For species like Whitethroat and Sedge Warbler the declines were caused by changes in their wintering grounds; a factor that was only beginning to be appreciated when Bailey wrote his paper. Little Grebe and Moorhen probably declined because of the arrival of American Mink, whilst Tufted Duck may have declined with competition for food from Roach. Reed Bunting might have declined as a result of agricultural change, albeit away from the immediate survey area. However the fact that so many species showed decreases over the 17-years of the River Lagan WBS might indicate that a common cause could be implicated as well. The only factor common to these species would be water quality for which, unfortunately, there is no direct evidence.

Afterword

Chris Bailey's contribution to bird monitoring must not be under-estimated; he contributed 34 years of field-work to CBC and WBS. Long-term studies of birds are rarely reported in *Irish Birds*, but one (Nairn *et al.* 2000) made specific mention of the value of long-term monitoring. There has never been such long-term monitoring of suburban and riparian birds in the greater Belfast area than those provided by Chris Bailey's work. His published River Lagan work shows the importance of longer-term data sets: his report only just managed to detect the declines that the full Lagan WBS shows so dramatically. The importance of Chris Bailey's CBC and WBS can also be seen in that six species are birds of conservation concern in Ireland (Colhoun & Cummins 2013) and occurred as breeding species: Grey Wagtail (Red-listed) and Little Grebe, Coot, Kingfisher, Sand Martin and Spotted Flycatcher (Amber-listed).

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I am grateful to the British Trust for Ornithology for providing Chris Bailey's CBC and WBS data for the express purpose of writing this paper; in particular to John Marchant for answering numerous queries regarding the surveys. I must thank Chris Bailey's daughter, Ruth McNabb, who wrote his short biography and Neville McKee who provided snippets regarding Chris's involvement at the Copeland Bird Observatory, and finally to two critical minds, Olivia Crowe and Jeremy Greenwood, who kept me on the straight and narrow when I had a tendency to veer off.

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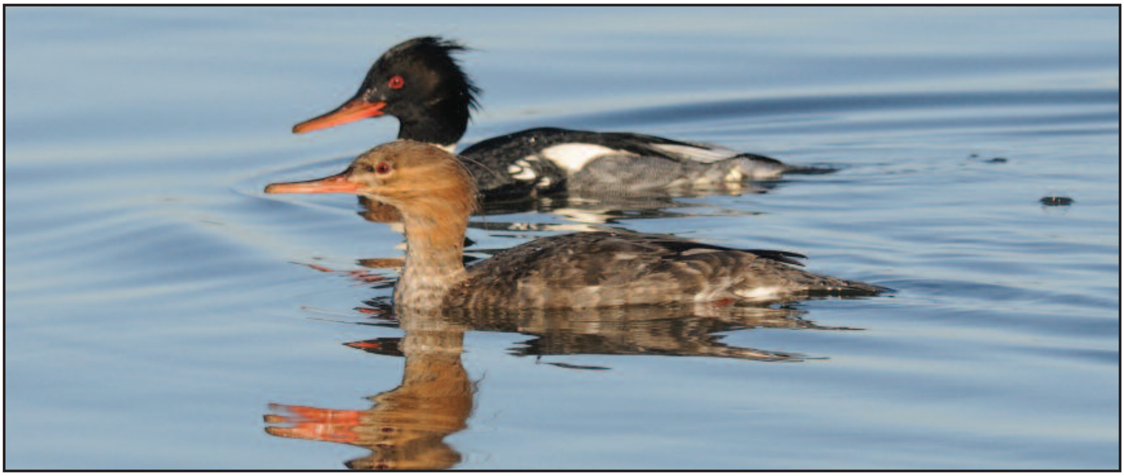
Appendix 1

Christopher W. Bailey: a biography by his daughter, Ruth McNabb.

Chris Bailey was born on 13 July 1912 in Par, Cornwall, and spent his childhood in the South West of England. In 1928, at the age of 16, he sailed to Australia and spent three years there, working on a farm in New South Wales. On his return to England he joined the army and spent six years in the Royal Horse Artillery before leaving to join Post Office Telephones as an engineer. He was posted to Glasgow where he met his wife Effie. They married in 1942 and subsequently had two daughters, Ruth and Joan. When war broke out in 1939 Chris joined the Coast Artillery and was posted to the island of Saint Helena for two years. Following that he spent some time in Dover before being posted to India and then Burma. When he returned to Post Office Telephones after the war he was promoted to Executive Engineer covering many of the islands on the west coast of Scotland.

Towards the end of 1954 he was promoted again and took up a post in Northern Ireland as Maintenance Area Engineer for the whole province. Chris had been interested in nature from an early age and shortly after moving to Belfast he joined the Belfast Naturalists' Field Club, initially interested in botany but later joining the zoological section where he developed his interest in ornithology. In 1958 he joined the Copeland Bird Observatory where he became a qualified ringer. He was treasurer and then secretary of the observatory for many years. He was a member of the Ulster Society for the Protection of Birds (now RSPB), the Northern Ireland Ornithologists' Club where he carried out a monthly count of wildfowl on the west coast of Lough Neagh and the British Trust for Ornithology for whom he started doing census work covering areas such as Stranmillis College, Belvoir Park Forest and the Lagan Towpath. He retired in September 1974 and became a voluntary worker at the Citizen's Advice Bureau and Meals on Wheels. In 1975 he began teaching a bird-watching class for the Workers' Educational Association (WEA). In 1984 the group left the WEA and set up the Phoenix Bird Group. Chris died on 4 December 1997 at the age of 85. Following his death, the Phoenix Bird Group funded a sea watching hide in his name at Copeland Bird Observatory.

Disturbance response of Red-breasted Mergansers *Mergus serrator* to boat traffic in Wexford Harbour



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Keywords: Boat traffic, disturbance, *Mergus serrator*, Red-breasted Merganser

Observations were made of 45 interactions between Red-breasted Mergansers *Mergus serrator* and boat traffic in Wexford Harbour; 71% of these observations showed a disturbance response and 49% showed a flush response. The percentage of observations showing a disturbance response increased from 29% at lateral distances of more than 500 m from the boat route, to 90% at lateral distances of less than 250 m. The present study shows that mergansers have a high degree of behavioural sensitivity to disturbance from marine traffic. This sensitivity should be taken into account in the assessment of any proposed activities that may cause increased levels of marine traffic during the winter period in areas frequented by important concentrations of this species.

Introduction

Wexford Harbour supports a nationally important population of Red-breasted Merganser *Mergus serrator,* with a mean annual peak count of 90, and a maximum count of 100, during the period 2009/10 to 2013/14 (Irish Wetland Bird Survey: www.birdwatchireland.ie/?tabid=111, accessed 7 February 2016), compared to the 1% national importance threshold of 20. Most Red-breasted Mergansers occur in the main harbour (i.e. downstream of Wexford Bridge) although small numbers occur in Ferrycarrig Reach (upstream of Wexford Bridge), as

Plate 164. Red-breasted Mergansers (John Fox).

well as offshore outside the harbour. Wexford Harbour also supports a large bottom mussel culture industry. This activity involves the relay of seed mussels from the Irish Sea for on-growing in subtidal habitat within the harbour and subsequent harvesting after a period of one to two years. There are six dredgers based in Wexford Harbour (with overall lengths of 28-45 m), that are used for relay, husbandry and harvesting of the mussels. Detailed information on patterns of boat activity in Wexford Harbour is not available, but, for example, during September 2015 to January 2016, dredger activity was recorded on seven of nine visits, with one to four dredgers active on these visits (TG, unpublished data; these visits were for an unrelated survey and were not timed to coincide with dredging activity). While the majority of winter boat activity within the harbour is related to mussel culture, additional boat activity in the harbour can include crab potting, a cot (a small flat-bottomed boat) used to access an oyster farm in the southern part of the harbour, and small inshore potting vessels entering and leaving the harbour along the navigation channel.

An Appropriate Assessment of the impact of aquaculture on the Wexford Harbour and Slobs Special Protection Area is currently being prepared, as part of a programme of measures to ensure compliance with the Court of Justice of the European Union's judgement in Case C418/04 Commission versus Ireland, "The Birds Case" - Appropriate Assessment (see DAHG 2015). During an initial survey visit for this assessment, it was noted that Red-breasted Mergansers in Wexford Harbour appear to show a high degree of sensitivity to disturbance from boat traffic. On this visit, we recorded some observations of the response of mergansers to a cot and to small inshore potting vessels. We subsequently made two additional visits with the specific aim of recording the response of mergansers to dredgers. Here we report our observations and provide the first detailed published evidence about the response of wintering Red-breasted Mergansers to marine traffic.

Methods

Observations were made on 4, 5 and 20 February and 2 March 2015. All observations were made by the same observer (TG). The vessel activity observed included: dredgers mussel dredging, starfish mopping (fishing for starfish using a dredge modified for operating in mussel cultivation beds) and travelling to and from the quays in Wexford Town; a cot travelling to and from an oyster farm in the southern part of the harbour; and small inshore potting vessels travelling along the navigation channel as they entered and left the harbour. The duration of the observation periods on each of these days, and the vessel activity observed, is shown in Table 1. The positions of the activities observed are shown in Figure 1. Four vantage points were used: Harbour View Road, Ferrybank (by Wexford Swimming Pool and Leisure Centre), Ardcavan Beach, and the observation tower in the Wexford Wildfowl Reserve. Observations of boats travelling to and from the quays in Wexford were mainly made from the Harbour View Road vantage point, with a small number made from the Ferrybank vantage point. Observations of dredgers actively mussel dredging or starfish mopping were made from the other three vantage points, depending upon their position.

For each boat observed travelling, the response of all mergansers within a distance of at least 500 m perpendicular to the boat's route was recorded. Where practical, the response of mergansers at greater distances was also recorded. For each dredger observed actively mussel dredging or starfish mopping, the response of all mergansers within a distance of around 1 km from the area being fished was

Table 1. Duration of observation periods and vessel activity in Wexford Harbour.

Date	Time	Vantage point	Vessel activity
04/02/2015	15:45	Harbour View	Cot returning from oyster trestles
05/02/2015	15:30-16:30	Harbour View	Cot returning from oyster trestles and small inshore potting vessel returning along navigation channel
20/02/2015	07:00-09:00	Harbour View	Two dredgers and three small inshore potting vessels departing along navigation channel
20/02/2015	09:45-10:50	Observation Tower	Dredger starfish mopping
20/02/2015	11:30-13:00	Observation Tower	Dredger starfish mopping and returning across northern side of harbour
02/03/2015	06:45-08:30	Harbour View	Small inshore potting vessel departing along navigation channel
02/03/2015	09:45	Harbour View	Dredger departing
02/03/2015	10:30	Ferrybank	Dredger returning
02/03/2015	11:00-12:30	Observation Tower	Dredger starfish mopping
02/03/2015	13:00-14:00	Harbour View	Two dredgers departing
02/03/2015	14:30-18:00	Ardcavan/Ferrybank	Three dredgers mussel dredging (one returned at 16:40)

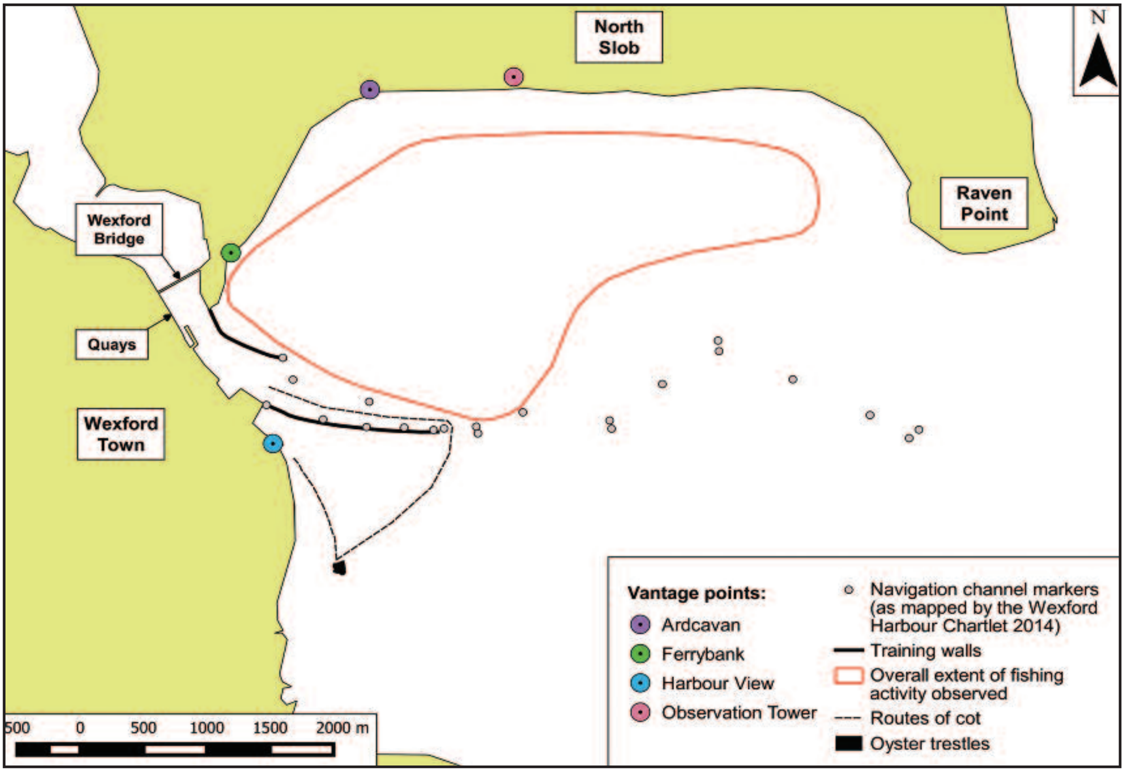

Figure 1. Vantage points used and locations of boat activity in Wexford Harbour.

recorded, in so far as was possible. However, when dredgers were working at long distances from the vantage points, birds within this zone on the far sides of the dredgers could have been missed. The responses were classified as 'no response' (continued normal behaviour as the boat passed), 'alert response' (stopped previous behaviour, sat up in water and stretched neck, often looking around at the boat), 'swam away' (purposefully swam away from the path of the boat, usually following an alert response), or 'flushed' (flew away from the path of the boat, often following alert and swam away responses). For each observation, the perpendicular distance of the birds from the path of the boat (lateral distance), and the straight line distance between the boat and the bird when any disturbance response occurred (closest distance) was estimated. For birds showing no response, the closest distance was equal to the lateral distance. For observations of boats travelling in, and around, the navigation channel, the positions of the birds and the boats relative to the navigation markers mapped on the Wexford Harbour Chartlet 2014 were used to help estimate distances. For observations of dredgers fishing, the position of the plot being fished (as supplied by the operators) was used to help estimate distances. Lateral distances were not estimated for responses to boats dredging or starfish mopping, as the boats were not following defined routes in these cases. Where possible, when birds were flushed, the duration of their flight was estimated and the approximate flight path was mapped. However, as the priority was to record the maximum number of disturbance responses, flushed birds were only followed when there were no other mergansers to monitor for disturbance responses.

Red-breasted Mergansers typically occur in small groups and, usually, the disturbance response was the same for all members of the group. For the analyses, the number of observations (where each observation represents the interaction of one group with one boat), rather than the total number of birds, have been used. In two cases, the response within the group was variable: and in these cases each response was used as a separate observation. It should be noted also that, in some cases, separate observations may refer to the same birds interacting with different boats. The Freeman-Halton extension of the Fisher exact probability test (Ruxton & Neuhäuser 2010) was used to test the significance of differences in the proportions of observations showing disturbance responses between the different lateral distance classes, and between the different vessel types.

Results

A total of 45 observations of Red-breasted Mergansers interacting with marine traffic were recorded in February and March 2015 (Table 2). Across all distances, 71% of the observations showed a disturbance response and 49% of observations showed a flush response. Swimming away was less frequently used as a disturbance response, while there were only two observations of birds showing an alert response without subsequently swimming away or flushing. Birds that swam away often made short dives while swimming, but we did not observe any birds making sustained dives as an apparent disturbance response. The disturbance response was clearly related to the lateral distance from the boat route, with an increase in the percentage of observations showing a disturbance response from 29% at lateral distances of more than 500 m to 90% at lateral distances of less than 250 m (Table 3). The proportions of observations showing disturbance responses, and the proportions showing flush responses, were significantly different between the three lateral distance classes (Freeman-Halton extension of the Fisher exact probability test: $P = 0.009$ for disturbance responses and $P = 0.023$ for flush responses). Birds that flushed, always flushed at more than 250 m from the boat (Table 2), and often flushed at very long distances. The maximum flush distance recorded was around 1.5 km.

Observations of interactions with the cot travelling to and from the oyster trestles showed the highest incidence of active disturbance responses, and the proportion of observations showing flush responses was significantly different between the three vessel types (Freeman-Halton extension of the Fisher exact probability test: $P = 0.099$ for disturbance responses and $P = 0.042$ for flush responses). However, when the distribution of observations in relation to lateral distances from the vessels are taken into account, and taking account of the small sample sizes, the evidence for differences in the nature of the disturbance responses between the three vessel types is not compelling (Table 4).

The flight durations of flushed mergansers ranged from 23 to 220 seconds (mean 86 seconds, n = 9). The flushed mergansers resettled at distances ranging between a few hundred metres to around 3 km from their original locations. However, because the birds followed circuitous flight paths the distances actually travelled were much greater. On two occasions the birds that had been originally flushed, subsequently flushed additional birds as they flew over.

Table 3. Summary of incidence of disturbance response type by lateral distance by Red-breasted Mergansers in Wexford Harbour.

Lateral distance	% of observations with:		n
	Any disturbance response	Flush response	
< 250 m	90%	70%	20
250-500 m	73%	36%	11
> 500 m	29%	14%	7
All	**71%**	**49%**	**45**

Table 2. Disturbance response of Red-breasted Mergansers in Wexford Harbour.

Lateral distance	Closest distance	Number of observations				
		No response	Alert response	Swam away	Flushed	Totals
< 250 m	< 250 m	2	0	3	0	5
	250-500 m	0	0	0	6	6
	> 500 m	0	0	1	8	9
250-500 m	250-500 m	3	0	4	3	10
	> 500 m	0	0	0	1	1
> 500 m	> 500 m	5	1	0	1	7
Not classified	< 250 m	0	0	0	0	0
	250-500 m	3	0	0	1	4
	> 500 m	0	1	0	2	3
Totals	**All**	**13**	**2**	**8**	**22**	**45**

The 'alert response' column only includes observations where no other disturbance response was recorded.

Table 4. Summary of disturbance responses by boat type of Red-breasted Mergansers in Wexford Harbour.

Boat type	Lateral distance	Number of observations			
		No response	Alert	Swam away	Flushed
Cot	< 250 m	0	0	0	6
	250-500 m	0	0	1	1
Small inshore potting vessel	< 250 m	1	0	4	3
	250-500 m	2	0	2	3
	> 500 m	3	1	0	1
Dredger	< 250 m	1	0	0	5
	250-500 m	1	0	1	0
	> 500 m	2	0	0	0
	Not classified	3	1	0	3

Discussion

While the impact of disturbance on breeding Red-breasted Merganser has previously been studied (Kahlert 1994), there appears to be very little information about the effects of disturbance on wintering populations. In the Baltic Sea, it has been reported that "the disturbance distance of red-breasted merganser with regard to vessels is large and the birds usually take flight when a ship is approaching" (www.helcom.fi/baltic-sea-trends/biodiversity/red-list-of-species/red-list-of-birds, accessed 12 February 2016). However, the report cited in support of this statement is not available. There are a few studies that report the response of other sea duck species to vessel traffic. Schwemmer *et al.* (2011) reported median flush distances for four species ranging from 208 m for Common Eider *Somateria mollissima* to 804 m for Common Scoter *Melanitta nigra* in the German North Sea and German Baltic Sea, while Skei (2014) reported a mean distance for "energy demanding responses" of 178 m for Common Eider in Norway (note that these figures refer to the straight line distance, not the lateral distance). Larsen and Laubek (2005) studied the effects of high speed ferries on wintering Common Eider in the Kattegat Sea in Denmark and concluded that the ferries had the potential to significantly reduce habitat use within 500 m of the ferry routes.

The Wexford Harbour observations are based on survey work over a limited period of time, but they show a very clear pattern of disturbance responses that are related to the lateral distance from the disturbance source. The results indicate that Red-breasted Mergansers in Wexford Harbour are very sensitive to disturbance from marine traffic with disturbance responses being recorded in around 84%, and flush responses in 58%, of observations when boats passed within a lateral distance of 500 m or less, and with birds flushing at distances of up to 1.5 km from the boats. While these observations are

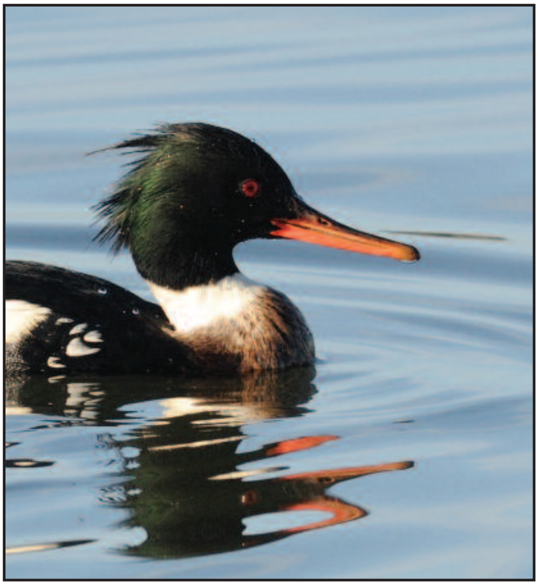

Plate 165. Red-breasted Merganser (John Fox).

in line with the perceived disturbance response reported from the Baltic Sea (see above), we found these observations to be surprising and unexpected: we are familiar with Red-breasted Merganser wintering populations in a number of coastal sites in Ireland and have not previously noted any evidence indicating a high degree of behavioural sensitivity to disturbance from marine traffic. However, we have not carried out systematic observations at any of these other sites. It may be that the proximity of an area regularly frequented by mergansers to the main navigation channel in Wexford Harbour has highlighted a disturbance sensitivity that may not be so obvious at other sites. Alternatively, there may be some site-specific factor that causes a higher degree of disturbance sensitivity in Wexford Harbour. For example, wildfowling from boats takes place in the harbour and, while Red-breasted Merganser is not a quarry species, such hunting may increase their sensitivity to disturbance from boats.

The use of measures of behavioural responses to disturbance (such as the distance at which birds flush) to assess the conservation significance of disturbance impacts has been criticised. Species responses to disturbance should reflect the costs of responding to the disturbance (Gill *et al.* 2001): if there is alternative habitat available and the costs of moving to this habitat are low, species may show a stronger avoidance of disturbed areas, compared to species with little alternative habitat available and or higher costs of moving to this habitat which may be forced to stay close to the source of the disturbance. In this study, Red-breasted Mergansers in Wexford Harbour regularly frequented areas adjacent to the navigation channel, indicating that these areas provided suitable foraging habitat. This pattern of regular occurrence adjacent to the navigation channel could also be interpreted as indicating that the costs to the mergansers of responding to disturbance by flushing are relatively low. However, the frequency of boat movements along the navigation channel in winter is low and there will be long periods each day without any vessel activity. Therefore, the occurrence of mergansers in the vicinity of the navigation channel does not necessarily indicate a high degree of habituation to, or tolerance of, disturbance impacts.

This study has shown that Red-breasted Mergansers in Wexford Harbour have a high degree of behavioural sensitivity to disturbance from marine traffic. This behavioural sensitivity will need to be taken into account in assessment of any proposed activities that may cause increased levels of marine traffic during the winter period. Further research in other Irish coastal sites is required to determine whether this sensitivity is a general pattern, or whether it is due to some site-specific factor at Wexford Harbour.

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Diet of coastal breeding Ravens *Corvus corax* in east County Cork

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Keywords: *Corvus corax*, diet, marine prey, Raven, terrestrial prey

A nest site on a coastal cliff at Ballycotton, County Cork, provided an opportunity to examine the diet of Ravens *Corvus corax* by analysing the contents of pellets collected near the nest. Most terrestrial prey was Rabbit *Oryctolagus cuniculus,* Brown Rat *Rattus norvegicus* and Field Mouse *Apodemus sylvaticus.* Birds (including Racing Pigeon *Columba livia*) and Common Frog *Rana temporaria* were taken sparingly. Several prey species of marine origin were also taken, mostly in small quantities. Stones and mortar pieces ranging up to 23 mm across in size were frequent, sometimes in large quantities, within individual pellets. Prey taken is compared with the only other Irish study of Raven diet.

Introduction

The Raven *Corvus corax* breeds throughout Ireland, and its range has increased considerably (by 95%) in the 40 years since 1968-72 (Balmer *et al.* 2013). This increase has been mirrored in County Cork, and breeding takes place at a variety of site types ranging from coastal cliffs, inland cliffs, quarries, ruined buildings and trees (Pat Smiddy, personal observation).

The Raven population has been studied at several locations in Ireland, notably in Northern Ireland and in County Wicklow and adjoining counties (see Hutchinson (1989) and Ratcliffe (1997) for details of these and other studies). The breeding populations in east County Cork and west County Waterford (Smiddy 1992) and south County Mayo have also been studied

Plate 166. Raven (John Carey).

Plate 167. Raven (Michael O'Clery).

(McGreal 2007), as have those in Connemara, County Galway (Breen 2014). However, apart from casual observations of feeding behaviour and prey taken (e.g. Thompson 1849, Ussher & Warren 1900, Ratcliffe 1997), only one Irish study of diet has been published, involving an examination of 39 pellets collected in April and May 1989 at two nearby west County Cork coastal breeding sites (Berrow 1992). This study examines diet through pellet analysis at a single coastal breeding site in east County Cork.

Methods and study area

Pellets of indigestible parts of Raven prey are often cast near nest sites and at roosts (Ratcliffe 1997, Berrow 1992). Sixty-three pellets were collected near an occupied nest site on a coastal cliff at Ballycotton, County Cork (W9863) (one in May 1997, 25 in April 1998, 37 in April and May 2003). Each pellet was teased apart and prey remains identified to species where possible. Results are expressed as percentage frequency of occurrence of each prey type within each pellet, identical to the methods employed by Berrow (1992). However, the volume content of each prey type within each pellet was not estimated.

The terrestrial habitat along the cliff either side of the nest site includes a narrow band of mostly steep-sloping coastal heath with vegetation including Western Gorse *Ulex gallii* and Bell Heather *Erica cinerea.* Farmland immediately inland from the coast consists of a mixture of pasture and tillage fields. Dairying (cattle) is the dominant animal based farming type, although sheep rearing occurs also, but is much less important locally and does not occur within 1-2 km of the nest site. The coastal cliffs within 1-2 km of the nest site are steep and are composed mostly of rock with little horizontal exposure at low tide, apart from a few small sandy beaches sandwiched between the rocks backed by low clay cliffs. At the time of the study, the next nearest Raven nest site along the coast was about 8 km away, while inland the nearest was about 5 km away. Ballycotton Island (3 km away) has a small seabird colony dominated by Cormorant *Phalacrocorax carbo,* Herring Gull *Larus argentatus,* Lesser Black-backed Gull *Larus fuscus* and Great Black-backed Gull *Larus marinus,* while the mainland coast is sparsely occupied by small colonies of Fulmar *Fulmarus glacialis.*

Results

The mean size of whole pellets was 43.82 x 22.09 (se = 2.86 x 0.71; range = 26-65 x 19-33; n = 11) (all measurements in mm) (most pellets could not be measured accurately as they crumbled easily due to the presence of stones and mortar; see below). The prey items taken are shown in Table 1. Despite the coastal nature of the site, most prey was terrestrial in origin. Among the terrestrial prey, Rabbit *Oryctolagus cuniculus,* Brown Rat *Rattus norvegicus* and Field Mouse *Apodemus sylvaticus* were taken in greatest numbers, while birds were taken sparingly, as was the Common Frog *Rana temporaria,* which was taken only in one season. Species of marine origin were also taken, but apart from the calcareous algae, *Corallina officinalis* and fish (Pisces), only in small numbers. Stones and mortar pieces ranging up to 23 mm across in size were frequent, sometimes in large quantities, within individual pellets.

Table 1. Percentage frequency of occurrence of prey (minimum number of each prey species in parentheses) identified in 63 Raven pellets from Ballycotton, County Cork, 1998 and 2003 (note that one pellet collected in May 1997 has been included in the April 1998 column.

Prey items	April 1998 (n = 26)	April & May 2003 (n = 37)
Terrestrial species		
Rabbit *Oryctolagus cuniculus*	76.9 (11)	16.2 (3)
Brown Rat *Rattus norvegicus*	19.2 (4)	16.2 (4)
Field Mouse *Apodemus sylvaticus*	34.6 (7)	51.4 (19)
Fox *Vulpes vulpes*	-	2.7 (1)
Common Frog *Rana temporaria*	-	10.8 (4)
Racing Pigeon *Columba livia*	3.8 (1)	2.7 (1)
Small passerine bird	3.8 (1)	-
Beetle (Coleoptera)	3.8	-
Marine species		
Fish (Pisces)	23.0	-
Saddle Oyster *Anomia ephippium*	-	2.7 (1)
Whelk *Buccinum undatum*	3.8 (1)	-
Crab (Crustacea)	-	2.7 (1)
Algae *Corallina officinalis*	34.6	5.4
Other items		
Eggshell	3.8	-
Grass	3.8	-
Stones and mortar	38.5	21.6

Discussion

The Raven is mainly a scavenger of prey that is already dead, or dying, although there is evidence that it behaves as a predator in some instances (Ratcliffe 1997). Prey remains recovered from pellets indicate the qualitative composition of diet, rather than actual diet in terms of the relative importance of each prey species (Marquiss & Booth 1986). Although few prey items are completely digested, the importance of large vertebrates may be underestimated since few bones are swallowed, and once a carcass is opened, only a little wool, fur or hair may be taken. On the other hand, smaller prey may be swallowed entirely, especially marine organisms such as crustaceans and molluscs whose hard shells may then be plentifully represented in pellets.

In this study there was no evidence that large domestic farm animals were consumed. Farming practice in the study area involves tillage as well as dairying, and dead or fallen animals are quickly removed, therefore leaving little or no carrion from this source available for scavenging birds such as the Raven. In contrast, large vertebrate animals, most of which were probably sheep, were identified as important prey in the west Cork study (Berrow 1992). Rabbit, Brown Rat and Field Mouse were the commonest prey in this study (Table 1). It is interesting to consider how these small animals were obtained by Ravens; Rabbit and Brown Rat may be taken either as carrion or as live animals, but although the Field Mouse is largely nocturnal, it is likely that this species, and Common Frog, were taken alive while Ravens were probing among vegetation. A single lower jaw bone of a Fox *Vulpes vulpes* was discovered in one pellet, and it is most likely that this large prey species was taken as carrion. The absence of Bank Vole *Myodes glareolus,* which is often active by day, may be explained by the fact that this species was then still in the process of colonising this part of southeast Cork, and had not reached the coast in the vicinity of the nest site by 2003 (Pat Smiddy, personal observation). Berrow (1992) identified Rabbit on only two occasions in his west Cork study, and he did not find either Brown Rat or Field Mouse.

Birds did not rate highly as a prey item for the Raven in this study, nor did they do so in west Cork (Berrow 1992). Remains of two Racing Pigeons *Columba livia* were recovered, identified by the presence of rings. Ratcliffe (1997) considered that Racing Pigeons found as prey of the Raven were probably mostly scavenged remains from Peregrine Falcon *Falco peregrinus* kills. Eggshell fragments were recovered from only one pellet, and possibly (although not definitely) belonged to Pheasant *Phasianus colchicus*. Eggs have been recorded in most studies of Raven diet, and coastal breeding birds often raid seabird colonies (Thompson 1849, Ussher & Warren 1900, Ewins *et al*. 1986, Marquiss & Booth 1986, Berrow 1992, Ratcliffe 1997). However, there was no evidence that the Ravens in this study visited the seabird colony at nearby Ballycotton Island.

Ravens mostly took prey of marine origin in small numbers in this study, apart from the calcareous algae, *Corallina officinalis* and fish, and the steep nature of most of the coast and the consequent scarcity of rock pools may be the reason for this. In contrast, Berrow (1992) recorded marine species in 77% of pellets collected from two nest sites, including an abundance of *Corallina officinalis*; this species also featured strongly in pellets of Ravens from Shetland (Scotland) (Ewins *et al.* 1986). Although it may be that Ravens preferentially take this seaweed in spring as a source of calcium for growing nestlings, it was also taken frequently in other months in Shetland (Ewins *et al.* 1986). Only a single example each of a gastropod and bivalve mollusc was recorded in this study (Table 1).

Given that Ravens at Ballycotton appear to have foraged in terrestrial habitats to a large extent, it is perhaps surprising that grass was found only in a single pellet. However, stones were found in many pellets, and this is in agreement with Berrow (1992) who found stones in 59% of pellets in west Cork. Among the stones in the Ballycotton study were an approximately equal amount of similar sized pieces of mortar. The clifftop fields nearby contain gate piers made of stone and mortar, and these are likely to be the source of the mortar pieces. While Ravens (and many other species) swallow stones to assist in the digestion of food (Ratcliffe 1997) the taking of pieces of mortar recorded in this study may have the added benefit of providing a source of calcium during the breeding season.

This study provides further evidence of the diversity of food taken by the Raven across its range, and it shows that while there are general similarities, there may be considerable variation between different coastal territories (e.g. Ewins *et al.* 1986) depending on the adjoining habitat, leading to differences in the amount of marine prey taken versus terrestrial prey. The adaptability of the Raven with respect to dietary diversity in different habitats may be, at least partly, responsible for the success of the species over the past 40 years in expanding its range into lowland habitats away from the remoter upland and coastal areas which became its refuges during periods of intense persecution in the past. A study of diet at inland sites in Ireland would make an interesting comparison with the two relatively small studies of coastal breeding birds published to date (Berrow 1992, this study), as would a study in winter at both coastal and inland sites.

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Post-breeding aggregations of roosting terns in south Dublin Bay in late summer

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Keywords: Dublin Bay, migration, roosting, Sternidae, terns

A series of 26 evening counts of terns was carried out in Dublin Bay in August and September between 2013 and 2016. The survey area encompassed the whole of Sandymount Strand in the south of Dublin Bay, while on two occasions Dollymount Strand, about 5 km to the north, was surveyed simultaneously. The maximum number of terns recorded at Sandymount Strand was 6,645 in 2013, 2,264 in 2014, 4,035 in 2015 and 17,440 in 2016. These data are broadly consistent with previous surveys undertaken between 1959 and 2010, but the peak of 17,440 terns recorded in 2016 is the second highest total ever recorded at the site. The numbers recorded at Dollymount Strand were much lower, but were not insignificant. Five tern species were recorded: Black *Chlidonias niger*, Sandwich *Sterna sandvicensis,* Common *Sterna hirundo,* Roseate *Sterna dougallii* and Arctic *Sterna paradisaea.* There were considerable fluctuations in the numbers recorded on consecutive surveys, indicative of a high turnover rate. The large numbers involved, ringing controls, and the presence of Black Terns, which do not breed locally, show that terns are drawn to the roost from further afield than the Dublin, or Irish Sea, colonies.

Introduction

Conservation of migratory species requires knowledge of key staging sites used on migration, and adequate protection being bestowed on these sites. Sandymount Strand forms part of the South Dublin Bay and River Tolka Estuary Special Protection Area, designated under the European Union Birds Directive. Roseate Tern *Sterna dougallii,* Common Tern *Sterna hirundo* and Arctic Tern *Sterna paradisaea* use Sandymount Strand as a post-breeding staging site, and they are listed as qualifying interests to the designation (NPWS 2015).

The use of south Dublin Bay as a major night roosting site for terns between the end of the breeding season and their departure to their wintering grounds was first noted in 1959 (Merne *et al.* 2008), and evening counts have taken place

Plate 168. Tern roost at Sandymount Strand, Co. Dublin (Dick Coombes).

sporadically since then. In 1960, 27 evening counts took place (Merne *et al.* 2008). More recently, a single count was conducted in 1996 (Newton & Crowe 1999), 13 counts between 2002 and 2004, 20 counts in 2006 and 22 counts in 2007 (Merne *et al.* 2008), with a further ten counts in 2010 (Merne 2010).

This post-breeding roost is located within about 30 km of three important breeding colonies; Rockabill (Common, Roseate and Arctic Tern), Dublin Port (Common and Arctic Tern) and Dalkey Islands (Common, Roseate and Arctic Tern), and it is likely that many of the terns that occur in the Sandymount roost originate from these. Typically, the number of terns using Sandymount Strand builds from late July onwards when birds disperse from the breeding colonies. There are roosting terns present at Sandymount Strand for up to two months each year, building reserves for migration and commencing their moult (Cabot & Nisbet 2013, Ginn & Melville 1983). Five species of tern, namely: Black Tern *Chlidonias niger,* Sandwich Tern *Sterna sandvicensis,* Common Tern, Roseate Tern and Arctic Tern have been recorded regularly, and Little Tern *Sternula albifrons* has been reported occasionally. A small number of other large gatherings of pre-migratory terns have been documented; in the southeast of Ireland close to the Lady's Island Lake colony (Wexford), on the east coast of England at Teesmouth, Cleveland and on the west coast of England at Seaforth, near Liverpool.

Surveys were carried out between 2013 and 2016 as part of the wider Dublin Bay Birds Project, which aims to collate baseline data for waterbirds and seabirds in Dublin Bay throughout the year. This paper summarises the results of these tern roost surveys, and puts these numbers in the context of the number of birds likely to be in the Dublin Bay area at this time of the year, based on the number of birds at the nearby colonies and the likelihood of turnover in the number of staging birds.

Methods

Sandymount Strand is located in south Dublin Bay (53.32° N, 6.19° W) on the coast of County Dublin. The intertidal sandflats between Poolbeg and Dun Laoghaire were searched for terns on 26 evenings during August and September between 2013 and 2016 (Table 1). The surveys were undertaken as the birds congregated to roost at dusk. Survey days were chosen when high water occurred within one to two hours of sunset. This ensured that the area available for roosting terns (i.e. exposed sand) was relatively small (compared with low water) and facilitated counting, as the birds were relatively close to the observers. All counts took place between 17:30 and 21:20, with counts starting two hours before sunset and continuing until light levels were insufficient for counting, which tended to be around 15 minutes after sunset. All counts took place in fair weather and in good light conditions. Each year, either six or seven surveys were conducted, between 11 August and 25 September. While the surveys in each year were spread throughout August and September, counts often took place on consecutive evenings in order to avail of suitable tidal conditions.

Table 1. Dates of dusk tern surveys at Sandymount Strand, south Dublin Bay, 2013–2016.

Date	2013	2014	2015	2016
11 August	-	-	-	✓
12 August	-	-	✓	✓
13 August	-	-	-	✓
15 August	✓	-	-	-
16 August	✓	-	-	-
18 August	-	✓	-	✓
19 August	-	✓	-	-
25 August	-	-	✓	-
26 August	-	-	✓	-
29 August	✓	-	-	✓
30 August	✓	-	-	✓
2 September	-	✓	-	-
3 September	-	✓	-	-
7 September	-	-	✓	-
8 September	-	-	✓	-
12 September	✓	-	-	-
13 September	✓	-	-	-
14 September	-	-	-	✓
15 September	-	✓	-	-
16 September	-	✓	-	-
21 September	-	-	✓	-
25 September	✓	-	-	-

At least two observers were present during each survey session. In 2013, three of the surveys (29 August, 12 September and 13 September) were conducted with one observer on the West Pier at Dun Laoghaire, a vantage point at the southern end of the bay, and the other on the strand at Merrion Gates located approximately mid-way along the strand. The first observer directed his/her telescope across the mouth of the bay (to the Bailey Lighthouse in Howth) and counted the terns as they flew into the bay. The second observer counted the terns as they settled on the beach. All counts were time-referenced so that both surveyors could compare their totals at the end of the survey to minimise double-counting of birds. In all other counts the observers took up vantage points on Sandymount Strand (Table 2), either working together from the same place, or if the flock settled in several disparate locations, counting in different areas of the strand and adding their totals later. All tern species were counted and included in the totals. The presence, but not the abundance, of different tern species was recorded on

Table 2. Approximate vantage points used during dusk tern surveys at Sandymount Strand, south Dublin Bay, 2013-2016 (in order of frequency of use).

Vantage point	Coordinates
Sandymount Baths	53.3238° N, 6.2037° W
Merrion Gates	53.3152° N, 6.2003° W
Booterstown Strand	53.3113° N, 6.1923° W
Irishtown Strand	53.3335° N, 6.2073° W
Dun Laoghaire west pier	53.3040° N, 6.1351° W

each survey, but due to difficulty in accurately identifying the different species at distance and in poor light, birds were not identified to species level during the counts. When time allowed, and when flocks had settled within suitable observation distances, the presence of Black, Sandwich, Common, Roseate and Arctic Terns was recorded.

The survey was concentrated in the core roosting area, but outlying areas were repeatedly scanned with telescopes to determine if there were satellite flocks present. When present, these flocks were counted to ensure that all birds were accounted for. The surveyors counted synchronously, each counting the same flock and comparing totals afterwards. However, on one occasion, due to the very large numbers of birds, surveyors counted different parts of the flocks and combined their totals. Binoculars were used to locate the flocks and telescopes (Swarovski and Opricron, each with 20-60 zoom) were used to carry out the counts. Birds were generally counted when settled on the ground in units of five. However, when birds were disturbed or 'dreading' (which was frequent), it was necessary to count in units of 50 or 100 individuals. When possible, these approximate counts were followed with more accurate counts when the birds resettled. In 2016, two counts were conducted on Dollymount Strand, which is about 5 km to the north of Sandymount Strand and located on the eastern side of North Bull Island, to determine if the area was being used as an alternative or additional roosting area. Three surveyors took part in these counts, which were conducted synchronously with the Sandymount Strand counts on 29 and 30 August.

During all counts where both surveyors counted the same flock of birds, each surveyor counted separately, logging the

Plate 169. Map of Dublin Bay with the primary area used by roosting terns on Sandymount Strand delineated.

totals and the time, so that they could be compared at the end of each survey evening. The average of the final count from both surveyors is considered the total number of birds in the roost on that evening. To maximise consistency in counting, at the beginning of each season, both surveyors counted several distinct flocks from the same vantage point and compared their respective totals for each.

The count period ranged from 30 days (in 2014) to 42 days (in 2013). There was a difference of seven days in the first survey of each season, with surveys starting earliest in 2016 (11 August) and latest in 2014 (18 August). The last count of each season took place in the second half of September, with the latest survey taking place on 25 September 2013.

Results

Surveyors obtained a high level of consistency when counting flocks, and differences were generally less than 10% of the total. The numbers of terns (all tern species combined) recorded at Sandymount Strand in south Dublin Bay in August and September between 2013 and 2016 are shown in Figure 1. The maximum number was 6,645 in 2013, 2,264 in 2014, 4,035 in 2015 and 17,440 in 2016. The total number of terns at Dollymount Strand on 29 and 30 August 2016 was 300 and 650, respectively.

Five tern species were recorded during the study period: Black, Sandwich, Common, Roseate and Arctic Tern. Common and Arctic Terns were recorded in each of the 26 dusk surveys. Sandwich Terns were recorded on 23 surveys, Roseate Terns on 24, and Black Terns on 15 surveys (Table 3). The vast majority of birds were Common Terns; Arctic and Roseate terns were the next most prevalent, and Sandwich and Black Terns were only recorded in numbers of less than ten.

On the ten occasions when the numbers on consecutive evenings could be compared (Table 4), there were often considerable differences in the totals; greater than 10% on nine occasions, greater than 40% on four occasions and greater than 70% on two occasions.

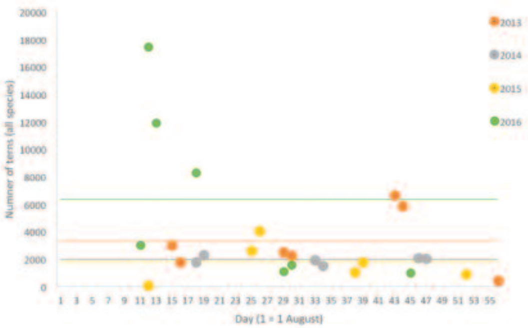

Figure 1. Total number of all species recorded during dusk tern surveys at Sandymount Strand, south Dublin Bay, 2013–2016 (horizontal lines represent the average for each year).

Discussion

During all counts the peak numbers of birds were recorded towards the end of the survey session at dusk, and there was no apparent reduction in the number of birds arriving at the roost as dusk approached. On each occasion it is highly likely that birds continued to arrive after the light levels had diminished and prevented further counts, therefore the figures presented in this paper must be considered minima. The average number of terns recorded in each year between 2013 and 2016 is broadly consistent with those recorded previously. An average of 2,845 terns was recorded in 2010 (Merne 2010), averages of 3,868 and 2,344 were recorded in 2006 and 2007 respectively (Merne *et al*. 2008), and an average of 1,230 was recorded between 2002 and 2004 (Merne *et al.* 2008). In 1998 and 1999 total counts of 2,000 and 5,040 were recorded (Newton & Crowe 1999), and an average of 1,850 was recorded in 1960 (Merne *et al.* 2008).

However, the number of surveys per season in the present study (6-7) was considerably lower than during many

Table 3. Number of dusk tern surveys at Sandymount Strand, south Dublin Bay, 2013–2016, in which each of the five tern species was recorded.

Year	Number of surveys	Black Tern	Sandwich Tern	Common Tern	Roseate Tern	Arctic Tern
2013	7	2	5	7	5	7
2014	6	6	6	6	6	6
2015	6	3	6	6	6	6
2016	7	4	6	7	7	7
Total	26	15	23	26	24	26

Table 4. Differences in numbers recorded on successive days during dusk tern surveys at Sandymount Strand, south Dublin Bay, 2013–2016, ordered by magnitude of the difference.

Initial survey date	Subsequent survey date	Year	Initial survey total	Subsequent survey total	Difference (%)
11 August	12 August	2016	3,003	17,440	83
15 August	16 August	2013	3,010	1,765	-71
12 August	13 August	2016	17,440	11,800	-48
7 September	8 September	2015	1,048	1,795	42
25 August	26 August	2015	2,617	4,035	35
2 September	3 September	2014	1,908	1,496	-28
18 August	19 August	2014	1,766	2,264	22
12 September	13 September	2013	6,645	5,835	-14
29 August	30 August	2013	2,500	2,252	-11
15 September	16 September	2014	2,065	2,040	-1

of the previous surveys (20-22) (Merne *et al.* 2008), which increases the likelihood that seasonal peaks could have been missed, and there is currently insufficient information to determine if the seasonal patterns of occurrence has changed over time.

The large number of terns recorded on 12 August 2016 was exceptional when compared with other recent years. However, it is not possible to comment on whether such large accumulations had occurred in other years but remained undetected. Perhaps such an aggregation is a rare and short-lived event, like the estimated 20,000 to 30,000 terns that were reported on 31 August 1996 (Newton & Crowe 1999).

Due to the distance that birds were observed from, often in suboptimal light conditions, priority was given to obtaining accurate counts rather than determining the proportion of the different species using the roost. Of the five tern species that were recorded, three (Common, Roseate and Arctic Tern) breed in Dublin, while the closest colonies for the other two species are the Netherlands (Black Tern, BirdLife International 2004)) and Counties Down and Wexford (Sandwich Tern). Little Terns, which breed about 30 km to the south at Kilcoole (Wicklow) and about 50 km to the north at Baltray (Louth), were not recorded in any of the surveys.

The presence of Black Terns suggests that some proportion of other tern species may come from at least as far away as the Netherlands. However, their presence can also inform us on the rate of turnover within the roosting flock. Black Terns were recorded in each season and in 15 out of the 26 surveys overall. This species is also reported to the Irish Birding website (www.irishbirding.com) by birdwatchers each August and September. The 71 Black Terns (51 at Merrion Strand and 20 at North Bull Island) reported to the Irish Birding website on 23 August 2015 was remarkable, and would undoubtedly have attracted the attention of many local birdwatchers. On the following four evenings, Black Terns were reported to the website, but the greatest number reported was just five, despite the presence of larger numbers of birdwatchers than usual. While all observers may not have submitted their records of Black Terns to the website, it is likely that if the same large numbers were present as were reported on 23 August, this fact would be unlikely to have gone unnoticed. This apparent short-lived peak possibly reflects how transient Black Terns (and possibly other tern species as well) are in Dublin Bay.

The extent of turnover of the other tern species in the roost is not known, but is likely to be considerable. The high level of disparity between consecutive counts, given the consistency in surveyors and survey methods, lends strong support to this suggestion. Surveys were conducted on three consecutive evenings, between 11 and 13 August 2016, and the total number of birds recorded was 3,003, 17,400 and 11,890 respectively. While it cannot be ruled out that the high number of birds that roosted at Sandymount Strand on 12 August were present in Dublin, away from the survey area, on the preceding and succeeding evenings, nevertheless, it suggests a very high rate of turnover with several thousand birds spending just a single night at the Sandymount roost. Whether this level of turnover reflects distinct migratory waves, or if the numbers at the roost on a given evening are driven by weather and other variables, or a combination of both, is not known. A higher frequency of surveys in Dublin Bay, in concert with similar survey effort at other night roosting sites in the Irish Sea would be required to elucidate this.

A colour-ringing scheme for Common and Arctic Terns was initiated in 2015 to investigate recruitment within the Dublin Port colony. A high level of re-sightings of these individually identifiable birds could be generated at the roost, and these data could go some way to revealing how long individual birds spend in the roost each season.

Based on the number of nesting terns at the three nearby tern colonies in Dublin (Rockabill, Dalkey Islands and Dublin Port), and the associated productivity values between the 2013 and 2016 breeding seasons, there has been an estimated 10,184 to 11,430 terns (juveniles and post-breeding adults) in Dublin at the end of each breeding season (Burke *et al.* 2014, 2015, 2016, Butler & Newton 2016, Kearney *et al.* 2013, Newton 2010, Newton *et al.* 2013, Tierney *et al.* 2014, 2015, 2016). However, there is evidence that some of the birds using the roost are coming from further afield than the Dublin colonies. The count of 17,440 terns on 12 August 2016 and the estimated 20,000 to 30,000 terns reported on 31 August 1996 (Newton & Crowe 1999) illustrate that the Dublin totals are at least occasionally augmented by birds from other colonies. Furthermore, while the majority of captures (for ringing) of ringed birds at the roost are those that were ringed in the Dublin breeding colonies, birds ringed in Norway, Scotland and England have also been captured (Niall Tierney, own data). It is also known that at Rockabill, where there is an extensive programme of ring-reading throughout the breeding season, individual terns from foreign colonies occur during the latter part of the breeding season (Rockabill dataset, unpublished).

The importance of Dublin Bay for post-breeding terns, and the protected status of the site, is endorsed by the current set of counts. It is recommended that the programme of counts, in conjunction with ringing (and colour-ringing) are continued to facilitate further detailed assessments of the origins and migratory patterns of the birds that are using this staging area, and to further demonstrate the importance of the site for discrete populations of terns in both a national and international context.

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Mother knows best? Interspecific fostering between Roseate Terns *Sterna dougallii* and Common Terns *Sterna hirundo* on Rockabill Island, County Dublin

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Several cases of interspecific fostering by Roseate Terns *Sterna dougallii* and Common Terns *Sterna hirundo* were identified on Rockabill Island, County Dublin in 2015 and 2016, and these are described here. Common Terns incubated a single Roseate Tern egg in a nest-box. The resulting chick was last seen alive when at least ten days old, and it is considered likely to have survived to fledging. Common Terns also incubated a mixed clutch of Common Tern and Roseate Tern eggs, but all young died due to heavy rainfall. A Roseate Tern was seen feeding a Common Tern chick of at least three weeks old, but no further sightings were obtained. Roseate Terns hatched a mixed brood of Common and Roseate Tern chicks, although only the Common Tern chick survived. Two nests have also been recorded with mixed clutches of Common Tern and Roseate Tern eggs, but both nests were abandoned before chicks could hatch.

Introduction

Rockabill is a small granite island (0.9 ha) around 6 km off the coast of north County Dublin. The island has been wardened each year since 1989 for the conservation of the breeding Roseate Terns *Sterna dougallii,* Common Terns *Sterna hirundo* and Arctic Terns *Sterna paradisaea* present (the Rockabill Tern Project). Total numbers of breeding terns on Rockabill have risen from 206 primary nests in 1989 to 3,645 in 2016 (Burke *et al.* 2016), making it the second largest mixed tern colony in Ireland.

The most numerous species on Rockabill are Roseate and Common Terns, with 1,556 and 2,029 nests respectively in 2016 (Burke *et al.* 2016). Roseate and Common Terns are broadly similar in appearance and ecology and are closely related (Bridge *et al.* 2005). Both are medium-sized white black-capped terns in the genus *Sterna* with a broad global distribution in tropical and temperate climates, and they often nest at the same colonies (Burger & Gochfeld 1988, Ramos &

Plate 170. Roseate Terns (John Fox).

del Nevo 1995). Both species forage by plunge-diving and there is significant overlap in their diet of small fish, generally including sandeels *Ammodytes* spp. and species of the family Clupeidae, although they may constitute different proportions of their respective diets (Robertson *et al.* 2014). In Europe, the Common Tern is listed on Annex I of the EU Birds Directive (2009/147/EC) and their conservation status in Europe is currently considered favourable (BirdLife International 2015). The Roseate Tern is also listed on Annex I but their conservation status is currently Red-listed in the United Kingdom (Eaton *et al.* 2015) and critically endangered in France (UICN France & MNHN 2016). Rockabill has been the largest of four regularly occupied colonies of Roseate Terns in Europe away from the Azores for much of the last thirty years (Cabot & Nisbet 2013).

Both species nest amongst each other over much of the island on Rockabill, with Roseate Terns preferring nest sites sheltered by vegetation or rocks and Common Terns using more open areas (Kelly 1998). Since the Rockabill Tern Project began, specially designed nest-boxes have been deployed to provide protected nesting sites for Roseate Terns (Casey *et al.* 1995). The number of nest-boxes deployed has been increased in line with the population, with around 650 boxes in recent years and in excess of 75% occupancy rates; 759 boxes were deployed in 2016 with 90% occupancy (Burke *et al.* 2016). In keeping with their nesting choices, Roseate Tern chicks tend to remain in shelter, where possible, while Common Tern chicks often move about freely in the vicinity of the nest site once they are old enough to do so.

A number of interspecies mixed clutches and broods were identified on Rockabill in 2015 and 2016, involving Roseate and Common Terns. We discuss the different permutations of the mixed clutches and broods recorded, the behaviour of the adults in caring for their heterospecific chicks in relation to the typical adult/chick behaviour of Roseate and Common Terns, and the ultimate success of the broods involved.

Methods

Two wardens are present on Rockabill during each breeding season (May to August). Nine subsections of the island are monitored as 'study areas' and surveyed each morning and evening during the breeding season to follow the progress of Roseate and Common Tern nests through the incubation and chick-rearing stages. A census of primary nests outside the designated study areas is carried out within 34 days of the first egg being laid (usually second week of June), and multiple visits to these areas are made thereafter for the purposes of ringing the chicks. During the 2015 and 2016 seasons any unusual nests found either during or after the nest census were followed up with subsequent visits when time allowed. No nest visits were carried out in strong wind or persistent rain and time spent in the colony was kept to a minimum to avoid chilling of eggs or chicks, and to ensure adults can feed their chicks. Chicks are coded based on the clutch size from which they originated and the order in which they hatched: A1 chicks are the product of a clutch of one egg, A2 and B2 are the first and second chicks to hatch from a clutch of two eggs, respectively.

For monitoring purposes chicks are considered to have fledged successfully if they were recorded on or after Day 15 and were not subsequently recorded dead (Day 0 is hatch day). Chicks fledge at 28-34 days, but the majority of mortality occurs before Day 15 when energy requirements are high and before chicks can independently thermoregulate (Klaassen 1994).

Results

The following are accounts of the different mixed-species clutches and broods recorded on Rockabill during the 2015 and 2016 breeding seasons:

(Case 1) Common Tern pair with Roseate Tern egg and chick in nest-box

In July 2015 an adult Common Tern was seen leaving a Roseate Tern nest-box. On inspection the box was found to contain a single Roseate Tern egg being incubated by a pair of Common Terns. A Roseate Tern chick hatched soon after and the adults continued to raise the chick inside the nest-box, with the chick behaving as it normally would in terms of begging for food, and it generally remained inside the nest-box. The chick was seen alive until it was at least ten days old, and was not subsequently found dead leading to the conclusion that it is likely to have survived to fledging.

(Case 2) Common Tern pair with mixed clutch of Common and Roseate Tern eggs and chicks

During July 2015 in a separate part of the island a clutch containing two Common Tern eggs and one Roseate Tern egg was found, being incubated by a pair of Common Terns. The presence of a Roseate Tern nest-marker in a sheltered corner nearby (about 0.6 m away) indicates that the Roseate Tern egg originated from there. All three eggs hatched within a few days of each other. The Roseate Tern chick was the runt of the brood, seemingly not getting fed as much as the other two. When the chicks were only a few days old a period of wet weather caused the death of the two Common Tern chicks and the Roseate Tern chick was not subsequently recorded (all were ringed).

In 2016 two nests were recorded as having three Common Tern eggs and one Roseate Tern egg. Both clutches were seemingly abandoned before hatching. In one nest the Roseate Tern egg had rolled from the rest of the clutch prior to abandonment.

Plate 171. Adult Common Tern and chick Roseate Tern in a Roseate Tern nest-box on Rockabill in 2015 (Case 1) (Brian Burke).

Plate 172. Brood of two Common Tern chicks and one Roseate Tern chick (bottom left) on Rockabill in 2015 (Case 2) (Brian Burke).

(Case 3) Roseate Tern pair with Common Tern chick
In late July 2015, during a feeding study, an adult Roseate Tern was seen feeding a Common Tern chick that was at least three weeks old. No further sightings or observations were made of this pairing.

In June 2016 two eggs showing physical characteristics of Roseate Tern eggs (shape, size, colour and pattern) were laid one day apart at a nest site which had been used by Common Terns the year before. The site was predominantly open with minimal cover from a rock face about 0.6 m away. Observation from a distance confirmed that Roseate Terns were attending the nest. The first chick hatched 23.5 days after the first egg was laid and displayed physical characteristics of that of a Common Tern chick, with the other egg producing a typical looking Roseate Tern chick two days later (24.5 days after laying). The average incubation period for the first egg in Common Tern clutches on Rockabill in 2016 was 22.48 ± 0.05 days (range: 21.0-28.5, n = 286; Burke *et al*. 2016), and that for the first and second eggs in Roseate Tern clutches was similar at 23.10 ± 0.07 days (range: 21.5-27.5, n = 281) and 23.13 ± 0.09 days (range: 21.5-28.5, n = 165) respectively (Burke *et al.* 2016). Within a few days the first (i.e. Common Tern) chick had left the nest while the Roseate Tern chick remained where it was. This often meant that the incoming Roseate Tern parents perched beside the Common Tern chick and away from the nest when returning with food. Survival of B2 chicks was very low at 5.17% and 4.47% for Roseate and Common Terns, respectively, on Rockabill in 2016, seemingly due to food shortages (Burke *et al.* 2016). The Roseate Tern chick from this clutch died on its third day, weighing around half (10.5 g) of what a chick of the same species and hatch order would be in years of adequate food supply (23.7 g in 2015, 19.2 g in 2013) (Burke *et al.* 2015, Kearney *et al.* 2013). The Common Tern chick continued to grow as normal (in the context of growth rates of other chicks that year) and was expected to fledge successfully having last been seen on Day 22 when wardening on the island ceased. It should be noted that neither the parents nor either of the chicks showed any indication in their plumage features of being hybrids or backcrosses of Roseate and Common Terns.

Discussion

The examples of mixed clutches and broods described here illustrate some of the possible permutations of mixed 'families' at a tern colony and how the differing and contrasting behaviours of the two species play out in somewhat unique contexts.

(Case 1) Common Tern pair with Roseate Tern egg and chick in nest-box
This example is notable not just because of the different species involved, but the fact that the Common Terns successfully incubated and brooded inside a nest-box. In recent years around 650 nest-boxes have been deployed for Roseate Terns on Rockabill, and despite the presence of about

Plate 173. Adult Roseate Tern brooding chick Roseate Tern (left) and chick Common Tern (right) on Rockabill in 2016 (Case 3) (Brian Burke).

Plate 174. Common Tern (M.O'Clery).

2,000 pairs of Common Terns none have nested in a box. Boxes are either 3- or 3.5-sided, with a 30 cm or 15 cm opening respectively, and Common Terns have occasionally nested right outside the opening. Common Terns will nest amongst stands of Tree Mallow *Malva arborea,* as do Roseate Terns, but perhaps the mallow stands still offer access and escape routes that the open-nesting Common Terns prefer. It seems most likely that the Common Terns in this instance lost their own egg nearby and took the Roseate Tern egg as their own. Previous work on guillemots *Uria* spp. found they were more likely to accept 'foreign' eggs or chicks if their own egg was lost or chicks had died, particularly within a short period of time after breeding failure (Gaston *et al.* 1995, Lefevre *et al.* 1998). Common Terns are more aggressive towards intruders than are Roseate Terns, and they also return to nests more quickly when flushed (personal observation). It may be that this pair found the egg when the Roseate Tern parents had been flushed, and were subsequently able to drive off the rightful owners if and when they returned. The nest-box used was a 3-sided one with a 30 cm opening partly blocked by a stone. Given their usual avoidance of nest-boxes it is also surprising that the Common Terns did not attempt to move the egg closer to the opening or even out of the nest-box over the course of incubation.

(Case 2) Common Tern pair with mixed clutch of Common and Roseate Tern eggs and chicks

In the case of mixed clutches incubated by Common Terns it seems that the Common Tern adult rolled the Roseate Tern egg into its own nest, either because it was blown or had rolled into close proximity, or that the Common Tern clearly saw the egg and assumed it to be its own. In the example from 2015 both nests were within about 0.5 m of each other and the Roseate Tern nest was at the higher end of a gradual slope in a corner between rocks, but with no overhanging shelter. Common Terns have been recorded stealing eggs from neighbouring conspecific nests on Rockabill (see Penland (1984) for similar behaviour in Caspian Terns *Hydroprogne caspia*), presumably assuming that any egg in close proximity must be their own. Common Tern clutches of eight and nine eggs have been seen in recent years, and the nesting material surrounding neighbouring clutches broken through where the eggs had apparently been actively rolled into the new host nest. Invariably none of the eggs in these clutches hatch as it is impossible for a Common Tern to incubate such a large surface area of eggs. All three eggs in this mixed-species clutch hatched within a short time of each other, leading to a brood of three, which is often seen in Common Terns. We suspect the Roseate Tern chick hatched last, meaning it was the runt of the brood in terms of growth and mass. However, it is possible that the Roseate Tern chick was not from the third egg to hatch, but the more strident and active nature of the Common Tern chicks meant they were better capable of claiming incoming food from the adults, although this remains speculation. Notably, for the first few days the Roseate Tern chick also stayed with its Common Tern siblings in the open area where they hatched rather than seeking out a sheltered spot, as Roseate Tern chicks would normally do. It is impossible to know whether this behaviour would have continued as the chick grew stronger and became more mobile.

The two similar nests in 2016, both with full Common Tern clutches and an additional Roseate Tern egg, appear to have occurred in a similar set of circumstances to the 2015 nest described above. The additional egg, making clutches of four, may have been enough to prevent the adults incubating the clutch effectively in these cases, and led to their eventual abandonment. In one of these nests the Roseate Tern egg had clearly rolled from the nest.

(Case 3) Roseate Tern pair with Common Tern chick

In the 2015 example the adult Roseate Tern was feeding a well-grown (>3 weeks old) Common Tern chick in a self-contained section of the island with 208 Common Tern nests and only eight Roseate Tern nests, the latter all situated under dense vegetation and not near any Common Tern nests. Bearing this in mind, a mix-up at the egg stage, as in Case 1, seems unlikely. It seems more likely they met at a later stage, perhaps with the adult having lost its egg or chick, and the chick having wandered away from its own nest area and imprinted on the adult. This behaviour has previously been observed amongst Roseate and Common Terns on Rockabill (although not between species) and is most likely to happen when the chick is still only a few days old (as in guillemot species; Gaston *et al.* 1995, Lefevre *et al.* 1998). A pair of Roseate Terns was observed feeding a Common Tern chick at Lady's Island Lake (Wexford) in 2011, having previously abandoned their own egg (Daly *et al.* 2011). We cannot rule out the possibility of a Roseate x Common Tern hybrid, which have been recorded previously (e.g. Robbins 1974, Hays 1975), and which often bear a strong resemblance to Common Terns (or to Arctic Terns in Roseate x Arctic Tern crosses). Whittam (1998) suggested that a skewed sex ratio, mis-imprinting on a chick that later affected its mate choice, or simply parental inexperience, could explain cross-breeding between Roseate and Arctic Terns. Nisbet *et al.* (2016) found a female-biased sex ratio in adult Roseate Terns at Bird Island (Massachusetts), and in most recorded cases of hybridization between Roseate and Common Terns, the Roseate has been the female (Ewins 1987, Zingo *et al.* 1994, Whittam 1998). However, the sex ratio of either species on Rockabill has yet to be investigated.

The 2016 example is the only one to have occurred in a monitored study area and was therefore observed twice daily from the time the first egg was laid until the wardens left and the remaining chick was 22 days old and deemed likely to fledge successfully. The nest site was used by Common Terns in previous years and on first impression appears to be more typical of a Common than a Roseate Tern nest site given the lack of surrounding shelter. Both eggs were paler with finer spots than the average Common Tern egg. Both eggs were also longer than the average Common Tern egg, measuring 44.4 x 28.8 mm and 45.4 x 29.3 mm compared to a mean of 40.7 x 29.8 mm (n = 125) for A and B Common Tern eggs measured on Rockabill in 2013 (similar in Vogrin 1998, Janzekovic *et al.* 2003, Kearney *et al.* 2013). Though uncommon, Common Tern eggs of similar length have been recorded elsewhere with a maximum of 45.8 mm reported in Janzekovic *et al.* (2003). On the day the second egg was laid it was confirmed that Roseate Terns were attending the nest. Had a Roseate Tern been the first to hatch it might be assumed that Common Terns had opportunistically laid or dumped an egg in an attempt at brood-parasitism. However, the hatch dates and incubation periods suggest the Common Tern egg was laid first. The most likely explanation, therefore, is that Roseate Terns took over a Common Tern nest, in contrast to the example outlined in Case 1 above. Given the close laying dates of the eggs, perhaps both Roseate and Common Terns had intended to nest in a similar space, and the Common Terns laid first. If the Common Terns had intended to lay a second egg they might have been absent for extended periods of time before initiating incubation when the clutch was complete, allowing the Roseate Tern pair to lay their own egg and takeover the nest and the mixed clutch. The clutch was laid relatively late in the season and most late Roseate Tern nests have only one egg (personal observation). It would have been interesting to see how the differing behaviours of Common and Roseate Tern chicks, in terms of wandering from the nest, would have played out in a 'normal' year when adequate food was available. The propensity for Common Tern chicks to wander and Roseate Tern chicks to remain in one place may have impacted provisioning rates to the latter if the parents were to perch where the Common Tern chick had moved to, reducing the opportunities for the Roseate Tern chick to compete for, or to receive food. However, the shortage of food in the 2016 season meant the second chick was always likely to perish within a few days of hatching.

Concluding remarks

We can rule out the possibility of brood-parasitism (i.e. egg dumping) in Case 1, Case 2 (2015) and Case 3 (2016), despite this being suspected in mixed tern families elsewhere (Cadiou

& Jacob 2010). Case 1 involved only one egg and it was clearly the adults who were out of place rather than the egg. In Case 2 (2015), the nearby Roseate Tern nest marker with no corresponding egg indicates that this was the origin of the Roseate Tern egg in the Common Tern clutch. In Case 3 (2016) the hatch dates of the eggs suggests egg dumping was unlikely.

Arctic Terns also nest on Rockabill, but use the peripheral and rocky areas of the colony, in contrast to the Roseate and Common Terns who nest among each other in the more central parts of the colony on soil and vegetation. Thus, there is little or no opportunity for interspecies mix-ups between Arctic Terns and the other tern species on Rockabill.

According to Craik (2010), whether an egg from one species in the clutch of another is successfully hatched and ultimately raised to fledging depends on the time of laying, incubation period, feeding compatibility and the degree to which the two species are related. Both species discussed here are very similar in all of these traits and all were thought likely to be capable of raising their adopted young to fledging. We strongly suspect in Case 1 that the Roseate Tern chick with Common Tern parents was raised successfully, although only a future resighting will confirm this beyond doubt. In Case 3 (2015) the Common Tern chick was raised by Roseate Tern parents to a level where fledging success seems almost certain and met the criteria usually used to deem fledging successful on Rockabill. Therefore, some of the cases of mixed species examined here were successful.

It has been highlighted (Craik 2010) that mixed clutches are likely to be more frequent than supposed, but individual reports are likely to be scattered in journals or written in reports with restricted distribution. As well as the difficulty in tracking down records, instances are likely to be regularly missed, even at monitored colonies, particularly between similar species, such as Common and Roseate Terns. At a large colony, such as Rockabill, there can be considerable variation between the eggs of a single species in terms of size, shape and pattern. Roseate Terns tend to lay eggs that are longer and with finer spotting than those of Common Terns (Ratcliffe *et al.* 2004), but in a colony of about 2,000 pairs of Common Terns with a mean clutch in excess of two eggs, many Common Tern eggs are found that bear a striking resemblance to those of Roseate Terns. Despite differing nesting preferences, there is much potential for competition between the two species for nesting sites, similar to the circumstances creating potential for cross-adoption and cross-fostering between Razorbills *Alca torda* and Common Guillemots *Uria aalge* (Harris & Wanless 2001).

On Rockabill, where there is uncertainty, nests are observed from a distance to see which species returns, although generally context clues such as location and the appearance of other eggs in the clutch are used to decide on nest identity. Unless nests are checked regularly (i.e. daily, or more often) or there is something particularly unusual about a nest (e.g. more eggs than expected, one particularly unusual egg, chicks of different species together) then many broods where the egg and or chick and parent species differ will go undetected. Furthermore, instances of intra-species mixed broods are also likely to be common, but yet may go unrecorded in the absence of genetic testing. This is more likely for the more numerous and open-nesting Common Terns, than for Roseate Terns whose nests tend to be in discrete and sheltered locations. We know from ringing and direct observation at Rockabill that a number of Common Tern chicks are adopted by different adults every year when their own chick gets lost or dies and a similar-sized chick is seen by the adults in the same area.

Overall, instances like these are likely to regularly occur at mixed species tern colonies, but may go largely unreported or unnoticed. However, their occurrence probably represents a tiny proportion of breeding attempts, and given the capability of similar species to bring an adopted egg or chick to fledging it is unlikely to cause any significant problems even amongst species of high conservation priority such as the Roseate Tern in Europe. Instances such as those reported here do, however, provide an insight into the parental behaviour of the species involved and the lengths they will go to in their determination to breed successfully.

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Whimbrel *Numenius phaeopus* migrations at North Bull Island, Dublin Bay: 2012 to 2015

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Whimbrel *Numenius phaeopus* migrations were monitored at North Bull Island in Dublin Bay from 2012 to 2015. A small but regular passage was recorded annually with more birds in spring than in autumn in each year. Earliest migrants arrived from mid-March but the main spring passage began abruptly in week 17 (22-28 April) with values remaining high in week 18 (29 April-5 May). Small numbers recorded frequently in mid-summer were considered likely to be late prenuptial migrants. Autumn passage was more protracted and lacked an obvious 'peak' week. There were two winter records. Based on other recent investigations into Whimbrel migration routes, it appears that the vast majority of spring migrants at North Bull Island belong to the large *islandicus* breeding populations in Iceland. A direct overseas route to West Africa in autumn by the majority of Icelandic birds possibly explains why smaller numbers are recorded at that time of year. However, exactly what proportion of autumn passage migrants are *islandicus* or *phaeopus* from Northern Europe is unknown at present and requires further investigation. The results also show that North Bull Island is a regular staging area in both spring and autumn for nationally important populations.

Introduction

There are seven subspecies of Whimbrel *Numenius phaeopus* of which two occur in Europe. The breeding range of the *islandicus* subspecies is from Iceland south to Scotland and *phaeopus* from Fennoscandia to the River Yenisey in Russia; both subspecies winter in the southern hemisphere (Van Gils *et al.* 2016). Small numbers also overwinter on the Atlantic coast of Europe (Velasco & Alberto 1993, Balmer *et al.* 2013). Ringing recoveries and biometrics indicate that the majority of Whimbrels recorded during migration in Ireland and west Britain are *islandicus,* whilst those in eastern Britain and mainland Europe are *phaeopus* (Ferns *et al.* 1979, Grant 2002, Carneiro *et al.* 2015, Robinson *et al.* 2016). During migration Whimbrels can be found in wetlands, saltmarshes and tidal flats.

In Ireland, their biannual migration patterns have been documented since the nineteenth century, with the heaviest passage thought to occur along the south and west coasts (Thompson 1850, Ussher & Warren 1900, Kennedy *et al.* 1954). They also occur inland during migration (Hutchinson 1989), while flocks are regularly observed at seabird migration watchpoints, especially along the west coast such as Annagh Head (Mayo) (personal observation) and Bridges of Ross (Clare) (Niall T. Keogh, personal communication). In spring, the earliest migrants usually arrive in March and the main passage is from mid-April into May but autumn passage is less intense and extends from June/July to September (Ruttledge 1966, Hutchinson 1989). Small numbers can occur in winter

Plate 175. Whimbrel (Shay Connolly).

and nesting has been suspected on one occasion, but not proven (Perry 2006). Despite being a familiar migrant, only one detailed study has been carried out to date. This reported 'unprecedented' numbers in spring at Cork Harbour in the 20 day period between 15 April (DOY 105) and 4 May (DOY 124) in the three years 1977 to 1979 (Pierce & Wilson 1980) (DOY = Day of Year; see Methods).

North Bull Island, Dublin Bay, is a stopover site for passage migrant waders, including Whimbrel, and this species has been known to occur regularly since the nineteenth century (Patten 1898). However, it would appear that numbers on passage were quite small with a maximum of only five or six reported by Kennedy (1953) up to the early 1950s, but this may, in part, be due to observer effort. Hutchinson (1975) stated that the highest count up to the early 1970s was 60 birds. There were also three November records and one over-wintering record from 11 November 1957 to 19 January 1958. Although the number of Whimbrels that migrate through Ireland annually is not known, it is not considered to be a species of conservation concern (Colhoun & Cummins 2013). Sites of national importance for Whimbrels in Ireland are those where a five-year mean exceeds 20 birds (Olivia Crowe, personal communication).

This study sought to establish the timing and scale of Whimbrel migrations at North Bull Island from 2012 to 2015 and to compare the results to other data for Ireland and Britain. The results provide additional insights into the migration patterns of this species in Ireland.

Study area

North Bull Island (53.3705° N, 6.1440° W) on the northern shore of Dublin Bay is a relatively new island having evolved through the accelerated accumulation of sediments following engineering works in Dublin Port in the nineteenth century (Jeffery 1977). It has since developed into a coastal ecosystem of sand dunes, saltmarshes and inter-tidal sediments. Due to the scientific importance of its habitats and species, the area is a National Nature Reserve (NNR), Special Protection Area (SPA) for birds and Special Area of Conservation (SAC) for habitats (NPWS 2016). The rich feeding grounds within the reserve attract large numbers of passage migrants and wintering waterfowl annually, including several species with populations of national and international importance (Crowe 2005).

Methods

The primary objective of field monitoring was to obtain one count per week during two ten-week periods, one in spring and one in autumn. Spring counts were conducted from mid-March to May (weeks 12 to 22) and autumn counts from early July to September (weeks 29 to 39). Data were obtained for a maximum of 80 weeks, with the occasional assistance of other observers (see Acknowledgements). The mudflats and saltmarshes were scanned up to one hour before high tide from the Wooden Bridge to Sutton (about 5.5 km) using a telescope (x30) and binoculars (x10). Outside of these main survey times in spring and autumn data on Whimbrel numbers were recorded during regular but less formal visits to the island.

Long term data on the earliest spring arrivals at North Bull Island from 1999 to 2016 were recorded by the author. These data were used to determine potential advancement in arrival times in spring. To assess potential trends in arrival times in relation to local air temperatures and the winter North Atlantic Oscillation (NAO), data were analysed using a linear regression model with Whimbrels as the dependant variable (Data Desk 6.0). January, February and March air temperature data for Phoenix Park (1999-2016) were obtained from the European Climate Assessment and Dataset (2016) website (http://www.ecad.eu/). The winter NAO index (DJFM) 1998-2015 were obtained from Hurrell *et al.* (2016). Models were checked for any breaches of the assumptions of the linear model. Calendar dates were converted to Julian days (DOY = Day of Year) and the data were assessed in weekly terms with the first week of the year starting with Julian day = 1. Data were adjusted for the 2012 leap year.

Winter records at North Bull Island were put into an Irish context using data extracted from the *Irish Birding* (2016) website (www.irishbirding.com). These data were used to assess potential changes in over-wintering populations in Ireland from 2011/12 to 2015/16 compared to data reported in the *Bird Atlas 2007-2011* (Balmer *et al.* 2013). Winter records were defined as those that occurred from 1 November to 28 February.

As the rates of turnover during pre- and postnuptial migrations at the site are unknown, it was not considered reasonable to calculate the ratio of birds between spring and autumn based on total or average numbers. Instead, the single highest weekly count in spring is compared to the highest weekly count in autumn for each of the four years.

Results

Intra-annual patterns

Apart from isolated early and late migrants, Whimbrels were recorded almost continuously between weeks 12 and 37 (Figure 1). All birds recorded were either feeding or roosting on the mudflats and saltmarshes of the island.

Although a few individuals arrived from week 11 onwards (mid-March), numbers remained in single digits until weeks 15 ($\tilde{x} = 7$, peak of 14) and 16 ($\tilde{x} = 16.3$, peak of 37). An abrupt

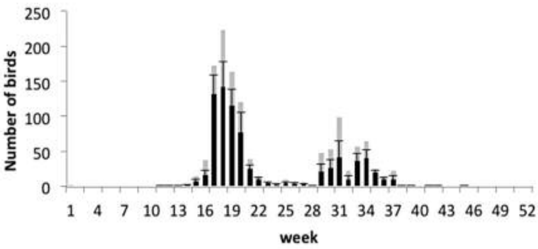

Figure 1. Weekly counts of Whimbrels at North Bull Island, 2012 to 2015. Black bars are mean number of birds (+S.E.); grey bars represent a maximum weekly count.

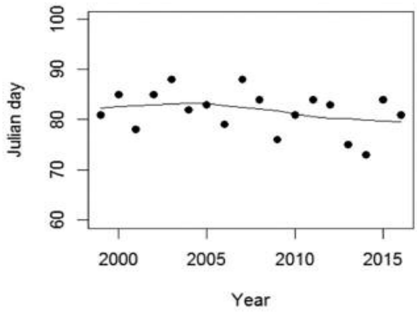

Figure 2. Earliest arrival dates of Whimbrels at North Bull Island, 1999 to 2016. Black line = lowess line.

increase occurred in week 17 ($\tilde{x}$ = 132) with peaks of 130 in 2013, 172 in 2014 and 142 in 2015. This event occurred in week 18 ($\tilde{x}$ = 141, peak of 223) in 2012. A steady decrease took place subsequently with numbers tapering off by week 22 ($\tilde{x}$ = 10, peak of 15). Up to nine birds were present in all four years in weeks 23 to 28 inclusive (June to early July). Autumn passage was lower and lacked a clearly defined 'peak'. Small numbers began to filter through from weeks 29 to 37 with maximum values in weeks 31 ($\tilde{x}$ = 41, peak of 98) and 34 ($\tilde{x}$ = 40, peak of 64). Migration usually ceased by week 38 (mid-September) but a few birds occurred sporadically into October. Within the suvey years 2012 to 2015, the earliest arrival date was DOY 73 (14 March 2014) and latest in autumn was DOY 289 (15 October 2012). Single birds were recorded on DOY 3 (3 January 2013) and DOY 313 (9 November 2015).

In excess of 20 birds (the threshold for national importance) were recorded regularly during spring and autumn in each year from 2012 to 2015. The mean peak count over this period in spring was 141 and that in autumn was 41. The ratio of the peak weekly counts from spring migration to autumn were 2.3x (2012), 2.3x (2013), 6.4x (2014) and 3.6x (2015).

Responses to changing climatic factors

The earliest arrival date over the years 1999-2016 was DOY 73 (14 March) and the mean earliest arrival date was DOY 82 (23 March). There was no significant trend in first arrival dates at North Bull Island (Figure 2) and no signifcant relationship between earliest arrival times and either local air teperatures or the winter North Atlantic Oscillation (NAO).

Although it is likely that some wintering birds in Ireland go unrecorded and there may be some duplication in those that are reported, it would appear that an average of 15 birds were present annually in winter at coastal sites in Ireland, including North Bull Island (Table 1). The majority of records refer to single birds, but ten were recorded on two occasions. Connaught was the only province without a winter record during 2011/12 to 2015/16.

Discussion

Whimbrels are common passage migrants in Ireland, occurring widely at coastal and inland sites on an annual basis (Kennedy *et al*. 1954, Hutchinson 1989). Although the numbers recorded at North Bull Island were small, a very clear biannual migration pattern was apparent, with higher numbers in spring compared to autumn in all four years. In

Table 1. Winter records of Whimbrels in Ireland, 2011/12 to 2015/16 (Irish Birding 2016).

	2011/12	2012/13	2013/14	2014/15	2015/16	Mean	S.E.
No. of reports	9	8	11	7	13	10	1.2
No. of birds	22	9	21	7	14	15	3.4
No. of sites	8	7	10	7	10	8	0.8
No. of 10km^2 recorded	8	6	9	6	7	7	0.6

Plate 176. Whimbrels (Colum Clarke).

each of the four years 2012 to 2015, spring migration occurred in a relatively short window with peak numbers usually in the last week of April and the first few days of May. In contrast, smaller numbers passed through over a longer period in autumn, and there was no clear 'peak' week. In addition, the regular presence of small numbers of birds during summer months in all years, and the two winter records, were unexpected, but not unprecedented.

To qualify as nationally important for Whimbrels the mean of five years peak counts at a site must exceed 20 birds. Although this threshold was exceeded regularly during spring and autumn annually, this survey was only carried out over a four year period. Although 2016 was not part of this survey, results from occasional counts between 14 April and 27 September also exceeded the threshold for national importance on five occasions in spring (peak of 103 on 1 May) and once in autumn (peak of 34 birds on 26 July) (Cooney 2017). Therefore, the five-year mean for 2012 to 2016 was 154 in spring and 54 in autumn. These results qualify North Bull Island as a site of national importance for Whimbrels.

Although much is known about Whimbrel breeding and wintering ranges, there are gaps in knowledge of the scale, temporal patterns and migration routes of the species in Western Europe, and particularly in Ireland. While data are sparse, Ireland is an important stopover location for the Icelandic populations in spring due to its location on the western fringe of Europe. This survey is the first time that pre- and postnuptial migrations have been systematically recorded at one location in Ireland. Apart from occasional records in bird publications, the only other survey of Whimbrel passage was carried out in Cork Harbour nearly four decades ago in the late 1970s (Pierce & Wilson 1980). Although that study reported large numbers over a three year period, it was confined to spring migration and the numbers recorded were at the time considered 'unprecedented'. Peak passage was recorded at Cork Harbour during week 17. The only annual data available on Whimbrel passage in Ireland since then is available on the Irish Birding website (www.irishbirding.com). In recent years, large numbers were reported in spring and autumn from many locations but highest counts were always in spring. For example, eight separate daily counts of over 200 birds were reported from 2009 to 2016. The five highest counts were in week 17 (22-28 April), two in week 18 (29 April–5 May) with one in week 35 (26 August–1 September). The highest daily total during this period was 600 at Rosscarberry (Cork) on 27 April 2011 (DOY 117). Peak migration occurred in week 17 at Cork Harbour (1977-1979), in Ireland as a whole (2009–2016) and at North Bull Island (2012–2015).

The seasonal bias towards higher numbers at North Bull Island in spring, compared to autumn, is consistent with the published status of the species in Ireland (Hutchinson 1989). The seasonal pattern at North Bull Island is also consistent with that reported from the Severn Estuary in the United Kingdom (Ferns *et al.* 1979). During spring migration the Severn Estuary was estimated to hold 74% of British migrant Whimbrels, but only 8% in autumn. This was in contrast to data from eastern England, where a higher proportion of birds occurred in autumn than in spring (Ferns *et al.* 1979, Prater 1981). A comparison of biometric data using wing lengths in that study confirmed that the majority of birds in the Severn Estuary were likely to be Icelandic, while those in eastern England were likely to be from Fennoscandian breeding populations.

Evidence on the migration routes of Whimbrels is available from ringing recoveries and, more recently, from the use of satellite transmitters and geolocators. Previous ringing recoveries in Ireland and west Britain included only one autumn bird and eight birds in spring (Gunnarsson & Guðmundsson 2016). These relative spring and autumn recovery rates did not fit the pattern that might be anticipated if post-breeding Icelandic Whimbrels, with their young, undertook the same migration route. More recently, a satellite transmitter fitted to a Whimbrel in northwest England showed that this bird flew directly from its Icelandic breeding ground to Guinea-Bissau (Whimbrel info 2016). This was the first evidence of a direct, non-stop Atlantic route for a Whimbrel to its wintering grounds in West Africa, a distance of 6,000 km. Estimates of energy expenditure required for such a long-distance migration were previously considered doubtful (Trolliet 2006). A more recent investigation using geolocators showed that this was not a one-off occurrence. All birds recovered in a study by Carneiro *et al.* (2015) (five of ten which were originally fitted with geolocators) followed the direct Atlantic route in autumn which took four or five days. The fact that all birds had undertaken the same direct route southwards is strong evidence that this is a major route for the Icelandic population and could explain the low autumn numbers recorded at North Bull Island and in Ireland in general. This study also showed that the birds fitted with geolocators undertook a completely different route in spring, with a non-stop flight from West Africa that was mainly over sea but included a stopover of approximately twelve days in Ireland and western Britain (Carneiro *et al.* 2015). At least two of these birds entered into the general area of the Irish Sea in April 2015 (Camilo Carneiro, personal communication) at the same time that a significant rise in numbers occurred at North Bull Island. This suggests that Whimbrels at North Bull Island in spring are of the *islandicus* race which breeds in Iceland. However, exactly what the proportion of the *islandicus* population stopover in Western Europe during prenuptial migration is unclear as there is new evidence that some, but possibly a significant proportion of the population, also undertake a continuous sea crossing directly from West Africa to Iceland (Alves *et al.* 2016).

There is mounting evidence that changing climatic conditions are impacting on wildfowl, wading bird and passerine distributions and migration patterns (Pearce-Higgins & Holt 2013, Pavón-Jordán *et al.* 2015, Miles *et al.* 2016) and that these impacts are likely to continue (Huntley *et al.* 2007). However, unlike some Icelandic breeding species, Whimbrels have not shown any advancement in earliest arrival dates or any relationship in arrival times with climatic drivers (Boyd & Petersen 2006, Gunnarsson & Tómasson 2011). This is consistent with the results from data collected for North Bull Island. A possible explanation for their lack of response compared to other Icelandic breeding wader species may be due to the fact that they are long-distance migrants with a short breeding season and therefore a more pressurised annual cycle (Gunnarsson 2010).

A possible response to milder conditions in winter might be an increase in the number of birds overwintering in Ireland. Although the *Bird Atlas 2007-2011* (Balmer *et al*. 2013) reported a 111% increase in 10 km squares with winter records this might not necessarily equate to an increase in the number of overwintering birds. From 2011/12 to 2015/16 about seven to 22 birds were reported annually during the winter months in Ireland (Irish Birding 2016), a figure that is remarkably close to the estimated 20 birds in the mid-1980s (Hutchinson 1989). Careful monitoring of wintering populations might indicate whether any significant change in winter numbers is occurring and whether this is related to climatic drivers.

In conclusion, although local in nature, the results from North Bull Island probably reflect a more general pattern for Ireland and west Britain. North Bull Island is a regular staging post for nationally important populations of Whimbrels in spring and autumn. Whilst it appears that the strong passage in spring belongs to the *islandicus* sub-species, the situation in autumn seems to be more complex. A direct route for all the Icelandic population in autumn, avoiding the west coast of Europe, would not explain the presence of birds at North Bull Island or at other sites in Ireland at that time of the year. At present, a single autumn ringing recovery from the Outer Hebrides provides the only evidence, albeit tenuous, that Icelandic birds do migrate southwards in autumn via Ireland and the United Kingdom. How many Icelandic birds take this route along the coast of Western Europe is at present unknown. However, given that *phaeopus* are more common during autumn migration than in spring in eastern Britain and mainland Europe (Fern *et al.* 1979, Prater 1981), it is not unreasonable to conclude that autumn flocks in Ireland possibly contain some *islandicus* from Iceland and *phaeopus* from the Faroese and Fennoscandian populations.

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